Oklahoma Scenic Rivers Joint Phosphorus Criteria Study

Statement of Qualifications Submitted by:

Walter K. Dodds
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Kansas State University
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Descriptions of experience with similar projects (proposed project leader):

We would create a team that has expertise in nutrient criteria and effects of nutrients on streams, including algae. Dodds will lead this project and has experience leading similar projects and doing research in the area. As evidence for the experience Dodds has published several papers on the topic, including some that have been highly cited and form the basis for current nutrient criteria in several states. These papers include very broad and general ones on nutrient pollution and algae in streams (Dodds et al. 2002), general effects of nutrients in streams (Dodds et al. 1998, Dodds and Welch 2000, Dodds 2006, Dodds 2007). Dodds was also involved in producing the nutrient criteria guidance document for rivers and streams produced by the US Environmental Protection Agency (http://www2.epa.gov/sites/production/files/documents/guidance_rivers.pdf). Other relevant publications by Dodds include studies examining background nutrient concentrations (Dodds and Oakes 2004), and case studies on nutrient control in specific rivers (Dodds et al. 1997, Suplee et al 2012). We have attached some of these papers to this statement, as well as Curriculum Vitae so general qualifications of the project team can be assessed.

Specific qualifications of research team:

Dodds will oversee the project, with much of the field and laboratory work done by a postdoctoral researcher, a graduate student and a field/ lab technician. Dodds has extensive experience managing large research teams, including quality assurance, data management, and personnel. Dodds also has decades of experience working with nutrients, chlorophyll, reference trophic state and system productivity.

The team will also include Dr. Rex Lowe (CV attached) as a consultant on the algal identification and bioassessment tools. Dr. Lowe is an accomplished freshwater ecologist. He is on the review board for the Diatoms of North America website and has participated in both the Algal Component of National Water Quality Assessment Program and the EPA's National Lakes Survey. Dr. Lowe has published on nutrient limitation of benthic algae (Carrick and Lowe 2007, Luttenton and Lowe 2006) as well as use of benthic algae in water quality monitoring (Lowe and Pan 1996). He has also published extensively on the taxonomy of freshwater algae.

Dr. Matt Whiles (CV attached) will oversee the invertebrate portion of the research, as macroinvertebrates are excellent indicators of overall biotic integrity and respond to nutrients and algae (Evans-White et al. 2009). Dr. Whiles has published on water quality and invertebrates (e.g., Whiles et al. 2000, Heatherly et al. 2007) and research linking nutrient cycling, primary production, and invertebrates (e.g., Whiles et al. 2012). Whiles has been extensively involved in the Cache River restoration project in southern Illinois, where he works closely with state and federal agency personnel on assessing ecological responses to river restoration activities (e.g., Walther and Whiles 2008). He is well versed in the use of invertebrate-based biological assessment methods for monitoring changes in water quality related to management actions.

Related citations:

Carrick, H.C. and R.L. Lowe 2007. Nutrient limitation of benthic algae in Lake Michigan: the role of silica. J. Phycol. 43:228-234.

Dodds, W., V. Smith, and K. Lohman. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. Canadian Journal of Fisheries and Aquatic Sciences 59:865-874.

Dodds, W., V. Smith, and B. Zander. 1997. Developing nutrient targets to control benthic chlorophyll levels in streams: a case study of the Clark Fork River. Water Research 31:1738-1750.

Dodds, W. K. 2006. Eutrophication and trophic state in rivers and streams. Limnology and Oceanography 51:671-680.

Dodds, W. K. 2007. Trophic state, eutrophication and nutrient criteria in streams. Trends in Ecology & Evolution 22:669-676.

Dodds, W. K., J. R. Jones, and E. B. Welch. 1998. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Research 32:1455-1462.

Dodds, W. K. and R. M. Oakes. 2004. A technique for establishing reference nutrient concentrations across watersheds affected by humans. Limnology and Oceanography Methods 2:333-341.

Dodds, W. K. and E. B. Welch. 2000. Establishing nutrient criteria in streams. Journal of the North American Benthological Society 19:186-196.

Evans-White, M., W. Dodds, D. Huggins, and D. Baker. 2009. Thresholds in macroinvertebrate biodiversity and stoichiometry across water-quality gradients in Central Plains (USA) streams. Journal of the North American Benthological Society 28:855-868.

Heatherly, T., M. R. Whiles, T. V. Royer, and M. B. David. 2007. Relationships between water quality, habitat quality, and macroinvertebrate assemblages in Illinois streams. Journal of Environmental Quality 36:1653-1660.

Lowe, R. L and Y. Pan. 1996. Use of Benthic Algae in Water Quality Monitoring. *In* Stevenson, R. J., M. L. Bothwell and R. L. Lowe, (eds.) Benthic Algal Ecology in Freshwater Ecosystems. Academic Press

Luttenton, M.R. and R.L. Lowe. 2006 Response of a lentic periphyton community to nutrient enrichment at low N:P ratios. J. Phycol.42:1007-1015.

Suplee, M. W., V. Watson, W. K. Dodds, and C. Shirley. 2012. Response of Algal Biomass to Large-Scale Nutrient Controls in the Clark Fork River, Montana, United States1. JAWRA Journal of the American Water Resources Association 48:1008-1021.

Walther, D. A. and M. R. Whiles. 2008. Macroinvertebrate responses to constructed riffles in a small temperate river. *Environmental Management* 41: 516-527.

Whiles, M. R., R. O. Hall, Jr., W. K. Dodds, P. Verburg, A. D. Huryn, C. M. Pringle, K. R. Lips, S. S. Kilham, C. Colón-Gaud, A. T. Rugenski, S. Peterson, and S. Connelly. 2012. Disease-Driven Amphibian Declines Alter Ecosystem Processes in a Tropical Stream. Ecosystems:1-12.

Whiles, M. R., B. L. Brock, A. C. Franzen, and S. Dinsmore II. 2000. Stream invertebrate communities, water quality, and land use patterns in an agricultural drainage basin of northen Nebraska. *Environmental Management*: 26: 563-576.

Specific related projects (see examples of attached papers)

Walter K. Dodds

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Education:

Ph.D.	1986 University of Oregon, Biology
B.S.	1980 University of Denver, cum laude, Biology and Chemistry
Advisors:	Ph.D., Dr. Richard Castenholz, University of Oregon
	Post-Doctoral, Dr. John Priscu, Montana State University

Professional Experience and Appointments:

2009-present	University Distinguished Professor of Biology, Kansas State University
2002-2009	Professor, Division of Biology, Kansas State University
1995-2002	Associate Professor, Division of Biology, Kansas State University
1990-1995	Assistant Professor, Division of Biology, Kansas State University
1990	Adjunct Assistant Professor, Department of Biology, Montana State University
1988-1990	NSF Postdoctoral Fellow, Department of Biology, Montana State University
1987-1988	Postdoctoral Research Associate, Department of Biology, Montana State University
1986-1987	Visiting Assistant Professor, University of Oregon
1985-1986	Research Assistant, University of Oregon
1981-1985	Teaching Assistant, University of Oregon

Research Goals:

My overarching goal is to provide a general and predictive understanding of aquatic ecology, and to promote the application of basic ecological science to water quality and conservation. A central focus of my research has been on river and stream ecosystems and how human influences affect water quality and biological integrity. Nutrient dynamics, especially nitrogen, and eutrophication are major thrusts of my program. Recent emphasis has also been on valuation of ecosystem services.

Professional Memberships:

American Association for the Advancement of Science
American Geophysical Union
American Institute of Biological Sciences
American Society of Limnology and Oceanography
North American Benthological Society
Sigma Xi

Awards and Professional Recognition:

2013 Fellow, American Association for the Advancement of Science

2009 University Distinguished Professor of Biology, Kansas State University

2008 Commerce Bank Distinguished Graduate Faculty Member Award, KSU Graduate School

2007 Outstanding Graduate Faculty Award, Biology Graduate Student Association, Kansas State University. Featured in Mentors and Protégés, Award Winning Commitment. *Nature* 447:610 (www.nature.com/naturejobs/2007/070531/pdf/nj7144-610b.pdf).

2006 Best Paper Award, North American Lake Management Society. Dodds, W.K., E. Carney and R.T. Angelo. 2006. Determining ecoregional reference conditions for nutrients, Secchi depth and chlorophyll *a* in Kansas lakes and reservoirs. Lake and Reservoir Management 22:151-159.

Teaching Experience:

1990-present	Kansas State University: Fresh Water Ecology (BIOL 612), Principles of Biology (BIOL 198), Conservation Biology (BIOL 642), Environmental Problems (BIOL 303), Microbial Ecology (BIOL 687), Origins of Life (BIOL 620), Herbivory (BIOL 890), Presentations in Ecology (BIOL 862), Advanced Aquatic Ecology (BIOL 812), Stream Ecology, Algal Identification (BIOL 890), Professional Skills in Biology (BIOL 890)
1994,1996	Flathead Lake Biological Station: Algal Ecology
1990	Montana State University: Algal Ecology
1986-1987	University of Oregon: Bacteriology, Origins of Life, Freshwater Biology
1985	Oregon Institute of Marine Biology: Measuring Primary Production
1981-1985	University of Oregon: Teaching Assistant in laboratory and field courses

Graduate Students Supervised:

Student	Degree	Year	Current Position	Location
Chris Edler	Ph. D.	deceased		
Eric Strauss	MS	1995	Associate Professor	University of Wisconsin, LaCrosse
Ken Fritz	M.S.	1997	Ecologist	US EPA
Michelle Evans-White	M.S.	2000	Associate Professor	University of Arkansas
Melody Bernot	Ph.D.	2001	Associate Professor	Ball State University
Randall Bernot	Ph.D.	2003	Associate Professor	Ball State University
Robert Oakes	M.S.	2003	Lawyer	Fish & Richardson
Nicole Gerlanc	Ph.D.	2004	Associate Professor	Frederick Community College
Kymberly Wilson	M.S.	2005	Laboratory Manager	Arizona Department of Water Resources
Jonathan O'Brien	Ph.D.	2006	Assistant Professor	Canisius College
Jessica Eichmiller	M.S.	2007	PhD candidate	University of Minnesota
Justin Murdock	Ph.D.	2008	Assistant Professor	Tennessee Tech University
Alex Reisinger	M.S.	2010	PhD Student	University of Notre Dame
Kyle Winders	M.S.	2010	Research Scientist	Missouri Department of Conservation
Alyssa Riley	Ph.D.	2011	Environmental Quality Analyst	Michigan Department of Environmental Quality
Danelle Larson	Ph.D		Current student	
Allison Veach	Ph.D		Current student	
Matthew Trentman	Ph.D		Current student	

Postdoctoral Associates:

Wilfred Singogo 1992-1994
Francisco Costa 1994-1995
Yiyong Zhou 1995-1996
Deb Walks 2006-2007
Michelle Evans-White 2006-2007
David Hoeinghaus 2006-2009
Justin Murdock 2008-2009
Janine Rueegg 2011-

Undergraduate Students Mentored:

John Brant (REU) 2013
Taylor Laskowski 2013
Adam Siders (REU) 2012
Lauren Bansbach (REU) 2011
Anika Bratt (REU) 2010
Dumi Presuma (REU) 2009
Sarah Mueting 2006, 2007
Loren Reinhardt (REU) 2006
Alyssa Standorf (REU) 2005
Eric Banner (REU) 2002, 2003
Amy Junglass 2001
Kajsa Stromberg 2000
Alex Corum, 2000
Janelle Riger 2000

Amanda López (REU) 1999
Molly McGill (REU) 1999
Jennifer Nelson (REU) 1998
Michelle Evans 1995-1997
Britta Culbertson 1995
Esra Aksoy 1995

Jeannie Skalsky 1993-1994
Angie Eichem 1991-1993
Eric Strauss 1991-1992
Clay Randel 1991-1992
Ruth Lehmann 1991-1992

Publications

Books Authored:

Dodds, W.K. and M.R. Whiles. 2010. *Freshwater Ecology: Concepts and Environmental Applications in Limnology* (2nd ed.). Academic Press. 839 pp.

Dodds, W.K. 2009. *Laws, Theories and Patterns in Ecology*. University of California Press. 232 pp

Dodds, W.K. 2008. *Humanity's Footprint: Momentum, Impact and Our Global Environment*. University of Columbia Press. 270 pp.

Dodds, W.K. 2002. *Freshwater Ecology: Concepts and Environmental Applications*. Academic Press. 569 pp.

Peer-reviewed articles:

Cunha, D.G.F., W.K. Dodds, and Md.C. Calijuri (in press). Trends in nutrient and sediment retention in Great Plains reservoirs (USA) *Environmental Monitoring and Assessment*. 10.1007/s10661-013-3445-3

Goring S, K. C. Weathers, W. K. Dodds, P. A. Soranno, L. C. Sweet, K. S. Cheruvelil, J. S. Kominoski, J. Rüegg, A. M. Thorn, and R. M. Utz. (in press) Improving the culture of interdisciplinary collaboration in ecology by expanding measures of success. *Frontiers in Ecology and the Environment*

Smith, V. H. W. K. Dodds, K. E. Havens, D. R. Engstrom, H. W. Paerl, B. Moss and G. E. Likens. (in press). Comment: Cultural eutrophication of natural lakes in the United States is real and widespread. *Limnology and Oceanography*

Ding, Y., Y. Yamashita, W.K. Dodds and R. Jaffé. 2013. Dissolved black carbon in grassland streams: Is there an effect of recent fire history? *Chemosphere* 90:2557–2562.

Dodds, W. K. J. S. Perkin and J. E. Gerken. 2013. Human impact on freshwater ecosystem services: A global perspective. dx.doi.org/10.1021/es4021052 Environ. Sci. Technol. 47: 9061−9068

Dodds, W. K., A. M. Veach, C. M. Ruffing, D. M. Larson, J. L. Fisher, and K. H. Costigan. 2013. Abiotic controls and temporal variability of river metabolism: multi-year analyses of Mississippi and Chattahoochee River Stations *Freshwater Science* 2(4):1073–1087

Larson, D.M., W.K. Dodds, K.E. Jackson, M.R. Whiles, and K.R. Winders. 2013. Ecosystem characteristics of remnant, headwater tallgrass prairie streams. *Journal of Environmental Quality* 42:239-249.

Larson, D.M., B. P. Grudzinski, W. K. Dodds, M. D. Daniels, A. Skibbe, and A. Joern. 2013 Blazing and grazing: influences of fire and bison on tallgrass prairie stream water quality. Freshwater Science 32:779-791

Reisinger, A.J., J.M. Blair, C.W. Rice and W.K. Dodds. 2013. Woody vegetation removal stimulates riparian and benthic denitrification in tallgrass prairie. *Ecosystems.* 16: 545-560

Riley, A. J. and W. K. Dodds. 2013. Whole-stream metabolism: strategies for measurement and modeling diel trends of dissolved oxygen. *Freshwater Science* 32:56-69.

Whiles, M.R., R.O. Hall, Jr., W.K. Dodds, P. Verburg, A.D. Huryn, C.M. Pringle, K.R. Lips, S.S. Kilham, C. Colón-Gaud, A.T. Rugenski, S. Peterson, and S. Connelly. 2013. Disease-driven amphibian declines alter ecosystem functioning in tropical streams. *Ecosystems* 16:146–157.

Dodds, W.K., C.T. Robinson, E.E. Gaiser, G.J.A. Hansen, H. Powell, J.M. Smith, N.B. Morse, S.L. Johnson, S.V. Gregory, T. Bell, T.K. Kratz, and W.H. McDowell. 2012. Surprises and insights from long-term aquatic datasets and experiments. *BioScience* 62:709-721.

Jaffé, R. Y. Yamashita, N. Maie, W. T. Coope, T. Dittmar, W. K. Dodds, J.B. Jones, T. Myoshi, J. R. Ortiz-Zayas, D. C. Podgorski, and A. Watanabe. 2012. Dissolved organic matter in headwater streams: Compositional variability across climatic regions. *Geochimica et Cosmochimica Acta* 95:94-108.

Riis, T., W.K. Dodds, P.B. Kristensen, and A.J. Baisner. 2012. Nitrogen cycling and dynamics in a macrophyte-rich stream as determined by a $^{15}N\text{-}NH_4^+$ release. *Freshwater Biology* 57:1579-1591.

Riley, A.J. and W.K. Dodds. 2012. Riparian woody expansion and subsequent restoration influences prairie stream metabolism. *Freshwater Biology* 57:1138-1150.

Suplee, M.W., V. Watson, W.K. Dodds and C. Shirley. 2012. Response of algal biomass to large scale nutrient controls in the Clark Fork River, Montana, U.S.A. *Journal of the American Water Resources Association* 48:1008-1021.

Beaulieu, J.K., J.L. Tank, S.K. Hamilton, W.M. Wollheim, R.O. Hall Jr., P.J. Mulholland, B.J. Peterson, L.R. Ashkenas, L.W. Cooper, C.N. Dahm, W.K. Dodds, N.B. Grimm, S.L. Johnson, W.H. McDowell, G.C. Poole, H.M. Valett, C.P. Arango, M.J. Bernot, A.J. Burgin, C. Crenshaw, A.M. Helton, L. Johnson, J.M. O'Brien, J.D. Potter, R.W. Sheibley, D.J. Sobota, and S.M. Thomas. 2011. Nitrous oxide emission from denitrification in stream and river networks. *Proceedings of the National Academy of Sciences*. 108:214-219.

Cunha, D.G.F., W.K. Dodds, and Md.C. Calijuri. 2011. Defining nutrient and biochemical oxygen demand baselines for tropical rivers and streams in São Paulo State (Brazil): a comparison between reference and impacted sites. *Environmental Management* 48:945–956.

Findlay, S., P. Mulholland, S. Hamilton, J. Tank, M. Bernot, A. Burgin, C. Crenshaw, C. Dahm, W. Dodds, N. Grimm, W. McDowell, J. Potter, and D. Sobota. 2011. Cross-stream comparison of substrate-specific denitrification potential. *Biogeochemistry* 104: 381-392.

Helton, A, G.C Poole, J.L. Meyer, W.M. Wollheim, B.J. Peterson, P.J. Mulholland, E.S. Bernhardt, J.A Stanford, C. Arango, L.R. Ashkenas, L.W. Cooper, W.K. Dodds, S.V. Gregory, R.O. Hall, Jr., S.K. Hamilton, S.L. Johnson, W.H. McDowell, J.D. Potter, J.L. Tank, S.M. Thomas, H.M. Valett, J.R. Webster, and L. Zeglin. 2011. Thinking outside the channel: Modeling nitrogen cycling in networked river ecosystems. *Frontiers in Ecology and Environment* 9:229–238.

Kohler, T.J, J.N. Murdock, K.B. Gido, and W.K. Dodds. 2011. Nutrient loading and grazing by the minnow *Phoxinus erythrogaster* shift periphyton abundance and stoichiometry in experimental streams *Freshwater Biology* 56:1133–1146.

Murdock, J.N., W.K. Dodds, K.B. Gido, and M.R. Whiles. 2011. Dynamic influences of nutrients and grazing fish on benthic algae during recovery from flood. *Journal of the North American Benthological Society* 30:331–345.

Reisinger A., D. Presuma, K. Gido, and W.K. Dodds 2011. Direct and indirect effects of central stoneroller (*Campostoma anomalum*) on mesocosm recovery following a flood: can macroconsumers affect denitrification? *Journal of the North American Benthological Society* 30:840-852.

Bernot, M.J., D.J. Sobota, R.O. Hall Jr., P.J. Mulholland, W.K. Dodds, J.R. Webster, J.L. Tank, L.R. Ashkenas, L.W. Cooper, C.N. Dahm, S.V. Gregory, N.B. Grimm, S.K. Hamilton, S.L. Johnson, W.H. McDowell, J.L. Meyer, B. Peterson, G.C. Poole, H.M. Valett, C. Arango, J.J. Beaulieu, A.J. Burgin , C. Crenshaw, A.M. Helton, L. Johnson, J. Merriam, B.R. Niederlehner, J.M. O'Brien, J.D. Potter, R.W. Sheibley, S.M. Thomas, and K. Wilson. 2010. Inter-regional comparison of land-use effects on stream metabolism. *Freshwater Biology.* 55:1874–1890.

Dodds, W.K., W.H. Clements, K. Gido, R.H. Hilderbrand, and R.S. King. 2010. Thresholds, breakpoints, and nonlinearity in freshwaters as related to management. *Journal of the North American Benthological Society* 29:988-997.

Gido, K.B., K.N. Bertrand, J.N. Murdock, W.K. Dodds, and M.R. Whiles. 2010. Disturbance-mediated effects of fishes on stream ecosystem processes: concepts and results from highly variable prairie streams. *American Fisheries Society Symposium* 73:593–617.

Gido, K.B., W.K. Dodds, and M.E. Eberle. 2010. Retrospective analysis of fish community change during a half-century of land-use and streamflow changes. *Journal of the North American Benthological Society* 29:970-987.

Graham D.W., C. Linacre, W.K. Dodds, J.M. O'Brien, E. Banner, I.M. Head, M.S. Smith, R.K. Yang, and C.W. Knapp. 2010. Correlations between in situ denitrification activity and nir-gene. abundances in pristine and impacted prairie streams. *Environmental Pollution* 158:3225-3229.

Murdock, J.N., W.K. Dodds, J.A. Reffner, and D.L. Wetzel. 2010. Measuring cellular scale nutrient distribution in algal biofilms with synchrotron confocal infrared microspectroscopy. *Spectroscopy* 25:32-41.

Murdock, J.M. K.B. Gido, W.K. Dodds, K.N. Bertrand, and M.R. Whiles. 2010. Consumer return chronology alters recovery trajectory of stream ecosystem structure and function following drought. *Ecology* 91:1048–1062.

O'Brien, J.M. and W.K. Dodds. 2010. Saturation of NO_3^- uptake in prairie streams as a function of acute and chronic nitrogen exposure. Journal of North American Benthological Society 29(2):627–635

Banner, E., A. Stahl, and W.K. Dodds. 2009. Stream discharge and riparian land use influence in-stream concentrations and loads of phosphorus from Central Plains watersheds. *Environmental Management* 44:552–565.

Bertrand, K.N., K.B. Gido, W.K. Dodds, J.N. Murdock and M.R. Whiles. 2009. Disturbance frequency and functional identity mediate ecosystem processes in prairie streams. *Oikos* 118:917-933.

Dodds, W.K., W. Bouska, J.L. Eitzmann, T.J. Pilger, K.L. Pitts, A.J. Riley, J.T. Schloesser and D.J. Thronbrugh. 2009. Eutrophication of U.S. freshwaters: Analysis of potential economic damages. *Environmental Science and Technology* 43:12-19.

Evans-White, M.A., W.K. Dodds, D.G. Huggins, and D.S. Baker. 2009 Threshold patterns in aquatic biodiversity across water quality gradients in Central Plains streams and rivers. *Journal of the North American Benthological Society* 28:855-868.

Hall, R.O. ,Jr., J.L. Tank, D.J. Sobota, P.J. Mulholland, J.M. O'Brien, W.K. Dodds, J.R. Webster, H.M. Valett, G.C. Poole, B.J. Peterson, J.L. Meyer, W.H. McDowell, S.L. Johnson, S.K. Hamilton, N.B. Grimm, S.V. Gregory, C.N. Dahm, L.W. Cooper, L.R. Ashkenas, S.M. Thomas, R.W. Sheibley, J.D. Potter, B.R. Niederlehner, L.T. Johnson, A.M. Helton, C.M. Crenshaw, A.J. Burgin, M.J. Bernot, J.J. Beaulieu and C.P. Arango. 2009. Nitrate removal in stream ecosystems measured by ^{15}N addition experiments: Total uptake. *Limnology and Oceanography* 54:563-665

Knapp, C.W, W.K. Dodds, K.C. Wilson, J.M. O'Brien and D.W. Graham. 2009. Spatial heterogeneity of denitrification genes in a highly homogenous urban stream. *Environmental Science and Technology* 43:4273–4279.

Mulholland, P.J., R.O. Hall, Jr., D.J. Sobota, W.K. Dodds, S.E.G. Findlay, N.B. Grimm, S.K. Hamilton, W.H. McDowell, J.M. O'Brien, J.L. Tank, L.R. Ashkenas, L.W. Cooper, C.N. Dahm, S.V. Gregory, S.L. Johnson, J.L. Meyer, B.J. Peterson, G.C. Poole, H.M. Valett, J.R. Webster, C.P. Arango, J.J. Beaulieu, M.J. Bernot, A.J. Burgin, C.L. Crenshaw, A.M. Helton, L.T. Johnson, B.R. Niederlehner, J.D. Potter, R.W. Sheibley and S.M. Thomas. 2009. Nitrate removal in stream ecosystems measured by ^{15}N addition experiments: Denitrification. *Limnology and Oceanography* 54:666–680.

Johnson, L.T., J.L. Tank and W.K. Dodds. 2009. The influence of land use on stream biofilm nutrient limitation across eight North American ecoregions. *Canadian Journal of Fisheries and Aquatic Sciences* 66:1081–1094.

Wilson, K.C. and W.K. Dodds. 2009. Centimeter-scale stream substratum heterogeneity and metabolic rates. *Hydrobiologia* 623:53-62.

Dodds, W.K., J.J. Beaulieu, J.J. Eichmiller, J.R. Fischer, N.R. Franssen, D.A. Gudder, A.S. Makinster, M.J. McCarthy, J.N. Murdock, J.M. O'Brien, J.L. Tank and R.W. Sheibley. 2008. Nitrogen cycling and metabolism in the thalweg of a prairie river. *Journal of Geophysical Research Biogeosciences* 113, G04029.

Dodds, W.K. and R.M. Oakes. 2008. Headwater influences on downstream water quality. *Environmental Management* 41:367-377.

Dodds, W.K., K.C. Wilson, R.L. Rehmeier, G.L. Knight, S. Wiggam, J.A. Falke, H.J. Dalgleish and K.N. Bertrand. 2008. Benefits of ecosystem goods and services associated with restored lands compared to native lands. *BioScience* 58:837-845.

Mulholland, P.J., A.M. Helton, G.C. Poole, R.O. Hall, Jr., S.K. Hamilton, B.J. Peterson, J.L. Tank, L.R. Ashkenas, L.W. Cooper, C.N. Dahm, W.K. Dodds, S. Findlay, S.V. Gregory, N.B. Grimm, S.L. Johnson, W.H. McDowell, J.L. Meyer, H.M. Valett, J.R. Webster, C. Arango, J.J. Beaulieu, M.J. Bernot, A.J. Burgin, C. Crenshaw, L. Johnson, B.R. Niederlehner, J.M. O'Brien, J.D. Potter, R.W. Sheibley, D.J. Sobota and S.M. Thomas. 2008. Stream denitrification across biomes and its response to anthropogenic nitrogen loading. *Nature* 452:202-207.

Murdock, J.N., W.K. Dodds and D.L. Wetzel. 2008. Subcellular chemical imaging of localized benthic algal nutritional content via HgCdTe array FT-IR. *Vibrational Spectroscopy* 48:179-188.

O'Brien, J.M. and W.K. Dodds. 2008. Ammonium uptake and mineralization in prairie streams: chamber incubation and short-term nutrient addition experiments. *Freshwater Biology* 53:102-112.

Williamson, C.E., W. Dodds, T.K. Kratz and M. Palmer. 2008. Lakes and streams as sentinels of environmental change in terrestrial and atmospheric processes. *Frontiers in Ecology and the Environment* 6:247-254.

Dodds, W.K. and J.J. Cole. 2007. Expanding the concept of trophic state in aquatic ecosystems: It's not just the autotrophs. *Aquatic Sciences* 69:427-439.

Dodds, W.K. 2007. Trophic state, eutrophication, and nutrient criteria in streams. *Trends in Ecology and Evolution* 22:670-676.

O'Brien, J.M. , W.K. Dodds, K.C. Wilson, J.N. Murdock and J. Eichmiller. 2007. The saturation of N cycling in Central Plains streams: ^{15}N experiments across a broad gradient of nitrate concentrations. *Biogeochemistry* 84:31-49.

Murdock, J.N. and W.K. Dodds. 2007. Linking benthic algal biomass to stream substratum topography. *Journal of Phycology* 43:449-460.

Dodds, W.K. 2006. Nutrients and the "Dead Zone": Ecological stoichiometry and depressed dissolved oxygen in the northern Gulf of Mexico. *Frontiers in Ecology and the Environment* 4:211-217.

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Dodds, W.K. and R.M. Oakes. 2006. Controls on nutrients across a prairie stream watershed: Land use and riparian cover effects. *Environmental Management* 37:634-646.

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Book Chapters:

Gray, L., G.L. Macpherson, J.K. Koelliker and W.K. Dodds. 1998. Hydrology and aquatic chemistry. In Knapp, A. K., J. M. Briggs, D. C. Hartnett and S. L. Collins (eds.), *Grassland Dynamics: Long-term Ecological Research in Tallgrass Prairie*. Oxford Press, pp 159-176.

Gray, L.J. and W.K. Dodds. 1998. Structure and dynamics of aquatic communities. In Knapp, A. K., J. M. Briggs, D. C. Hartnett and S. L. Collins (eds.), *Grassland Dynamics: Long-term Ecological Research in Tallgrass Prairie*. Oxford Press, pp. 177-189.

Reviews, Reports and Popular Articles:

Dodds, W. K. 2013. Pure headwaters in the Flint Hills. Symphony in the Flint Hills Field Journal, Volume V, 2013: Fort Riley

Dodds, W.K. 2007. Review of D. P. Batzer and R. R. Sharitz (eds.) Ecology of Freshwater and Estuarine Wetlands, University of California Press. The Quarterly Review of Biology 82:431.

Dodds, W.K. 2000. Review of Gary E. Dillard, Common Freshwater Algae of the United States. Journal of Phycology 36:622.

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Buck, S., G. Denton, W. Dodds, J. Fisher, D. Flemer, D. Hart, A. Parker, S. Porter, S. Rector, A. Steinman, J. Stevenson, J. Stoner, D. Tillman, S. Wang, V. Watson, E. Welch. 2000. Nutrient Criteria Technical Guidance Manual, Rivers and Streams. United States Environmental Protection Agency. EPA-822-B-00-002.

Pimentel, D. and W. Dodds. 1999. Human resource use, population growth, and environmental destruction. Bulletin of the Ecological Society of America 80:88-91.

Dodds, W.K. 1996. Assessment of blue-green algal toxins in Kansas. Report no. G2020-02, Kansas Water Resources Research Institute, contribution no. 320. U.S. Department of the Interior Geological Survey.

Meyer, J., T. Crocker, D. D'Angelo, W. Dodds, S. Findlay, M. Oswood, D. Repert and D. Toetz. 1993. Stream research in the LTER network. LTER publication No. 15.

Research Grants Funded :

2013-2014 *Doctoral Dissertation Research: The Impact of Historical Logging Activities on Mountain Stream Ecogeomorphology (Claire Ruffing).* National Science Foundation. $15,972 (co-PI)

2011-2016 *Collaborative Research: Scale, Consumers and Lotic Ecosystem Rates (SCALER): Centimeters to Continents.* National Science Foundation, $3,304,097 total, $ 1,198,082 to KSU (National Project lead PI)

2011-2014 *MRI: Acquisition of a Hybrid GPU Computing Cluster High-End Applications in Science and Engineering.* National Science Foundation, $700,000 (co-PI)

2009-2013 *Collaborative Research: EPSCoR R 11 Track 2 Oklahoma & Kansas: A cyberCommons for Ecological Forecasting.* National Science Foundation EPSCoR Program, $1,608,168 to KSU (KSU-lead PI, Kansas co-PI)

2008-2013 *Ecological integrity of prairie streams as influenced by patch-burn grazing and riparian protection*. Missouri Department of Conservation, $281,512 to KSU (KSU-lead PI).

2008-2014	*Konza Prairie LTER VI: Grassland dynamics and long-term trajectories of change.* National Science Foundation, $5,640,000 (co-PI).
2006-2010	*Forecasting ecological change in the Central Plains,* National Science Foundation EPSCoR Program, $3,373,478 to KSU (PI).
2005-2007	*Ecosystem thresholds and alternate states in Great Plains rivers and streams: cascading effects of anthropogenic hydrologic disturbance.* U.S. Environmental Protection Agency, $299,566 (PI).
2004-2007	*Interactive effects of disturbance frequency and species composition on ecosystem functioning of intermittent streams: a test of future climate change scenarios.* National Science Foundation, $320,000 (co-PI).
2004	*Funds for workshop on N transport.* LTER Network Office, *$*9,264 (PI).
2003	*Funds for workshop on lotic denitrification.* LTER Network Office, $2,400 (PI).
2003-2005	*Water quality and nitrogen loading in KS streams (Genomic approaches to study organismal response to global environmental change).* National Science Foundation EPSCoR, subcontract- $25,000 (PI).
2003-2005	*REU Site: Conservation of the tallgrass prairie ecosystem.* National Science Foundation, $169,954 (co-PI).
2002-2005	*Development of aquatic GAP analysis in Kansas.* United States Department of the Interior, $183,217 (co-PI).
2002-2008	*Konza Prairie LTER V: Long-term research on grassland dynamics and global change.* National Science Foundation, $4,680,000 (co-PI).
2001-2006	*Nitrate uptake and retention in streams: mechanisms and effects of human disturbances from stream reaches to landscapes.* National Science Foundation. $3,000,000 (co-PI).
1999-2002	*Research experience for undergraduates in grassland ecology at Konza Prairie.* National Science Foundation, $168,612 (PI).
1999-2001	*Acquisition of an isotope ratio mass spectrometer in the Kansas State University-University of Kansas-Creighton University Consortium.* National Science Foundation, $169,400 (co-PI).
1999-2000	*Assessing the impact of exposure of periphyton to linear alkylbenzene sulfonate with microscale sensors at the Procter & Gamble experimental stream facility.* Proctor and Gamble, $6,000 (PI).
1999-2001	*Quality and quantity of suspended material in Kansas Rivers: Demonstrating the influence of management practices.* Kansas Department of Health and Environment, $96,997 (co-PI).
1999-2003	*Stocking success and factors influencing survival and growth of stocked Walleyes.* Kansas Department of Wildlife and Parks, $160,378 (co-PI).
1997	*Research in modeling metabolism of attached stream algae.* National Science Foundation, Division of International Programs, $14,576 (PI).
1996-1999	*Nitrogen uptake, retention and cycling in stream ecosystems: An intersite N-15 tracer experiment.* National Science Foundation, $1,389,335 (co-PI).
1996-2002	*Long-term ecological research in tallgrass prairie: the Konza Prairie LTER Program.* National Science Foundation. $3,600,000 (co-PI).

1996-1999 *Research experience for undergraduates in grassland ecology at Konza Prairie Research Natural Area.* National Science Foundation. $163,679 (co-PI).

1995-1996 *Assessment of blue-green algal toxins in Kansas.* U.S. Geological Survey, $44,206 (PI).

1995 *LTER supplement,* National Science Foundation, $36,236 (co-PI).

1994-1997 *Use of remotely sensed data on phenological activity and heterogeneity to detect changes in grassland species composition in response to stress*. U.S. Environmental Protection Agency. $240,842 (co-PI).

1994-1995 *Estimation of effects of ultraviolet irradiance on periphyton primary production in streams*. National Science Foundation, $49,935 (PI).

1993 *LTER* e*quipment supplement: DOC Analyzer.* National Science Foundation, $18,000 (co-PI).

1992-1995 *Biotic and abiotic factors controlling nitrogen flux in subsurface systems.* National Science Foundation, EPSCoR, $594,965 (co-PI).

1992 *Density controls on ecological communities: Relationships between complexity and stability.* National Science Foundation, 30 service units on Cray YM-P Super Computer; 5 units on Connection Machine (PI).

1991-1992 *Nutrient Removal Bioassay Methods for Assessment of the Effects of Decreased Nutrient Loading on Phytoplankton Communities in Aquatic Ecosystems*. Soap and Detergent Association. $28,202 (PI).

1991-1996 *Fire, grazing and climatic interactions in tallgrass prairie (*Konza Prairie LTER program). National Science Foundation, ~$18,000/yr (co-PI).

1991 *Diatoms of ephemeral pools as air quality indicators*. National Park Service, Air Quality Division, $2,500 (PI).

1987 *Grazing of epiphytes from Cladophora: Biological, physical and chemical interactions.* National Science Foundation Postdoctoral Fellowship, $56,000 (PI).

1987 *Understanding the ecological* r*elationship between* Nostoc parmelioides *and its mutualistic midge larva*. Whitehall Research Foundation, $9,865 (PI).

1984 Sigma Xi Grants-in-Aid of Research. $300.

1982 Oregon Biomedical Research Fund. $600.

Invited Seminars:

2013 *Understanding freshwaters in the anthropocene: long-term and large-scale approaches.* Keynote talk. Mississippi River Research Consortium 45th annual meeting. Lacrosse, WI

Teaching strategies. The New Faculty Institute. Kansas State University, Manhattan.

Value of freshwater, Public talk, Weese Distinguished Lecture, University of Oklahoma, Norman

Understanding freshwaters in the anthropocene: long-term and large-scale approaches, Technical talk, Weese Distinguished Lecture, University of Oklahoma, Norman

Prairie Streams. Keynote talk, Annual Konza Docent Roundup, Manhattan, KS

The Kaw River Informational talk, Symphony on the Prairie. Fort Riley, KS

2012 *The unique characteristics of grassland streams.* North American Benthological Society, Louisville, Kentucky

Valuation of ecosystem goods and services in aquatic ecosystems. University of Nebraska, Lincoln

Valuation of ecosystem goods and services in aquatic ecosystems. Oregon State University, Corvallis

Scale Consumers and Lotic Ecosystem Rates: a successful Macrosystems proposal. NEON annual members meeting, Washington, DC

2011 *The worst problems in the world.* Kansas State University, Manhattan, Kansas

Valuation of ecosystem goods and services in aquatic ecosystems, Grand Valley State University, Allendale, Michigan

*Global human impacts on freshwaters scaled by relative influence on ecosystem goods and service*s. Special Session: North American Benthological Society, Providence, Rhode Island

Grassland Streams. Grasslands in a Global Context, International Symposium. Kansas State University, Manhattan, Kansas

2010 Graduate Commencement Speaker. Kansas State University, Manhattan, Kansas

The worst problems in the world. University of California, Santa Cruz

Valuation of ecosystem goods and services in aquatic ecosystems. University of Arkansas, Fayetteville, Arkansas

Measuring stream metabolism. The University of Copenhagen, Copenhagen, Denmark

The ecology of prairie streams. University of Aarhus, Aarhus, Denmark

Nutrient Criteria in the Midwestern United States Iowa Water Conference

Thresholds, non-linearity and prediction in freshwater ecosystems. Iowa State University

Valuation of ecosystem goods and services in aquatic ecosystems. Oklahoma State University

Valuation of ecosystem goods and services in aquatic ecosystems. University of North Texas

2009 *Thresholds, non-linearity and prediction in freshwater ecosystems.* Ecology group, Eawag, Swiss Federal Institute of Aquatic Science and Technology, *Dübendorf* Switzerland.

Prairie streams. Eawag, the Swiss Federal Institute of Aquatic Science and Technology, Kastanienbaum – Lucerne, Switzerland.

Humanity's Footprint. Eawag, the Swiss Federal Institute of Aquatic Science and Technology, *Dübendorf* Switzerland.

Properties of Water. Eidgenössische Technische Hochschule Zürich, Switzerland.

Laws Theories and Patterns in Ecology. Centre d'Estudis Avangats de Blanes (C.S.I.C.), Spain.

Thresholds, non-linearity and prediction in freshwater ecosystems. University of Barcelona, Spain.

Valuation of ecosystem goods and services in aquatic ecosystems. Universidad de Girona, Girona Spain.

2008 *Thresholds, non-linearity and prediction in freshwater ecosystems* NSF EPSCoR/ Water Dynamics/ Workshop. Keynote presentation. Burlington, VT.

Ecology of prairie streams. Plenary talk. Great Plains Limnology Society. Lake Texoma, OK.

STREON: Stream experimental and observational network. North American Benthological Society Annual Meeting, Salt Lake City, UT.

STREON: Stream experimental and observational network. Consortium of Universities for the Advancement of Hydrologic Science (CUAHSI), Boulder, CO.

2007 *Ecology of prairie streams. Southwest Missouri State University, Springfield, MO.*

Aquatic research on Konza Prairie. EPSCoR Reverse site visit, Washington, D.C.

Ecological forecasting: building infrastructure for the Central Plains. Water & Society Seminar Series, Kansas State University, Manhattan, KS.

It is not just phosphorus that controls trophic state in fresh waters. US EPA Webcast.

Ecological forecasting and aquatic resources. Kansas Water Authority.

Ecological forecasting and aquatic resources. Consortium for Environmental Stewardship and Sustainability.

Laws in ecology. Southern Illinois University, Carbondale, IL.

Laws in ecology. University of Florida, Gainesville, FL.

2005 S*ome things I was wrong about in stream ecology, and why they matter*. Fordham University, Fordham, NY.

Some things I was wrong about in stream ecology, and why they matter. Stroud Water Research Center, Philadelphia, PA.

Nutrient criteria in streams. Great Plains Center for Bioassessment. Lawrence, KS.

2004 *Laws in ecology*. University of New Mexico. Albuquerque, NM.

Ecological stoichiometry in streams. University of New Mexico. Albuquerque, NM.

2003 *Nitrogen cycling rates and carbon and nitrogen stoichiometry in streams.* North American Benthological Society, Athens, GA.

Nitrogen in streams. University of Oklahoma, Norman, OK.

2002 *Nitrogen in streams*. Iowa State University, Ames, IA.

Nitrogen in streams. Wichita State University, Wichita, KS.

2001 C*atapult: Momentum of human impact on earth*. Division of Biology, Kansas State University. Manhattan, KS.

1999 *Nitrogen cycling at Konza Prairie*. Institute of Ecosystem Studies, Millbrook, NY.

Establishing nutrient criteria in streams. North American Benthological Society, Duluth, MN.

R*elative importance of N and P limitation of stream periphyton.* American Society of Limnology and Oceanography, Santa Fe, NM. (co-authors: Tank, J., Lohman, K. and Smith, V.)

1998 N*itrogen cycling in some aquatic environments.* University of Montana, Missoula, MT.

Attenuation of flow in macrophytes and periphyton. North American Benthological Society, Prince Edwards Island, Canada.

Asking Large-Scale Ecological Questions: Some Possible Approaches for Individuals. University of Kansas, Lawrence, KS.

1997 *Microhabitat and aquatic microorganisms.* Monash University, Melbourne, Australia, National Institute of Water Quality and Atmospheric Research, Christchurch, New Zealand, Otago University, Dunedin, New Zealand.

Aquatic ecology on Konza Prairie. Canterbury University, Christchurch, New Zealand, Otago University, Dunedin, New Zealand, National Institute of Water Quality and Atmospheric Research, Hamilton, New Zealand, University of New England, Armidale, Australia.

1996 *How good was water quality in native tallgrass prairie streams?* Kansas Water Environment Association, 51st Annual Conference, Hutchinson, KS.

1994 *Mutualism in Communities and Ecosystems: Theory and Aquatic Examples*. Division of Biology, Kansas State University.

Water Quality and Extreme Flows on Pristine Tallgrass Prairie. Water and the Future of Kansas Conference, Manhattan, KS.

1992 *Nutrient Pollution and Surface Water Quality*. Water and the Future of Kansas Conference, Manhattan, KS.

Is Photosynthesis in Cyanobacterial Surface Blooms Limited by Flux of Atmospheric CO_2? American Society of Limnology and Oceanography, Santa Fe, NM.

1991 *Mutualism and Aquatic Primary Producers*. Ecology Group, University of Nebraska, Lincoln, NE.

1990 *Mutualism and Aquatic Primary Producers*. Department of Zoology, University of Oklahoma, Department of Zoology and Division of Biology, Kansas State University.

Microenvironment and Microbial Processes. Institute of Arctic Biology, University of Alaska and College of St. Benedicts, MN.

1989 *Mutualism and Aquatic Primary Producers*. Department of Biology, Florida International University.

Eutrophication in Flathead Lake. State Department of Health and Environmental Sciences, Helena, MT.

1988 *Influence of* PO_4^{3-} *on* $^{15}NH_4^+$ *and* $^{15}NO_3^-$ *Fluxes in an Oligotrophic Lake: Results from Mesocosm Studies*. American Society of Limnology and Oceanography, San Francisco, CA.

Microhabitat and Interrelationships between Current Velocity, Photosynthesis, O_2*, Light and* N_2 *Fixation in a Benthic Cyanobacterium*. American Society of Limnology and Oceanography, San Francisco, CA.

Nutrient-Phytoplankton Interactions in Flathead Lake. Department of Fish and Wildlife and Parks, Kalispell, MT.

Mutualism between Nostoc *and* Cricotopus. Department of Microbiology, Montana State University.

1987 *Nitrogen and Phosphorus Physiology of Phytoplankton in Flathead Lake*. University of Montana Biological Station.

The Nitrogen Budget and Community Interactions in a Cold Water Pool Dominated by a Nitrogen Fixing Cyanobacterium. Department of Botany, University of Washington.

1985 *Grazing and Season Effects on* Nostoc *Growth*. Stream Team, Oregon State University.

The Nitrogen Cycle of Mare's Egg Spring. Oregon Institute of Marine Biology.

Selected Professional Activities and Service Contributions:

Professional Activities:

Elections and Place Committee Society for Freshwater Science (2013-2016)

Leader, Review Team for Environmental Protection Agency draft report: Connectivity of Streams and Wetlands to Downstream Waters- A Review and Synthesis of the Scientific Evidence (2012)

Group Leader, STREON: Stream Research and Observational Network, National Ecological Observatory Network, (2007-2009)

Member, STREON Working Group. Advisory board to NEON (2009 to present)

Editorial Board, Freshwater Biology (2008-present)

Associate Editor, Journal of Geophysical Research (2006-2008)

Associate Editor, Journal of the North American Benthological Society (2004-2006)

2005 Annual Meeting Co-Chair, North American Benthological Society (2003-2005)

Panelist, National Science Foundation- Ecosystems Program (2001, 2004)

Associate Editor, Journal of Phycology (1999-2003)

Chair, Intersociety and International Interactions Committee of the North American Benthological Society (1997)

Member, Intersociety and International Interactions Committee of the North American Benthological Society (1998-2001)

Member, Future Meetings Committee of the American Society of Limnology and Oceanography (1997-present)

Chair, Council for Aquatic Sciences-Aquatic Sciences Meeting Committee (2005)

Member, Council for Aquatic Sciences-Aquatic Sciences Meeting Committee (1999-2002)

Member, North American Benthological Society- Elections and Meetings Committee (2000-2003)

Member, United States Environmental Protection Agency, Nutrient Criteria in Streams Work Group Region 7 (1999-present)

Member, Water and the Future of Kansas Conference Committee (1990-1997)

University Activities:

Chair, Konza LTER Aquatic and Hydrological Research Group (1990-present)
Chair, Division of Biology Seminar Committee (1992-present)
Chair, Division of Biology Chris Edler Outstanding Research Award Committee (1995-present)
Chair, Konza Management Plan: Aquatic Subgroup (1999-2001)
Chair, Division of Biology Search Committees- Fisheries position (2001); Wildlife position (2002); Kansas Cooperative Unit and Division of Biology, Assistant Leader Fisheries Unit (2003) (member of 9 additional search committees)
Member, Division of Biology Undergraduate Curriculum Committee (1995-2002)
Member, Division of Biology Tenure and Promotions Committee (2002 -2008)
Member, Division of Biology BRIEF Committee (2002-present)
Member, Konza Education Advisory Committee (1999-present)
Member, Konza Prairie Advisory Committee (1994-present)
Member, Division of Biology Ecology and Evolutionary Biology Section, (1990-present)
Member, Division of Biology Microbiology and Immunology Section, (1990-present)
Treasurer, Sigma Xi, Kansas State University Chapter (1998-2000)

PROFESSIONAL RESUME
REX L. LOWE

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Academic Year
Professor of Biological Sciences
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I. Academic Degrees and Positions

A. Degrees

1. B. S., Iowa State Univ., Ames, IA. November, 1966. Botany major; Zoology minor.
2. Ph.D., Iowa State Univ., Ames, IA. August, 1970. Phycology major. Dissertation Title: Taxonomic and Ecological Analyses of Diatom Communities in Drainage Ditches.

B. Positions (Past)

1. Graduate Teaching Assistant in General Botany and Elementary Plant Physiology. Iowa State University, 1966-67.
2. Graduate Teaching Assistant in Freshwater Algae and Phytoplankton Ecology. The University of Michigan Biological Station, Douglas Lake, Pellston, MI, summer, 1969
3. Assistant Professor of Biology, Central Michigan Univ., NSF Summer Institute, C. U. M. Biological Station, Beaver Island, MI, June 1971-August 1971.
4. Assistant Professor of Biology, Bowling Green State Univ., Bowling Green, OH, 1970-75.
5. Visiting Assistant Professor, Biological Station, College of Literature, Science and the Arts, Univ. of Michigan, June-August 1974-1976.
6. Visiting Associate Professor, Biological Station, College of Literature, Science and the Arts, Univ. of Michigan, June-August 1977-1981.
7. Associate Professor of Biology, Bowling Green State Univ., Bowling Green, OH, 1975-1981.
9. Visiting Professor, Virginia Polytechnic Institute and State University. 1982-1983 (One-year Faculty Improvement Leave from BGSU).
11. Visiting Scientist, Hydrology Centre, Department of Scientific and Industrial Research, Christchurch, New Zealand. 1991-1992. (One-year Faculty Improvement Leave from BGSU).

12. Fellow, Cooperative Institute for Limnology and Ecosystems Research, Ann Arbor, MI. 1991-present.
13. Visiting Professor, Stone Laboratory, The Ohio State University, summer 2009 - 2011.

