PROCEEDINGS

Central Plains Aquatic Bioassessment & Biocritera Symposium

A symposium on aquatic issues of the Central Plains

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18 & 19 September 2002

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The University of Kansas, Lawrence, Kansas

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PROCEEDINGS OF THE CENTRAL PLAINS AQUATIC BIOASSESSMENT & BIOCRITERIA SYMPOSIUM

A Symposium on Aquatic Issues of the Central Plains

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CONTENTS

Foreword ---------- iii.
Reviewer Comments ---------- iv.
Ecoregions of the Central Plains ---------- v.

SESSION 1 BIOASSESSMENT AND BIOMONITORING

Response of Stream Biological Communities to Agricultural Disturbances in Kansas: an Historical Overview with Comments on the Potential for Aquatic Ecosystem Restoration
R.T. Angelo, M.S. Cringan, and S.G. Haslouer ---------- 1

Developing Linkages Between Clean Sediment Indices and Biological Impairment
R.A. Kuhnle, A. Simon, and S.S. Knight ---------- 23

Biomonitoring Intermittent Streams Using Benthic Invertebrates
C. Rabeni ---------- 35

Big Muddy Bioassessment: Can Biocriteria Be Developed for the Lower Missouri River?
B.C. Poulton and R.J. Sarver ---------- 36

Dissolved Oxygen Flux, Nutrients, and Community Productivity in Some Central Plains Streams: What We Know and What We Think We Know!
D. Huggins, J. Anderson, and S. Wang ---------- 44

Development of a Multimetric Aquatic Stressors Index for the Central Mixed Grass Prairie Ecoregion
P. Gagnon, M. Lammert, J. Higgins, and T. FitzHugh ---------- 45

Environmental Impact Assessment of Three Anti-Microbial Chemicals
B.A. Wilson, V.H. Smith, F. DeNoyelles, Jr., and C.K. Larive ---------- 46

SESSION 2 LAKES AND WETLANDS

Development of a Comprehensive Lake and Reservoir Classification Strategy for Nebraska as a Model for Agricultural Dominated Ecosystems
J.C. Holz ---------- 47

An Overview of Harmful Algal Blooms (HABS): Diversity and Dynamics of Toxic Algae in Freshwater Lakes and Marine Habitats
R.G. Rhodes ---------- 48

Application of Wetland Bioassessment Protocols for Making Aquatic Life Beneficial Use Support Determinations in Montana
R. Apfelbeck ---------- 49

Wetlands Monitoring: The Big Picture
K. Mulder ---------- 53

The Kansas Wetland Survey
E. Carney ---------- 54

Using Aerial Photography and Satellite Imagery to Monitor Changes in the Vegetation of Cheyenne Bottoms Wildlife Area
M.E. Houts, K.P. Price, J.K. Oliver, and K. Sexton ---------- 56

Reservoir Classification in Agriculturally Dominated Eco-systems
A. Severn ---------- 63

SESSION 3 RIVERS AND STREAMS

Food Webs of the Great Rivers of the Central Basin: Application of Stable Isotopes in Bioassessment
M.D. Delong ---------- 64

An Assessment of Freshwater Biodiversity in the Central Mixed Grass Prairie Ecoregion
M.L Khoury, J. Higgins, T. FitzHugh, P. Gagnon, J. Hall, and B. Schreurs ---------- 74

Classification of Oklahoma Rivers and Preliminary Development of an Index of Biotic Integrity
J. Gerritsen, M.T. Barbour, B.D. Snyder, and C.A. Potts ---------- 78

Assessment of Ecological Condition in Headwater Streams of the Central Plains
R.W. Bouchard, Jr. and D.G. Huggins ---------- 100

Biological Assessment of South Logan Creek: A Service Learning Project in Northeastern Nebraska
B. Hayford ---------- 124

Gypsum Creek Stream Restoration Project
N. Davis ---------- 136

Foreword

The September 2002 Central Plains Bioassessment and Biocriteria Symposium, held in Lawrence, Kansas, followed in the wake of six workshops that brought aquatic scientists together to address regional issues of biocriteria and bioassessment. The workshops began in 1999 as an initiative of The Central Plains Center for BioAssessment (CPCB) to facilitate cooperation among the biological assessment community throughout the region. This initiative began with the formation of a biocriteria workgroup consisting of federal, state, and tribal agencies interested in the pursuit of information sharing, multi-state cooperation, standardization of bioassessment methodologies, and biocriteria development. The workgroup and additional scientists participate in the CPCB-hosted workshops one to two times per year. Workshop topics include: reference sites and conditions; sampling methodologies in the Central and Great Plains region; programmatic and other applied uses of biological information in protecting aquatic resources; stream habitat assessment; and stream reference conditions and landscape-scale bioassessment. CPCB posts the activities and outcomes of these workshops on its website (www.cpcb.ku.edu).

Though our work is primarily centered in USEPA Region 7, CPCB embraces an approach to aquatic ecology that transcends geopolitical boundaries and focuses on ecoregions found within and adjacent to the Central Plains. Thus, the September 2002 Central Plains Bioassessment and Biocriteria Symposium focused on research and application of biological data and bioassessments of water environs of the Central Plains area of the Great Plains Region of the United States with the goal of reaching a broad audience from beyond the borders of USEPA Region 7. Audience and presenters included 70 people from 12 states: Arkansas, Colorado, Illinois, Iowa, Kansas, Maryland, Minnesota, Mississippi, Missouri, Montana, Nebraska, and Oklahoma. What follows in this proceedings are the results of the hard work and efforts of the presenters, participants, and coordinators.

USEPA Region 7, KU Office of the Provost and Continuing Education, and BioDevices Corporation provided funding for this symposium. We thank the CPCB and KUCE staff that brought this symposium together. And most importantly we thank the presenters and attendees of the symposium for making it successful.

Reviewer Comments

Session 1 BioAssessment and Biomonitoring
TBA

Session 2 Lakes and Wetlands
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Session 3 Rivers and Streams

The six papers dealing with rivers and streams offered a diversity of research and application in aquatic biological assessment in terms of technique and scope, the latter ranging from great rivers to headwater streams. The intended and potential relevance of each to specific management issues was also diverse including some potentially new applications such as resource management and preservation, documentation of nuisance conditions, validation of restoration actions, in addition to the more commonplace Clean Water Act applications. This testifies to the broad applicability and relevance of biological assessment in the central U.S. While much of the science and technique of aquatic bioassessment has emphasized structural attributes, one paper in this session presented an interesting case for the inclusion of functional indicators. While several of the papers focused on some of the unique challenges offered by the aquatic faunas of central plains streams and rivers, there was a strong under girding of the approaches and techniques in each based on those that have been successfully used in the eastern and western U.S and elsewhere. Even though each paper made a genuine effort to reference and even emulate some of the more celebrated and contemporary examples, some recent and important contributions were overlooked. This may well be evidence that the field of aquatic bioassessment is making even more rapid progress, which highlights the need for improved technical awareness and transfer. Several other important issues were evident as requiring further development and research and include the selection and delineation of reference sites, metric testing and development, index derivation and calibration, sampling methods, depauperate faunas, and the appropriate spatial extent within which those activities take place. However, it was less than satisfying that we still seem to be grappling with baseline issues such as reference condition, sampling methods, and index choice and development, issues which by now should be a matter of routine. Perhaps the absence of detailed compendia, which chart the proper sequence and execution of these processes in regional bioassessment, contributes to this problem. Regional bioassessment centers like CPCB can play an important role in facilitating the type of applied research, collaboration, and dissemination of technology that is needed to achieve the kind of progress that is urgently needed in these areas.

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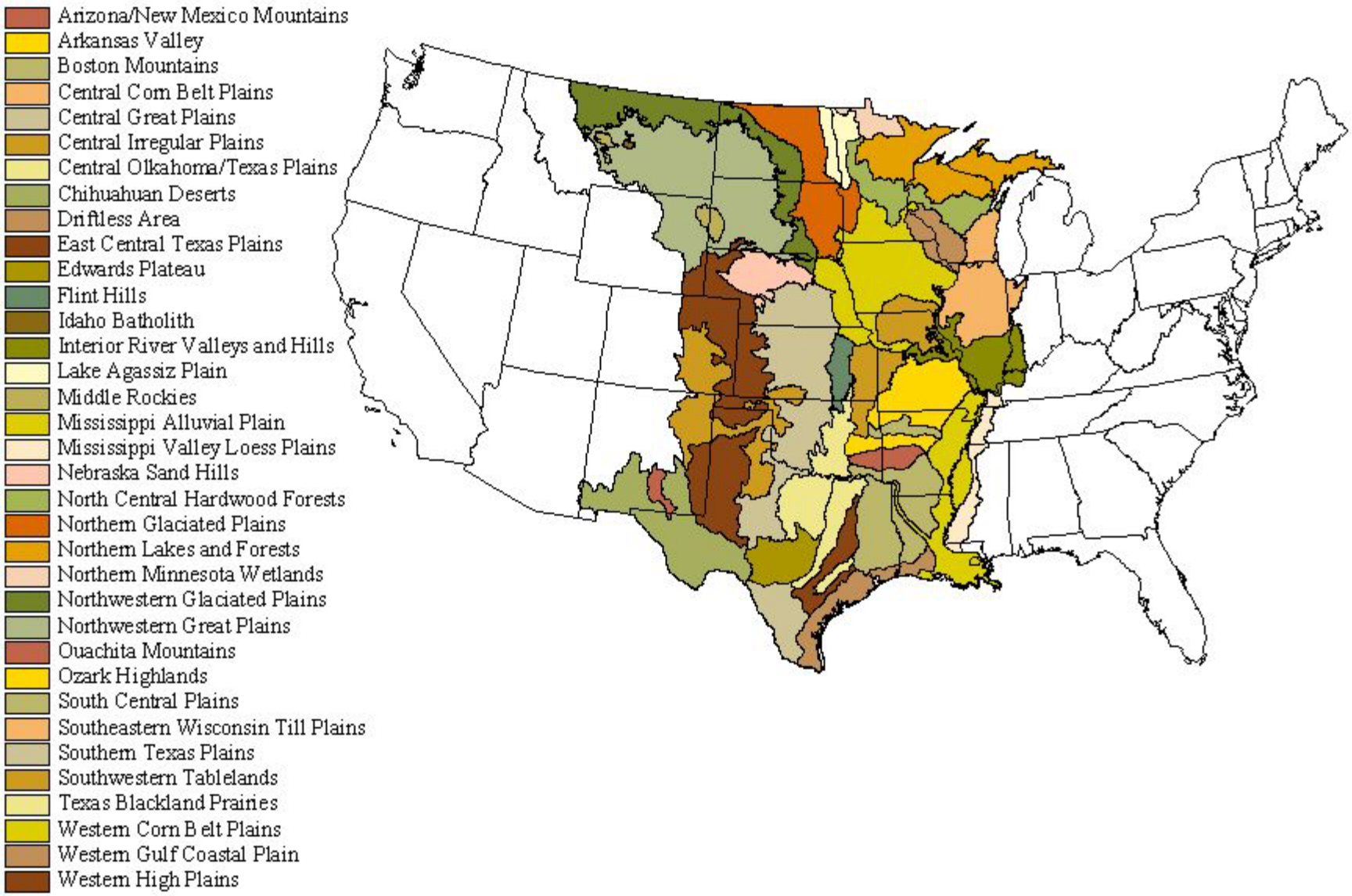

Ecoregions of the Central Plains of the United States (Omernik Level III).

RESPONSE OF STREAM BIOLOGICAL COMMUNITIES TO AGRICULTURAL DISTURBANCES IN KANSAS: AN HISTORICAL OVERVIEW WITH COMMENTS ON THE POTENTIAL FOR AQUATIC ECOSYSTEM RESTORATION

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Abstract. Environmental changes resulting from agricultural development and supporting water use policies have profoundly altered the composition of stream biological communities in Kansas. This is reflected, most notably, in the extirpation of several native fish and freshwater molluscan species and in the declining distribution and abundance of many other indigenous aquatic taxa. Although governmental programs for reducing the environmental impacts of agriculture have received strong public support and expanding budgetary allocations, their ultimate aim has not been articulated with respect to anticipated improvements in biological condition. The Clean Water Act seeks "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." In the context of this federal objective, some knowledge of the pre-settlement stream characteristics and the identification and study of minimally disturbed (i.e., reference) ecosystems are necessary precursors to the development of meaningful biological restoration goals. We briefly explore the available literature on historical stream conditions and biological communities in the central plains, discuss changes in the aquatic environment that accompanied the arrival and expansion of intensive agriculture, and consider the general attributes of minimally disturbed water bodies in this region. Based on this review, we address the potential role of historical information and reference data in the development of biological restoration goals for streams in Kansas.

Key words: ecological integrity, reference condition, rivers, siltation, habitat loss, extirpation, bioassessment, fish, mussels, macroinvertebrates, water quality standards, Clean Water Act.

Introduction

During the late nineteenth century, the grassland interior of North America was transformed from a veritable wilderness into one of the largest and most productive agricultural regions on Earth. This process wrought fundamental changes in the character of the land and, ultimately, the rivers and creeks that drained the land. Newly plowed soils were exposed to the erosive forces of wind and water, streams received heavy influxes of silt, and a general decline occurred in the more vulnerable native fish and shellfish populations (Mead 1896, 1903, Doze 1924, Franzen and Leonard 1943, Williams 1954, Murray and Leonard 1962, Metcalf 1966, Warren 1974). Further declines in aquatic life accompanied the proliferation and concentration of domestic livestock; the construction of dams, ditches, and levees; the channelization of streams; the loss of riparian woodlands and wetlands; the introduction of chemical fertilizers and biocides and the advent of center pivot irrigation (Mead 1896, Franzen and Leonard 1943, Fitch and Lokke 1956, Gray 1968, Nicholson 1968, Cross and Braasch 1969, Prophet and Edwards 1973, Huggins and Moss 1975, Metcalf 1983, Cross et al. 1985, Dahl 1990, Pigg

1991, Hoke 1996, 1997, Obermeyer et al. 1997, Eberle et al. 2002, Mammoliti 2002). Today, few streams in this region have remained unaffected by large scale agriculture. Nearly all have experienced some change in their original physical structure, hydrology, water quality, and biology as a result of food production practices and policies implemented during the past 150 years.

Pre-settlement stream conditions

Written descriptions of streams in the central plains date from the Spanish military expeditions of the mid sixteenth century. In 1541, Lieutenant Juan Jarmillo characterized what is now central Kansas as a region of "table-lands, plains, and charming rivers with fine waters" while acknowledging the agricultural potential of the lush and seemingly endless prairie (Barry 1972). Nearly two hundred years later, French explorers described the watershed of the Kansas River as "the most beautiful land in the world" and the stream itself as "a beautiful river" abounding in beaver, otter, and other fur bearing animals (Barry 1972). Sporadic references to fish, waterfowl, and other aquatic and semiaquatic biota accompanied the early nineteenth century accounts of Lewis and Clark, Zebulon Pike, Stephen Long, and other travelers in this region (Cutler 1883, Thwaites 1905, 1959, 1966). Railroad surveys and related investigations yielded additional information on the aquatic flora and fauna of the central plains and generated more accurate maps and the earliest known photographs of many streams (e.g., Girard 1858, Cope 1865, Caldwell 1937, Charlton 2000). More detailed biological surveys were initiated in the mid 1870s and continued well into the next century (Snow 1875, Aughey 1877, Wheeler 1878, Gilbert 1884, 1885, 1886, 1889, Call 1885a, 1885b, 1885c, 1885d, 1886, 1887, Cragin 1885, Graham 1885, Faxon 1885, Popenoe 1885, Evermann and Fordice 1886, Hay 1887, Harris 1901, Scammon 1906, Baker 1909, Hanna 1909, Sampson 1913, Utterback 1915, 1916, Isely 1925). Although most of these surveys followed the initial onset of intensive agriculture, they documented the occurrence of several freshwater species that would soon be extirpated from specific watersheds or the region as a whole (Table 1, Fig. 1). Archeological and paleontological studies conducted in the latter half of the twentieth century shed further light on historical and prehistorical stream conditions and aquatic biological communities in the central plains (e.g., Wedel 1959, Hibbard and Taylor 1960, Miller 1966, 1970, Bradley 1973, Warren 1974, Thies 1981, 1996, Witty 1983). The early explorers and settlers in this region encountered many streams with shifting sand bottoms, shallow braided channels, wide seasonal fluctuations in flow, and a general paucity of woody riparian vegetation (Hutchinson 1872, Smyth 1885, Gregg 1952). With headwaters in the Rocky Mountains, the largest rivers (Missouri, Arkansas, Platte) generally attained their peak flows during early summer in response to snowmelt runoff. Much of this water infiltrated the sandy alluviums and underlying aquifers, but a portion returned to the river channels as base flow during drier seasons. These large streams were deceptively swift and maintained an obvious yellow or milky turbidity during periods of normal flow owing primarily to the suspension and transport of fine sand (Smyth 1885, Mead 1896, Clark and Gillette 1911, Gregg 1952; cf., Allan 1995). At lower

flows, groundwater intrusion into the river channels effectively minimized silt deposition and sediment compaction, producing a semi-buoyant substrate or easily yielding "quicksand" (Marcy 1859, Gregg 1952, Dodge 1959; cf., Langsdorf 1950, Cross and Moss 1987). Fish and invertebrate species endemic to these rivers possessed morphological and reproductive adaptations to the turbid water, shallow channels, high current velocities, and unstable stream bottoms (Cross 1967, Pflieger and Grace 1987; cf., Burks 1953, pg. 80). In contrast, many smaller rivers and creeks in the central plains were noted for their transparency, alternating pools and riffles, and variable substrate composition (Smucker 1856, Thwaites 1905, Caldwell 1937, Barry 1972). These supported few endemic taxa but many peripheral (primarily eastern) species requiring clear water and stable stream bottoms (Isely 1925, Cross 1967). Naturally saline streams, saltwater and freshwater marshes, and oxbow, playa, and sand dune lakes also supported their own distinctive assemblages of plants and animals and comprised important stopover points for migratory waterfowl (e.g., Smucker 1856, Cutler 1883, Mead 1906, Gregg 1952, Barry 1972). References to heavily silted water bodies in this region were uncommon initially but increased in frequency with the passage of time and westward expansion of agriculture (Mead 1896, 1903, Doze 1924, Metcalf 1966). Persistent blooms of algae and other manifestations of nutrient enrichment also were seemingly rare or at least seldom mentioned by the early travelers. Although the residual pools of intermittent streams were occasionally described as stagnant or "strongly seasoned" with buffalo urine, the great herds of bison and elk roamed incessantly in search of food and exerted only transient influences on local water quality (cf., Mead 1896, Gregg 1952, Thwaites 1966). Much of this region was prone to recurrent drought, and even some of the larger rivers could be reduced at times to a few isolated pools crowded with fish and other aquatic life (Smyth 1885, Mead 1896). Recovery of biological populations following the return of normal flows was expedited by the migration and drift of organisms from environmental refuges such as spring-fed tributaries, beaver dam ponds, deeper pools created by channel irregularities, and distant stream reaches less coupled to the local weather conditions. This capacity for rapid recovery from natural perturbations was one of the original hallmarks of stream biological communities in the central plains; however, persistent droughts (lasting for years or decades) and climatological fluctuations (spanning centuries or millennia) did induce compositional shifts in native plant and animal assemblages (Weaver and Albertson 1936, Franzen and Leonard 1943, Deacon 1961, Cross 1970, Bryson 1980, Cross and Collins 1995, Distler and Bleam 1995, Kay 1998). Permanent springs, spring-fed streams, and artesian marshes were less influenced by fluctuations in weather and climate. Some of these systems supported relict fish and invertebrate assemblages traceable to earlier glacial periods and ancestral stream linkages (Franzen 1944, Metcalf 1966, Cross and Moss 1987, Fausch and Bestgen 1997, Angelo et al. 2002; cf., Knerr 1901, Hibbard and Taylor 1960) (Fig. 2).

Contemporary stream conditions

Reference streams.—A few isolated creeks and river segments in the central plains have largely escaped the environmental vagaries of intensive agriculture and other modern human disturbances. These so-called reference streams facilitate the study of minimally altered ecological systems and provide convenient benchmarks for gauging the impacts of modern civilization on similar water bodies located in the same geographical area. Reference streams also provide insight into the natural resources that supported aboriginal cultures for centuries and ultimately attracted the earliest settlers to this region. Some provide critical habitat for threatened and endangered species and play an important role in fish and wildlife conservation efforts. Nearly all are appreciated for their outstanding scenic qualities. Systematic attempts to inventory reference streams in the central plains have been initiated only recently (CPCB 1999), but it is clear that the best candidates are restricted to a few smaller water bodies flowing through exceptionally well managed native grasslands. These possess water quality and hydrological characteristics approaching the historical norm, retain nearly all known elements of their original biological communities, and support few exotic species (Table 2). Larger streams in this region generally have experienced more substantive changes in their biotic and abiotic attributes. Although some pass through essentially intact corridors of native vegetation and superficially resemble the historical condition over restricted reaches, the natural flow regime has been modified dramatically in nearly all such ecosystems (Tomelleri 1984, Cross et al. 1985, Pflieger and Grace 1987, Sanders et al. 1993, Fausch and Bestgen 1997). The need to protect and maintain highly valued streams is recognized in the antidegradation provisions of state and federal water quality regulations (EPA 1983, KDHE 2002a); however, the scope of impending changes to reference systems extends well beyond the existing purview of these laws. Factors other than agriculture and conventional pollution sources (e.g., proposed drinking water reservoirs and diversions, landfills, highways, pipelines, residential developments, industrial parks, introduced species) pose the most immediate threats to the biological integrity of some of these water bodies.

Streams deviating from the reference condition.—Most streams in the central plains deviate markedly from the regional (or ecoregional) reference condition. With respect to the interim goals of the Clean Water Act, these streams may be partitioned into (1) those that fully support their designated aquatic life support (ALS) uses under state surface water quality standards and (2) those that are deemed only partially supportive or non-supportive of these uses. The second category includes water bodies listed as biologically impaired systems under section 303(d) of the Clean Water Act, based on noncompliance with applicable numeric or narrative water quality criteria. Traditionally, biological indicators utilized by the Kansas Department of Health and Environment (KDHE) for translating narrative criteria have included four community-based measures of pollution tolerance, among them the macroinvertebrate biotic index (MBI) and Ephemeroptera-Plecoptera-Trichoptera (EPT) index (Fig. 3). Recent statewide assessments also have considered documented declines in local

freshwater mussel assemblages (KDHE 2000). Based on this combination of biological indicators, only about half (54%) of the monitored streams in Kansas are deemed fully supportive of their designated ALS uses (cf., KDHE 2002b). Community-based biological measures, and particularly macroinvertebrate indicators such as EPT and mussel assemblage decline, are responsive to a wide variety of environmental stressors and provide integrated measures of ecological condition over time frames ranging from weeks to years (or much longer if consideration is given to the collection and identification of molluscan shell material) (Rosenberg and Resh 1993, KDHE 2000). Because short-term fluctuations in rainfall and stream low may elicit transient changes in the relative abundance of certain fish and aquatic invertebrate species, at least 3-5 years of monitoring data normally are needed in this region to confidently assign streams to their respective biological condition categories (KDHE 2000). Future ALS evaluations in Kansas will consider the use of more sophisticated multimetric indices and additional statistical models for categorizing streams in this manner (e.g., Davies et al. 1999, Karr and Chu 1999). Supplemental monitoring data obtained from various cooperating state and federal agencies and academic institutions also are expected to play an increasingly important role in statewide evaluations of stream condition.

Restoration goals

Efforts to alleviate the impacts of modern agriculture on the aquatic environment have focused primarily on the abatement of soil erosion and proper management of chemical fertilizers, biocides, and livestock wastes (e.g., Devlin 2000). Although the wider adoption of agricultural best management practices should lead to measurable reductions in stream contaminant levels (Fig. 4), water quality is not the only factor limiting the survival and propagation of aquatic species in this region. Throughout much of western Kansas, decades of irrigated crop production have exacted a heavy toll on stream life by lowering groundwater tables, reducing base flows, and transforming formerly perennial water bodies into intermittent or ephemeral systems (Jordon 1982, Tomelleri 1984, Cross et al. 1985, Cross and Moss 1987, Sanders et al. 1993, Hoke 1997, Eberle et al. 2002, Fausch and Bestgen 1997) (Fig. 5; cf., Fig. 1D). In some areas of northeastern Kansas and southeastern Nebraska, stream channelization has radically simplified the original aquatic habitats and decimated a formerly diverse fish and shellfish fauna (Witt 1970, Bliss and Schainost 1973, Delich 1983, Hoke 1996). Impoundments (large and small) throughout this region have encouraged the establishment of predominantly nonnative fish assemblages, fragmented the remaining stream habitats, and diminished the seasonal peak flows needed by certain native fishes for spawning and egg development (Cross and Moss 1987, Fausch and Bestgen 1997, Dean et al. 2002, Gido et al. 2002, Mammoliti 2002). The complete restoration of these degraded aquatic ecosystems would require massive habitat rehabilitation efforts and sweeping changes in the laws, policies, and attitudes currently controlling the use and allocation of water in this region (cf., Sherow 2002). Less effective (but more readily implemented) options for

partially offsetting the historical effects of agriculture would include the establishment of legally binding minimum stream flows, the expansion of hatchery restocking programs for native fish and shellfish (e.g., Barnhart 1999), the selective removal of lowhead dams and other barriers to fish migration, the installation of fish ladders and elevators on larger dams, and other related management initiatives – all in addition to concurrent improvements in agricultural practices. Most of these concepts are not new. For example, the importance of maintaining migrational corridors for fish was emphasized repeatedly by Kansas officials during the late nineteenth century but never seriously considered in the course of water resource development (Long 1878, 1880, 1883, Gile 1885, Fee 1886, 1888, Brumbaugh 1891, Wampler 1894, 1895). Biological restoration goals for flowing waters normally should be based on the documented attributes of regional reference streams and stream reaches. Consistent with the intent of the Clean Water Act, each reference system should exhibit a high degree of physical and chemical integrity and support "a balanced, integrated, adaptive community of organisms having a composition and diversity comparable to that of the natural habitats of the region" (Frey 1977). This stipulation does not preclude the absence of a few historically occurring peripheral species, but it does sharply limit the loss of historically dominant taxa and those species deemed integral to the function and identity of the community as a whole. Similarly, the presence of a few non-indigenous forms in low densities may be acceptable for reference purposes, provided the measured attributes of the biological community are otherwise representative of the wider body of reference systems. Large areas of the central plains probably no longer retain true reference streams as a result of decades of intensive agricultural development and other modern human disturbances. In such cases, restoration goals should be set initially at the highest level of biological integrity that can be supported by the prevailing land use, the adoption of all appropriate agricultural best management practices, and the completion of all readily implemented physical (restorative) improvements to the aquatic environment. Longer term planning efforts should draw upon historical narrative accounts and photographs, early biological survey reports, museum fish and shellfish collections, published archeological studies, and similar sources of information to ensure that the projected changes in aquatic plant and animal assemblages trend in the direction of the pre-settlement biological condition. Recovery goals should be reexamined periodically and modified, as needed, to reflect new scientific and historical findings and ongoing advances in the field of environmental restoration. Such an intensive, iterative, and persistent planning process is in keeping with the magnitude of the environmental changes that have taken place in this region during the past 150 years.

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Literature Cited

Allan, J. D. 1995. Stream ecology: structure and function of running waters. Chapman and Hall, London.

Angelo, R. T. 1994. Impacts of declining stream flow on surface water quality. Pages 1-2 *in* Proceedings of the Eleventh Annual Conference on Water and the Future of Kansas, Kansas State University, Manhattan, Kansas.

Angelo, R. T., and M. S. Cringan. 2002. Delta hydrobe: a newly discovered snail in Kansas. Kansas Pearly Mussel Newsline 7:3.

Angelo, R. T., and M. S. Cringan. 2003. Rediscovery of the black sandshell, *Ligumia recta* (Lamarck, 1819), in Kansas. Transactions of the Kansas Academy of Science (in press).

Angelo, R. T., M. S. Cringan, and J. E. Fry. 2002. Distributional revisions and new and amended occurrence records for prosobranch snails in Kansas. Transactions of the Kansas Academy of Science 105:246-257.

Aughey, S. 1877. Catalogue of the land and fresh-water shells of Nebraska. Bulletin of the United States Geological and Geographical Survey of the Territories 3:697-704.

Baker, F. C. 1909. Mollusks from Kansas and Oklahoma. Nautilus 23:91-94.

Barnhart, M. C. 1999. Successful stocking of Neosho muckets in the Fall and Verdigris rivers. Kansas Pearly Mussel Newsline 7:1-2.

Barry, L. 1972. The beginning of the West: annals of the Kansas gateway to the American West 1540-1854. Kansas State Historical Society, Topeka, Kansas.

Bliss, Q. P., and S. Schainost. 1973. Nemaha Basin stream inventory report. Nebraska Game and Parks Commission, Lincoln, Nebraska.

Bradley, L. E. 1973. Subsistence strategy at a late archaic site in south-central Kansas. Unpublished Master's thesis, University of Kansas, Lawrence, Kansas.

Branson, B. A. 1963. Additions to and distributional annotations on the Kansas gastropod fauna. Transactions of the Kansas Academy of Science 66:72-75.

Branson, B. A. 1966. A partial biological survey of the Spring River Drainage in Kansas, Oklahoma and Missouri. Part I. Collecting sites, basic limnological data, and mollusks. Transactions of the Kansas Academy of Science 69:242-293.

Brumbaugh, J. M. 1891. Seventh biennial report of the State Fish Commissioner. Kansas Publishing House, Topeka, Kansas.

Bryson, R. A. 1980. Ancient climes on the Great Plains. Natural History 89:64-73.

Burian, S. K. 2001. Revision of the genus *Leptophlebia* Westwood in North America (Ephemeroptera: Leptophlebiidae: Leptophlebiinae). Bulletin of the Ohio Biological Survey, New Series 13:vi + 80 pp.

Burks, B. D. 1953. The mayflies, or Ephemeroptera, of Illinois. Bulletin of the Illinois Natural History Survey 26:1-216.

Caldwell, M. B. 1937. Exploring the Solomon River Valley in 1869. Kansas Historical Quarterly 6:60-76.

Call, R. E. 1885a. Contributions to a knowledge of the fresh-water Mollusca of Kansas, I: freshwater bivalves. Bulletin of the Washburn College Laboratory of Natural History 1:49-51.

Call, R. E. 1885b. Contributions to a knowledge of the fresh-water Mollusca of Kansas, II: freshwater univalves. Bulletin of the Washburn College Laboratory of Natural History 1:51-54.

Call, R. E. 1885c. Contributions to a knowledge of the fresh-water Mollusca of Kansas, III: freshwater bivalves. Bulletin of the Washburn College Laboratory of Natural History 1:93-97.

Call, R. E. 1885d. Contributions to a knowledge of the fresh-water Mollusca of Kansas, IV. Bulletin of the Washburn College Laboratory of Natural History 1:115-124.

Call, R. E. 1886. Fifth contribution to a knowledge of the fresh-water Mollusca of Kansas. Bulletin of the Washburn College Laboratory of Natural History 1:177-184.

Call, R. E. 1887. Sixth contribution to a knowledge of the fresh-water Mollusca of Kansas. Bulletin of the Washburn College Laboratory of Natural History 2:11-25.

Chapman, S. S., J. M. Omernik, J. A. Freeouf, D. G. Huggins, J. R. McCauley, C. C. Freeman, G. Steinauer, R. T. Angelo, and R. L. Schlepp. 2001. Ecoregions of Nebraska and Kansas (1:1,950,000 scale map with color poster, descriptive text, summary tables, and photographs). U. S. Geological Survey, Reston Virginia.

Charlton, J. 2000. Kaw River Valley scenes revisited after 130 years. Transactions of the Kansas Academy of Science 103:1-21.

Clark, H. W., and G. H. Gillette. 1911. Some observations made on Little River, near Wichita, Kansas, with reference to the Unionidae. Proceedings of the Biological Society of Washington 24:63-68.

Cope, C. H. 1979. Survey of the Unionidae considered for conservation status in Kansas. Final report to the Kansas Fish and Game Commission, Pratt, Kansas.

Cope, E. D. 1865. Note on the fishes brought from the Platte River, near Fort Riley, by Dr. Wm. A. Hammond. Proceedings of the Philadelphia Academy of Natural Sciences 17:78-85.

CPCB. 1999. Workshop on reference sites and conditions, 14-16 April 1999. Central Plains Center for BioAssessment, Kansas Biological Survey, University of Kansas, Lawrence, Kansas (http://www.cpcb.ku.edu/workshops/html/index.htm).

Cragin, F. W. 1885. Preliminary list of Kansas fishes. Bulletin of the Washburn College Laboratory of Natural History 1:105-111.

Cross, F. B. 1967. Handbook of fishes of Kansas. Miscellaneous Publication No. 45, Museum of Natural History, University of Kansas, Lawrence, Kansas.

Cross, F. B. 1970. Fishes as indicators of Pleistocene and Recent environments in the central plains.

Pages 241-257 *in* Pleistocene and Recent environments of the central Great Plains. Special Publication 3, Department of Geology, University of Kansas, Lawrence, Kansas.

Cross, F. B., and M. Braasch. 1969. Qualitative changes in the fish-fauna of the upper Neosho River System, 1952-1967. Transactions of the Kansas Academy of Science 71:350-360.

Cross, F. B., and J. T. Collins. 1995. Fishes in Kansas. Natural History Museum, University of Kansas, Lawrence, Kansas.

Cross, F. B., O. T. Gorman, and S. G. Haslouer. 1983. The Red River shiner, *Notropis bairdi*, in Kansas with notes on depletion of its Arkansas River cognate, *Notropis girardi*. Transactions of the Kansas Academy of Science 86:93-98.

Cross, F. B., and R. E. Moss. 1987. Historic changes in fish communities and aquatic habitats in plains streams of Kansas. Pages 155-177 *in* W. J. Matthews and D. C. Heins (editors). Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman, Oklahoma.

Cross, F. B., R. E. Moss, and J. T. Collins. 1985. Assessment of dewatering impacts on stream fisheries in the Arkansas and Cimarron rivers. Final report to the Kansas Fish and Game Commission, Pratt, Kansas.

Cutler, W. G. 1883. History of the State of Kansas. A. T. Andreas, Chicago.

Dahl, T. E. 1990. Wetlands losses in the United States 1780's to 1980's. U. S. Fish and Wildlife Service, U. S. Department of the Interior, Washington, D. C.

Davies, S. P., L. Tsomides, J. L. DiFranco, and D. L. Courtemanch. 1999. Biomonitoring retrospective: fifteen year summary for Maine rivers and streams. Bureau of Land and Water Quality, Division of Environmental Assessment, Maine State Department of Environmental Protection, Augusta, Maine.

Deacon, J. E. 1961. Fish populations, following a drought, in the Neosho and Marais des Cygnes rivers of Kansas. University of Kansas Publications, Museum of Natural History 13:359-428.

Dean, J., D. Edds, D. Gillette, J. Howard, S. Sherraden, and J. Tiemann. 2002. Effects of lowhead dams on freshwater mussels in the Neosho River, Kansas. Transactions of the Kansas Academy of Science 105:232-240.

Delich, C. 1983. River portraits: the Nemahas. NebraskaLand Magazine 61:115-118.

Devlin, D. L. 2000. Water quality protection: best management practices for cropland. Agricultural Experiment Station and Cooperative Extension Service, Kansas State University, Manhattan, Kansas (http://www.oznet.ksu.edu/library/h20ql2/MF2462.pdf).

Distler, D. A., and D. E. Bleam. 1995. Decline in the diversity of Bivalvia, Ninnescah River, Kansas. Transactions of the Kansas Academy of Science 98:156-159.

Dodge, R. I. 1959. The plains of the great West and their inhabitants. Archer House, New York.

Doze, J. B. 1924. Fifth biennial report of the Kansas State Fish and Game Department. Pratt, Kansas.

