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IOWA WATER CENTER at IOWA STATE UNIVERSITY

Assessing the Health of Streams in Agricultural Landscapes:

The Impacts of Land Management Change on Water Quality

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The Science Source for Food, Agricultural, and Environmental Issues

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The development and printing of this publication were funded in part by the Iowa Department of Agriculture and Land Stewardship.

Assessing the Health of Streams in Agricultural Landscapes: The Impacts of Land Management Change on Water Quality

Layout and cover design by Richard Beachler, Studio 2172,
Boone, Iowa
Cover photos source: Shutterstock Images LLC, New York

ISBN 978-1-887383-34-9
ISSN 0194-407x
17 16 15 14 13 5 4 3 2 1

Special Publication

No. 31 March 2012

Council for Agricultural Science and Technology

Ames, Iowa, USA

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Foreword

In February 2010, the Board of Directors of the Council for Agricultural Science and Technology (CAST) approved partnering with the Iowa Water Center (IWC) on a three-part project: (1) establishing a Task Force of scientific experts to prepare a Special Publication on the topic "Assessing the Health of Streams in Agricultural Landscapes," (2) managing the editorial and publishing processes to produce a hardcopy document, and (3) presenting the publication through CAST's rollout and distribution processes.

An eminent group of six experts was selected as the writing Task Force, led by Dr. Rick Cruse of the IWC as project manager. Three highly qualified scientists were invited to serve as peer reviewers, and a member of the CAST Board of Representatives served as project liaison. The authors prepared an initial draft of this document and reviewed and revised all subsequent drafts based on reviewers' comments. The CAST Board of Directors reviewed the final draft, and all Task Force members reviewed the galley proofs. The CAST staff provided editorial and structural suggestions and published the document in cooperation with the IWC. The Task Force authors are responsible for the publication's scientific content.

On behalf of CAST, we thank the Task Force members who gave of their time and expertise to prepare this publication as a contribution by the scientific community to public understanding of the issue. We also thank the employers of the scientists, who made the time of these individuals available at no cost to CAST.

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Multiple agricultural land management policies and government-funded programs target improving surface water quality in Iowa and other agriculturally dominated states. Program costs have been and remain high, with improvements in stream water quality infrequently well correlated with program investments. The Iowa Water Center is partnering with CAST to address the relationship between land management and stream water quality, and in particular factors that seem to negate or minimize the impacts of farm management practice changes on stream water quality improvements.

The Iowa Water Center, one of 54 Water Resources Research Institutes in the United States and its territories, is working with partners such as CAST to improve water resources for Iowans and other stakeholders. This publication and others (see http://www.water.iastate.edu/homepage.htm) are developed to inform policymakers, educate the water-literate public, and improve the water resources for a wide array of stakeholders.

Comments regarding this document or suggestions for future science-based reports are welcome and can be sent to:

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Interpretive Summary

Streams and rivers are among society's most valued natural resources. Human activities, especially farm production activities in agriculturally dominated watersheds, have a significant potential to alter the health and well-being of those aquatic systems. The response of streams to the physical and chemical alterations caused by human activities determines their health, sustainability, and value to society. This publication concludes that although agricultural land use and practices play a key role in determining stream water quality, a number of other factors (e.g., geology, climate, and land use history) may strongly influence water and biological quality under a variety of scenarios.

A healthy stream contains high-quality physical habitat that meets the requirements of the aquatic life the stream is capable of supporting. This could include the substrate on the streambed for aquatic invertebrates, woody debris for fish cover, a mixture of pool and riffle habitat, and a robust zone of native riparian vegetation. The biological diversity of a healthy stream is influenced by the flow conditions, physical habitat, and water chemistry within the stream, which in turn are determined largely by the geologic and geomorphic setting in which the stream occurs. The water chemistry of healthy streams can range widely based on the geology and soil characteristics of the watershed. Factors such as excessive nutrients, overly warm water temperatures, low dissolved oxygen, and toxic compounds (e.g., pesticides) contribute to unhealthy conditions by pushing streams outside the natural range in water chemistry.

As population has increased, so too has human influence on stream condition. With increased urban development in agricultural watersheds comes a multitude of stressors on streams and rivers independent of agriculturally related activity. Separating or isolating urban- vs. agricultural-related impacts can be, and often is, challenging. Because of the large land surface that agriculture influences and the total volume of crop management materials applied to these lands, agriculture's assumed role in the consideration of stream health, or lack thereof, remains large. Additionally, urban areas are highly regulated by the Clean Water Act, so most of those areas have treatment or storage requirements. Streams are closely linked to their surrounding landscapes and reflect their watersheds, which, when adversely affected by agricultural practices, may cause hydrologic alteration and/or decrease the quality of all water flowing in the area.

The causes of poor water quality must be identified before stream health can be repaired. Changes in agricultural practices to accommodate, for instance, the Clean Water Act have had significant positive impacts in restoring the integrity of the nation's waterways. But broad management alterations may be insufficient. One critical element in promoting healthy streams and rivers is understanding both when and where stream water quality should respond to changes in agricultural practice. Expectations that land management practice changes alone will result in an immediate water or biological response may be unrealistic, and even expectations of long-term impacts may not be reasonable. By nature, some streams may be more sensitive to impacts from agricultural practice than other stream systems. Additionally, changes are more readily detected in small watersheds than in rivers drained by larger, more complex watersheds.

As people focus on agriculture and even urban impacts on stream water quality and stream health, they must be aware that a variety of human alterations to streams as well as natural factors may cause streams to be large sources of sediment and nutrients. Stream straightening and damming, and removal of natural vegetation in the riparian areas, exemplify human-induced changes that negatively impact stream health and water quality. In those cases where stream habitat is highly degraded by human actions (rather than naturally high in sediment), stream restoration and management can help improve water quality and may actually be prerequisites for solving water quality issues.

Much money has been spent for soil conservation and water quality practices, primarily in the area of agriculture. Still, improving the quality of streams and rivers remains a challenge. Enacted conservation practices should theoretically have solved many of the problems with water quality, but factors such as lag time (the amount of time between practice implementation and observation of a response) and buffering capacity (the natural resistance of some streams that slows responses to management activities) have to be considered in monitoring for improvements. Changes in agricultural practices may not be enough in some cases, demonstrating that land management is only a piece of the puzzle.

This publication considers that a whole systems approach, one that takes into account the many factors responsible for water degradation, may be necessary to repair the quality of the nation's streams and rivers. A more integrated watershed management approach and a systematic understanding of the role land and stream elements play in stream water quality and function are critical to (1) identifying the causes of water degradation, and (2) devising a strategy to improve water conditions in the nation's streams in agriculturally dominated watersheds.

1 Introduction

Awareness of decreased river water quality and function, production and maintenance of healthy in-stream biota, and processing of materials entering rivers has existed for decades in many, if not most, agriculturally dominated watersheds (USEPA 1972). To improve water quality and stream function in these watersheds, it is paramount to identify causes of impairment and develop measures to adequately mitigate those causes (Figure 1.1).

Current efforts that target nonpoint source pollution focus primarily on altering agricultural management in agriculturally dominated watersheds. Based on the perceived strong connection between agricultural practices and surface water quality, state and federal governments are investing substantial financial resources through multiple programs to modify existing agricultural production activities. From 1990 to 2009, more than \$2.6 billion was spent in the Section 319–Nonpoint Source Management Program created by the Environmental Protection Agency to support and provide technical and financial assistance, educate, train, transfer technology, demonstrate projects, and monitor outcomes to assess the success of specific nonpoint source implementation projects (USEPA 2011a). In Iowa alone, approximately \$435 million has been invested annually to implement land use practices that, based on sound science, should favor soil conservation and water quality (Kling et al. 2007). The Mississippi River Basin Healthy Watersheds Initiative will infuse \$320 million in fiscal years 2010 to 2013 for projects in states contributing water to the Mississippi River (USDA–NRCS 2010a).

In spite of these large investments, evidence is quite strong that improving stream water quality, at least in the Mississippi River Valley Basin, remains a challenge (Sprague, Hirsch, and Aulenbach 2011; Turner, Rabalais, and Justic 2008); this occurs while modeling estimates indicate that nutrient, sediment, and pesticide loading from agricultural land to streams should have been decreased through application of conservation practices (USDA–NRCS 2010b). In Iowa, a state in which nearly all watersheds are agriculturally dominated, computer modeling estimates that hypothetical placement of a broad set of conservation practices on the agricultural landscape should have the following effects: sediment, phosphorus, and nitrate delivery to surface water would be 35 to 94%, 31 to 72%, and 80 to 95%, respectively, of that observed with the baseline (Secchi et al. 2005). These broad projected reductions, however, may not be realized or may be insufficient to improve the biological integrity of water bodies, meet water quality criteria, or significantly increase functions of aquatic ecosystems.

Figure 1.1. Filamentous algal bloom during July in an Indiana stream. Dense blooms of filamentous algae are common in nutrient-rich, open canopy streams in agricultural regions. (Photo courtesy of Todd Royer.)

In reality, land management practices are among a number of factors, ranging from stream condition to distance of practice from the stream, that impact stream water quality and function. This suggests that assuming agricultural practice change alone will improve surface water quality, at least in the short term, needs to be reconsidered, even though modeling efforts suggest impairment loads may be decreased through conservation practice implementation. A whole systems approach can offer greater insight into pathway and process. Systematic understanding of the role land and stream factors play in stream water quality and function is critical to identifying the cause of water degradation and devising a strategy to improve water conditions in the nation's streams in agriculturally dominated watersheds.

Both the structure and function of streams are degraded by increases in urban activities and land use and as such can exacerbate and mask nonurban impact on aquatic ecosystems within watersheds having even small towns and cities (Booth and Jackson 1997; Burcher and Benfield 2006; Chadwick et al. 2006; Moore and Palmer 2005; Paul and Meyer 2001). Whereas impacts associated with urban development tend to be great, however, they are somewhat restricted to small geographic areas (e.g., urban areas, sub-basins) as opposed to large-scale alterations that variously affect the whole basin (Morley and Karr 2002). Impervious cover (IC), whether measured as total impervious area, effective impervious area, or an impervious surface coefficient (ISC), in an urban environment or watershed is a well-documented indicator (i.e., predictor) of urban impacts on the hydrology, geomorphology, and biology of streams, with significant impairments typically occurring at or above an ISC of 10% (Brabec, Schulte, and Richards 2002; May et al. 1997; McMahon and Cuffney 2000; Paul and Meyer 2001; Schueler 1994; Wang, Lyons, and Kanehl 2001). Total impervious area percentages as low as 4%, however, were noted to affect macroinvertebrate communities (Walsh et al. 2007).

While IC is the most often identified indicator of urban impacts, many urban-related factors collectively contribute to alterations of the physical, chemical, and biological conditions of streams (Barringer, Reiser, and Price 1994; Kalkhoff et al. 2000; Klein 1979; Omernik 1976; Porcella and Sorenson 1980; USGS 1999). Aside from urban land use, contributions from combined sewer overflows and wastewater treatment plants, as well as habitat degradation, were also factors associated with reductions in biological integrity (Miltner, White, and Yoder 2004; Ourso and Frenzel 2003). In summary, the authors feel that urban areas most often mimic point source impacts within the larger framework of agriculture watersheds; represent areas of intense, near-irreversible land development; and as such, demand separate attention and programs, many of which are already in place.

In order to be complete in the discussion of factors that are contributing to stream impairments and complicating their biological recovery, the authors have included pesticides and anthropogenic compounds in their synopsis but not in the general text. Pesticides are important pollutants in agricultural watersheds, but this topic is so complex with new products and uses changing over time that its inclusion is minimized in this document. Instead, the authors have focused more on nutrients and other factors that currently are in the forefront of debates and concerns. Any future review of pesticides and other man-made chemicals in agricultural watersheds needs to include discussions of existing water quality standards, the complex exposure regimes that occur with annual and episodic applications, the co-occurrence of pesticides and their metabolites, and the changing field of new compounds. In addition, future works need to address the legacy impacts of older, more persistent compounds usually associated with sediments that are not often collected and analyzed. Given all these concerns and the fact that there are virtually hundreds of man-made compounds occurring or that can occur in our aquatic ecosystems, covering this topic in a separate effort seemed most prudent.

Land Surface Change

The character of the Midwest landscape contributing water to streams has changed dramatically during the past 150 years from that of predominantly tallgrass prairie and forested riverine corridors with marshy headwaters before Euro-American settlement to today's extensively altered agricultural landscape. Widespread prairie plow-down in the mid-to-late 1800s to facilitate agriculture development transformed the landscape. Changes in agriculture continued throughout the twentieth century, including increased use of mechanization, artificial drainage, extensive ditch drainage in some regions, and commercial fertilizers, converting a more diversified land cover to an agricultural landscape dominated by annual row crops of corn (*Zea mays* L.) and soybeans (*Glycine max* [L.] Merr.). Extensive drainage systems brought more land into row crop production, although eliminating ecologically and hydrologically important wetlands.

Stream Degradation

Science suggests that more conservation-oriented agricultural activities will decrease nonpoint source contaminant transport to associated streams (Kling et al. 2007) and therefore improve water quality. It is less frequently recognized that stream water quality and function in agriculturally dominated watersheds may be difficult to improve because streams have been degraded through straightening, past sediment and nutrient deposition, dam construction, major flooding, altered hydrology, and urban influences. The impact of stream modifications can last for decades (Urban and Rhoads 2003). In other words, ongoing internal adjustments of streams to historic alterations may be significant contributors to water impairment, masking improvements associated with land management changes. Because of differences in flow regimes, bed characteristics and grade, soils, bank and channel materials, and historic damage, some streams may respond favorably to agricultural practice change whereas others may not. It is also necessary to recognize that practices distant from a stream may not, or only marginally, impact water quality in the receiving stream. Understanding when, where, and how stream water quality should respond to agricultural practice change is critical. Expectations that land management practice changes alone will be reflected in an immediate water response may be unrealistic, and even expectations of long-term impacts may be unreasonable. Nonpoint source pollution research has typically failed to deal adequately with both temporal and spatial scale (Magner and Brooks 2008).

Policy Implications

It seems that strategies and/or policies to improve water quality in agriculturally dominated watersheds should be developed based on reasonable expectations grounded in science that includes both land management and stream conditions, such as integrated water resource or watershed management (Brooks et al. 2003). Furthermore, if improving water quality and stream function is the goal, then exclusively targeting farm practices based on their surface water impacts seems wise only if and when conditions exist for which these practices alone are driving water quality change. Recognizing that stream water quality may not respond adequately to changing today's agriculture practices because of factors unrelated, or only marginally related, to current farming practices is an essential part of this strategy development.

