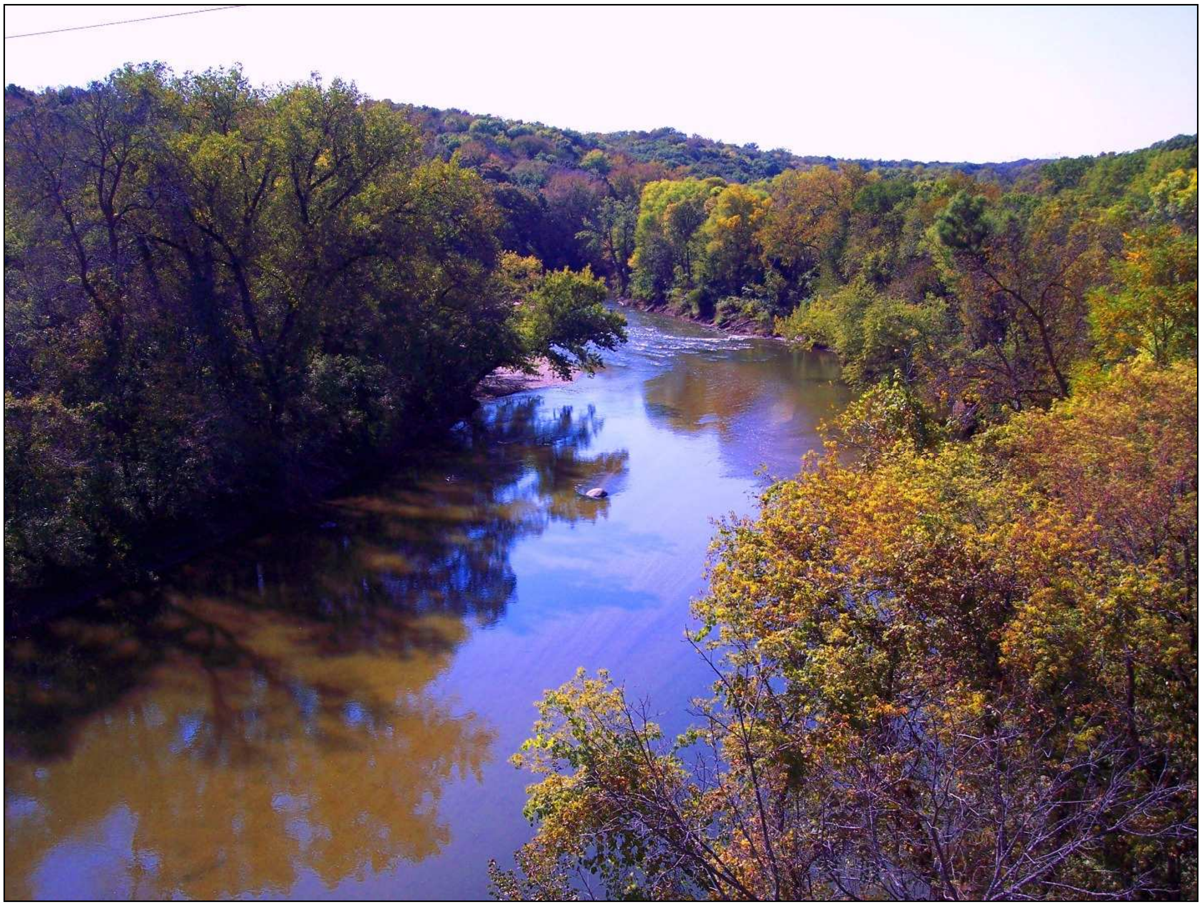
Boone River Watershed
CONSERVATION ACTION PLAN



Narrative

May 31, 2008

Kristen Blann, PhD
The Nature Conservancy
Iowa Chapter

The Nature Conservancy

Protecting nature. Preserving life.™

Boone River Conservation Action Plan (CAP)

Acknowledgments

Rachel Neugarten
David Braun
Inger Lamb
Phil Gassman
Silvia Secchi
Jim Cooper
Todd Sutphin
Anthony Seeman
Sonya Krogh
Anita Maher-Lewis
Tom Isenhart
Jennifer Filipiak
Dave DeGeus
Laura Norian
Jan Glendening
Nick Walters
Boone River Watershed Association
Tom Wilton
Kelly Poole
Keith Schilling
IOWATER

This document was produced by Kristen Blann for the Iowa Chapter of The Nature Conservancy with funding from the U.S. Environmental Protection Agency and Cargill, Inc. to support the development of a comprehensive watershed plan for the Boone River watershed. The document was produced with substantial assistance, input, analysis, and/or data from the individuals and organizations listed above, particularly Rachel Neugarten.

The Boone River watershed partnership seeks to improve the environmental performance of agriculture in the watershed in ways that best support both a healthy farming economy and the native biological communities of the watershed.

Table of Contents

Executive Summary **7**
Background and Intent **7**
Introduction and Summary of the CAP process **9**
Status of the watershed **9**
Summary of the Ecological Assessment 13
Boone River Watershed Planning Activities since the Ecological Assessment 16
(1) The NRCS Rapid Watershed Assessment 16
(2) Iowa Soybean Association Water Quality Monitoring 16
(3) The Boone River Soil and Water Assessment Tool (SWAT) Model 17
(4) Update to the Ecological Assessment 18
Water Quality Regime 18
Hydrologic Regime and Connectivity 37
Channel geomorphic regime 45
Physical Habitat and Riparian Condition 46
Biotic Condition 47
Benthic Macroinvertebrate Assemblage Composition 48
Freshwater Mussel Assemblage and Composition 52
Fish and Assemblage Composition and Health 57
Topeka Shiner 63
Aquatic Mammal Population Status 64
Priority Resource Concerns, threat assessment and situation analysis **65**
Threat Assessment 66
Putting it All Together 68
Impending and Future Threats 79
Summary of Threats Across Targets 82
Situation Analysis 83
Potential Solutions, Actions & Strategies **86**
Restoration Goals 86
Actions & Strategies 88
Potential actions and practices are elaborated below: 89
On-farm modifications to Drainage Systems 91
Off-farm Landscape and Hydrology Management 93
Increasing the capacity of the landscape to process and remove contaminants 93
Redesign drainage system with fluvial and nutrient cycling processes in mind 94
Urban and suburban nonpoint source control 96
Point source control 96
Strategic Analysis of Alternatives **97**
Targeting 97
Onfarm changes in practices 100
Nutrient Management -- Rate and Timing 100
Conservation Tillage 101
Cropping systems 102
Cover cropping 103
Controlled Drainage / Drainage Water Management 103
Filtered tile intakes 104
Off-farm practices: Landscape restoration 106

Compound channels and 2 stage ditches 107
Buffers 108
Wetlands 110
Recommended Strategic Approaches 120
Implementation Strategy 121
1. Monitoring and Assessment 121
2. Planning & Targeted Implementation 125
3. Communications/Education Plan 126
Literature Cited 128

Tables & Figures

Table 2.1. Summary status of Boone River surface waters as reported by the USEPA National Assessment Database.
Table 2.2. Status of individual assessed water bodies in the Boone River watershed as reported to the USEPA National Assessment Database.
Table 2.4. Mean percent land use within 100m of stream channels by stream order.
Table 2.5. Mussel species status in the Boone watershed in successive surveys (modified from Poole 2005).
Table 2.6. Mussel species richness and CPUE in the Boone River by site.
Table 2.7. Fish species sampled in the Boone, sorted by distribution (number of subwatersheds).and descending frequency of occurrence.
Table 3.1. Priority resource concerns.
Table 3.2. Stresses and Sources of Stresses.
Table 3.3. Summary of Threats Across Targets
Table 4.2. Potential Actions and Practices for Addressing Critical Threats
Table 5.3. Reported effectiveness of riparian buffers for reducing nonpoint-source pollutants (runoff, sediment, nutrients and pesticides).
Table 5.4. Estimates of minimum effectiveness areas for riparian buffers and treatment areas by subwatershed area based on source area (cropland acres) and estimated nutrient loads.
Table 5.5. Anticipated benefits associated with different agricultural management options (multiple sources)**.**

Table 6.1. Summary of strategic approach
Table 6.2. Current status of Boone River planning & assessment activities
Table 6.3. Action steps and measures for assessing their effectiveness
Table 6.4. Recommended monitoring activities.

Figure 1.1. Boone River Watershed, Iowa.
Figure 1.2. The Nature Conservancy's Conservation Action Planning process.
Figure 2.1. 30 year average annual N loads predicted by the SWAT baseline scenario, shown in comparison with statewide load estimates published by Libra and Wolter (2004).
Figure 2.2. Mean and median NO3-N concentrations measured in 2007 ISA monitoring.
Figure 2.3. Percentage of the nitrogen export from interior watersheds delivered to the Gulf of Mexico

Figure 2.4. Distribution of confined animal feeding units (by animal units) overlayed on average nitrate concentrations determined for each 12-digit watershed during the 2007 growing season in the Boone River watershed.
Figure 2.5. Mean and median ortho-phosphate (OP) concentrations measured in 2007 ISA monitoring.
Figure 2.6. Estimated 30-yr avg. annual P load (kg/ha) based on SWAT model baseline scenario (average OP [ug/L] from 2007 ISA monitoring (a) and predicted P loads from Libra and Wolter (2004) (b) for comparison).
Figure 2.8. Sediment rating curve for the Boone River based on IOWATER / STORET data from 1999- 2006 near the outlet of the Boone River.
Figure 2.9. Sediment rating curve based on IOWATER/STORET TSS data in relation to discharge in the Boone (1999-2005).
Figure 2.10. Mean erosion losses predicted by the USLE for UMR watersheds (Gowda et al.).
Figure 2.11. 2007 Weekly ISA monitoring results for *E. coli* at Boone River sample locations.
Figure 2.12. Indicators of Hydrologic Alteration Range of Variability Analysis (RVA) results, comparing 1940-1970 to 1971-2008 discharge records.
Figure 2.13. Indicators of Hydrologic Alteration analysis of monthly median flows, comparing 1940-1970 to 1971-2008 discharge records.
Figure 2.14. Indicators of Hydrologic Alteration Range of Variability Analysis (RVA) for environmental flow parameters, comparing SWAT perennial switchgrass (PRE) to baseline (current conditions) hydrologic scenarios.
Figure 2.15. Indicators of Hydrologic Alteration analysis of monthly median flows, SWAT perennial switchgrass (PRE) to baseline (current conditions) hydrologic scenarios
Figure 2.16. Influence of flow disturbance regime (predictability vs. variability of flow and temperature) on life history adaptation of aquatic organisms.
Figure 2.17. Mean benthic macroinvertebrate IBI scores for Boone River sites (Wilton 2004) by subwatershed.
Figure 2.18. Mean number of intolerant macroinvertebrate taxa collected by volunteers at IOWATER volunteer monitoring sites.
Figure 2.19. Relationship between water quality variables, BM-IBI, and FIBI for sites in the Boone River watershed measured in the REMAP study (Wilton 2004).
Figure 2.20. HQI, F-IBI, and BM-IBI scores for White Fox Creek 1994-2001.
Figure 2.21. Map of 2005 mussel sampling locations.
Figure 2.22. Fish species richness by 12 digit watershed.
Figure 2.23. Date and location of fish kills reported to / on file with IA DNR 1985-2005.
Figure 2.24. Distribution and estimated population density of river otter in Iowa.

Figure 3.1. Conceptual model of hydrology (flow regime) as the driver for all other key ecological attributes in freshwater stream systems.
Figure 3.2. Chain of events due to disturbance.
Figure 3.3. Sources of nitrogen in the Mississippi River Basin and measured nitrogen discharge to the Gulf of Mexico from the basin.
Figure 3.4. Seasonal patterns of precipitation, water use and uptake by crops, and N loss potential.
Figure 3.5. Relative location of the Boone and Blue Earth River Basins and ecoregion boundaries.
Figure 3.6. Ethanol facilities as of 2007 (planned & operational). Source: Renewable Fuels Association, modified from Gassman et al. 2008.
Figure 3.7. Situation Analysis: Identifying places to take action.

Figure 3.8. Pathways of nutrient loss.

Figure 4.1. Controlled Drainage (Drainage Water Management) vs. Conventional Drainage [reprinted from University of Minnesota extension publication]
Figure 4.2. Conventional versus two-stage ditch morphology for a 2 mi^2 drainage area in N.W. Ohio, showing stages of various recurrence interval storms ranging from very frequent (0.2-year recurrence interval) to infrequent (100-year storm)
Figure 4.3. Relationship between storm recurrence interval and bedload potential for alternative channel designs.

Figure 5.1. Croplands with < 0.5-1% slope (potential for cost effective controlled drainage/ drainage water management).
Figure 5.2. Relationship between sediment trapping efficiency (i.e. percent of input load retained by the buffer) and buffer-area ratio developed for conditions on four farms in Nebraska.
Figure 5.3. Wetness index and landscape depressions ("potholes" or "sinks") that may meet wetland siting criteria.
Figure 5.4. Modeled and measured nitrate removal efficiencies from CREP wetlands.

Appendices

Appendix A. Additional Figures & Tables
- **Table A.1.** Updated water quality summary tables from STORET and IOWATER
- **Table A.2.** Suspended sediment effects on stream fishes.
- **Table A.3.** Physical habitat descriptions for sampled locations in the Boone.
- **Map A.1.** Critical habitat for Topeka Shiners in the Boone River watershed.

Appendix B. Boone River Ecological Assessment and Appendices, Rachel Neugarten and David Braun, 2005.

Appendix C. Boone River Rapid Watershed Assessment, NRCS, 2008.

Appendix D. Assessments of Practices to Reduce Nitrogen and Phosphorus Nonpoint Source Pollution of Iowa's Surface Waters. USDA-ARS National Soil Tilth Laboratory, for the Iowa Department of Natural Resources, Dana Dinnes, 2005.

Executive Summary

The Boone River conservation action plan (CAP) is intended to provide guidance for the sustainable future of the Boone River watershed. The plan builds upon the ecological assessment undertaken from 2003 -2006 by The Nature Conservancy (included as Appendix B). The Boone River CAP package consists of an ecological assessment, a monitoring plan for expanded/continued watershed assessment, and a strategic analysis of potential alternative actions that could be implemented in the Boone to meet the goals identified in the viability assessment. These actions include partnership work, land acquisition and easements for upland, wetland, and riparian buffer restoration in targeted areas, and best management practices (BMPs), as well as support for education and outreach programs for landowners in targeted portions of the watershed. The Boone River Watershed Association and the project led by the Prairie Rivers RC&D can play a critical role in involving and educating local farmers in these practices.

Many of the key ecological attributes in the Boone—from water quality to macroinvertebrate and fish index of biotic integrity scores (IBIs) to the condition of mussel beds and communities—either do not score well on standard indicators for environmental quality (Wilton 2004; Neugarten and Braun 2005; Krogh et al. 2008), or show signs of significant degradation based on best available data and weight of evidence (Neugarten and Braun 2005; Poole 2005).

The condition of agricultural watersheds such as the Boone River is often related to the complex interaction of altered hydrology and nutrient regimes (Poff et al. 1997, Arbuckle and Downing 2000). Successful restoration of aquatic biodiversity and water quality in the Boone River, as in other agricultural watersheds, depends not only on what is done instream and in the riparian corridor, but also requires improved management and restoration of upland watershed hydrology.

This document, describing a strategic analysis of action alternatives and recommended actions to pursue, is a companion piece to the Boone River Rapid Watershed Assessment, produced in 2008 by Sonya Krogh, Tom Isenhart, and others for the Natural Resources Conservation Service; the Boone River Ecological Assessment produced in 2005 by Rachel Neugarten and David Braun of The Nature Conservancy; and the Excel spreadsheet planning tool developed by The Nature Conservancy for developing landscape level conservation plans. The narrative first discusses the current status and work done to date, then goes into threats, recommended solutions, etc

Background and Intent

The mission of the Nature Conservancy is to preserve the plants, animals, and natural communities that represent the diversity of life on Earth by protecting the lands and waters they need to survive. Thus, the specific interest of TNC as a stakeholder in the Boone River watershed activities is to maintain, restore, and sustain terrestrial and aquatic biodiversity in the Boone River watershed, to the maximum extent practical and possible.

The goals proposed for the Boone River watershed in this CAP include:

- Restoration and maintenance of biodiversity in the Boone River watershed, with a focus on aquatic systems
- Restoration of water quality and aquatic habitat to fully support aquatic life, human health and recreation.
- Development of capacity for the sustainable adaptive management of the Boone River watershed into the future, robust to social, economic, climatic, and other large-scale changes

Specific objectives articulated in this plan include the intent to:

- Establish baseline data and long-term monitoring capacity
- Develop capacity to fully and regularly assess the status of all ten key ecological attributes for both the Upper and Lower watershed zones
- Restore conditions such that all ten key ecological attributes can be rated as good or better in both zones.
- Reduce loading & concentrations of total N and P to meet designated uses, aquatic life and human health water quality standards
- Develop understanding and restore condition of key ecological attributes & indicators, especially the evidently poor and declining status of freshwater mussels
- Increase residence time of agricultural drainage (surface runoff and subsurface drainage) in the landscape before it enters surface waters; as well as storage and watershed retention of water in general
- Identify and better quantify the nature, severity, and causes of sedimentation and instream bank erosion in the Boone River watershed
- Restore natural hydrology, channel and fluvial processes
- Retain and restore important landscape features, pattern, & processes

Introduction and Summary of the CAP process

Status of the watershed

The Boone River Watershed is a 237,000 ha (~580,000 acres) watershed in north central Iowa. The Boone River itself originates in Hancock County, Iowa and flows nearly 100 miles south before joining the Des Moines River just north of Stratford. The Boone River watershed* incorporates the Boone River itself and numerous tributary streams, including Prairie, Otter, Eagle, Buck, White Fox, and Brewers Creeks, as well as many smaller tributaries and drainage ditches (See Figure 1 and Appendix B and C.)

The entire Boone River watershed encompasses approximately 900 square miles extending over six central Iowa counties. It is located entirely within the Des Moines Lobe, the dominant landform of North-Central Iowa. The Des Moines Lobe is an area of hummocky, poorly drained morainal soils that corresponds to the southernmost extent of the last glacial advance in the Upper Midwest. Des Moines Lobe terrain is young (about 12,000 years since glacial retreat), and consists largely of glacial till deposits in moraines and flat to rolling uplands, clay and peat in depressional "prairie pothole" areas, and sand and gravel deposits in floodplains of rivers and streams.

Corn and soybean production accounts for more than 84% of the land use. Fertilizer and livestock manure applications to cropland are major sources of nutrient loads to surface waters in the watershed. The deep rich soils deposited by the glaciers have proven highly productive as cropland, but to achieve that level of production with existing production systems has required extensive investments in drainage. Much of the landscape is characterized by low relief and poor surface drainage. Soil wetness is a major concern for agricultural production. Hydric soils (indicative of soil saturation on at least a seasonal basis) occupy about 54% of the watershed, and artificial tile drainage has been extensively implemented to lower the water table and allow crops to be grown.

Watersheds draining the Des Moines Lobe today may yield as much water as those draining fractured carbonate bedrock such as that of northeast Iowa's karst country (Schilling and Wolter 2005). For example, in the neighboring South Fork watershed to the east of the Boone (a similarly sized watershed in a similar landscape), about 70% of the stream flow in derives from subsurface drainage (Green et al. 2006, Schilling et al. 2007), with most tile discharge occurring during spring and early summer. These watersheds have been identified as contributing disproportionately to nutrient loads delivered to the Gulf of Mexico, and are therefore a significant focus of efforts to reduce Gulf hypoxia. Thus, water quality in the Boone River is an issue of both local and national significance. As with other tile-drained landscapes of the Corn Belt, nitrate losses are some of the highest in the country, due to leaching of soil nitrogen via subsurface tile drains.

---

* We distinguish between the *Boone River*, which is a single waterway, and the *Boone River Watershed*, which includes the river, its watershed, and the network of tributary streams flowing out of this watershed into the river. Confusingly, neither the town of Boone nor Boone County, Iowa, lies even partially within the watershed.

Although from 1992-2000 row crop acreage declined 5% (from 88% to 83%) (IDNR http://wqm.igsb.uiowa.edu/activities/stream/monthly%20sites/booneriv.htm), recent increases in corn prices associated with the ethanol appear to have reversed, driving increased corn acreage, including an increase of corn-corn rotations.

The lower portion of the river is less suitable for agriculture, due to the more dissected, erodible river valley, and significant sections have been acquired for recreation and conservation purposes. In 1985 the lower 25 miles of the river was designated as a Protected Water Area by the State of Iowa. This section of the river is characterized by relatively good water quality and high fish diversity, and is a popular destination for canoeing and sport fishing. Portions of the Boone River watershed have also been designated critical habitat for a federally endangered fish, the Topeka shiner (*Notropis topeka*), by U.S. Fish and Wildlife Service (USFWS 2004).

NatureServe and The Nature Conservancy have also identified the Boone River and its tributary streams as an aquatic system priority for the conservation of freshwater biological diversity within the Upper Mississippi River Basin overall (Weitzell *et al.* 2003). The Boone River watershed was identified as a priority freshwater biodiversity conservation area based on evidence and expert advice, which indicated that the watershed still supports a relatively un-degraded stream ecosystem despite facing a high likelihood of future degradation (Khoury 2004; Neugarten and Braun 2005). Positive attributes of the Boone River include good sand and riffle habitat, historically rich mussel communities, high aquatic Index of Biotic Integrity (IBI) scores, presence of sensitive aquatic invertebrates, and high native fish diversity. Bald eagles (*Halieaetus leucocephalus*), which are still federally listed, but scheduled for de-listing, have been observed nesting and feeding in the lower Boone River, near Bell's Mill. Blanding's turtles (*Emydoidea blandingii*) have been observed in the area around Big Wall Lake (Buck Creek and White Fox Creek headwaters). Protected areas in the lower watershed also harbor two state threatened wetland species (oval ladies tresses, *Spiranthes ovalis*, and showy lady's slippers, *Cypripedium reginae*) along with two state species of concern (tall cotton grass, *Eriophorum angustifolium* and small white lady's slippers, *Cypripedium candidum*).

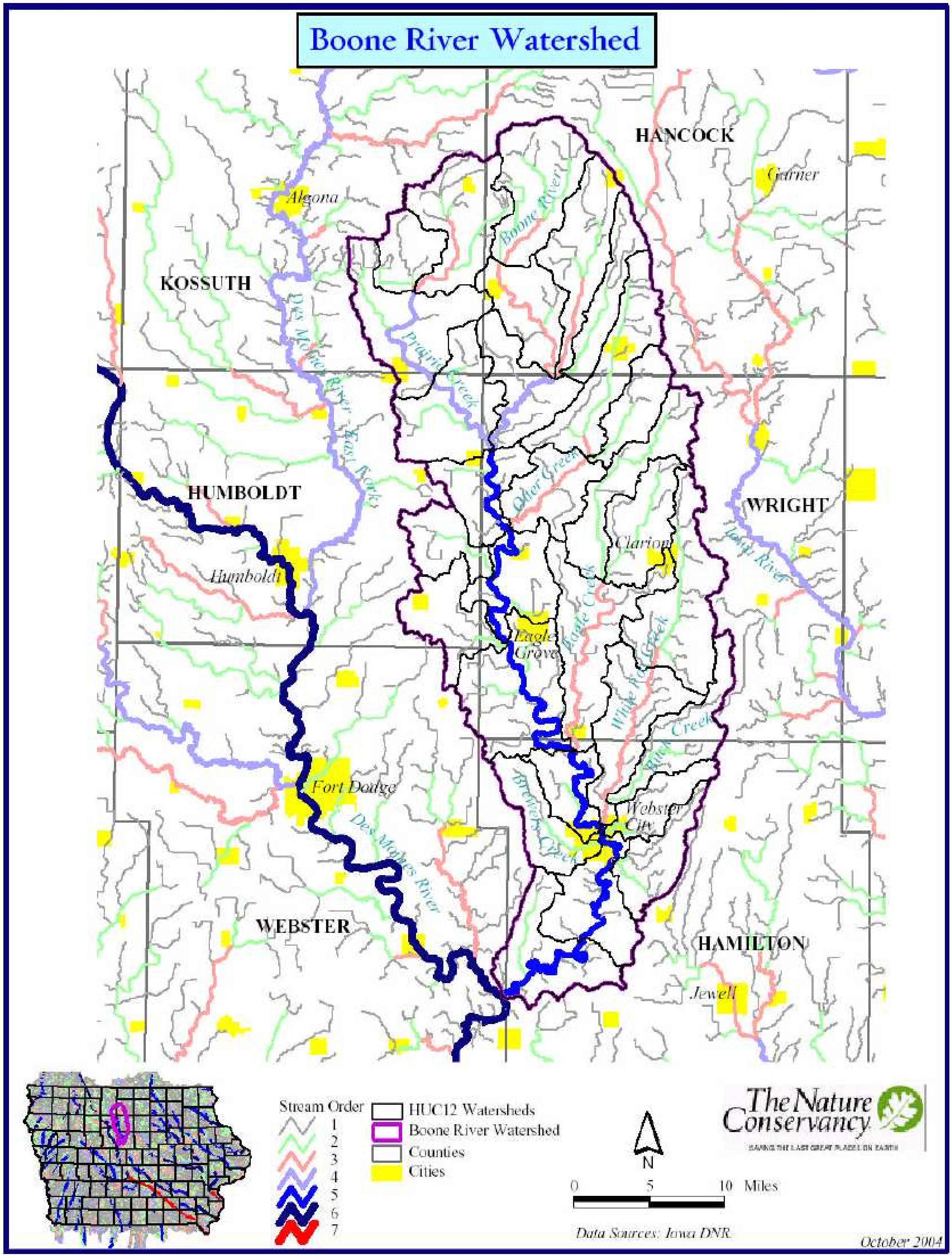


Figure 1.1. Boone River Watershed, Iowa. (Reprinted from Neugarten and Braun 2005.)

In the headwater areas, the Boone River and its tributaries are generally small, shallow streams and ditches draining wide, low-relief valleys with little or no timber (Iowa Conservation Commission 1985, Harlan et al. 1987). The bottom substrate of these streams is a combination of silt and sand, and some streams are artificially straightened and lengthened in their extreme upper reaches (Neugarten and Braun 2005). Fish diversity is relatively low in these smaller creeks and streams, and there are few or no protective riparian buffers along their banks.

Threats to the river ecosystem and its native biodiversity include chronically high nutrient concentrations, nonpoint source contaminants from agricultural operations, and insufficient wastewater treatment (Krogh et al. 2008). As in other watersheds of Iowa, intensive agriculture, urban development, artificial tile drainage, soil erosion, deforestation, channelization of streams and rivers, and an extensive grid of transportation corridors have significantly reshaped the Boone River watershed since the beginning of European settlement.

Prior to settlement by Europeans, much of the north central Iowa landscape was a complex of “pothole” wetlands, with poorly developed stream networks. The headwaters of many streams more resembled grassy swales and interconnected wetlands than true streams. Natural drainage was poor, therefore excess rainfall tended to collect in surface depressions. The Government Land Office historic vegetation survey map of Boone watershed indicates 21,900 acres of swamp, slough, wetland, marsh, or pond was present prior to European settlement and conversion to agriculture. This represents roughly 4% of the watershed.

Conversion of the prairie to farmland has largely involved the draining of wetland and wet soils by means of ditches and buried “tile” lines. Of the 60% of watershed soils classified as poorly or very poorly drained, for example, 93% of these are cropped. Thus many channels and/or drainage ditches today occur in locations where no channelized flow previously occurred. Modern subsurface tile drainage with greater depth and reduce spacing (“pattern tile”) has further accelerated the routing of rainfall water off the land. Currently, less than 0.2% of the watershed is composed of wetlands, according to land use analysis in the RWA (Krogh et al. 2008). However, National Wetland Inventory (NWI) data do indicate more than 5100 acres of seasonally or temporarily inundated forests and shrublands, primarily along the riparian corridor in the lower watershed, that most likely support some wetland vegetation and functions.

The upland landscape has also been substantially altered. Prior to conversion to agriculture, the upland portion of the Boone River watershed was dominated by a prairie community that was maintained by fires (both natural and set by Native Americans). During the late 19th and early 20th centuries, as European settlers converted the prairie to farmland, prairie fires were suppressed on remaining grasslands. Where the fires were suppressed, woodlands grew. Throughout Iowa, < 1% of prairie remains. Today there are fewer than 80 acres of quality prairie remnants within the Boone River watershed, based on inventories by plant ecologists conducted for Iowa DNR. Most records noting high quality prairie remnants, native plant communities, and rare wetland plants are located in the lower watershed in association with protected wetlands or wildlife areas.

Summary of the Ecological Assessment

The Nature Conservancy has developed a planning process for helping conservation practitioners to develop strategies, take action, measure success, and adapt and learn over time. Conservation Action Planning (CAP) involves an iterative sequence of action steps, designed to increase conservation effectiveness and implement adaptive management by refining and improving the linkage between actions and project goals (Figure 1.2).

Figure 1.2. The Nature Conservancy's Conservation Action Planning process.

For the Boone River watershed, this process was initiated with an Ecological Assessment conducted in 2004-5 in collaboration with the Boone River Watershed Project, a multi-partner initiative under the overall coordination of Prairie Rivers of Iowa RC&D, Inc (PRRCD). The assessment provided an overview of the Boone River freshwater ecosystem and the ways in which changes to the landscape over the past 150 years have likely affected this system, and an overview of The Nature Conservancy's conservation planning approach. The report described the characteristics that make the Boone River Watershed a center of both productive agriculture and native aquatic diversity; and in identifying the kinds of improvements in environmental conditions needed to fully support the native freshwater wildlife and habitat of the watershed.

Once the scope of the planning effort has been defined, the planning process developed by TNC involves defining the main species, natural communities or systems that represent the biodiversity of the planning area. These species or systems are called the "conservation targets," or "targets."

Conservation targets identified in the UMRB assessment for the Boone included small river and perennial headwater creek systems, as well as two aquatic species targets: Plain pocketbook mussel (*Lampsilis cardium*), and black sandshell mussel (*Ligumia recta)* (Weitzell et al. 2003). The Boone River itself was classified as a small river system, low gradient, with low to moderate gradient, largely intermittent tributaries, in fine ground and end moraine with isolated areas of lake sand and clay. The Boone River headwaters, as well as Eagle Creek, Otter Creek, Prairie Creek and White Fox Creek systems are all described as perennial creek systems, with low to moderate gradient headwaters of mixed intermittency, in fine ground and end moraines, with localized areas of outwash, sand, and alluvium along the main channels. In their lower reaches, these creeks grade into outwash, sand, alluvium, and ultimately outwash, the primary substrate that underlies the mainstem of the Boone River itself. Lyons Creek was classified as a different system type, one where the lower reaches are in outwash and colluvium and connect directly to much larger downstream systems. The UMRB assessment also noted that despite very low natural cover (0.7-2.4%) in the Basin, the mainstem supports quality riffle habitat, good invertebrate and native fish diversity, and historically rich mussel beds.

For the purpose of the ecological assessment, the Boone River watershed was divided into two ecologically distinct zones, representing two distinct systems or "targets" – 1) an Upper Boone River Watershed zone, covering the area of the watershed formerly covered in prairie and drained by smaller streams, and currently dominated primarily by cropland, drainage ditches and small headwater streams, and 2) a Lower Boone River Watershed zone, including the larger streams of the watershed with currently or formerly woody riparian vegetation (stream order of 3 or greater, including Prairie Creek, the Boone River mainstem and the Middle Branch).

<u>Target viability</u>

Ten "key ecological attributes" were deemed to be most significant to maintaining the Boone River and its headwater streams as healthy ecosystems. These include:

1. Freshwater Mussel Assemblage Composition
2. Topeka Shiner (*Notropis topeka*) Population Status
3. Fish Assemblage Composition and Health
4. Benthic Macroinvertebrate (Non-Mussel) Assemblage Composition
5. Riparian Community Vegetative Structure
6. Aquatic Mammal Population Status
7. Hydrologic Regime
8. Water Quality Regime
9. Channel Geomorphic Regime
10. Hydrologic Connectivity

Each of these ten key ecological attributes was assessed for the two watershed zones by bringing together existing sources of information. This information consisted of published literature, published and unpublished datasets, and the knowledge of experts from numerous organizations and agencies including the Iowa Department of Natural Resources, Iowa State University, the University of Iowa, USGS, and others. The information was then integrated using The Nature Conservancy's standard conservation approach, which incorporates data for one or more indicators for each key ecological attribute in order to estimate its acceptable (desired) ecological condition and rate its current status. Information on each of the ten key ecological attributes for the two watershed zones, including an explanation for the selection of each as a "key" ecological attribute, a description and explanation of the selected indicators, an assessment of the ecologically acceptable range of variation for each indicator, an assessment of the current status of these indicators relative to their acceptable ranges of variation, and recommendations for further investigations.

Findings concerning the status of key ecological attributes in the two watershed zones are presented in the table below. Complete documentation and support for these findings and definitions of the rating categories are provided Neugarten and Braun (2005).

Key Ecological Attribute	**Upper Watershed Rating**	**Lower Watershed Rating**
1. Freshwater Mussel Assemblage Composition	**Poor**	
2. Topeka Shiner (*Notropis topeka*) Population Status	**?**	*(probably n/a)*
3. Fish Assemblage Composition and Health	**Fair**	**Fair**
4. Benthic Macroinvertebrate Assemblage Composition	**Fair**	**Fair**
5. Riparian Community Vegetative Structure	**Fair**	**Very Good**
6. Aquatic Mammal Population Status	**? (Fair)**	**? (Fair)**
7. Hydrologic Regime	**? (Poor)**	**Fair**
8. Water Quality Regime	**Fair**	**Fair**
9. Channel Geomorphic Regime	**? (Poor)**	**? (Fair)**
10. Hydrologic Connectivity	**Fair**	**Good**

The Ecological Assessment concluded that the Boone River watershed requires action to address the undesirable (Poor or Fair) status of many of the ten key attributes in the Upper and Lower watershed. Overall, the upper watershed (headwater streams) is in poorer condition than the lower watershed. Nitrate, phosphorus, and sediment routinely exceed water quality criteria. Fish and macroinvertebrate sampling indicate fair to moderate habitat quality and conditions for fish and benthic organisms. However, the status of the freshwater mussels in the watershed appears to be a significant concern. The most recent surveys failed to record any live individuals – juvenile or adult – for several species that were historically present.

Limitations in the available data prevented a full assessment of all ten key ecological attributes for the Boone River watershed. Data were considered insufficient to establish condition ratings for the hydrologic regime, channel geomorphic regime, aquatic mammal population status, and Topeka shiner status in the upper watershed, and for the channel geomorphic regime and aquatic mammal population status in the lower watershed. The Ecological Assessment therefore identified a number of priority research, analysis and data needs.

Boone River Watershed Planning Activities since the Ecological Assessment

(1) The NRCS Rapid Watershed Assessment

The NRCS Rapid Watershed Assessment (RWA) conducted 2006-2008 provides additional information for assessment of Boone River Watershed resources (NRCS 2008; Appendix C). The RWA is a detailed description and inventory of the Boone River watershed, including land, water, biological, and cultural resources. The report is organized in sections beginning with a physical description, including geographic and geologic setting, land ownership, land use/land cover, common resource areas, precipitation, elevation, soils, and landforms. Biological resources described by the RWA include vegetation, fish and wildlife, important habitats, and threatened and endangered species. Water resources of the watershed include groundwater; streams, rivers, and drainage ditches; and wetlands, floodplains, and lakes. Streams and rivers were also summarized according to their designated uses.

The RWA also identifies general threats to resource health in the watershed. Existing data and information on threats and status of land, water, and biological resources were assembled, including inventory of subsurface drainage, impaired waters, water erosion, manure application areas, environmental facilities, major air facilities, biofuel plants, and groundwater, as well as water use withdrawals.

Cultural and economic sections describe demographic census data, social survey, and farm census data, as well as recreation areas and cultural resources. The RWA also identifies ongoing watershed projects and monitoring initiatives in the Boone River watershed, as well as a thorough inventory of conservation practices based on field-by-field surveys, work that was conducted as input for the SWAT model.

(2) Iowa Soybean Association Water Quality Monitoring

Additional baseline water quality data has been identified as a critical need for the Boone River in order to help determine where protection is needed, develop a strategic plan, and ultimately to assess the effectiveness of conservation actions and activities implemented in the watersehd. In April 2007, the Iowa Soybean Association (ISA), on behalf of the newly forming Boone River Watershed Association, with funding from The Nature Conservancy and partnership support from the PRRCD, and other Boone River partners, initiated Phase I of a multi-year watershed monitoring plan. The purpose of Phase I was to develop a comprehensive baseline understanding of the relative nitrogen and microbial contribution of each sub-watershed (HUC12) within the Boone River basin. Biweekly data samples were collected from April through August at 30 sample sites located in each Huc12 subwatershed.

Phase II, scheduled for the summer of 2008, is designed to implement targeted sampling from areas of interest identified via the baseline sampling. Both wet weather sampling (via automated samplers) and grab sampling methods will be employed. In Phase III, monitoring will be implemented at the field level to evaluate the effects/impacts of management change and implementation via a set of microwatershed/paired watershed studies.

(3) The Boone River Soil and Water Assessment Tool (SWAT) Model

A modeling framework using the Soil and Water Assessment Tool (SWAT) model (version 2005) has been constructed at the Center for Agriculture and Rural Development (CARD) to support analyses of alternative management practice and/or cropping system scenarios for the Boone River Watershed (Gassman et al. 2007, 2008). Phase I of this modeling has been completed, providing initial baseline calibration/validation results and a range of initially proposed scenarios. These include baseline SWAT simulation scenario, constructed based on a 2002 Iowa land use data layer, designed to represent current conditions; a set of future scenarios based on a range of nutrient management and nutrient reduction practices; a reference "all perennial cover" scenario, to provide a picture of how the watershed would function under a hypothetical condition of all grassland. A full description of the modeling framework is provided in Gassman et al. (2007) and Gassman (2008).

Scenarios that differ in the types, mixtures, and magnitudes of alternative (including simply improved) land, cover, soil, and drainage management practices that they incorporate (subject to the ability of the SWAT model to adequately portray these practices) are designed as a “sensitivity analysis” to map out the range of potential water quality and hydrologic responses to changes in land use and cropping systems, and to provide a realistic approximation of what changes would be needed to achieve desired water quality and ecological goals. The baseline scenario is designed to represent current conditions. It is also a method for identifying "hot-spots" within the watershed that are predicted by the model to contribute disproportionately to alterations to the hydrologic regime, sediment loads, or nutrient loads. The “all perennial” scenario is designed as a “bounding” scenario representing the magnitude of possible effects expected under maximum land use change. However, it does not technically represent a “presettlement” scenario, in that the surface and subsurface drainage network in the model remain unchanged. Comparing these model predictions to monitoring data--identifying critical source areas for N, P, and sediment, for example--may identify areas in the watershed where different land-use or conservation practices will be most effective and best-suited for reducing impacts—particularly if model predictions are consistent with actual patterns observed from the monitoring data.

To date, several workshops coordinated by the PRRCD and partners have been conducted with stakeholders and experts to review these initial runs and analyses, and to solicit feedback on simulation scenarios. Phase I results show the degree of improvement that is likely to result from a variety of changes in land, cover, and nutrient soil, and drainage management practices.

The workshop participants have met several times to review and discuss the results of the scenario runs. However, additional improvements to the model are needed. In the next phase of the model CARD will assess the cost-effectiveness of the appropriate strategies for achieving any given level of environmental benefit. They will also be modeling the potential for wetland treatment at tile outlets to treat tile drainage water and to achieve desired levels of nitrate reduction and other benefits.

(4) Update to the Ecological Assessment

Several of the recommended next step actions identified in the Ecological Assessment have been completed as part of the NRCS Rapid Watershed Assessment. Others are addressed in this document as part of TNC's ongoing CAP and the activities of the Boone River Watershed Association. These include improved mapping and spatial analysis to identify differences between biological, hydrological, chemical and physical features of the Upper and Lower watershed zones (i.e. the system level conservation targets in this CAP) as well as additional analysis of existing data. Results and updates to the viability assessment are described below for each of the key ecological attributes for both the Lower and the Upper watershed.

Water Quality Regime

Water quality sample locations in the Boone River include 18 STORET surface water sample locations, 23 IOWATER volunteer monitoring sites, and 29 ISA sample locations.

Of the 61% of 225 stream or river miles that have been assessed within the Boone River watershed, 31% have been classified as "Good", 41% as "threatened," and 27% as "impaired", according to the USEPA National Assessment Database online at http://iaspub.epa.gov/tmdl/w305b_report_V4.huc?p_huc=07100005&p_state=IA , which summarizes electronic information submitted by the states to EPA through 2004 (Table 2.1). The 38 miles of stream listed as "impaired" includes 22 miles of White Fox Creek, 7.7 miles of Lyons Creek, 6.6 miles of Otter Creek, and 1.2 miles of Buttermilk Creek (Table 2.2).

Four water bodies – one stream and three lakes – are listed on the Iowa state impaired waters list (303d list), requiring a TMDL[1] plan under the federal Clean Water Act. A biological assessment conducted by the Iowa DNR on Buttermilk Creek in 2006 as part of the EPA's Regional Environmental Monitoring and Assessment Program (REMAP) found low dissolved oxygen (DO) and organic impairments that were attributed to wastewater discharges.

Table 2.1. Summary status of Boone River surface waters as reported by the USEPA National Assessment Database (2004).

LAKES	ACRES	PERCENT
IMPAIRED	1287	71%
THREATENED	522	29%
	1809	

RIVERS	MILES	PERCENT
GOOD	42.6	31%
THREATENED	56.6	41%
IMPAIRED	37.5	27%
ASSESSED MILES	136.8	61%
NOT ASSESSED	88.2	39%
TOTAL MILES	224.8	

Lyons Creek has recently been added to the list for biological impairment, and is scheduled for development of a water quality improvement plan in 2009. For Briggs Woods Lake and Lake Cornelia, nutrient loading is the primary concern for aquatic life, whereas water level management is listed as the primary source of impairment for Big Wall Lake.

[1] TMDL= Total Maximum Daily Load, is a calculation of the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards, and an allocation of that amount to the pollutant's sources; as established by EPA under the Clean Water Act.

Table 2.2. Status of individual assessed water bodies in the Boone River watershed as reported by the USEPA National Assessment Database.

Water Name	Assessment Unit ID	Location	Acres	Water Status
Big Wall Lake: entire wetland	IA_02-IOW-00860-L_0	Wright County, S14,T90N,R24W, 8 mi WSW of Dows.	935	**IMPAIRED**
West Twin Lake: entire wetland	IA_02-IOW-04045-L_0	Hancock County, S30,T94N,R24, 4 mi E of Kanawha.	109	**IMPAIRED**
Lake Cornelia: entire lake	IA_04-UDM-02290-L_0	Wright County, S16,T92N,R24W, at Cornelia.	243	**IMPAIRED**
Elm Lake: entire wetland	IA_02-IOW-00870-L_0	Wright County, S21,T92N,R24W, 1 mi. S of Cornelia.	463	THREATENED
Briggs Woods Lake: entire lake	IA_04-UDM-01880-L_0	Hamilton County, S17,T88N,R25W near Webster City.	59	THREATENED

Water Name	Assessment Unit ID	Location	Miles	Water Status
Lyons Creek: Mouth (At Webster City) To Headwaters	IA_04-UDM-0215_0	mouth (NW 1/4, S6, T88N, R25W, Hamilton Co.) to headwaters in S18, T89N,R24W, Hamilton Co.	7.7	**IMPAIRED**
White Fox Cr	IA_04-UDM-0220_2	from Hamilton/Wright co. line (N line, SS3, T89N, R25W, Hamilton Co.) to confluence with unnamed tributary in E 1/2, SE 1/4, S36, T91N, RR25W, Wright Co.	8.4	**IMPAIRED**
White Fox Cr	IA_04-UDM-0225_0	from confluence with unnamed tributary (E 1/2, SE 1/4, S36, T91N, R25W, Wright Co.) to headwaters in S5, T92N, R24W, Wright Co.	13.5	**IMPAIRED**
Buttermilk Creek	IA_04-UDM-0247_0	mouth (T92N, R26W, Sec 33) to headwaters (T92N, R26W, Sec 34), Wright County	1.2	**IMPAIRED**
West Otter Creek	IA_04-UDM-0253_1	mouth (S31, T93N, R25W, Wright Co.) to the Wright-Hancock county line (north line, S4, T93N, R25W, Wright Co.	6.6	**IMPAIRED**
Boone River	IA_04-UDM-0180_1	mouth (Webster Co.) to Hwy 17 in S18, T88N, R25W, Hamilton Co.	21.2	THREATENED
White Fox Cr	IA_04-UDM-0220_1	mouth (S33, T89N, R25W, Hamilton Co.) to Hamilton/Wright county line at N line, SS3, T89N, R25W, Hamilton Co.	8.9	THREATENED
Eagle Creek	IA_04-UDM-0240_1	mo to DD 9 in S30,T91N,R25W, Wright Co.	13.3	THREATENED

Drainage Ditch 49	IA_04-UDM-0244_0	mouth to headwaters	3.7	THREATENED
Otter Creek	IA_04-UDM-0250_0	mouth (NW 1/4, S28, T92N, R26W, Wright Co.) to confluence with West Otter Cr. in S31, T93N, R25W, Wright Co.	9.5	THREATENED
Boone River	IA_04-UDM-0180_2	from Hwy 17 (S18, T88N, R25W, Hamilton Co.) to confluence with Brewers Cr. at Webster City in SW 1/4, S6, T88N, R25W, Hamilton Co.	3.8	GOOD
Boone River	IA_04-UDM-0190_0	White Fox Cr to Otter Cr (Wright Co.)	38.8	GOOD
Boone River	IA_04-UDM-0180_3	from confluence with Brewers Cr. (SW 1/4, S6, T88N, R25W, Hamilton Co.) to confluence with White Fox Cr. in S33, T89N, R25W, Hamilton Co.	1.1	NOT ASSESSED
Boone River	IA_04-UDM-0200_1	Otter Cr. to M Br Boone R, Wright Co.	19.3	NOT ASSESSED
Boone River	IA_04-UDM-0200_2	M Br Boone R-Wright to DD-10 Hancock Co.	12.5	NOT ASSESSED
Brewers Creek	IA_04-UDM-0210_0	[Formerly Class B(w); assessed general.]	5.0	NOT ASSESSED
Buck Creek	IA_04-UDM-0230_0	mo to DD-144 S11,T89N,R25W Hamilton Co.	4.2	NOT ASSESSED
Eagle Creek	IA_04-UDM-0240_2	DD 9->L Eagle Cr S9,T91N,R25W Wright Co	5.6	NOT ASSESSED
Drainage Ditch 94	IA_04-UDM-0245_0	mouth to W line S3,T90N,R26W Wright Co.	1.1	NOT ASSESSED
West Otter Cr	IA_04-UDM-0253_2	from the Wright-Hancock county line (north line, S4, T93N, R25W, Wright Co.) to headwaters in S35, T95N, R25W, Hancock Co.	8.0	NOT ASSESSED
Prairie Creek	IA_04-UDM-0260_0	mo to DD 116 S24,T94N,R28W Kossuth Co.	16.7	NOT ASSESSED
Middle Branch Boone River	IA_04-UDM-0265_0	mo to trib S31,T95N,R25W Hancock Co.	7.3	NOT ASSESSED

Nitrate

Monitoring data from IOWATER, STORET, and ISA all show that nitrate levels in surface waters of the upper and lower watershed routinely exceed the drinking water standard and greatly exceed levels considered to be fully protective of aquatic life (Camargo et al. 2005). Median concentrations of Nitrite + Nitrate (NO_2+NO_3) for the watershed as a whole continue to exceed the acceptable range of variation for every season and for the annual cycle overall. The annual median value of 5.225 mg/L is roughly 2-3 times greater than the recommended annual median of 1.965 mg/L. The Boone River mainstem exceeds the acceptable range of variation on an annual basis and for all seasons except the fall. Among subwatersheds in the Upper zone with adequate sample sizes, Buttermilk Creek exceeded the acceptable range of variation on an annual basis and for all seasons except winter; Eagle Creek exceeded the standard in all seasons and for the annual cycle as a whole; Drainage Ditch 4, Little Eagle Creek, West Otter Creek, and White Fox Creek exceeded the standard for all seasons for which sufficient samples are available.

In addition, both nitrate and nitrite are present individually in concentrations that routinely exceed state and federal criteria for being considered harmful to human health if consumed in drinking water. Nitrate levels equaled or exceeded the state health criterion of 10 mg/L in 45 of 174 samples (26%) for which it was analyzed separately. Nitrite levels equaled or exceeded the state health criterion of 1 mg/L in 8 of 168 samples (3.6%) for which it was analyzed separately.

Monitoring by Iowa Soybean Association in 2007 at 29 sites throughout the watershed confirmed that nitrate levels routinely exceed drinking water and aquatic life standards, particularly in spring and early summer. Nitrate levels at all 29 sample sites exceeded 10 mg/L drinking water standard in April 2007, and in all but one or two sample locations in May and June. Although levels for most locations were below 10 mg/L by July during the peak crop growth period, all subwatersheds were back above 5 mg/L by the end of August. 5 mg/L is more than twice the level considered protective of sensitive aquatic life. Nitrate levels in the Boone are similar to those in many other streams and rivers of the Des Moines Lobe, and some of the highest in the country, typical of the most intensively cropped, tile–drained watersheds of the Mississippi River Basin (Kalkhoff et al. 2000, Schilling et al. 2007).

Concentrations and per acre loads seem to be most acute in the eastern and uppermost portions of the watershed (Figure 2.2). It is perhaps notable that these are both the watersheds with the lowest percentage of riparian areas in grass or forested vegetation. Subwatersheds in the eastern portion of the basin also have the highest numbers of animals in concentrated animal-feeding operations (AFO), and are hypothesized to receive higher applications of manure as fertilizer (Figure 2.3).

The Boone SWAT model estimated an annual 30 year average load of 6-7,000,000 kg at the outlet of the Boone River under the baseline scenario, or roughly 25-30 kg/ha. The statewide nutrient budget analysis developed by Libra and Wolter (2004) predicted comparable loads in the range of 20-35 lbs/acre (roughly 18-23 kg/ha) for the Boone. However, although nitrate loads from the Boone are some of the highest in Iowa, the Boone accounts for moderate annual loads to the Gulf of Mexico equivalent to just 7.5-12 kg/ha N, after accounting for instream uptake and processing (Figure 2.4; Alexander et al. 2000). Instream processing during the residence and travel time of waters en route to the Gulf of Mexico removes an estimated 50-70% of the N load derived from the Boone. In other words, only 30-50% of N lost from the Boone is ultimately delivered to the Gulf of Mexico.

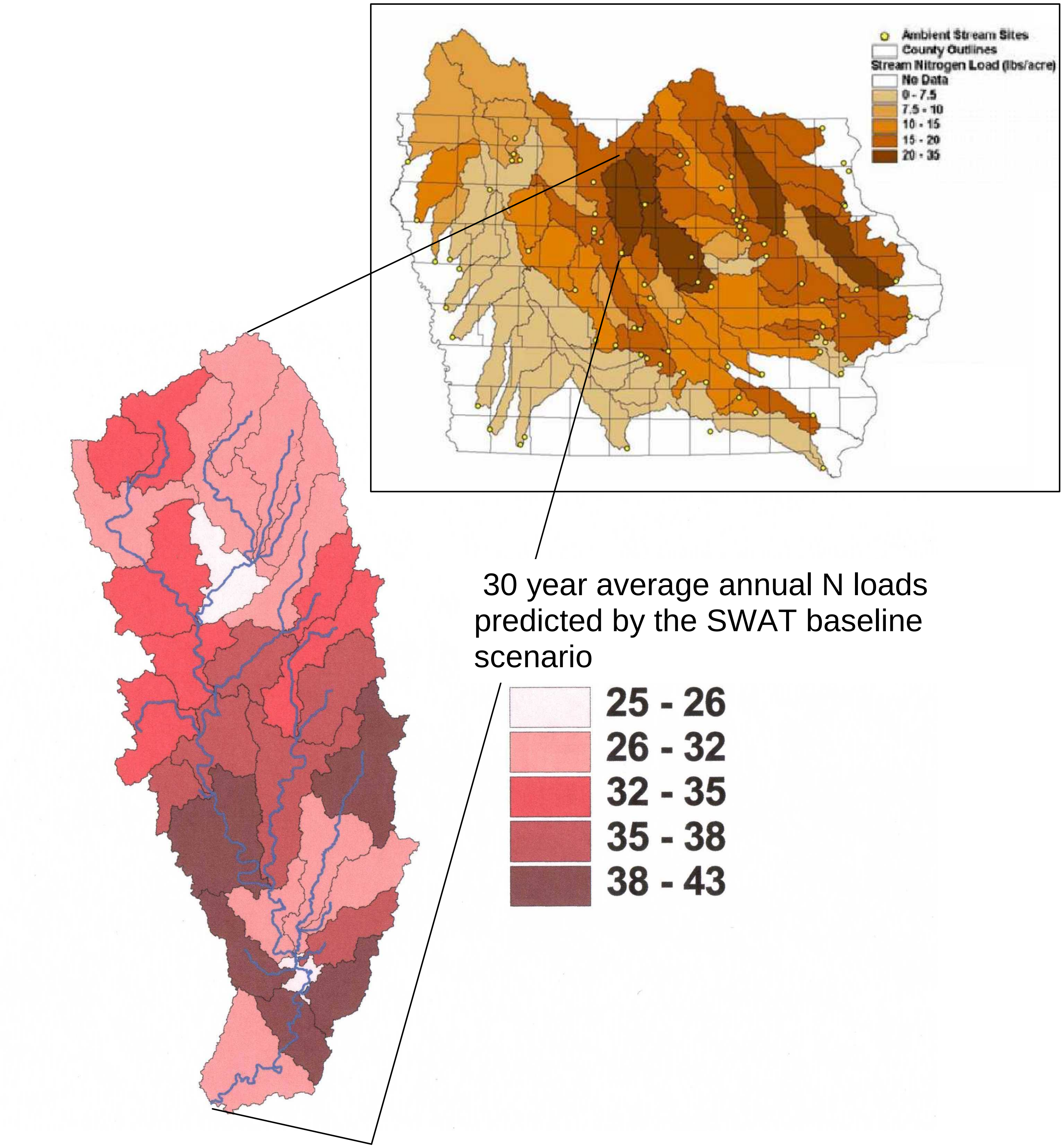


Figure 2.1. 30 year average annual N loads predicted by the SWAT baseline scenario, shown in comparison with statewide load estimates published by Libra and Wolter (2004).

