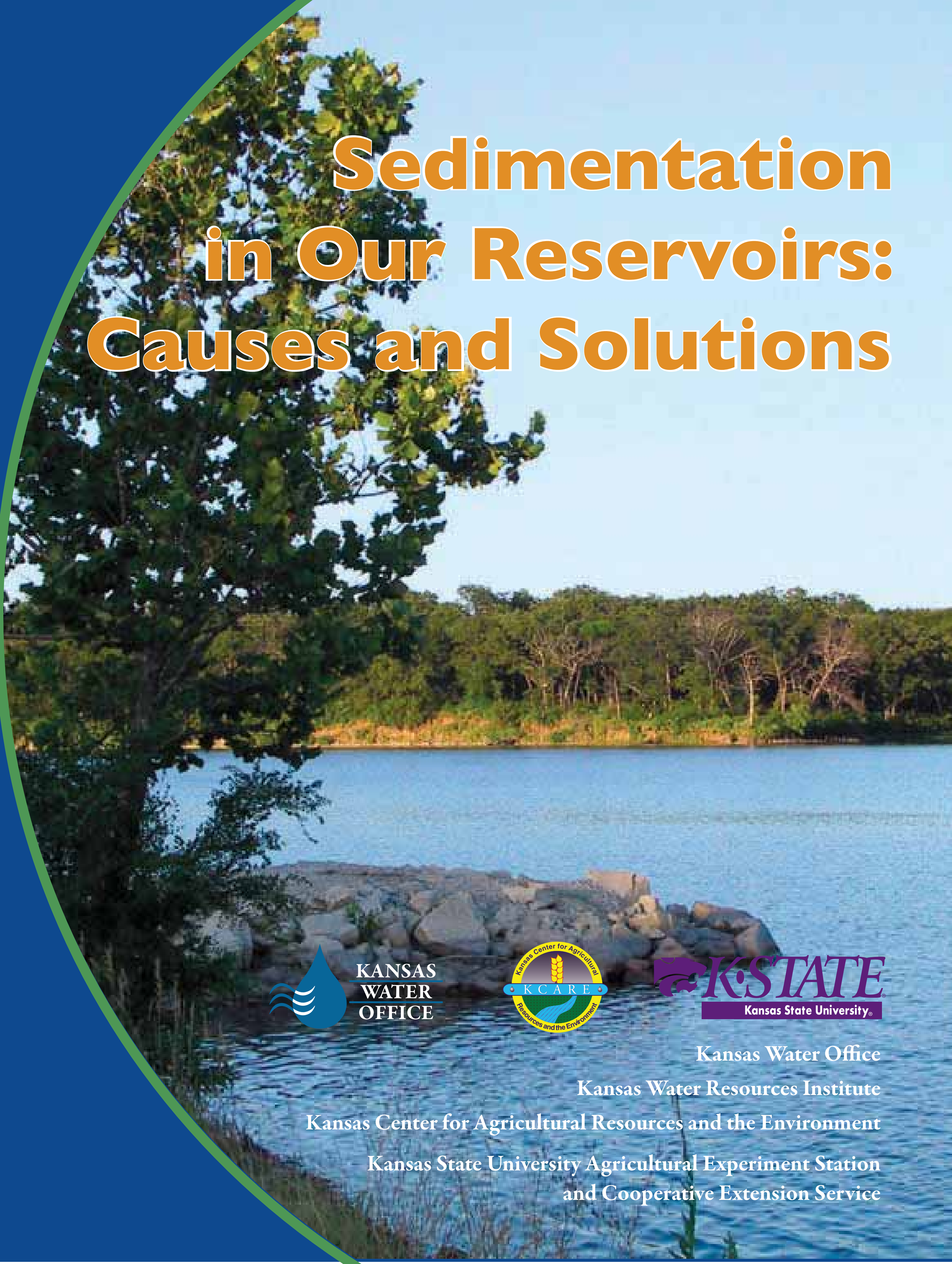

Sedimentation in Our Reservoirs: Causes and Solutions

Kansas Water Office

Kansas Water Resources Institute

Kansas Center for Agricultural Resources and the Environment

Kansas State University Agricultural Experiment Station
and Cooperative Extension Service

Sedimentation in Our Reservoirs: Causes and Solutions

Table of Contents

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Reservoirs: Infrastructure for Our Future

Federal reservoirs in Kansas serve as the source of municipal and industrial water for more than two-thirds of the state's population. They are recreational destinations and provide a reserve to supplement streamflow for water quality, aquatic life, and related activities. These reservoirs were built from the 1940s through the 1980s by the U.S. Army Corps of Engineers and the Bureau of Reclamation primarily for flood control. State and local users saw value in adding water supply storage to the purpose of those reservoirs.

Reservoirs are integral to Kansas' water supply infrastructure, but like all infrastructure, reservoirs age. By their nature, reservoirs act as settling basins; they gradually fill with sediment, which reduces their capacity to store water to meet our needs. Although erosion is natural, our actions often accelerate this process. Human activities such as urbanization, agriculture, and alteration of riparian and wetland habitats have changed flow regimes, increasing the concentrations and rates at which sediment enters streams and rivers.

Kansas' economic landscape is changing. A viable economy depends on well-managed natural resources. Too often we take for granted that the foundation of our lives and livelihoods will be there forever. Future demand for water supply from federal reservoirs is projected to increase. Increasing demands coupled with decreasing supplies will eventually result in water supply shortages during severe drought conditions. Preliminary studies indicate that if a multi-year, severe drought occurred in the foreseeable future, water supply shortages could occur because of diminished storage in several basins. Models are currently being developed to more effectively use available storage and optimize use of reservoir water to meet current and future needs.

At the same time, study and research should be directed toward determining sources and movement of sediment in our streams and rivers. This knowledge will allow resource managers to improve the effectiveness of programs and practices to reduce sedimentation rates, improve riparian and aquatic habitats, and derive the most value from dollars spent and resources invested.

Protecting and making the best use of reservoirs and the streams and rivers that feed them requires an investment today to assure they will be sustained for future generations. The Kansas Water Office is committed to that investment.

Tracy Streeter

Tracy Streeter
Director, Kansas Water Office

Sedimentation and the Future of Reservoirs in Kansas

The U.S. government made significant investments in building reservoirs in the 1950s and 1960s, which changed much of the rural environment in Kansas. Although many reservoirs were built with a projected lifespan of 150 to 200 years, current projections indicate these lifespans could be cut short by 50 to 100 years. Sedimentation is reducing water-storage capacity of these reservoirs, and deposited sediments containing nutrients, trace metals, and endocrine disrupting compounds are significantly affecting reservoir water quality. Scientists have documented changes in sediment load and water quality, and citizens have watched reservoirs "shrink" over past decades. Bridges that once spanned water now sit above a "mud flat" of sediment.

The Dust Bowl of the early 1900s had dramatic social, biological, and physical consequences in Texas, Oklahoma, and Kansas and resulted in dramatic technological changes in land management. The "Mud Bowl" resulting from reservoir sedimentation poses an even larger threat that demands corrective action based on sound science and practical, affordable technologies.

Protecting reservoirs from sedimentation will:

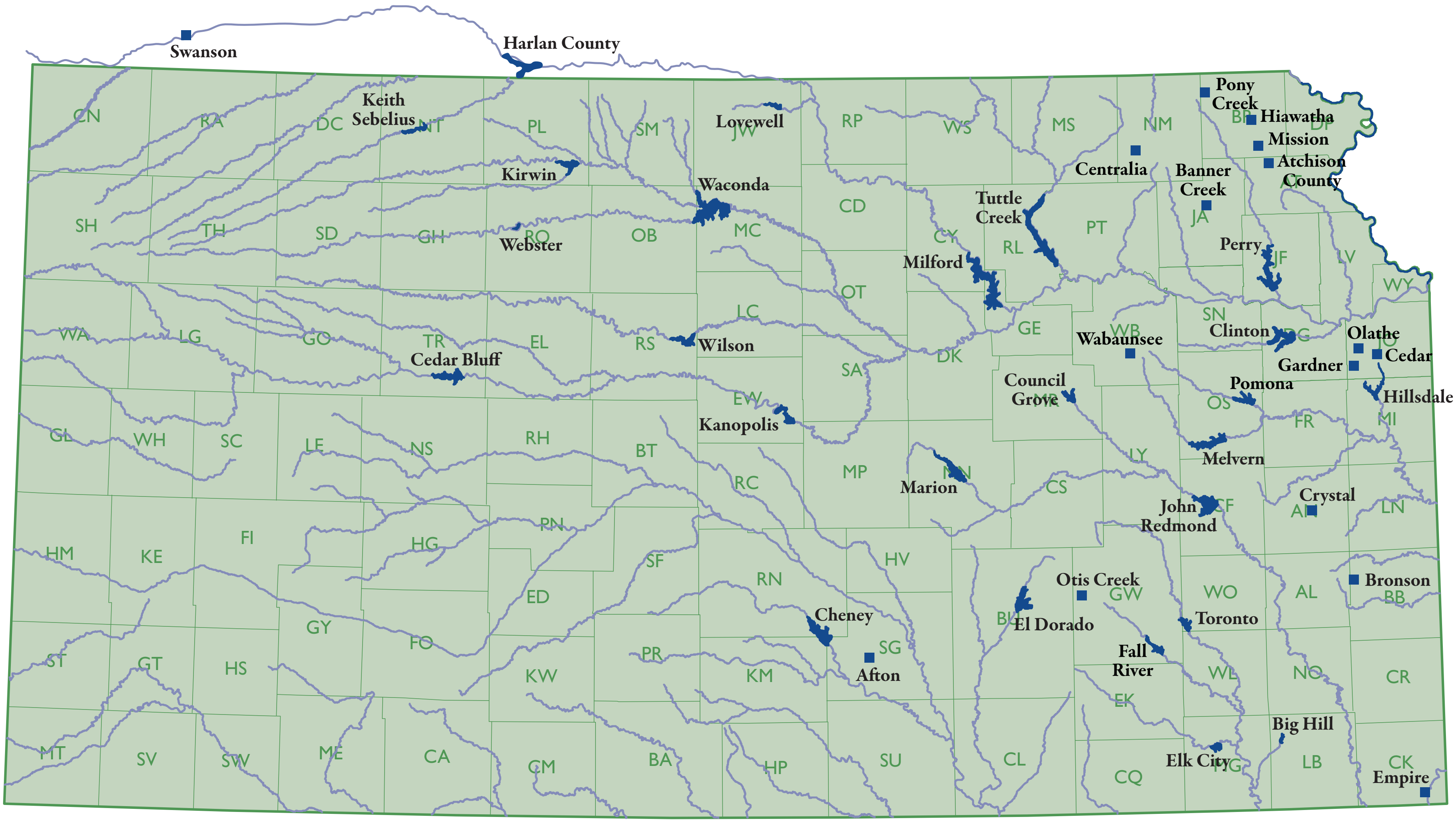
- result in overall water conservation (i.e., maximize reservoir water storage, minimize water loss during storm events, and improve water conservation management);
- require widespread implementation of conservation measures; this requires us to evaluate, understand, and influence producer management behaviors that affect implementation of conservation measures as well as sedimentation and future functioning of reservoirs;
- involve participants from a variety of disciplines including agriculture, engineering, hydrology, sociology, economics, and others;
- affect water savings on a large scale not only by conserving and protecting existing reservoir resources but also by retaining more soil and water on land; and
- be crucial to agriculture and rural life, especially in Kansas, and encompass a variety of community, economic, environmental, health, and social issues.

This publication brings together leading scientific knowledge from many academic disciplines and identifies technological solutions that will protect and conserve federal reservoirs. The following white papers evaluate threats to sustainability of federal reservoirs, causative factors behind these threats, and technological solutions along with their scientific underpinnings and propose future research needed to improve sustainability of these vital water resources and landscapes to which they are connected. Our aim is to advance interdisciplinary science, research, collaboration, and problem solving to achieve a key goal: sustaining supplies of abundant, clean water in Kansas.

W. L. Hargrove

W.L. Hargrove
Director, Kansas Center for Agricultural Resources and the Environment (KCARE)

Reservoirs in Kansas



Map from USGS; Kansas Geological Survey. Adapted with permission.

Kansas has more than 120,000 impoundments ranging in size from small farm ponds to large reservoirs. The 24 federal reservoirs in Kansas range in size from 1,200 to 15,314 surface acres; 21 of these provide drinking water for more than half the state's population. Smaller, state- and locally owned reservoirs are vital resources for drinking water, flood control, and recreation and are distributed across nearly every county in the state.

This map shows the 24 federal reservoirs in Kansas and several smaller basins referenced throughout this publication.

Current State, Trend, and Spatial Variability of Sediment in Kansas Reservoirs

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Kansas Biological Survey, University of Kansas

Introduction

The more than 300,000 acres of public and private reservoirs and ponds constructed in Kansas during the past century are steadily filling with silt. These water resources were constructed at great expense. For example, cost of a typical Kansas reservoir (≈7000 acres in size) constructed in the 1970s was $50 million to $60 million ($200 million to $250 million in 2007 dollars). Yet, reservoirs provide significant economic value to the state through flood control, irrigation, recreation, wildlife support, power generation, and high-quality water for human and livestock consumption. More than half the U.S. and Kansas population receives some drinking water from reservoirs.

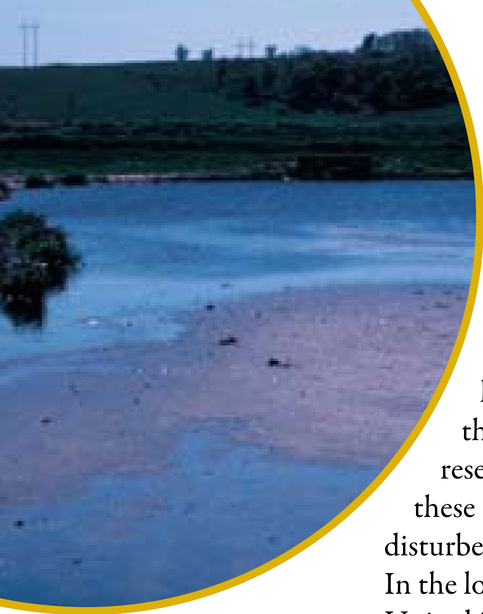
It is becoming increasingly complicated and costly to manage these crucial water resources; inevitably, silt will fill these water bodies entirely unless removed periodically. Silt removal will be an enormous task, even more so than original construction, but there is still time to prepare. Although a number of state agencies are beginning to examine this long term management problem, new efforts must be directed at controlling the currently declining quality of aging reservoirs.

The Reservoir as a Resource

During the 20th century, more than 2 million reservoirs of all sizes, including smaller ponds, were constructed in the United States, and many more were constructed worldwide. Nearly 1,000 U.S. reservoirs are larger than 1,000 acres, and about half of these are federally operated. The lower half of the mid-continental United States, particularly the central states of Kansas, Missouri, Oklahoma, Arkansas, and Texas, has the greatest number of reservoirs. The National Recreation Lakes Study Commission (1999) determined that the 490 federal reservoirs larger than 1,000 acres had an annual economic impact of $44 billion and provided employment for 637,000 persons. Several thousand smaller reservoirs provide recreation opportunities, and all reservoirs provide flood control that protects lives and property; economic impacts of these benefits are incalculable.

Reservoirs and lakes are basins of standing water; flow of water through them is slower than that in entering streams and rivers. Reservoirs are constructed by human means, but lakes form naturally. Both range greatly in size, function similarly, are affected by the same environmental conditions, and provide similar resources. Most reservoirs have a normal operation depth and pool volume for recreation and water supply with additional flood control depth and pool volume above the normal pool and below the spillway to temporarily absorb floodwaters (i.e., minimize prolonged added pressure on the dam). Reservoirs and lakes require similar management and renovation practices, but these efforts often are focused on reservoirs, which typically are constructed to serve particular continuing needs.

Reservoir problems requiring particular management actions usually involve quality of drinking water and recreation and water



storage capacity for flood control and power generation. We build reservoirs in areas with few natural lakes, but we also recognize that these environments do not support reservoirs' continued existence. Soils in these areas are very erodible and can be disturbed even more by human activities. In the lower half of the mid-continental United States, where many reservoirs have been constructed, surface soils and clays are deep. For thousands of years, these materials moved naturally into valleys and stream channels; now they move into reservoirs. Thus, reservoirs act as settling basins in which the sedimentation process deposits soil, clay, and smaller rock particles. The upper regions of reservoirs, where streams enter, fill with sediment three to five times more rapidly than deeper areas. Expanding shallow zones reduce quality of water and wildlife habitat as well as operation storage capacity for drinking water and recreation. Sediment can fill the basin in 100 to 200 years, the projected life expectancy of most reservoirs. In contrast, most natural lakes exist for tens of thousands of years.