In other words, the question isn't "Is conservation and land management making a change?" but rather "Are the changes being made making a difference?" This paper seeks to (1) examine issues linking land management to tangible environmental change, (2) address the time scale for change, and (3) identify factors that may be limiting water quality and biological improvements in the presence of improved practices within agriculturally dominated watersheds.

Abbreviations and Acronyms

IC	Impervious cover
ISC	Impervious surface coefficient

2 Stream Ecology Primer

Streams and rivers have played a vital role in human history and continue to be among society's most valued natural resources. Unfortunately, in many regions streams and rivers are highly modified and impaired because of a variety of human activities (see Table 2.1). Following is a brief overview of the structure and function of streams, the benefits they provide, and how they respond to physical and chemical alterations.

Table 2.1. Principle mechanisms by which land use affects stream systems (adapted from Allan 2004, Table 1)

Land Use Impact	Effects
Sediment delivery	• Increases turbidity • Impairs function of stream-groundwater interaction zone • Decreases primary production and food quality • In-filling of interstitial crevices harms crevice-occupying organisms and decreases availability of suitable substrate for gravel-spawning fishes • Coats gills and respiratory surfaces • Decreases stream depth heterogeneity and leads to a decrease in pool species
Nutrient enrichment	• Increases autotrophic biomass and production, resulting in proliferation of filamentous algae, especially if adequate light is available • Accelerates litter breakdown rates • Synergistic effects may cause decrease in dissolved oxygen and a shift to more tolerant species
Contaminant pollution	• Increases heavy metals, synthetics, and toxic organics in suspension and in bed materials and increases concentrations of dissolved pharmaceuticals and pesticides • Increases deformities • Increases mortality rates and negatively impacts drift and emergence in invertebrates • Depresses growth, reproduction condition, and survival among fishes • Endocrine system disruption leads to physical avoidance
Hydrologic alteration	• Alters runoff-*evapotranspiration*[1] relationship, promoting increases in flood magnitude and frequency • Changes *baseflow* contribution to stream flow • Alters channel dynamics • Increases nutrient, sediment, and contaminant transport efficiency, thereby negatively impacting downstream areas
Riparian clearing/canopy thinning	• Decreases shading, which increases stream temperatures and the magnitude of stream diurnal temperature variation during low flow • Increases light penetration and in-channel plant growth • Decreases bank stability and inputs of litter and wood • Decreases trapping of sediment from adjacent landscape • Alters quality and character of dissolved organic carbon reaching streams • Lowers retention of benthic organic matter by decreasing direct input and loss of retention structures • Alters trophic structure
Loss of *large woody debris*	• Decreases substrate for feeding, attachment, and cover • Decreases energy dissipation • Decreases bank stability • Decreases fine-grained sediment and organic material storage • Disrupts habitats by altering flow hydraulics • Has deleterious effects on invertebrate and fish diversity and alters community function

[1]Italicized terms (except genus/species names and published material titles) are defined in the Glossary at the end of each chapter.

Stream as Reflection of Watershed

Streams transport water, sediments, and dissolved constituents from their catchment. The hydrologic and geomorphic conditions under which this transport takes place are dictated by regional climate, geologic materials, and human land use patterns. For example, regional variations of stream characteristics across the agriculturally productive Midwest occur because of broad climatic gradients that influence precipitation patterns and potential native vegetation, differences in geologic materials and geologic history, and land use. Annual precipitation amounts increase from about 0.5 to 1.3 meters (m; 20 to 50 inches) northwest to southeast across the region, whereas mean annual temperatures decrease from south to north. Potential natural vegetation roughly reflects the precipitation and temperature gradients with grasslands dominant west of the Missouri River Valley and across the Prairie Peninsula in Missouri, Iowa, and Illinois; deciduous forest in the humid east and south; and mixed deciduous/coniferous forest in the north.

The upper Midwest has a diverse geologic history reflected in its variety of parent materials. Thick-to-thin wind-blown silt (loess) overlying weathered glacial till is the most common situation across the western, central, and southern part of the region (Figure 2.1). Glacial till and related deposits are at the land surface in southern Minnesota, central Iowa, and northern Illinois, where the last advances of Pleistocene glaciers occurred after regional loess deposition had ceased (Bettis et al. 2003). In northeast Iowa, southeast Minnesota, southwest Wisconsin, and northwest Illinois (see Figure 2.1), a thinner mantle of loess and patchy weathered glacial till overlies Paleozoic rocks (primarily limestone, dolomite, and sandstone) that crop out extensively on the landscape.

The defining characteristic of streams is the downstream flow of water through a defined channel. Streams and rivers are referred to as *lotic* systems, as opposed to *lentic*, or nonflowing systems such as wetlands and lakes. The organisms found in streams display a wide range of adaptations to life in flowing water (see Allan and Castillo 2007). Indeed, many stream organisms have specific flow requirements that preclude their existence in lentic systems. Flowing water also imparts a dynamic nature to streams and rivers that results in areas of natural erosion, transport, and deposition within a stream channel. Unconstrained low-gradient streams often meander and form sinuous channels that can migrate across a valley floor. The extent to which a stream meanders across the valley is controlled by the valley gradient, topography, sediment load, and geologic landforms of the watershed in which the stream occurs. The science of *fluvial geomorphology* describes the interactions among landforms, stream channels, and sediment transport (Leopold, Wolman, and Miller 1964). It is important to note that a certain amount of erosion, deposition, and sediment transport is to be expected in all streams because these are natural processes and part of the dynamic nature of streams.

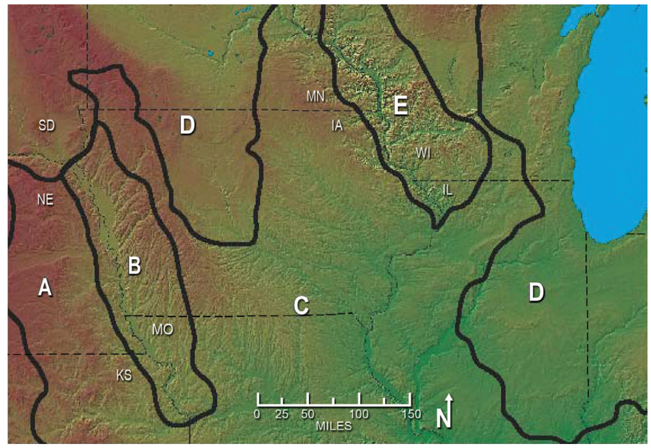

Figure 2.1. Hydrogeomorphic regions of the upper Midwest discussed in the text. A—glaciated plains; B—thick loess region; C—loess-mantled drift plain; D—recently glaciated plains; E—area of extensive Paleozoic rock outcrop. (Source: Art Bettis, University of Iowa.)

Streams are closely linked to the surrounding landscape in a variety of ways (Frissell et al. 1986). Stream flow, or *discharge*, can respond quickly to precipitation because the watershed serves to collect rainfall or snowmelt and route it to stream channels through a variety of mechanisms including artificial drainage systems, shallow and deep groundwater pathways, and *overland flow*. Each of these mechanisms can transport material to the stream. From upland areas, however, it generally is overland flow that erodes soil and transports sediment to streams. Overland flow occurs when precipitation falls on soils that already are saturated with water or when the rate of precipitation exceeds the rate at which water infiltrates the soil, or *hydrophobic* soil conditions. Whether a given precipitation event will generate overland flow is dependent on soil conditions, the presence of subsurface drainage tiles, the type and extent of vegetation, and the intensity and duration of the precipitation.

Compounds that are sorbed to soil particles, such as phosphorus (P), will also be transported to streams by overland flow and bank and bed erosion.

During dry periods, water seeping through saturated sediment and rock is the source water for streams. When the discharge in a stream consists only of inputs from shallow and deep subsurface flow, the stream is said to be at baseflow. During baseflow, erosion and sediment transport are minimal and streams tend to have high water clarity. Compounds that have infiltrated the soil and entered the shallow groundwater, such as nitrate, will be transported to the stream with the movement of groundwater. When the level of the shallow groundwater drops below the streambed, streams may go dry. Such streams are referred to as *ephemeral*, as opposed to *perennial streams*, which flow continuously. Ephemeral streams occur commonly in arid regions but also can exist in more humid regions, such as the central United States. Channels of ephemeral streams often have well-developed vegetation during the dry period, and such vegetation can slow the movement of water and decrease erosion if heavy precipitation should occur.

The soils and geologic material of a watershed have a strong influence on a stream. Streams vary naturally in many aspects of water chemistry, and this is largely controlled by the underlying geology of the stream's watershed. The geology of a watershed also influences the types and sizes of materials found on a streambed. The streambed of a low-gradient stream flowing through glacial till often will consist mainly of small particles, such as silt/clay and sand. In higher-gradient channels, streambeds are often composed of larger gravel and cobbles. The substrate on the streambed serves as critical habitat for algae, aquatic insects, and fish. The interface between the streambed and the water column is the *benthic zone*, and it is within the benthic zone of a stream that most aquatic organisms occur. The benthic zone extends from the sediment-water interface to a depth of approximately 10 centimeters (cm; 4 inches) into the streambed. The benthic zone supports many of the important chemical and biological processes that occur within a stream.

Terrestrial vegetation contributes food resources in the form of plant material and insects to streams, particularly from the *riparian zone*, the area immediately adjacent to the stream channel (see Gregory et al. [1991] for an overview of riparian zones). Riparian zones often are characterized by a unique set of vegetation that prefers the moist environment along a stream. Climate exerts a strong influence on the type of vegetation surrounding a stream, with deciduous trees dominant in humid regions and a transition to grasses and shrubs in drier regions. Riparian vegetation can stabilize stream banks, help maintain cooler water temperatures in the stream, and contribute woody debris to the channel, which in turn provides important habitat for fish and invertebrates. Riparian zones also serve to influence the movement of water and sediment via trapping and redistributing before reaching the stream channel. The riparian zone subsurface water usually converges en route to the stream, sometimes passing through organic-rich soils. The chemical conditions within the riparian soils can vary but, given the right conditions, serve to promote *denitrification*, a biological process that converts nitrate to nitrogen gas and thus removes nitrate from groundwater. Denitrification can also occur within the benthic zone of a stream. The extent to which processes such as denitrification occur within a watershed is largely determined by the soil microorganisms present, the *soil reduction/oxidation* conditions, and the geology of the region.

Defining and Assessing Stream Health

When speaking of human health, one generally understands "health" to include a number of factors, some of which are obvious, such as body mass, and others that are invisible to the casual observer, such as blood pressure. Likewise, stream health (or integrity) is a general term that encompasses many individual variables. Some characteristics of stream health, such as bank erosion, are visually obvious, whereas others, such as water chemistry, must be quantified through sampling and laboratory analysis. The Clean Water Act addresses the physical, chemical, and biological attributes of a water body and many aspects of these attributes that contribute to stream health. A healthy stream contains high-quality physical habitat that meets the requirements of the aquatic life the stream is capable of supporting. This could include the substrate on the streambed for aquatic invertebrates, woody debris for fish cover, a mixture of pool and riffle habitat, and a robust zone of native riparian vegetation. The water chemistry of healthy streams can range widely based on geologic and soil characteristics, but in most cases compounds such as pesticides, high nutrient levels, and overly warm temperatures contribute to unhealthy water chemistry and poor stream health. Throughout this document, the term "stream health" is used to indicate the overall physical, chemical, and biological condition of a stream and is synonymous with other common terms such as "integrity" or "condition."

Biologically, a healthy stream supports a diverse and productive community of *primary producers* (algae and aquatic plants), invertebrates, and fish (Giller and Malmqvist 2000). A healthy stream ecosystem can also support semi-aquatic or terrestrial species in adjacent areas, such as amphibians and birds in the riparian zone. The density and diversity of invertebrates and fish can vary among streams, and there is no single, definitive value for fish or invertebrate diversity that signifies a healthy or unhealthy stream. Streams vary in many physical and chemical characteristics (as influenced by the geomorphologic setting of the stream), and these differences are reflected in the types of fish and invertebrates that occur in the streams. Thus, the biological diversity of a healthy stream is determined by the flow conditions, physical habitat, and water chemistry within the stream, which are in turn determined largely by the geologic and geomorphic setting in which the stream occurs. Importantly, the biological health of a stream can be decreased by the presence of exotic and invasive species or an abundance of undesirable species, such as nuisance aquatic plants or low-value fish. Other biological metrics that often indicate a water quality problem include diseases and deformities in fish, amphibians, or waterfowl.

Not all fish and aquatic invertebrates are equally sensitive to changes in water quality or physical habitat. For example, some aquatic invertebrates can tolerate very low dissolved oxygen concentrations, whereas others disappear from a stream if dissolved oxygen becomes low for even a short period of time. Similarly, some fish species feed in turbid, sediment-laden water, whereas others require clear water for identifying and catching prey. This variability in pollution responses among organisms provides a means for assessing the health of streams. *Bioassessment* is a method in which biological attributes, such as the diversity of fish and invertebrate species, are used to determine the health of a stream (Hughes et al. 2010). The index of biotic integrity is used in many states today as an indicator of ecological health (Karr 1981). By understanding the pollution tolerance and habitat requirements of individual species (or groups of related species), the overall health of a stream can be accurately assessed based on the abundance and diversity of the various organisms found in the stream. Such assessments are typically made in comparison to *reference sites*, which are streams known or believed to be minimally degraded by human activities. It is important to note that reference sites usually are not "pristine" streams flowing through a watershed of only native vegetation. Rather, reference sites represent the best attainable water quality and biological condition of a region, given the land use and land cover characteristics of that region.

Bioassessment of streams most often is based on fish and invertebrates, but occasionally other organisms are used, such as algae and diatoms. The chemical and biological samples collected during field sampling can be analyzed in a variety of ways, depending in part on the specific questions and goals of the assessment. Likewise, there are several ways by which the physical habitat of the stream and riparian zone can be documented and analyzed. The Environmental Protection Agency has developed detailed protocols for use in stream bioassessment (Barbour et al. 1999).