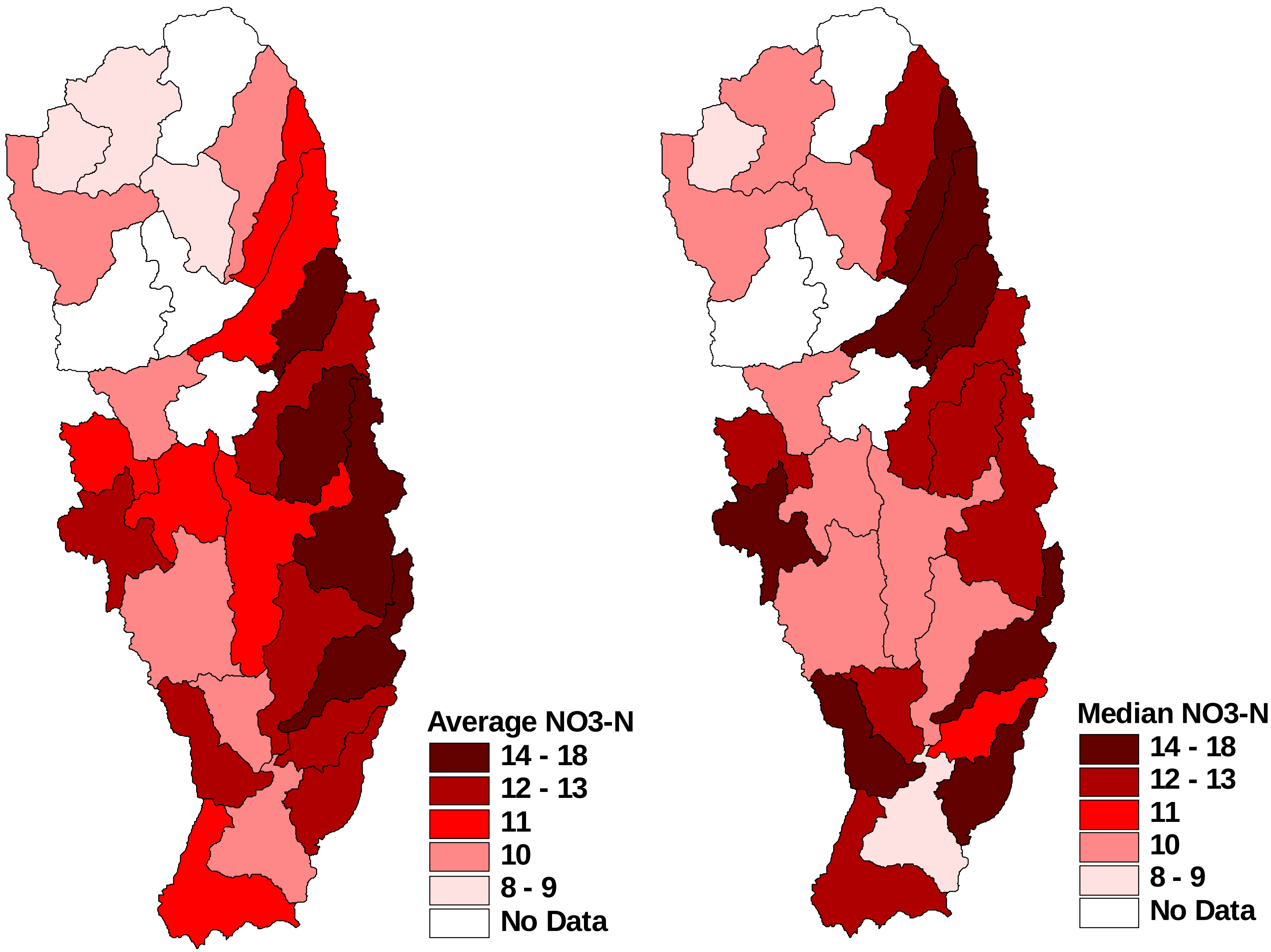


Figure 2.2. Mean and median NO_3-N (mg/L) concentrations measured in 2007 ISA monitoring.

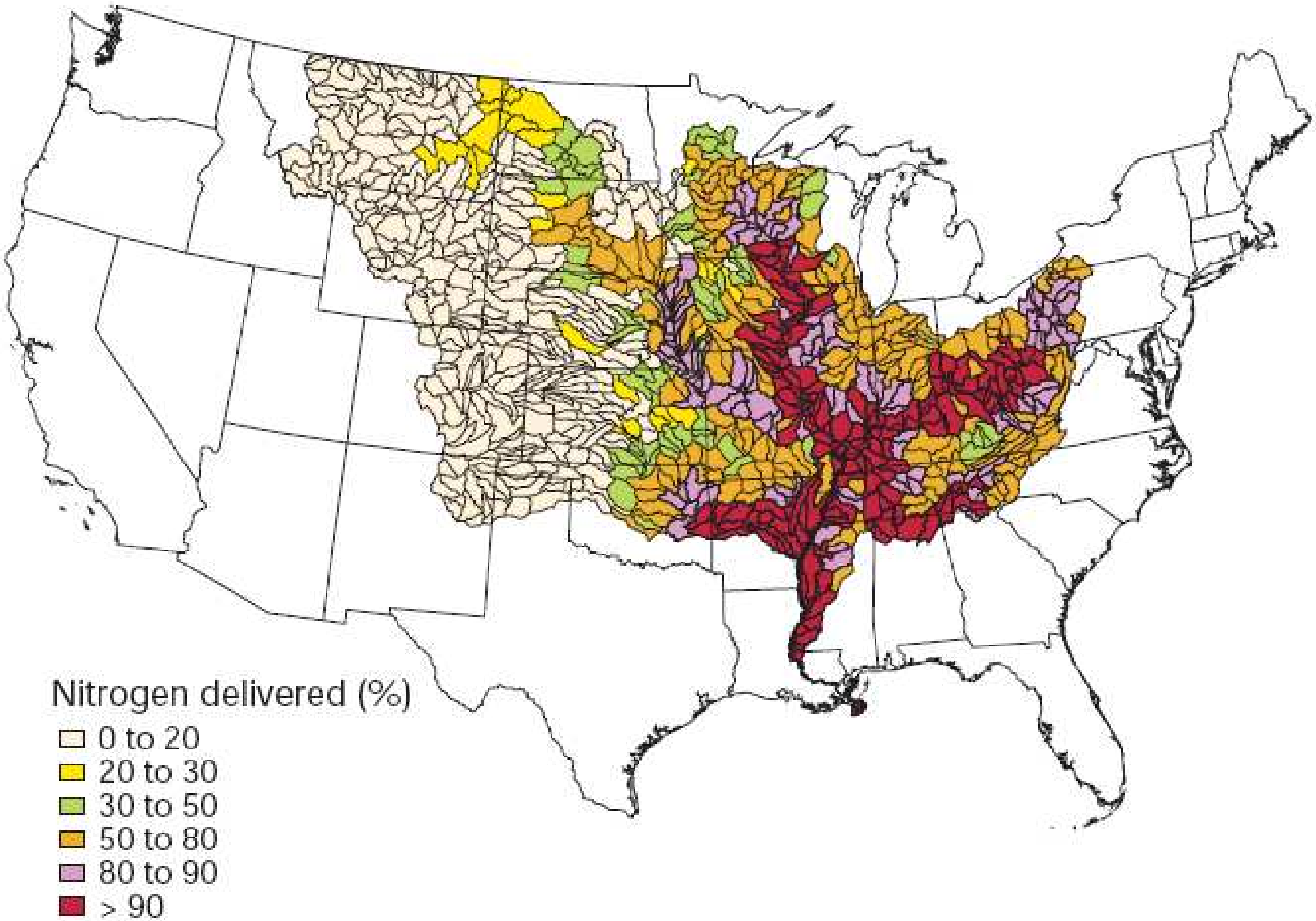


Figure 2.3. Percentage of the nitrogen export from interior watersheds delivered to the Gulf (reprinted from Alexander et al. 2000). The delivery percentage is the fraction of the nitrogen exported from inland watersheds that remains after in-stream transport to the Gulf, and is computed as an estimate of in-stream nitrogen loss for four stream sizes based on mean water travel times from each watershed outlet to the Gulf.

Phosphorus

Although phosphorus (P) is a naturally occurring and necessary nutrient for plant growth, elevated levels of P in aquatic systems derived from anthropogenic sources (e.g. fertilizer, livestock manure, wastewater treatment effluent, etc) can significantly alter aquatic plant and animal communities. Because N as nitrogen is readily available in the environment in forms easily converted to those required for plant growth, P is generally the primary nutrient limiting primary production in surface waters (Daniel et al. 1998), and is therefore the nutrient responsible for driving freshwater eutrophication – i.e. a complex set of trophic changes induced by nutrient enrichment. Common sources of phosphorus in freshwater environments include certain soils and bedrock; human and animal wastes; detergents; decomposing plants; and runoff from fertilized lawns and cropland.

Phosphorus has several fates once it enters the aquatic environment depending upon its form (Dinnes 2005). Particulate P may be deposited with sediments in stream or lake beds where it may either be stored and unavailable (a P "sink"), or dissolve and become available (a "source"), depending upon the physical and chemical properties of the system. Whether sediments serve as a P source or sink varies on annual cycles of flow, depending on the ratio of the concentration of P in sediments relative to that in the water column (Sharpley et al. 2006). Phosphorus may eventually leave a particular waterbody by flow transport, especially during high flow periods, or by deep burial within bed sediments. High flow periods can also add P, continuing the cycle. Dissolved reactive P (also referred to as soluble P) may either be adsorbed by sediments or assimilated by algae as concentrations increase.

Typical concentrations in Iowa streams today are 0.1 to 0.4 mg/L. Although the state of Iowa has not yet developed nutrient criteria for streams, these levels exceed the 0.13 mg/L annual and seasonal criteria established by EPA for TP in streams of the ecoregion. These values are also considerably higher than background reference levels of 0.03-0.07 mg/L TP estimated for natural streams of the Corn Belt Plains ecoregion (Smith et al. 2003). Estimated background levels are based on empirical models developed for each ecoregion in the U.S. to establish reference nutrient levels under pristine (premodern) conditions, prior to agricultural development of the region as well as prior to widespread atmospheric deposition of anthropogenically derived nutrients.

In the Boone River, TP levels measured at the IDNR site near Stratford from 2000-2005 ranged from 0.05-1 mg/L, exceeding the acceptable range of variation for every season and for the annual cycle overall (Neugarten and Braun 2005). Seasonal median TP only slightly exceeded the acceptable range of variation during the spring, but was roughly 1.5 times greater during the summer, twice as great during the fall, and three times greater during the winter. Seasonal maximum TP values exceeded the acceptable range of variation by a factor of 4 to nearly 10.

Some monitoring protocols measure *orthophosphorus* (OP) levels in addition to or rather than TP. TP is strongly adsorbed to sediment, whereas OP is soluble reactive phosphorus, and is therefore readily available for biological uptake. Dissolved orthophosphorus (DOP) in particular tends to stimulate excess algae growth, often leading to subsequent depletion of dissolved oxygen. In 2007, Iowa Soybean Association collected OP readings at the 29 sites outleting each subwatershed in the Boone River watershed. At 25 out of 29 sites, concentrations exceeded the 0.13 mg/L EPA annual standard for *total* P on at least one sample date. Mean, median, and maximum levels of OP across all sample locations were 0.19, 0.16, and 0.57 mg/L, respectively. These results are consistent with average and median levels reported for streams throughout Iowa, suggesting than in the Boone River as in other Iowa streams and rivers, plant-available phosphorus is frequently present at levels sufficient to drive nutrient enrichment instream and influence aquatic biota. IOWATER volunteers frequently describe anecdotally abundant blooms of algae at sample sites where they observed other signs of impairment.

At a subwatershed scale, Figure 2.5 depicts mean and median OP concentrations as measured by ISA in 2007. Schilling et al. (2007) have reported that phosphorus and suspended-sediment concentrations are typically larger in streams that drain the Des Moines Lobe than in other Iowa streams. The median OP concentration of 0.16 mg/L in the Boone River is more than twice what EPA has proposed as the standard for Midwest streams. Concentrations of P in the South Fork watershed, by comparison, had a median of 0.07 mg/L during three years of weekly-biweekly sampling.

Figure 2.6 estimates per acre TP loads based on output from the SWAT model, shown in comparison to loads predicted by Libra and Wolter (2004). The bulk of the phosphorus load is typically delivered to surface waters from a small proportion of the landscape during high flow events. This is discussed further in the section on strategic analysis of actions in the context of targeting.

Groundwater can also be a P contributor to streams. Recent groundwater sampling from 24 wells located throughout the South Fork watershed has shown median and maximum total P concentrations of 0.030 and 0.340 mg/L, respectively. These groundwater P concentrations are found in similar materials and landscapes in Iowa (Burkart et al. 2004).

Chlorophyll-a is another water quality measure that is often measured as a more proximate indicator of nutrient enrichment impacts based on total algal growth. At the IDNR site on the Boone River near Stratford--the only site with consistent sampling for this parameter—median values exceeded the acceptable range of variation for three out of four seasons (fall, winter, and spring) and for the annual cycle overall. The pattern was the same when scattered data from a few other sites in the watershed were added, indicating that chlorophyll-a levels in the watershed were consistently high during non-summer months (Neugarten and Braun 2005).

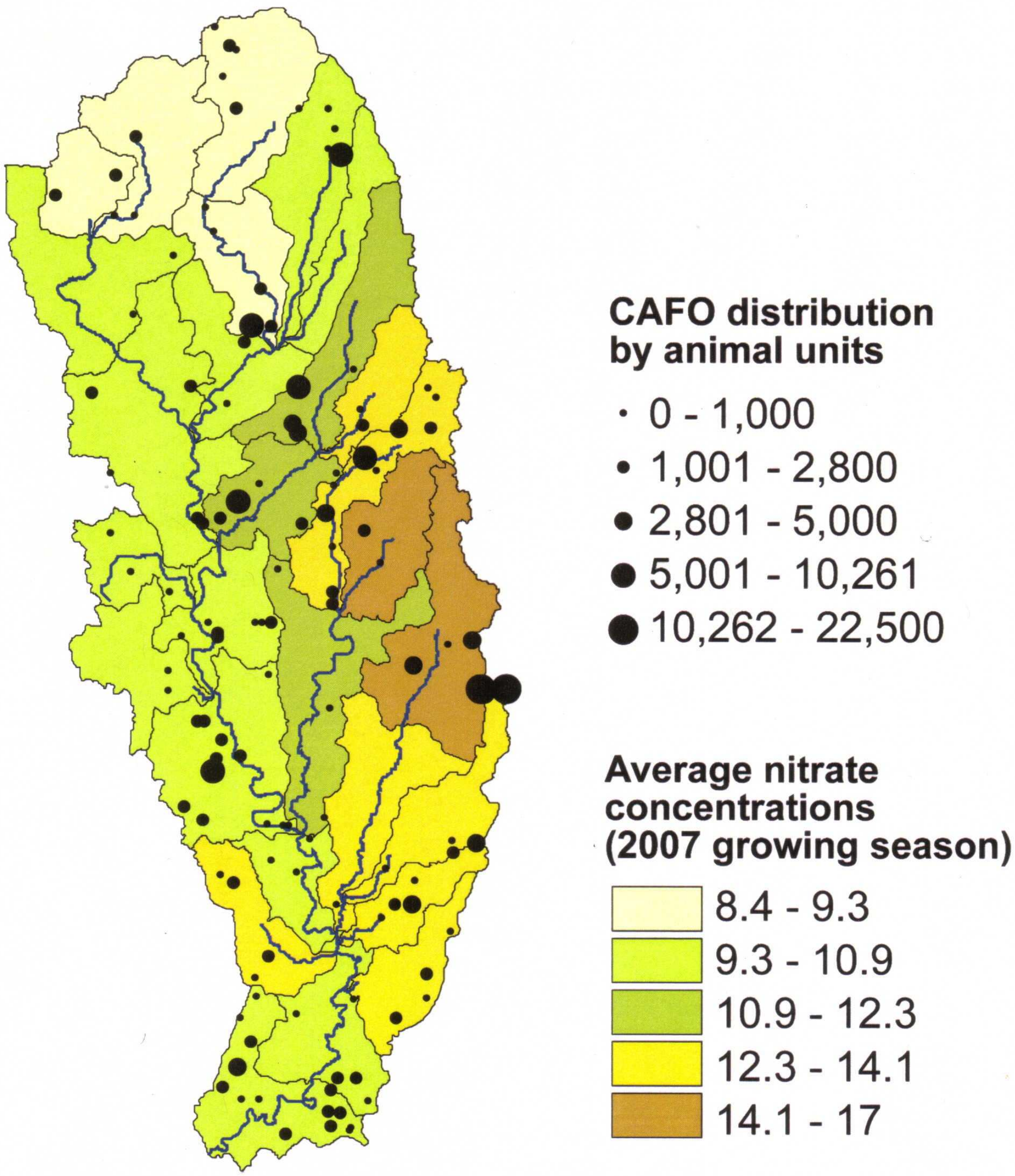


Figure 2.4. Distribution of confined animal feeding units (by animal units) overlayed on average nitrate concentrations determined for each 12-digit watershed during the 2007 growing season in the Boone River watershed. (reprinted from Krogh et al. 2008, Rapid Watershed Assessment).

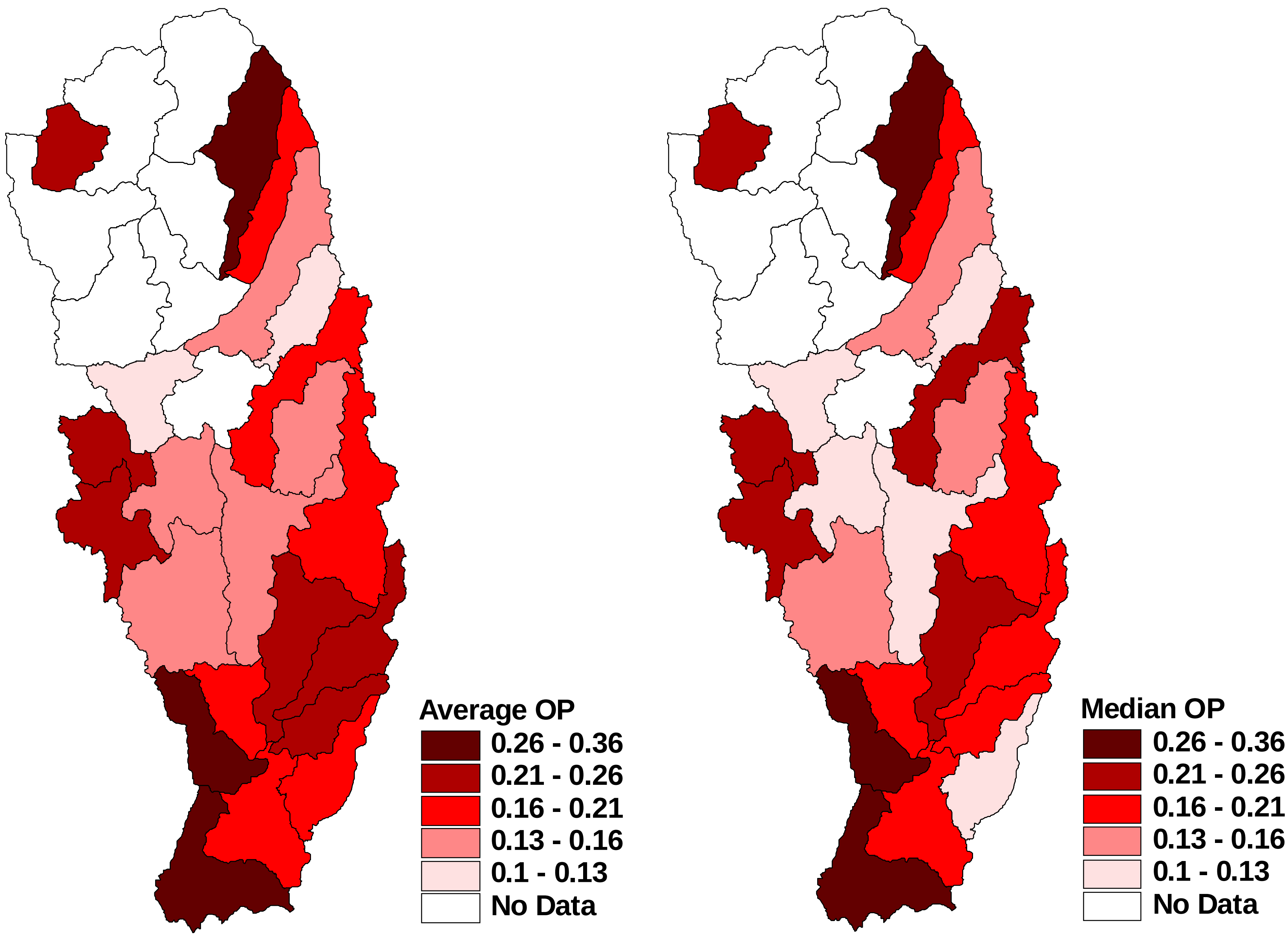


Figure 2.5. Mean and median ortho-phosphate (OP) concentrations measured in 2007 ISA monitoring.

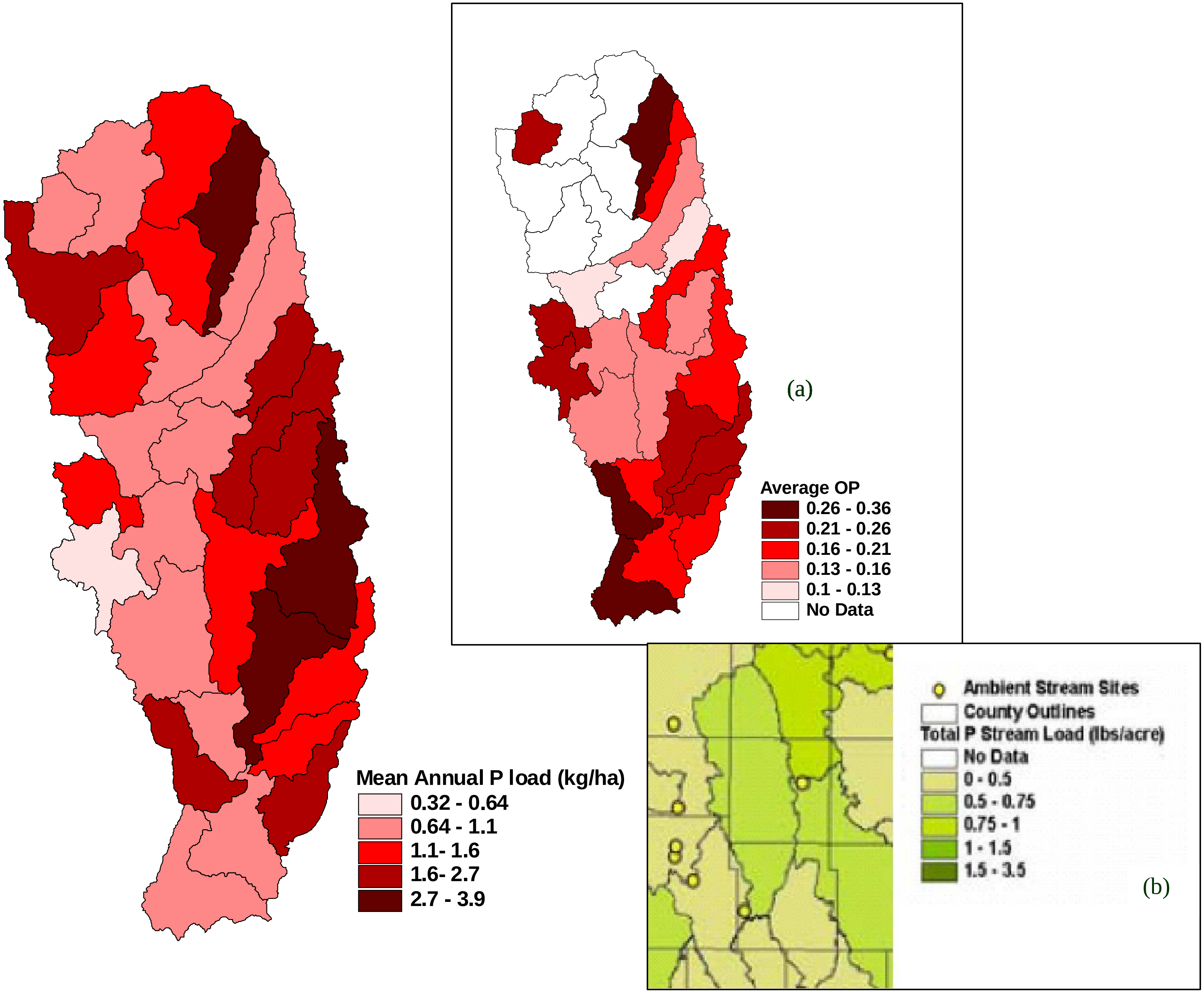


Figure 2.6. Estimated 30-yr avg. annual P load (kg/ha) based on SWAT model baseline scenario (average OP [ug/L] from 2007 ISA monitoring (a) and predicted P loads from Libra and Wolter (2004) (b) for comparison).

Sediment

In the U.S., sediment—including excess turbidity, suspended solids, erosion and sedimentation—is routinely cited as the number one water quality problem for surface waters (Simon and Darby 1999), and the leading cause of water quality impairment on state 303(d) (TMDL impairment) lists (U.S. Environmental Protection Agency (USEPA 2002). Excess suspended sediment results in reduced diversity and abundance of aquatic biota, reduced reservoir capacity, increased drinking water treatment costs, and serves as a carrier for contaminants such as phosphorus, bacteria, heavy metals and pesticides. Among stream fishes, excess suspended sediment and/or turbidity can induce physiological stress, impair feeding rates, and reduce reproductive success (Newcombe and Jensen 1996). Suspended sediment also reduces the amount of sunlight available to aquatic biota, impairs vision of visual feeders, block fish gills, and causes changes in habitat and biodiversity (Dils and Heathwaite 1999, Newcombe and Jensen 1996). In addition to being a source of phosphorus, which readily adsorbs to sediment, suspended sediment may carry pesticides, pathogens, heavy metals, and other pollutants. Sedimentation of habitats occurs as suspended sediment settles out of the water column as regular or baseflow conditions return.

Sediment regime indicators include turbidity, total suspended solids or total suspended sediment (TSS), as well as analysis of total sediment yield in relation to pre-disturbance conditions. Turbidity data analyzed in the ecological assessment suggested that turbidity values for the watershed as a whole did not exceed acceptable ranges of variation for either season or for the annual cycle overall. Some individual sample values did exceed the acceptable range of variation for turbidity in both the Lower and Upper watershed zones. The ecological significance of these individual elevated values is not known; the relationship of turbidity to flow conditions was not considered in the EPA (2000) analysis and is difficult to assess.

Average and median TSS concentration from measured data at the outlet of the Boone (1999-2007) are 52 and 11 mg/L, respectively. This is much lower than the median suspended sediment concentration of 82 mg/L found in the USGS study of eastern Iowa watersheds located just to the east of the Boone, occurring on the more erodible Driftless and Iowan Surface landforms rather than the Des Moines Lobe (Becher et al. 2001).

A review of quantitative effects of TSS appears in Table 2.3. In a review of the effects of sediment on stream fishes of the Missouri Basin, Doisy and Rabeni (2004) defined “excess sediment” as “the concentration of particles < 2 mm in size entrained in the water column of a stream for a period that deviates from the normal concentrations and durations for that stream type to the extent that it has a detrimental effect on native aquatic life.”

Sediment yield is perhaps the most important indicator of sediment regime, as it relates to altered hydrology and channel geomorphic regime. Because there is a proportional relationship between stream sediment load and stream discharge (Lane 1955; Simon et al. 2004), it is possible to estimate sediment yield by regressing TSS on discharge using the sediment transport curve. The sediment -discharge relationship varies between ecoregions due to differences in slope and particle size, and between stable and unstable streams (Glysson 1987, Simon et al. 2004).

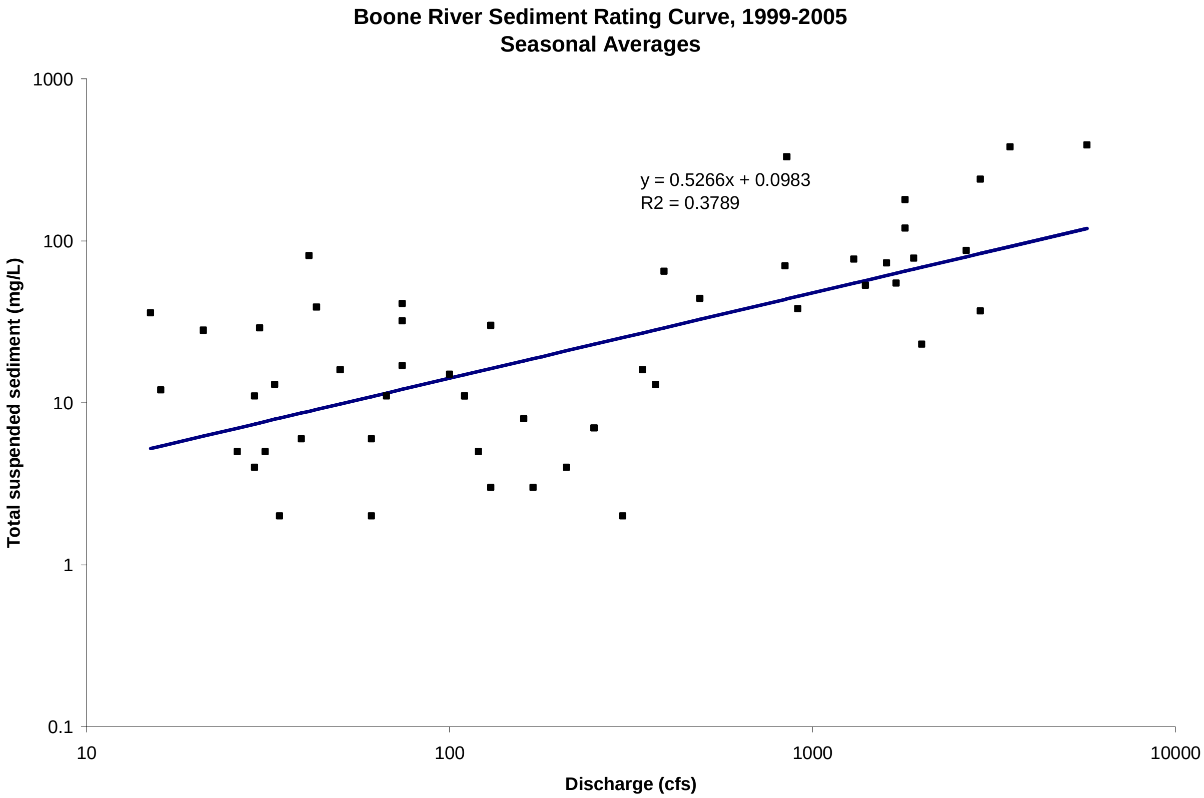


Figure 2.8. Sediment rating curve for the Boone River based on IOWATER / STORET data from 1999- 2006 near the outlet of the Boone River.

Simon et al. (2004) attempted to establish ecoregionally specific 'background" or "reference condition" sediment yield estimates for establishing sediment TMDLs. The methodology began by characterizing sediment yields by quartile for all streams in each ecoregion for which data were available. For a subset of ecoregions, sediment yield values for geomorphologically stable stream channels were then compared to those for unstable channels. The median value for stable sites within a given ecoregion is generally at least an order of magnitude lower than for nonstable sites. They observed a four order-of magnitude range of median ''reference'' values for the eight ecoregions, indicating that background levels for sediment may vary at a finer scale even than Level III ecoregion[2], requiring a considerable data collection effort to establish reliable standards. The median TSS concentration at the $Q_{1.5}$[3] for rivers and streams in Ecoregion 47 (Western Corn Belt Plains) can be expected to vary between 401-2000 mg/L (Simon et al. 2004).

Application of the sediment rating curve developed for the Boone River to estimate historical sediment yields at the outlet of the Boone River suggest lower yields than Simon et al. (2004) or

---

[2] Level III is the third level in the hierarchical classification of national ecosystems used by U.S. EPA and others for planning purposes, defining ecoregions based on spatial patterns in geology, physiography, vegetation, climate, soils, land use, etc. (Omernik 1986)
[3] See discussion of $Q_{1.5,}$ bankfull and effective discharge in section on Hydrologic Regime, pgs 42-43.

than that predicted by the SWAT model baseline scenario (average annual yield of 120,000 tons when applied to 1980-2007 discharge data). Only six measurements of TSS are available for the Boone at flows above 2000 cfs, averaging 390 mg/L. Based on measured data near the outlet of the Boone, the linear sediment rating curve for the Boone (Figure 2.8) would predict a TSS at 4000 cfs of around 443 mg/L, which falls within the lower end of the range predicted by Simon et al. (2004). However, the log linear estimate—the conventional approach—gives a much lower prediction of just 82 mg/L. However, the r^2 value for the linear sediment rating curve has a higher r^2 value (0.35) relative to the log linear curve (0.27), and gives an annual yield estimate of 175,000 tons/year—much closer to the value predicted by the SWAT baseline scenario for a similar time period (see discussion below).

The SWAT model baseline scenario estimated the 30 year average annual sediment yield at the watershed outlet of 236,500 tons. This corresponds to an average annual yield at the watershed outlet of 1 ton/ha, equivalent to 0.4 tons/acre, or < 0.1 T/km2. Since most annual sediment is transported on the few days when flows are at or above the mean annual flood (~2000 cfs), the median sediment yields reported for Ecoregion 47 by Simon et al. (2004) ranging from 0.81-6.5 tons/day/km^2 would seem to place the Boone River well below the median range for the ecoregion. Although this is certainly within the median estimated for the region, there are few if any reference streams in the Western Corn Belt Plains representative of pre-settlement hydrologic conditions, and interpreting this data is difficult without more information.

Whether estimating sediment yields based on the SWAT model or the sediment rating curve developed from limited TSS data, it appears per acre sediment yields for the Boone River are generally lower than Iowa watersheds draining other landforms. Sny Magill, an eastern Iowa watershed of more erodible Paleozoic Plateau landform (in the Driftless Area) has an average annual sediment yield of 0.26 tons/acre, despite having only 25% of the watershed in row crops. Schilling et al. (2007) reported average annual sediment yield of 0.69 tons/acre for the Squaw Creek watershed, located to the south of the Boone in the Southern Iowa Drift Plains. From 1996-2000, Squaw Creek and Walnut Creek—two smaller watersheds located in the Southern Iowa Drift Plains landform--carried sediment loads of 40,357 and 42,000 tons, equating roughly to 2.5 and 4 tons/ha (1- 1.6 tons/acre) respectively. Figure 2.9 shows the log rating curve for the Boone based on IDNR data at the Boone River watershed outlet, compared with sediment rating curves developed for Squaw Creek and Walnut Creek subwatersheds as reported in Shilling et al. (2006).

However, the annual yield of 0.4 tons/acre/year for the Boone predicted by the SWAT model is higher than average yields reported for another highly row cropped basin of the Des Moines Lobe, the South Fork of the Iowa River, estimated at approximately 0.27 tons/acre (Schilling et al. 2007). This is significant, because Schilling et al. (2007) also reported that sediment losses in the South Fork watershed are actually about three times higher than typically measured in the Des Moines Lobe region. They suggest that the bulk of sediment losses for the South Fork watershed may originate in the lower third of the watershed, within a more erodible landscape of hilly moraines near the edge of the Des Moines Lobe, where the river erodes its banks as it meanders across an alluvial valley. The lower portion of the Boone River also flows through a more erodible and dissected landscape.

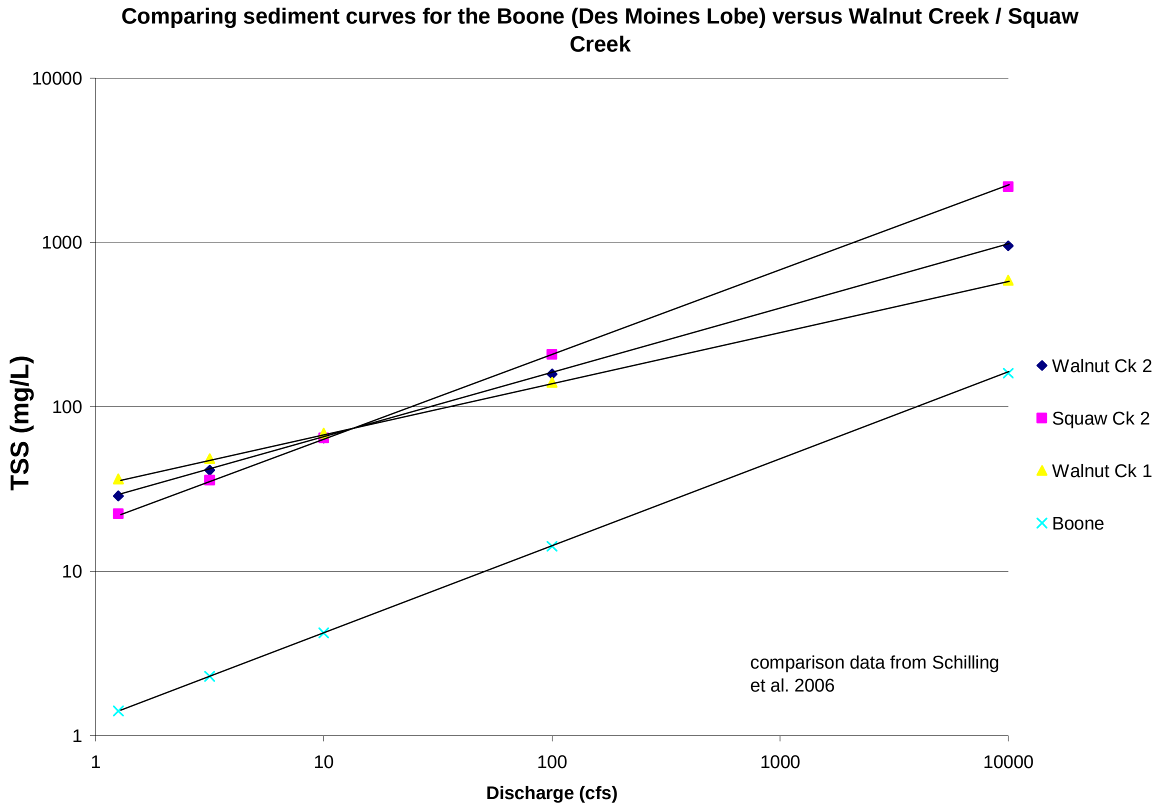


Figure 2.9. Sediment rating curve based on IOWATER/STORET TSS data in relation to discharge in the Boone (1999-2005).

Physical habitat assessment data available for the Boone come primarily from three sources: the Iowa Wadeable Streams Biological Assessment (5 sites), Kelly Poole's 2005 mussel survey, and anecdotal descriptions provided by IOWATER volunteers. A visual survey of the Boone river streams conducted as part of Charlie Kiepe's field practices inventory mapped 37 miles of stream experiencing moderate stream bank erosion, and 2.6 miles (4.2 km) of severe streambank erosion, or about 5% of the watershed by total stream miles. In addition, more than 95 miles of moderate or severe gully erosion were mapped, corresponding with intermittent channels in fields.

Historically, concerns about soil erosion have focused almost exclusively on surface runoff as the major transport pathway for sediment. SWAT model assumptions in fact treat all sediment and most adsorbed contaminants as entirely delivered to surface waters via surface runoff, and the model does not route sediment and other sorbable contaminants through subsurface flow.

Increasingly, however, watershed research and restoration is beginning to focus on the role of both subsurface drainage and on stream channel erosion in contributing to sediment impairments to surface waters. The role of streambank erosion in stream sediment load and transport, and its relationship to hydrology, row crop land use, and tile drainage is further discussed in the context of channel geomorphic regime and in the section on threats and resource concerns.

The relationship between sediment transport and discharge is nonlinear—i.e. the bulk of sediment transport occurs during relatively infrequent high flow events. Therefore, episodic transport of sediment is responsible for a disproportionate share of sediment and contaminant yields to surface waters. Schilling (2000) found that in Walnut Creek, 98% of the annual sediment load occurs over the 6 month period from February to July (also the period of highest nitrate loss), and 60-80% of the annual sediment load occurs on average in just 5 days of high flows. Gentry et al. (1998) and David et al. (1997) also showed that brief episodes of high discharges of water, sediment, and agrochemicals in response to heavy rainfall make up a significant fraction of the total annual discharges. For streams in the western United States, Whiting et al. (1999) determined 57% of the annual bedload is transported at flows between the mean annual flood and the effective discharge (~bankfull, the flow with recurrence interval of 1.5 years), and 37% occurs in flows above bankfull (Whiting et al. 1999).

The sediment yield per unit land area than for the Boone, as well as for Squaw Creek and Walnut Creek, is also considerably lower than the 5.1-10 tons/acre (12-24 tons/ha) predicted by USLE models of field erosion on cropland for the region of the Upper Mississippi River Basin that encompasses the Boone (Gowda 1998, Figure 2.10). This is not surprising, given that estimates of upland erosion and field losses generally overestimate sediment loads and delivery in streams and rivers (Trimble and Crosson 2000). Much sediment reported as lost from fields is stored in upland areas of catchments and does not reach streams (Wilkin and Hebel 1982, Knox 2001, Trimble and Crosson 2000). Likewise, upland soil that is delivered to streams during storm runoff events does not all immediately travel downstream as sediment yield at the mouth of the waterbody. Rather, much of this sediment is deposited within the channel or floodplain as bedload or alluvium, causing sedimentation of instream habitats. During subsequent high flow events, these sediments are remobilized or deposited downstream according to the sediment transport capacity of any given flow event. Thus, the movement of stored sediments from the channel, streambank, and

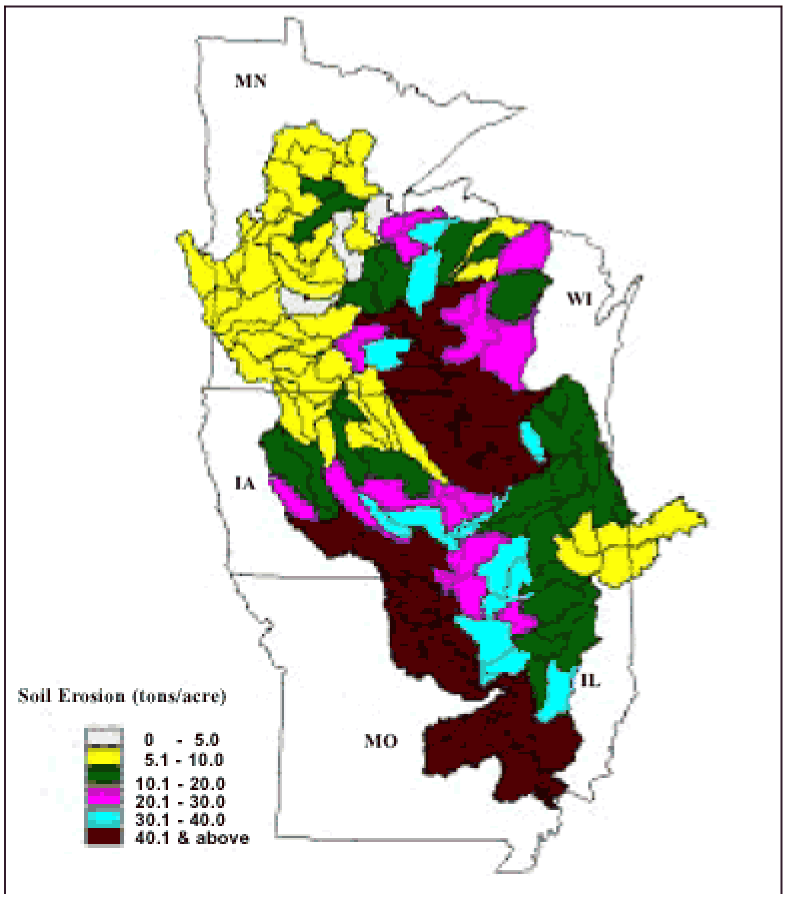


Figure 2.10. Soil erosion from agricultural sources by USGS 8-digit hydrologic units in the Upper Mississippi River Basin (Gowda et al. 1995)

valley floodplains has long-term implications for stream habitats throughout a channel network (Knox 1987, Magilligan and Stamp 1987).

Alluvial deposits from periods of high sediment aggradation may require timescales of decades to centuries to migrate downstream (Beach 1994). Sources, sinks, and fluxes vary widely over time and space, depending on how recently channels have been destabilized (Faulkner 1998, Trimble 1999). In river basins of southwestern Wisconsin, for example, geomorphologic investigations have demonstrated that the rate of alluvial sedimentation has greatly decreased since the period of maximum erosion during the 1930s, when conversion to agriculture generated a 3-5 fold increase in annual flood volumes. However, sediment yield has remained fairly constant as streams have redistributed bedload downstream, and most sediment has moved only short distances (Fitzpatrick et al. 1999). In southern Minnesota, Wisconsin, and Pennsylvania, the movement of large amounts of topsoil from the uplands into stream floodplains and bottomlands during the early part of the 20th century during the initial conversion of the landscape from perennial prairie to agriculture and timber resulted in deposition of sediments in stream valleys that are sometimes several meters thick. Studies of sediment footprints in the upper Mississippi River valleys in Minnesota and Wisconsin show that in 137 years since European settlement, 38-73% of all eroded sediment has travelled no more than 4 km (Beach 1994). Similarly, in the Squaw Creek and Walnut Creek subwatersheds of Iowa, located south of the Des Moines Lobe in the more erodible Southern Iowa Drift Plain, up to 50% of the annual total sediment load comes from Holocene alluvium and post-settlement materials.

<u>*Emerging/Toxic Contaminants*</u> **–** Water and fish tissue samples from the Boone River have been evaluated for presence and levels of a number of contaminants known to be toxic and potentially present based on patterns of use. Levels detected in sampling from 1980-2003 were reported in the 2005 Ecological Assessment. A few – hexachlorobenzene, Nitrate (NO_3) as N, and Nitrite (NO_2) as N – have been detected during at least one sampling episode at concentrations that exceed state criteria for acute exposure. Subsequent sampling years are summarized in Appendix A. Numerous pesticides and herbicides and their byproducts continue to be found at detectable levels in surface waters of the watershed. Although data are insufficient to assess whether concentrations of contaminants exceed Iowa criteria for chronic exposure to toxic contaminants in Class B Warm Waters, average concentrations – based on small numbers of samples – of several herbicide and pesticide compounds do exceed the state criteria for chronic exposure. Therefore it is possible that many locations exceed state criteria for chronic exposure. Several pesticides or their byproducts are present in fish tissues, indicating that these contaminants are moving through the food web.

For comparison purposes, the USGS investigated water quality in watersheds located just to the east of the Boone River from 1996-1998 (Kalkhoff et al. 2000). Although the use of herbicides and insecticides in this area was among the most intensive nationwide, herbicide concentrations in streams were not among the highest 25 percent nationally, and insecticide concentrations were in the lowest 25 percent nationally. However, breakdown compounds (degradates), whose widespread occurrence has only recently been discovered and about which little is known about human and environmental effects, generally accounted for the majority of the pesticide compounds present in rivers and streams. The most commonly used herbicides were the most frequently detected and were generally present in the greatest concentrations. Atrazine and metolachlor were detected in all stream samples. Atrazine concentrations generally ranged from 0.1 to 1.0 μg/L (microgram per liter), and exceeded the USEPA 3.0-μg/L drinking-water standard in about 10 percent of the samples; mainly during late-spring runoff. Acetochlor, a conditionally registered herbicide that is intended to replace several other commonly used herbicides, was frequently detected, but concentrations were less than 0.1 μg/L in 75 percent of the samples. Mean annual acetochlor concentrations did not exceed the 2.0-μg/L USEPA registration requirement at any site, but concentrations did exceed that level in about 3 percent of the individual samples. The maximum concentration measured during the study (10.6 μg/L) exceeded the level that would require biweekly sampling by water-supply systems. Alachlor, metolachlor, and acetochlor degradates were present in relatively high concentrations throughout the year; thus they appear to be more persistent than their parent compounds. Carbofuran and chlorpyrifos, insecticides that have been identified as posing a high risk to aquatic insects and mussels, were present in as much as 60 percent of the monthly samples during the summer when these insecticides are normally applied.

E. coli. Certain types of bacteria, including *E. coli* (a type of fecal coliform), normally found in the digestive systems of animals including humans, livestock, and wildlife, serve as indicators of fecal contamination of water. While these indicator bacteria themselves do not always cause human or wildlife disease, their presence indicates the likelihood that other contaminants, harmful bacteria and pathogens are also present. Because it is unclear how long *E. coli* can survive in the environment, their presence in surface waters suggests a direct pathway of contaminants from livestock operations or other sources, such as failing septic systems or sewage treatment. Because the scale of livestock production in the region dwarfs other potential sources, the bulk of the problem of bacterial contamination in streams is generally attributed to improper handling of waste from confinement feeding operations (primarily swine and poultry), including inappropriate rate or timing of land application of manure as fertilizer.

Although not initially identified as a key water quality attribute for the Boone River, *E. coli* have been found at elevated levels at stream locations within the Boone River watershed, and have been identified as a source of impairment for Buttermilk and Lyons Creeks. Elevated bacteria counts have also been detected in drinking water wells. Bacteria levels are therefore of concern to watershed residents, and bacteria were identified as one of the key resource concerns under the NRCS RWA.

The amount of bacteria in water is expressed as the number of Colony Forming Units per 100 milliliters (CFU/100 ml) or as Most Probable Number per 100 milliliters of water (MPN/100 ml). Under Iowa's water quality standard, the one-time maximum value is 235 CFU/100 ml for Class A swimmable and Class A wadeable waterbodies. For Class B (CW) or HQ (high quality) resource streams, the maximum value is 2880. Between 2000-2005, the Class A standard was exceeded in 4 out of 64 samples (6%) on the lower Boone River, including a maximum level of 21,000 for one measured summer event. *E. coli* levels well above the water quality standard were recorded at least 5 different IDNR stream sample locations in the Boone River watershed. Expanded sampling conducted by the Iowa Soybean Association in 2007 found that the Class A 235 cfu/100ml standard was exceeded in 35-100% of summertime samples at all 30 sample sites, and in 13-75% of fall/winter samples. The Class B 2880 cfu/100 ml standard was exceeded in summer in 10-33% of samples at 24 of 30 sites sampled (Figure 2.11).

The highest levels of *E. coli* were associated with high flow events, suggesting that bacterial contamination of surface waters occurs during episodic flow events and that loads are being delivered in large flushes to streams via surface runoff or subsurface tile lines. However, there is often considerable uncertainty in tracing bacterial loads to their source. In Minnesota, various fingerprinting techniques are being developed to accurately identify the origin of bacteria for the purposes of more equitable load allocation in TMDL plan implementation.

Hydrologic Regime and Connectivity

Analysis of USGS gage data shows that bankfull or effective discharge for the Boone River, generally the flow frequency that can be expected to occur on average once every 1.1 - 2 years ($Q_{1.5}$), is between 2000 and 4000 cfs, depending on the period of record. Based on Simon et al. (2004)'s regional reference curves for Ecoregion 47, the Western Corn Belt Plains, $Q_{1.5}$ discharge for the Boone River is 4092 cfs, which is slightly higher than that actually observed based on the period from 1971-2006.

To help understand and quantify the impact of hydrologic change, The Nature Conservancy has developed a software tool called the Indicators of Hydrologic Alteration (IHA). The program is used by water resource managers, hydrologists, ecologists, researchers and policy makers from around the world to assess how rivers, lakes, and groundwater basins have been affected by human activities over time. The IHA program assesses 67 ecologically-relevant statistics derived from daily hydrologic data, including the timing and maximum flow of each year's largest flood or lowest flows, as well as mean and variance of these values over some period of time. Comparative analysis can then help statistically describe how these patterns have changed for a river as a result of land- and water-use changes. USGS daily discharge gage data from the Boone River gaging station were used in the RVA to compare the period from 1940-1970 to the period from 1970-2007.

To analyze the change between two time periods, IHA software uses a Range of Variability Approach (RVA) described in Richter et al. (1996). The RVA uses IHA parameter values from a "pre-impact" period as a reference for defining the extent to which natural flow regimes have been altered. The pre-impact variation can be used as a basis for defining initial environmental flow goals. However, in the Boone, gage station records are not available prior to 1940, despite the fact that substantial land use change had already occurred in the Boone River by 1940. The period from 1940-1960 corresponds to a second major period of drainage activities in the Upper Midwest (Zucker and Brown 1998, Wilson 2000).

RVA analysis generates a series of Hydrologic Alteration (HA) factors which quantify the degree of alteration of the 33 IHA flow parameters between two time periods. Parameters are assigned to categories of "low", "medium", or "high" based on comparison to the set of values observed during the pre-impact period (the default is 1/3 in each category). The HA factor is computed using the number of years in the category during the post-impact period (the observed frequency) and the number of years that would be expected to be in the category in the post-impact period if flows were un-impacted. A positive Hydrologic Alteration (HA) score means that values in the category have increased from the pre- to the post- period, while a negative score means that the values have decreased.

In an RVA analysis for the Boone River comparing the period from 1940-1970 to 1970-2007, the majority of indicators (19 out of 33 indicator variables) showed increases in the “high” RVA category from the pre- to the post- period (Figure 2.12). The analysis shows significant increases in the 1-day, 3-day, and 7-day minimum flows. Other indicators that experienced significant flow alterations were median flows for the months of December, May, June, and July; 90 day maximum flows, and 30 day minimum flows. The median, 25th, and 75th percentile flow rates increased in the latter period in every month of the year (Figure 2.13). The RVA results are consistent with the observations by Schilling and Libra (2003) that baseflows have increased throughout Iowa in the second half of the 20th century, albeit at a lower rate of change in the Boone than in many other watersheds.

As mentioned above however, the 1940-1970 flow period does not represent “presettlement” flow conditions. Substantial land use change, agricultural development, and drainage activities had already occurred in much of the region by 1940. To approximate a more accurate “pre-impact” hydrologic scenario, we used daily hydrologic output predictions from the SWAT model “all perennial” scenario that assumed 100% switchgrass with no fertilizer inputs. To estimate the degree of hydrologic alteration in the Boone between these two scenarios, we compared these outputs over the exact same flow and climate conditions to the baseline conditions (current conditions) scenario. This comparison provides a control on climate variability not usually available in IHA analyses. However, because it does not assume any differences in the drainage network layout (i.e. all tile drain and surface drainage channels are the same in both scenarios) it is not a true “pre-settlement” scenario. Furthermore, a presettlement vegetation scenario would involve a mix of species with different patterns of growth and water use that would likely lead to greater evapotranspiration than the monoculture of switchgrass (which is a limitation of the SWAT model) and therefore these hydrologic scenarios. Nevertheless, the comparison is instructive.