Two hundred of the largest reservoirs in the United States are now more than 40 years old. What will we do when most of our existing reservoirs are filled enough to end their useful life? We already built reservoirs in nearly all of the best places. Excavating old reservoirs will require moving 15 to 30, even up to 100, times more material than originally was moved to construct the dam. We also need to find a location for the removed material, ideally one that is nearby and will withstand this environmental disturbance. Further, because urban and rural development steadily surrounded our reservoirs, we cannot continually raise the height of the original dam and the contained water level or build new reservoirs nearby. Obviously, we must develop and implement new management strategies to maintain current reservoirs for their intended uses and extend their life expectancy.

Kansas Reservoirs: Number, Size, Distribution, Ownership, Uses

Kansas has more than 120,000 impoundments, although most (> 80%) are farm ponds smaller than 1 acre. Nearly 6,000 reservoirs are large enough to be regulated by the state (Figure 1). Approximately 585 reservoirs are owned by state or local governments; these average 30 years in age. The 93 Kansas reservoirs used as water supplies are an average of 51 years old; 63 of these are state or locally owned. The 21 federal reservoirs used for drinking water in Kansas have watersheds that cover 23% of the state and contain more than 4,000 miles of stream channels. Many reservoirs serve multiple purposes (e.g., domestic water supply, flood control, recreation, and irrigation).

Responding to increasing occurrences of water quality problems affecting use of Kansas reservoirs is an enormous challenge. The most pressing issue is ensuring the quality of water received by drinking water suppliers, who provide treated water to more than 60% of Kansas residents. Flood control, recreation, irrigation, and other uses also must be protected. Sediment accumulation and other factors continue to create immediate problems for water and habitat quality. But, siltation is just one part of the problem; reservoirs experience many problems long before they are completely filled (deNoyelleys et al., 1999). For example, sedimentation produces shallow water zones. This leads to increased cyanobacteria (blue-green algae) production, which, in turn, often causes taste and odor problems in drinking water (Figure 2). Numerous Kansas reservoirs are already experiencing problems. Cheney Reservoir (Smith et al., 2002; Wang et al., 2005b), Clinton Lake (deNoyelles et al., 1999; Mankin et al., 2003; Wang et al., 1999, 2005a), and Marion Lake (Linkov et al., 2007) all experienced massive algae blooms that triggered shutdowns of drinking water intakes. The near-complete siltation of the north end of Perry Lake (Figure 3) led to abandoned recreation areas and boat ramps and loss of fish habitat.

Particular Challenges of Smaller Reservoirs

Smaller, state- and locally owned reservoirs are vital resources for drinking water, flood control, and recreation and are distributed across nearly every county in the state (Figure 4). Small reservoirs are more likely than large reservoirs to exhibit serious

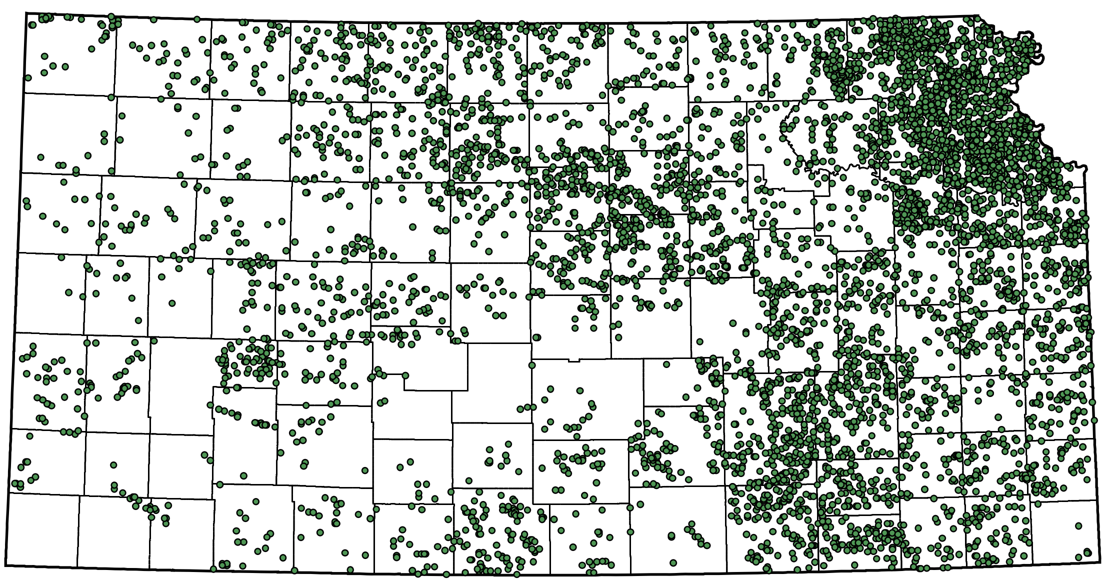


Figure 1. Reservoirs and impoundments in Kansas
Data analysis and map preparation: Kansas Biological Survey
Data source: USACE (200

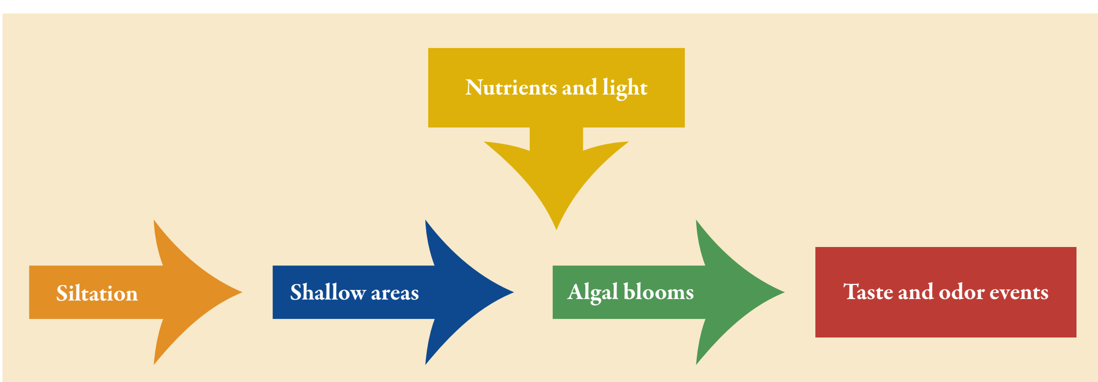
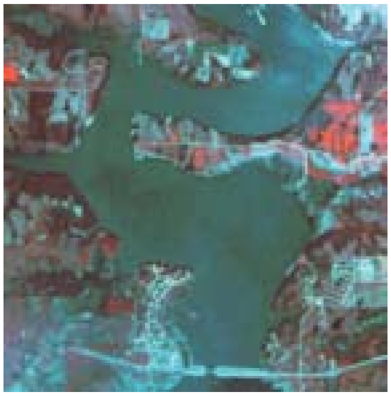


Figure 2. Sedimentation triggers a series of problems



April 24, 1974

October 25, 2001

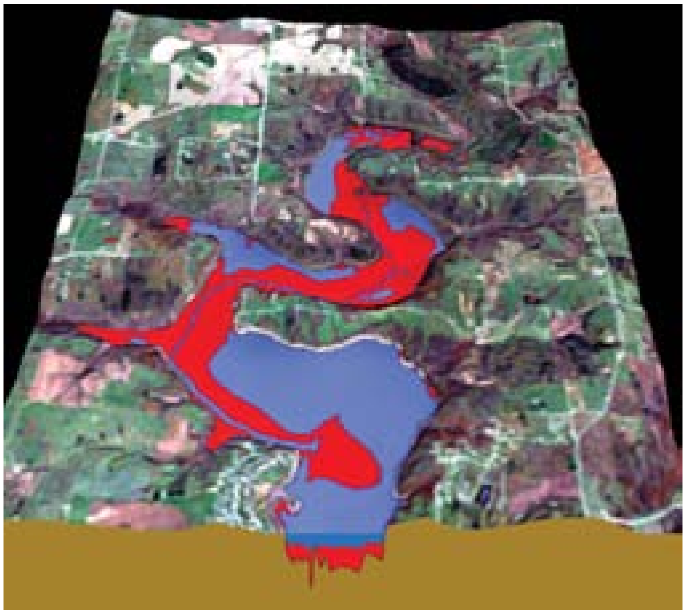


Figure 3. Siltation in Perry Lake, 1974-2001
An estimated 91.5 million cubic yards of sediment have accumulated leading to loss of more than 1,000 acres of surface area
Images courtesy of Kansas Biological Survey