B. Positions (Current)

1. Professor Emeritus of Biology, Bowling Green State Univ., Bowling Green, OH.
2. Visiting Professor, Biological Station, College of Literature, Science and the Arts, Univ. of Michigan, June-August 1982-present.
3. Associate Editor, The Scientific World, 2000-present
4. Fellow and Lifetime Member, Ohio Academy of Science
5. Membership Chair and Institutional Representative, Great Lakes Aquatic Ecosystem Research Consortium
6. BGSU Institutional Representative, Fish and Wildlife Section, National Association of State Universities and land-grant Colleges.
7. Editorial Board Member, Diatoms of the United States

II. Honors and Awards

1. Bowling Green State University Distinguished Teacher in Sciences Award, 1974
2. Bowling Green State University College of Arts and Sciences, Citation for Instructional Excellence, 1991
3. Bowling Green State University College of Arts and Sciences, Citation for Instructional Excellence, 1994
4. Bowling Green State University College of Arts and Sciences, Faculty Excellence Award for Teaching and Research, 1996
5. Bowling Green State University, Olscamp Research Award 1998.
6. G. P. Wilder Distinguished Chair in Botany at the University of Hawaii2007-2008

III. Consultantships

1. Ichthyological Associates
2. N.U.S. Corp.
3. Michigan Department of Natural Resources
4. Industrial Biotest
5. U.S. EPA
6. Nalco Environmental Sciences
7. U.S. Park Service
8. Westinghouse, Environmental Systems Division
9. Metropolitan Sanitary District of Greater Chicago
10. Environmental Impact Associates
11. AWARE
12. Commonwealth Associates
13. Birchwood Farm Estates
14. Bristol-Meyers
15. Design Management Associates
15. Proctor and Gamble

16. Toledo Edison
17. The South Florida Water Management Dist.
19. Kilkelly Environmental Associates
20. Chadwick and Associates
21. Thunderbird Hills Country Club
22. The Academy of Natural Sciences of Philadelphia
23. Hull and Associates
24. U. S. Geological Survey
25. Ecology III
26. Eckenfelder Inc.
27. The Nature Conservancy
28. Midwest Consultants
29. AATA International, Inc. Environmental Consultants
30. Leelanau Nature Conservancy,
31. Detroit Edison

IV. Research Grants and Contracts

1. Diatoms of Northwest Ohio, $1,274, Faculty Research Committee, BGSU, 7/1/71.
2. A proposal for the establishment of facilities for light and scanning electron microscopy in the Department of Biology at Bowling Green State University, $4,000, Knight Foundation, 10/3/71. (With R. E. Crang).
3. Bacillariophyta of Miller Blue Hole, $860, Ohio Biological Survey, 1972.
4. Investigations of Golf Course Ponds, $2,980, Belmont Country Club, Rossford, OH, 2/1/1972. (With G. Acker and W. B. Jackson).
5. The fine structure and morphological variability of some planktonic centric diatoms, $474, BGSU Faculty Research Committee, 1973.
6. Diatoms as water quality indicator organisms, $5,532, U.S. Environmental Protection Agency, 1973.
7. Bacillariophyta of Oak Openings, Lucas County, Ohio, $450, Ohio Biological Survey, 1973.
8. Periphyton diatoms of the Sandusky River, $800, Ohio Biological Survey, 1975.
9. Diatom communities from natural and disturbed streams in the Great Smoky Mountains National Park, $4,725, U.S. National Park Service, 1976-78.
10. Characterization of epilithic attached algae communities in Grand Traverse Bay, Lake Michigan, $50,000, National Science Foundation, January 15, 1978-June 30, 1980.
11. Distribution of blue-green algal endosymbionts in diatoms of the family Epithemiaceae. $2,820. BGSU Faculty Research Committee. June 1986-June 1987.
12. Reservoir phytoplankton distribution. $18,900. U. S. Army Corps of Engineers. Oct. 1988-1989.
13. Periphyton Bioassays. $11,000. The Procter & Gamble Co. 1990.

14. The fate of phytoplankton following processing by the zebra mussel. $6,000. Ohio Sea Grant. May, 1990.
15. Stream Laboratory at the Biological Station. $92,228 plus matching funds from The University of Michigan. The National Science Foundation. August, 1990.
16. Purchase of scanning electron microscope & image acquisition capability. $280,000, National Science Foundation, Jan. 1990.
17. Investigations into temporal shifts in nutrient limitation in New Zealand stream algae. $2,950, BGSU Faculty Research Committee, July 1991.
18. Analyses of periphyton from the Procter & Gamble experimental stream facility. $15,000. The Procter & Gamble Company. April, 1991
19. Potential impact of climate change on northern temperate littoral algae. Consortium for International Earth Science Information Network. $30,000, Funded May, 1991.
20. The impact of zebra mussels on the biology of the benthic algae in Saginaw Bay, Lake Huron. $40,000. The Cooperative Institute for Limnology and Ecosystems Research, May 1991
21. Analyses of differential periphyton growth on tiles and cobble from the experimental stream facility. $15,000 The Procter & Gamble Co. August, 1991
22. The Impact of Zebra Mussels on the Benthic Food Web in Saginaw Bay, Lake Huron. National Oceanic and Atmospheric Administration, Sea Grant Program, $150,811 Sea grant 1992-1994.
23. Periphyton Response To Alkylethioxysulphate in Streams. The Procter and Gamble Company, $22,680, Sept. 1992
24. Environmental Monitoring and Assessment Using Periphyton as an Indicator of Stream Conditions, U. S. Environmental Protection Agency, $10,000, October, 1993
25. Periphyton Communities from the Big Darby Basin Project. The Nature Conservancy. $10,890, 1993-1994.
26. Spatial and temporal distribution of sponges (Spongillidae) in Lake Erie: Potential competitors with zebra mussels (*Dreissena polymorpha*). Ohio Sea Grant, $5,000, 1994. (Co-PI)
27. Occurrence and Distribution of Freshwater Sponges (Spongillidae) on Zebra Mussels in Lake Erie. Ohio Sea Grant, $5,000, 1994. (Co-PI)
28. Zebra mussel-mediated shifts in benthic algal communities in Saginaw Bay, Lake Huron. $140,387, Sea grant 1995-1997.
29. Ultrastructural analyses of periphyton from the Procter and Gamble Experimental stream facility. The Procter and Gamble Company, $2,500 1994.
30. 7th Intermittent Great Lakes Algal Foray. BGSU Faculty Development Committee, $268, 1994.
31. Benthic Gastropod Community in the Great Lakes: Regulating Mechanisms in a Perturbed System. BGSU Graduate College, Research Assistantship (submitted 1994)
32. Analyses of periphyton from the Procter and Gamble Experimental stream facility. The Procter and Gamble Company, $20,160 1994.

33. Periphyton Communities in Stream Ecosystems: an Analysis of the Disturbance-Grazer-Resource Control Model. National Science Foundation. $11,300, 1995-1996.
34. Analyses Of Periphyton From The Procter And Gamble Experimental Stream Facility. The Procter and Gamble Company, $11,250 1995.
35. Long-Term Periphyton Colonization Of The Procter & Gamble Experimental Stream Facility, The Procter And Gamble Company, $9,000, 1995.
36. Geographic Variation In Periphyton Community Sensitivity To Surfactants, The Procter And Gamble Company, $10,000 1995.
37. Great Lakes Aquatic Ecosystem Research Consortium, Ohio Board of Regents, $62,400. submitted 1995
38. The Effect Alcohol Ethoxylate On The Distribution Of Cells In Periphyton Mats From The Procter & Gamble Experimental Stream Facility. $4,800, November 1995.
39. The impact of sewage effluent an benthic algae of the Little Miami River, Ohio. The Procter & Gamble Company, $6,750, 1996.
40. Co. P. I. Jeff Miner. Numerical Analysis Workshop. BGSU Faculty Development Committee. $700, 1996
41. Impact of linear alkyl benzene sulfonate (LAS) on periphyton from The Procter & Gamble Experimental Stream Facility, including an analysis for the potential effects of over sampling. The Procter & Gamble Company, $13,500, 1996
42. The impact of linear alkyl benzene sulfonate (LAS) on fine structure of periphyton mats: an analysis algal mat fine structure following periphyton recovery from LAS exposure at The Procter & Gamble Experimental Stream Facility. The Procter & Gamble Company $10,000, 1996
43. Regulatory mechanisms of benthic macroinvertebrate community structure in western Lake Erie: direct and indirect effects of zebra mussels and large fish. Ohio Department of Natural Resources, Division of Wildlife, $3,725 (Co-PI. J. Miner), 1996
44. Impact of linear alkyl benzene sulfonate (LAS) on periphyton from The Procter & Gamble Experimental Stream Facility, including an analysis for the potential effects of over sampling, supplemental funding. The Procter & Gamble Company, $1,875, 1997
45. Zebra mussels as determinants of benthic macroinvertebrate community composition in western Lake Erie: A systematics approach to species responses. (Miner and Stewart Co-P.I.s) National Fish and Wildlife Foundation through Ohio Sea Grant. $13,500, 1997.
46. Impact Of C14/15-Alkyl Sulfate On The Sediment Community At The Procter & Gamble Experimental Stream Facility. The Procter & Gamble Company, $10,000, 1997-98
47. Benthic-Pelagic Coupling: Community Responses to Round Goby Predation on Zebra Mussels. Co-P.I. Ohio Sea Grant (NOAA) $118,966, 3/98-2/01
48. Impact Of C14/15-Alkyl Sulfate On The Sediment Community At The Procter & Gamble Experimental Stream Facility. The Procter & Gamble Company, $5,500, 1998-99

48. Impact of Alkyl Dimethyl Ammonium Chloride (DHAC) on Tile and Cobble Periphyton. The Procter & Gamble Company, $21,375, 1998-99
49. Impact of Alkyl Dimethyl Hydroxyethyl Ammonium Chloride (DHAC) on the Sediment Community at the Procter & Gamble Experimental Stream Facility. The Procter & Gamble Company, $10,000, 1998-99
50. Impact of Alkyl Dimethyl Ammonium Chloride (DHAC) on Tile and Cobble Periphyton. The Procter & Gamble Company, Contract supplement $4,500, 1999
51. Algal Analyses from the Pacific Northwest. U. S. Geological Survey. $14,280 4/99-5/00
52. Impact of High Solubility Alkyl Sulfate (HSAS) on Periphyton in The Procter and Gamble Experimental Stream Facility (ESF)$32,050 1999-00
53. Algal Analyses from the Pacific Northwest. U. S. Geological Survey. $3,000 99-00
54. Biodiversity and Biogeography of Diatoms (Bacillariophyceae) in New Zealand. National Science Foundation $19,340. 00-02
55. Impact Of Ramped vs. Instantaneous Exposure of Periphyton Linear Alkyl Sulfate at The Procter & Gamble Experimental Stream Facility. The Procter & Gamble Company, $25,550, 00-01
56. Biosphere-Atmosphere Research and Training, National Science Foundation Integrative Graduate Education and Research Training Program Grant (IGERT), BGSU is one of 12 participating institutions. $36,000, 01-03
57. Algal Biodiversity In The Great Smoky Mountains National Park. The National Science Foundation, $400,000, 10/1/03-10/1/07
58. Phytoplankton Sample Analysis In The Orange Creek Basin And Middle St. Johns River Basin, St Johns River Water Management District, $32,384, 10/1/03-12/31/04.
59. Phytoplankton comparison study, St. Johns River Water management District. $1,600, 8/20/03-12/31/03.
60. Cornerstone Experience. The National Science Foundation , $27,337, 8/03-8/04.
61. Phytoplankton Analysis of Historical Samples In The Orange Creek Basin And Middle St. Johns River Basin, St Johns River Water Management District, $20,565.22 10/1/03-12/31/04.
62. Algal Biodiversity in the Great Smoky Mountains National Park-Research Experience for Undergraduates Supplemental Funding Request. The National Science Foundation, $7,018.00, 5/1/04-5/1/05.
63. Phytoplankton Sample Analysis In The Middle St. Johns River Basin and Orange Creek Basin And, St Johns River Water Management District, $26,353, 10/1/03-12/31/04.
64. Algal Biodiversity in the Great Smoky Mountains National Park-Research Experience for Undergraduates Supplemental Funding Request. The National Science Foundation, $7,580.00, 5/1/05-5/1/06.
65. Phytoplankton In The Middle St. Johns River Basin and Orange Creek Basin, St Johns River Water Management District, $38,941, 10/1/05-12/31/06.

66. Phytoplankton analyses of communities in The Middle St. Johns River Basin and Orange Creek Basin, St Johns River Water Management District, $45,601, 10/31/06-10/31/07.
67. Freshwater Algae of Hawaii. (Co-PI) National Science Foundation, $980,000 January 2009-December 2011

V. Publications

A. Books and Monographs

1. Lowe, R. L. 1974. Environmental requirements and pollution tolerance of freshwater diatoms. EPA-670/4-74-007, 340 pp.
2. Stevenson, R. J., M. L. Bothwell and R. L. Lowe. 1996. Benthic Algal Ecology in Freshwater Ecosystems. Academic Press, San Diego, CA, 753pp.
3. Fučiková, K., Hall, J. D., Johansen, J. R. and Lowe, R. L. 2008. Desmid Flora of the Great Smoky Mountains National Park, USA. Bibliotheca Phycologica 113:1-59.
4. Furey, P. C., Lowe, R. L. and Johansen, J. R. 2011. *Eunotia* Ehrenberg (Bacillariophyta) of the Great Smoky Mountains National Park, USA. Bibliotheca Diatomologica, Band 56 132 pp, 2 tables, 34 plates.

B. Book Chapters

1. Pryfogle, P .A. and R.L. Lowe. 1979. Sampling and interpretation of epilithic lotic diatom communities. IN R. Weitzel, ed., Methods and Measurements of Attached Microcommunities: A Review. American Soc. Testing and Materials, pp. 77-89.
2. Lowe, R.L. 1980. Phytobenthic ecology and regulated streams. Proc. of 1st Int. Symp. on Regulated Streams. IN Stanford and Ward, eds., The Ecology of Regulated Streams, Plenum Press, NY, pp. 25-34.
3. Schelske, C. L. and Lowe, R.L. 1982. Algal nuisances and indicators of pollution. IN Selected Papers in Phycology II. Rosowski and Parker, eds., Phycol. Soc. Amer., Book Div., Lawrence, Kansas. pp. 799-819.
4. Stevenson, R. J. and Lowe, R.L. 1986. Sampling and interpretation of algal patterns for water quality assessments. IN B .G. Isom, Editor, Rationale for sampling and interpretation of ecological data in the assessment of freshwater ecosystems. A.S.T.M., STP894, 118-149.
5. Wallace, B. J., Webster, J. and Lowe, R.L. 1992 High-gradient streams of the Appalachians. IN Biodiversity of the Southeastern United States/Aquatic Communities. C. T. Hackney, S. M. Adams, and W. A. Martin (Eds.), John Wiley & Sons Inc., pp. 133-191.
6. Belanger, S. E., J. B. Barnum, D. M. Woltering, J. W. Bowling, R. M. Ventullo, S. D. Schermerhorn and R. L. Lowe 1994. Algal periphyton structure and function in response to consumer chemicals in stream mesocosms. IN Aquatic Mesocosm Studies in Ecological Risk Assessment. R. L. Graney, J. H. Kennedy and J. H. Rogers Eds., SETAC Special Publication Series, Lewis Publishers.

7. Johnson, D. W. and R. L. Lowe. 1995. Diatoms and the information "super highway". IN A Century of Diatom Research in North America. J. P. Kociolek Ed., Koeltz Scientific Books USA, Champaign, IL, p. 175-182.
8. Lowe, R. L. 1996. Periphyton patterns in lakes. *In* Stevenson, R. J., M. L. Bothwell and R. L. Lowe, (eds.) Benthic Algal Ecology in Freshwater Ecosystems. Academic Press, pp. 57-76.
9. Lowe, R. L and Y. Pan. 1996. Use of Benthic Algae in Water Quality Monitoring. *In* Stevenson, R. J., M. L. Bothwell and R. L. Lowe, (eds.) Benthic Algal Ecology in Freshwater Ecosystems. Academic Press
10. Lowe, R. L. and G. D. LaLiberte. 1996. Benthic Stream Algae: Distribution and Structure. *In* Lamberti, G. and F. R. Hauer, (eds.), Stream Ecology: Field and Laboratory Exercises. Academic Press, pp. 269-293.
11. Lowe, R. L. 2003 The Keeled genera of diatoms. *In* Sheath, B. and J. Wher, (eds.), Freshwater Algae of the United States. Academic Press, pp. 669-684
12. Lowe, R. L. and G. D. LaLiberte. 2006. Benthic Stream Algae: Distribution and Structure. *In* Lamberti, G. and F. R. Hauer, (eds.), Methods in Stream Ecology. Academic Press, pp. 327-356.
13. Lowe, R. L. 2006 Algology and algologists at Bowling Green, a short history. *In* Stevenson, R. J., Y. Pan and J. P. Kociolek (eds), Advances in Algal Biology: A Commemoration to the work of Rex Lowe, Hydrobiologia pp 1-11.
14. Greenwood, J. L. and Lowe, R. L. 2006 The effects of pH on a periphyton community in an acidic wetland, USA. . *In* Stevenson, R. J., Y. Pan and J. P. Kociolek (eds), Advances in Algal Biology: A Commemoration to the work of Rex Lowe, Hydrobiologia pp 71-82.
15. Lowe, R., Pillsbury, R. and Schrank, A. 2009. Aquatic ecosystems of northern Michigan. In *The Changing Environment of Northern Michigan.* Nadelhoffer, K., Hogg, A. and Hazlett, B. eds. University of Michigan Press, Ann Arbor, MI, USA pp 50-60.
16. J.P. Kociolek R.L. Lowe & E.F. Stoermer 2009. Algae of Northern Michigan and the Douglas Lake Region. In *The Changing Environment of Northern Michigan.* Nadelhoffer, K., Hogg, A. and Hazlett, B. eds. University of Michigan Press, Ann Arbor, MI, USA pp 85-92.
17. Thomas, E.W., J.P. Kociolek, R.L. Lowe and J.R. Johansen. 2009. Taxonomy, ultrastructure and distribution of gomphonemoid diatoms (Bacillariophyceae) from Great Smoky Mountains National Park (U.S.A.). *Nova Hedwigia, Beiheft* 135:201-237.
18. Lowe, R. L. 2011 The importance of scale in understanding the natural history of diatom communities. In *The Diatom World*, Seckbach, J. and Kociolek, J. P, eds. Springer, NY, NY. pp 293-311.

C. Published Journal Articles

1. Gale, W.F. and R.L. Lowe. 1971. Phytoplankton ingestion by the fingernail clam, Sphaerium transversum (Say), in Pool 19, Mississippi River. Ecology 52: 507-513.
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131. Porcupine Mountains Wilderness State Park, Michigan. Michigan Botanist.
132. Lowe Rex L., Kociolek, Patrick, Johansen, Jeff R., Van de Vijver Bart, Lange-Bertalot, Horst and Kopalová Kateřina (in press) *Humidophila* gen. nov., a new genus for a clade of diatoms (Bacillariophyta) formerly within the genus *Diadesmis*: species from Hawai'i, including one new species. Diatom Research.
133.

VI. Teaching

A. Formal courses

Bowling Green State University

1. Algology (30 years)
2. Limnology (35 years)
3. Environment of Life (10 years)
4. Great Lakes Ecosystems (12 years)* Taught through "Distance Learning" to three campuses.
5. Concepts in Biology I. (10 years)
6. Aquatic Ecology (8 years)
7. Diatom Systematics (5 years)

University of Michigan Biological Station

1. Algae of Freshwater Ecosystems (33 summers)

Ohio State University at Stone Lab

1. Phytoplankton IdentificationWorkshops (4 summers)

Central Michigan Biological Station on Beaver Island

1. Algology (1 summer)

Cass Field Station, New Zealand

1. Freshwater Algae Identification

University of Hawaii, Manoa

Limnology (1 year)

Michigan State University, Kellogg Biological Station

Algal Biology (1 summer)

B. Thesis and Dissertation Students Trained by R.L. Lowe

1. **J. Michael McCullough**, M. A., 1971. The effect of sewage-treatment-plant effluent on diatom communities in the North Branch of the Portage River, Wood County, Ohio. BGSU.
2. **Robert Reitz,** M. S., 1973. Phytoplankton periodicity in two Northwestern Ohio ponds. BGSU.
3. **Bill Brower**, M. S., 1973. Phytoplankton and periphyton diatom relationships in two highly eutrophic lakes. BGSU.
4. **David E. Busch**, M. S., 1974. Vertical and seasonal distribution of the Bacillariophyta in the Miller Blue Hole, Sandusky Co., Ohio. BGSU.
5. **Terrance L. Breyman**, M. S., 1974. Bangia in Western Lake Erie. BGSU.
6. **David C. Jackson**, M. S., 1975. Distribution and morphology of members of the diatom genera *Gyrosigma* Hassal and *Pleurosigma* W. Smith in the Portage River Drainage System. BGSU.
7. **Phillip A. Kline**, M. S., 1975. Survey of the phytoplankton of the Sandusky River at Fremont, Sandusky Co., Ohio. BGSU.
8. **Ronald J. Bockelman**, M. S., 1975. The seasonal productivity of zooplankton and benthic macroinvertebrate populations in six northwest Ohio ponds. BGSU.
9. **R**. **Jan Stevenson**, M. S., 1976. The periphytic diatoms of the Sandusky River. BGSU.
10. **P.A. Pryfogle**, M. S., 1976. Seasonal distribution of periphytic diatoms on natural substrates in Tymochtee Creek. BGSU.
11. **Keith Camburn**, M. S., 1977. The haptobenthic diatom flora of Long Branch Creek, South Carolina. BGSU.
12. **Frank Acker**, M. S., 1977. The phytoplankton of the Maumee River between Grand Rapids, Ohio and Maumee, Ohio. BGSU.
13. **J. L. Rohr**, M. S., 1977. Changes in diatom community structure due to environmental stress. BGSU.
14. **R. F. Andritsch**, M. S., 1977. Seasonal photosynthetic rates of *Chara globularis* in Steidtmann Pond. BGSU.
15. **Mary Bruno**, M. S., 1978. Distribution and periodicity of desmids and diatoms in a Northwestern Ohio bog lake. BGSU.
16. **David F. Millie**, M. S., 1979. An analysis of epiphytic diatom assemblages of three species of aquatic vascular plants in three Lake Erie marshes. BGSU.
17. **Robert Foster**, M. S., 1980. Selected toxic metal concentrations in several species of western Lake Erie fish with respect to age. BGSU.
18. **Daniel Z. Fisher**, M. S., 1980. Autumn periphyton and phytoplankton diatom communities in relation to depth and current velocity in the Maumee River, Ohio. BGSU.
19. **John C. Kingston**, Ph.D., 1980. Characterization of benthic diatom communities in Grand Traverse Bay, Lake Michigan. BGSU.
20. **Earl Chilton**, M. S., 1982. A comparison of macroscopic invertebrates living in *Bangia atropurpurea* and *Cladophora glomerata* beds in Lake Erie. BGSU.
21. **Pat Kociolek**, M. S., 1982. Diatoms from two streams in Great Smoky Mountains National Park. BGSU.
22. **Charles G. Maurice**, M. S., 1982. Effects of acidification on the periphyton of an artificial stream. BGSU.

23. **David R. Beeson**, M. S., 1982. Epiphytic diatom (Bacillariophyceae) community structure in a wetland continuum Sugar Island, Michigan. BGSU.
24. **Barry H. Rosen**, Ph.D., 1982. Physiological and ultrastructural responses to light intensity and nutrient limitation in the planktonic diatom *Cyclotella meneghiniana*. BGSU.
25. **Mark E. Lamb**, M. S., 1983. The effects of current velocity on the structuring of diatom communities. BGSU.
26. **Ann R. Miller**, M. S., 1983. Temporal and spatial relationships in the epipsammic diatom community. BGSU.
27. **Robert Genter,** M. S., 1983. The effects of different initial colonists on the outcome of periphyton succession in a small stream. BGSU.
28. **Kevin A. Karl**, M. S., 1983. The effects of fly ash extract on periphyton community structure in field enclosures. BGSU.
29. **Elaine D. Keithan**, Ph.D., 1983. Primary productivity and structure of phytolithic communities in streams in the Great Smoky Mountains National Park. BGSU
30. **Norlida Anis**, M. S., 1985. Effects of water chemistry on the distribution of diatom communities. BGSU.
31. **Mark Krejci**, Ph.D., 1985. Spatial patterns of epipsammic diatoms in a spring-fed brook with emphasis on the effect of sand grain mineralogy on diatom occurrence. BGSU.
32. **Hunter J. Carrick**, M. S., 1985. The response of Lake Michigan benthic algae to an in situ nutrient manipulation. BGSU.
33. **Gail Blake**, M. S., 1987. The effects of the agricultural herbicide alachlor on total biomass and community structure of algal periphyton in artificial streams. BGSU.
34. **Mark R. Luttenton**, Ph.D. 1989. In situ manipulation of factors affecting periphyton community structure. BGSU.
35. **Lisa E. Barnese**, Ph.D. 1989. A survey and experimental study of algal drift in the Maple River, Pellston, Michigan. BGSU.
36. **Douglas Deutschman**, M. S., 1990. Response of an algal community to temporal variability of resources. BGSU.
37. **Jane C. Marks**, M. S. 1990. The independent and interactive effects of nitrogen, phosphorus and light on structuring periphyton in Flathead Lake, Montana. BGSU.
38. **Craig D. Layne** M. S. 1990. The algal mat of Douglas Lake, Michigan: Its composition, role in lake ecology, and response to chemical perturbations. BGSU.
39. **Hudson DeYoe** Ph.D. 1991. Preliminary characterization of the relationship between *Rhopalodia gibba* (Bacillariophyceae) and its cyanobacterial endosymbiont. BGSU
40. **Susan Hardman** Ph.D. 1992. Environmental components influential in epipelic algal community structure. BGSU.
41. **James C. Sferra** M. S. 1992 Potential effects of the zebra mussel, *Dreissena polymorpha* (Pallas) on the Western Basin of Lake Erie. BGSU.
42. **Carmen Pedraza-Silva** M. S. 1992 A description of the algal floras of *Guzmania berteroniana* and *Vriesia sintenisii* (Bromeliaceae) and preliminary investigation of bromeliad-algal interactions.

43. **Robert W. Pillsbury** Ph.D. 1993. Factors influencing the structure of benthic algal communities in acid lakes. BGSU
44. **Diane Longanbach** M. A. T. 1993.
45. **Yangdong Pan** Ph.D. 1993. The effects of nutrients on periphyton. BGSU
46. **LouAnne Reich** M. S. 1994. An examination of Douglas Lake, Cheboygan County, Michigan as suitable habitat for the zebra mussel (*Dreissena polymorpha*): food quality and attachment site preference. BGSU
47. **Bret Gargasz** M. S. 1994. Non-thesis, plan II.
48. **Steven Francoeur** M. S. 1997. The effect of in-stream flow refugia on the recovery of stream periphyton communities following flooding disturbance.
49. **Sophia Passy** Ph.D. 1997. Ecology and systematics of the periphytic diatoms from the Mesta River system, Bulgaria.
50. **Joanne Rhoers** M. S. 1997. The Impact of the Crayfish Orconectes propinquis on Benthic Algae and Zebra Mussels".
51. **Rebecca Visnyai** M. S. 1997. Wetland restoration: the need to base restoration on function and landscape-level processes.
52. **Randy Litteral**, 1998. Benthic algal community structure and the compensation point.
53. **Jennifer L. Greenwood**, M. S. 1998. The effects of pH light on periphyton communities in a Michigan Wetland.
54. **Todd A. Clason**, M. S. 1999. Diurnal migration and community ultrastructure of benthic algae in Douglas Lake.
55. **Timothy Stewart**, Ph.D. 1999. Evidence and mechanisms for *Dreissena* effects on other benthic macroinvertebrates in western Lake Erie. (Co-advisor)
56. **Julianne Heinlein**, M. S. 2000. Flood disturbance mechanisms in stream periphyton: individual and interactive effects of shear stress perterbations and suspended sediment concentration.
57. **Agnieszka Pinowska**, Ph.D. 2001. Indirect effect of sediment nutrient enrichment on epiphytic algal communities.
58. **Amy Kireta**, M. S. 2001. Benthic algal shifts in response to the round goby.
59. **Wanda Gooden**, Ph.D. 2002. Periphyton responses to surfactants: Community structure and mat architecture
60. **Jennifer Ress**, M. S. 2003. Grazing efficiency of three aquatic grazers and their impact on periphyton recovery.
61. **Sarah Zeiler**, M.S. 2004. The ratio if periphyton to plankton under variable nutrient regmens in a fen peatland.
62. **Jennifer Wearly**, M.S. 2004. Changes in algal communities due to zebra mussel invasion of an oligotrophic inland lake.
63. **Jessie Knapp**, M.S. 2005. Effects of moss morphology on epiphyte diatom distribution in the Great Smoky Mountains National Park.
64. **Linda Novitski** M.S. 2006. The effects of chemical and physical characteristics on benthic algal community structure along an elevational gradient in Hazel Creek watershed, Great Smoky Mountains National Park.
65. **Evan Thomas** MS. 2007. The role of wave disturbance on lentic benthic algal community structure and diversity.
66. **Xiaozhao Song** MS. 2007
67. **Paula Furey PhD**. 2008. Taxonomy (morphology and ultrastructure) and ecology of the diatom genus *Eunotia* Ehrenberg (Bacillariophyta) of the Great

Smoky Mountains National Park, U.S.A. and seep-wall community response to nutrient and moisture manipulation at high and low altitudes.

68. Meg Woller-Skar Phd 2009. Zebra mussel promotion of cyanobacteria in oligotrophic lakes, and the subsequent production and fate of microcystin.
69. Phd**Ted Bambakidis**, MS 2009. The role of desiccation in structuring benthic diatom assemblages.
70. Jennifer Ress in progress,

MATT ROWLAND WHILES

Department of Zoology
Southern Illinois University
Carbondale, Illinois 62901-6501
Phone: (618) 453-7639

PERSONAL INFORMATION

Born December 4, 1964; Kansas City, Missouri.

Married 1997, 1 daughter and 1 son

EDUCATION

9/91-6/95 University of Georgia, Athens, Georgia; **Ph.D. Ecology**.
Dissertation: Disturbance, recovery, and invertebrate communities in southern Appalachian headwater streams.

9/88-9/91 University of Georgia, Athens, Georgia; **M.S. Entomology**.
Thesis: First-year recovery of a southern Appalachian headwater stream following an insecticide induced disturbance.

8/84-8/88 Kansas State University, Manhattan, Kansas; **B.S. Biology**.

AREAS OF SPECIALIZATION

Freshwater ecosystem ecology with an emphasis on the structure and function of streams and wetlands, the roles of consumers in ecosystem processes, ecological consequences of extinction, linkages between aquatic and terrestrial systems, disturbance ecology, and bioassessment and restoration of freshwater habitats.

PROFESSIONAL EXPERIENCE

2011- **Director**, SIUC Center for Ecology
Leading development of the SIUC Center for Ecology, a research center facilitating interdisciplinary ecological research and education at SIUC. Duties include overseeing operations of the Middle Mississippi River Wetlands Field Station.

2008- **Professor of Zoology**, Southern Illinois University
Teaching Freshwater Invertebrates, Stream Ecology, and General Ecology. Advising graduate research in freshwater ecosystem ecology.

PROFESSIONAL EXPERIENCE (continued)

2005-11 **Director**, Middle Mississippi River Wetlands Field Station
Led development of a field station for research and education on large river floodplain ecology.

2003-08 **Associate Professor of Zoology**, Southern Illinois University
Teaching Freshwater Invertebrates, Stream Ecology, and General Ecology. Advising graduate research in freshwater ecosystem ecology.

2000-03 **Assistant Professor of Zoology**, Southern Illinois University
Teaching Freshwater Invertebrates, Stream Ecology, and General Ecology. Advising graduate research in freshwater ecosystem ecology.

2000- **Adjunct Assistant Professor of Entomology**, Kansas State University
Serving as a graduate committee member for students pursuing studies in the area of aquatic invertebrate ecology

1997-00 **Assistant Professor of Entomology** (non-tenure track), Kansas State University
Taught Insect Ecology, Insects and People, Economic Entomology, and an interdisciplinary Environmental Concerns course. Advised graduate research in invertebrate ecology.

1996- **Adjunct Graduate Faculty**, University of Memphis
Graduate committee member for students working in aquatic ecology.

1995-97 **Assistant Professor of Biology**, Wayne State College
Taught Introductory Zoology, Invertebrate Zoology, Entomology, Vertebrate Zoology, Ecology, and General Biology (majors and non-majors). Advised undergraduate research in freshwater invertebrate ecology.

FUNDING

2013-15 Assessing recovery of tallgrass prairie headwater streams following experimental patch burn grazing treatments. **M. R. Whiles**. **$72,000**. Source: Missouri Department of Conservation.

2012 Center for Ecology/IDNR Undergraduate student internships 2012. C. Nielson, S. G. Baer, J. Groninger, and **M. R. Whiles**. Amount: **$12,353** Source: Illinois Department of Natural Resources.

2011-16 Scale, Consumers and Lotic Ecosystem Rates (SCALER): Centimeters to continents. Dodds, W.K., K. Gido, F. Ballantyne, A. Rosemond, J. Kominoski, **M. R. Whiles**, M. B. Flinn, B. Bowden, W. McDowell. Amount: **$3,304,097**, **SIU amount $267,469**. Source: NSF macrosystems

2011-12 RAPID: A massive floodplain reconnects: physical and biotic responses of the Birds Point levee breach in the Mississippi River. Garvey, J. E., D. Glover, **M. R. Whiles**, N. Pinter. Amount: **$74,994**. Source: NSF Ecosystems

2011-12 Stream biomonitoring program for the Shawnee National Forest. **M. R. Whiles**. Amount: **$3,000**, source: USDA Forest Service

FUNDING (continued)

2011	Development of a moveable bed river model campus-wide curriculum: enhancing graduate education in river science and engineering. Garvey, J, **M. R. Whiles**, L. Chevalier, and N. Pinter. Amount: **$3,000**. Source: SIU Graduate Technology Enhancement Grant.
2011	Center for Ecology/IDNR Undergraduate student internships 2011. Groninger, J., S. G. Baer, D. Gibson, and **M. R. Whiles**. Amount: **$12,353** Source: Illinois Department of Natural Resources.
2010-11	Workshop: Use of ^{15}N tracer addition datasets to quantify and synthesize relationships between stream biodiversity and ecosystem function across environmental and hydrologic gradients. **M. R. Whiles**, W. K. Dodds, and S. Johnson. Amount: **$47,790** Source: NSF
2010-13	Intensive basin survey assistance with the Illinois EPA. **M. R. Whiles**. Amount: **$54,747**, Source: IEPA
2010-14	Demonstrating the benefits of stream restoration to aquatic communities in the Cache River basin. H. Rantala, **M. R. Whiles**, and G. Wilkerson. Amount: **$399, 218** Source: IL State Wildlife Grant (US Fish and Wildlife Service).
2010-12	Training Illinois' future natural resource managers through a statewide natural areas internship Program. Baer, S. G., J. Groninger, D. Gibson, and **M. R. Whiles**. Amount: **$33,320** Source: Illinois Clean Energy Foundation.
2010	Center for Ecology/IDNR Undergraduate student internships 2010. Groninger, J., S. G. Baer, D. Gibson, and **M. R. Whiles**. Amount: **$12,353** Source: Illinois Department of Natural Resources.
2009-14	IGERT: Multidisciplinary, team-based training in watershed science and policy. N. Pinter, M. Davenport, **M. Whiles**, L. Chevalier, and C. Lant. Amount: **$3,179,181**. Source: NSF.
2009-14	Riparian influences on macroinvertebrate production in streams on the Konza Prairie Biological Station. **M. R. Whiles**. Amount: **$150,327**. Source: NSF: Konza Long Term Ecological Research (LTER) program subcontract.
2008-11	Demonstrating the benefits of in-stream restoration to riparian wildlife in the Cache River Basin. **M. R. Whiles** and C. Roy. Amount: **$111,786**. Source: IL State Wildlife Grant (US Fish and Wildlife Service).
2008-11	Illinois Conservation Opportunity Areas: Coordination and planning in support of the Illinois Wildlife Action Plan. **M. R. Whiles** and D. M. Day. Amount: **$383, 320**. Source: IL State Wildlife Grant (US Fish and Wildlife Service and Illinois Nature Conservancy).
2008-12	Biotic integrity of prairie streams as influenced by patch burn grazing and riparian protection. W. K. Dodds and **M. R. Whiles**. Total amount: **$267,294, SIU amount: $102,500**. Source: Missouri Department of Conservation.
2008-10	Continuation of a statewide natural land management internship program. S. G. Baer, D. J. Gibson, J. Groninger, and **M. R. Whiles**. Amount: **$56,266**. Source: Illinois Clean Energy Foundation.
2008-09	Center for Ecology IDNR internships. **M. R. Whiles**, D. Gibson, S. G. Baer. Amount: **$10,000**. Source: Illinois Department of Natural Resources.

FUNDING (continued)

2007-12 Collaborative research: Ecosystem level consequences of extinction: Quantifying the ecological effects of catastrophic amphibian declines in neotropical streams. **M. R. Whiles**, K. R. Lips, C. Pringle, and S. Kilham. **Total amount: $710,000, SIU amount: $364,900**. Source NSF Ecosystems

2006-07 Assessing the ecological consequences of amphibian declines: emergency sampling along a moving disease front in Panama. **M. R. Whiles** and K. R. Lips. Amount: **$75,136**. Source: NSF Ecosystems

2006-08 Evaluation of soils and associated belowground resources in central Platte River slough wetland restorations. **M. R. Whiles**, S. G. Baer, and C. K. Meyer. Amount **$33,500**. Source: NE Game and Parks and USFWS.

2006-07 Analysis of water quality and quantity in the Kaskaskia River watershed K. Williard, **M. R. Whiles**, and J. Schoonover. Amount: **$30,000**. Source: Illinois Southwest Resource Conservation and Development Area, Inc.

2006-07 Center for Ecology IDNR internships. D. Gibson, **M. R. Whiles**, S. G. Baer. Amount: **$10,000**. Source: Illinois Department of Natural Resources.

2006-09 Statewide natural land management internship program. D. Gibson, **M. R. Whiles**, and S. G. Baer. Amount: **$43,734**. Source: Illinois Clean Energy Foundation

2006-08 Development of a hydrologic monitoring network for the Middle Mississippi River Wetland Field Station. **M. R. Whiles**, S. G. Baer, L. Battaglia, J. E. Garvey, E. Hellgren, K. Williard, and G. Whitledge. Amount: **$54,554**. Source: SIU Office of Research and Development (interdisciplinary seed grant).

2005 Center for Ecology/IDNR undergraduate internships. D. Gibson, **M. R. Whiles**, and S. G. Baer. Amount: **$10,000**. Source: Illinois Dept. of Natural Resources.

2004-08 Interactive effects of disturbance frequency and species composition on ecosystem functioning of intermittent streams: a test of future climate change scenarios. K. Gido, W. K. Dodds, and **M. R. Whiles**. **Total: $320,000, SIU: $138,700**. Source: NSF Ecology.

2004-08 Cycling of novel allochthonous carbon in midwestern agricultural streams. J. Tank, E. Rosi-Marshall, T. V. Royer, and **M. R. Whiles**. **Total amount: $530,000. SIU: $117,700**. Source NSF Ecosystems.

2003-07 Collaborative research: response of tropical stream ecosystem structure and function to amphibian declines. **M. R. Whiles**, K. Lips, C. Pringle, S. Kilham. **Total: $800,000, SIU: $400,000**. Source: NSF Ecosystems.

2003-04 Distribution, abundance, and dynamics of stream macroinvertebrates: metapopulation dynamics and metacommunity structure. D. J. Gibson, **M. R. Whiles**, and S. L. Collins. Amount: **$6,000**. Source: NSF LTER.

FUNDING (continued)

2003-06 Spatial and temporal relationships among nutrients, dissolved oxygen, and biotic integrity and controls on dissolved reactive phosphorus and particulate phosphorus in Illinois streams. M. B. David, T. V. Royer, **M. R. Whiles**, R. G. Darmody, G. F. McIssac. **Total: $475,000, SIU: $91,494**. Source: Illinois Council For Agricultural Research.

2003-06 Biological evaluation of central Platte River slough wetland restorations. **M. R. Whiles**, B. S. Goldowitz, S. G. Baer, and C. K. Meyer. Amount: **$41,332**. Source: US Fish and Wildlife Service

2003-04 Conservation assessment of sensitive reptiles and amphibians in the Shawnee National Forest. K. Lips and **M. R. Whiles**. Amount: **$20,000**. Source: USDA Forest Service.

2002-04 Fitting metapopulation models of community structure to stream macroinvertebrate data: issues and problems of taxonomic resolution. D. Gibson and **M. R. Whiles**. Amount: **$6,000**. Source: National Science Foundation LTER program.

2002-05 Demonstrating the biological benefits of stream channel restoration in the Cache River, southern Illinois. M. Geutersloh, **M. R. Whiles**, and D. Hankla. **Total: $320,000, SIU: $58,000**. Source: Illinois C2000 Ecosystems Program.

2002-08 Riparian influences on macroinvertebrate production in Kings Creek. **M. R. Whiles**. Amount: **$140,927**. Source: NSF: Konza Long Term Ecological Research (LTER) program subcontract.

2002-04 Environmental assessment of the Sparta Illinois National Guard training facility. J. E. Garvey, R. C. Heidinger, **M. R. Whiles**, and M. Lydy. Amount: **$230,406**. Source: Illinois Dept. of Military Affairs.

2002-03 Evaluation of macroinvertebrate diversity and productivity in central Platte River slough restorations. **M. R. Whiles**. Amount: **$20,000**. Source: Platte River Trust and USFWS.

2002-03 Evaluation of Environmental Pool Management in Mississippi River Navigation Pool 25. R. Sheehan, **M. R. Whiles**, B. Dugger, B. Burr. Amount: **$67,000**. Source: US Army Corps of Engineers.

2001-02 Assessment of the aquatic resources of the Shawnee and Hoosier National Forests. B. M. Burr and **M. R. Whiles**. Amount: **$30,000**. Source: USDA Forest Service.

2001-02 Fish, invertebrate, and water quality responses to environmental pool management: Mississippi River pool 25. R. J. Sheehan, B. Dugger, B. M. Burr, S. Reid Adams, and **M. R. Whiles**. Amount: **$66,446**. Source: US Army Corps of Engineers.

2000-02 Biological response to Newbury Weir construction in Big Creek, Illinois. **M. R. Whiles**. Amount: **$10,808**. Source: Illinois Dept. of Natural Resources.

2001-02 Diets of larval and adult *Ambystoma cingulatum*. **M. R. Whiles**. Source: USFWS. Amount: **$3,000**.

2000-01	Land management influences on biological diversity of the Smoky Hills KSARNG training facility. Charlton, R. E., **M. R. Whiles**, and J. Cully. Amount: **$98,000**. Source: Kansas Army National Guard.
2000-01	Influence of riparian forests on water quality, in-stream habitat, and biotic integrity in agriculturally impacted southern Illinois streams. **M. R. Whiles** and K. Williard. Amount: **$20,000**. Source: SIU Research and Development.
2000-01	Aquatic insect diversity and productivity in Cache River wetlands. **M. R. Whiles**. Amount: **$3,000**. Source: Illinois Nature Conservancy.
1999-01	Quantity and quality of suspended particles in the Kansas River system: demonstrating the influence of land management practices. W. K. Dodds and **M. R. Whiles**. Amount: **$31,323**. Source: Kansas Dept. of Health and Environ.
1999-00	Assessment of physicochemical, biological, and landscape features influencing Topeka shiner distribution in Kansas streams. Guy, C., and **M. R. Whiles**. Amount: **$18,000**. Source: USFWS.
1999-00	Integrated natural resource plan and land condition trend analysis for the Smoky Hills KSARNG training facility. Charlton, R. E., **M. R. Whiles**, J. Cully, and D. Kaufman. Amount: **$119,206**. Source: Kansas Army National Guard
1998-00	Influence of golf course construction and management on aquatic invertebrate, amphibian, and fish communities of the Colbert Hills area. **M. R. Whiles** and R. Charlton. Amount: **$62,000**. Source: KSU Institutional Advancement
1998-01	Influence of military training activities on aquatic communities in Fort Riley streams. R. E. Charlton and **M. R. Whiles**. Amount: **$77,000**. Source: Oak Ridge Institute for Science and Education.
1997-98	Biotic inventory in support of land condition trend analysis on KSARNG training area, Salina KS. R. Charlton, J. Cully, G. Kaufman, D. Kaufman, and **M. R. Whiles**. Amount: **$114,909**. Source: Kansas National Guard
1997	Development of an Aquatic Entomology/Bioassessment teaching and research laboratory. **M. R. Whiles** and R. Charlton. Amount: **$2,615,** Source: KSU Agricultural Experiment Station
1996-98	Hydrologic connectivity between river discharge, aquatic habitats, and biota of the middle Platte River. B. S. Goldowitz and **M. R. Whiles**. Amount: **$50,000**. Source: USEPA
1996-97	Use of invertebrate bioassessment techniques for assessing aquatic ecosystem health in the Willow Creek Reservoir drainage system. **M. R. Whiles** and R. Wozniak. Amount: **$15,000**. Sources: USEPA and Nebraska DEQ

HONORS AND AWARDS

2012	Invited Keynote Speaker, Odum School of Ecology Graduate Student Symposium, University of Georgia
2011	Outstanding Scholar, SIUC College of Science
1997	Professor of the Year, Math and Sciences Division, Wayne State College.
1996	Professor of the Year, Math and Sciences Division, Wayne State College.
1995	Outstanding Teaching Assistant, University of Georgia.
1994-1995	University-Wide Assistantship Award, University of Georgia.

1994-1995 Merit Assistantship Award; Outstanding Teaching and Research, Univ. of GA.
1993-1994 Merit Assistantship Award; Outstanding Teaching and Research, Univ. of GA
1987 Hydrolab Award; best poster, North American Benthological Society meetings

PROFESSIONAL PUBLICATIONS – PEER REVIEWED ARTICLES

Heinrich, K. K, **M. R. Whiles**, and C. Roy. *In press*. Cascading ecological responses to an in-stream restoration project in a Midwestern River. *Restoration Ecology:*

Scholl, E. A., H. M. Rantala, **M. R. Whiles**, and G. V. Wilkerson. *In Press*. A quantitative framework for predicting biological responses to a proposed river restoration. *Freshwater Science*:

Vanderymyde, J. M. and **M. R. Whiles**. *In press*. Effects of woody vegetation encroachment and experimental forest removal on macroinvertebrate production and functional structure in tallgrass prairie streams. *Freshwater Science*:

Mountjoy, N. J., E. Seekamp, M. Davenport, and **M. R. Whiles**. *In press*. The best laid plans: CBNRM group capacity and resource management planning success. *Environmental Management*:

Wilkerson, G. V., D. R. Kandel, L. A. Perg, W. E. Dietrich, P. R. Wilcock, and **M. R. Whiles**. *In revision*. Continental-scale relationships between bankfull width and drainage area for single-thread alluvial channels. *Water Resources Research:*

Mountjoy, N. J., **M. R. Whiles**, M. Davenport, and E. Seekamp. *In press*. Identifying the Top Ten Capacity Indicators for Community-based Natural Resource Management Initiatives: Focus Group Results from Conservation Practitioners across Illinois. *Journal of Environmental Planning and Management:*

Whiles, M. R. 2013. Freshwater Invertebrate Ecology. Oxford Bibliographies in Ecology, Oxford University Press. DOI: 10.1093/OBO/9780199830060-0132

Frauendorf T. C., C. Colon-Gaud, **M. R. Whiles**, T. R. Barnum, K. R. Lips, C. M. Pringle, and S. S. Kulham. 2013. Energy flow and trophic basis of macroinvertebrate and amphibian production in a neotropical headwater stream. *Freshwater Biology:*58:1340-1352.

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DiRenzo, G. V. R. Brenes, **M. R. Whiles**, C. M. Pringle, S. S. Kilham, and K. R. Lips. Submitted. Community disassembly as a homogenizing process: Tropical stream tadpole community dynamics before and after a fungal disease invasion. Submitted to: *Ecology*

Barnum, T. R., J. M. Drake, C. Colón-Gaud, A. T. Rugenski, T. C. Frauendorf, S. Connelly, S. S. Kilham, **M. R. Whiles**, K. R. Lips, and C. M. Pringle. Submitted. Food web structure persists after amphibian extirpation in a Neotropical stream. Submitted to: *Ecology Letters*

Connelly, S., C. M. Pringle, M. Hunte-Brown, T. Barnum, S. Kilham, **M. R. Whiles**, K. R. Lips, and C. Colon-Gaud. *Submitted*. Initial versus long-term effects of tadpole extirpations on algal resources and nitrogen cycling in a neotropical stream. Submitted to *Freshwater Biology*

Montgomery, C., P. Verburg, **M. R Whiles**, C. M Pringle, S. S Kilham, J. M Ray, M. Hunte-Brown, K. R Lips. *Submitted*. Responses of riparian and terrestrial snake predators to a catastrophic amphibian decline in Central America. Submitted to: *Biological Conservation*

Rantala, H. E., D. Glover, J. E. Garvey, Q. Phelps, D. Herzog, R. Hrabik, J. Crites, and **M. R. Whiles**. Submitted. Ecological responses to a floodplain reconnection during the 2011 Mississippi River flood. Submitted to: *River Research and Applications*

Rantala, H., E. Scholl, **M. R. Whiles**, and G. Wilkerson. *Submitted*. Predicting the ecological benefits of stream reconnection: a case study of the Cache River, Illinois. Submitted to: *Environmental Management*

Rothert, A., A. Harwood. P. Landrum, **M. R., Whiles**, and M. J. Lydy. *Submitted*. Examining the role of the rapidly desorbing fraction in the bioavailability of sediment-associated permethrin. Submitted to: *Environmental Toxicology and Chemistry*.

ORAL PRESENTATIONS (since 2007 only)

Whiles, M. R. 2013. Distribution and ecosystem significance of invertebrate decomposers in freshwater habitats. Invited special session speaker, Annual meetings of the Entomological Society of America, Austin, TX.

Muria, C., A. T. Rugenski, M. R. Whiles, and A. P. Vogler. 2013. Whole community DNA bar coding of aquatic insects reveals high endemism and limited movement among neotropical highland streams. International Biogeographical Society meeting, Montreal.

Whiles, M. R. 2013. Assessing the ecological consequences of declining freshwater biodiversity: impacts of catastrophic amphibian declines on Central American streams. Invited seminar speaker, Kansas State University Division of Biology.

W. K. Dodds, K. Gido, **M. R. Whiles**, M. D. Daniels, and N. B. Grimm. 2012. The unique qualities and global significance of grassland streams. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Rugenski, A. T., **M. R. Whiles**, M. J. Vanni, K. R. Lips, C. M. Pringle, and S. S. Kilham. 2012. Influence of amphibian declines in Panamanian headwater streams on leaf decomposition processes and consumer stoichiometry. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Rantala, H. M., E. A. Scholl, A. K. Kennington, **M. R. Whiles**, and G. Wilkerson. 2012. The effects of hydrologic fragmentation on stream ecosystem processes in the Cache River, IL, Illinois Water Conference, Springfield, Illinois, 24-25 September.