DuBois, M. B. 1981. Additions and corrections to the freshwater mussel

fauna from Kansas (Pelecypoda: Eulamellibranchia: Unionidae). Technical Publications of the State Biological Survey of Kansas 10:89-94.

Eberle, M. E., E. G. Hargett, T. L. Wenke, and N. E. Mandrak. 2002. Changes in fish assemblages, Solomon River Basin, Kansas: habitat alterations, extirpations, and introductions. Transactions of the Kansas Academy of Science 105:178-192.

EPA. 1983. Water quality standards (regulations promulgated by U. S. Environmental Protection Agency). Federal Register 48:51405-51413.

Evermann, B. W., and M. W. Fordice. 1886. List of fishes collected in Harvey and Cowley counties, Kansas. Bulletin of the Washburn College Laboratory of Natural History 1:184-186.

Fausch, K. D., and K. R. Bestgen. 1997. Ecology of fishes indigenous to the central and southwestern Great Plains. Pages 131-166 *in* F. L. Knopf and F. B. Samson (editors). Ecology and conservation of Great Plains vertebrates. Ecological Studies 125, Springer- Verlag, New York.

Faxon, W. 1885. Preliminary catalogue of the crayfishes of Kansas. Bulletin of the Washburn College Laboratory of Natural History 1:140-142.

Fee, S. 1886. Fifth biennial report of the State Fish Commissioner. Kansas Publishing House, Topeka, Kansas.

Fee, S. 1888. Sixth biennial report of the State Fish Commissioner. Kansas Publishing House, Topeka, Kansas.

Fitch, H. S., and D. H. Lokke. 1956. The molluscan record of succession on the University of Kansas Natural History Reservation. Transactions of the Kansas Academy of Science 59:442-454.

Franzen, D. S. 1944. New state records of Mollusca from Kansas. Transactions of the Kansas Academy of Science 47:261-273.

Franzen, D. S., and A. B. Leonard. 1943. The Mollusca of the Wakarusa River Valley. University of Kansas Science Bulletin 29:363-439.

Frey, D. G. 1977. Biological integrity of water: an historical approach. Pages 127-140 *in* R. K. Ballentine and L. J. Guarraia (editors). The integrity of water: a symposium. U. S. Environmental Protection Agency, Washington, D. C.

Gido, K. B., C. S. Guy, T. R. Strakosh, R. J. Bernot, K. J. Hase, and M. A. Shaw. 2002. Long-term changes in the fish assemblages of the Big Blue River Basin 40 years after the construction of Tuttle Creek Reservoir. Transactions of the Kansas Academy of Science 105:193-208.

Gilbert, C. H. 1884. Notes on the fishes of Kansas. Bulletin of the Washburn College Laboratory of Natural History 1:10-16.

Gilbert, C. H. 1885. Second series of notes on the fishes of Kansas. Bulletin of the Washburn College Laboratory of Natural History 1:97-99.

Gilbert, C. H. 1886. Third series of notes on Kansas fishes. Bulletin of the Washburn College Laboratory of Natural History 1:207-211.

Gilbert, C. H. 1889. Fourth series of notes on the fishes of Kansas. Bulletin of the Washburn College Laboratory of Natural History 2:38-43.

Gile, W. S. 1885. Fourth biennial report of the State Fish Commissioner. Kansas Publishing House, Topeka, Kansas.

Girard, C. 1858. General report upon the zoology of the several Pacific railroad routes: fishes. 400 pp. *in* Reports of explorations and surveys, to ascertain the most practicable and economical route for a railroad from the Mississippi River to the Pacific Ocean. Volume 10. U. S. War Department, Washington, D.C. (http://cprr.org/Museum/Pacific_RR_Surveys/).

Graham, I. D. 1885. Preliminary list of Kansas fishes. Transactions of the Kansas Academy of Science 9:69-78.

Gray, M. W. 1968. Water pollution and agriculture. Supplement to the Transactions of the Kansas Academy of Science 70:30-38.

Gregg, K. L. 1952. The road to Santa Fe: the journal and diaries of George Champlin Sibley and others pertaining to the surveying and marking of a road from the Missouri frontier to the settlements of New Mexico, 1825-1827. University of New Mexico Press, Albuquerque, New Mexico.

Hacker, R. 1980. Unionid mussels of the Walnut River Basin, Kansas. Unpublished Master's thesis, Wichita State University, Wichita, Kansas.

Hanna, G. D. 1909. Mollusca of Douglas County, Kansas. Nautilus 23:81-82, 94-96.

Harris, A. 1901. Annotated catalogue of the crayfishes of Kansas. Transactions of the Kansas Academy of Science 17:115.

Hay, O. P. 1887. A contribution to the knowledge of the fishes of Kansas. Proceedings of U. S. National Museum 10:242-253.

Hibbard, C. W., and D. W. Taylor. 1960. Two late Pleistocene faunas from southwestern Kansas. Contributions from the University of Michigan Museum of Paleontology 16:1-223.

Hoke, E. 1996. The unionid mollusks of the Big and Little Nemaha river basins of southeastern Nebraska and northeastern Kansas. Transactions of the Nebraska Academy of Sciences 23:37-57.

Hoke, E. 1997. The unionid mollusks of the upper Kansas Basin of northwestern Kansas and southwestern Nebraska. Transactions of the Nebraska Academy of Sciences 24:35-62.

Huggins, D. G., and R. E. Moss. 1975. Fish population structure in altered and unaltered areas of a small Kansas stream. Transactions of the Kansas Academy of Science 77:18-30.

Hutchinson, C. C. 1872. The Arkansas Valley. April 12, Wichita City Eagle, Wichita, Kansas.

Isely, F. B. 1925. The fresh-water mussel fauna of eastern Oklahoma. Proceedings of the Oklahoma Academy of Science 4:43-118.

Jordon, P. R. 1982. Rainfall-runoff relations and expected streamflow in western Kansas. Bulletin of the Kansas Water Office 25:1-42.

Karr, J. R., and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington, D. C.

Kay, M. 1998. The Great Plains setting. Pages 16-47 *in* W. R. Wood (editor). Archaeology on the Great Plains. University of Kansas Press, Lawrence, Kansas.

KDHE. 2000. Stream biological monitoring program quality assurance management plan. 69 pp. *In* Division of Environment quality management plan. Part III: program level quality assurance management plans. Technical Services Section, Bureau of Environmental Field Services, Kansas Department of Health and Environment, Topeka, Kansas.

KDHE. 2002a. Surface water quality standards (regulations promulgated by Kansas Department of Health and Environment). Kansas Register 21:2096-2109.

KDHE. 2002b. Kansas water quality assessment (305(b) report). Bureau of Environmental Field Services, Kansas Department of Health and Environment, Topeka, Kansas.

KDWP. 2000. Article 15: nongame, threatened and endangered species. Kansas Administrative Regulations 5:575-578.

KDWP. 2002. Locational records for threatened and endangered species. Unpublished data. Kansas Department of Wildlife and Parks, Pratt, Kansas.

Knerr, E. B. 1901. The Muscotah artesian wells. Transactions of the Kansas Academy of Science 17:53-54.

Langsdorf, E. 1950. The first survey of the Kansas River. Kansas Historical Quarterly 18:146-158.

Layher, B. 2002. Recovery plan for the Scott riffle beetle, *Optioservus phaeus* Gilbert, in Kansas. Final report to the Kansas Department of Wildlife and Parks, Pratt, Kansas.

Liechti, P. M. 1984. Population study of *Pomatiopsis lapidaria* (Say), a small amphibious snail of endangered status in Kansas. Report No. 28, Kansas Biological Survey, University of Kansas, Lawrence, Kansas.

Liechti, P. M., and D. G. Huggins. 1977. Unionacean mussels of Kansas. Technical Publications of the State Biological Survey of Kansas 4:17-30.

Long, D. B. 1878. First biennial report of the Commissioner of Fisheries, of the State of Kansas, for the years 1877-8. Kansas Publishing House, Topeka, Kansas.

Long, D. B. 1880. Second biennial report of the State Fish Commissioner. Kansas Publishing House, Topeka, Kansas.

Long, D. B. 1883. Third biennial report of the State Fish Commissioner. Kansas Publishing House, Topeka, Kansas.

Mammoliti, C. S. 2002. The effects of small watershed impoundments on native stream fishes: a focus on the Topeka shiner and hornyhead chub. Transactions of the Kansas Academy of Science 105:219-231.

Marcy, R. B. 1859. A hand-book for overland expeditions. Harpers and Brothers, New York.

Mead, J. R. 1896. A dying river. Transactions of the Kansas Academy of Science 14:111-112.

Mead, J. R. 1906. The Saline River country in 1859. Transactions of the Kansas State Historical Society 9:8-19.

Mead, S. G. 1903. The drying-up of pools and streams in central Kansas. Transactions of the Kansas Academy of Science 18:213-214.

Metcalf, A. L. 1966. Fishes of the Kansas River System in relation to zoogeography of the Great Plains. University of Kansas Publications,

Museum of Natural History 17:23-189.
Metcalf, A. L. 1980. Unionacean mussels, past and present, from six streams in Kansas and Oklahoma. Transactions of the Kansas Academy of Science 83:1-19.
Metcalf, A. L. 1983. Mortality in unionacean mussels in a year of drought. Transactions of the Kansas Academy of Science 86:89-92.
Miller, B. B. 1966. Five Illinoian molluscan faunas from the southern Great Plains. Malacologia 4:173-260.
Miller, B. B. 1970. The Sandahl molluscan fauna (Illinoian) from McPherson County, Kansas. Ohio Journal of Science 70:39-50.
Minckley, W. L., and F. B. Cross. 1959. Distribution, habitat and abundance of the Topeka shiner, *Notropis topeka* (Gilbert) in Kansas. American Midland Naturalist 6:210-217.
Murray, H. D., and A. B. Leonard. 1962. Handbook of unionid mussels in Kansas. Miscellaneous Publication No. 28, Museum of Natural History, University of Kansas, Lawrence, Kansas.
Nicholson, H. P. 1968. Pesticides: a current water quality problem. Supplement to the Transactions of the Kansas Academy of Science 70:39-44.
Obermeyer, B. K., D. R. Edds, E. J. Miller, and C. W. Prophet. 1997. Range reductions of southeast Kansas unionids. Pages 108-116 *in* K. S. Cummins, A. C. Buchanan, C. A. Mayer, and T. J. Naimo (editors). Conservation and management of freshwater mussels II: initiatives for the future. Upper Mississippi River Conservation Committee, Rock Island, Illinois.
Obermeyer, B. K., D. R. Edds, and C. W. Prophet. 1995. Distribution and abundance of federal candidate mussels (Unionidae) in southeast Kansas. Final report to the Kansas Department of Wildlife and Parks, Emporia, Kansas.
Pflieger, W. L. , and T. B. Grace. 1987. Changes in the fish fauna of the lower Missouri River, 1940- 1983. Pages 166-177 *in* W. J. Matthews and D. C. Heins (editors). Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman, Oklahoma.
Pigg, J. 1991. Decreasing distribution and current status of the Arkansas River shiner, *Notropis girardi*, in the rivers of Oklahoma and Kansas. Proceedings of the Oklahoma Academy of Science 71:5-15.
Popenoe, E. A. 1885. List of Unionidae, collected in Kansas rivers, with localities. Transactions of the Kansas Academy of Science 9:78-79.
Prophet, C. W., and N. L. Edwards. 1973. Benthic macroinvertebrate community structure in a Great Plains stream receiving feedlot runoff. Water Resources Bulletin 9:583-589.
Rosenburg, D. M., and V. H. Resh. 1993. Introduction to freshwater biomonitoring and benthic macroinvertebrates. Pages 1-9 *in* D. M. Rosenburg and V. H. Resh (editors). Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York.
Sampson, F. A. 1913. A preliminary list of the Mollusca of Missouri (exclusive of the Unionidae).

Transactions of the Academy of Science of Saint Louis 22:67-108.
Sanders, R. M., D. G. Huggins, and F. B. Cross. 1993. The Kansas River system and its biota. Pages 295-326 *in* L. W. Hesse, C. B. Stalnaker, N. G. Benson, and J. R. Zuboy (editors). Proceedings of the symposium on restoration planning for the rivers of the Mississippi River ecosystem. Biology Report 19, National Biological Survey, U. S. Department of the Interior, Washington, D. C.
Scammon, R. E. 1906. The Unionidae of Kansas, part I: an illustrated catalogue of the Kansas Unionidae. Kansas University Science Bulletin 3:279-373, pl. 52-85.
Schloss, J. A., R. W. Buddemeier, and B. B. Wilson (editors). 2000. An atlas of the Kansas High Plains Aquifer. Educational Series 14, Kansas Geological Survey, University of Kansas, Lawrence, Kansas.
Schrank, S. J., C. S. Guy, M. R. Whiles, and B. L. Brock. 2001. Influence of instream and landscapelevel factors on the distribution of Topeka shiners *Notropis topeka* in Kansas streams. Copeia 2001:413-421.
Schuster, G. A., and M. B. DuBois. 1979. Additional new records of freshwater mussels (Bivalvia: Unionidae) from Kansas. Technical Publications of the State Biological Survey of Kansas 8:1-11.
Sherow, J. E. 2002. The art of water and the art of living. Kansas History 25:52-71.
Smucker, S. M. 1856. The life of Col. John Charles Fremont, and his narrative of explorations and adventures in Kansas, Nebraska, Oregon, and California. Miller, Orton and Mulligan, New York.
Smyth, B. B. 1885. The age of Kansas. Transactions of the Kansas Academy of Science 9:129-136.
Snow, F. H. 1875. The fishes of the Kansas River, as observed at Lawrence. State Board of Agriculture Annual Report 4:139-141.
Thies, R. M. 1981. Archeological investigations at John Redmond Reservoir, east central Kansas, 1979. Kansas State Historical Society, Topeka, Kansas.
Thies, R. M. 1996. The testing of 14LV92 and borrow area #1: phase III archeological investigations of KDOT project K-4441, Leavenworth County, Kansas. Kansas State Historical Society, Topeka, Kansas.
Thwaites, R. G. 1905. Account of an expedition from Pittsburgh to the Rocky Mountains performed in the years 1819, 1820. Volume 1. Arthur H. Clark, Cleveland, Ohio.
Thwaites, R. G. 1959. Original journals of the Lewis and Clark expedition 1804-1806. Volume 6. Antiquarian Press, New York.
Thwaites, R. G. 1966. Early western travels 1748-1846. Volume 16. AMS Press, New York.
Tomelleri, J. R. 1984. Dynamics of woody vegetation along the Arkansas River in western Kansas, 1870-1983. Unpublished Master's thesis, Fort Hays State University, Hays, Kansas.
Utterback, W. I. 1915. The naiades of Missouri. American Midland Naturalist 4:41-53, 69-152, 189-204, 244-273.
Utterback, W. I. 1916. The naiades of Missouri. American Midland

Naturalist 4:311-327, 339-354, 387-400, 432-464.
Wampler, J. W. 1894. Annual report of the Fish Commissioner, State of Kansas. Hamilton Printing, Topeka, Kansas.
Wampler, J. W. 1895. Second annual report of the Fish Commissioner, State of Kansas. Hamilton Printing, Topeka, Kansas.
Warren, R. E. 1974. Pre-Columbian unionacean (Mollusca, Pelecypoda) distribution in north-central Kansas. Abstract. Proceedings of the Nebraska Academy of Sciences and Affiliated Societies 84:17.
Weaver, J. E., and F. W. Alberton. 1936. Effects of great drought on the prairies of Iowa, Nebraska, and Kansas. Ecology 17:567-639.
Wedel, W. R. 1959. An introduction to Kansas archeology. Bulletin 174, Bureau of American Ethnology, Smithsonian Institution, Washington, D. C.
Wheeler, W. 1878. A partial list of the fishes of the Marais des Cygnes, at Ottawa. Transactions of the Kansas Academy of Science 6:33-34.
Williams, A. B. 1954. An explanation for the distribution of a North American crayfish. Ecology 35:573-575.
Witt, L. A. 1970. The fishes of the Nemaha River, Nebraska. Transactions of the Kansas Academy of Science 73:70-88.
Witty, T. A. 1983. Four archeological sites of the Perry Lake, Kansas. Kansas State Historical Society, Topeka, Kansas.

Table 1. Fishes and aquatic mollusks native to Kansas but now extirpated in the state, or nearly so. Fish listings were compiled originally by F. Cross and S. Haslouer on 5 February 1998. Mollusk listings represent the consensus opinion of the authors. Species (common names) marked with a single asterisk are not known to be vouchered by museum specimens, and original reports are vague or questionable with respect to collection locale. Species with a double asterisk have been collected in the state on one or more occasions during the past decade, but the presence of reproductively viable populations is considered doubtful. In addition to certain taxa listed below, 28 fish species and 23 aquatic molluscan species in Kansas are designated as endangered, threatened, or in need of conservation by the Kansas Department of Wildlife and Parks (KDWP 2000).

Class/family	Scientific name	Common name
Cephalaspidomorphi		
Petromyzontidae	*Ichthyomyzon castaneus* Girard, 1858	Chestnut lamprey
Actinopterygii		
Acipenseridae	*Acipenser fulvescens* Rafinesque, 1817	Lake sturgeon
	Scaphirhyncus albus (Forbes and Richardson, 1905)	Pallid sturgeon
Amiidae	*Amia calva* Linnaeus, 1766	Bowfin *
Hiodontidae	*Hiodon tergisus* Lesueur, 1818	Mooneye *
Cyprinidae	*Hybognathus argyritis* Girard, 1856	Western silvery minnow
	Hybopsis amblops (Rafinesque, 1820)	Bigeye chub *
	Macrhybopsis gelida (Girard, 1856)	Sturgeon chub
	Macrhybopsis meeki (Jordan and Evermann, 1896)	Sicklefin chub
	Notropis girardi Hubbs and Ortenburger, 1929	Arkansas River shiner
	Notropis heterolepis Eigenmann and Eigenmann, 1893	Blacknose shiner
	Notropis shumardi (Girard, 1856)	Silverband shiner
	Opsopoeodus emiliae Hay, 1881	Pugnose minnow *
	Platygobio gracilis (Richardson, 1836)	Flathead chub
Catostomidae	*Moxostoma carinatum* (Cope, 1870)	River redhorse
Lotidae	*Lota lota* (Linnaeus, 1758)	Burbot
Fundulidae	*Fundulus catenatus* (Storer, 1846)	Northern studfish
	Fundulus sciadicus Cope, 1865	Plains topminnow
Percopsidae	*Percopsis omiscomaycus* (Walbaum, 1792)	Trout-perch *
Percidae	*Ammocrypta clara* Jordan & Meek, 1885	Western sand darter *
	Etheostoma exile (Girard, 1859)	Iowa darter *
	Etheostoma microperca Jordan and Gilbert, 1888	Least darter
Bivalvia		
Margaritiferidae	*Cumberlandia monodonta* (Say, 1829)	Spectaclecase
Unionidae	*Alasmidonta marginata* Say, 1818	Elktoe **
	Alasmidonta viridis (Rafinesque, 1820)	Slippershell
	Epioblasma triquetra (Rafinesque, 1820)	Snuffbox
	Lasmigona compressa (I. Lea, 1829)	Creek heelsplitter
	Ligumia recta (Lamarck, 1819)	Black sandshell **
	Obovaria olivaria (Rafinesque, 1820)	Hickorynut
	Quadrula fragosa (Conrad, 1835)	Winged mapleleaf
	Quadrula quadrula form *nobilis* (Conrad, 1854)	Mapleleaf (variant)
Gastropoda		
Viviparidae	*Campeloma crassulum* Rafinesque, 1819	Ponderous campeloma
Hydrobiidae	*Amnicola limosus* (Say, 1817)	Mud amnicola

Table 2. Selected biological and chemical attributes of two candidate reference streams in Kansas. Thompson Creek is a small, sandy bottomed, spring-fed stream located in the Southwestern Tablelands. Cedar Creek is a somewhat larger, gravelly bottomed stream located in the Flint Hills. For comparative purposes, summary data are presented for all other streams routinely monitored in the state by KDHE. Fish tissue contaminant data (composited whole-fish samples) are available for Thompson Creek but not Cedar Creek. Fish contaminant concentrations in Thompson Creek are among the lowest yet documented in the central plains (B. Littell, U. S. Environmental Protection Agency, personal communication), whereas nitrate levels are suggestive of groundwater contamination from recent or historical agricultural sources (e.g., chemical fertilizer use, livestock waste). Collectively, the measured biological attributes of both streams place them among the highest quality surface waters in their respective level III ecoregions (cf., Chapman et al. 2001).

Metric or parameter		Thompson Creek	Cedar Creek	Statewide
Biological:		N = 11	N = 17	N = 1169
Macroinvertebrate biotic index	Range	3.91 - 4.35	3.90 - 4.34	3.37 - 9.96
	Median	4.20	4.13	4.47
Kansas biotic index	Range	2.29 - 2.62	2.35 - 2.67	2.09 - 4.63
	Median	2.44	2.53	2.69
EPT index	Range	10 - 13	12 - 20	0 - 25
	Median	12	16	12
EPT% abundance	Range	60 - 75	37 - 77	0 - 87
	Median	70	53	52
Mussel taxa decline (% historical spp.)	Range	(No native spp.)	6 (1 of 18 spp.)	0 - 100
	Median	–	–	20
Chemical:				
Total suspended solids (mg/L)	Range	3 - 26	3 - 620	<1 - 60,700
	Median	10	20	42
	N	7	22	32,001
Nitrate, as N (mg/L)	Range	1.98 - 2.56	0.03 - 0.92	<0.01 - 75
	Median	2.28	0.36	0.70
	N	7	19	34,905
Phosphorus, total (mg/L)	Range	0.025 - 0.058	<0.010 - 0.63	<0.010 - 60
	Median	0.027	0.052	0.188
	N	7	22	34,742
Fish contaminants: (mg/Kg)				
	N:		N:	
Total chlordane	7	<0.002 - 0.003	–	875 <0.011 - 1.34
p,p'- DDE	7	<0.002 - 0.02	–	500 <0.002 - 0.99
Total PCB	7	<0.03	–	575 <0.02 - 1.13
Mercury	7	0.02 - 0.16	–	460 0.01 - 0.52
Lead	7	<0.5 - 0.15	--	175 <0.05 - 2.35

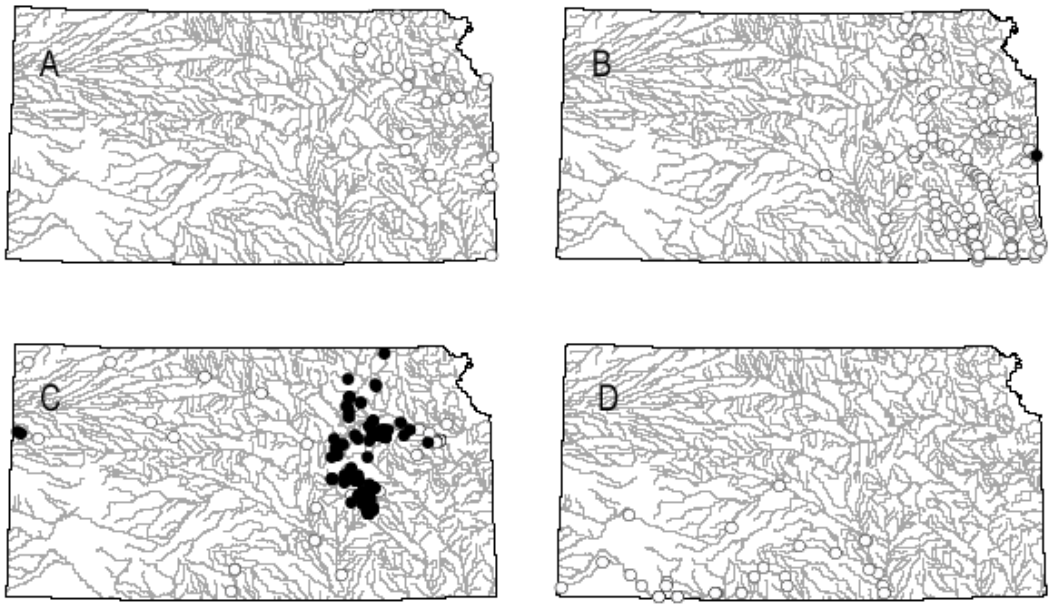

Fig. 1. Historical changes in the geographical distributions of four aquatic animals in Kansas. Solid circles represent recently verified extant populations. Open circles represent formerly documented populations, archeological shell collection sites (map B only), or stream sites now yielding only weathered shell material (maps A and B only). A - *Campeloma crassulum*, a viviparid snail (Call 1885b, 1885d, 1886, Hanna 1909, Franzen and Leonard 1943, Branson 1963, Angelo et al. 2002). B - *Ligumia recta*, a unionid mussel (Call 1885a, 1885c, 1885d, 1886, 1887, Popenoe 1885, Scammon 1906, Isely 1925, Murray and Leonard 1962, Branson 1966, Bradley 1973, Liechti and Huggins 1977, Cope 1979, Schuster and DuBois 1979, Hacker 1980, Metcalf 1980, DuBois 1981, Thies 1981, Witty 1983, Obermeyer et al. 1995, Angelo and Cringan 2003; T. Nickens, National Museum of Natural History, personal communication). C - *Notropis topeka*, a cyprinid fish of small streams with permanent pools (Minckley and Cross 1959, Cross 1967, Schrank et al. 2001, KDWP 2002). D - *Notropis girardi*, a cyprinid fish endemic to sandy rivers of the central Arkansas River Basin (Cross 1967, Cross et al. 1983, 1985).

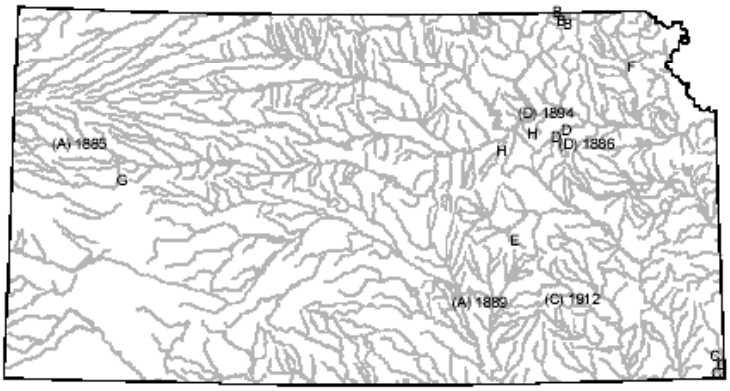

Fig. 2. Kansas fish and aquatic invertebrate populations indicative of earlier glacial climates and ancestral stream linkages (or prolonged genetic isolation and differentiation; see G-H, below). Letters without parentheses correspond to known extant populations. Letters within parentheses correspond to extirpated populations (dates refer to last known occurrences). A - *Notropis heterolepis*, a cyprinid fish of cool, clear, weedy pools with sandy bottoms (Cross 1967). B - *Rhinichthys atratulus*, a cyprinid fish of clear headwater brooks and streams with gravelly bottoms (Cross and Collins 1995). C - *Cyprinella spiloptera*, a cyprinid fish of clear, clean streams with sandy or gravelly bottoms (also occurs in lakes in other regions) (Cross 1967). D - *Percina maculata*, a percid fish of clear streams with shallow pools and gravelly bottoms (Cross 1967). E - *Probythinella emarginata*, a hydrobiid snail of spring-fed streams with deep, well-aerated, essentially silt-free pools (also occurs in lakes in other regions) (Angelo and Cringan 2002, Angelo et al. 2002). F - *Pomatiopsis lapidaria*, a semiaquatic pomatiopsid snail of marshlands and permanently vegetated stream banks (Liechti 1984, Angelo et al. 2002). G - *Optioservus phaeus*, an elmid beetle endemic to a single permanent spring and gravelly spring run in western Kansas (Layher 2002). H - *Leptophlebia konza*, a leptophlebiid mayfly endemic to a few clustered springs in the Flint Hills of eastern Kansas (Burian 2001). Many other isolated relict and endemic populations probably were extirpated early in the history of the state, precluding their later documentation (cf., Fausch and Bestgen 1997).

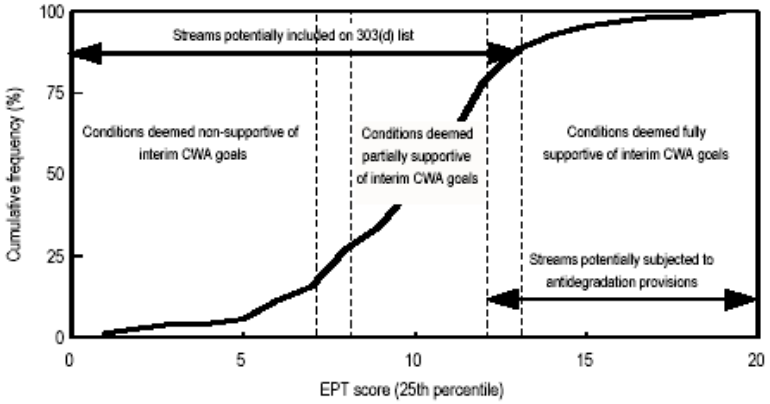

Fig. 3. Distribution of Kansas streams among biological condition categories based on lower quartile EPT scores. This graph summarizes data from 72 monitoring sites included in the KDHE stream biological monitoring network and surveyed for five or more years under a seasonally rotating schedule. Delineation thresholds (EPT scores of 7-8 and 12-13) correspond approximately to inflection points on the graph. Best professional judgement (e.g., trend evaluation) is employed in allocating sites to biological condition categories if lower quartile EPT scores (ties) fall between 7.0-8.0 or 12.0-13.0. Four additional biological metrics (percent mussel loss, EPT percent count, Kansas Biotic Index, and MBI) also are considered by KDHE before assigning individual sites to their respective (aggregate) biological condition categories (KDHE 2000).

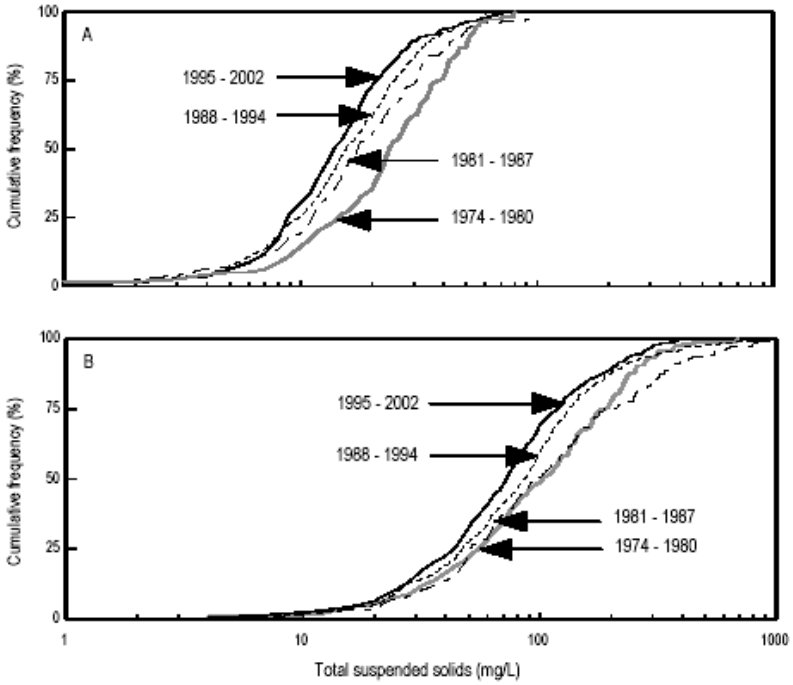

Fig. 4. Total suspended solids (TSS) levels in Kansas streams, 1974-2002. Graph A depicts a progressive statewide decline in lower quartile TSS concentrations over four discrete time intervals. Graph B depicts a similar decline in upper quartile TSS concentrations. These graphs may be interpreted as corresponding to periods of moderately low flow and moderately high flow, respectively. They represent 31,858 observations distributed among 390 KDHE stream chemistry monitoring sites. Measured differences between the 1974-1980 and 1995-2002 time intervals are significant in both the lower quartile and upper quartile comparisons ($p < 0.001$, Mann-Whitney test). Factors contributing to these declines are currently unknown but probably include soil conservation efforts implemented during the past three decades.

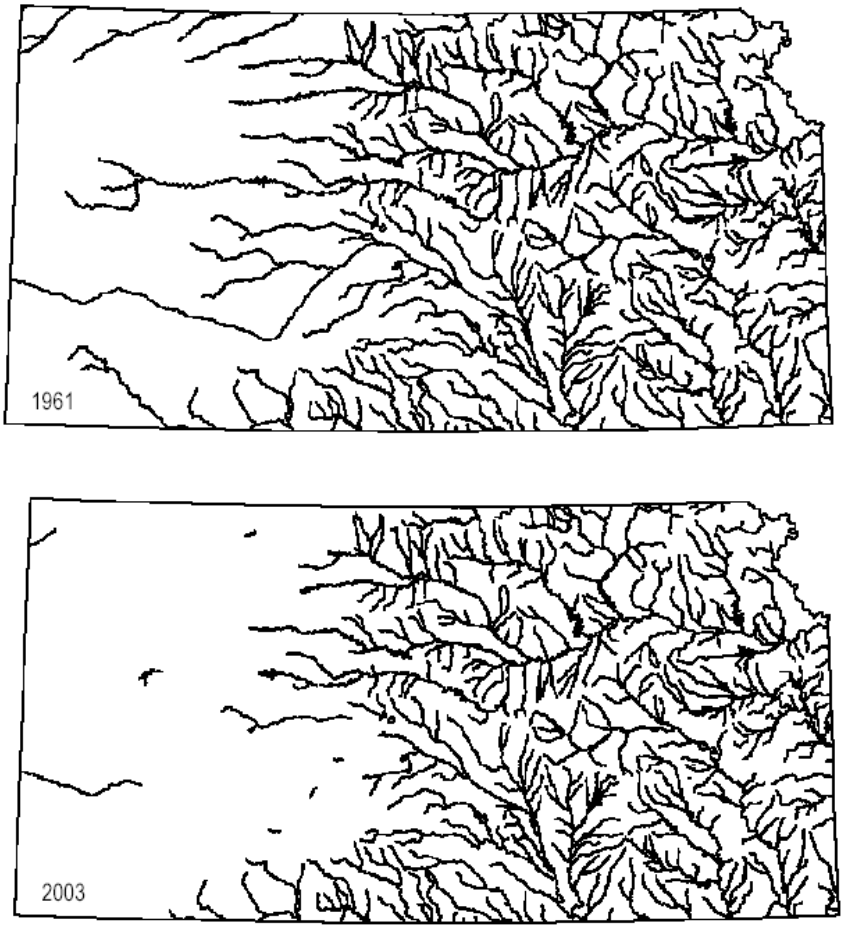

Fig. 5. Major perennial streams in Kansas, 1961 versus 2003. The upper illustration is adapted from a U. S. Geological Survey 1:500,000 scale base map compiled in 1961. The lower illustration reflects bimonthly flow observations made by KDHE stream chemistry monitoring personnel from January 1990 through February 2003. Streams observed to be dry (or pooled) more than 10% of the time, and in four or more years during this period, were regarded as non-perennial. Pooled conditions accounted for less than 25% of the zero-flow occurrences. The 2003 map represents an updated version of a similar illustration prepared nearly a decade ago, based on a considerably shorter (52-month) period-of-record (Angelo 1994). Crop irrigation has been implicated as a primary factor in the loss of stream base flows over much of western Kansas (e.g., Jordon 1982, Cross et al. 1985, Schloss et al. 2000).

DEVELOPING LINKAGES BETWEEN CLEAN SEDIMENT INDICES AND BIOLOGICAL IMPAIRMENT

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Abstract. Clean sediment has been identified as the largest named pollutant in the 303(d) listed sites in the United States. The methods used by states to list streams as impaired by sediment is variable. Standard scientifically-based assessment tools are needed to determine the likelihood streams are impaired by clean sediments. In this study, linkages were sought between sediment indices and biologic indices for streams with detailed records of flow discharge, suspended sediment transport, and biological data to use as analogues in the evaluation of sites lacking detailed data. Preliminary analyses show that as durations of suspended sediment concentration at or above 1000 mg/l increase, the total number of organisms and the number of taxa tend to decrease for benthic organisms. The data for this determination was from streams in the Mississippi Valley Loess Plains in the state of Mississippi.