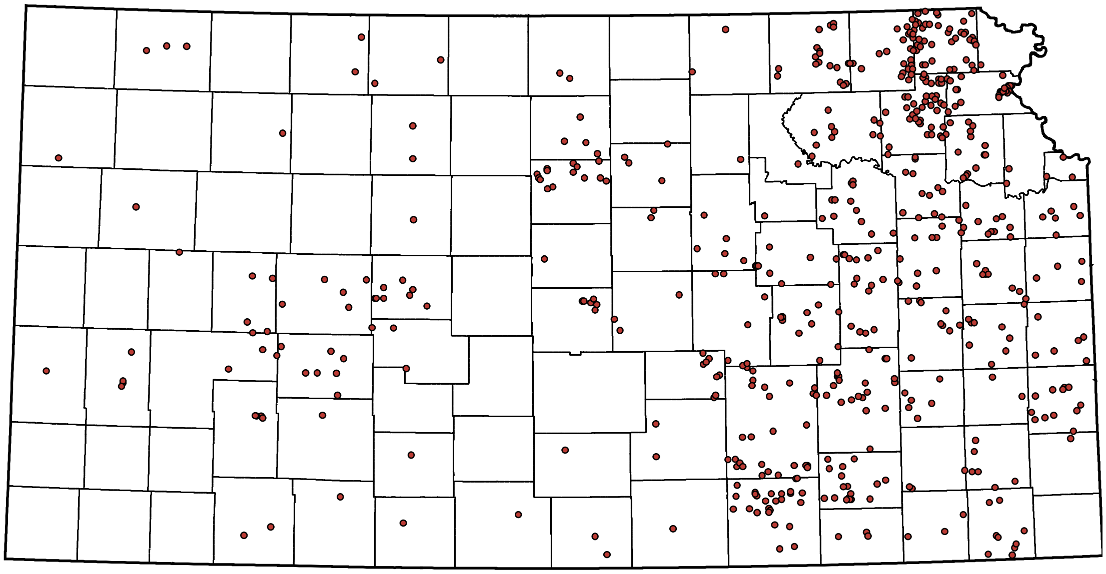


Figure 4. Reservoirs owned by the state of Kansas or local governments
Average age: 30 years; Average normal storage: 639 acre-feet
Data analysis and map preparation: Kansas Biological Survey
Data source: USACE (2005)

impairments in water quality and quantity and wildlife habitat due to siltation. For example, Cedar Lake in Johnson County (54-acre surface area) lost 50% of its volume since its construction in 1938. Cedar Lake is upstream from Lake Olathe, a water supply for Johnson County, and intercepts much of its sediment load.

Kansas currently is losing value, resources, and benefits from all its impoundments to varying degrees and will experience more rapid losses in the future, but the vast number of small reservoirs in Kansas is a challenge for state agencies charged with managing them. Unfortunately, most nonfederal reservoirs are not mapped and monitored for changes that could signal the onset of conditions that lead to water supply impairment. Water managers lack basic physical and biological data that can help identify impaired reservoirs, prioritize reservoirs in terms of impairment and need for renovation, or assess the current state of a reservoir.

Current State, Trend, and Conditions of Sedimentation in Kansas Reservoirs

Large Reservoirs

Current information on sedimentation is not available for most large, federal reservoirs in Kansas. In most cases, these reservoirs have not been surveyed for 10 to 20 years (Table 1). Available information (projected through 2005) indicates that

Table 1. Bathymetric surveys of 18 federal reservoirs in Kansas

Reservoir	Year of closure	Year of most recent survey	Years since most recent survey[a]
Kanopolis	1948	1982	25
Marion	1968	1982	25
Wilson	1964	1984	23
Council Grove	1964	1985	22
Melvern	1972	1985	22
Pomona	1963	1989	18
Fall River	1949	1990	17
Toronto	1960	1990	17
Clinton	1977	1991	16
Big Hill	1981	1992	15
Elk City	1966	1992	15
Milford	1967	1994	13
Hillsdale	1981	1996	11
Cheney	1964	1998	9
Tuttle Creek	1962	2000	7
Perry	1969	2001	6
El Dorado	1981	2005	2
John Redmond	1964	2007	0

[a] As of 2007

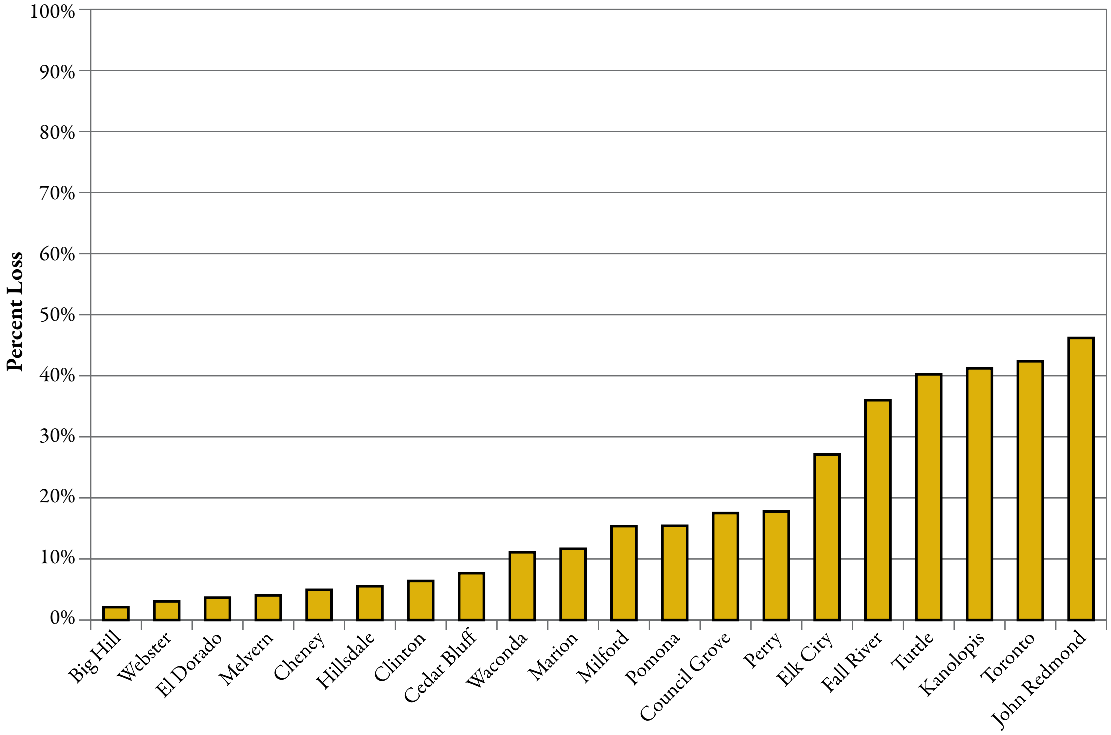


Figure 5. Loss of multipurpose pool water-storage capacity in Kansas federal reservoirs
Data source: KWO (2008)

Table 2. Mean annual sediment yield and mean annual precipitation for selected reservoir basins in Kansas[a]

Reservoir basin	Sediment yield (acre-feet/ square mile per year)	Mean annual precipitation (in.)
Small reservoir basins		
Mound City Lake	2.03	40
Crystal Lake	1.72	40
Mission Lake	1.42	35
Gardner City Lake	.85	39
Otis Creek Reservoir	.71	33
Lake Afton	.66	30
Large reservoir basins		
Perry Lake	1.59	37
Hillsdale Lake	.97	41
Tuttle Creek Lake	.40	30
Cheney Reservoir	.22	27
Webster Reservoir	.03	21

[a] Data source: Juracek (2004)

multipurpose pool water-storage capacity lost because of sedimentation ranges from less than 10% for Cheney Reservoir, Hillsdale Lake, and Webster Reservoir to more than 40% for the Tuttle Creek, Kanopolis, Toronto, and John Redmond Reservoirs (Figure 5; KWO, 2008). Approximately 18% of the original multipurpose pool of Perry Reservoir, one of the largest reservoirs in Kansas, which was constructed in 1969 with a 12,200-acre operation pool and a 25,300-acre flood control pool, has been lost to sediment deposition (Figure 5). Mean annual sediment yields from basins of five large reservoirs range from 0.03 acre-feet/square mile for Webster Reservoir to 1.59 acre-feet/square mile for Perry Lake (Table 2; Juracek, 2004).