Rantala, H. M., E. A. Scholl, and **M. R. Whiles**. 2012. The importance of spatial scale for assessing ecological responses to stream restoration. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Scholl, E. A., H. M. Rantala, **M. R. Whiles**, and G. V. Wilkerson. 2012. Predicting macroinvertebrate responses to a hydrologic restoration in a southern Illinois river. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Martí, E., J. L. Tank, T. Riis, P. J. Mulholland, W. K. Dodds, **M. R. Whiles**, B. M. Cheever, S. M. Collins, C. L. Crenshaw, T. A. Crowl, N.A. Griffiths, N.B. Grimm, S.K. Hamilton, S.L. Johnson, W.H. McDowell, E.J. Rosi-Marshall, K.S. Simon, S.A. Thomas, J.R. Webster, W.M. Wollheim. 2012. Unpacking the black box of in-stream assimilatory nitrogen uptake: insights from 15N ammonium tracer additions conducted in different biogeoclimate regions. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Vandermyde, J. M. and **M. R. Whiles**. 2012. Effects of riparian woody vegetation removal on macroinvertebrate production and functional structure in tallgrass prairie streams. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Rantala, H. M., E. A. Scholl, A. K. Kennington, **M. R. Whiles**, and G. Wilkerson. 2012. Effects of stream diversion on oxygen dynamics and macroinvertebrate community structure and function in a southern Illinois agricultural watershed. Mississippi River Research Consortium, La Crosse, Wisconsin, 26-27 April 2012.

Rantala, H. M., E. A. Scholl, T. Egdorf, A. K. Kennington, **M. R. Whiles**, and G. Wilkerson. 2012. Effects of hydrologic fragmentation on stream ecosystem structure and function, Cache River, IL. American Fisheries Society, North Central Division, Rivers and Streams Technical Committee Meeting, Milan, Illinois, 26 March 2012.

H. M. Rantala, E. Scholl, **M. R. Whiles**, and G. V. Wilkerson, 2011. Effects of hydrologic alteration on macroinvertebrate community structure and ecosystem metabolism in a southern Illinois river. Midwest fish and wildlife conference, Des Moines, Iowa, 4-7 December 2011.

Whiles, M. R. 2011. Assessing the ecological consequences of declining freshwater biodiversity: impacts of catastrophic amphibian declines on Central American streams. Invited Keynote Address, University of Georgia Odum School of Ecology Graduate Student Forum.

Whiles, M. R. 2011. Assessing the ecological consequences of amphibian declines in Central America. Invited Keynote Speaker, Illinois Junior Science and Humanities Symposium, Southern Illinois University, Carbondale, IL

Frauendorf, T. C, Colon-Gaud, J. C, **Whiles, M. R**, Lips, K. R, Pringle, C. M, Kilham, S. S. 2011. Trophic basis of production in a neotropical headwater stream. Annual meetings of the North American Benthological Society, Providence, RI.

Whiles, M. R, Lugthart, G. J, Cuffney, T. F. 2011. Sans bugs: invertebrate removal manipulations demonstrate the significance of invertebrates to ecosystem functioning of headwater streams. Invited special session speaker, Annual meetings of the North American Benthological Society, Providence, RI.

Rugenski, A. T, Whiles, M. R, Vanni, M. J, Lips, K. R, Pringle, C. M, Kilham, S. S. 2011. Ecological stoichiometry of macroinvertebrates in neotropical streams before and after amphibian declines: are biodiversity losses altering stream ecosystem processes? Annual meetings of the North American Benthological Society, Providence, RI.

Whiles, M. R. 2011. Tallgrass prairie herpetofauna and invertebrate communities. Invited speaker. Missouri Department of Conservation Grassland Review. Aug. 3, Nevada, MO.

Scholl, E. A, Rantala, H. M, Whiles, M. R, Wilkerson, G. V. 2011. Infuence of water velocity on snag-dwelling invertebrates in a southern Illinois river. Annual meetings of the North American Benthological Society, Providence, RI.

Vandermyde, J. M, Whiles, M. R. 2011. Impact of woody vegetation removal on tallgrass prairie stream macroinvertebrate communities. Annual meetings of the North American Benthological Society, Providence, RI.

Rantala, H. M, **Whiles, M. R**, Wilkerson, G. V. 2011. Effects of stream diversion on ecosystem metabolism in a southern Illinois agricultural watershed. Annual meetings of the North American Benthological Society, Providence, RI.

Heinrich, K. K, **Whiles, M. R**, Roy, C. 2011. In-stream restoration enhances aquatic subsidies to riparian food webs: insect emergence production and riparian bird responses to rock weir construction. Annual meetings of the North American Benthological Society, Providence, RI.

Rantala, H., E. Scholl, **M. R. Whiles**, and G. Wilkerson. 2011. Effects of Hydrologic Alteration on Macroinvertebrate community structure and ecosystem metabolism in a Southern Illinois river. Invited symposium presentation, Midwest Fish and Wildlife Conference, Des Moines, IA.

Whiles, M. R. 2010. Adventures in stream ecosystem ecology: from tallgrass prairies to Central American cloud forests. Invited seminar speaker, Virginia Tech University, Department of Biology

Kilham, S. S., P. Verburg, C. Pringle, **M. R. Whiles**, and K. R. Lips. 2010. Stable isotope niches of predatory stream macroinvertebrates in a neotropical headwater stream. Society of International Limnologists meetings, Cape Town, South Africa

Whiles, M. R., H. Rantala, D. Walther, and J. C. Colon-Gaud. 2010. Local and regaional factors influencing aquatic invertebrate production. Invited special symposium talk, annual meetings of the Entomological Society of America, San Diego, CA.

Vandermyde, J. M., **M. R. Whiles**, S. G. Baer, D. J. Gibson, J. Groninger, and J. Shimp. 2010. Importance of internship opportunities for undergraduate students; SIU as a model program. Natural Areas Association conference, Branson, MO.

Whiles, M. R. 2010. Assessing the ecological consequences of declining freshwater biodiversity: impacts of catastrophic amphibian declines on Central American streams. Invited seminar presentation, Virginia Tech University Department of Biology.

Whiles, M. R., P. Verburg, W. K. Dodds, R. O. Hall, A. D. Huryn, A. T. Rugenski, S. Peterson, S. S. Kilham, K. R. Lips, C. M. Pringle, S. Connelly, E. Griffiths, H. Ross. 2010. Use of a ^{15}N tracer to quantify changes in nitrogen cycling associated with a massive amphibian decline in a tropical headwater stream. Annual meetings of the North American Benthological Society, Santa Fe, NM.

Barnum, T. R., C. M. Pringle, S. S. Kilham, K. R. Lips, C. Colon-Gaud, T. C. Frauendorf, and **M. R. Whiles**. 2010. Structure of an algal-based food web in a neotropical stream before and after amphibian extirpation. Annual meetings of the North American Benthological Society, Santa Fe, NM.

Benke, A. C. and **M. R. Whiles**. 2010. Life tables vs. secondary production analyses – relationships and usage in ecology. Annual meetings of the North American Benthological Society, Santa Fe, NM.

Frauendorf, T. C., C. Colon-Gaud, T. R. Barnum, T. R., **M. R. Whiles**, K. R. Lips, K. R., C. M. Pringle, C. M., and S. S. Kilham, 2010. Trophic basis of production in a neotropical headwater stream: implications for the ecological consequences of amphibian declines. Annual meetings of the North American Benthological Society, Santa Fe, NM.

Rugenski, A. T., **M. R. Whiles**, M. J. Vanni, K. R. Lips, C. M. Pringle, and S. S. Kilham. 2010. Quantifying the roles of larval amphibians in tropical stream nutrient cycling: are amphibian declines altering ecosystem processes? Annual meetings of the North American Benthological Society, Santa Fe, NM.

Vandermyde, J. M., and **M. R. Whiles**. 2010. Macroinvertebrate responses to removal of riparian woody vegetation along tallgrass prairie streams. Annual meetings of the North American Benthological Society, Santa Fe, NM.

Rantala, H. E. Scholl, **M. R. Whiles**, and G. Wilkerson. 2010. Predicting ecological responses to reconnection of the Cache River. Cache River Symposium, Vienna, IL

Heinrich, K., **M. R. Whiles**, and C. Roy. 2010. Biological responses to in-stream restoration in the upper Cache River. Cache River Symposium, Vienna, IL

Heinrich, K., **M. R. Whiles**, and C. Roy. 2010. Biological responses to in-stream restoration in the upper Cache River. Illinois Water 2010 conference, Champaign, IL.

Whiles, M. R. 2009. Quantifying the ecological consequences of declining biodiversity: responses of stream invertebrates to changes in vertebrate diversity. Invited symposium presentation, Annual Meetings of the Entomological Society of America, Indianapolis, IN.

Whiles, M. R., H. Rantala, D. Walther, and K. Heinrich. 2009. Biological responses to constructed rock riffles in the upper Cache River, Illinois. Invited symposium presentation, Midwest Fish and Wildlife Conference, Springfield IL.

Whiles, M. R. 2009. Assessing the ecological consequences of amphibian declines. Invited seminar speaker. Cary Institute of Ecosystem Studies.

Zandona, E., S. S. Kilham, C. M. Pringle, M. Marshall, R. El-Sabaawi, S. Thomas, A. Flecker, **M. R. Whiles**, K. Lips, and D. Reznick. 2009. Stable isotope analyses as a tool in conservation biology. Annual Meetings of the Society For Conservation Biology, Beijing, China.

Bertrand, K. N., **M. R. Whiles**, K. B. Gido, and J. N. Murdock. 2009. Trajectories of invertebrate community development following disturbances in an intermittent prairie stream. Annual Meetings of the American Fisheries Society, Nashville, TN.

Colon-Gaud, C., **M. R. Whiles**, S. S. Kilham, K. R. Lips, C. M. Pringle, S. Connelly, and S. D. Peterson. 2009. Macroinvertebrate community responses to catastrophic amphibian declines in neotropical streams: Potential for functional redundancy among tadpoles and grazing mayflies. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Myers, D. J., **M. R. Whiles**, and G. W. Whitledge. 2009. Use of $^{\delta}$D and $^{\delta 18}$O to trace the origins and movements of macroinvertebrates in large river floodplain water bodies. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Whiting, D. P., and **M. R. Whiles**. 2009. Macroinvertebrate production and food web structure along a tallgrass prairie stream continuum. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Peterson, S. D., **M. R. Whiles**, M. I. Gladyshev, K. J. Regester, N. N. Sushchik, and O. N. Makhutova. 2009. Use of fatty acid analyses to assess trophic relations of omnivorous larval amphibians. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Schultheis, R. Eichholz, M. W., **Whiles, M. R**., and T. Yerkes. 2009. Invertebrate availability in flooded agricultural habitats of the upper Midwest. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Schultheis, R. Eichholz, M. W., T. Yerkes, and **M. R. Whiles**. 2009. Patterns in invertebrate availability for waterfowl consumption during the spring migratory period. North American Duck Symposiun, Toronto.

Whiles, M. R. 2008. Biological responses to hydrologic disturbance and restoration activities in the central Platte River valley, Nebraska. Invited seminar speaker, Truman State University.

Tank, J. L., T. V. Royer, E. J. Rosi-Marshall, T. C. frauendorf, N. A. Griffiths, M. L. Stephen, and **M. R. whiles.** 2008. The widespread occurrence of toxin from transgenic corn in a stream network of an agricultural landscape. Annual Meetings of the American Society of Limnologists and Oceanographers.

Whiles, M. R., K. R., Lips, S. Kilham, C. Pringle, P. Verburg, R. Brenes, S. Connelly, C. Colon-Gaud, M. Hunte-Brown, C. Montgomery, and S. Peterson. 2008. Effects of amphibian population declines on the structure and function of neotropical streams. Invited special symposium presentation, American Society of Ichthyologists and Herpetologists, Montreal.

Anderson, R. V., T. J. Newton, **M. R. Whiles**, and J. W. Grubaugh. 2008. 100 years of macroinvertebrate sampling on the Mississippi River: what have we learned? Annual meetings of the Mississippi River Research Consortium, LaCrosse, WI.

Whiting, D. P. and **M. R. Whiles**. 2008. Macroinvertebrate production and functional structure along a tallgrass prairie stream continuum. Annual meetings of the North American Benthological Society, Salt Lake City.

Whiles, M. R., P. Verburg, W. K. Dodds, R. O. Hall, A. D. Huryn, S. S. Kilham, K. R. Lips, C. M. Pringle, S. Connelly, E. Griffith, and H. Ross. 2008. Use of a ^{15}N tracer addition to assess the ecosystem-level significance of amphibians and their extirpations in neotropical headwater streams. Annual meetings of the North American Benthological Society, Salt Lake City.

Griffiths, N. A., J. L. Tank, E. J. Rosi-Marshall, T. V. Royer, **M. R. Whiles**, J. J. Beaulieu, L. T. Johnson, S. S. Roley, and M. L. Stephen. 2008. Inter- and intra-system variation in whole-stream metabolism within an agricultural landscape. Annual meetings of the North American Benthological Society, Salt Lake City.

Royer, T. V., A. R. Childers, M. B. David, and **M. R. Whiles**. 2008. Towards the development of nutrient standards: Relationships among nutrients, algal biomass, and dissolved oxygen in illinois streams and rivers. Annual meetings of the North American Benthological Society, Salt Lake City.

Kilham, S. S., P. Verburg, C. M. Pringle, **M. R. Whiles**, and K. R. Lips. 2008. How important is omnivory in tropical stream food webs? Annual meetings of the North American Benthological Society, Salt Lake City.

Colón-Gaud, C., **M. R. Whiles**, S. S. Kilham, K. R. Lips, C. M. Pringle, S. Connelly, and S. D. Peterson. 2008. Food web structure and energy flow in panamanian headwater streams: Assessing responses to catastrophic amphibian declines. Annual meetings of the North American Benthological Society, Salt Lake City

Connelly, S. J. C. M. Pringle, S. S. Kilham, P. Verburg, **M. R. Whiles**, K. R. Lips, and J. Checo Colon-Gaud. 2008. Changes in quality and quantity of stream algae and seston resulting from neotropical amphibian declines. Annual Meetings of the Ecological Society of America, Milwaukee.

Kilham, S. S., P. Verburg, C. M. Pringle, S. J. Connelly, **M. R. Whiles**, and K. R. Lips. 2008. Comparisons of niche diversification in an upland neotropical stream food web pre- and post- frog extinction. Annual Meetings of the Ecological Society of America, Milwaukee.

Whiles, M. R. 2007. Biological responses to hydrologic gradients and restoration activities in the central Platte River valley, Nebraska. Invited seminar speaker, Washington University Tyson Research Center.

Whiles, M. R. 2007. Quantifying the ecological consequences of amphibian declines. Invited seminar speaker, Kansas State University Division of Biology.

Meyer, C. K., S. G Baer, and **M. R Whiles**. 2007. Belowground carbon and nitrogen dynamics following wet meadow restoration. Annual Meetings of the Ecological Society of America, San Jose, CA.

Connelly, S. J., C. M. Pringle, K. R. Lips, S. S. Kilham, **M. R. Whiles**, R. Brenes, and J. C. Colon-Gaud. 2007. Neotropical tadpole extirpations cause long-term changes in periphyton quantity and quality. Annual Meetings of the Ecological Society of America, San Jose, CA.

Kilham, S. S., M. Hunte-Brown, P. Verburg, C. M. Pringle, **M. R. Whiles**, and K. R. Lips. 2007. Challenges for interpreting stable isotope fractionation of carbon and nitrogen in tropical freshwater ecosystems. Annual Meetings of the Society of International Limnologists, Montreal.

Marshall, E. J., J. Tank, M. Evans-White, T. V. Royer, **M. R. Whiles**, J. Pokelsek, N. Griffiths, and C. Chambers. 2007. Crop residues and stream carbon budgets: examining the ecological significance of allochthonous carbon in Midwestern agricultural streams. Annual meetings of the American Soceity of Limnology and Oceanography, Santa Fe, NM.

Bertrand, K. N., J. N. Murdock, K. B. Gido, W. K. Dodds, and **M. R. Whiles**. 2007. Fishes and Floods in Prairie Streams. Annual Meetings of the North American Benthological Society, Columbia, SC.

Griffiths, N. A., J. L. Tank, T. J. Warrner, T. V. Royer, T. C. Frauendorf, C. P. Chambers, J. D. Pokelsek, M. A. Evans-White, E. J. Rosi-Marshall, and **M. R. Whiles**. 2007. The contribution of corn detritus to metabolism and carbon cycling in Midwestern agricultural streams. Annual Meetings of the North American Benthological Society, Columbia, SC.

Royer, T. V., M. B. David, L. E. Gentry, C. A. Mitchell, K. M. Starks, and **M. R. Whiles**. 2007. Is Chlorophyll the Appropriate Indicator of Nutrient Enrichment in Illinois Streams? Annual Meetings of the North American Benthological Society, Columbia, SC.

Colon-Gaud, C., **M. R. Whiles**, S. S. Kilham, K. R. Lips, and C. M. Pringle. 2007. Secondary production in four Panamanian highland streams: Assessing macroinvertebrate responses to catastrophic amphibian declines. Annual Meetings of the North American Benthological Society, Columbia, SC.

Warrner, T. J., T. V. Royer, N. A. Griffiths, M. A. Evans-White, J. L. Tank, E. J. Rosi-Marshall, and **M. R. Whiles**. 2007. Biogeochemistry of Dissolved Organic Carbon in Midwestern Agricultural Streams. Annual Meetings of the North American Benthological Society, Columbia, SC.

Meyer, C. K. and **M. R. Whiles**. 2007. Macroinvertebrate communities in natural and restored floodplain wetlands in the central Platte River basin. Annual Meetings of the North American Benthological Society, Columbia, SC.

Whiles, M. R. A. D. Huryn, R. O. Hall, K. R. Lips, C. M. Pringle, S. S. Kilham, W. K. Dodds, and C. E. Montgomery. 2007. Quantifying the Roles of Neotropical Tadpoles in Headwater Stream Nitrogen Cycling: the Influence of Starvation and Stress on Excretion Estimates. Annual Meetings of the North American Benthological Society, Columbia, SC.

Chambers, C. P., **M. R. Whiles**, N. A. Griffiths, M. A. Evans-White, E. J. Rosi-Marshall, J. L. Tank, and T. V. Royer. 2007. Assessing the impacts of transgenic Bt corn detritus on macroinvertebrate communities in agricultural streams. Annual Meetings of the North American Benthological Society, Columbia, SC.

Csoboth, L. A., D. L. Martin, J. E. Garvey, and **M. R. Whiles**. 2007. Lateral Springtime Invertebrate Drift between a Restored Backwater and a Large Midwestern River. Annual Meetings of the North American Benthological Society, Columbia, SC.

Gido, K. B., K. N. Bertrand, J. N. Murdock, W. K. Dodds, J. R. Bengtson, **M. R. Whiles**, M. A. Evans-White, and D. J. Hoeinghaus. 2007. Disparate effects of grazing minnows on ecosystem processes in field and mesocosm experiments

Pokelsek, J. D., E. J. Rosi-Marshall, C. P. Chambers, N. A. Griffiths, M. A. Evans-White, J. L. Tank, **M. R. Whiles**, and T. V. Royer. 2007. Effects of Bt corn pollen on caddisfly growth rates in Midwestern agricultural streams. Annual Meetings of the North American Benthological Society, Columbia, SC.

Schultheis, R. D., M. W. Eichhholz, **M. R. Whiles**, and T. Yerkes. 2007. Invertebrate availability for waterfowl consumption during the spring migratory period. Annual Meetings of the North American Benthological Society, Columbia, SC.

Murdock, J. N., K. N. Bertrand, K. B. Gido, W. K. Dodds, and **M. R. Whiles**. 2007. Recovery trajectories of prairie streams following drought. Annual Meetings of the North American Benthological Society, Columbia, SC.

<u>POSTER PRESENTATIONS</u> (since 2007 only)

*Frauendorf, T., **M. R. Whiles**, A. T. Rugenski, J. C. Colon-Gaud. 2012. Ontogenetic diet shifts among neotropical stream macroinvertebrates. Annual meetings of the Society for Freshwater Science, Louisville, KY. ***best student poster presentation in basic research**

Vandermyde, J. and **M. R. Whiles**. 2012. Effects of riparian forest removal on macroinvertebrate functional structure and production along tallgrass prairie streams. LTER All Scientists Meeting, Estes Park, CO.

Jackson, K. E. and **M. R. Whiles**. 2012. Establishing seasonal baseline conditions for benthic macroinvertebrates and organic matter in remnant tallgrass prairie streams. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Erndt, K. C., **M. R. Whiles**, and W. K. Dodds. 2012. Terrestrial insect inputs to tallgrass prairie headwater streams. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Kennington, A. K., E. A. Scholl, G. Garcia, H. M. Rantala, G. Wilkerson, and **M. R. Whiles**. 2012. Predicting ecological responses to a proposed river restoration: influence of

enhanced flow on duckweed cover, light penetration, and dissolved oxygen in an Illinois river. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Egdorf, T. C., H. M. Rantala, and **M. R. whiles**. 2012. The effects of stream diversion on leaf litter breakdown in the Cache River, IL. Annual meetings of the Society for Freshwater Science, Louisville, KY.

Heinrich, K. K., **M. R. Whiles**, and C. Roy. 2010. Influence of an in-stream restoration practice on insect emergence and riparian birds. Annual meetings of the North American Benthological Society, Santa Fe, NM.

Peterson, S. D., A. T. Rugenski, C. Colon-Gaud, **M. R. Whiles**, S. S. Kilham, K. R. Lips, and C. M. Pringle. 2010. Influence of catastrophic amphibian declines on storage and export of fine particular organic matter in neotropical headwater streams. Annual meetings of the North American Benthological Society, Santa Fe, NM.

Walther, D. A., and **M. R. Whiles**. 2009. Relationships between benthic organic matter standing stocks and secondary production in temperate North American streams. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Rothert, A. K., P. F. Landrum, M. J. Lydy, and **M. R. Whiles**. 2009. Bioavailability of slower desorbing fractions of sediment-associated permethrin. Annual meetings of the Society of Environmental Toxicology and Chemistry.

Frauendorf, T. C., Colon-Gaud, C., and **M. R. Whiles**. 2009. Macroinvertebrate diet shifts along a neotropical stream continuum. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Rugenski, A., **M. R. whiles**, Lips, K. R., Pringle, C. M. Kilham, S. S., and M. J. Vanni. 2009. Quantifying the roles of larval amphibians in nutrient cycling in neotropical headwater streams. Annual meetings of the North American Benthological Society, Grand Rapids, MI.

Meyer, C. K., **M. R. Whiles**, and S. G. Baer. 2009. Assessment of ecosystem recovery in restored Platte River Wetlands. Platte River Basin Science and Resource Management Symposium, Kearney, NE.

Peterson S. D., C. Colon-Gaud, **M.R. Whiles**, S. S. Kilham, K.R. Lips, and C.M. Pringle. 2008. Changes in seston quality and quantity associated with amphibian declines in central american streams. Annual meetings of the North American Benthological Society, Salt Lake City.

Myers, D. J., **M. R. Whiles**, and G. W. Whitledge. 2008. δD and $\delta^{18}O$ as tracers of macroinvertebrate origins and movements in large river floodplain water bodies. Annual meetings of the North American Benthological Society, Salt Lake City.

Walther, D. A., and **M. R. Whiles**. 2007. Macroinvertebrate responses to in-stream restoration structures in a midwestern river. Ecological Society of America, San Jose, CA.

Whiting, D. P., M. R. Whiles, K. N. Bertrand, K. B. Gido, and W. K. Dodds. 2007. Influence of flood frequency and fish on invertebrate communities in experimental stream mesocosms. Annual Meetings of the North American Benthological Society, Columbia, SC.

Meyer, C. K., S. G. Baer, and **M. R. Whiles**. 2007. Belowground recovery across a chronosequence of restored wetlands in the Platte River Valley. Soil Ecology Society meetings, Moab, UT.

Peterson, S. D., C. K. Meyer, and **M. R. Whiles**. 2007. Quantitative assessment of the efficiencies of three macroinvertebrate sampling techniques in Platte River wetlands. Annual Meetings of the North American Benthological Society, Columbia, SC.

Frauendorf, T. C., N. A. Griffiths, J. L. Tank, M. A. Evans-White, E. J. Rosi-Marshall, T. V. Royer, and **M. R. Whiles**. 2007. The rate of Bt leaching from corn detritus and the effects on microbial respiration on stream sediments. Annual Meetings of the North American Benthological Society, Columbia, SC.

PROFESSIONAL SERVICE and MEMBERSHIPS

2012	Chair, SIUC Honorary Degree & Distinguished service award committee
2011	Platte River Caddisfly endangered species listing review
2010-	SIU Graduate Council
2009	Salt Creek Tiger beetle endangered species listing review
2009	Luquillo Long-Term Ecological Research Site review panel member
2009	Platte River Ecosystem Monitoring Program review panel Member
2008-11	SIU Judicial Review Board
2008-	STREON steering committee
2006-	Organization of Biological Field Stations member
2006-08	SIU Faculty Senate
2005-	American Institute of Biological Sciences member
2004-	Associate Editor, Journal of the North American Benthological Society
2004-05	Co-chair, 2005 annual meetings of the North American Benthological Society, New Orleans, LA
2004-05	Treasurer, SIU chapter of Sigma Xi
2002-03	Program Committee, North American Benthological Society
2000-	Entomological Society of America member
1997- 2005	Sigma Xi member
1986-	North American Benthological Society member

GRADUATE STUDENTS

C. K. Meyer	MS	2000	KSU	Associate Prof., Simpson College
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LIMNOLOGY
and
OCEANOGRAPHY: METHODS

Limnol. Oceanogr.: Methods 2, 2004, 333–341
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A technique for establishing reference nutrient concentrations across watersheds affected by humans

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Abstract

Establishing reference nutrient conditions for rivers and streams is necessary to assess human impact on aquatic ecosystems and protect water quality and biotic integrity. Several methods have been proposed: (1) percentiles from statistical distributions of all rivers and streams in a region or dataset, (2) reference stream approaches, and (3) modeling river networks from existing reference streams. We propose an additional statistical method to estimate the influence of anthropogenic land uses on lotic nutrient concentrations. First, we quantify regional variation by using analysis of covariance, where total nitrogen or total phosphorus is the dependent variable, region is the categorical predictor, and percentage of anthropogenic land use (e.g., cropland, urban land) is the covariate. This allows for the aggregation of regions if there is not a significant regional effect, or if there is a significant regional effect, identifies the need to analyze regions separately. Second, we develop multiple linear regression models with best-model techniques in which anthropogenic land-use classifications are the independent variables, and the logarithms of in-stream nutrient concentrations are the dependent variables. The intercept of these regression models (i.e., expected nutrient concentration in the absence of human activities assuming linear extrapolation to the origin) represents the reference nutrient concentrations. This analysis suggests that larger percentages of cropland and urban land have strong positive influences on in-stream nutrient concentrations, both in eastern Kansas and across the conterminous United States. The most appropriate method for regions may depend on the relative availability of reference sites and other data sources. The covariance/reference approach offers a potential method for regions with limited numbers of reference sites.

One of the central issues of pollution control is the establishment of reference conditions. This is particularly true of efforts to control anthropogenic eutrophication because some amount of nutrient enrichment of aquatic systems from land is a natural part of pristine ecosystems. Identifying reference conditions is an especially difficult issue when establishing nutrient criteria in rivers and streams. Water quality conditions downstream are affected by the entire watershed upstream, and few watersheds are minimally affected by humans (Lewis 2002). Reference conditions (i.e., background nutrient concentrations) provide an indication of the maximum obtainable water quality if human impacts are completely controlled and may approximate the natural trophic state of a lotic ecosystem. Reference conditions in aquatic ecosystems also provide insight into the abiotic habitat in which the biotic community evolved. In some instances, it may be necessary to establish pristine natural conditions to maintain biotic integrity (Dodds and Welch 2000).

Establishing nutrient criteria has recently assumed greater importance (Dodds and Welch 2000) as regulatory agencies in numerous developed countries have extended efforts to control eutrophication. Originally, efforts were aimed toward controlling eutrophication in lakes and reservoirs, but efforts have now expanded to streams and wetlands. As part of this strategy, several approaches have been used to establish reference conditions. Most approaches rely on the concept that nutrient concentrations in waters may be based upon site-specific characteristics that are in some way independent of anthropogenic impacts. One approach to delineating ecoregions for North America is based on soils, vegetation, and dominant land uses (Omernik 1995) and explicitly recognizes that reference conditions may vary spatially across landscapes. The United States Environmental Protection Agency (USEPA) has

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Acknowledgments

We thank Dolly Gudder for technical assistance and members of the LAB journal club and Ed Carney for helpful criticism. Jim Omernik provided useful insights, and the Kansas Department of Health and Environment, the Konza Prairie Long Term Ecological Research project, the USGS, and the USEPA (with special thanks to Jeffrey Comstock) provided data. Support for this project came from the USGS GAP program and from the NSF Long Term Ecological Research program. This is contribution 04-286-J from the Kansas Agricultural Experiment Station.

aggregated Omernik's ecoregions to suggest geographically delineated nutrient criteria (USEPA 1998).

The USEPA suggests using one of three strategies for determining reference conditions (Buck et al. 2000). The first approach characterizes reference reaches for each stream class within a region according to best professional judgment, and then uses these reference conditions to develop criteria. This method fails when reference reaches are not available, as is the case in some agriculturally dominated regions of the midwestern United States. The second approach identifies the 75th percentile of the frequency distribution of reference streams for a class of streams and uses this percentile to develop the criteria. The third approach calculates the fifth to 25th percentile of the frequency distribution of the general population of a class of streams and uses the selected percentile to develop the criteria. In regions that are extensively affected by human nutrient loading, the method of using the 25th percentile can lead to the establishment of artificially large values for nutrient criteria (Smith et al. 2003), perhaps leading to failure to protect water quality.

A biologically based approach would use the functional response between biotic indicators and stream nutrient concentrations. This approach could be based on community analyses of algae (e.g., ordination analyses of diatom assemblages; Pan et al. 1996, 2000). Alternatively, the natural breakpoint beyond which no additional benthic chlorophyll yield is expected (Dodds et al. 2002) could be used to set upper limits on in-stream nutrient concentrations. While some regions (e.g., England, northeastern United States) have well defined biotic indices, community responses of biota to elevated nutrients are not well developed for many specific regions. To our knowledge, biologically based index approaches have received little general application to nutrient control strategies employed by regulatory agencies. Such approaches may ultimately provide important data as biotic responses at the species level to nutrients are documented for each specific region of interest.

A promising approach to estimate reference conditions involves using data from small, moderately affected systems and applying statistical modeling techniques to obtain reference nutrient values for large rivers (Smith et al. 2003). This method uses available reference reaches, predicts in-channel nutrient removal rates from existing data, and attempts to model in-channel nutrient concentrations in the absence of human impact. This technique is useful in that it provides an alternative to the difficult (or impossible) task of identifying large reference watersheds, and it accounts for in-stream nutrient processing that may remove nutrients from the water column as water flows through drainage systems. This method can also account for atmospheric deposition. But the technique works only if small, moderately affected systems are available, and if modeling assumptions are met.

The considerations described in the previous paragraphs identify the need for a method of determining baseline nutrient concentrations in areas with different degrees of human impact. Such a method would be most useful in regions that have few or no minimally affected sites, such as heavily agricultural or urban areas in the United States. In the examples we present in this paper, digital land cover and population density data, which are available for the entire contiguous United States, are used to generate reference nutrient concentrations. We will discuss the potential applicability and limitations of this approach.

In this paper, we explore statistical techniques to distinguish human land-use effects from naturally variable nutrient concentrations. We use three data sets to explore this approach: a detailed water-chemistry data set from central and eastern Kansas, data from an extensively monitored pristine watershed in the Flint Hills region of northeast Kansas, and a national dataset generated from the United States Geological Survey (USGS) survey network (Alexander et al. 1998).

Materials and procedures

Kansas data—The mean of the nutrient concentration for all samples at each site was used as the response variable. Chemical data were taken from the ambient monitoring network maintained by the Kansas Department of Health and Environment (KDHE). Data for total phosphorus (TP) were collected from 1990 to 2002. Data for total nitrogen (TN) were collected from January 2000 to May 2003. Total nitrogen and phosphorus were analyzed by a colorimetric automated phenate method, after digestion by metal-catalyzed acid and persulfate techniques, respectively, according to USEPA standard methods. Nitrate concentrations were added to the amount of N determined by metal-catalyzed digestions to represent TN. Minimum detection limits were 0.1 and 0.01 mg L^{-1} for TN and TP, respectively. Samples lower than the minimum detection were encountered less than 1% of the time, and values were set to the detection limit if this occurred. Data were taken from watersheds in four ecoregions from central-eastern Kansas (Central Great Plains, Central Irregular Plains, Corn Belt, and Flint Hills). Only watersheds that were entirely within a USEPA level III ecoregion were selected (Oakes 2003). No watersheds were selected that had large numbers of confined animal feeding operations or permitted sewage outfalls close to the sampling location. We were unable to control for animal feeding operations holding fewer than 200 animals.

Additional TN and TP data were collected on Kings Creek, a pristine prairie stream that has been monitored for nutrient chemistry. We used stream site data from 1994 to 2001 (Dodds 2003). Within the Kings Creek site, there are 4 small pristine watersheds where samples were taken. Total N and TP samples were collected three times a week during the period when flow was occurring, leading to 1727 individual samples, which were analyzed as reported previously (Dodds 2003). The TN and TP values were never below the limit of detection (4 μg L^{-1} P and N).

USGS data for United States—Nutrient data were obtained from a large data set compiled by the USGS for sampling that occurred from 1970 to 1983 (Alexander et al. 1998). All sampling dates when both TN and TP were collected were included in these analyses. Samples were below the detection limits less than 0.3% of the time for TP, and these values were set to zero. There were no values below detection for TN. The dataset was further restricted to sites sampled more than 20 times, those that included land-cover data as of 1987, and sites that did not appear to be statistical outliers. The mean of the nutrient concentrations for all samples at each site was used as the response variable. Land-cover classifications were obtained from the U.S. Soil Conservation Service (now officially titled the Natural Resources Conservation Service of the U.S. Department of Agriculture). The classifications assumed to include anthropogenic impacts were cropland (areas used for production of crops for harvest), pasture land (land managed primarily for the production of introduced forage plants for livestock grazing), range land (plant cover that is principally grasses or small plants suitable for grazing), farm land (buildings and livestock-holding facilities), and urban land (residential, industrial, commercial, and institutional land). Population data from 1990 were also included in models. These areas were classified by hydrologic units in the USGS database. Monitoring stations that met the criteria listed above for both TN and TP were used to create a spatially explicit Geographic Information System (GIS) coverage of sampling site locations that was overlaid on a digital stream network (USEPA river reach file, RF1) and a GIS layer of the USEPA level III ecoregions (USEPA 1998). Hydrologic units are not defined solely by the watershed above a single point. For example, a hydrologic unit on the Mississippi river may only cover a portion of the watershed above that point (i.e., the tributaries entering for a relatively short distance upstream from that point). Thus, hydrologic units do not always encompass the entire watershed above a sampling station (Omernik and Bailey 1997). Therefore, as a final data refinement step, stations were only included where 2/3 or more of the entire watershed draining into the station was contained in one ecoregion. This last selection avoids problems with stations that may be influenced by multiple upstream ecoregions (Griffith et al. 1999). A total of 519 stations were used in analyses, with 5 to 113 stations present in each of the 13 USEPA level III aggregate National Ecoregions (USEPA 1998). The aggregate National Ecoregions are hereafter referred to simply as ecoregions.

Statistical methods—Statistical analyses were performed by using Statistica 6.1 (Statsoft). All nutrient data were $\log_{10}$ transformed for two reasons: (1) it led to normal data distributions (Kolmogorov-Smirnoff test, $P > 0.05$) and (2) data distributions did not yield intercept estimates that were less than zero once data were log transformed. This is particularly important when applying the regression approach described next, inasmuch as a negative nutrient concentration is not possible, but the method could potentially yield a negative intercept when using non–log-transformed nutrient concentrations as the dependent variable.

Statistical analyses were accomplished in two steps. Analysis of covariance (ANCOVA) was first used to test for significant differences among ecoregions while accounting for the effects of land use variables on water column nutrients (Sokal and Rolhf 1981). Second, within ecoregions, or across ecoregions that were not significantly different ($P > 0.05$) as determined by ANCOVA, multiple linear regression was used to establish relationships between land use and nutrient concentrations. All possible subsets regression was used to determine the best fit model with Mallow's Cp as an index to control for the effect of adding additional variables into the model.

The regression approach is reflective of that used by Omernik (1977). In that study, however, the percentage of urban land and cropland were simply summed, whereas the approach outlined herein finds the best regression model of all potential model subsets, assigns relative weights to each land-use category, and considers additional land-use categories that were not available to Omernik (1977). Results from the multiple linear regression analyses were examined in two ways. First, the intercept (β_o) and associated error were used to extrapolate nutrient concentrations in the absence of predominantly anthropogenic landscapes, and the variance in this prediction could be characterized by the 95% prediction band around the estimate for β_o. Second, the values for β, excluding the intercept, were condensed into a single axis to visualize the influence of multiple significant land uses. For example, if the regression model yielded a relationship of $\log_{10}TP = \beta_o + \beta_1 \times \%\ \text{cropland} + \beta_2 \times \%\ \text{urban land}$, then β_o was taken as the expected reference level and its 95% prediction band was used to establish the uncertainty in the estimate for reference nutrient concentrations. In addition, a two-dimensional plot was created in which the x value for each site was calculated as $\beta_1 \times \%\ \text{cropland} + \beta_2 \times \%\ \text{urban}$ and was plotted against TP concentrations at sites on the y axis.

Methods developed by the USEPA for determining reference nutrient concentrations (Buck et al. 2000) were also used; the fifth to the 25th percentile of the frequency distribution of the general population of a class of streams were calculated for Flint Hills data (using data from the state of Kansas only) to allow comparison with reference and regression methods.

Assessment

Kansas data—When data taken from KDHE were used, the Flint Hills were characterized by relatively small percentages of cropland and urban land, the Corn Belt by large percentages of cropland, and the Central Irregular Plains by the largest proportion of urban lands (Fig. 1). The Flint Hills were characterized by the lowest TN and TP concentrations (Fig. 1).

Analyses related to TP are used first to exemplify the methodology developed in this article. Analysis of covariance demonstrated that there was not a significant ecoregion effect, or a significant interaction effect on TP between ecoregion

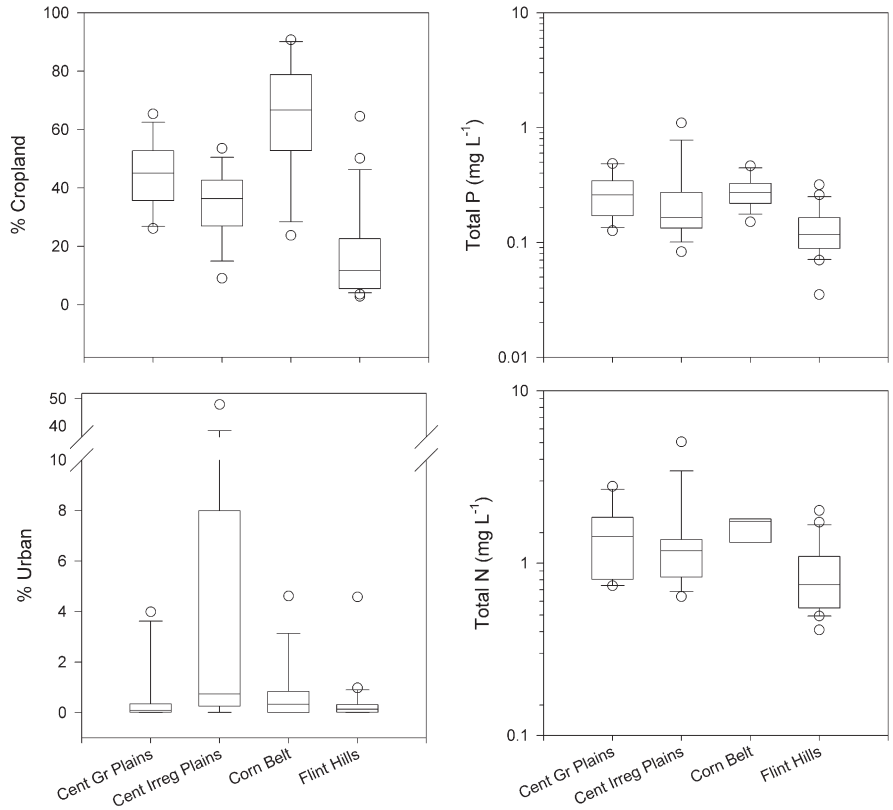

Fig. 1. Box plots of cropland, urban land, TP, and TN for water quality sites in eastern Kansas by level III ecoregion. Lines in center of boxes are the medians, tops and bottoms of boxes are 75th and 25th percentiles, respectively. Bars are 95% confidence intervals, and outliers are plotted as open points.

and percentages of cropland or urban land (Table 1). This lack of an ecoregion effect indicated that the ecoregions could be combined; therefore, data were pooled across ecoregions for the multiple regression analysis. Both the percentage of cropland and urban land, as well as the intercept, were significantly related to TP concentrations in the regression model (Table 2). The strength of this relationship can be visualized by creating an x-axis value from the results of the regression across all sites with varying land use (Fig. 2). In this example, the regression equation was Log_{10} TP = –0.724 + 0.00668 × % cropland + 0.1465 × % urban land (Table 2).

Therefore, for each sampling site, an x variable was created by using the equation: x = 0.00668 × % cropland + 0.1465 × % urban land. This plot also includes the individual ecoregion-specific regression lines to allow visual verification of the statistical result from ANCOVA that slopes did not vary significantly across ecoregion. Although the line for the Corn Belt has a slightly lower slope, this was not statistically significant.

The intercept (β_o = –0.724) is then used to establish reference criteria in the TP regressions. The intercept represents the point where percentages of cropland and urban land are zero. Given the regression, a 95% prediction band could be calculated around this predicted value such that the reference level, the lower confidence interval, and the upper confidence interval were 97, 38, and 251 µg P L^{-1}, respectively.

In contrast to the results from the analyses of TP, TN did vary significantly across Kansas ecoregions (Table 3, ANCOVA with significant ecoregion effect and ecoregion × % cropland interaction). Thus, it was necessary to perform the regressions separately by ecoregion in this example.

Table 1. Analysis of covariance of log transformed total P, with four Kansas ecoregions as categorical predictors and with percentages of urban and cropland as the covariates

	Sum of squares	d.f.	Mean square	F	P
Intercept	2.331	1	2.331	71.59	<0.0001
Ecoregion	0.175	3	0.059	1.80	0.1593
% crop	0.161	1	0.161	4.96	0.0303
% urban	0.112	1	0.112	3.43	0.0697
Ecoregion × % crop	0.175	3	0.058	1.80	0.1595
Ecoregion × % urban	0.223	3	0.074	2.29	0.0896
% crop × % urban	0.051	1	0.051	1.56	0.2169
Ecoregion × % crop × % urban	0.155	3	0.056	1.58	0.2043
Error	1.693	52	0.033		

Table 2. Results of regression analysis for relationship between percentages of urban land, cropland, and $\log_{10}$ transformed TP (mg L^{-1}), with aggregated data across four Kansas ecoregions (R^2 = 0.431).

	B	Standard error	t(65)	*P* level
Intercept	–0.72376	0.048511	–20.8900	<0.000001
% crop	0.00668	0.001097	6.0872	<0.000001
% urban	0.01465	0.003280	4.4652	0.000033

The regression method was compared with two other methods suggested by USEPA, the fifth and 25th quartile of a general population of streams sampled by KDHE and comparison with a known reference stream (Table 4). In this instance, the upper 75% of data taken from a known reference stream (Kings Creek on Konza Prairie) had substantially less TP than results arrived at by any other method. The quartile methods had slightly smaller reference values than did the TP regression method across the four ecoregions. The regression method for TN yielded very similar results to the percentile method, and results from both methods were slightly more than two times greater than those of the pristine reference site (Table 4).

United States Geological Survey (USGS) data—Analysis of covariance indicated significant differences in TN and TP across ecoregions when using percentages of cropland and urban land as the covariates ($F_{12,466}$ = 8.8693, P < 0.00001 for TN; $F_{12,466}$ = 8.1841, P < 0.00001 for TP), so each of the USEPA aggregate level III ecoregions were analyzed separately for both TN and TP. Ecoregion 6 (Corn Belt and Northern Great Plains) had, on average, substantially greater mean TN and TP values than the other ecoregions, and ecoregions 2 (Western forested mountains) and 8 (Nutrient poor glaciated upper midwest and northeast) had significantly smaller mean values (P < 0.05) than other ecoregions.

All possible subsets regression using Mallow's Cp for model selection of anthropogenic land use classifications (population, % urban land, % cropland, % pasture land, % range land, and % farm land) was used to predict TP (Table 5). Regression analysis for TP suggested that percentage of cropland was a strong predictor of TP within ecoregions (6 of 13 models), as were percentages of urban land and total population above the sampling site (6 of 13 cases for both predictor variables). In ecoregion 4 (Great Plains grass and shrublands), there was a weak relationship between land-use factors and TP concentrations, which contrasts with the results of the Kansas analyses (Table 2) that included smaller portions of this ecoregion. In general, the TP values predicted in the absence of human land use were less than 50 µg L^{-1} TP, except when the relationship between

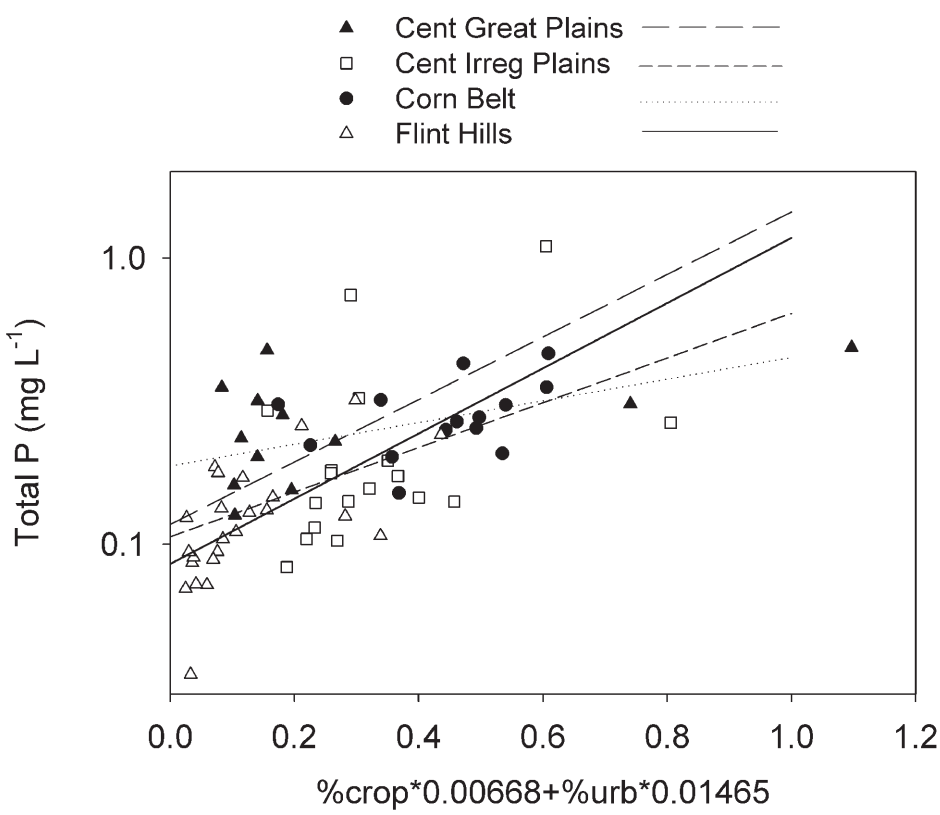

Fig. 2. Relationships between land use and TP by ecoregion from ecoregions of eastern Kansas. Data on x axis were constructed from results of regression analysis across ecoregions (*see Materials and procedures* for details). Individual lines represent regressions within ecoregions.

Table 3. Analysis of covariance of $\log_{10}$ TN, with percentage of cropland as the covariate and ecoregion as the categorical predictor

	Sum of squares	d.f.	Mean square	F	P
Intercept	0.013	1	0.013	0.402	0.5291
Ecoregion	0.632	3	0.210	6.457	0.0009
% crop	0.011	1	0.011	0.327	0.5701
Ecoregion × % crop	0.469	3	0.156	4.793	0.0053
Error	1.600	49	0.033		

land use and TP was weak (i.e., the Xeric west and the Great Plains grass and shrublands) and in the Texas-Louisiana coastal and Mississippi alluvial plains ecoregion.

All possible subsets regression of anthropogenic land-use factors was also used to predict TN (Table 6). As with the regression analyses for TP, the percentage of cropland was the variable most often included in the model that best predicted TN within ecoregions (9 of 13 cases), followed by watershed population (7 of 13 cases) and the percentage of urban land in the watersheds (5 of 13 cases). In ecoregion 12 (Southern coastal plains), there were no significant relationships between land-use factors and TN concentrations, and in ecoregion 4 (Great Plains grass and shrublands), the predictive ability of the model was very weak. Predicted background TN concentrations were less than 600 μg L^{-1} TN in 8 of 12 instances.

Discussion

In a broad sense, the regression method for determining reference nutrient concentrations in the absence of anthropogenic land uses provided numbers comparable to those generated with the USEPA 25% method and that of Smith et al. (2003). The data suggest that generally the reference values should be suspect if they fall above 60 μg L^{-1} TP or 600 μg L^{-1} TN (Table 7). There were some instances of serious divergence among methods, however, with each approach having its own limitations and benefits. Correlation analyses and paired-difference *t* tests failed to find significant correlations among the methods ($P > 0.05$) or significantly higher or lower values predicted by any of the methods ($P > 0.05$).

The regression model we present here has the primary limitation of not quantifying all sources of human impacts because such data were not readily available. Quantification of impacts such as atmospheric deposition of N, confined animal feeding units, effects of riparian buffer strips and other best-management practices, and point discharges from sewage treatment likely would have improved model accuracy. Specifically, accounting for other anthropogenic nutrient inputs would further lower the estimated reference nutrient concentrations. Accounting for additional factors such as drainage area and slope could improve the accuracy of the models. In-channel processing (i.e., the effect of stream size) cannot be assessed with the regression approach.