Key Words: suspended sediment concentration, frequency, duration, biological impairment, clean sediment TMDLs.

Introduction

Excessive erosion, transport, and deposition of sediment in surface waters is a major problem in the United States. The 1996 National Water Quality Inventory (Section 305(b) Report to Congress) indicates sediments are ranked as a leading cause of water quality impairment of assessed rivers and lakes. The impact of sediment in many of these listed streams is from too much or too little clean sediment. Clean sediment is defined here as sediment uncontaminated by other substances. Methodologies are needed to evaluate the likelihood that a given stream is impaired by clean sediment and the sediment conditions for an unimpaired stream (reference conditions) are needed to serve as a target for restoration.

The determination of unimpaired reference conditions and the magnitude of change necessary for impairment due to clean sediment will likely vary in the different physiographic or eco-regions of the country. A related study (Simon et al., 2001) using data collected by the United States Geological Survey (USGS), has shown that sediment yields vary dramatically in different physiographic provinces of the United States. Reference conditions of undisturbed or stable streams and the ability of the biota to handle changes in the sediment regime will likely also vary with physiographic region.

The linkage between changes in the sediment regime of a stream and the impact on the biota is poorly known for streams in most areas of the country. Knowledge of the linkage between change in sediment loading, however, is essential for the assessment of streams suspected to be affected by clean sediment and for the development of

clean sediment TMDLs. The United States Environmental Protection Agency (USEPA) has defined a seven-step procedure for the development of clean sediment TMDLs in impacted waterbodies (USEPA, 1999a, Fig. 1-2). This study will present data towards defining the first two steps of the TMDL process for clean sediments: Problem Identification, and Development of Numeric Targets. Kuhnle and Simon (2000, 2001) have outlined a methodology to identify streams impacted by sediment and a methodology to develop numeric targets for impairment. In this study, linkages between suspended sediment variables (frequency and duration) and indices of biological assessment (USEPA, 1999b) were sought for streams in the Demonstration Erosion Control (DEC) watersheds of Mississippi. Detailed flow discharge and suspended sediment transport data have been collected at all of these sites and it is the long term goal of this research to develop methodologies to assess the likelihood that a site is impaired because of clean sediments over the United States without requiring each site have an extensive record of sediment transport and flow measurements.

Previous work

Notable work to relate the range of suspended sediment conditions that can be handled by aquatic organisms has been done by Newcombe and others (1991, 1996). They have related sediment intensity (the product of concentration and duration) to severity of impact on fish and aquatic invertebrates (Fig. 1). Newcombe and Jensen (1996) have also related sediment concentration, duration, to severity of impact on fish in 3-dimensional plots (Fig. 2). Their work has demonstrated that fish and invertebrates are definitely affected by the concentration and duration of suspended sediment in the water column. While the negative effects of sediment on fish, primarily salmonids, and invertebrates has been demonstrated, there is still much to be learned on the effects of sediment on other types of fish and aquatic organisms.

Study Sites

The study sites were located on DEC watersheds in the northern half of the state of Mississippi. The DEC project was organized as a cooperative project involving the United States Department of Agriculture (USDA) Natural Resources Conservation Service, USDA - Agricultural Research Service, National Sedimentation Laboratory (NSL), US Army Corps of Engineers, Vicksburg District, and the USGS Mississippi Office. The main goals of the DEC project were to develop and implement new erosion-control techniques for implementation and testing on the highly erodible Mississippi Valley Loess Plains in the state of Mississippi (Little and Murphey, 1981; Cooper and Knight, 1986). Detailed records of flow discharge and suspended sediment transport have been collected by the USGS for eleven of the sites and one was collected by the NSL (Table 1). The periods of record at the sites range from 6 to 16 years and the number of sediment samples from 91 to 3686. The location of the data collection sites and the level III eco-regions of Mississippi are shown in Figure 3. Most of the sites are in the Mississippi Valley Loess Plains eco-regions (Fig. 3).

Analysis of Data

The flow data from all sites except Goodwin Creek station 2, consist of stage values collected every 15 minutes at USGS gauging stations. The stage values were converted to discharge using a calibrated relation. The flow data from Goodwin Creek station 2 consists of break point stage data collected in a supercritical flow flume (Bowie and Sansom, 1986). When flows were changing rapidly, stage data was collected at one minute intervals. When flow rates were steady, stage was collected as infrequently as every 24 hours. Stage data was converted to discharge using a calibrated relation. Frequencies of the discharge data were calculated using 35 classes defined by base 10 logarithms (Searcy, 1959). The instantaneous annual peak discharge for each site was calculated from the annual series of peak flows (Haan, 1977).

The sediment transport data from the eleven U.S. Geological Survey sites consist of all sizes of the sediment suspended in the water. These samples were collected using standard USGS collection procedures (Edwards and Glysson, 1999). The sediment transport samples collected at Goodwin Creek station 2 were collected using a strut-mounted DH-48 sampler through the entire flow depth at the downstream end of a supercritical flow structure (Willis et al., 1986). Power functions were fit to the suspended sediment and flow discharge data pairs. Where necessary, multiple power functions were used to accurately represent the trend of the data.

Using the flow and sediment records (Table 1), frequency and duration for sediment concentrations of a given magnitude were calculated. This was accomplished by using the power function rating curves to calculate sediment concentration for a given flow. Durations of a given sediment concentration were calculated similarly by first calculating the duration of a given flow using the time records and then converting it to a sediment concentration using the relations between flow and concentration.

Sediment Frequency and Duration

Frequency and duration values of suspended sediment were calculated for each of the twelve sites (Table 2). Despite the similarities of these streams, appreciable differences in frequency and duration values were found. Suspended sediment concentrations of 1000 mg/l or greater were present for between 0.4 to 10 percent of the time annually, with the expected annual duration of 1000 mg/l ranging from 0 to 3044 minutes. The ranges in the suspended sediment concentration and duration at the one year flow had similar magnitudes. Another useful way to compare the durations of sediment concentrations at the study sites is shown in Figure 4. These curves show the expected annual durations (minutes) for a wide range of sediment concentrations. The location of curves in duration versus sediment concentration space (Fig. 4) may be important for determining whether a given stream is impaired by clean sediment.

Relating Sediment Variables to Biologic Indices

The relation, if any, between the sediment parameters contained in Table 2 and several indices describing the community of benthic invertebrates were explored using scatter plots. The eight sites from this study which have biologic

data available are indicated in Table 2. The list of the benthic indices is contained in USEPA (1999b). The only two benthic indices which yielded evidence for a relation were number of taxa and total number of organisms (Fig.5). The best sediment variable to relate to these indices was determined by inspection to be the duration above 1000 mg/l.

The relation of a bed disturbance index was also explored using the benthic data for the DEC streams. A bed disturbance index was defined as t_0/t_c, where t_0 is the flow strength (expressed as bed shear stress) at the one year flow, and t_c is the critical flow strength (bed shear stress) at which the median grain size of the bed material just begins to move. Large values of this index would indicate a high degree of bed disturbance and would be expected to be detrimental to the benthic invertebrates. There is some evidence for a relation between the bed disturbance index and the total number of benthic organisms and number of taxa (Fig. 6).

Discussion and Future Work

Figure 5 demonstrates evidence for a relation between number of taxa and the total number of organisms to the annual duration of suspended sediment above 1000 mg/l. This relation is remarkable in light of the complexity of the physical and biological systems in these streams. There is also some evidence for a relation between the bed disturbance index (t_0/t_c) and number of benthic organisms and taxa. In future studies, frequency and duration of bed material movement and indices for other organisms, such as fish, will be considered. It is possible that vertebrates that live in the water column will be more susceptible to changes in suspended sediment concentrations and benthic dwelling organisms will be more susceptible to instability of the bed material. Identification of the critical combinations of magnitude, frequency, and duration for sediment related indices that cause impairment in streams of the major eco-regions of the country is the long term goal of this study.

Literature cited

Bowie, A. J., Sansom, O. W. (1986). Innovative techniques for collecting hydrologic data. Fourth Federal Interagency Sedimentation Conference, Las Vegas, Nevada, p.1-59 - 1-69.

Cooper, C. M., Knight, S. S. (1986). Status Report: Ecological aspects - Demonstration Erosion Control Project in the Yazoo basin. Mississippi Chapter American Fisheries Society 11: 20-26.

Edwards, T. K., Glysson, G. D. (1999). "Field methods for measurement of fluvial sediment." U. S. Geological Survey Techniques of Water-Resources Investigations, Book 3, Chapter C2, 89 pp.

Haan, C. T. (1977). Statistical Methods in Hydrology. The Iowa State University Press, Ames, Iowa, 378 p.

Kuhnle, R. A., Simon, A. (2000). Evaluation of Sediment Transport Data for Clean Sediment TMDLs. USDA-ARS, National Sedimentation Laboratory Report 17.

Kuhnle, R. A., Simon, A. (2001). Evaluation of sediment transport data for clean sediment TMDLs. Seventh Federal Interagency Sedimentation Conference, Reno, Nevada.

Little, W. C., Murphey, J. B. (1981). Evaluation of streambank

Demonstration Erosion Control projects in the bluffline of northwestern Mississippi; Project Objective 2, Appendix A, Stream Channel Stability, Report to US Army Corps of Engineers, Vicksburg District, 342 pp.

Newcombe, C. P., Jensem, J. O. T. (1996). Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries and Management, 16(4):693-727.

Newcombe, C. P., MacDonald, D. D. (1991). Effects of suspended sediment on aquatic ecosystems. North American Journal of Fisheries and Management, 11:78-82.

Searcy, J. K. (1959). Flow-duration curves. Manual of Hydrology: Part 2. Low-flow Techniques. United States Geological Survey, Water-Supply Paper, 1542A, 33 p.

Simon, A., Hupp, C. R. (1986). Channel evolution in modified Tennessee streams. Fourth Federal Interagency Sedimentation Conference, March, 1986, Las Vegas, Nevada, v. 2, p. 71-82.

USEPA, (1999a). Protocol for developing sediment TMDLs. United States Environmental Protection Agency, Office of Water, Washington DC, EPA 841-B-99-004.

USEPA (1999b). Rapid bioassessment protocols for use in wadeable streams and rivers. United States Environmental Protection Agency, Office of Water, Washington DC, EPA 841-B-99-002.

Willis, J. C., Darden, R. W., and Bowie, A. J. (1986). Sediment transport in Goodwin Creek. Fourth Federal Interagency Sedimentation Conference, Las Vegas, Nevada, p.4-30 - 4-39.

Table 1. Study Sites and Years of Record.

Name	USGS station ID	years of record	no. of suspended sed. samples
Abiaca Creek at Cruger, MS	07287160	1992-1999	1128
Abiaca Creek near Seven Pines, MS	07287150	1992-1999	3686
Batupan Bogue at Grenada, MS	07285400	1986-1993	393
Fannegusha Creek near Howard, MS	07287355	1987-1999	933
Goodwin Creek Station 2 near Batesville, MS	na.	1981-1996	962
Harland Creek near Howard, MS	07287404	1987-1993	413
Hickahala Creek near Senatobia, MS	07277700	1986-1993	476
Hotophia Creek near Batesville, MS	07273100	1986-1991	207
Long/Peters Creek near Pope, MS	07275530	1987-1993	223
Otoucalofa Creek Canal near Water Valley, MS	07274252	1986-1993	91
Topashaw Creek near Calhoun City, MS	07282100	1988-1993	682
Yalobusha near Calhoun City, MS	07282000	1988-1993	504

Table 2. Frequency and duration values for the study sites.

Name	1	2	3	4	5
Abiaca Creek at Cruger, MS *	0.015	1407	2032	242	5
Abiaca Creek near Seven Pines, MS*	0.019	1441	2272	188	5
Batupan Bogue at Grenada, MS*	0.032	101	992	222	5
Fannegusha Creek near Howard, MS*	0.060	2075	3525	276	5
Goodwin Creek Station 2	0.012	668	3591	85	5
Harland Creek near Howard, MS*	0.100	3044	4043	57	5
Hickahala Creek near Senatobia, MS	0.030	1396	2070	58	5
Hotophia Creek near Batesville, MS*	0.008	237	1405	111	5
Long/Peters Creek near Pope, MS*	0.030	1123	1664	275	5
Otoucalofa Creek Canal near Water Valley, MS*	0.090	766	1548	465	5
Topashaw Creek near Calhoun City, MS	0.019	354	1071	342	5
Yalobusha near Calhoun City, MS	0.004	0	320	604	5

1 - Fraction of time suspended sediment equals or exceeds 1000 mg/l.
2 - Expected annual continuous duration of suspended sediment at 1000 mg/l or greater (min).
3 - Concentration of suspended sediment at 1.0 year flow (mg/l).
4 - Expected duration of 1 year flow (min).
5 - Stage of channel evolution (Simon and Hupp, 1989).

- - indices available on benthic community at these sites.

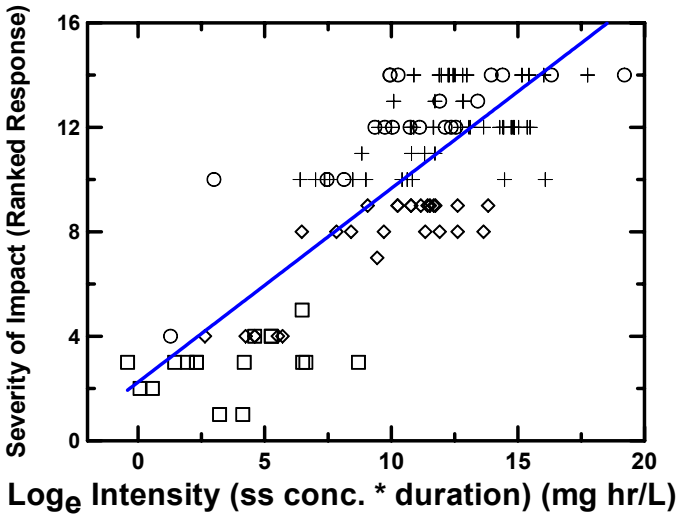

Figure 1. Effect of suspended sediment on salmon fish and aquatic invertebrates (Newcombe and MacDonald, 1991).

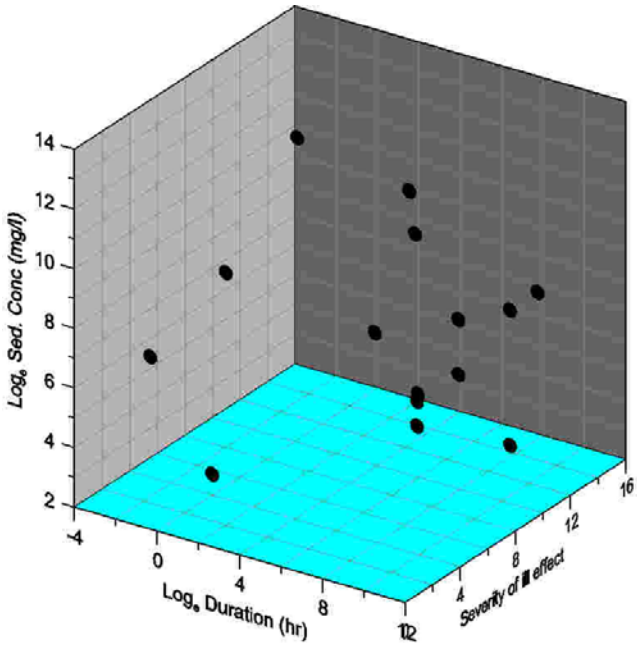

Figure 2. Effect of suspended sediment concentration and duration on severity of ill effect to adult freshwater nonsalmonid fish (Newcombe and Jensen, 1996).

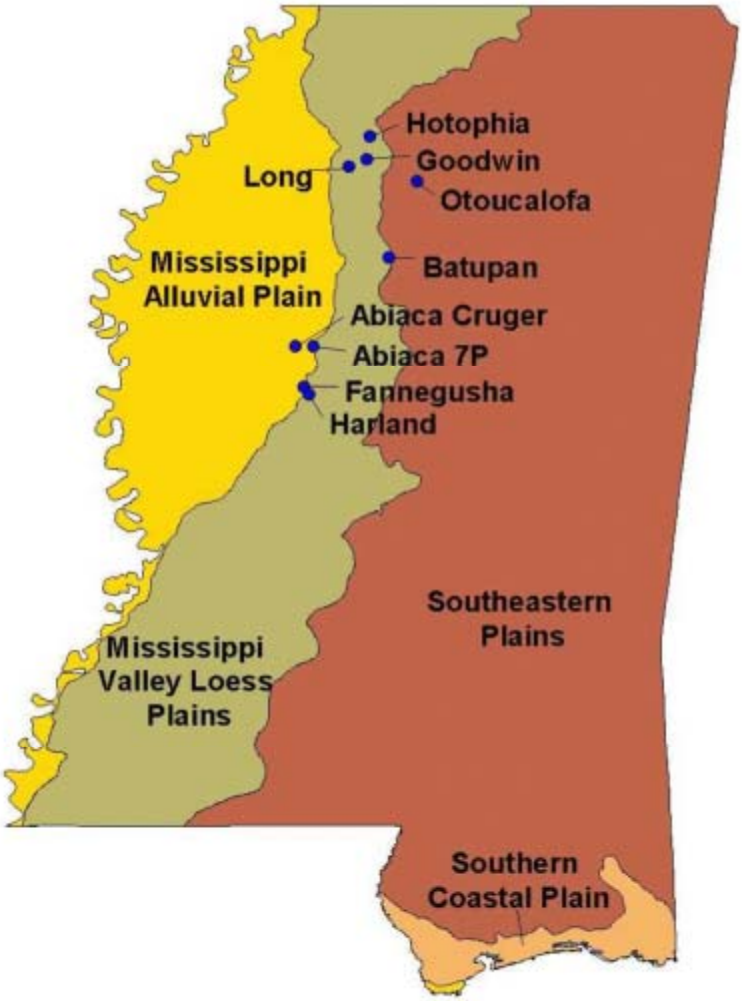

Figure 3. Locations of study sites in the level III ecoregions of the state of Mississippi.

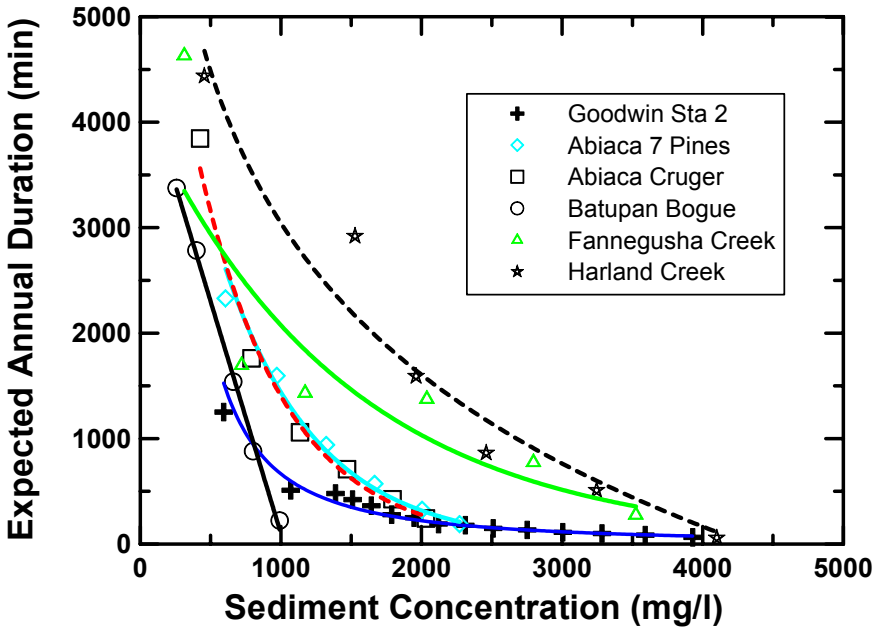

Figure 4. Curves of annual duration versus sediment concentration for several of the streams demonstrate the variability of this relation.

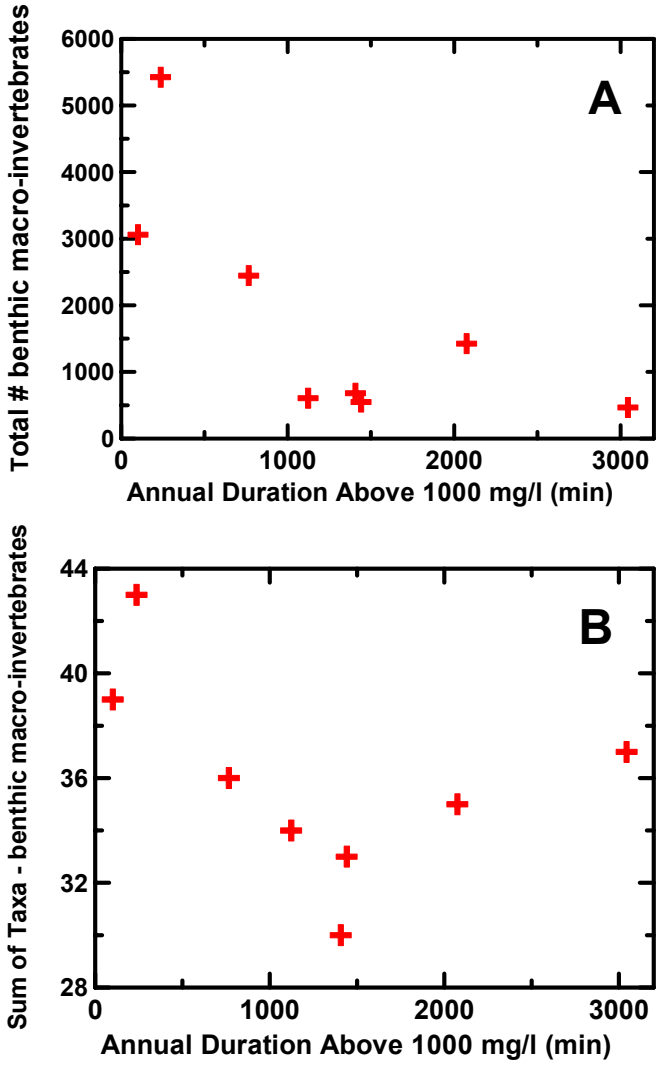

Figure 5. Total number of macro benthic organisms (A) and number of taxa (B) as related to annual duration above 1000 mg/l.

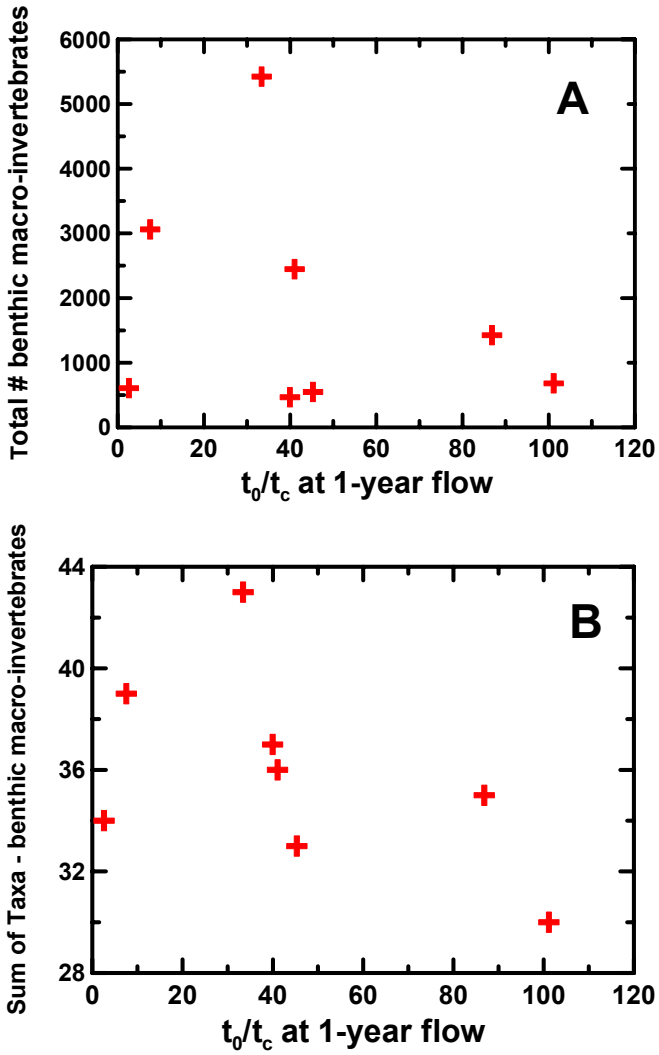

Figure 6. Total number of macro benthic organisms (A) and number of taxa (B) as related to excess bed shear stress at one-year flow.

BIOMONITORING INTERMITTENT STREAMS USING BENTHIC INVERTEBRATES

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Abstract. I investigated the feasibility of biomonitoring "very small" streams by studying benthic invertebrate communities at 15 sites of varying flow permanence within a single drainage for seven years. Streams were assigned to intermittent, intermediate and permanent categories based on relative discharge over time. Each category of stream had a distinctive invertebrate community structure, due more to differences in relative abundances than to presence or absence. The intermittent stream community could be considered a subset of the permanent stream community. Indices of community structure were positively related to flow permanence, and variability of indices increased as flow decreased. However variance in reference metrics was reduced if season of sampling and habitat types sampled were taken into account. I conclude that biomonitoring "intermediate" streams is appropriate using current standards. Biomonitoring intermittent streams is possible if appropriate reference standards are developed, and if reduced metric sensitivity is acceptable.

Keywords: biomonitoring, streams, metrics, indices, benthic invertebrates.

BIG MUDDY BIOASSESSMENT: CAN BIOCRITERIA BE DEVELOPED FOR THE LOWER MISSOURI RIVER?

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Abstract. Aquatic macroinvertebrate biological assessments are widely used for evaluating the status and condition of flowing waters. The methods and approaches already in place for wadeable streams, need to be modified and fine-tuned for application towards great river systems such as the channelized lower Missouri. Over the last 6 years of research on this system, we have generated extensive species lists for multiple habitats, descriptive comparisons of the efficiency of sampling methods, and a preliminary evaluation of longitudinal site differences. We are in the process of validating large river metrics for the invertebrate communities in the lower Missouri River by sampling a larger number of sites in autumn 2002 and including the simultaneous collection of water and sediment quality data. To aid in future analysis and interpretation of the data, we will present some preliminary findings and the important considerations that need to be addressed in order to develop large river biocriteria for the lower Missouri. Definition of reference condition and metric expectations, establishment of impairment categories, and outlining approaches used for quantifying longitudinal site differences, will be discussed. An important research need for the Lower Missouri system is the determination of overall water resource status and biological condition. However, other large river system goals can be addressed with bioassessment endpoints, including evaluation of the success of habitat rehabilitation projects and the identification of broad-scale indicators for monitoring ecosystem function.

Keywords: bioassessment, biocriteria, macroinvertebrates, Missouri River, biological condition.

Introduction

The Missouri River is classified as one of the "great" rivers of North America (Simon and Lyons 1995), and contains an aquatic fauna that includes unique species only found in the largest systems (Pflieger 1971, Poulton et. al., in press). Referred to as "Big Muddy" because of its historically high turbidity, this system has suffered from habitat loss, altered organic matter dynamics, and drastic changes in hydrological characteristics. The lower 1211 km of river from above Sioux City, Iowa to the mouth at St. Louis, Missouri, has been channelized into a narrow, deep channel confined by an extensive system of levees, wing dikes, and rock revetments. Problems related to public health and cumulative water quality-related impacts also continue to be a major concern in the lower Missouri River. In response to the need for water resource status evaluations, bioassessments are being implemented nation-wide to address the "Ecological Integrity" goal (USEPA 1990) of the Clean Water Act (CWA). These assessments, which utilize community-level endpoints as measures of relative condition or impairment, are being successfully used to achieve this goal in small streams and wadeable rivers. However, biological evaluation of great rivers has only recently begun to receive attention. Further, most great rivers in the U.S. have not been included in the funding of national bioassessment programs due to disagreements concerning sampling methods and approaches, definable restoration goals, and the lack of agency coordination. In

many cases, larger rivers have been ignored due to lack of resources, perceptions that biological evaluations cannot be accomplished on these systems, and the presumption that assessment of interjurisdictional rivers should be a federal responsibility. There is also a lack of coordination in governance and oversight regarding policy-making and changes in the management and restoration of natural resources within the Missouri Basin.

Large River Biocriteria Issues

Biological criteria are narrative or numeric statements that represent aquatic community expectations within a particular watershed, geographic area or ecoregion. Biocriteria serve as benchmarks from which all sites are compared to, and are developed based on aquatic community endpoints resulting from biosurvey data. Bioassessments and biocriteria development are part of a concise framework for evaluating relative biological condition, while also identifying realistic targets for system recovery, and providing the basis for categorizing impairment levels. This process relies on the establishment of reference community characteristics based on stream sites that represent the most pristine or least disturbed condition. However, applying this process to great rivers poses many logistical problems as compared to that of wadeable streams, including: 1) a resident fauna that is more poorly known ecologically, 2) a severely altered habitat condition, 3) lack of available historical data for defining reference communities or expectations in response endpoints, 4) the inability to separate simultaneous or cumulative effects caused by the multiple sources of stressors that often exist, and 5) sampling designs and equipment choices that are more operationally-defined (upstream / downstream site selection in key habitats using substrate-specific equipment). Further, large river biocriteria development is hampered by the lack of consistent designations in aquatic life uses among states, which creates problems in establishing relationships between relative condition and the actual degree of impairment.

Status of Macroinvertebrate Studies on the Lower Missouri System

Within the last 6 years, a series of macroinvertebrate pilot studies were conducted in the channelized lower Missouri River system. These studies examined aquatic macroinvertebrate community composition within different habitats and substrate types, identified key habitats and reliable invertebrate sampling methods, and generated a comprehensive species list. Our currently funded study, which began in 2001, includes measurement of water and sediment quality parameters and macroinvertebrate community data from a population of sites or reaches (N = 18). We are attempting to validate the effects of cumulative anthropogenic perturbations on community-level macroinvertebrate endpoints (i.e. “metrics”), and identify a longitudinal continuum or response gradient for the lower Missouri River. These metrics include those commonly used in wadeable streams, and others that have been modified to help explain responses to these impacts in large rivers (Table 1). Sites have been selected based on longitudinal features such as major metropolitan areas, large tributaries, and other potential sources of cumulative stressors. The two key habitats that have been chosen for this study include coarse substrate with flow (rock revetments on outside bends), and depositional mud substrate in backwater areas behind wing deflectors (dike fields). Both of these macroinvertebrate habitats are repeatable throughout the lower Missouri system,

harbor the most diverse macroinvertebrate community (Poulton et al, in press), and can be sampled with standard methods and gear (artificial substrates and a petite Ponar). As a result of this process, it is hoped that our evaluation will identify the best large river macroinvertebrate community endpoints that can be eventually combined into a multi-metric index for evaluating relative biotic condition in the lower Missouri system. This study also has the potential to outline impact or impairment categories, and characterize the community composition that might be expected under prevailing water and habitat quality scenarios in the lower Missouri system.

The ongoing macroinvertebrate research on the lower Missouri system relies on several assumptions that are based on preliminary findings from our pilot studies conducted in the 1990's. It is assumed that wadeable stream bioassessment approaches and endpoint metrics will be appropriate and useful for great rivers, with some modifications. It is also assumed that large river macroinvertebrate communities are integrators of cumulative anthropogenic effects, and that these effects can be separated from other, more natural influences such as biogeography and latitude. We have also assumed that community composition at a sampling site represents an integration of the cumulative exposure of conditions within that site or reach. Based on the results of our pilot studies, we also postulate that most macroinvertebrate species in the lower Missouri system are substrate-oriented, and that a habitat-stratified sampling design with upstream-downstream site selection are viable options for evaluating relative biological condition in large rivers.

Discussion and Conclusions

Among the macroinvertebrate approaches recently proposed to evaluate the condition and status of great rivers, some represent modifications of those used for smaller systems, and/or are parallel to applications with fish communities. The pilot studies conducted in the 1990's identified significant differences in individual macroinvertebrate attributes and overall metric scores when sites above Kansas City, MO were compared to those directly downstream (Poulton et al., in press). However, this longitudinal study could not provide definitive conclusions or validate individual metrics because of the low number of sites (Figure 1). Ongoing macroinvertebrate research on the lower Missouri River is attempting to identify a biological response gradient among a larger number of sites, and validate endpoint metrics based on water and sediment quality variables using a habitat-stratified sampling design.

Multi-metric macroinvertebrate evaluations such as the ICI (Invertebrate Community Index, Karr and Kerans 1991) may include a more unique list of attributes when applied to each of our great rivers. This approach is parallel to the modifications that have been proposed for the large river fish community IBI (Index of Biotic Integrity) recently applied to the Ohio River system (Simon and Emery 1995). Because true reference conditions cannot be defined for great rivers, benchmarks or criteria could be determined based on the best available or best attainable condition, rather than relying on data from smaller stream systems or an EDU (ecological drainage unit). In other words, community expectations can be based on the highest metric value, or reach with the highest overall score (Figure 2). Determination of impact or impairment categories can also be accomplished by using percentiles as cutoff ranges, as long as sufficient data is available from many sites. There are problems associated with these approaches. Although

large data sets do not yet exist for macroinvertebrate communities in these systems, these approaches currently have the best potential for defining community-level criteria, restoration targets, or recovery benchmarks in great rivers. Due to differences in management, regulation, and degree of alterations between the largest rivers in the U.S. (Missouri, Ohio, Mississippi, Columbia, Rio Grande, Colorado, etc.), each of these systems may need to be evaluated individually, rather than using a common reference point for comparisons or evaluations of overall system health.

Literature Cited

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish (2nd edition). USEPA Report, EPA 841/B-99/002.

Boesch, D.F. 1977. Application of numerical classification in ecological investigations of water pollution. U.S. EPA, EPA-600/3-77-033, Environmental Research Laboratory, Corvallis, OR. 113 pp.

Camargo, J.A. 1992. Macroinvertebrate responses along the recovery gradient of a regulated river (Spain) receiving an industrial effluent. Arch. Environ. Contam. Toxicol. 23: 324-332.

Ferrington, L.C. 1987. Collection and identification of floating exuviae of Chironomidae for use in studies of surface water quality. SOP No. FW130A USEPA, Region VII, Kansas City, KS.

Hilsenhoff, W.L. 1982. Using a biotic index to evaluate water quality in streams. Tech. Bull. No. 132, Dept. of Natural Resources, Madison, WI. 48 pp.

Karr, J.R., and B.L. Kerans. 1991. Components of biological integrity: Their definition and use in development of an invertebrate IBI. USEPA Report 905-R-92-003 Environmental Sciences Div, Chicago, IL 16 p.

Klemm, D.J., P. A. Lewis, F. Fulk, and J. M. Lazorchak. 1990. Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters. Environmental Monitoring Series, USEPA-ORD, EPA/600/4-90/030.

Lenat, D.L. 1983. Chironomid taxa richness: Natural variation and use in pollution assessment. Freshwat. Invertebr. Biol. 2(4): 192-198.

Lenat, D.R. 1993. A biotic index for the Southeastern United States: derivation and list of tolerance values, with criteria for assigning water-quality ratings-1993. J.N. Am. Benthol. Soc. 12:279-290.

Pflieger, W.L. 1971. A distributional study of Missouri Fishes. Occ. Pap. Mus. Nat. Hist. Univ. Kansas 20: 225-570.

Poulton, B.C., M.L. Wildhaber, C.S. Charbonneau, J.F. Fairchild, B.G. Mueller, and C.J. Schmitt. In press. A longitudinal assessment of the aquatic macroinvertebrate community in the channelized lower Missouri River. Environm. Monit. & Assess.

Schloesser, D.W., T.A. Edsall, B.A. Manny, & S.J. Nichols. 1991. Distribution of Hexagenia nymphs and visible oil in sediments of the upper Great Lakes connecting channels. Hydrobiologia 219: 345-352.