Small Reservoirs

Current information on sedimentation also is lacking for most small reservoirs in Kansas. However, the U.S. Army Corps of Engineers recently completed a resurvey of 34 small reservoirs (KWA, 2001). Results indicated that water-storage capacity lost because of sedimentation ranged from negligible for Wellington New City Lake (4 years old at the time of the resurvey) to 62% for Alma City Lake (34 years old at the time of the resurvey) (Table 3). In another study, Juracek (2004) determined that mean annual sediment yields from six small reservoirs ranged from 0.66 acre-feet/square mile for Lake Afton to 2.03 acre-feet/square mile for Mound City Lake (Table 2).

Statewide Variability in Reservoir Sedimentation

The combined influence of several factors determines the sedimentation rate for a given reservoir. Collins (1965) created a generalized map of sediment yield in Kansas using available information on areal geology, topography, soil characteristics, precipitation, runoff, reservoir sedimentation, and measured suspended-sediment loads in streams (Figure 6). In the Collins map, mean annual sediment yields ranged from less than 50 tons/square mile in parts of southwestern and south-central Kansas to more than 5,000 tons/square mile in the extreme northeastern part of the state. More than 4,000 of the nearly 6,000 major reservoirs in the state are located in areas with the three highest sediment yield classes. A recent comparison of basin-specific sediment yields for eight reservoirs using regional estimates provided by Collins (1965) indicated that basin-specific yields tended to be smaller. This difference could be due to implemented conservation practices and information used to estimate yields (Juracek, 2004).

To explain differences in sediment yields among reservoir basins in Kansas, Juracek (2004) compared estimated mean annual sediment yields for 11 reservoirs with factors that affect soil erosion—precipitation, soil permeability, slope, and land use. Only the relationship between mean annual sediment yield and mean annual precipitation (Table 2) was statistically significant. As mean annual precipitation increased, mean annual sediment yield also increased. For the 11 reservoirs studied, mean annual precipitation was the best predictor of sediment yield. Given the pronounced decrease in precipitation from east to west across Kansas, a similar east to west decrease in reservoir sedimentation rates is likely.

Table 3. Characteristics of small municipal reservoirs in Kansas[a]

Reservoir	Community served	Year built	Original capacity (acre-feet)	2000 capacity (acre-feet)
Alma City Lake	Alma	1966	1,013	383
Augusta City Lake	Augusta	1940	2,358	2,100
Blue Mound City Lake	Blue Mound	1957	---	165
Buffalo City Reservoir	Buffalo	1960	---	1,631
Council Grove City Lake	Council Grove	1942	8,416	7,346
Crystal Lake	Garnett	1879[b]	229	104
Eureka Reservoir	Eureka	1939	3,690	3,125
Fort Scott City Lake	Fort Scott	1959	---	7,200
Gardner Lake	Gardner	1940	2,301	2,020
Harveyville City Lake	Harveyville	1960	235	222
Herington Reservoir	Herington	1982	5,759	5,750
Lake Kahola	Emporia	1936	6,600	5,500
Lake Miola	Paola	1957	2,960	2,760
Louisburg City Lake	Louisburg	1984	---	3,750
Lyndon City Lake	Lyndon	1966	948	930
Madison City Lake	Madison	1970	1,445	1,333
Mission Lake	Horton	1924	1,866	940
Moline Reservoir	Moline	1937	---	1,590
Mound City Lake	Mound City	1979	1,773	1,525
Olathe Lake	Olathe	1957	3,330	3,300
Parsons Lake	Parsons	1938	10,050	8,500
Pleasanton Reservoir	Pleasanton	1968	---	1,180
Polk Daniels Lake	Howard	1935	777	640
Prairie Lake	Holton	1948	---	495
Prescott City Lake	Prescott	1964	138	---
Richmond City Lake	Richmond	1955	---	220
Sedan City South Lake	Sedan	1965	780	770
Severy City Lake	Severy	1938	---	115
Strowbridge Reservoir	Carbondale	1966	3,371	2,902
Thayer New City Lake	Thayer	1960	---	560
Winfield City Lake	Winfield	1970	19,800	19,500
Wabaunsee Lake	Eskridge	1937	4,175	3,600
Wellington New City Lake	Wellington	1996	3,250	3,250
Westphalia Lake	Anderson RWD[c]	1963	278	130
Yates Center City Lake	Yates Center	1990	2,720	2,241

[a] Data source: KWA (2001)
[b] Date incorrectly listed as 1940 in KWA (2001)
[c] RWD = rural water district

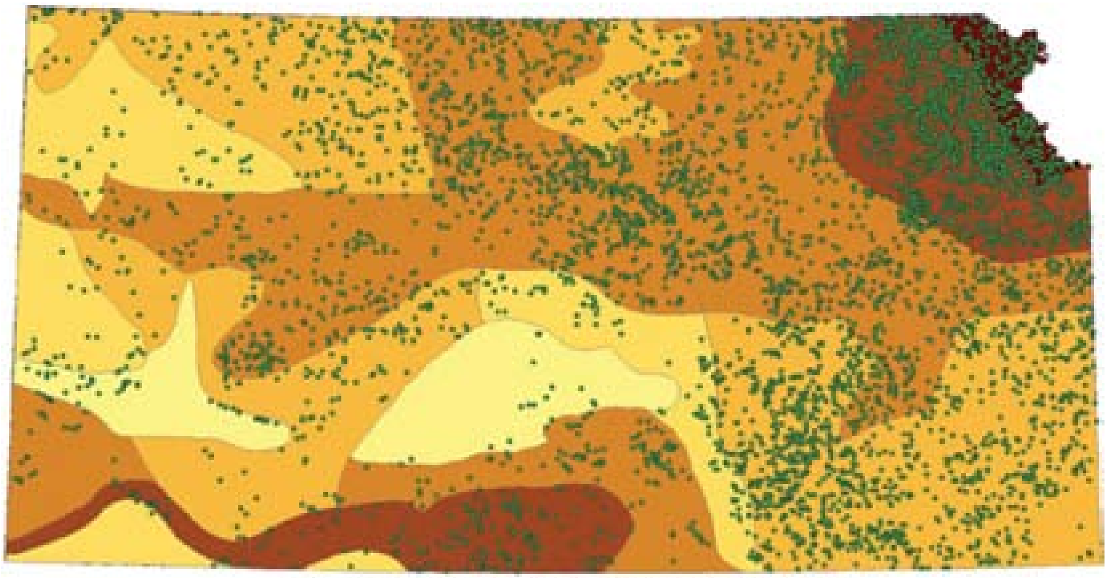


Figure 6. Sediment yield regions in Kansas and 5,847 major Kansas reservoirs
Data analysis and map preparation: Kansas Biological Survey
Sediment map: Collins (1965); Reservoir data: USACE (2005)

Information Needs for Reservoir Management and Restoration

Estimates of Sediment Volume, Mass, Load, and Yield

Effective reservoir sedimentation management requires knowing the amount of sediment deposited (i.e., volume and mass) as well as the rate (i.e., load and yield) at which sediment deposition is occurring. This information provides a baseline to assess changes in sedimentation and the effectiveness of implemented sediment reduction management practices. Federal reservoirs are surveyed most frequently, typically along range lines, and an increasing number of federal reservoirs have been mapped using acoustic echosounding to produce whole-reservoir maps of water depth. However, most of the nearly 6,000 regulated reservoirs in Kansas do not have bathymetric (lake bottom contour) data. Data for state and local reservoirs in Kansas are even rarer, collected on an as-needed or ad-hoc basis, and often incomplete.

Estimates of Reservoir Sediment Trap Efficiency

Reservoir sediment trap efficiency is a measure of the effectiveness of a reservoir for trapping and permanently storing the inflowing sediment load. Trap efficiency typically is greater than 90% for large reservoirs (Brune, 1953; Vanoni, 2006), less for smaller reservoirs. For example, estimated trap efficiency of Perry Lake is 99% (Juracek, 2003). Trap efficiency declines with increasing sedimentation (Morris and Fan, 1998); therefore, obtaining trap efficiency estimates is crucial, especially for reservoirs that are rapidly filling with sediment.