The regression approach relies upon the percentage of land use in each category to compare anthropogenic impacts across watersheds. This leads to potential problems with non-normal data distribution (Sokal and Rohlf 1981). Unfortunately, the transformations generally employed to correct non-normal proportional data are not defined at zero. Because the point of the approach is specifically to determine the intercept when the dependent variable (degree of anthropogenic impact) is zero, we used percentages in our analyses. Perhaps the greatest weakness of our regression approach is the need to extrapolate beyond known data points. However, where reference reaches are not available, this may be the only option for determination of reference conditions.

The regression approach does not require data from a large number of reference or low impact sites. If there are no low impact sites however, this method requires prediction of data far from the data points that anchor the relationship. Predictions made farther away from the observed data result in broader prediction bands. Thus, the accuracy of this approach should be greatest when sites include a relative continuum of land-use intensity.

The use of ANCOVA before application of the regression approach allowed for a statistically defensible aggregation of

Table 4. Mean and 95% confidence intervals for estimated reference-nutrient concentrations in a pristine Flint Hills site (Kings Creek), and intercepts and 95% prediction bands according to regression analysis of TP across four ecoregions in Kansas and TN across the Flint Hills (Kansas Department of Health and Environment data), and using the 25th and fifth percentile of all Flint Hills data (Kansas Department of Health and Environment data)

Method	Parameter	Concentration (μg L^{-1})	*n*	Low 95% (μg L^{-1})	High 95% (μg L^{-1})
Flint Hills reference site (Kings Creek)	TP	6.56	1146	6.28	6.87
Regression across ecoregions	TP	97	65	38	251
25th percentile Flint Hills data	TP	90	24	—	—
Fifth percentile Flint Hills data	TP	70	24	—	—
Flint Hills reference site (Kings Creek)	TN	223	1727	214	232
Regression across Flint Hills	TN	575	22	285	1148
25th percentile Flint Hills data	TN	569	24	—	—
Fifth percentile Flint Hills data	TN	493	24	—	—

Table 5. Best model regression results (Mallow's CP) for TP

Ecoregion number	Ecoregion name	*n*	R^2	β_0	Standard error	Best model land uses
1	Willamette and Central Valleys	5	0.90	–1.710	0.134	% crop
2	Western forested mountains	39	0.34	–1.348	0.137	pop, % urban, % past, % range
3	Xeric west	84	0.20	–0.820	0.099	pop, % urban
4	Great Plains grass and shrublands	36	0.19	–1.226	0.427	pop, % urban
5	Central cultivated Great Plains	35	0.30	–1.645	0.359	% urban, % range, % farm
6	Corn belt and Northern Great Plains	42	0.47	–1.640	0.314	pop, % urban, % crop, % range, % farm
7	Mostly glaciated dairy region	32	0.56	–1.630	0.101	pop, % crop
8	Nutrient poor glaciated upper Midwest and Northeast	41	0.32	–1.555	0.044	% crop, % range
9	Southeastern temperate forested plains and hills	113	0.48	–1.514	0.071	% urban, % crop, % range, % past
10	Texas-Louisiana coastal and Mississippi alluvial plains	30	0.64	–0.950	0.162	pop, % range
11	Central and Eastern forested uplands	27	0.32	–1.370	0.097	% urban, % crop
12	Southern coastal plain	15	0.54	–1.316	0.239	% range, % farm
14	Eastern coastal plain	18	0.19	–1.397	0.127	% farm

β_0 is calculated for mg L^{-1} phosphorus.

ecoregions. This is beneficial because it may be possible to aggregate ecoregions with few or no reference or low-impact sites with other ecoregions that do include reference sites, and subsequently decreases the prediction bands around estimates of reference concentrations.

Another benefit of the regression approach is that it identifies specific anthropogenic land use practices that contribute to nutrient pollution in an ecoregion and, thus, may guide management efforts. Finally, the graphical method used to represent the regression results can be used to visualize the data in two dimensions, even when more than one independent variable is significantly related to the dependent variable.

The modeling approach taken by Smith et al. (2003) used reference data to model the expected increase in N in the stream channel as stream order increased in order to estimate reference conditions. Smith et al. (2003) also accounted for atmospheric deposition of N into watersheds and for the loss of N and P from the water column of rivers as the water moved down the drainage network. However, their approach requires using reference reaches in which human impacts are relatively moderate as a starting point for modeling, requires extrapolation from small streams to larger rivers, and requires substantial modeling expertise to implement.

The establishment of reference watersheds to delineate nutrient criteria may be most desirable because it allows true

Table 6. Best model regression results (Mallow's CP) for total N*

Ecoregion number	Ecoregion name	*n*	R^2	β_0	Standard error	Best model land uses
1	Willamette and Central valleys	5	0.98	–0.583	0.048	% crop
2	Western forested mountains	39	0.41	–0.319	0.080	pop, % urban, % past, % range
3	Xeric west	84	0.32	–0.037	0.089	pop, % urban, % crop, % past, % range
4	Great Plains grass and shrublands	36	0.11	–0.181	0.291	pop, % urban
5	Central cultivated Great Plains	35	0.40	–0.247	0.179	pop, % range, % farm
6	Corn belt and Northern Great Plains	42	0.67	–0.668	0.205	% urban, % crop, % past, % range
7	Mostly glaciated dairy region	32	0.69	–0.248	0.069	pop, % crop
8	Nutrient poor glaciated upper midwest and northeast	41	0.62	–0.230	0.031	% crop
9	Southeastern temperate forested plains and hills	113	0.68	–0.432	0.040	% urban, % crop, % past, % range
10	Texas-Louisiana coastal and Mississippi alluvial plains	30	0.61	–0.128	0.069	pop, % crop, % range
11	Central and Eastern forested uplands	27	0.52	–0.091	0.042	% crop, % range
12	Southern coastal plain	15	NA†			
14	Eastern coastal plain	18	0.53	–0.445	0.114	pop, % crop, % farm

*β_0 is calculated for mg L^{-1} nitrogen.
†NA signifies that regression was not significant.

Table 7. Comparison of results from this paper and with 25% values suggested by the United States Environmental Protection Agency and values modeled by Smith et al. (2003) corrected for atmospheric N loading*

Ecoregion number	Ecoregion name	TP †	TP ‡	TP §	TN †	TN ‡	TN §
1	Willamette and Central valleys	20	47	16	261	310	156
2	Western forested mountains	45	10	19	479	120	157
3	Xeric west	151	22	24	918	380	44
4	Great Plains grass and shrublands	59	23	60	659	560	95
5	Central cultivated Great Plains	23	67	58	566	880	258
6	Corn belt and Northern Great Plains	23	76	54	215	2180	355
7	Mostly glaciated dairy region	23	33	22	565	540	147
8	Nutrient poor glaciated upper midwest and northeast	28	10	13	589	380	165
9	Southeastern temperate forested plains and hills	31	37	48	370	690	150
10	Texas-Louisiana coastal and Mississippi alluvial plains	112	128	48	745	760	439
11	Central and Eastern forested uplands	43	10	20	1102	310	156
12	Southern coastal plain	48	40	24	NA	900	548
14	Eastern coastal plain	40	31	15	359	710	561

*All values reported in μg L^{-1} N or P; NA = not applicable.
† This study.
‡ United States Enviromental Protection Agency.
§ Smith et al. (2003)

assessment of baseline, or reference, nutrient concentrations. But reference reaches simply are not available in many areas. As watershed size increases, the probability of identifying a suitable reference watershed decreases, as reference reaches tend to be in small headwater streams. Therefore, the reference reach method has limited applicability for medium-to-large rivers. Additionally, reference sites may be unaffected because of specific characteristics that make them undesirable for human uses (e.g., rocky terrain unsuitable for cropland), and may not be reflective of reference conditions across a region.

The USEPA 25% method is entirely dependent on the data that are incorporated, because it relies on the statistical distribution of all existing data. Thus, the USEPA method may present a moving target of criteria as nutrient pollution amounts increase with population growth and development (Dodds et al. 1998). The benefit of the 25% method is that it allows the maximum number of data points available to be used in the analysis, which may be important in regions without extensive sampling. Additionally, it requires the least technical expertise (i.e., no local determination of what is a reference was reached by experts, only moderate statistical training, and no modeling expertise) to implement.

The Corn Belt and Northern Great Plains ecoregions included few reference sites, and applying the USEPA 25% method to these sites yielded very large values. Most of the region is dominated by agricultural practices, and strong correlations existed between TN, TP, and row-crop agriculture when these relationships were analyzed using the USGS nationwide dataset. The USEPA 25% method resulted in reference values of 76 and 2180 μgL^{-1} for TP and TN, respectively, whereas the regression method with USGS data yielded substantially smaller numbers (3.3 and 10.1 times less for TP and TN, respectively). In this instance, the regression model was in better accord with the modeling approach of Smith et al. (2003).

Comments and recommendations

There is no optimal method for determining reference conditions for nutrient levels in rivers and streams. Fewer and fewer minimally affected streams are available to sample, particularly with regard to atmospheric contamination by N and agricultural influences. There is no better option than using detailed local data collected at pristine sites to establish baseline conditions, but many instances arise in which such data are not available. In these instances, it is important that multiple approaches are available to assess reference conditions. Individuals undertaking such analyses should be aware of the strengths and weaknesses of the datasets they have to work with and supplement their analyses with data from a variety of sources. Reference values provide regulators with a theoretical framework for best- and worst-case nutrient-control scenarios for a specific region.

Whereas this paper uses EPA ecoregions as a framework in which to use the regression approach, other spatial characteristics could be considered (e.g., geology, slope, drainage area). Accounting for any additional variance would allow for tightening of the confidence limits on the predicted reference concentrations. Thus, the regression method can be tailored for specific areas. It may also be possible to use the various techniques in concert with each other. For example, ANCOVA could indicate that it is reasonable to aggregate ecoregions, allowing for inclusion of more than a few reference sites.

This paper underscores the concept that there are multiple approaches to determining reference nutrient conditions. Each method has strengths and weaknesses; the regression

method presented here provides yet another tool for estimating reference conditions. Perhaps the most useful aspect of the numbers generated with this analysis is to examine the variations between the data presented by Smith et al. (2003) and the methodology currently recommended by the USEPA. Unless a reasonable mechanism can be found to explain why certain values are high (e.g., TN and TP values in the Corn Belt using the USEPA approach, or high values in the Xeric west using the regression method), the two lower agreeing values should probably be considered. The availability of multiple techniques to establish reference nutrient conditions allows analyses to be tailored to the individual circumstances of each situation and gives scientists more options for determining suitable nutrient targets for surface waters.

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Submitted 5 March 2004
Revised 24 June 2004
Accepted 5 July 2004

J. N. Am. Benthol. Soc., 2000, 19(1):186–196
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BRIDGES

BRIDGES is a recurring feature of J-NABS intended to provide a forum for the interchange of ideas and information between basic and applied researchers in benthic science. Articles in this series will focus on topical research areas and linkages between basic and applied aspects of research, monitoring policy, and education. Readers with ideas for topics should contact Associate Editors, Nick Aumen and Marty Gurtz.

Criteria for setting nutrient levels in lotic ecosystems are relevant to US states and other countries in the process of setting water-quality regulations. There are few articles in the peer-reviewed literature on this topic, and policy makers have had little information from which to base their decisions for streams. This lack of information is particularly troublesome because of the large number of streams and rivers that have impaired water quality, and the ever-increasing pace of urban and agricultural development. In addition to the effects of high nutrient concentrations on stream ecosystem structure and function, high nutrient concentrations, particularly nitrate, may have adverse effects on human health.

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Establishing nutrient criteria in streams

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"Every child deserves to grow up with water that is pure to drink, lakes that are safe for swimming, rivers that are teeming with fish. We have to act now to combat these pollution challenges with new protections to give our children the gift of clean, safe water in the 21st century."

—President Clinton, 23 February 1999, Baltimore

The US Environmental Protection Agency (USEPA) and the US Department of Agriculture (USDA) recently have been directed to set criteria for nutrients in rivers, lakes, and estuaries. State reports compiled by the USEPA (National Water Quality Inventory: 1996 Report to Congress) claim that 40% of streams or rivers surveyed were impaired because of the nutrients N and P, but no well-defined standards have been proposed to determine if nutrients impair flowing waters (USEPA 1998). A rational framework for determining criteria is necessary because the USEPA has been charged with establishing maximum acceptable levels of nutrients in streams and rivers by 2001 as part of the Clean Water Action Plan. State and tribal governments will use these criteria to set total maximum daily loads (TMDLs) for nutrients and adopt their own standards by 2003. The USEPA, the USDA, and other national governmental agencies (e.g., US Geological Survey [USGS], US Army Corps of Engineers), state and tribal officials, and private parties will set these criteria.

Data analyses are needed to explain the relationships between stream algae and nutrients, which previously have received attention from researchers. Given the potential economic impacts of nutrient control, the process by which nutrient levels are set likely will engender controversy. Basing the criteria on the best scientific data available will minimize conflict and maximize the potential benefits related to controlling nutrients in streams.

Although the question of how to set nutrient criteria is framed above in terms of US politics and policies, other countries also are interested in nutrient criteria for streams, particularly developed countries where industrialization, urbanization, and modern agriculture have resulted in extensive nutrient discharge into water courses. The following discussion will be based

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primarily on examples from the US, but the general principles could apply to any watershed.

We 1st will address why nutrient criteria are needed. Next, we will discuss the scientific basis behind possible criteria. Last, we discuss ways to account for variability in streams (i.e., factors that may decouple nutrient concentrations from biomass). The main thrust of this paper is to outline what scientific methods currently are available to managers for setting nutrient criteria, given specific reasons for setting the criteria.

Why do we need nutrient criteria?

Reasons for nutrient criteria include: 1) adverse effects on humans and domestic animals, 2) aesthetic impairment, 3) interference with human use, 4) negative impacts on aquatic life, and 5) excessive nutrient input into downstream systems. Each of these will be discussed in sequence; they share several characteristics, but they also have facets that may result in criteria being set at different levels.

High levels of some nutrients may have adverse effects on human health. Control of NO_3^- levels particularly is important to avoid metheamoglobemia. Furthermore, NO_3^- consumption has been correlated with stomach cancer (Hartman 1983). Although correlation does not guarantee causation, NO_3^- could be regulated as a carcinogen in the future.

Eutrophication from N and P causes proliferation of algal masses, some of which may be toxic. In one of the worst cases, eutrophication caused Cyanobacteria to bloom in the stagnant Murray-Darling River system (Australia) during a drought, leading to livestock deaths and concerns about impacts on humans (Bowling and Baker 1996). Such toxic blooms are most likely to occur in very enriched, slow-moving, and nonturbid rivers.

Eutrophication causes taste and odor problems in lakes (Arruda and Fromm 1989, Wnorowski 1992), but these negative effects have not been linked directly to trophic state of streams and rivers. Such problems often can be traced to production of odorous metabolites by Cyanobacteria (e.g., geosmin) and other algae and their subsequent leakage into surrounding waters. Algae that cause taste and odor problems can reach high biomass in eutrophic streams and rivers, both in the phytoplankton of slow-moving rivers and the periphyton of shallow streams.

Aesthetic impairment is more difficult to quantify, but usually is associated with filamentous algal forms. Nuisance levels may be reached somewhere between 100 and 200 mg/m^2 chlorophyll (Horner et al. 1983, Nordin 1985, Welch et al. 1988, Quinn 1991). Enriched waters often have benthic chlorophyll concentrations >150 mg/m^2, and many stream users find high levels of algal growth objectionable (Welch et al. 1989, V. Watson, University of Montana, personal communication). A link also may exist between property values and trophic state in lotic waters, as has been documented for lakes (Michael et al. 1996). However, to our knowledge, such analyses have not been conducted for rivers and streams.

Excessive growth of algae and macrophytes can interfere with human uses of flowing waters. Such interference is exemplified by problems caused by the filamentous green alga *Cladophora.* Exorbitant amounts of this alga can slow water flow in canals (decreasing delivery rates and increasing water losses), interfere with swimming, and snag fishing lures (Dodds and Gudder 1992). Furthermore, excessive algal growth may clog screens on water intakes for water treatment plants and industries.

High concentrations of NH_3 in the water column clearly are toxic to aquatic animals (Russo 1985). For example, levels of >1 mg/L NH_3-N in Ohio streams have negative impacts on the fish communities (Miltner and Rankin 1998). Negative impacts on aquatic life related to stimulation of algal biomass by increased nutrients are subtler. As a system becomes more productive, different species of algae may become more competitive and species composition can shift (Kelly and Whitton 1995, Pan et. al. 1996, Kelly 1998). However, unless such species shifts cause specific water-quality symptoms (e.g., toxic algae) or aesthetic problems (e.g., very long streamers of filamentous algae), the public is unlikely to be concerned.

Nutrient enrichment may adversely affect stream animal communities. Enriched streams have increased invertebrate biomass and altered invertebrate communities (Bourassa and Cattaneo 1998). Community structure has been correlated directly with P concentration (Miltner and Rankin 1998). Excessive levels of algae were damaging to invertebrates (Nordin 1985).

Changes in community structure may be viewed as problematic, particularly if game fish are affected. In extreme cases, levels of primary production can be stimulated by nutrients; organic C will build up in the system and cause a subsequent low dissolved O_2 (DO) and high pH event. Fish and invertebrates will grow poorly and even die if the O_2 depletion and pH increases are severe (Welch 1992).

Because streams drain into lakes and oceans, eutrophication caused by influx of nutrients from flowing waters is a concern for downstream lake and coastal areas. Examples of marine eutrophication are the zone of hypoxia that develops in the Gulf of Mexico (Rabalais et al. 1998) and the production of toxic estuarine dinoflagellate blooms (Burkholder and Glasgow 1997). Eutrophication problems in lakes are well documented, and the control of external and internal nutrient loading necessary to minimize eutrophication can be calculated (Cooke et al. 1993). Requirements for control of nutrient loading to lakes and coastal marine systems may lead to more stringent nutrient criteria in rivers than those required for controlling instream eutrophication, especially in localities where stream algae are limited by factors other than nutrients.

What are the scientific bases for criteria?

In this section we discuss what nutrients and what forms of those nutrients should be used to set criteria. We also describe some models and approaches that can be used to set criteria. Last, we offer some discussion on how criteria may vary depending upon the reason for the criteria.

What nutrients and forms should be used to set criteria?

The traditional view is that P limits primary production in fresh waters (e.g., Correll 1998), and N limits it in the ocean. However, nutrient bioassays and correlation analyses do not substantiate this point of view. Data were compiled from 158 bioassays reported in the literature in which the response of stream periphyton to nutrient fertilization was measured. Of the studies, 13% showed stimulation by N alone, 18% by P alone, 44% by simultaneous N and P additions, and 25% by neither nutrient (W. K. Dodds, unpublished data). The absolute proportions associated with each type of limitation should not be viewed as a general guide to nutrient limitation in streams. However, bioassay results do suggest that both N and P can limit primary producers in streams.

Correlation analyses also do not support the idea of P as the sole limiting nutrient in rivers and streams. Mean and maximum benthic chlorophyll correlated better with total N (TN) than total P (TP) in the water column in several hundred streams. Nitrogen and P occur in several forms in rivers and streams, including dissolved organic and inorganic forms and in particulate material. All of these forms together are referred to as TN and TP. Total N does not include dissolved N_2 gas. A regression model using both nutrients explained the highest proportion of the variance in biomass (Dodds et al. 1997). Thus, both N and P can control primary production in at least some streams and rivers.

Control of P alone may cause P to limit and lower algal biomass, as has occurred in many lakes (Sas 1989, Cooke et al. 1993). However, if pulses of P occur, they can be taken up in excess of requirements and stored inside algal cells in a process called *luxury consumption*. This stored P can allow algae to grow even if P concentrations are low in the water column. If controlling such P pulses is impossible (e.g., pulses associated with high runoff events in spring), control of N could become necessary. For example, Lohman and Priscu (1992) demonstrated that intracellular P concentrations in *Cladophora* increased, while P availability in the surrounding water decreased. Thus, *Cladophora* was N limited, even though analyses of available N and P in the river water column suggested P limitation. Given the bioassay and correlation data, and that periphyton can consume P in excess of immediate needs, setting nutrient criteria for both N and P makes sense.

Unless clear limitation by other nutrients has been demonstrated in a particular system, N and P should be assumed to be the dominant nutrients controlling the trophic states of streams and rivers. Fortunately, nuisance and some toxic heterocystous Cyanobacteria that can use N_2 gas as a N source generally are not part of eutrophic stream periphyton, but may occur in the plankton of slowly flowing rivers (Bowling and Baker 1996). The decreased dominance of heterocystous Cyanobacteria in streams leads to some situations where N control alone may

lead to decreases in algal biomass. Controlling N and not P inputs in lakes can encourage blooms of nuisance Cyanobacteria (Stockner and Shortread 1988). However, the strategy of controlling N alone should be viewed with caution, especially in plankton-dominated rivers. More data on cyanobacterial problems in eutrophic streams are necessary before we can be certain that N control will not lead to cyanobacterial dominance.

Control based on measured levels of dissolved inorganic N and P may not be effective because these pools are replenished rapidly by remineralization in surface waters (Dodds 1993). Correlation of algal biomass with dissolved inorganic nutrients was poor in some studies (Dodds et al. 1997), but not all (Biggs and Close 1989). Also, lake managers are aware of problems with using dissolved inorganic nutrient concentrations to set nutrient criteria. Last, most of the data linking land-use practices to N and P loading have been reported in TN and TP (Loehr 1974), so basing criteria on total nutrients for calculating TMDLs is more practical than using dissolved inorganic nutrients.

Two caveats are necessary to the generalization that TN and TP should be emphasized. First, if nutrients are released directly into streams in dissolved inorganic form, their influence may be more intense and localized near the point source of release. Dissolved inorganic nutrients will be taken up rapidly, which can lead to a very high, localized concentration of biomass (Hynes 1969). Second, some models using seasonal means of dissolved inorganic nutrients to predict algal biomass have been very successful (Biggs 1995, 2000), and some sites have considerably more data on dissolved than total nutrients on which to base decisions.

At what concentrations should criteria be set?

One difficulty in setting criteria involves assessment of the trophic state of a stream or river. Stated another way, how can we declare that a river or stream is in an unacceptable trophic state if there is no basis for scaling the trophic state relative to other rivers? A generally accepted system for classifying the trophic states of streams and rivers is lacking (Dodds et al. 1998). In general, trophic state is classified by nutrients and algal biomass. System metabolism may be more relevant to ecosystem function, but difficulties with methods and limited data have precluded use of production and respiration to classify trophic states of lakes and streams. One classification system proposed for streams relies upon the cumulative frequency distributions of chlorophyll and nutrients. The lower 1/3 of the distribution sets the range for oligotrophic streams, and the upper 1/3 for eutrophic streams. This approach is consistent with the convention of classifying trophic state into 3 categories, while basing classification on the actual distribution of biomass and nutrient levels found in streams (Dodds et al. 1998). The published classification was based on only 286 temperate streams. More data are necessary to determine how well this classification scheme applies to rivers from different ecoregions, how distributions of nutrients correlate to algal biomass, and how well such classification represents pristine conditions. Analyses of existing databases may provide a valuable tool in extending this approach to trophic classification.

A few models directly link TN and TP to benthic algal biomass in streams (e.g., Lohman et al. 1992, Dodds et al. 1997, Bourassa and Cattaneo 1998, Chételat et al. 1999). Such models can be applied to estimate algal biomass as a function of water column nutrients. A similar correlation approach has been very successful in managing eutrophication in lakes and reservoirs. Extension of these models to link instream nutrient concentrations to known sources of nutrient loading also has been described (Dodds et al. 1997).

Models describing the correlation between nutrients and chlorophyll in lakes differ from those for streams because benthic chlorophyll may be much more variable in streams as a result of the effects of floods, turbidity, and grazing. This difference is exemplified by the ratios of maximum to mean chlorophyll. This ratio describes the variance in level of chlorophyll, with high numbers denoting a high variance. The maximum/mean chlorophyll ratio is 4.5 for stream benthos compared to 1.7–2.6 for lake phytoplankton (Dodds et al. 1998). Furthermore, total water column nutrients usually are correlated strongly with chlorophyll because phytoplankton contain chlorophyll, N, and P. This linkage leads to high correlation coefficients between total nutrients and algal biomass in lakes. This relationship is not as highly coupled in streams when *benthic* chlorophyll and *water col-*

umn TN and TP are considered. Thus, the correlation models developed for stream benthic algae contain a much greater degree of uncertainty than those for lakes.

Biggs (2000) proposes a correlation method that considers hydrodynamic disturbance and inorganic nutrients in New Zealand streams, that is pertinent for predicting benthic algal biomass. Such an approach may prove useful within an ecoregion, and could be used to provide a sliding scale of nutrient criteria, with higher nutrient content allowed in more hydrodynamically unstable rivers (i.e., criteria may be more lenient because of regular scouring of algal biomass in rivers that flood frequently).

An alternative approach to correlation models also has been developed. This method consists of sampling nutrients in reference stream reaches where chlorophyll levels are deemed acceptable. Gary Ingman (Montana Department of Environmental Quality) and Vicki Watson (University of Montana) proposed this technique for use in the Clark Fork River in Montana (Dodds et al. 1997). General regional criteria have yet to be established using this method. In systems where the entire stream receives nutrient loading, or regions where all watersheds are enriched, locating suitable reference reaches may be impossible. Data from other similar streams should be used to identify the obtainable baseline nutrient concentrations in those cases.

A regression model linking TP to river phytoplankton is available (Van Niewenhuyse and Jones 1996). This model can be used to set TP criteria. The TP levels can be used to calculate corresponding TN concentrations with the Redfield ratio (Harris 1986). This model captures additional variance when watershed area is considered.

Setting nutrient criteria is difficult based on subjective impressions of what constitutes excessive levels of benthic algae. However, >200 mg/L of benthic chlorophyll generally produces a very green stream bottom (Welch et al. 1988). To further complicate matters, filamentous green algae have a less desirable appearance than brown-colored diatoms, even when the biomass of the 2 is similar. Moreover, a large amount of the variance in benthic chlorophyll levels in streams is not related to nutrient levels. We simply do not have the data in the US to predict when benthic algal community structure will shift to more nuisance forms with changes in nutrients. Preliminary data from Canada indicate that rhodophytes make up a large portion of the algal community when biomass is low, and *Cladophora* and *Melosira* prefer high nutrient water (Chételat et al. 1999). More research clearly is needed in this area, both original research and analysis of existing data. Thus, criteria based on current data will need to be set based on what amount of chlorophyll is acceptable, not on how nutrient amounts and ratios will influence algal communities.

Dissolved O_2 deficit and high pH are perhaps the most severe algal-related problems affecting the aquatic life-support characteristics of a river or stream. Deficits of DO can occur when respiration of organic C produced by photosynthetic processes in the stream exceeds the ability of reaeration to supply DO. Depletion of DO in streams was described years ago (Odum 1956). However, the severity of the deficit is difficult to predict in specific situations. Deficits of DO are most likely to occur in rivers with laminar flow (slow, non-turbulent flow), when a large algal biomass is present, with high water temperature, and during times of low light (early morning or after protracted cloudy periods). Given that such events rarely are recorded (though they may occur frequently), and that so many factors are related to DO depletion rates, existing data for most streams are insufficient to develop nutrient criteria for avoiding DO deficits. Such models probably will be developed in the future and development will be facilitated by recent improvements in tools for measuring and storing temporal data on instream DO concentrations. As more data become available, it will be possible to directly link frequency and severity of low DO events with nutrient loading.

Similar problems exist for predicting pH excursions. High pH is promoted by laminar flow and sunny conditions that, respectively, minimize atmosphere-to-water transport of CO_2 and maximize photosynthetic uptake of CO_2. Again, limited data for most streams hamper prediction of the degree of pH excursions as a function of TMDLs of N and P.

Nutrient criteria also could be set relative to other streams on a regional or national basis. Dodds et al. (1998) combined data from the EPA eutrophication survey (Omernik 1977) and several hundred streams and rivers in the US and analyzed the resulting cumulative frequency distributions. Half of the systems had TP >0.04

mg/L, and ½ had TN >0.9 mg/L. If the target is to bring streams and rivers to nutrient levels at or below current means, then using frequency distributions would be a viable approach to setting nutrient criteria. Problems with using such frequency distributions are discussed below.

Nutrient criteria may be more stringent when potential eutrophication of systems fed by rivers is a factor driving adoption of criteria. A common classification system suggests that 35 μg/L TP and a mean of 8 μg/L chlorophyll constitutes the dividing line between eutrophic and mesotrophic lakes (Organization for Economic Cooperation and Development [OECD] as cited in Rast et al. 1989). In contrast, data from Dodds et al. (1997) suggest that maximum benthic chlorophyll values are likely to exceed 200 mg/m^2 at 90 μg/L TP, and mean values of chlorophyll of 50 mg/m^2 are likely with 55 μg/L TP. Thus, unacceptable levels of chlorophyll may occur at much lower nutrient concentrations in lakes than streams.

Streams and rivers are less likely to accumulate as much algal biomass as lakes, given the same TP, because the lentic planktonic habitat is considerably more benign. Thus, there is fairly low chlorophyll yield per unit nutrient in streams. Comparing streams that flood at moderate frequency to more hydrodynamically stable artificial and spring-fed streams substantiates this view. Much higher benthic chlorophyll yield per unit TP than predicted by Dodds et al. (1997) is possible in controlled laboratory streams, outdoor artificial streams, or spring-fed rivers (Welch et al. 1992, Walton et al. 1995, Anderson et al. 1999, Welch et al., in press).

Likewise, planktonic chlorophyll yield is less in flowing waters than in lakes. A river with 8 μg/L chlorophyll would have ~48 μg/L TP, using the relationship proposed by Van Niewenhuyse and Jones (1996) for suspended chlorophyll in rivers as a function of TP. This value is ~1.4 times greater than the proposed mesotrophic/eutrophic boundary value for lakes and reservoirs (OECD as cited in Rast et al. 1989).

Last, a missing link in the above discussion is how to relate instream TN and TP concentrations to nonpoint and point sources of nutrients (i.e., to set TMDLs). Models predicting nutrient loading in streams need to be developed if mitigation strategies based on water column nutrients are to be successful. A method for determining instream TN and TP concentrations based on loading from point sources has been developed for use in the Clark Fork River (Dodds et al. 1997). Simple correlation techniques using data available in various regions may yield a relationship that can be used to predict what management strategies are necessary to bring nutrients from point sources, and consequently algal biomass, to target levels.

What factors may alter responses to nutrient control?

Variation of benthic algal biomass occurs among areas with different geology, land-use practices, and as a function of other biotic and abiotic factors. In this section, we discuss how regional differences (ecoregions) may play a role in setting nutrient criteria. In general, the relationships described above that can be used to set criteria based on algal biomass response, represent average responses.

Nutrient criteria should be set after considering the natural state of streams and rivers in an ecoregion. For example, in watersheds with high-PO_4^{3-} rock that is weathering at significant rates, low P concentrations may never occur. Large rivers will have higher TP, and yield of suspended algae will be different than in smaller streams (Van Niewenhuyse and Jones 1996). Furthermore, some watersheds have very high natural NO_3^- weathering rates (Halloway et al. 1998). Such areas naturally high in nutrients occur in several places in the US (Omernik 1977). Clearly, if nutrient levels naturally are high in a watershed, restrictive nutrient criteria cannot be met. Furthermore, when pristine systems are absent, determining natural baselines could be impossible.

Considerably greater levels of accuracy for prediction of benthic algal biomass with regression models are possible if region-specific data are available. For example, the general data sets used in regression models relating water column nutrients to benthic algae developed by Dodds et al. (1997) have a maximum r^2 of 0.43. Data from Missouri streams alone have r^2 values ranging from 0.47–0.60, depending upon year and whether TN or TP is used to predict algal biomass (Lohman et al. 1992). Biggs (1995) was able to construct a model for algal biomass with an r^2 of 0.89 in a region of New Zealand by normalizing for the effect of floods and using conductivity as a surrogate for nutrients. Fur-

TABLE 1. Various potential nutrient criteria set using different outcomes of concern related to instream nutrient concentrations. TN = total N, DIN = dissolved inorganic N.

Outcome	N (mg/L)	Total P (mg/L)	Comments
Toxicity, human	10 NO_3		US national standard
Toxicity, aquatic life, acute	0.03–5 NH_3		Fish and invertebrate data (Russo 1985)
Toxicity, aquatic life, chronic	0.005–1 NH_3		Fish data (Russo 1985, Miltner and Rankin 1998)
Oxygen deficit, pH excursion	?	?	Probably greater than levels presented below
Mean benthic chlorophyll <50 mg/m^2	0.47 TN	0.055	Large data set (Dodds et al. 1997)
Mean benthic chlorophyll <50 mg/m^2	0.25 TN	0.021	Lohman et al. (1992)
Maximum benthic chlorophyll <200 mg/m^2	3.0 TN	0.415	Calculated form Dodds et al. (1997)
Significant effect on biotic integrity index using invertebrates and fish	1.37 inorganic N	0.17	Headwater streams, Ohio (Miltner and Rankin 1998); effects less apparent in larger rivers
Systems with nutrient concentrations in upper ½	0.9 TN	0.04	Dodds et al. (1998)
Planktonic stream chlorophyll <8 μg/L	0.29 TN	0.042	Calculated from Van Nieuwenhuyse and Jones (1996); chlorophyll level from Organization for Economic Cooperation and Development (OECD, as cited in Rast et al. 1989); TN set by Redfield ratio (Harris 1986)
Lake mesotrophic/eutrophic boundary (planktonic chlorophyll <8 μg/L)	0.25 TN	0.035	OECD (as cited in Rast et al. 1989); TN set by Redfield ratio
Values set by State of Montana and cooperators	0.30 TN	0.020	Tri-State Implementation Council, Clark Fork Voluntary Nutrient Reduction Program
Levels leading to periphyton and macrophyte control	1.0 DIN	<0.020 (total dissolved)	Bow River, Alberta (A. Sosiak, Alberta Environmental Protection, personal communication)
Levels set to control summer phytoplankton		0.07	Tualatin River, Oregon (R. Burkhart, Oregon Department of Environmental Quality, personal communication).
Levels recommended to control maximum periphyton below 200 mg/m^2 for 50 d accrual	0.019 DIN	0.002 (soluble reactive)	(Biggs 2000)

thermore, all relationships that have been developed to date are from temperate regions, with most data from North America and New Zealand. Subtropical or polar regions could have quite different relationships. Thus, if data are available for an ecoregion, they should be used to set criteria for that region. Extant data such as state and tribal water quality records, USGS National Water Quality Assessment Program data, and Environmental Monitoring and Assessment Program results may serve as sources for such analyses.

Streams in a local region also may exhibit different relationships between TN or TP and benthic chlorophyll than those observed with larger-scale data sets. Thus, large, generalized data sets should not be the 1st choice for setting criteria, if local data are available. For example, the TN and TP values that yield a mean benthic chlorophyll of 50 mg/m^2, were lower for the detailed data set from Missouri than those from a larger data set (Table 1).

Nevertheless, one should not expect that the nutrient concentration yielding a given peri-

phytic biomass will be markedly different among regions if other factors (i.e., light, grazing, etc.) are similar. For example, most regression relationships for chlorophyll-TP in lakes show slopes or chlorophyll:TP ranging from 0.5–1.0 (Ahlgren et al. 1988). Invertebrate grazing may result in low chlorophyll yield per unit nutrient in streams regardless of ecoregion (Bourassa and Cattaneo 1998), as is the case in lakes.

One potential problem with the ecoregion approach is that variation over time and space within a small area may be as great as the variation among ecoregions. The nutrient bioassays of Wold and Hershey (1999) demonstrate high variation of responses to N or P additions in 6 watersheds within 100 km of each other. The responses also were variable across season. Similar seasonal responses have been documented in New Zealand streams (Francoeur et al. 1999).

All the data sets that have been published linking algal biomass to water column nutrients in rivers and streams have a potential statistical problem (Lohman et al. 1992, Biggs 1995, Van Niewenhuyse and Jones 1996, Dodds et al. 1997, Chételat et al. 1999). Investigators may have introduced bias in site selection because sites were not selected randomly. In many cases, study sites are selected specifically to represent the broadest possible range of site types. Thus, extremely eutrophic and oligotrophic systems may be overrepresented. Such models may work well for the streams used to construct the models, but their application should be viewed with caution. For example, Dodds et al. (1997) reported relationships among nutrients and chlorophyll derived from literature values. The investigators who conducted this literature analysis had no way of knowing why investigators choose to investigate particular sites or if all data were reported. Streams with low amounts of periphyton may have been excluded, or researchers may have preferred to work in pristine systems. Last, much ecological investigation has concentrated on temperate, forested streams, which may have low levels of nutrients and where canopy cover may have restricted algal growth. Temperate forested streams may not be globally representative of all streams because they provide $<1/3$ of the runoff from the earth's continents (Dodds 1997). Thus, future sampling strategies to generate data that will be used to link stream eutrophication with nutrients should attempt to avoid investigator-specific biases. The models for setting criteria should be based on representative streams with data taken from the full population of streams and with each type of stream sampled in proportion to its relative occurrence. Such an approach has been taken in lakes (Peterson et al. 1999). Large data sets such as those collected by the USGS water quality monitoring network of the National Water Quality Assessment Program may be useful because sites could be selected from the databases to provide data specific to individual ecoregions.

If streams and rivers are turbid as a result of suspended particles, nutrient enrichment will have less influence on trophic status of the entire system. Sediments attenuate light, which becomes the factor limiting ecosystem production. However, even in turbid systems, enrichment may increase periphyton and macrophyte production in shallow portions of the river. Similarly, extensive shading by a riparian canopy will inhibit algal growth. Both conditions reduce chlorophyll yield per unit nutrient.

If macrophyte production predominates in streams and rivers, setting nutrient criteria will be difficult. We are not aware of any general published relationships between water column nutrients and macrophyte biomass. Such relationships may be very difficult to establish for macrophytes that are able to acquire nutrients from sediments through their root systems. However, nutrient control resulted in lowered macrophyte biomass in the Bow River, Alberta (A. Sosiak, Alberta Environmental Protection, personal communication), so future work on macrophyte-nutrient relationships could yield useful predictive models.

Conclusions

Many factors can regulate primary producers in streams, including nutrient availability, hydrodynamics, grazing, turbidity, riparian shading, and human impacts (e.g., addition of toxic compounds, global change, introduced species, watershed development). However, nutrient inputs are usually the most effectively managed factor. Factors in addition to nutrients need to be considered mainly because they can lead to cases of low algal biomass with high nutrients. Although these additional factors may decouple nutrient enrichment from algal biomass, most of these (e.g., flooding, grazing, turbidity) are not

easily controlled at most sites. Thus, we are left with setting nutrient criteria as the primary way to mitigate problems of excessive algae.

Developing a single value that can be used for nutrient criteria in streams and rivers will be difficult, given the variety of reasons for setting the criteria (Table 1). To protect human health, no more than 10 mg/L NO_3^--N should be present. To avoid chronic toxicity by NH_3, no more than 0.02 mg/L NH_3^--N should be present. If the concern is eutrophication, then setting criteria for TN and TP is most reasonable.

If streams are not turbid, preventing maximum benthic chlorophyll levels from exceeding 200 mg/m^2 is reasonable because streams with higher levels are not aesthetically pleasing, and their recreational uses may be compromised. For benthic chlorophyll to remain below 200 mg/m^2 at the very least, TN should remain below 3 mg/L, and TP below 0.4 mg/L. Based on cumulative frequency distributions of nutrients, and assuming that ~½ the systems in the US have been impaired by excessive nutrients, levels of TN and TP would be set at 0.9 and 0.4 mg/L, respectively. If a mean of 50 mg/m^2 chlorophyll is the target (thus ensuring chlorophyll is <100 mg/m^2 most of the time), TN should be 0.47 and TP 0.06 mg/L. Lower levels for nutrient criteria should be considered for regions with more pristine systems (e.g., TN and TP levels of 0.3 and 0.02 mg/L, respectively, were chosen for the Clark Fork River in Montana, Table 1). If systems downstream are to be protected, even lower stream nutrient concentrations will be necessary in some situations.

A significant amount of monitoring data are necessary to refine recommendations for nutrient criteria. Some regions and agencies have data that can be used for this purpose. Data that would be useful to collect or glean from existing sources for many more systems include seasonal means and maxima for benthic and planktonic chlorophyll, associated water column nutrients, and diurnal DO concentrations for a variety of stream types. Such data should be collected in a way that avoids sampling bias. Data on macrophyte abundance related to nutrients, reference streams with acceptable algal and macrophyte biomass, and factors related to dominance by nuisance algal and macrophyte species also are sorely lacking for many regions.

Establishing rational criteria will require bridging the gap between managers and scientists. The managers will provide the realistic assessment of what needs to be accomplished, whereas the scientists can suggest the best available means to reach the management goals. Continued interplay between applied and basic approaches will be necessary if eutrophication in streams is to be controlled in an efficient manner.

Acknowledgements

W. D. thanks Kirk Lohman for early conversations and subsequent collaborations that led to much of this work. Jack Jones, Vicki Watson, Val Smith, and Jan Stevenson also provided useful input into developing these ideas. Barry Biggs, Debra Hart, Marty Gurtz, Amanda Parker, and 2 anonymous reviewers commented on the manuscript and Dolly Gudder provided technical assistance. This research was supported in part by the Konza LTER grant from the National Science Foundation. This paper is contribution no. 99-375-J from the Kansas Agricultural Experiment Station.

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Pergamon

PII: S0043-1354(96)00389-2

Wat. Res. Vol. 31, No. 7, pp. 1738–1750, 1997
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Printed in Great Britain
0043-1354/97 $17.00 + 0.00

DEVELOPING NUTRIENT TARGETS TO CONTROL BENTHIC CHLOROPHYLL LEVELS IN STREAMS: A CASE STUDY OF THE CLARK FORK RIVER

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(Received April 1996; accepted in revised form November 1996)

Abstract—Approaches for assessing the effects of lowering nutrients on periphyton biomass in streams and rivers are poorly developed in contrast to those for lakes. Here we present two complementary approaches to assess target nutrient concentrations in streams, given desired mean and maximum standing crops of benthic algal chlorophyll. In the first approach, a reference portion or reach of the river that typically exhibits acceptable levels of benthic chlorophyll is identified (i.e. seasonal mean and maximum values do not exceed desirable levels), and the target levels for instream nutrient concentrations are defined by mean nutrient levels in the reference region. In the second approach, regression and graphical analyses of a large stream database are used to identify acceptable levels of instream total N and total P. The first approach supplies site-specific nutrient targets, whereas the second places nutrient control into a broader, more comparative perspective. In order to link these target concentrations to specific nutrient control measures, we describe a spreadsheet model that can be used to translate changes in external loading by point sources into predicted new instream nutrient concentrations. These quantitative methods are applied here to the control of nuisance algal growth in the Clark Fork River, Montana. We suggest that, in general, maintenance of mean instream total N concentrations below 350 $\mu g\ l^{-1}$ and total P below 30 $\mu g\ l^{-1}$ will result in mean benthic algal chlorophyll *a* density below nuisance levels of 100 mg m^{-2} in most streams.
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Key words—eutrophication, periphyton, streams, rivers, nitrogen, phosphorus, benthic algae

INTRODUCTION

Eutrophication of most fresh waters is dependent upon supplies of nitrogen and phosphorus (Vollenweider, 1968). In the case of lakes and reservoirs, a strong quantitative framework has been developed, over the past three decades, that allows the prediction of algal biomass and other water quality parameters from nutrient loading and water column nutrient concentrations (OECD, 1982; Smith, 1982; Canfield, 1983; Reckhow and Chapra, 1983; Ryding and Rast, 1989). These tools are employed with great success in water quality management of lakes worldwide (Sas, 1989; Cooke *et al.*, 1993).

In contrast, the development of a comparable quantitative framework for algae in flowing waters has lagged far behind. Although nuisance algal growth in nutrient-enriched streams and rivers is common in North America and elsewhere (Wong and Clark, 1976; Lembi *et al.*, 1988; Welch *et al.*, 1988), the general quantitative relationships between nutrient supplies and algal biomass in lotic systems are not well characterized. It is clear from laboratory studies (Klewer and Holm, 1980; Horner *et al.*, 1983) and from nutrient enrichment studies in outdoor artificial streams (Stockner and Shortreed, 1976; Bothwell, 1985; Watson *et al.*, 1990), that nutrients stimulate the growth of stream periphyton. However, empirical models for periphyton biomass in natural streams and rivers are rare (Welch *et al.*, 1988; Biggs and Close, 1989; Lohman *et al.*, 1992) and may be specific to localized areas. Consequently, managers find it difficult to make informed decisions regarding the nutrient control of nuisance algal biomass in lotic systems. The purposes of this paper are to present a series of approaches that can be used to help establish nutrient control criteria in rivers and streams where eutrophication has been deemed excessive, and to illustrate the application of these techniques to the Clark Fork River, Montana (Fig. 1).

Benthic chlorophyll *a* routinely in excess of 100 mg m^{-2} is observed upstream from Alberton in the Clark Fork of the Columbia River, Montana (Watson, 1989; Watson *et al.*, 1990; Ingman, 1992a, b). These densities are generally undesirable from a water quality point of view (Horner *et al.*, 1983; Welch *et al.*, 1988). The State of Montana was

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interested in four eutrophication management questions: (1) what concentrations of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) and ratios of N:P are required to yield chlorophyll *a* densities of 50, 100 and 200 mg m^{-2}?; (2) should total phosphorus and nitrogen be managed instead of DIN and SRP, when assessing potential algal density?; (3) will P removal make a difference in total algal biomass and relative abundance of *Cladophora* (a common nuisance filamentous green alga) in apparently N-limited upstream reaches?; (4) would nearly complete removal at upstream point sources only during the summer irrigation season result in significant reductions in river-wide periphyton levels? We assume that similar questions, typically, will be asked by those attempting to control eutrophication in other flowing waters and suggest that the general approach presented here provides a quantitative framework that has broad applicability.

METHODS

Site description

The Clark Fork River drains approximately 57,000 km^2 of western Montana (Fig. 1) and this paper considers roughly the upper 370 km. There are nine major tributaries and nine towns along this portion of the river. Land use is predominantly forest, rangeland and agriculture in this basin (Ingman, 1992a).

Previous assessments of nutrient limitation on the Clark Fork include artificial stream measurements of the effect of N and P additions (Watson, 1989), and nutrient uptake measurements made using *Cladophora* collected from the river (Lohman and Priscu, 1992). Nutrient deficiency has also been evaluated using cellular N and P concentrations in *Cladophora* (Lohman and Priscu, 1992) and ratios of DIN:SRP (Watson *et al.*, 1990). These studies together indicated that N limitation is common in the Clark Fork system; at times a balance may occur between N and P limitations; and N limitation appears to decrease downstream.

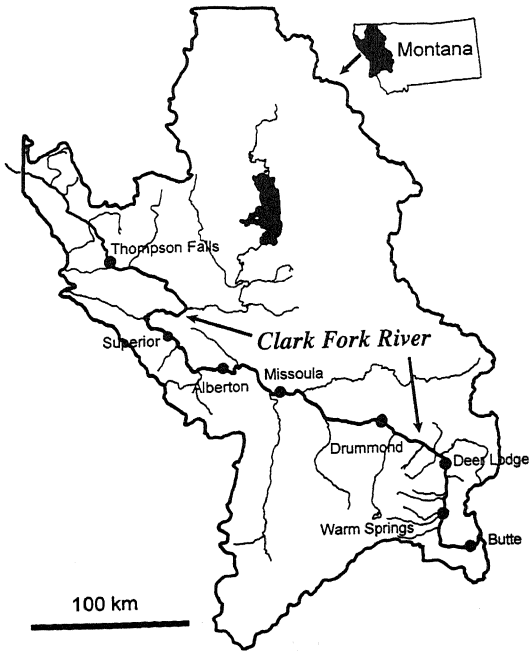

Fig. 1. The Clark Fork River basin, Montana.

Analysis and modeling

Models relating TN, TP, DIN, SRP and other variables to benthic chlorophyll *a* were constructed from a cross-sectional database derived from over 200 distinct sites or rivers throughout North America, Europe and New Zealand (Table 1). Means were determined for 2–3 month periods from all seasons, but summer samples were the most common. Values of chlorophyll *a* in these studies were derived from artificial (about 1/3 of the data) and natural substrates. Stepwise multiple regression was used to construct a series of eutrophication models. Different combinations of variables were forced in different orders (SAS, 1988) and the 95% prediction confidence levels for chlorophyll *a* were calculated for each combination of variables. Models for seasonal mean and maximum chlorophyll *a* are reported here because they are likely to be most relevant to users and to those concerned with controlling stream eutrophication. A graphical probabilistic method based on the procedures of Heiskary and Walker (1988) was also developed. In this method, the percentages of chlorophyll *a* values (mean and maximum) exceeding a series of defined levels (50, 100, 150 and 200 mg m^{-2} chlorophyll *a*) were calculated for a series of specified ranges of nutrient concentrations. Other factors influencing benthic algal biomass (latitutde, temperature, stream gradients, discharge and light) were also investigated using nonparametric correlation analysis, but none of these variables were found to be as useful as a predictor of stream chlorophyll *a* as TN or TP (Dodds *et al.*, unpubl.), so they were not included in our final model development.

We defined nuisance levels of benthic algal chlorophyll *a* as mean values exceeding 100 mg m^{-2} and maximum values exceeding 150 mg m^{-2}. Similar criteria have been used previously (Horner *et al.*, 1983; Welch *et al.*, 1988), and we consider benthic chlorophyll *a* of values less than these critical limits to be indicative of acceptable conditions with regard to benthic algal biomass. Our analysis does not apply to macrophyte growth, a separate water quality parameter that we have not addressed here.

The bulk of the data on nutrient inputs to the Clark Fork were derived from monitoring programs conducted by the State of Montana Department of Health and Environmental Sciences (Ingman, 1992a, b). These data include measurements of DIN, SRP, total N (TN), and total P (TP) both for the river itself and for major point sources and tributaries to the river. Information was also available on benthic chlorophyll *a* levels (Watson, 1989; Watson *et al.*, 1990; G. Ingman, personal communication; V. Watson, personal communication), and the composition and abundance of attached algal species (Bahls, 1989; Weber, 1991; Weber, 1993).