Stream Response to Degradation

In agricultural landscapes, stream degradation tends to involve changes in the watershed, riparian vegetation, water chemistry, and in-channel habitat. Activities that cause degradation are called *stressors* because they place a stress on the health of the stream. The type, intensity, and location of the stressors will determine the extent to which the stressors degrade the health of the stream. Often, multiple stressors occur simultaneously. Likewise, activities to improve stream health can address more than one stressor. For example, restoration of riparian vegetation might lower water temperature, decrease sediment inputs, and stabilize bank erosion—three common stressors to agricultural streams.

Nitrogen (N) and P have the same fertilizing effect on algae and aquatic plants that these nutrients have on terrestrial plants. The result of nutrient loading to streams is excess growth of algae and aquatic plants. This is the process of *eutrophication*, and it is one of the more common stressors to streams in agricultural regions. Severe eutrophication can deplete dissolved oxygen concentrations at night and kill fish, facilitate harmful algal blooms, cause odor problems, and decrease the recreational and aesthetic value of a stream (Miltner and Rankin 1998). Eutrophication also decreases the diversity of pollution-sensitive invertebrates and fish but may increase the abundance of undesirable species of aquatic organisms. Under normal flow conditions, streams have a natural capacity to process and retain nutrients because of the biological activity of microorganisms in the benthic zone, but excessive input of nutrients from the watershed, or some high flow events, will result in loads that exceed the natural assimilative capacity of the stream. When this occurs, streams become a conduit for transporting N and P to downstream water bodies, such as the Mississippi River and the Gulf of Mexico.

Excessive sediment load degrades the benthic habitat of streams by clogging the spaces between larger streambed material (see Waters [1995] for a review of the effects of sediment on streams). These spaces, known as *interstices*, provide a critical habitat for aquatic insects and juvenile fish. Species that rely on interstitial spaces are quickly lost from a stream suffering from excessive sedimentation. The degree to which the interstices on the streambed are clogged can be determined with a measurement called *embeddedness*. Habitat assessments often estimate the degree of embeddedness in a stream to determine if the benthic habitat has been impaired by sedimentation.

Bank erosion not only contributes sediment to streams but also decreases the quality of fish habitat in a stream. Well-vegetated, overhanging banks provide important cover and feeding opportunities for fish. Bank erosion and sedimentation are frequent stressors to the fish communities in agricultural streams. Loss of riparian vegetation decreases the input of natural organic matter to streams, decreases shading, and can accelerate bank erosion. All of these factors contribute to an overall loss of species diversity within streams.

Many of the characteristics of streams, such as temperature, turbidity, and sediment size distribution, are dependent on the flow of water. As a result, changes to the natural *hydrology*, or patterns in discharge, act as a stressor to stream organisms. The life histories of many aquatic invertebrates and fish are closely tied to particular water temperature and flow conditions. Modifications to the hydrology of a watershed, such as channelization and tiling, change the flow conditions in the stream. Physically, a stream responds to hydrologic change by channel adjustments such as *downcutting*, filling, widening, narrowing, or pattern shift. Downcutting can lead to bank erosion, whereas filling will lead to the loss of channel capacity and flooding. Biologically, changes in hydrology can result in loss of aquatic species for which the stream no longer supports suitable flow and temperature conditions.

Streams that are degraded or impaired are typically suffering from multiple stressors, and it can be very difficult to isolate the impact of any one stressor. The activities that affect water quality tend also to affect physical habitat conditions, both of which contribute to the biological status of a stream. In the case of some stressors the origin may be in the upland areas of a watershed, whereas other stressors may originate within the channel itself. Stream ecologists often refer to the four dimensions of a stream: longitudinal (upstream to downstream), lateral (channel to upland), vertical (water column to benthic zone), and temporal (seasonality, or changes through time). Designing, implementing, and assessing programs to improve stream health should, therefore, consider all the interacting components of a stream and its watershed as an integrated, temporally dynamic system.

Stream Assessment Issues and Landform/ Sediment Associations

Midwestern streams, stream channels, and the hydrologic system that links surface and subsurface water across the valley landscape have been significantly altered by agricultural and suburban land use in the Midwest. Channel, bed, and bank conditions reflect a legacy of response to catchmentwide increases in runoff and sediment delivery and *reach*-scale adjustments to channel straightening. These legacy effects frustrate attempts to link channel and bank behavior at the reach and cross-section scales to present land use conditions. Assessment approaches that evaluate stream channels in a context of local bed and bank materials, reach-scale channel behavior, and inputs of water and sediment are most likely to provide information about the trajectory of change.

Bed, bank, vegetation, and watershed geologic materials have profound effects on channel response to shifts in hydrologic conditions. For example, erosion of deposits, such as glacial till, that contain sand and gravel (hydraulically controlled sediment) provides the coarse material necessary for downstream aggradation and the initial phase of channel recovery (Simon and Rinaldi 2000). The thick loess regions have a paucity of these materials, and as a result many channels in these areas are still undergoing dramatic incision and widening. In general, channels in areas with less loess, such as the loess-mantled drift plain and the recently glaciated plains, seem to be closer to attaining forms adjusted to present conditions (Simon and Rinaldi 2000). The resistance of bed and bank materials to fluvial erosion and mass wasting also varies across the region, within watersheds, and along a given reach. Quantifying and inventorying these properties will allow better assessment of reach-scale channel and bank dynamics by placing them in a catchment perspective.

Properties of soils and subsoil materials have profound effects on the fate of nutrients and synthetic compounds in infiltration water that recharges groundwater and contributes to stream baseflow (Carlyle and Hill 2001; DeVito et al. 2000; Kalkhoff et al. 2000; Schilling et al. 2009a; Van der Peijl and Verhoeven 2000). Regional alluvial stratigraphic sequences stand to serve as an important guide for assessing how and to what degree valley geologic materials and soils influence the hydrogeochemistry of stream water at cross-section, reach, and catchment scales. In their research on central Iowa, Schilling and colleagues (2009a) showed that DeForest Formation alluvium contained nutrient concentrations at levels known to negatively impact shallow groundwater resources and suggested that these materials may be a significant source of nutrient losses to streams. The regional occurrence of these relatively nutrient-rich deposits and the fact that properties of the formation's members vary suggest that documenting the occurrence of the deposits in future riparian zone studies may improve understanding of important nutrient sources and sinks in the valley landscape.

To better understand present conditions and to predict future trajectories of stream health assessment and monitoring, strategies must consider (1) differences in channel types and bank materials; (2) reach- and catchment-scale variations in bed, bank, and valley materials; (3) the nature of hydrologic linkages among runoff, infiltration, shallow groundwater, and channel flow; (4) how floodplain, bank, and channel bed materials and conditions may influence stream water quality; and (5) spatial and temporal variability of these properties and processes and their effects in a watershed. The characteristics of today's streams reflect response and adjustments to present conditions in the context of longer-term adjustments to major land use changes during the past century and a half. The challenge for assessing present impacts and for arriving at effective mitigation measures is to identify legacy effects, to place various scales of stream condition observations in a catchment perspective, and to recognize where a given stream is in the stability/instability spectrum.

Abbreviations and Acronyms

cm	centimeter
m	meter
N	nitrogen
P	phosphorus

Glossary

Baseflow. Stream discharge consisting of groundwater and other water storage zones.

Benthic zone. The interface between the stream bed and the water column.

Bioassessment. A method in which biological attributes are used to determine the health of a stream.

Denitrification. A biological process that removes nitrate from water.

Discharge. The volume of water flowing through a stream channel per unit of time.

Downcutting. Creating a deeper and contained channel.

Embeddedness. The degree to which the interstices in the stream bed are clogged with sediment.

Ephemeral streams. Streams that do not flow continuously throughout the year.

Eutrophication. Excess growth of algae and aquatic plants resulting from nutrient loading to streams.

Evapotranspiration. The loss of water from soil both by evaporation and by transpiration from growing plants.

Fluvial geomorphology. The interactions between landforms, stream channels, and sediment transport.

Hydrology. The patterns and processes involved with the distribution and movement of water across the landscape.

Hydrophobic. Condition that occurs when the rate of precipitation exceeds the rate at which water infiltrates the soil.

Interstices. The spaces between large substrate that provide critical habitat for aquatic insects and juvenile fish in streams.

Large woody debris. Logs with a minimum diameter of 10 cm (4 inches) and a minimum length of 1.83 m (6 feet) that protrude or lie within a stream channel.

Lentic. Nonflowing systems such as wetlands and lakes.

Lotic. Flowing systems such as streams and rivers.

Overland flow. Occurs when precipitation falls on soils that already are saturated with water or when the rate of precipitation exceeds the rate at which water infiltrates into the soil.

Perennial streams. Streams that flow continuously.

Primary producers. Photosynthetic organisms, mainly algae and aquatic plants in streams.

Reach. A straight portion of a stream or river.

Reference sites. Streams known or believed to be minimally degraded by human activities.

Riparian zone. The area immediately adjacent to the stream channel.

Soil reduction/oxidation (redox). A change in the oxidation state of soil that alters the biochemistry.

Stressors. Activities that cause stream degradation.

3 Influence of Time Lags on Land Management Change

Watershed Hydrology and Material Transport to Streams

Lag time is defined as "the amount of time between implementing a practice and observing a response" (Meals and Dressing 2006). Although the concept seems simple, embedded in it are several issues adding to the challenge of detecting water quality changes resulting from land management alterations. Lag time involves not only the time needed to produce an effect at a specific location, but also the time needed for the localized effect to be observed in the water resource of interest. Once the effect has been delivered, time is needed for the water resource to respond to the effect (Meals, Dressing, and Davenport 2010). Management changes implemented on a single field may result in water quality change in or close to the field relatively soon, but timing of water quality changes at a greater distance will be controlled by the manner of pollutant delivery and response time in the water body. Furthermore, the lag time for observing a response varies considerably depending on the parameter of interest; there is no "one size fits all" answer to the question of "How long will it take for my management change to make a difference?" Lag time issues may frustrate stakeholders who often demand rapid water quality improvement for their conservation investment of time and money.

Although lag times may vary among parameters, specific characteristics related to each provide some guidance for evaluating lag times for water quality response. In this section the authors explore how lag times of common agricultural pollutants vary.

Infiltration Capacity/Water Runoff

Land surface modifications have changed the soil's capacity to infiltrate rainfall. With adequate water infiltration, rainfall soaks into the ground like water into a sponge, decreasing or eliminating runoff. Infiltration rates are closely related to soil structure and organic matter content, although factors such as topography, soil texture, vegetation cover, and management systems are also important (Bharati et al. 2002). Infiltration rates of many upland agricultural soils have been substantially decreased because of severe erosion of organic matter-rich topsoils (Trimble 1983). Soil erosion and loss of infiltration capacity are still occurring today, but conservation practices such as terraces, decreased or no-tillage, cover crop, sod-based rotations, and introduction of perennials are serving to decrease the rate of upland soil loss. Despite implementation of practices that decrease runoff and increase landscape-scale infiltration of rainfall, increasing the infiltration capacity of the soil "sponge" requires improving soil structure and organic matter content (USDA–NRCS 1998).

Surface Water Runoff

Excessive runoff and subsequent agriculturally related soil erosion provided much of the impetus for today's conservation programs. Conservation practices designed to decrease runoff from cultivated fields are well understood and include installation of terraces, decreased or no-till cropping, strip cropping, conservation buffers and filter strips, and installation of wetlands or ponds. All of these practices are intended, in part, to increase surface roughness and water storage on the landscape, thereby slowing runoff and increasing the time for excess water to infiltrate. Hence, given the short duration of a typical rainfall event, the time needed to observe a change in runoff characteristics following installation of a runoff-control practice may be very short—on the order of minutes, hours, or days. For example, following terrace or pond installation, runoff downslope or downstream of the installed feature would decrease (assuming a proper design) in the timescale of the rainfall event.

Over the long term (decades), combining enough runoff control measures in a watershed has been shown to measurably affect watershed hydrology. Conservation practices implemented to decrease soil erosion during storm events increase infiltration, which could lead to more groundwater recharge and greater stream baseflow. In Wisconsin, a decrease in flood peaks and an increase in baseflow were observed because of improved land management and change in land cover (Gebert and Krug 1996). Similarly, in Iowa research has shown more precipitation routed to baseflow than storm runoff in the second half of the twentieth century, which is associated with less perennial cover and more intensive row crop production (Schilling and Libra 2003). In addition to conservation practices, increasing subsurface drainage, increasing row crop production, and channel incision also are responsible for the significant increase in baseflow discharge in Iowa. The trend of increasing baseflow since the 1940s extends across the upper Mississippi River basin (Zhang and Schilling 2006).

Despite conservation practice effectiveness in decreasing surface runoff at a local and regional scale, rainfall amount, timing, and intensity are ultimately the drivers of runoff. Runoff is part of the natural water cycle, and significant rainfall will produce saturated conditions and occasional flooding regardless of field conservation implementation. Unfortunately, conservation practices seldom are installed across the landscape for maximum runoff mitigation. Installation of a practice may decrease local-scale runoff downslope, but if other runoff-prone areas are not treated, detection of any improvement from the practice would be lost in the hydrograph signal dominated by untreated areas. The greater the concentration of practices installed in a basin, the more likely a change in runoff routing would be observed. Likewise, the more targeted the control measure to runoff-producing regions of a basin (i.e., steeper slopes), the greater the potential for measureable reduction in rainfall runoff to occur. Thus, the lag time to observe a reduction in runoff from a basin can vary by location and be dependent on effective placement and targeting. It should be viewed as encouraging that reductions in rainfall runoff can be observable at short time frames given the correct circumstances.

Sediment

Soil erosion and sediment deposition are natural processes of the earth-water cycle, but human activities have accelerated these processes in many agricultural regions (Montgomery 2007). Sediment loads in streams are largely derived from erosion of upland soils, collapse of stream banks, and resuspension of bed materials. Conservation practices during the last half-century have been primarily concerned with decreasing soil erosion at the plot or edge-of-field scale (Richardson, Bucks, and Sadler 2008). A growing body of evidence, however, suggests the source of sediment in streams may be shifting from uplands to near-channel sources (Nieber et al. 2009; Wilson et al. 2008). Overall, there have been few studies of sufficient scale and duration to assess the effects of conservation practices on watershed-scale sediment export (King et al. 2008).

Detecting changes in sediment export from watersheds due to conservation practice implementation is difficult for several reasons. First, discharge and sediment transport in watersheds is highly variable and may be exceptionally flashy. The majority of sediment is transported during intermittent storm events; in some cases one or two events per year may contribute the majority of the annual total sediment export from watersheds. Unless high-temporal resolution sampling is conducted during the events, the true sediment export from a watershed may be unknown.