Shackleford, B. 1988. Rapid bioassessments of lotic macroinvertebrate communities: Biocriteria development. Arkansas Dept. of Pollut. Contr. & Ecology, Little Rock, AR., 44 pp.

Simon, T.P., and E.B. Emery. 1995. Modification and assessment of an index of biotic integrity to quantify water resource quality in great rivers. Regulated Rivers :Research and Management 11: 283-298.

Simon, T.P., & J. Lyons. 1995. Using fish community attributes for evaluating water resource integrity in freshwater ecosystems. In W.S. Davis and T.P. Simon (eds), Biological Assessment and Criteria: Tools for water resource planning and decision making. Lewis Pub., Ann Arbor pp 179-206.

USEPA. 1990. Biological Criteria: National program guidance for surface waters. USEPA Office of Water, EPA-440/5-90-004, Washington, DC. 57 p.

Washington, H.G. 1984. Diversity, biotic and similarity indices: a review with special reference to aquatic systems. Water Res. 18(6): 653-694.

Table 1. List of possible endpoint metrics to be evaluated during the 2001-2003 Lower Missouri River indicator validation study with aquatic macroinvertebrate communities.

Invertebrate Metric (Indicator or Endpoint)	Habitat Used In		Expected Response to Perturbation	References
	Coarse Substrate	Mud (sediment)		
Taxa Richness	X	X	decrease with corresponding increase in tolerant groups	Barbour et al. (1999)
EPT Taxa Richness	X		decrease; tolerant EPT species may increase in abundance	Klemm et al. (1990)
%EPT organisms	X		decrease, often with corresponding increase in tolerant groups	Barbour et al. (1999)
% Chironomidae	X	X	increase in flowing water habitats but may decrease in depositional habitats	Lenat (1983)
% Oligochaeta	X	X	increase in both depositional and flowing water habitats	Lenat (1993)
% Ephemeroptera		X	decrease, especially in depositional habitats	Schloesser et al. (1991)
% Filtering Trichoptera	X		decrease when bound toxicants prevail, with corresponding increase in tolerant groups	Camargo (1992)
Density (#/m^2)	X	X	decrease within intolerant groups, but tolerant groups may increase	Klemm et al. (1990)
Hilsenhoff Biotic Index	X		increase with higher organic enrichment	Hilsenhoff (1982)
% Large River Taxa	X	X	may decrease with increasing numbers of habitat and/or substrate generalists	Poulton et al. (in press)
Shannon-Wiener Index	X	X	decrease with corresponding reduction in species richness and evenness	Washington (1984)
Scraper/Filterer Ratio	X		increase if bound toxicants expected; decrease if organic enrichment prevails	Barbour et al. (1999)
EPT/Chironomid Ratio	X		decrease with increase in stress; balance indicator	Ferrington (1987)
Indicator Assemblage Index	X	X	range 0.0->1.0; decrease may indicate community imbalance	Shackleford (1988)
Jaccard Similarity Index	X	X	range 0.0-1.0; relatively low values may indicate stress	Boesch (1977)
% Dominant Taxon	X	X	increases; indicator of evenness and dominance	Shackleford (1988)
Large River Invertebrate Community Index	X	X	Multimetric Index with unit-less score; declines as biotic condition decreases	To Be Developed

Figure 1. Relative biological condition scores for 6 lower Missouri River sites based on rock basket artificial substrate data and 10 aquatic macroinvertebrate metrics. Each point represents a replicate sample (n = 5 at each site). RM = River Mile.

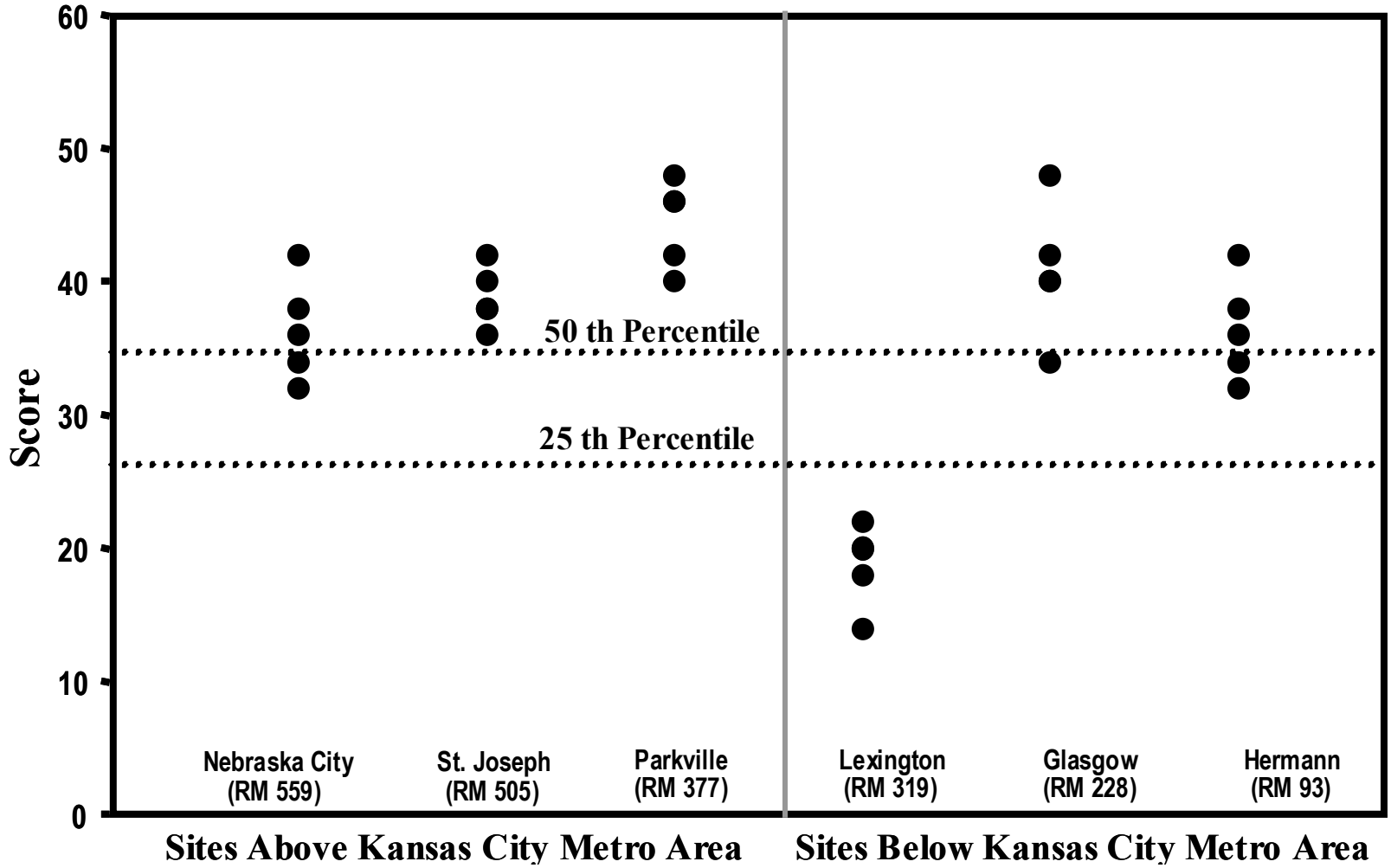
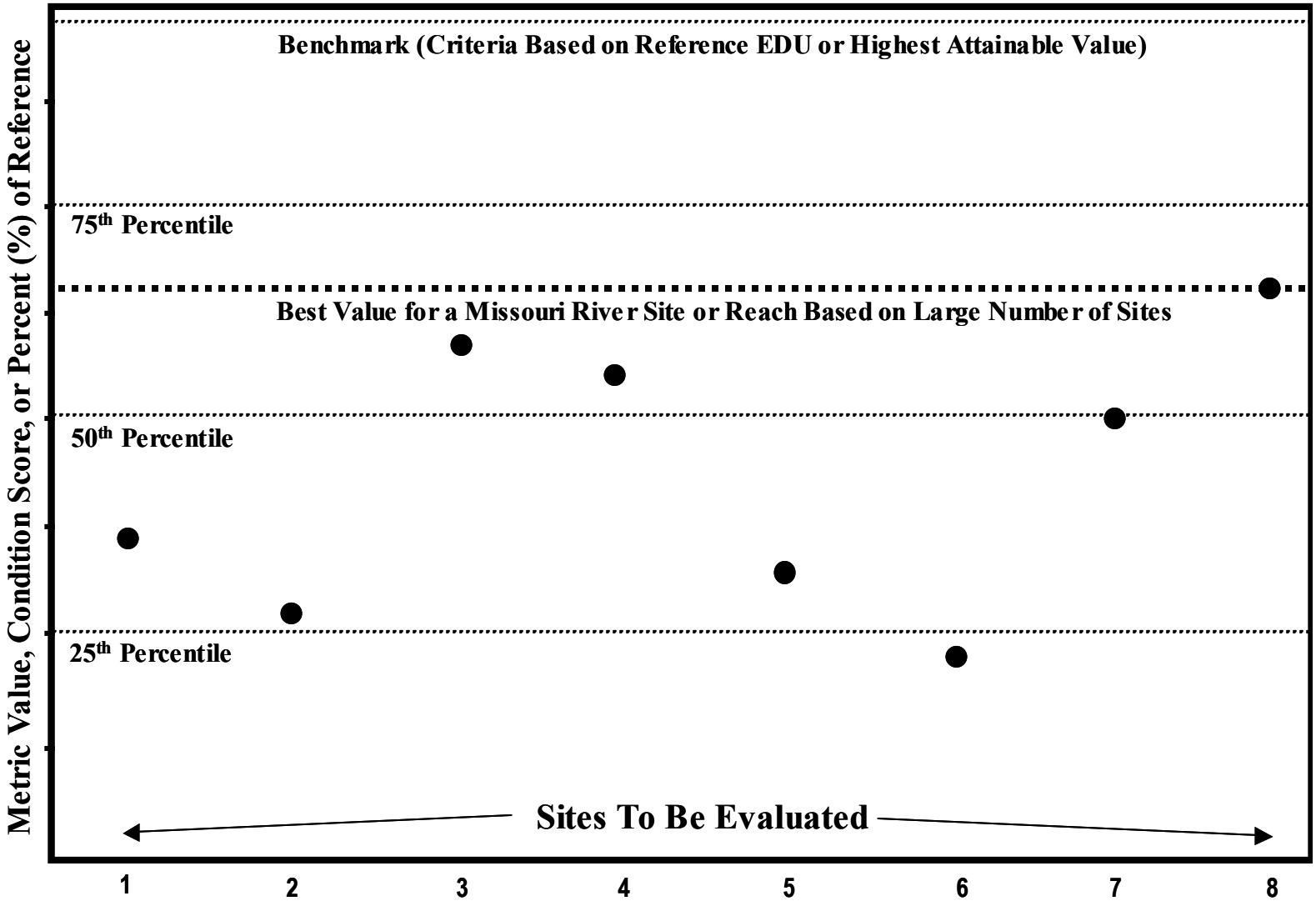


Figure 2. Representation of biological assessment data for evaluating relative biotic condition at numerous sites in a large river such as the Lower Missouri. In this example, cut-off ranges for impairment categories are defined using percentiles based on reference conditions for an Ecological Drainage Unit (EDU).



DISSOLVED OXYGEN FLUX, NUTRIENTS AND COMMUNITY PRODUCTIVITY IN SOME CENTRAL PLAINS STREAMS: WHAT WE KNOW AND WHAT WE THINK WE KNOW!

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Abstract. The Central Plains Center for BioAssessment is currently studying the relationships between nutrients, primary production and diel dissolved oxygen flux in wadeable streams of the Western Corn Belt Plains ecoregion. Over the last three years, we have deployed Aqua 2002 ® dissolved oxygen and temperature data loggers in a total of 36 study streams for time periods of 10 or more days during low flow conditions.

In situ measures of pH, conductivity, turbidity as well as dissolved oxygen and temperature which were used as QA checks against concurrent logger values were obtained at the start and finish of each deployment period. In addition, a series of physical stream measures (e.g. velocity, depth, width, discharge) and grab samples of stream water were collected at these same times for laboratory analysis of nutrients and water column chlorophyll and phaeophytin *a* concentrations. Lastly, three to five replicate periphyton samples were collected both at the beginning and end of each deployment period to determine benthic chlorophyll and phaeophytin *a* levels. Plots of dissolved oxygen (DO) concentrations verses time clearly show a high degree of daily and weekly fluctuation in the DO curves. Statistically significant correlations were found that suggested strong relationships between various DO variables and in-stream nutrient concentrations and other stream variables (e.g. periphyton & chlorophyll *a*) existed in these stream systems during low flows.

Initial findings indicate that relationships between in-stream nutrient concentrations, algal biomass, and DO regimes can be quantified during low flow periods but a number of yet to be quantified stream factors control these relationships. Daily and weekly estimates of gross primary production and community respiration calculated from the DO and temperature data are being examined as possible indicators of stream function as it relates to nutrient enrichment and other stream factors.

Key words: dissolved oxygen, nutrients, primary productivity, streams, Central Plains, Western Corn Belt Plains.

DEVELOPMENT OF A MULTIMETRIC AQUATIC STRESSORS INDEX FOR THE CENTRAL MIXED GRASS PRAIRIE ECOREGION

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Abstract. Information on the types and distributions of anthropogenic stresses to aquatic biodiversity (e.g., dams, point sources and incompatible land management) may enhance efforts to identify aquatic conservation sites with high ecological integrity, low conservation cost, and a high likelihood of conservation success. Additionally, these data may be used to identify or validate potential reference locations for biocriteria development. We designed a multimetric index that quantified and ranked the intensity (in terms of frequency, density or distribution) of multiple types of threats across aquatic ecosystems in the Central Mixed Grass Prairie (CMGP) ecoregion. Component metrics in the index (e.g., catchment dam density, % riparian agriculture landuse, catchment point source density) included threats identified by local experts as the predominant activities impairing aquatic ecosystem integrity. To validate the index and component metrics, metric scores were compared between expert-nominated conservation areas, and those systems not nominated as conservation areas. Metric values between nominated and non-nominated system types differed markedly, demonstrating potential thresholds above which conservation areas may not be suitable. The effectiveness of most metrics was dependent on the size of the systems: distinct thresholds were observed for small systems and were not observed for large systems. The index demonstrated that nominated conservation areas may be useful reference sites for biocriteria development, based on the relatively low degree of stressors observed in these freshwater systems.

Key words: biodiversity, biocriteria, conservation, metrics, indices, Central Mixed Grass Prairie.

ENVIRONMENTAL IMPACT ASSESSMENT OF THREE ANTI-MICROBIAL CHEMICALS

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Abstract. All finished water produced by wastewater treatment plants in the United States is released into stream ecosystems. Although these treated wastewaters meet U.S. effluent standards, they nonetheless contain detectable quantities of surfactants, antibiotics, and other types of anti-microbial chemicals contained in pharmaceutical and personal-care-products (PPCPs). These chemicals minimally biodegrade, but they are currently not mandated for removal under the U.S. Clean Water Act. The ecological effects of three representative PPCPs on natural algal communities were individually tested in this study using a series of laboratory dilution bioassays: Ciprofloxin (an antibiotic), Triclosan (an anti-microbial agent), and Tergitol (a surfactant). Each of these compounds caused strong shifts in community structure, and genus diversity was reduced for attached and suspended algae from sites located just upstream and just downstream of the Olathe, Kansas, WWTP. Although these compounds did not significantly alter rates of exponential growth in vitro, treatment effects on the lag phase appeared to have resulted in significant differences in final biomass yields in the bioassays.

Key words: wastewater treatment plants, surfactants, antibiotics, personal-care-products, Ciprofloxin, Triclosan, Tergitol, algae.

DEVELOPMENT OF A COMPREHENSIVE LAKE AND RESERVOIR CLASSIFICATION STRATEGY FOR NEBRASKA AS A MODEL FOR AGRICULTURAL DOMINATED ECOSYSTEMS

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Abstract. In agriculturally dominated regions, land use practices have an unusually large impact on water bodies and, therefore, may reduce the utility of current ecoregion-based approaches to lake classification by dampening the signals which underlie the ecoregion framework. This research focuses on developing a comprehensive classification scheme for agriculturally dominated ecosystems by (1) establishing a protocol for aggregating water bodies in agricultural ecosystems into classification strata and identifying reference conditions for these classes and (2) establishing the role of remote sensing and GIS in a classification strategy.

A water quality database for nearly 325 Nebraska lakes and reservoirs has been established with all water bodies sampled monthly from May through September for common limnological parameters (e.g., nutrients, clarity, chlorophyll). From this database, lakes and reservoirs are being classified hierarchically using a combination of rule-based and data-based approaches. Thus far, lake and reservoir classes have been defined by performing a factor analysis on the limnological data and plotting the significant factors to identify groups with similar water quality characteristics. Interesting, the Level IV Ecoregions based on soil type, native vegetation cover, topography, and geology do not accurately represent water quality of Nebraska's reservoirs or natural Sand Hills lakes. Limited surface water inputs, combined with local hydrology, reduce the utility of landscape classification approaches in the Sand Hills region and the dominant impact of land use minimizes the effectiveness of ecoregional classification for the reservoirs.

Improved methods are also being developed for integrating field data, data collected via airborne and close range remote sensing, data collected via operational and near future satellite remote sensing systems, and ancillary geospatial data in a multistage approach to lake classification. Biological indicators that integrate the lake conditions of each stratum will be developed from summer phytoplankton and zooplankton collections and special emphasis has been placed on developing methods to remotely sense biological indicators of water quality based on a the optical phytoplankton pigment structures of lakes. Close range remote sensing techniques were very sensitive to even slight variation in chlorophyll concentration (as well as in turbidity and Secchi depth), with standard errors of <5 mg/m3 over a range of chlorophyll concentrations from 10 to 194 mg/m3.

Key words: lake classification, nutrients, chlorophyll, ecoregions, phytoplankton, zooplankton, Nebraska, Sand Hills.

AN OVERVIEW OF HARMFUL ALGAL BLOOMS (HABS): DIVERSITY AND DYNAMICS OF TOXIC ALGAE IN FRESHWATER LAKES AND MARINE HABITATS

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Abstract. Reports of the occurrence of toxic and harmful freshwater and marine algae are becoming more frequent. Not only have they been reported in the scientific community, but also in the media, and numerous websites have been established that include regular updates on population studies on the abundance of these forms of algae. This reporting is due, in part, to an increased awareness of the impact of algae on biota in the aquatic communities.

With the concerns of federal, state, and local agencies and interested persons sponsoring activities such as Stream Teams, mariculture managers, and knowledgeable laypersons, there is a greater assessment of the environment. Knowledge of the diversity of the kinds of algae that can have a harmful impact on the biota and on humans, while requiring some microscopic abilities and equipment, will be presented. The dynamics of the ways that such algae have an impact on the biota that are the causative agents, and potential preventative measures will be also be presented. Thus an overview of the problems resulting from blooms of toxic and harmful and toxic algae lends to the continuing education of those who have a need to be aware of the potential for algal impacts serve as stewards of our aquatic communities.

Key words: algae, blooms, toxic, aquatic communities.

APPLICATION OF WETLAND BIOASSESSMENT PROTOCOLS FOR MAKING AQUATIC LIFE BENEFICIAL USE SUPPORT DETERMINATIONS IN MONTANA

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Abstract. All Montana (MT) state waters are classified in accordance to their designated beneficial uses. State waters determined to be impaired are placed on the 303(d) list of water quality limited segments and require Total Maximum Daily Load (TMDL) plans to control pollutants. The MT legislature has directed our agency to evaluate all available chemical, physical and biological water quality information when making beneficial use-support determinations (BUDs). The information that we receive is often difficult to assess, as it often comes in diverse forms and from many sources. For this reason, we developed guidelines that are used to screen the information content for sufficient credible data (SCD) for making BUDs. We have also developed decision tables that help reviewers to consistently interpret and apply numeric and narrative water quality standards when making BUDs. The guidelines and decision tables incorporate the use of bioassessment data for making aquatic life use-support (ALUS) determinations for all state waters, including wetlands. All of the data that we receive is categorized as chemical, physical or biological. The data within each category is individually assessed and scored. The cumulative scores of all three data categories are evaluated to determine if there is SCD for making ALUS determinations. Over 400 streams and lakes, and eight wetlands, were placed on MT's 2000 303(d) list using this decision process. Biological data and information were used to assess 94% of these waters.

Key words: wetland, bioassessment, aquatic life use, sufficient credible data, beneficial use support determinations, 303(d) list, TMDL.

Introduction

The fundamental goal of the federal clean water act is to "restore and maintain the chemical, physical and biological integrity of the Nation's waters." Therefore, EPA has mandated a strategy that States are required to follow for protecting water quality in lakes, rivers, streams, and wetlands. A vital step in the strategy involves monitoring state waters to identify those, which do not meet state water quality standards and, therefore, do not fully support their beneficial uses. As required by the federal Clean Water Act, these "impaired" waterbodies are placed on the "303(d) List." The assessment method Montana uses for placing waters on the 303(d) List establishes the degree to which individual state waters support all, some, or none of their beneficial uses.

The 1997 MT Legislature amended state water quality law to require that placement of waterbodies on the 303(d) List must be supported by "sufficient credible data" to ensure that such listings are justified. The sufficient credible data threshold applies both to the reassessment of waters previously on the list and to the consideration of any additional waters for listing.

In response to the legislative mandate, DEQ has developed a two-step process for assessment of impaired waters. First, DEQ searches out the available data for a waterbody and evaluates whether there is sufficient credible data to make a valid and reliable determination of beneficial use support. Then, if the data is adequate, DEQ compares it with the applicable water quality standards to make a beneficial use-support determination. The following paragraphs provide an overview of this process. Readers wanting a detailed explanation of the process, along with the tables and criteria used in making the SCD assessments and beneficial use determinations, will find these on our web site (Montana Department of Environmental Quality 2002).

Identification of Available Water Quality Data

In recent years DEQ's water quality monitoring data, along with information from other selected *s*ources, has been incorporated into computerized water quality databases. These records and databases provided a basic foundation to which materials from external sources were systematically added. DEQ sent out more than 2,700 letters requesting information from individuals, organizations and agencies, identified as possibly having water sampling data or other relevant information. Responses to this mailing provided much useful information, including references to additional materials available from other sources. Specific and general searches for these references and water quality information were conducted on the library catalogs of all the university system and resource agency libraries in the state.

Sufficient Credible Data (SCD) assessment.— MT law defines sufficient credible data (SCD) as "chemical, physical, or biological monitoring data, alone or in combination with narrative information, that supports a finding as to whether a water body is achieving compliance with applicable water quality standards" (75-5-103 MCA). This definition is consistent with a model developed by EPA for assessing the beneficial uses of streams on the basis of a combination of physical (habitat), biological, and chemical monitoring (U. S. EPA 1997). For example, EPA recommends that monitoring for ALUS include the collection of habitat and community level biological data and the measurement of chemical parameters in water and sediment.

MT DEQ drew on the EPA model to develop SCD criteria and decision tables to evaluate data adequacy for lakes and wetlands, as well as for streams. Methods and criteria are specified to evaluate SCD for the MT Water-Use Classification System beneficial uses. These uses are: 1) drinking, culinary use, and food processing; 2) aquatic life support for fishes, associated aquatic life, waterfowl, and furbearers; 3) bathing, swimming, and recreation; 4) agriculture supply; and, 5) industrial supply.

The SCD review focuses on four components that contribute to data validity and reliability for water quality assessment: 1) Technical soundness of methodology, 2) Spatial/temporal coverage, 3) Data quality, and 4) Data currency.

In most cases, a finding that there is SCD will result when several types of

data have been collected over a period of time using sound technical methods and there are no indications of recent changes to the water body that would invalidate previously-obtained results.

Aquatic life and fisheries support SCD.— The MT Water-Use Classification System requires that all waters support the "growth and propagation of fishes and associated aquatic life, waterfowl, and furbearers" (ARM 17.30.604-624). Based on this requirement, the “aquatic life” assessment considers fish, invertebrates, aquatic plants, and associated wildlife. Therefore, the aquatic life SCD assessment entails an evaluation and scoring of the following data categories:

Habitat/physical.— Includes qualitative and /or quantitative riparian and aquatic vegetation information, and hydrogeomorphic characteristics and functions.

Biology.— Includes chlorophyll *a* data; and aquatic biological community data such as fish, macroinvertebrates and algae; and wildlife community characteristics.

Chemistry/toxicity.— Includes bioassay, temperature and total suspended sediment data and chemistry data such as toxicants, nutrients, and dissolved oxygen.

Ideally, SCD for aquatic life would include data pertaining to all three categories; but very strong evidence relating to one or two data categories can constitute SCD for an aquatic life beneficial use support determination.

Beneficial use support determination (BUD).— Once it is ascertained that SCD is available for a waterbody, the assessment process moves to determine the level of beneficial use support. The degree of support for each beneficial use is rated using four categories: 1) Full support, 2) Partial support, 3) Non-support, and 4) Threatened.

A use is fully supported when all water quality standards applicable to that use are met. When one or more standards are not met due to human activities, the water body is either "not supporting" or "partially supporting" the beneficial use tied to that standard. A use that is currently fully supported, but for which observed trends or proposed new sources of pollution indicate a high probability of future impairment, may be rated as "threatened." Because the standards for determining use support are different for each use, the determinations for the various uses of a waterbody are often not the same. Only those beneficial uses that apply to the particular water-use classification of a waterbody are evaluated for that waterbody.

Beneficial use determination, aquatic life and fisheries.— Making aquatic life and fisheries use support determinations can be a complex process because of the amount and variety of information that may bear on the decision. In some cases the reviewer will evaluate, compare, and weigh many bits of physical, biological, chemical, and habitat in reaching the aquatic life and fisheries use support determinations for a waterbody. In other cases clear evidence of use impairment or support is provided from only one or two of the aquatic life data categories (habitat/physical, biology, and chemistry). Where there is a wide variety of data with no single element that by itself supports a conclusion, the evaluator follows a process employing criteria that lead to a determination based on the overall weight of evidence. A slightly different process is followed

when data is not available for all the categories, yet there is clear evidence to support a particular determination. Whatever the process used, data showing that a waterbody's aquatic life and fisheries uses are “moderately impaired” results in a “partially supporting” determination. Data indicating that aquatic life and fisheries uses are “severely impaired” results in the waterbody being listed as “not supporting” these uses.

Beneficial use determination, other uses.— Reaching beneficial use determinations for the drinking water, contact recreation, agriculture supply, and industrial supply uses is a relatively straightforward process. For each of these uses, criteria based on water quality standards are listed in a table, the available data for a waterbody is evaluated using the listed criteria, and an overall use support determination is made based on consideration of all the criteria for which data is available. In some situations the overall rating will result from clear evidence of support or impairment associated with one or two criteria; other determinations may be derived from indications of water quality derived from the entire set of criteria that apply to a particular use.

The 303(d) list of impaired waters.— The primary product of the SCD assessment and beneficial use support determination process is an updated list of state waters, which are impaired or threatened by human activity in their ability to fully support the beneficial uses specified by the MT Water-Use Classification System.

Literature Cited

75-5-103 to 75-5-703 Montana Codes Annotated (MCA). 1999 edition. The applicable Water Quality Statutes as codified in the Statutes of Montana.

Montana Department of Environmental Quality. 2002. Appendix A: Water Quality Assessment Process and Methods. Montana 303(d) List. http://nris.state.us/wis/environet/2002_303dhome.html

U.S. EPA. 1997. Guidelines for Preparation of the Comprehensive State Water Quality Assessments, (305(b) Reports), EPA-841-B-97-002A.

WETLANDS MONITORING: THE BIG PICTURE

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Abstract. Wetlands monitoring is a national priority of the U.S. Environmental Protection Agency. States and tribes are being encouraged to ensure that wetlands, like other waters of the United States, are part of their monitoring program. In the past, the physical, chemical, and biological monitoring of remaining wetlands in some areas within the plains region has been wanting, and region-wide there is a need to develop more robust monitoring efforts. Information from such monitoring can benefit many other programs, including water quality standards, TMDLs, and mitigation under Section 404 of the Clean Water Act and "Swampbuster" provisions of the farm bill.

Key words: wetlands, USEPA, monitoring, TMDL, Section 404, Clean Water Act.

THE KANSAS WETLAND SURVEY

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Abstract. As a follow up to earlier wetland inter-agency activities, the Kansas Department of Health and Environment undertook a survey of 32 public wetland areas in the state in the 1990s. This survey included collection of baseline water quality data as well as determination of wetland functional potential for several categories of wetland functions and uses. The results represent the first state wide wetland water quality survey and should be of value for several areas of water resources management.

Keywords: wetlands, water quality, wetland trophic state, wetland functions.

In the early 1990s, the Kansas Department of Health and Environment (KDHE) was involved in an interagency project called the Wetland and Riparian Areas Project (WRAP). Based on the publicly managed wetland areas identified during WRAP, the current project was undertaken via an Environmental Protection Agency (EPA) grant. The three-fold goal of this current project was to (1) examine wetland trophic state and reference trophic condition, (2) examine wetland water quality in comparison to state water quality criteria, and (3) to estimate the level of wetland function for a suite of ecological and functional (i.e., wetland processes that contribute to the environmental infrastructure) categories. Out of the 32 wetland areas examined, all were surveyed for their attainment of environmental functions but only 17 had sufficient water depth during the summer to allow consistent collection of water quality samples for analysis and comparison to criteria.

The majority (65%) of these wetlands were classed as hypereutrophic, in terms of planktonic algae. The wetlands with the worst (highest) trophic states and nutrient levels were also dominated by blue-green algae communities. As has been postulated in years past, and as a suite of metrics suggests, nitrogen is likely to be the primary limiting factor in Kansas wetlands, with phosphorus and hydrology being likely secondary factors. Contemporary reference conditions were estimated based on published EPA methodologies. Reference (least impacted) water quality for Kansas wetlands equates with moderately low nutrient levels (around 80 ppb total phosphorus and 865 ppb total nitrogen), algae levels at the low end of the eutrophic category (around 12 ppb chlorophyll-a), reasonably good water clarity, and general abundance of submersed and emersed macrophytes. Reference wetland quality would be supportive of virtually all beneficial uses recognized by Kansas law and water quality regulations (K.A.R. 28-16-28b et seq.).

The majority of water quality criteria/goal/guideline exceedences were for aquatic life uses (55%), with 55% of total exceedences due to nutrient levels, trophic status, and secondary effects of eutrophication such as low dissolved

oxygen, elevated pH, and dramatic diel changes in dissolved oxygen and water chemistry. The principal sources of wetland loss and impairment in Kansas are hydrologic loss due to both agriculture and urban activities, and sediment/nutrient pollution from these same two activities.

Many Kansas wetlands either perform basic ecological functions in an adequate manner, or have the potential to do so. Many others have a great deal of room for improvement. The most disturbing aspect of the estimates for wetland functions involved nutrient/sediment/toxicant retention. The majority of the wetlands exhibited high existing or potential capacity for materials retention. Combined with a generally high potential for downstream production transport, this has obvious implications for downstream water quality in streams and lakes. The suggestion is that many of these wetlands act both as "sinks" for these pollutants, which will lead to their impairment, but also likely act as "transformers" of pollutants, which make them a likely source of additional downstream impacts. Despite the popular view that wetlands are pollutant "traps," many of these anticipated downstream impacts would not be positive for either the environment or human activities that depend on water quality. Protection of these water resources becomes an important need for their future and our enjoyment of their benefits.

Literature Cited

Adamus, P.R., L.T. Stockwell, E.J. Clairain Jr., M.E. Morrow, L.P. Rozas, and R.D. Smith. Wetland Evaluation Technique Volume 1: Literature Review and Evaluation Rationale. United States Army Corps of Engineers, Waterways Experiment Station, 1991.

Environmental Protection Agency. Water Quality Standards for Wetlands National Guidance. United States Environmental protection Agency, Office of Water Regulation and Standards, Washington, D.C., EPA-440/S-90-011, 1990.

Kansas Department of Health and Environment. Surface Water Quality Standards. Kansas Administrative Regulations 28-16-28b through 28-16-28f, Kansas Register Vol. 13, No. 28, 1999 and 2000.

Kansas Department of Health and Environment. Kansas Wetland Survey: Water Quality and Functional Potential of Public Wetland Areas. 2002.

Kansas Department of Health and Environment. Kansas Water Quality Assessment (305(b) Report), 2000a.

Wetland and Riparian Areas Project. Wetland and Riparian Areas Program Directory Manual, 1992a.

Wetland and Riparian Areas Project. Wetland and Riparian Areas in Kansas: Resources in Need of Conservation, 1992b.

Wetland and Riparian Areas Project. Kansas Wetland and Riparian Resources: Conservation Goals and Strategies, 1992c.

USING AERIAL PHOTOGRAPHY AND SATELLITE IMAGERY TO MONITOR CHANGES IN THE VEGETATION OF CHEYENNE BOTTOMS WILDLIFE AREA

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Abstract. The Cheyenne Bottoms Wildlife Area (CHBWA) is a 19,857 acre wetland located in central Kansas, and is a critical stopover point in the central fly-way for approximately half of the migrating waterfowl and shorebirds of North America. In 1998, a five year project began that used Geographic Information Systems (GIS) technology to create a database of vegetation communities to support park personnel with management of the CHBWA. With the maintenance of vegetation communities and water levels being critical for preserving habitat function, special emphasis was placed on monitoring the extent and condition of cattail patches. This was important because cattails were encroaching on saturated soils and shallow water areas, thereby decreasing the area available to wading birds and waterfowl.

Each summer, near infra-red (NIR) photography was acquired over CHBWA and processed to produce a NIR digital orthorectified photograph with a one meter spatial resolution. Personnel collected ground truth data within the CHBWA each summer, then digitized vegetation communities from visual interpretation of the orthophotograph. Now, as the project draws to a close, the results of the project are examined, and new techniques are being explored that may assist in the continued monitoring of CHBWA.

Keywords: Cheyenne Bottoms, wetlands, cattail, land cover mapping.

The Cheyenne Bottoms Wildlife Area (CHBWA) covers 19,857 acres within a larger natural land sink that covers approximately 41,000 acres between the cities of Great Bend and Hoisington in Barton County, Kansas (Figure 1). In addition to the state managed CHBWA, The Nature Conservancy manages 7,269 acres of wetlands to the north and west CHBWA. Together, these managed areas serve a very important role for wildlife, and the area was dedicated a wetland of international importance by the Ramsar Convention on Wetlands in 1988. This wetland complex makes up the largest inland freshwater marsh network in the central United States, and it serves as a critical stopover point for many birds as they migrate along the central flyway between Canada and Mexico. It is estimated that 45% of the North American shore bird population stops at the Bottoms during the spring migration (Zimmerman 1990). Surveys have identified at least 320 species of birds including more than 25 species of ducks and geese, and several threatened and endangered species including the whooping crane, bald eagle, peregrine falcon, least tern, and piping plover (Hoffman 1983). The quantity of bird life that the Cheyenne Bottoms supports is astounding. The fall migration of ducks and geese have numbered in excess of 600,000 birds, and the spring migrations can bring thousands of sand hill cranes and tens of thousands of shorebirds.

Since its creation, the Cheyenne Bottoms basin has been slowly filling with sediment (marsh succession) at a historical rate of

approximately 12 inches every 800 years (Von Loh et al. 1998). It could be argued that this rate has increased recently (last 100 years) due to the abundant agricultural disturbances in the surrounding watershed. This increased sedimentation in filling the deeper pools and reducing the overall water holding capacity of the wetlands.

Additionally, cattail (narrow leaf (*Typha angustifolia*) and common cattail (*Typha latifolia*)) have recently established themselves in the area. Surveys from 1929 made no mention of cattails (Uhler 1929), and cattails were a minor occurrence in Sonnenbergs (1961) and Hastings (1970) reports. Cattail conditions changed quickly, and by 1987, cattail covered 15% of pool 3, 45% of pool 2, and 50% of pool 5 (Figure 2) (Brooks and Kuhn 1987). The cattail population thrived in the shallow waters and moist soils, and as a result, the area of cattail coverage has increased and the area available to wading and shore birds has decreased. To address this issue, the Kansas Department of Wildlife and Parks (KDWP) has been trying to control the cattail expansion and reduce its area of coverage. Park personnel have tried numerous method of cattail control including mowing, disking, burning, grazing, and herbicides.