A spreadsheet model was developed to predict the fate and transport of nutrient loadings throughout the study area. This one-dimensional, steady-state, spreadsheet model can be used to calculate the effects of changes in external loadings on instream nutrient concentrations. It uses a mass-balance approach to calculate the spatial distributions of TN and TP over a section of the river, as well as to predict changes in water chemistry that result from external load reductions at different locations. The model portrays conditions during June, July and August for a broad geographic area (Fig. 1) from Warm Springs down to Superior (approximately 370 river km). The results of the model are reported here as a longitudinal profile of instream concentrations and loadings of TN.

The spreadsheet model consists of 28 river segments ranging from 0.5 to 80 km in length. The length of each segment is defined by the locations of known inputs (i.e. point sources and major tributaries) and the locations of existing sampling stations. The TN concentrations in each

Table 1. Location, number of sites, source and type of data collection sites. N = natural substrate; A = artificial substrate

Location	No.	Type	Source
Washington II, Montana 4, Sweden 6	21 rivers	N	Welch *et al.*, 1988
Ozark, Missouri, 2 yr	21 streams	N	Lohman *et al.*, 1992, Lohman, 1988
Pennsylvania, Michigan, Idaho, Oregon, (four streams each)	16 streams	N	Bott *et al.*, 1985
New Zealand	16 streams	N	Biggs 1995
Missouri urban, forest and pasture catchements, 2 yr	13 streams	N	Smart *et al.*, 1985; Smart, 1980; Jones *et al.*, 1984
New Zealand	9 streams	N	Biggs and Close, 1989
New Zealand	9 streams	A	Biggs, 1988
North America (Southeastern U.S. to Arctic)	8 streams	N	Meyer *et al.*, 1993
Spokane River, Washington, 2 yr, both substrate types	8 sites	N, A	Funk *et al.*, 1983; Nielsen *et al.*, 1984
New Zealand, 2–4 sites each stream	7 streams	N	Welch *et al.*, 1992
North Carolina	7 streams	N	Smith (unpublished)
Virginia and New Hampshire	6 streams	N	Hornberger *et al.*, 1977
Idaho	6 sites	A	Delong and Brusven, 1992
Quebec	5 streams	N	Naiman, 1980; Naiman, 1981; Naiman, 1983
Artificial streams	5 treatments	A	Horner *et al.*, 1990
Montana	4 sites	A	Marcus, 1980
Kansas artificial streams	4 treatments	A	Tate, 1990
Oregon Cascades	3 streams	N	Lyford and Gregory, 1975
British Columbia	3 streams	N	Stockner and Shortreed, 1976
Upper Mississippi River navigation pools, four sites each, 2 yr	2 pools	A	Luttenton, 1982; VanSteenberg, 1983
Fertilized and unfertilized Arctic stream	1 stream	N	Peterson *et al.*, 1985
Tennessee, 2 yr	1 stream	N	Rosemond, 1994
New Jersey, six sites	1 stream	A	Flemer, 1970
Michigan, four sites	1 stream	A	Lenon *et al.*, 1979
New Zealand grazed and ungrazed	1 stream	A	Biggs and Lowe, 1994
Danish lowland	1 stream	N	Sand-Jensen *et al.*, 1988
British Columbia	1 stream	A	Perrin *et al.*, 1987

segment were calculated from all known nutrient inputs (point sources, plus direct inflows from the segment above) and nutrient losses (loss downstream, dilution by tributaries). An additional correction factor was calculated for each segment to estimate changes in nutrient levels that could not be measured directly (e.g. nonpoint sources, gain or loss associated with groundwater exchange, biotic uptake or release, sedimentation and burial). The correction factor was calculated as the proportion of nutrients gained or lost in the segment that could not be accounted for by known inputs and losses. Thus, for cases in which the water in a segment exhibited a 50% decrease in TN concentration during its transit through the segment that could not be accounted for by known inputs and exports, we assumed that 50% of TN entering the segment similarly would be lost under all future management scenarios.

A summer period of 21 June–21 September was chosen for our spreadsheet modeling because: (1) it is the time period in which past data suggest that benthic algal biomass problems are most likely to occur; (2) the use of annual mean values would not adequately reflect the period when critical low flows may occur; and (3) this is the time period used in previous stream water modeling efforts for the State of Montana. The model was calibrated using longterm (1988–1992), summer mean TN concentrations available from the State of Montana for both normal summer flows and for critical low flows (7Q10 hydrologic regimes). The 7Q10 flow regime is minimum discharge summed over a 7 day period having an expected return time of 10 yr.

RESULTS

Dissolved inorganic nutrient criteria

The State of Montana wished to know what concentrations of streamwater, DIN and SRP and ratios of N:P were necessary to yield chlorophyll *a* densities of 50, 100 and 200 mg m^{-2}. General relationships between benthic chlorophyll and dissolved inorganic nutrients are characterized by extremely high variance. Seasonal mean values for SRP or DIN explain a very low proportion of the variance in observed chlorophyll *a* (Table 2 and Fig. 2A, B) from our entire database although values from the Clark Fork alone show a closer relationship. Thus, these data suggest that practical regulations for general external nutrient loading for stream eutrophication control should not be based upon instream SRP and DIN levels, because the prediction uncertainty inherent in such an approach may preclude the satisfactory management of benthic chlorophyll *a*. To illustrate the weakness of modeling approaches based on dissolved nutrients, we used empirical models for DIN and SRP (equations (10) and (20), Table 2) to estimate three desired levels of control (50, 100 and 200 mg m^{-2}). Note that the 95% confidence levels for the predicted seasonal mean chlorophyll *a* densities span 1–2 orders of magnitude (Table 3).

The relationships between the summer mean chlorophyll *a* values found in the Clark Fork in 1990 (V. Watson, personal communication) and their associated mean DIN and SRP values (G. Ingman, personal communication) are plotted with values compiled from other sites in Fig. 2. Using data from the Clark Fork alone to extrapolate to lower values of DIN and SRP would be difficult because chlorophyll *a* measurements were taken from artificial substrates and a limited range of DIN and SRP concentrations. These data do suggest that lower inorganic nutrients may lead to lower chlorophyll *a* densities in the Clark Fork.

Total N and P criteria

Another question posed by Montana was: should instream total P and N concentrations be managed instead of DIN and SRP when assessing potential algal biomass? Our analyses revealed that both total N and total P are related more strongly with benthic algal biomass than are dissolved inorganic N or P

(Table 2). We used three complementary approaches to predict the instream TN and TP concentrations that should correspond to desired chlorophyll *a* densities in the Clark Fork: (1) the regression models (Table 2) were used to calculate acceptable target concentrations of TN and TP and their associated confidence intervals; (2) a graphical probabilistic approach was used to predict target TN and TP levels; and (3) measurements of water quality from six reference sites consistently exhibiting acceptable levels of benthic algal biomass were averaged to develop independent estimates of target instream TN and TP concentrations.

Regression model results. Chlorophyll *a* densities and TN and TP concentrations in the Clark Fork River lie well within the range of reported values throughout the world (Fig. 2C, D). Regression analyses of the full data set revealed that the greatest amount of variance in benthic chlorophyll *a* was accounted for by instream concentration of TN, followed by TP (Table 2).

We subsequently determined target instream TN and TP concentrations corresponding to the three critical mean and maximum chlorophyll *a* values (50, 100 and 200 mg m^{-2}) with a three step process. (1) The appropriate TN and TP values were calculated using the Redfield ratio of 7.23 g N:1 g P, by mass (Ryding and Rast, 1989); these values were assumed to result in balanced algal growth. Setting either TN or TP in these combined models is necessary, because otherwise an enormous number of solutions can satisfy the equations. (2) Equations (7) and (17) (Table 2, mean and maximum chlorophyll *a*, respectively) were used, and TN was varied by iteration to find the values of TN and associated TP that would result in the desired target chlorophyll levels. The associated 95% confidence interval was calculated for each of the TN values corresponding with the targets. (3) Equations (4) and (14) (Table 2, mean and maximum chlorophyll *a*, respectively) were used with TN set according to the iterated TP value using the Redfield ratio, and the TP values that correspond to the values of 50, 100 and 200 mg chlorophyll *a* m^{-2} were found by iteration.

Considerable statistical variance exists in the TN-TP-chlorophyll *a* models that we developed (Tables 4 and 5). However, these confidence limits are on average 40% less than those for the DIN and SRP chlorophyll *a* predictions (Table 3). If instream mean TN and TP concentrations in the Clark Fork are reduced by external nutrient loading controls such that a target mean chlorophyll *a* of 50 mg m^{-2} is attained, then the 95% confidence intervals suggest that the resulting mean chlorophyll *a* generally would not exceed 170 mg m^{-2}. Maximum chlorophyll *a* levels could still, periodically, exceed 150 mg m^{-2}. If the seasonal mean TN concentration is reduced to 275 $\mu g\ l^{-1}$, equations (7) and (17) predict that maximum chlorophyll *a* will be 100 mg m^{-2} (Table 4). Similar results were obtained using equations (4) and (14); with TP up to 35 $\mu g\ l^{-1}$, and TN at 252 $\mu g\ l^{-1}$, the maximum chlorophyll *a* would be about 100 mg m^{-2} (Table 5).

Probabilistic modeling approaches and results. This probabilistic analysis was based on methods developed by Heiskany and Walker (1988) to assess the risk of exceeding user-specified chlorophyll *a* levels. We calculated the frequency with which the three critical chlorophyll *a* levels posed in question 1 (50, 100 and 200 mg m^{-2}) were exceeded in our water quality database across defined ranges of instream mean TN concentrations (Fig. 3A, B). This analysis revealed that neither the seasonal mean nor the maximum densities of benthic chlorophyll *a* exceed

Table 2. Regressions with mean and maximum chlorophyll *a* as the dependent variables. Units are: mean chlorophyll *a* and maximum chlorophyll *a* in mg m^{-2}; TP, TN, DIN and SRP in $\mu g\ l^{-1}$; TN:TP ratio by mass. The value for the coefficient of regression for each independent variable is given in parentheses. Statistical significance for each variable is indicated by the superscripts.* $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$. This analysis is based upon values compiled for 205 streams or sites throughout North America and New Zealand

Dependent variable	Independent variables				Adjusted r^2
	intercept	1st	2nd	3rd	
log (mean Chl *a*)					
(1)	1.091***	logTP (0.2786***)	—	—	0.089
(2)	0.7950*	logTP (0.6021)	$(logTP)^2$ (−0.08113)	—	0.086
(3)	− 0.1238	logTP (1.3418***)	$(logTP)^2$ (−0.19749***)	TN:TP (0.00727*)	0.412
(4)	− 0.4285	logTP (0.92178***)	$(logTP)^2$ (−0.16468**)	logTN (0.37408**)	0.429
(5)	0.01173	logTN (0.59490***)	—	—	0.348
(6)	− 2.36566*	logTN (2.28959**)	$(logTN)^2$ (−0.29390*)	—	0.371
(7)	− 3.22360**	logTN (2.82630**)	$(logTN)^2$ (−0.431247**)	logTP (0.25464**)	0.430
(8)	1.02412***	logSRP (0.2327**)	—	—	0.053
(9)	0.83531***	logDIN (0.29576***)	—	—	0.140
(10)	0.37366	logDIN (0.42093**)	logSRP (−0.03812)	—	0.166
log (max Chl *a*)					
(11)	1.4995***	logTP (0.28651***)	—	—	0.071
(12)	1.25846**	logTP (0.55015)	$(logTP)^2$ (−0.06611)	—	0.067
(13)	0.21620	logTP (1.47096***)	$(logTP)^2$ (−0.22211***)	TN:TP (0.007238*)	0.368
(14)	0.00652	logTP (1.10067***)	$(logTP)^2$ (−0.19286**)	logTN (0.3129*)	0.370
(15)	0.47022	logTN (0.60252***)	—	—	0.284
(16)	− 1.80885	logTN (2.22712*)	$(logTN)^2$ (−0.28175)	—	0.299
(17)	− 2.70217*	logTN (2.78572**)	$(logTN)^2$ (−0.43340**)	logTP (0.30568**)	0.354
(18)	1.71065***	logSRP (0.09080)	—	—	0
(19)	1.47894***	logDIN (0.22488***)	—	—	0.068
(20)	1.06686***	logDIN (0.41661**)	logSRP (−0.177211)	—	0

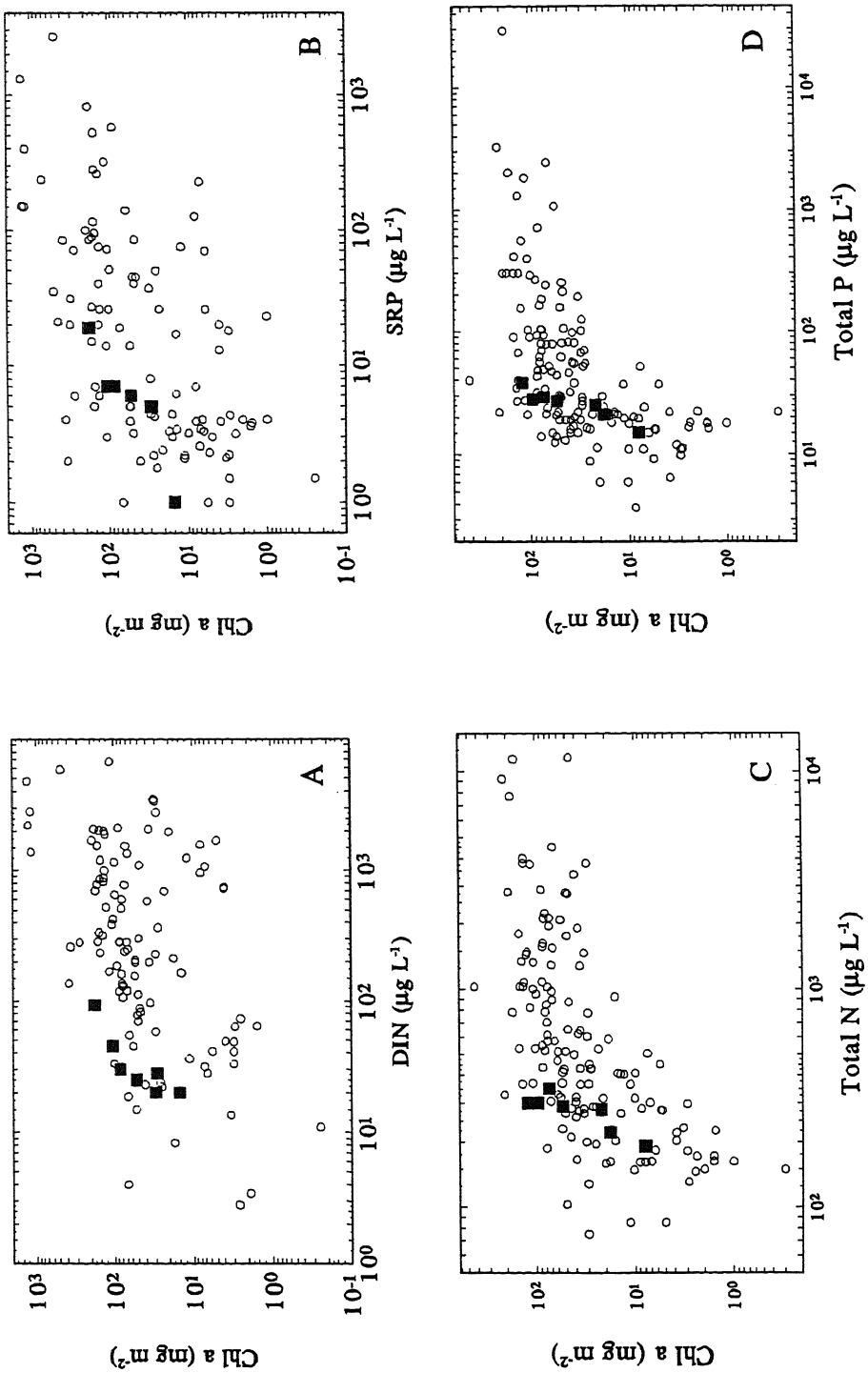

Fig. 2. Mean summer (June–September) chlorophyll *a* for the Clark Fork River and seasonal mean data from other sites throughout the world as a function of DIN (A); SRP (B); TN (C); and TP (D). The filled squares are data points from the Clark Fork (courtesy of V. Watson and the Montana Department of Health and Environmental Sciences).

Table 3. Estimates of SRP and DIN levels required to yield mean and maximum chlorophyll *a* levels of 50, 100 and 200 mg m^{-2}. Calculated from relationships presented on Table 1. — = not possible to calculate from our model

Target Chl *a* (mg m^{-2})	SRP (mg l^{-1})	Lower 95% confidence (mg m^{-2})	Upper 95% confidence (mg m^{-2})	DIN (mg l^{-1})	Lower 95% confidence (mg m^{-2})	Upper 95% confidence (mg m^{-2})
Mean 50	0.79	2.44	889	0.83	4.75	505
100	15.6	3.2	1383	8.67	7.92	883
200	307	—	—	90.3	11.1	1292
Max. 50	0.0008	1.98	1333	0.01	3.79	693
100	1.54	3.51	3152	0.21	7.55	1338
200	—	—	—	4.53	14.7	2776

100 mg m^{-2} in the majority of cases, when TN was kept below 200 μg l^{-1} (Fig. 3A, B). Similarly, when mean TN concentrations remained at or below 500 mg m^{-2}, mean benthic chlorophyll *a* densities exceeded 150 mg m^{-2} in only 5% of the cases. A similar approach using TP plots suggested mean chlorophyll of 100 mg m^{-2} were present in over half the cases where TP is <50 μg l^{-1} (Fig. 3C). Maximum chlorophyll exceeded 150 mg m^{-2} at least half the time when TP exceeded 50 μg l^{-1} (Fig. 3D). These plots also further confirmed a clearer relationship between TN and chlorophyll *a* than between TP and chlorophyll *a*.

Reference station approach. In this analysis we used a reference station approach, analogous to that used by Ingman (1992b), to develop target instream DIN and SRP levels for the Clark Fork River. In this approach, six sampling stations (stations 8.5, 9, 13, 15.5, 24 and 25 from Ingman, (1992a, b)) were selected that exhibit subjectively determined acceptable levels of benthic chlorophyll *a*. Target nutrient concentrations then were calculated as the 1988–1992 means of summer (21 June–21 September) TN and TP. The mean summer TN and TP concentrations averaged 318 and 20.5 μg l^{-1}, respectively, during this period. These mean concentrations from the reference reaches are consistent with the target values obtained from both the regression and the probabilistic analyses.

Nutrient removal and Cladophora *dominance*

The State of Montana wished to know if P removal would make a difference in total algal biomass and relative *Cladophora* abundance in apparently N-limited upstream reaches. As is true of many over-enriched streams, nuisance levels of *Cladophora* biomass constitute a water quality variable of concern in the Clark Fork River. We, thus, sought to find the nutrient conditions that correlated most strongly with *Cladophora* growth. No consistent relationship could be found between *Cladophora* abundance and instream TN:TP ratios, but the lowest *Cladophora* abundance was observed primarily at extremely high TN and TP concentrations (Fig. 4B, C). Therefore, it is difficult to predict on the basis of our current database whether a policy of P removal will alter the relative abundance of *Cladophora* in upstream reaches of the Clark Fork. However, a loading management policy favoring high instream TN and TP concentrations designed to reduce *Cladophora* abundance would lead, in turn, to excessive total levels of chlorophyll *a* and would not be a viable management option.

Reach specific management options

The final issue of concern to the State of Montana was that if nearly complete removal at upstream point sources only occurred during the summer irrigation season, it would result in significant reductions in river-wide periphyton levels. Our one-dimensional, steady-state, spreadsheet model was used to create predictions for: (1) baseline TN concentrations with no loading controls imposed; (2) a scenario in which a 100% removal of the Deer Lodge inputs was imposed; and (3) control at Butte and 100% removal of Deer Lodge inputs. The second two control scenarios were modeled because they were identified previously as the most viable management approaches by the State of Montana. Both sets of predictions were run first for typical summertime flow conditions and then for conditions of critical low flows over a 7 day period expected to occur every 10 yr on average (the 7Q10 conditions). Here, we present only the typical summertime flow results.

Under normal summer baseline flow conditions, the 350 μg l^{-1} TN target is exceeded at many stations under normal summer flow conditions (Fig. 5A). A 100% removal of nutrient inputs from Deer Lodge alone results in compliance with a 350 μg l^{-1} TN target at all stations downstream (Fig. 5B). A 100% elimination of the Deer Lodge inputs coupled with a

Table 4. Levels of instream TN and TP required to reach target mean and maximum chlorophyll *a*. Levels calculated from equations (7) and (17) (Table 2) with TP set to Redfield ratio

Target Chl *a* (mg m^{-2})	TN (μg l^{-1})	TP (μg l^{-1})	Lower 95% (mg m^{-2})	Upper 95% (mg m^{-2})
Mean 50	450	62.3	6.56	170
100	1600	221	26	420
200	3000	415	15.5	436
Max. 50	145	20.1	7.8	407
100	275	38.1	29	1380
200	650	90	63	3310

Table 5. Levels of instream TN and TP required to reach target mean and maximum chlorophyll *a*. Levels are calculated from equations (4) and (14) (Table 2) with TN set to Redfield ratio

Target Chl *a* (mg m^{-2})	TN (μg l^{-1})	TP (μg l^{-1})	Lower 95% (mg m^{-2})	Upper 95% (mg m^{-2})
Mean 50	470	65	9	234
100	1423	197	11	295
200	7570	1020	33	1072
Max. 50	115	16	7	352
100	252	35	15	710
200	650	90	28	1356

7.0 mg l^{-1} effluent TN limit at the Butte WWTP would extend compliance with a 350 μg l^{-1} TN target to all stations upstream of Deer Lodge and downstream of the mouth of Warm Springs Creek (Fig. 5C). Results from critical low flows show that more stringent loading controls are required to reach the TN target (data not shown). Similar results were also obtained for TP reduction simulations. Less stringent controls are likely to lead to a lack of compliance at one or more downstream stations.

DISCUSSION

Utility of models using dissolved inorganic nutrients and TN and TP

The poor correspondence between levels of algal biomass and DIN and SRP seen in Fig. 2(A) and (B) is a direct consequence of the biotic processes that control upstream concentrations of DIN and SRP. At any given time, the concentrations of dissolved inorganic nutrients are determined by the balance between uptake and regeneration (remineralization), which act simultaneously to control dissolved nurients at approximately equilibrium levels and are very resistant to perturbations (Dodds, 1993). Dissolved inorganic nutrients are also poorly correlated with the biomass or activity of algae in lakes (Brylinsky and Mann, 1975). Lake managers use TP rather than SRP to make recommendations for target chlorophyll *a* levels (e.g. OECD 1982; Reckhow and Chapra, 1983; Ryding and Rast, 1989; Sas, 1989). Similarly, we believe that using SRP in streams probably would not be fruitful in the absence of explicit site-specific models for nutrient uptake and remineralization.

Although some published data on uptake rates of inorganic N and P are available for the Clark Fork, we have been unable to obtain the corresponding direct estimates of N and P remineralization rates. A similar lack of data on these two important parameters is also likely to be the case for most, if not all, of the rivers in the world. Thus, it is extremely difficult to estimate, with confidence, how changes in nutrient loading would influence instream DIN or SRP concentrations, and the high variance (Fig. 2A, B) makes it difficult to relate changes in DIN or SRP to subsequent changes in benthic algal biomass.

The most parsimonious explanation for the stronger relationship observed here between chlorophyll *a* and TN and TP is that instream TN and TP concentrations are more indicative of the nutrients that are ultimately biologically available for benthic algal growth than are DIN and SRP. We tested a variety of other factors in our global data set (including DIN, SRP, latitude, stream slope, mean and maximum flows and temperature), and none were correlated as strongly with benthic chlorophyll *a* as TN and TP (Dodds *et al.* unpublished). Thus, management criteria based on TN and TP probably offer the best option for managers wishing to control nuisance algal growth in streams and rivers. We recognize that water column TN and TP values include N and P in algal cells suspended from the benthos. However, much of the TN and TP is ultimately biologically available. Even if all the TN and TP (Bradford and Peters, 1987) is not available, TN and TP are probably the best predictors of available nutrients.

Other investigators have also used water column nutrients to predict algal biomass in streams. For example, local TP-based models have been presented for Missouri (Lohman *et al.*, 1992) and New Zealand (Biggs and Close, 1989). As far as we know, the general application of these models as a management tool for benthic chlorophyll has not been explored. In addition, suspended chlorophyll has been positively correlated with TP in streams as a function of stream catchment area (Van Nieuwenhuyse and Jones, 1996), but this model cannot be applied to benthic algal biomass.

Possible targets for TN and TP

The results presented above provide three separate, but complementary, estimates of target concentrations for TN and TP in the Clark Fork. Regression methods suggest that 275 μg l^{-1} TN generally will yield acceptable (100 mg m^{-2} mean and 150 mg m^{-2} maximum) chlorophyll *a* levels. The probabilistic approach suggests that when TN is in the range of 200–500 μg l^{-1}, chlorophyll will be acceptable in most cases. The reference station approach demonstrated that, generally, an acceptable site had an average TN concentration of 318 μg l^{-1}. Given the strong similarities among the target values derived from these three different approaches, we adopted a value of 350 μg l^{-1} TN as a provisional target level that would allow for some external input of TN to the Clark Fork system and yet should avoid frequent episodes of excessive benthic algal growth.

A corresponding phosphorus target value of 38 μg l^{-1} TP (Table 4) is suggested, when the Redfield

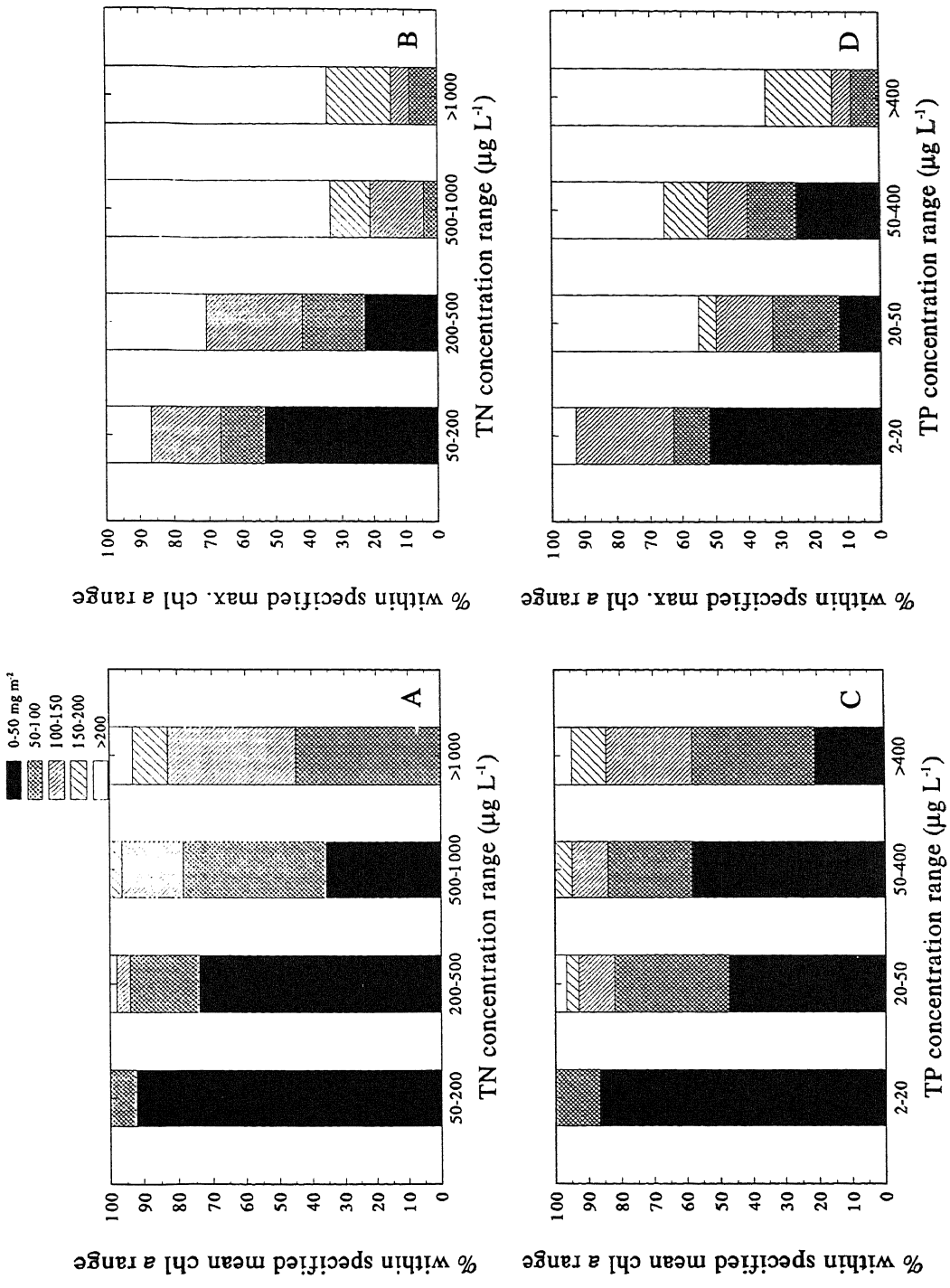

Fig. 3. Percentage of mean or maximum chlorophyll *a* values falling within predefined ranges for each of four separate TN or TP ranges. Mean (A) and maximum (B) as functions of TN, and mean (C) and maximum (D) as functions of TP.

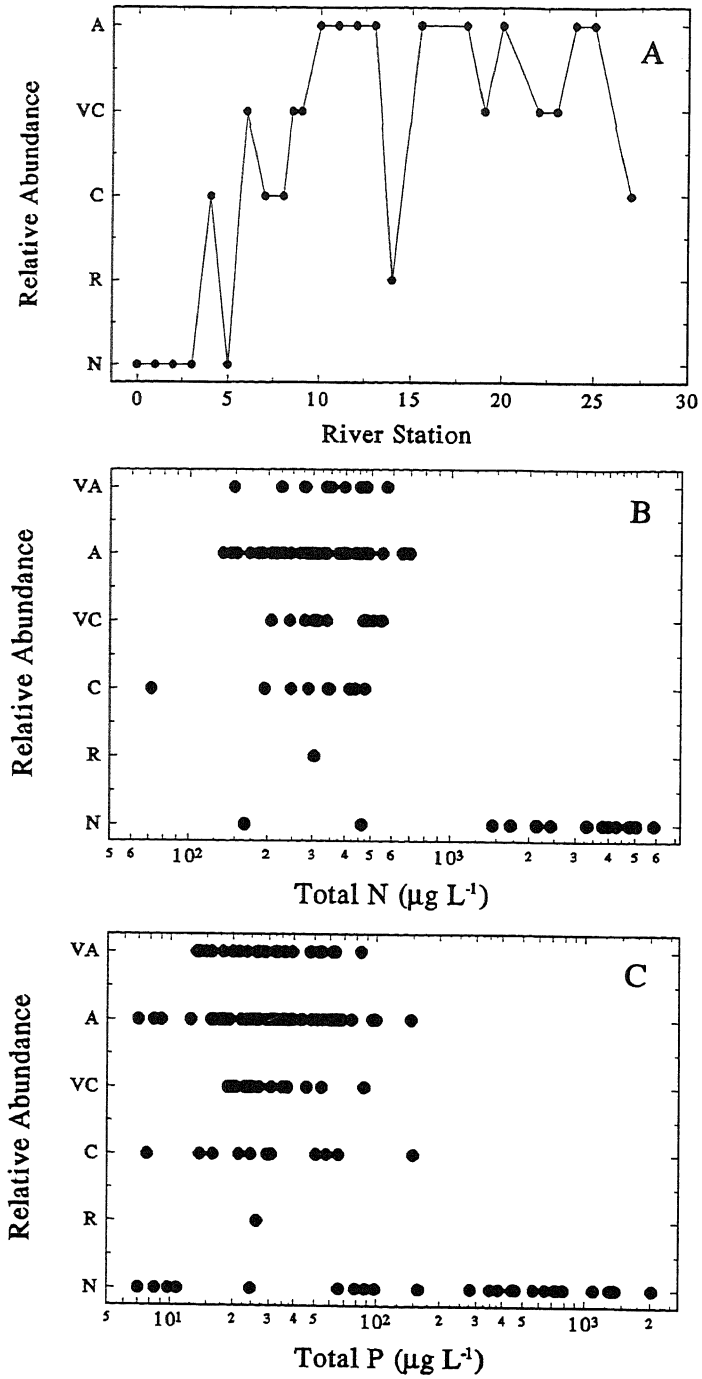

Fig. 4. Relative *Cladophora* abundance as functions of stream reach (A); TN (B); and TP (C). Abundance scores: VA = very abundant, A = abundant, VC = very common, C = common, R = rare, and N = not present.

ratio is used to set TP, and TN is varied to reach target chlorophyll *a* concentrations according to equation (17) (Table 2). A similar approach using equation (14) (Table 2), setting TN by the Redfield ratio, and instead varying TP to reach target chlorophyll *a* concentrations suggests that 35 $\mu g\, l^{-1}$ TP correspond with maximum values around 100 mg m^{-2} chlorophyll *a* (Table 5). In contrast, the target TP value (20.5 $\mu g\, l^{-1}$) derived from actual instream nutrient measurements at the reference stations was somewhat lower. We, thus, conclude that a provisional target of 30 $\mu g\, l^{-1}$ TP may control peak algal biomass below 150 mg m^{-2} chlorophyll *a*.

Management of Cladaphora *growth.*

Cladophora is a common nuisance alga in enriched streams, rivers and lakes (Whitton, 1970; Dodds and Gudder, 1992), and control of its growth is very

difficult (Dodds, 1991). In the Clark Fork River, *Cladophora* was occasionally abundant at some stations (Fig. 4A). *Cladophora* appears to be N-limited in the Clark Fork (Lohman and Priscu, 1992), and research suggests that this is also true in many other systems (Dodds and Gudder, 1992). However, we were unable to develop models that could be used to predict management scenarios that would lower algal biomass and simultaneously reduce *Cladophora* dominance. Control of *Cladophora* in the

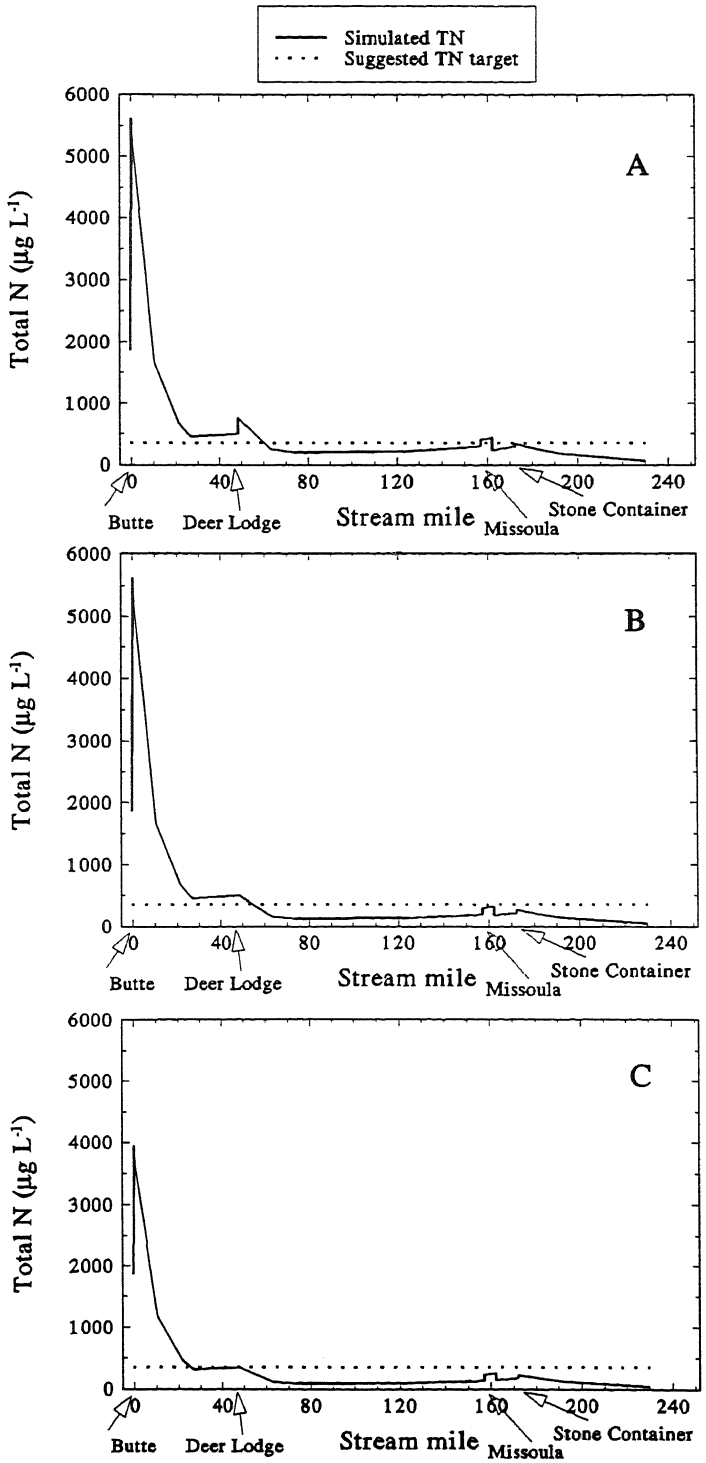

Fig. 5. TN loading scenarios used to calculate instream TN concentrations during normal summer low flows for no loading controls (A); complete removal of sewage from the city of Deer Lodge (B); and city of Butte limited to 7 mg l^{-1} and complete removal at Deer Lodge (C).

Clark Fork will be as difficult as it has been in other systems.

Instream summer loading policies

We conclude that significant reductions are needed in certain segments of the Clark Fork to reach instream targets of 350 $\mu g\,l^{-1}$ for TN and 30 $\mu g\,l^{-1}$ for TP. The spreadsheet nutrient model developed here can be used to define different combinations of controls that would maintain these targets throughout the basin under critical flow conditions. The spreadsheet model can also be used to evaluate the relative importance of point and nonpoint sources of nutrients, and to aid in the design of control strategies. For example, one alternative control strategy may be to augment flow during periods of low flow to dilute inflowing nutrients. The spreadsheet model could easily be modified to simulate such a control strategy. However, the model currently includes only nutrient loading and concentrations and does not predict instream biological responses such as periphyton growth levels.

SUMMARY

We conclude that strategies to control external nutrient loading and manage stream eutrophication should not be based on instream DIN and SRP levels, because instream dissolved inorganic nutrient concentrations are poorly related to benthic algal biomass. Management of external nutrient loading instead should be based upon instream TN and TP concentrations. Our three approaches (regression, probabilistic and reference reaches) converge on similar suggested targets for nutrient control. A general probabilistic analysis using a global data set suggests that, if TN values are maintained below a target concentration of 350 $\mu g\,l^{-1}$, chlorophyll *a* should not exceed 100 mg m^{-2} in most cases (Fig. 3). The validity of this target TN concentration is supported independently by water chemistry data from six reference reaches of the Clark Fork that generally do not exhibit subjectively determined excessive levels of chlorophyll *a*; the mean TN for these reference reaches is 317 $\mu g\,l^{-1}$ TN.

If the instream TP concentrations are maintained below 13% of the target TN value (target = 45.5 $\mu g\,l^{-1}$ TP), a high probability exists that the effectiveness of TN control measures will not be impaired by the nuisance growth of nitrogen-fixing algal species. By comparison, the mean TP concentration from the six reference reaches on the Clark Fork was 20.5 $\mu g\,l^{-1}$. The regression calculation using the global data set suggests that keeping TP below 35 $\mu g\,l^{-1}$ and TN below 252 $\mu g\,l^{-1}$ will control algal biomass below 100 mg m^{-2} chlorophyll *a* (Table 4). We suggest 30 $\mu g\,l^{-1}$ TP as a target level that will control algal biomass.

We stress that our recommended TN and TP targets may not apply to all systems. The final target values of TN and TP that are chosen for water quality protection in streams in other geographical locations ultimately must be based upon the objective criteria that local managers consider to be most relevant. In addition, the proposed levels of TN and TP may be unattainable in some ecoregions depending upon local geology, sediment contamination and other factors.

An additional benefit of using TN and TP values to control nutrients in the river as opposed to DIN and SRP levels involves consideration of nonpoint sources. Most information linking land use practices with nutrient input to aquatic systems uses TN and TP, not DIN or SRP. Building a regulatory framework for stream nutrients around TN and TP input will assist in future efforts to control nonpoint sources.

Acknowledgements—This reserch was supported by the Montana Department of Health and Environmental Sciences through a grant from Stone Container Corporation. Discussions with V. Watson, G. Ingman and J. Priscu improved the focus of this work, P. McCormick and three anonymous reviewers provided valuable critiques of the manuscript. K. Lohman helped conceptualize and construct the global data set, J. Jones provided data. D. Gudder provided technical assistance. This is contribution no. 96-439-J from the Kansas Agricultural Experiment Station.

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PII: S0043-1354(97)00370-9

Wat. Res. Vol. 32, No. 5, pp. 1455–1462, 1998
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Printed in Great Britain
0043-1354/98 $19.00 + 0.00

SUGGESTED CLASSIFICATION OF STREAM TROPHIC STATE: DISTRIBUTIONS OF TEMPERATE STREAM TYPES BY CHLOROPHYLL, TOTAL NITROGEN, AND PHOSPHORUS

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(First received January 1997; accepted in revised form August 1997)

Abstract—Aquatic scientists and managers have no conventional mechanism with which to characterize and compare nutrients and algal biomass in streams within a broader context analogous to trophic state categorization in lakes by chlorophyll (chl) and nutrients. We analyzed published data for a large number of distinct, temperate, stream sites for mean benthic chl (n = 286), maximum benthic chl (n = 176), sestonic chl (n = 292), total nitrogen (n = 1070), and total phosphorus (n = 1366) as a first effort to establish criteria for trophic boundaries. Two classification systems are proposed. In the first system, the boundary between oligotrophic and mesotrophic categories is defined by the lower third of the cumulative distribution of the values. The mesotrophic–eutrophic boundary is defined by the upper third of the distribution. In the second system, individual streams are placed more precisely in a broad geographic context by assessing the proportion of streams that have greater or lesser nutrient and chl values. The proposed relationships for streams were compared to trophic criteria published for lakes. The proposed trophic boundaries for streams generally include a broader range of values in the mesotrophic range than conventional criteria for lakes. The ratio of maximum to mean benthic chl for streams was significantly higher than that found for planktonic chl in lakes, reflecting the greater variance in streams. This high variance in streams suggests that the proposed stream trophic criteria should be viewed only as a general first approach to categorizing stream ecosystems. © 1998 Elsevier Science Ltd. All rights reserved

Key words—chlorophyll, eutrophic, mesotrophic, nitrogen, nutrients, oligotrophic, periphyton, phosphorus, rivers, streams

INTRODUCTION

Classification of ecosystems by an index of trophic state is common in the aquatic sciences. Streams occasionally are classified as eutrophic or oligotrophic (e.g. Hornberger *et al.*, 1977; Kelly and Whitton, 1995), but no conventional criteria exist for these terms when applied to lotic systems. Stream enrichment often leads to increases in algal biomass (e.g. Dodds *et al.*, 1997; Lohman *et al.*, 1992, Van Niewenhuyse and Jones, 1996; Welch *et al.*, 1992), and thus, a trophic classification using both nutrients and algal biomass seems useful as it has been for lakes. Autotrophic biomass is important in many streams as a food source for organisms (Lamberti, 1996). Being able to place a stream in a continuum of nutrient concentrations and producer biomass from a variety of temperate streams should aid stream researchers and managers in characterizing ecosystems and facilitate comparative research and management.

Conventional systems exist for classifying lakes into trophic categories using nutrients and algal biomass (e.g., OECD, 1982; Porcella *et al.*, 1980; Ryding and Rast, 1989). Trophic classifications for lakes have a rich history and stem from differences in lake ecosystem function and phytoplankton communities over the range of lake types (Hutchinson, 1967). General functional characteristics exist among lakes within each of the major trophic state categories. Simply put, oligotrophic lakes have low nutrients, low algal biomass, high clarity, and deep photic zones, and may support a cold-water fishery. Eutrophic lakes can have frequent cyanobacterial blooms, high total nutrients, and wide swings in dissolved oxygen (DO) concentrations (potential anoxia) and pH. Mesotrophic lakes have intermediate characteristics. The boundaries placed between these categories by aquatic scientists are similar but not universal (e.g., Forsberg and Ryding, 1980; OECD, 1982; Porcella *et al.*, 1980), in part because

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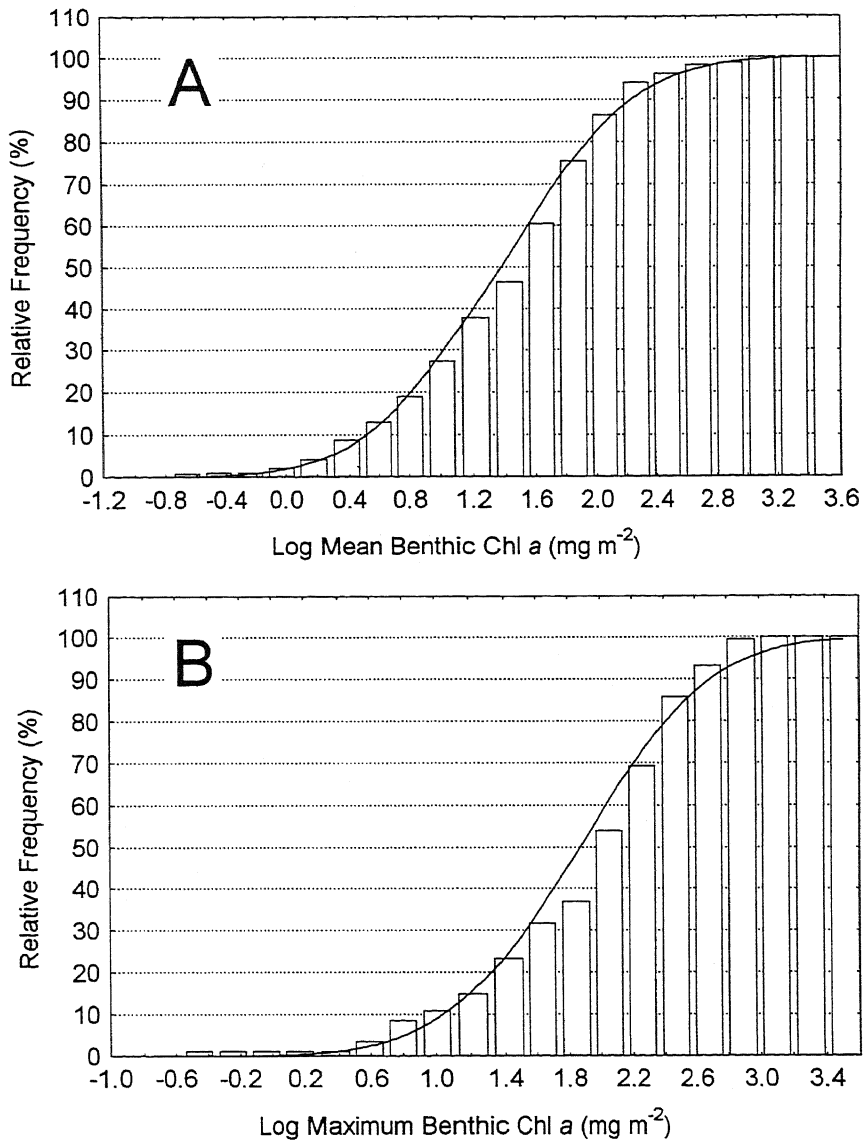

Fig. 1. Cumulative frequency diagram of seasonal mean (A, $n = 286$) and maximum (B, $n = 176$) benthic chl *a*. The line indicates the log–normal distribution. The distribution of mean benthic chl was not significantly different from the log–normal distribution ($p < 0.15$), but that of maximum chl was ($p < 0.005$, Kolmogorov–Smirnov).

the perceptions of researchers have provided the basis for most schemes and true geographic differences exist among lakes that justify modifying criteria for specific regions (Jones and Knowlton, 1993). Alternatively, Carlson (1977) constructed a Trophic State Index (TSI) on a scale of 0 to 100 based on interrelations among chlorophyll (chl), total phosphorus (TP), and Secchi depth. Each 10 units within this system represent a doubling of TP, a one-third increase in chl, and a half decrease in Sechhi depth, thus providing a basis for the scale and a means to identify nutrient-limited conditions (Carlson, 1991).

Stream ecosystems have been described on the basis of carbon sources (Dodds, 1997) and position in the watershed (Vannote *et al.*, 1980), but boundaries separating stream types based on producer biomass and nutrients across watersheds are lacking. Stream nutrients usually have been observed to increase as a function of intensity of land use within the basin (Omernik, 1977; Smart *et al.*, 1985) and with human population density in the watershed (Peierls *et al.*, 1991). Historically, specific zones within streams have been identified in relation to point source inputs of organic pollution and its impact on dissolved O_2, nutrient content and stream

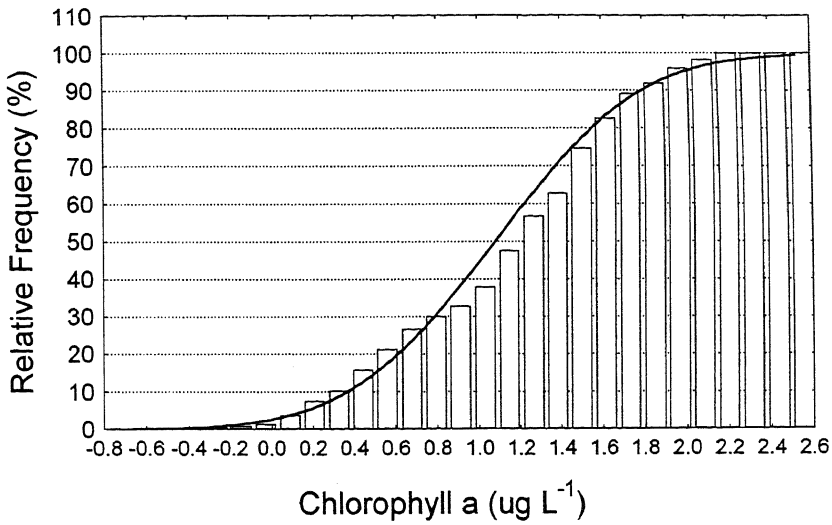

Fig. 2. Cumulative frequency diagram of sestonic chl for temperate streams (n = 292). The line indicates the log–normal distribution, which was not significantly different from this distribution (p < 0.07, Kolmogorov–Smirnov).

biota (Bartsch and Ingram, 1959). These identifications dealt primarily with longitudinal zones of degradation and recovery but did not allow characterization of the fertility and productivity of individual streams relative to others. A trophic index based on diatoms was proposed for use in the United Kingdom (Kelly and Whitton, 1995), but this index is specific to a limited region, requires sophisticated taxonomic skills that are not generally available, and excludes filamentous green algae which are most commonly associated with nuisance conditions. Photosynthesis/respiration ratios also have been suggested as a method to characterize trophic state in streams (Hornberger *et al.*, 1977), but this characterization requires intensive monitoring of diel variations in dissolved oxygen concentrations and has received little use.