Sediment Quality

Sediment quality is an environmental concern because sediment can act as a sink for various contaminants and, under certain conditions, a source of contaminants for the overlying water column and biota (Baudo et al., 1990; Zoumis et al., 2001). Examples of sediment-associated contaminants include phosphorus, trace elements, certain pesticides, and polychlorinated biphenyls. Once in the food chain, some sediment-derived contaminants pose a greater concern because of bioaccumulation. Even after the source of a particular contaminant is eliminated from a basin, it could take several decades before newly deposited sediment recovers to baseline contaminant concentrations (Van Metre et al., 1998; Juracek and Ziegler, 2006). When considering dredging as a sediment management strategy, it is important to ascertain the quality of reservoir bottom sediment before determining where to store dredged material (Morris and Fan, 1998). Sediment quality information is available for several large and small Kansas reservoirs (Juracek and Mau, 2002; Juracek, 2003, 2004, 2006).

Sediment Sources

Nationally, billions of dollars have been spent over the past several decades to control erosion and mitigate its effects (Pimentel et al., 1995; Morris and Fan, 1998). Determining sediment sources is essential for designing cost-effective sediment management strategies that will achieve meaningful reductions in sediment loads and yields (Walling, 2005). A fundamental question is whether the sediment load in streams originates mostly from erosion of channel banks or surface soils within a basin. Using a combination of several chemical tracers, Juracek and Ziegler (2007) determined that the majority of sediment now being deposited in Perry Lake originated from channel-bank sources.

Sedimentation Dynamics

Repeated bathymetric surveys can provide significant insight into the nature of sedimentation within a reservoir (e.g., is the rate of sedimentation a continuous or episodic process?). Changes in reservoir bottom topography can be monitored over time to provide an overall estimate of the sediment accumulation rate and a spatially explicit representation of sediment accumulation and movement across a reservoir. Similarly, bathymetric surveys before and after major rain events can provide information on whether significant sediment movement occurred.

Sedimentation Patterns

Similar to bathymetric data, sediment thickness information for Kansas reservoirs is limited. Federal reservoirs have the most complete data sets, state and local reservoirs have the poorest. Sediment thickness and volume can be estimated by several direct or indirect approaches (e.g., topographic and acoustic differencing and sediment coring). But even in the best cases, sediment thickness and distribution data likely are limited to a few point samples or transects across a reservoir, which provides a very limited representation of actual sediment accumulation patterns and rates.

Statewide Suspended-Sediment Monitoring Network

A suspended-sediment monitoring network can provide valuable information for managing sediment loads in streams. Information could include instantaneous concentrations, long-term variability, seasonality, and relation to streamflow and turbidity. Moreover, data from the monitoring network could be used to document and explain differences among sites and provide baseline information to assess effectiveness of implemented erosion control practices. A USGS suspended-sediment monitoring network provided data for several sites from the 1950s through the 1980s (Jordan, 1985). However, at present, few if any suspended-sediment data are being collected routinely.

Reservoir Information Systems for Decision Support

Multiple constituencies in Kansas need or desire information from state agencies on water depth, sediment accumulation, and related conditions affecting reservoirs. This need is expressed in many ways: a fisherman desiring a reservoir depth map, a neighborhood association faced with the difficult decision of whether to dredge their reservoir, and state officials grappling with major issues of drinking water quality and quantity in reservoirs.

Critical decisions about reservoir management must be made at numerous times and places across the state, yet information on the current status and trends of Kansas reservoirs is not readily accessible and is dispersed among federal, state, and local entities. This prevents timely and efficient identification of currently impaired reservoirs and reservoirs that could become impaired. No comprehensive database exists to identify these water bodies and determine their size, age, location, proximity to urban areas, current level of impairment, or potential future physical or chemical impairment; and existing data and information are of little use unless accessible to a wide variety of users. Therefore, a reservoir decision-support system should be developed as a resource for Kansans. This system should incorporate a suite of physical, chemical, geospatial, and other data gathered from a variety of sources.

Reservoir Restoration: Issues Related to Sediment Removal

Unique Aspects of Sediment Removal Projects

Removing sediment from a reservoir typically is performed by dredging. Unique among earthmoving projects, dredging requires removing material from beneath a water surface. Excavated material is out of sight of both the contractor and stakeholders until deposited on land. Generally, dredging projects in Kansas involve pumping sediment from the reservoir bottom as a slurry and placing it on land behind levees, which allow water to drain back to the reservoir. It is difficult to quantify the amount of excavated sediment and impossible to determine if removal achieved the desired reconfiguration of the reservoir bottom. The end product of dredging is out of view with only the spoils as evidence of progress and completion.

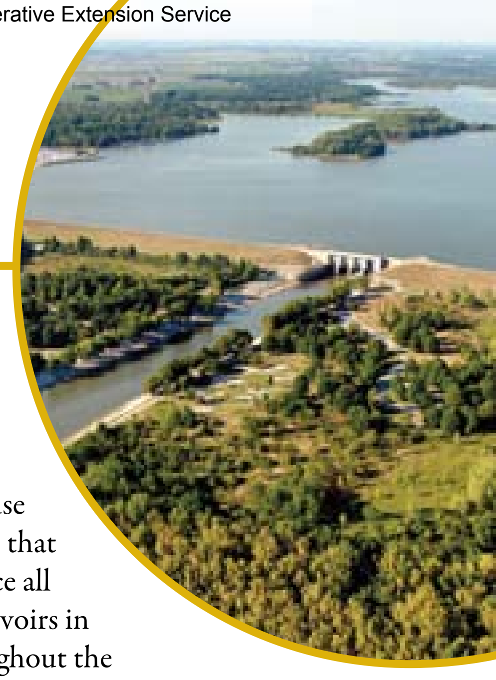
Also unique to dredging is a basin of water (with more water flowing in and out) that is highly disturbed by the excavation process. Observers, particularly those living nearby, expect to see sediment deposits on land. However, they will also witness changes in the reservoir—waters becoming increasingly cloudy, heavier than normal growth of aquatic plants, and impaired fishing and other activities. Failing to identify or address these effects and issues can hinder satisfactory project completion. Examples of potential problems include:

- Inability of stakeholders to adequately develop project goals because they cannot accurately identify the extent and location of sediment
- Higher bids from contractors to cover contingencies because they are not able to adequately assess sediment conditions beneath the water surface
- Stakeholder concerns including unexpected project costs, difficult-to-view progress, and unexpected appearance of site disturbances
- Impeded progress or equipment damage as unexpected rocks, tree stumps, compacted sediment, or other impediments are encountered
- Uncertainty between contractors and stakeholders regarding the new bottom configuration as each area of the reservoir bottom is completed
- Disenchantment among financial investors, particularly citizen stakeholders, due to continuing site disturbances and perceived slow progress
- Disagreements between contractors and stakeholders regarding project completion resulting from contractors judging contract commitments only by rough estimates of removed materials
- Diminished credibility between contractors and stakeholders, whether justified or not, leading to contentious final contract completion settlement
- Lingering questions among stakeholders: Will the reservoir meet future expected needs? Was the investment worth it?
- Dissatisfaction of stakeholders and contractors leading to discouraging projections for the future with no other restoration options available

Managing These Issues

To address the above-mentioned issues and resulting effects, a management plan should be developed based on accurate mapping of the reservoir bottom before, during, and after the sediment removal project. The Kansas Biological Survey, a state agency, can provide this mapping service through a newly developed bathymetric mapping capability. Simultaneously measuring water quality conditions can help address other related issues. State and federal agencies with expertise in measuring particular water and sediment quality conditions of interest can work together to provide this information.

Before sediment removal. Of primary importance are high-resolution (less than 1 square meter) contour maps of the bottom configuration for the entire reservoir and for specific sites. Comparing this information with pre-impoundment contours and selected sediment coring to verify thickness in certain locations will enable stakeholders to develop well-defined project goals and work plans to support the bidding process. All interested contractors can receive clear project goals and an accurate view and quantification of the reservoir bottom contour conditions to be reconfigured. This will minimize unknown factors and encourage preparation of the most accurate, cost-effective bids and most mutually acceptable work plan.



During sediment removal. Excavated sediment can be quantified most accurately with mapped contour changes at each bottom site before and after excavation. During the sediment removal process, bottom configuration information should be available immediately before the dredge moves into a new area and immediately after the new area is completed. Such data allows contractors to more accurately quantify sediment removal continually during the project, a determination that is difficult, if not impossible, to make based only on excavated slurry on land that might still be combined with an undetermined volume of water. Quantifying excavated sediment will improve contractors' sediment removal efficiency and provides contractors and stakeholders an ongoing measure of progress related to the original goals and work plan.