Recognizing that some treatments (and combinations of treatments) worked better than others, KDWP and the Bureau of Reclamation (BOR) initiated a five year study to monitor the vegetation of CHBWA.

In 1998, the Remote Sensing and Geographic Information Group (RSGIG), a division of the BOR, began working with KDWP to develop a land cover database to assist park personnel with land management (cattail) decisions. The first three years of the project were conducted by the RSGIG, the fourth year the project was shared with the Kansas Applied Remote Sensing (KARS) program as project control was transferred to KDWP, and the fifth year of the project was performed by the KARS program. During each year of the project, the goals and methodology remained focused on accurately mapping the vegetation and creating a database to assist with management of the CHBWA. In the summer of each year, near infra-red (NIR) aerial photography over the study area was acquired and processed by Horizons Inc. to produce a digital ortho-photograph with a one meter spatial resolution. Field personnel conducted surveys each year to record the geographic position and vegetative conditions at approximately 100 sites to assist with interpreting the NIR photograph. Heads up digitizing was then performed to classify the area into one of 28 land cover/land use classes (Figure 3).

The classification of the vegetation of CHBWA was not always clear. Vegetation changed between and within years as a result of changing water availability and vegetation phenology. When two or more vegetation classes were mixed, certain classes were given increased importance. Cattails (Typha) were dominant over other classes because of their ability to expand and replace surrounding vegetation, while spikerush (Eleocharis) wetlands were considered dominant because they were an important food source for ducks. The mixing of classes was fairly common, especially in the intermittently submerged areas where both wetland and upland plants occurred. These seasonally wet areas were categorized as undifferentiated emergent wetland, and were a mix of species including bullrush (Scirpus), smartweed (Polygonum), saltmarsh aster (Symphyotrichum), dock (Rumex), and saltgrass (Distichlis). This constant flux in vegetation made mapping the CHBWA a challenge, and more research should be done comparing vegetation patterns between wet and dry years.

Results from the research indicate that the total area of cattail has decreased substantially from 6,841 acres in 1998, down to 3,865 acres in 2001 (Table 1). Most of this decrease is a result of mowing, discing, and burning, though some herbicides were also used. Park personnel have found that cattail were best controlled when submerged in greater than two feet of water for at least two consecutive years, however this was not usually possible with the limited water flowing into CHBWA. A combined treatment of burning and discing the cattail provided a temporary solution, but if the disced area became wet again before winter, the cattail would quickly regenerate from the root tubers. Herbicides worked well for areas of cattail that could not be accessed and/or dried for burning, and worked best when the treated cattail remained in the water for several years. According to park personnel, these cattail management techniques worked well for temporary removal of cattail, but the real problem is the natural process (and increased rate) of marsh succession which would be best countered by the extremely expensive and time consuming process of scraping out and removing the accumulation of sediment and cattail roots (Grover 2002).

With the dredging of Cheyenne Bottoms being an unlikely scenario, it seems more likely that KDWP will be continuing their control efforts against the cattail for years to come. To assist with future cattail management options (now that this five year project is over), alternative monitoring techniques were explored that could be used for future monitoring of CHBWA. Two basic imaging options were investigated, aerial photography, and satellite imagery. Near infra-red aerial photography was used successfully in this project, and is indeed an option for future monitoring efforts. Satellite imagery is another option that could be used to monitor the extent of cattail in Cheyenne Bottoms. While satellite imagery does not have the one-meter spatial resolution of the aerial photography, single date satellite data from the ASTER (15 meter resolution) and Landsat 7 (30 meter resolution) did provide sufficient information to map cattail patches but was less effective for mapping the highly heterogeneous transitional shorelines and uplands (Figure 4). Analysis of multitemporal Landsat Thematic Mapper 7 (30 meter resolution) satellite imagery was investigated to see if changes in vegetation phenology could help improve classification. The results showed that the multitemporal analysis did not work well because changes in water level between the image dates led to increased classification errors.

In summary, using GIS and aerial photography to map the vegetation of Cheyenne Bottoms provided KDWP personnel with a database of cattail area and provide insights into the effectiveness of different management practices. This new database can be used for years to come and additions can easily be made to track changes in the vegetation patterns and help with future assessments of the wetland. Additionally, using aerial photography or remote sensing can provide a lasting visual record of the conditions that can be reviewed at a later date giving this method an advantage over visual analysis and field notes alone.

Acknowledgments

Special recognition should be given to Jan Oliver at the GIS/RS group of the BOR, from whom I inherited this project. She provided me with a great deal of information that she had acquired during the first two years of the project. Special thanks are also deserved by the CHBWA park personnel (Karl Grover, Field Supervisor; Helen Hands, Wildlife Biologist; and Gene

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Literature Cited

Brooks, R.. E., C. Kuhn. 1987. The Vegetation of Cheyenne Bottoms, ch.12 in Cheyenne Bottoms, an Environmental Assessment. Performed by the Kansas Biological Survey and the Kansas Geological Survey, submitted to Kansas Fish and Game Commission, Jan 20, 1987.

Grover, Karl. 2002. Personal communications with Cheyenne Bottoms Field Supervisor.

Hastings, D.W. 1970. Distribution and Phenology of Vegetation in Pool 3, Cheyenne Bottoms Waterfowl Management Area: M.A. thesis. Fort Hays Kansas State College, Hays, 49pp.

Hoffman, Wayne. 1987. The Birds of Cheyenne Bottoms, ch.16 in Cheyenne Bottoms, an Environmental Assessment. Performed by the Kansas Biological Survey and the Kansas Geological Survey, submitted to Kansas Fish and Game Commission, Jan 20, 1987.

Oliver, K. Jan. and Michael E. Houts. 2001 Annual Report - Geographic Information System Database; Cheyenne Bottoms Wildlife Area, Kansas. Technical Memorandum No. 8260-02-04. U.S. Bureau of Reclamation. Technical Service Center. Denver, CO.

Sonnenberg, D. 1961. Some Ecological Studies of the Vegetation at the Cheyenne Bottoms Waterfowl Refuge: M.A. Thesis, Fort Hays Kansas State College, Hays, 54pp.

Uhler, F. M. 1929. Cheyenne Bottoms, Barton County, Kansas: Unpublished report to the Kansas Fish and Game Commission. (Original in possession of Mr. Marvin Schwilling, Kansas Fish and Game Commission, Emporia, Kansas.

Von Loh, Jim, K. Jan Oliver, and M. Pucherelli. 1998 Annual Report - Geographic Information System Database; Cheyenne Bottoms Wildlife Area, Kansas. Technical Memorandum No. 8260-99-04. U.S. Bureau of Reclamation. Technical Service Center. Denver, CO.

Zimmerman, John. 1990. Cheyenne Bottoms: Wetland in Jeopardy. University of Kansas Press, Lawrence, KS.

Table 1. A comparison of cattail area by pool through four decades of observations. Modified from the 2001 annual report.

Comparison of Cattail Area by Pool (1998-2001)						
	1960	**1986**	**1998**	**1999**	**2000**	**2001**
Pool 1	ND	40%	17%	11%	9%	8%
Pool 2	0%	43%	71%	71%	71%	70%
Pool 3	10%	15%	82%	41%	11%	6%
Pool 4	0%	5-10%	40%	43%	35%	31%
Pool 5	0%	50%	90%	86%	78%	41%

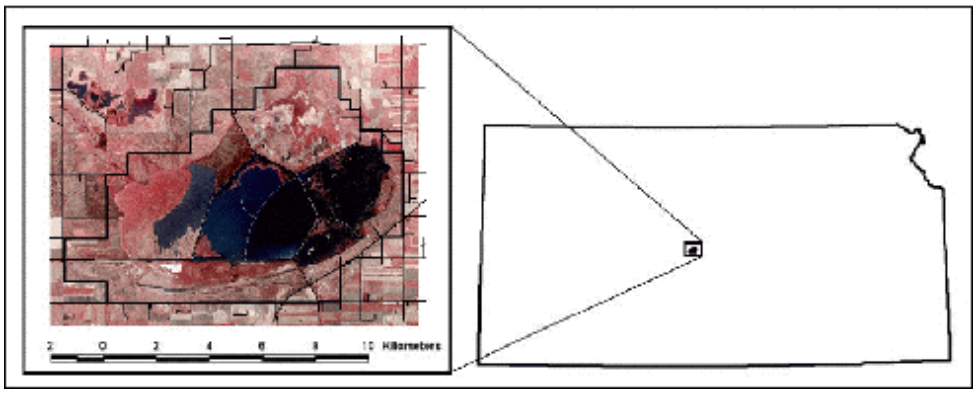

Figure 1. Location of Cheyenne Bottoms within the state of Kansas, and a near infra red image of the area showing the pools and levees within the state managed wildlife area.

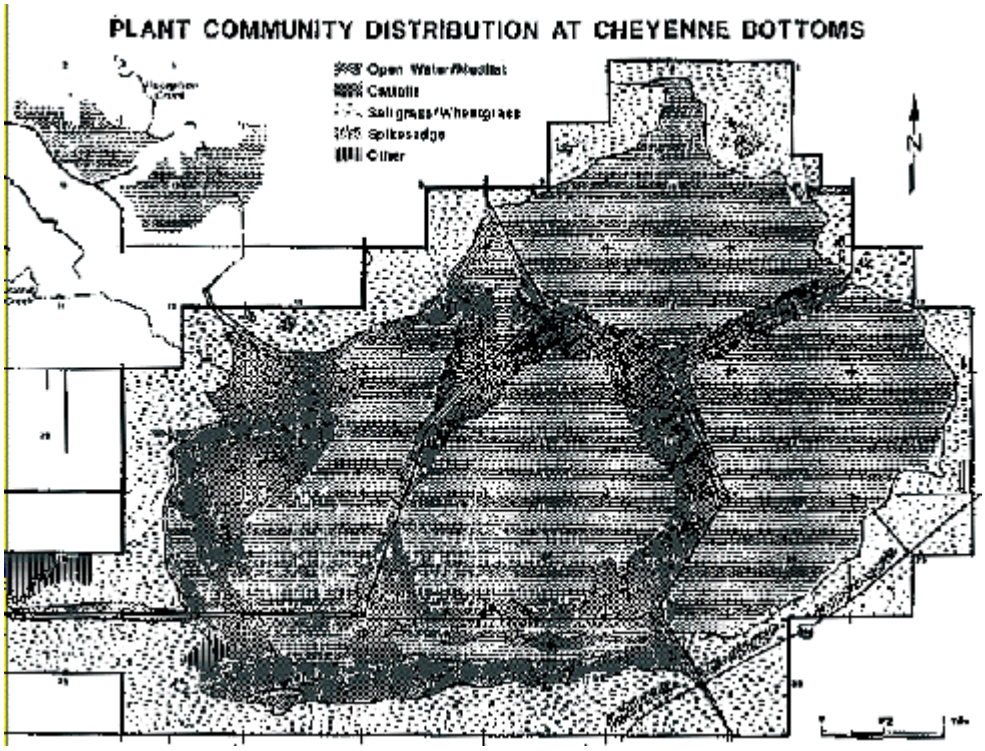

Figure 2. Vegetation survey of the Cheyenne Bottoms Wildlife Area conducted in 1986 provides a visual aid in assessing the historical distribution of cattail. (Reproduced from Brooks and Kuhn 1987).

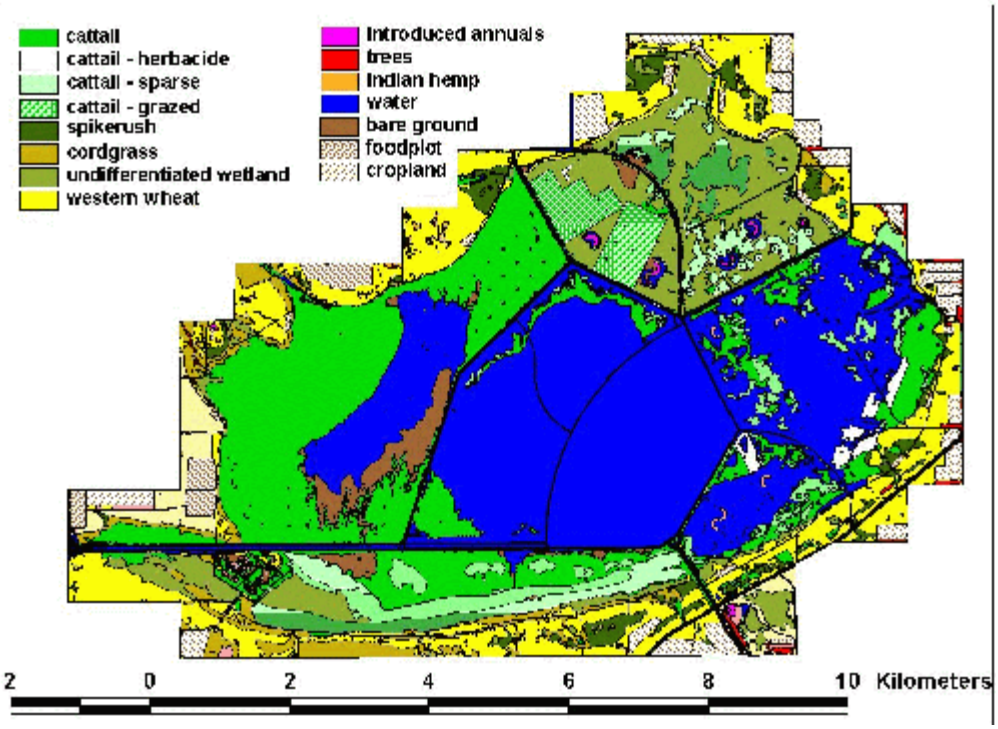

Figure 3. Land cover map derived from interpretation of 2001 aerial photography. The current monitoring project using GIS and aerial photography provided significantly more detail about vegetation patterns (especially upland classes) than the 1986 survey shown in Figure 2.

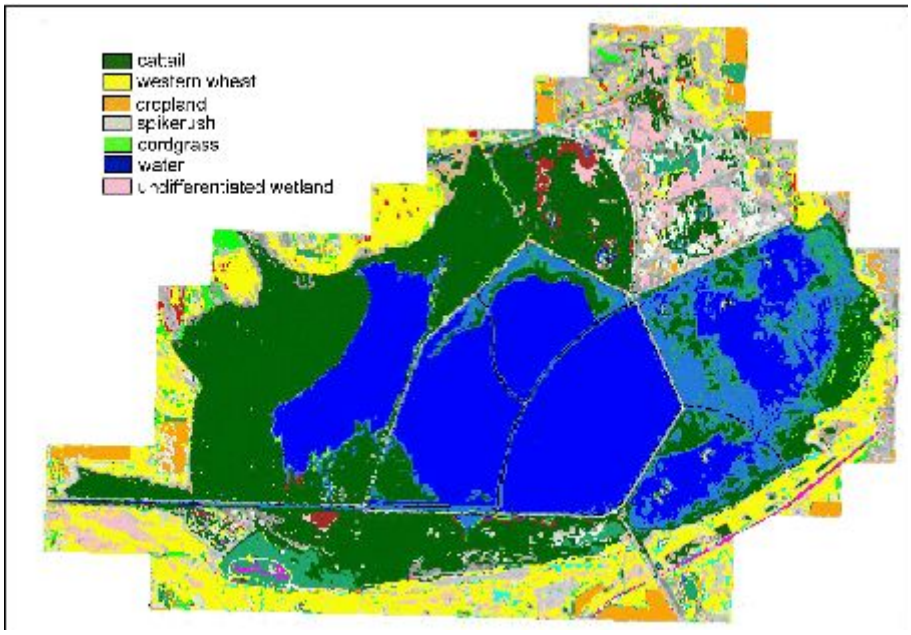

Figure 4. Results from unsupervised classification of Cheyenne Bottoms vegetation using single date Landsat Thematic Mapper satellite imagery. Areas of cattail are pretty clearly defined, while the heterogeneous uplands are a little less accurate.

RESERVOIR CLASSIFICATION IN AGRICULTURALLY DOMINATED ECO-SYSTEMS

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Abstract. Understanding regional patterns of lake conditions through lake classification is critical to enhancing our ability to manage the water quality of reservoirs and to set a framework for protection and restoration goals. Reliable indicators of aquatic ecosystems function have been developed in an agriculturally dominated area by examining the relationship between organisms and their associated abiotic environment. A protocol has been developed for grouping reservoirs into appropriate classification strata based on in-lake water quality parameters. Data-based statistical analyses of 100 reservoirs across Nebraska indicate at least five classes exist. Cluster analyses, coupled with factor analyses, indicate that total nitrogen, chlorophyll *a*, and turbidity are most important in defining these reservoir classes. Landscape ecosystem (eco-region) classification approaches use soil, native vegetation cover, geology, and topography data for grouping reservoirs. Comparing these classification methodologies, we find reservoirs belonging to the same class do not necessarily group together within an eco-region. Land use may contribute substantially to differences in classification strategies.

Key words: lake classification, nitrogen, chlorophyll, turbidity, ecoregion, Nebraska.

FOOD WEBS OF THE GREAT RIVERS OF THE CENTRAL BASIN: APPLICATION OF STABLE ISOTOPES IN BIOASSESSMENT

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Abstract. The long-term relationship between humans and large river ecosystems has lead to marked alteration of both river and river basin, including degradation of water quality. Interest in restoration and development of better management plans for large rivers has increased of late, but the problem remains as how to best assess the health of these large, spatially and temporally complex systems. Structural measures of density and abundance have been the most commonly used approach for developing ecological indicators, but these measures often contribute little information about how an ecosystem works. A more recent approach has been the use of indicators based on measures of ecosystem function, such as trophic dynamics, because functional processes are influenced by a number of biotic and abiotic factors, most of which are impacted by ecosystem disturbance. Indicators based on functional dynamics, therefore, would be integrative measures, summarizing the effects of disturbance for many ecosystem components into a single or minimal number of metrics. While a number of methods for developing aquatic food webs are available, the use of stable isotope ratios has the greatest potential in development of ecosystem indicators. Using isotopic ratios of potential food sources and consumers, it is possible to create linkages to construct a food web that allows for comparison of different systems as to the state of trophic dynamics. This paper describes how evidence was found during an unrelated study that indicates commonalities exist in the trophic functions of large rivers, but that deviations from these commonalities are evident in rivers that have been subjected to extreme alterations in hydrological and morphological attributes (e.g., Missouri River). In addition to descriptive models, techniques are available for generating quantitative measures using stable isotope ratios of nitrogen and carbon of consumers to generate scaleable measures for assessing ecosystem health.

Key words: food web, stable isotope, large river, Upper Mississippi River, Ohio River, Missouri River, trophic position, ecosystem function, trophic dynamic, ecosystem health.

Human civilization developed in proximity of large rivers, taking advantage of their productivity and their capacity to function as a transport for goods and refuse. Human activity along large rivers, however, came at a price. Point source pollution, while largely abated since the implementation of the Clean Water Act, has left a legacy of contamination in many rivers (Arnold et al. 1999, Lyons et al. 2001). Conversion of forests and prairies to agriculture and urban development has made nonpoint source pollution through increased nutrient and sediment loads a growing concern (Patrick 1998) not only for large rivers, but also for estuaries receiving riverine flow. The presence of the hypoxia zone in the Gulf of Mexico is testimony to the effects of increased nutrient transport from the Mississippi basin (Battaglin et al. 2001).

Concern for the health of large rivers in the U.S. has been demonstrated by implementation of monitoring programs such as the Long-Term Resource Monitoring Program (LTRMP) and the National Stream Quality Accounting Network (NASQAN) of the U.S. Geological Survey. There have been

some efforts to develop measures of ecosystem health for rivers, primarily through development of indices of biotic integrity. Still, the application of IBIs in large/great rivers has been limited (Ohio EPA 1987, Lyons et al. 2001). Most typically, IBIs for rivers have been used to address specific issues, such as the effects of dams (e.g., Scott 1999) or to represent a specific region or state (Ohio EPA 1987, Lyons et al. 2001). While benefits and shortcomings, including difficulty of interpretation where multiple stressors exist, have been debated extensively (Sutter 1993, Simon 1999), their use has become a mainstay in the evaluation of ecosystem health.

Structural measures of density and abundance have been the most commonly used approach in the development of IBIs. While structural measures are useful measures of system health, they often contribute little information about how an ecosystem works (Harris 1994). There may also be occasions where structural measures of diversity and abundance may not tell the whole story if the effect of a stressor leads to reduced vigor of organisms but not necessarily a decline in densities or diversity. An additional concern is that structural measures, because they often require intensive sampling efforts, might be limited to local questions and not be applicable across broader spatial scales (Whittier et al. 2002). An alternative to density- and abundance-based metrics is the application of metrics based on functional processes (e.g., primary and secondary production) as measures of ecosystem health. The use of indicators of ecosystem function has been on the increase in recent years (Bunn et al. 1999, Bunn and Davies 2000) and ecosystem function-based indicators have been established in monitoring protocols in Australia (Smith and Storey 2001). Recent attention has been given to the use of ecological indicators based on conceptual views of functional dynamics of rivers because functional processes in ecosystems are influenced by a number of biotic and abiotic factors, most of which are, in turn, influenced by ecosystem disturbance (Lorenz et al. 1997). Indicators based on functional dynamics, therefore, would be integrative measures, summarizing the effects of disturbance for many ecosystem components into a single or minimal number of metrics.

Food web ecology has become an increasingly useful discipline for addressing ecological questions. It has been used to consider: direct and indirect effects of predation (Power et al. 1985, Spiller and Schoener 1996), intraguild predation (Polis and Holt 1992, (Melian and Bascompte 2002), metapopulations (Holyoak 2000), competition (Holt 1984), community structure (Winemiller 1990), ecosystem dynamics (DeAngelis 1992), and landscape effects on ecosystem structure (Hershey et al. 2000). Food web ecology has also been identified as a critical component in the development of management strategies for tropical fisheries (e.g., Winemiller and Jepsen 1998). Food web ecology has also been applied to questions relating to the assessment of potential environ-mental impacts, particularly with regard to aquatic pollution (e.g., Van Dover et al. 1992). Food webs have been successfully applied in these situations because they represent biotic interactions across population, community and ecosystem scales because trophic dynamics involve interactions between resources and consumers.

The goal in trophic studies is to establish linkages between potential food sources and consumers to establish the food sources critical to the food web and ascertain trophic levels of representative fauna. A common approach in developing food web models is the use of gut content analysis. Because this approach requires the removal and examination of the digestive tract of a large number of individuals for an accurate quantitative assessment, gut content analysis can be quite labor intensive in the field and laboratory. Moreover, gut contents reflect everything that a consumer has ingested. In some situations, food items may be digested rapidly and not be detectable. A number of other materials may be digested or variously processed to the point that they are no longer recognizable, thus resulting in their placement in a sweeping category referred to as "amorphous detritus". Finally, what is present in the gut is a reflection of what has been ingested. Much of this matter may pass through the gut undigested, thereby not contributing to the actual energy assimilated by a consumer.

An alternative to the development of food web models is the use of ratios of stable isotopes. Stable isotope analysis measures the ratio of elements found in the organic molecules transferred from food source to consumer. The primary advantage of stable isotopes over gut content analysis is that isotopic ratios reflect what has actually been assimilated by a consumer and converted to biomass; undigested materials are not measured through isotopic examination of the tissues of a consumer. Preparation of samples for stable isotope analysis is also less labor-intensive in the laboratory. The two most common stable isotope ratios used in food web studies are for carbon and nitrogen, although sulfur is sometimes included as an additional marker. Carbon and nitrogen are used because their stable isotope ratio exhibits a predictable change from food source to consumer. The ratio of 13C/12C, referred to as δ13C, allows for a linkage between food source and consumer because the ratio changes very little (+ 1o/oo) from source to consumer (Peterson and Fry 1987, Keough et al. 1996). The ratio of 15N/14N, referred to as δ15N, allows for the determination of the number of trophic levels present because the ratio increases 2 – 3o/oo from a lower to higher trophic level (DeNiro and Epstein 1981, Hansson et al. 1997, Thorp et al. 1998). The use of δ13C and δ15N also provides better overall resolution when determining linkages between sources and consumers particularly when the degree of separation of δ13C of potential food sources is small.

In work with colleagues, we have been able to address how channel type (constricted vs. floodplain; Thorp et al. 1998) and catastrophic flooding (Delong et al. 2001) influence trophic dynamics in large rivers. Critical in both of these studies was being able to isotopically discriminate between the numerous food sources present in large rivers. While some overlap between instream primary producers (principally benthic filamentous algae and aquatic macrophytes), we were able to conclude that C3 terrestrial plants, such as silver maple and cottonwood, in the floodplain were not contributing to the food web (Figure 1). While C3 terrestrial plants have a δ13C within 1o/oo of most of the primary consumers (e.g., filterers, grazers), the difference between the δ15N of C3 plants and primary

consumers is considerably greater than the 3o/oo difference expected if terrestrial vegetation was a major component of the diet of primary consumers. Based on the expected differences in isotopic ratios of primary consumers and potential food sources, it was concluded that the most important food sources in the Ohio and Upper Mississippi River were fine (1 mm – 100 μm diameter) and ultrafine (100 – 1 μm diameter) organic matter transported in the water column and dissolved nutrients were the forms of organic matter driving the food web of these systems (Delong et al. 2001). The C:N ratio of TOM suggested that these fractions were composed primarily of organic matter that was riverine in origin, including phytoplankton.

While the food webs of floodplain reaches of the Ohio and Upper Mississippi Rivers appear to be driven by similar food sources, the food web of the lower Missouri River appeared to function very differently (Figure 2). The δ13C of transported organic matter and other potential food sources in the Missouri River appeared to be more 13C-enriched than similar sources in the Upper Mississippi, suggesting a difference in composition. This is also supported by the lower δ15N of potential sources in the Missouri and the differences in the sources are evident throughout the consumer groups, even up to the piscivorous fishes. While not within the original scope of the study, the question was raised as to why the food web of the Upper Mississippi and Ohio Rivers would be so similar, while that of the Missouri, which is also a floodplain river, would be different. Of the three, the lower Missouri has undergone the most extensive alteration. The flood pulse has been greatly diminished through the construction of large reservoirs, the once numerous braided channels have been reduced to a single channel, and the floodplain has been largely disconnected from the floodplain through the construction of levees (Galat et al. 1998). Are these differences a result of natural differences between the river basins or is there a commonality to the functioning of large river food webs that has been disrupted in the Missouri? This question could be easily addressed by examining components of the food web in minimally impacted reaches of the Missouri and comparing it to the food web of more heavily impacted reaches using a stable isotope approach.

Application of food web ecology using stable isotope techniques does not require that a complete food web model be produced. Recent studies have demonstrated that measures of trophic position, particularly for top predators, can be used to assess the status of a food web (Vander Zanden and Rasmussen 1996, Post 2002). In contrast to trophic level models, which provide a simple identifier of trophic level, trophic position models recognize that consumers feed at multiple levels, thereby incorporating omnivory into the model. The result is that trophic position is a continuous variable that will change in response to food quality and availability, in contrast to fixed-integer values of trophic levels (Figure 3). To reliably measure trophic position, it is only necessary to obtain stable isotope data from: a top predator; a primary consumer feeding primarily from the water column; and a primary consumer using benthic organic matter. From this, then, it is possible to calculate the maximum trophic position in a food web and compare this measure from each site

to assess the condition within a system (*sensu* Vander Zanden and Rasmussen 1996).

In summary, food web models represent an integration of critical biotic and abiotic components in an ecosystem. As such they have the capacity to serve as a simple means of assessing ecosystem health by simultaneously incorporating multiple measures into a single entity. The use of stable isotopic data is a cost-effective means of generating food web models because they require only straightforward measures in the field and are easily prepared in the laboratory for analysis. While often viewed as a descriptive tool, approaches are available where quantitative measures, such as trophic position of a consumer, can be used to generate "scaleable" measures of ecosystem health. Further advances and refinements in the use of stable isotopes for examining trophic relationships will serve to increase their utility as a bioassessment tool.

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Literature Cited

Agency, O. E. P. 1987. Biological criteria for the protection of aquatic life, colume II: Users manual for biological field assessment of Ohio surface waters. Ohio EPA, Division of Water Quality Monitoring and Assessment, Columbus, OH.

Arnold, T. L., D. J. Sullivan, M. A. Harris, F. A. Fitzpartick, B. C. Scudder, P. M. Rul, D. W. Hanchar, and J. S. Stewart. 1999. Environmental setting of the upper Illinois River Basin and implications for water quality. Water Resources Investigations Report 98-4268 U.S. Geological Survey.

Battaglin, W. A., C. Kendall, C. C. Y. Chang, S. R. Silva, and D. H. Campbell. 2001. Isotopic and chemical composition of inorganic and organic water-quality samples from the Mississippi River basin, 1997-98. Water-Resources Investigations Report 01-4095 U.S. Geological Survey, Denver, CO.

Bunn, S. E., and P. M. Davies. 2000. Biological processes in running waters and their implications for the assessment of ecological integrity. Hydrobiologia 422/423:61-70.

Bunn, S. E., P. M. Davies, and T. D. Mosisch. 1999. Ecosystem measures of river health and their response to riparian and catchment degradation. Freshwater Biology 41:333-345.

DeAngelis, D. L. 1992. Dynamics of nutrient cycling and food webs. Chapman & Hall, New York.

Delong, M. D., J. H. Thorp, K. S. Greenwood, and M. C. Miller. 2001. Responses of consumers and food resources to a high magnitude, unpredicted flood in the Upper Mississippi River basin. Regulated Rivers: Research and Management 17:217-234.

DeNiro, M. J., and S. Epstein. 1981. Influence of diet on the distribution of nitrogen isotopes in animals.

Geochimica et Cosmchimica Acta 45:341-351.
Galat, D. L., and coauthors. 1998. Flooding to restore connectivity of regulated, large-river wetlands. Bioscience 48:721-733.
Hansson, S., J. E. Hobbie, R. Elmgren, U. Larsson, B. Fry, and S. Johansson. 1997. The stable nitrogen isotope ratio as a marker of food-web interactions and fish migration. Ecology 78:2249-2257.
Harris, G. P. 1994. Pattern, process, and prediction in aquatic ecology. A limnological view of some general ecological problems. Freshwater Biology 32:143-160.
Holt, R. D. 1984. Spatial heterogeneity, indirect interactions, and the coexistence of prey species. American Naturalist 124:377-406.
Holyoak, M. 2000. Habitat subdivision causes changes in food web structure. Ecology Letters 3:509-515.
Keough, J. R., M. E. Sierszen, and C. A. Hagley. 1996. Analysis of a Lake Superior coastal food web with stable isotope techniques. Limnology and Oceanography 41:136-146.
Lorenz, C. M., G. M. Van Dijk, A. G. M. Van Hattum, and W. P. Cofino. 1997. Concepts in river ecology: implications for indicator development. Regulated Rivers: Research and Management 13:501-516.
Lyons, J., R. R. Piette, and K. W. Niermeyer. 2001. Development, validation, and application of a fish-based index of biotic integrity for Wisconsin's large warmwater rivers. Transactions of the American Fisheries Society 130:1077-1094.
Patrick, R. 1998. Rivers of the United States. IV. Part A: The Mississippi River and tributaries north of St. Louis. John Wiley & Sons, New York.
Peterson, B. J., and B. Fry. 1987. Stable isotopes in ecosystem studies. Annual Review of Ecology and Systematics 18:293-320.
Polis, G. A. and R. D. Holt. 1992. Intraguild predation: the dynamics of complex interactions. Trends in Ecology and Evolution 7:151-154.
Post, D. M. 2002. Using stable isotopes to estimate trophic position: models, methods, and assumptions. Ecology 83:703-718.
Power, M. E., W. J. Matthews, and A. J. Stewart. 1985. Grazing minnows, piscivorous bass, and stream algae: dynamics of a strong interaction. Ecology 66:1448-1456.
Scott, E. M., Jr. 1999. Tailwater fish index (TFI) development for the Tennessee River tributary tailwaters. Pages 507-522 *in* T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, FL.
Simon, T. P. 1999. Introduction: biological integrity and use of ecological health concepts for application to water resource characterization. Pages 3-16 *in* T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, FL.
Smith, M. J., and A. W. Storey. 2001. Developing an ecosystem health monitoring program for rivers and streams in southeast Queensland. Final Report Southeast Queensland Regional Water Quality Management Strategy.

Spiller, D. A. and T. W. Schoener. 1996. Food web dynamics on some small subtropical islands: effects of top and intermediate predators. Pages 160-169 *in* G. A. Polis and K. O. Winemiller (ed.), Food webs: integration of patterns and dynamics. Chapman and Hall, New York.

Sutter, G. W., II. 1993. A critique of ecosystem health concepts and indexes. Environmental Toxicology and Chemistry 12:1533-1539.

Thorp, J. H., M. D. Delong, K. S. Greenwood, and A. F. Casper. 1998. Isotopic analysis of three food web theories in constricted and floodplain regions of a large river. Oecologia 117:551-563.

Van Dover, C. L., J. F. Grassle, B. Fry, R. H. Garritt, and V. R. Starczak. 1992. Stable isotope evidence for entry of sewage-derived organic material into a deep-sea food web. Nature 360:153-156.

Vander Zanden, M. J., and J. B. Rasmussen. 1996. A trophic position model of pelagic food webs: impact on contaminant bioaccumulation in lake trout. Ecological Monographs 66:451-477.

Whittier, T. R., S. G. Oaulsen, D. P. Larsen, S. A. Peterson, A. T. Herlihy, and K. P.R. 2002. Indicators of ecological stress and their extent in the populations of northeastern lakes: a regional-scale assessment. Bioscience 52:235-247.

Winemiller, K. O. 1990. Spatial and temporal variation in tropical fish networks. Ecological Monographs 60:331-367.