Second, climate and historical sediment storage can greatly influence the sediment monitoring record. Climatic effects, including variable location and intensity of precipitation, can mask reductions in discharge and sediment loads for many years. Detecting changes against a backdrop of climate variability often requires a long-term record (Potter 1991). Sediment eroded from uplands is transported downslope and stored at the edge of fields and on floodplains as “postsettlement” deposition. Whereas some have used the distribution and age of the postsettlement sediment to prepare sediment budgets to estimate long-term changes in sediment transport at a watershed scale (Beach 1994; Trimble 1983), sediment storage also has been observed to introduce additional lag time in watersheds. Some estimates suggest that 8 to 25 years may be needed simply to flush sediment stored in the channel bottom (Marutani et al. 1999; Schilling and Wolter 2000). This would vary greatly by stream slope and discharge.

Finally, detecting changes in suspended sediment transport may be complicated by different sediment sources contributing to sediment loads. Channel-derived sediment may contribute 40 to 80% of the total sediment load in rivers (Sekely, Mulla, and Bauer 2002; Wilson et al. 2008).

Given the challenges in detecting changes in sediment transport, a monitoring time of decades may be needed to measure changes in sediment export from management practices (Figure 3.1). This does not mean, however, that changes in sediment export cannot be documented, though changes in small watersheds are more likely to be observed quickly. In Ohio, a 30-year record of daily suspended sediment was used to show sediment decreases because of successful implementation of conservation practices (Richards et al. 2008). Decreasing trends in total suspended solids from 1976 to 2003 compared to the previous 25–50 years also were noted in the Minnesota River and linked to conservation measures implemented in

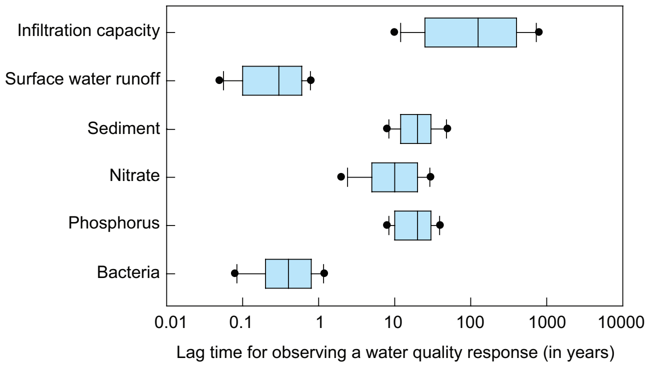

Figure 3.1. Hypothetical range of lag time responses for conservation practice effects on various water quality parameters.

the watershed (Johnson et al. 2009). In the Raccoon River in Iowa, suspended sediment concentrations have been decreasing despite an increasing trend in discharge, the product of which is a decrease in suspended sediment load (Jones and Schilling 2011). The long-term sediment record available for the Raccoon River suggests that conservation efforts to decrease sediment export from the highly agricultural watershed seem to be working (Figure 3.2). On a global scale, the majority of decreasing trends in suspended sediment loads is thought to be due to reservoir construction (Walling and Fang 2003).

Nitrate

Nitrate is a naturally occurring constituent found in surface water and groundwater. In agricultural regions, nitrate is primarily derived from nitrogen (N) fertilizers, livestock manure, and mineralization of soil N (organic matter). Nitrate is very soluble and readily leached from cropped field soils, moving as a dissolved constituent in shallow groundwater to streams. Groundwater discharge as baseflow and discharge of groundwater as artificial subsurface drainage provide the main source of nitrate to many midwestern rivers and streams (Hallberg 1987).

Because of the subsurface delivery of nitrate to streams, the rate of groundwater movement can be a controlling factor in the time lag between a change in management practice and change in nitrate concentrations in a stream. Depending on the aquifer, groundwater flow rates can be variable, ranging from less than 1 meter (m) per year in fine-textured glacial sediments to more than 100 m per day in sand. Groundwater flow rates in fractured limestone and karst may be greater still. Nitrate losses in watersheds are impacted by wet and dry years, with storage of N that occurs during a dry year becoming rapidly mobilized during a subsequent wet year (Lucey and Goolsby 1993). Hence, it is not surprising that nitrate concentrations exhibit long-term memory of up to two years in agricultural watersheds (Zhang and Schilling 2005).

Results from various monitoring and modeling studies suggest that the time needed for conservation practices to decrease nitrate concentrations in receiving streams and rivers is often on the order of years to decades (Figure 3.1). At a western Iowa deep loess site, more than 30 years was needed for changes in fertilizer applications at the watershed divide to travel to a nearby stream (Tomer and Burkart 2003). Likewise the time needed for groundwater to residence at a watershed scale may dictate the pace of nitrate concentration reductions measured at the watershed outlet. Schilling and Wolter (2007) estimated that the groundwater travel time was 10 years in one Iowa watershed and noted that unrealized stream nitrate reductions from land use changes had simply not reached the stream during the period of monitoring. In the Chesapeake Bay, time of travel estimates suggested that 50% reduction in baseflow nitrate concentrations would be observed about five years after all N applications were halted (Meals,

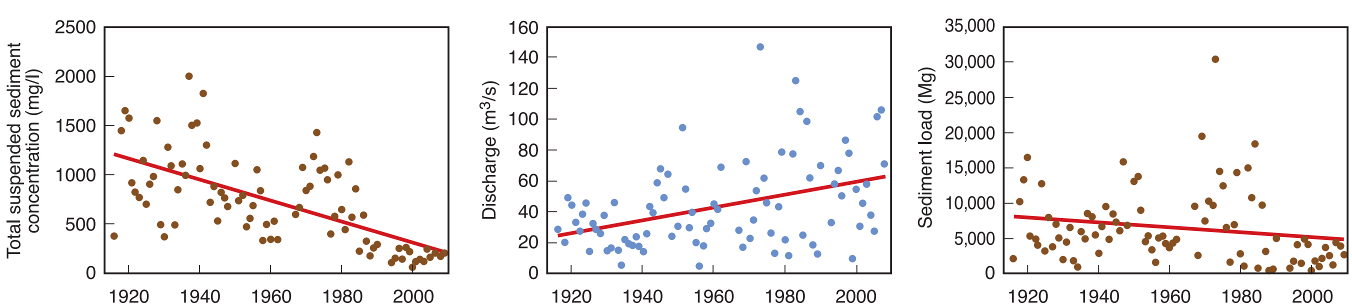

Figure 3.2. Suspended sediment concentrations are decreasing in the Raccoon River in west-central Iowa (left), but river discharge is increasing (middle). Sediment load (product of concentration and discharge) shows a significant decrease over time (right) (Jones and Schilling 2011). (Left: milligrams per liter; Middle: cubic meters per second; Right: megagrams)

Dressing, and Davenport 2010). This estimate is similar to the entire Mississippi River basin, where it has been suggested that reductions in anthropogenic inputs would be evident in from two to five years (McIsaac et al. 2002).

Subsurface tile drainage (Figure 3.3), although contributing to nitrate movement from the field and watershed, may actually accelerate the detection of nitrate management changes in some watersheds by lowering the travel time needed for soil drainage water to reach the stream network. Recent modeling suggested that increasing tile drainage density may decrease the groundwater travel time by approximately 50% (Jindal 2010).

Phosphorus

Phosphorus (P) is a naturally occurring plant nutrient typically found at low concentrations in soils and water. In agricultural regions, higher concentrations of P are primarily sourced from applications of fertilizer and manure. In Iowa, 54% of P inputs have been associated with fertilizer applications, whereas 45% of inputs have been from manure. The remaining 1% of P inputs to the state has been associated with human and industrial activities (Libra, Wolter, and Langel 2004). Des Moines lobe till typically contains higher P than alluvial or other parent material. Phosphorus can be bound in soils and transported during the erosion process or it can be dissolved in water and transported with water export. With both natural and anthropogenic sources, and variable mode of transport in total or dissolved forms, detecting changes in P transport from management changes is problematic on several fronts.

Sediment-bound P will typically follow the same pattern of detection as sediment, with highly variable concentrations during water runoff events and elevated streamflow conditions. Also, like sediment storage in a watershed, there can be a substantial legacy of P accumulation in soils due to historical agricultural practices. Soils in some agricultural areas already contain sufficient P for years of crop growth without additional fertilizer amendments (Mallarino, A., B. Hill, and K. Culp. 2010. Personal communication). Sediment accumulation along footslopes and drainageways in watersheds may provide a long-term source of high P concentrations in watersheds. High concentrations of P in soils provide a source of high P concentrations in surface runoff water during erosion events. Under the right subsurface conditions, sediment-bound P may be released to groundwater as dissolved P and subsequently discharged to streams with baseflow and tile drainage. Gentry and colleagues (2007), citing research in Illinois, stated that overland flow was the dominant P source to streams, but tile drains were an important P transport mechanism even in years with below-average precipitation.

Figure 3.3. Typical tile drain outlet discharging to a headwater stream or ditch. Tile drainage is common in the upper Midwest, where poorly drained soils and low topographic relief necessitate artificial drainage of wetlands and mesic prairie when the land is farmed. (Photo courtesy of Todd Royer.)

A lag time of decades may be needed to measure changes in P export from management practices at a watershed scale (Figure 3.1). On a plot scale, according to research, changing the P balance between application and crop removal may affect P concentrations in soils within a decade (Sharpley et al. 2007); however, time lags of 10 to 30 years before changes in management can decrease elevated soil P concentrations to background levels have been reported in other studies (Zhang et al. 2004). Beyond the plot scale, detecting short-term improvements in P transport in watersheds is hampered by widespread P storage and availability. Improvements made at a field scale will inevitably be lost at watershed scale against the background of other P sources. Although reductions in soil P and dissolved P may be evident within a decade or two, concentrations often remain higher than allowable environmental thresholds for an extended period of time (Meals, Dressing, and Davenport 2010).

Bacteria

There are many different bacteria present in the natural environment, but considerable water quality focus has been on coliform bacteria such as *Escherichia coli* (*E. coli*). Bacteria such as *E. coli* are naturally found in humans and other warm-blooded animals, but exposure to high concentrations can result in waterborne disease. In agricultural regions, where the overwhelming source of bacteria is animal manure from livestock, other sources include contributions from septic systems and wildlife. For example, in the 9,389 square kilometer (3,625 square mile) Raccoon River watershed in west-central Iowa, manure from hogs, cattle, and poultry was the source of 99.9% of the total amount of *E. coli* generated in the watershed (Schilling et al. 2009b). Manure from confined livestock is typically land applied to cropland as fertilizer. Bacteria are then commonly delivered to streams from rainfall runoff or snowmelt events via overland flow or, in some instances, discharged through subsurface drainage systems. In pasture systems, bacteria may be delivered to streams through runoff or directly deposited when livestock spend time in or near streams. Bacteria from septic systems, though composing a minor component of the overall bacteria population in the watershed, may be a larger problem than the total numbers would suggest. Failing septic systems may deliver bacteria and human contagion to surface waters and cause local water quality concerns.

The lag time needed to measure changes in bacteria transport from management practices is relatively short—on the order of days to months (Figure 3.1). Because elevated bacteria concentrations are primarily associated with rainfall runoff events, efforts to decrease surface water runoff from bacteria-prone areas would also serve to decrease bacteria transport. For example, installing a pond or terrace downstream of a pasture or manured field would not only decrease discharge, the feature would also capture and settle bacteria transported with runoff. Targeted practices that decrease the direct delivery of bacteria into surface runoff or streams may result in a rapid reduction in bacteria load. Keeping livestock from entering the stream and riparian zone or eliminating direct septic discharge into a stream would result in immediate water quality improvement. Applying manure within agronomic limits and avoiding applications during rainy periods would further result in reducing bacteria delivered to streams. Finally, unlike nutrients such as N and P, bacteria generally do not persist for long periods in the environment, with a typical half-life ranging from hours to several days, although extended bacteria survival rates in sediments have been documented (Byappanahalli et al. 2006). It is not surprising that bacteria concentrations in surface water exhibit a short-term memory of only four days, essentially the duration of a rainfall runoff event and time of significant bacteria decay (Schilling et al. 2009b).

Studies have documented the effectiveness of management practices in decreasing bacteria runoff from unrestricted livestock grazing (e.g., Line and Jennings 2002; Meals 2001). It should be noted, however, that these studies have typically required a sampling program specifically designed to detect the change, including event-based sampling and before/after monitoring designs. So, although it is possible to detect changes in bacteria transport within a short time frame, actually measuring these changes is difficult because they are associated with measuring rapid changes in flow and concentration during a storm event.

Lag Time Relationships among Parameters

Although it is tempting to view the hydrologic and water quality parameters as individual concerns to address, it is more common that watershed managers are faced with addressing multiple issues simultaneously. If streams suffer from excessive nutrient enrichment, conservation practices may be focused on reducing N, P, and sediment losses to streams concurrently. Unfortunately, efforts to decrease losses of one parameter may exacerbate losses for another parameter. This is particularly true when parameters of interest are delivered with different components of the stream hydrograph, as when one pollutant is lost through surface water runoff while another is exported with baseflow.

For example, using terraces to decrease surface water runoff and lower sediment and P concentrations in a watershed may have an unintended consequence of increasing water infiltration behind the terrace, thereby increasing groundwater recharge and nitrate leaching. In one example, implementation of tile-drained terraces in northeast Iowa lowered stream turbidity levels but likely increased the efficiency of routing leached nitrate to the stream system (Gassman et al. 2010). In poorly drained areas, subsurface tile drainage may cause soils to dry and become more receptive to infiltrating rainfall, but tile drainage is also a major source of nitrate loads in streams. Because nitrate is primarily transported in tile drainage and shallow groundwater discharge, efforts to lower pollutant losses from surface water runoff (e.g., sediment, P, bacteria) may result in increasing nitrate losses. These competing pollutant delivery mechanisms may introduce additional lag time in monitoring conservation effectiveness.