The results of this RVA analysis show a more subtle pattern of differences between the perennial versus the baseline scenarios (Figure 2.14). For April flows, the number of values in the “high” RVA category greatly increased, as did the values in the “high” RVA category for duration of high pulses (although the count of such pulses decreased). However, the number of years that fell into the high and middle RVA categories for July, August and September flows decreased, while the low RVA category flows increased. This means that more years fell into the low RVA category for summer flows under current conditions than would be expected under a perennial scenario.

Analysis of monthly flows reveals the pattern (Figure 2.15). Median flows from October through February are very similar. However, the baseline scenario shows much higher median flows in March, April, May, and June, followed by lower median flows in July, August and September.

In sum, hydrologic alteration in the Boone River since the early part of the 20th century seems to have led to an overall pattern of increased flows, both baseflow and peak flows, which is particularly pronounced in April and May. The pattern is consistent with the widespread implementation of subsurface tile drainage and partial shift in water budget from evapotranspiration to baseflow. While it is difficult to establish a quantitative acceptable range of variation with respect to the impacts on aquatic biota, the magnitude of the changes is large (-10 - 35%) and suggests that altered hydrology is a likely suspect accounting for some of the observed changes in and stresses for aquatic biota. Higher flows could account for excessive sediment loads and increased bank and bed erosion in the Boone. High flows in spring and early summer followed by decreased flows July-September may also have negative effects on stream biota, via effects on hydraulic habitats.

Ecological impacts associated with altered flow magnitude and altered patterns of high and low flows are discussed further in the section on threats. Implications of these changes for mussels, fish, and other biota are discussed in other sections, but potentially include increased scour and stranding, channel erosion and sedimentation, as well as increased loads of sediment, nutrients, and contaminants. Other potential impacts include chronic effects on life history and habitat selection by mussels and their host fishes.

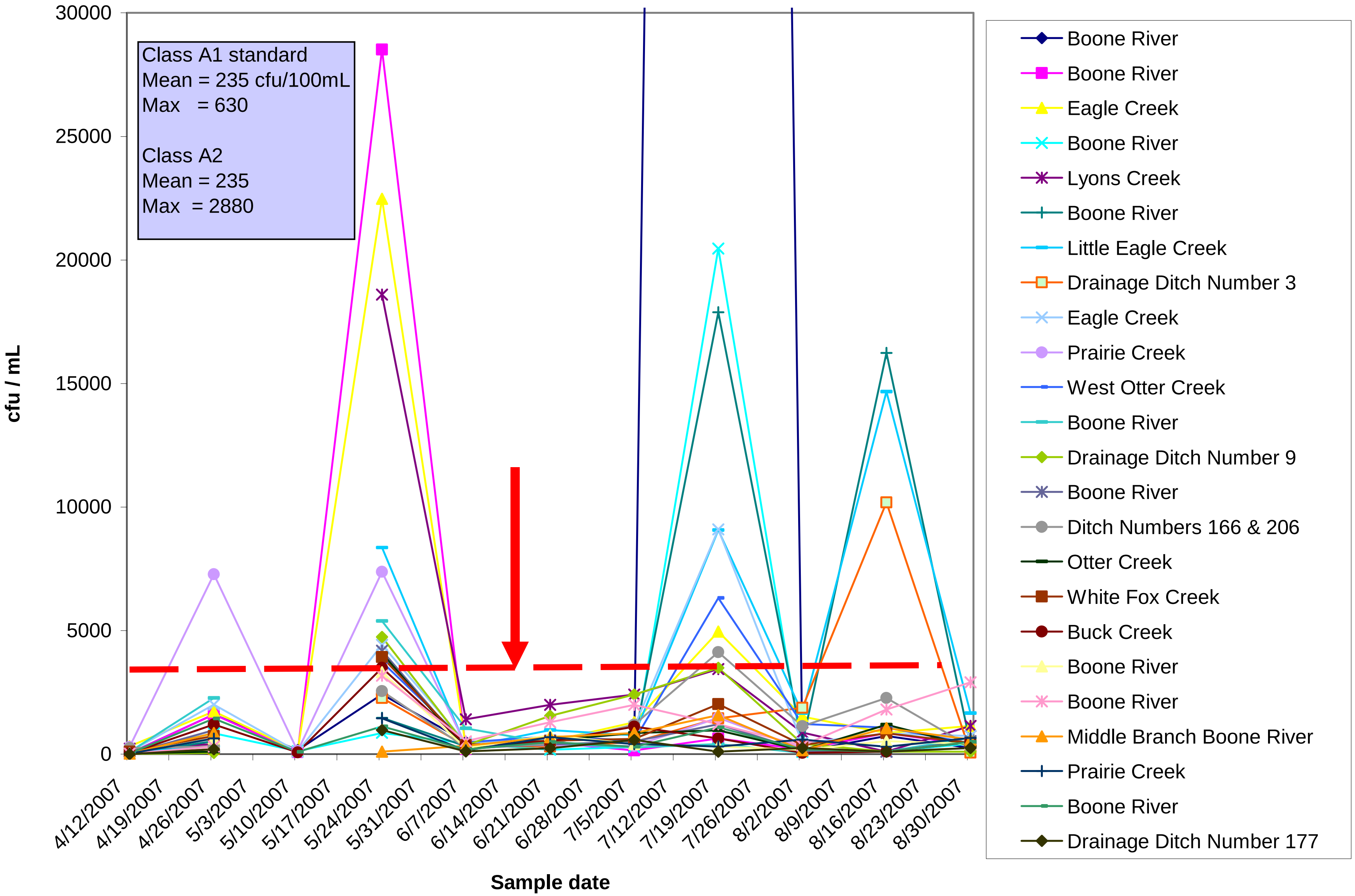


Figure 2.11. 2007 Weekly ISA monitoring results for *E. coli* at Boone River sample locations.

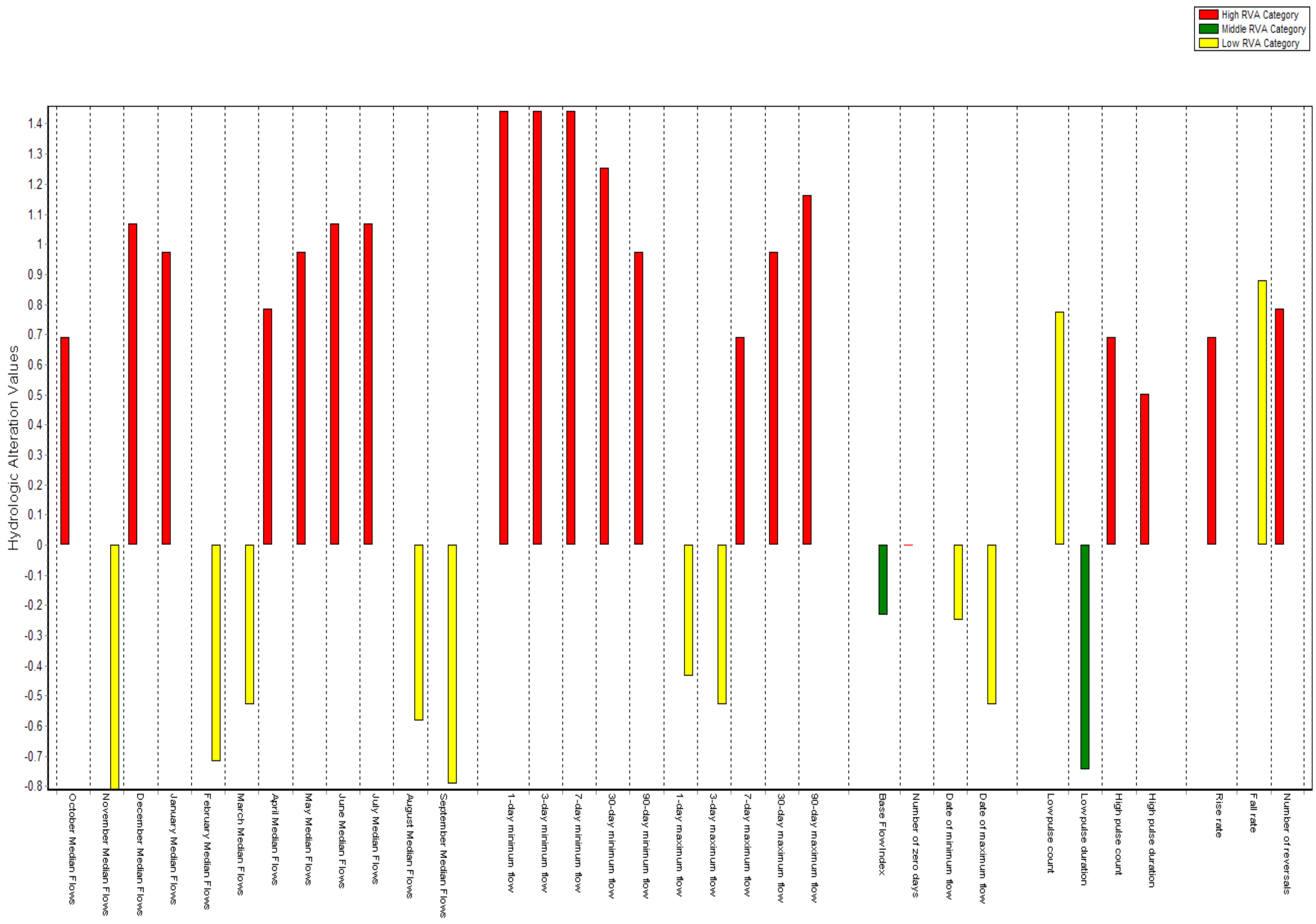


Figure 2.12. Indicators of Hydrologic Alteration Range of Variability Analysis (RVA) results, comparing 1940-1970 to 1971-2008 discharge records.

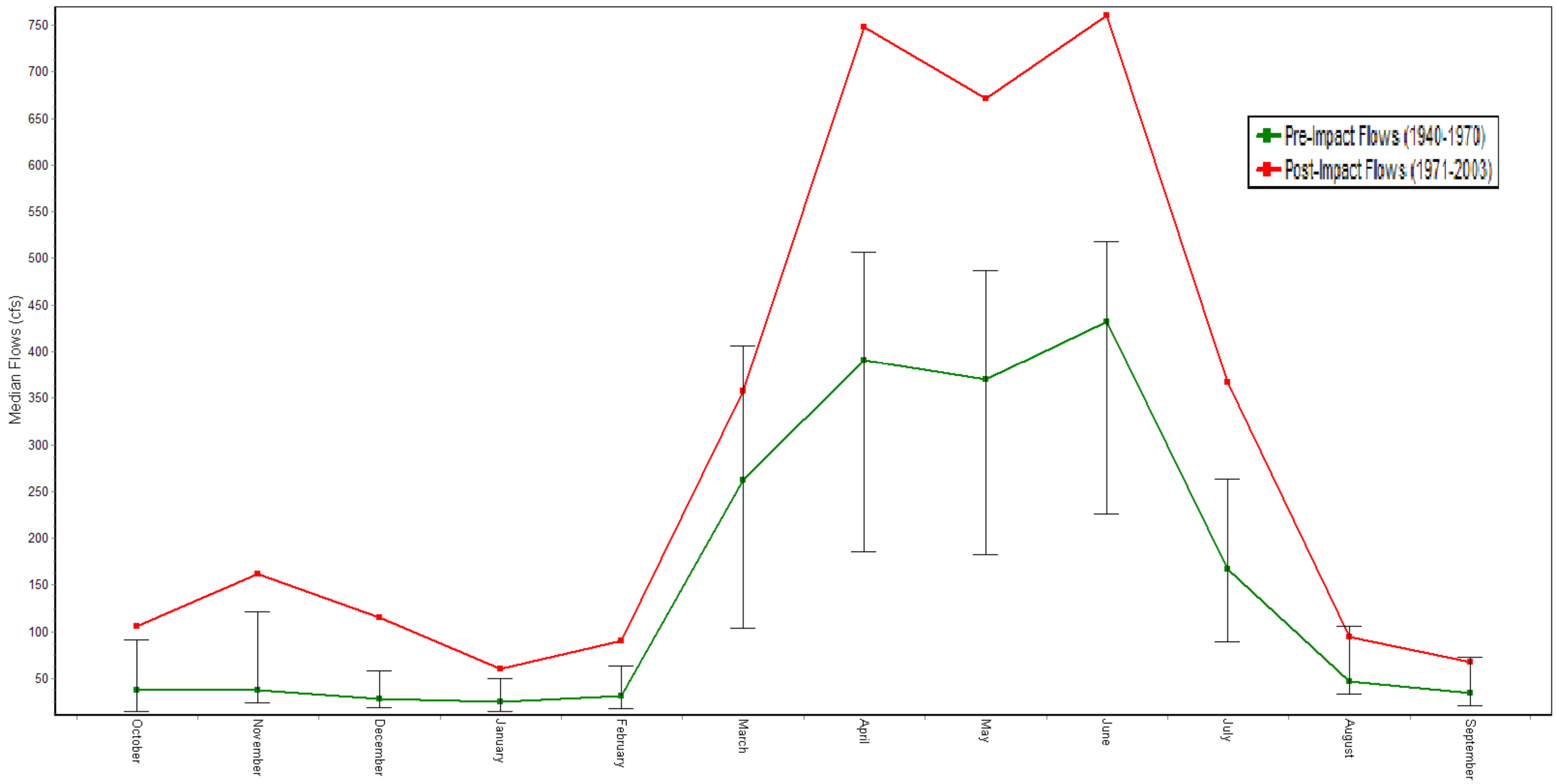


Figure 2.13. Indicators of Hydrologic Alteration analysis of monthly median flows, comparing 1940-1970 to 1971-2008 discharge records.

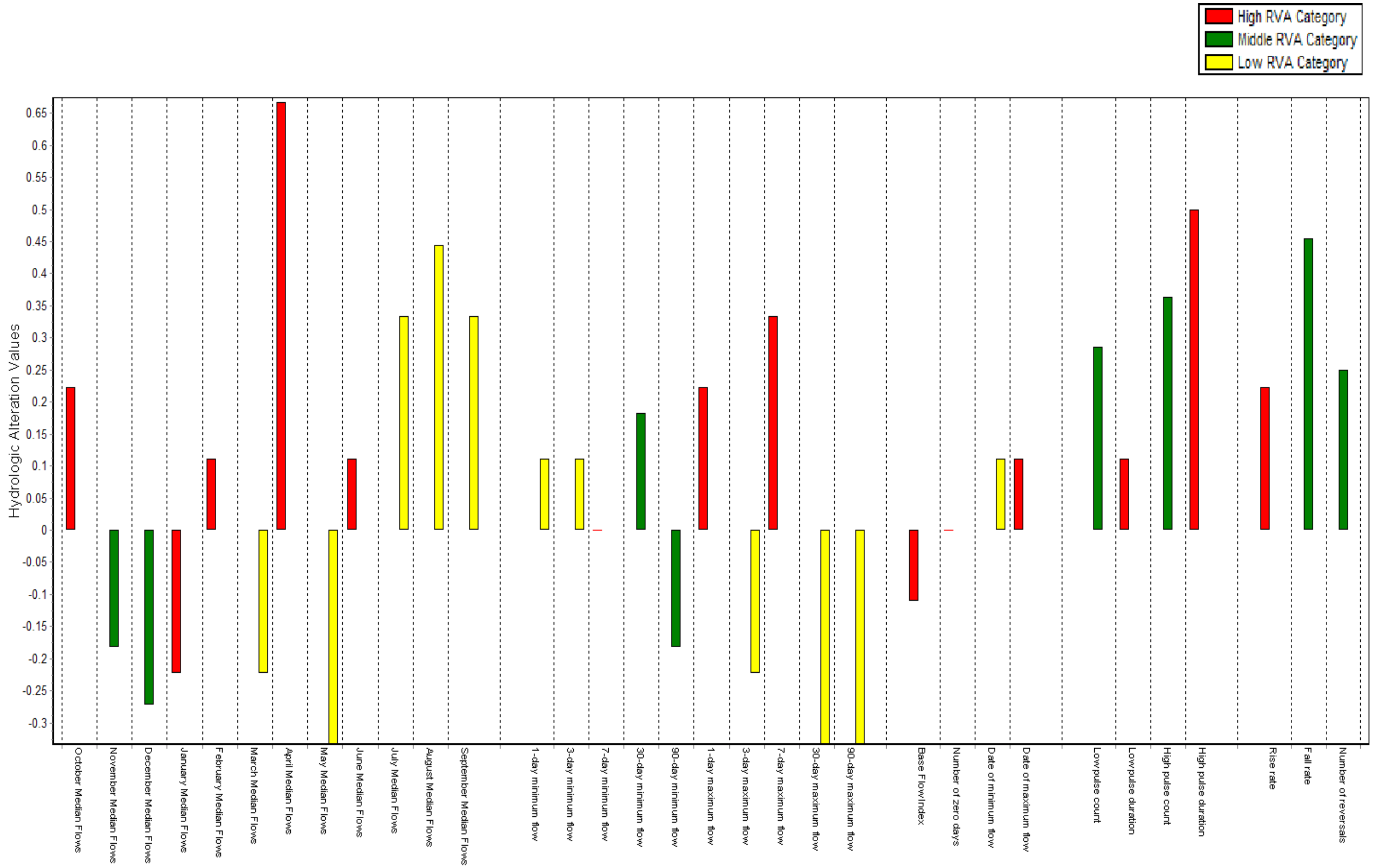


Figure 2.14. Indicators of Hydrologic Alteration Range of Variability Analysis (RVA) for environmental flow parameters, comparing SWAT perennial switchgrass (PRE) to baseline (current conditions) hydrologic scenarios.

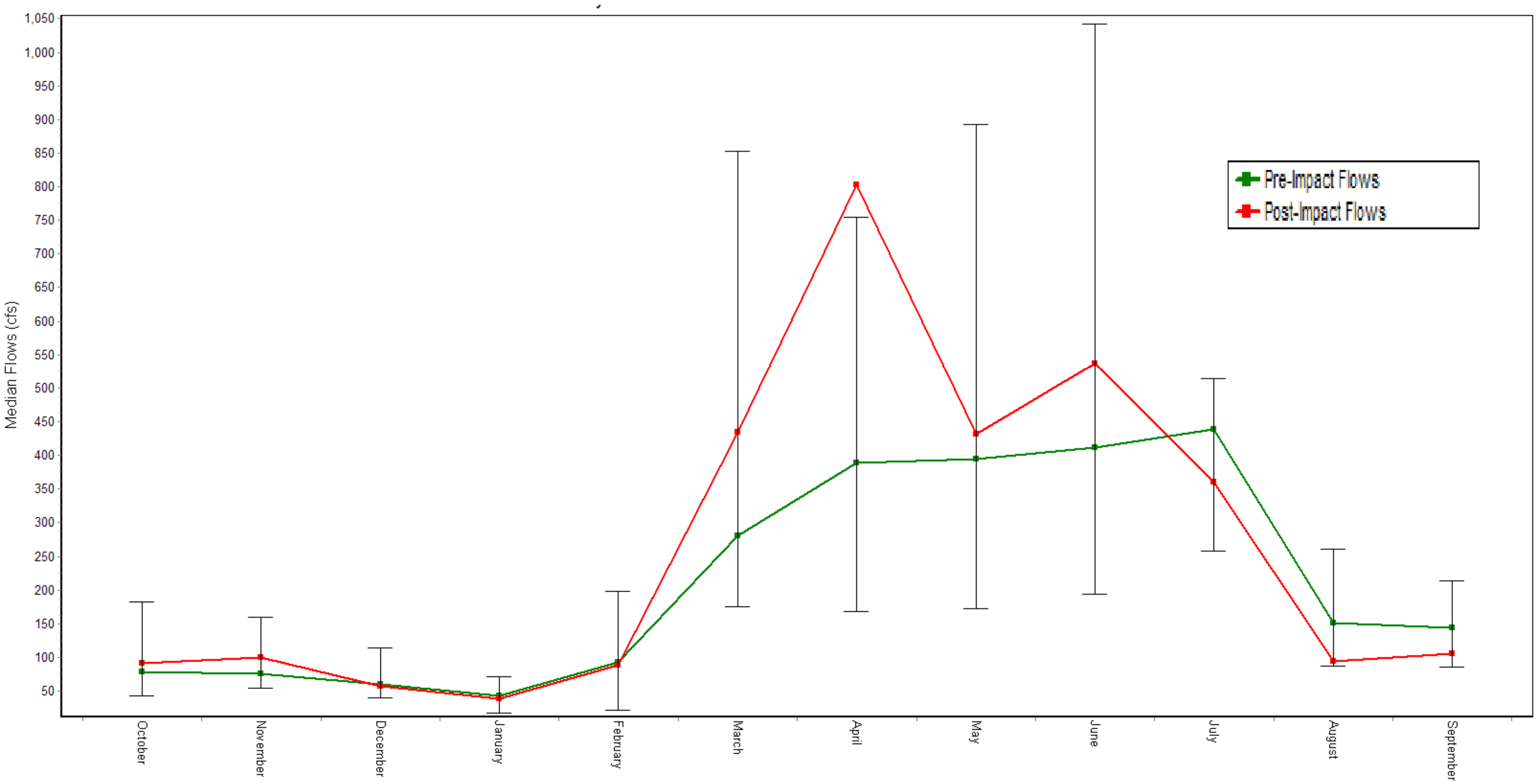


Figure 2.15. Indicators of Hydrologic Alteration analysis of monthly median flows, SWAT perennial switchgrass (PRE) to baseline (current conditions) hydrologic scenarios.

Channel geomorphic regime

While considerable effort has been directed toward reducing erosion from agricultural and urban lands, stream channel degradation has only recently been acknowledged as a major contributor to sediment impairments in surface waters. Studies have shown that sediment from streambanks can account for as much as 85% of watershed sediment yields, and bank retreat rates as great as 1.5 m – 1100 m/year have been documented (Simon et al. 2000). In addition to water quality impairment, streambank retreat impacts riparian ecosystems, as well as developed infrastructure along streams and in floodplains, including homes, roads, and bridges (ASCE 1998, Wynn 2006).

Channel instability, bank/bed erosion and phosphorus. Stream bank erosion has been estimated to account for anywhere from 45-90% of the sediment load in streams in Iowa (Odgaard 1984; Schilling and Wolter 2000, Schilling 2000, Zaimes et al. 2004, 2006), and up to 80-90% in other regions of the US (Lawler et al. 1999, Simon et al. 1996) and other countries (Kronvang et al. 1997). In Minnesota, stream bank erosion contributed only 7-10% of the total phosphorus load in the stream (Sekely et al. 2002), while in Illinois (Roseboom 1987) and Denmark (Kronvang et al. 1997) the percentages were much higher with 56% and 90%, respectively. Bank and/or bed erosion is likely to be a major contributor of stream suspended sediment and P loads, particularly in small channelized lowland streams in agricultural or tile-drained catchments where P concentrations in sediment stored in streambanks are at or near the same level as in soils of surrounding agricultural lands (Kronvang et al. 1997, Laubel et al. 2003). Over a 2-year period, bank-derived clay-silt sediment and TP to streams represented 48-59% and 40-48% of mean annual suspended sediment (SS) and total P losses from similarly sized Danish agricultural catchments. Streambank slumping is also estimated to account for 36-84% of TSS load in the Blue Earth River (Bauer 1998). Sekely et al. (2002) estimated that 31-44% of the sediment and 7-10% of the annual TP load in flows discharged to the Minnesota River from the Blue Earth River comes from streambank erosion and slumping.

Because stream sediment loads are proportional to discharge, even if soil erosion on the upland landscape is completely eliminated, altered watershed hydrology has implications for stream sediment loads and channel instability. Higher annual and peak stream discharges in the agricultural Midwest have increased the scouring potential and sediment transport capacity of stream channels, leading to extensive incision and stream bank erosion (Menzel 1981). Even when sediment in stream is not derived from the uplands, increases in basin yield, peak flows, and total annual discharge can account for the increased instream erosion (Magner et al. 2004). In natural streams, the stream channel develops an equilibrium that can accommodate the movement of water and the movement of sediment over time (Lane 1955, Leopold 1994). Thus, changes in either the flow regime or sediment loads tend to produce compensatory adjustments in channel morphology as stream channels adjust to the new hydrologic and sediment regimes (Poff et al. 1997).

Therefore, even when the bulk of sediment load comes from instream erosion, much of this sediment can ultimately be traced to agricultural lands—either as the source of the re-deposited alluvial soils themselves, the hydrologic change that has generated their stream channel erosion, or both. Because of the time lag in channel adjustments to hydrology, even if pre-disturbance hydrology is restored to a watershed, it may take decades or longer for the channels to recover and for sediments deposited during the disturbance period to migrate out of the system. If stream bank erosion problems are driven by channel instability that is caused by altered watershed hydrology, the solution may require hydrologic restoration, redesign and restoration of eroding stream channels that are properly sized to the current hydrologic regime, or both.

Restoration of wadeable streams therefore requires scale appropriate management focusing on hydrology at catchment scales and erosion control and bank stabilization at reach scales (Rowe et al. 2006). For example, in the Pecatonica River watershed of Wisconsin, as a result of post-settlement erosion and alluvial deposition in the floodplain, stream incision is occurring in postsettlement alluvial deposits. Stream restoration in such settings – such as the Pecatonica – involves removal of floodplain sediments and re-establishing a more stable natural channel designed to mimic presettlement stream stability. But because the entire watershed has not been hydrologically restored, the restoration also requires upstream sediment traps.

Physical Habitat and Riparian Condition

Physical habitat assessment data available for the Boone come from three main sources: the statewide biological assessment of wadeable streams, Kelly Poole's 2005 mussel survey, and Charlie Kiepe's visual survey of stream and gully erosion (Kiepe 2005). Results suggest moderate impairment in many areas of the watershed.

The riparian area of the Boone River mainstem, even in the upper reaches, is predominantly woodlands on both sides of the river at most sites (Poole 2005). Analysis of land use data in GIS shows that the size and extent of natural vegetation in the riparian area increases with increasing stream size. 81% of land within 100 m of first order headwater channels in the Boone is in row crop production, whereas for the mainstem of the Boone River (a 5th order stream), only 28% of land within 100m of either side of the river is in row crop production. 68% is in natural vegetation (forest, grass or wetland).

Table 2.4. Mean percent land use within 100m of stream channels by stream order.

ORDER	Total miles of stream channel	% Natural	% Crop	% Developed	% Pasture/hay
1	376	28.6%	68.4%	1.3%	1.7%
2	168	43.6%	52.6%	1.4%	2.4%
3	80	62.2%	32.9%	2.1%	2.9%
4	32	72.8%	25.3%	0.3%	1.6%
5	73	85.6%	10.3%	2.2%	2.0%

Average stream width in the Boone River is 7 m in the Upper Zone and 22 m in the Lower Zone. In the upper portion of the river, the substrate is predominantly a mix of sand and gravel (72 %) with silt accumulating in pockets along the river edges. In the lower reach, the substrate is a mix of sand (34%), gravel (29%), and cobble (22%) with silt accumulating along the edges and in pooled areas.

Few data are available to assess the quality or condition of riparian plant communities along the Boone River and tributaries, in terms of wildlife habitat value, relative dominance by native versus introduced species, stream bank condition, etc.

A visual survey of the Boone river watershed conducted by Charlie Kiepe for the RWA identified 37 miles of stream experiencing moderate stream bank erosion, and 2.6 miles (4.2 km) of severe streambank erosion, or about 5% of the watershed by total stream miles (Krogh et al. 2008; Gassman et al. 2008). In addition, more than 95 miles of moderate or severe gully erosion were mapped, corresponding with intermittent channels in fields.

Biotic Condition

Aquatic, semi-aquatic, and riparian-dependent species—whether plant, invertebrate, or fish—have evolved life history strategies in response to particular flow regimes (i.e. the pattern of daily, seasonal, and annual flows) and the habitat conditions created by them (Lytle and Poff 2004). Numerous empirical studies have demonstrated that the structural and functional organization of species assemblages in lotic systems varies across a gradient of hydrological stability (Horwitz 1978, Schlosser 1985, Bain et al. 1988, Aadland 1993, Poff and Allan 1995, Frenzel and Swanson 1996). Some species are adapted to stable flows, whereas others can tolerate extreme fluctuations in flow (Schlosser 1985, Bain et al 1988, Poff and Allan 1995, Poff et al. 1997). Lytle and Poff (2004) recently reviewed more than 30 studies of life history, behavioral, or morphological adaptations to flow variability in fish and aquatic invertebrates. Specific behavioral, physiological/morphological, or life history adaptations are designed to cope and/or exploit both the magnitude and pattern of variability as well as the predictability of the disturbance (Figure 2.17). In groundwater-dominated systems, for example, species tend to be adapted to relatively stable flows and thermal regimes. In systems where flows may be seasonably variable but the pattern of annual or seasonal variation is fairly predictable, *fluvial specialists* may time life history events (e.g., emergence, spawning, feeding, or migration) to coincide with or avoid annual high and low flow events (Poff et al. 1997, Lytle and Poff 2004). Where flow regimes are both highly variable and somewhat unpredictable, adaptations are needed to increase the probability of egg or larval survival. For example, fish species from hydrologically variable sites often have

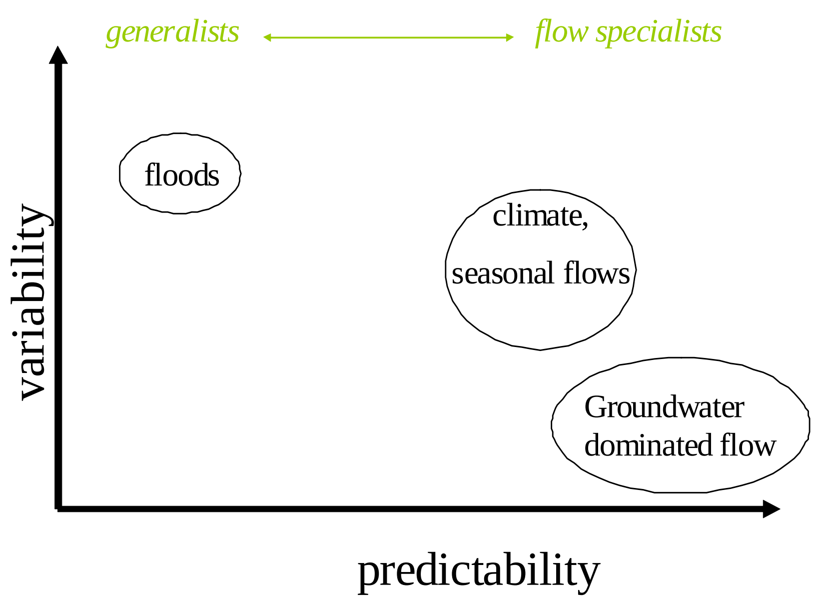


Figure 2.16. Influence of flow disturbance regime (predictability vs. variability of flow and temperature) on life history adaptation of aquatic organisms.

generalized feeding strategies, and are more tolerant to silt (Poff and Allan 1995). Larval fish of several species are also relatively mobile just after hatching to cope with high flows, and the eggs of some species are semi-buoyant to avoid shifting sediments during high flows (Poff et al. 1997).

Benthic Macroinvertebrate Assemblage Composition

There are six sites in the Boone River watershed that are part of the US EPA's Regional Environmental Monitoring and Assessment Program (REMAP) in Iowa (Rowe and Pierce 2006), a continuation of Iowa DNR's 1994-2003 IDNR biological assessment of wadeable streams (Wilton 2004) and also part of the USEPA National Wadeable Streams Assessment. The six sites are White Fox Creek, Otter Creek at Holmes and Goldfield, the Boone River at Webster City and Bells Mill Park, and Drainage Ditch 49. Figure 2.18 depicts the results of mean benthic macroinvertebrate IBI scores by subwatershed. BM-IBI scores for the Boone of 35-70 were slightly favorable compared to other sites throughout the Des Moines Lobe, with several sites in "Good" condition. The statewide stream assessment conducted by Rowe and Pierce (2007) found that 85% of streams in western Iowa were in fair to poor condition, while streams of the Des Moines Lobe were mostly "fair". Diversity and condition of the lower Boone River is slightly better than that of the upper watershed, primarily due to the greater amount and diversity of habitat available.

Biological Condition Rating	Benthic Macroinvertebrate Index of Biotic Integrity (BMIBI)	Fish Index of Biotic Integrity (FIBI)
Poor	0 – 30	0 – 25
Fair	31 – 55	26 – 50
Good	56 – 75	51 – 70
Excellent	76 – 100	71 - 100

Negative relationships were observed between BM-IBI scores and several habitat and water quality variables at REMAP sites within the Boone watershed, including nitrate + nitrite, dissolved oxygen (DO), and HQI score (Figure 2.19a-c). Relationships were less clear for F-IBI scores (Figure 2.20 d-f), though a negative slope was also observed for the relationship between F-IBI and HQI (Figure 2.20f). Although fish IBI scores in the Boone did not show a strong response to nutrient concentrations, Heiskary and Markus (2003) found that fish index of biotic integrity (IBI) scores throughout Minnesota were found to be inversely correlated with summer-mean TP.

In addition to the negative relationships between fish IBI reported in Figure 2.19d-e, our analysis showed that fish IBI score was negatively correlated with cropland in the riparian buffer. Riparian habitat coded as "bluff" in the REMAP study was also negatively correlated with benthic macroinvertebrate IBI score. Analysis of physical habitat data from statewide wadeable stream assessment (Wilton 2004) showed that mussel species richness and % gravel substrates were both significantly positively related to stream order. Gravel was negatively correlated with stream depth and percent fines.

Although there are relatively few repeat sample sites within the REMAP study, White Fox Creek is the exception, with an 8 year time series from 1994-2001. Figure 2.20 depicts Habitat Quality Index (HQI), Fish IBI (F-IBI), and Benthic Macroinvertebrate (BM-IBI) scores for repeat samples taken at White Fox Creek over time. The HQI is based on Barbour and Stribling (1991), ranging from 0 (poor) to 180 (maximum). The White Fox Creek scores of 110-130 are moderate, and higher than other sites in the watershed.

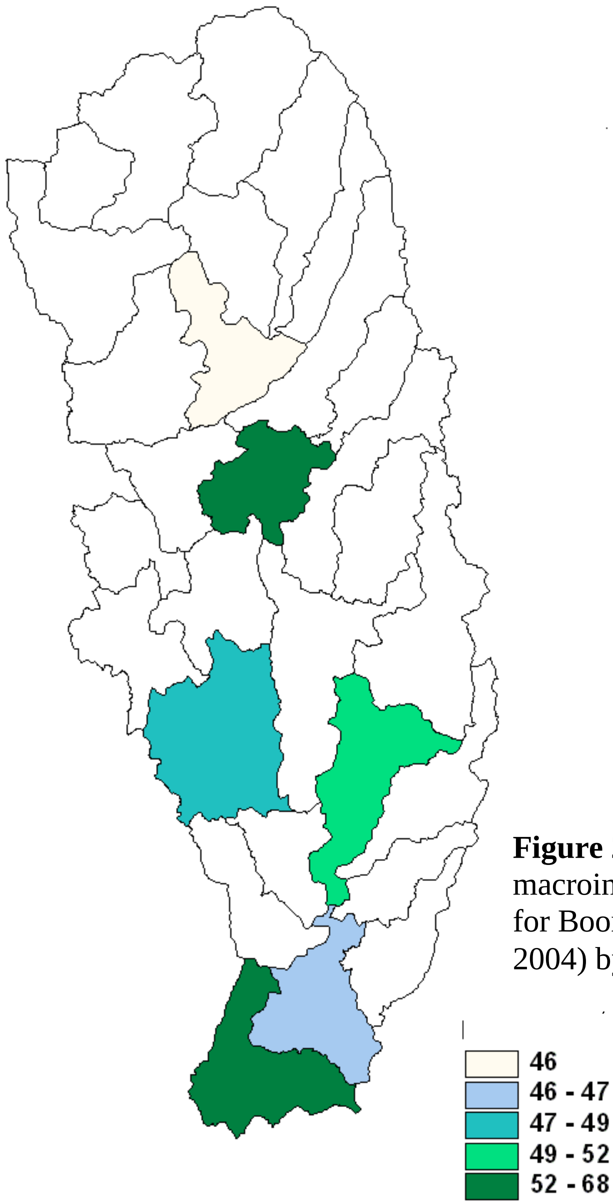


Figure 2.17. Mean benthic macroinvertebrate IBI scores for Boone River sites (Wilton 2004) by subwatershed.

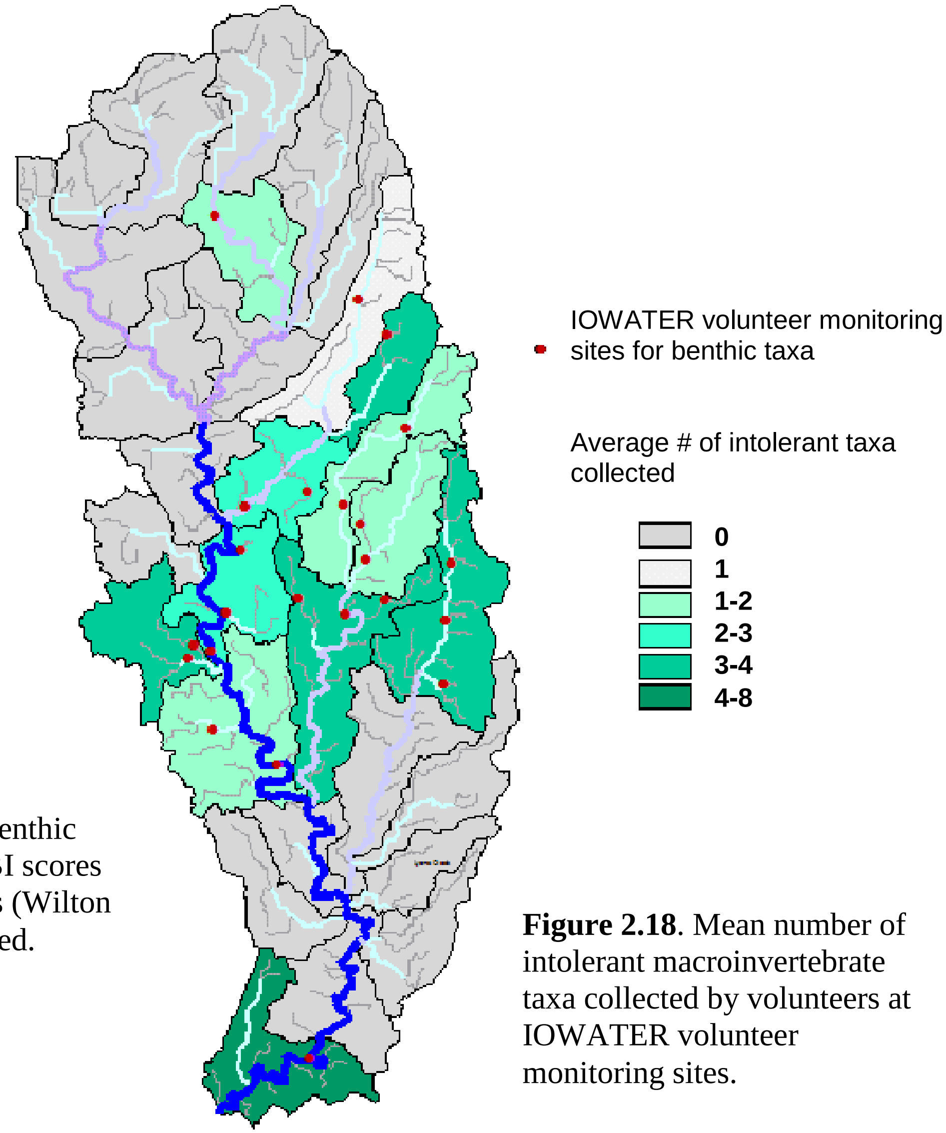


Figure 2.18. Mean number of intolerant macroinvertebrate taxa collected by volunteers at IOWATER volunteer monitoring sites.

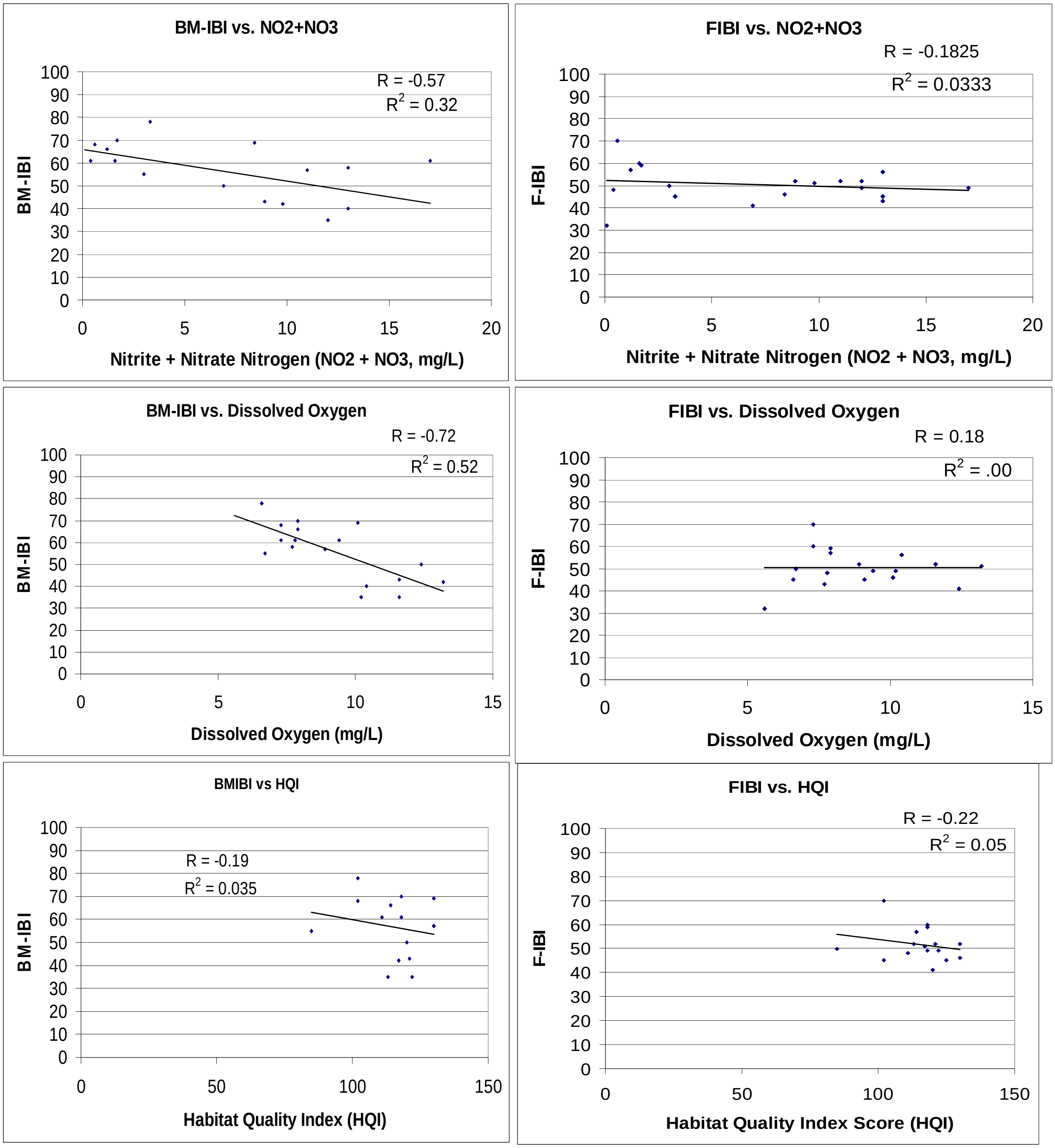


Figure 2.19. Relationship between water quality variables, BM-IBI, and FIBI for sites in the Boone River watershed measured in the REMAP study (Wilton 2004).

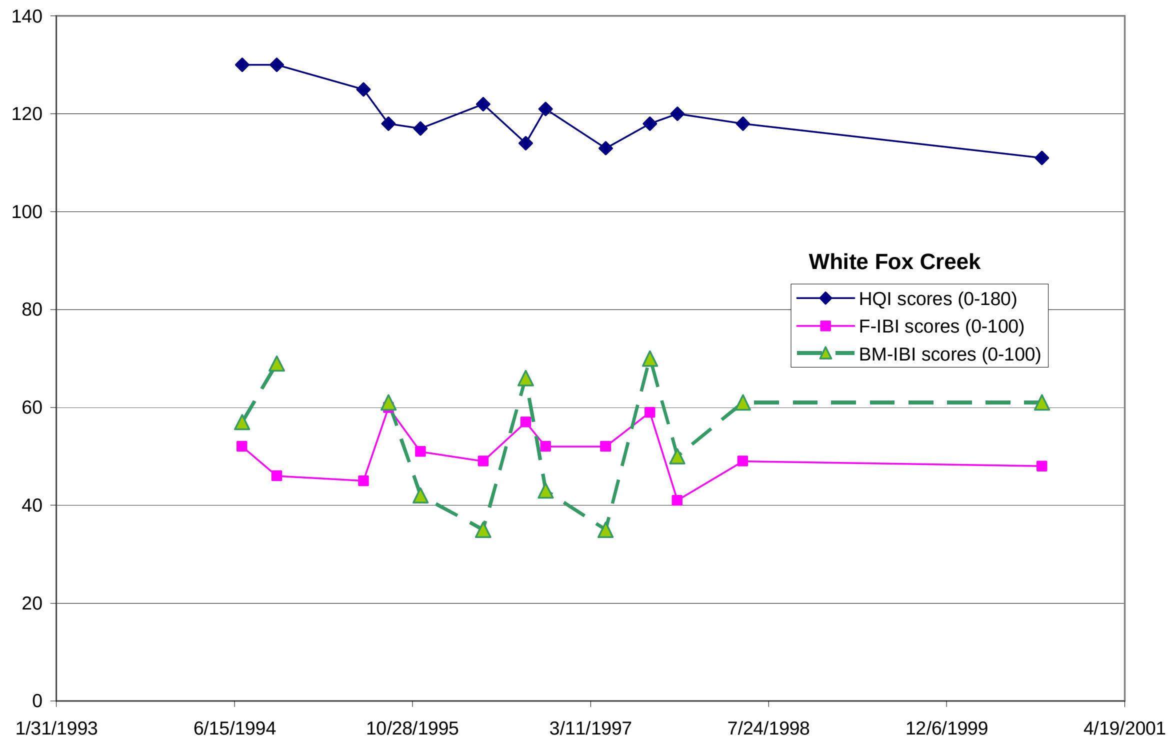


Figure 2.20. HQI, F-IBI, and BM-IBI scores for White Fox Creek 1994-2001.

Additional biological assessment data collected by IOWATER volunteers at Boone River sites was obtained but not analyzed by Neugarten and Braun (2005). Because IOWATER reports generally assign organisms only to broad taxonomic categories, and do not identify organisms to the level of genus or species as required by the IBI metrics, IBI scores cannot be calculated. However, Figure 2.19 attempts to convey a rough estimate of site quality by subwatershed, integrating IOWATER site report information on water clarity, condition, and number of sensitive taxa.

No additional macroinvertebrate sampling data have been obtained since Neugarten and Braun (2005). The Ecological Assessment recommended developing a methodologically consistent sampling design for future macroinvertebrate monitoring throughout the Boone River watershed to establish a baseline of data and facilitate comparative analyses.

Freshwater Mussel Assemblage and Composition

As a group, freshwater pearly mussels are among the most endangered freshwater taxa in North America, with 40-60% of species listed in some category of imperilment. Mussel declines have been substantial in freshwater aquatic systems throughout the continent. Of 297 species that comprised the North American freshwater fauna north of Mexico, 19 are presumed extinct, 44 species listed or proposed as federally endangered, and another 69 species that may be endangered (Bogan 1993, Master et al. 2000). Several species listed as “endangered” are believed functionally extinct (individuals of a species surviving but not reproducing). The majority of North American unionid bivalve extinctions are linked to impoundment and inundation of riffle habitat in major rivers such as the Ohio, Tennessee and Cumberland and Mobile Bay Basin, that historically also had the richest and most endemic fauna. However, throughout North America, other causes of unionid declines include hydrologic alteration, commercial exploitation, exotic zebra mussels, local losses of obligate host fish, sedimentation and siltation, and various types of industrial and domestic pollution. In Iowa, high rates of mussel disappearance are probably linked to large-scale watershed alterations that are not as amenable to short-term solutions or local restoration projects. In some cases, recreational impacts may even be significant, although this has rarely been studied (Watters 2000).

In the Boone River and tributary streams, mussel species richness has declined sharply since initial surveys (Hoke 2004, Poole 2005). The declines are of concern for regional biodiversity in Iowa streams because the Boone River system has been identified in the past as having had some of the best mussel habitat in the state (Hoke 2004). Although there are no comparable large-scale surveys elsewhere in the literature, these declines are consistent with other reports of local population declines in freshwater mussels (e.g., Master et al. 2000, Watters 2000, Strayer et al. 2004). The Boone River was flagged for inclusion in the UMRB biodiversity assessment on the basis of the mussel community. At the time, local experts noted that chronic nitrate levels posed a threat to long-term persistence of species and populations.

Tables 2.5 and 2.6 show the results of three mussel surveys conducted in the Boone since 1982 (Hoke 2004, Frest 1987, and Poole 2005). The Poole 2005 survey appears to have confirmed the declining condition of mussels in watershed. Live individuals were found representing only 4 of 13 species historically present in the watershed. Included in the list of species of which live individuals were not found in 2005 is one of TNC's UMRB aquatic species targets, the black sandshell mussel. Figure 2.20 also presents a map of the 2005 mussel survey results. The elktoe (*Alasmidonta marginata)*, found on the Boone River near Webster City in 1937, has not been reported in any of the four survey reports in nearly 25 years and is likely extirpated from the system (Hoke 2004). Five additional mussel species have not been recorded (living or in shell collections) since 1982 and may also be extirpated from the Boone River. The majority of living individuals were observed on main stem of the Boone River. However, two tributaries, White Fox Creek and Otter Creek, appear to have had had diverse populations of mussels historically based on shell collections, but had fairly low species diversity in the 2005 survey. Poole (2005) concluded, based on few live individuals and reduced species diversity at individual sites, that conditions on the Boone River (and its tributaries) are unfavorable for mussel persistence.

Exploitation (usually by humans for buttons and other commercial uses) has been cited as a factor in some mussel population declines, but generally only as a partial contributing cause, and one that operates locally over small patches. Ecologically, mussels are an important food source for raccoons, muskrats, and river otters. Given the recent reintroduction of river otter to the state, some watershed residents have hypothesized that these mammals may be impacting mussel populations along the river. There is, however, no evidence of this, and exploitation by meso-predators has rarely been cited in the literature as a factor contributing to mussel declines. The most frequently cited factors in mussel declines are altered hydrology and sediment regimes (Watters 2000, Strayer et al. 2004).

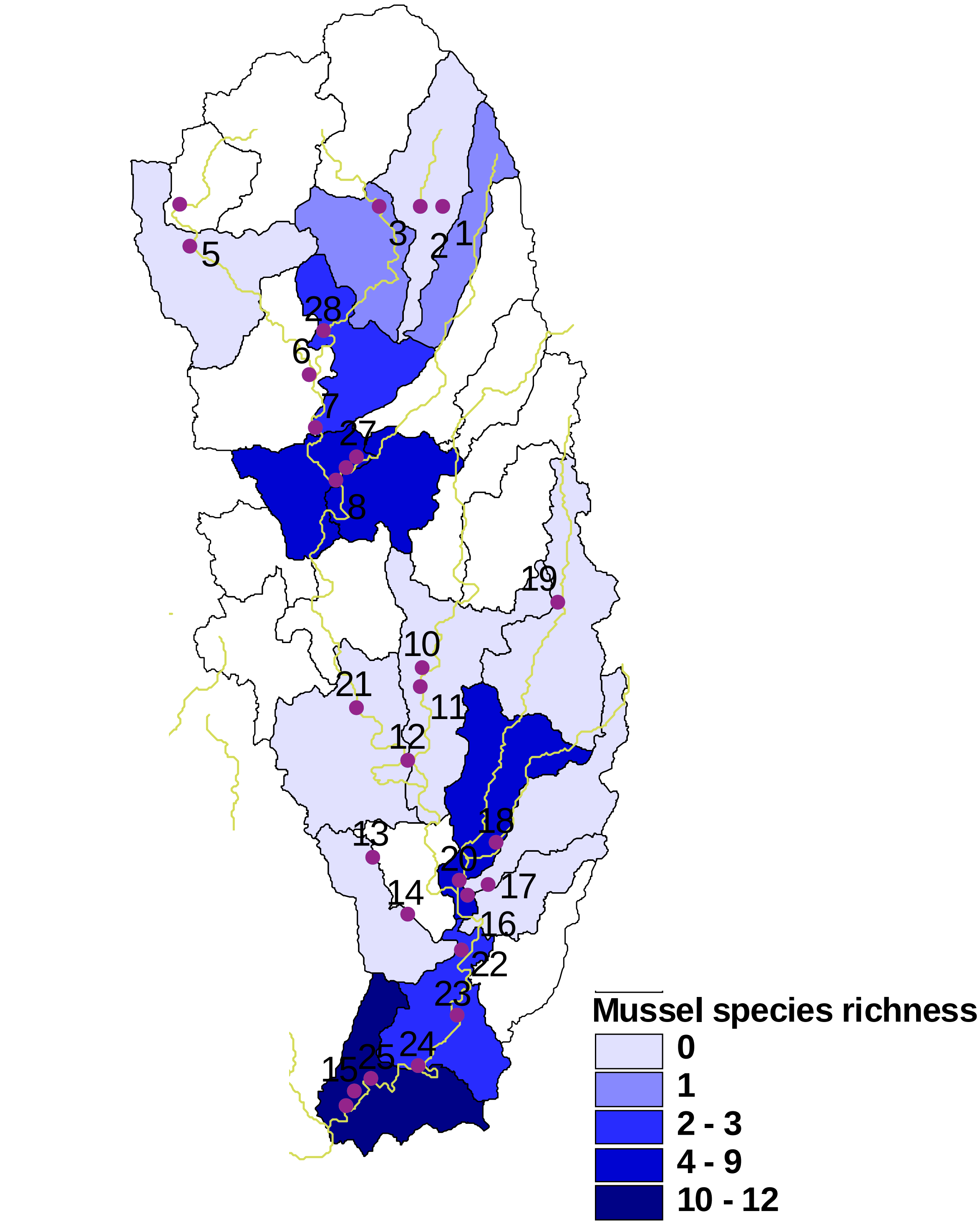


Figure 2.21. Mussel species richness (including both living mussels and shell material); plus locations of 2005 mussel sampling by site number.

Table 2.5. Mussel species status in the Boone watershed in successive surveys (modified from Poole 2005).