Keeping Stakeholders Informed. Other issues can be addressed by disseminating useful information (e.g., excavation progress and changing water quality) to stakeholders. State agencies should maintain an information network to continually document progress and changing water quality conditions resulting from excavation or water returning from the spoils area. Periodic stakeholder meetings, some on site, should be convened. However, this level of oversight and communication among all parties, particularly dredging contractors who might not have previously worked with this level of stakeholder input, requires conscientious management to ensure continued progress.

Summary

Many reservoirs have been constructed in locations where their lifespans are threatened by natural conditions as well as human land use activities. It is impossible to expect that we could someday restore or replace all these reservoirs. Hundreds of reservoirs in Kansas and thousands more throughout the United States already require restoration or replacement. Eventually, all reservoirs will require some action to maintain, restore, or replace their ability to provide resources as intended. Most reservoirs worldwide were constructed at about the same time (post-1930s) and have similar lifespans. This creates a time period for renovation or replacement that is similarly condensed and too short to ensure successful rehabilitation of all reservoirs. Replacement is difficult because reservoirs have already been constructed in most of the best locations. Raising dam height to compensate for lost water storage is structurally impossible for many reservoirs, and it is not feasible to lose all of the urban development surrounding many reservoirs. Renovation by dredging requires moving material and will cost 15 to 100 times more than original dam construction. Dredging one 7,000-acre reservoir nearly filled with sediment would cost about $1 billion today. We must continue to preserve quality of reservoirs and watersheds with better management until renovation or replacement is feasible. It is imperative that we protect these vital public resources, first by responding to immediate problems affecting water quality and wildlife habitat and then by addressing progressive siltation.

Recommendations

Reservoir management is an enormous task requiring considerable investment. Our actions and procedures must be successful. Therefore, both now and in the future, we should:

- Determine rates of sedimentation by bathymetric mapping, coring, and isotopic dating
- Manage reservoirs, their watersheds, and stream channels more effectively to delay filling
- Address declining environmental quality of water and habitat in reservoirs
- Identify and refine renovation or replacement strategies for particular reservoir situations
- Prioritize particular reservoirs for types of eventual treatments
- Explore alternative water collection, holding, and distribution systems
- Accept that all reservoirs eventually will fill with sediment and prepare to address the consequences

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Additional Resources

ASTRA Initiative, Kansas Biological Survey: http://www.kars.ku.edu/astra

USGS Reservoir Sediment Studies: http://ks.water.usgs.gov/Kansas/studies/ressed/

Methods for Assessing Sedimentation in Reservoirs

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Introduction

Sediment accumulation and other factors continue to create water quality problems that affect the many uses of Kansas reservoirs. The most pressing issue is ensuring the quality of drinking water supplies. Flood control, recreation, irrigation, and other reservoir uses also must be protected, and renovation to ensure reservoirs' long-term viability is becoming increasingly necessary. Solving these problems is an enormous challenge that requires gathering crucial information about physical, chemical, and biological conditions in reservoirs and watersheds. Bathymetric (lake bottom contour) mapping and reservoir assessments are becoming particularly important as federal, state, and local agencies contemplate and initiate sediment management projects to renovate Kansas reservoirs.

Current State of the Science: Bathymetric Mapping

Traditional Approaches to Water Depth Measurement

Information on water depth has been important for thousands of years. Until the 20th century, water depth measurements were obtained manually from the side of a boat with a sounding line and lead weight (Figure 1) or, in shallower waters, a pole with depth markings. Sounding weights and poles often were tipped with an adhesive substance, such as wax or lard, to capture a sample of sediment. The location of each sounding (depth measurement) was determined by estimation or direct measurement in smaller water bodies or harbors and by celestial navigation (sextant or astrolabe) in oceans. Thus, horizontal accuracy

Figure 1. A 19th century sounding boat
Image from NOAA Central Library

of these ad-hoc spot positions generally was low. A measure of control could be imposed in areas where range lines could be established between identifiable landmarks on shore. This permitted repeat visits to sounding positions over time to monitor sedimentation or erosion.

Manual approaches to depth measurement are labor intensive, have relatively low accuracy and precision, and have considerable limitations, particularly for mapping detailed bottom contours or estimating whole-lake sedimentation volumes, rates, and changes. Development of acoustic echosounding systems that use global positioning systems technology for horizontal position location enabled "whole-lake" approaches that build detailed representations of depth contours based on mathematical interpolation of thousands of geographically referenced depth measurements.

Whole-Lake Acoustic Echosounding for Lake Depth (Bathymetric) Mapping

By the 20th century, advances in acoustic science and technology permitted development of sonar systems, originally used for military purposes but adapted for civilian mapping operations. During the past decade, acoustic echosounding systems became sufficiently self-contained and portable, allowing for use even on small lakes and ponds.

Acoustic echosounding relies on accurate measurement of time and voltage. A sound pulse of known frequency and duration is transmitted into the water, and the time required for the pulse to travel to and from a target (e.g., a submarine or the bottom of a water body) is measured. The distance between sensor and target can be calculated using the following equation:

$$D = \tfrac{1}{2}(S \times T)$$

Where D = distance between sensor and target, S = speed of sound in water, and T = round-trip time.

To acquire information about the nature of the target, intensity and characteristics of the received signal also are measured. The echosounder has four major components: a transducer, which transmits and receives the acoustic signal; a signal generation computer, which creates the electrical pulse; the global positioning system, which provides precise latitude/longitude coordinates; and the control and logging computer. Typical acoustic frequencies for environmental work are:

- 420 kHz – plankton, submerged aquatic vegetation
- 200 kHz – bathymetry, bottom classification, submerged aquatic vegetation, fish
- 120 kHz – fish, bathymetry, bottom classification
- 70 kHz – fish
- 38 kHz – fish (marine), sediment penetration

Prior to conducting a bathymetric survey, geospatial data (including georeferenced aerial photography) of the target lake are acquired, and the lake boundary is digitized as a polygon shapefile. Transect lines are predetermined based on project needs and reservoir size. Immediately before or after the bathymetric survey, elevation of the lake surface is determined. For large reservoirs (e.g., U.S. Army Corps of Engineers or Bureau of Reclamation lakes), elevation is determined using local gages. For smaller reservoirs that are not gaged, a laser line is established from a surveyed benchmark to the water surface at the edge of the lake.

System parameters are set after boat launch and echosounder initialization. Water temperature at a depth of 1 to 2 meters is recorded (°C) and used to calculate the speed of sound in water for the given temperature and depth. A ball check is performed using a tungsten-carbide sphere, which is supplied specifically for this purpose with each transducer. The ball is lowered to a known distance below the transducer face. The position of the ball in the water column (distance from the transducer face to the ball) is clearly visible on the echogram, and the echogram distance is compared with the known distance to ensure parameters are set properly.

A typical survey procedure for smaller lakes is to run the perimeter of the lake, maneuvering as close to shore as permitted by boat draft, transducer depth, and shoreline obstructions to establish near-shore lake bottom dropoff. Then, predetermined transect patterns are followed, and data are automatically logged by the echosounding system.

Raw acoustic data are processed through proprietary software to generate ASCII point files of latitude, longitude, and depth. Point files are ingested to ArcGIS and merged into a master point file, and bad points and data dropouts are deleted. Depths are converted to elevations of the lake bottom based on the predetermined lake elevation value. Lake bottom elevation points are interpolated to a continuous surface by generation of a triangulated, irregular network or simple raster interpolation. Elevation of the digitized lake perimeter is set to the predetermined value and used in the interpolation as the defining boundary of the lake. Then, area-volume-elevation tables can be computed from the lake bottom surface model.

Current State of the Science: Sediment Classification and Thickness Assessment

Acoustic Characterization of Sediment Types

The acoustic echosounding system has a proprietary software suite that classifies reservoir bottom sediment (e.g., rock, sand, silt, or mud) based on characteristics of the acoustic return signal (Figure 2). Ideally, this process would be used to collect acoustic data from known bottom types to provide a "library" of Kansas-specific classification data. Sediment sampling and coring also provide bottom composition data for calibration and accuracy assessment.