Buffering Capacity of Streams

Streams are dynamic ecosystems with a high degree of temporal and spatial variability in nearly all attributes. Streams are also subject to random events, such as droughts and floods. Natural variation and random events can mask or delay improvements in stream health. As a result, documenting responses in water quality or overall stream health following a particular management activity can be challenging because the physical, chemical, and biological characteristics of a stream are driven by multiple and interacting factors, such as climate, land use, geology, and biogeography. If the concept of water quality is extended to include biotic integrity as it is by many states, then certain aspects of upland best management practices may have less influence than the many in-stream factors influencing biotic integrity, such as the role of *large woody debris*. Trends or changes in stream health must be separated from natural variation before such changes can be attributed to a management or restoration activity, and this requires rigorous and long-term monitoring (Kondolf 1995; Magner and Brooks 2008).

Because it is often difficult to observe improvements in water quality and stream health, the question of whether or not streams have a natural resistance (or *buffering*) that slows responses to management activities warrants consideration. This section addresses the physical, chemical, and biological factors within the stream channel itself that may lead to buffering, adding to the lag time between management activities and responses in water quality and stream health. Particular emphasis is given to identifying, when possible, the mechanisms that may be responsible for time lags between management activities and in-stream responses.

Physical Factors

The erodibility of channel materials is a primary determinant of channel response to watershed management. If the streambed is composed of bedrock, boulder, or cobbles, stream response to reductions of suspended sediment will be more rapid. In regions where channels flow across finer-textured glacial and alluvial materials, the chemical and biological response will be slower because of the availability of more erodible bed sediment to fuel turbidity.

Impaired stream health usually is the result of multiple stressors. Elevated nutrient concentrations and degraded physical habitat generally occur together in streams, as shown in research done in Illinois by Heatherly and colleagues (2007). Long-term habitat degradation is difficult to repair but necessary for biological recovery (e.g., Moerke and Lamberti 2003; Moerke et al. 2004). Sediment load reduction, lowering water temperatures, and restoring woody debris and other forms of cover are long-term and difficult challenges, but without these habitat improvements, efforts to decrease nutrient and pesticide concentrations may have little impact on the biotic integrity of streams. The authors of this paper stress the following:

> Degraded physical habitat within a stream can thus create an impediment that prevents improvement in overall stream health following management activities aimed at improving water quality. Channel entrenchment also lowers a stream's propensity for water quality improvement by decreasing flood plain connectivity. If flood frequency is decreased by channel incision, less sediment is deposited on the floodplain, degrading water quality in the stream.

Chemical Factors

Certain compounds, such as ammonium, P, and some pesticides, can adsorb to sediment particles. This provides a mechanism for transporting these compounds from fields to streams and also allows for the compounds to potentially accumulate in the benthic zone of streams. Phosphorus bound to stream sediments can have an influence on the concentration of P in the water (e.g., Haggard, Stanley, and Hyler 1999; McDowell, Sharpley, and Folmar 2003). It is possible that P in stream sediments could maintain elevated water column concentrations even if inputs of P from the watershed were decreased. McDaniel, David, and Royer (2009) investigated this possibility for streams in Illinois and concluded that P release from stream sediments likely occurred but was minor compared to the watershed inputs.

Sediments clearly have the potential to buffer changes in water column P concentrations, but for how long and to what extent remain open questions. Furthermore, the strength of the buffering will certainly vary among streams in relation to the particle size, type, and chemistry of the sediments and the amount of microbial activity within the sediments (Haggard, Smith, and Brye 2007; McDaniel, David, and Royer 2009). Bottom-feeding fish such as the common carp (*Cyprinus carpio*) can suspend sediments and sediment-bound P through their foraging activities, and in lakes this can cause increased P availability and algal biomass (e.g., Søndergaard, Jensen, and Peppesen 2003).

Tile drains, however, also provide a mechanism for seasonal inputs of agri-chemicals to streams, including N, P (Gentry et al. 2007), and pesticides (David et al. 2003; Gentry et al. 2000). Tile drains effectively bypass riparian buffers (including soils of the buffers) and deliver water and associated nutrients directly to waterways. Streams receiving water and nutrients through tile drainage may appear buffered against changes in water quality following land management practices targeted at overland flow (Osborne and Kovacic 1993).

Seasonal inputs of pesticides may affect critical life stages of aquatic organisms or cause changes in food web structure that alter aquatic communities (Relyea and Diecks 2008). The extent to which seasonal inputs of agri-chemicals slow biotic recovery of impaired streams is unknown, but in theory this could cause a buffering effect between management activities and stream health.

Biological Factors

A wide range of aquatic organisms exists in streams, some of which are capable of dispersing long distances (fish) whereas others have limited dispersal abilities (snails, mussels). Improvements in stream habitat and water quality will not be reflected in the biotic communities until organisms can disperse into the improved stream reaches and establish viable populations. The time required for this to occur will depend on many factors, such as the types of organisms involved, the distances to be traveled, the size of the source population, impediments to natural dispersal (e.g., dams or culverts), and whether or not there is human intervention (e.g., fish stocking).

Most aquatic invertebrates have limited dispersal capabilities (Bilton, Freeland, and Okamura 2001), and even aerial adults may travel only a few tens of meters from their *nascent* stream, with only occasional long-distance dispersal (Petersen, Hildrew, and Ormerod 2004). If the source population and the new habitat are near each other, dispersal of aerial adults can provide a source of colonists for the new habitat (Masters et al. 2007). Even if adults reach the previously uninhabited stream, an appropriate habitat for oviposition (egg laying) must exist for successful colonization (Blakely et al. 2006). The dispersal of aquatic invertebrates, particularly by aerial adults, is a relatively unstudied aspect of stream ecology but certainly a mechanism that could impact time lag between management activities and improvement in the biotic community of a stream.

Other factors that may delay or prevent recovery of biotic communities are the presence of exotic species that compete with native species and the availability of appropriate food resources, including specific prey species. Invertebrates that feed on the terrestrial leaf litter that falls into streams may never return unless the riparian zone is planted with trees. Excluding terrestrial leaf litter can significantly affect the ecology of streams (Wallace et al. 1997), and full recovery of biotic communities may require development of a forested or wooded (multispecies) riparian zone—a process that could take several years. Many predators are specialized to feed on a particular type of prey, and such predators will not return to a stream until the appropriate prey species have established sufficiently large populations. Overall, the recovery of the biotic community in a stream is highly unpredictable and susceptible to random events, such as scouring floods or droughts that may decrease populations and reset the recovery process.

Characteristics Responsible for Stream Response to Land Management Changes

Streams are a reflection of the watersheds in which they occur. The variation among streams in responsiveness to management activities arises, in part, from variation among watersheds in climatic, geologic, and geomorphic characteristics. Also important is the extent to which the stream (or its watershed) has been degraded and the number of stressors responsible for the degradation. Improving stream health in an efficient manner requires knowledge about the particular factors responsible for the *impairment*; otherwise management activities may miss the mark and result in little or no improvement. In many cases, poor stream quality may be due to several factors acting independently or the interaction between such factors. Teasing apart the role of multiple interacting factors can be challenging and may not be possible outside of experimental approaches (e.g., Matthaei, Piggot, and Townsend 2010).

When considering factors that might cause variable responses among streams in agricultural landscapes, the following are likely to be important:

1. **The extent to which historic land management activities have occurred in the watershed**. Watershed areas are unique in their land management history, and these historic practices have left a legacy of soil and water conditions on the landscape. Because the starting point of impairments varies across regions, stream responses to similar land management practices will vary.
2. **The extent to which the *geomorphology* of a stream channel has been altered by dredging, straightening, bank and bed armoring, etc., and the degree to which channel adjustment has occurred**. With greater geomorphic alteration, it is likely the time lag between management activities and stream response will increase. These activities also disrupt the benthic sediments of a stream and slow the recovery of aquatic communities.
3. **The extent to which the hydrologic pathways to a stream are influenced by baseflow, overland flow, and interflow tile drainage**. Each of these pathways has unique influences on the hydrology and transport of materials to streams, and the relative contribution of each will impact how (or if) the water chemistry in a stream responds to land management activities.
4. **The extent of each management practice within the watershed**. How much of a watershed does a particular management practice need to cover in order to affect conditions in the watershed streams? This question cannot be answered in most situations, and it must be assumed that "some is better than none" when implementing land management activities. This assumption simply may not be valid in many cases. At a watershed scale, there may be a threshold response such that increasing adoption of a management practice is not reflected in stream health until some minimum, or threshold, amount of the watershed is under that practice. Other types of nonlinear response are also possible (Allan 2004). Effective targeting of practices to locations producing a disproportionate share of the impairment may result in more cost-effective water quality improvement than dispersed practices across a watershed (Walter et al. 2007).
5. **The spatial arrangement of management practices in relation to the location of a stream and its riparian zone**. Assessment of stream habitat, fish, and macroinvertebrate communities in relation to land use has revealed that spatial scale and spatial arrangement are critical factors that determine how streams respond to the impacts of various land use (Allan, Erickson, and Fay 1997; Roth, Allan, and Erickson 1996; Sponseller, Benfield, and Valett 2001). For example, the integrity of macroinvertebrate communities may be relatively insensitive to land use changes in the upland areas but highly sensitive to changes within a local buffer near a stream (Sponseller, Benfield, and Valett 2001). The effectiveness of a land management practice may vary considerably depending on where it is implemented in relation to the stream. Even extensive implementation of a practice may not elicit a strong response in stream health if the practice is not targeted at the appropriate spatial scale or the appropriate location within the watershed. This could give the appearance of the stream being buffered against the management practice.
6. **The dynamics of ecosystem recovery.** Long-term stress on an aquatic ecosystem can cause significant changes in the structure and function of that system. When the stressor is removed or decreased, the response (or recovery) of the ecosystem often is not a simple return to the prestressor condition (Magner and Brooks 2008). Instead, the ecosystem may have transitioned to a new stable state that will persist even in the absence of the original stressor (Scheffer et al. 2001). In streams that have been severely altered by channelization, dredging, sedimentation, or nutrient loading, the implementation of land management practices may never return the stream to a prestressor condition (Lake, Bond, and Reich 2007; Sarr 2002) (Figure 3.4). In most cases it is not possible to predict how an ecosystem, such as a stream or river, will respond when a stressor is removed. It is clear, however, that ecosystems can have multiple stable states that may confound efforts to restore a system to a particular ecological endpoint (Scheffer et al. 2001).

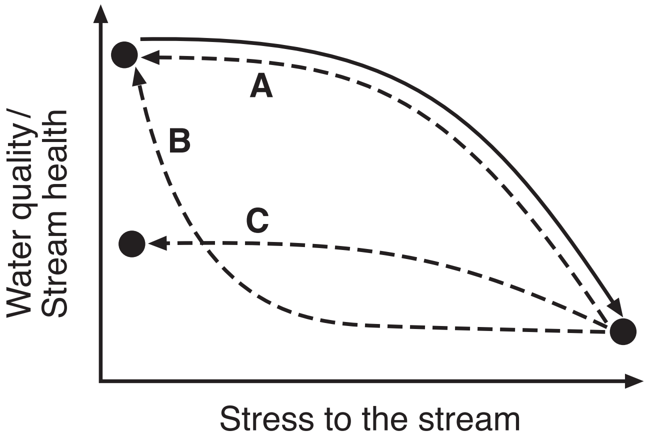

Figure 3.4. Effect of stress on water quality and stream health (adapted from Lake, Bond, and Reich 2007; Sarr 2002). Solid line indicates degradation of water quality and stream health because of stress on the ecosystem. Dashed lines indicate various trajectories of recovery following management actions to decrease stress to the system. Trajectories A and B result in water quality and stream health similar to the prestress condition, although trajectory B displays a time lag and threshold response. Trajectory C shows some recovery, but the system never returns to the prestress condition.

Abbreviations and Acronyms

l	liter
m	meter
Mg	megagram
mg	milligram
N	nitrogen
P	phosphorus
s	second

Glossary

Buffering. A stream characteristic's resistance to change.

Geomorphology. The science that deals with the relief features of the earth or another celestial body and seeks a genetic interpretation of them.

Impairment. The physical, chemical, or biological consequence of an action that results in a stream not attaining water quality standards necessary to meet its designated beneficial use.

Large woody debris. Logs with a minimum diameter of 10 centimeters (4 inches) and a minimum length of 1.83 m (6 feet) that protrude or lie within a stream channel.

Nascent. Coming or having recently come into existence.

4 Alteration and Restoration of Stream Health

Introduction

Despite implementation of numerous conservation activities in the Minnesota River basin (i.e., the Conservation Reserve Enhancement Program and the Wetland Reserve Program), reductions to sediment and nutrient load have not been observed through current monitoring strategies. This demonstrates how standard stream water quality parameters can be, at least in a relatively short time period, an insensitive indicator of the effects of land management change, particularly in larger rivers, which can be slow to respond to land use change in their basin. Smaller watersheds, composed of first- and second-order streams, will respond more quickly to watershed management, as evidenced by a wetland restoration and perennial planting program on two small agricultural watersheds in Martin County, Minnesota (Lenhart 2008), on a tributary within the Minnesota River basin.

Human Causes of Stream Degradation

Stream Characteristics before Anthropogenic Impact

Details about the conditions of the region's streams before anthropogenic impact are virtually unknown. Impacts from logging, snag removal, and grazing began in the early to mid-nineteenth century across most of Iowa and the Midwest, long before systematic measurements of channel dimensions and stream discharge were undertaken. Anecdotal accounts generally mention clear water in many of the region's streams (Mutel 2010 and references therein), and the threatened state of many fish, invertebrate, and insect species in today's lotic environments suggests general degradation in stream quality. Channel straightening, drainage ditch construction, and artificial drainage that began in the late nineteenth century and continue today have altered nearly all streams and shallow groundwater connections among streams, floodplains, and valley margins in the region.

As mentioned earlier in this paper, the stratigraphic record of the region's valleys indicates that the behavior of streams changed during the past in response to climate, vegetation, and sediment inputs. The physical, chemical, and biological characteristics of streams at any point in time reflect attempts of those systems to adjust to diurnal, seasonal, and longer-term inputs of water, sediment, organic matter, nutrients, and disturbance related to environmental conditions (in the broadest sense of the term "environmental"). These inputs always have varied, meaning that stream properties also always have varied. What sets modern stream systems apart from their pre-impact counterparts is that the extent and rate of anthropogenic alterations of the midwestern landscape by agricultural and urban land use have resulted in unprecedented systemwide instability.