Species	Common Name	Hoke 1982	Frest 1984-85	Arbuckle & Downing 1998-99	Poole 2005	Tolerance (Kopplin 2002)	Status in Iowa**
Species							
**Actinonaias ligamentina*	Mucket	□	□	□	✓	Intermediate	Vulnerable
**Lampsilis cardium* (formerly *L. ventricosa)*	Plain pocketbook	✓	✓	✓	✓	Intermediate	Vulnerable
Lasmigona complanata	White heelsplitter	✓	✓	✓	✓	Tolerant	Imperiled
Quadrula pustulosa	Pimpleback	✓			✓	?	Imperiled
Amblema plicata	Threeridge	✓	✓	✓	□	Tolerant	Not Ranked
Leptodea fragilis	Fragile papershell			✓	□	Intermediate	Imperiled
Potamilus ohiensis	Pink papershell			✓		?	Not Ranked
Utterbackia imbecillis	Paper pondshell			□		?	Not Ranked
Potamilus alatus	Pink heelsplitter			□		?	Imperiled
Elliptio dilatata	Spike	□	□		□	Intermediate	Imperiled
Fusconaia flava	Wabash pigtoe	✓	✓	✓	□	Intermediate	Imperiled
**Lampsilis siliquoidea* (formerly *L. radiata luteola)*	Fat mucket	✓	✓	□	□	Tolerant	Not Ranked
Lasmigona costata	Fluted shell	□	□		□	Intermediate	Imperiled
Ligumia recta	Black sandshell	✓	□	✓	□	Intermediate	Not ranked
Pyganodon grandis (formerly Anodonta grandis)	Giant floater	✓	□	□	□	Tolerant	Imperiled
Strophitus undulatus	Strange floater	✓	✓		□	Tolerant	Threatened
Anodontoides ferussacianus	Cylindrical papershell or cylinder	✓		□		Intermediate	Threatened
Lampsilis teres	Yellow or slough sandshell	□				?	Endangered
Lasmigona compressa	Creek heelsplitter	□				Intermediate	Threatened
Toxolasma parvus	Lilliput	□	□			?	Critically Imperiled
Uniomerus tetralasmus	Pondhorn	□	□			?	Not Ranked
Pleurobema sintoxia	Round pigtoe		□				?
**Alasmidonta marginata*	Elktoe	(From 1937 collection)				Intolerant	Vulnerable
TOTALS	**Live (✓)**	**10**	**6**	**7**	**4**		
	Recently dead (□)	**7**	**5**	**6**	**9**		
	TOTAL	**17**	**11**	**13**	**13**		

Table 2.6. Mussel species richness and CPUE in the Boone River by site (Poole 2005).

SITEID	STREAM NAME	Species richness[1]	Mussel CPUE[2]
1	East Branch Boone River	1	0
2	Middle Branch Boone River	0	0
3	Boone River	1	0
8	Otter Creek	6	0
9	Otter Creek	2	0
10	Eagle Creek	0	0
11	Eagle Creek	0	0
12	Eagle Creek	0	0
13	Brewers Creek	0	0
14	Brewers Creek	0	0
15	Prairie Creek	0	0
16	Lyons Creek	1	0
17	Lyons Creek	0	0
18	Buck Creek	0	0
19	White Fox Creek	0	0
20	White Fox Creek	7	0.67
27	Otter Creek	0	0
	Average for the Upper Zone	**1.1**	**0.039**
4	Prairie Creek	0	0
5	Prairie Creek	0	0
6	Boone River	0	0
7	Boone River (Oakdale Recreation Area)	9	6
21	Boone River (Troy Safety Rest Area)	0	0
22	Boone River (Briggs Woods Access)	10	3.6
23	Boone River (Albrights Access)	3	1.2
24	Boone River (Tunnel Mill)	12	0
25	Boone River (Bells Mill)	2	0
26	Boone River (Boone Forks Wildlife Area)	0	0
28	Boone River - Helmke Wildlife Area	3	0
	Average for the Lower Zone	**3.5**	**1.0**

[1]including living and shell records
[2]Catch Per Unit Effort, defined according to timed searches.

Neugarten & Braun (2005) suggested a review of fish host population data to see whether loss of fish hosts limits mussel persistence in the Boone River and its tributaries. All of the fish hosts listed in the literature and in Table 2.7 are still present in the Boone River in more recent surveys, particularly in the lower reaches that have historically supported the most diverse mussel beds. However, given the patchiness of mussel beds and the complexity of mussel reproductive biology—i.e. the intricate and complex relationship they have developed with fish hosts-- relatively small changes in habitat use or behavior by fish hosts may have significant impacts. Furthermore, host compatibility has been shown to vary with temperature and other ecological factors that may have been altered in subtle ways (Strayer et al. 2004). Host fish use by Iowa mussels is the subject of current research led by Kelly Poole and John Downing at the Iowa State University Cooperative Fish and Wildlife Research Unit.

As a sensitive and long-lived species, mussels are an excellent long-term barometer of overall watershed health. However, the considerable uncertainty still surrounding mussel life cycles, life history, and reproductive ecology, and the likelihood that there are multiple factors contributing to their declines suggests that further research and consultation is needed with mussel ecologists to understand mussel population dynamics and status in the Boone River watershed. Research and monitoring should focus on identifying the most important mechanisms driving stress and declines of beds. Further studies of mussel life history, reproduction, local population dynamics, and mortality may provide excellent research opportunities for university undergraduate or graduate students. Permanent monitoring sites should be established on the Boone River mainstem, White Fox Creek, and Otter Creek to assess community changes over time. Given that headwater streams and tributaries have been identified as important habitats within the framework of the Upper Mississippi River System Network with the UMRS mussel recovery plan, there may be additional opportunities for funding and cooperative research on the Boone with federal and state biologists.

Fish and Assemblage Composition and Health

The Boone River supports at least 55 fish species, including five nonnative species (Neugarten and Braun 2005). In agricultural areas of the Midwest, historic fish assemblages were probably similar to present communities; however, many species have undergone documented shifts in distribution, range, and abundance (TerHaar and Herricks 1989). The introduction of game fish species, alterations of physical habitat, and the loss of endemics has resulted in homogenization of fish communities throughout North America (Rahel 2000, 2002)

Fish sampling data records for the Boone River are available from multiple sources, yielding a dataset with 2656 sample records from 194 different sample locations throughout the Boone covering a period from 1932 to 2002 (median sample year = 1999). From presence/absence data collected over many years using a variety of different sampling methods for different purposes, it is not possible to establish statistically significant evidence of fish community change. However, there have not been significant changes in species richness or diversity. Table 2.6 shows no apparent changes in species diversity pre- and post - 1990.

Lower watershed sites (n=78) have a mean of 12 species, maximum of 37, minimum of 3, and standard deviation of 6 species. Upper watershed sites (n=142) have a Mean of 12, maximum of 34, and min of 3, with a standard deviation of 6. Total number of species sampled in the watershed is 60.

All of the species present in the Boone that have been classified as “Intolerant” of disturbance and habitat degradation are present in recent surveys (since 1990). The composition of the 10 most commonly detected species has changed, and a few species that appear to have been numerous based on their frequency of appearance in surveys prior to 1990 have been relatively less common in recent sampling. These include the silver redhorse and highfin carpsucker. However, it is not possible to determine without additional monitoring whether this is a real trend, or an artifact of sampling, as sampling methods are not statistically comparable.

Current condition should be viewed in context of historical ecological changes. Contemporary presence/absence data in relation to early historical collections and surveys in the Midwest often show few extirpations, although there may be moderate range reductions for many intolerant species. Sensitive or intolerant species have all seen shifts in distribution and abundance despite being locally abundant in parts of their range (Menzel et al. 1984, Rahel 2000). Sensitive stream fishes are often limited to small populations present only in some subwatersheds.

Fish communities tend to be the most diverse and stable where high-quality habitat—i.e. complex heterogenous riffle: pool morphology, coarse substrate, adequate cover-- is available (Rowe and Pierce 2006, 2007). Streambed composition is important to fish species diversity (Schlosser 1985). Many stream fishes dependent on riffle habitats with gravel or cobble substrates that remain well-aerated and free of sediment for spawning and feeding. Fish indices of biological integrity (IBI) scores often correlate with river gradient, perhaps because of the ability of higher velocity waters to maintain DO, flush fine sediment and keep important interstitial habitats in coarse substrates free of sediment and silt. Important riffle habitats with cobble or boulder substrates are most abundant in the lower reaches.

IDNR data report 18 documented fish kills in the Boone River watershed since 1981 (Figure 2.22). Documented causes, although often reported as unknown, included low DO and several ammonia spills. Fish kill data provide clues to significant events that may be influencing aquatic communities that otherwise may not detected in routine water quality sampling. Fish and other mobile aquatic organisms often recover quickly from events due to their ability to seek refugia and then recolonize suitable environments. However, long-lived, sessile (nonmobile) organisms such as mussels may recover more slowly. Successful recruitment for some species of mussels may only occur under infrequent sets of conditions (e.g. 2-5 years; Strayer et al. 2004).

Table 2.7. Fish species sampled in the Boone, sorted by distribution (number of subwatersheds).and descending frequency of occurrence.

COMMON NAME	**SCIENTIFIC NAME**	**TOL[1]**	**Years present**	**Sites present**	**Sites (pre 1990)**	**Sites (post 1990)**	**Sub water-sheds where present**	Sub water sheds <1990	Sub water sheds >1990
Common shiner	*Luxilus cornutus*		20	170	31	161	26	12	26
Bluntnose minnow	*Pimephales notatus*	T	18	155	22	153	26	11	26
Bigmouth shiner	*Notropis dorsalis*		17	131	20	131	25	11	25
Fathead minnow	*Pimephales promelas*	T	17	90	19	80	25	10	25
Creek chub	*Semotilus atromaculatus*	T	18	136	23	132	24	11	24
White sucker	*Catostomus commersoni*		18	98	25	92	24	24	24
Johnny darter	*Etheostoma nigrum*		17	104	17	104	23	23	23
Central stoneroller	*Campostoma anomalum*	T	17	90	15	91	22	7	22
Blacknose dace	*Rhinichthys atratulus*	T	12	79	8	83	22	7	22
Spotfin shiner	*Cyprinella spiloptera*	T	17	91	14	91	21	8	21
Sand shiner	*Notropis stramineus*		18	103	20	101	21	21	21
Brassy minnow	*Hybognathus hankinsoni*		12	63	11	57	21	21	21
Hornyhead chub	*Nocomis biguttatus*	I	15	85	11	88	19	6	19
Emerald shiner	*Notropis atherinoides*		8	43	5	38	18	18	18
Rosyface shiner	*Notropis rubellus*	I	14	41	9	45	15	4	15
Black bullhead	*Ameiurus melas*	T	9	44	16	29	14	5	14
Common carp	*Cyprinus carpio*	T	16	56	29	39	13	5	13
Green sunfish	*Lepomis cyanellus*	T	17	52	17	52	13	6	13
Northern pike	*Esox lucius*	I	12	27	10	18	13	5	13
Largemouth bass	*Micropterus salmoides*	T	8	24	1	31	13		13
Northern rock bass	*Ambloplites rupestris*	I	18	39	17	35	12	3	12
Blackside darter	*Percina maculata*		16	35	13	35	11	5	11
Brook stickleback	*Culaea inconstans*	I	4	21	2	19	11	1	11
Smallmouth bass	*Micropterus dolomieu*	I	20	43	20	38	10	6	10
Orangespotted sunfish	*Lepomis humilis*		10	29	11	21	10	5	10
River redhorse	*Moxostoma carinatum*		3	17		17	10		10
Northern hog sucker	*Hypentelium nigricans*	I	19	35	18	33	8	5	8
Quillback carpsucker	*Carpiodes cyprinus*	T	11	21	14	12	8	5	8
Shorthead redhorse	*Moxostoma macrolepidotum*		15	29	17	20	6	8	8
Bluegill	*Lepomis macrochirus*		9	22	6	17	3	8	8

COMMON NAME	SCIENTIFIC NAME	TOL[1]	Years present	Sites present	Sites (pre 1990)	Sites (post 1990)	# subwater-sheds where present	Subwater sheds <1990	Subwater sheds >1990
Topeka shiner	*Notropis topeka*		6	16	4	12	8	1	8
River shiner	*Notropis blennius*		2	7		7	7		7
Stonecat	*Noturus flavus*	I	13	22	11	21	6	4	6
River carpsucker	*Carpiodes carpio*		12	17	10	8	6	4	6
White crappie	***Pomoxis annularis***	**T**	**5**	**9**	3	6	**6**	2	6
Golden redhorse	*Moxostoma erythrurum*		17	28	18	27	4	7	7
Channel catfish	*Ictalurus punctatus*		15	34	25	14	3	6	6
Slenderhead darter	***Percina phoxocephala***		**8**	**13**	10	5	3	**5**	5
Fantail darter	*Etheostoma flabellare*		10	14	3	23	2	5	5
Suckermouth minnow	*Phenacobius mirabilis*		16	27	10	28	2	6	6
Walleye	***Stizostedion vitreum***		**6**	**10**	5	6	2	**3**	3
Slender madtom	*Noturus exilis*		6	7	1	8	2		2
Highfin carpsucker	***Carpiodes velifer***	**I**	**5**	**6**	5	2	**2**	1	2
Flathead catfish	*Pylodictis olivaris*		6	6	4	3	2	2	2
Yellow perch	*Perca flavescens*		5	5	1	7	2		2
Banded darter	*Etheostoma zonale*	I	2	4	2	2	2	1	2
Silver chub	***Macrhybopsis storeriana***		**2**	**3**		3	2	**2**	2
Yellow bullhead	*Ameiurus natalis*	T	2	2		2	2		2
Gizzard shad	*Dorosoma cepedianum*	T	2	2		2	2		2
Bullhead minnow	*Pimephales vigilax*		2	2		2	2		2
Bigmouth buffalo	***Ictiobus cyprinellus***		**8**	**13**	9	5	1	**3**	3
Silver redhorse	***Moxostoma anisurum***		**8**	**12**	11	2	1	**2**	2
Freshwater drum	***Aplodinotus grunniens***		**4**	**7**	1	6	1	**4**	4
Black crappie	*Pomoxis nigromaculatus*		6	7	3	6	1	3	3
Tadpole madtom	*Noturus gyrinus*	I	3	3	2	1	1		1
Red shiner	*Cyprinella lutrensis*	T	2	2	1	1	1		1
Warmouth	***Chaenobryttus gulosus***		**1**	**1**	1		1	**1**	
Smallmouth buffalo	*Ictiobus bubalus*	I	1	1	1		1	1	
Speckled chub	*Macrhybopsis aestivalis*	I	1	1		1	1		1
Spottail shiner	***Notropis hudsonius***	**I**	**1**	**1**	1		**1**		

TOL = tolerance (I=intolerant, T=tolerant) in the literature. Bold = species primarily occurs in the Lower watershed (>= 70%) ; Highlighted – species that occur only in the Lower watershed

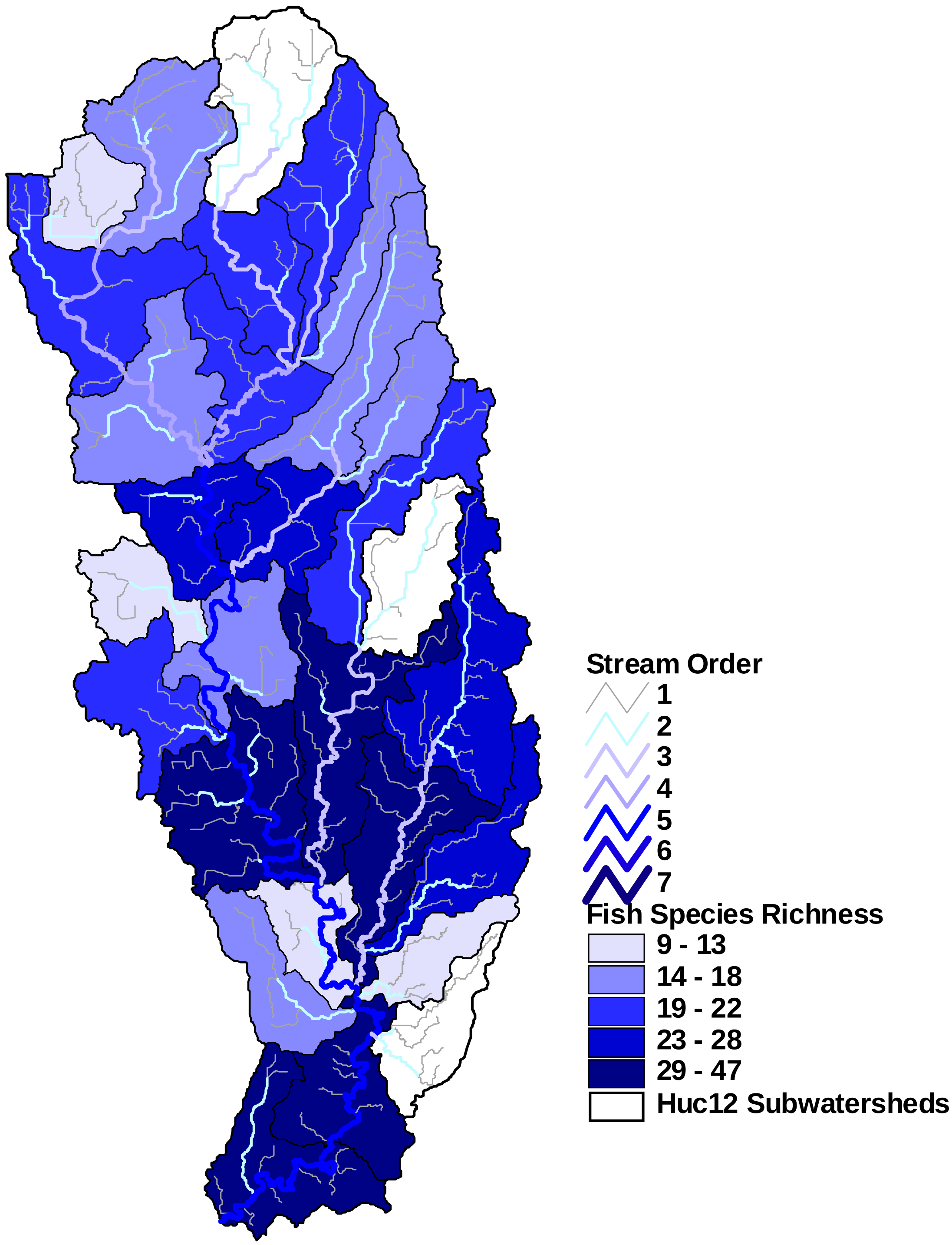


Figure 2.22. Fish species richness by 12 digit watershed.

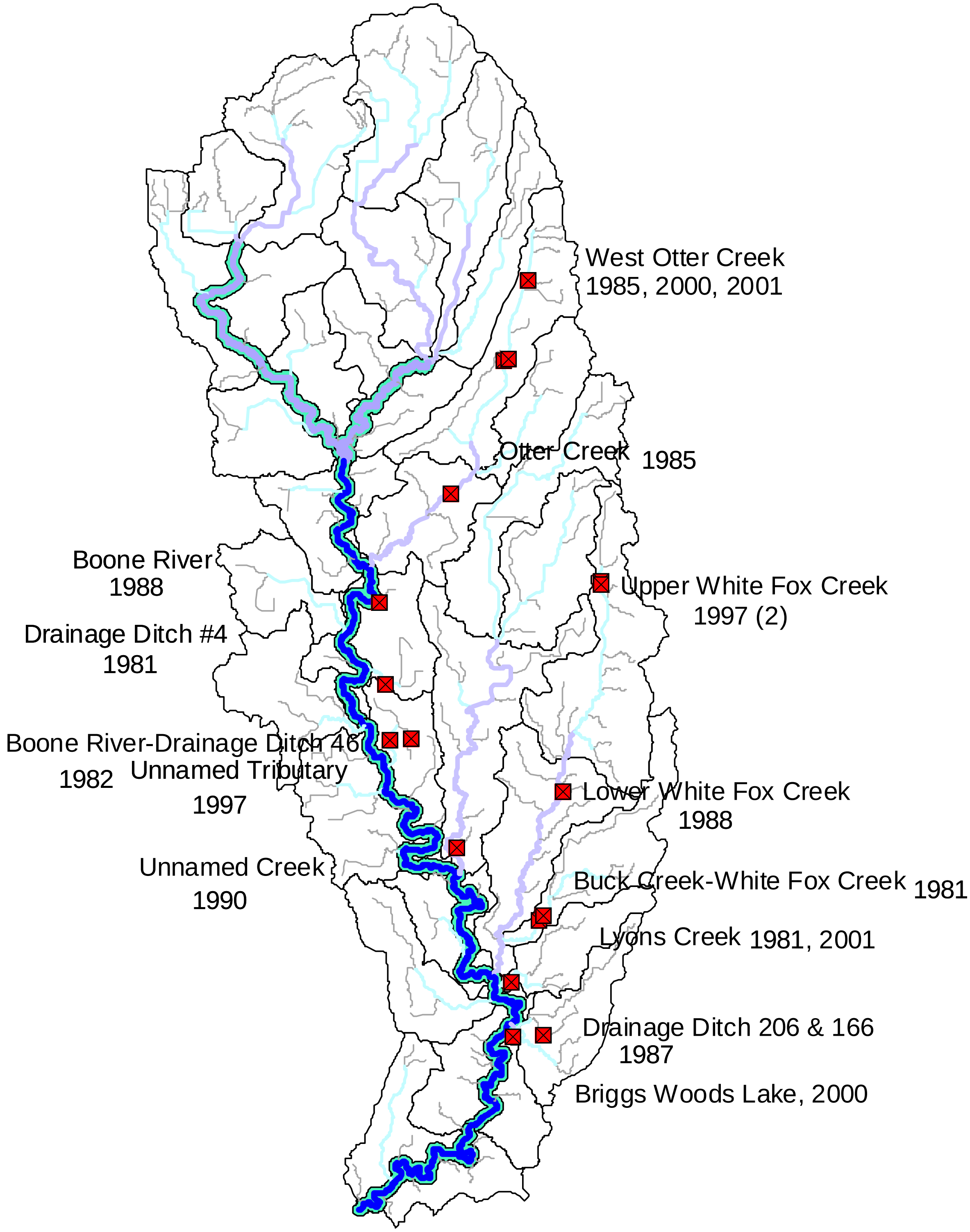


Figure 2.23. Date and location of fish kills reported to IDNR 1985-2005.

Topeka Shiner

Nongame fishes have been possibly more impacted by stream degradation than game fishes. Bailey (1956) speculated that "it is doubtful that any other state has experienced such extensive reduction in its original fish fauna." A list of endangered and threatened species in Iowa identifies 22 species as extirpated, threatened, endangered, or of special concern—out of roughly 150 species native to the state (IDNR 2008).

In the Boone River, the Topeka shiner (*Notropis topeka)* is the only federally-listed fish that has been recorded occurring in the Boone since 1939. Although the NRCS Rapid Watershed Assessment listed 3 state-listed fish as potentially occurring in the Boone River based on county level occurrence data (including blacknose shiner *Notropis heterolepis*, orangethroat darter, *Etheostoma spectabile*, and western sand darter, *Ammocrypta clara*), there are no records of these species from fish sampling records in the Boone River.

Several areas of the Boone River watershed have been designated as critical habitat for the Topeka Shiner under the Federal Endangered Species Act. Areas designated as critical habitat are areas occupied by the species or are short segments that provide critical links between habitats. Appendix A shows known and potential critical habitat for Topeka shiners in Iowa, considered essential for the conservation of the Topeka shiner that may require special management and protection.

Designation of an area as critical habitat through the federal regulatory process does not set up a preserve or refuge and has no specific regulatory impact on landowners' actions on lands that do not involve federal agency funds, authorization, or permits. Identification of critical habitat is intended to help guide priorities for conservation actions in the affected areas. Fisheries biologists with Topeka shiner populations in their management areas in Iowa work with the USFWS on critical habitat and habitat restoration on private land. For example, in 2004-2005, endangered species recovery funds paid for the design and construction of a habitat restoration project for the Topeka shiner along Cedar Creek in Greene County, Iowa. The project, a collaboration between USFWS (Partners for Fish and Wildlife Program, Rock Island, Illinois Field Office), the Iowa Natural Heritage Foundation and two private landowners, was designed to restore the hydrology of an oxbow in the Cedar Creek floodplain, provide permanent off-stream réfugia and potential spawning habitat for Topeka shiners, and reconnect the downstream end of the oxbow to Cedar Creek to allow Topeka shiners to disperse into the watershed (Mirando-Castro 2006). Similar projects are underway in the Cedar Creek and West Buttrick Creek watersheds in Greene and Calhoun counties, as a result of another partnership between private landowners and staff at USFWS, TNC, IDNR, and Greene County Soil and Water Conservation District. As noted in the 2007 Iowa Wildlife Action Plan, Iowa DNR staff are represented on the federal Topeka shiner recovery team that includes representatives from the U.S. Fish and Wildlife Service, National Park Service, South Dakota State University, University of Minnesota, private consultants and staff from state natural resources agencies of Kansas, South Dakota, and Minnesota.

Neighboring states South Dakota and Missouri have also developed recovery plans (Shearer 2003). The Boone River Watershed was identified by IDNR as a LIP priority area because of Topeka shiners. However, we are unaware of any current activities focusing on Topeka shiner recovery in the Boone River watershed.

Aquatic Mammal Population Status

Aquatic mammals proposed as key ecological attributes for the Boone River are the beaver, *Castor canadensis,* due to the beaver's keystone role as an ecosystem engineer (Wright and Jones 2006) and the river otter, *Lutrensis canadensis,* the largest mammal state listed as a species of special concern. Surveys have not been conducted specifically in the Boone River to assess populations and status of beaver or river otter. However, Clark (2006) reported that estimated population density of river otters at between 0.54-1 per 10 mi^2 for the Lower Boone, and 0-0.55 per 10 mi^2 for the upper Boone. Across the entire state, otter populations have made a substantial recovery from reintroductions and subsequent expansions. Factors important to sustaining river otter populations include the availability of connected, diverse riparian habitats (i.e. wetlands and wooded areas along streams and rivers), as well as high water quality capable of sustaining fish populations. Restoration of otter in Iowa is considered to have been successful, leading to a slowly increasing population, and IDNR has recently approved managed harvest of the population through licensing of trapping (Clark 2006).

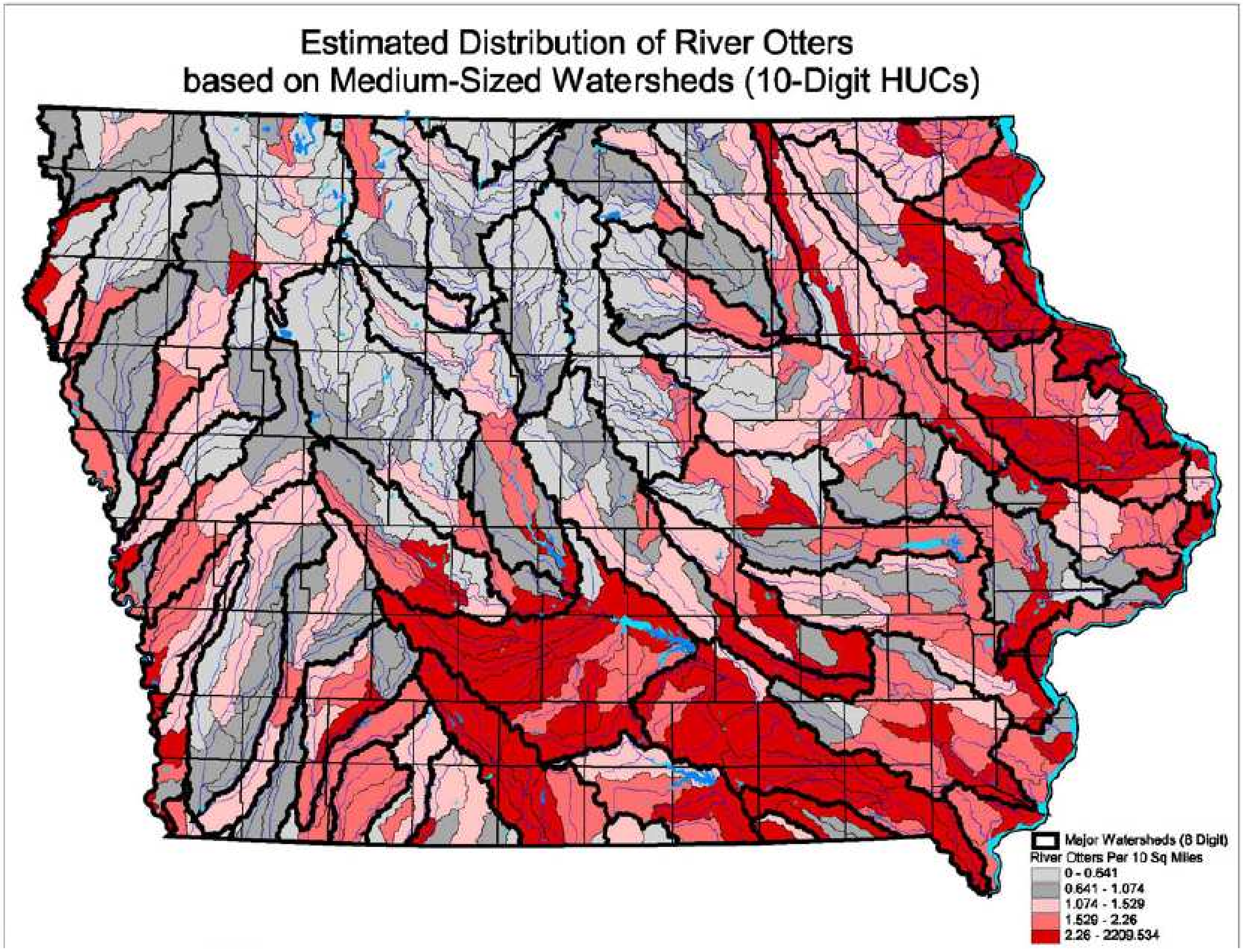


Figure 2.24. River otter distributions in watersheds throughout Iowa. [reprinted from Clark 2006]

Priority Resource Concerns, threat assessment and situation analysis

As part of the Rapid Watershed Assessment commissioned by NRCS, watershed experts and residents were asked to identify priority resource concerns in the Boone River watershed. Responses were compiled based on interviews with the seven NRCS District Conservationists, ten Iowa DNR experts, and two different public meetings where participants completed surveys, as well as surveys distributed to others within the DNR. Priority resource concerns are summarized in Table 3.1.

Table 3.1. Priority Resource Concerns
1) Point source pollution
• Municipal discharge
• Facilities
• Unsewered communities
2) Non point source pollution of surface and groundwaters
a. Nitrate
b. Phosphorus
c. Bacteria / *E. coli*
d. Agricultural chemicals
e. Soil erosion and sedimentation
3) Stream and gully erosion
4) Riparian Development
5) Aquatic ecosystem integrity
Land use priorities
6) Row crop
7) Rural Development
a. Developments happening right alongside the river (contaminants, septic)
i. Combined developments
ii. Raw sewage connected to tile lines (failing septic)
iii. Performing septic
b. Fragmentation & aesthetics
c. Farmland loss

Threat Assessment

The conservation action plan methodology developed by TNC characterizes threats to conservation targets as ***stresses*** and ***sources of stress***. A "threat" is actually a combination of a stress and a source of stress (TNC 2000). Stresses can be thought of as the proximate causes that impact the targets directly, whereas sources of stress are the ultimate drivers of the proximate stresses, and may be one or more steps removed from the impact on the targets. Distinguishing between stresses and sources is designed to help lead to effective strategies for addressing critical threats. Separating threats in this way allows for potentially more creative ways to alleviate the stress than simply defining the source of stress as the threat. However, it may also allow identification of conservation actions that effectively treat not just symptoms, but causes. For example, elevated nutrient levels in streams can be thought of as a "stress" driving ecological changes instream, whereas agricultural land use activities resulting in nutrient losses would be considered a "source of stress". Actions might be designed either to address the stress directly (e.g. intercept nutrients before they reach the stream with buffers), and/or eliminate the source of stress (reduce fertilizer applications that lead to nutrient losses on-site). Whether it will be most feasible or cost-effective to address stresses or sources of stress varies depending on the context.

Priority resource concerns for the Boone River have been reorganized into stresses and sources of stress for the purposes of the Conservation Action Plan as follows.

Stresses	***Sources of Stress***
Nutrient loading	Cropping systems (e.g. corn-soybeans) and practices (amount & timing of fertilizer) that result in significant N and P losses in surface runoff and subsurface tile drains
	Application of fertilizer or animal manure at inappropriate times
	Inadequate storage & handling of animal waste / manure from livestock confinement operations
	Leaching and macropore flow into tile drains that discharge directly to surface waters
	Long-term shift towards in “leakier” cropping systems in response to economic drivers, including increased soybeans and increased corn acreage and decreased perennial / grass / CRP cover
	Long-term accumulation of anthropogenic P in landscape sinks such as streambanks and release to surface waters
	Municipal and industrial discharges; unsewered communities
Chemical and bacterial contaminants	Herbicide, pesticide and manure loss from croplands in surface runoff events
	Leaching and macropore flow into tile drains
Stream and gully erosion	Alteration of watershed hydrology; increased baseflow and storm flows associated with increased drainage density Altered stream morphology: destabilized stream channels with high banks that readily release large pulses of sediment
Habitat loss / degradation	Historical alteration of landscape hydrology and aquatic/wetland habitat loss Land use and conversion (especially riparian areas) from natural vegetation to agricultural, suburban or developed uses Stream channelization and artificial surface drainage Water quality and hydraulic impacts of runoff and channel alteration on aquatic habitat quality
Groundwater contamination	Agricultural Drainage Wells
	Surface tile inlets

The final step in the assessment of stresses and sources is a synthesis of the individual stress and
source analyses to identify the critical threats and persistent stresses to the conservation targets.
Critical threats are those highly ranked threats that have an active source of stress. Conservancy planning guidance suggests that highly ranked threats that have an historical source be considered as persistent stresses, if the source component of the threat is no longer active. In such cases, reducing persistent stresses requires restoration strategies.

Putting it All Together

Many of the ten key attributes identified as key ecological attributes (KEAs) for the Upper and Lower Boone are ecologically interrelated. As the major driver of physical form and habitat in streams, the flow regime is the driver for all the other KEAs, from water quality to stream geomorphology (Figure 3.1, 3.2). Changes in basin hydrology affect fish and other aquatic organisms through almost all hydrogeomorphic processes (Hupp 1992). Hydrology also determines the successional evolution of riparian plant communities and ecological processes (Nilsson and Svedmark 2002.)

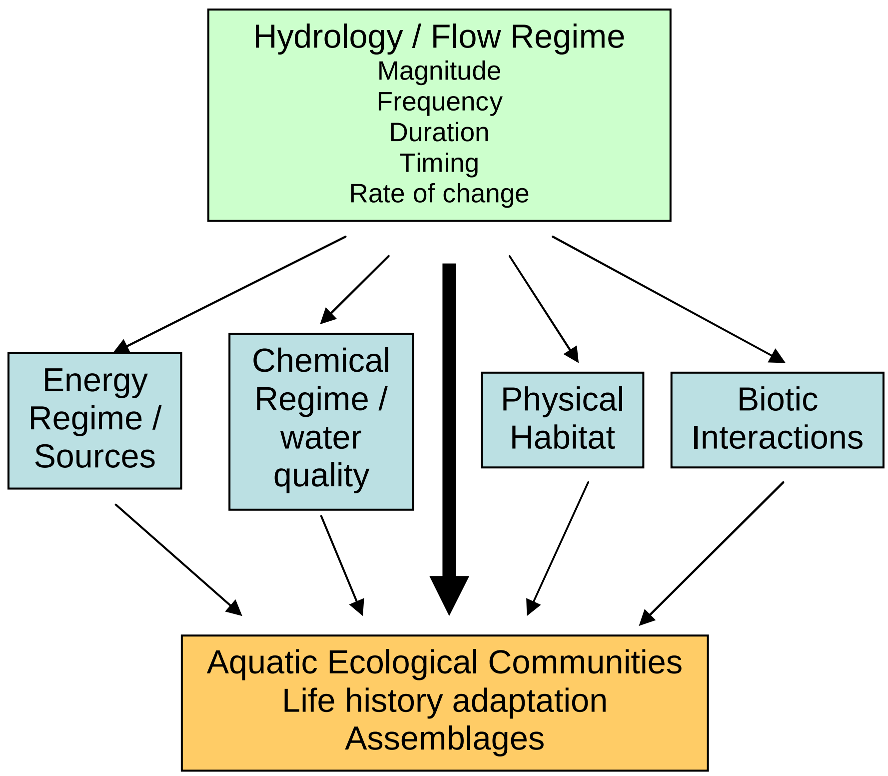


Figure 3.1. Hydrology (flow regime) is the driver for all other key ecological attributes in freshwater stream systems. (Modified from Poff et al. 1997).

The hydrologic effects of land use and drainage system modifications operate via a number of pathways. These include the impact of drainage and land use change on stream hydrographs, flood frequencies, nutrient cycling, and basin water yield. Land use and drainage modifications also affect soil water storage at field, catchment, and regional scales, scale hydraulic connectivity and drainage response (i.e., how rapidly water is able to travel across the landscape), as well as drainage density, configuration and layout of the drainage network, conveyance capacity, groundwater recharge, evapotranspiration, depressional storage, and interaction of water table and baseflows. These hydrologic changes in turn have impacts on stream habitats through impacts on stream channel stability, erosion, and geomorphology (Figure 3.6).

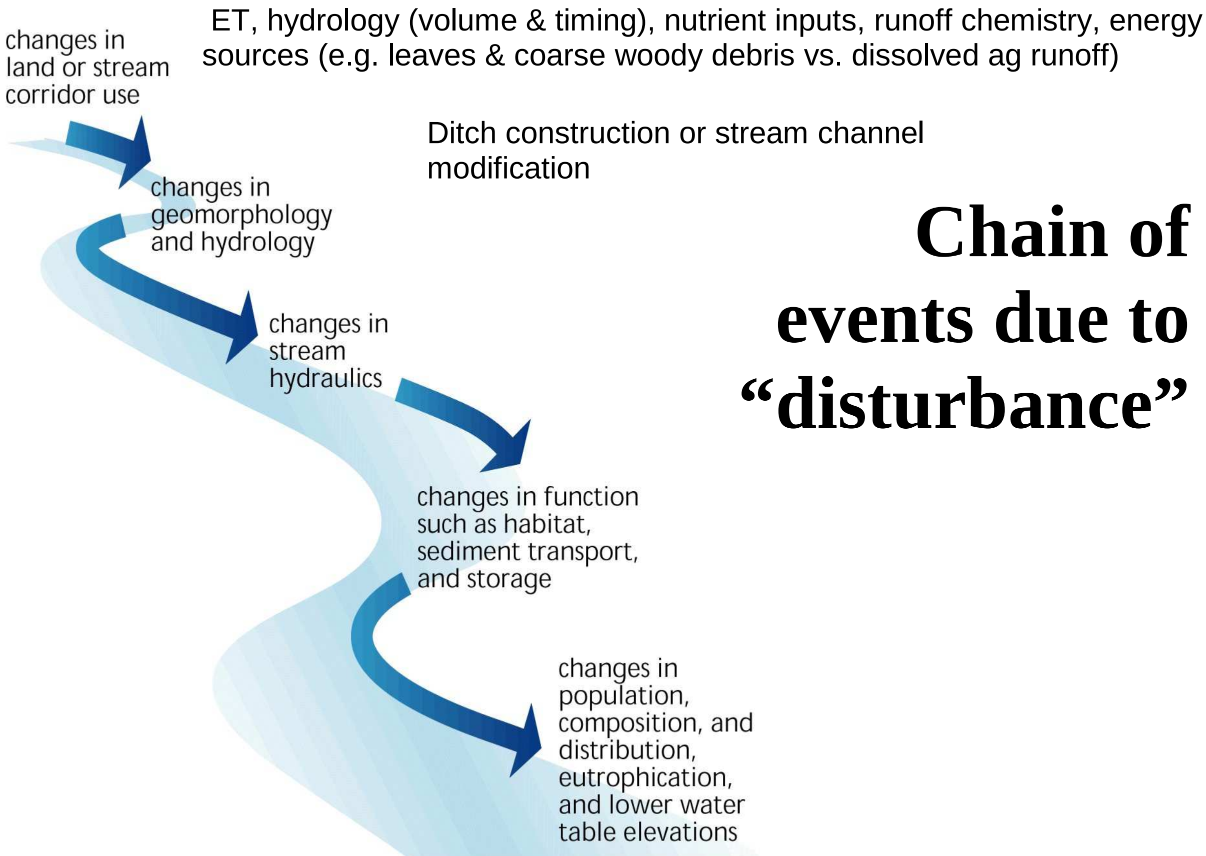


Figure 3.2. Chain of events due to disturbance. Disturbance to a stream corridor system typically results in a causal chain of alterations to stream corridor structure and functions. Reprinted from Stream Corridor Restoration: Principles, Processes and Practices (10/98) produced by the Federal Interagency Stream Restoration Working Group (FISRWG, 15 US agencies)

Disruption of hydrologic regimes, e.g. channelization, low head dams, etc., results in loss of riparian habitat and connectivity, altered sediment transport, and either magnified or reduced differences between baseflow and flood stages (Sprenger 2001). The combination of changes in flow, sediment dynamics, and habitats drives changes in water quality and trophic dynamics that ultimately drive shifts in the organization of biological communities, often from specialists to a few competitively-dominant generalists. Changes in species composition in turn generate feedback loops, altering nutrient uptake and energy flows within and across trophic levels, i.e. phytoplankton, macroinvertebrate, and fish communities, via competition, predation and food consumption (Schlosser 1985, Niyogi et al. 2004).

Hydrologic changes associated with agricultural development over the past century have resulted in increased total discharge and fluxes of carbon and nutrients to rivers and streams of the Mississippi River Basin and the Gulf of Mexico (Donner et al. 2002, Raymond et al. 2008). At least 75 percent of the nitrate in the Mississippi River today is anthropogenic in origin (Goolsby et al. 2001). Decades of research have concluded that agriculture is by far the largest source of nutrients in streams and rivers of the Mississippi River. In particular, the Upper Mississippi River Basin, including the heavily-tiled corn growing regions of Minnesota, Iowa, and Illinois, contributes 22% of the flow and 31% of the nitrates in the entire Mississippi River system (Figure 3.3; Goolsby et al. 1999). Load analysis of sources of N to the Mississippi River Basin show that fertilizer inputs, legume nitrogen fixation, and livestock operations combined account for almost 50 times as much N as municipal and industrial point sources. The 900,000 metric tons of nitrate discharged by the Mississippi River into the Gulf of Mexico in 1991 was therefore equivalent to about 16% of the nitrogen fertilizer applied to cropland in one year (Goolsby et al. 1999). Nutrients released to the atmosphere by volatilization (e.g., loss of NH_3 gas), gaseous nutrient release (e.g., N_2O), or through wind erosion from crops and soils can also be added to surface waters via precipitation or by dry deposition/adsorption. Atmospheric transport of N from land to water has been measured between 3 to 20 kg/ha/year, depending on location, source, and weather (Hatfield et al. 1996). Stream bank sediments derived from agricultural lands and redeposited in the floodplain can also be a long-term source of P and other contaminants (see sections on sediment and hydrology).

Modeling of contaminant sources for the nitrate TMDL on the Raccoon River, to the west of the Boone River, shows that nearly 90% of the nitrates and more than 90% of the E. coli contamination comes from nonpoint sources (row crop and livestock operations), as compared with point source discharges (van Gorp 2007). It is likely that load allocations would be similar for the Boone, a similarly sized agricultural watershed.

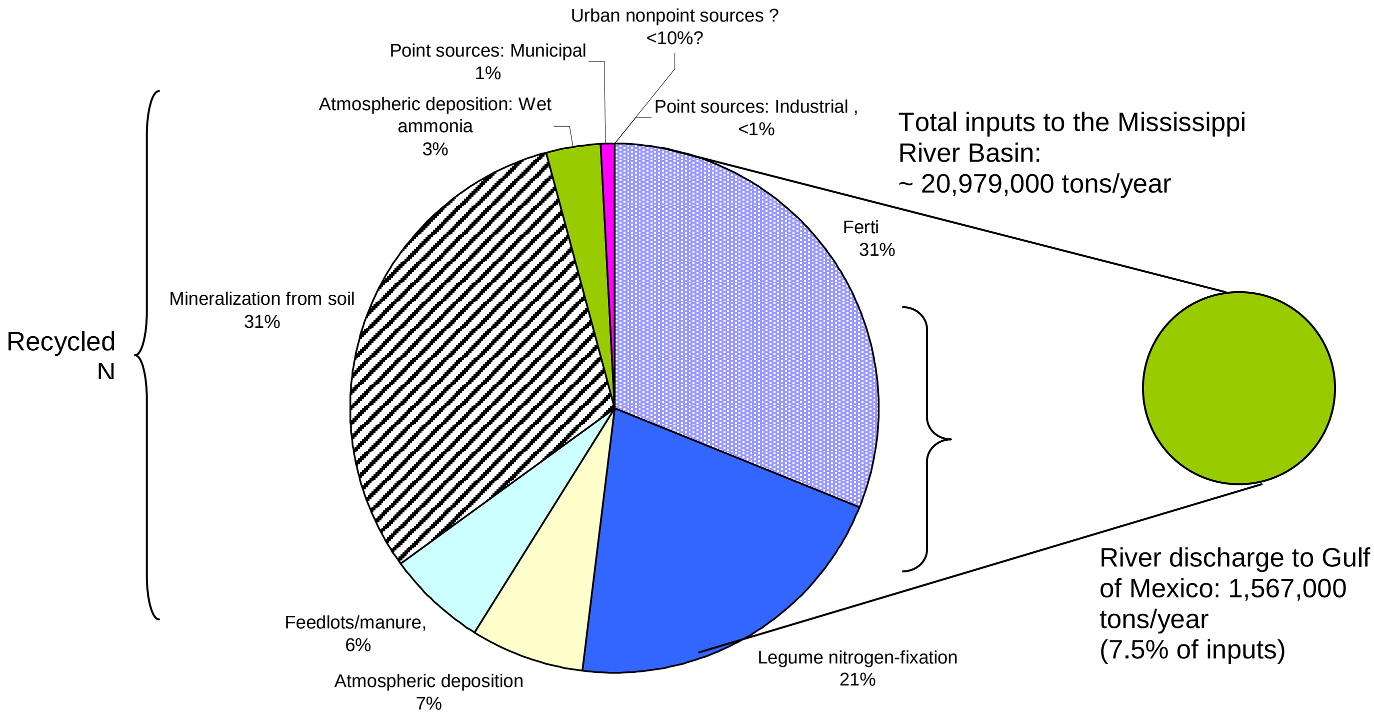


Figure 3.3. Sources of nitrogen in the Mississippi River Basin and measured nitrogen discharge to the Gulf of Mexico from the basin (Goolsby et al. 1999).

Consensus is fairly well established that wetland and prairie-lake conversions to cropland have in general increased runoff and contributing drainage areas and reduced storage. The hydrologic response of the Mississippi River Basin is the result of both continued intensification of water management and observed changes in climate (Simpkins et al. 2004; Raymond et al. 2008). In the upper Midwest, increases in stream discharge have coincided with temporal increases in amount and intensity of precipitation over the past 40-80 years, a scenario that is predicted to continue across much of the region according to most climate change model predictions. Precipitation increases can not entirely account for the increased discharge (Schilling and Libra 2003, Baker et al. 2004, Raymond et al. 2008). Over the same period, peak streamflows have increased disproportionately (Kunkel et al. 1999, Garbrecht et al. 2004). In Illinois, a 27% increase in intense rainfall events from 1921-1980 led to a 77% increase in floods (Changnon 1983). Only one third the observed increases in annual flow volume for four Illinois river basins from 1940-1990 was attributed to increased precipitation (Changnon and Demissie 1996). Studies in the Minnesota River Basin, a predominantly agricultural tributary basin to the Upper Mississippi have also shown a link between agricultural land use and increased flows independent of precipitation changes (Mallawatantri et al. 1999).

In tile-drained, row cropped agricultural basins, increases in baseflows have been tied specifically to increased drainage density created by drainage infrastructure. Schilling and Libra (2003) for example have demonstrated that a greater proportion of precipitation has been routed into streams as baseflow than as stormflow in the latter half of the 20th century. Miller (1999) modelled the hydrological effects of land use and drainage for an extensively tile-drained tributary basin to the Minnesota River. Changes in land use over the past century have increased average annual water yield from 50 mm to 170 mm, an increase in the discharge/precipitation ratio from 5% to 19% Flood frequency analysis also showed that under a scenario designed to reflect presettlement conditions prior to drainage, average annual peak discharges would have been 82-88% lower, and the magnitude of peak flows would have been lower up to the 50-year flood recurrence interval. The influence of drainage and land use change is more important for the smaller, more frequent flood events. Compared with presettlement perennial vegetation scenario, Miller (1999) found that row crop land use with tile drainage roughly doubled the discharge/precipitation ratio and average annual water yield, while it nearly tripled the annual peak discharge. Schilling and Helmers (in press) acknowledge that while hydrograph separation of streamflow suggests widespread increases in baseflow in many Midwestern rivers, a significant fraction of the baseflow signal may not be related to natural groundwater seepage at all, but rather to increasing contributions from subsurface drainage tiles. In short, changes in watershed hydrology from presettlement to present have increased total flow in streams, both peak flows (making streams more "flashy") as well as the baseflows.

The major impact of agriculture on watershed budgets and streamflows in the Midwest has been the changes in seasonal and annual evapotranspiration associated with conversion of the landscape from prairie to crops. The rate of evapotranspiration (ET) and its role in watershed hydrologic cycling is greatly affected by the type of vegetation present on the landscape. Historically, ET accounted for the major proportion of water budgets in northern prairie, and the pattern of water use by native prairie vegetation more closely matched the seasonal availability of water. Significant changes in regional ET accompanied the conversion of prairie grasslands and forested lands to cropland (Woo and Rowsell 1993, Poiani et al. 1996, Brye et al. 2000). Although ET rates on cropland during the peak growing season often exceed that of natural grasslands, most runoff in the upper Midwest now occurs during early spring before crops are planted, when ET rates are typically higher for lands in perennial crops or native vegetation. Studies of water use by crops throughout the growing season generally show higher rates of transpiration than native perennial vegetation during midsummer, whereas prairie grassland and wetland ET is higher than that of cropland earlier in spring and later into the fall (post-harvest) (Schaffer 2005). This is related to why the highest rates of nitrate leaching now occurs in early spring, before annual row crops are able to utilize and transpire available water (Figure 3.4).

Changes in evapotranspiration, vegetation, and annual water budgets associated with agricultural land use may even drive alterations in local and regional microclimate through effects on convection and latent heat flux (the heat energy involved in the phase change of water during evaporation). Schaffer (2005) has suggested that increased intensity of mid-summer storm energy and precipitation events over much of the corn growing region of the Midwest may be related to the increased latent heat flux generated by the extreme peaks in ET for corn relative to native perennial vegetation. University of Minnesota climatologist Mark Seeley has also noted that in recent years, the climate in northwestern Minnesota has been characterized by warmer winters, higher minimum temperatures, more tropical dew points, and more frequent heavy rains. The increased intensity of storms is related to higher convection energy more than to absolute temperature increases (Seeley 2003).

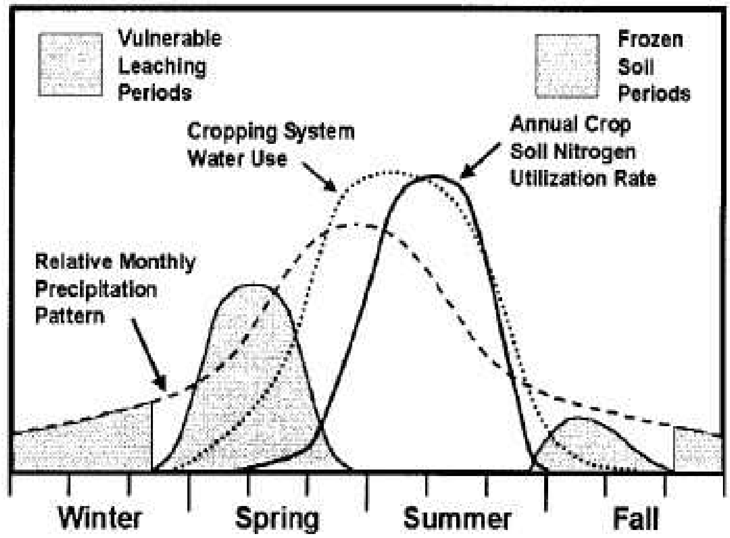


Fig. 3. General seasonal patterns for precipitation, N uptake rate by a corn crop, cropping system water use, and periods potentially favorable for NO_3 leaching from midwestern corn production (adapted from Fig. 4 of Power et al., 1998).

As noted earlier, the presettlement landscape of the Boone was about 4% swamp, marsh, wetland, and slough, and 59-60% of the watershed is characterized by hydric soils that would have supported hydrophytic vegetation. Currently, < 0.2% of the watershed is composed of wetlands, based on land use analysis. Because prairie wetlands have higher soil water retention capacity and evapotranspiration, the presettlement landscape would have had higher storage of water in soils and surface water and much longer retention times. The 1994 USACOE study in the Boone concluded based on modelling that the frequency of small flood flows has probably increased as a result of wetlands losses. Unfortunately there are no flow records from presettlement.

An instructive comparison is provided by a study of stream channel changes, nitrate loads, and hydrology in a very similar basin located less than 100 miles to the north of the Boone River, the Blue Earth River Basin. The Blue Earth River (BER) Basin is part of the same EPA Level III ecoregion as the Boone River (the Western Corn Belt Plains; also the North Central Glaciated Plains in Bailey's ecoregion classification; Figure 3.5), and has similar soils, land use, and topography. Similar to the Boone, the highest nitrate-N concentrations are typically

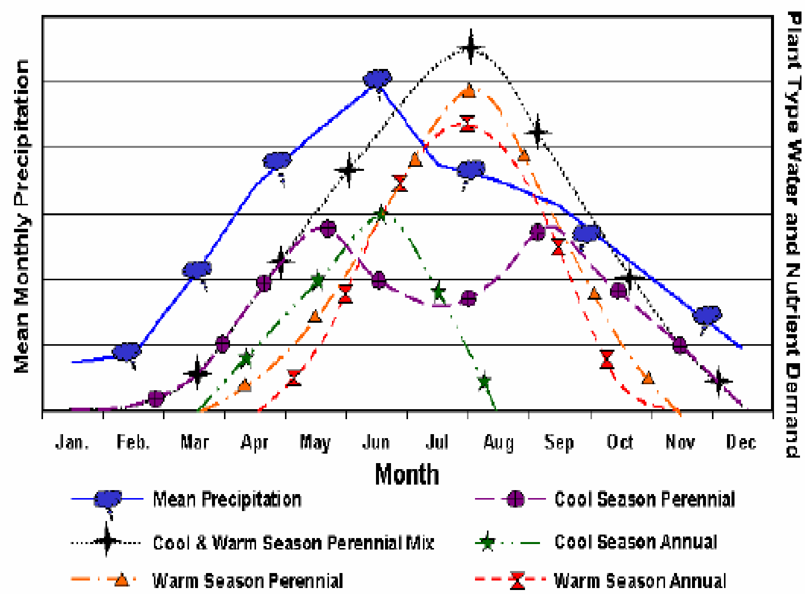


Fig. 6 General annual water and nutrient demand curves of cool and warm season

Figure 3.4. (a) Seasonal patterns of precipitation, water use and uptake by crops, and N loss potential. (from Dinnes et al. 2002)

observed in May and June, and typically range from 7–34 mg/L. Magner and Steffen (2000) and Magner et al. (2004) have shown that in the BER, land use and hydrologic changes made to the watershed associated with crop production and modern drainage have increased peak flows 20-206%, contributing to channel incision, increasing bank erosion, and substantially increasing nitrate loads to the Minnesota River.