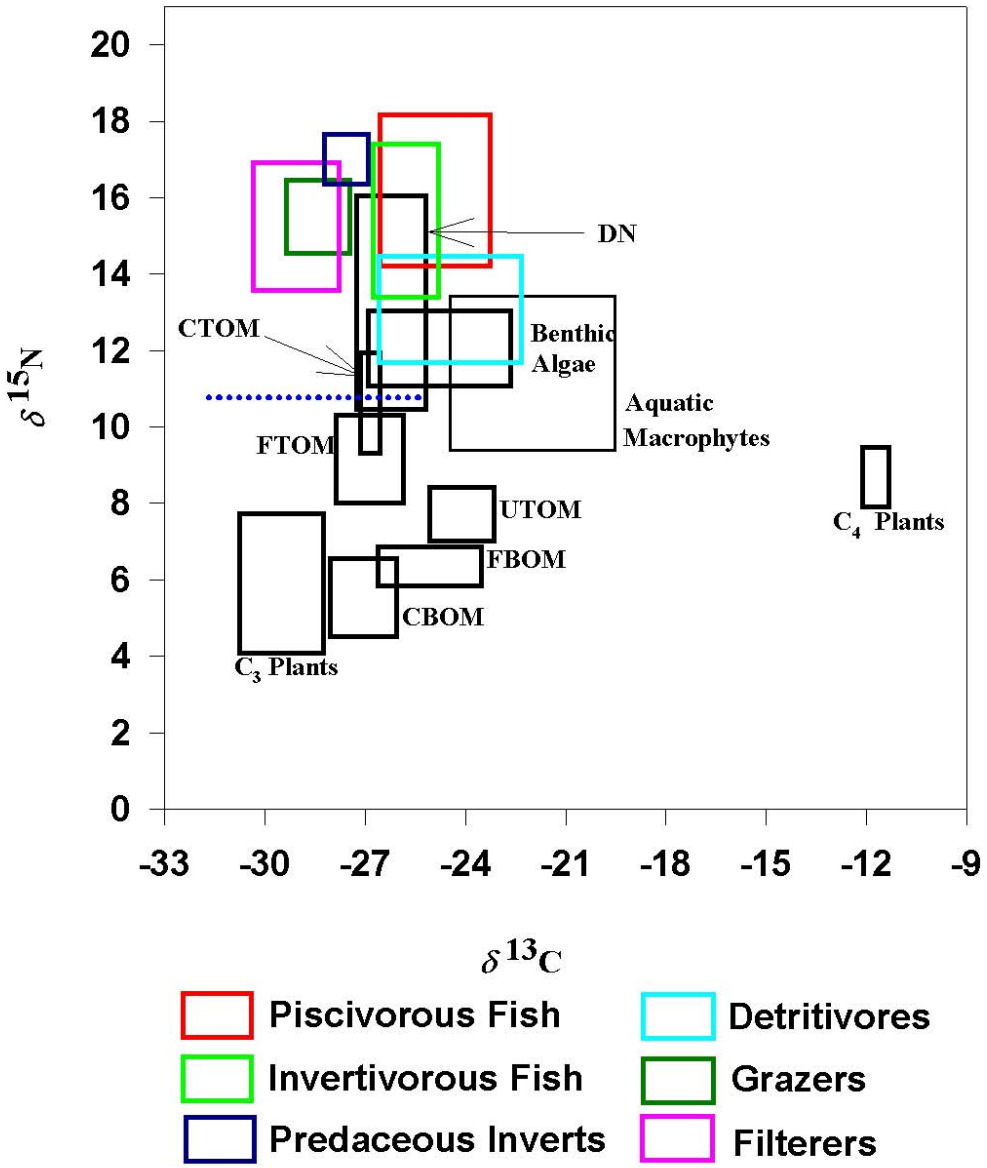

Figure 1. Stable isotope ratios of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) for consumers (color boxes) and potential food sources (black, labeled boxes) from the Upper Mississippi River near Louisiana, Missouri, July 1994. Boxes encompass the mean $\pm$ 1 standard deviation for observed values. Food sources represent size fractions of coarse, fine, and ultrafine transported organic matter (TOM) and benthic organic matter (BOM). DN represents dissolved nutrients. If consumers were using C_3 terrestrial plants as a food source, part of their box should fall below the dashed line.

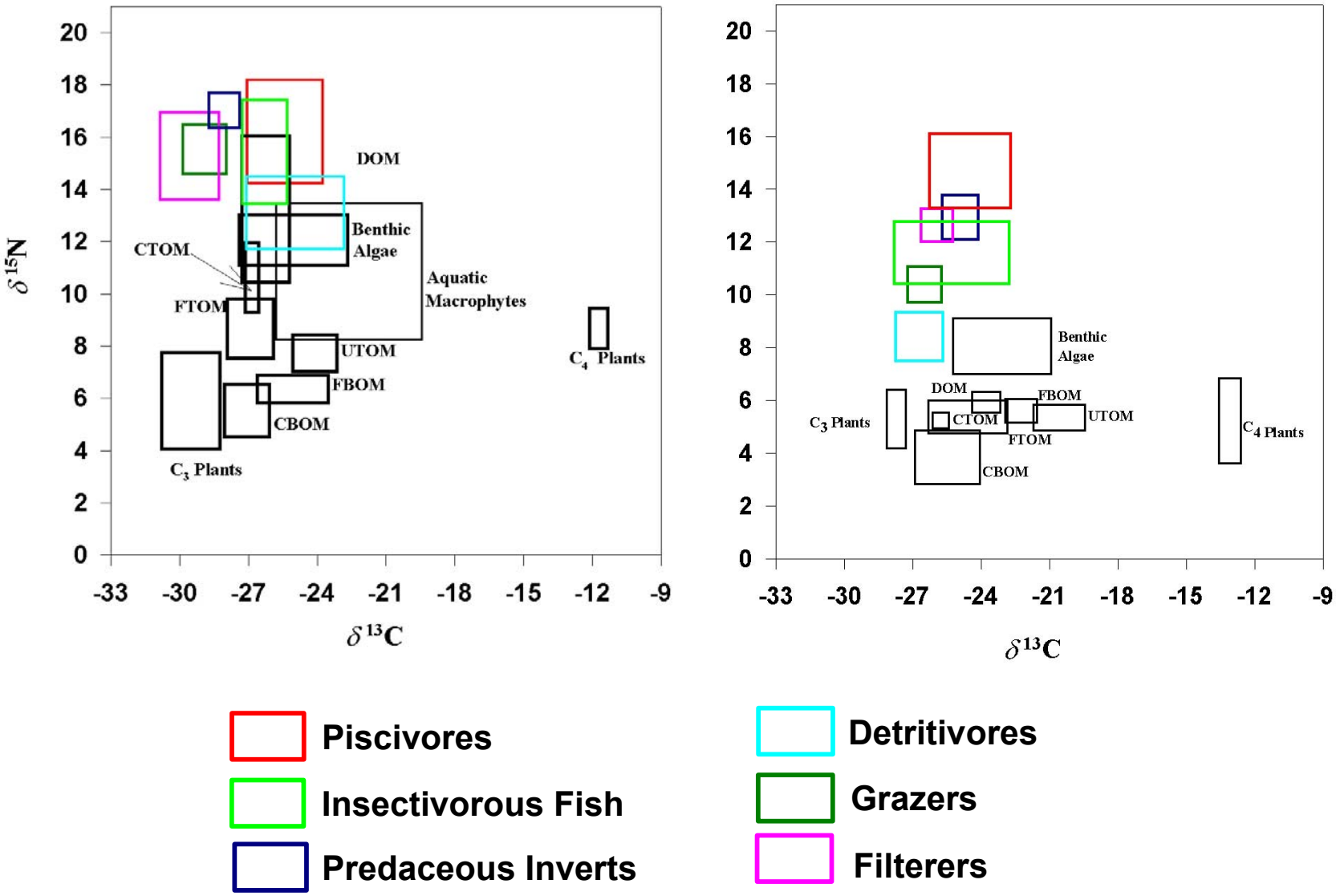

Figure 2. Comparison of the relationship between consumers and potential food sources in the Upper Mississippi River (A) and lower Missouri River (B) near New Haven, Missouri. The more ^{13}C-enriched and ^{15}N-depleted nature of resources and consumers in the Missouri cause the boxes for these categories to shift down and to the right relative their counterparts in the Upper Mississippi River.

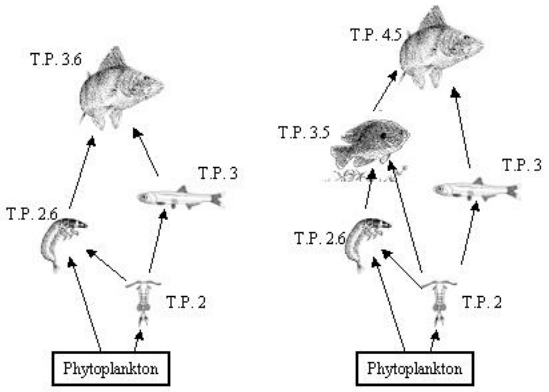

Figure 3. Example of the use of trophic position, as determined using carbon and nitrogen stable isotope ratios in food chains. The number of trophic levels in both models remains 4; however, the absence of an insectivorous fish (left) reduces the trophic position of the top predator from 4.5 to 3.6 because it has to supplement its diet with organisms feeding closer to the base of the chain.

AN ASSESSMENT OF FRESHWATER BIODIVERSITY IN THE CENTRAL MIXED GRASS PRAIRIE ECOREGION

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Abstract. To focus its conservation efforts, The Nature Conservancy has undertaken the identification of areas of biodiversity significance within ecoregions within North America, Latin America, and Asia. As part of this effort, we have identified priority river systems in the Central Mixed-Grass Prairie Ecoregion (CMGP), a planning area occupying roughly 59 million acres in the central portions of Nebraska, Kansas and Oklahoma. The CMGP overlaps parts of three ecoregional sections defined by the U.S. Forest Service: the Sand Hills, the South Central Great Plains, and the Red Bed Plains (McNab and Avers 1994). The goal of this planning effort was to represent the elements of aquatic diversity across the major environmental gradients in the CMGP, and to select a suite of places that together represent the best opportunities to protect the aquatic diversity of the region. The first step in creating the plan was to define hydrologic units that capture environmental gradients and zoogeographic distinctions. Based on physiographic differences, we divided the major drainages that intersect the CMGP (Missouri River and Arkansas River) into ten Ecological Drainage Units (EDUs). Using the Conservancy's aquatic classification framework, we also developed a taxonomy of about 80 Freshwater Ecological Systems that describes the physical diversity of riverine ecosystems. The second step was to select the elements of biological diversity (species and ecological systems) that would be addressed in the plan (i.e., targets). For species, we followed expert recommendations to include thirteen imperiled or declining fish and five imperiled or declining aquatic macroinvertebrates. We also targeted intact natural assemblages or groups of species. We then applied an integrity assessment to select the best occurrences of each ecosystem target (See Gagnon et al., poster presentation). This presentation will discuss the results of this assessment and the methods used to classify aquatic ecosystems and select the set of rivers that meet the conservation goals for each target in the ecoregion.

Key words: freshwater, ecological classification, conservation, ecoregional planning, biodiversity, priority setting, Great Plains, prairies, Kansas, Nebraska, Oklahoma.

Introduction

The Central Mixed Grass Prairie (CMGP) ecoregion is one of 80 planning units in the United States for which The Nature Conservancy is developing a conservation vision. The CMGP encompasses 59 million acres of central Nebraska (the Sand Hills region), central Kansas, and north central Oklahoma. This paper will provide a brief overview of the freshwater priority setting aspect of the conservation plan for the CMGP, a project conducted from Fall 2001- Fall 2002. First the paper will address the purpose of the Conservancy's ecoregional planning approach, then outline the five steps involved in the freshwater assessment for this one specific ecoregion.

Purpose and goal of Ecoregional planning

Ecoregional planning is the approach we take to set priorities for conservation, i.e., identifying areas of biodiversity significance (Groves et al. 2000). Briefly, ecoregional

planning is a comprehensive process that entails selecting and designing networks of areas which best capture representative biodiversity native to each ecoregion. In other words, we try to identify the suite of places within each ecoregion which, if managed appropriately, would ensure the viability of all native biodiversity over the long-term.

Five steps to identify priority aquatic conservation areas

1. Identify Stratification Units

Ecological Drainage Units (EDUs) are groups of watersheds that share a common zoogeographic history and physiographic and climatic characteristics. We expect that each EDU will contain sets of aquatic system types with similar patterns of drainage density, gradient, hydrologic characteristics, and connectivity. We use EDUs to stratify our selection of conservation targets so that we represent these targets in our conservation plans across the environmental gradients on which they array.

The fish fauna of the Great Plains is fairly uniform and not very distinct from nearby regions. It has over 77 native species – but nearly all of these species are all found in the Central Lowlands. Species diversity decreases and the fauna become more ubiquitous between drainages as you move north and west from the mainstem Mississippi River (Cross et al. 1986). Cross et al. (1986) name two main zoogeographic divisions in the Central Mixed-Grass Prairie ecoregion (CMGP) – those watersheds in the Missouri River drainage that represent the northern plains/central lowland fauna and those in the Arkansas River drainage that represent the fauna of the southern plains. This division is also represented in the aquatic zoogeographic units delineated by the World Wildlife Fund (Abell et al. 2000) – the Middle Missouri River subregion and the Southern Plains subregion.

We further divided these two regions based on physiography to identify a total ten EDUs that intersect the CMGP. The CMGP comprises three ecoregional sections: the Sand Hills, the South Central Great Plains, and the Red Bed Plains (McNab and Avers 1994). These sections are distinct in terms of their lithology and landform. The Sand Hills are characterized by rolling to steep irregular sand dunes, long narrow valleys, and abundant groundwater (McNab and Avers 1994). The South Central Great Plains is a gently sloping area with loess-capped ridge tops, rivers with broad flood plains, and smaller streams occurring in narrow bottomlands. The third main section is the Red Bed Plains. These shale dominated plains are highly irregular, with a low density of small to medium intermittent streams.

2. Identify Conservation Targets

In the CMGP we focused on two sets of conservation targets – fine filter and coarse filter. The fine filter comprises imperiled species – those that are globally rare, endemic, or known to be declining. The aquatic species targets included thirteen fish, five insects and two reptiles. In addition, experts identified seven natural assemblages that added to the selection of priority areas. We applied an abiotic classification to the CMGP to define 81 aquatic ecological system (AES) types which serve as the coarse filter – representing the common species and diversity above the level of species. The factors used to define the AES types included geology, permanence of flow, size, position in the drainage and gradient.

3. Conduct Expert Review of Ecologically Significant Areas

In November 2001, we met with regional experts to identify the best occurrences of the target species as well as places that maintain good examples of natural assemblages or aquatic system types. They nominated 25 streams and provided information on the viability of the targets at the those locations as well as the primary threats.

4. Identify the Highest Quality Occurrences of the Conservation Targets and Select Conservation Areas

Following the November meeting, we analyzed the proposed streams to see how well they captured the full array of aquatic system types. To fill in the gaps in what was captured and to provide additional information on the systems that were selected, we assessed the quality of the aquatic systems using watershed analysis of land use/land cover patterns, densities of point sources, and state government assessments of water quality. Given the high level of alteration to this region, it was difficult to distinguish the highest quality systems. One challenge is that we don't have clear thresholds of land use change that indicate when a systems is no longer viable. Efforts to correlate the land use and point source data with biomonitoring data or expert opinion was inconclusive. In the end we used the stream reaches nominated by the experts to select our system examples.

5. Evaluate Goals and Summarize Threats

The final set of aquatic conservation areas represents a subset of the fine filter and coarse filter targets. There are large geographic gaps due to patterns of land use. For example, agriculture and water withdrawal has dramatically altered the streams of the Smoky Hills basin in Central Kansas to the point that we could not recommend any areas – so the plan does not represent the stream types found there. The best opportunities for aquatic conservation appear to be the large rivers that still maintain viable populations of target species. However, these aquatic systems also are highly threatened by the cumulative impacts of agricultural practices, flow regulation and water consumption.

For additional information on the Central Mixed Grass Prairie Ecoregion and forthcoming report please contact Al Steuter, Director of Science and Stewardship, Nebraska Field Office. 431 N. Maple Street, Ainsworth NE 69210. (402) 387-1061, asteuter@tnc.org. For additional information on the aquatic classification and prioritization, contact the author.

Literature Cited

Abell, R. A, D. M. Olson, E. Dinerstein, P. T. Hurley, J. R. Diggs, W. Eichbaum, S. Walters, W. Wettengel, T. Allnutt, C. J. Loucks, and P. Hedao. 2000. Freshwater Ecoregions of North America: a conservation assessment. World Wildlife Fund – US, Island Press, Washington, DC. 319pp.

Cross, F. B, Mayden, R. L. and J. D. Stewart. 1986. Fishes in the western Mississippi Basin (Missouri, Arkansas and Red Rivers). *In* Hocutt, C. H. and E. O. Wiley, eds. The Zoogeography of North American Freshwater Fishes. John Wiley and Sons: New York, p.p. 363-411.

Gagnon, P. J., Lammert, M., Higgins, J. V. and T. W. FitzHugh. 2002. Development of a multimetric aquatic stressors index for the Central Mixed Grass Prairie Ecoregion. Poster presented at September 2002 Central Plains Aquatic Bioassessment and Biocriteria Symposium. Lawrence, Kansas.

Groves, C., L. Valutis, D. Vosick, B. Neely, K. Wheaton, J. Touval, and B. Runnels. 2000. Designing a Geography of Hope: a practitioner's handbook for ecoregional conservation planning. Arlington, VA: The Nature Conservancy.

McNab, W. H. and P. E. Avers. 1994. Ecological Subregions of the United States: Section Descriptions. U.S. Department of Agriculture, Forest Service. WO-WSA-5. Washington, DC.

CLASSIFICATION OF OKLAHOMA RIVERS AND PRELIMINARY DEVELOPMENT OF AN INDEX OF BIOTIC INTEGRITY

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Abstract. The Oklahoma Department of Environmental Quality has monitored fish in Oklahoma rivers annually since 1976 with standardized beach seining methods at 12 to 80 fixed sites. Multivariate analysis of the species abundance data suggested that there are three classes or types of Oklahoma rivers related to size and geography: (1) large rivers, including the Arkansas and lower Red Rivers; (2) prairie rivers, including most of the North Canadian, Canadian, Washita, and upper Red Rivers; and (3) saline prairie rivers, including most of the upper Cimarron, upper North Canadian (Beaver River) Rivers, and some upper Red River tributaries. Each river basin traverses several ecoregions across the state. Neither ecoregion nor river basin adequately accounted for differences among Oklahoma fish assemblages. Our analyses indicated that fish community composition is best explained by mean annual flow and salinity.

Metrics were selected for development of an Index of Biotic Integrity (IBI) by comparison of reference sites believed to be least stressed, with known impaired sites. The IBI metrics were responsive only in the large river sites, and an IBI was derived from seven metrics (numbers of total taxa, darter species, intolerant species, sucker species, sunfish species, percent insectivorous cyprinids, and total individuals). Further analysis of IBI scores in all three river types showed that several sites had consistently high or consistently low scores throughout the 20-year period of record, but no apparent temporal trends were found

Results of this study are being used by Oklahoma to expand its fish sampling methods in nonwadeable rivers, defined by criteria of greater than 1.25m depth and 10 cfs, and to develop a technically sound framework for large river biocriteria. Ongoing efforts are focused on evaluating the use of ecoregions in Oklahoma and the river classes suggested by this study.

Key words: rivers, fish assemblage, bioassessment, index of biotic integrity, biological metrics.

Introduction

The Oklahoma Department of Environmental Quality (DEQ) developed an Index of Biotic Integrity (IBI) for use in assessing the quality of Oklahoma's large rivers, and for eventual development of biological criteria for river water quality. *Biological integrity* is commonly defined as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of the natural habitat of the regions" (Karr and Dudley 1981; Gibson et al. 1996). This concept refers to the natural assemblage of indigenous organisms that would inhabit a particular area if it had not been affected by human activities. Biological integrity or naturally occurring diversity becomes the primary reference condition used to measure and protect waterbodies in a particular region. In the Central plains there is extensive agriculture and ranching, urban development and other land use, so identification of natural, unstressed waterbodies is problematic.

Careful measurement of key attributes of the aquatic ecosystem and its constituent biological communities can determine the overall biological condition. These key attributes or biological endpoints are critical to quantifying and ranking the quality of the waters of concern. The biological endpoints are established by biosurveys based on sampling of fish, invertebrates, plants, and other flora and fauna. Such biosurveys establish the endpoints or measures used to summarize several community characteristics such as taxa richness, number of individuals, sensitive or insensitive species, observed pathologies, and the presence or absence of essential habitat elements.

The Index of Biotic Integrity (IBI), developed for stream fish assemblages (Karr and Dudley 1981, Karr et al. 1986, Karr 1991), has now been applied in many regions of North America (e.g., Miller et al. 1988, Yoder 1991, Simon 2000, Halliwell et al 2000) A regional application of IBI and other such indexes requires calibration of metrics to regional reference conditions and region-specific responses of the metrics to perturbation (Gerritsen 1995). The classification may determine patterns that do not follow any previously described regional framework (Fausch et al. 1984; Simon and Lyons 1995). This classification process, applied to Oklahoma fish fauna, will allow the development of a set of biological expectations that will increase the utility of the data for assessments of ecosystem condition or biological integrity. This paper proposes a regional classification of fish assemblages in Oklahoma rivers, and develops a preliminary IBI for use in Oklahoma, based on data from 20 years of monitoring Oklahoma rivers.

The Oklahoma DEQ has maintained a fish monitoring program that consists of ichthyofaunal surveys at permanent sampling sites in the major rivers of Oklahoma, as well as their headwaters and major tributaries. The Oklahoma fish monitoring data set consists of extensive, repeated measures (12-80 sites annually) of fish community composition and abundance at 96 total sites (Figure 1). The major rivers of Oklahoma (Arkansas, Canadian, Cimarron, Red, and Washita) traverse as many as five ecoregions from west to east (Omernik 1987). Researchers in other sections of the country (Hughes et al. 1986; Larsen et al. 1986; Hughes and Larsen 1988; Lyons 1989; Gallant et al. 1989) have shown that fish fauna differ among ecoregions or physiographic provinces, and in some cases ecoregions may be more appropriate for developing regional expectations for bioassessments than drainage basins. Differences in fish assemblages among ecoregions may include shifts in species composition, as well as shifts in relative abundance of different functional groups of fish.

Our approach was to derive a classification scheme for biological assessments for Oklahoma rivers, thus increasing the regional precision and sensitivity of those bioassessments for detecting impairment. The central task was to examine the Oklahoma fish data for differences in assemblage attributes among rivers, major drainages, and among ecoregions or physiographic provinces.

Methods

The Oklahoma fish monitoring program began in 1976 with sampling at 12 permanent sites. Sites were added annually until 1990, reaching a current total of over 80 permanent sites. During selected years, sampling was carried out at other targeted sites on the mainstems of the rivers. The original monitoring sites were downstream

of major discharges on Oklahoma's large rivers, and were placed near USGS gauging stations so that flow information would be available. A concurrent ambient water chemistry monitoring program existed through 1985, and consisted of semi-monthly sampling and analyses at all permanent sites. Physical-chemical parameters measured during fish sampling events from 1986 to the present have included dissolved oxygen, temperature, conductivity, pH, turbidity, and Secchi transparency.

All fish samples (from 1976 to the present) were collected with 3.05m x 1.22m (4.76mm mesh) seine nets. Sampling effort was standardized at each monitoring site and consisted of 20 seine-hauls, each being 10m in length, resulting in a total sample reach of 200m (J. Pigg, personal communication). All fish were preserved, transported to the laboratory, and identified to species (including hybrids).

Data files were obtained from the Oklahoma DEQ, and were entered into a relational database (Access) so that queries could be developed to create reduced data files for statistical analysis. To limit the potential variability of the fish data, analyses were restricted to those samples obtained in an index period of May 1 to September 1. Furthermore, because 1 to 3 samples were taken at a site within the index period in any one year, a seasonal mean value (of the 1 to 3 samples) was used for all analyses. Oklahoma DEQ conducted 1,652 sampling events at the 96 sites (consisting of from 1 to 40 events at any one site during the period), yielding a total of 2,758,706 individual fish representing 133 species.

Because the concept of "natural" or undisturbed rivers in the central plains is likely nonexistent, the least disturbed conditions were identified by "best professional judgment" (BPJ) of the Oklahoma DEQ. Ideally, quantitative non-biological criteria would be used to avoid circularity in establishing robust regional reference conditions. However, these criteria and associated background information were not available at the time of this study. As an alternative, we used the BPJ as a first step, then refined the list of reference sites through evaluation of the biological data. These reference sites were subjected to a classification analysis.

The objective of classification is to group similar waterbodies together, so that reference conditions will reflect reasonable expectations for assessing waterbodies. The basic assumption of classification is that physical habitat and water quality largely determine the composition of biological communities in waterbodies. Therefore, if waterbodies are classified adequately, reference biological community types should correspond to the classification. Classification is often an iterative process of refining the classification scheme as new data are obtained, until a satisfactory classification emerges that accounts for variation in the reference site biological data.

Classification consisted of multivariate ordination to observe whether gradients or discontinuities occur in the fish assemblage structure, followed by examination of potential associations of environmental variables with the observed gradients in assemblage structure. For ordination and classification, relative species composition and abundance data were used instead of metric values. Species abundances were cumulative for all samples within the index period in a year. A matrix of dissimilarities, chord distance (Kenkel and Orloci 1986; Ludwig and Reynolds 1988), was calculated from the species abundance-by-sites matrix. Ordination consisted of nonmetric multidimensional scaling (NMDS). NMDS

works on a matrix of ranked distances among sites, and thus is distribution-free and unaffected by non-normality and nonlinearity in the data (Ludwig and Reynolds 1988).

Using the ranked distances, NMDS attempts to create a "map" of the data points in two or three dimensions, similar to creating a map from a set of distances among cities. The objective of NMDS is to obtain a "best fit" between the original dissimilarity measures (the distances) and the estimated ordination (the map). The dissimilarities have as many dimensions as there are sites, but the ordination reduces these to a smaller number, usually two or three. The procedure is to rank the distances in the matrix from smallest to largest, then to calculate an initial starting configuration directly from the dissimilarity matrix. A second set of distances ("map" distances) are calculated from the initial configuration, and their ranks are compared to the "true" original distances. A best solution is sought iteratively, changing the configuration so that the rankings of "map" and "true" distances become more similar. Goodness of fit of the configuration is measured by Kruskal's stress coefficient, and the iteration stops when the stress coefficient converges to a minimum value (SYSTAT 1992). The result of NMDS is a final configuration, consisting of coordinates for each site in the dimensions chosen. Points close to each other in ordination space represent sites with similar species composition. Correlation of environmental variables with each axis of the ordination can provide insight on environmental gradients that may be associated with species composition of the sites.

The NMDS ordination was repeated for three separate fish monitoring program sample years: 1979, 1986, and 1994. River basins were examined for consistent values of environmental variables that were correlated with the ordination. Each basin of the *a priori* classification was also examined in ordination space to find those basins that coincided with each other and could be aggregated.

An Index of Biotic Integrity (IBI) was developed using sixteen fish community attributes (i.e., metrics) to determine the biological condition of Oklahoma streams. The sixteen metrics (Table 1) were examined to determine their suitability for the assessment of river condition using the Oklahoma fish monitoring database. The final selection of candidate metrics was made based on several criteria. First, and foremost, the metrics must be ecologically relevant, and must be responsive to environmental changes or degradation. The chosen metrics should reflect insights from individual, population, community, ecosystem and zoogeographic perspectives of the region.

To date, relatively little work has been directed toward the modification of the IBI for large rivers in the United States. Difficulties in the ease or accuracy of sampling river fish assemblages, and the paucity of appropriate least-impaired reference sites for setting metric expectations have hampered IBI development for large river systems (Simon and Lyons 1995, Simon 2000). Modifications that have been documented in the literature, e.g., for rivers in Ohio (Ohio EPA 1987) and Indiana (Simon 1992) have involved unique metrics specially tailored for large river fauna. However, these metric adaptations (e.g., number of specialized large-river species; percent of the total that are round-bodied suckers) are not suitable for inclusion in the Oklahoma IBI, because they were developed based on data gathered using large-scale boat electrofishing methods. The Oklahoma fish monitoring

data reflect the results of river seining methods, a collection method necessitated by the physical nature of most monitoring sites (i.e., not suited to boat sampling during low-flow periods).

Desirable characteristics of IBI metrics included: (1) A consistency of response among river groups; (2) At least one response (among the river groups) where the median of test sites was beyond a quartile of proposed reference sites – a test median below the reference quartile is equivalent to 50% or more of test observations below the reference quartile, showing a measurable response in half or better of test observations; (3) at least one metric representing each of three relevant classes of metrics, i.e., taxonomic richness and community composition, trophic composition, and fish abundance, and (4) minimal redundancy with other metrics.

The first two desiderata were rarely met by candidate metrics. Taxonomic diversity metrics (including total species richness, Shannon diversity, and family-level richness metrics), and fish abundance metrics may be affected by stream size (Karr et al. 1986; Ohio EPA 1989). These metrics were plotted against average flow (measured or interpolated) for a site to determine if adjustment was necessary for smaller rivers or tributaries. If a metric was influenced by flow, then flow or river size was accounted for in scoring the metric. Metrics were also examined to determine differences among geographic river types, after flow or river size had been accounted for.

Results And Discussion

River Classification.— Ordination by NMDS revealed a cluster of sites consisting of many of the intermediate-sized rivers of Oklahoma, such as the Washita, Canadian and lower Cimarron (Figure 2). There were also two more scattered groupings, one consisting of larger river sites of the Arkansas and lower Red; and the other consisting of low flow, smaller and saline streams, such as the upper Cimarron, Beaver River, and Prairie Dog Branch. The pattern repeated in all three years examined with NMDS analysis (Figure 2).

On the basis of the ordination, three groups of streams were identified:

- *Large rivers* (Arkansas River mainstream; Salt Fork of the Arkansas River; and Red River below, but not including, Arthur City, Texas)

- *Prairie rivers* (Canadian; Washita; Red River to Arthur City, Texas; Deep Fork of the Canadian River; Arkansas River at Ponca City and Ralston; and Cimarron River sites above Englewood and below Dover)

- *Low flow or saline prairie streams* (Beaver River at Turpin and Beaver; Prairie Dog Branch; Red River at Waurika; and Cimarron River from Englewood to Dover)

The classification can be due to nearly coincident geographic and physical factors. The river classes comprise a geographic gradient of small intermittent streams (e.g., upper Canadian/ Beaver, upper Cimarron, Red tributaries) in western Oklahoma to large, impounded rivers in eastern Oklahoma (e.g., Arkansas mainstream and lower Red). However, the three river types were not distinct but represented more of a gradient. In particular, the saline rivers appeared to be arrayed on a linear gradient from prairie streams to the most saline sites, especially the Cimarron River in 1994

(Figure 2c). Although the river groups could generally be separated in each year, there was considerable year-to-year variability (Figure 2). For example, in 1979, the Cimarron River at Guthrie and Perkins was more similar to other saline river sites, but in 1986 and 1994, these sites were more similar to other prairie river sites. Similarly, the Arkansas River at Ponca and Ralston was often intermediate between the prairie and large river groups.

Ecoregions do not account for biological differences among these long, arid-land rivers because each river traverses several ecoregions from its headwaters to its mouth (Fig. 1). All of the rivers of Oklahoma flow through some combination of the Southwestern Tablelands, the Western High Plains, the Central Great Plains, and the Central Oklahoma/Texas Plains ecoregions. Exceptions include: the South Canadian River does not flow through the Western High Plains; the Arkansas River also flows through the Central Irregular Plains and the Arkansas Valley ecoregions in eastern Oklahoma; and the Red River enters the South Central Plains ecoregion near DeKalb, Texas. Finally, headwaters of both the Arkansas and South Canadian rivers are in the Southern Rocky Mountain ecoregion in Colorado and New Mexico.

The three fish community types were identified via composition and abundance data (Table 3). The "large river" community type is more accurately predicted by median flow than by ecoregion (Table 2) and had more species at each site, but fewer individuals. Large river sites were dominated by gizzard shad, red shiner, and western mosquitofish. Salinity and low-flow tolerant species (e.g., plains killifish) dominated in the saline streams, and fishes typical of large rivers and reservoirs (e.g., gizzard shad, threadfin shad, sunfishes, and temperate basses) were common at the Arkansas and lower Red sites. Prairie rivers were dominated by red shiner and western mosquitofish, and saline streams were dominated by plains killifish, plains minnow, Red River shiner and sand shiner.

Metric Selection and Scoring.— The selection of fish metrics (appropriate for calculation using the Oklahoma data) was based on a review of metrics reported in the literature, an examination of the DEQ metrics, and the selection criteria listed in the Methods. That selection process resulted in the inclusion of thirteen candidate metrics (Table 4). Certain metrics were deleted from the DEQ list to reduce redundancy, eliminate those that lack ecological relevance, and strive to increase the sensitivity of the index for the detection of environmental perturbation. For example, the DEQ metric "percentage of rough fishes" may be important from a fisheries management standpoint; however, the classification of "rough fishes" is ecologically artificial.

Candidate metrics were calculated from the Oklahoma fish composition and abundance data. Because sites were visited one to three times during a season, metrics were calculated for each site visit, and averaged over the seasonal index period. All metrics are thus on the same basis (not cumulative), to allow for different sampling frequency among years and sites, and all information collected in the index period (May 1 to September 1) was used. Species richness metrics for a site could therefore have fractional values.

Precision of Metrics.— Most sites were sampled 2-3 times in the index period (May-August), allowing an estimate of precision of the metrics from multiple visits. Standard deviations of multiple measures is estimated from the root mean square error of a 2-way analysis of variance, where site and year are

the factors in the ANOVA (Snedecor and Cochran 1967), and are shown in Table 5.

Variability of most fish metrics was high, especially given the large area sampled ($610m^2$) and the large numbers of fish captured (average 1670 fish per sample). For example, the standard deviation of total taxa in a sample was 3.7 taxa (Table 5). In general, variance of metrics was slightly greater in large rivers than in the other two classes.

Metric Scoring.— Each metric was scored for the IBI using the 95th percentile of the entire metric distribution (Fausch et al. 1984; Karr et al. 1986; Ohio EPA 1989). The three river types were discriminated by flow, thus, no adjustment was made for flow because the effect of flow on the metrics within each river type was negligible (Table 2). The 95th percentile approach was used because all sites are affected by some combination of water removal, impoundments, discharges, agricultural runoff, urban runoff, or industrial runoff; and least-impaired reference areas could not be easily defined (Gibson et al. 1996). Multiple or cumulative stressors may affect all of the river sites to some extent. This approach has been widely used by other researchers when large databases were available and when it was difficult to identify least-impaired sites (Fausch et al. 1984; Karr et al. 1986; Hite and Bertrand 1989; Simon 1991; Lyons 1992).

Metric Response.—The response of individual metric values for sites identified by ODEQ (J. Pigg; pers. comm.) as potential reference sites and probable impaired sites were examined. We found that the potential reference sites lacked consistent separation from those categorized as "probable impaired", especially in prairie and saline rivers. Differences were only ascertained among reference and probable impaired sites in the large rivers (Arkansas and lower Red). Metrics uniformly failed to distinguish between reference and probable impaired sites in prairie rivers and saline rivers. Accordingly, metrics were selected based on apparent responses in large rivers only. Seven metrics were responsive to probable impairment and were included in IBI calculations (Table 4).

Responsive metrics were assigned scores (non-dimensional values) by trisecting the area beneath the 95th percentile following Fausch et al. (1984). Scores of 5, 3, or 1 were assigned to each metric according to whether its value approximated, deviated somewhat from, or deviated strongly from reference expectations, respectively. The final step in the IBI sequence was summing of each of the seven metric scores for each site. The maximum score possible was 35 (i.e., representing no perturbation) and the minimum score was 7 (i.e., representing extremely degraded conditions).

IBI Performance.— The IBI has been widely applied and accepted as a valuable tool for assessing environmental condition, and has been adapted for numerous regions and calibrated for varying fish assemblages across the country. In each region, maintenance of an IBI program is a dynamic process, involving continuous testing and validation of data, metrics, and reference benchmarks.

Total IBI scores for the entire data set (collected within the targeted index period and chosen period of record) are summarized in Figure 3. The IBI results, grouped by river classification type, indicated a gradient in condition from large rivers and prairie rivers (medians = 23), to low flow or salinity-impaired prairie streams (median = 18).

In the large river sites (Arkansas and lower Red), the "probable impaired" sites consistently scored lower on the IBI and component metrics than the "candidate

reference" sites (Figure 4). However, the IBI failed to discriminate "probable impaired" from reference sites in the prairie rivers and in saline rivers. This failure of the IBI in these river types could be due to a combination of several factors:

- Prairie and saline rivers may be naturally highly stressed environments (extreme low summer flows, flash floods, homogeneous sand substrate, high salinity and high temperature during low flow periods).

- Prairie and saline rivers may be uniformly stressed in Oklahoma by anthropogenic activities, including irrigation pumping of groundwater, salt and nutrient enriched irrigation return flows, disturbance by cattle, and insufficient recovery downstream of stressor inputs.

- Sampling with a seine, as necessitated by the physical nature of most monitoring sites, may restrict the capture of large, mobile fish (e.g., top carnivores large, tolerant fishes) that are indicators in the IBI, resulting in reduced ability to discriminate reference from impaired streams. Seining could be augmented with electroshocking (backpack or barge-mounted) to determine if large species are captured.

- The potential reference and "probable impaired" sites in Oklahoma streams may be similar with respect to the total stressors acting on the fish community (habitat, substrate, water quality, toxic chemicals) so that metrics do not differentiate potential reference from probable impaired sites.

- An IBI-type index requires a relatively species-rich fauna to be able to detect faunal degradation. If the fauna is depauperate in the reference sites, then the index will be less effective. Fish fauna in rivers of western North America are depauperate in comparison to eastern rivers (Lee et al. 1980). Accordingly, IBI development has most often been successful east of the Great Plains.