Stream channels destabilized by various disturbances systematically pass through a sequence of channel forms with time (Schumm, Harvey, and Watson 1984; Simon 1989). Stream channels across the Midwest responded in a similar fashion to alterations in runoff, sediment delivery, and channelization that occurred during the nineteenth and twentieth centuries (Baker et al. 1993; Knox 1977; Simon and Rinaldi 2006). Channel straightening, construction of drainage ditches, and increases in runoff and peak discharge that occurred during the shift to agricultural land use destabilized the region's stream channels and initiated a complex sequence of channel responses that has continued to the present. Early in the period, channels were not able to adjust to increased runoff with larger and more frequent flood flows, and alluvium accumulated rapidly on floodplains. With or without subsequent straightening, channels soon responded to these more erosive flows (greater stream power) by rapid bed degradation and channel widening that destabilized channel banks (Figures 4.1 and 4.2).

In the areas with thick to moderately thick loess, these responses led to channel widening by mass-wasting processes (Figure 4.1A). In areas with thinner or no loess, channels did not incise as deeply and responded to the larger and more erosive peak flows through widening and increased meander growth

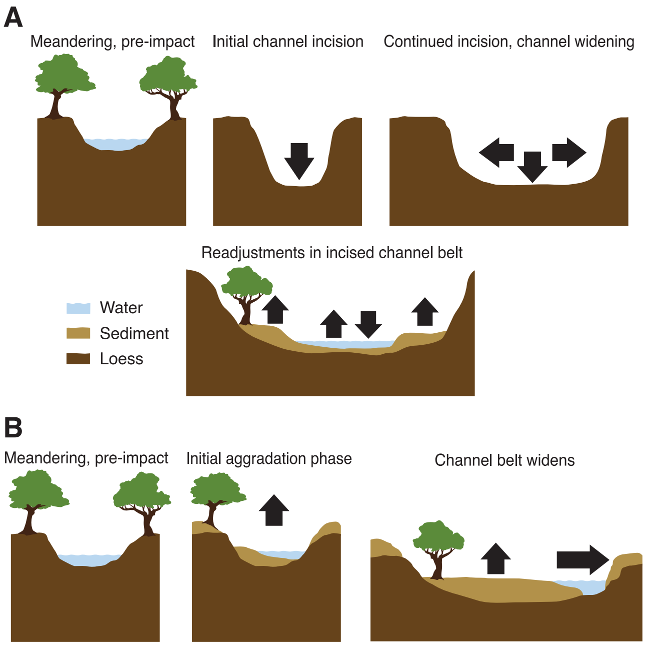

Figure 4.1. Stages of channel erosion in midwestern stream after historic land use changes (modified by Art Bettis from Paul and Meyer [2001], Simon [1989], and Simon and Rinaldi [2006]).

Figure 4.2. Lack of a riparian buffer and row cropping to the stream edge have led to stream bank failure and massive erosion along this stream in northwestern Indiana. Buffers of riparian vegetation stabilize stream banks and decrease sediment inputs to streams. (Photo courtesy of Todd Royer.)

rates, thereby widening the meanderbelt (Figure 4.1B). Continued widening of channels in both situations eventually produced relatively wide channel belts incised below the presettlement floodplain. These new channel belts contained most floods and significantly decreased overbank sedimentation. As degradation moved into upper reaches of the drainage system, sediments began to aggrade in the widened channel belts downstream. Similar, though less dramatic, adjustments involving aggradation, incision, and changes in channel sinuosity and sediment movement through the drainage network continue to occur within this historic floodplain.

These changes in the region's stream channels during the past 100 to 150 years have had significant impacts on the physical characteristics, quality, and function of streams. Through the use of stratigraphic data, historical accounts, and information from the least-altered streams in today's landscape, generalized conditions of the region's streams before the historic channel changes outlined in this section can be estimated. These estimates provide a useful perspective on the kinds of changes the streams have experienced and the potential instabilities in present streams related to legacy effects (Foster et al. 2003).

Hydrologic Regime

Before European settlement in the Midwest in the 1800s there was significantly less runoff, streamflow was less flashy, and flood recession curves likely were longer. Both small and large magnitude floods have increased, with the greatest change to the smaller floods. Stream power was generally lower than in today's stream channels, and *large woody debris* significantly dissipated the energy of stream flow (Cordova et al. 2007).

Characteristics of Channels and Banks

Channels were freely meandering (except where constrained by bedrock outcrops), significantly more sinuous, and narrower before anthropogenic disturbance (Knox 1999; Simon and Rinaldi 2000, 2006). Incised channels were common only in the thick loess region adjacent to the Missouri River Valley, and those channels were not incised as deeply nor did the incised channel network extend as far into headwater areas as in the modern drainage network (Bettis and Mandel 2002). Stream profiles were in general less steep and smoother than their modern counterparts. Less silt and clay were present on the streambed, more interstitial space was present in bed materials, and coarse-grained bars (point bars and longitudinal bars) were more common. Large woody debris (tree trunks and branches) that was common in channels passing through forested riparian corridors made significant contributions to erosional resistance of channels and produced habitat variation not common in most modern streams in the region (Langendoen and Simon 2000).

Stream banks were lower, more often vegetated, and significantly more stable before anthropogenic disturbance. Materials exposed in cut banks on the outside bend of meanders and along incised stream reaches were more resistant than the postsettlement deposits that are commonly exposed in today's cut banks. The lower predisturbance stream banks were less susceptible to toppling and seepage erosion than are steeper modern banks (Lutenegger, Lamb, and Bettis 1981; Wilson et al. 2007). Root systems of the riparian forests found along many predisturbance channels also probably contributed to bank stability.

Physical and Chemical Characteristics of Stream Waters

Important characteristics of past stream water such as its temperature, chemistry, and source (runoff vs. groundwater) are not recorded in alluvial deposits, but inferences about them can be made by observations of the few relatively unaltered or moderately altered reaches of streams that still remain (see Magner 2001). Surface runoff from adjacent slopes, floodplain surfaces, and impermeable surfaces was significantly less than in modern streams. Because of this, predisturbance streams were less turbid, had significantly lower nutrient loads (especially sediment-adsorbed phosphorus [P]), and did not contain the pesticides, pesticide degradates, pharmaceuticals, volatile organics, and other manufactured compounds that occur in midwestern streams today (Henley et al. 2000; Kalkhoff et al. 2000; Kolpin et al. 2002). Because of lower nutrient loads, autotropic biomass and production were lower in streams, filamentous algae were less abundant, litter breakdown rates were lower, and dissolved oxygen content was higher (Carpenter et al. 1998; Niyogi, Simon, and Townsend 2003). The proportion of the nitrate-nitrogen (N) load removed by sediment denitrification was probably higher than in modern impacted streams (Inwood, Tank, and Bernot 2005). Because riparian trees and shrubs were common on unaltered floodplains, streams were generally more shaded and experienced less diurnal fluctuation in water temperature, and a large proportion of the organic matter on the streambed was derived from riparian trees and shrubs.

Shallow Groundwater/Stream Interactions

The majority of pre-alteration streams had lower baseflow than their modern counterparts. The water table across the floodplain was likely higher, especially near the channel area. The hydrochemistry of streams, riparian corridor, floodplain, and valley slopes was connected by shallow groundwater flow. This had significant effects on the rate and timing of nutrient delivery to streams (Schilling et al. 2009a; Van der Peijl and Verhoeven 2000). The stream-groundwater exchange zone (*hyporheic zone*) of sand- and gravel-bed streams was not clogged with fine sediment and maintained aerobic conditions that supported a diverse fauna and allowed the zone to effectively filter water that passed through it into alluvial aquifers (Hancock 2002).

Direct Physical Modifications to Streams

The physical characteristics of many streams in agricultural watersheds have been directly modified by human actions ranging from channelization for drainage (Figure 4.3) to construction and reconstruction of bridges, culverts, and dams for various water management purposes. Nowhere else in the United States is the extent of land drainage more evident than in the upper Midwest, where rainfall and the broad expanses of flat terrain in such states as Indiana, Illinois, and Iowa led to the extensive ditching and draining for agriculture during the late nineteenth and early twentieth centuries (Pavelis 1987; Rhoads and Herricks 1996; Winsor 1975; Zucker and Brown 1998). Stream channelization (i.e., stream straightening, realignment, relocation, and ditching) was such a common practice by private entities, states, and the federal government that by 1971 its use came under congressional investigation, resulting in more than 3,200 pages of Senate and House of Representatives

Figure 4.3. Typical headwater stream or ditch in the glaciated, agricultural landscape of the upper Midwest. Such streams and ditches are periodically dredged to ensure drainage of the landscape during wet periods. (Photo courtesy of Todd Royer.)

testimony, comments, and documentation (U.S. Congress 1971a, 1971b). As a result of these hearings, federal support for large channelization projects has all but disappeared due in part to the recognition of the often unavoidable and severe environmental and downstream hydrological impacts.

Yet these practices continue at a small scale, and legacy channelization projects have left some areas of the Midwest with as much as 80 to 100% of their headwater streams channelized (Mattingly, Herricks, and Johnston 1993). More than one-third of streams in such states as Idaho, Illinois, Kentucky, Montana, and Ohio have been experiencing some sort of channel alterations, while many states have seen an actual loss in stream miles (Swift 1984). Iowa has lost between 1,000 and 3,000 stream miles (Bulkley et al. 1976), and nationwide more than two-thirds of all woody riparian ecosystems have been converted to some other land use (Swift 1984).

Channelization of Streams and Construction of Ditches

Channelization is often done for agricultural drainage or to make space for the addition of less flood-prone farmland. Ditch construction is the creation of a straight, trapezoidal channel by dredging where a stream did not previously exist. Both practices are typically done to minimize crop damage from overbank flooding and provide drainage of excess water away from the farmed landscape. Streams are also frequently channelized to place them at perpendicular angles under bridges at road crossings to prevent damage to bridge and road infrastructure. In the semi-arid Great Plains, ditches or canals were often created to divert water from streams for irrigation of crops, reservoir construction, or cattle watering holes (Wohl 2001).

Drainage practices, such as channel dredging or cutting riparian vegetation, may lead to acute habitat damage and limit biological recovery. Dredging can mobilize sediment-bound P within the channel and provide a pulse of this nutrient to the stream (Smith et al. 2006). Dredging also is a major disturbance to the benthic habitat in the stream that can slow or prevent recovery of the biological community. The long-term impact from dredging and channelization is the removal of many habitat features within a stream, such as meanders, overhanging banks, riffles and pools, and woody debris (Rhoads and Herricks 1996; Urban and Rhoads 2003). Streams that have been deepened and straightened lack these critical habitat features, and consequently such streams also lack the aquatic organisms that rely on such habitat. For example, channelization decreased geomorphic and hydraulic complexity in streams in central Illinois with corresponding reductions in the diversity and quality of the fish communities (Rhoads, Schwartz, and Porter 2003). Channelized streams may show little biological improvement following management activities that decrease nutrient, sediment, or pesticide inputs to streams if no effort is made to improve physical habitat and restore important habitat derived from geomorphic features of the channel.

The lessened geomorphic and hydraulic complexity of channelized streams also impacts the ability of microorganisms in the benthic sediments to process N. A meandering reach of a stream in central Illinois removed significantly more nitrate than did a channelized reach of the same stream (Opdyke, David, and Rhoads 2006). Powell and Bouchard (2010) in Ohio as well as Christner and colleagues (2004) and Magner and colleagues (submitted) in the Midwest examined streams, and/or drainage ditches, that had not been maintained but were instead allowed to develop benches within the main channel (referred to as two-stage ditches). The two-stage ditches supported greater rates of denitrification than did ditches that had been maintained (dredged) within the past two years (Kramer, G. 2011. M.S. thesis in preparation). The realization that channel complexity is an important feature of healthy streams has led to the examination of two-stage ditches as means of increasing nutrient retention, decreasing overbank flooding, and improving overall stream and riparian habitat for headwater streams and ditches in agricultural regions (e.g., Magner et al. 2010; Powell et al. 2007a, 2007b). As with the recovery of fish and invertebrate communities, the recovery of in-stream nutrient processing may require the restoration of critical geomorphologic and hydrologic features that are lost when streams are channelized.

Ditches have a trapezoidal design that has traditionally been thought to minimize maintenance costs by decreasing the bank angle to 45 degrees for stability. This creates uniform channel width and depth, causing a reduction of the depth variability typically found in pool-riffle streams in low-gradient environments. Loss of habitat heterogeneity has negative impacts on the biotic diversity and ecological integrity of fish and invertebrate communities. Minnesota and Ohio have invested in the index of biotic integrity (IBI) as the primary means of assessing stream health. The Clean Water Act of 1972 specifically called out biological integrity, and the IBI represents a direct measure of the biology.

Biological integrity is influenced by multiple factors, but typically in the Midwest habitat is an important driver of IBI. For example, in the intensive row crop area of west-central Minnesota the habitat changed in Judicial Ditch #8 (JD#8) as the channel adjusted to overwidening by the county highway department. Coarse-grained sediment aggraded in the channel because of inadequate stream power to move the sand and gravel contained in the geologic substrate, causing JD#8 to overwiden. Subsequently, a narrow low-flow channel formed as water carved through the deposited sand and gravel. The low-flow channel also developed some pools because large portions of cohesive upper-bank material fell into the channel; the cohesive silts and clays presented a resistant soil mass, forcing downward scour of the weaker noncohesive sand. After several years of natural forces working on the ditch bed and banks, a more diverse set of habitat features formed throughout the channel, resulting in the second-best fish IBI score in the Minnesota River basin in 2002 (Christner et al. 2004).

In some agricultural regions of the United States, stream length has been increased by construction of canals or ditches, adding to the contributing watershed area of a stream (Quade et al. 1980). Particularly in glaciated regions with their poorly integrated drainage network, drainage ditches have expanded the contributing drainage areas of smaller watersheds (Ward and Trimble 2004). An increase in contributing watershed area increases flow to streams, often leading to channel instability and larger suspended sediment loads (Kuehner 2004; Lenhart 2008). Most of the drainage ditch network was constructed in the late 1800s to the early 1900s in the Midwest, so the greatest rates of channel adjustment occurred decades ago. More recent expansion of subsurface drainage, however, has changed the timing and volume of water delivery from the field to the channel. More rapid delivery of water along with shallower groundwater has resulted in ditch-channel degradation (Magner, Payne, and Steffen 2004; Magner et al. submitted; Magner et al. 2010).