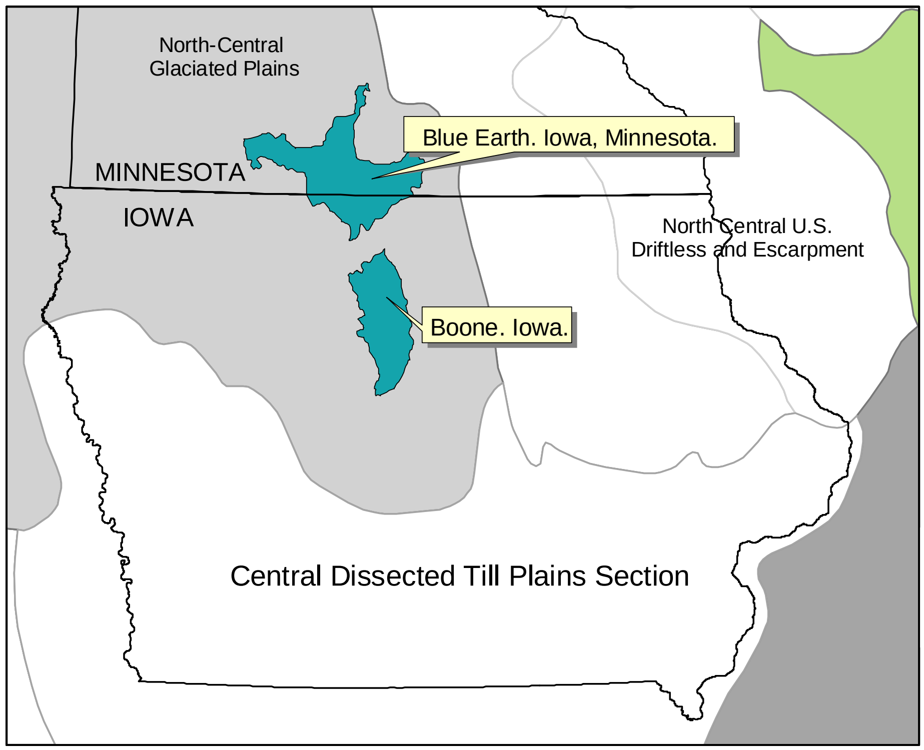


Figure 3.5. Relative location of the Boone and Blue Earth River Basins and Bailey's ecoregions.

Annual peak flows for the 1.01-yr, 1.5-yr and 2.0-yr discharge recurrence intervals at the USGS gage station on the BER (Rapidan, MN) have increased by 130%, 28% and 17%, respectively, comparing the periods from 1940-1960 and 1974-1998. Similar patterns were observed for another neighboring basin, the LeSueur River, with annual peak flows for the 1.01-yr, 1.5-yr and 2.0-yr recurrence intervals increasing by 343%, 78% and 48%, respectively. By comparison, the Minnesota River at Mankato (downstream, a much larger basin including other less agricultural land uses in its headwaters) showed relatively more moderate increases of 52%, 31% and 28%, respectively. On the Little Cobb Creek (a subwatershed within the BER), Miller (1999) estimated with a model that annual peaks associated with the 1.5-yr recurrence interval have likely increased by 1.5–2.5 times the pre-drainage flow.

Mallawatantri et al. (1999) attributed a portion of the increase in annual peak flows observed throughout the Minnesota River Basin primarily to increased precipitation. However, Magner and Steffen (2000) conclude that increased precipitation in southern Minnesota only explains a portion of the increased stream discharge, and that most of the observed increased stream discharge to changes in drainage density. Modern tile drainage changes contributing drainage area by decreasing "micro landscape storage"—i.e. soils and small (less than a hectare) swales or wet portions of fields that historically held water and even sent a small amount water back to the atmosphere in May or June via evaporation (Magner et al. 2004).

Because of the natural tendency of channels to evolve towards an equilibrium between flow and sediment transport, the instream hydrologic and geomorphologic effects of drainage are confounded when they occur in conjunction with ditch construction, maintenance, and other modifications of surface drainage channels and networks. In a stable natural stream, bed scour and aggradation occur in equilibrium, and channel dimensions—including plan-form, width:depth, and floodplain connections—are determined by the effective discharge. All else being equal, in response to increased discharge, channels must either downcut and/or widen to accommodate the increased discharge. Increases in channel forming flows may lead to increase in channel cross sectional area or decreased sinuosity. Although many channels have been entrenched by design, through ditching and straightening, many unmodified natural channels have incised in response to increased effective discharge associated with surface and subsurface drainage (Knox 1987, Faulkner 1998, Fitzpatrick et al. 1999, Magner and Steffen 2000).

In addition to channel incision, increased effective discharge or flashiness of flows can also lead to loss of the channel-floodplain connection, and headward erosion of stream channels (Magner and Steffen 2000, Knox 2001). Subsurface drainage may also contribute to channel and bank erosion if it alters stream and watershed hydrology. Because waters discharging into streams from drainage tiles are relatively sediment-free relative to their sediment transport capacity (Leopold 1994, Hubbard 2005), delivery of low sediment water via tile drains may cause effects analogous to creation of sediment hungry water by impoundments. The role of low-sediment water from tile drains in causing downcutting has not been well-studied specifically as it relates to subsurface drainage, and the effect may be difficult to separate from the effect of increased effective discharge. However, ongoing work in this area suggests that this may be a significant phenomenon in some landscapes (Ward et al. 2003, Magner 2006).

Hydrologic and geomorphic alterations of streams have implications for nutrient cycling and nutrient loads as well. Because sediment and P are primarily a problem of surface runoff, upland runoff may be a relatively minor source of sediment to surface water in tile-drained watersheds. A significant loading legacy may result from P that has accumulated in already eroded sediments within the channel and floodplain, which means that even if watershed P loading is reduced, there may be a lag time before surface waters respond to reduced loading. Streambanks, riparian zones and wetlands, once effectively saturated, can become a long-term source of P. In the Minnesota River Basin, much of which is part of the same ecoregion as the Boone River, load analysis by the Minnesota Pollution Control Agency suggests that as much as 16% of the P load is derived from bank erosion and bank sediments. Exploration of the role of streambanks as a source of P in the Boone River under different land use and restoration scenarios could be developed combining instream physical habitat measures, including analysis of field and streambank soil test P, with channel models such as CONCEPTS using measured hydrological data or model output from SWAT (Langendoen et al. 2006).

Channel alterations and channel incision have implications for nitrate loads and riparian communities as well. Channel degradation disconnects the channel from the active floodplain and the riparian corridor, as well as lowering the water table below the riparian corridor. This destabilizes riparian vegetation and habitat, as well as the riparian corridor's ability to buffer environmental stress, as shallow groundwater movement into the channel begins to bypass the roots of floodplain and riparian vegetation.

Subsurface flow and tile drainage. Dissolved nutrients transported in subsurface drainage water (leaching loss) are dominated by nitrate- N. Nutrient forms such as ammonium-N and phosphate are largely adsorbed (or precipitated) from solution as they move with water through the soil to the saturated zone in subsoils, where excess water can move laterally to streams as base flow and/or artificial subsurface drainage. The most important factors in determining leaching loss are the amount of nitrate- N in the soil and the timing and amount of precipitation or irrigation that drives subsurface drainage. Therefore, N management in terms of form, rate, method, and timing of application is critical in determining concentrations and losses. Annual nitrate-N leaching losses from row-crop land with artificial subsurface drainage range from 2 to 130 kg/ha/year (Randall et al. 1997, Doering et al. 1999, Kladivko et al. 2001). Timing of precipitation and snowmelt has a great effect on the timing of nitrate-nitrogen exports from crop fields. The majority of nitrate loss in most watersheds of the Midwest occurs from January to June, with the peak coinciding with snowmelt and spring rains of April-June.

Although sediment and sorbed contaminants such as P are considered primarily a problem of surface runoff, preferential flow—i.e. the movement of water through cracks, “macropores”, and other open flow pathways below the soil surface—has been found in many studies to play the dominant role in transport of contaminants to surface waters. Sediment and other contaminants in runoff that does find its way into subsurface drains—whether via macropore flow or surface tile inlets, primarily during episodic storm event—is discharged directly to surface waters, bypassing buffers, hedgerows, upland depressions, fencelines, grassed roadside ditches, and other “upland” sinks. In some cases, preferential flow accounts for substantial losses of sediment (Chapman et al. 2003, Foster et al. 2003), phosphorus (Gachter et al. 1998, Sims et al. 1998), liquid manure (Dean and Foran 1992, Fleming and Bradshaw 1992, Geohring et al. 2001), pesticides and herbicides (Kladivko et al. 1999, Elliott et al. 2000, Nieber 2001).

Because sediment in subsurface drainflow is discharged directly to surface waters, it may be an important contributor to potentially algal-available P in surface waters. Uusitalo et al. (2004) found that suspended soil material in subsurface runoff from clayey soils in Sweden contained as much P (47–79 mg kg-1) as did the sediment in surface runoff (45–82 mg kg-1).

Implications for aquatic biota in the Boone River watershed

The modification of geomorphological conditions in fluvial systems that drives homogenization of habitats is one of the major threats to aquatic biodiversity (TerHaar and Herricks 1989, Frothingham et al. 2002). Stream channel incision—whether driven by fluvial processes or by construction and maintenance of ditches for surface drainage--typically leads to reduced spatial habitat heterogeneity and greater temporal instability instream (Shields et al. 1995, 1998). Compared to a channel formed by natural fluvial processes, where benches, woody debris, and other channel or floodplain features may create refugia for fish and other organisms, the oversized trapezoidal channel typical of a surface ditch provides very few refugia. An altered channel can lead to scour, stranding, and/or wash-out of benthic organisms, eggs, and smaller fishes. Shields et al. (1998) found, for example, that small fish species with restricted microhabitats were eliminated from a study site with high flow fluctuation, whereas the relative abundance of habitat generalists that could tolerate deep, fast flows peaked at sites with greatest flow variability. In Michigan streams, increased channel incision was associated with reduced biomass of total, game, and intolerant fish species (Infante et al. 2000, Infante 2001). McRae et al. (2004) found that sites with the richest mussel assemblages in Michigan streams had greater flow stability, lower percentage of fine sediments in the streambed substrate, and lower channel incision. In the Red River Basin of Minnesota, Meneks et al. (2003) found channelized reaches tended to have greater overall variation and greater daily fluctuation in temperature and dissolved oxygen (DO), as well as a higher level of intermittent flow than unchannelized reaches. Channelized reaches also exhibited lower diversity of larval fishes, and are often dominated by highly tolerant species such as the fathead minnow (*Pimephales promelas*).

As the ultimate indicator of watershed health and sustainability, the decline of aquatic species is the principle conservation concern for The Nature Conservancy. Nationally, dams have been the principle factor behind widespread declines of fishes, mussels, and other aquatic taxa, causing local extirpation of migratory and anadromous species such as channel catfish (*Ictalurus punctatus)* and redhorse (*Moxostoma* spp.). In the agricultural Midwest, the main stressors affecting aquatic communities have been the combination of aquatic habitat loss, flow and channel alterations, sedimentation, and water quality changes associated with settlement and agricultural development (Menzel et al. 1984).

Potential water quality effects on mussels. Both N and P drive degradation and eutrophication in stream systems. Available P stimulates additional algal and aquatic plant growth, which is exacerbated by warm water temperatures. This can lead to eutrophic and hypoxic conditions in freshwater systems when production peaks, algae and aquatic plants die, and biological oxygen demand (BOD) is created to support breakdown and decomposition of excess organic matter. In addition to causing fish kills, it also can cause fish population changes. Rough fish species are more tolerant to low dissolved oxygen conditions than game fish and can become dominant. Although P is traditionally viewed as the driver of freshwater trophic alteration and low DO, recent research does present evidence of significant effects of N in freshwater due to nitrate toxicity (Camargo et al. 2005), and some specifically to mussels (Sharpe 2005). Ohio EPA found a strong correlation between P and instream biodiversity; but did not demonstrate similar correlations for N, and have not therefore released a similar report (Rankin et al. 1999). A growing number of studies have also demonstrated that ammonia toxicity poses a chemical threat to freshwater mussels, particularly to juveniles (Augspurger et. al. 2003, Newton et. al. 2003, and Mummert et. al. 2003). Recent evaluation of water quality criteria for ammonia and nitrate suggest that these criteria levels are not sufficiently protective for mussels (Camargo et al. 2005, Sharpe 2005).

Toxic and/or bacterial contaminants. The potential impacts of bacteria, pesticides and herbicides on mussels are more poorly understood. Chemicals in particular are very poorly monitored and expensive to sample. Although herbicides including atrazine, metalochlor, prometon, and simazine are frequently detected in agricultural watersheds, most ambient concentrations of detected organic contaminants are at or near the method detection limit, well below any aquatic life criteria. Fitzhugh (2005) suggested that agricultural insecticide and herbicide runoff is likely responsible for some of the association between agricultural land use and stream biota (Cooper 1993, Skinner et al. 1997); however, evidence comes primarily from localized toxicity tests rather than from landscape-scale investigations. For example, field enclosures using caged amphipods and laboratory tests that exposed midge larvae to stream sediments showed pesticide toxicity in an agricultural catchment in the United Kingdom (Crane et al. 1996). The disappearance of 8 of the 11 most abundant invertebrate taxa from a reach of headwater stream after surface runoff from arable land was attributed to an insecticide (Schulz and Liess 1999), although most species recovered within 6–11 months, indicating a pulse disturbance. Because the concentrations of agricultural pesticides and herbicides are seldom measured in studies relating agricultural land use to stream biota, their role may be more widespread than is recognized. Relatively infrequent releases, many of which may never be reported or detected via grab sampling, could cause infrequent but severe damage to instream organisms, effects that in the case of mussels are likely to persist for decades.

Potential hydrologic/geomorphologic effects on aquatic biota. Scour and sediment deposition associated with altered hydrology and increased flashiness ranks as one of the larger threats to mussel populations nationwide, and has been hypothesized to be a significant problem for mussels in Iowa. Mussels are vulnerable to scour at high flows (during which they can ripped from stream bed and washed downstream) and low flows (high temperatures and lack of oxygen). Because mussels are less mobile than fish and their populations are less able to respond to disturbance than most benthic invertebrates due to their complex life history, they are thus very vulnerable to hydrologic changes.

Baker et al. (2004) found that many Iowa streams, particularly tile-drained watersheds of the Des Moines Lobe, seem to have experienced ***decreased*** flashiness and increased baseflow from 1975-2001, during the same period when it appears there has been a continued decline in mussels in the Boone River. The IHA analysis from the Boone River comparing 1940-1970 to 1970-2003 show that flows have increased, across all the indicators and recurrence intervals. Low flows do not appear to be a critical issue, nor do existing data on the condition of the banks, substrates, and the habitat in the lower watershed point immediately to a significant channel erosion or turbidity problem. However, the increases in flow magnitudes across all recurrence intervals, as well as the fact that a significant fraction of the water discharged into the Boone system is relatively sediment-free, suggest that altered hydrology may indeed be driving bank and bed erosion, scour, and altered deposition in the Boone.[4]

Despite the apparent absence of a turbidity problem in the Boone, extensive research done on the Blue Earth River Basin to the north suggests that sediment and channel change in response to intensification of the drainage infrastructure from 1970 to the present time has been a major driver of habitat changes in that basin. As in the Boone River, although the only major land use / cropping system change since the 1970s has been a substantial increase in soybean acreage (mainly replacing hay and other grain crops), intensification of drainage—specifically pattern tile beginning in the 1980s—has been proposed as a contributing factor to the observed patterns (Magner and Steffen 2000; Magner et al. 2004). Thus, scour, stranding, sedimentation, and increased sheer stress may indeed be contributing to impacts on mussels.

It is unlikely that observed declines in mussels in the Boone River can be attributed to a single stressor. More likely, mussel beds in the Boone River are being impacted by cumulative effects from multiple stressors. Slow moving, slow maturing mussel populations may require a very long time to recover from even a brief short-term acute events, such as an ammonium or pesticide runoff event, or a high flow event resulting in scour. Poole et al. (2005) suggested that mussel conservation efforts are most critical in highly sloping landscapes with less permeable soils, where low groundwater flows might lead to unfavorable conditions for reproduction. Statewide, they recommended restoration of riparian woodlands and the protection of streams from agricultural impacts such as agro-chemical flux and siltation.

Neugarten and Braun (2005) suggested the creation of predictive (simulation) models and indices for a number of watershed features. The SWAT model has made progress towards greater understanding of the relationship between land use, precipitation patterns, hydrology, and water quality. However, many of these relationships remain unexplored. Additional models could be developed in conjunction with SWAT model hydrologic outputs to explore channel versus upland sediment contribution, channel stability or migration, and implications for habitat quality, biological community response, and restoration. The potential benefits from additional modeling are further discussed in the sections on potential and recommended actions.

Impending and Future Threats

Several ongoing and anticipated trends also have the potential to drive significant change in the Boone River watershed. These changes may either exacerbate or mitigate current stresses, as well as introducing new threats to biodiversity in the Boone River.

Climate change is anticipated to be a major driver of ongoing and future ecological change at global, regional, and local scales. The Nature Conservancy encourages conservation planners and practitioners to assess the impact of climate change and other important drivers for conservation goals and targets (Aldous et al. 2007). Shifts in precipitation and temperature will drive a range of interrelated changes and system stresses, including but not limited to, increased eutrophication, altered instream nutrient processing, altered hydrology and thermal regimes, altered groundwater availability, and increased disturbance frequencies, all of which are likely to result in significant shifts or changes in species composition, habitat availability and use. Climate change also has the potential to drive global, regional, and local economic feedback loops in ways that are virtually impossible to predict. However, learning to anticipate, monitor, detect and adapt constructively to these changes, as well as recognize chains of causality, will become increasingly important.

One economic change that is already driving significant physical changes on the landscape in the Boone River is the increase in corn prices driven by the ethanol boom. The rapid growth in corn demand associated with ethanol expansion can even be seen, in fact, as indirectly related to climate change, in that it has been partially driven by the concerns about climate change, oil independence, and government action to encourage the development of renewable fuels.

The Boone River is located in the heart of the area that has experienced the most rapid ethanol development (Figure 3.6). There is currently one operational ethanol plant in the Boone River Watershed located in Goldfield, IA that has a current capacity of 50 million gallons/year (Krogh et al. 2008). Fourteen other operational ethanol facilities are located within 10 miles of the Boone River watershed boundary (four producing ethanol for in-house use), with three more currently under construction (IDNR 2007). Water demand for a 50 million gallon ethanol facility is estimated at between 100-200 million gallons of water per year based on a low estimate of 2-4 gallons of water per gallon of ethanol.

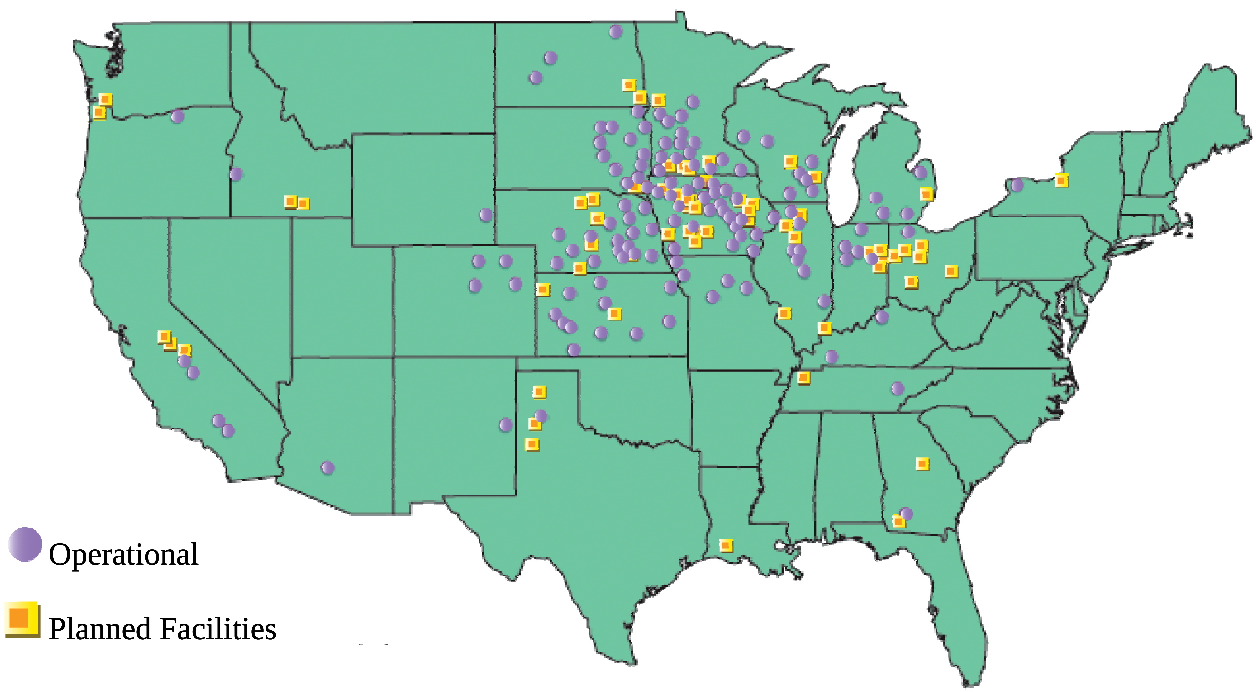


Figure 3.6. Ethanol facilities as of 2007 (planned & operational). Source: Renewable Fuels Association, modified from Gassman et al. 2008.

At the current average production of 2.7 gallons of ethanol per bushel of corn (Iowa Corn Growers Association 2008), a 50 million gallon ethanol facility would require 18.5 million bushels of corn. At 175 bu/acre average yield, the facility could consume the production output from 100,000-110,000 acres of croplands—or roughly 20-30% of potential annual production in the Boone River watershed. Each bushel of corn also yields about 18 pounds of animal feed as a byproduct of ethanol production, providing a local source of low cost animal feed. This has the potential to stimulate expansion and development of animal feeding operations in the region. The watershed currently has more than 100 animal feeding operations, estimating a total of ~250,000 animal units. A large combined increase in corn-based ethanol plants and livestock operations would have potential implications for both water quantity and quality, as livestock facilities also require significant amounts of water. Although the ethanol expansion has recently showed signs of leveling off, as rising cost of corn for fuel and feed has squeezed both the ethanol and livestock industries, it is impossible to predict the demand for corn and subsequent economic effects.

The SWAT model was developed partially to aid in exploring potential future scenarios such as these, assessing how sensitive water quality and quantity responses may be to potential changes in land use and cropping systems, whether driven by economics or policy. For example, the SWAT model predicts that a significant increase in continuous corn acreage, replacing soybeans in corn-soybean rotations in response to high corn prices and demand, may result in nearly double the nitrate loading to the Boone River (Gassman et al. 2007). SWAT model scenarios suggest that expansion of continuous corn would result in sediment decreases of 2 to 11% (because soybeans experience greater erosion than corn) relative to the baseline (Gassman et al. 2008).

Jha et al. (2003, 2006) also used SWAT, coupled with a regional climate model (RCM), to explore the effects of potential future climate change on streamflows in the Upper Mississippi River Basin (UMRB). The potential impacts of climate change on water yield and other hydrologic budget components were then quantified by driving SWAT with current and future climates. A 21 percent increase in future precipitation simulated by the climate model produced an 18 percent increase in snowfall, a 51 percent increase in surface runoff, and a 43 percent increase in groundwater recharge. The net result was a 50 percent annual increase in total water yield in the UMRB. Confidence in model predictions was good, given that the combined performance of the SWAT and RCM using observed weather data was good, and that uncertainty analysis showed that the simulated change in stream flow were robust against known model biases. Increased intensity of rainfall events and increased water yields both have negative implications for sediment and nutrient loads to surface waters in the Upper Mississippi River Basin, and could undermine conservation efforts to achieve N and P load reductions recommended in the Gulf Hypoxia Action Plan (USEPA Science Advisory Panel 2007).

Climate change could also potentially drive improvements in water and environmental quality. If concerns about climate change result in policy or economic changes that reward carbon storage and increased perenniality in cropping systems, this could generate habitat benefits, restore watershed hydrology, and reduce nutrient loading to surface waters. Gassman et al. (2008) also used SWAT to explore the potential water quality impacts of cellulosic biofuels production scenarios, by increasing the acreage of switchgrass and other perennial grasses relative to corn and soybeans. Compared to the baseline (current conditions), the perennial grass (biofuels) scenarios resulted in sediment decreases ranging from 5 to 39% and nitrate loss decreases of 3 to 26%.

Overall, climate change scenarios for the Upper Midwest and the Great Plains reveal a large degree of uncertainty in current climate change forecasts for the region (Covich et al. 1997, Easterling et al. 2001). The SWAT and RCM model results furthermore indicate that simulated streamflows are very sensitive to current forecasted future climate changes (Jha et al. 2006). Although less well studied, it is likely that groundwater levels may be similarly responsive.

Summary of Threats Across Targets

Assessment of critical threats is based on *severity, scope, contribution,* and *irreversibility*, based upon the best available knowledge and judgments, for each stress and priority conservation target. For each target and stress, severity and scope are assessed based on best available knowledge as “Very High”, “High”, “Medium”, or “Low”. Sources of stress are assessed based on their contribution and irreversibility. Conservation strategies should be designed to reduce or eliminate those stresses that have high severity combined with wide scope. Strategies are less important for stresses with very severe impacts to only a small area—unless that area is critical to a conservation target--or stresses that are widespread but low severity.

Table 3.7 shows the summary of stresses and threats across conservation targets as they have been summarized in the CAP Excel workbook. Threats are listed in the summary table in order of importance according to best available knowledge, with priorities for strategy development listed at the top. Some stresses, while not seemingly widespread or severe, may actually be at or near a threshold of irreversibility. That is, the severity and/or scope of the stress may remain relatively small over the next ten years but in the future will increase inexorably and be impossible to reverse if the source of stress is not abated within the next ten years. The threat posed by climate change falls into this category, and accounts for its high ranking. Climate change is essentially irreversible at the scale of the Boone River watershed, thus strategies will ultimately be geared towards anticipation, adaptation and mitigation of the threats posed to system and species conservation targets.

Table 3.7. Summary of Threats Across Targets

Threats Across Targets		Upper Watershed (uplands and headwater streams)	Lower Watershed (larger streams, rivers, and riparian-floodplain)	Overall Threat Rank
Project-specific threats		1	2	
1	Climate Change	High	High	High
2	Nutrient loading from point and nonpoint sources	High	High	High
3	Habitat loss & degradation	High	Medium	Medium
4	Streambank and cropland erosion	High	Medium	Medium
5	Chemical and bacterial contaminant loading to surface waters	Medium	Low	Low
6	Groundwater contamination	Low (?)	Low (?)	Low (?)
Threat Status for Targets and Project		High	High	High

Situation Analysis

After evaluating the status of conservation targets and identifying critical threats, the next step in the CAP process is situation analysis. Situation analysis involves drilling further down into the "situation" describing the best current understanding of how threats, targets, and potential strategies are linked. This step is not meant to be an unbounded analysis, but instead probes the root causes of critical threats and degraded targets to bring explicit consideration to contributing factors – i.e. what is the scope and magnitude of current and anticipated threats, and indirect threats? What are the drivers, key actors, and opportunities for successful action? Specific questions addressed in this step include: "What factors positively & negatively affect the system, conservation targets, and key attributes?" "Who are the key stakeholders linked to each of these factors?"

A situation analysis is designed to identify the threats and opportunities linked to the planning targets, including direct threats, indirect threats and opportunities. Each factor can typically be linked to one or more stakeholders, i.e. those individuals, groups, or institutions that have an interest in or will be affected by project activities. Through this process, a fuller understanding can emerge of what is really driving those critical threats, what would motivate these conditions to change, and who the partners might be in the efforts to change that trajectory.

Many of the stresses and impacts on the landscape are interrelated, and serve as linked drivers of the health of the system. A system analysis can capture this information and put it into a visual, easier-to-understand format. This system analysis depicts how the threats can be related back to the targets. In particular, the process of developing a conceptual model, or “picture” – either in narrative form or a simple diagram – of hypothesized linkages between indirect threats and opportunities, critical threats, and targets, showing in particular where intervention would have the most impact, can help members of the project team create a common understanding of the project’s context – including the biological environment and the social, economic, political, and institutional systems that affect the biodiversity targets. It can also aid in gaining understanding of the relative scope and magnitude of a range of threats and opportunities. The attempt to depict chains of causality in visual form allows us to assess our conceptual models of how the system is functioning, how we may affect it, and where there are areas of shared understanding as well as uncertainty and/or disagreement.

An example situation analysis for the Boone is the problem of nitrate loading of surface waters via agricultural subsurface drainage and tile outlets (Figure 3.4). The NO_3 problem in the Boone River is both local—i.e. elevated levels are well above those that have been determined safe for aquatic life--and downstream in scope--Iowa is one of the leading states as a source of NO_3 to the Gulf, and the Des Moines Lobe tile-drained watersheds have been identified as some of the priority source watersheds contributing to the Mississippi River loading. The figure illustrates how driving factors at different scales are connected to the key ecological attributes and indicators, as well as how water quality, geomorphology, and ecological status indicators are connected to hydrology. It also provides a well-studied and clear example of why a landscape, functional, adaptive management approach is needed for a component problem such as nitrate loss, restoring hydrology and landscape function at the watershed scale. A comprehensive landscape approach to nitrate reduction has the potential to go a long way towards addressing P, sediment, and aquatic habitat as well.

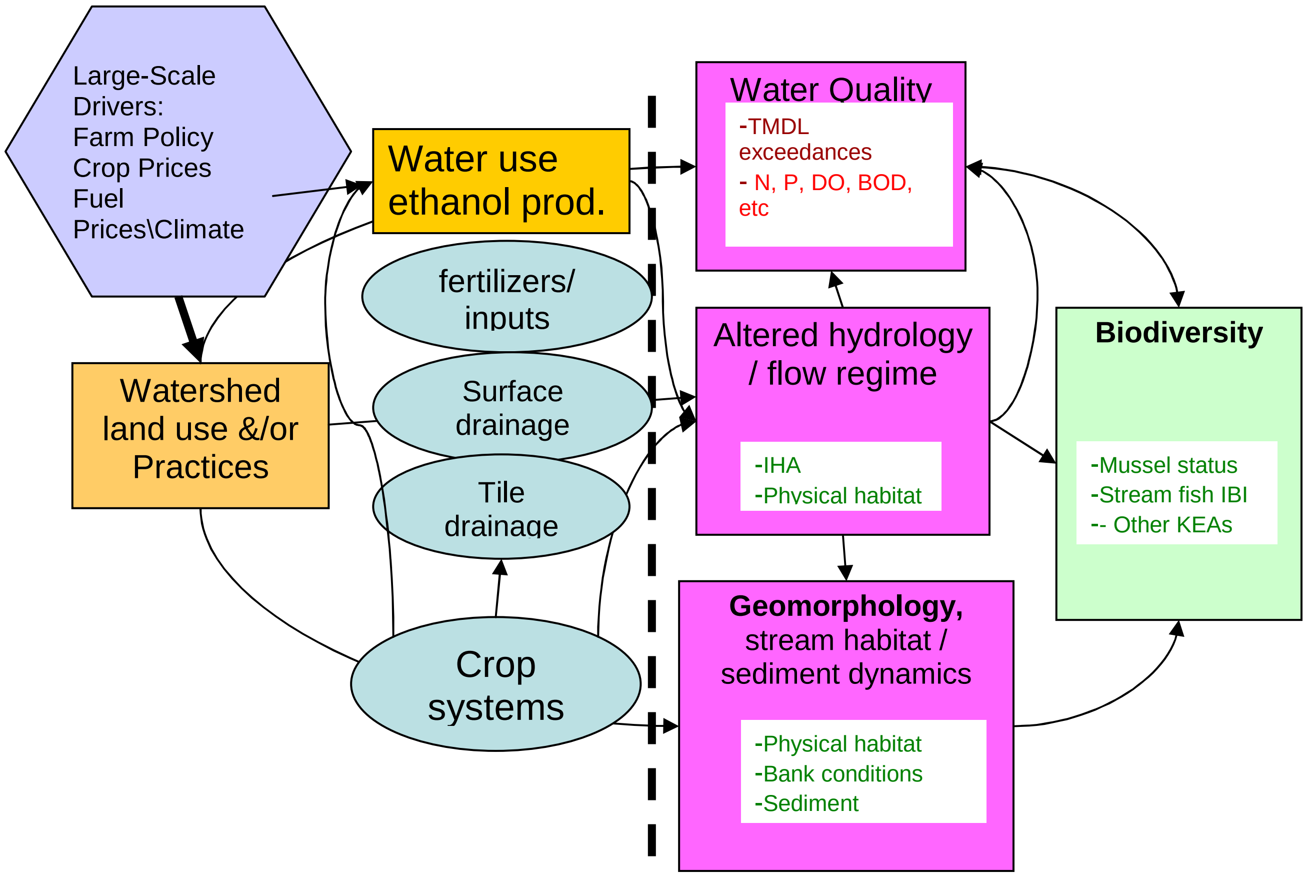


Figure 3.4. Situation Analysis: Identifying places to take action.

Potential Solutions, Actions & Strategies

Strategies to sustain and restore the ecological health of the Boone River will require a multi-pronged approach that addresses the interrelated impacts of watershed land use on aquatic habitat, water quality and altered hydrology. A variety of drainage and conservation techniques have been identified that can be implemented to improve water quality and instream aquatic habitat within agricultural watersheds. There are two primary approaches to reducing, mitigating, and controlling agricultural impacts on aquatic systems:

1. reducing nutrient and contaminant losses to surface waters in the Boone River, including losses of nutrients from croplands and loads from point sources and other sources; and
2. restoring the ecological function of the landscape to process and remove contaminants, including enhancing natural denitrification and nutrient retention processes in each subwatershed throughout the watershed and reducing the total contaminant loads exported from the outlet of the Boone River.

Restoration Goals

1. Reduce loading & concentrations of total N and P to meet aquatic life and human health water quality standards
    a. Nitrate
    b. Phosphorus
2. Target load reductions of other contaminants to meet acceptable range of variation
    a. Sediment
    b. Pesticide
    c. Bacteria
3. Restore streamflows to acceptable range of variation for natural hydrologic regime
    a. Identify and better quantify the nature, severity, and causes of sedimentation and instream bank erosion in the Boone River watershed
    b. Quantify the effects of different potential restoration actions on hydrograph and sediment regime
4. Reverse the decline of mussels
5. Maintain biodiversity and abundance of Boone River species and ecological communities

The specifics of each goal above are as follows:

1a. Nitrate reduction. Under current baseline conditions, the SWAT model predicts 30 year annual average NO_3 load at the outlet of the Boone River of roughly 6-7 million kg. Based on annual average discharge of 18-20 cfs, the mean annual concentration is predicted to be about 10-11 mg/L. To bring the mean concentration down to the 1.95 mg/L aquatic life standard—i.e. the level recommended for protection of sensitive aquatic life--would require reducing riverine loads at the watershed outlet by 85%. Similarly, based on analysis of IOWATER and STORET data, nitrogen concentrations would have to be reduced 60% for median samples to fall below the ecoregional water quality standard of 1.95 mg/L. Table 3.2 shows the results for each subwatershed based on SWAT model output.

1b. Phosphorus reduction. Recent work on addressing hypoxia acknowledges that not only N, but reductions in P loads are needed to effectively address hypoxia. Using the SWAT model predictions to back-calculate the load reductions needed to reduce NO_3 concentrations to 2 mg/L suggests that N loads would need to be reduced by 80-85% (e.g. from 7,000,000 kg N per year at the outlet to less than 800,000, assuming overall flows remain the same). For phosphorus, average concentration at average annual load of 228,000 kg of TP is 0.36 mg/L. The load reduction needed to reduce the average concentration to the 0.1 mg/L aquatic life standard is therefore ~72%.

P removal is a more complex challenge, because permanent conversion of P through benign, inert atmospheric forms is not as simple as with N (which is converted to N_2, the most abundant gas in the atmosphere, through denitrification). Rather, excess P stored in soils and sediments remains a long-term source of P release to surface waters unless permanently removed from the system. Watersheds that have experienced long-term P loading in excess of outputs can experience long lag times before a response to P controls may be observed in surface waters. The problem has in fact become global (Bennett et al. 2001), with atmospheric P deposition a significant source of P in many systems. The potential problem of P loss from agricultural fields is compounded in Iowa because soil-test summaries show that approximately 60% of the soils of Iowa have soil test phosphorus (STP) levels that meet or exceed the level needed to optimize crop production (Baker et al. 2004). High STP levels increase the potential that P will leach via dissolved forms.

2a. Sediment reduction. Further study and analysis is needed to establish specific targets for sediment reduction in the Boone. Research is needed to identify sediment sources and to better quantify the nature, severity, and causes of sedimentation and instream bank erosion in the Boone River watershed.

3. Restoring streamflows to acceptable range of variation. Restoration of aquatic systems in agriculturally-modified watersheds such as the Boone River requires the restoration of the watershed hydrograph to natural flow volume and timing; and/or reduction of the magnitude of peak flow events to a level that existing, modified channels can accommodate without further destabilization. Analysis of hydrologic data from the Boone River indicate that base flows and flood discharges across a range of recurrence intervals have increased 20-30% since permanent gages were established in the early 1940s. Regional research and analysis of the baseline discharge scenario in comparison to the all perennial scenario also suggests that peak and base flows are well above historic levels. Although further analysis is needed to establish specific targets for hydrologic restoration, actions that serve to increase the residence time in the watershed of storm runoff and precipitation, whether in watershed soils, pothole depressions, or surface water bodies is needed. Increasing residence time of waters, particularly during the months of April, May, and June, is needed to restore more natural hydrograph and moderate peak flow volumes to stream channels, returning the processes of scour, sediment transport, and deposition to equilibrium. Actions that increase infiltration, reduce the flashiness of storm runoff events, and maintain groundwater recharge will be particularly beneficial.

4 & 5. Reversing the decline of mussels and maintaining overall aquatic biodiversity. Conserving and maintaining aquatic system health in the Boone River requires implementation of adaptive management to mitigate ongoing or impending threats to biodiversity in the Boone River. This includes action to identify, monitor, and clarify ongoing or impending threats to biodiversity; the need to identify and reverse the major factors driving continued mussel declines in the Boone River; and the strategy of implementing restoration activities as actively monitored experiments that acknowledge and reduce uncertainty.

Actions & Strategies

This section analyzes recommended actions needed to meet goals and objectives above. It also summarizes literature, research, and best professional judgment regarding the effectiveness, feasibility, costs and benefits of potential actions and practices.

Because the most significant threats to aquatic systems are interrelated and largely driven by hydrology, solutions hinge on increasing the residence time in the watershed of water that falls as precipitation. By increasing residence time, effectively slowing flow velocity, the power of the water to transport sediment and attached phosphorus is reduced. By reducing flow velocities and increasing residence time in the watershed, fields, contaminants can settle out of the water column, and denitrification and nutrient uptake will be increased in soils, riparian zones, and wetlands.

Table 4.2. Potential Actions and Practices

Reducing soil, water, and agrichemical losses from croplands

- On-farm nutrient management
    - Reducing nitrogen fertilizer application rates
    - Managing the timing of nitrogen application
    - Managing manure and manure spreading
- On-farm changes in practices
    - Cropping systems and cover crops
    - Changing tillage methods
- On-farm modification of drainage systems
    - Modification of tile drainage depth and spacing
    - Drainage Water Management/Controlled drainage
    - Filtering tile intakes
- Off-farm landscape management and management of hydrology
    - Riparian zones and buffers
    - Stream and wetland restoration

Urban and suburban nonpoint source control

Point source control

- Environmental technology
- Design and treatment

A partial reduction in the quantity of nutrients in surface waters, especially chronically elevated nitrate–nitrogen, can be accomplished through several general approaches and specific techniques (Table 4.2). These include modification of agricultural practices, construction and restoration of riparian zones and wetlands as buffers between agricultural lands and waterways, control of urban and suburban nonpoint sources, and use of environmental technologies such as tertiary treatment for point sources.

Reducing nutrient, sediment, and agrichemical losses from croplands can be achieved via a number of strategies focused on on-farm practices designed to prevent contaminants from escaping or "leaking" from farm fields (Dinnes 2005). Preventing contaminant losses from the farm requires focus on both surface and subsurface drainage pathways (see Figure 3.3).

Potential actions and practices are elaborated below:

Onfarm Nutrient Management includes a range of strategies addressing the rate, timing, and retention of nutrients and fertilizer designed to increase the efficiency of crop uptake during the periods of peak crop growth, and thereby reduce losses to surface waters, which also represent a financial loss to farmers. A variety of techniques and recommendations are being studied and developed, targeted to the specific setting and cropping systems for different farms. These include matching crop needs to fertilizer rates based on crediting (i.e., testing the soil to evaluate N and P residual levels), spring rather than fall application, and rate reductions. The Iowa Soybean Association has been conducting nutrient management trials in the Boone River watershed to evaluate the effectiveness and feasibility of different methods and timing of fertilizer application with respect to both crop yields and nutrient losses.

Nutrient Timing and Rate Management conservation practices include using soil-tests, remote sensing, and split season timing of applications to better match fertilizer application to crop needs across fields and throughout the growing season. This can in theory be achieved with little or no impact on yields.

Buffer strips are areas of land planted to native or perennial vegetation-- rather than annual row crops -- to prevent sediment and contaminants from entering waterways. Buffer strips can be used to line cultivated fields, line the banks of open ditches and stream, and border wetlands.

Grassed Waterways are uncultivated strips of grass sited in areas of the field where water naturally tends to be channeled after rainfall events, and where without cover and root structure, field erosion is a problem. The root systems of grasses, forbs, and other perennial plants hold the soil in place, while the above-ground growth acts to trap sediment and filter contaminants. Velocity of runoff water is slowed, and many contaminants can be decomposed, transformed, or taken up by the buffer vegetation before reaching surface waters.

Manure management. Iowa law requires that all manure from animal feeding operations of any size—including open feedlots and confinement feeding operations -- be disposed so that it does not cause surface or groundwater pollution. Various separation distances must be maintained between areas of land application and protected buildings or other locations such as sinkholes, wells and agricultural drainage wells, described at the IDNR web site. Recent changes in Iowa law have added water sources and high quality water resources as protected areas.

Land application of manure as fertilizer or for disposal requires a certified applicator. Manure disposal is generally prohibited within 800 feet of a high quality resource water (lakes, rivers, streams, ditches, etc.), or within 200 feet of a well, ag drainage well, cistern, surface water inlet or regular water source (lakes, rivers, streams, ditches, etc.), unless the manure is injected or is incorporated in the soil on the day it is applied, or unless permanent vegetation covers the area within 50 feet of the designated area, and no manure is applied within the 50-foot area (IDNR website).

Despite these precautions, in practice, manure enters surface waters via noncompliance, accidental releases, underperforming or inadequate storage operations, as well as inappropriate timing of manure applications to croplands. Manure releases to surface water, groundwater, a drainage tile line or intake, or other designated areas must be reported to the DNR within six hours after occurring or being discovered so that potentially affected parties (including the public) may be notified, as well as to limit the extent of the spill or prevent more extensive damage. Manure can also be transported into tile drains under conditions of preferential flow (i.e, when runoff enters tile drains through soil cracks or open tile intakes).

Several programs and practices are recommended to reduce the probability of manure discharges to surface waters. Although winter application of manure-- i.e. application on snow-covered or frozen ground-- is not regulated, it is not a recommended practice due to significant risk of runoff and contamination of surface waters.

Tillage practices. Conservation tillage has the potential to reduce surface runoff significantly, resulting in reduced erosion and transport of sediment and adsorbed contaminants, improving water infiltration and nutrient adsorption, and reduced in-field volume of runoff water. Leaving residue on the field protects the soil from raindrop impacts and slows sheet and rill erosion. Conservation tillage practices vary in their effects on runoff, ranging from moderate and reduced till to mulch till and ridge till, to minimum or no-till.

Cropping systems. Differences among cropping systems can significantly impact the volume of surface runoff, drain flows, and associated contaminant losses. The influence of cropping systems is related to the effect on surface and subsurface flow volumes, as well as the implications of crop choice for nutrient inputs and management. Several studies have shown that average annual runoff is significantly less under perennial cropping systems, such as alfalfa, than under annual crops such as corn and corn-soybeans (Chung et al. 2001, Randall and Mulla 2001). Perennial crops are able to reduce quantities of gravitational water that would otherwise be lost via subsurface drains under annual row cropping (Huggins et al. 2001). Employing crop rotations where water demand is more suitably matched to available water can reduce tile flows and potentially adverse effects to surface waters.

Cover crops. In the Midwest, fall cover crops have been proposed as a strategy to reduce N leaching by extending the growing season and the uptake of N beyond that for corn and soybean (Strock et al. 2004). A **cover crop** is any annual, biennial, or perennial plant grown as a monoculture (one crop type grown together) or polyculture (multiple crop types grown together) to prevent soil erosion and nutrient losses, and/or to manage soil fertility, soil quality, water, weeds, pests, diseases, or habitat. Because of the cold climate of the Midwest, fall cover crops are generally limited to small grain cover crops (e.g. winter wheat, winter rye) that can take up residual N, released by mineralization during fall and spring, and N released from fall - applied anhydrous ammonia (NH_3) (Strock et al. 2004). The cover crops then release this N as their residue decays the next spring or summer. A potential disadvantage or limitation of this practice is the need to kill off the cover crop before planting in the spring, which could be difficult in a wet spring. Also this practice may require a net increase of chemicals (herbicide), use of fuels, etc.

On-farm modifications to Drainage Systems

Many researchers have concluded that the only hope for a permanent and effective solution to the problem of nitrate loss from croplands involves “structural modifications” of drainage systems (Lemke 2007). Improving tile drainage-water management on farms is the first step in reducing nitrate runoff.

Depth and spacing of subsurface drainage tile systems significantly affects nitrate losses on drained lands. The effect on nutrient losses is driven primarily by the impact of depth and spacing on total subsurface flow volumes (Sands et al. 2003). In clay soils, the depth of drains may have a lesser impact on the hydrograph, because there is less subsurface flow in the pre-drained condition (Trafford 1973, Robinson 1990). Choice of drain tile depth and spacing during installation is typically determined by drainage needs, and may be sub-optimal from the standpoint of N management (Skaggs and Chescheir 2003). Denitrification requires contact time with organic matter. In tile-drained fields, the hydraulic residence time of runoff in and contact time with organic matter is decreased, because the primary water management objective is rapid and efficient movement of water out of the crop rooting soil zone. For this reason, reducing nitrogen loss by control of drain depth and spacing may have limited potential as a management tool without additional financial incentives.

Drainage Water Management (DWM), aka "Controlled Drainage", or "Conservation Drainage", involves drainage outlet control structures that allow the water table to be drawn down gradually throughout the growing season. DWM has the potential to significantly abate nitrate losses, and has been recommended as a best management practice (BMP) in some states. In many years, drainage water management may provide an additional agronomic benefit relative to conventional drainage by retaining water for crop use later in the growing season when the crop has greater need for water and rains are less reliable. The practice involves using a water control structure in a main, submain, or lateral drain to vary the depth of the drainage outlet. The water table must rise above the outlet depth for drainage to occur, as illustrated in diagram. The outlet depth, as determined by the control structure, is raised after harvest to limit drainage outflow and reduce the delivery of nitrate to ditches and streams during the off-season; lowered in early spring and again in the fall so the drain can flow freely before field operations such as planting or harvest; then raised again after planting and spring field operations to create a potential to store water for the crop to use in midsummer (Figure 4.1 (?)). Controlled drainage has been found to reduce nitrate loss by 14-40% under a range of conditions (Tan et al. 1998, Sands et al. 2003, Skaggs and Chescheir 2003).

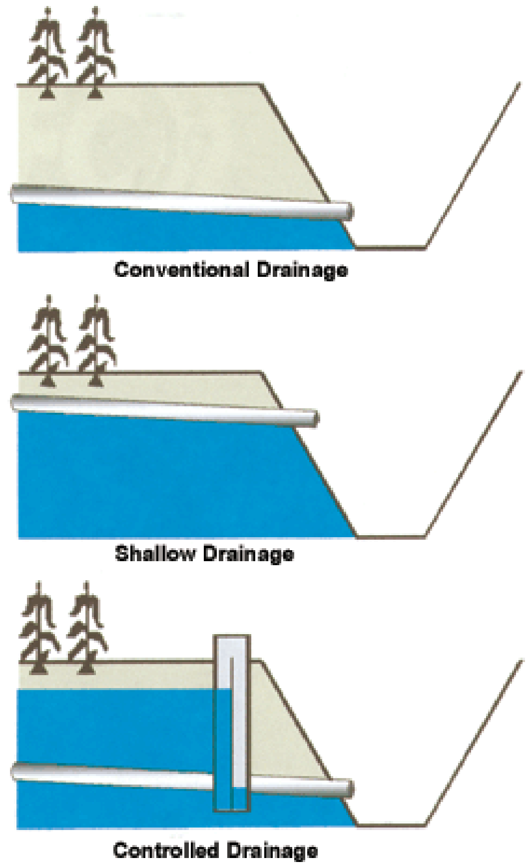


Figure 4.1. Controlled Drainage (Drainage Water Management) vs. Conventional Drainage [reprinted from University of Minnesota extension publication]

In the past, research has shown that effective controlled drainage systems were limited by physical and economic feasibility of design to very flat landscapes, i.e. fields with slopes < 0.5-1%. However, Agrem (www.agrem.com), a drainage contractor in Illinois, has developed a design for controlled drainage on the contour, or subirrigation. Subirrigation is a type of drainage water management that can provide controlled drainage benefits even in fields with significant slope. In these systems, a tile system on rolling terrain provides drainage, sub-irrigation, fertilization, and nitrate removal. Water from the tiles is collected and retained in a reservoir or wetland basin for nitrate removal, and can be recycled through the tile drain system during dry periods for crop use via a pump system. This technique is being pioneered in the Mackinaw River Basin in Illinois by the Conservancy's Mackinaw River Basin project (Lindenbaum and Kirkham, pers. comm.; www.agrem.com).

Filtering open tile intakes. A significant pathway for direct delivery of contaminants to tile drains, including not just nitrate, but P and sediment, are open tile intakes located in cultivated fields, designed to drain areas that tend to pond. During storm events, a pit often develops through which large amounts of soil and sediment can be lost. Tile intakes can be filtered using a grass buffer or through an artificial structure that fits over the mouth of a tile line intake. A filtered intake can be designed to trap sediment and debris while allowing water to pass through at a sufficient rate during a flooding event. A *rock inlet*, in which the intake is buried beneath a pile of gravel, is another option. During runoff events, water is able to percolate down through the gravel and into the tile line, but soil, phosphorus, and other debris gets caught up. Various filtration designs and techniques have been successfully used in other watersheds, and could be potentially used to reduce the amount of upland erosion directly contributing to the contamination of Boone River.

Off-farm Landscape and Hydrology Management

Increasing the capacity of the landscape to process and remove contaminants

It is unlikely that significant reductions in nutrient loading to surface waters will be achieved through traditional, in-field management alone (Schultz et al. 2004, Baker) Denitrification in subsurface soils, wetland outlets, and restored reaches instream can remove much of the nitrate load from drainage waters before it is delivered to downstream surface waters. Denitrification is a significant pathway for N removal in midwestern tile-drained streams during low flow, warm periods (summer and autumn). This accounts for improvements in water quality observed in mid to late-summer (Royer et al. 2004; Schaller et al. 2004), also observed in the Boone River in 2007. Thus, much of the excess N delivered to surface waters locally can be removed by aquatic processing in streams, rivers, wetlands, lakes, reservoirs, and estuaries, given sufficient residence time (i.e., water does not move too rapidly into larger downstream systems).

Riparian and wetland buffers

Riparian vegetation plays many important roles in both natural systems and human-altered ecosystems, acting to buffer surface waters from the impacts of upstream land use. Both above-ground and below-ground biomass can trap sediment, remove and take up excess N and P from runoff and shallow groundwater, and stabilize streambanks. Riparian buffer zones therefore have a significant influence on the quality of water in streams and rivers (Kalkhoff et al. 2000). Riparian buffers located along smaller waterways and streams also provide important habitat and habitat corridors for terrestrial and aquatic wildlife. They provide food, cover, and nesting sites for a variety of bird species.

Riparian buffer zones play an important role in the structural development of habitat conditions instream, providing a source of woody debris, shade, habitat heterogeneity and hydraulic roughness helping to trap sediments and increase residence time of water. Many studies have shown invertebrate taxa indicative of good stream quality are associated with increased numbers of trees. Shade provided by trees can help to limit rapid algal production, suppressing effects of eutrophication.

Redesign drainage system with fluvial and nutrient cycling processes in mind

Compound Channels (aka 2-stage ditch) A compound channel is a naturally occurring feature of any river or stream, referring to the channel's ability to handle two stages of flow: normal flow and flood flows. Normal flows are confined to the meandering banks of the channel, while flood flows are associated with the more expansive, flat, straighter channel of the floodplain.

Surface drainage ditches are typically constructed so as to contain flows as large as the 100 year recurrence interval within the ditch, and thus the bottom of the ditch is typically wider than the channel bottom that would form by natural fluvial processes (Mecklenburg et al. 2001, Ward et al. 2003). Because the ditch provides no floodplain for large flows, many discharges above the 1-2 year recurrence interval that would have been bankfull flows in the natural channel occur as relatively wide and shallow flows within the oversized channel. In areas that are extensively drained by both surface and subsurface drainage, subsurface drainflow contributes the bulk of effective discharges, and the dominant main channel and benches may be formed by discharges which occur much more frequently than discharges associated with natural channels (Mecklenburg et al. 2001).