We propose a simple approach for initially characterizing trophic state of streams using the frequency distributions of nutrients and chl to define the three trophic categories. This approach is appropriate given the absence of natural trophic boundaries based on stream ecosystem characteristics. Using the distribution of values across a large number of temperate streams, the lowest third represents the oligotrophic category, the middle third the mesotrophic category, and the top third is proposed to constitute the eutrophic category. These frequency diagrams also allow individual streams or streams within a specific geographic area to be placed in a broader context of stream types (Omernik, 1977). For example, it may be more persuasive to argue that a specific stream is eutrophic if 70% of the streams in this data set have less chl *a*. We used previously published data for our analyses.

SOURCES OF DATA

Data on mean and maximum benthic stream chl, and some of the mean water-column total nitrogen (TN), and mean TP were taken from a previously published data set for temperate streams (Dodds *et al.*, 1997). These data were collected for more than 200 streams or sites in North America and New Zealand. The majority of the systems were from temperate habitats. Seasonal means from a single year (for 2 to 3-month periods) generally were used for TN, TP and mean benthic chl. These were not always restricted to summer, because some streams (e.g., those with deciduous tree cover) may have more biomass or production at other times of the year. Seasonal maxima also were taken from this data set. The data set included sites from low order streams up to large rivers, and from nutrient enriched to those with pristine vegetative cover.

Data on planktonic (sestonic) chl and some of the water column TP data were taken from a different data set (Van Niewenhuyse and Jones, 1996). These data came from 115 streams for multiple sites and years (almost 300 separate data points) from a variety of watershed types and areas in the temperate zone (primarily North America, with some European streams represented). All TP and chl in this data set were calculated as growing season means.

Additional data on water column TN and TP were taken from a compilation of stream data from the United States Environmental Protection Agency National Eutrophication Survey (Omernik, 1977). These values are represented by means from approximately monthly sampling of 928 streams over a period of a year. The streams sampled in this study were generally small, with watershed use distributed from pristine forests to fully agricultural.

RESULTS AND DISCUSSION

Wide distributions of mean and maximum benthic chl, and water column chl, TN, and TP were observed within the data sets (Figs 1–3). Some of the distributions were not significantly different from a log–normal distribution, but those for maximum chl and TN were. For this reason, the actual data distributions were used to set trophic classification boundaries. The approximate values marking the divisions between the lower and middle thirds, and between the middle and upper thirds of each distribution were used as boundaries between oligotrophic and mesotrophic and between mesotrophic and eutrophic, respectively (Table 1). Sharp, natural boundaries in trophic state and functional relations among the variables have yet to be identified for streams, so these boundaries should be viewed as provisional. The boundaries will likely change somewhat as the database expands, sites from tropical and polar regions are added, macrophytes are considered, and functional relationships among nutrients and algal biomass are developed further in lotic systems. Nevertheless, the suggested boundaries provide general guides to divide the trophic continuum.

Our trophic categories match another trophic characterization proposed on the basis of watershed characteristics for 16 New Zealand streams (Biggs, 1996). The 16 New Zealand streams were divided into unenriched, moderately enriched, and enriched categories based upon the amount of agriculture and geology in each individual watershed. Then the cumulative frequency distributions were plotted for benthic chl. About 90% of the values in the unen-

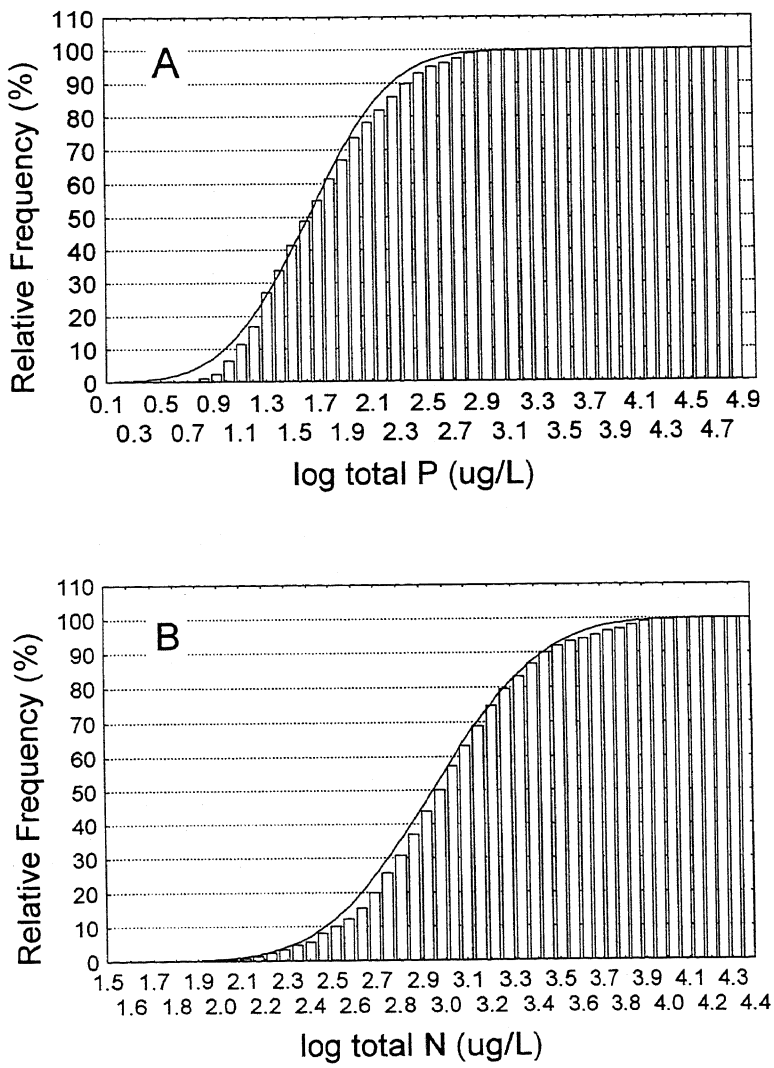

Fig. 3. Cumulative frequency diagram of TP (A, $n = 1366$) and TN (B, $n = 1070$) for temperate streams. The line indicates the log–normal distribution. The TN distribution was significantly different than the log–normal distribution ($p < 0.02$) but the TP distribution was not ($p < 0.07$, Kolmogorov–Smirnov).

Table 1. Suggested boundaries for trophic classification of streams from cumulative frequency distributions in Figs 1–3. The boundary between oligotrophic and mesotrophic systems represents the lowest third of the distribution, and the boundary between mesotrophic and eutrophic marks the top third of the distribution

Variable (units)	Oligotrophic–mesotrophic boundary	Mesotrophic–eutrophic boundary	N
Mean benthic chlorophyll (mg m^{-2})[a]	20	70	286
Maximum benthic chlorophyll (mg m^{-2})[a]	60	200	176
Sestonic chlorophyll (μg l^{-1})[b]	10	30	292
TN (μg l^{-1})[a,c]	700	1500	1070
TP (μg l^{-1})[a,b,c]	25	75	1366

[a]Data from Dodds *et al.* (1997).
[b]Data from Van Niewenhuyse and Jones (1996).
[c]Data from Omernik (1977).

riched and moderately enriched watersheds fell below 20 and 100 mg m^{-2} chl respectively. These categories compare favorably to our proposed upper boundaries of 20 and 70 mg m^{-2} chl for oligotrophic and mesotrophic systems respectively.

The cumulative frequency diagrams (Figs 1–3) depict the broad variation inherent in stream chl and nutrient chemistry when many streams are considered. The diagrams also provide a relative scale to locate individual streams of interest along a larger continuum of nutrients and chl. For example, a stream may be considered to be moderately enriched if TN is 1000 μg/l because about 50% of the streams in our data set have less N (Fig. 3B). Comparisons of this type may also help identify functional processes in streams. If a stream has TP levels in the top 10% of our distribution and either benthic or sestonic chl in the bottom 10%, some factor other than nutrients likely is controlling accrual of biomass. Specific functions to determine the expected relationships between TN:TP and benthic chl (Dodds *et al.*, 1997) and TP and sestonic chl (Van Niewenhuyse and Jones, 1996) are available, if more detailed analysis is required.

Previous studies have suggested that benthic chl above 50–200 mg m^{-2} may represent a cut point for nuisance conditions (Table 2). These values of chl have been determined subjectively and there may be a regional basis for the four-fold range. However, most of these objectionable levels fall above the proposed lower boundary for eutrophic streams (Table 1). About 85% of the streams analyzed in our database had mean benthic chl below 100 mg m^{-2}, and 50% had maximum values below this level. Therefore, a general suggestion that a mean of 150 mg m^{-2} represents nuisance levels agrees with values suggested by other authors (Table 2).

A positive relationship occurred between seasonal mean and maximum stream benthic chl (Fig. 4). The ratio of maximum to mean benthic chl for our data set was 4.52 (n = 178, std. dev. = 7.61, lower 95% confidence level 3.39). The ratio of maximum to mean chl in lakes has been reported to be 1.7–1.8 by Jones *et al.* (1979), 2.5 by Chapra and Tarapchak (1976), and 2.6 by OECD (1982). This significantly higher ratio for streams and rivers than for lakes indicates the higher potential variability in chlorophyll in streams. Probably this higher ratio for streams can be explained by the influence of scouring floods (Allan, 1995; Peterson, 1996). Such hydrologic fluctuations between flood and base flow have been demonstrated to greatly influence stream chemical constituents and autotrophic biomass (Lohman *et al.*, 1992; Perkins and Jones, 1994), and these temporal changes complicate stream characterization.

The papers that analyze nutrient chlorophyll relationships in streams more thoroughly (Dodds *et al.*, 1997; Van Niewenhuyse and Jones, 1996) have demonstrated relationships between TN or TP and chl for streams that are weaker than they are in lakes. This is probably because high turbidity (non-algal TN and TP) may be more common in streams than in lakes. Also, our cumulative frequency distributions for TN and TP contained a large number of points for which no chl data were available. Therefore, it may not be advisable to attempt to tightly link distributions of TN or TP concentrations to the benthic chl distributions presented here.

We evaluated our trophic criteria for streams against those published for lakes because trophic state characterization for lakes is more established through actual use. We compared the areal phytoplankton chl content in lakes to benthic chl in

Table 2. Suggested criteria from various studies for maximum benthic biomass (as chl) levels to avoid problems for recreational and aesthetic use in streams

Suggested value or range (mg chl m^{-2})	Comment	Reference
150–200	based on perceived impairment	Welch *et al.*, 1989
100–150	based on 19 enrichment cases and surveys	Welch *et al.*, 1988, Horner *et al.*, 1983
150	guidelines for Clark Fork River Montana, U.S.A.	Tristate Implementation Council, 1996
50–100	British Columbia Environment guideline	Nordin, 1985

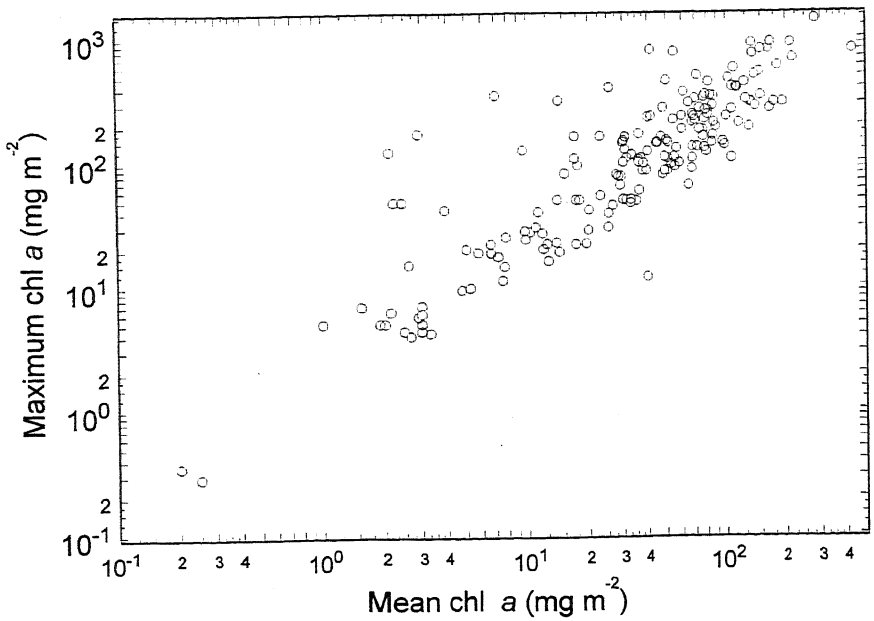

Fig. 4. Relationship between seasonal mean and maximum benthic chl for 176 temperate streams.

streams and volumetric planktonic chl concentrations in lakes and streams. A popular fixed boundary system for trophic classification (OECD, 1982), using chl, TN, TP, and Secchi depth, was employed for this comparison (Table 3). Assuming that chl is distributed evenly down to two times the Secchi depth (roughly the compensation point for photosynthesis), we also calculated areal chlorophyll values for lakes. These data suggest that the OECD (1982) mesotrophic range for areal chl in lakes is slightly narrower than the range for streams. This could be because chl (on an areal basis) is distributed differently in lakes than in streams or the methods used to define trophic state in lakes led to drawing boundaries at different levels. The reader should keep in mind how the OECD (1982) system was constructed. Data were provided for a large number of lakes by a number of investigators. Each investigator signified if their lakes were oligotrophic, mesotrophic or eutrophic. The boundaries for each trophic classification were then derived for all lakes proposed for each of the three categories.

We used our cumulative frequency method to define boundaries for lakes in a similar fashion as that used for streams. When Smith (1982) data for approximately 309 lakes and Jones and Knowlton (1993) data for 94 reservoirs were used, the trophic boundaries for chl *a*, TN, and TP of lakes were similar to those proposed for streams in this paper. In fact, TP boundaries for lakes and streams from cumulative frequency distributions were virtually identical. Thus, our trophic classifications for streams may vary from the OECD values for lakes largely because of methodology. That is, a statistical approach to classification of lake trophic state

Table 3. Trophic boundaries for lakes from OECD (1982) and lake data reanalyzed using methods in this paper

Variable	Oligotrophic–mesotrophic boundary	Mesotrophic–eutrophic boundary
OECD-based values		
Volumetric mean planktonic chl ($\mu g\ l^{-1}$)[a]	2.5	8
Volumetric maximum chl ($\mu g\ l^{-1}$)[a]	8	25
TP ($\mu g\ l^{-1}$)[a]	10	35
Mean Secchi (m)[a]	6	3
A real mean planktonic chl (mg m^{-2})[b]	30	48
Based on cumulative frequency distributions from lakes		
Planktonic mean chl ($\mu g\ l^{-1}$)[c]	8	25
TP ($\mu g\ l^{-1}$)[c]	25	71
TN ($\mu g\ l^{-1}$)[c]	500	1260

[a]OECD, 1982 values.
[b]Calculated from OECD (1982) assuming chlorophyll is distributed evenly to 2 times the Secchi depth.
[c]Smith (1982) data, n = 309, and Jones and Knowlton (1993) data, n = 94 for Missouri Reservoirs, with the cumulative frequency method proposed here applied to determine trophic state.

yielded categories for lakes more similar to those proposed here for streams, whereas the more subjective methods used by others for lakes yielded trophic classifications less similar to ours.

The distributions presented here can be refined as more data become available, particularly for benthic and suspended chl. Tropical streams and rivers may vary from their temperate counterparts analyzed here. Eutrophic streams may be more abundant in particular geographic regions. For example, areas with considerable agricultural activity typically have streams with high nutrient content (Omernik, 1977). Detailed analysis in specific regions may lead to categorization of streams that has more geographic relevance (e.g. Biggs, 1995; Biggs, 1996). It is also certain that the distributions we report are influenced by anthropogenic nutrient sources and do not represent pristine, background conditions. The past can not be re-visited and there are virtually no large, completely pristine temperate rivers left to construct a data base from, so the next best approach is to provide a baseline using data collected over the past few decades. We suggest that frequency distributions can be used to assist stream scientists and managers in determining the trophic states of their individual streams relative to others.

Acknowledgements—We thank D. Gudder for technical assistance. Discussions with V. Smith, K. Lohman and V. Watson helped improve the focus of the research. V. Smith and E. Van Niewenhuyse graciously provided their published data in electronic form. This research was financially supported in part by National Science Foundation IBN 9632851. This is contribution no. 97-257-J from the Kansas Agricultural Experiment Station.

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Relationships between Water Quality, Habitat Quality, and Macroinvertebrate Assemblages in Illinois Streams

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The influence of specific stressors, such as nutrient enrichment and physical habitat degradation, on biotic integrity requires further attention in Midwestern streams. We sampled 53 streams throughout Illinois and examined relationships between macroinvertebrate community structure and numerous physicochemical parameters. Streams were clustered into four major groups based on taxa dissimilarity. Habitat quality and dissolved nutrients were responsible for separating the major groups in a nonmetric multidimensional scaling ordination. Furthermore, the alignment of environmental factors in the ordination suggested there was a habitat quality–nutrient concentration gradient such that streams with high-quality habitats usually had low concentrations of nutrients. Discrimination by community measures further validated the major stream groups and indicated that forested streams had generally higher biological integrity than agricultural streams, which in turn had greater integrity than urban streams. Our results demonstrate that physical habitat degradation and nutrient pollution are important and often confounded determinants of biotic integrity in Illinois streams. In addition, we suggest that management of Midwestern streams could benefit from further implementation of multivariate data exploration and stream classification techniques.

The dramatic impacts that humans have had on lotic ecosystems are exemplified in the midwestern USA, where agricultural and urban activities dominate the landscape. Although anthropogenic impacts in this region have long been known to negatively affect aquatic biota (DeWalt et al., 2005), the impacts of specific stressors, and combinations thereof, that most influence biological integrity at large scales require more attention. For example, physical habitat degradation may be most responsible for impairment of biological integrity in some streams (Master et al., 2000), yet some researchers have found that nutrients have the largest effect on stream organisms (e.g., Wang et al., 2007).

Biological monitoring techniques represent a departure from more standard chemical monitoring of systems, which may not account for stressor variability or nonpoint sources of impairment (Scrimgeour and Wicklum, 1996; Gerritsen et al., 1998). Biomonitoring approaches directly measure biological components of systems, such as community and functional structure. This is advantageous because biota integrate conditions over time and space and are directly linked to ecosystem function and integrity (Fausch et al., 1990; Loeb and Spacie, 1994; Resh et al., 1996).

Relationships between physicochemical characteristics and macroinvertebrates in wadeable streams are generally not well documented (Wang et al., 2007), and this is especially true in Illinois. Degraded physical habitat quality has been linked to decreased diversity and altered stream function (e.g., Newbold et al., 1983; Webster and Ehrman, 1996). Reduced substratum size and stability and decreased organic matter retention have historically been identified as major detrimental characteristics of habitat degradation (e.g., Egglishaw, 1964; Minshall, 1984). Elevated nutrient concentrations have been linked to poor biotic integrity in streams (Miltner and Rankin, 1998), and the mechanism is generally considered to be stimulation of excess primary production, which can degrade habitat, alter food resources, and deplete dissolved oxygen (Wang et al., 2007). However, the interplay of these factors and their ultimate influence on biotic integrity are in need of further study, and understanding these relationships is central to managing and assessing streams.

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Published in J. Environ. Qual. 36:1653–1660 (2007).
doi:10.2134/jeq2006.0521
Received 30 Nov. 2006.
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Abbreviations: DRP, dissolved reactive phosphorus; NMDS, nonmetric multidimensional scaling; MBI, macroinvertebrate biotic index.

Table 1. Sample stream locations, Illinois Environmental Protection Agency (IEPA) codes, a priori predominant land use codes, and dates sampled for macroinvertebrates in 2005.

Stream	Location	IEPA code	a priori Code†	Date sampled
Apple River	near Hanover	MN-03	ag6	8 Mar.
Bankston Creek	near Harrisburg	ATGC-01	ag13	5 Mar.
Bay Creek	at Nebo	KCA-01	ag16	7 Mar.
Bear Creek	near Marcelline	KI-02	ag33	8 Mar.
Big Bureau Creek	near Princeton	DQ-03	ag15	8 Mar.
Big Creek	near Balcom	IXJ-02	for7	5 Mar.
Big Creek	near Bryant	DJB-18	ag24	7 Mar.
Big Ditch	near Dewey	EZU-01a	ag7	14 Mar.
Big Muddy River	near Mt. Vernon	N-08	ag14	6 Mar.
Black Slough	near Philo	BETA	ag10	15 Mar.
Bonpas Creek	near Browns	BC-02	ag1	15 Mar.
Cache River	near Forman	AD-02	for2	5 Mar.
Cahokia Canal	near Collinsville	JN-02	urb2	7 Mar.
Canteen Creek	near Collinsville	JNA-01	urb3	7 Mar.
Clear Creek	near Elbridge	BM-02	ag3	15 Mar.
Crab Orchard Creek	at Marion	ND-04	ag17	5 Mar.
Crooked Creek	near Newton	BEG-01	for1	15 Mar.
Dupage River	at Shorewood	GB-11	urb12	10 Mar.
E. Fk. Kaskaskia River	near Sandoval	OK-01	ag30	6 Mar.
E. Fk. Lamoine River	near Colchester	DGL-03	for8	8 Mar.
Elkhorn Creek	near Penrose	PH-16	ag23	9 Mar.
Elm River Drainage Ditch	near Tom's Prairie	CD-01	ag2	15 Mar.
Farm Creek	at E. Peoria	DZZP-03	urb4	8 Mar.
Hickory Creek	near Bluff City	ON-01	ag29	6 Mar.
Hurricane Creek	near Mulberry Grove	OL-02	ag26	5 Mar.
Hutchins Creek	near Bald Knob	ICE-01	for4	6 Mar.
Indian Creek	near Wyoming	DJL-01	ag5	8 Mar.
Kankakee River	at Momence	F-02	urb6	13 Mar.
Kickapoo Creek	at Waynesville	EIE-04	ag12	14 Mar.
Kishwaukee River	at Marengo	PQ-10	for6	9 Mar.
Little Kickapoo Creek	near Bloomington	–‡	ag11	14 Mar.
Lusk Creek	near Eddyville	AK-02	for3	5 Mar.
McDonald Creek	at Mt. Prospect	GR-01	urb7	10 Mar.
McKee Creek	near Chambersburg	DE-01	ag25	7 Mar.
Mid. Fk. Vermillion River	near Penfield	BPK-07	for9	13 Mar.
Midlothian Creek	near Midlothian	–	urb1	10 Mar.
N. Fk. Saline River	near Texas City	ATF-04	ag27	5 Mar.
N. Fk. Vermillion River	near Bismark	BPG-09	ag21	13 Mar.
Plum Creek	near Crete	HBE-01	for5	13 Mar.
Rayse Creek	near Waltonville	NK-01	ag31	6 Mar.
S. Br. Kishwaukee River	near Fairdale	PQC-06	ag8	9 Mar.
S. Fk. Saline River	near Carrier Mills	ATH-05	ag27	5 Mar.
S. Fk. Vermillion River	near Fairbury	DS-06	ag22	13 Mar.
Saline Branch Salt Fork	near Mayview	BPJC-06	ag18	14 Mar.
Salt Creek	at Western Springs	GL-09	urb11	10 Mar.
Shoal Creek	near Breese	OI-08	ag32	6 Mar.
Sisinawa River	near Galena	–	ag19	9 Mar.
Skokie River	at Lake Forest	–	urb8	10 Mar.
Spring Creek	at Algonquin	–	urb9	10 Mar.
Spring Creek	at Springfield	EL-01	urb5	7 Mar.
Spring Creek	near Gilman	–	ag9	13 Mar.
Thorn Creek	at Thornton	HBD-04	urb10	10 Mar.
Whitley Creek	in Moultrie County	–	ag4	15 Mar.

† ag, agricultural; for, forested; urb, urban.

‡ A dash indicates that a code does not exist for a stream.

The objectives of this study were to identify stressors that most strongly influence the biotic integrity of stream macroinvertebrate communities and to determine what stressors most influence the macroinvertebrate communities of specific stream types. We used multivariate techniques to identify patterns of macroinvertebrate composition and factors influencing them. We used metrics of community structure, function, and pollution tolerance to assess biotic integrity. We predicted that the degree of nutrient enrichment and the amounts and types of habitat degradation would vary across Illinois in relation to land use and that this would be reflected in macroinvertebrate community assemblages. We further hypothesized that, across all sites, physical habitat quality would be the most important factor controlling biotic integrity among the streams.

Materials and Methods

Sampling and Processing

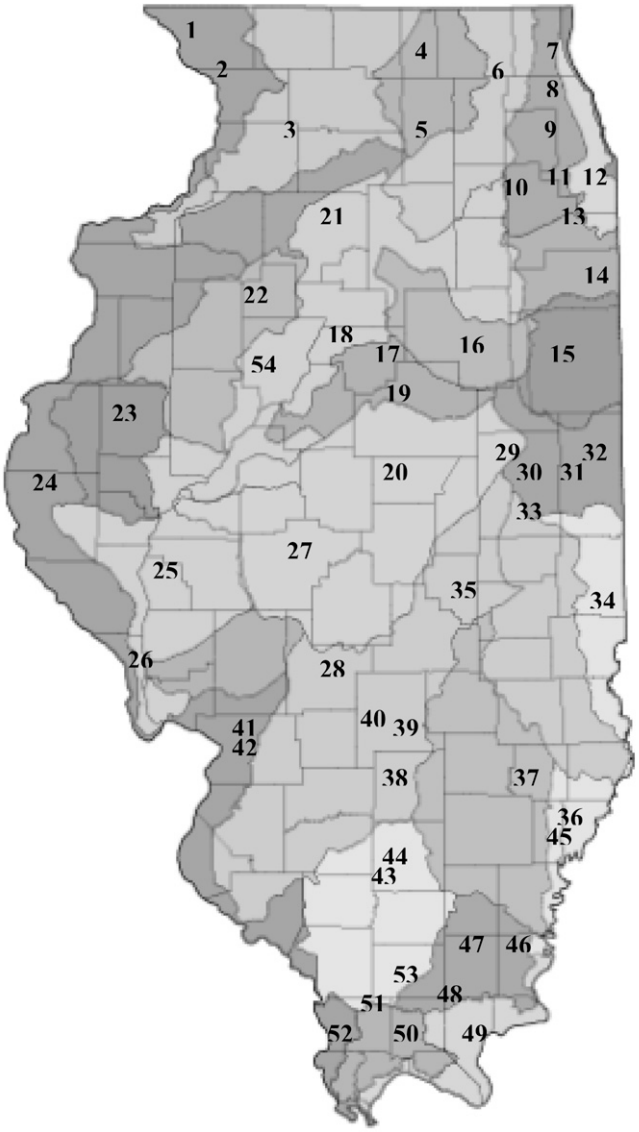
This study was designed to gather representative data from gauged, wadeable streams from each major physiogeographic region of Illinois (Table 1; Fig. 1). A total of 53 streams were analyzed from May 2004 to March 2005. At each site, water temperature, pH, specific conductance, and dissolved oxygen concentration were measured with calibrated, portable meters. Water samples for analysis of sestonic chlorophyll-*a* and dissolved nutrients were collected from the center of the channel. Water samples were kept in the dark and on ice until they could be processed and preserved according to standard methods (American Public Health Association, 2005). Samples were analyzed for dissolved organic carbon, dissolved reactive phosphorus (DRP), nitrate-nitrogen (NO_3–N), ammonium-nitrogen (NH_4–N), and chloride after filtration through a precombusted 0.45-μm membrane filter. Total P and total N concentrations were determined on unfiltered water samples after appropriate digestion (American Public Health Association, 2005). Chloride and NO_3–N were determined with ion chromatography (DX-120; Dionex, Sunnyvale, CA). Dissolved reactive P, total P, NH_4–N, and total N were determined colorimetrically using a QuikChem 8000 (Lachat, Loveland, CO). Dissolved organic carbon was determined on a Dohrmann total organic carbon analyzer. Additionally, we compiled historical water quality data that were collected between 1972 and 2005 by the Illinois Environmental Protection Agency, Illinois State Water Survey, United States Fish and Wildlife Service, or University of Illinois Department of Natural Resources and Environmental Sciences. The number of samples from each site ranged from 22 to 321, with 150 to 200 samples being most common. A linear regression between pre- and post-1990 N and P showed that concentrations were consistent through both time periods ($r^2 > 0.9$). Therefore, we averaged all available nutrient data for each site to provide a robust assessment of nutrient availability.

For physical habitat assessments, we first quantified the composition of the substrata of each study reach by taking 100 random grabs (Wolman, 1954) and classifying substratum size using a modified Wentworth scale (Cummins, 1962). Second, we conducted semi-quantitative habitat sur-

veys by scoring physical features (riffle-run-pool morphology, sinuosity, substrata, and riparian condition) of the reach to obtain overall habitat scores based on procedures of Barbour et al. (1999). Each of nine habitat features was given a score of 0 to 20, with a total score of 180 being the best possible score.

We sampled benthic macroinvertebrates from 8 to 18 Mar. 2005 following USEPA multihabitat Rapid Bioassessment Protocols for wadeable streams (Barbour et al., 1999). We took 20 jabs with a 0.3 by 0.5 m, 500-µm mesh dip net from each 100-m study reach. A jab consisted of disturbing the substrata for 0.5 m upstream of the net to a depth of ~10 cm in riffle and run habitats. In pools, the net was bumped through ~10 cm of substrata for 0.5 m. The 20 jabs were partitioned among pool, riffle, and run habitats within each study reach (e.g., a reach with 50% riffle habitat had 10 jabs taken from riffles). All jabs were combined into one composite sample, which was immediately rinsed through a 500-µm sieve and preserved in a 10% formalin solution.

A two-stage subsampling procedure was used to remove invertebrates from samples in the laboratory (Vinson and Hawkins, 1996). First, relative abundances were estimated by randomly removing 300 organisms, when available, using a numbered, gridded pan and random number generator. Next, the unsearched portion of the sample was searched for taxa not found in the original 300 to obtain a comprehensive assessment of richness. With the exception of chironomids and several non-insect taxa (e.g., oligochaetes), organisms were identified to genus.



1. Sisinawa River
2. Apple River
3. Elkhorn Creek
4. Kishwaukee River
5. S. Br. Kishwaukee River
6. Spring Creek
7. Skokie River
8. McDonald Creek
9. Salt Creek
10. Dupage River
11. Midlothian Creek
12. Thorn Creek
13. Plum Creek
14. Kankakee River
15. Spring Creek
16. N. Fk. Vermillion River
17. Farm Creek
18. Big Creek
19. Little Kickapoo Creek
20. Kickapoo Creek
21. Big Bureau Creek
22. Indian Creek
23. E. Fk. LaMoine River
24. Bear Creek
25. McKee Creek
26. Bay Creek
27. Spring Creek
28. Shoal Creek
29. Big Ditch
30. Middle Fk. Vermilion River
31. Saline Branch Salt Fork
32. N. Fk. Vermilion River
33. Black Slough
34. Clear Creek
35. Whitley Creek
36. Bonpas Creek
37. Crooked Creek
38. E. Fk. Kaskaskia River
39. Hickory Creek
40. Hurricane Creek
41. Cahokia Canal
42. Canteen Creek
43. Rayse Creek
44. Big Muddy River
45. Elm River Drainage Ditch
46. N. Fk. Saline River
47. Bankston Creek
48. S. Fk. Saline River
49. Lusk Creek
50. Cache River
51. Big Creek
52. Hutchins Creek
53. Crab Orchard Creek

Fig. 1. Map of Illinois showing the approximate locations of the 53 wadeable streams sampled. Colors indicate major watersheds. Map created by Sally A. McConkey and Kathleen J. Brown, Illinois State Water Survey. Map and legend for watersheds can be found at http://www.watershed.uiuc.edu/getting_involved/map.cfm.

Data Analyses

We organized the 53 streams into discrete groups using an agglomerative hierarchical cluster analysis (PC-ORD; MJM Software Design, 2005). Bray-Curtis dissimilarity (Bray and Curtis, 1957) was used for the dissimilarity matrix due to its robustness for community data. The flexible beta method (value = 0.25) was used to link clusters because it is compatible with Bray-Curtis dissimilarity and because space-distorting properties can be controlled (McCune and Grace, 2002). We used Wishart's (1969) distance function to scale the resulting dendrogram based on the amount of information lost at each step in the analysis. We pruned the dendrogram in what in our estimation was the most ecologically meaningful manner.

We compiled a nonmetric multidimensional scaling (NMDS) ordination on presence-absence macroinvertebrate data using DECODA (Minchin, 1989). Ordinations are effective for summarizing multidimensional community data (McCune and Grace, 2002), and NMDS is more robust and effective for ordination of community data than previous methods, such as detrended correspondence analysis or canonical correspondence analysis (Minchin,

Table 2. Macroinvertebrate metrics calculated for Illinois streams and expected direction of change with increasing anthropogenic disturbance.

Metric	Definition	Expected change
Richness	number of distinct taxa	decrease
Shannon diversity	index that reflects richness and evenness of a community	decrease
MBI	macroinvertebrate biotic index, based on Hilsenhoff (1987), modified for Illinois; reflects tolerance to pollution	increase
EPT	number of distinct taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera	decrease
% EPT	relative abundance of members of the orders Ephemeroptera, Plecoptera, and Trichoptera	decrease
% Oligochaeta	relative abundance of aquatic worms	variable
% Chironomidae	relative abundance of midges (Diptera: Chironomidae)	increase
% Gatherer	relative abundance of the functional group gatherers	variable
% Filterer	relative abundance of the functional group filterers	increase
% Scraper	relative abundance of the functional group scrapers	decrease
% Shredder	relative abundance of the functional group shredders	decrease
% Dominance	relative abundance of the most common taxon	increase

1987; Battaglia et al., 2002). The distance matrix was calculated using Bray-Curtis dissimilarity. We chose the best ordination (lowest stress solution) from 100 random starting configurations and 1000 iterative adjustments of each starting configuration. The proper number of dimensions was determined by plotting stress values versus dimensions. We also used the vector-fitting procedure in DECODA. This is a type of correlation analysis that gives a confident estimate of the variables responsible for separating sites in ordination space (Kantlivas and Minchin, 1989).

To evaluate the relative biotic integrity of our groups of streams, we used assemblage-level bioassessment metrics. Metrics were chosen to reflect diversity, richness, tolerance, and functional structure and composition (Table 2). The macroinvertebrate biotic index (MBI) is a version of the Hilsenhoff Biotic Index (Hilsenhoff, 1987) that has been calibrated for Illinois taxa (Fitzpatrick et al., 2004). An ANOVA in conjunction with a Tukey-Kramer honestly significant difference test was used to test for differences in metric scores among the groups of streams.

Results

A total of 14,015 macroinvertebrates, comprising 85 taxa, were identified. We were able to remove 300 macroinvertebrates from the composite samples in most cases. Samples containing less than 300 total macroinvertebrates were typically from highly degraded streams. For example, Farm and Canteen creeks drain predominantly industrial areas, and samples from these sites contained only 42 and 53 macroinvertebrates, respectively. The number of macroinvertebrates identified was not a significant vector-fitting variable to the ordination, indicating that sampling bias was not a factor in our results.

Richness of the study streams ranged from 7 to 29. Taxa present in nearly every stream included *Tubifex* (Oligochaeta: Tubificidae), several subfamilies of midges (Diptera: Chironomidae), *Simulium* (Diptera: Simuliidae), *Stenelmis* (Coleoptera: Elmidae), *Baetis* (Ephemeroptera: Baetidae), and *Cheumatopsyche* (Trichoptera: Hydropsychidae). In contrast, taxa such as *Palaemonetes* (Decapoda: Palaemoniidae), *Nigronia* (Neuroptera: Corydalidae), *Attenella* (Ephemeroptera: Ephemerellidae), *Ptilostomus* (Trichoptera: Phryganeidae), and *Amphinemura* (Plecoptera: Nemouridae) were collected from only one or two streams.

Four groups of streams were assembled according to their compositional similarity (Fig. 2). Most of the sites we examined drained watersheds that were dominated by a single land use, which allowed us categorize the sites by this parameter and examine general patterns in land use among the four groups identified in the dendrogram (Fig. 2). Most of the sites draining agricultural land were associated with Group 1, and most of the forested sites occurred in Group 2. Group 3 contained most of the urban streams, although two highly degraded urban streams were identified as a distinct group (Group 4) and were dissimilar from other streams. Streams in geographic proximity often did not cluster together, although there was tendency for this to happen with forested sites in southern Illinois and with urban sites in the northeastern portion of the state.

Concentrations of nutrients and dissolved ions were generally lowest in Group 2, followed by Groups 1, 4, and 3 (Table 3). Group 3 had a much higher concentration of DRP than the other groups. Taxa richness ($F_{[3,30]} = 9.40$; $P < 0.0001$) was higher in Group 1 and 2 streams than in Groups 3 and 4 (Table 4). Similarly, Shannon diversity ($F_{[3,30]} = 5.03$; $P = 0.004$) was greater in Groups 1 and 2 than in Group 4. Group 2 had a lower MBI score ($F_{[3,30]} = 11.04$; $P < 0.0001$) than Groups 3 and 4, and Group 1 had a lower MBI score than Group 4. The percent Oligochaeta metric ($F_{[3,30]} = 10.97$; $P < 0.0001$) was higher in Group 4 than in Groups 1 and 3, which in turn were higher than in Group 2. Groups 1 and 2 had more Ephemeroptera, Plecoptera, and Trichoptera taxa ($F_{[3,30]} = 2.99$; $P = 0.039$) than Groups 3 and 4. Overall, patterns suggested the highest biotic integrity in forested streams and the lowest in urban streams.

We identified the same NMDS solution for 98 of 100 random starting configurations. Three dimensions were necessary to achieve an acceptable stress level of 0.182. Additional dimensions did not greatly reduce this stress. Of 18 measured environmental variables, seven had significant effects on the ordination (Table 5; Fig. 3). Habitat measures (survey score and substrate stability) as well as nutrient and dissolved ion measures correlated in an antagonistic manner to the ordination (Fig. 3). Groups 1 and 2, determined from the cluster analysis, aligned best with habitat quality and stable substrata, whereas Group 3 aligned best with higher nutrient and dissolved ion concentrations. Group 4 was notably isolated in the ordination and seemed to be aligned best with the percentage of run habitat.

Discussion

As predicted, differences in macroinvertebrate assemblages of Illinois streams seemed to be a function of physical habitat and

nutrient concentrations. However, we were unable to support our hypothesis that habitat quality would be the primary factor governing biotic integrity because habitat degradation was generally evident in streams with elevated nutrient concentrations. Our analyses indicate that habitat quality and nutrient concentrations were of equal importance in structuring macroinvertebrate communities across Illinois streams. In a similar study, Wang et al. (2007) found that macroinvertebrate assemblage measures were strongly linked to physical parameters and nutrient concentrations in wadeable Wisconsin streams. Our study provides further evidence that nutri-

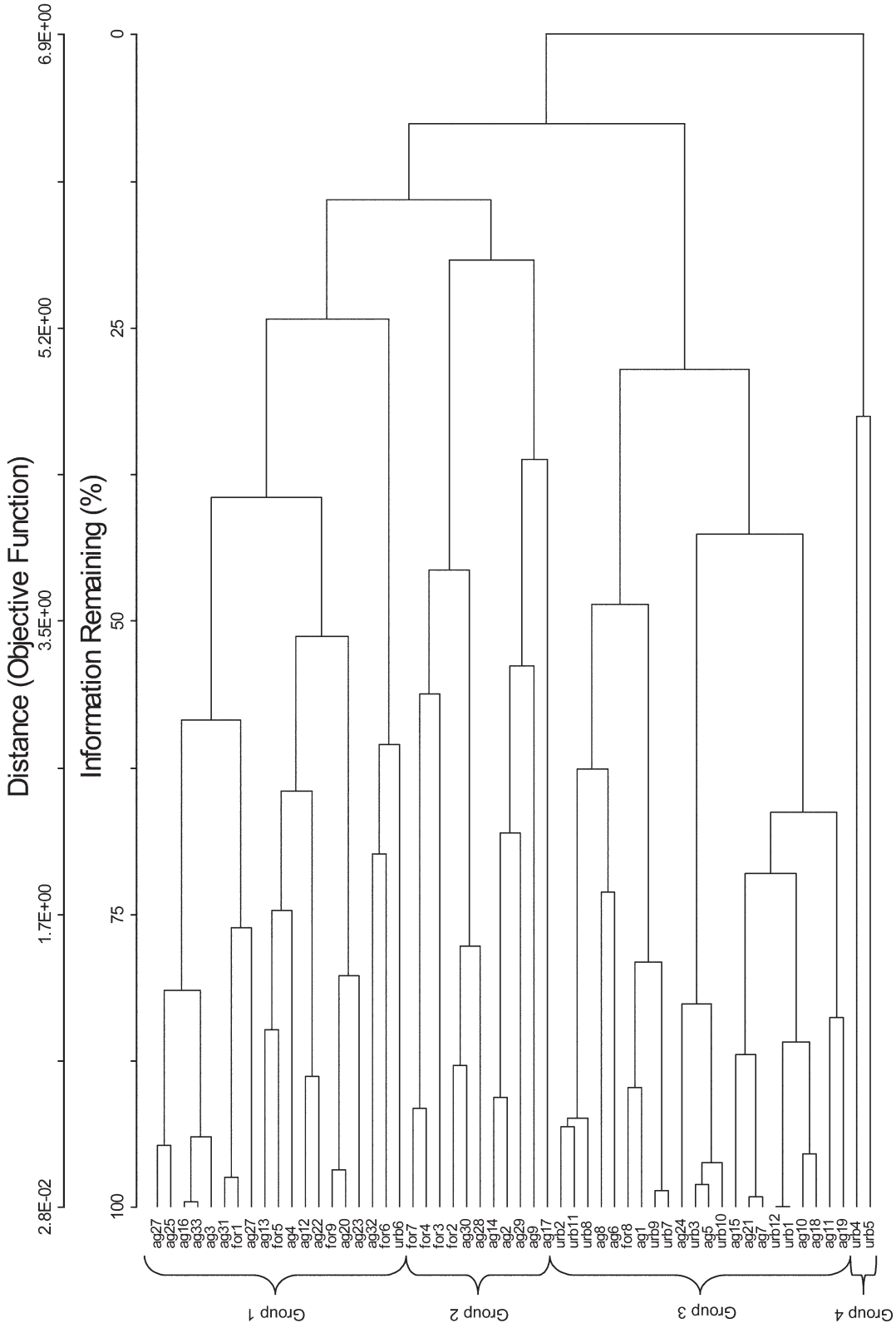


Fig. 2. Dendrogram of Illinois study streams resulting from a hierarchical cluster analysis of benthic macroinvertebrate presence–absence data. Streams were separated into the four groups shown based on Bray-Curtis dissimilarity. Stream site codes are from Table 1.

Table 3. Mean (SE) values of water chemistry parameters for four groups of streams as determined by cluster analysis.

Parameter	Group 1	Group 2	Group 3	Group 4
Chloride, mg L^{-1}	40 (12)	35 (16)	92 (17)	56 (17)
Sulfate, mg L^{-1}	38 (14)	12 (4)	44 (7)	NA†
NO_3–N, mg L^{-1}	3.7 (0.8)	1.1 (0.6)	5.1 (0.6)	4.9 (0.8)
NH_4–N, mg L^{-1}	0.2 (0.0)	0.2 (0.0)	0.4 (0.1)	0.2 (0.1)
DRP,‡ mg L^{-1}	0.08 (0.01)	0.06 (0.01)	0.53 (0.18)	0.17 (0.04)
Total P, mg L^{-1}	0.23 (0.04)	0.22 (0.04)	0.64 (0.03)	0.3 (0.01)
DOC,§ mg L^{-1}	3.9 (0.5)	5.8 (1.2)	3.8 (0.5)	NA
Silica, mg SiO_2 L^{-1}	10.4 (1.0)	9.3 (1.7)	8.7 (0.9)	NA
pH	7.5 (0.2)	7.3 (0.1)	7.8 (0.1)	7.9 (0.3)
Specific conductivity, μS cm^{-1}	668 (47)	581 (123)	895 (87)	724 (115)

† NA indicates data were unavailable.

‡ DRP, dissolved reactive P.

§ DOC, dissolved organic C.

ent pollution is related to degraded biotic integrity in streams (e.g., Miltner and Rankin, 1998), although separating the effects of nutrients per se from other factors remains problematic.

The inverse relationship that we observed between habitat quality and nutrient concentrations is intuitive considering the impacts of row crop agriculture, which constitutes ~95% of the land use in the northern two thirds of Illinois (DeWalt et al., 2005). A combination of channelization and tile drainage results in habitat impairment through scouring, sedimentation, habitat homogenization, and altered riparian vegetation (Cooper, 1993). Furthermore, widespread subsurface tile drainage in Illinois enhances transport of N (Gentry et al., 1998; Royer et al., 2006), P (Gentry et al., 2007; Stamm et al., 1998), and numerous agrochemicals (David et al., 2003) into streams that also receive nutrient inputs from surface runoff. Both N and P were identified as predictors of macroinvertebrate assemblages, but in many Illinois streams, N concentrations are sufficiently high to preclude N limitation of algae (Morgan et al., 2006). This suggests P may be the critical nutrient affecting biotic integrity among the sites, particularly in agricultural regions of the state.

Streams with the lowest biotic integrity were located in urban/residential areas. This has major implications for the state of Illinois because nearly seven million people live within a six-county area around Chicago (DeWalt et al., 2005). We also sampled urban streams near Peoria, Springfield, East St. Louis, and numerous smaller population centers. As would be expected in streams draining watersheds with extensive impervious area, most had high concentrations of nutrients (Paul and Meyer, 2001; Meyer et al., 2005; Moore and Palmer, 2005). Channelization is the most common form of habitat modification in urban streams (Tavzes et al., 2006), and the urbanized streams in our study had the poorest physical habitat quality. Among other detrimental effects, channelization may cause habitat homogenization and reduced biodiversity because of reduced niche potential (Beisel et al., 2000). Farm Creek and Spring Creek provided clear examples of high degrees of impairment from urbanization, as indicated by their poor metric values and status as outliers in the ordination. Farm Creek drains much of East Peoria, and Spring Creek drains the city of Springfield. These two streams were dominated by a few pollution-tolerant, non-insect taxa (e.g., sowbugs [Isopoda]), whereas agricultural streams typically were dominated by chironomids and oligochaetes.

Streams in southern Illinois represented the opposite end of the habitat–nutrient gradient. These streams drain primarily forested catchments in the Shawnee National Forest, which experience relatively little anthropogenic disturbance, have high-quality physical habitat, and have much lower nutrient concentrations than streams from other areas in the state. In particular, Lusk and Hutchins creeks have high substratum and habitat heterogeneity, intact riffle-run-pool morphology, and forested riparian zones. These factors likely contributed to the high richness and diversity scores, the high numbers of pollution intolerant taxa, and the low nutrient concentrations observed in Lusk and Hutchins creeks. Wang et al. (1997) found that fish community integrity improved with increasing forested land area in Wisconsin, and our study indicates a similar relationship for stream macroinvertebrates in Illinois.

The pattern we observed of lower nutrient concentrations in streams with better physical habitat quality was in some cases (e.g., streams draining undisturbed forested catchments) clearly related to a lack of human impacts on each factor. However, in some cases this relationship may also be related to direct and indirect interactions between physical habitat structure and nutrient dynamics. In addition to directly increasing biological activity and thus nutrient uptake (e.g., Newbold et al., 1983; Webster and Ehrman, 1996), high physical habitat and substratum quality can indirectly result in decreased nutrient export through enhanced physical retention, which allows for increased biological uptake (Peterson et al., 2001).

An important implication of this study for Illinois stream bioassessment is that that similarity of macroinvertebrate communities in streams was not linked to the geographic proximity of the

Table 4. Mean (SE) values of bioassessment metrics that significantly discriminated among four groups of streams as determined by cluster analysis.

Parameter	Group 1	Group 2	Group 3	Group 4
Richness	18.9a† (1.1)	20.8a (1.4)	14.4b (1.0)	7.0b (3.2)
MBI	7.1ab (0.2)	6.2a (0.3)	7.8b (0.2)	9.5c (0.7)
EPT richness	6.1a (0.5)	6.9a (0.7)	2.1b (0.5)	0.5b (1.6)
Shannon diversity	1.8a (0.1)	1.8a (0.1)	1.5ab (0.1)	0.8b (0.3)
% Oligochaeta	17.3abc (4.1)	9.1b (5.4)	31.2c (3.8)	79.9d (12.7)

† Letters after mean values indicate significant differences among stream groups (Tukey-Kramer's honestly significant difference test, $P < 0.05$). Descriptions of metrics appear in Table 2.

Table 5. Significant vectors of the nonmetric multidimensional scaling ordination performed on macroinvertebrate presence-absence data for Illinois streams and correlation coefficients (*r*) and significance (*P*).

Vector	*r*	*P*
Habitat quality	0.64	<0.0001
% stable substrate	0.62	<0.0001
% run habitat	0.45	0.009
NO_3–N concentration	0.60	<0.0001
DRP† concentration	0.50	0.005
pH	0.64	0.001
Silica concentration	0.65	0.012

† DRP, dissolved reactive P.