Conclusions And Recommendations

The Oklahoma DEQ fish monitoring database consists of a long-term record of extensive, repeated measures of fish community composition and abundance. To increase the utility of the data for assessments of biological condition and to allow the development of an acceptable set of biological expectations, the data set was examined for patterns in faunal structure. We utilized statistical approaches in the classification process to determine whether patterns were present that may not follow any previously prescribed regional framework. The resulting classification scheme should improve Oklahoma IBI assessments, and should increase the regional precision and sensitivity of these bioassessments for detecting impairment.

Classification consisted of multivariate ordination to observe whether gradients or discontinuities occur in fish assemblage structure, followed by examination of potential associations of environmental

variables with the observed gradients in assemblage structure. Three groups of rivers were identified on the basis of the ordination:

- *Large rivers —*
  - Arkansas River mainstem below Ralston
  - Salt Fork of Arkansas River
  - Red River below (but not including) Arthur City, Texas

- *Prairie rivers —*
  - Canadian River
  - Washita River
  - Red River to Arthur City, Texas
  - Deep Fork of the Canadian River above Englewood and below Dover Cimarron River sites

- *Saline prairie rivers —*
  - Beaver River at Turpin and Beaver
  - Prairie Dog Branch
  - Red River at Waurika
  - Cimarron River from Englewood to Dover

The classification is due to nearly coincident geographic and physical factors. The river classes, topography, vegetation, mean annual precipitation, and ecoregions generally follow an east-west gradient.

The Oklahoma fish composition and abundance data indicated that fishes differentiated among the three river classifications. Large river sites generally had more species but fewer individuals than the other river classes. Salinity and low-flow tolerant fishes dominated in the saline streams, whereas species typical of large rivers or reservoirs were common in the Arkansas and lower Red rivers.

Fish community attributes or metrics were examined to determine their suitability for application in the assessment of river condition using the Oklahoma fish monitoring database. Seven metrics were selected, and each was scored for the IBI using the 95th percentile of the entire metric distribution. Total IBI scores, grouped by river classification type, indicated a gradient in condition that was generally higher for the large rivers, and the prairie rivers, and lower for the saline streams. The IBI successfully discriminated reference sites from probable-impaired sites in the Arkansas and lower Red rivers (large river class), but it was unable to discriminate reference from probable-impaired sites in either prairie or saline rivers.

Our recommendations from this study are:

1. An index of biotic integrity (IBI) for fish, measured with current seining methods, appears to be successful in the Arkansas River and lower Red River, but was not successful in smaller rivers of the state. Although the IBI with current methods show consistent differences among sites, it would need to be tested and confirmed with careful, *a priori,* definitions of reference and impacted sites before the observed differences can be interpreted as impairment of the fish community.

2. To develop a fish IBI for the prairie rivers, we recommend the careful selection of least impacted reference sites and most impacted probable impaired sites. A minimum of 15-20 reference and impaired sites, respectively, would have sufficient power to test candidate metrics. The reference sites should be among the least impacted sites in the state, and as far away as possible from roads and

bridges. Probable-impaired sites should include some of the most-impacted river sites in Oklahoma, including stressors such as habitat impairment, channelization, toxic sediments, discharges, urban/industrial runoff, or heavy agricultural and livestock runoff.

3. Given the difficulties of developing a fish IBI in rivers of western arid lands, which have a relatively depauperate fish fauna, the addition of invertebrates as an indicator assemblage may be warranted.

Acknowledgments

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Literature Cited

Conquest, L.L., S.C. Ralph, R.J. Naiman. 1994. Implementation of large-scale stream monitoring efforts: Sampling design and data analysis issues. *In*: S.L. Loeb and A. Spacie, eds., Biological Monitoring of Aquatic Systems. Lewis Publishers, Boca Raton, FL.

Fausch, K.D., J.R. Karr, and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Trans. Am. Fish. Soc. 113:39-55.

Gallant, A.L., T.R. Whittier, D.P. Larsen, J.M. Omernik, and R.M. Hughes. 1989. Regionalization as a Tool for Managing Environmental Resources. EPA-600-3-89-060. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

Gerritsen, J. 1995. Additive biological indices for resource management. J.N. Am. Benthol. Soc. 14:451-457.

Gibson, G.R., M.T. Barbour, J.B. Stribling, J. Gerritsen, and J.R. Karr. 1996. Biological Criteria and Technical Guidance for Streams and Small Rivers. EPA 822-B-96-001. USEPA, Office of Science and Technology, Health and Ecological Criteria Division, Washington, DC.

Halliwell, D.B., R.W. Langdon, R.A. Daniels, J.P. Kurtenbach, and R.A. Jacobson. 2000. Classification of freshwater fish species of the northeastern United States for use in the development of indices of biological integrity, with regional applications. Pages 301-337 *In* T.P. Simon (editor). *Assessing the sustainability and biological integrity of water resources using fish communities.* CRC Press, Boca Raton, Florida.

Hite, R.L. and B.A. Bertrand. 1989. Biological Stream Characterization (BSC): A Biological Assessment of Illinois Stream Quality. IEPA/WPC/89-275. Special Report Number 13, Illinois Environmental Protection Agency, Illinois State Water Plan Task Force, Springfield.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: a method for assessing stream potential. Environ. Manage. 10:629-635.

Hughes, R.M. and D.P. Larsen. 1988. Ecoregions: an approach to surface

water protection. J. Wat. Poll. Control Fed. 60:486-493.

Karr, J.R. 1991. Biological Integrity: A long neglected aspect of water resource management. Ecol. Appl. 1:66-84.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspective on water quality goals. Environ. Manage. 5:55-68.

Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. Assessing Biological Integrity in Running Waters: A method and its rationale. Illinois Natural History Survey, Spec. Publ. 5.

Kenkel, N.C., and L. Orloci. 1986. Applying metric and non-metric multidimensional scaling to ecological studies: some new results. Ecology 67:919-928.

Larsen, D.P., J.M. Omernik, R.M. Hughes, C.M. Rohm, T.R. Whittier, A.J. Kinney, A.L. Gallant, and D.R. Dudley. 1986. The correspondence between spatial patterns in fish assemblages in Ohio streams and aquatic ecoregions. Environ. Manage. 10:815-828.

Lee, D.S., C.R. Gilbert, C.H. Hocutt, R.E. Jenkins, D.E. McAllister and J.R. Stauffer, Jr. 1980. Atlas of North American Freshwater Fishes. North Carolina State Museum of Natural History, Raleigh, North Carolina.

Ludwig, J.A., and J.F. Reynolds. 1988. Statistical Ecology. John Wiley and Sons, Inc., New York, New York.

Lyons, J. 1989. Correspondence between the distribution of fish assemblages in Wisconsin streams and Omernik's ecoregions. Am. Midl. Nat. 122:163-182.

Lyons, J. 1992. Using the Index of Biotic Integrity (IBI) to Measure Environmental Quality in Warmwater Streams of Wisconsin. General Technical Report NC-149. North Central Forest Experiment Station, U.S. Department of Agriculture, St. Paul, Minnesota.

Miller, D.L., P.M. Leonard, R.M. Hughes, J.R. Karr, P.B. Moyle, L.H. Schrader, B.A. Thompson, R.A. Daniels. 1988. Regional applications of an index of biotic integrity for use in water resource management. Fisheries 13:12-20.

Ohio EPA. 1987. Biological Criteria for the Protection of Aquatic Life. Volume II: Users Manual for Biological Field Assessment of Ohio Surface Waters. Ohio EPA, Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio.

Ohio EPA. 1989. Addendum to Biological Criteria for the Protection of Aquatic Life, Volume II: Users Manual for Biological Field Assessment of Ohio Surface Water. Ohio EPA, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, Ohio.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Ann. Assoc. Am. Geogr. 77:118-125.

Simon, T.P. 1991. Development of Ecoregion Expectations for the Index of Biotic Integrity. I. Central Corn Belt Plain. EPA 905-9-91-025. U.S. Environmental Protection Agency, Region 5, Chicago, Illinois.

Simon, T.P. 1992. Development of Biological Criteria for Large Rivers with an Emphasis on an Assessment of the White River Drainage, Indiana. EPA 905-R-92-026. U.S. Environmental Protection Agency, Region 5, Chicago, Illinois.

Simon, T.P. and J. Lyons. 1995. Application of the Index of Biotic Integrity to Evaluate Water Resource Integrity in Freshwater Ecosystems. pp. 245-262 *In* Biological Assessment and

Criteria: Tools for Water Resource Planning and Decision Making (W.S. Davis and T.P. Simon, eds.), Lewis Publishers, Boca Raton, Florida.

Simon, T.P. 2000. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, FL. 671 pp.

Snedecor, G.W., and W.G. Cochran. 1967. Statistical Methods. Iowa State University Press. Ames, Iowa.

SYSTAT. 1992. Statistics, Version 5.2 edition. SYSTAT, Inc., Evanston, Illinois.

USEPA. 1990. Biological Criteria: National Program Guidance for Surface Waters. EPA 440-5-90-004. U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Washington, DC.

Yoder, C.O. 1991. The integrated biosurvey as a tool for evaluation of aquatic life use attainment and impairment in Ohio Surface Waters. In Biological Criteria: Research and Regulation. EPA-440/5-91-005. Off. Water, US Environ. Prot. Agency, Washington, DC.

Table 1. Candidate fish metrics for Oklahoma rivers.

Metric	Expected Response to Stressors
Number of species per sample	decrease
Accumulated species per sampling period	decrease
Shannon species diversity	decrease
Shannon biomass diversity	decrease
Mean biomass per fish	decrease
Number of darter species	decrease
Number of sunfish species	decrease
Number of sucker species	decrease
Intolerant species	decrease
Percent green sunfish total sunfish	increase
Mean individuals per species	increase
Percent omnivorous cypernids	increase
Percent insectivorous cyprinids	decrease
Top carnivore species	decrease
Number of species making up 75% of individuals	decrease
Percent rough fish	increase

Table 2. Physical and biological characteristics of river types.

	Large Rivers	**Prairie Rivers**	**Saline Rivers**
Median flow (cfs)	7237.42	412.76	156.22
Number of sites	18	68	10
Number of annual observations	141	470	117
Medians of selected fish assemblage metrics			
Total taxa	13.5	12.5	8
Number darter species	0	0	0
Number sunfish species	2	2	1
Number sucker species	1	1	0
Number intolerant species	1.5	1.5	1
Percent tolerant individuals	1.8%	5.0%	1.4%
Percent omnivores	74.0%	43.0%	10.0%
Percent insectivorous cyprinids	0.33	0.65	0.37
Log total individual without tolerant species	2.74	2.95	2.86
Species diversity	1.90	1.64	1.74
Percent dominance	0.53	0.61	0.53

Table 3. Occurrence of fish species in Oklahoma rivers, example year 1994.

		Large Rivers		Prairie Rivers		Salinity Prairie Rivers	
Common Name	Species	%	N (sites)	%	N (sites)	%	N (sites)
Central stoneroller	*Campostoma anomalum*	<1	6	<1	7	<1	3
Blacktail shiner	*Cyprinella venusta*	<1	3	<1	2		
Red Shiner	*Cyprinella lutrensis*	14	14	52	48	6	6
Common carp	*Cyprinus carpio*	1	9	<1	18	<1	4
Cypress minnow	*Hybognathus hayi*					<1	1
Mississippi silvery minnow	*Hybognathus nuchalis*	<1	2				
Plains minnow	*Hybognathus placitus*	1	5	2	18	18	6
Redfin shiner	*Lythrurus umbratilis*	<1	1				
Speckled chub	*Macrhybopsis aestivalis*	<1	1	<1	10	<1	4
Silver chub	*Macrhybopsis storeiania*	<1	2	<1	2	<1	1
Golden shiner	*Notemigonus crysoleucas*	<1	5	<1	8	<1	1
Emerald shiner	*Notropis atherinoides*	8	12	3	20	5	3
Red River shiner	*Notropis bairdi*	<1	1	2	10	14	6
River shiner	*Notropis blennius*	1	5	<1	2		
Ghost shiner	*Notropis buchanani*	<1	4	<1	3	<1	1
Arkansas River shiner	*Notropis girardi*			2	5	<1	1
Chub shiner	*Notropis potteri*			<1	4	<1	2
Silverband shiner	*Notropis shumardi*	<1	1				
Sand shiner	*Notropis stramineus*	<1	5	4	28	13	4
Sand shiner X Red shiner	*N. Stramineus x C. lutrensis*	<1	1			<1	1
Mimic shiner	*Notropis volucellus*	<1	1	<1	1		
Suckermouth minnow	*Phenacobius mirabilis*	<1	3	<1	28	<1	3
Bluntnose minnow	*Pimephales notatus*	<1	3	<1	1		
Fathead minnow	*Pimephales promelas*	1	3	3	25	3	6
Bullhead minnow	*Pimephales vigilax*	4	13	8	43	<1	2

Table 3. (cont'd).

		Large Rivers		Prairie Rivers		Salinity Prairie Rivers	
Common Name	Species	%	N (sites)	%	N (sites)	%	N (sites)
River carpsucker	*Carpiodes carpio*	<1	10	<1	26	<1	1
Quillback	*Carpiodes cyprinus*	<1	1				
Smallmouth buffalo	*Ictiobus bubalus*	1	11	<1	14	<1	2
Bigmouth buffalo	*Ictiobus cyprinellus*	<1	6	<1	4		
Black buffalo	*Ictiobus niger*	<1	2				
Golden redhorse	*Moxostoma erythrurum*	<1	3				
Skipjack herring	*Alosa chrysochloris*	<1	4				
Gizzard shad	*Dorosoma cepedianum*	22	16	<1	20	<1	1
Threadfin shad	*Dorosoma petenense*	5	13	<1	4	<1	1
Black bullhead	*Ameiurus melas*			1	4		
Yellow bullhead	*Ameiurus natalis*	<1	1	<1	5		
Blue catfish	*Ictalurus furcatus*	<1	1	<1	4		
Channel catfish	*Ictalurus punctatus*	<1	9	<1	31		
Tadpole madtom	*Noturus gyrinus*	<1	1				
Freckled madtom	*Noturus nocturnus*			<1	3		
Flathead catfish	*Plyodictis olivaris*	<1	4	<1	7		
Redfin pickerel	*Esox americanus*	<1	1				
Paddlefish	*Polyodon spathula*	<1	2				
Spotted gar	*Lepisosteus oculatus*			<1	1		
Longnose gar	*Lepisosteus osseus*	<1	6	<1	12	<1	2
Shortnose gar	*Lepisosteus platostomus*			<1	6		
Goldeye	*Hiodon alosoides*			<1	1		
American eel	*Anguilla rostrata*	<1	1				

Table 4. Summary of biological metric inclusion from original IBI to candidate revised IBI.

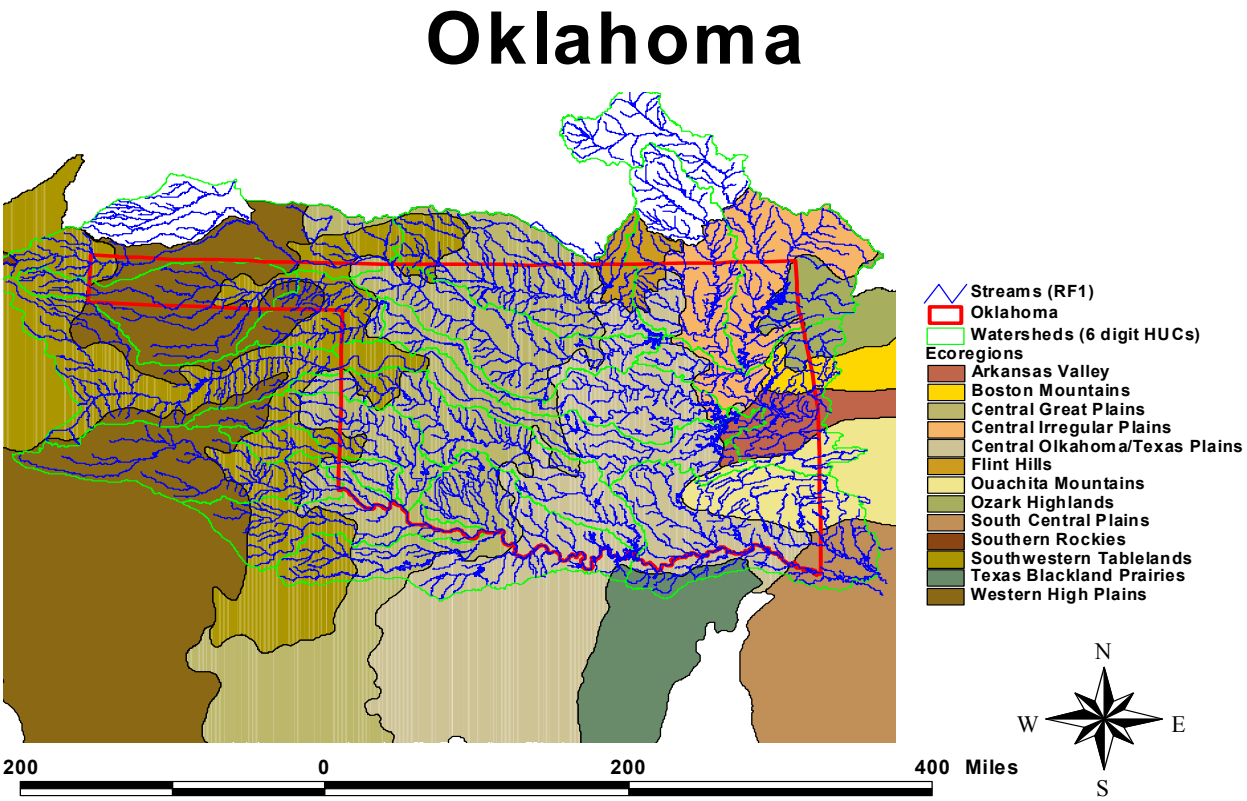
	Original OSDH/DEQ IBI Metrics	Candidate Metrics (based on selection criteria)	Core Metrics
1	Number of native species	Number of native species	X
2	Number of darter species	Number of darter species	X
3	Number of sunfish species	Number of sunfish species	X
4	Number of sucker species	Number of sucker species	X
5	Number of intolerant species	Number of intolerant species	X
6	Proportion of individuals as green sunfish	Proportion of individuals as tolerant species[a]	
7	Accumulated number of species per sampling period		
8	Numerical species diversity	Numerical species diversity	
9	Proportion of individuals as omnivores	Proportion of individuals as omnivores	
10	Proportion of individuals as insectivorous cyprinids	Proportion of individuals as insectivorous cyprinids	X
11	Proportion of individuals as top carnivores (piscivores)	Proportion of individuals as top carnivores (piscivores)	
12	Biomass species diversity	Total Biomass[a]	
13	Proportion of individuals to number of species	Number of individuals per sample[a]	X
14	Number of species that make up 75% of the population	Percent abundance of the predominant species[a]	
15	Proportion of biomass to number of fish		
16	Percentage of rough fishes		

[a] Metrics modified from original.

Table 5. Standard deviations of core fish metrics, Oklahoma rivers (from sites sampled at least four years and at least two determinations per year during the index period).

Metric	All Sites	Large Rivers	Prairie River	Saline River
Taxa	3.74	4.13	3.64	.21
Log individuals	0.474	0.515	0.457	.459
Darter species	0.385	0.522	0.358	.202
Sunfish species	0.502	0.518	0.516	.405
Sucker species	0.722	0.895	.694	.524
Intolerant species	0.756	0.903	.732	.581
Insectivorous cyprinids	0.232	0.263	.205	.239

a.



b.

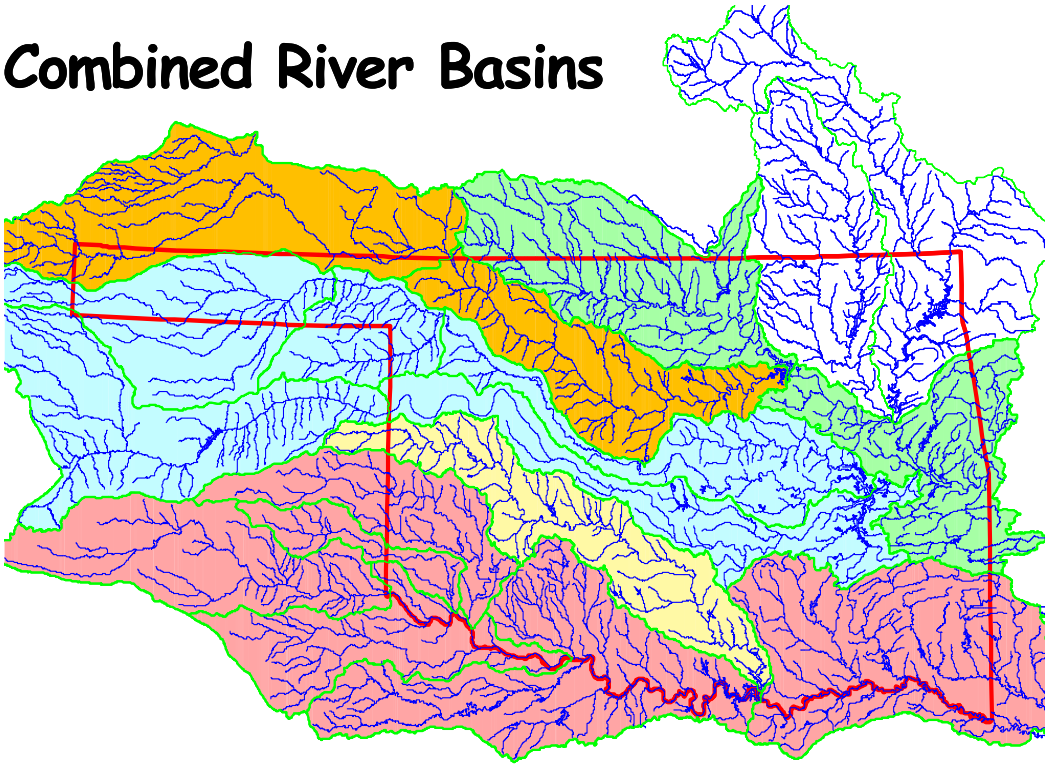
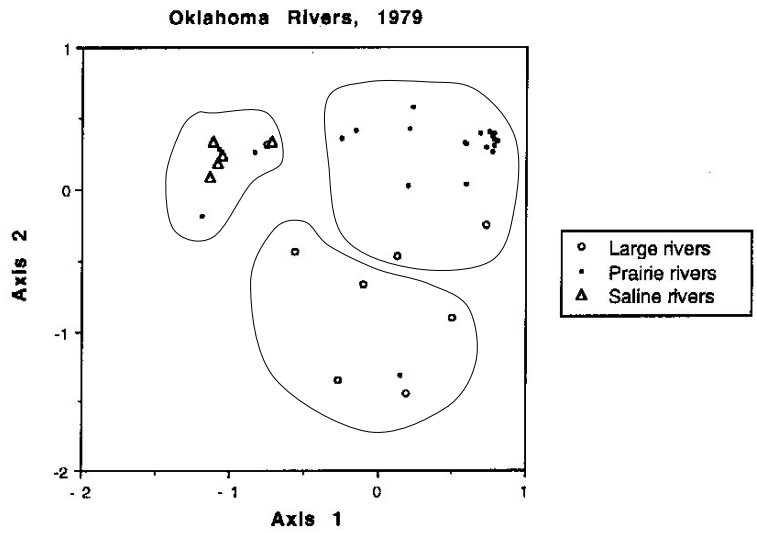
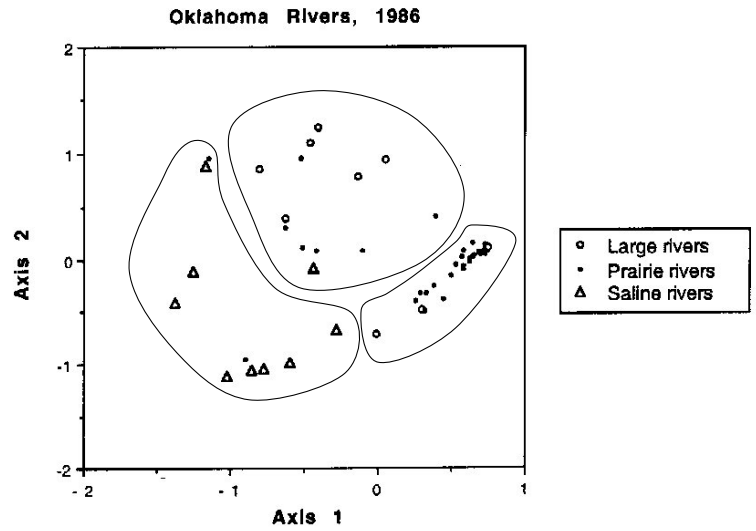


Figure 1. Physiographic delineation of Oklahoma, illustrating ecoregions (1a) and river basins (1b).

a.



b.



c.

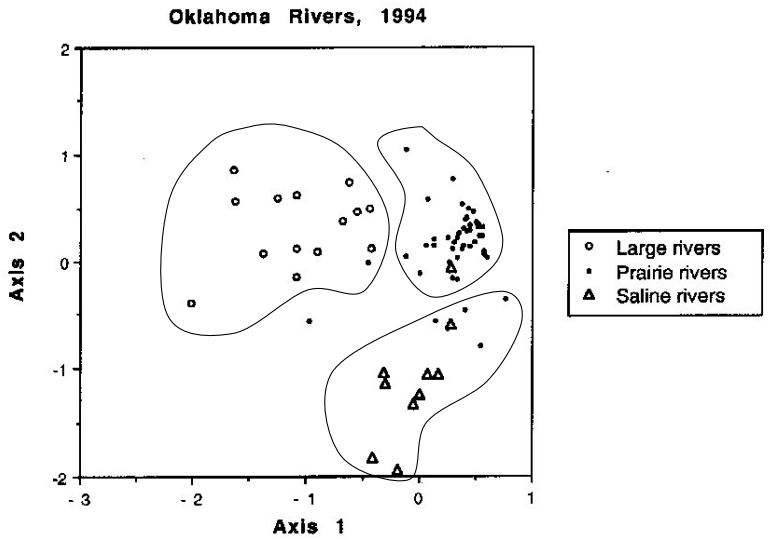


Figure 2. Non-metric multidimensional scaling (NMDS) ordination of river sites sampled in 1979 (2a), (2b), and (2c), showing first two axes (of 3).

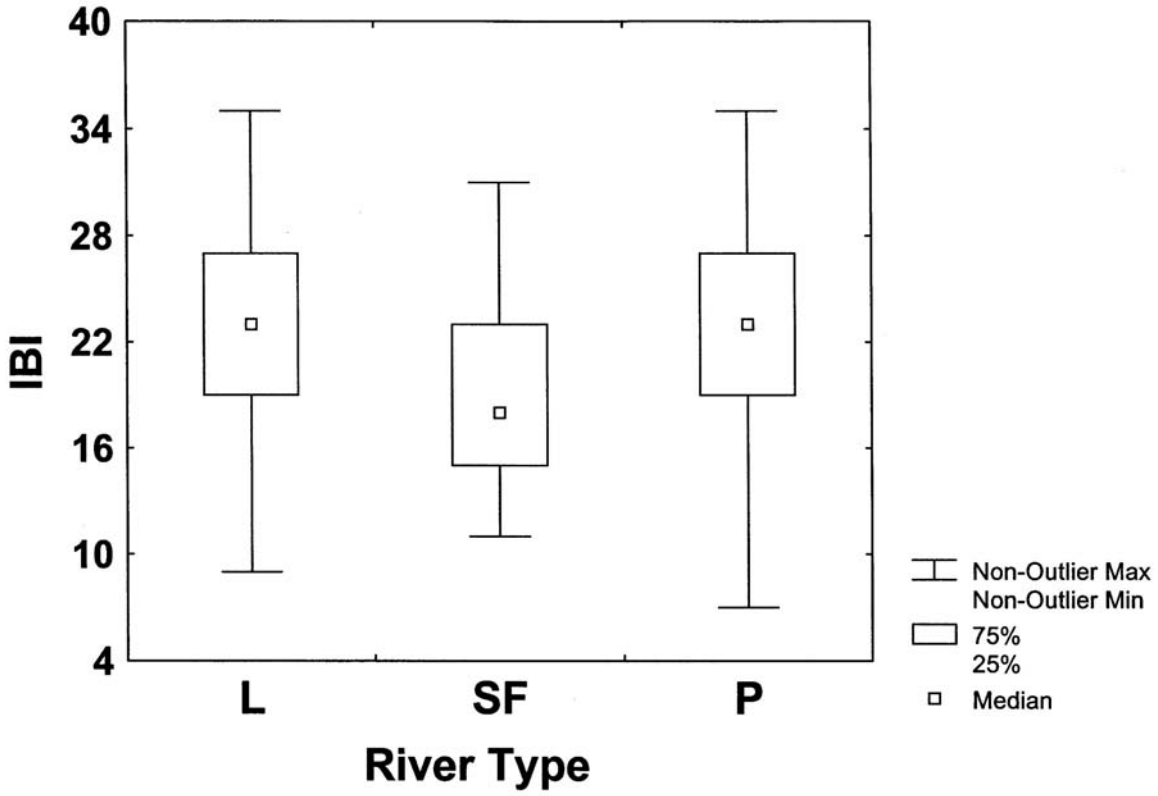


Figure 3. Comparison of IBI values for Oklahoma river types, 1976-1995. L = Large Rivers, SF = Saline Prairie, and P = Prairie.

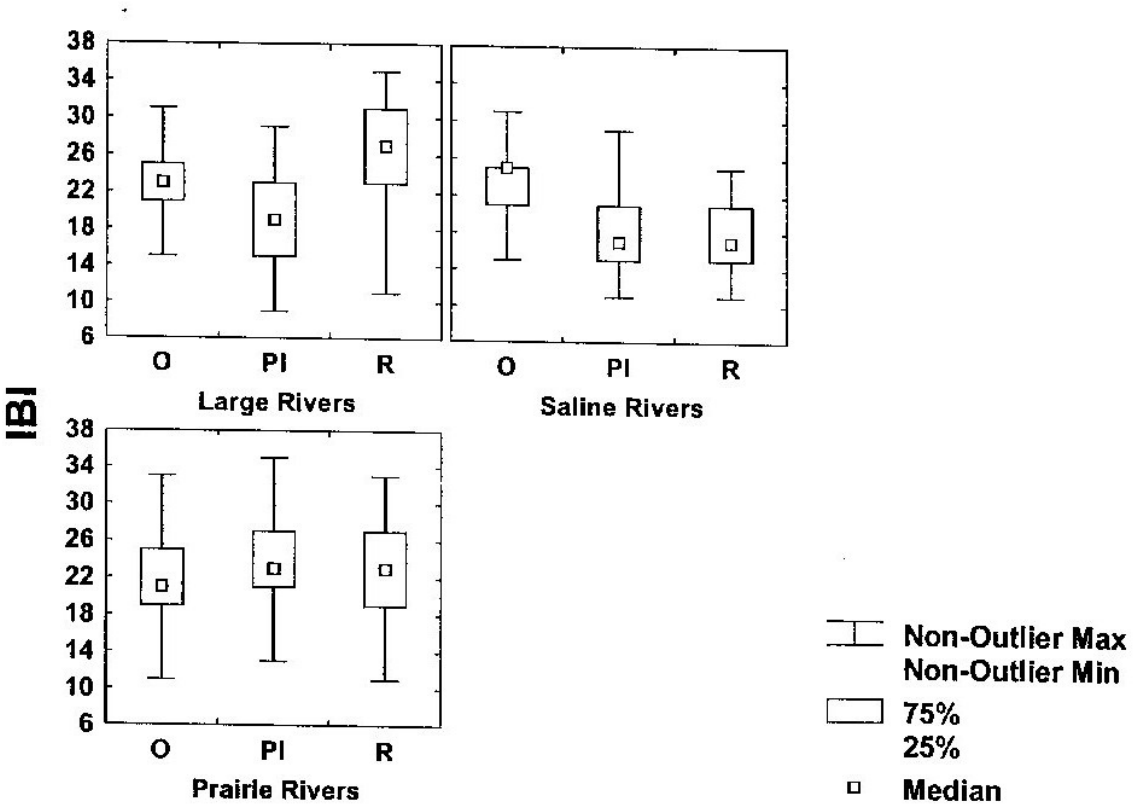


Figure 4. Comparison of IBI values between reference and probable impaired rivers, all sites, 1976-1995. R = *a priori* known reference, PI = probable impaired or stressed, and o = unknown.

ASSESSMENT OF ECOLOGICAL CONDITION IN HEADWATER STREAMS OF THE CENTRAL PLAINS: EVALUATION OF MULTIMETRIC AND PREDICTIVE MODELING APPROACHES

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Abstract. Multimetric analysis is the most common technique currently employed to assess the health of aquatic systems in North America. However, predictive modeling, a standard technique in the United Kingdom, has been tested in some regions of North America with promising results. Using a watershed assessment procedure to predict stream condition, the abilities of multimetric and predictive modeling techniques to distinguish reference from impacted conditions in headwater streams within the Western Corn Belt Plains ecoregion were compared. Eighteen reference sites were used to develop models with reference site replicate samples and samples from eight test sites used to evaluate the models. Predictive modeling techniques were better at assessing site condition and genus resolution models performed better than family models. Although predictive modeling performed better in this study, both approaches have shown they are useful tools in the biological assessment of streams and each has advantages that should be emphasized. For example, the basis for assessing impact in the multimetric approach has been better tested, but predictive modeling is a logistically more appealing approach that permits better standardization of techniques. Despite their differences, predictive modeling and multimetric approaches are based on similar theories and require similar data and therefore could be used in conjunction to provide additional assessment tools or a hybridization of the two methods could be used to develop better biological assessment techniques. The success of each approach depends on numerous factors including region and study purpose, which still requires a great deal of study to determine how these methods can best be applied.

Key words: multimetric, predictive modeling, Central Plains, Western Corn Belt Plains ecoregion, reference condition, biological assessment, Index of Biotic Integrity.

Introduction

Protection and remediation of streams is inherently dependent on the ability to assess the health of streams, but currently, the bewildering array of bioassessment methods complicate decisions (Davis 1995, Southerland and Stribling 1995). Multimetric and predictive modeling approaches are the two most commonly used biological assessment techniques, but which method provides the better determination of aquatic system health is unclear and the subject of debate. Multimetric techniques, developed and used in the United States, employ an array of metrics to quantify the health of several segments of the biotic community (Barbour et al. 1995). Predictive modeling, a multivariate technique developed in the United Kingdom, measures the divergence of test site fauna from a predicted reference community. To aid the development of biological assessment techniques in the Great Plains, this study compared these two popular biological assessment approaches and taxonomic resolutions (i.e., family and generic).

Methods

Eighty sub-watersheds in the Western Corn Belt Plains (WCBP) ecoregion were sampled on a seasonal basis (spring, summer, autumn, winter) beginning in the winter of 1992 and ending in the spring of 1994 (10 sampling events). Data collection consisted of water quality, habitat condition, biota, and land use/land cover. Macroinvertebrates were sampled using a one-minute, kick-net sample with three replicates per site.

Eighteen reference sub-watersheds were selected using an abiotic classification scheme, employing land use/land cover, in-/near-stream habitat, and water quality. Additionally, eight test sites were selected randomly from the remaining non-reference sub-watersheds. Macroinvertebrate data from autumn 1992 sampling events were used for development and testing of both multimetric and predictive model approaches.

Nine metrics from Donley (1999) were selected for this study to be used in an Invertebrate Community Index (ICI) (Tables 1 and 2). Methods for the ICI followed methods of Reynoldson et al. (1997) and produced an index with a potential range of values from 9 to 45, with higher numerical scores indicative of better community condition.

Predictive model calculations were derived from several predictive model versions described by Wright (1995), Reynoldson et al. (1997), Wright et al. (1998), and Simpson and Norris (2000), although some modifications were made to tailor the basic approach to small streams of the Great Plains and the limited data available in this study. A major modification was the use of an ecoregion classification in place of multivariate techniques.