Open drainage ditches can be constructed to mimic compound stream channels. By widening the ditch beyond the needs of normal flow, the stream can reestablish natural meanders and fluvial features that provide greater diversity of aquatic habitats. Most of the time a portion of the channel will not be inundated, but that area remains available to handle the high flows of spring thaw and storm events.

"Passive restoration" to compound channels may be an option where ditches are already adequately sized. In the absence of regular maintenance, fluvial processes often result in the development of small meandering channels and re-establishment of aggraded sediment benches within the confines of the ditch. Drainage ditches can and do therefore reestablish fluvial features over time, including meandering main channels, benches, small floodplains, riffles and pools. Some artificial channels over time begin to support relatively higher quality aquatic communities than might be expected.

Mecklenburg (2004) provides a brief overview of typical two-stage ditch characteristics and design considerations. Figure 4.2 compares conventional versus two-stage ditch morphology for a 2 mi^2 drainage area in N.W. Ohio. Stages of various recurrence interval storms ranging from very frequent (0.2-year recurrence interval) to infrequent (100-year storm) are shown. While the flow depths of large events are lower in the two-stage design, depths of more frequent events are deeper. Deeper flow has a greater ability to scour and reduce the accumulation of fine sediment building up on the bed.

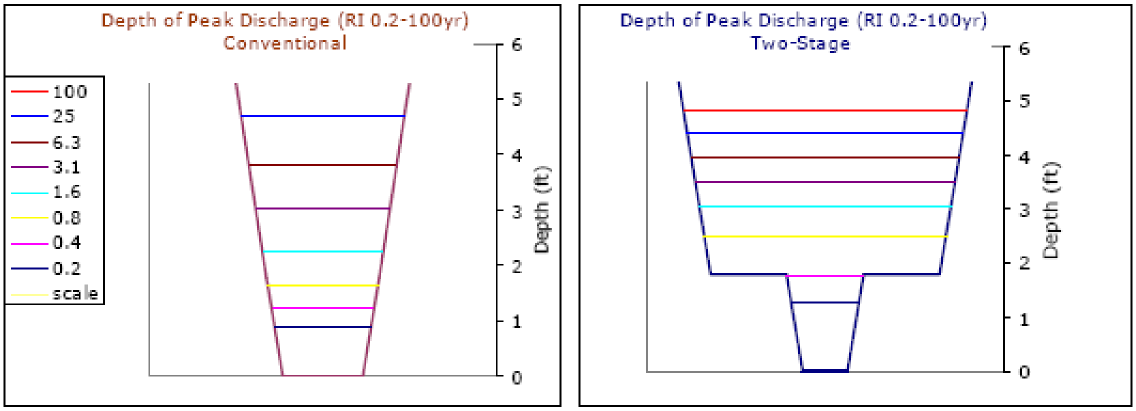


Figure 4.2. Conventional versus two-stage ditch morphology for a 2 mi^2 drainage area in N.W. Ohio, showing stages of various recurrence interval storms ranging from very frequent (0.2-year recurrence interval) to infrequent (100-year storm). [Reprinted from Mecklenburg 2004]

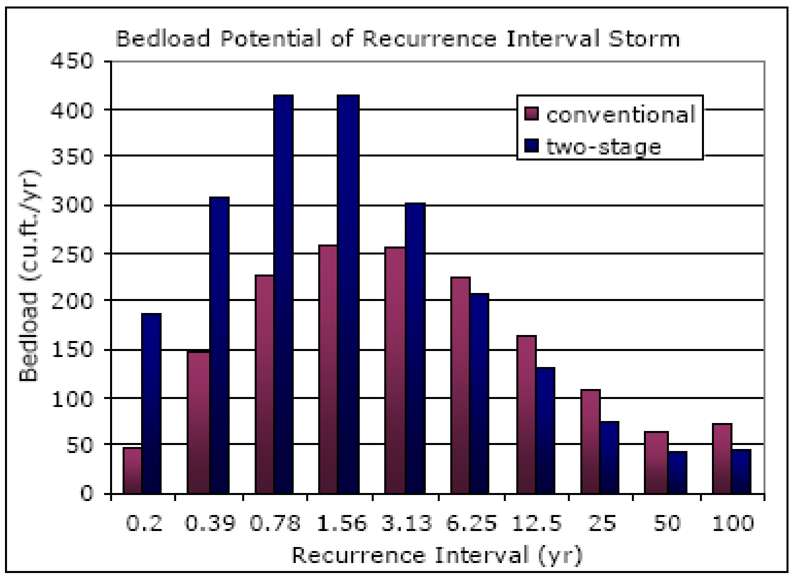


Figure 4.3. Relationship between storm recurrence interval and bedload potential for alternative channel designs. [Reprinted from Mecklenburg 2004]

The two-stage configuration also moderates the bed shear stress (Figure 4.3). During more frequent flows when accumulation of fine sediment is of concern, the shear stress is higher (helping to flush fine sediments). At high flows, when erosion is of greater concern, the shear stress is lower, reducing the total bedload and most erosive events.

Wetland treatment of tile outlet drainage

Nitrate in surface waters is primarily transported from crop land via subsurface drainage, especially in extensively tile drained areas like the Corn Belt. Studies suggest that better nutrient management has some potential to reduce nitrate losses from crop land, but that potential is probably limited to 25% or less (Baker et al. 1997, Crumpton and Helmers 2006).

Although grass buffer strips, woody riparian buffers, and many other practices have been demonstrated to filter nutrients, sediment, and contaminants in surface runoff, these landcape features have little opportunity to intercept nitrate loads in tile-drained landscapes because the drainwaters are delivered directly to surface waters in pipes--completely bypassing landscape filters. Wetlands sited to intercept tile drainage have the potential to significantly reduce nitrate loads, and this approach is particularly promising for heavily tile drained areas like the Corn Belt (Crumpton et al. 2004). Restoration of wetlands has a significant potential to provide the residence time needed to naturally clean contaminated water. Water that is high in dissolved nitrates can be denitrified through biological processes before it seeps out of the wetland and enters groundwater or stream flow.

Wetlands also remove phosphorus and sediment from the water cycle. Slow flow through the wetland causes these solids to settle out of suspension. Movement of fecalcoliform in the wetland environment can be slowed sufficiently to allow microbial processes to break down the harmful bacteria. Water exiting the constructed wetland basin is either routed out through tile lines into ditches, or seeps naturally into the ground below.

The Iowa Drainage District Association (IDDA) has developed an active strategy to address concerns about drainage through efforts to support, advise, and assist the implementation of NO_3-removal wetland technologies through the Iowa Conservation Reserve Enhancement Program (CREP). The IDDA has agreed to partner and collaborate with the Iowa CREP project by providing the necessary linkage to drainage districts, boards of trustees, and affected landowners.

CREP is a pilot program in Iowa under CRP, designed to provide economic incentives to farmers to re-establish wetlands. Farmers choosing to enroll in the program receive monetary subsidies for the acreage they take out of cultivation and turn into a wetland. To prepare a location, a buffer zone is planted around the perimeter and tile lines are routed into the basin. Wetland restoration for a low area near an edge or corner of a field may help make it practical to continue farming around the area. The closer a suitable site is located to a drainage network can also ensure that the wetland site is down gradient from most of the field so that tile lines can be routed toward it.

Urban and suburban nonpoint source control

The Boone watershed is not an area experiencing particularly rapid development pressures. However, the RWA did identify riparian development along the river corridor as a potential priority resource concern for the Boone. Planning is needed to ensure that developments have minimal hydrologic and water quality impacts on the river.

Point source control

Point source discharges from permitted municipal, industrial, and agricultural sources are regulated under the Clean Water Act and facility permits. However, in some cases failing or underperforming municipal or industrial facilities, unsewered communities, and failing septic systems may be responsible for discharges responsible for stream impairments. These should be identified during load analysis required in the development of TMDL plans for state-listed impaired waters. New environmental technologies can be identified that can help achieve additional reductions from point sources.

Strategic Analysis of Alternatives

Three important factors merit consideration in strategic analysis of potential actions and alternatives: (1) benefits, (2) feasibility, and (3) costs.

Benefits

- ***Sufficiency*** towards achieving the threat abatement or target enhancement outcome
- ***Duration*** of outcome
- ***Leverage*** towards achieving *another* important outcome within the conservation area, or elsewhere

Feasibility

- Ability of lead individual/institution to implement strategy
- Ability to motivate key constituencies
- Degree of complexity/difficulty

Cost

- Staffing & direct costs (one-time & recurring)

Sufficiency assesses whether the proposed solutions are in fact adequate to achieve the threat abatement or protection outcome. For example, a practice that results in a very small but significant reduction to a threat may be necessary but not sufficient. Sufficiency is therefore related to whether the scale of the solution matches the scale of the problem.

Feasibility refers primarily to social and technical capacity to implement proposed actions.

Costs should be assessed both in terms of discretionary dollars and human capital. There are opportunity costs involved in both dollars and human capital in terms of funds and time that could be applied for some other purpose.

Costs can also be assessed in terms of both private and public net costs and benefits. The private benefits of reducing sediment and nutrient losses include improved land productivity and sustainability from lowered erosion and fertilizer loss. For example, Goolsby et al. estimated that agricultural fertilizer's share (about 55%) of the 1.8×10^9 kg per year of N added to the Mississippi River's outflow in the 1990s would be worth about $410 million annually if applied as anhydrous fertilizer.

In general, most of the benefits to the water quality and biodiversity are social benefits, while the costs of practices borne by the farmer/landowner may exceed the private benefits. It is therefore often necessary to provide funding to mitigate the private financial costs of socially and ecologically beneficial practices and to achieve adoption at the scale necessary to see biodiversity, aesthetic, and other public benefits.

Targeting

Critical source areas. Research has shown that generally some areas of a watershed contribute disproportionately to contaminant loads, whereas others are potentially better sited to intercept and "treat" pollutants, or are important as critical habitat for conservation targets. Just as the nature of agricultural and land use impacts on conservation targets can vary geographically and temporally depending on the specific characteristics of a given watershed, so too do strategies for reducing the impacts vary in their effectiveness, depending on local soils, cropping systems, tillage practices, topography, rainfall, and agricultural practices. The costs of implementing conservation practices also vary across the landscape; therefore, "targeting" practices where the cost benefit ratio is highest is often the most cost-effective strategy for maximizing benefits from finite conservation dollars (Walter et al. in Schnepf and Cox 2006). As a result, it is critical to target where, when and what actions are pursued in a watershed to maximize their performance and cost effectiveness. Targeting has the potential to increase benefits (sufficiency, duration and leverage), feasibility, and/or to reduce costs.

Due to the size of the Boone River watershed, and given the administrative and on-farm costs of most conservation practices, it is likely prohibitive to apply treatment and programming to the entire basin. In theory, funds can be used more efficiently by targeting resources to the areas needing the greatest attention and to programs and practices that most closely match the type of mitigation needed at a given stage (e.g., farm and area wide planning, wetland restoration, technical assistance, conservation practices). Findings from initial baseline sample collection can be used to define future monitoring efforts.

Areas to be targeted depend on the nature of the problem (e.g. hydrologically sensitive areas, soil test P, nitrate). For nitrate reduction at the scale of the Mississippi River, the natural places to target are identified by the tile-drained landscapes of the Midwest dominated by corn-soybean rotations that have been shown to contribute by far the highest nitrate loads to the Gulf. But within the landscape, there are areas that have much greater natural uptake and processing of nutrients.

Substantial reductions in surface water contamination by sediments and bound nutrients requires changes in tillage practices, crop rotations, and idling of cropland acreage—all of which reduce yields, at least in the short term. Prato and Wu (1996) found that field-level targeting (i.e. targeting conservation compliance to specific fields responsible for the bulk of inputs) was more effective for reducing the level of nonpoint source pollutants and erosion than farm or watershed-scale targeting, but reduced net private and social benefits due to more unequal distribution of-- and lower overall--expenditures. Yang et al. (2003) found under multiple policy scenarios that land parcels targeted for retirement should be those that are highly sloping and adjacent to a water body. Yang et al. (2005) also estimated that current acreage enrollment in the Conservation Reserve Enhancement Program (CREP) in an Illinois watershed was insufficient to achieve the 20% goal set for reducing erosion /sediment, and was four times a least cost solution that involved optimal targeting.

At the scale of the Boone River watershed, the SWAT modeling exercise has been designed to facilitate targeting of appropriate and effective actions towards the least cost/highest benefit lands. Phase II of the SWAT modeling will include a policy analysis that will assess the minimum set of preferred alternatives needed to achieve stated goals.

For example the model could analyze which alternatives could reduce nitrogen & P by the amounts required to meet federal and state water quality standards and aquatic life criteria.

Water quality monitoring conducted at 30 sites by the Iowa Soybean Association (ISA) in 2007 confirms and supports the analysis of STORET data at 46 sites in the Boone River watershed. Combined with the outputs from the SWAT model, these have helped us identify areas of the Boone watershed that are responsible for the highest nutrient loading, and/or where the most abatement can be achieved for least cost.

At subwatershed scales, additional assessment and targeting work is needed for sediment and phosphorus. Identifying critical source areas for soil and phosphorus losses is essential for correct allocation of BMPs (Strauss et al. 2007). Field scale practice data developed for the SWAT model could be adapted to target critical source areas for P at the field level, using the Iowa P index methodology developed by NRCS. The index is a tool designed to help evaluate the current risk from P reaching surface water from a specific site, based on the relative weight of factors which dominate the risk of P transport to surface waters. Description of the method and an Excel spreadsheet for calculating site-level risk is available for download at www.ia.nrcs.usda.gov/technical/Phosphorus/phosphorusstandard.html .

Minimum areas. Although the landscape has a great capacity to absorb, filter, and process nutrients, biological processes in riparian buffers and wetlands have natural limits; thus there is a saturation capacity beyond which additional nutrient uptake and processing does not occur. Wetlands, riparian soils, and other natural landscape features may in fact become a long-term source of phosphorus after reaching saturation with anthropogenic inputs. Thus, the efficacy of wetlands and buffers is subject to threshold / minimum area requirements. I.e., how big does a wetland have to be to treat a sub-watershed of a specific size? The effectiveness of these off-site features in filtering contaminants depends on hydraulic loading rate (a measure of both the amount of water moving through the system and how much nutrients it contains; therefore how likely terrestrial/wetland vegetation and microorganisms will be able to process it) and the source area: treatment area ratio.

At the landscape scale, studies of watershed land use and on-the-ground paired watershed studies in the Midwest suggest that adverse biological effects on aquatic ecosystems can be detected when 20-30% of a watershed is in agricultural land use (Yoder and Rankin 2003). Arbuckle and Downing's (2000) statewide survey of mussel status in Iowa rivers and streams suggested adverse impacts on mussel species richness and communities occur at a much lower threshold, at > 25% agriculture. Signs of degradation of water quality and stream biota become more pronounced at >50-80% agriculture (Wang et al. 1997, Yoder and Rankin 2003). Schilling and Libra found a linear relationship in drained Iowa watersheds (subsurface drainage) for nitrate concentrations, with mean annual nitrate concentrations in mg/L corresponding roughly to 10x the percentage of the watershed in row crop land use (i.e., 10% row crop = 1mg/L nitrate; 90% row crops = 9 mg/L). This would suggest that only watersheds with less than 20% row crop could meet water quality criteria of 2 mg/L nitrate.

At more than 80% row crop plus at least 4% pasture, the Boone River watershed is well above all these thresholds. However, Yoder and Rankin (2003) also conclude that with effective targeting of appropriate best management practices, buffers, and other conservation land uses, clean water act goal uses can still be attained at up to 70-80% agricultural land use.

Below we include a literature review and analysis exploring the sufficiency and effectiveness of proposed conservation strategies described in the previous section. Further information and evaluation of effectiveness of these practices in Iowa can be found in a comprehensive assessment of conservation practices produced by Dana Dinnes of the National Soil Tilth Lab for the Iowa Department of Natural Resources (Dinnes 2005). A copy of this publication is included in Appendix D. The reference section includes an online web address for obtaining the complete document in Adobe Reader (pdf) electronic format. Further discussion of the feasibility and applicability of different conservation practices in the Boone River is discussed in the context of the analysis, including additional research and analysis needs.

Onfarm changes in practices

Nutrient Management -- Rate and Timing

Any field that receives fall N fertilizer applications may benefit from shift in timing to spring, late-spring or early summer time periods, when there is less time for the nutrients to be lost before the crop reaches the stage where it can use them (Dinnes 2005).

Practice	Range of Effectiveness
Timing	
Spring Pre-Plant vs. Fall Application	-25% to +50%
Soil-Test Based Split In-Season vs. Fall Application	-25% to +70%
Soil-Test Based Split In-Season vs. Spring Pre-Plant	-50% to +70%
Rate	
Yield Goal or Crop Removal Based vs. Excessive	+10% to +90%
Soil-Test Based vs. Excessive	+10% to 90%
Soil-Test Based vs. Yield Goal or Crop Removal Based	-50% to +70%

Source: Dinnes 2005.

The effectiveness of shifts in rate, timing or form is subject to a substantial range of variation, based on seasonal climatic variability in temperature as well as timing, duration, and intensity of rainfall, especially following application. Soil conditions are often too wet for equipment trafficking in the spring, and greater than normal precipitation may lead to N deficiencies in corn if N is applied based on normal conditions. There are potential constraints on availability and cost of high-clearance equipment for practices that include late-season N application, as well as on farm labor during spring planting season. Commercial N fertilizers are also typically more expensive in the spring and late-spring/early summer time periods than in the fall

Saleh et al. (2007) used measured values of water quality indicators from the Walnut Creek watershed (WCW) in central Iowa to verify the capability of SWAT-M to predict the impact of late-spring nitrate test (LSNT) and rye cover crop management on NO_3-N reduction at the subbasin level. The results obtained from SWAT-M simulation results, similar to field measurement data, indicated a 25% reduction in NO_3-N under the LSNT scenario. A farm level economic model was also used to estimate the cost of practices, and showed a corresponding increased annual cost of $6/ha across all farms in the watershed for the LSNT.

Nitrogen management trials conducted in the Boone River under the auspices of the Iowa Soybean Association (ISA) using the corn stalk nitrogen test have suggested that the optimal fertilization rate and form may differ in the Boone River from those promoted by Iowa State University based on research throughout Iowa (Blackmer, 2008). Additional work is needed to refine recommended nutrient BMPs for the Boone to improve efficiency of fertilizer application in the Boone.

Conservation Tillage

The effectiveness of tillage methods for reducing P losses depends on numerous factors, including crop rotation and crop present at time of consideration, soil types, slope and slope length, climate, conditions during rainfall events, event duration and intensity, timing of applications, etc. Especially on fields where there are relatively high erosion rates, reducing tillage can be more beneficial for reducing P losses as long as P fertilizers and manure are knifed or injected into the soil with minimal soil disturbance.

The degree of P loss reduction depends on type of tillage systems being compared; more P loss reduction is possible when changing from a moldboard plow tillage system to no-till than from a chisel plow tillage system to no-till. Reductions in TP that can be achieved with conservation range from 25-80% moving from intensive tillage to moderate, 30-60% from moderate to no-till, and 50-90% from intensive to no-till (Dinnes 2005). Large rainfall events following P fertilizer or manure application in soils characterized by tile drainage or macropores may lead to elevated soluble P leaching losses via preferential flow, though sediment-bound P losses from reduced runoff and erosion will still be reduced. Zimmerman et al. (2003) used the ADAPT model to estimate a watershed-wide reduction in runoff of 35% from a combination of increased conservation tillage, riparian buffers, and permanent vegetative cover.

There may be significant potential for increased adoption of conservation tillage in the Boone River watershed because a large percentage of croplands are still managed under conventional tillage. This is partly because poor field drainage in heavy soils such as those prevalent in the Boone pose management difficulties for no-till (Dinnes 2005). These problems can be overcome with proper practices (see Appendix D). There is a transition period from conventional and reduced tillage systems to no-till as field soils develop improved physical properties under no-till. Increased adoption of conservation tillage may be accelerated with financial incentives such as those being offered by NRCS (see Appendix D).

Cropping systems

Choice of cropping system significantly affects nutrient losses from agricultural lands. Smith et al. (1993) showed that nitrogen yields in streams draining corn and soybean agriculture were twice that of streams draining urban areas, and streams draining wheat agriculture carried less nitrogen than either urban or corn soybean catchments. High nitrate concentrations were found in streams with the highest percentages of corn and soybeans, and probabilities of exceeding the 10 mg/L water quality standard increased dramatically as the percentage of corn increased. Mueller et al. (1993) found the most important basin-scale variable explaining variation in nitrate concentrations throughout the corn-growing region of the Midwest was the percent of land upstream used for growing corn and soybeans. McIsaac and Hu (2004) also reported the largest N fluxes in the Mississippi River from agricultural basins dominated by corn-soybean production with extensive subsurface drainage, i.e., southern Minnesota, Iowa, Illinois, Indiana and Ohio.

The influence of cropping systems is related to the effect on drainflow volumes, and to a lesser extent implications of crop choice for long-term nutrient management. Reduced nitrate loads observed under alfalfa, CRP, or other perennial crops relative to corn or soybeans are closely tied to differences among crops in water use (Drury et al. 1996, Randall et al. 1997, Chung et al. 2001, Bahksh et al. 2002, Kanwar et al. 2005). All else being equal, total annual basin water yield is generally greater from corn/soybeans and other row crops than from perennial crops or pasture. Decreases in perennial crops (i.e., pasture and alfalfa) typically results in less evapotranspiration in April, May, and June and larger nitrate-N losses (Randall et al. 1997).

Studies across the humid Midwest consistently show significant reductions in runoff and associated contaminants with conversion from annual row crops to perennial vegetation. Updegraff et al. (2004) applied a field-scale runoff, sediment and nutrient transport model (Agricultural Drainage and Pesticide Transport, ADAPT; Ward et al. 1988) to simulate the hydrologic effects of converting 10, 20 and 30% cropland to short rotation woody crops, grown on a 5-year rotation, in a Minnesota River sub-watershed. At the highest conversion level, mean annual runoff was reduced by up to 9%, sediment loads by 28% and nitrogen (N) loads by 15%, although total phosphorus (P) loads increased by 2%. The benefits of conversion at the field level were contingent on soil type, drainage status and alternative crop. For a 52 km^2 low-relief watershed in central Iowa that is 90% farmed and 75% tile-trained, Chaplot et al. (2004) used SWAT to explore 9 different cropping system and N management scenarios over a 30 year simulation period. Converting from corn-soybeans to pasture reduced discharge, sediment, and NO_3-N by 58, 50, and 97% respectively. Prato (1995) found that cropping systems that efficiently decreased sediment loss were less expensive and differed from those that efficiently decreased nitrate concentrations.

The Boone River SWAT model showed potential water quality benefits from shifts in cropping systems that might be possible if markets develop in the future for cellulosic biofuels. Converting 50% of croplands to perennial grass cropping systems reduced losses of sediment by 20-25%, nitrate and mineral P by 10-15%, and organic N and organic P by nearly 50% (Gassman et al. 2008). The UMRB scale SWAT model developed by CARD was also adapted to explore water quality impacts of future cellulosic biofuels scenarios (Secchi et al. 2008). Several biofuels scenarios resulted in water quality improvements (i.e. decreases in sediment and nutrient losses relative to corn-soybean and continuous corn rotations from expansion of perennial grass cropping systems). However, the model also showed that the most economically efficient targeting of biofuel expansion was in watersheds closer to the mainstem of the Mississippi River with more highly erodible land, and thus did not include the Boone (Secchi et al. 2008). The report acknowledged that much is unknown regarding development of markets for cellulosic feedstocks, and that subsidies will almost certainly be needed to stimulate the development of a cellulosic fuels industry if it is to compete with established corn-based ethanol.

Cover cropping

Fall cover cropping in corn-soybean rotations has the potential to be an effective management tool for reducing NO_3–N loss from subsurface drainage discharge, despite challenges to establishment and spring growth in the north-central states (Logsdon et al. 2002, Strock et al. 2004). Rye cover crops in particular can result in significant reduction in NO_3-N in tile drains (Jaynes et al. 2004, Strock et al. 2004). Research conducted on a moderately well-drained soil in southern Minnesota found that the rye cover crop did not reduce soybean yields, but did reduce drainage discharge, flow-weighted mean nitrate concentration (FWMNC), and NO_3–N loss (13%) relative to winter fallow. The magnitude of the effect varied considerably with annual precipitation. Three-year average drainage discharge was 11% lower with a winter rye cover crop than without ($p = 0.06$), while nitrate loss was reduced 13%.

Saleh et al. (2007) used SWAT-M to model scenarios combining multiple practices, including combinations of late spring nitrate testing and winter cover cropping. Results showed a progressive reduction in sediment and nutrient losses as adoption rates of both practices increased. Use of the rye cover crop added about $25/ha to $35/ha to the annual cost of the average farm, indicating that some cost-share support may be necessary to encourage farmers to use winter cover crops.

The Boone River SWAT model also showed significant (50%) reductions in annual organic-N losses with a cover crop (Gassman et al. 2008). Reductions were considerably more modest for sediment, organic and mineral P.

Controlled Drainage / Drainage Water Management

Controlled drainage, aka drainage water management (DWM), creates wet, anaerobic environments upstream of the restriction that can result in beneficial denitrification. DWM has been found to reduce nitrate loss at the field scales by 20-40% percent over conventional (free flowing) subsurface drainage. However, Randall (2004) suggested the practice may be of limited value in northern regions where the majority of drainflow and nitrate loss occurs during the winter or early spring when field operations require lowered water tables. Retention of water to minimize nitrate loss earlier in the year (fall or late winter) would also potentially conflict with the goal of minimizing flood risk by lowering water tables to maximize soil storage capacity to accommodate spring snowmelt and precipitation.

To evaluate the potential for controlled drainage (drainage water management) to reduce NO_3 contribution to Gulf hypoxia, Dan Jaynes, a soil scientist at the Soil Tilth Lab in Ames, and colleagues (2007) recently developed a regional scale GIS spatial model. The model predicts that if drainage water management was implemented on all suitable land in the UMR, 8% of the total NO_3 load to the Gulf could be removed. Assuming a subsidy to the landowner for the extra cost involved, the cost for N removal was estimated at roughly $1/kg N removed, which is comparable to other costs for N removal.

The model suggests that controlled drainage may be applicable to a relatively small fraction of tile drained land in Iowa. Accurate estimates were available for a few large drainage districts in north central Iowa for which very high resolution topography have been developed. Although 50 to 75% of the cropland in these drainage districts is tile drained, only about 10% has a slope less than 1% and only about 3% has a slope less than 0.5% (Matt Helmers, Iowa State University, Ag Drainage Website, http://www3.abe.iastate.edu/agdrainage). However, the model may not be representative of other regions of the Corn Belt. Based on soils information maintained by NRCS, Illinois, Indiana, and Ohio may have twice as much cropland suitable for controlled drainage as Iowa (Jaynes 2007).

Figure 5.1 shows a map of cropland areas within the Boone River watershed that potentially satisfy criteria for Drainage Water Management. Analysis of the statewide DEM (at 30 m resolution) indicates that perhaps just under 5000 acres of currently cropped lands within the watershed have slopes under 0.5 – 1%, and are therefore potentially suitable for controlled drainage, or just 1-2% of the watershed. Higher resolution topography (e.g. LIDAR) will soon be available, and could provide a much better basis for this assessment.

A larger portion of the watershed may be eligible for economically feasible drainage water management-- especially if crop prices remain high-- using the subirrigation (controlled drainage on the contour) design being pioneered in the Mackinaw River, IL by Agrem in cooperation with The Nature Conservancy.

Filtered tile intakes

Research at a demonstration farm in southern Minnesota has shown that approximately 20% of the sediment bound contaminants delivered to a depression by runoff enter an open inlet (Rainaivoson et al. 2002). By replacing the open inlet with a gravel filter, losses of sediment bound contaminants was reduced 20-28% (Ranaivoson et al. 2002) The gravel filter preferentially trapped sediment with higher P concentrations. However, survey evidence suggests that farmers are extremely skeptical about filtered intakes, as they consider infeasible any practice that will potentially interfere with field drainage (Casey et al. 1995). ISA and Prairie Rivers RC&D have developed a proposal to implement tile intake filters in the northern Boone. However, the proposal has not yet successfully been funded.

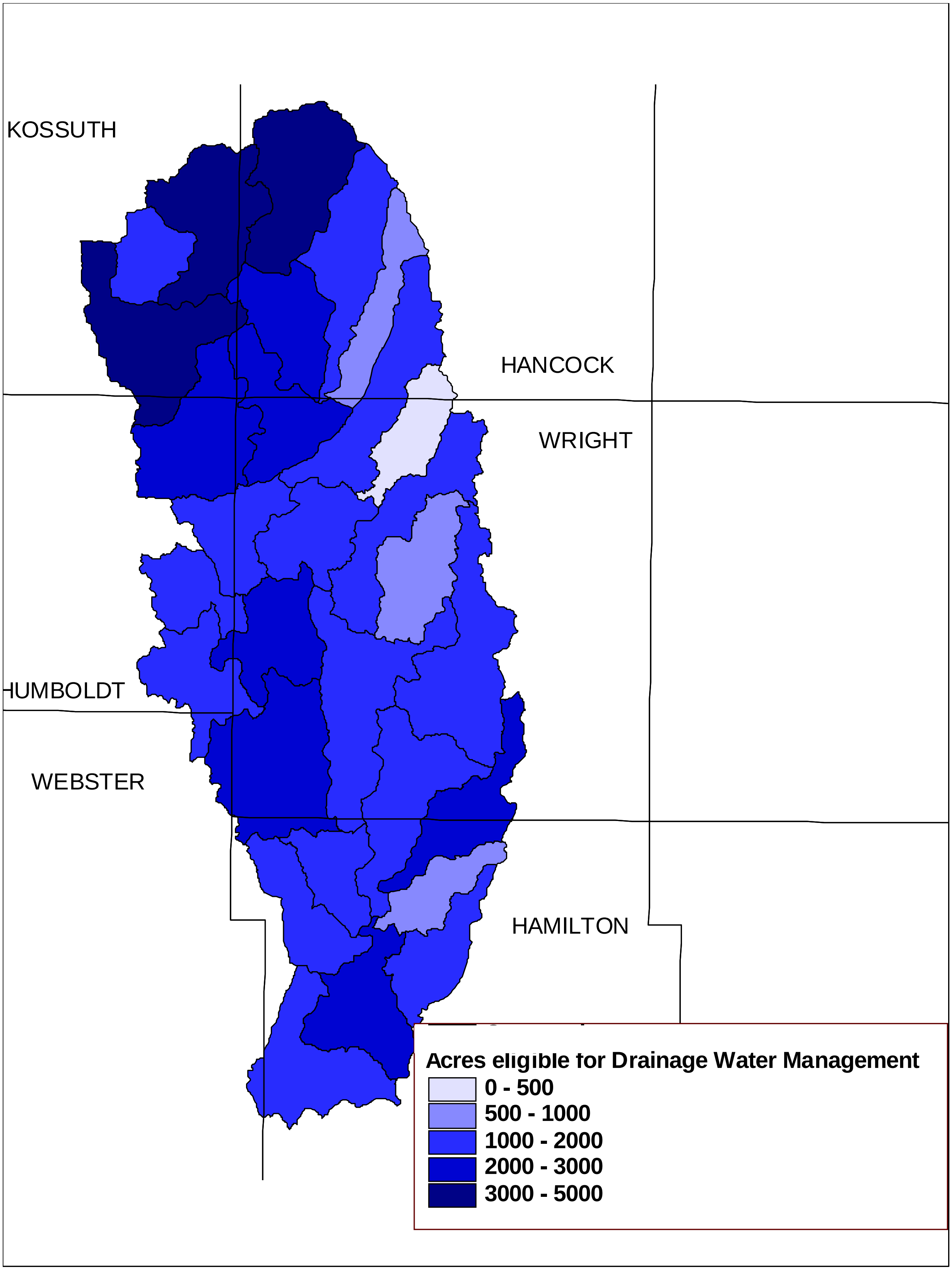


Figure 5.1. Estimated acreage with potential for cost effective controlled drainage (cropped lands with < 0.5-1% slope), by 12-digit watershed.

Off-farm practices: Landscape restoration

Many researchers have concluded that although important, on-farm nutrient management can not achieve the reductions needed to meet local water quality standards or to address hypoxia. For example, most cost effective BMP's generally reduce nutrient and sediment losses by less than 20% in their current condition (Bracmort et al. 2006), yet the 2008 Gulf Hypoxia Action Plan requires dual nutrient removal of 45% of the N and P load.

Wu et al (2004) developed a Mississippi River scale empirical model to estimate the effects of farmers' production practice decisions in response to alternative conservation policies. Although payments for conservation tillage and crop rotations did increase adoption of those practices, the programs were not cost effective on their own for addressing hypoxia. The Boone River SWAT model indicated that even a 100% reduction in N fertilizer could not achieve 45% reduction in N and P loss. Clearly this is an unrealistic scenario given the negative impacts on crop yields.

The limits to effectiveness of on-farm BMPs—with the possible exception of cropping system changes—is related to nutrient dynamics both on-farm and at the landscape scale. At the farm scale, despite the use of best management practices for nitrogen (N) application rate and timing, significant losses of nitrate nitrogen in drainage discharge continue to occur from row crop cropping systems (Strock et al. 2004). The extensive network of subsurface drainage pipes means that leaching through the soil profile will continue to deliver excess nitrate directly to surface waters, even on fallow ground with no fertilizer inputs (Randall 2004, Tan et al. 2002, Dinnes et al. 2002).

At the landscape scale, the problem of nutrient loading to surface waters is driven by the alteration of hydrology and reductions in landscape function resulting from wetland drainage, riparian cover removal, land use changes, and drainage system modifications including channelization, ditching, and tiling. These changes are at least as important as nutrient additions from fertilizer.

Research on streams has shown that, in comparison to larger rivers, small streams remove a higher proportion of their incoming nitrogen per unit of water travel time (Alexander et al., 2000), per stream reach (Seitzinger et al. 2002), and per unit length (Wollheim et al. 2006; Helton 2006). However, larger streams remove larger masses of nitrogen because more nitrogen passes through them (Seitzinger et al. 2002, Wollheim et al. 2006, Helton 2006). Small streams receive and transport a significant amount of N to larger rivers, because N loads to headwaters account for 45% of the load delivered to the entire river network in the northeastern U.S. (Alexander et al. 2007). Improvements due to small decreases in agricultural N losses might be amplified downstream by augmented processing of lower concentration, slower moving drainage waters through restoration of landscape features such as stream channels, riparian buffers, and onstream wetlands.

Most nitrate is exported from croplands during high flows from January to June (Royer et al. 2006), and denitrification removes an insignificant fraction of this flux (Royer et al.

2004, 2006). It is critical to focus on enhancing in-stream removal during this period of high flows, because removal during low flows (Q < median) has a relatively insignificant impact on annual loads (Royer et al. 2006).

The following "off-farm", or landscape restoration practices are recommended as part of an overall strategy for the improvement of water quality and aquatic biodiversity in the BRW.

Compound channels and 2 stage ditches

In agricultural landscapes that have been extensively altered and channelized, agricultural drainage ditches may provide a majority of the available headwater stream habitat in a watershed. In Ohio, Smiley and King (2006) found that drainage ditches represent 25% of stream habitat miles, and suggested that management actions that alter the hydrology of ditches will exhibit a greater impact on fish communities than other types of management actions.

Three environmental benefits can be achieved by constructing compound channel ditches:

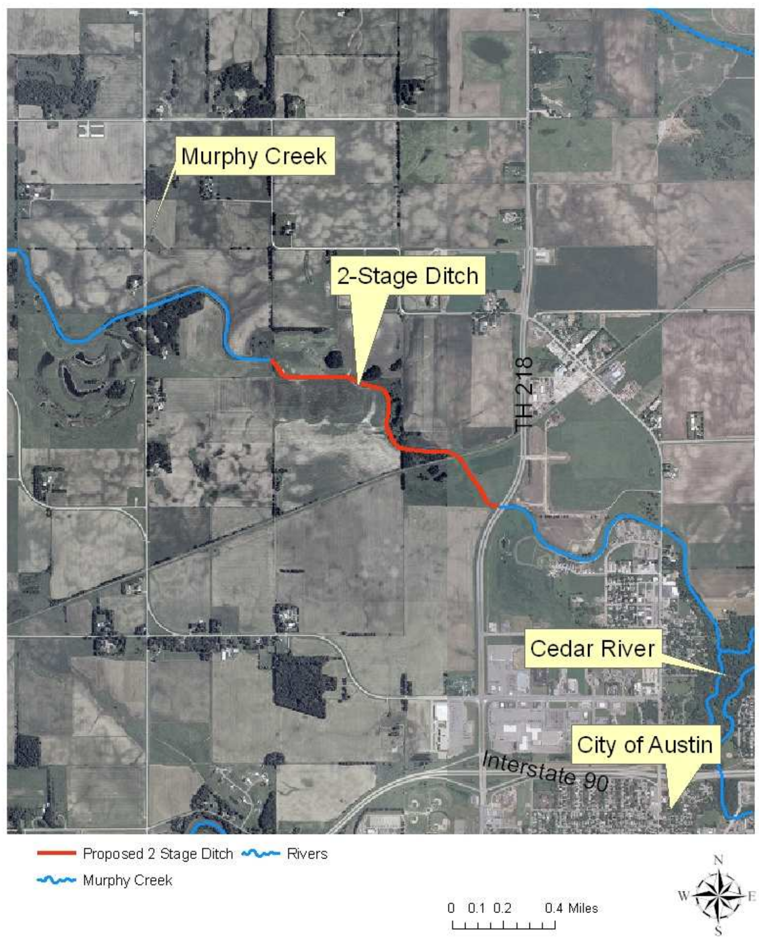
1. Channel complexity and fluvial features such as meanders increase storage and decrease flow velocity. By dissipating the water's energy around every bend, the erosive capacity of flows is reduced.
2. Slower moving water can not carry such high sediment loads. Rather than transporting sediment from uplands to the river and its outlet, portions of the load will be deposited in buffers, throughout the meanders on accretionary or depositional point bars. The floodplain provides another surface onto which sediment can be deposited during the recessional curve of flood flows, without causing sedimentation problems for benthic organisms in the channel.
3. As flow velocity is dissipated throughout the channel, the residence time of water in the channels is increased, allowing for greater instream processing and uptake of nutrients and contaminants.

Economic benefits from compound channels include potential reductions in long-term ditch maintenance costs, fewer and less frequent need for ditch clean-outs, and reduced ditch bank erosion (Mecklenburg et al. 2001, Ward 2004). In Michigan, two-stage ditch designs are showing up to 45% reduction in nitrates and up to 40% reduction in TP load (Ward et al. 2006). Compound channel experiments in Indiana are showing 10-30% reductions of nutrients.

When evaluating the need for and feasibility of 2-stage ditch designs versus natural stream restoration or simply passive recovery and natural re-establishment of riparian buffer vegetation, several factors should be considered. Can restoration of a riparian buffer initiate passive restoration via natural stream processes? If so, structural restoration may not be the most cost-effective solution in the long run. Is the channel incised and therefore cut off from the floodplain via high terraces and previous bank failures? If so, a 2-stage ditch may be the most cost-effective approach, requiring the least amount of land to be taken out of production.

In southern Minnesota, a consortium of groups has recently implemented several experimental compound channel projects. The University of Minnesota is conducting a paired-ditch evaluation project at its Southwest Research and Outreach Center in Lamberton to evaluate the performance of two similar ditches to remove nitrogen under varying physical and flow characteristics (Busman and Sands 2002).
Additional ditch studies are occurring at the Southern Research and Outreach Station at Waseca, MN. The Nature Conservancy in Minnesota is also involved in a proposed 2-state ditch study in the Minnesota portion of the Cedar River, just upstream of Austin (see map). Research results are only just beginning to be obtained and analyzed.

Murphy Creek 2-Stage Ditch



Buffers

Riparian buffers have the potential to remove as much of 90% or more of a range of contaminants in a variety of landscapes (Table 5.3; Simpkins et al. 2002, Schultz et al. 2004, Dinnes 2005, Gregory et al. 2006). Research shows, however, that the effectiveness of riparian buffers in trapping or filtering contaminants can vary significantly depending on a number of site-specific factors. The width and vegetative composition of the buffer, whether the buffer is continuous or patchy, and whether it is located so as to intercept the bulk of runoff or drainage from cropland and other factors may affect the proportion of runoff water that is filtered through the buffer and the length of time water is in contact with the buffer. In highly sloping landscapes where runoff events may occur very rapidly, for example, large amounts of runoff water may pass so rapidly through the buffer that little of the sediment or nutrients contained in the runoff are removed. Likewise, in tile drained landscapes or settings where groundwater flows to the stream pass below the root zone, drainage waters may bypass the buffer almost completely. Where groundwater passes through riparian zones, they have substantial denitrification potential (Lowrance et al. 1997, Groffman et al. 2002, Gregory et al. 2006). Therefore, targeting buffers where they can be the most effective is important (Bentrup and Kellerman 2004, Burkart et al. 2004, Dosskey et al. 2005), along with better coordination of restoration efforts at larger scales (Gregory et al 2006).

Table 5.3. Reported effectiveness of riparian buffers for reducing nonpoint-source pollutants (runoff, sediment, nutrients and pesticides).

Parameter	Range (%)	Mean (%)	n	Source
Runoff	21-88%	51	8	Gregory et al. 2006
Biological oxygen demand	18	18	1	“
Ammonium	28-87%	65	9	
Nitrate (runoff)	9-99%	69	13	
Nitrate (subsurface)	49-91%	72	6	
Phosphate	36-98%	73%	8	
Total Kjeldahl nitrogen	11-79%			

From Gregory et al. 2006

Buffer strips are more effective in decreasing transport of sediment and sediment-carried nutrients in overland flow than they are in removing dissolved nutrients. Transport of soluble nutrients can be decreased if runoff water infiltrates the buffer strip area.

Buffer strips can also be effective in removing nutrients from subsurface drainage if shallow groundwater move laterally through subsoils in the riparian zone, in other words, if no “short circuiting” or bypassing is provided by artificial drain pipes.

Riparian buffers can also play an important role in stabilizing stream sediments (Wynn 2006). Vegetation indirectly affects soil erosion by changing soil physical and chemical properties including soil organic matter, aggregate stability and bulk density (Wynn 2006). Woody and herbaceous roots significantly increase slope stability over bare conditions, acting to stabilize banks by increasing soil shear strength (Simon and Collison, 2001).

Odgaard (1987) studied erosion along meander bends of two major rivers in Iowa and determined that erosion along wooded streambanks was half that along sparsely vegetated banks. In Bear Creek, Iowa, Zaimes et al. (2006) riparian forest buffers had significantly lower magnitude of streambank erosion and total soil loss than other riparian land uses. Establishment of riparian forest buffers along all of the nonbuffered subreaches would have reduced stream-bank soil loss by an estimated 77 to 97 percent, significantly decreasing sediment in the stream.

Qiu and Prato (1998) used SWAT to estimate water quality benefits of riparian buffers by combining experimental data and simulated water quality impacts of farming systems. Results found net economic value of riparian buffers in reducing atrazine and greater savings in government cost, and strongly supported efforts that encourage farmers to develop or maintain riparian buffers adjacent to streams.

Lovell (2006) recently reviewed the environmental benefits of conservation buffers in the U.S. and explored the reasons why they have been underutilized, and suggested that many important questions related to performance and implementation have not been answered. They suggest additional multidisciplinary research on aesthetic and economic issues related to buffer adoption, and recommended modifying policies to better reflect the preferences of landowners and local conditions.

Minimum area effectiveness for nutrient removal. Most research on nutrient removal by buffers has been conducted on small areas with small source-area-to-buffer area ratios not representative of actual conditions in most of the watershed. Across the studies depicted in Table 5.3, the source-area-to-buffer ratio ranged from 0.4:1 to 55:1 (median: 5.5:1). Gregory et al. (2006) estimated that 18% of agricultural land would have to be converted to buffers to achieve this ratio. Dosskey et al. (2002) also attempted to depict the performance of buffers at a range of source-area-to-buffer ratios (Figure 5.2).

Targeting. Cumulative water quality benefits downstream also depend on placement of buffer restoration in the landscape. Buffer performance is improved where shallow groundwater flow system channels water through the buffer at velocities that allow for denitrification and uptake (Simpkins et al. 2002). Burkart et al (2004) developed hydrologic and terrain analysis methods to aid in strategic location of riparian buffers, using elevation and streamflow data to develop indices of sediment trapping efficiency, groundwater interception, and flow. These indices can be used to target areas along the drainage network where buffers would be likely to provide the greatest water quality benefits. In general, areas along first order streams had a much greater opportunity to intercept significant proportions of water than did areas adjacent to larger streams (Burkart et al. 2004). Significantly smaller values of the sediment transport index along smaller streams also provide enhanced opportunities for deposition of sediment and associated contaminants.

Wetlands

Wetlands --constructed, restored, reconstructed, or natural-- have the potential to remove N in flow-through water and also can remove some P. Although empirical studies show 30-90% nitrate reductions from Iowa wetlands, Crumpton (2007) feels 70-80% is possible if wetlands are appropriately sited. At $1.32/lb removed, they concluded wetlands are much cheaper than the “next best” option. With the exception of P associated with suspended solids, wetlands are generally less effective at retaining P than at removing NO_3 (Reddy et al. 1999). Unlike N, P cannot be permanently converted to inert atmospheric forms, but rather continually cycles through the landscape being released and taken up. Thus a critical factor for P reduction is overall watershed balance (input from sources versus output in crops and vegetation).

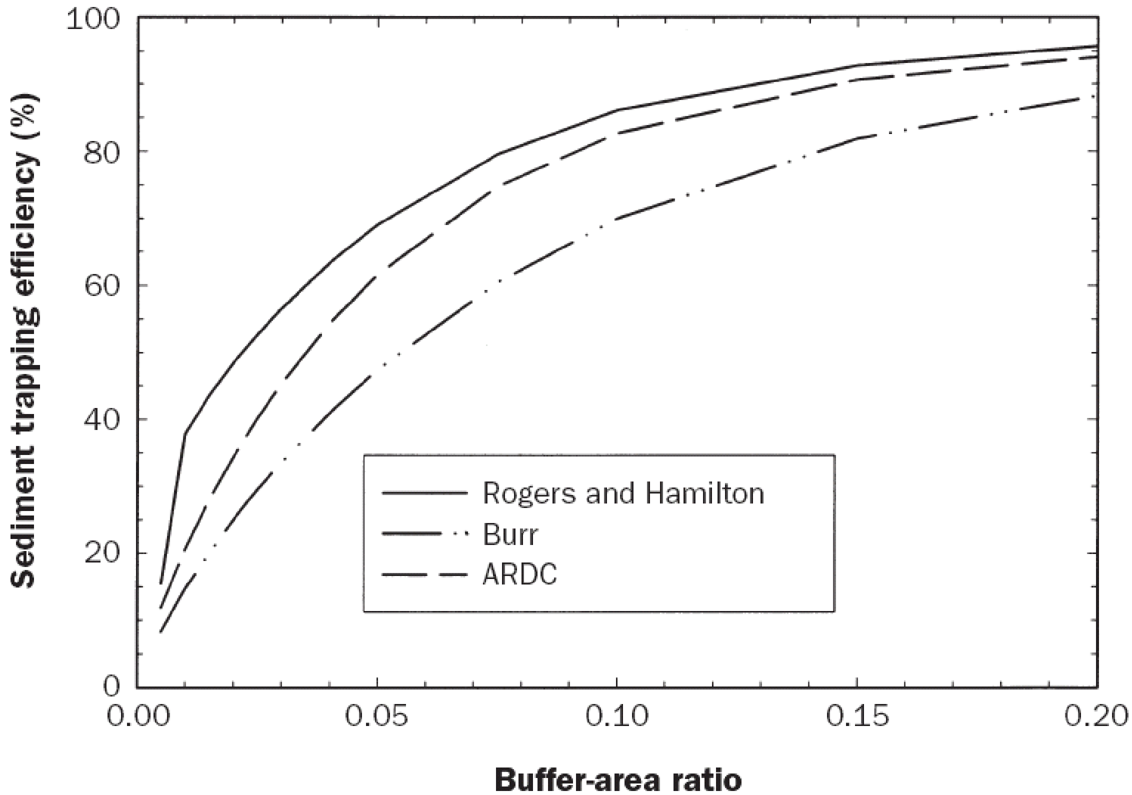


Figure 5.2. Relationship between sediment trapping efficiency (i.e. percent of input load retained by the buffer) and buffer-area ratio developed for conditions on four farms in Nebraska. Buffer-area ratio = (Buffer area/field runoff area.) Figure from Dosskey et al. 2002.

If water storage, retention and wetland treatment were to be pursued in the Boone River, where might this water be stored? The CREP wetland program has several criteria or requirements for siting. Wetlands must be sited where they can intercept tile drain flows and pollutants downstream of agricultural lands. They must represent at least 0.5-2% of the drainage area, with a minimum area drained of 500 acres. To ensure that wetlands are sited appropriately in the landscape, areas with hydric soils are preferred.

Because the region was historically rich in wetlands, opportunities for wetland restoration are numerous, and there is considerable topographic potential for restored wetlands to intercept tile flow. Potential sites are defined by existing topography and hydrology. Aerial photographs taken in a series after a period of heavy rain may help identify portions of the watershed that tend to pond after a storm event, and could be used to map out areas in the upland region that are naturally susceptible to ponding. Areas that represent natural depressions in the landscape, aka, “sinks” representing natural “pothole” depressions, may also be identified from a DEM using standard ArcView Spatial Analyst tools (ESRI 2007; see example in Figure 5.3). Topographic analysis of the 90m Iowa Digital Elevation Model (DEM) shows there are ~22,000 acres of “sinks”, or areas representing natural depressions in the Boone River watershed (compare to the 21,900

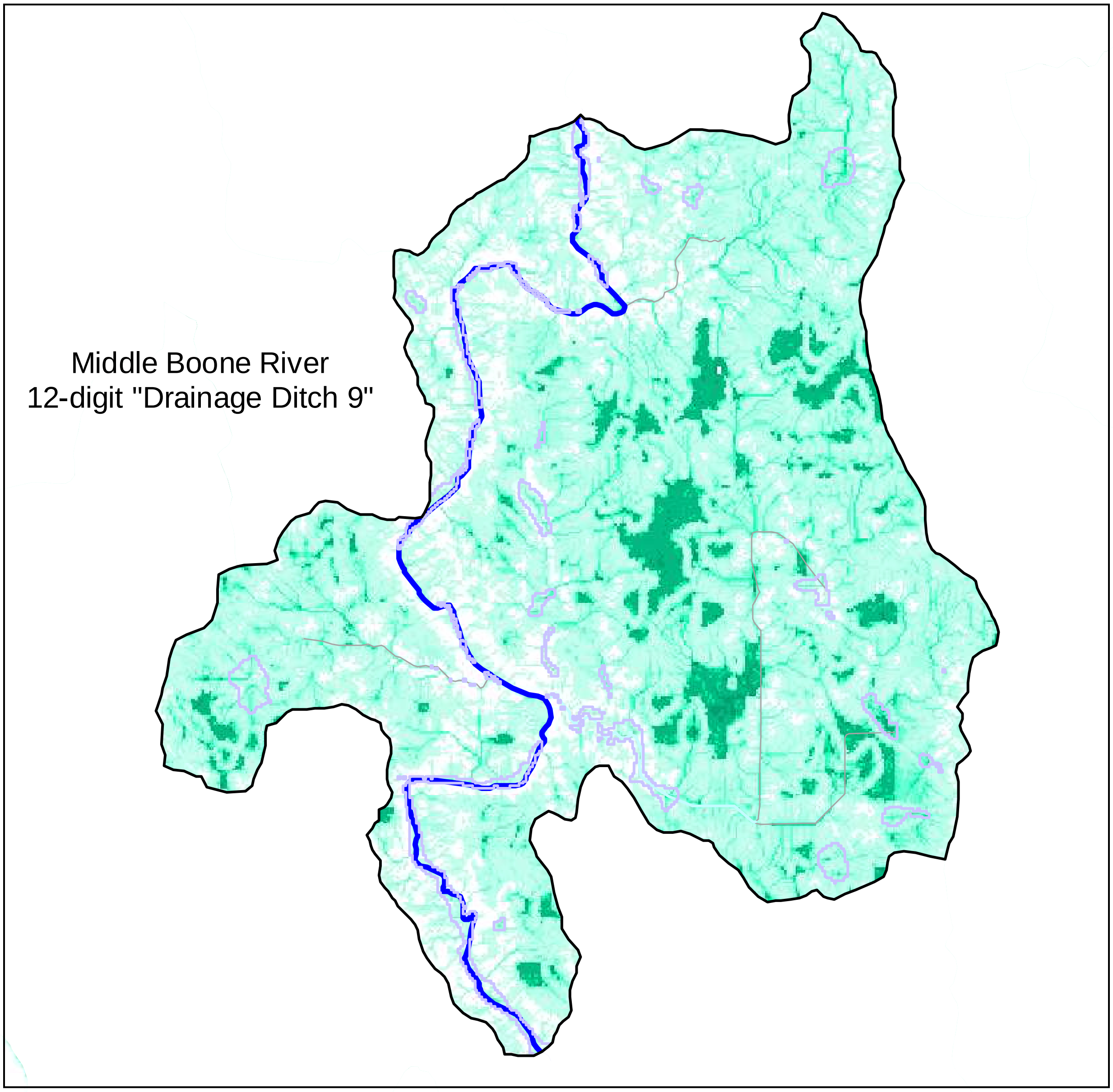


Figure 5.3. Example from a Middle Boone River 12-digit watershed, showing, as predicted based on the Digital Elevation Model, (a) soil wetness (shown in graduated shades of green) and (b) landscape depressions ("potholes" or "sinks", shown outlined in lavender) that potentially meet wetland/buffer siting criteria.

acres identified as wetlands by the GLO estimate of presettlement cover). These depressional areas already represent areas of topographically constrained drainage, given that only 67% of these acres are in row crops. The majority of these depressional areas meet the CREP criteria for upstream watershed area greater than 500 acres. The stream network intersects depressional areas totalling 11,475 acres; the rest occur in headwater locations upstream of the existing drainage network. These areas may also represent a starting point for exploring the possibility of riparian and wetland restoration.

Standard Arc GIS hydrologic tools can also generate a “soil wetness index”, predicting areas most likely to have saturated soils based on flow paths, defined as significantly ($p<0.05$) greater probability of saturation. Because riparian areas along first order streams have greater potential to intercept groundwater or runoff than similar areas along larger streams (Burkart et al. 2004), the “wetness index” may help identify areas where riparian buffers are likely to provide the greatest benefit. Figure 5.3 depicts the soil wetness index for a sample 12-digit subwatershed in the Boone, generated based on the DEM. Targeting buffers in these areas enhances the potential for groundwater interception. Evaluated in relation to existing land cover, the model suggests that existing natural cover in near-riparian areas is already highly associated with soil wetness.