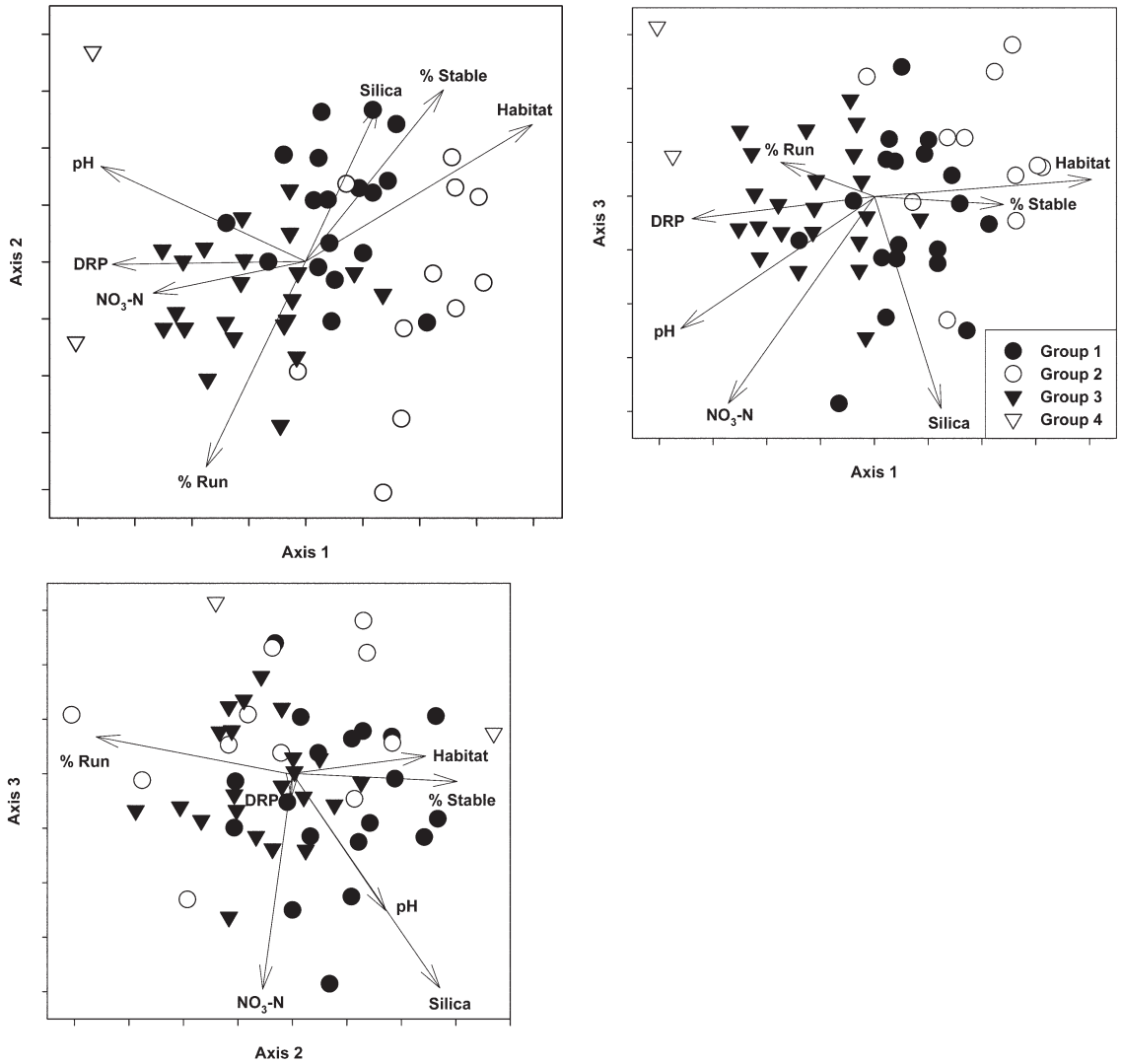


Fig. 3. Nonmetric multidimensional scaling ordination of presence–absence data for benthic macroinvertebrate taxa in Illinois streams. Graphs represent Axis 1 vs. Axis 2, Axis 1 vs. Axis 3, and Axis 2 vs. Axis 3. The stream groups were determined by cluster analysis (Fig. 2). Correlation coefficients and *P* values for vectors are listed in Table 4. DRP, dissolved reactive P.

streams. Previous studies also have reported that streams in the same geographic region do not necessarily contain similar macroinvertebrate assemblages (Parsons and Norris, 1996; Reynoldson et al., 1997; Turak et al., 2000), which is evidence that a strictly ecoregional approach to stream classification and reference choice may prove problematic. We therefore suggest using compositional dissimilarity measures with multivariate analyses to avoid problems that could arise if streams in geographic proximity are assumed to have the same biotic potential.

Acknowledgments

This research was funded by the state of Illinois through the Council on Food and Agricultural Research (C-FAR) Water Quality SRI. We thank L. Battaglia and P. Minchin for statistical guidance and L. Gentry, T. Marusarz, C. Mitchell, M. McDaniel, and A. Morgan for field assistance. We also thank J. McPherson, J. Reeve, and three anonymous referees for constructive comments on the manuscript.

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Environmental Management (2008) 41:516–527
DOI 10.1007/s00267-007-9058-2

Macroinvertebrate Responses to Constructed Riffles in the Cache River, Illinois, USA

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Published online: 9 January 2008
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Abstract Stream restoration practices are becoming increasingly common, but biological assessments of these improvements are still limited. Rock weirs, a type of constructed riffle, were implemented in the upper Cache River in southern Illinois, USA, in 2001 and 2003–2004 to control channel incision and protect high quality riparian wetlands as part of an extensive watershed-level restoration. Construction of the rock weirs provided an opportunity to examine biological responses to a common in-stream restoration technique. We compared macroinvertebrate assemblages on previously constructed rock weirs and newly constructed weirs to those on snags and scoured clay streambed, the two dominant substrates in the unrestored reaches of the river. We quantitatively sampled macroinvertebrates on these substrates on seven occasions during 2003 and 2004. Ephemeroptera, Plecoptera, and Trichoptera (EPT) biomass and aquatic insect biomass were significantly higher on rock weirs than the streambed for most sample periods. Snags supported intermediate EPT and aquatic insect biomass compared to rock weirs and the streambed. Nonmetric multidimensional scaling (NMDS) ordinations for 2003 and 2004 revealed distinct assemblage groups for rock weirs, snags, and the streambed. Analysis of similarity supported visual interpretation of NMDS plots. All pair-wise substrate comparisons differed significantly, except recently constructed weirs versus older weirs. Results indicate positive responses by macroinvertebrate assemblages to in-stream restoration in the Cache River. Moreover, these responses were not evident with more common measures of total density, biomass, and diversity.

Keywords Macroinvertebrates · Biological assessment · EPT · Stream restoration · Artificial riffle · Rock weir

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Introduction

Ecological restoration of rivers and streams has become an important focus in natural resource management. Considerable effort has been devoted to restoring the ecological and social functions of rivers and streams, even as they continue to degrade (Palmer and Allen 2006). Costs of river restoration in the United States alone are estimated at greater than $1 billion/year (1990–2003; Bernhart and others 2005). Despite significant resources dedicated to river restorations, post-construction assessment or monitoring remains limited (Bernhart and others 2005). Studies of the effects of restoration activities on lotic and riparian ecosystems are necessary to refine criteria for successful ecological restoration, improve the efficacy of techniques applied in river restoration, and promote future restoration projects.

In-stream restoration usually involves engineering of the channel based on geomorphologic principles (Brookes and Sear 1996). Newbury and Gaboury (1993, 1994) developed an approach to channel design that combines geomorphologic and hydraulic elements of the stream at multiple scales with fish habitat requirements to achieve ecologically sound restoration of river channels. Constructed riffles have been used extensively to reestablish pool–riffle sequences that are characteristic of many natural streams (e.g., Newbury and others 1997; Harper and others 1998; Harrison and others 2004) and are currently a common

technique used to stabilize stream channels in Illinois. These sequences are generally formed by constructing a series of riffles from natural materials that create stable substrate and turbulent flows similar to those of natural riffles and impound water into upstream pools.

Riffles constructed over a multiyear period in the upper Cache River in southern Illinois, USA, provided an opportunity to examine biological, specifically macroinvertebrate, responses to in-stream restoration activities. The constructed riffles (hereafter called rock weirs to avoid confusion with natural riffle habitat) were designed to control channel incision in the restored stream section, improve in-stream habitat, and to protect the hydrologic integrity of ecologically important riparian wetlands as part of an extensive watershed-level restoration project. Although these structures were not designed to directly repair the damage sustained by the channel over the past several decades, we anticipated that they would set the river and its associated riparian wetlands on a trajectory of ecosystem recovery as the stream channel stabilized. We hypothesized that the rough, stable substrate and hydraulic heterogeneity of the rock weirs would result in increased macroinvertebrate diversity and biomass and alter macroinvertebrate assemblage structure relative to unrestored sections of the river.

Our objective was to quantify macroinvertebrate responses to rock weirs in restored segments of the upper Cache River by comparing macroinvertebrate assemblages on rock weirs to those on the clay streambed and snags in unrestored reaches of the river. We predicted that diversity and biomass would be highest on the rock weirs. We also hypothesized that biomass of aquatic insects in the orders Ephemeroptera, Trichoptera, and Plecoptera (EPT) and aquatic insect taxa with aerial adults would be highest on the rock weirs. Ephemeroptera, Trichoptera, and Plecoptera are generally regarded as taxa sensitive to poor habitat quality and emerging insects represent a source of stream subsidies to the riparian ecosystem (Huryn and Wallace 2000; Baxter and others 2005); thus we focused on these groups as ecologically important subsets of aquatic macroinvertebrate assemblages. We assessed differences in overall assemblage structure among the three habitats using nonmetric multidimensional scaling (NMDS) and indicator species analysis.

Methods

Study Site

The Cache River is highly modified and is divided into upper and lower sections based on differences in hydrology. We conducted our study in the upper Cache River (Fig. 1) in the Cache River State Natural Area. The upper Cache River drains approximately 632 km^2 of forest, agricultural land, and wetlands in southern Illinois. Mean annual discharge is about 8.2 m^3/s (1984–2003). United States Geological Survey Station 03612000, Cache River at Forman, Illinois, is located immediately downstream of the restored stream section (latitude 37°20'11", longitude 88°55'26"; Fig. 1). Additional physicochemical data for the upper Cache River can be found at the USGS Station 03612000 website (http://waterdata.usgs.gov/il/nwis/nwismap/?site_no=03612000&agency_cd=USGS).

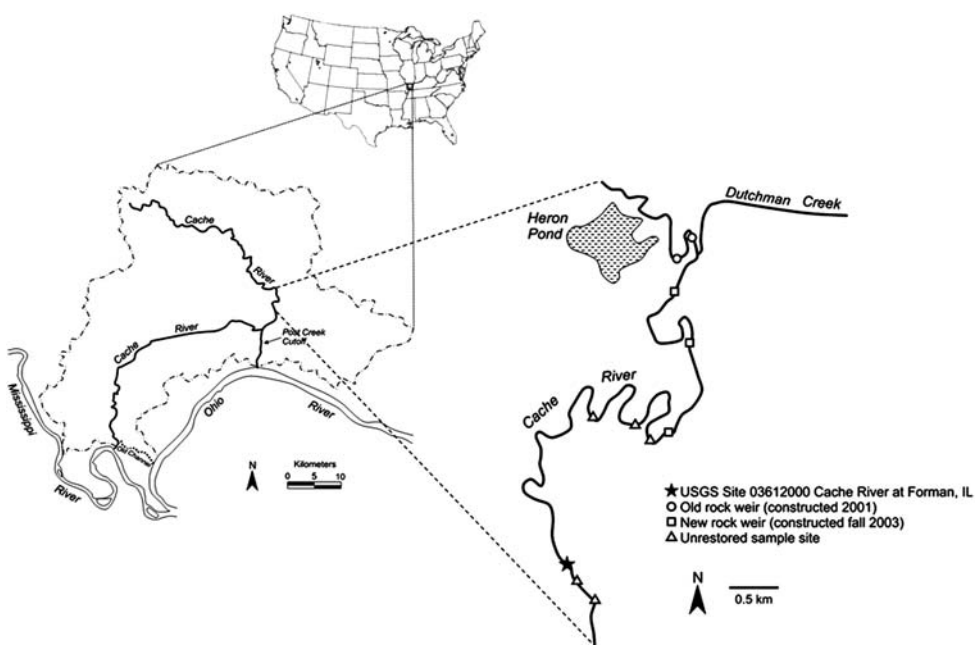

Fig. 1 Map of the study area in Johnson County, Illinois showing locations of sampled rock weirs and unrestored sampling sites

A series of channel modifications over the past century has substantially altered the hydrology of the Cache River (Demissie 1997). The Post Creek Cutoff, constructed around 1915, caused the greatest change in river hydrology. The Cutoff diverted most of the water from the upper Cache River into a more direct route to the Ohio River. Consequently, higher flow velocities caused severe channel erosion. The channel upstream of the cutoff has severely incised and widened. Although once highly connected to its floodplain, the upper Cache River now rarely leaves its banks and entrenched gullies drain critical riparian wetlands. The watershed encompasses most of the remaining bald cypress/tupelo swamps in southern Illinois (Dorge and others 1984) and protecting the integrity of these wetlands was a major goal of the restoration project.

Restoration activities in the Cache River watershed included construction of 25 rock weirs similar to those designed by Newbury and Gaboury (1993) along a degraded segment of the upper Cache River. The Illinois State Water Survey (ISWS) constructed six riffles in the vicinity of Heron Pond (Fig. 1) in 2001, creating a series of riffles and pools that extended approximately 1.7 km upstream of the confluence of the Cache River and Dutchman Creek. Construction of a second series of 19 riffles and pools spanning approximately 6.7 km of the Cache River downstream of a segment restored in 2001 was completed in fall of 2003 (seven riffles) and 2004 (12 riffles).

Study Design

Our study of the effects of in-stream restoration in the Cache River on macroinvertebrate assemblages began in spring 2003. We did not establish reference sites upstream of the series of rock weirs constructed in 2001 because of logistical constraints. Instead, we compared macroinvertebrate assemblages on the two habitat types that predominated the unrestored reaches of the upper Cache River downstream of the restored section, scoured clay streambed and snags (Fig. 2a), to assemblages on both established and newly constructed rock weirs (Fig. 2b).

In the first year of our study (2003), we compared macroinvertebrate assemblages on rock weirs constructed in 2001 (old weirs) to those on snags and clay streambed at sites scheduled for rock weir construction in the following year. In the second year of our study (2004), we compared assemblages on rock weirs constructed in fall 2003 (new weirs) to the previously sampled upstream rock weirs, as well as snags and the streambed at unrestored downstream sites.

Macroinvertebrate Sampling

We sampled macroinvertebrates on rock weirs, snags, and clay streambed on three occasions in 2003 (April, July, and October) and four occasions in 2004 (February, April, July, and October). In 2003, we sampled macroinvertebrate communities on two old weirs (constructed in 2001) and on the streambed at three downstream pre-restoration sites where construction of new weirs was scheduled for fall 2003. We also sampled macroinvertebrates on snags at two of the pre-restoration sites. In 2004, we sampled macroinvertebrates on the same two old weirs, on the three new weirs constructed in fall 2003, and on the streambed and snags at two unrestored sites downstream of the restored stream section. The unrestored sampling sites were shifted further downstream in July 2004 because suitable and accessible snags were scarce and again in October because downstream progress of weir construction unexpectedly surpassed our unrestored sites.

We collected three samples from each weir. Weir substrates were generally too coarse to use traditional Surber or Hess samplers. Instead, we removed 1–8 rocks from the surface of the weir using a 0.09 m^2 Surber sampler as a catch-net (250-μm mesh) for dislodged invertebrates. We

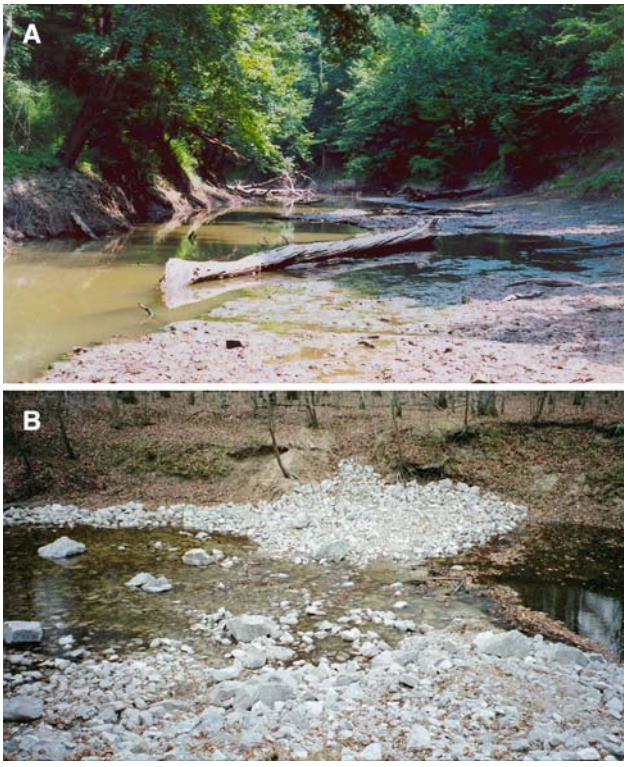

Fig. 2 Photos showing (**a**) representative snag and clay streambed habitats (looking upstream) and (**b**) a constructed riffle (age approximately 1 year; flow from right to left in photo) in the upper Cache River in southern Illinois, USA. Both photos show base flow conditions in the river

then placed dislodged rocks in an approximately 60 L container of stream water and scrubbed them with a brush to remove remaining invertebrates. Contents of the Surber net and the container were concentrated using a 250-μm sieve and preserved in 8% formalin. We photographed sampled rocks next to a meter stick and used ImageJ digital imaging software (Rasband 2005) to calculate the streambed surface area occupied by each rock. Mean sampled surface area was 0.063 $m^2 \pm 0.015$ SD ($n = 78$) and mean substrate size (longest axis) was 175 mm ± 74 SD ($n = 250$). We selected sample locations for each weir haphazardly from within accessible areas. It was necessary to restrict sampling to wadeable locations on one side of each weir during February and April because high flows were concentrated in the center of the weir, precluding both sampling in the center of the weir and access to the opposite bank. The presence of boulders larger than the Surber frame also restricted the availability of potential sample locations within the weirs.

We collected three core samples from the streambed at each unrestored site in 2003 and 2004. A 20-cm-diameter stovepipe corer was forced into the substrate. All water and up to 10 cm of substrate (less if extremely hard clay was encountered) was removed from the corer and placed in a 19-L bucket. All material collected from the corer was elutriated through a 250-μm sieve. Elutriation was repeated with fresh stream water until rinse water was clear. Contents of the sieve were preserved in 8% formalin. The streambed was sampled in April 2003 using a Surber sampler because water was generally too deep and swift to use a coring device. In this case, we pressed the downstream edge of the Surber frame into the clay substrate and disturbed the substrate within the frame by hand. This technique was similar to the procedure used by Grubaugh and others (1997) to sample deep cobble areas of the Little Tennessee River.

We sampled three snags at two unrestored sites in 2003 and 2004. We covered up to approximately 60 cm of a submerged branch with a 250-μm-mesh bag and secured the opening of the bag around the branch. We then removed the covered section by sawing or breaking the branch. We emptied the contents of the bag into an approximately 60 L container of stream water. Invertebrates were scrubbed from the snag surface with a brush and removed with forceps. We then concentrated the contents of the container using a 250-μm sieve and preserved them in 8% formalin. We measured total length and diameter (three measurements) of the snag and calculated the sampled area using the formula for surface area of a cylinder. Mean sampled surface area for snags was 0.082 $m^2 \pm 0.05$ SD ($n = 42$).

We rinsed preserved samples from all substrate types through stacked 1-mm and 250-μm sieves in the laboratory. Macroinvertebrates were removed from debris in the > 1 mm (coarse) fraction by eye and from the < 1 mm > 250 μm (fine) fraction using a dissecting microscope (6–30×). When necessary, we subsampled fine fractions only (up to 1/64) using a Folsom plankton splitter. Subsamples contained a minimum of 75 individuals. Macroinvertebrates from each fraction were then measured (total body length) and identified to the lowest practical taxonomic level (genus for most taxa) using keys in Merritt and Cummins (1996) and Smith (2001). Taxa were assigned to functional feeding groups (FFG) according to Merritt and Cummins (1996) or Barbour and others (1999). Crayfish were excluded from all analyses because they are extremely large relative to other stream invertebrates and because our collection methods were not effective for sampling crayfish.

Density of each taxon was standardized to numbers of individuals/m^2. Biomass within each 1-mm size class was calculated as mg ash-free dry mass (AFDM) using taxon-specific length-mass relationships from Benke and others (1999). We developed our own length-mass regressions for some taxa following procedures of Benke and others (1999). Dry mass estimates were multiplied by 0.9 to estimate AFDM (Benke 1993). Taxon richness was calculated as the total number of identified taxa for a site on each date. Shannon diversity (H′) was calculated using natural logs following standard procedures provided by Brower and others (1990). Only aquatic insect taxa in the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT), which are generally considered sensitive environmental indicators (Barbour and others 1999), were included in EPT biomass. Our analyses of aquatic insect biomass included the orders Ephemeroptera, Odonata, Plecoptera, Megaloptera, Trichoptera, and Diptera. We excluded Coleoptera (essentially Elmidae in our study) from our estimates of aquatic insect biomass because their contribution to emergence production is unclear. Elmidae are known to disperse by flight after emergence, but are generally aquatic as adults.

Statistical Analyses

Effects of habitat (old weir versus snag versus clay in 2003; old weir versus new weir versus snag versus clay in 2004) and sample date (interval) on macroinvertebrate richness, percent dominance, Shannon diversity, density, biomass, EPT biomass, and aquatic insect biomass were analyzed using 2-way ANOVA with habitat and date as fixed effects and included the habitat × date interaction. Density, biomass, and EPT and aquatic insect biomass were transformed ($\log_{10}[x + 1]$) to reduce heteroscedasticity and improve normality of the data (Zar 1996). Data from each year were analyzed separately because we had insufficient replication in our study to include inter-annual variation as a factor in our analyses. Pair-wise tests based on the *t*-statistic were

used to evaluate differences between habitat pairs when the habitat × date interaction was not significant. If the interaction was significant, then habitat pairs were evaluated by pair-wise contrasts within each sample date (Jaccard 1998). Results of these contrasts were considered significant at $\alpha = 0.05$ and marginally significant at $\alpha = 0.1$. We used JMP statistical software (version 5, SAS Institute, Cary, North Carolina) for all univariate analyses.

We used NMDS (Minchin 1987) ordinations to visualize differences in assemblage structure among rock weirs, the streambed, and snags. Ordinations were constructed separately for macroinvertebrate assemblages collected in 2003 and 2004 based on Bray-Curtis dissimilarity matrices. Bray-Curtis coefficients were calculated using transformed ($\log_{10}[x + 1]$; McCune & Grace 2002) taxon densities. Ordinations were constructed using original samples (3 per site), not site averages. We performed NMDS using 100 random initial configurations for each of one to six dimensions. We examined scree plots of resulting stress values versus dimensionality to select the appropriate number of dimensions for our data.

We used analysis of similarity (ANOSIM; Clarke 1993) as a complement to NMDS to statistically test differences in assemblage structure among habitats and sample dates. Bray-Curtis coefficients were calculated using average density of each taxon for each site because samples within sites could not be considered independent. The ANOSIM statistic R denotes the magnitude of the difference among groups. R equals 1 when groups differ completely and equals 0 when there are no differences detected among groups. We used the software DECODA (Minchin 1998) to perform NMDS and ANOSIM analyses.

Indicator species analysis (Dufrene and Legendre 1997) was used to identify taxa that were characteristic of assemblages inhabiting each of our study habitats. Indicator values (IndVal) range from 0 to 100, where the maximum value indicates that a taxon is restricted entirely to the designated group and occurs at all sites of that group. Monte Carlo tests (1000 permutations) were used to determine the significance of the maximum indicator value for each taxon. Indicator value calculations and Monte Carlo tests were performed using PC-ORD version 4.0 (McCune and Mefford 1999).

Results

Invertebrate Diversity, Density, and Biomass

We collected 90 macroinvertebrate taxa over the entire study, including 26 EPT taxa. Mean taxa richness across all sampled habitats was 22.0 ± 1.2 (mean ± 1 SE) in 2003 and 23.8 ± 1.1 in 2004.

Richness and diversity relationships between habitats varied with sample date. Streambed and snag habitats supported higher taxa richness compared to rock weirs in both July 2003 and July 2004 ($p < 0.05$). Taxa richness on snags was greater than on old weirs in April 2003 and old weirs and the streambed in April 2004 ($p < 0.05$). However, no significant differences between habitats were detected in October 2003, February2004, or October 2004. Shannon diversity and percent dominance did not show notable patterns among habitats. Mean Shannon diversity was 1.73 ± 0.07 in 2003 and 1.74 ± 0.06 in 2004 across all habitats, and mean percent dominance was 46.1 ± 2.7 in 2003 and 47.2 ± 2.7 in 2004 across all habitats.

Two-way ANOVAs on total macroinvertebrate densities (Table 1) revealed significant effects of both habitat and date in both 2003 (habitat, $F_{[2,12]} = 6.67$, $p = 0.01$; date, $F_{[2,12]} = 24.51$, $p < 0.0001$) and 2004 (habitat, $F_{[3,20]} = 5.03$, $p = 0.0092$; date, $F_{[3,20]} = 18.94$, $p < 0.0001$), as well as a significant interaction in 2004 (habitat × date; $F_{[9,20]} = 2.52$, $p = 0.041$). Pair-wise comparisons of habitats for 2003 indicated that densities on rock weirs were significantly higher than those on snags ($p = 0.0033$) and marginally higher than densities on the streambed ($p = 0.084$). Pair-wise differences between habitats were tested separately within each date for 2004 because the interaction was significant. However, we did not detect clear patterns in total density among habitat types in 2004.

Analysis of total biomass (Table 1) for 2003 showed a significant effect of sample date ($F_{[2,12]} = 11.75$, $p = 0.0015$), but no effect of habitat ($F_{[2,12]} = 1.86$, $p = 0.20$) or the habitat × date interaction ($F_{[4,12]} = 0.93$, $p = 0.48$). Similarly, in 2004 there was a significant effect of date ($F_{[3,20]} = 8.53$, $p = 0.0008$) and no effect of habitat ($F_{[3,20]} = 1.87$, $p = 0.17$). The interaction was significant in the 2004 analysis ($F_{[9,20]} = 4.02$, $p = 0.0046$). Results of pair-wise comparisons of habitat within each date were variable. In February 2004, old weirs and snags supported significantly higher biomass than the new weirs ($p < 0.05$) and biomass of old weirs was marginally higher than streambed biomass ($p = 0.055$). No significant differences were found between habitat pairs in April 2004. In July 2004, both new and old rock weirs supported significantly higher biomass than snags and streambed ($p < 0.05$); and, in October 2004, streambed biomass was significantly higher than that of all other habitats ($p < 0.05$).

Two-way ANOVA showed significant effects of habitat ($F_{[2,12]} = 11.22$, $p = 0.0018$) and date ($F_{[2,12]} = 9.68$, $p = 0.0031$) on EPT biomass in 2003. The habitat × date interaction was not significant ($F_{[4,12]} = 0.16$, $p = 0.95$). Tests of pair-wise habitat effects indicated that rock weirs supported significantly higher EPT biomass than the streambed ($p = 0.0005$) and snags ($p = 0.029$). Biomass

of EPT taxa was marginally higher on snags over the streambed ($p = 0.067$).

Analysis of EPT biomass for 2004 also indicated significant effects of habitat ($F_{[3,20]} = 50.20$, $p < 0.0001$) and date ($F_{[3,20]} = 18.69$, $p < 0.0001$), and the interaction between habitat and date was significant for 2004 ($F_{[9,20]} = 3.63$, $p = 0.0078$). Thus, pair-wise differences between habitats were tested separately within each sample date. Streambed habitats supported consistently lower EPT biomass than rock weirs and snags (Table 2). These differences were significant for all comparisons of new and old rock weirs to the streambed ($p < 0.05$; see Appendix 1 for pair-specific p-values), while statistical differences between snags and other habitat types were variable.

Results of analyses of aquatic insect biomass (Table 2) were very similar to those we observed for EPT biomass. Habitat ($F_{[2,12]} = 13.04$, $p = 0.0010$) and date ($F_{[2,12]} = 25.15$, $p < 0.0001$) showed significant effects on aquatic insect biomass in 2003 and the habitat × date interaction was not significant ($F_{[4,12]} = 1.34$, $p = 0.31$). Rock weirs supported significantly higher aquatic insect biomass than both snags ($p = 0.012$) and the streambed ($p = 0.0003$). Analysis of aquatic insect biomass in 2004 showed significant effects of habitat ($F_{[3,20]} = 31.88$, $p < 0.0001$), date ($F_{[3,20]} = 17.37$, $p < 0.0001$), and the habitat by date interaction ($F_{[9,20]} = 4.37$, $p = 0.0029$). Streambed habitats supported significantly lower aquatic insect biomass than old weirs, new weirs, and snags for all dates, excluding the streambed–new weir comparison for October 2004 (see Appendix 1 for pair-specific p-values).

Table 2 Mean (1 SE in parentheses) EPT biomass and total aquatic insect biomass collected from four habitats sampled in the Cache River, Johnson Co., Illinois in 2003–2004

	old weirs	new weirs	snags	streambed
EPT biomass (mg AFDM/m^2)				
April 2003	187 (17)	—	34 (25)	16 (6)
July 2003	1357 (342)	—	489 (193)	252 (213)
October 2003	899 (315)	—	235 (14)	111 (50)
February 2004	134 (43)	171 (60)	231 (9)	13 (12)
April 2004	631 (31)	785 (170)	764 (103)	14 (0)
July 2004	2430 (54)	2581 (443)	276 (29)	76 (25)
October 2004	1155 (756)	518 (161)	233 (71)	113 (42)
Total aquatic insect biomass (mg AFDM/m^2)				
April 2003	241 (19)	—	108 (42)	25 (10)
July 2003	1882 (169)	—	623 (209)	379 (186)
October 2003	1856 (856)	—	431 (24)	688 (235)
February 2004	801 (33)	405 (35)	478 (82)	55 (2)
April 2004	1198 (116)	1072 (197)	912 (64)	389 (35)
July 2004	2714 (161)	3234 (646)	470 (15)	346 (79)
October 2004	2261 (1448)	790 (171)	605 (308)	552 (186)

Assemblage Structure

Nonmetric multidimensional scaling ordinations revealed distinct groups of assemblages on rock weirs, snags, and the streambed (Figs. 3 and 4). Analyses from 2004 showed substantial overlap in the positions of samples from new and old rock weirs. We selected a two-dimensional solution for both 2003 and 2004 analyses. Minimum stress values

Table 1 Mean (1 SE in parentheses) density and biomass of macroinvertebrates collected from four habitats sampled in the Cache River, Johnson Co., Illinois in 2003–2004

	old weirs	new weirs	snags	streambed
Density (individuals/m^2)				
April 2003	6413 (1187)	—	1296 (1113)	3869 (1673)
July 2003	50959 (6999)	—	16416 (2789)	20195 (3779)
October 2003	45599 (6193)	—	13674 (3667)	30664 (6826)
February 2004	11689 (6822)	5826 (853)	5371 (271)	5829 (1774)
April 2004	10910 (36)	14729 (857)	8152 (2474)	29861 (13092)
July 2004	62374 (16960)	22356 (7064)	14181 (1069)	16286 (3735)
October 2004	22775 (981)	20826 (3609)	15434 (6238)	53585 (3441)
Biomass (mg AFDM/m^2)				
April 2003	685 (175)	—	330 (172)	232 (109)
July 2003	2317 (422)	—	1410 (41)	1269 (274)
October 2003	1942 (872)	—	1070 (262)	1908 (35)
February 2004	1421 (362)	523 (33)	1143 (137)	669 (243)
April 2004	1510 (318)	1417 (188)	1410 (230)	1012 (328)
July 2004	2929 (339)	3266 (657)	1158 (92)	1162 (13)
October 2004	2441 (1495)	1265 (208)	1361 (625)	4528 (520)

Old weirs were constructed prior to our study in 2001 and new weirs were constructed in fall 2003 after the October 2003 samples were collected

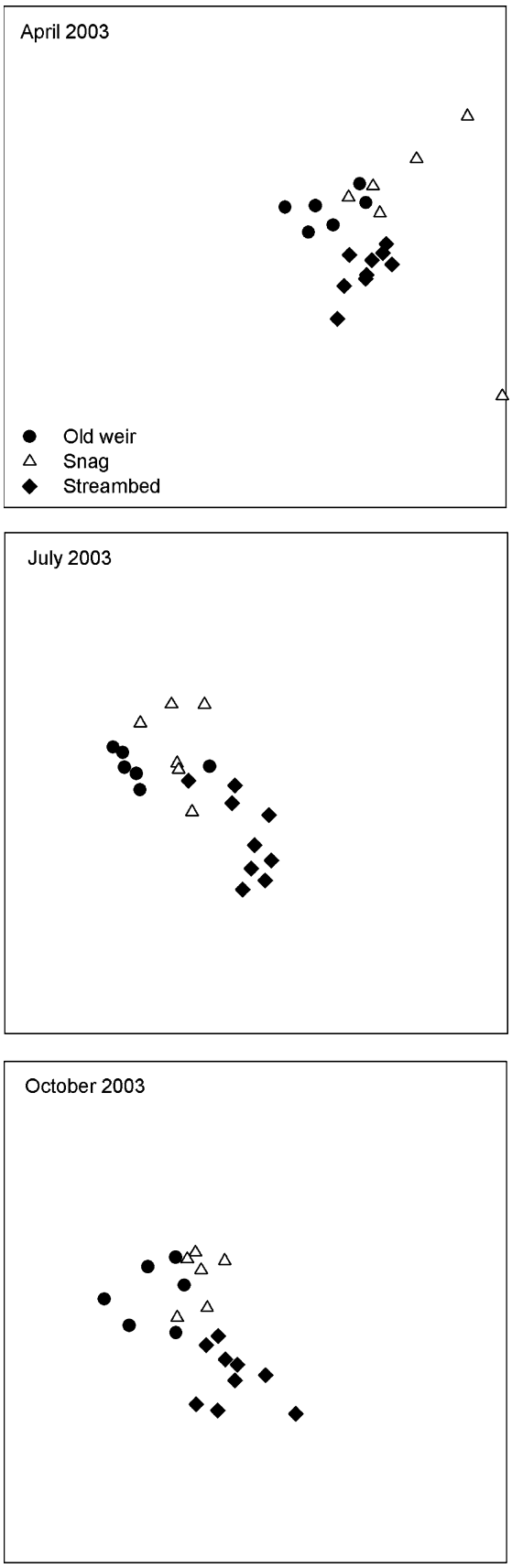

Fig. 3 NMDS ordination based on density of macroinvertebrate assemblages sampled on old rock weirs (constructed in 2001), snags, and streambed substrates in the Cache River, Illinois, during 3 sample intervals in 2003. Each sample interval is presented on a different panel. Minimum stress = 0.15

were 0.15 and 0.22 for 2003 and 2004, respectively. The same solution was obtained for all 100 random initial configurations.

Significant differences among habitats and dates ($p < 0.01$; Table 3) were detected by ANOSIM. Pair-wise comparisons in 2003 indicated that assemblages on old weirs differed significantly from those on the streambed ($R = 0.33$; $p = 0.014$) and were marginally different from those on snags ($R = 0.21$; $p = 0.075$). In 2004, assemblages on new rock weirs were similar to those of old weirs ($R = -0.05$; $p = 0.73$) while all other comparisons revealed significant differences between habitats (R, 0.32–0.51; $p < 0.01$), Strong effects of sample date on assemblage structure were seen in both 2003 and 2004, except for July–October comparisons.

Indicator species analysis showed high specificity of several taxa for each habitat (Table 4). Overall, maximum indicator values (among the 3 groups) for 37 taxa were > 25% ($p < 0.01$). Highest indicator values (>80%; $p = 0.001$) were for the filter-feeders *Simuliium* (blackflies) and *Cheumatopsyche* (Hydropsychidae) on rock weirs and collector-gatherers Oligochaeta and Ostracoda on the streambed. *Macronychus* (Elmidae [riffle beetles]) larvae, *Dubiraphia* (Elmidae) adults, and *Paranyctiophylax* (Polycentropodidae) (IndVal = 70–78%; $p = 0.001$) were the most specific indicators for snags.

Discussion

Invertebrate Diversity, Density, and Biomass

Analyses of macroinvertebrate diversity, density, and biomass among Cache River habitats produced equivocal results. Patterns were either undetectable or inconsistent between sample dates making interpretation difficult. Rock weirs did not have higher taxa richness and did not result in higher diversity at the reach level despite differences in assemblage structure between rock weir, snag, and streambed habitats. Although total macroinvertebrate density was higher on rock weirs over snags and the streambed in 2003, this trend was not consistent through 2004. Biomass did not differ significantly among habitats in 2003 and biomass relationships between habitats varied among sample dates in 2004.

Inconclusive or negative responses of macroinvertebrate diversity or densities have been reported for other in-

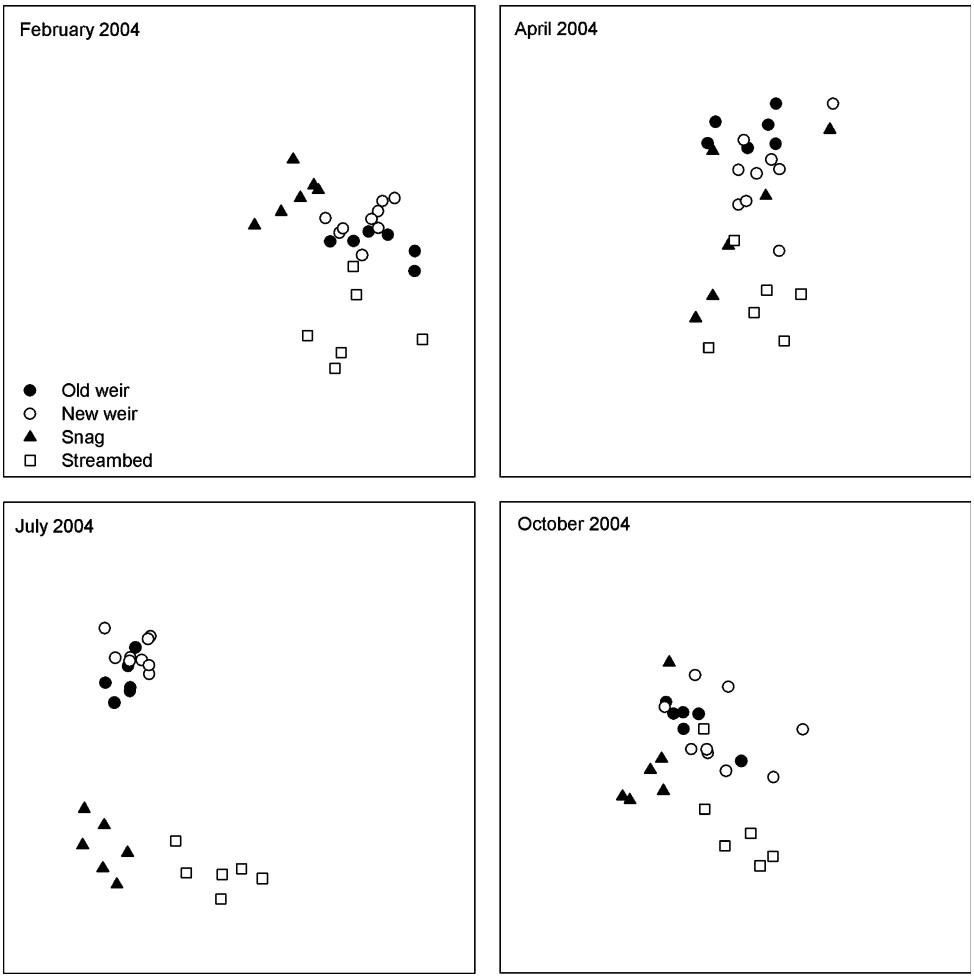

Fig. 4 NMDS ordination based on density of macroinvertebrate assemblages sampled on old rock weirs (constructed 2001), new rock weirs (constructed 2003), snags, and streambed substrates in the Cache River, Illinois, during 4 sample intervals in 2004. Each sample interval is presented on a different panel. Minimum stress = 0.22

Table 3 Results of ANOSIM using Bray-Curtis coefficients calculated from densities (individuals/m^2) of macroinvertebrate taxa

	2003		2004	
	R	*p*	*R*	*p*
Habitat	0.28	0.009	0.32	<0.001
old weir × streambed	0.33	0.014	0.51	<0.001
old weir × snag	0.21	0.075	0.33	0.005
snag × streambed	0.32	0.016	0.49	<0.001
new weir × old weir	–	–	−0.050	0.73
new weir × streambed	–	–	0.48	<0.001
new weir × snag	–	–	0.32	0.005
Sample date	0.62	<0.001	0.60	<0.001
February × April	–	–	0.71	<0.001
February × July	–	–	0.75	<0.001
February × October	–	–	0.84	<0.001
April × July	0.86	<0.001	0.42	0.001
April × October	0.93	0.001	0.74	<0.001
July × October	0.15	0.10	0.28	0.006

Samples were collected from the Cache River, Illinois in April, July, and October of 2003 and in February, April, July, and October of 2004. Data were transformed by log (x + 1)

stream restoration studies. Moerke and others (2004) reported no differences in Shannon diversity among two artificial riffles formed by addition of coarse substrate and an unrestored reach of Juday Creek (Indiana, USA) up to five years after restoration. Macroinvertebrate densities on the artificial riffles in Juday Creek were often greater than densities in the unrestored reach, but were widely variable over the five-year study (Moerke and others 2004). Assessment of headwater stream restorations in northeastern Finland, which included construction of boulder dams and flow deflectors, also showed only moderate macroinvertebrate responses. Total macroinvertebrates recovered to levels comparable to those of unrestored streams but did not approach those of natural streams for 1, 3, 8, and 16-year-old restorations (Laasonen and others 1998). However, older restorations studied by Laasonen and others (1998; 8 and 16-yr) were not as extensive as more recent improvements and left the mid-channel area intact for timber floating, possibly limiting the potential for long-term recovery in these streams. Muotka and Laasonen (2002) likewise found that total invertebrate abundances generally did not increase 3 yr following headwater stream

Table 4 Indicator values (IndVal) of selected taxa collected from the Cache River, IL on rock weir, streambed and snag habitats

	Observed IndVal(% perfect indication)			Monte Carlo tests		
	rock weirs	streambed	snags	mean IndVal	S.D.	p-value
Simulium	**90**	0	1	29.6	6.57	0.001
Cheumatopsyche	**87**	3	6	41.1	8.74	0.001
Hemerodromia	**66**	1	6	35.2	8.17	0.005
Corydalus	**48**	0	3	22.5	7.45	0.008
Baetis	**48**	2	4	26.4	7.48	0.011
Hydropsyche	**47**	0	2	26.9	8.13	0.024
Hydracarina	**44**	17	2	32.7	7.12	0.078
Dugesia	**38**	3	21	29	6.69	0.098
Calanoida	**29**	2	0	16.8	5.95	0.045
Oligochaeta	5	**90**	4	45.7	8.47	0.001
Ostracoda	8	**86**	1	40.2	9.16	0.001
Corbicula	2	**59**	1	24.3	7.63	0.003
Stenelmis (larva)	10	**56**	10	39.9	8.29	0.051
Probezzia	0	**53**	2	18.3	6.22	0.001
Cyclopoida	20	**52**	3	35.5	7.78	0.033
Dubiraphia (larva)	0	**51**	2	15.1	5.71	0.001
Nematoda	0	**50**	2	18	6.77	0.002
Chydoridae	20	**37**	0	26	7.1	0.089
Sphaerium	0	**35**	0	17.6	7.15	0.026
Trichocorixa	0	**33**	0	11.7	5.46	0.003
Caenis	1	**33**	15	22.5	7.33	0.091
Sialis	0	**29**	0	10.3	4.98	0.005
Cernotina	0	**28**	1	16.7	6.6	0.064
Helobdella	0	**24**	0	8.2	4.12	0.004
Macronychus (larva)	0	0	**78**	17.1	6.43	0.001
Paranyctiophylax	0	4	**71**	23.9	7.17	0.001
Dubiraphia (adult)	0	1	**70**	16.2	6.02	0.001
Macronychus (adult)	0	0	**64**	13.3	5.32	0.001
Ancyronyx (larva)	0	0	**55**	14.8	6.27	0.001
Ancyronyx (adult)	0	0	**49**	12.6	5.59	0.001
Stenacron	40	6	**46**	37.2	6.1	0.097
Pycnopsyche	10	2	**37**	22.3	6.98	0.043
Stenelmis (adult)	23	1	**37**	25.7	6.29	0.06
Bezzia/Probezzia	2	12	**34**	22.7	7.21	0.082
Argia	1	2	**33**	15.3	5.56	0.009
Hyalella	1	0	**24**	9.2	4.65	0.013
Atrichopogon	0	0	**21**	6.7	3.88	0.012

Values in bold are those that have a maximum IndVal > 20 with an associated *p*-value of < 0.1 and are grouped by their associated habitat. IndVal's for other habitats are shown for comparison

restorations in northeastern Finland. In their study, scrapers were the only functional invertebrate group that increased significantly.

In contrast to our results, other studies have demonstrated positive effects of in-stream restoration on macroinvertebrate diversity, density, and biomass. Edwards and others (1984) reported higher total macroinvertebrate densities, number of benthic families, and biomass on a series of five artificial riffles and an upstream natural riffle compared to a channelized downstream reference in the Olentangy River in north central Ohio. A study of several rivers in the United Kingdom also showed greater abundance and taxon richness of macroinvertebrates on artificial riffles compared to reference sediments (Harrison and others 2004). Likewise, Ebrahimnezhad and Harper (1997) concluded that the artificial riffles in a

restored reach of a third-order channelized stream in England that achieved hydrologic conditions similar to those of natural riffles increased macroinvertebrate diversity. Muotka and others (2002) reported higher densities of detritivorous invertebrates in 8-yr restoration & natural streams compared to 4 and 6-yr restorations and unrestored channelized streams in northeastern Finland.

Differences in effects on macroinvertebrate densities and diversity among in-stream restoration studies may reflect variability in local and regional macroinvertebrate species pools and the degree of pre-restoration impairment. Harrison and others (2004) concluded that the primary effect of artificial riffles sampled in their study was to promote greater relative abundance of rheophilic taxa already common in other channel habitats because the increase in taxon richness at the reach level was low. We observed a similar response by macroinvertebrates to rock weirs in the Cache River where certain taxa present at relatively low levels and probably habitat-limited before the restoration responded positively to rock weirs, but concurrent increases in overall diversity of the river were not seen. These studies indicate that diversity measures may not be suitable as biological endpoints of river restoration in all cases. Other factors that may influence results of in-stream restoration studies include watershed-level constraints on water quality (e.g., Harper and others 1998) and seasonal shifts in macroinvertebrate assemblages.

EPT and Aquatic Insect Biomass

We found consistent, positive responses to rock weirs for biomass of EPT and aquatic insect taxa, two ecologically important subsets of the Cache River's macroinvertebrate assemblage. Patterns of aquatic insect biomass in our study were generally similar to those of EPT taxa alone, with the greatest biomass found on rock weirs. Because EPT and Diptera are often dominant components of drift (Brittain and Jan Eikeland 1988), we expect that the rock weirs will benefit downstream fish communities, as well as riffle-dwelling fish. Responses of these aquatic insects to restoration efforts in the Cache River could also have important direct and indirect effects on the riparian ecosystem (Baxter and others 2005). Emergence of aerial adult insects results in export of stream resources to riparian food webs. Most (> 80%) emerging adult production does not return to the stream and available studies suggest that a substantial portion of these adults are consumed by riparian vertebrate predators (Huryn and Wallace 2000). We compared potential differences in export of adult insects among the habitats using a rough, but reasonable, estimate of adult insect emergence for habitats sampled in the Cache River in 2004. We estimated aquatic insect secondary production using mean annual biomass (from 2004 seasonal samples), a production to biomass ratio (P/B) of five for univoltine taxa and P/B ratios specific to Big Creek, a tributary of the Cache River, for multivoltine and semivoltine taxa (DAW and MRW unpublished; Walther and others 2006). Secondary production was then multiplied by 0.24 to estimate emergence production based on reviews by Statzner and Resh (1993) and Huryn and Wallace (2000). Resulting estimates of adult insect emergence production were about 2x higher on rock weirs (3.1 and 2.9 g AFDM/m^2 on old and new weirs, respectively) compared to snags (1.3 g AFDM/m^2) and the streambed (1.5 g AFDM/m^2). Although these are rough estimates, they do suggest an important role for rock weirs in subsidizing riparian food webs and possibly influencing the dynamics of reciprocal flows of invertebrate prey between linked river and riparian food webs (Baxter and others 2005). We predict that export from pools impounded by the rock weirs will also increase emergence production (e.g., from increased production of Chironomidae and mayflies in the families Ephemeridae and Caenidae) as organic matter retention improves and sediment accumulates over areas that were formerly dominated by scoured clay substrate.

Assemblage Structure

Structure of macroinvertebrate assemblages differed markedly between dominant substrate types sampled in the Cache River. Separate groups of samples from rock weirs, snags, and the scoured clay streambed evident in NMDS ordinations, as well as significant pair-wise comparisons between habitats detected by ANOSIM, indicated that macroinvertebrate assemblages on weirs were distinct from those of the other two dominant substrate types in the Cache River. Macroinvertebrate assemblages on new and old weirs overlapped in ordination space for each date in 2004 and ANOSIM showed no difference between the two sets of weir samples ($R = -0.05$), indicating that newly constructed weirs are colonized within only a few months by assemblages nearly identical to previously constructed weirs (2 years).

We attribute differences in assemblages on rock weirs compared to the other habitats to increased stability and rough substrate, hydraulic variability, and overall habitat heterogeneity available to macroinvertebrates on the weirs. These factors can exert strong effects on macroinvertebrate distributions. For example, hydraulic variables such as roughness and shear velocity significantly affected macroinvertebrate assemblages in natural riffles studied by Brooks and others (2005). Shear velocity and Shields entrainment function (a measure of substrate mobility) were negatively correlated to macroinvertebrate densities

in a large gravel-bed river for assemblages dominated by taxa that typically avoided flow (Rempel and others 1999). Other taxa showed positive correlations to bed roughness (*Rhithrogena* and *Hydropsyche*), turbulence intensity (*Rhithrogena*), and Reynolds number (*Hydropsyche*; Rempel and others 2000).