To evaluate biological assessment approaches and taxonomic resolutions an arbitrary level of the 25th percentile was used to set the lower limit for reference sites. Sites scoring below the reference site 25th percentile were considered impacted and sites scoring at or above the reference site 25% percentile were considered reference (Figures 1 and 2). The accuracy of a model was expressed as the ratio of the number of samples identified correctly as reference or impacted based on the abiotic classification.

Results

Biological assessment models correctly identified eight to 17 of 22 of test site samples as impacted (Table 3). The multimetric approach correctly identified eight (36%) and 10 (45%) of 22 samples as impacted for family and generic resolutions, respectively. The predictive modeling approach using family resolution identified 11 of 22 (50%) test site samples correctly, and genus resolution identified 17 of 22 (77%) correctly. Nine to 14 of the 17 replicate reference sites were identified correctly as reference condition (Table 4). The multimetric approach using family resolution correctly identified nine (53%) samples, and the genus model assessed 13 (77%) samples correctly. Predictive modeling techniques using family and generic resolution correctly assessed 13 (77%) and 14 (82%) samples, respectively.

Discussion

Tristate test and reference sub-watersheds were most accurately and precisely identified with predictive models using generic resolution. These results indicate that predictive models may be the most accurate method to detect impact in the Great Plains when compared to multimetric methods. Despite the better performance of predictive modeling compared to the multimetric approach, both have advantages

and disadvantages which must be examined in order to select appropriate bioassessment techniques. Table 5 outlines factors that differ between predictive modeling and multimetric approaches, but discussion will focus on three major differences between these approaches.

Development of Reference Condition

Both multimetric and predictive modeling approaches rely on professional judgment and existing information to select reference sites, but once these sites are selected, methods between these approaches vary considerably. Predictive models typically use a set of multivariate techniques to select reference stream groupings and to match test sites to reference groups, which does not vary greatly among regions or waterbody types or sizes. These standardized techniques reduce effort and allow the development of models without relying on human decisions to remove natural effects on the biota (Boulton 1999). The development of reference criteria is the most complicated and difficult part of the implementation of a multimetric approach (Simon and Lyons 1995) and is contingent on experienced personnel able to make informed decisions where standard multimetric methods are ambiguous. The selection of appropriate metrics and the calibration of scoring criteria in multimetric programs can also be complicated and time consuming as many of these steps are based on arbitrary decisions and must be calibrated for ecoregion and stream size.

Basis for Assessing Impact.—The methods used to assess impact in multimetric and predictive models are different and are often a source of contention between biological assessment investigators. Predictive modeling was originally developed to assess biodiversity for conservation purposes and therefore relies on the composition of specific taxa. In contrast, multimetric approaches are designed to assess the structure and function of stream communities without relying on specific taxa. The use of the structure and function of biotic communities as a measure of health has several advantages: the structure and function of stream communities are well established measures of stream quality; structure and function measures the fundamental biological integrity of a stream; rare taxa can be included in analyses, and natural shifts in taxonomic composition do not affect measures because specific taxa are not used. In contrast, taxonomic composition, as used in predictive models, may not be truly indicative of stress and/or may be affected by natural factors. The appropriateness of the use of predictive modeling to assess the biological integrity is less well understood and may not necessarily correlate to stream health. However, the results of this study suggest that predictive modeling can successfully assess biological integrity.

Approach Calculation.— Both methods use the reference condition approach and require similar data permitting easy comparison or the use of both approaches together to increase analytical tools available to investigators. Particular advantages of each model could also be selected to improve assessment capabilities. For example, predictive modeling is more easily standardized across regions, so the development of reference criteria could employ predictive modeling techniques and then used to calculate both predictive modeling and multimetric measures. These methods would remove the need to base reference groups on ecoregional or watershed groupings and the need to manually scale metric criteria to stream size which may be more prone to error.

The intellectual and time commitments associated with learning how to implement

predictive models can be daunting, but are simple to duplicate once the procedures are learned and/or appropriate software is used to perform the computations. Most established predictive modeling programs have developed software that greatly simplifies the calculation of predictive models and allows users to bypass learning complicated multivariate procedures. In contrast, widely available platform for the calculation of multimetric approaches has not been developed (Boulton 1999). It is hoped that software will be developed for biological assessment approaches in the Great Plains in order to simplify the construction of initially complex models by allowing users to input data and retrieve model outputs for both multimetric and predictive models. Widespread computer software will improve accessibility of biological criteria to agencies unable to collect the huge amount of data needed to develop such programs. An indirect effect of widespread computer software and online programs is their prevalence may create increased field and laboratory standardization as standardized data will be needed to utilize these programs.

Summary

The current study was a baseline analysis of the development of appropriate biological assessment techniques for Great Plains headwater streams. Although future analyses will likely employ a combination of multimetric and predictive modeling approaches, there is presently a need to test available techniques. In this study, predictive models better identified anthropogenic impact in the headwater streams of the WCBP compared to multimetric approaches. However, both approaches have been shown to be effective measures of biotic health with differing strengths and weaknesses. Because data requirements for both techniques are similar, these approaches should be used together in biological assessment programs to test the applicability and accuracy of each technique and to improve the capabilities of biological assessment programs. Future programs could employ predictive models as a platform to incorporate both approaches into a more easily developed and comprehensive biological assessment tool as well as the use of computer programs to improve standardization and calculation of approaches. There is no biological assessment "silver bullet" for streams and rivers, but methods that best serve a region's needs, study objectives, and agency goals and can most accurately and precisely identify degradation of streams and rivers need to be identified.

Literature Cited

Barbour, M.T., J.B. Stribling and J.R. Karr. 1995. Multimetric approach for establishing biocriteria and measuring biological condition. pp. 63-77. *In:* W.S. Davis and T.P. Simon (ed.), Biological assessment and criteria: tools for water resource planning and decision making, Lewis, Boca Raton, FL.

Boulton, A.J. 1999. An overview of river health assessment: philosophies, practice, problems and prognosis. Freshwater Biology 41: 469-479.

Davis, W.S. 1995. Biological assessment and criteria: building on the past. pp. 15-29. *In:* W.S. Davis and T.P. Simon (ed.) Biological assessment and criteria: tools for water resource planning and decision making, Lewis Publishers, Boca Raton, FL.

Donley, S.R. 1999. Development of biological criteria for macroinvertebrates for streams of the Western Corn Belt Plains ecoregion. M.S. Thesis, Ecology and Evolutionary

Biology, University of Kansas, Lawrence, KS. 82 pp.
Reynoldson, T.B., R.H. Norris, V.H. Resh, K.E. Day and D.M. Rosenberg. 1997. The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. Journal of the North American Benthological Society 16: 833-852.
Simon, T.P. and J. Lyons. 1995. Application of the Index of Biotic Integrity to evaluate water resource integrity in freshwater ecosystems. pp. 245-262. *In:* W.S. Davis and T.P. Simon (ed.), Biological assessment and criteria: tools for water resource planning and decision making. Lewis, Boca Raton, FL.
Simpson, J.C. and R.H. Norris. 2000. Biological assessment of river quality: development of AUSRIVAS models and outputs. pp. 125-142. *In:* J.F. Wright, D.W. Sutcliffe and M.T. Furse (ed.), Assessing the biological quality of fresh waters, Freshwater Biological Association, Ambleside, Cumbria, UK.
Southerland, M.T. and J.B. Stribling. 1995. Status of biological criteria development and implementation. pp. 81-96. *In:* W.S. Davis and T.P. Simon (ed.), Biological assessment and criteria: tools for water resource planning and decision making, Lewis Publishers, Boca Raton, FL.
Wright, J.F. 1995. Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. Australian Journal of Ecology 20: 181-197.
Wright, J.F., M.T. Furse and D. Moss. 1998. River classification using invertebrates: RIVPACS applications. Aquatic Conservation: Marine and Freshwater Ecosystems 8: 617-631.

Table 1. Selected macroinvertebrate biological criteria metrics used in the WCBP ecoregion Invertebrate Community Index (ICI), their expected response to stress, and the scoring criteria used for family resolution tests.

Metric	**Response**	**Family Scoring**		
Richness		**1**	**3**	**5**
Taxa Richness	Decrease	<7	7-8	>8
Mayfly Richness	Decrease	<1	1	>1
EPT Richness	Decrease	<2	-	>2
Diptera Richness	Decrease	0	-	>0
Sensitive Taxa Richness	Decrease	<4	4	>4
Abundance				
Abundance	Variable	<68	68-139	>139
EPT Abundance	Decrease	<10	10-14	>14
Diversity				
Margalef	Decrease	<1.26	1.26-1.82	>1.82
Shannon	Decrease	<0.95	0.95-1.26	>1.26

Table 2. Selected macroinvertebrate biological criteria metrics used in the WCBP ecoregion Invertebrate Community Index (ICI), their expected response to stress, and the scoring criteria used for genus resolution tests.

Metric	**Response**	**Genus Scoring**		
Richness		**1**	**3**	**5**
Taxa Richness	Decrease	<11	11-16	>16
Mayfly Richness	Decrease	<2	2	>2
EPT Richness	Decrease	>2	-	<2
Diptera Richness	Decrease	<2	2-5	>5
Sensitive Taxa Richness	Decrease	<6	6-7	>7
Abundance				
Abundance	Variable	<68	68-131	>131
EPT Abundance	Decrease	<10	10-13	>13
Diversity				
Margalef	Decrease	<1.86	1.87-3.22	<1.87
Shannon	Decrease	<1.14	1.14-1.70	>1.70

Table 3. Classification accuracy of multimetric and predictive models for macroinvertebrate family and genus level resolutions from autumn 1992 test (i.e., impacted) site samples [R = reference (incorrect assessment); I = impacted (correct assessment)].

Taxonomy	**Family**		**Genus**	
Approach	**Multimetric**	**Predictive**	**Multimetric**	**Predictive**
Site (Replicate)				
Wolf 4(1)	R	R	R	R
Silver 2(1)	R	R	R	R
Silver 2(2)	R	R	R	R
Silver 2(3)	R	R	R	R
Bear 5(1)	R	I	I	I
Bear 5(2)	R	I	R	I
Bear 5(3)	R	R	R	R
Beemis 2(1)	I	I	I	I
Beemis 2(2)	R	R	R	I
Beemis 2(3)	R	R	I	I
Four Mile 5(1)	R	R	R	I
Four Mile 5(2)	I	I	I	I
Four Mile 5(3)	I	I	R	I
Powell 4(1)	R	R	R	I
Powell 4(2)	I	I	I	I
Powell 4(3)	I	I	I	I
Seven Mile 5(1)	R	I	I	I
Seven Mile 5(2)	R	I	I	I
Seven Mile 5(3)	I	I	I	I
Walnut-J 3(1)	R	R	R	I
Walnut-J 3(2)	I	R	R	I
Walnut-J 3(3)	I	I	I	I
Accuracy	**8 of 22**	**11 of 22**	**10 of 22**	**17 of 22**
% Accuracy	**36%**	**50%**	**45%**	**77%**

Table 4. Classification accuracy for multimetric and predictive models using family and genus resolutions macroinvertebrate samples from autumn 1992 for reference sites [R = reference (correct assessment); I = impacted (incorrect assessment].

	Family		**Genus**	
Site (Replicate)	**Multimetric**	**Predictive**	**Multimetric**	**Predictive**
Straight 1(2)	R	R	R	R
Straight 3(2)	I	I	I	I
Straight 4(2)	I	R	R	R
Straight 5(2)	I	I	I	I
Straight 5(3)	I	I	I	I
French 3(2)	I	I	R	R
French 4(2)	I	R	R	R
French 5(2)	I	R	R	R
French 5(3)	I	R	I	R
Buck 1(2)	R	R	R	R
Buck 1(3)	R	R	R	R
Rock 5(2)	R	R	R	R
Rock 5(3)	R	R	R	R
Three Mile 4(2)	R	R	R	R
Three Mile 4(3)	R	R	R	R
Walnut-S 4(2)	R	R	R	R
Walnut-S 4(3)	R	R	R	R
Accuracy	**9 of 17**	**13 of 17**	**13 of 17**	**14 of 17**
% Accuracy	**53%**	**77%**	**77%**	**82%**

Table 5. Comparison of predictive modeling and multimetric attributes important in this study [from Karr and Chu (2000), Norris and Hawkins (2000)].

	Predictive Modeling	**Multimetric**
Required Dataset	Large	Smaller than predictive modeling, but still considerable
Sampling Period	Multiple or single season	Defined sampling period
Faunal Groups	Macroinvertebrates (although other groups should be applicable)	Macroinvertebrates, fishes, algae
Reference Sites	Site without serious human disturbance	Site with little or no human disturbance
Development of Reference Criteria	Standardized	Can be complicated and variable
Site Classification	Suite of environmental factors (stream size, geography, water chemistry, substrate, date) minimally influenced by human activities	Ecoregion (geology, land use, stream size, temperature, altitude)
Approach Calculation	Complicated and difficult initially (multivariate techniques)	Straightforward, but no widespread computer platform available
Rare Taxa	Excluded	Included
Basis for Assessing Impact	Presence/absence of taxa	Structure and function of community
Communication	Concept easy to explain	Concept less clear

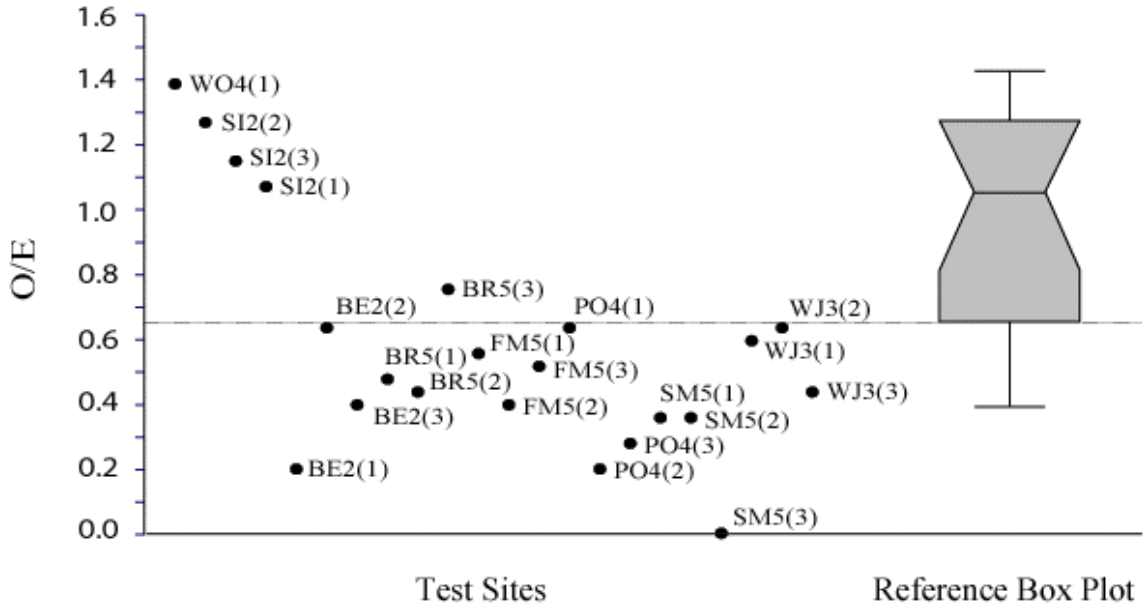

Figure 1: Example of box plot comparison for determining assessment capabilities of predictive model using independent data using genus resolution test site data from autumn 2001 (- - - = reference site 25th percentile).

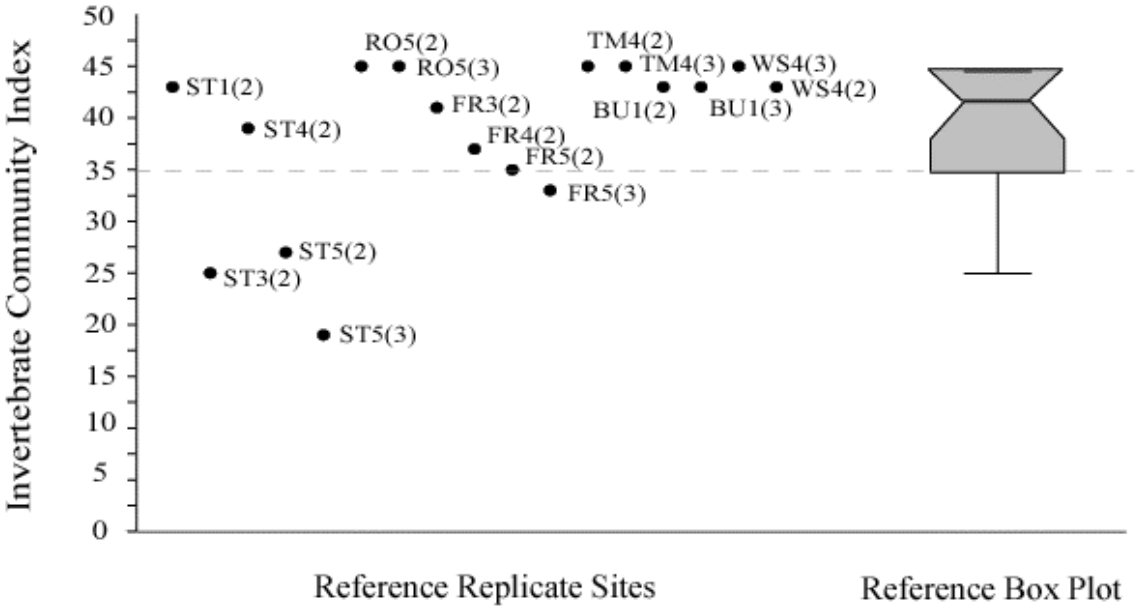

Figure 2: Example of box plot comparison for determining assessment capabilities of multimetric techniques using genus resolution replicate reference site data from autumn 1992. (- - - = reference site 25th percentile).

BIOLOGICAL ASSESSMENT OF SOUTH LOGAN CREEK: A SERVICE LEARNING PROJECT IN NORTHEASTERN NEBRASKA

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Abstract. A Service Learning Biological Assessment was performed for South Logan Creek in Wayne, Nebraska. Students assessed the possible impact and impairment of the Wayne Waste Water Treatment Plant effluent on South Logan Creek. All community measures were nearly identical between the reference site and the effluent site, indicating no impact or impairment from the effluent. A comparison between the Taxonomic Richness and Shannon-Weaver Diversity of South Logan Creek and the Reference Condition for the Western Cornbelt Plains Ecoregion indicates that, although suffering from considerable habitat degradation and possible non-point source pollution, the macroinvertebrates of South Logan Creek are relatively unperturbed.

Key words: volunteer stream monitoring, biological assessment, northeast Nebraska.

Introduction

The Waste Water Treatment Plant (WWTP) in Wayne, Nebraska received complaints from nearby residents, during the Summer of 2001, about high numbers of non-biting midges or Chironomidae (Diptera) believed to originate from the WWTP. High numbers of emerging Chironomidae have been associated with the organic discharge from wastewater treatment plants (Wiederholm 1984, Ferrington 1990). Many Chironomidae possess hemoglobin as larvae and are commonly called blood worms. The hemoglobin in blood worms allows them to extract oxygen more efficiently from water, thereby allowing them to live in habitats with low concentrations of dissolved oxygen (DO) (Armitage et al. 1994). Increased temperature and organics resulting from discharges from wastewater treatment plants lead to a decrease in levels of dissolved oxygen. Many macroinvertebrates in the receiving stream or lake can no longer live in the low dissolved oxygen habitat and blood worms come to be the dominant organisms. Due to lack of competition their numbers increase until they are very abundant and produce large emergences of the adult flies.

A Service Learning project was initiated for the Advanced Ecology Class at Wayne State College. Service Learning is a combination of students serving the community and learning from the service through critical thought and reflection of the service experience (Service Learning 2000). The goals of this project were: 1) to determine whether waste effluent from the Wayne WWTP impacted or impaired South Logan Creek; 2) to determine whether the Wayne WWTP effluent caused an increase in chironomids in South Logan Creek; 3) to engage the students in local ecosystem studies; 4) to teach students the value of stream ecosystems; and 5) to serve the local community by providing the community with stream biological monitoring data.

Methods and Materials

Study Site

The Wayne WWTP is located on South Logan Creek bordering the south side of Wayne, Nebraska. South Logan Creek is a third order stream and is formed from the

confluence of South Logan Creek and Deer Creek. Where the stream runs through Wayne, it is a perennial spring fed creek. The South Logan Creek Watershed covers approximately 190 square miles in Northeastern Nebraska and is located in the northwestern margin of the Western Cornbelt Plains Ecoregion (WCBP) as defined by Omernik (1987).

Two sites were used for collection of macroinvertebrates and for measurement of physical, chemical, and habitat parameters. Site 1 was located approximately 50 m upstream of the Wayne WWTP effluent discharge and was designated the reference site and represented the unperturbed nature of South Logan Creek, relative to Site 2. Site 2 was located right at or below the WWTP effluent discharge and represented the site of greatest possible impact and impairment from the plant. Both sites were studied for 25 meter reaches each and were characterized by a deeply cut, U-shaped, channelized stream channel. Banks were approximately 7-15 meters deep and were very steep, showing a high degree of erosion. Row crop agriculture was planted up to the south bank of South Logan Creek and residences and some trees lined the north bank. Little or no riparian zone or buffer strip was present.

Both Site 1 and Site 2 had a riffle, run, and pool sequence. The stream flow was characterized by considerable sediment deposition. There were few vascular plants and algae were the predominant flora in the stream. Sediment was composed primarily of silt and mud with some concrete debris. Mud concretions formed in the stream and acted as large cobble and rocks and were colonized by macroinvertebrates.

Average stream width and depth were nearly identical at both Site 1 and Site 2 (Table 1). Average velocity of the runs in Site 1 and Site 2 were identical but average velocity was slightly greater in the Site 2 riffle than the Site 1 riffle (Table 1). The very similar physical structure of Sites 1 and 2 allowed for direct comparison of the macroinvertebrate communities.

Methods and Materials

Physical, chemical, and habitat parameters were measured at every site using methods outlined in the U.S. EPA Rapid Bioassessment Protocol (Barbour et al. 1999). Water temperature, Conductivity, Dissolved Oxygen, and Turbidity were measured *in situ* using a YSI® portable probe. Macroinvertebrates were collected using semi-quantitative methods outlined in the U.S. EPA Rapid Bioassessment Protocol (Barbour et al. 1999).

Taxa were sorted, enumerated, and identified in the laboratory. Chironomidae were subsampled using the following methods. Specimens were placed in a petri dish that was divided into 12 equal parts and the specimens were swirled and let to settle. One of the 12 divisions was randomly selected and all the chironomids were taken from that division for identification and enumeration. This process was repeated until 100 specimens were subsampled. If a sample had less than 100 specimens it was identified in its entirety. Identifications were made using Pennak (1983), Merritt and Cummins (2000), and Wiederholm (1983). All samples that were sorted and identified were verified by the author and instructor of the class.

Descriptive statistics, total abundance, Shannon-Weaver Diversity Index, and the Hilsenhoff Biotic Index (HBI) (Hilsenhoff 1986) were calculated using MS Excel®. The HBI tolerance values were calculated using Huggins and Moffett (1988) tolerances for organic pollution.

Results

South Logan Creek is a highly perturbed stream as it flows into the City of Wayne, Nebraska. It is channelized and shows considerable sedimentation, resulting in high turbidity ranging from 71% in Site 2 to 76% in Site 1 (Table 1). The temperature in the creek was fairly high during the evening and morning when the collections were made (Table 1). Dissolved oxygen at both Site 1 and Site 2 was very low, but not low enough to be lethal to the macroinvertebrates. Conductivity readings were high in both sites (Table 1).

A total of 1510 macroinvertebrates were collected, enumerated, and identified for this study. Altogether, 52 different taxa of macorinvertebrates were identified (Table 2).

Chironomidae had the greatest number of individuals at the family level and *Baetis* had the greatest number of individuals at the genus level. The greatest number of genera was found in the chironomid family.

The total number and average number of macroinvertebrates was higher at Site 1 than at Site 2, however the number of different taxa or the taxonomic richness was higher at Site 2 than at Site 1 (Table 3). Standard deviations for average numbers of macroinvertebrates from both sites were very high indicating considerable variability in both samples. The maximum number of specimens for a particular taxon was 223 individuals of *Baetis* in Site 1 and 119 individuals of *Physa* in Site 2. Several taxa were represented by only one individual in both Sites 1 and 2 (Table 3).

Index values were virtually identical at both sites (Table 3). In fact the Shannon-Weaver Diversity Index was identical at both sites. The HBI was very similar. The V-evenness was higher at Site 2 than at Site 1 indicating a greater degree of unequal distribution of diversity at Site 2 over Site 1 (Table 3).

Discussion

Even though there were some minor differences, dissolved oxygen, turbidity, conductivity, and water temperature were very similar between Site 1 and Site 2, indicating little to no impact on South Logan Creek from the WWTP effluent discharge (Table 1). In addition, most metric and index values used to compare Site 1 and Site 2 were very similar (Table 3). These similarities indicate that there is very little or no impact on South Logan Creek from the effluent of the Wayne Waste Water Treatment Plant. Furthermore, there is little or no impairment of South Logan Creek from the effluent. If anything, the effluent may actually cause an increase in diversity of the taxa.

Typically below a WWTP there will be a decrease in macroinvertebrate diversity and abundance with a corresponding increase in the abundance of only a few species of blood worms, chironomids that possess hemoglobin in the subfamily Chironomini (Ferrington 1990). Blood worms or Chironomini did not predominate either above or below the Wayne WWTP effluent in South Logan Creek. Instead, the subfamily Orthocladiinae, which are not blood worms, predominated. Collections in South Logan Creek were made in late summer of 2001. Typically, Chironomini would predominate in a warm water stream at this time of the year. Although they were not few in number, they certainly did not dominate diversity in South Logan Creek.

Interestingly, Taxonomic Richness and Shannon-Weaver Diversity of the South Logan Creek macroinvertebrate community are within or above the range of the Reference Condition for the WCBP ecoregion (Donley 1999) (Table 4). This holds true when assessing the chironomid community separately, as well (Table 4). Considering the habitat degradation and possible enrichment of Nitrogen and

Phosphorus and possible elevated levels of pesticides, this result is amazing. It is possible that springs dilute any possible non-point source pollution in the area and the naturally high levels of calcium carbonate recorded in the area buffer the macroinvertebrate populations. Future studies will be directed at investigating this phenomenon.

As a Service Learning project, this biological assessment was an incredible success. Students learned many of the methods associated with stream ecology, they answered a question for the City of Wayne and thus felt that their work was of service. City administrators have thanked them publicly for the service they have performed. Some of the students have decided to continue on in aquatic ecology and biological monitoring. Future plans for stream biological monitoring in the region include more Service Learning activities and community based Volunteer Stream Monitoring.

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Literature Cited

Armitage, P., P. Cranston, and L. Pinder. 1994. Biology and ecology of non-biting Chironomidae. 572 pp.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. U.S. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Donley, S. 1999. Development of Biological Criteria for Macroinvertebrates for Streams of the Western Corn Belt Plains Ecoregion. Master's Thesis. University of Kansas, Lawrence, KS.82pp.

Ferrington, L.C., Jr. 1990. Assessment of the longitudinal distribution, population densities and life cycle dynamics of *Chironomus riparius* in Indian Creek near the Middle Basin plant. Rep. No. 43. Kansas Biol. Surv., Lawrence, KS. 56 pp

Hilsenhoff, W. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society. 7(1):65-68.

Huggins, D.G. and M.F. Moffett. 1987. Proposed biotic and habitat indices for use in Kansas streams. Rep. No. 35. Kansas Biological Survey, Lawrence, KS. 128 pp + appendices

Merritt. R.W. and K.W. Cummins. 1996. An Introduction to the Aquatic Insects of North America. Kendall/Hunt. Dubuque, IA. 862 pp.

The Nebraska Consortium for Service Learning in Higher Education. 2000. http://www.unl.edu/ncslhe/service-learning.htm

Omernik, J. 1987. Ecoregions of the conterminous United States. Annals of

the Association of American Geographers. 77: 118-125.
Pennak, R. 1989. Fresh-Water Invertebrates of the United States: Protozoa to Mollusca. John Wiley and Sons. New York. 628pp.
U.S. EPA. Office of Water. 1998. National Strategy for the Development for Regional Nutrient Criteria. U.S. EPA 822-R-98-002.
Wiederholm, T. (Ed.). 1983. Chironomidae of the holarctic region. Keys and diagnoses. Part 1. Larvae. Entomologica Scandinavica Supplement No. 19. 457 pp.
Wiederholm, T. 1984. Responses of aquatic insects to environmental pollution. Pages 508-557 *In* Resh, V. and D. Rosenberg (eds) The Ecology of Aquatic Insects. Praeger, New York.

Table 1. Physical and Chemical Conditions of South Logan Creek.

Parameter	Site 1	Site 2
Stream Reach	25 m	25 m
Steam Width	6.2 m	5.5 m
Average Depth	24 cm	23.7 cm
Average Velocity/Riffle	0.5 m/sec	0.75 m/sec
Average Velocity/Run	0.33	0.33 m/sec
Water Temperature	22 C	18.7 C
Conductivity	930 uS	735 uS
Dissolved Oxygen	6.7 mg/L	6.6 mg/L
Turbidity	76%	70.70%

Table 2. Taxa identified from South Logan Creek, Wayne, Nebraska.

Class	Order	Family	Genus and species
Insecta	Odonata	Coenagrionidae	Genus
			Enallagma
			Chromagrion
		Calopterygidae	*Heterina*
			Calopteryx
	Ephemeroptera		Genus
		Baetidae	Genus
			Baetis
		Caenidae	*Caenis*
		Heptageniidae	
		Trichorythidae	*Trichorythodes*
	Hemiptera	Veliidae	*Microvelia*
	Trichoptera	Hydroptilidae	*Hydroptila*
			Stactobiella
		Hydropsychidae	Genus
			Parapsyche
			Cheumatopsyche
			Hydropsyche
			Ceratopsyche
	Lepidoptera	Arctiidae	*Estigmene*
		Pyralidae	
	Coleoptera	Elmidae	*Stenelmis*
			Dubriaphria
		Dytiscidae	*Hydrovatus*
		Staphylinidae	*Stenus*
	Diptera	Chironomidae	*Conchapelopia*
			Cricotopus
			Eukiefferiella
			Nanocladius
			Paratrichocladius
			Rheocricotopus
			Thienemanniella
			Chironomus
			Cryptochironomus
			Dicrotendipes
			Polypedilum
			Paratanytarsus
			Tanytarsus
		Simuliidae	*Simulium*
		Tipulidae	
		Dolichopididae	
		Culicidae	
		Psychodidae	
		Ceratopogonidae	*Atrichopogon*
Crustacea	Amphipoda	Talitridae	*Hyalella azteca*
	Isopoda		
Gastropoda			Genus
		Physidae	*Physa*
		Planorbidae	*Radix auricularia*
		Lymnaeidae	
Pelycopoda			
Oligochaeta			

Table 3. Descriptive statistics and community measures for South Logan Creek.

	Site 1	Site 2
Total	992	518
Count	30	43
Average	33.07	12.05
Standard Deviation	49.56	25.19
Max	223	119
Min	1	1
Median	10.5	3
Shannon Weaver Div.	2.65	2.65
V evenness	0.38	0.42
HBI	2.88	2.84

Table 4. Macroinvertebrate and chironomid community metrics compared to WCBP Reference Condition.

	Richness	Diversity
South Logan Creek Total Taxa	52	2.65
WCBP Reference Condition Total Taxa	37.5 (25-51)	2.34 (1.69-2.76)
South Logan Creek Chironomids	13	1.87
WCBP Reference Chironomids	12 (7-15)	1.83 (1.13-2.21)

GYPSUM CREEK STREAM RESTORATION PROJECT

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Abstract. Gypsum Creek is a second-order urban stream in Wichita, KS that has been channelized in the past for flood control and has been selected as a sight for a stream restoration project. The initial phase of the project was to document the current conditions of the chemical, physical, and biological components of the stream and use it to compare to data that will be collected following restoration measures to determine if improvements to the stream's health and condition occur. Three separate 300m reaches have been chosen as collection sites along Gypsum Creek for sampling of water quality, pesticides, physical habitat, and benthic macroinvertebrate and fish communities. Data collected to date indicates anthropogenic stressors have degraded the ecological conditions at Gypsum Creek. Monitoring documented the presence of the herbicide simazine in water samples, as well as the insecticide diazinon at levels above recommended water quality criteria for protection of aquatic life (0.08µg/L). Planktonic chlorophyll *a* concentrations averaged more than 12 µg/L indicating eutrophic conditions exist in the stream. Physical habitat assessments indicate degradation in both riparian and in-stream habitat as a result of the altered hydrology associated with flood control management. Analysis of macroinvertebrate communities using a family level biotic index classifies the water quality conditions in Gypsum Creek as non-supportive. Ichthyofaunal communities assessed with a regionally based Index of Biotic Integrity (IBI) where categorized as poor, with the majority of species and individuals collected characterized as being tolerant to anthropogenic stressors. Data indicates that Gypsum Creek would benefit from stream restoration procedures to improve the ecological conditions of the stream.

Key words: water quality, chlorophyll, physical habitat, macroinvertebrates, fish communities, Gypsum Creek Kansas.

About the Central Plains Center for BioAssessment

The Central Plains Center for BioAssessment (CPCB), which resides within the Kansas Biological Survey at the University of Kansas, was created in 1998 to be a center of aquatic expertise and to facilitate the exchange of information among individuals and organizations involved with aquatic issues within the Central Plains. Much of our work is concentrated within USEPA Region 7 (Kansas, Iowa, Missouri, and Nebraska). However, our research and assessment approaches are based on the ecoregion concept to allow for natural variation and similarity in recognizing differing biological potentials. Since ecological conditions and trends transcend geopolitical boundaries, our focus extends into the Central Plains area of the Great Plains Region of Canada and the US. As a non-regulatory scientific entity, one of our primary goals is to facilitate cooperation among academicians, scientists, the States, Tribes, and other public entities in the region that will result in collaborative research on issues of aquatic ecology and water quality. To this end, we host workshops to encourage open exchange of information and ideas, and make the workshop proceedings available over our web page (www.cpcb.ku.edu). We are also a member of two workgroups, the CPCB Biological Criteria Workgroup and the USEPA Region 7 Nutrient Criteria Workgroup, and coordinate meetings for the members of these workgroups. We are involved with research projects which include the analyses of historical lake and stream data to determine ecoregional trends in water quality; supplementation of this data through collection of biological, physical, and chemical data from streams and lakes in KS, IA, MO, and NE; and the modeling of lake/watershed interactions to estimate potential effects of nutrient loads on eutrophication.

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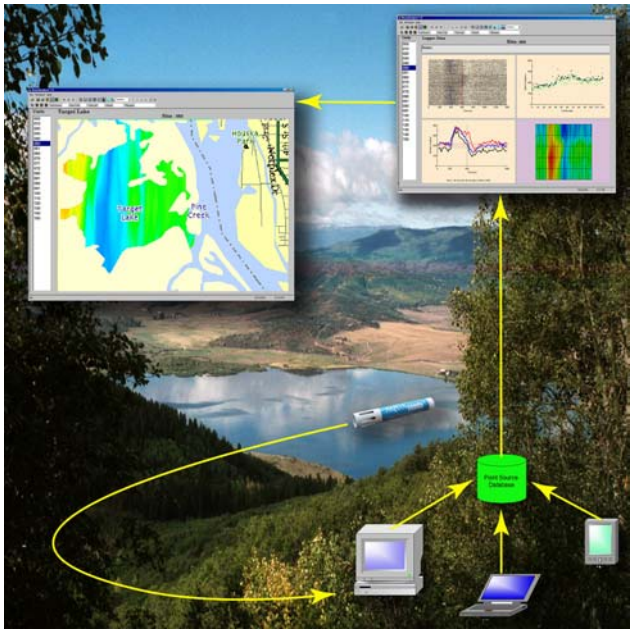

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