Minimum effectiveness based on source area: treatment area ratio

The Conservancy’s paired watershed demonstration projects in the Mackinaw River, Illinois subwatersheds are showing ~40-90% reductions in nitrate from multiple wetlands and some early success at removing phosphorus as well. Initial results from the wetland treatments in the Mackinaw River suggest that a treatment area representing 5-9% of the source area may be needed to achieve the 80-90% reductions necessary to meet local water quality objectives. Research in Iowa (Crumpton and Helmers 2006) suggests wetland area as small as 2% is sufficient to treat cropland runoff for nutrient runoff, if wetlands are appropriately sited.

Based on a restoration criteria requiring that the treatment area must represent at least 1%, and probably closer to 5-9%, of the upstream watershed cropped or urban area to achieve effective wetland area: source area ratios, then between 5000 – 45,000 acres of wetlands or riparian areas are needed downstream of croplands in the Boone River watershed.

Minimum effectiveness area based on hydraulic loading rate (HLR)

To estimate potential nitrate removal by wetlands across the same grid area, Crumpton and Helmers (2006) used mass balance simulations to estimate percent nitrate reduction for hypothetical wetland sites distributed across the UMR and Ohio River basins. Results were used to develop a nonlinear model for percent nitrate removal as a function of hydraulic loading rate (HLR) and temperature. Mass nitrate removal for potential wetland restorations distributed across the UMR and Ohio River basins was estimated based on the expected mass load and the predicted percent removal (Figure 5.4). These functions explained most of the variability in percent and mass removal reported for field scale experimental wetlands in the UMR and Ohio River basins. They concluded 1-5% of the watershed would be sufficient to handle the majority of the N load, and that across the UMR and Ohio River Basins a 30% reduction in nitrate load could be achieved using wetlands targeted towards the highest nitrate contributing areas.

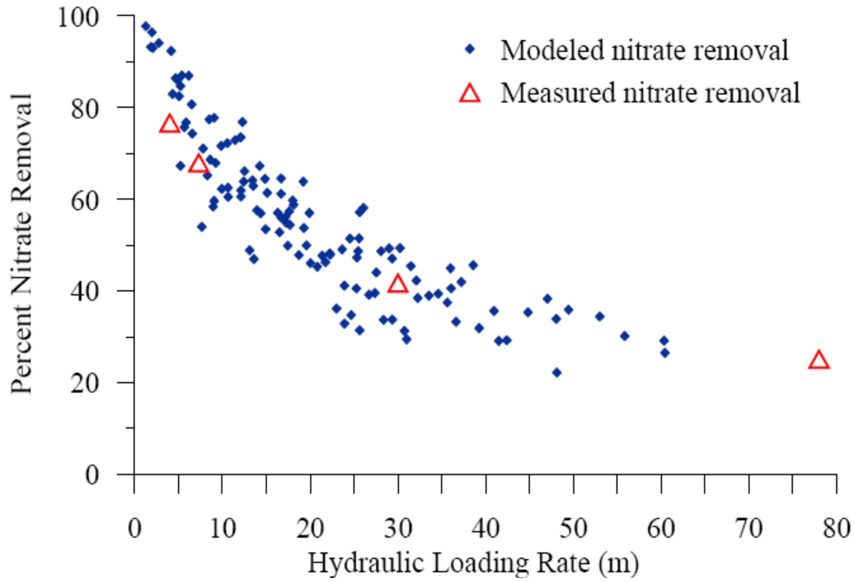


Figure 5.4. Modeled nitrate removal efficiencies for CREP wetlands based on 1996 to 2005 input conditions and measured nitrate removal efficiencies for CREP wetlands in 2004 & 2006. [Reprinted from Crumpton & Helmers 2006).

Implications for the Boone River

Table 5.4 contains estimates for the Boone River of the total area within each 12-digit subwatershed, what percent of that is currently in cropland, and how much would be needed to achieve a 1%, 5%, and 9% treatment area ratio of buffer or wetlands to source area (cropland upstream). We used two different methods to estimate the total amount of riparian or wetland buffers needed along stream corridors to achieve water quality and biodiversity goals:

1) a 100 m buffer around all stream channels
2) wetland treatment area: source area of 1, 5, and 9%, assuming wetlands and riparian zones would be located in natural depressions with surface tile outlets, as well as in sinks along riparian corridors

We then estimated the percent nitrate removal for each treatment area: source area ratio (1, 5, and 9%) based on the Hydraulic Loading Rate and expected mass load for each scenario using two equations provided by Crumpton and Helmers (2006):

a. Equation 1: % nitrate removed = -0.45*log (X) + 1.23, where X = NO_3-N loading in g-N/m2-year
b. Equation 2: Mass nitrate removed = 10.3 * HLR(^0.67)*FWA (HLR= Hydraulic Loading Rate in m/year; FWA = flow weighted average nitrate concentration in mg/L)

Using these methods, we developed a range of estimates of the minimum riparian land in the Boone currently in cropland that would be implied would be converted to natural cover or wetlands to achieve water quality criteria. Based on those acreage ranges, we develop a rough estimate of the implied cost of land purchase or rental. Based on the 2002 land use cover, total area of cropland in the Boone is 2161 km^{2}, of which 245 km^2 (60,476 acres) of the watershed are within 100 m of a 1st-6th order perennial stream (i.e., slightly more than 10% of the watershed area). Much of this land is already in grass or perennial vegetation; only 54% of the land in this riparian area is cropped.

Under the most optimistic scenario, assuming that efficient targeting of field-scale BMPs could reduce existing loads by 20%, average loads at the mouth of the Boone River would decline to 4.8 million kg. To achieve a total reduction of 45% (to 3 million kg/ha) an additional 1.5 million kg load reduction would be needed. At the most optimistic estimated removal rate for high performing wetlands of 200 kg/ha N, the minimum additional wetland acreage needed in the Boone would be 7500 ha, or 3125 acres (~0.6%).

Assuming a more realistic scenario involving a wetland/riparian treatment area: cropland ratio of 1:20 (5%) in each subwatershed, 40-200% of buffer area in each sub watershed would be needed, or ~8700 hectares (21,500 acres) (Table 5.4). Based on current nitrate loading estimated by the SWAT baseline scenario and using the HLR equations to estimate percent nitrate removal, a 5% treatment area: source area ratio of riparian buffer or wetlands would result in an estimated 38-55% nitrate reduction at the watershed outlet.

Assuming that all the land needed for such a project would have to be purchased at 2006 farmland values for an acre of Iowa corn land of $3200, the cost would be $65-75 million. Annual rental rates for corn/soybean land in the Boone River watershed counties range from $140-$160 per acre. Price tags like this often lead analysts to conclude that land retirement is not cost-effective or feasible, even if it results in the greatest reductions in runoff and nonpoint source pollution (Wu et al 2004, Petrolia et al. 2005). However, the fact that only 54% of riparian area in the Boone River watershed is currently cropped suggests that recreation, aesthetic, and suitability benefits are already playing a role in land use decision-making. Furthermore, the cost of targeted restoration on an additional ~5% of the landscape represents a small fraction of gross economic activity in the watershed. For example, if we assume at least 400,000 acres planted to corn in the Boone watershed per year (about 75% of current cropland), with an average yield of 175 bushels/acre, gross receipts for corn in the Boone River watershed easily exceed $250 million per year at a price of $4/bushel.

SWAT model results for wetland treatment

Future targeting analysis involves use of the SWAT model to evaluate the feasibility and effectiveness of wetland treatment of tile drainage at subwatershed outlets, and to refine the estimate of wetland acreage needed. Limitations of the SWAT model are such that only one wetland can be simulated at each subwatershed outlet; thus a proxy approach must be used for 30 subwatershed configuration. The initial baseline simulation estimates 50% long-term nitrate reduction predicted using a 30 subwatershed wetland approach (Gassman et al. 2007). The alternative approach involves dividing the watershed into

405 subwatersheds to identify suitable subwatersheds that meet wetland siting criteria, without assuming a single large aggregated wetland at the outlet of larger subwatersheds. Although not spatially explicit, the methodology involves identifying the percentage of subwatershed that drains to each theoretical wetland. Currently, refinements to the SWAT wetland module are needed (Gassman et al. 2007). The initial analysis suggested there is a need to improve the baseline nitrate simulation, nitrogen transformation routines, test the model with sediment and P data, and reconstruct an alternative wetland delineation (requiring some measured data). However, this work is currently not funded.

Summary – Conservation Practices

Targeting conservation practices to fields and subwatersheds where the greatest benefit can be achieved for the least cost is the goal of much nonpoint source water quality improvement and watershed restoration research. However, environmental benefits and trade-offs are notoriously difficult to measure appropriately. Feather et al. (1999) demonstrated how estimates of nonmarket values provide a more robust set of information for the targeting of agricultural conservation programs, facilitating cost comparisons against a full range of benefits, and leading to better evaluation of programs. Table 5.5 attempts to qualitatively summarize benefits, costs and feasibility of proposed actions and strategies discussed in the previous sections, across a range of potential benefits.

Table 5.4. Estimates of minimum effectiveness areas for riparian buffers and treatment areas by subwatershed area based on source area (cropland acres) and estimated nutrient loads.

				(a)	(b) Wetland/ riparian ha needed for				(b) % NO_3 removed by HLR equation 1 at:			(c) % NO_3 removed based on HLR equation 2 at:		
HUC_12	SWATNO	Cumulative upstream area (ha)	% Cropped	Area of 100m buffer	1%	5%	9%	NO3 load (kg/year)	1%	5%	9%	1%	5%	9%
071000050702	28	4481	84%	481	38	188	339	130,571	9%	40%	52%	30%	51%	62%
071000050201	19	4689	91%	156	43	213	384	73,711	22%	54%	65%	34%	58%	71%
071000050501	16	4940	88%	368	43	217	391	180,323	5%	37%	48%	30%	51%	62%
071000050304	10	5152	89%	1078	46	229	413	292,991	-	28%	40%	24%	41%	49%
071000050703	26	5912	87%	742	51	257	463	59,303	30%	62%	73%	32%	55%	67%
071000050701	25	6151	85%	399	52	261	471	38,063	39%	71%	82%	33%	56%	68%
071000050401	1	6824	85%	769	58	290	522	141,442	16%	47%	59%	31%	52%	63%
071000050601	21	7198	84%	1387	60	302	544	133,443	18%	49%	60%	27%	46%	56%
071000050301	15	7316	86%	1215	63	315	566	174,574	13%	45%	56%	33%	56%	68%
071000050402	2	7501	88%	715	66	330	594	206,843	11%	42%	54%	32%	55%	67%
071000050102	7	9157	89%	500	82	408	734	96,068	30%	61%	73%	42%	71%	86%
071000050602	22	9637	83%	783	80	400	720	50,763	42%	73%	85%	35%	60%	73%
071000050103	12	10336	87%	986	90	450	809	415,646	3%	35%	46%	27%	46%	55%
071000050303	11	10793	90%	606	97	486	874	118,267	29%	61%	72%	41%	70%	85%
071000050101	8	10924	87%	1249	95	475	855	159,173	23%	54%	66%	39%	67%	81%
071000050202	20	11703	89%	1133	104	521	937	169,588	23%	55%	66%	38%	64%	78%
071000050603	24	12042	85%	1139	102	512	921	274,367	14%	45%	57%	24%	41%	50%
071000050503	4	12255	86%	406	105	527	949	82,299	38%	69%	81%	53%	89%	109%
071000050302	9	14725	87%	671	128	641	1153	429,657	9%	41%	52%	32%	54%	65%
071000050203	18	26299	88%	813	231	1157	2083	379,205	23%	55%	66%	37%	63%	77%
071000050403	6	26889	83%	1417	223	1116	2009	559,743	15%	46%	58%	32%	54%	66%
071000050104	13	36031	88%	846	317	1585	2854	774,129	16%	47%	58%	35%	60%	73%
071000050305	14	39320	87%	773	342	1710	3079	969,848	13%	44%	56%	34%	58%	70%
071000050306	17	81753	83%	731	679	3393	6107	1,944,114	12%	44%	55%	34%	58%	70%
071000050502	3	115446	77%	750	889	4445	8000	2,976,462	9%	41%	52%	33%	56%	68%
071000050504	5	139561	78%	1706	1089	5443	9797	3,559,667	10%	41%	53%	34%	57%	69%
071000050505	23	179611	69%	648	1239	6197	11154	4,579,219	7%	39%	50%	32%	55%	67%
071000050704	29	225949	71%	800	1604	8021	14438	6,118,286	7%	38%	50%	32%	54%	65%
071000050705	30	235198	74%	1208	1740	8702	15664	6,593,333	7%	38%	50%	32%	54%	66%
				24474	**1740**	**8702**	**15664**	**6,593,333**	**7%**	**38%**	**50%**	**32%**	**54%**	**66%**

Table 5.5. Anticipated benefits associated with different agricultural management options (multiple sources)**.** Some management practices offer minimal (or even negative) improvements at very high cost to farmers. On the other hand, there are many highly effective conservation practices that are either relatively inexpensive or highly cost-effective. Some very cost-effective examples are alterations to fertilizer application methods to decrease surface runoff and/or leaching losses of N, and alterations to tillage regimes to decrease sediment/runoff losses.

Management Option	Reduce N loads to downstream surface waters	Reduce P loads in surface waters	Reduce upland erosion	Reduce bank erosion	GW quality	Carbon seques-tration	Local wildlife habitat	Bio-diversity	Aesthetics/ recreation	Multiple benefits	Cost	Feasi-bility
On-farm												
Nutrient Rate and Timing Management												
Reduce fertilizer N and/or P application	+	+	na	0	+	0	+	+	+	6	M	M
Spring fertilizer N and/or P application	+15%	+30%	na	0	+	0	0	0	0	3	M	M
Soil test based Split-in-season vs. fall	+30%	na		0	+	0	+	+	+	4	M	M
Soil test based Split-in-season vs. spring	+15%	na		0	+	0	+	+	+	4	M	M
Soil test based vs. excessive	+60%	+40%		0	+	0	+	+	+	4	L	H
Yield Goal or Crop Removal based (vs excessive)	+35%	na		0	+	0	+	+	+	4	L	H
Deep Tillage vs broadcast (P)	na	-15%								1	L	M
Shallow tillage vs broadcast (P)	na	-10%								1	L	M
Knife/injection vs broadcast (P)	na	+35%								1	L	M
Nitrification/urease inhibitors	+10%	na								1	M	M
Improve manure management	+	+	na	na	+	0	0	0	0	3	L to H	required
Conservation Tillage												
Moderate vs. intensive	+3%	+50%	+			+				4	L	H
No-till vs intensive	+10%	+70%	++			++				4	L	H
No-till vs moderate	+5%	+45%	+			+				4	L	H

Management Option	Reduce N loads to downstream surface waters	Reduce P load in surface waters	Reduce upland erosion	Reduce bank erosion	GW quality	Carbon seques -tration	Local wildlife habitat	Bio-diversit y	Aesthet ics/ recreati on	Multipl e benefit s	Cost	Feasi-bility
Cropping systems												
Cover crops	+50%	+50%	+	+	0	+	0	0	0	3	M	M
Diverse cropping systems	+50%	+50%									H	L to M
Perennial cropping systems	+	+	+	+	+	+	+	+	+	9	H	L to M
Drainage Water Management	+	-	+ / -	+	0	0	0	0	0	2	M	
Decrease drainage intensity	+	-	+	+	0	0	0	+	+	3	H	L
Shallow/wide vs. Standard	+20%	- 5%	-	+	0	0	0	0	0	1	M	H
Controlled vs uncontrolled	+25%	- 5-10%	-	+	0	0	0	0	0	1	M	M
Subirrigation with treatment wetland	+30%	+10%	+	+	0	0	0	0	0	1	M	L to M
Pasture/grassland management												
Rotational grazing	+20%	+25%	+	-							L	H
Seasonal grazing	+20%	+50%		-							L	H
Livestock exclusion	+30%	+75%	+/-	-							L	M
Increase acres of farmland retired	+	+	+	+	+	+	+	+	+	9	H	L
Edge-of-field / Off-farm												
2-stage ditch & stream channel restoration	20-60%	+20-75%	+	+	+	+	+	+	+	7	M to H	H
Increase freshwater wetlands	30%	-50% to +80%	+	+	0/?	+	+	+	+	6	H	M
Forested riparian bufffers	40%	45%	+	+	+	+	+	+	+	7	L to M	H
Herbaceous riparian buffers	+	+	+	+	+	+	+	+	+	9	L to M	H

Recommended Strategic Approaches

Table 6.1 Summary of strategic approach

Retain and restore important landscape features, pattern, & processes
- riparian corridors and buffers
- wetland sinks, outlet treatment, spatial & temporal habitat connectivity

Minimize "leakage" / maximize retention
- Controlled drainage
- Nutrient management (timing & amount)
- Perennial & cover crops
- Treatment of tile runoff
- Wetlands/buffers at tile outlets
- Bioreactors/bioremediation

Redesign drainage with fluvial and nutrient cycling processes in mind
- 2 stage ditch / compound channels

Build capacity for sustainable management of the watershed
  - Assessment and Monitoring
  - Planning and Targeted Implementation
  - Communication and Outreach Plan

Table 6.2. Current Status of planning & assessment

	Status
Adaptive assessment & management of watershed resources	
▪ Ecological Assessment	*Completed and available online*
▪ Rapid Watershed Assessment	*Completed; soon to be available online*
▪ SWAT model	*Phase I modeling completed; results include baseline scenario and multiple cropping system / nutrient management scenarios*
▪ Ongoing spatial / targeting analysis	*ISA monitoring (3 years); see separate monitoring plan*

Table 6.3 Action steps and measures for assessing their effectiveness

Action Steps	**Measures / Notes**
Build Capacity for Sustainable Watershed Management	
Work with watershed organizations to implement habitat and water quality restoration	*Formation of the BRWA* *# working projects in collaboration with RC&D, etc.*
Increase awareness of watershed condition	*Survey (?)*
▪ Demonstration, education and outreach programs (also see #3)	*# Field days / # attendees*
▪ Outreach workshops to communicate assessment/monitoring results	*# of workshops / # attendees*
Develop monitoring capacity ▪ Build volunteer monitoring working with ISA, IOWATER, IDNR and other programs: ▪ Conduct assessment & monitoring training workshops	*# volunteers trained & participating* *20-25 sites – Fish, habitat assessment & macroinvertebrate*

▪ Conduct public water quality “snapshots” presentations	*monitoring* *2 (visual survey) and 5 year (intensive monitoring) rotations*
Future research needs	
Geomorphology – Evaluate instream sediment and estimate stream bank erosion o integrate CONCEPTS and SWAT o conduct RASCAL habitat assessment for Lyons Creek subwatershed	*Conduct sediment workshop / meeting with key researchers to design next generation sediment study*
Groundwater Studies - understand the pathways of groundwater-surface water interactions, impacts of restoration on water tables and adjacent lands	*Incorporate water table monitoring as part of paired watershed studies*
Multi-species monitoring • Mussels • River otters • Fish	*Possible coordination with multiple partners for summer 2008-2009*

Implementation Strategy

The Implementation Strategy involves three plan components:

1. Monitoring and Assessment
    a. Status assessment/monitoring
    b. Effectiveness monitoring
2. Planning and Targeted Implementation
3. Communication and Outreach

1. Monitoring and Assessment

As identified in the Boone River Ecological Assessment, additional research and monitoring is needed in the short-term to provide a more complete understanding of the status of biodiversity in the Boone River. Additional monitoring and assessment is needed, including both further development of monitoring capacity – through training additional staff and further development of the IOWATER monitoring network of certified samplers – as well as specific assessment needs.

Status versus effectiveness monitoring. Status monitoring is monitoring designed to track the overall health and biodiversity of the watershed over time. Such sampling is critical to assessing the acceptable ranges of variation and current status of each of these key attributes, individually and in relation to one another. Effectiveness monitoring, by contrast, is monitoring designed to detect evidence that actions taken as part of the conservation action plan for the Boone are having the desired or hypothesized effects. Are our strategies working? Both status and effectiveness monitoring are needed in the long-term. Table 6.4 lists a summary of recommended monitoring activities for the Boone River watershed, broken down by status versus effectiveness monitoring goals, action steps, and indicators.

The Conservancy and Boone River partners should work to establish a long-term monitoring network for periodic, watershed-wide sampling of key ecological attributes and indicators. The goal of this sampling effort would be to augment baseline understanding of conditions throughout the watershed as a basis for assessing effectiveness of activities and projects as well as to track the long-term status of the key ecological attributes. The Conservancy and partners should work with IOWATER network trainers, volunteers, and ISU to train a corps of watershed residents to conduct macroinvertebrate and physical habitat assessments as well as "snapshot" water quality sampling. This could be initiated in the summer of 2008 through a 1-2 day workshop on water quality, benthic macroinvertebrate, and physical habitat assessment, followed by a 1-day data sampling event (July or August) designed to collect information simultaneously across the watershed ("snapshot"). Training would involve introduction to water quality protocols, macroinvertebrate identification, and volunteer collection procedures with expert trainers[5]. The RC& D and TNC would coordinate organization of the volunteers.

Long-term effectiveness monitoring requires specific plans to assess implications and results of implemented actions. The Conservancy should partner with BRWA, NRCS, RC&D, etc. to encourage and track ongoing implementation of on-farm best management practices (e.g., buffer strips). Effectiveness monitoring should also be designed to evaluate any experimental and "adaptive management" strategies implemented in the watershed, such as riparian buffers, channel/drainage system alterations, or restoration projects to monitor environmental effects. Continuous monitoring of such projects is needed to ensure they are making progress towards stated goals.

Mussels as the "Canary in the Coal Mine" watershed status indicator. Because mussels are long-lived, slow to reproduce, and sensitive to environmental stress, they may serve as an excellent long-term indicator of Boone River watershed health. Understanding the causes of mussel declines in the Boone may require considerable additional research. Mussels are relatively difficult to identify. Because of the threatened status of many species in the Boone, care should be taken in conducting other sampling and assessment work to avoid disturbing mussel beds. The impacts of recreation and scientific research may themselves be significant, and should be avoided (Watters 2000). Periodic sampling should be conducted based on methodology developed for statewide mussels assessment by Kelly Poole. A repeat study by Ellet Hoke (who conducted the 1984 baseline work) is recommended for summer 2008 or 2009. This could involve additional sampling in the Upper and Lower Zones. Appropriate involvement of watershed residents in sampling events may help to build awareness and concern for the condition of Boone River mussels. Additional ISU faculty, such as Kevin Roe, may be recruited to conduct a mussel ID workshop.

It would also be helpful to evaluate the Boone mussel population condition in the context of an ecoregional analysis of mussel species diversity and distribution, developed from the statewide mussel survey. At the watershed scale, mapping would indicate distribution of species within the watershed, distance between colonies, and identify important mussel

[5] (IOWATER/Mary Pat Heiman and Lynette Seigley; Wade Roe and Greg Courtney, ISU

beds on which to focus protection. Because the location of rare species is potentially sensitive information, this information should be somewhat protected.

Hydraulic models and studies may be helpful to assess hypotheses regarding observed mussel declines, in particular whether mussel response is influenced by sediment transport and scour during increasingly frequent high flow events, or whether exploitation by raccoons, river otters, and other predators may be playing a role in declines. Alternative hypotheses include chronic water quality impairments, periodic acute mortality events and timing of events, or all the above. Mussel and aquatic mammal sampling could potentially be funded through state wildlife action grants.

Additional short-term assessment and monitoring needs include the following:

- Sediment/Erosion/Geomorphology/Hydrology Assessment
  - Continuous flow monitoring equipment is needed to begin developing stage-discharge-sediment transport curves and channel evolution stage determinations in targeted subwatersheds
  - Streambank erosion assessments should be conducted in conjunction with stream water quality biological monitoring. This may include visual assessment of channel and bank conditions; as well as assessment using habitat assessment protocols developed for IDNR watershed programs[6]
  - Streambank erosion and sediment loads should be evaluated during both low and high flow conditions to address the variability and uncertainty associated with the estimates presented here. More TP data should be collected from eroding streambanks both within the Boone and in other watersheds in the Des Moines Lobe and other landforms to evaluate how much P loading enters streams from upland versus fluvial processes (Wilson 2003).
  - Stream visual assessments of riparian conditions procedures could be adapted for use in the Boone River watershed. Many watershed restoration initiatives are in the process of developing rapid, standardized protocols for physical habitat assessment, modeled after EPA, USDA, and NRCS protocols. Modeling work for sediment using CONCEPTS with SWAT outputs could help elucidate relationship of sediment to channel erosion and instability.
- Water Quality: A load analysis is needed to estimate the role of point source contributions to water quality impairments in the Boone River watershed, including potentially failing municipal and/or septic performance and unsewered communities.
- Iterative assessment of the implications of ongoing economic and other drivers of change that impact conservation targets in the future, for example:
  - Impacts of expanded corn acreage under ethanol scenarios for Upper and Lower watershed key attributes
  - Impacts of climate change, ethanol, cellulosic ethanol and other emerging technologies for groundwater sustainability, basin water yield and stream hydrology

[6] RASCAL

Table 6.4. Recommended monitoring activities

Status Monitoring	**Action Steps & Indicators**
Watershed-scale, coordinated, spatially representative sampling effort focused on key ecological attributes.	• Water quality at subwatershed outlets • IHA indicators • Channel geomorphology and riparian/wetland/aquatic habitat
Systematic field surveys needed to assess status and health of the following: • beaver and otter populations in the watershed - data from DNR on otter/fur trapping tags • distribution of beaver and effects on ecosystem structure • baseline reptiles & amphibian surveys	Coordinate with Karen Kinkead --Iowa Multispecies inventory & monitoring program: • Baseline population estimates • Species richness • Relative abundance • Distribution
Mussel assessment	Contract with Ellet Hoke, Karen Kinkead, Kevin Roe, and/or Kelly Poole
Additional fish & macroinvertebrate surveys for IBI tracking.	Work with Tom Wilton (IDNR) to establish repeat sampling of REMAP sites for fish IBI and macroinvertebrates (3-5 year sampling rotation).
Topeka shiner assessment	
Climate Change detection	• Establish continuous thermal and flow recording at 1-5 permanent flow gage locations • Summer and low flow DO measurements • Groundwater monitoring
Effectiveness monitoring	**Action Steps & Indicators**
Develop Paired Watershed (control & treatment)	Huc12 outlets and paired subbasins with projects
BMP's - Innovations in Nutrient and Soil Management • ISA on-farm field trials • Encourage adoption of Nutrient Best Management Practices • Grassed waterways • Cover crops	# Farmers adopting / % of acres in practices #/% Farmers adopting / practices # Farmers adopting / % of acres in practices Acres in grassed waterways Acres in cover crops ; cost of incentive
Manure Management	# projects / water quality performance
Tile drainage management • alternatives to open tile intakes • controlled/conservation drainage innovations	# installed water quality performance at outlet # projects and acreage % water quality performance at outlet
Ditch & Stream Habitat Restoration & Rehabilitation ▪ Compound Channels / 2 stage ditches	 Miles of restored stream Water quality performance upstream-downstream Macroinvertebrate IBI Fish IBI
▪ Streambank Stabilization / restoration	Miles of eroding streambanks Sediment yield at subwatershed outlet Sediment-discharge relationship
Riparian Buffer Zones	# stream miles protected by varying buffer widths Work with Tom Isenhart / Keith Schilling / Demonstration Farm
Wetland Restoration ▪ Farm-scale treatment wetlands modeled on Mackinaw project ▪ Tile outlet treatment wetlands modeled on Iowa CREP and/or Mackinaw project	Acres of restored wetlands # projects % eligible farmers participating % basin drainage area "treated" water quality at outlet

2. Planning & Targeted Implementation

- Water quality improvement plans for 303d listed impaired waters are required under consent decree from the U.S. Environmental Protection Agency. Watershed improvement planning activities in the Boone River watershed scheduled for 2008 include Buttermilk Creek (low DO, organic matter). Lyons Creek (fish kill of unknown origin) is scheduled for 2009. The Des Moines River nitrate plan is also scheduled for 2008, and because the Boone is a tributary to the Des Moines River, nitrate reductions targeted in the Boone may be potentially considered remedial actions under the Des Moines TMDL plan.
- Identify and target phosphorus critical source areas
  - Develop an implement the NRCS P index at field scale in top 1-3 watersheds with highest per acre loading, using field data collected by Charlie Kiepe for SWAT model if possible.
  - Assess soil test P levels in eroding streambanks and bank erosion contribution to P loads
- Erosion and sediment reduction
  - Identify heavily eroding banks in the main channel and pursue cooperators and funding for active bank stabilization/stream restoration demonstration
  - Identify potential 2 stage ditch/channel restoration in areas where maintenance is planned or landowners have stream bank erosion loss concerns
- Nitrate reduction and hydrologic retention
  - Targeting largest contributing subwatersheds based on ISA monitoring data and SWAT model outputs
  - Inventory planned drainage projects: contact contractors, county and district engineers, private landowners planning improvements/ upgrades, etc.
  - Explore feasibility of expanding controlled drainage/drainage water management on eligible lands (< 0.5% slope)
  - Explore feasibility of implementing tile outlet treatment wetlands in low slope areas or in eligible sinks in the landscape (based on hydraulic loading and nutrient loads/willingness to participate)
  - Assess willingness to participate in wetland treatment schemes at various payment and delivery levels
  - Explore opportunities to expand and augment existing riparian buffers; willingness to participate
- On the Ground Restoration/demonstration projects: The Conservancy should work with District Conservationists & RC&D to identify opportunities in existing farm and water quality programs.
  - Paired watershed – The Conservancy should develop proposals to implement a demonstration projects in the Boone River. As part of the implementation, the Conservancy should consider contracting with local farmers and/or agricultural outreach professionals to conduct outreach through Boone River watershed association specifically to increase adoption and publicize incentives for adoption of the following practices:
    - controlled drainage / outlet treatment / bioreactors
    - 2 stage ditch design / design of compound channels for ditches scheduled for maintenance

        - Stream restoration for unstable banks with upstream sediment traps where needed (a la Pecatonica)
        - Range of wetland restoration designs a la Mackinaw River with modified adaptive siting criteria
        - Riparian buffers
- Funding and fund-raising:
    - The Conservancy should work with District Conservationists & RC&D to identify funding sources available to help apply for and implement grants.
        - Watershed Improvement Review Board (WIRB)
        - IDNR Watershed Planning & Assistance Grants
        - Clean Water Action EPA 319 / TMDL planning grants
        - NRCS Farm Bill Conservation Program Funds
    - The Conservancy should consider developing a delivery program of economic incentives and outreach designed to achieve implementation at a scale above the minimum threshold required to detect measurable outcomes in targeted subwatersheds. For example, a potential role for the Conservancy is to work with NRCS to make up the gap in USDA CREP payments between the federal price and what the producer/landowner needs to participate in programs acknowledged to be the most environmentally effective. Outreach staff, whether employed by the Conservancy or partners, should facilitate communication throughout the process with potential landowner cooperators.

3. Communications/Education Plan

The Conservancy and partners should develop outreach materials summarizing the assessment, threats, and potential solutions. Field days and media events, including press releases to local newspapers and radio stations, should be a component of activities. In targeted watersheds, surveys soliciting information from producers and landowners about the feasibility and willingness to accept or implement different practices may be helpful.

An interactive, comprehensive website should be established to serve as a one-stop repository/directory index for monitoring data, analysis, reports, and plans, as well as updates on ongoing and scheduled activities. The web site could be maintained by The Nature Conservancy, Prairie Rivers RC&D, or the Boone River Watershed Association to provide a one-stop information to managers, watershed residents, and volunteers, organizing links to other sources of information on the Boone, such as the DNR, NRCS, and USGS, as well as the completed Rapid Watershed Assessment conducted for NRCS.

A media and outreach plan should be developed designed to disseminate the results of activities, both online, and through conventional and traditional media outlets in the area. This could involve public meetings/workshops in at least 4 communities distributed throughout the Boone River watershed to present the assessment & CAP.

Partners involved in Boone River watershed planning should continue to solicit and incorporate lessons learned from other watershed projects. Outreach efforts will benefit from inviting project leaders involved in similar watershed efforts (Iowa and neighboring states) to talk about their successes and experiences and lessons learned. For example, representatives from TNC projects on the Mackinaw and Pecatonica Rivers could be invited to discuss successes, lessons learned, and ongoing challenges.

The Conservancy and partners should, where possible, attempt to hire and/or work with local residents, opinion leaders, or existing outreach professionals to conduct outreach and publicize events and programs.

The Conservancy should help enable key local watershed professionals and producers to attend select field days at conservation practice demonstration sites in neighboring watersheds and states.

For subwatersheds targeted for projects and practice implementation, a survey tool could be designed to gauge the level of public concern/awareness over Boone River watershed condition, and to identify the acceptability of proposed actions, incentives, and voluntary management practices. Many such survey and focus group tools have been developed to move projects forward (Casey et al. 1995, Tisl and Palas 1998). For example, a survey of watershed residents and landowners was carried out early in the Bear Creek restoration project to assess willingness to adopt buffers and other practices (Isenhart et al. 1997).

Literature Cited

Aadland, L.P. 1993. Stream habitat types: their fish assemblages and relationship to flow. North American Journal of Fisheries Management 13:790- 806.

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

Aldous, A., P. Gonzales, and K. Popper. 2007. A Method for Incorporating Climate Change into Conservation Action Plans. The Nature Conservancy in Oregon.

Alexander, P.J. Johnes, E.W. Boyer and R.A. Smith. 2002. A comparison of models for estimating the riverine export of nitrogen from large watersheds. Biogeochemistry 57/58: 295–339.

Alexander, R.B., R.A. Smith and G.E. Schwarz. 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. Nature 403, 758-761.

Allan, J. David. 1995 Stream Ecology. Island Press, Washington, D.C.

Angermeier, P. L., and I. J. Schlosser. 1995. Conserving aquatic biodiversity. Am. Fish. Soc. Symp. 17: 402-414.

Arbuckle, K.E. and J. A. Downing. 2000. Statewide assessment of freshwater mussels Bivalvia: Unionidae in Iowa streams. Iowa Department of Natural Resources, Des Moines.

Arbuckle, K.E.and J. A. Downing. 2002. Freshwater mussel abundance and species richness: GIS relationships with watershed land use and geology. Can. J. Fish. Aquat. Sci. 59: 310-316.

ASCE. 1998. River width adjustment. I: Processes and mechanisms. Journal of Hydraulic Engineering. 124(9):881-902.

Augspurger, Tom, Anne E. Keller, Marsha C. Black, W. Gregory Cope, and F. James Dwyer. 2003 Water quality guidance for protection of freshwater mussels Unionidae from ammonia exposure. Environmental Toxicology and Chemistry 2211: 2569-2575.

Bailey, R.G., W.H. McNab, P.E. Avers, and T. King, T. 1994. Ecoregions and Subregions of the United States (Lower 48). Vector digital data, Washington, VA. USDA Forest Service. URL: http://www.fs.fed.us/institute/ecoregions/eco_download.html

Bailey, R.M. 1956. A Revised List of the Fishes of Iowa: With Keys for Identification. Iowa State Conservation Commission, Des Moines, Iowa.

Bain, M.B., J.T. Finn, and H.E. Booke. 1988. Streamflow regulation and fish community structure. Ecology 69: 382-392.

Baker, D.B., R.P. Richards, T.T. Loftus, J.W. Kramer. 2004. A New Flashiness Index: Characteristics and Applications to Midwestern Rivers and Streams. Journal of the American Water Resources Association 40(2): 503-522.

Baker, J. 2007. Drainage Water Quality: Continuous Corn versus Corn-Soybean Rotation. 2007 IA-MN Drainage Research Forum, 11-27-08, Ames, Iowa.

Bakker, K.K, D.E. Naugle, K.F. Higgins. 2002. Incorporating landscape attributes into models for migratory grassland bird conservation. Conservation Biology 16 6, 1638–1646.

Barbour, M.T. and J.B. Stribling. 1991. Use of habitat assessment in evaluating the biological integrity of stream communities. In, Biological criteria: research and regulation, proceedings of a symposium. EPA-440/5-91-005. U.S. Environmental Protection Agency, Office of Water. Washington, D.C. 171p.

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

Beach, T. 1994. The fate of eroded soil: sediment sinks and sediment budgets of agrarian landscapes in Southern Minnesota, 1851-1988. Annals of the Association of American Geographers 84:1, 5-28

Becher, Kent D., Stephen J. Kalkhoff, Douglas J. Schnoebelen, Kimberlee K. Barnes, and Von E. Miller. 2001. Water-Quality Assessment of the Eastern Iowa Basins—Nitrogen, Phosphorus, Suspended Sediment, and Organic Carbon in Surface Water, 1996–98. U.S.G.S. Water-Resources Investigations Report 01–4175." 56 pp.

Bennett, E.M., Carpenter, S.R. and Caraco, N.F. 2001. Human impact on erodable phosphorus and eutrophication: a global perspective. BioScience 51: 227–234.

Berg, J.A., T.A. Petersen, Y. Anderson, and R. Baker. 2004. Hydrogeology of the Rock River watershed, Minnesota and associated off-channel habitats of the Topeka shiner. Final report submitted by the Minnesota Department of Natural Resources. 13+ pp.

Bernot, M.J., J.L. Tank, T.V. Royer, and M.B. David. 2005. Nutrient processing in streams draining agricultural watersheds of the Midwestern U.S. Freshwater Biology in press.

Bogan, AE. 1993. Freshwater bivalve extinctions (Mollusca: Unionoida): A search for causes. American Zoologist [AM. ZOOL.] 33, no. 6, pp. 599-609.

Booth, D.B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association, v. 38:835-845 2002

Borah, D.K. G. Yagow, A. Saleh, P. L. Barnes, W. Rosenthal, E. C. Krug, L. M. Hauck. 2006. Sediment and nutrient modeling for TMDL development and implementation. Transactions of the ASABE. Vol. 494: 967-986. @2006

Burkart, M.R., Simpkins, W.W., Morrow, A.J., & Gannon, J.M. 2004. Occurrence of total dissolved phosphorus in surficial aquifers and aquitards in Iowa. Journal of the American WaterResources Association, 40(3), 827-834.

Camargo, J.A. and Álvaro Alonso. 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. Environment International Volume 32, Issue 6, August 2006, Pages 831-849

Camargo, Julio A., Alvaro Alonso, Annabella Salamanca. 2005. Nitrate toxicity to aquatic animals: a review with new data for freshwater invertebrates. Chemosphere 58: 1255-1267.

Casey, M.A., Gary Wyatt, Amy Rager, Cindy Arnevik, Dave Pfarr, Jim Anderson, Les Everett, and Lowell Busman. 1995. Addressing Nonpoint Source Agricultural Pollution in the Minnesota River Basin: Findings from focus groups conducted with Farmers, Agency Staff, Crop Consultants, and Researchers. December 1995. Study conducted for the Minnesota Department of Agriculture as part of the Minnesota River Agricultural Research Base Studies Project, University of Minnesota. St. Paul, MN. URL last accessed 3/20/2008: http://www.soils.umn.edu/research/mn-river/doc/fgrptweb.html

Changnon, S.A. and K.E. Kunkel 1995. Climate-related fluctuations in Midwestern flooding. Journal Water Resources Planning and Management, 121: 326-334.

Changnon, S.A., and D. Winstanley 2000. Temporal Changes in Corn-weather Seasons in Illinois. Climatic Change, 47: 353-372.

Changnon, S.A., and M. Demissie. 1996. Detection of changes in streamflow and floods resulting from climate fluctuations and land use-drainage changes. Climatic Change (Historical Archive) 32(4):411 – 421.

Clark, W.R. 2006. River Otter Ecology and Management in Iowa. Ecology, Evolution and Organismal Biology, Iowa State University. Ames, Iowa. URL: http://www.eeob.iastate.edu/faculty/ClarkW/homepage.html

Covich, A.P., S. C. Fritz, P. J. Lamb, R. D. Marzolf , W. J. Matthews, K. A. Poiani, E. E. Prepas, M. B. Richman, T. C. Winter. Potential effects of climate change on aquatic ecosystems of the Great Plains of North America. 1997. Hydrological Processes Volume 11, Issue 8 , Pages 993-1021.

Crumpton, W. G., G. A. Stenback, B. A. Miller, and M. J. Helmers. 2006. Potential Benefits of Wetland Filters for Tile Drainage Systems:Impact on Nitrate Loads to Mississippi River Subbasins. Final project report to U.S. Department of Agriculture, Project number: IOW06682. Iowa State University, Ames, IA.

Crumpton, W.G. and Matthew Helmers. Integrated Drainage-Wetland Systems for Reducing Nitrate Loads from Tile Drained Landscapes. American Society of Agricultural and Biological Engineers, St. Joseph, Michigan www.asabe.org. Citation: Paper number 701P0904.

Daniel, T. C., A. N. Sharpley, and J. L. Lemunyon. 1998. Agricultural phosphorus and eutrophication: A review. J. Environ. Qual. 27:251-257.

Darby, S.E. and A. Simon Eds. 1999. Incised River Channels: Processes, Forms, Engineering and Management. S.E.. Wiley, Chichester, 1999. 442 pp. 134 line diagrams, 59 photographs, 24 tables, and 8 color plates.

DeWalt, R.E, C. Favret, and D.W. Webb. 2005. Just How Imperiled Are Aquatic Insects? A Case Study of Stoneflies Plecoptera in Illinois. Ann. Entomol. Soc. Am. 986: 941-950

Diamond, J.M. VB Serveiss. 2001. Identifying sources of stress to native aquatic fauna using a watershed ecological risk assessment. Environ. Sci. Technol., 35 24: 4711 - 4718.

Dils, R.M. and A.L. Heathwaite. 1999. The controversial role of tile drainage in phosphorus export from agricultural land. Water Science and Technology Volume 39, Issue 12, Pages 55-61.

Dinnes, D.L. 2004. Assessments of Practices to Reduce Nitrogen and Phosphorus Nonpoint Source Pollution of Iowa's Surface Waters. USDA-ARS National Soil Tilth Laboratory, for the Iowa Department of Natural Resources, In Cooperation with the USDA-ARS National Soil Tilth Laboratory.

Dinnes, D.L., D.L. Karlen, D.B. Jaynes, T.C. Kaspar, J.L. Hatfield, T. S. Colvin, and Cynthia A. Cambardella. 2002. Nitrogen Management Strategies to Reduce Nitrate Leaching in Tile-Drained Midwestern Soils. Agronomy Journal 94:153–171.

Dodds, W.K., and E.B. Welch. 2000. Establishing nutrient criteria in streams. Journal of the North American Benthological Society 19(1): 186-196.

Dodds, W.K., Val H. Smith, and Kirk Lohman. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. Can. J. Fish. Aquat. Sci. 59: 865–874 (2002)

Doering, O.C., F. Diaz-Hermelo, C. Howard, R. Heimlich, F. Hitzhusen, R. Kazmierczak, J. Lee, L. Libby, W. Milon, T. Prato, and M. Ribaudo. 1999. Evaluation of the Economic Costs and Benefits of Methods for Reducing Nutrient Loads to the Gulf of Mexico. Topic 6 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico.

Doisy, K.E., C.F Rabeni. 2004. Effects of Suspended Sediment on Native Missouri Fishes: A Literature Review and Synthesis. Missouri Cooperative Fish and Wildlife Research Unit. St. Louis, Missouri.

Donner, M.T. Coe, J.D. Lenters, T.E. Twine, and J.A. Foley. 2002. Modeling the impact of hydrological changes on nitrate transport in the Mississippi River Basin from 1955 to 1994. Global Biogeochemical Cycles 163: 1396.

Dosskey, M.G., D.E. Eisenhauer, and M.J. Helmers. 2005. Establishing conservation buffers using precision information. Journal of Soil and Water Conservation 60(6): 349-354.

Downing, J.A. 1997. Relationship between habitat characteristics and the extinction of lake mussels in Minnesota. Final report submitted to the Minnesota Department of Natural Resources. 10 pp.

Downing, J.A., Hendrik Van Leeuwen, L. Antonio Di Paolo. 2000. Substratum patch selection in the lacustrine mussels Elliptio complanata and Pyganodon grandis grandis Freshwater Biology 44 4, 641–648.

Eash, David. 2001 Techniques for estimating flood-frequency discharges for streams in Iowa. USGS Water Resources Investigations Report 00-4233. Available online: http://ia.water.usgs.gov/pubs/reports/WRIR_00-4233.pdf.

Easterling, D.R., and T.R. Karl. 2001. Potential consequences of climate variability and change for the Midwestern United States, Chapter 6. In: Climate change impacts on the United States: the potential consequences of climate variability and change, report to Congress of the National Assessment Synthesis Team. New York: Cambridge University Press, 167-189.

Environmental Protection Agency EPA. 2000 Ambient Water Quality Criteria Recommendations: Information Supporting the Development of State and Tribal Criteria, Rivers and Streams in Nutrient Ecoregion VI: Corn Belt and Northern Great Plains. EPA Report 822-B-00-017. Available online: http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/rivers/rivers_6.pdf.

Environmental Protection Agency EPA. 1995. Case Study: Bear Creek, Iowa. Chapter 6 in: Ecological Restoration, EPA 841-F-95-007, November 1995. Available online: http://www.epa.gov/OWOW/NPS/Ecology/chap6bea.html.

ESRI. 2007. Creating a depressionless DEM Release 9.2 http://webhelp.esri.com/arcgisdesktop/9.2/index.cfm?id=3914&pid=3910&topicname=Creating_a_depressionless_DEM

Euliss, Jr. Ned H., David M. Mushet. 2004. Impacts of water development on aquatic macroinvertebrates, amphibians, and plants in wetlands of a semi-arid landscape. Aquatic Ecosystem Health & Management, Volume 7, Issue 1 January 2004 , pages 73 - 84

Faulkner, D.J. 1998. Spatially Variable Historical Alluviation and Channel Incision in West-Central Wisconsin. Annals of the Association of American Geographers 88 (4): 666-685.

Fitzhugh, T. 2005. Watershed Characteristics and Aquatic Ecological Integrity: A Literature Review. http://www.nature.org/initiatives/freshwater/resources/art17025.html#docs

Fitzpatrick, F.A., J.C. Knox, and H.E. Whitman. 1999. Effects of historical land-cover changes on flooding and sedimentation, North Fish Creek, Wisconsin. Water-Resources Investigations Report 99–4083. U.S. Department of Interior, U.S. Geological Survey.

Frenzel, S. A. and R. B. Swanson. 1996. Relations of fish community composition to environmental variables in streams of central Nebraska, USA. Environmental Management Volume 20, Number 5: 689–705.

Frest, T.J. 1987. Mussel survey of selected interior Iowa streams. University of Northern Iowa, Final Report to Iowa Department of Natural Resources and U.S. Fish & Wildlife Service. 354 pp.

Frothingham, K. M., B. L. Rhoads, and E.E. Herricks. 2002. A multiscale conceptual framework for integrated ecogeomorphological research to support stream naturalization in the agricultural Midwest. Environmental Management 29(1):16-33

Garbrecht, J., M. Van Liew, and G.O. Brown. 2004. Trends in precipitation, streamflow, and evapotranspiration in the Great Plains of the United States. J. Hydrol. Engin. 9, 360-367.

Gassman, P.W. 2008. A simulation assessment of the Boone River watershed: baseline calibration/validation results and issues, and future research needs. PhD. Dissertation, Iowa State University, Ames, Iowa.

Gassman, P.W., M.R. Reyes, C.H. Green, and J.G. Arnold. 2007. The Soil and Water Assessment Tool: Historical Development, Applications, and future research directions. Trans. ASAE. 50(4):1211-1250

Gassman, P.W., Silvia Secchi, and Manoj Jha. 2008. Assessment of Bioenergy-Related Scenarios for the Boone River Watershed in North Central Iowa.

Gelwicks, G. T. 1999. Evaluation of the importance of specific in-stream habitats to fish populations and the potential for protecting or enhancing Iowa's interior river resources (F-160-R). Iowa Department of Natural Resources, Federal Aid to Fish Restoration, Annual Performance Report-Study 7006, Des Moines.

Gelwicks, G. T. 2000. Evaluation of the importance of specific in-stream habitats to fish populations and the potential for protecting or enhancing Iowa's interior river resources (F-160-R). Iowa Department of Natural Resources, Federal Aid to Fish Restoration, Annual Performance Report-Study 7006, Des Moines.

Gentry, L.E., M.B. David, K.M. Smith, and D.A. Kovacic. 1998. Nitrogen cycling and tile drainage nitrate loss in a corn/soybean watershed. Agric. Ecosyst. Environ. 68:85–97.

Geohring, L.D., O.V. McHugh, M. T. Walter, T.S. Steenhuis, M. S. Akhtar, M.F. Walter. 2001. Phosphorus transport into subsurface drains by macropores after manure applications: implications for best manure management practices. Soil Science 166(12): 896-909.

Geohring, L.D., Sr., Tammo S. Steenhuis, Andrea S. Brooks, Melissa N. Rosenwald, Jennifer Chen, Cornell University and Victor J. Putnam, Essex County Planning Department. December 1999. Cost-Effective Phosphorus Removal from Secondary Wastewater Effluent through Mineral Adsorption. Larry D. LCBP Technical Report #36.

Glysson, G.D., 1987. Sediment-transport curves. U.S. Geological Survey Open-File Report 87-218. 47 pp.

Goolsby, D.A., W.A. Battaglin, B.T. Aulenbach, and Richard P. Hooper. 2001. Nitrogen input to the Gulf of Mexico. J. Environ. Qual. 30:328-336.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.P. Hooper, D.R. Keeney, and G.J. Stensland. 1999, Flux and sources of nutrients in the Mississippi-Atchafalaya River Basin: Topic 3 Report. White House Office of Science and Technology Policy Committee on Environment and Natural Resources, Hypoxia Work Group, Washington, D.C. Online: http://www.nos.noaa.gov/products/pubs_hypox.html.

Gowda, P.H. 1998. Agricultural Non-point Pollutant Loading Patterns in Upper Mississippi River Watersheds. USGS, Upper Midwest Environmental Science Center. Long Term Monitoring Program Project Status Report 98-12. http://www.umesc.usgs.gov/reports_publications/psrs/umesc_psr.html

Green, C.H., Tomer, M.D., DiLuzio, M., & Arnold, J.G. (2006). Hydrologic evaluation of the Soil and Water Assessment Tool for a large tile-drained watershed in Iowa. Transactions American Society of Agricultural and Biological Engineers, 49(2), 413-422.

Greenhalgh, Suzie , Faeth, Paul. 2001. A Potential Integrated Water Quality Strategy for the Mississippi River Basin and the Gulf of Mexico. TheScientificWorld Vol 1 Supplement no.:s2 976-983

Hanson, M., K. D. Zimmer, M. G. Butler, B.A. Tangen, B.R. Herwig, and N. H. Euliss, Jr.. 2005. Biotic interactions as determinants of ecosystem structure in prairie wetlands: an example using fish. Wetlands 25:3

Harlan, James, Everett Speaker, and James Mayhew. 1987 Iowa Fish and Fishing. Iowa Department of Natural Resources, Des Moines. 323 pp.

Heiskary, S. and H. Markus. 2003. Establishing Relationships Among In-stream Nutrient Concentrations, Phytoplankton and Periphyton Abundance and Composition, Fish and Macroinvertebrate Indices, and Biochemical Oxygen Demand in Minnesota USA Rivers. Minnesota Pollution Control Agency, St. Paul, MN.

Heitke, J.D., C.L. Pierce, G.T. Gelwicks, G.A. Simmons. 2006. Habitat, Land Use, and Fish Assemblage Relationships in Iowa Streams: Preliminary Assessment in an Agricultural Landscape. American Fisheries Society Symposium 48:287–303.

Hey, D. L. and N. S. Philippi. 1995. Flood reduction through wetland restoration: The Upper Mississippi River Basin as a case history. Restoration Ecology 31.

Higgins, J. V., M. T. Bryer, M. L. Khoury, snd T. W. Fitzhugh. 2005. A freshwater classification approach for biodiversity conservation planning. Conservation Biology 192: 432.

Hoke, Ellet. 2004 The freshwater mussels Mollusca: Bivalvia: Unionidae of the Boone River, North Central Iowa. Journal of the Iowa Academy of Science 1111,2: 43-47.

Horwitz, R.J. 1978. Temporal Variability Patterns and the Distributional Patterns of Stream Fishes. Ecological Monographs, Vol. 48, No. 3 : 307-321.

Hoyer, B.E., and Hallberg, G.H., 1991, Groundwater Vulnerability Regions of Iowa: Department of Natural Resources, Special Map Series 11.

Hupp, C. R. 1992. Riparian vegetation recovery patterns following stream channelization : a geomorphic perspective. Ecology 73:1209-1226.

Infante, D.M. 2001. The effects of channel shape on fish assemblage structure. M.S. thesis, University of Michigan, Ann Arbor, MI.

Infante, Dana M., M.J. Wiley, and Paul W. Seelbach. 2000. The Effects of Channel Incision and Low Flow Hydraulic Radius on Stream Fish Assemblages, University of Michigan, School of Natural Resources and Environment, 430 E. University, Ann Arbor MI

Iowa Administrative Code, Environmental Protection Commission Section 567 Chapter 61: Water Quality Standards effective 1977. Online: http://www.legis.state.ia.us/Rules/2003/iac/567iac/56761/56761.pdf.

Iowa Conservation Commission ICC. 1985 Boone River Protected Water Area Management Plan, Des Moines. 65 pp.

Iowa Department of Agriculture and Land Stewardship (IDALS). 1998. Iowa Wetlands and Riparian Areas Conservation Plan. Division of Soil Conservation, Des Moines.

Iowa Department of Natural Resources IDNR. 2003 Bald Eagle restoration. Available online: http://www.iowadnr.com/wildlife/pdfs/eagle%20log%202003%20complete.pdf.

IOWATER volunteer water monitoring data: http://www.iowater.net

Isenhart, T.M., J.D. Colletti, and R.C. Shultz. 1997 Watershed restoration and agricultural practices in the Midwest: Bear Creek in Iowa. Chapter 19 in Williams, J.E. ed. Watershed Restoration: Principles and Practices. American Fisheries Society. Beltsville, MD. Available online: http://www.ag.iastate.edu/centers/iawetlands/Bearcrk.html.

James, A. 1999. Time and the persistence of alluvium: River engineering, fluvial geomorphology, and mining sediment in California. Geomorphology 31, Issues 1-4, December 1999, Pages 265-290.

Jha, M., Jeffrey G. Arnold, Philip W. Gassman, Filippo Giorgi, Roy R. Gu (2006) Climate Change Sensitivity Assessment On Upper Mississippi River Basin Streamflows Using SWAT. Journal of the American Water Resources Association 42 (4) , 997–1015

Jha, M., Zaitao Pan, Eugene S. Takle, and Roy Gu. 2003. The Impacts of Climate Change on Stream Flow in the Upper Mississippi River Basin: A Regional Climate Model Perspective. Working Paper 03-WP 337.