Passive filter-feeders *Simulium* and *Cheumatopsyche* were the most specific indicators for rock weir habitats, followed by the predators *Hemerodromia* and *Corydalus*. These taxa are characteristic of high velocity habitats such as those provided by rock weirs. Tolerant collector-gatherers, Oligochaeta, Ostracoda, and *Corbicula*, were most specific to the streambed. *Stenelmis* (Elmidae) larvae were also strong indicators for the streambed. Although *Stenelmis* is considered a scraper, it seems likely that these larvae were functioning as gatherers in streambed habitats because stable substrate that would support growth of biofilm was essentially absent. Elmid beetles in the genera *Dubiraphia* (adults only), *Macronychus*, and *Ancronyx* were strongly indicative of snags. Drury and Kelso (2000) also reported an association of *Macronychus* and *Ancronyx* with wood substrate. The predaceous caddisfly, *Paranyctiophylax*, was also highly indicative of snag habitat.

Conclusions

Past and ongoing restoration efforts in the Cache River represent a unique opportunity to further explore linkages between geomorphological, hydraulic, and biological components of the river ecosystem in the context of an extensive watershed-level project. Our two-year study of the response of macroinvertebrate assemblages to rock weirs in the Cache River demonstrated positive responses by EPT taxa and aquatic insects, while results for more commonly reported measures of response (diversity, density, and biomass) were inconclusive. Differences in overall assemblage structure among the sampled macrohabitats also underscores the strong influence of rock weirs in shaping the river's macroinvertebrate communities. Such changes to assemblage structure are often obscured by broader measures of macroinvertebrate response, but can influence important ecosystem level processes such as secondary production, trophic interactions, and subsidies to riparian food webs.

Acknowledgments This research was funded by the Illinois Conservation 2000 program (C2000) and the Southern Illinois University Department of Zoology. Jordan Rosenfeld and two anonymous reviewers provided comments that improved earlier drafts of this manuscript. We thank J. Beardsley and M. Guetersloh for logistical support throughout the study. S.D. Peterson, J. Rowlett, T. Heatherly, and D. Martin assisted in macroinvertebrate sampling and laboratory processing.

Appendix 1

p-values resulting from pair-wise contrasts of biomass (mg AFDM/m^2) of aquatic insects in the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) and total aquatic insect biomass within each sample month in 2004 from the Cache River, Illinois. Pair-wise tests used the *t*-statistic (not reported). *p*-values less than 0.05 are in bold. Two-way ANOVAs are summarized in the text.

	February	April	July	October
EPT biomass				
new weir, old weir	0.73	0.74	0.95	0.27
new weir, snag	0.46	0.99	**0.0005**	0.17
new weir, streambed	**<0.0001**	**<0.0001**	**<0.0001**	**0.011**
old weir, snag	0.32	0.76	**0.0014**	**0.031**
old weir, streambed	**<0.0001**	**<0.0001**	**<0.0001**	**0.0019**
snag, streambed	**<0.0001**	**<0.0001**	**0.035**	0.22
Total aquatic insect biomass				
new weir, old weir	0.069	0.7	0.72	**0.03**
new weir, snag	0.66	0.71	**<0.0001**	0.32
new weir, streambed	**<0.0001**	**0.012**	**<0.0001**	0.32
old weir, snag	0.19	0.5	**0.0003**	**0.0062**
old weir, streambed	**<0.0001**	**0.0095**	**<0.0001**	**0.006**
snag, streambed	**<0.0001**	**0.042**	**0.021**	**<0.0001**

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DOI: 10.1007/s002670010113

ENVIRONMENTAL AUDITING

Stream Invertebrate Communities, Water Quality, and Land-Use Patterns in an Agricultural Drainage Basin of Northeastern Nebraska, USA

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ABSTRACT / We used invertebrate bioassessment, habitat analysis, geographic information system analysis of land use, and water chemistry monitoring to evaluate tributaries of a degraded northeast Nebraska, USA, reservoir. Bimonthly invertebrate collections and monthly water chemistry samples were collected for two years on six stream reaches to identify sources contributing to reservoir degradation and test suitability of standard rapid bioassessment methods in this region. A composite biotic index composed of seven commonly used metrics was effective for distinguishing between differentially impacted sites and responded to a variety of disturbances. Individual metrics varied greatly in precision and ability to discriminate between relatively impacted and unimpacted stream reaches. A modified Hilsenhoff index showed the highest precision (reference site CV = 0.08) but was least effective at discriminating among sites. Percent dominance and the EPT (number of Ephemeroptera, Plecoptera, and Trichoptera taxa) metrics were most effective at discriminating between sites and exhibited intermediate precision. A trend of higher biotic integrity during summer was evident, indicating seasonal corrections should differ from other regions. Poor correlations were evident between water chemistry variables and bioassessment results. However, land-use factors, particularly within 18-m riparian zones, were correlated with bioassessment scores. For example, there was a strong negative correlation between percentage of rangeland in 18-m riparian zones and percentage of dominance in streams ($r^2 = 0.90$, $P < 0.01$). Results demonstrate that standard rapid bioassessment methods, with some modifications, are effective for use in this agricultural region of the Great Plains and that riparian land use may be the best predictor of stream biotic integrity.

Use of biota as indicators of environmental quality in aquatic systems has increased greatly in recent years (see recent reviews by Ford 1989, Fausch and others 1990, Rosenberg and Resh 1993, Loeb and Spacie 1994). This increase has resulted partly from realization of the limitations associated with standard chemical monitoring schemes; improvements to and simplification of bioassessment methods [e.g., development of rapid bioassessment methods (Karr 1981, Plafkin and others 1989, Lenat and Barbour 1994)]; and increased awareness of the benefits and relevance of biological data for assessing natural resource quality. Additionally, testing and refinement of individual metrics and formation of comprehensive multimetric indices has resulted in bioassessment methods that can detect a variety of perturbations and, in some cases, can be applied appropriately across a wide array of geographical settings (Karr 1981, Fausch and others 1984, Karr and others 1986, Plafkin and others 1989, Barbour and others 1997).

Invertebrate communities are commonly chosen for aquatic bioassessment investigations. Aquatic invertebrates respond to a variety of perturbations, are present in a wide array of aquatic habitat types, are relatively easy to sample and process, and great progress has been made towards standardized methods of collection and analysis. Numerous studies have demonstrated the effectiveness of invertebrate bioassessment for detection

KEY WORDS: Biological assessment; Rapid bioassessment; Aquatic invertebrates; Riparian zone; Benthic invertebrates; Freshwater; River

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of stream reaches impaired by a variety of point- and nonpoint-source pollutants (Chutter 1969, Berkman and others 1986, Peterson 1991, Lenat 1993, Barton and Farmer 1997). Furthermore, Wallace and others (1996) recently demonstrated that invertebrate bioassessment metrics can reflect changes in ecosystem-level processes such as organic matter processing rates in streams.

Agricultural activities are primary contributors to aquatic ecosystem degradation in the United States (US EPA 1994). The influence of agricultural practices on streams can vary greatly with practice and region, and a variety of physicochemical and structural habitat alterations are often evident in streams draining agricultural areas. Documented impacts include changes in riparian vegetation and stream morphology, sedimentation, nutrient additions, organic enrichment, and pesticide contamination (see Cooper 1993 for recent review). These disturbances are often nonpoint and episodic in nature, making detection of specific problems and sources difficult. Bioassessment is a viable means for identification of impaired stream reaches and sources of degradation, and biologically based data can provide a more relevant and easily interpreted assessment of stream ecosystem health. However, effective bioassessment of streams in a given region requires development of regionally appropriate methods and analysis techniques based on comprehensive knowledge of communities in local systems (e.g., Barton 1996).

Northeastern Nebraska, USA, is a region of intense agricultural activity. Land use in the Willow Creek drainage basin examined during this study, which spans 29,000 ha of two counties in northeast Nebraska, is 62% cropland. Most land that is not cropped is used for cattle grazing (31% range, and pastureland). As a result, a majority of the low-gradient streams characteristic of this region appear impacted to some degree. Because of recent expiration of many Conservation Reserve Program (CRP) contracts in the region, the percentage of land in agricultural production, and thus degradation of streams, will probably increase. Surface water assessment programs in this region are, unfortunately, still in their infancy, and bioassessment methods have not been adequately tested or implemented. In 1983–1988, the Nebraska Department of Environmental Quality (NDEQ) performed a statewide stream survey, which included numerous sites on streams in the northeastern portion of the state, and constructed biotic integrity indices based on fish and invertebrate samples from 350 streams representing a wide array of conditions (NDEQ 1991). Aside from this NDEQ study, we know of no other studies that have addressed bioassessment of northeastern Nebraska streams, and the effectiveness of standard methods in agriculturally impacted streams in this region remains poorly investigated.

Our overall objective during this study was to use standard invertebrate rapid bioassessment methods, based on those outlined by the US EPA (Plafkin and others 1989, Barbour and others 1997), in an attempt to identify stream reaches contributing to water-quality degradation in a northeastern Nebraska reservoir. In doing so, we also addressed the following questions: (1) Are standard invertebrate bioassessment methods effective for discriminating among differentially impacted, low-gradient streams draining an intensely agricultural region of the Northern Plains? (2) Which individual metrics will perform best in this region? (3) Will seasonal patterns of bioassessment scores differ from those of other regions? and (4) What variables will most influence bioassessment results, and thus aquatic ecosystem health, in this agriculturally impacted drainage basin?

Study Area

Willow Creek Reservoir is located in northeast Nebraska, ~3 km southwest of the town of Pierce (Pierce County), and drains approximately 29,000 ha. This 290-ha reservoir was constructed in 1982 and is owned by the Nebraska Lower Elkhorn Natural Resource District. The Nebraska Game and Parks Commission manages the facility primarily for recreation. However, water quality in this facility has dropped in recent years. As a result, nuisance algal blooms and foul odors are occasionally evident, and negative impacts on fisheries resources have been documented (Nebraska Clean Lakes Program 1992).

Willow Creek, the major source to the reservoir, originates on the eastern edge of the Nebraska Sand Hills in Antelope County and flows west into the Glaciated Plains region, where it empties into the Elkhorn River, a tributary of the Platte River. This region of Nebraska was historically tallgrass prairie but is now an agricultural landscape dominated by corn and soybean row crops, with only small remnants of native tallgrass prairie present in the form of managed CRP tracts and rangeland. The topography features gently rolling hills, and elevation ranges from 518 m asl in the western reaches of the drainage to 457 m asl in the east. Soil texture ranges from sandy to very sandy, with mostly loess and sand parent materials (Bentall and others 1971). Climate is typical of the central Great Plains with warm, dry summers (July mean air temperature is 24°C) and cold winters (January mean air temperature is −7°C). Precipitation averages 58–64 cm/yr across

Table 1 Physical characteristics of Willow Creek study streams[a]

Parameter	Study stream reach					
	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Catchment area (ha)	901	9942	34,418	31,828	2145	4343
Discharge (liters/sec)	65.6	53.5	643.0	590.0	43.9	109.8
Substrate composition (%)						
Gravel	0	0	0	0	0	10
Sand	85	75	85	75	65	65
Silt	15	25	15	25	35	25
Temperature (°C)	18	19	19	19	20	19
pH	8.4	8.2	8.3	8.2	8.3	8.4
Conductivity (μs/cm)	491	325	340	356	544	553
Dissolved oxygen (mg/liter)	9.5	9.9	9.1	9.5	9.0	9.4

[a]Discharge, temperature, pH, and conductivity values are averages of monthly midday values measured May–September 1996 and April–October 1997. Site 4 served as a reference for bioassessments.

the drainage basin, with ~75% falling in spring–early summer (Bentall and others 1971).

Streams in the Willow Creek basin are typical low-gradient systems of the region that are fed by calcareous groundwater and dominated by shifting sand substrates (Table 1). Occasional debris dams, undercut banks, root mats, gravel bars, algal mats, and macrophyte beds provide some stable substrate. Because larger substrate particle sizes are scarce, riffle habitats are rare and runs and pools dominate in-stream habitat. Riparian vegetation along undisturbed streams in this region is typically a mixture of grasses, forbs, and shrubs, with scattered willow (*Salix* sp.), cottonwood (*Populus deltoides*), and sycamore (*Platanus occidentalis*) trees. Thus, canopy cover is generally minimal and these systems are primarily autotrophic.

Six study reaches, 50 or 100 m in length (sites 1–6, Figure 1), on tributaries to the reservoir in the Willow Creek drainage basin were established for water chemistry monitoring and invertebrate bioassessment during this study. Fifty-meter study reaches were established on two of the smaller streams (sites 2 and 5), and 100-m study reaches were established on others. Although tributaries to the reservoir represent a range of stream sizes, streams are otherwise similar because substrate and in-stream habitats are quite homogenous across different orders of low gradient streams in this area (Table 1). Because preliminary examinations by us and the NDEQ (1991) showed that site 4 had fewer apparent anthropogenic disturbances, an invertebrate community typical for the region, and regionally representative habitat quality and water chemistry, this reach of Willow Creek proper was designated as a reference for comparison during this study (Figure 1). Locating a reference site within our drainage basin was intended to minimize variability associated with comparisons across larger geographic regions, and our standard for comparison was the best current conditions within the Willow Creek basin.

Sites 1 and 2 are located on streams that drain relatively small catchments on the south side of the reservoir (Figure 1 and Table 1). Site 1 has an unusually dense riparian growth of trees and shrubs for this region, but upper reaches of this stream flow through an extensively cropped area, and a small feedlot is adjacent to the stream. Site 2 is located on a stream that flows through a mixture of rangeland and row crops, and occasionally water is removed from this stream for irrigation during summer dry periods. Thus, a seasonal hydrologic disturbance was evident at site 2 during summers of both study years. Site 3 and the reference site 4 are located on Willow Creek proper, which is bordered primarily by rangeland with some row crops, although the drainage basin for these sites as a whole includes extensive areas of cropland (Figure 1). Sites 5 and 6 are located on the main northern tributary to the reservoir that flows through a heavily cropped region. The upper reach of this stream (site 5) appears physically degraded and is channelized for much of its length, whereas the lower reach (site 6) retains natural channel sinuosity and flows through some parcels of rangeland and pasture. Based on initial examinations of land use, channel morphology, and instream habitat, sites 1 (feedlot drainage) and 5 (channelization and no riparian buffer) were deemed most impacted.

Methods

Water Chemistry

Beginning in May 1996, monthly grab samples were collected at all study reaches during nonfreezing

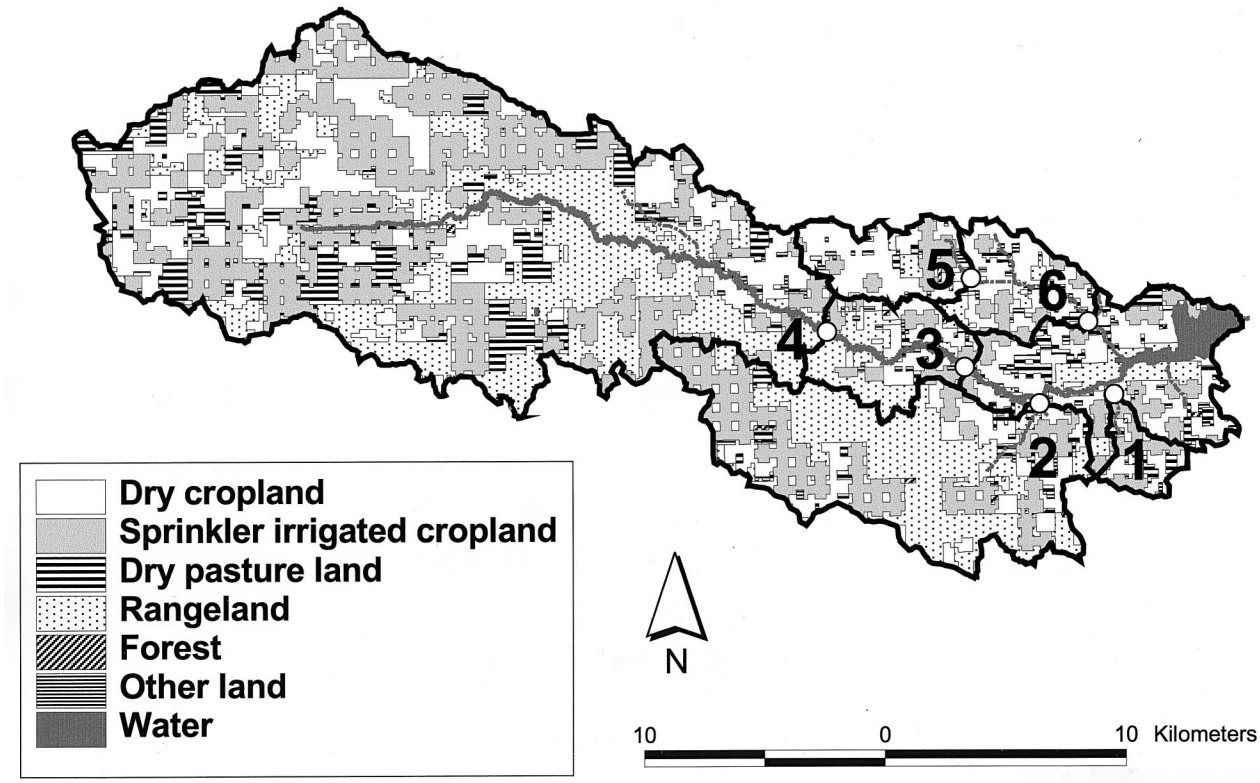

Figure 1. Map of the Willow Creek reservoir drainage basin (Pierce and Antelope counties, Nebraska) showing study site locations on streams (white circles), catchment boundaries (bold black lines), and land use. Willow Creek reservoir is located on the eastern edge of the catchment. Numbers 1–6 correspond to study sites (located at downstream points of each numbered catchment) on streams. For sites that are located on the same stream, land-use analysis for downstream study sites included upstream portions of catchments. Site 4 is located on Willow Creek proper and served as a basin reference. The "other" land use category includes barren and built-up land.

months (May–September 1996 and April–October 1997). Samples were collected from all sites on the same day. Samples were chilled and immediately shipped to E&I Laboratories in Lincoln, Nebraska, where they were analyzed for total nitrogen, nitrate, ammonia, total phosphorus, orthophosphate, chloride, and total suspended solids (TSS). Duplicate and blank (deionized H_2O) samples were included periodically for quality control purposes. During each grab sampling event, standard handheld meters were used to record temperature, dissolved oxygen, pH, and conductivity at each site. Discharge estimates also were obtained on each sampling date by calculating wetted cross-sectional area and measuring current velocity with a handheld flowmeter across three transects (upstream, middle, and downstream points) on each sampling reach. Differences in water chemistry values among stream study reaches were assessed with repeated measures analysis of variance and Tukey's multiple comparison performed on monthly values for each site (N = 12).

Physical Habitat and Land Use

Habitat quality was assessed at each study reach according to protocols for low gradient streams provided by Barbour and others (1997). Assessments included examination of in-stream and riparian features and were performed once each year in August. Numerical scores and overall assessments based on percent comparability to the reference were assigned according to Barbour and others (1997) and Plafkin and others (1989), respectively. Substrate composition was estimated at each site by examining substrate cover (particle size) across 1-m-wide transects at 10-m intervals along the length of each study reach.

Following completion of fieldwork, a geographic information system (GIS) for the Willow Creek drainage basin was constructed. The GIS then was used to calcu-

late catchment size and percent land use for each study stream. Watershed boundaries for each sample site and the entire Willow Creek drainage were derived from 1:24,000 USGS digital elevation model (DEM) data using the hydrologic modeling tools available in ArcView 3.0a with Spatial Analyst (ESRI, Redlands, California, USA). Land-use coverage, developed by the Nebraska Natural Resources Conservation Service (NRCS), for Pierce County, Nebraska, was obtained from the Nebraska Natural Resources Commission Data Bank. No land use coverage was available for Antelope County, Nebraska, so it was developed. The USGS 1:12,000 Digital Ortho Quarter Quadrangles (DOQQs) for the Antelope County portion of the Willow Creek drainage were overlaid with a grid containing 3.2-ha cells. Each cell was assigned a land-use code corresponding to codes used by the NRCS. This method is comparable to the methods used by NRCS to develop the Pierce County land-use coverage. Our method differed mainly in using DOQQ images rather than unrectified aerial photographs. A region of overlap between the two independently derived coverages provided a comparison. Land-use classifications in this area of overlap were virtually identical between the two methods, so the two covers were merged to provide a continuous land-use cover for the entire Willow Creek drainage. For our analyses, land-use classifications were combined into five major categories: cropland (all types of cropland combined), rangeland, forest, built-up land (developed), and other (bare ground and small waterbodies). For rangeland, we combined rangeland (native grassland) and pastureland (managed cool-season grassland).

Stream riparian zone coverages for Willow Creek and its tributaries were created using Arc/Info ver. 7.1.2 (ESRI). Three riparian coverages were generated using the following formulas: 18-m, 50-m × stream width at study reach, and 100-m × stream width at study reach. Riparian widths were meters from each side of the stream. Choices of riparian widths were based on the 18-m undisturbed buffer zone suggested for the highest score on physical habitat analysis in low gradient streams by Barbour and others (1997), and 50× and 100× widths were chosen arbitrarily to examine substantial increases in riparian width analysis. Catchment boundaries for each sample site and the three different widths of riparian covers were overlaid on the land-use cover for calculation of percent area land use for each catchment and riparian zone.

Invertebrate Communities

Composite invertebrate samples were obtained from each study site once every two months during nonfreezing conditions (May, July, September 1996 and April, June, August, October 1997). Samples were collected with a 1-mm mesh kick net. Sampling was performed according to the US EPA's low-gradient stream protocol for rapid bioassessment (Barbour and others 1997), where all habitats present are sampled in proportion to their availability along each 50- to 100-m study reach. This procedure was performed twice on each sampling date, and the two samples were combined and preserved in 8% formalin solution in the field.

In the laboratory, a 100-count random subsample of invertebrates was removed from each sample using a sorting pan with numbered grids and a table of random numbers. For evaluation of this 100-count subsample recommended by Plafkin and others (1989), we analyzed 1996 samples (May, July, and September sampling dates) with a 100-count subsample, 200-count subsample, and no subsample. For this procedure, analyses were performed after the first 100 individuals were randomly removed from the sample; then 100 more individuals were randomly removed, and the analyses were repeated based on the 200 individuals, and finally the whole sample was analyzed. In some cases, no more than 200 total individuals were present in samples, so only 100-count and 200-count procedures could be compared. Invertebrates removed from samples were identified to genus and species when possible. Notable exceptions were Chironomidae (identified to family), and some noninsect taxa such as oligochaetes. All taxa were assigned to functional feeding groups according to Merritt and Cummins (1996). Hilsenhoff's (1987) biotic index was modified by the addition of tolerance values from Lenat (1993) and values derived for the state of Nebraska (NDEQ 1991).

A composite biotic index (CBI) was developed with metrics used previously by the NDEQ during their statewide stream survey (NDEQ 1991). These metrics include those that reflect diversity and community structure [taxa richness and numbers of Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT index)]; community balance [modified Hilsenhoff index (Hilsenhoff 1987), EPT taxa abundance/EPT abundance + Chironomidae abundance, and % dominance]; functional structure (scraper abundance/filterer abundance); and community similarity (Jaccard coefficient comparing each site to the site 4 reference site). Because we did not identify chironomids to species, we did not include the chironomid species number metric also used by the NDEQ. Use of these particular metrics allowed us to correct for stream size (based on discharge) using relationships generated from a prior investigation of 350 streams across Nebraska, including this region (NDEQ 1991). Metric results were scored 1,

Table 2 Average water chemistry values for Willow Creek study streams during 1996–1997[a]

Parameter (mg/liter)	Study stream reach					
	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
PO_4^{-3}	0.07 (0.03)	0.10 (0.02)	0.15 (0.02)	0.12 (0.01)	0.06 (0.04)	0.19 (0.04)
Total P	0.27 (0.13)	0.24 (0.07)	0.21 (0.02)	0.20 (0.02)	0.14 (0.04)	0.27 (0.04)
NO_2^-/NO_3^-	12.71 (1.67)a	5.01 (1.65)b	6.70 (1.02)b	8.58 (1.59)ab	6.48 (0.98)b	9.02 (1.66)ab
NH_4^+	0.13 (0.03)ab	0.23 (0.12)a	0.17 (0.09)ab	0.16 (0.09)ab	0.08 (0.01)b	0.11 (0.01)ab
Total N	14.28 (1.63)a	6.30 (1.46)b	8.21 (1.28)b	9.54 (1.54)b	7.32 (0.90)b	10.10 (1.59)ab
Cl^-	9.61 (1.22)ac	6.46 (0.73)ab	5.22 (0.92)b	5.04 (0.43)b	7.20 (1.37)ab	12.03 (0.74)c
TSS	73.6 (46.7)	24.0 (7.0)	49.9 (11.3)	35.2 (5.4)	33.8 (16.2)	42.3 (10.1)

[a]Values in parenthesis are 1 standard error ($N = 12$ for all). Values followed by different letters are significantly different (repeated measures ANOVA and Tukey's multiple comparison, $P < 0.05$). Site 4 served as a reference for bioassessments.

3, or 5 based on regression equations generated by the NDEQ (1991) that divided scatter plots of stream size vs metric scores into thirds. These values then were summed into a CBI that could range in value from 7 (poorest) to 35 (best). To facilitate direct comparisons of scores in all streams including the reference, the Jaccard coefficient was not included in many analyses; scores without Jaccard could range from 6 (poorest) to 30 (best). Because we used the NDEQ data set and associated stream size corrections to assign values to metric scores, CBI scores actually are based on a "reference condition" for Nebraska rather than the reference stream (site 4) in our basin. However, all comparisons of physical habitat, water chemistry, and bioassessment scores within the drainage were compared to site 4 as a reference within the basin.

All metrics and CBI scores were calculated separately for each sampling date in each stream. Bimonthly CBI values were compared for differences among streams with repeated measures analysis of variance and Tukey's multiple comparison procedure ($N = 7$). The performance of individual metrics for discriminating among sites was examined by comparing scores in the reference site (site 4) and the two sites that appeared most impacted based on preliminary investigations (sites 1 and 5). Metric precision was evaluated by examining coefficients of variation (CV) through time in the reference site. Simple linear correlation was used to examine relationships between average bioassessment scores for streams and physical, chemical, and land-use variables. Variables that were percentages and not normally distributed were arcsine transformed prior to analysis (Zar 1996).

Results

Water Chemistry

Basic water quality parameters were generally similar among sites, although conductivity values were somewhat higher in sites 1, 5, and 6 than in the others (Table 1). No significant differences in orthophosphate or total P concentrations were evident between sites (Table 2). Significant differences were evident for nitrate, ammonia, and total N concentrations, indicating potential N enrichment in at least one study reach. Average concentrations of nitrate and total N were significantly higher in site 1 than in most other sites ($P < 0.05$) (Table 2), and peaked at 22.0 and 23.0 mg/liter, respectively in October, 1997. Site 6 also had somewhat elevated nitrate and total N levels, but no significant differences from other sites were evident. In contrast, average ammonia concentrations were highest in site 2 (0.23 mg/liter), but few significant differences were detected among sites. Chloride concentrations showed a similar pattern to nitrate and total N, with highest average concentrations at sites 1 and 6. Total suspended solids were also highest at site 1, but variability was high and no significant differences were detected among sites (Table 2). The reference site (site 4) was unique only in having the lowest average concentrations of chloride and total suspended solids during this study.

Physical Habitat Analysis

Although we used protocols for low-gradient streams, all streams scored somewhat low on habitat analysis (highest score = 138 of 200). This was likely a result of the generally homogenous nature of these sandy-bottomed channels, the lack of woody vegetation or brush in riparian zones, and at least some degree of anthropogenic disturbance at all sites, including the reference. Nonetheless, differences among sites were evident (Table 3). In sharp contrast to water chemistry results, site 1 scored highest on habitat analysis and exceeded the reference site (site 4) score, primarily due to well-developed growth of riparian trees and shrubs along the study reach, an abundance of woody debris in the channel, and relatively high channel sinuosity. Site

Table 3 Habitat assessment scores, percent of reference score, and habitat condition for Willow Creek streams[a]

Stream reach	Score	Percent of reference	Condition
Site 4 (reference)	130	—	—
Site 1	138	106	comparable
Site 2	113	87	supporting
Site 3	128	98	comparable
Site 5	59	45	nonsupporting
Site 6	111	85	supporting

[a]Scores are 2-year averages of assessments performed during August 1996 and 1997. Low-gradient stream protocols (Barbour and others 1997) were followed to obtain scores, and condition categories are according to Plafkin and others (1989). Site 4 served as a reference for habitat assessment and bioassessment.

3 scored comparable to the reference site, and sites 2 and 6 ranked somewhat lower (Table 3). Only site 5, which was severely impacted by agricultural activities for much of its length and scored only 45% of the reference score, ranked as nonsupporting. This indicated that site 5 was not capable of supporting an acceptable level of biological health (Table 3).

Invertebrate Communities

The average CBI score for the reference site during this study was 22 out of a possible 30 (Jaccard coefficient not included with reference scores). Average CBI scores at sites 1, 2, and 5 were significantly lower than that of the reference site ($P < 0.05$) (Table 4). Despite the highest score on physical habitat analysis, site 1 scored lowest of all sites (15.3 of 35 with Jaccard similarity included), and invertebrate taxa indicative of degraded conditions, such as *Eristalis* sp. (Diptera: Syrphidae), were occasionally present in samples from site 1. Site 5, which exhibited extensive physical habitat degradation in the form of channelization and row cropping up to the wetted margin in some reaches, scored similar to site 1 (15.6 with Jaccard included). Site 2 scored high during some sample dates (spring of both years), but low scores during periods of dewatering resulted in a low average score for this site. Site 6, which appeared relatively undisturbed except for cattle access points near the study site, scored slightly lower than the reference. Site 3 had the highest average CBI score (29 of 35 possible with Jaccard included), and exceeded the reference site score (Table 4).

Subsample size analysis performed on invertebrate samples from 1996 showed nearly identical CBI scores were obtained with 100-count, 200-count, and no subsampling. Changes in final biotic index scores as a result of examining sequentially greater portions of each sample were evident only for sites 2 and 4 in May, but overall scores in these two sites changed by only +2 (from 200-count to whole sample) and −2 (from 100-count to 200-count), respectively. For July samples, the CBI score for site 4 increased by +2 (from 100-count to 200-count). No differences were evident with September samples. These results indicated that 100-count subsamples were sufficient for reliable bioassessments in these streams.

Seasonal trends in CBI scores among all sites were similar, with values generally highest in mid to late summer of each year and lowest in spring (Figure 2 and see Appendix). Only site 2, which experienced significant drying and pooling during summer months as a result of dewatering, had lower CBI scores in summer. CBI scores (including Jaccard coefficient) in site 2 dropped from 23 in May to 13 in July during 1996 and from 25 in April to 17 in August during 1997 (Appendix).

Of metrics used in the CBI during this study, percent dominance showed the greatest discrimination between the reference site and the sites where water chemistry analyses and habitat assessments indicated impairment (sites 1 and 5) (Table 5). Two other metrics (EPT and EPT/EPT + chironomids) were >50% different between impaired sites and the reference. The scraper/filterer metric showed high discrimination (65%) between site 1 (average = 1.37) and the reference (average = 3.9), but values between site 5 (average = 3.35) and the reference differed by only 14% (Table 5). Although not as effective for discrimination between the reference and known impaired sites, the Hilsenhoff index and taxa richness metrics displayed highest precision (lowest CV in reference site) over the course of the study (Table 5). Percent dominance and the EPT index, which displayed highest discrimination, also showed relatively high precision, whereas the EPT/EPT + chironomids and scraper/filterer metrics were much less precise (CV > 1.0).

Correlations between average CBI scores for each stream site and physical/chemical variables were generally weak. No significant correlations were evident for CBI scores and any water chemistry variables examined. Relationships between CBI scores and physical habitat scores were slightly better, but still nonsignificant. Stronger relationships were evident with land-use variables determined from GIS, particularly within riparian zones. Of the land-use variables examined, percent rangeland and percent row-cropped land in catchments had the highest correlations with average CBI scores. These two land-use types dominated the landscape and were directly proportional to each other in this system. Thus, we chose percent rangeland, which

Table 4 Average composite biotic index scores (±1 standard error) for Willow Creek study stream reaches[a]

Site 1	Site 2	Site 3	Site 4 (reference)	Site 5	Site 6
11.7 (1.0)a	14.3 (1.7)ab	24.0 (1.1)c	22.0 (0.8)cd	12.0 (1.2)a	17.9 (1.8)bd

[a]Scores do not include Jaccard coefficient of similarity to the reference site (site 4). Values are averages of 1996–1997 sample dates ($N = 7$ for all). Values with different letters are significantly different (repeated measures ANOVA and Tukey's multiple comparison, $P < 0.05$). Site 4 is indicated as the reference site.

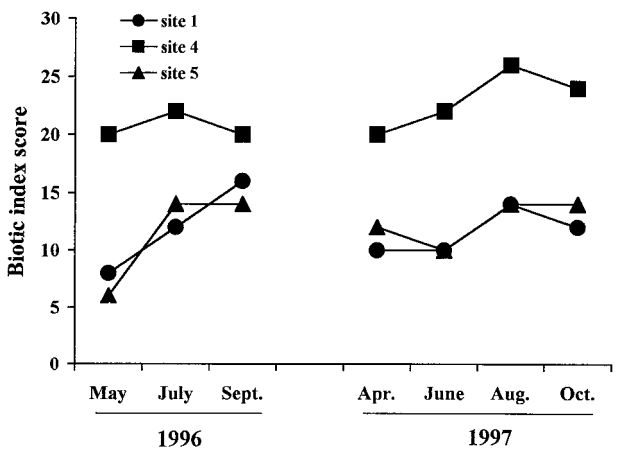

Figure 2. Seasonal patterns of composite biotic index scores in two impacted sites (sites 1 and 5) and the reference site (site 4). Samples were not collected during winter months.

showed slightly stronger correlations with CBI scores, for examination of relationships between bioassessment results and land-use patterns in riparian zones. Correlations between CBI scores and land use increased as riparian zone width analyzed was reduced, and average CBI scores were significantly positively correlated with percent rangeland in 18-m riparian zones of each stream ($r^2 = 0.71$, $P < 0.05$). Examination of individual metrics used during this study revealed a similar trend of highest correlations with land-use variables, particularly within 18-m riparian zones. Of all metrics examined, percent dominance consistently showed the strongest relationships with land-use variables, and the single strongest relationship found was a negative correlation between average percent dominance and percent rangeland in 18-m riparian zones for all streams ($r^2 = 0.91$, $P < 0.01$). (Figure 3).

Discussion

This study demonstrates that the US EPA's Rapid Bioassessment Protocols (Plafkin and others 1989, Barbour and others 1997), with modifications for low-gradient systems, are suitable for at least preliminary identification of relative impacts on stream reaches in this region and appear to be sensitive to a variety of disturbances. For example, low scores obtained from sites 1 and 5 reflect poor biotic integrity in streams with good and very poor physical habitat, respectively. Although physical habitat scores in site 1 were highest of all stream reaches examined during this study, water chemistry monitoring indicated poor conditions that most likely resulted from extensive row cropping and runoff from a small feedlot upstream of this site. Conversely, water chemistry monitoring in site 5 failed to indicate poor conditions, but the extensive physical habitat degradation as a result of channelization and row cropping up to the stream margin clearly affected the biota. Water chemistry monitoring and habitat assessments at site 2 did not reflect significant impairment, but a hydrologic disturbance was evident during summer of both years, and CBI scores dropped during summer months only in this site (see Appendix). Despite different disturbances, CBI scores indicated similar degrees of impairment in sites 1, 2, and 5.

Combined bioassessment, water chemistry, and physical habitat results suggest that site 1 may be the greatest contributor to degraded reservoir water quality, especially when its close proximity to the reservoir is considered. Although sites 2 and 5 had bioassessment scores similar to those of site 1, water chemistry and physical habitat analyses indicated that factors degrading biotic integrity in these sites (hydrologic disturbance and physical habitat degradation) are less likely influencing processes in the reservoir. Results also indicate that site 6, the downstream reach of site 5, also may be contributing to problems in the reservoir; somewhat higher nutrient concentrations and intermediate bioassessment scores reflect some degree of impairment, and this reach is also in close proximity to the reservoir.

Responses of individual metrics to different types of disturbances in our study sites were generally similar, except for the scraper/filterer metric (see Appendix). Despite major differences in stream habitat quality, scraper/filterer values were similar between the reference site and site 5. Despite higher N concentrations, scraper/filterer values were generally lower in site 1

Table 5 Metric discrimination and precision in descending order of performance[a]

Discrimination	Difference (%) from reference: Site 1	Difference (%) from reference: Site 5	Precision	Reference CV
Percent dominance	110	150	Hilsenhoff index	0.08
EPT index	74	73	Taxa richness	0.22
EPT/EPT + chironomids	71	60	% dominance	0.28
Scrapers/filterers	65	14	EPT index	0.31
Taxa richness	38	30	EPT/EPT + chironomids	1.53
Hilsenhoff index	24	32	Scrapers/filterers	1.83

[a]Discrimination is expressed as percent difference between two-year average values in the reference site and the two most impaired sites (sites 1 and 5). Precision is expressed as the coefficient of variation (CV) in the reference site over two years (N = 7 sampling dates).

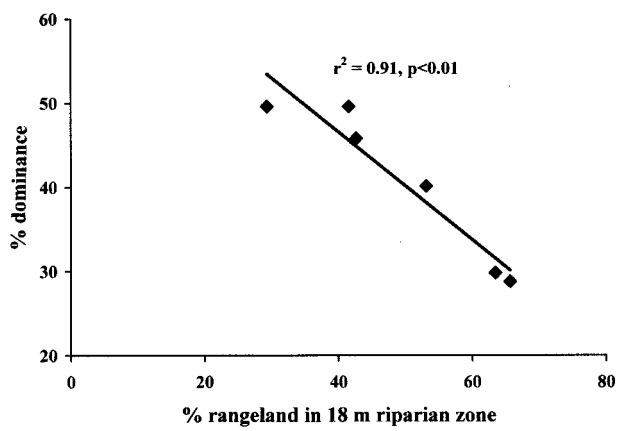

Figure 3. Linear correlation showing the negative relationship between percent rangeland (rangeland + pasture) in 18-m riparian zones of Willow Creek basin study sites and average percent dominance. Values are arcsine-transformed percentages.

than at other sites. This suggests shading from the unusually dense riparian forest at site 1 may have hindered primary production and thus the relative abundances of these functional groups was similar (avg. = 1.4). In contrast, scrapers were generally four to ten times more abundant than filterers in all other sites (see Appendix). Scraper/filterer values in site 2 were extremely variable, but tended to increase greatly following dewatering, resulting in the highest average score for this metric of all sites. Much of this shift in the relative abundance of these two groups was a result of large increases in density of *Physa* sp., a grazing gastropod, during periods of reduced flow. Although some patterns were evident, scraper/filterer values were extremely variable among and within sites. Since both of these functional groups are generally associated with stable substrates, use of this metric may not be appropriate in the sandy-bottomed streams of this region. Barbour and others (1992) also found the scraper/filterer metric to be variable and unreliable, and Hannaford and Resh (1995) reported that this metric did not discriminate site differences among differentially impacted streams in California. Thus, further testing of this metric appears necessary. Our results indicate a simplified metric, such as the relative abundance of *Physa*, may be more appropriate in this region.

Although apparently not appropriate for identification of specific disturbance types in our streams, percent dominance and the EPT metrics performed best for identification of impacted stream reaches. These two metrics showed the highest discrimination among our sites and relatively low variability in the reference system. Lenat and Barbour (1994) reported the EPT index was the single most reliable metric employed by state biologists in North Carolina. Percent dominance and the EPT index are also relatively simple to calculate and require minimal taxonomic skills compared to many others. Other metrics, including the more complex Hilsenhoff biotic index, exhibited either poor discrimination or precision, but none performed poorly in both categories. Thus, none of the metrics examined during this study appeared ineffective, although their performance varied in our system. We examined six physically similar streams subjected to a few disturbance types during this study, and relative performance of these metrics may be expected to vary with region and disturbance regime (e.g., Barbour and others 1992).

Seasonal patterns of CBI scores during this study differed from those observed in other regions. Our results show somewhat higher CBI scores in these streams during summer, except for site 2 while it was drying. In contrast, Lenat (1993) and Hilsenhoff (1988) both observed higher biotic integrity during spring and autumn periods compared to summer, and recommended seasonal correction factors. This discrepancy most likely is related to differences in climate, physical habitat, and trophic basis of the systems. The

growing season in northeastern Nebraska is relatively short, and streams often freeze extensively for long periods from October to April. During spring thaw, when much of the annual precipitation in this region falls, discharge and turbidity are high and these sandy-bottomed streams are often turbid, unstable environments. Lower, more stable flow generally occurs in summer months. Furthermore, streams in northeastern Nebraska are primarily autotrophic. Consumer groups in these systems should be synchronized with primary producer biomass, which appears highest during summer months (personal observation), rather than with autumn-shed leaves, which are not a major energy source for grassland streams. Although we did not sample during winter months when ice and snow covered the streams, our results indicate seasonal correction factors for northeastern Nebraska should differ from those recommended for other regions. Patterns in the two least impacted streams we examined (reference and site 3) suggest that adding 2 points to spring CBI values may be sufficient for standardizing across spring, summer, and autumn investigations (see Appendix).

This examination of six stream reaches in a single drainage basin could not possibly have incorporated all possible disturbances that streams in this region may experience. For example, we did not examine urbanization because it was not a factor in this basin. Although this may limit the conclusions and geographical relevance of our results, the strength of our study is that all streams were part of the same drainage basin, so baseline differences in physical/chemical features should be negligible. This study also represents an intensive effort on a few stream reaches, as opposed to a reduced sampling effort on numerous stream reaches, and each approach has inherent benefits. Repeated sampling of streams during this study allowed for elucidation of seasonal patterns, within-stream metric variability, and response of metrics to a disturbance that occurred only seasonally in one site (dewatering in site 2). Further, perennial streams in this region are not as numerous as in many other areas of the country, and our efforts covered the majority of perennial streams in a large drainage basin representative of the region as a whole.

Because our study sites included streams of different sizes and the small streams in this basin were more highly disturbed than the larger ones we examined, an apparent relationship between stream size and CBI scores was evident. In general, smaller streams had lower CBI scores. However, our analyses and assignment of metric scores incorporated stream size correction factors based on scatter plots of metrics vs stream size for 350 streams across the state of Nebraska (NDEQ 1991). Relationships generated by the NDEQ showed very little or no change for two of the metrics we used (Hilsenhoff index, percent dominance) among streams in the range of sizes we examined. Other metrics we included (taxa richness, scraper/filterer) are expected to decrease with increasing stream size in this region, based on NDEQ results. One of the smaller streams we examined, site 2, scored relatively high in spring periods (tied for highest CBI score in April, 1997) before summer dewatering degraded it. Thus, we are confident that bioassessment results reflect a disturbance regime rather than stream size differences.

Results of this study indicate riparian land use exerted the greatest influence on biotic integrity, that increasing amounts of rangeland resulted in higher biotic integrity, and that riparian land use may be the best predictor of stream health in this region. These results are consistent with studies from other regions that have also demonstrated strong links between stream communities and land-use patterns (Sallenave and Day 1991, Davies and Nelson 1994, Barton and Farmer 1997, Quinn and others 1997). Land use, particularly in riparian zones, influences instream habitat (Sweeney 1993, Quinn and others 1997), water quality (Cooper 1990, Gregory and others 1991, Jordan and others 1993), organic matter inputs (Sweeney 1993), and thus stream communities and biotic integrity. Although the majority of studies to date have focused on forested riparian zones, our results indicate similar relationships exist for prairie streams draining an agricultural landscape and that riparian zones should be a focal point for stream management in this prairie region.

Although this study indicates stream biotic integrity is heavily influenced by land use within a relatively narrow riparian corridor, these results should be interpreted within the context of our study. Land use in the Willow Creek basin is a mosaic of cropland and range/pastureland, and some degree of anthropogenic disturbance is evident in all terrestrial and aquatic habitats in the basin. Thus, as is the case in most regions, none of the streams we examined were undisturbed or pristine. Rather, we studied stream reaches subjected to various types and degrees of disturbance and compared them to a relatively undisturbed reference. We cannot predict how our results might have differed if a wider array of conditions had been present (e.g., pristine through highly degraded) or a more complex landscape were examined. The combination of significant disturbances associated with row cropping [e.g., sedimentation, nutrient and pesticide runoff, loss of riparian vegetation (reviewed by Cooper 1993)] may have overshadowed more subtle stressors in our system. For example,

higher proportions of range- and pastureland in riparian zones resulted in the highest biotic integrity in our streams, despite apparent negative impacts of cattle we observed, including bank erosion and waste inputs. Negative impacts of cattle grazing on stream ecosystems are well documented (Meehan and Platts 1978, Buckhouse and others 1981, Schultz and Leininger 1990, Armour and others 1991, Shimura and Tibuchi 1994, Trimble and Mendel 1995), and our results most likely reflect the relatively greater impacts of row crops on streams in this region, rather than indicating lack of impacts from cattle grazing. Additionally, stream systems in this region evolved under the influence of ungulate grazers (*Bos bison*) and may be more tolerant of their associated impacts than other anthropogenic disturbances.

The Willow Creek basin is representative of a region that is highly impacted by agricultural activity but understudied in terms of surface water bioassessment. This study demonstrates that standard methods applied in other regions of the country are appropriate for initial assessment of impairment, with some modifications including use of regionally specific seasonal correction factors. Our results also indicate landscape-level variables in this region may serve as the most accurate predictors of stream biotic integrity. However, a variety of data, including physical and chemical attributes of a system or better understanding of the environmental requirements of various benthic species, are necessary for identification of specific stressors. Future investigations in other drainages of the region and across a wider array of disturbances will further development of an accurate and comprehensive biologically based monitoring program for this part of the country.

Acknowledgments

This research was supported by funds from the Lower Elkhorn Nebraska National Resource District, NDEQ, US EPA, and Wayne State College. R. Wozniak (Lower Elkhorn Nebraska Natural Resource District) assisted with all aspects of this study and was instrumental in coordinating many of the activities. D. Sutherland and S. Wessel (Nebraska Game and Parks Commission) assisted with study site selection, transportation, and obtaining landowner permission for sampling. D. Alexander, D. Barton, R. Charlton, and an anonymous reviewer provided valuable comments on early drafts of this manuscript. This is contribution 99-176-J of the Kansas Agricultural Experiment Station.

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Appendix. Metric and composite biotic index (CBI) scores for each sampling date in Willow Creek stream reaches, and annual averages of each[a]

Metric and date	Site 1	2	3	4	5	6
Richness						
May 1996	13	12	19	15	11	8
July 1996	8	8	21	15	11	12
Sept 1996	13	8	21	14	10	13
April 1997	10	9	19	15	15	15
June 1997	9	8	17	17	13	15
Aug 1997	11	12	17	24	10	17
Oct 1997	11	14	23	21	15	19
Average	11	10	20	17	12	14
EPT taxa						
May 1996	0	4	12	8	1	3
July 1996	2	0	11	9	2	5
Sept 1996	5	1	9	8	3	5
April 1997	1	5	9	5	3	3
June 1997	1	0	9	8	1	5
Aug 1997	3	1	13	14	2	5
Oct 1997	4	3	13	10	5	5
Average	2	2	11	9	2	4
Hilsenhoff index						
May 1996	6.8	4.7	5.7	5.9	7.1	6.9
July 1996	6.6	4.5	5.3	5.2	7.8	7.3
Sept 1996	7.2	6.9	5.7	5.4	7.6	6.7
April 1997	6.6	5.1	5.0	5.9	7.5	6.4
June 1997	6.8	6.8	6.1	6.4	7.2	6.0
Aug 1997	7.2	6.8	5.2	5.3	7.4	6.3
Oct 1997	7.2	6.5	5.6	5.2	7.0	6.7
Average	6.9	5.9	5.5	5.6	7.4	6.6
EPT/Chironomidae						
May 1996	0.0	0.8	1.0	0.8	0.1	0.8
July 1996	0.8	0.0	0.8	0.9	0.8	0.8
Sept 1996	0.9	0.5	0.7	0.8	0.6	0.9
April 1997	0.0	0.9	0.9	0.9	0.8	0.4
June 1997	0.5	0.0	0.9	1.0	0.8	0.9
Aug 1997	0.9	0.2	0.9	0.9	0.9	0.8
Oct 1997	0.9	0.3	0.7	0.9	0.9	0.8
Average	0.6	0.4	0.8	0.9	0.7	0.8
Percent dominance						
May 1996	65	25	26	30	73	84
July 1996	62	29	18	19	53	58
Sept 1996	48	58	17	29	71	30
April 1997	44	50	36	19	57	41
June 1997	58	71	24	22	40	86
Aug 1997	60	28	36	12	63	30
Oct 1997	65	32	20	28	39	23
Average	57	42	25	23	57	50
Scraper/Filterer						
May 1996	0.03	0.79	34.50	3.38	0.02	0.02
July 1996	0.18	0.00	1.68	0.89	5.17	7.00
Sept 1996	2.36	28.00	1.32	0.57	9.50	20.00
April 1997	0.00	2.43	0.63	0.82	1.75	5.50
June 1997	0.23	0.79	10.25	20.00	0.62	5.92
Aug 1997	6.00	31.50	1.95	0.77	5.75	2.04
Oct 1997	0.79	2.33	0.19	0.90	0.67	0.75
Average	1.37	9.41	7.22	3.90	3.35	5.89

Appendix (*Continued*)

Metric and date	Site 1	2	3	4	5	6
Jaccard coefficient						
May 1996	0.21	0.29	0.36	—	0.24	0.28
July 1996	0.15	0.05	0.35	—	0.13	0.37
Sept 1996	0.29	0.16	0.36	—	0.33	0.29
April 1997	0.32	0.33	0.36	—	0.29	0.50
June 1997	0.30	0.16	0.50	—	0.36	0.41
Aug 1997	0.21	0.26	0.58	—	0.17	0.21
Oct 1997	0.23	0.30	0.42	—	0.33	0.32
Average	0.24	0.22	0.42		0.26	0.34
Composite biotic index						
May 1996	8	18	26	20	6	12
July 1996	12	12	28	22	14	16
Sept 1996	16	14	26	20	14	25
April 1997	10	20	20	20	12	12
June 1997	10	10	22	22	10	20
Aug 1997	14	14	24	26	14	20
Oct 1997	12	12	22	24	14	20
Average	12	14	24	22	12	18

[a]For purposes of comparison among all sites including the reference (site 4), Jaccard coefficient scores are not included in computation of CBI scores. See text for explanation of metric and composite biotic index calculations. Site 4 served as a reference during this study.

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