At the same time, there has been loss of headwater channels (first- to second-order streams) in agricultural watersheds to expand farmable areas and make fields level for crops. Sloughs (wide, shallow meandering vegetated swales) and other types of first-order streams were often graded and their drainage routed into ditches or subsurface tiles in low relief, glaciated landscapes of the upper midwestern United States (Lenhart 2008). West of the prairie-forest boundary (Figure 2.1; areas A, B, and the western part of D), drier conditions result in more ephemeral streams.

Channelization of natural streams alters the sediment transport regime by increasing the channel slope through shortening its distance, initially increasing its sediment transport capacity. If the channel widens, it will tend to decrease sediment transport capacity, inducing deposition (Landwehr and Rhoads 2003). Because most public ditches were constructed nearly a century ago, the dominant process occurring today is overwidening from maintenance dredging. In very wide, shallow streams with little shade, high temperature levels may decrease biotic diversity and contribute to algal growth, biological oxygen demand, and low oxygen levels. Aggradation of fine sediment affects water quality by increasing the internal loading of suspended sediment and P (James and Larson 2008).

Construction of ditches also lowers the frequency of overbank flooding as sediment is placed along the stream banks during ditch excavation and maintenance, raising the bank height relative to the streambed (Hansen et al. 2006). Entrenchment increases the percentage of sediment and nutrients that is transported or deposited within the stream rather than being deposited over the banks in the floodplain. In gentle topography typical of agricultural landscapes in the Midwest, however, slope increases may be counteracted by channel evolutionary changes (see Figure 4.1) over time. As channels widen, aggradation increases on the channel margins as described by Simon (1989). Channelized streams may return to an equilibrium state over time, but actively maintained streams will be dredged again every five to ten years, returning the stream to an unbalanced state. Yet, given time, ditch systems can evolve to a relatively stable form (Magner et al. submitted). The loss of headwater streams decreases nutrient uptake, as small streams are considered hotspots for nutrient uptake (particularly N) and processing that occurs less in larger rivers (Bernhardt et al. 2005).

Dams

Dams are structures built across a stream to impound water for power generation, recreation, or flood control. Thousands of in-stream dams were constructed in agricultural watersheds of the United States after European settlement until about 1950 or so, at which time their construction declined (Reisner 1993). They were built originally for milling purposes, hydropower, and occasionally flood control (with the larger dams) or recreation. Most of the small dams were built for milling purposes and operated as "run-of-the-river," thus not impounding large volumes of water. These dams block the passage of aquatic organisms and may trap large amounts of sediment (Heinz Center 2002). Smaller check dams were also built for sediment capture in headwater areas with gully erosion problems (Potter 1991).

The construction of dams alters the natural flow and sediment transport regime of a creek or river. By trapping sediment within the pool, dams may improve water quality in close proximity downstream of the pools. In agricultural regions, dams often serve as large sediment/nutrient traps. When the dams are removed, fish and aquatic life passage is restored, but sediment and nutrients may be mobilized during dam removal (Heinz Center 2002). Should dam failure occur, as occurred on Lake Delhi in eastern Iowa on July 24, 2010 (ASCE 1996–2012), large releases of sediment and materials contained within the sediment can have major negative impacts on downstream conditions.

Road Crossings and Armoring of Stream Banks

Many streams have been relocated at road crossings, effectively channelizing them to prevent erosion or excessive shear forces on bridges and road abutments. Although this human impact has gone largely unnoticed, its widespread occurrence and consequences for stream sediment transport and water quality warrant greater attention (Lenhart et al. 2010a).

Another indirect impact of roads is the placement of *riprap* or other armor to prevent erosion where streams laterally migrate up against road embankments or other human infrastructure (MNDOT 2006). The placement of riprap decreases riparian vegetation and large woody debris input to the stream while also locking the channel in place, which is undesirable from a sustainable management standpoint because channels naturally migrate over time. Armoring of channel boundaries also may narrow channel width, leading to scouring and/or channel incision along the armored reach, and may increase bed and bank erosion immediately downstream (Lagasse, Schall, and Richardson 2001). This is a common occurrence in channels running through narrow valleys, particularly in mountainous areas (Wohl 2001). The use of stream bank bioengineering and root wads serves as a viable alternative to riprap.

Altered Hydrology, Sediment, and Nutrient Loading in Streams

Hydrologic alteration is the modification from the "natural" or long-term hydrologic regime reflected by changes to the amounts of the various water budget components. Richter and colleagues (1996) defined hydrologic alteration as changes to the magnitude, timing, frequency, duration, and rate of change in stream flow.

Changes to any component of the water budget (not just precipitation) may alter stream flow by altering the hydrologic processes that lead to stream flow from a watershed. For example, changes to evapotranspiration (ET) rates from land cover change can drastically increase or decrease stream flow even if the watershed is mostly vegetated (NRC 2008). The conversion of forest land to agricultural row crops has demonstrated this point, as it usually increases water yield because of decreased ET and increased peak flow (Brooks et al. 2003). Less obvious than forest clearance, an increased row crop coverage at the expense of perennial crops and pastures has increased water yield over much of the midwestern and Great Plains states in the United States (Schilling and Libra 2003). The fact that row crops have greater water yield than hay or native perennial plants has actually been known for decades (Hays, McCall, and Bell 1949).

Although peak stream flows may directly increase channel erosion, the increased volume of stream flow can degrade stream water quality through a more subtle process. If the total volume of stream flow increases but not the peak, there may not be large sediment input through erosion from cultivated fields; but widespread stream bank saturation (particularly in the larger rivers with big contributing drainage areas) leads to more frequent and widespread bank failure through mass-wasting and thus greater sediment load (Lenhart et al. 2010a).

Human Stream Degradation vs. Land Management

The influence of stream degradation on water quality relative to land management practices depends on the extent and types of human alteration as well as the natural characteristics of a given stream and watershed. Generally, streams with widespread human alterations tend to have greater instability and thus higher channel-derived sediment loads, degrading water quality and making them less responsive to management.

Climate also strongly influences watershed sediment yield. Semi-arid regions produce more sediment per unit area than well-vegetated humid to semi-humid regions (Bull and Kirkby 2002). In areas with uplands sensitive to disturbance, including many of the semi-arid plains regions, poor land management can greatly increase sediment yield from uplands. In the 1920s and 1930s, the Dust Bowl was caused by the plowing of sensitive prairies followed by an extended drought. Flatter landscapes and humid climates with watersheds that are naturally well vegetated tend to have lower upland erosion rates (Bull and Kirkby 2002).

Topography strongly influences stream response to storm events. For example, on the Paleozoic Plateau located in southeastern Minnesota, western Wisconsin, and northeastern Iowa, large amounts of sediment moved from the agricultural landscape to flood plains in the late 1800s to mid-1900s (Knox 1977; Potter 1991). Most of the sediment was retained in the valley bottoms (Beach 1994; Trimble 1983). Today, with upland erosion rates greatly decreased in the region from soil conservation efforts (Argabright et al. 1996), channel sediment has become a larger percentage of the suspended sediment load than cultivated farm fields in many parts of the northern Midwest (Engstrom, Almendinger, and Wolin 2009; Odgaard 1987). (See Textbox 4.1 for case studies.)

A legacy of poor land use/management in a watershed is still the main cause of stream degradation in many agricultural regions. The Dust Bowl disaster in the western Great Plains is one of the best-known examples. During the Dust Bowl era, many streams in the Great Plains were smothered with wind-blown sediment. Poor land management, including conversion of wind erosion-sensitive semi-scrub desert or dry prairie, from cattle grazing in the 1920s to crops, fueled this disaster (Happ, Rittenhouse, and Dobson 1940).

Other extreme examples of highly erodible uplands include the loess hills in the midwestern United States (Argabright et al. 1996) and the Loess Plateau in the Yellow River basin in China, which have some of the highest sediment yields in the world. The high-sediment yielding uplands of the Badlands in North and South Dakota are another case where it is difficult to improve stream water quality through watershed management (Foreman and McCutcheon 2010; Gonzalez 2001). The Badlands have low vegetative cover, very erodible geologic materials, and high erosion rates and sediment supply, a situation typical of many semi-arid regions of the world. Streams in geologic and climatic settings like this tend to have naturally higher suspended sediment loads (Bull and Kirkby 2002).

Restoration and Management

From the previous examples, it is clear that human alterations to streams have influenced water quality. So, can channel degradation be decreased to improve water quality and biotic integrity? In many situations

Textbox 4.1. Case studies

Case Studies

Nemadji River

There are numerous examples of watersheds that have good land use management practices but have high rates of channel erosion because of the legacy of human impacts and natural factors. For example, the Nemadji River in the Lake Superior drainage of Minnesota and Wisconsin confounded managers for years. The Nemadji River, located in a relatively undeveloped area, is a primarily forested watershed with high sediment yield. After much investigation, it was found that the high sediment yield is a product of channel erosion due to historic hydrologic alterations from forestry and the watershed's unique geologic setting. The upwelling groundwater combined with the clayey parent materials fosters mass wasting, which delivers sediment to the streams (Reidel, Verry, and Brooks 2005.)

Minnesota River

Another example in which sediment load from channel erosion exceeds that derived from upland erosion is the Minnesota River (Engstrom, Almendinger, and Wolin 2009). In the Minnesota River basin, watershed management practices have been implemented at least since the 1980s (Minnesota River Assessment Project) with little improvement to in-stream sediment load. This is primarily because of the large sediment load that is coming from bluffs (Sekely 2001) and stream banks (Nieber et al. 2009) accelerated by increased flows. Stream bank erosion rates are also high because the river runs across noncohesive alluvial sediments with little erosional resistance.

Fish Creek Watershed

In contrast with the Nemadji basin, many northern forested watersheds of the upper Midwest have recovered from poor land use management of the early twentieth century. In the Fish Creek watershed (near the Nemadji in northwestern Wisconsin), forest clearcutting and agriculture in the early 1900s increased sediment supply to the creek, which has since improved because of enhanced watershed management and revegetation of some farmed areas (Fitzpatrick, Knox, and Whitman 1999).

Hubbard Brook Watershed

Similarly, Likens and colleagues (1978) found that stream water quality responded positively to revegetation following clearcutting of the Hubbard Brook watershed in Massachusetts, with in-stream declines in nitrate and other nutrients within a few years of clearcutting, further supporting time responses identified in Figure 3.1. Although the stream recovered, upland productivity declined and had not recovered to prelogging levels after many years.

stream restoration, or more accurately "re-creation," can improve water quality and biotic integrity. Small streams that are not highly entrenched are more easily restored to a stable condition than larger rivers and/or highly entrenched streams (Rosgen 1996). Small prairie streams with cohesive stream banks can be effectively restabilized if deep gullies did not develop during the period of increased stream flow. In some cases, such as the Badlands, the answer is probably that stream restoration could not improve water quality to levels found in "cleaner" streams because of the naturally high sediment loads.

Sediment Reduction and Nutrient Loading

Increasing sinuosity in channelized reaches can help to reestablish the natural sediment transport regime, leading to improvements in water quality by reduction in sediment and P loading. As the natural meander pattern is reestablished, the point bars on the inside of the bends can once again become natural depositional areas for sediment, especially coarser sands and gravels.

Restoration or enhancement of floodplain connectivity/function can help to remove large amounts of sediment from the system (Lenhart et al. 2010a). Whereas sandbars remove some medium-sized sediment, floodplains typically store large volumes of finer sediments (silt and clays). Reconnection of floodplains in areas where streams have been cut off by levee construction, channelization, or entrenchment can increase fine sediment deposition, thus improving downstream water quality, particularly in incised channels with decreased sediment removal capability.

Control of excessively high rates of stream bank and bluff erosion can also improve water quality by removing a large source of fine sediment and P. Natural channel design using native vegetative materials with some rock at the toe of the slope (referred to as stream bank bioengineering) can decrease reaches of excessive bank erosion (MNDOT 2006). Bioengineering can also provide a habitat for riparian wildlife species, including birds, and create overhang for shelter and temperature reduction for fish in the stream.

Reduction of N load is more difficult to achieve by stream restoration because most of the supply is external (watershed sources), primarily in tile drainage systems in the midwestern agricultural states. One approach may be to restore headwater streams that are more efficient at denitrification (Bernhardt et al. 2005). Many of these have been graded completely for cultivation in farm fields or have been replaced with ditches. Managing ditches to enhance sediment and nutrient removal may be effective as well, using two-staged channels or other alternative designs (Magner et al. 2010; USDA–NRCS n.d.).

Restoration of Biotic Integrity

The restoration of biotic integrity in streams entails a wide variety of methods and procedures beyond the scope of this chapter. Two general categories of restoration projects that may help improve IBI scores are discussed briefly here. Removal of barriers by dam removal and fish passage can help restore connectivity for fish and aquatic life, such as mussels, that is important for the maintenance of aquatic biodiversity. This is particularly true in regions with highly migratory fish, such as the coastal regions of the United States. Fish passage, however, is also important in the midwestern United States because many freshwater species migrate seasonally to feeding or refuge areas.

Replacement of channel habitat features or management to restore the processes that created them where they have been artificially removed or decreased in abundance may improve stream biotic integrity as well. Replacement of large woody debris and other semi-structural tools, such as those that use tree root wads (Lenhart et al. 2010b), may help restore the variety of stream depth and substratum that benefits biodiversity.

Summary

A number of human alterations and natural factors may cause streams to be large sources of sediment and nutrients, independent of watershed conditions. In those cases where streams are highly degraded by human actions (rather than naturally high in sediment), stream restoration and management can help improve water quality, particularly in streams with highly erodible channel boundaries.

When is stream water quality a poor indicator of land management success? In situations where channels are highly unstable or contribute high sediment loads due to natural factors or human impacts, streams may not reflect land management practices. For example, many streams in the Paleozoic Plateau of Minnesota, Wisconsin, and Iowa had extensive floodplain deposition from poor land management practices in the mid-to-late 1800s and early 1900s, creating streams with very high stream banks that are now large sources of sediment. Some examples of this are listed in Table 4.1.