Jha, P.W. Gassman, J.G. Arnold, S.S. Secchi, T. Campbell, and C.L. Kling. 2004. Hydrologic Modeling of the Upper Mississippi River Basin Using SWAT. Paper number 042069, ASAE Annual Meeting.

Jones, P.M. and Winterstein, T.A. 2000. Characterization of rainfall-runoff response and estimation of the effect of wetland restoration on runoff, Heron Lake Basin, Southwestern Minnesota, 1991-1997: U.S. Geological Survey Water-Resources Investigations Report 00-4095, 160 p.

Kadlec, R. H. 2005. Nitrogen Farming for Pollution Control. Journal of Environmental Science and Health, Part A, Volume 40, Issue 6 & 7: 1307 - 1330

Kalkhoff, S.J., Barnes, K.K., Becher, K.D., Savoca, M.E., Schnoebelen, D.J., Sadorf, E.M., Porter, S.D., and Sullivan, D.J., 2000, Water Quality in the Eastern Iowa Basins, Iowa and Minnesota, 1996–98: U.S. Geological Survey Circular 1210, 37 p., on-line at http://pubs.water.usgs.gov/circ1210/

Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. Illinois Natural History Survey, Special Publication 5. 28 p.

Keeney, D. and M. Muller. 2006. Water Use by Ethanol Plants: Potential Challenges. Institute for Agriculture and Trade Policy. Minneapolis, MN.

Khoury, M. 2004. Expert opinion cited in Weitzell et al. 2003, Conservation Priorities for Freshwater Biodiversity in the Upper Mississippi River Basin.

Kiepe, C. 2005. Field-level survey of the Boone River watershed including current land use, crop rotation, tillage practice, conservation practices; livestock operations; and eroded gullies or stream banks. Mr. Kiepe is a former USDA-NRCS employee and has performed similar surveys for several smaller watersheds in different parts of Iowa.

King, K.W., Smiley, P.C., Baker, B.J., Fausey, N.R. 2005. Assessing the impact of conservation management practices on water and soil quality in Upper Big Walnut Creek watershed [abstract]. ASAE Annual International Meeting. Paper No. 05-2128.

King, K.W. xxx. Impact assessment of conservation management practices on water quality in Upper Big Walnut Creek Watershed.

King, K.W., Fausey, N.R. 2006. Assessing Conservation Benefits and Costs. Ohio Country Journal. p. 47.

Kladivko, E.J., L.C. Brown, and J.L. Baker. 2001. Pesticide transport to subsurface tile drains in humid regions of North America. Critical Reviews in Environmental Science and Technology 31(1): 1-62.

Klein, R. 1979. Urbanization and stream quality impairment. American Water Resources Association. Water Resources Bulletin. 154.

Knight, G. L. and K.B. Gido. Habitat Use and Susceptibility to Predation of Four Prairie Stream Fishes: Implications for Conservation of the Endangered Topeka Shiner. Copeia Article: pp. 38–47

Knox, J.C. 1987. Historical valley floor sedimentation in the Upper Mississippi Valley. Annals of the Association of American Geographers 77(2) : 224-244

Knox, James C. 2001. Agricultural influence on landscape sensitivity in the Upper Mississippi River Valley. CATENA Volume 42, Issues 2-4 : Pages 193-224.

Kronvang, B., R. Grant, and A. L. Laubel. 1997. Sediment and phosphorus export from a lowland catchment: Quantification of sources. Water Air Soil Pol. 99:465-476.

Kunkel, K.E., K. Andsager, and D. Easterling. 1999. Long-term trends in extreme precipitation events over the conterminous U.S. J. Climate 12, 2515-2527.

Lane, E.W. 1955. The importance of fluvial morphology in hydraulic engineering. Proceedings of the American Society of Civil Engineers 81:1-17.

Langendoen, E.J., Simon, A., Pollen, N.L., Williams, R.G., Lowrance, R.R. 2006. A comprehensive stream-riparian corridor model to study the impact of riparian buffers on channel and edge-of-field processes: simulation of streambank hydrology. In: Proceedings of the 3rd Federal Interagency Hydrologic Modeling Conference, April 2-6, 2006, Reno, Nevada. CDROM.

Laubel, A., O.H. Jacobsen, B. Kronvang, R. Grant, and H.E. Andersen. 1999. Subsurface drainage loss of particles and phosphorus from field plot experiments and a tile-drained catchment. Journal of Environmental Quality. 28(2): 576-584.

Leopold, L.B. 1994. A View of the River. Harvard University Press, Cambridge, MA.

Lewis, D. B., and J. J. Magnuson. 1999. Landscape spatial patterns in freshwater snail assemblages across northern highland catchments. Freshwater Biology 41:1-12.

Libra, Robert D. and Calvin F. Wolter. 2004. Nitrogen and Phosphorus Budgets for Iowa and Iowa Watersheds. Iowa Department of Natural Resources - Geological Survey Technological Information Series 47, Des Moines. 47 pp.

Loan-Wilsey, A.K., C.L. Pierce, K.L. Kane, P.D. Brown and R.L. McNeely. 2005. The Iowa Aquatic Gap Analysis Project Final Report. Iowa Cooperative Fish and Wildlife Research Unit, Iowa State University, Ames, Iowa.

Logsdon, S.D. Thomas C. Kaspar, David W. Meek, and John H. Prueger. 2002. Nitrate Leaching as Influenced by Cover Crops in Large Soil Monoliths Agron. J. 94:807–814.

Lovell, S.T. 2006. Environmental benefits of conservation buffers in the United States: Evidence, promise, and open questions. Ecosystems & Environment Volume 112, Issue 4: 249-260

Lytle, D.A. and N.L. Poff. 2004. Adaptation to natural flow regimes. Trends in Ecology and Evolution 19(2):94-100

Mac, M.J. 1995. Aquatic ecosystems overview. In: Our living resources: a report to the nation on the distribution, abundance, and health of U. S. plants, animals, and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, DC. http://biology.usgs.gov/s+t/index.htm

Magilligan, F.J. and M.L. Stamp. 1997. Historical Land-Cover Changes and Hydrogeomorphic Adjustment in a Small Georgia Watershed. Annals of the Association of American Geographers 87 (4), 614-635.

Magner, J.A. and L. Steffen. 2000. Stream morphological response to climate and land-use in the Minnesota River Basin. In: American Society of Civil Engineers, Water Resources Engineering, Planning and Management. 2000. Reston VA.

Magner, J.A., G.A. Payne, L.J. Steffen. 2004. Drainage effects on stream nitrate-N and hydrology in south-central Minnesota (USA), Environmental Monitoring and Assessment 91, Issue 1-3: 183-198.

Mallawatantri, A.P., D.J. Mulla, and M.W. Seeley. 1999. Long-term flow trends within the Minnesota River. Hydrologic impacts of drainage. Technical Forum, Minnesota-Iowa Drainage Forum, St. Peter, MN.

Master, L. L., S. R. Flack, and B. A. Stein (Eds). 1998. Rivers of life: Critical watersheds for protecting freshwater biodiversity. The Nature Conservancy, Arlington, Virginia.

Master, L.L., B.A. Stein, L.S. Kutner, and G.A. Hammerson. 2000. Vanishing assets: Conservation status of U.S. species. Pages 93–118 in Stein BA,Kutner LS, Adams JS, eds. Precious Heritage: The Status of Biodiversity in the United States. Oxford (United Kingdom): Oxford University Press.

Mathews, W. J. 1998. Patterns in Freshwater Fish Ecology. Chapman and Hall, N.Y., NY.

Maxwell, J. R., C. J. Edwards, M. E. Jensen, S. J. Paustain, H. Parrot, and D. M. Hill. 1995. A Hierarchical Framework of Aquatic Ecological Units in North America Nearctic Zone. General Technical Report NC-176. St. Paul, MN: USDA Forest Service, North Central Forest Experimental Station.

Mayerfeld, D., Hallberg, G., Miller, G., Wintersteen, W., Hartzler, R., Brown, S., Duffy, M., and DeWitt, J., 1996, Pest Management in Iowa: Planning for the Future: Iowa State University publication IFM 17, University Extension, Iowa State University, Ames, IA, 89 p..

McIsaac, G.F. and X. Hu. 2004. Net N input and riverine N export from Illinois agricultural watersheds with and without extensive tile drainage. Biogeochemistry, 70(2), 253-273.

McRae, S. E., J . D. Allan and J. B. Burch. 2004. Reach- and catchment-scale determinants of the distribution of freshwater mussels (Bivalvia: Unionidae) in south-eastern Michigan, U.S.A. Freshwater Biology (2004) 49, 127–142

Meek, S.E. 1893. Fishes of the Cedar River basin. Proc. Iowa Acad. Sci. 1: 105-112.

Meneks, M.L., B. Vondracek, and J. Hatch. 2003. Larval fish as indicators of reproductive success in unchannelized and channelized tributaries of the Red River Basin, Minnesota. Journal of Freshwater Ecology 18(1): 141-154

Menzel, B. W. 1981. Iowa's waters and fishes: A century and a half of change. Proc. Iowa. Acad. Sci. 88(1): 17-23.

Menzel, B.W., J.B. Barnum, and L.M. Antosch. 1984. Ecological alterations of Iowa prairie-agricultural streams. Iowa State Journal of Research 5-30.

Miller, R.C. 1999. Hydrologic effects of wetland drainage and land use changes in a tributary watershed of the Minnesota River Basin: A modeling approach. M.S. Thesis, University of Minnesota, St. Paul, Minnesota.

Miranda-Castro, Leopoldo. 2006. Forging Partnerships for Habitat Restoration: Endangered Species Update.

Mitsch, W.J. and J. W. Day. 2004. Thinking big with whole-ecosystem studies and ecosystem restoration: a legacy of H.T. Odum. Ecol Modell 178 133-155.

Morton, L.W., S.S. Brown, J. Leiting. 2008. Water Issues in Iowa: A Survey of Public Perceptions and Attitudes about Water. Presentation at February 2008 Iowa Water Monitoring Conference.

Mueller, D.K., B.C. Ruddy, and W.A. Battaglin. 1993. Relation of nitrate concentrations in surface water to land use in the upper Midwestern United States,1989-90, in Goolsby, D.A., Boyer, L.L., and Mallard, G.E., eds., Selected papers on agricultural chemicals in water resources of the midcontinental United States: U.S. Geological Survey Open File Report 93-418, p. 41-50.

Mummert, Andrea K., Richard J. Neves, Tammy J. Newcomb, and Donald S. Cherry. 2003 Sensitivity of juvenile freshwater mussels Lampsilis fasciola, Villosa iris to total and un-ionized ammonia. Environmental Toxicology and Chemistry 2211: 2545-2553..

NatureServe. 2004 NatureServe Explorer: An online encyclopedia of life [web application]. Version 4.0. NatureServe, Arlington, Virginia. Available online: http://www.natureserve.org/explorer

Neugarten, R. and D. Braun. 2005. Boone River Watershed Ecological Assessment. The Nature Conservancy, Des Moines, Iowa.

Newcombe, C. P., and J. O. Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16:693–727.

Newton, T.J. 2003 Letter to the Editor: The effects of ammonia on freshwater unionid mussels. Environmental Toxicology and Chemistry 22: 2543-2544.

Newton, T. J., John W. Allran, Jonathan A. O'Donnell, Michelle R. Bartsch, and William B. Richardson. 2003. Effects of ammonia on juvenile Unionid mussels Lampsilis cardium in laboratory sediment toxicity tests. Environmental Toxicology and Chemistry 2211: 2554-2560.

Nilsson, C. and M. Svedmark. 2002. Basic principles and ecological consequences of changing water regimes: riparian plant communities, Environmental Management 30(4): 468–480.

Niyogi, D.C., K.S. Simon and C.R. Townsend. 2004. Land use and stream ecosystem functioning: nutrient uptake in streams that contrast in agricultural development. Arch. Hydrobiol. 160 (4): 471–486.

Odgaard, A. J. 1987. Streambank erosion along 2 rivers in Iowa. Water Resources Research 23(7): 1225-1236.

Omernik, J.M. 1987. Ecoregions of the Conterminous United States. Annals of the Association of American Geographers 77(1): 118 - 125.

Poff, N. LeRoy, J. David Allan, Mark B. Bain, James R. Karr, Karen L. Prestegaard, Brian D. Richter, Richard E. Sparks, and Julie C. Stromberg. 1997 The natural flow regime: A paradigm for river conservation and restoration. BioScience 47: 769-784.

Poff, N.L. and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology. Vol. 76, no. 2, pp. 606-627.

Poole, K.E. 2005. Assessment Of Stream Habitat And Survey For Mussel Species Boone River Watershed, Iowa. The Nature Conservancy. DECEMBER 2005

Porter, S.D., 2000. Upper Midwest river systems—Algal and nutrient conditions in streams and rivers in the upper Midwest region during seasonal low-flow conditions, in Nutrient criteria technical guidance manual, rivers and streams: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, EPA–822–B–00–002, p. A–25—A–42.

Porter, S.D., Harris, M.A., and Kalkhoff, S.J., 2001, Influence of natural factors on the quality of midwestern streams and rivers: U.S. Geological Survey Water-Resources Investigations Report 00–4288.

Postel, Sandra and Brian D. Richter. 2003. Rivers for Life. Island Press, Washington, D.C.

Prato, T. and Shunxiang Wu. 1996. Alternative Spatial Criteria for Targeting Soil and Water Quality Improvements in an Agricultural Watershed . Review of Agricultural Economics 18, No. 2 (May, 1996), pp. 293-301

Published by the American Society of Agricultural and Biological Engineers, St. Joseph, Michigan www.asabe.org

Rahel, F. J. 2000. Homogenization of fish faunas across the United States. Science 288(5467):854-6.

Rahel, F. J. 2002. Homogenization of freshwater faunas. Annual Review of Ecology and Systematics Vol. 33: 291-315

Randall, G.W., D.R. Huggins, M.P. Russelle, D.J. Fuchs, W.W. Nelson, and J.L. Anderson. 1997. Nitrate losses through subsurface tile drainage in Conservation Reserve Program, alfalfa, and row crop systems. Journal of Environmental Quality 26:1240-1247.

Rankin, E., B. Miltner, C. Yoder and D. Mishne 1999. Association Between Nutrients, Habitat, and the Aquatic Biota in Ohio Rivers and Streams. Technical Bulletin MAS/1999-1-1. Ohio EPA, Columbus, OH.

Richter, Brian D., Jeffrey V. Baumgartner, Jennifer Powell, and David P. Braun. 1996 A method for assessing hydrologic alteration within ecosystems. Conservation Biology 10: 1163-1174.

Richter, Brian D., Jeffrey V. Baumgartner, Robert Wigington, and David P. Braun. 1997 How Much Water Does a River Need? Freshwater Biology 37:231-249.

Roseboom, D. P. 1987. Case studies of stream and river restoration. Management of the Illinois River system: The 1990s and beyond. Illinois River Resource Management. A Governor's Conference held April 1-3. Peoria, Ill.

Rowe, D. and C.L. Pierce. 2006. Evaluations of Physical Habitat, Stressor, and Biological Response Indicators for Wadeable Streams in Iowa's Regional Environmental Assessment and Monitoring Program. Annual Progress Report to Iowa Department of Natural Resources. U.S. Geological Survey Iowa State University by Department of Natural Resource Ecology and Management and Iowa State University U.S. Geological Survey Iowa Cooperative Fish and Wildlife Research Unit.

Saleh, A. E. Osei, D. B. Jaynes, B. Du, J. G Arnold. 2007. Economic and environmental impacts of LSNT and cover crops for nitrate?nitrogen reduction in Walnut Creek Watershed, Iowa, using FEM and enhanced SWAT models. Transactions of the ASABE 504: 1251-1259.

Samson, F.B. and F.L. Knopf editors. 1996. Prairie Conservation: Preserving North America's most Endangered Ecosystem. Island Press, Covelo, California. 340 pages.

Schilling, K.E. 2000. Patterns of discharge and suspended sediment transport in the Walnut and Squaw Creek watersheds, Jasper County, Iowa: Water years 1996-1998. Iowa Department of Natural Resources, Geological Survey Bureau, Technical Information Series 42, Des Moines. 47 pp.

Schilling, K.E. 2002. Chemical Transport from Paired Agricultural and Restored Prairie Watersheds. Journal of Environmental Quality 31:1184-1193 2002

Schilling, K.E. 2004. Preliminary analysis of USGS gauge data at Webster City Word document sent via e-mail 10/12/2004.

Schilling, K.E. and C.F. Wolter, 2001. Contribution of Baseflow to Nonpoint Source Pollution Loads in an Agricultural Watershed. Ground Water 39:49-58.

Schilling, K.E. and C.F. Wolter. 2005. Estimation of streamflow, baseflow, and nitrate-nitrogen loads in Iowa using multiple linear regression models. Journal of the American Water Resources Association 41 Issue 6, 1333-1346.

Schilling, K.E. and D.S. Lutz. 2004. Relation of Nitrate Concentrations and Baseflow in the Raccoon River, Iowa. Journal of the American Water Resources Association 40:889-900.

Schilling, K.E. and J. Spooner. 2006. Landscape and Watershed Processes: Effects of Watershed-Scale Land Use Change on Stream Nitrate Concentrations. J Environ Qual 35:2132-2145

Schilling, K.E. and M. Helmers.

Schilling, K.E. and R.D. Libra. 2000. The relationship of nitrate concentrations in streams to row crop land use in Iowa. J. Environ. Qual. 29,1846-1851.

Schilling, K.E. and R.D. Libra. 2003. Increased baseflow in Iowa over the second half of the 20th century. Journal of the American Water Resources Association 39:851-860.

Schilling, K.E. and Y.K. Zhang. 2004. Contribution of Baseflow to Nitrate-Nitrogen Export in a Large Agricultural Watershed, USA. Journal of Hydrology 295:305-316.

Schilling, K.E., M.D. Tomer, Philip W. Gassman, Cathy L. Kling, Thomas M. Isenhart, Thomas B. Moorman, William W. Simpkins, and Calvin F. Wolter. 2007. Tale of Three Watersheds: Nonpoint Source Pollution and Conservation Practices across Iowa. Choices Magazine 2nd Quarter 2007.

Schilling, K.E., T. Hubbard, J. Luzier, J. Spooner. 2006. Walnut Creek Watershed Restoration And Water Quality Monitoring Project: Final Report. Iowa Geological Survey Technical Information Series 49. Iowa DNR, Iowa City, IA.

Schlosser, I.J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303: 71-81.

Schnepf, M., C. Cox, and P. Groffman (Editors). 2006. Environmental benefits of conservation on cropland. Soil and Water Conserv. Soc., Ankeny, IA , pp

Schnepf, M., C. Cox, and P. Groffman (Editors). 2006. Managing Agricultural Landscapes for Environmental Quality. Soil and Water Conserv. Soc., Ankeny, IA , pp

Schnoebelen, Douglas J., Stephen J. Kalkhoff, Kent D. Becher, and E.M. Thurman. 2003 "Water-Quality Assessment of the Eastern Iowa Basins: Selected Pesticides and Pesticide Degradates in Streams, 1996–98." USGS Water-Resources Investigations Report 03–4075. 61 pp.

Schrank, S.J., C.S. Guy, M.R. Whiles, and B.L. Brock. 2001 Influence of instream and landscape-level factors on the distribution of Topeka shiners Notropis topeka in Kansas streams. Copeia 20012: 413-421.

Schultz, R.C. T.M. Isenhart, W.W. Simpkins and J.P. Colletti. 2004. Riparian forest buffers in agroecosystems – lessons learned from the Bear Creek Watershed, central Iowa, USA. Agroforestry Systems Volume 61-62, Numbers 1-3 .

Science Assessment and Strategy Team SAST. 1994 Science for Floodplain Management into the 21st Century. Part V of the Report of the Interagency Floodplain Management Review Committee to the Administration Floodplain Management Task Force. Available through the USGS, Reston, VA, or online at http://edc.usgs.gov/sast/.

Sekely, A.C., D.J. Mulla, and D.W. Bauer. 2002. Streambank slumping and its contribution to the phosphorus and suspended sediment loads of the Blue Earth River, Minnesota. Journal of Soil and Water Conservation 575: 243-250.

Sharpe, A.J. 2005. What factors influence freshwater molluscan survival in the Conasauga River? MS Thesis NCSU.

Sharpley, A.N. and H. Tunney. 2000. Phosphorus research strategies to meet agricultural and environmental challenges of the 21st century. Journal of Environmental Quality 29: 176–181.

Shearer, J.S. 2003. Topeka shiner Notropis topeka management plan for the state of South Dakota. South Dakota Department of Game, Fish and Parks, Pierre, Wildlife Division Report No. 2003-10, 82 pp.

Shields, F. D., S. S. Knight, and C. M. Cooper. 1998. Rehabilitation of aquatic habitats in warmwater streams damaged by channel incision in Mississippi. Hydrobiologia 382, Numbers 1-3: 63–86.

Shields, F.D., Jr, S. S. Knight and C. M. Cooper. 1995. Effects of channel incision on base flow stream habitats and fishes. Biological Conservation 72(3): 411

Simon, A. 2006. Flow energy, time, and evolution of dynamic fluvial systems: implications for stabilization and restoration of unstable systems. In: Proceedings of the 2006 World Environmental and Water Resources Congress R. Graham, Ed., May 21-25, 2006, Omaha, Nebraska. CDROM.

Simon, A. and A. Collison. 2001. Scientific basis for streambank stabilization using riparian vegetation. Proceedings of the 7th Federal Interagency Sedimentation Conference, Reno, NV. pp. V-47-54.

Simon, A. and A. Collison. 2002. Quantifying the mechanical and hydrologic effects of riparian vegetation on stream bank stability. Earth Surface Processes and Landforms. 27:527-546.

Simon, A. and M. Rinaldi. 2000. Channel instability in the loess area of the Midwestern United States. Journal of the American Water Resources Association 361:133-150.

Simon, A., W. Dickerson, and A. Heins. 2004. Suspended-sediment transport rates at the 1.5-year recurrence interval for ecoregions in the United States: transport conditions at the bankfull and effective discharge? Geomorphology 58: 243-262.

Simpkins, W.W., T.R. Wineland, R.J. Andress, D.A. Johnston, G.C. Caron, T.M. Isenhart and R.C. Schultz. 2002. Hydrogeological constraints on riparian buffers for reduction of diffuse pollution: examples from the Bear Creek watershed in Iowa, USA. Water Science and Technology 45(9): 61–68.

Smiley, P.C., King, K.W. 2006. Fish-habitat relationships in drainage ditches within a predominantly agricultural watershed in central Ohio. Ecological Society of America Abstracts. p. 364.

Smith, R.A., R.B. Alexander, and G. E. Schwarz. 2003. Natural background concentrations of nutrients in streams and rivers of the conterminous United States. Environmental Science & Technology 3714: 3039-3047.

Soil Use and Management 23 (s1) , 5–15

Sprenger, M. D., L. M. Smith, and P. Taylor. 2001. Restoration of riparian habitat using experimental flooding. Wetlands 22:49-57.

Strahler, A. 1964. Quantitative geomorphology of drainage basins and channel networks. In: Chow, editor. Handbook of Applied Hydrology. NewYork: McGraw-Hill.

Straka, J.R. and J.A. Downing. 2000 Distribution and abundance of three freshwater mussel species Bivalvia: Unionidae correlated with physical habitat characteristics in an Iowa reservoir. Journal of the Iowa Academy of Science 1072:25-33.

Strauss, P. A. Leone, M. N. Ripa, N. Turpin, J.-M. Lescot, R. Laplana (2007) Using critical source areas for targeting cost-effective best management practices to mitigate phosphorus and sediment transfer at the watershed scale. Soil Use and Management 23 (s1) , 144–153

Strayer, David L., John A. Downing, Wendell R. Haag, Timothy L. King, James B. Layzer, Teresa J. Newton, and S. Jerrine Nichols. 2004 Changing perspectives on pearly mussels, North America's most imperiled animals. BioScience 54: 429-439.

Strock, J.S., P. M. Porter and M. P. Russelle. 2004. Cover Cropping to Reduce Nitrate Loss through Subsurface Drainage in the Northern U.S. Corn Belt. Journal of Environmental Quality 33:1010-1016.

Sullivan, D.J. 2000. Fish Communities and Their Relation to Environmental Factors in the Eastern Iowa Basins in Iowa and Minnesota, 1996. U.S.G.S. Water-Resources Investigations Report 00–4194. 27 pp.

Taylor, C.A., Guenter A. Schuster, John E. Cooper, Robert J. DiStefano, Arnold G. Eversole, Premek Hamr, Horton H. Hobbs III, Henry W. Robison, Christopher E. Skelton, and Roger F. Thoma. 2007. A Reassessment of the Conservation Status of Crayfishes of the United States and Canada after 10+ Years of Increased Awareness. Fisheries Vol 32 no 8

TerHaar, M.J., and E. E. Herricks. 1989. Management and development of aquatic habitat in agricultural drainage systems. UILU-WRC-89-212. Water Resources Center, University of Illinois, Urbana, 145 pp.

The Nature Conservancy (TNC). 2000. The Five-S Framework for Site Conservation: A Practitioner's Handbook for Site Conservation Planning and Measuring Conservation Success. Volume I, Second Edition.

The Nature Conservancy TNC. 2005 Indicators of Hydrologic Alteration, Version 7.0.0. Available online: http://www.freshwaters.org/tools.

Thomason, S. K., C. R. Berry, Jr., C. A. Niehus, and S. S. Wall. Constructed Impoundments in the Floodplain: A Source or Sink for Native Prairie Fishes, in Particular the Endangered Topeka Shiner Notropis Topeka?

Tisl, J., and Palas, E., 1998, Sny Magill Creek Watershed Project Final Report: Clayton County Soil and Water Conservation District, 54 p.

Tomer, M.D., D. W. Meek, D. B. Jaynes and J. L. Hatfield. Evaluation of Nitrate Nitrogen Fluxes from a Tile-Drained Watershed in Central Iowa. Journal of Environmental Quality 32:642-653 2003 © 2003 American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America

Tornes, L.H. 1986. Suspended sediment in Minnesota streams. U.S. Geological Survey. Water Resources Investigation Report 85-4312. St. Paul, MN.

Trimble, Stanley W. and Pierre Crosson. 2000. U.S. Soil Erosion Rates--Myth and Reality. Science 289 (5477): 248-250

U.S. Environmental Protection Agency (USEPA), Science Advisory Board. 2007. SAB Hypoxia Panel Draft Advisory Report. 8-30-2007.

U.S. Fish and Wildlife Service USFWS 2002. Designation of critical habitat for the Topeka shiner. Federal Register 67162, p. 54262-54306.

U.S. Fish and Wildlife Service. 2004. FINAL ENVIRONMENTAL ASSESSMENT. DESIGNATION OF CRITICAL HABITAT FOR TOPEKA SHINER Notropis topeka. Prepared by Kansas Ecological Services Field Office, Manhattan, Kansas.

Ulén, M. Bechmann, J. Fölster, H. P. Jarvie, H. Tunney (2007) Agriculture as a phosphorus source for eutrophication in the north-west European countries, Norway, Sweden, United Kingdom and Ireland: a review

United States Army Corps of Engineers USACOE 1994 Impacts of Wetlands on Floods, Boone River Basin. U.S. Army Corps of Engineers, Hydrologic Engineering Center, Report PR-23. On file, Hydrologic Engineering Center and Rock Island District offices.

United States Department of Agriculture USDA. 1999. Stream Visual Assessment Protocol. National Water and Climate Center, Technical Note 99–1.

United States Environmental Protection Agency. 2004. USEPA National Assessment Database online at http://iaspub.epa.gov/tmdl/w305b_report_V4.huc?p_huc=07100005&p_state=IA

United States Fish and Wildlife Service USFWS 1996. Review of plant and animal taxa that are candidates for listing as endangered or threatened species. Federal Register, 61, p.7596.

United States Fish and Wildlife Service USFWS 1998. Endangered and threatened wildlife and plants; final rule to list the Topeka shiner as endangered. Federal Register, 63, p.69008.

United States Fish and Wildlife Service USFWS 2004. Endangered and Threatened Wildlife and Plants; Final Designation of Critical Habitat for the Topeka Shiner. Federal Register, 69, p.44736 36 pp..

USEPA. Tolerance and trophic guilds for selected fish species. Appendix C. RBP guidance online http://www.epa.gov/owow/monitoring/rbp/app_c-3.html

van Gorp, C. 2007. Improving the Raccoon River.

Wall, S.S., C,R. Berry, Jr., C, M. Blausey, J, A. Jenks, and C,J. Kopplin. 2004. Fish-habitat modeling for gap analysis to conserve the endangered Topeka shiner Notropis topeka. Can. J. Fish. Aquat. Sci. 616: 954-973.

Ward, A., E. Powell, A. Jayakaran, J. Witter, J. D'Ambrosio, and L. Williams. 2003. Why Small Floodplains (Benches) Sometimes Build In Agricultural Drainage Ditches -Are They Beneficial?Andy

Watters, G.T. 2000. Freshwater mussels and water quality: A review of the effects of hydrologic and instream habitat alterations. Proceedings fo the First Freshwater Mollusk Conservation Society Symposium, 1999, pp. 261-274.

Weitzell, R. E, M.L. Khoury, P. Gagnon, B. Schreurs, D. Grossman, and J. Higgins. 2003 Conservation Priorities for Freshwater Biodiversity in the Upper Mississippi River Basin. NatureServe and The Nature Conservancy, Arlington, VA.

Whiting, P.J., J.F. Stamm, D.B. Moog and R.L. Orndorff, 1999, Sediment transporting flows in headwater streams: Bull. Geol. Soc. America, v. 111, p. 450-466.

Wilkin, D.C. and S.J. Hebel. 1982. Erosion, redeposition, and delivery of sediment to Midwestern streams. Water Resources Research 18(4):1278-1282.

Williams, J.D., M.L. Warren, Jr., K.S. Cummings, J.L. Harris, and R.J. Neves. 1993 Conservation status of freshwater mussels of the U.S. and Canada. Fisheries 189: 16-22.

Wilson, B.N. 2000. History of Drainage Research at the University of Minnesota. University of Minnesota and Iowa State University Drainage Research Forum.

Wilton, T. F. 2004 Biological Assessment of Iowa's Wadeable Streams. Project Report. Iowa Department of Natural Resources, Environmental Protection Division, TMDL and Water Quality Assessment Section. Des Moines, Iowa. Available online at http://www.iowadnr.com/water/tmdlwqa/wqa/streambio/index.html

Winston, MR Spatial and Temporal Species Associations with the Topeka Shiner Notropis topeka in Missouri. Journal of Freshwater Ecology [J. Freshwat. Ecol.]. Vol. 17, no. 2, pp. 249-262. Jun 2002.

Winter, T.C. 1999. Relation of streams, lakes, and wetlands to groundwater flow systems. Hydrogeology Journal 7 1: 1435-0157

Withers, A.J., P. M. Haygarth (2007) Agriculture, phosphorus and eutrophication: a European perspective. Soil Use and Management 23 (s1) , 1–4

Wright, J.P. and C.G. Jones. 2006. The Concept of Organisms as Ecosystem Engineers Ten Years On: Progress, Limitations, and Challenges. BioScience 56(3): 203-209.

Wright, J.P., W.S.C. Gurney, and C.G. Jones. 2004. Patch dynamics in a landscape modified by ecosystem engineers. Oikos 105: 336-348.

Wu, J., E.N. Castle, K. Tanaka. Reducing Nitrogen Runoff from the Upper Mississippi River Basin to Control Hypoxia in the Gulf of Mexico: Easements or Taxes?

Wu, J., Richard M. Adams, Catherine L. Kling, and Katsuya Tanaka. 2004. From Microlevel Decisions To Landscape Changes: An Assessment Of Agricultural Conservation Policies. Amer. J. Agr. Econ. 861 February 2004: 26–41

Wynn, T. 2006. Streambank retreat: a Primer. Watershed Update 4(1). AWRA Hydrology and Watershed Management Technical Committee.

Yang, W., M. Khanna, R. Farnsworth and Hayri Önal. 2003. Integrating economic, environmental and GIS modeling to target cost effective land retirement in multiple watersheds. Ecological Economics 46(2): 249-267.

Yang, W., M. Khanna, R. Farnsworth, H. Önal. 2005. Is Geographical Targeting Cost-Effective? The Case of the Conservation Reserve Enhancement Program in Illinois. Review of Agricultural Economics, Volume 27, Number 1, Spring 2005 , pp. 70-8819

Yuan, Y., Bingner, R.L., Langendoen, E.J., Wells, R.R., Simon, A., Alonso, C.V. 2006. Determination of watershed sediment sources using USDA watershed and channel models. American Society for Agricultural Engineers. Paper No. 062048, St. Joseph, Michigan, 17 pp.

Zaimes, G. N., Schultz, R. C., Isenhart, T. M. 2004. Stream bank erosion adjacent to riparian forest buffers, row-crop fields, and continuously-grazed pastures along Bear Creek in central Iowa. Journal of Soil and Water Conservation Ankeny, 2004 Vol. 59 No. 1 19-27

Zaimes, G.N., R.C. Schultz, T.M. Isenhart. 2006. Riparian land uses and precipitation influences on stream bank erosion in central Iowa. Journal of the American Water Resources Association 42 1, 83–97.

Zedler, J. B. and S. Kercher. 2005. Wetland resources: status, trends, ecosystem services, and restorability. Annual Review of Environment and Resources 30: 39-74.

Zucker, L.A.and L.C. Brown. 1998. Agricultural Drainage: Water Quality Impacts and Subsurface Drainage Studies in the Midwest Bulletin 871-98. Ohio State University Extension.

Appendices

Appendix A. Additional Data, Figures, and Tables

Table A.1. STORET water quality data for Boone River sites, 1999-2006

a. Total Phosphorus

		Median	Max
10400001	Boone River near Stratford	0.19	1.3
11400001	Boone River at Woolstock	0.14	0.19
11400002	Lyons Creek in Webster City (Site LC1)	0.20	0.49
11990003	Buttermilk Creek near Goldfield (Site BMC1)	0.50	1.3
22400004	Briggs Woods Lake	0.05	0.08
22990001	Lake Cornelia (maximum water depth)	0.07	0.13

b. Nitrogen, Nitrite (NO2) + Nitrate (NO3) as N

STORET ID		Average	Maximum
10400001	Boone River near Stratford	8.0	28
11400001	Boone River at Woolstock	8.5	14
11400002	Lyons Creek in Webster City (Site LC1)	8.7	27
11400003	Lyons Creek near Webster City (Site LC2)	0.4	0.44
11990001	Buttermilk Creek near Goldfield (Site BMC1)	3.1	12
11990003	Buttermilk Creek near Goldfield (Site BMC3)	4.1	15
22400004	Briggs Woods Lake	10.2	16
22990001	Lake Cornelia (at maximum water depth)	0.0	0.3
22990002	Lake Cornelia (mean water depth)	0.0	0.11
22990003	Lake Cornelia (shallow water depth)	0.0	0.12

c. E. coli

		Median	N	Min	Max
10400001	Boone River near Stratford	580	86	0	21000
11400001	Boone River at Woolstock	204	5	50	400
11400002	Lyons Creek in Webster City (Site LC1)	4739	9	280	23000
11400003	Lyons Creek near Webster City (Site LC2)	4200	1	4200	4200
11990001	Buttermilk Creek near Goldfield (Site BMC1)	1075	10	220	5400
11990003	Buttermilk Creek near Goldfield (Site BMC3)	540	7	150	860
21400001	Briggs Woods Park & Golf Course Beach	158	30	0	2200
21990001	Lake Cornelia Park Beach	60	29	0	1100

d. Atrazine – Boone River Near Stratford

Year	N samples	Average	Max
1999	3	0.087	0.26
2000	12	0.145	1.3
2001	12	0.139	0.66
2002	12	0.187	0.92
2003	12	0.097	0.29
2004	12	0.13	0.28
2005	12	0.114	0.31
2006	11	0.118	0.49

Table A.2. Suspended sediment effects on warmwater stream fishes (adapted from Doisy and Rabeni 2004).

Species	Lifestage	Conc(mg/L)	Duration (h)	Class	Effect	Source
Warmwater fishes	Adult (A)	620 *	48	lethal	Fish kills downstream	Hesse & Newcomb 1982
Warmwater fishes	A	22 *	8760	lethal	Fish populations destroyed	Menzel et al. 1984
Warmwater fishes	A	40	8760	chronic	Depressive effect on populations	Gammon 1970
Bluegill	A	423 *	0.05	chronic	Rate of feeding reduced	Gardner 1981
Bluegill	A	15 *	1	chronic	Reduced prey detection	Vinyard & O'Brien 1976
Bluegill	A	144.5 *	720	chronic	Growth retarded	Buck 1956
Bluegill	A	62.5 *	720	chronic	Weight gain reduced	Buck 1956
Bluegill	A	144.5 *	720	reproductive	Unable to reproduce	Buck 1956
Bluegill	I (Juvenile)	195 **	24	chronic	Increased coughing	Carlson 1984
Bluegill	I (J)	76	336	chronic	25% reduction in biomass	Sweeten 1996
Bluegill	I (J)	244	336	lethal	50% reduction in biomass	Sweeten 1996
Bluegill	Immature (I)	11	96	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Bluegill	I	20.4	48	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Bluegill	I	11.7	168	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Bluegill	I	36	168	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Bluegill	I	200	96	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Bluegill	I (J)	315	336	chronic	>20 – 40% mortality	Sweeten 1996
Bluegill	I (Larval)	39	168	lethal	>40 – 60% mortality	Sweeten 1996
Bluegill	I (L)	79	168	lethal	>60 – 80% mortality	Sweeten 1996
Bluegill	I (L)	158	168	lethal	>60 – 80% mortality	Sweeten 1996
Bluegill	I (L)	315	168	lethal	>40 – 60% mortality	Sweeten 1996
Bluegill	I	500	24	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Bluegill	I	500	48	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Bluegill	I	32	48	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Bluegill	I	16	168	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Bluegill	I	16	96	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Black bullhead	A	100000	24	chronic	Change in behavior	Wallen 1951
Black bullhead	A	225000	2	lethal	>80 – 100% mortality	Wallen 1951
Black crappie	A	85000	2	lethal	>80 – 100% mortality	Wallen 1951
Black crappie	A	200000	2	lethal	>80 – 100% mortality	Wallen 1951

Species	Lifestage	Conc(mg/L)	Duration (h)	Class	Effect	Source
Blackstripe topminnow	A	175000	2	lethal	>80 – 100% mortality	Wallen 1951
Creek chub	A	4.5 *	168	chronic	Change in behavior	Gradall & Swenson 1982
Darters	A	2045 *	8760	lethal	Darters absent	Vaughn 1979
Emerald shiner	A	11,750 **	1	chronic	Decreased feeding	Bonner & Wilde 2002 (sb)
Fathead minnow	A	39	1	chronic	Change in behavior	Abrahams & Kattenfeld 1997
Fathead minnow	A	738	96	lethal	>20 – 40% mortality	Smith et al. 1965 (cg)
Fathead minnow	A	2000	96	lethal	>60 – 80% mortality	Smith et al. 1965 (cg)
Fathead minnow	A	2000	96	lethal	>20 – 40% mortality	Smith et al. 1965 (ck)
Fathead minnow	A	100	24	chronic	Reduced activity, metabolism change	MacLeod & Smith 1966 (cg)
Gizzard shad	A	53 - 92 *	8760	lethal	Population absent	Gammon 1970
Golden shiner	A	20000	24	chronic	Change in behavior	Wallen 1951
Golden shiner	A	200000	2	lethal	>80 – 100% mortality	Wallen 1951
Golden shiner	A	50000	24	chronic	0 – 20% mortality	Wallen 1951
Golden shiner	A	338 **	0.3	chronic	Increased activity	Chiasson 1993
Green sunfish	A	9,600 *	1	chronic	Rate of ventilation increased	Horkel & Pearson 1976
Green sunfish	A + J	20000	24	chronic	Change in behavior	Wallen 1951
Green sunfish	A + J	50000	24	chronic	0 – 20% mortality	Wallen 1951
Green sunfish	A + J	210000	2	lethal	>80 – 100% mortality	Wallen 1951
Largemouth bass	A	62.5 *	720	chronic	Weight gain reduced	Buck 1956
Largemouth bass	A	144.5 *	720	chronic	Growth retarded	Buck 1956
Largemouth bass	A	144.5 *	720	reproductive	Unable to reproduce	Buck 1956
Largemouth bass	A	51	8760	chronic	Growth retarded	Hasting & Cross 1962
Largemouth bass	A	84 **	1	chronic	Change in food habits	Foster 1980
Largemouth bass	A	20000	24	chronic	Change in behavior	Wallen 1951
Largemouth bass	A	101000	2	lethal	>40 – 60% mortality	Wallen 1951
Largemouth bass	A	115000	2	lethal	>80 – 100% mortality	Wallen 1951
Largemouth bass	A	150000	2	lethal	>80 – 100% mortality	Wallen 1951
Largemouth bass	I (J)	30 **	720	chronic	Reduced activity	Heimstra et al. 1969
Largemouth bass	I (J)	150 **	1	chronic	Decreased feeding	Reid et al. 1999
Largemouth bass	I	108	96	chronic	25% reduction in biomass	Sweeten & McCreedy 2002

Species	Lifestage	Conc(mg/L)	Duration (h)	Class	Effect	Source
Largemouth bass	I	53	168	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Largemouth bass	I	116	96	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Largemouth bass	I	138	168	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Largemouth bass	I (L)	39	168	lethal	>40 – 60% mortality	Sweeten 1996
Largemouth bass	I (L)	79	168	lethal	>40 – 60% mortality	Sweeten 1996
Largemouth bass	I (L)	158	168	lethal	>40 – 60% mortality	Sweeten 1996
Largemouth bass	I (L)	315	168	lethal	>40 – 60% mortality	Sweeten 1996
Largemouth bass	I	500	168	lethal	>40 – 60% mortality	Sweeten & McCreedy 2002
Largemouth bass	I	125	96	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Largemouth bass	I	250	96	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Largemouth bass	I	500	96	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Orange spotted sunfish	A	200000	2	lethal	>80 – 100% mortality	Wallen 1951
Plains minnow	A	50000	24	chronic	Change in behavior	Wallen 1951
Plains minnow	A	150000	336	lethal	>60 – 80% mortality	Wallen 1951
Quillback	A	53 - 92 *	8760	lethal	Population absent	Gammon 1970
Redear sunfish	A	62.5 *	720	chronic	Weight gain reduced	Buck 1956
Redear sunfish	A	144.5 *	720	chronic	Growth retarded	Buck 1956
Redear sunfish	A	144.5 *	720	reproductive	Unable to reproduce	Buck 1956
Red shiner	A	11,750 **	1	chronic	Decreased feeding	Bonner & Wilde 2002 (sb)
Red shiner	A	100000	24	chronic	Change in behavior	Wallen 1951
Red shiner	A	190000	2	lethal	>80 – 100% mortality	Wallen 1951
Rock bass	A	38250	2	lethal	>80 – 100% mortality	Wallen 1951
Sand shiner	A	11,750 **	1	chronic	Decreased feeding	Bonner & Wilde 2002 (sb)
Smallmouth bass	A	53 - 92 *	8760	lethal	Population absent	Gammon 1970
Smallmouth bass***	A	8 – 81**	0.3	chronic	Reduced prey detection	Sweka and Hartman 2003
Smallmouth bass	I (J)	35	168	chronic	25% reduction in biomass	Sweeten 1996
Smallmouth bass	I (J)	305	168	lethal	50% reduction in biomass	Sweeten 1996
Smallmouth bass	I (J)	315	168	lethal	>20 – 40% mortality	Sweeten 1996
Smallmouth bass	I	11.4	24	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Smallmouth bass	I	5.8	48	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Smallmouth bass	I	58	24	lethal	50% reduction in biomass	Sweeten & McCreedy 2002

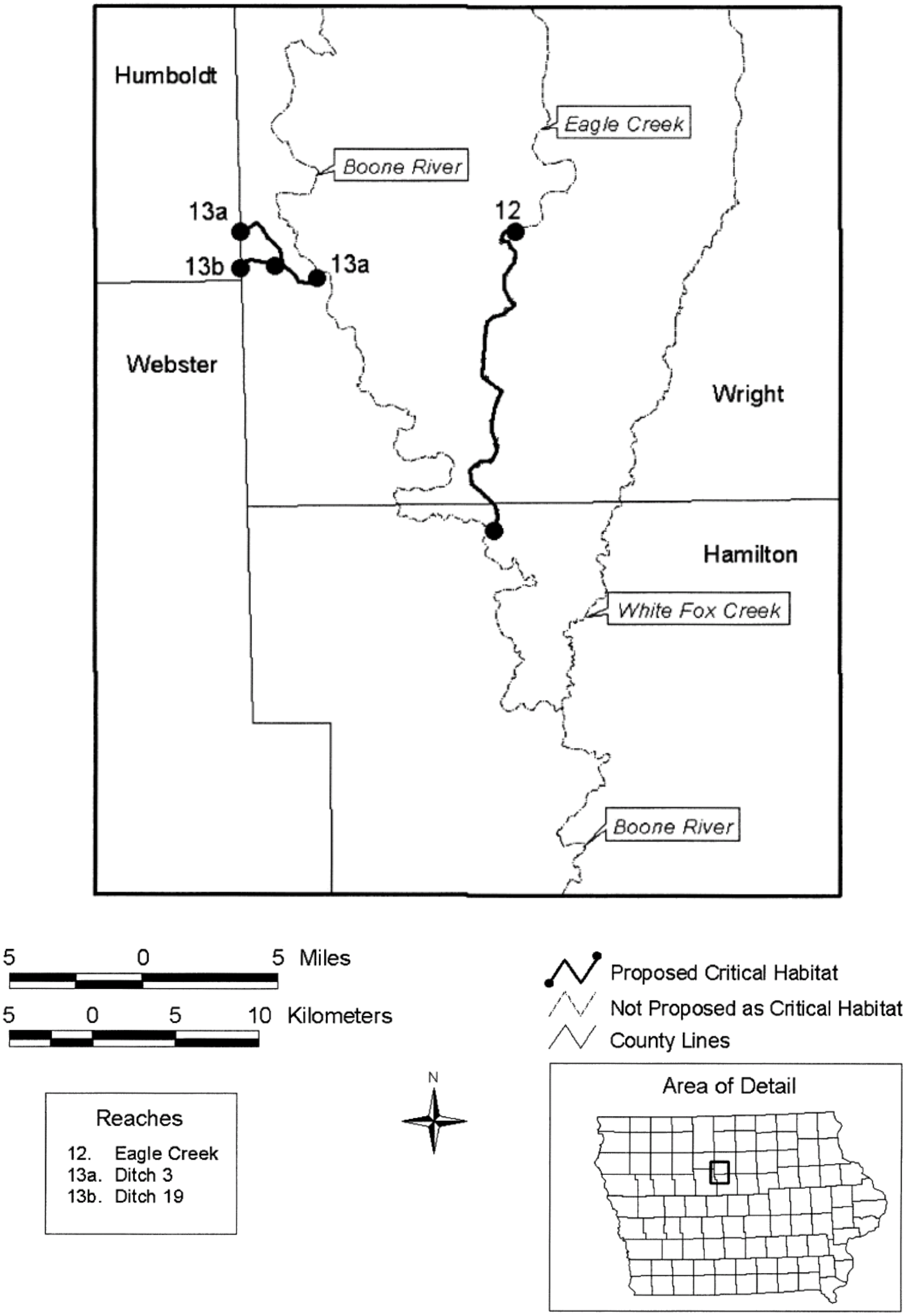
Species	Lifestage	Conc(mg/L)	Duration (h)	Class	Effect	Source
Smallmouth bass	I	9	48	lethal	50% reduction in biomass	Sweeten & McCreedy 2002
Smallmouth bass	I	25	24	lethal	50% reduction in biomass	Sweeten & McCreedy 2002
Smallmouth bass	I	282	96	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Smallmouth bass	I	32	168	chronic	25% reduction in biomass	Sweeten & McCreedy 2002
Smallmouth bass	I	8	24	chronic	0 – 20% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	16	24	lethal	>20 – 40% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	32	24	lethal	>20 – 40% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	64	24	lethal	>20 – 40% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	125	24	lethal	>40 – 60% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	250	24	lethal	>40 – 60% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	500	24	lethal	>60 – 80% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	8	48	lethal	>20 – 40% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	16	48	lethal	>40 – 60% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	32	48	lethal	>60 – 80% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	64	48	lethal	>60 – 80% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	125	48	lethal	>60 – 80% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	250	48	lethal	>80 – 100% mortality	Sweeten & McCreedy 2002
Smallmouth bass	I	500	48	lethal	>60 – 80% mortality	Sweeten & McCreedy 2002
Walleye	I (J)	74	72	lethal	>40 – 60% mortality	Smith et al. 1965 (ag)
Walleye	I (J)	100	72	lethal	>40 – 60% mortality	Smith et al. 1965 (ag)
Walleye	I (J)	272	72	lethal	>40 – 60% mortality	Smith et al. 1965 (ag)
Walleye	I (J)	2000	72	lethal	>40 – 60% mortality	Smith et al. 1965 (ag)
Walleye	I (J)	100	72	lethal	>20 – 40% mortality	Smith et al. 1965 (cg)
Walleye	I (J)	272	72	lethal	>80 – 100% mortality	Smith et al. 1965 (cg)
Walleye	I (J)	738	72	lethal	>80 – 100% mortality	Smith et al. 1965 (cg)
Walleye	I (J)	2000	72	lethal	>80 – 100% mortality	Smith et al. 1965 (cg)
Walleye	I (J)	220 **	1	chronic	Decreased feeding	Vandenbyllaardt et al. 1991
Western mosquitofish	A + J	40000	24	chronic	Change in behavior	Wallen 1951
Western mosquitofish	A + J	150000	2	chronic	0 – 20% mortality	Wallen 1951
Western mosquitofish	A + J	225000	2	lethal	>80 – 100% mortality	Wallen 1951

Table A.3. Physical habitat parameter (average values) for STORET sample sites in the Boone River watershed.

Habitat parameter	Boone River	Drainage Ditch 49	Otter Creek	White Fox Creek
Buffer Vegetation Type - Field/Sprayed/Lawn				
Buffer Vegetation Type - Herbaceous				
Buffer Vegetation Type - Mixed Grassy/Woods				
Buffer Vegetation Type - Woody				
Buffer Width - Average (feet)	100.00	45.75	93.75	100.00
Canopy - Average Percent of Channel Shaded	41.31	9.50	42.50	49.83
Canopy - Standard Deviation - Percent of Channel Shaded	17.97	21.30	24.60	31.24
Canopy - Transect Maximum Percent of Channel Shaded	73.88	52.00	70.50	93.00
Canopy - Transect Minimum Percent of Channel Shaded	10.81	0.00	20.50	40.00
Coarse Rock Embededness - Average	2.20			
Fish Cover - Large Features % Areal Cover - EPA Method	9.50			
Fish Cover - Large Features % Areal Cover - IDNR Method	16.50			
Fish Cover - Natural Concealment Features % Areal Cover - EPA Method	18.75			
Fish Cover - Total Percent Areal Cover - IDNR Method	45.00			
Fish Cover - Total Percent Areal Cover - EPA Method	33.25			
Instream Cover - Artificial Structure - Average Percent	0.25			
Instream Cover - Boulders - Average Percent	7.00			
Instream Cover - Depth/Pool - Average Percent	10.50	16.50	4.00	8.17
Instream Cover - Filamentous Algae - Average Percent	0.00			
Instream Cover - Macrophytes - Average Percent	0.00			
Instream Cover - Overhanging Vegetation - Average Percent	9.25			
Instream Cover - Small Brush - Average Percent	11.00			
Instream Cover - Trees/Roots - Average Percent	1.75			
Instream Cover - Undercut Banks - Average Percent	0.50			
Instream Cover - Woody Debris - Average Percent	11.67	4.00	64.00	28.75
Macrohabitat - Percent Pool	54.00	9.00	68.00	27.75
Macrohabitat - Percent Riffle	6.47		4.00	11.00
Macrohabitat - Percent Run	39.20	91.00	30.50	65.92
Maximum Depth (feet)	4.27	3.90	2.55	3.41
Maximum Depth Exceeds Measuring Capacity				
Reach - Total Habitat Reach Length (feet)	1180.00	513.00	855.00	1184.46
Segment Sinuosity	1.57			1.88

Habitat parameter	Boone River	Drainage Ditch 49	Otter Creek	White Fox Creek
Stream Gradient (feet/mile)	3.00			6.00
Stream Width - Average (feet)	86.71	11.82	27.20	38.36
Streambank - Percent Bare	67.17	54.25	70.00	58.27
Streambank Angle - Percent Horizontal (0-15 degrees)	55.00	0.00	35.00	13.33
Streambank Angle - Percent Moderate (20-50 degrees)	31.67	37.50	22.50	11.15
Streambank Angle - Percent Undercut (115-180 degrees)	0.00	0.00	2.50	2.69
Streambank Angle - Percent Vertical (55-110 degrees)	13.33	32.50	32.50	9.62
Substrate - Percent Bedrock	0.00			
Substrate - Percent Boulder	1.33		2.00	4.00
Substrate - Percent Clay	0.00	5.50	2.00	3.50
Substrate - Percent Cobble	12.33	3.00	2.00	14.92
Substrate - Percent Detritus/Muck	0.50	4.00		4.40
Substrate - Percent Gravel	13.67	31.50	18.50	30.15
Substrate - Percent Other	0.00			2.00
Substrate - Percent Rip-Rap	0.00			
Substrate - Percent Sand	58.33	33.00	71.00	38.00
Substrate - Percent Silt	14.00	16.50	4.50	11.17
Substrate - Percent Soil	0.00	20.00	2.00	
Substrate - Percent Wood	0.00		4.00	2.40
Thalweg Depth - Average (feet)	1.57	2.36	0.87	1.40
Thalweg Reach Transect - Percent Soft Sediment	75.00			
Transect Depth - Standard Deviation (feet)	0.63	0.74	0.44	0.63
Width - Thalweg Depth Ratio	40.77	4.33	24.50	22.92

Map A.1. Critical habitat for Topeka Shiners in the Boone River watershed.



Reprinted from the **Federal Register** / Vol. 69, No. 143 / Tuesday, July 27, 2004 / Rules and Regulations / 44765

Appendix B. Boone River Ecological Assessment and Appendices, Rachel Neugarten and David Braun, 2005.

Appendix C. Boone River Rapid Watershed Assessment, NRCS, 2008.

Appendix D. Assessments of Practices to Reduce Nitrogen and Phosphorus Nonpoint Source Pollution of Iowa's Surface Waters. USDA-ARS National Soil Tilth Laboratory, for the Iowa Department of Natural Resources, Dana Dinnes, 2005.