Table 4.1. Factors influencing responsiveness of streams as indicators of watershed management

Region or stream	Cause	Reference
Streams that respond slowly or very ineffectively to watershed management		
Nemadji River, Lake Superior basin, Wisconsin/Minnesota	Unstable geologic materials—lake clays; groundwater discharge flow path creates unstable bank conditions	Reidel, Verry, and Brooks 2005
Minnesota River basin	Increased rates of stream flow; steep bluffs and ravines along valley walls; highly erodible alluvial materials along main channel lead to a high channel-derived sediment load	Nieber et al. 2010; Sekely 2001
Badlands of South and North Dakota	Unstable channel materials; upstream migration of headcut from past land use change is driving channel erosion	Foreman and McCutcheon 2010; Gonzalez 2001
Streams that are responsive to effective watershed management		
Boulder Creek, Colorado	Improved watershed management with stable boulder-bedrock stream bottom led to water quality improvement in Boulder Creek	Wohl 2001
Lake Superior tributary streams on northern shore	Steep boulder-bedrock streams on the slope above Lake Superior are more responsive to watershed management than streams in more erodible channel materials further inland	Fitzpatrick et al. 2006
Hubbard Brook, Massachusetts	Appalachian Mountain stream water quality responded quickly to reforestation with declining in-stream nutrient levels within years of revegetation of the watersheds	Likens et al. 1978
Streams with mixed response to watershed management		
Fish Creek, Wisconsin	Decreased peak flows from period of peak agricultural activity and decreased sediment loading; however, sedimentation of lower main stem has decreased biotic integrity, limiting ecological recovery	Fitzpatrick, Knox, and Whitman 1999
Driftless area, Wisconsin	Bank erosion continues from high postsettlement alluvium (sediment deposited in river valleys after European settlement)	Knox 1977

When is stream water quality a good indicator of effective watershed management? Stream water quality is more resilient in regions with limited noncohesive, fine-sediment supply. In stable channels that are highly resistant to erosion, such as the bedrock- or boulder-lined channels on the front range of the Rocky Mountains, the Appalachian Mountains, or many of the tributary streams to Lake Superior, water quality in streams is more likely to reflect the condition of the watershed (Table 4.1). In these cases, excess sediment load must come primarily from poor upland management practices, such as forest clear-cutting or construction of improperly designed dirt roads, rather than from channel erosion. Most agricultural watersheds of the central United States, however, lie in glaciated regions with finer-textured geologic materials, providing an abundant supply of erodible, fine sediment. Consequently, water quality in these watersheds may take longer to show a positive response to land management efforts.

Abbreviations and Acronyms

ET	evapotranspiration
IBI	index of biotic integrity
JD#8	Judicial Ditch #8
N	nitrogen
P	phosphorus

Glossary

Hyporheic zone. The stream-groundwater exchange zone.

Large woody debris. Logs with a minimum diameter of 10 centimeters (4 inches) and a minimum length of 1.83 meters (6 feet) that protrude or lie within a stream channel.

Riprap. A layer of stones or chunks of concrete thrown together without order on an embankment slope to prevent erosion.

5 Synthesis

Most present agricultural land use promotes a landscape where increased wind and water erosion of soils results in concurrent increases in the transportation of pollutants, sediment, and runoff water into receiving water bodies. Seldom are there watershed streams that haven't experienced concurrent changes in sedimentation, nutrient enrichment, anthropogenic compounds, habitat loss, and even hydrology. While examining the impacts of land use and land management, the fact that the watersheds themselves have become altered and a source of recent stored material pollutants such as nutrients should not be overlooked. For example, soils in both developing and developed countries have accumulated and are accumulating total phosphorus (Bennett, Carpenter, and Caraco 2001).

For developed countries, soil phosphorus inputs in fertilizer and manure have exceeded removal of phosphorus by crops and animal products, resulting in continual accumulation of phosphorus in soil over the past four decades. Reasons for the noted increases in phosphorus include intensification and specialization of farming systems that have led to regional surpluses of phosphorus imported in fertilizer and animal feed compared with phosphorus exported in farm produce (Carpenter et al. 1998). Now many farms have soil phosphorus concentrations well in excess of plant needs with increased potential for phosphorus loss (Sharpley 2000). The dominant source(s) of phosphorus loss to streams, however, typically is associated with small areas within basins and watersheds (McDowell et al. 2004). It is necessary to understand, nevertheless, that legacy nutrients may be constraining stream water nutrient reductions associated with land management changes at the watershed level.

Improving the water and biological quality of surface waters located in predominately agricultural watersheds is related to several determinate factors that collectively create and sustain the ecological health of streams, rivers, and other water resources. Karr and Dudley (1981) defined biological integrity and listed four determinants of ecological integrity in flowing water ecosystems—water quality, flow regime, habitat structure, and energy relationships. If the term "flow regime" is broadened to mean the hydrology of the system, these determinants apply to all aquatic systems. Thus, restoring the integrity of aquatic ecosystems means that all of these factors must be assessed and managed because any one of these determinants can become the limiting factor in restoration of the watershed and aquatic system. Although the Clean Water Act has focused attention on the importance of water quality, it remains just one means to the greater goal of maintaining and restoring the integrity, use, and function of the nation's waters.

Land management can often improve specific water quality conditions such as reductions in bacteria, turbidity, and nutrients, but results are often mixed and reductions to meet specific goals such as water criteria are seldom addressed. Improvements in water quality are only meaningful if all constraining water quality parameters are improved and no other ecological determinate is limiting restoration. Complying with established water quality standards provides some guidance in setting management goals, but quite often toxic pollutants in agricultural watersheds co-occur and their joint toxicities within watersheds are not known or understood.

Gilliom (2007) found that in agricultural watersheds most streams were contaminated with pesticides (97%) and many had concentrations greater than aquatic-life benchmarks for water (57%) and bed sediment (31%). Aquatic life support criteria or benchmarks are typically derived from single organism/single toxicant bioassay tests, yet Gilliom showed that more than 80% of all agricultural streams sampled had pesticide mixtures (≤4 compounds). Few studies have examined joint toxicity of agricultural pesticides, but when it has been examined, additive and greater than additive responses were noted, suggesting existing criteria values may not be protective (Belden and Lydy 2000; Belden et al. 2007; Blackburn 1985; Carder and Hoagland 1998; Lydy et al. 2004; Mohammad and Itoh 2011; Pape-Lindström and Lydy 1997).

Joint occurrences of toxicants and their resulting impacts are a special case representing a more prevalent agricultural watershed condition—multiple stressors/cumulative effects. In fact, cumulative effects from both incremental effects of past conditions and co-occurring containments are a well-documented but difficult to quantify environmental phenomena associated with nonpoint source pollution (CEQ 1997; Duinker and Greig 2006; Loftis et al. 2001; MacDonald 2000; Reid 1998; Seitz, Westbrook, and Noble 2011).

Although water quality has long been the focus of efforts to protect and manage aquatic resources in agriculture watersheds, hydromodification and habitat degradation/loss are now seen as major causes of the continued impairment of aquatic ecosystems. Whereas most hydromodification activities are intended to provide societal benefits (e.g., decreased flooding, better drainage for agricultural land, power production, drinking water supply), there are often many unintended consequences resulting from these activities (e.g., Bunn and Arthington 2002; Mantel, Hughes, and Muller 2010; USEPA 1989, 2007). These factors directly and indirectly affect aquatic biota as well as the pollutant assimilation capabilities and water quality of these same systems. The *National Water Quality Inventory: 2004 Report to Congress* (USEPA 2009) identified hydromodification (e.g., dams, levees, channelization) as a leading source of water quality and biological impairment in assessed surface waters. Within this report, hydromodification was ranked the second leading source of impairment in all freshwater ecosystems excluding wetlands, which were not assessed in the report.

Recent studies of National Water Quality Assessment data substantiate Environmental Protection Agency findings and indicate that as much as 70% of U.S. streams and rivers are biologically impacted by altered hydrological conditions within their watersheds (Carlisle, Wolock, and Meador 2011). Only agriculture as a pollutant source category was ranked higher than hydromodification as a major source of impairment to streams, rivers, ponds, lakes, and reservoirs. Agricultural land use and management techniques themselves, however, can result in hydromodifications that then contribute to biological and water quality impairments (e.g., Nejadhashemi, Wardynski, and Munoz 2011; Poff, Bledsoe, and Cuhaciyan 2006; Potter 1991; Ramireddygari et al. 2000).

Channelization and channel modification, dams/impoundments, and stream bank/shoreline erosion are commonly occurring types of hydromodification that can alter both a waterbody's physical structure (e.g., fish habitat, bank height, sinuosity) and its natural functions (e.g., flow retention time, nutrient assimilation, energy inputs), all of which can affect water and biological quality. Direct loss, degradation, and disconnection (loss of floodplain connectivity) of both instream and near-stream habitats directly impact aquatic communities and contribute to the overall reduction in stream quality within agricultural watersheds (Amoros and Bornette 2002; Naiman, Decamps, and McClain 2005; Soman et al. 2007). Overall, it seems that restoration of aquatic ecosystems is increasingly proposed as an approach for improving water quality (Jorgensen and Yarbrough 2003), yet this approach remains understudied and underfunded.

It is increasingly understood that land use legacies can affect both the rate and extent of ecological recovery in terms of stream structure and function (see Foster et al. 2003). In the United States, these historic (i.e., legacy) land use changes are most often measured in hundreds of years; for example, the Iowa landscape was completely transformed in a period of 100 to 150 years (Gallant et al. 2011). Huggins (unpublished) plotted landscape disturbance curves based on the collective density of people, domestic animals, and tilled land/404.7 hectares (1,000 acres) for major ecological regions in Kansas and found that most regional peaks occurred around 1902. The suggestion from these and other findings is that most aquatic ecosystems in agricultural watersheds have experienced >100 years of excessive nutrient and sediment exposure and deposits that can compromise and extend expected timelines for pollutant reductions and ecosystem recovery (Francis and Foster 2001; Jackson et al. 2005; Schelske and Hodell 1995; Schubauer-Berigan, Minamyer, and Hartzell 2005).

On a global scale, most legacy sediments (and some nutrients) are stored within third- to sixth-order tributaries and their floodplains (Wilkinson and McElroy 2007), where they can be remobilized through floodplain and bank erosion processes and add to the pollutant loads of the streams and downstream lakes and impoundments (Allmendinger et al. 2007; Laubel et al. 2003; Walter, Merritts, and Rahnis 2007; Zaimes et al. 2005). It has been suggested that legacy sediment storage has occurred within even smaller tributaries (i.e., second order) in many agricultural landscapes (Bettis, A. 2012. Personal communication).

In addition to overcoming legacy effects, there are often significant time lags between employment of management practices and both terrestrial and aquatic system response to these management efforts, whether they are reductions in pollutant transportation, improvements in specific water quality parameters, or ecological integrity. Lag time has been defined as the sum of time for practice to produce effect, effect to travel to waterbody, and waterbody to respond (Meals, Dressing, and Davenport 2010).

Whereas nonstructure management practices can be implemented in a day, the development of a woody riparian buffer, for example, can take decades. Timelines for pollutant reductions and ecosystem recovery are both pollutant and system dependant, varying from perhaps less than a year to centuries. Meals, Dressing, and Davenport (2010) listed recovery time from microbial pollution as relatively short (bacteria themselves are short lived), but decades for nutrients and some stream community elements. When considering geomorphological changes to waterbodies and their watersheds that often result from hydromodifications, ecosystem recovery could take much longer. For instance, the time required for stream channels to stabilize after landscape alteration is on a scale of decades to centuries, if not longer (Brunsden and Thornes 1979; Schumm 1977; Trimble 1974, 1999).

Agriculture activities in many watersheds tend to be the dominant activity, both in terms of extent and intensity; thus massive broad-scale management efforts must be brought to bear to invoke the magnitude of change that will stimulate water quality transformation and ecosystem recovery. In the process of examining factors affecting progress in developing and implementing agricultural management to improve water quality, researchers must remain aware that they are aiming at a moving target. More than a decade ago Tilman (1999) noted that since the mid-1960s global agricultural food production has doubled, resulting in at least a six- and three-fold increase in nitrogen and phosphorus fertilizer, respectively. Additionally, this intensification of agriculture has resulted in the near doubling of irrigated cropland and a 10% increase in cultivated land.

Assuming that these global figures are representative of agricultural trends in the United States, scientists must also consider the possibility that environmental consequences (i.e., improvements) of past and current advancements in land management are being masked by concurrent increases in the intensity of agricultural use. Progress has been made, but cumulative effect, legacy conditions, intensified agricultural production, and lag times between implementation of management practices and system responses collectively affect the extent and trajectory of ecosystem recovery. Aquatic systems are dependent on a number of determinant factors, and management planning and efforts must address many differing impairments because ecosystem recovery will ultimately be governed by the last limiting factor or condition, whether it is sedimentation, nutrients, or habitat loss and channel instability.

In review of this "management effort versus stream improvement" question, the authors would recommend examining several policy, planning, and implementation considerations that individually and collectively could improve their comprehensive efforts to move more quickly and successfully toward stream improvements in agriculturally dominated watersheds. Their initial recommendations include the following:

1. Examine the use and benefits of biological criteria and biological goal setting to establish both meaningful objectives and measured responses (e.g., success, improvements, no change). Continued reliance on water quality standards alone to achieve "biological integrity" goals is short sighted and in many cases invites failure if nonpollutant impairments prevent or hinder overall stream quality improvements.
2. Better understand aquatic ecosystem alterations and impacts related to legacy conditions such as erosion and sedimentation, nutrient accretions, and floodplain alterations. By determining legacy contributions and accounting for current, intensive agricultural land use in many watersheds and basins, discussions of the attainability of specific designated uses assigned to particular aquatic waterbodies or stream segments can be better facilitated. The U.S. Environmental Protection Agency's *Use Attainability Analysis* (USEPA 2011b) is a structured scientific assessment of the factors affecting the attainment of uses specified in Section 101(a)(2) of the Clean Water Act.
3. Adopt and incorporate long-term, large-scale, coordinated stream restoration planning, evaluating, and monitoring. It is clear that many aquatic ecosystems are hydrologically and geomorphologically impaired, and these "physical" as well as "chemical" disturbances in many cases are constraining biological improvements.
4. Focus funding and monitoring on implementation of the management tools at hand, and measure system responses to determine what works and what does not. Whereas many programs that assist stakeholders in defraying management costs for "pollution" reduction are already in place, few stream restoration programs are linked to water quality improvement projects. Unfortunately, most restoration efforts still focus on small-scale projects with ill-defined or limited goals that fail to monitor for accomplishments.

5. Agricultural management goals aimed at improving water and stream quality should be built upon the understanding that ecological integrity is dependent on multiple factors that include but are not limited to water quality, habitat quality, and proper hydrological function. Team building and policy structure should include a broader suite of expertise in goal setting, practice(s) implementation, and restoration efforts.

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