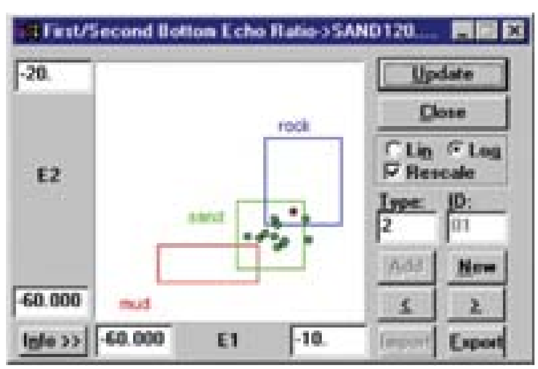

Figure 2. Acoustic signal classification for bottom type mapping
Image courtesy of Mark Jakubauskas, Kansas Biological Survey

Approaches for Estimating Sediment Thickness

Estimating thickness of accumulated sediment in a reservoir is not a simple process. Three techniques—sediment coring, topographic differencing, and acoustic estimation—show promise for estimating the spatial distribution, thickness, and volume of accumulated sediment in Kansas reservoirs. Each technique has strengths and limitations, and an ideal methodology uses all three approaches in concert to calibrate and cross-check results.

Sediment Coring. Sediment cores typically are taken from a boat using a gravity corer or vibrational coring system. In either case, an aluminum, plastic, or steel tube is forced into the sediment, ideally until pre-impoundment substrate is reached. The tube is withdrawn and sliced longitudinally, or the sample is carefully removed from the tube, allowing for sediment thickness measurement and sample collection. The interface between pre-impoundment substrate and post-impoundment sediment is fairly distinct in Kansas lake sediment samples (Figure 3).

Several companies manufacture and distribute sediment coring systems. However, most systems are intended for deep water marine use in the ocean and are not suitable for smaller, shallower lakes and reservoirs. Sampling inland reservoirs requires a portable, self-contained unit with an independent power supply that is small enough to fit on an outboard motorboat or pontoon boat, which disqualifies pneumatic, hydraulic, or high-voltage systems commonly used on larger marine vessels. Smaller systems have been developed and are used in Kansas (e.g., the VibeCore System, Specialty Devices Inc., Texas).

Figure 3. Sediment core from Mission Lake in Brown County, Kansas, showing pre-impoundment substrate (left) and post-impoundment sediment (right)
Photo courtesy of Kansas Biological Survey

A benefit of the sediment coring approach is that cored material can be preserved and analyzed for sediment classification or chemical composition. However, core sampling is time and labor intensive; only a small number (≈10 to 25) of point samples can be taken per day. Although sediment core data are likely highly accurate for a given location, the overall result is an incomplete and fragmentary representation of sediment thickness and volume across the reservoir.

Topographic Differencing. The topographical approach computes the difference between pre-impoundment and present-day lake bottom topographic data and uses that information to create a spatially-explicit, three-dimensional representation of sediment accumulation (Figure 4). Data from archived topographic maps, reservoir blueprints, or "as-built" pre-impoundment topographic surveys are used to create a pre-impoundment surface, and data from new bathymetric surveys are used to create a map of current reservoir bottom topography. Unlike spot measurements of sediment thickness, topographic differencing can display a "whole-lake" representation of sediment accumulations, facilitating estimates of sediment volume (Figure 5).

However, quality of sediment thickness data produced by this approach depends on quality of data used to create pre-impoundment maps. Archival topographic data can have one or more of the following limitations: no information on horizontal or vertical projection of data used, referenced to an arbitrary local elevation (i.e., nonstandard/nongeodetic vertical control), or of inappropriate spatial scale to produce meaningful comparisons with present-day topographic data.

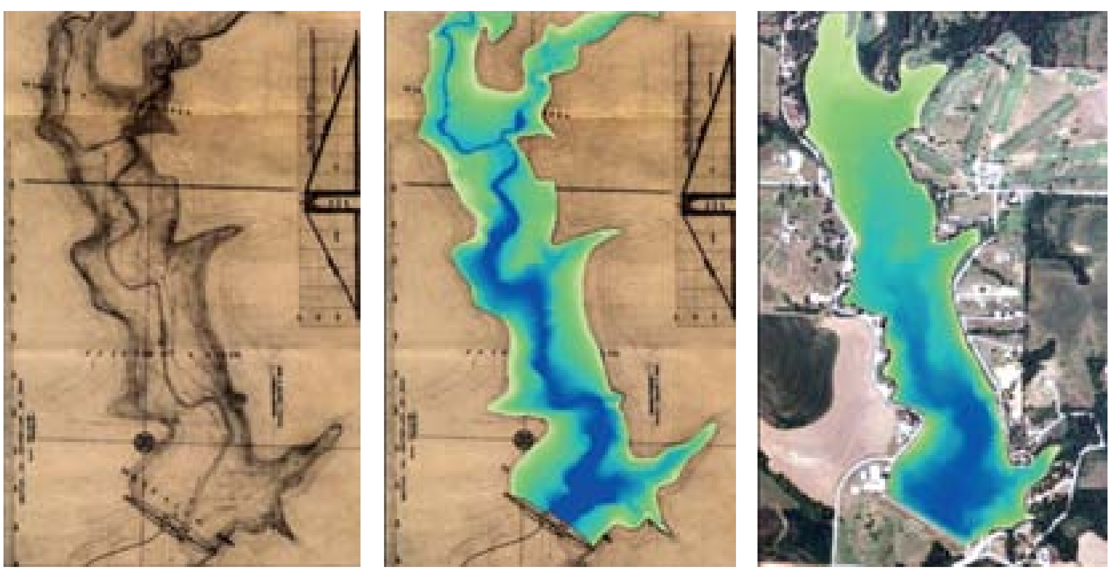

Figure 4. Topographic differencing of pre-impoundment and present-day reservoir topography
Left: 1923 engineering contour map of Mission Lake in Brown County, Kansas; Center: Digital elevation model created from 1923 map; Right: Present-day lake bottom topography created from analysis of acoustic echosounder data.
Images courtesy of Kansas Biological Survey

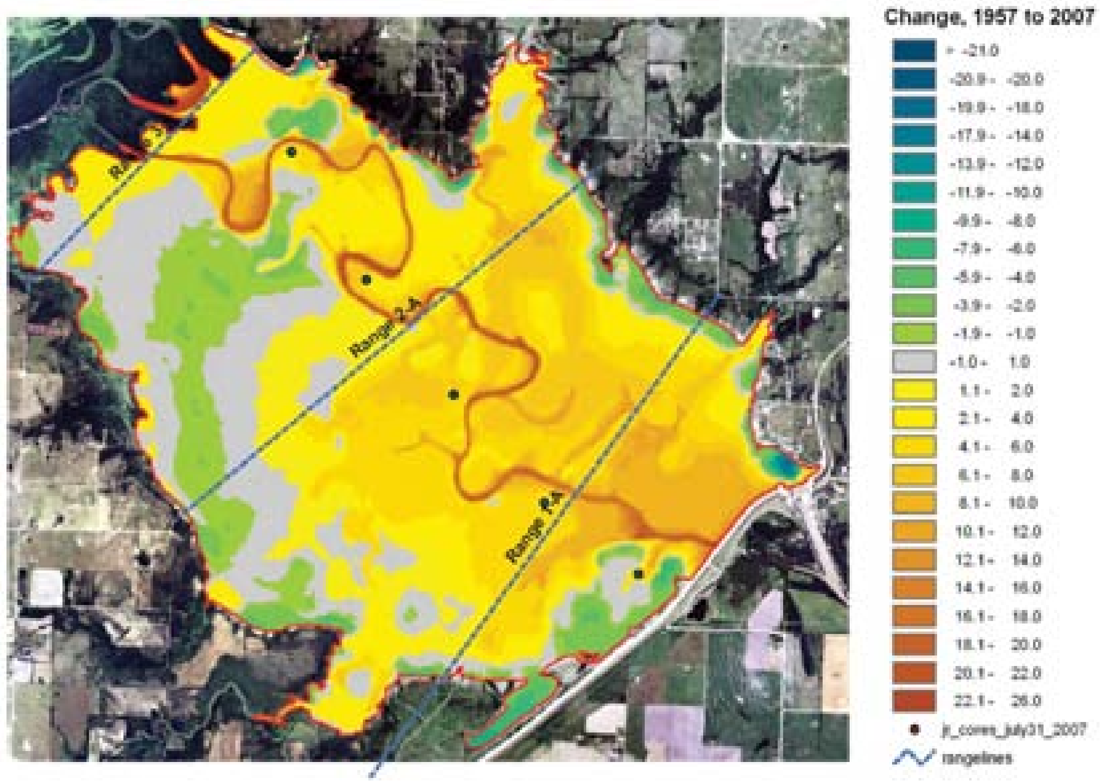

Figure 5. Elevation map of John Redmond Reservoir showing the difference between 2007 bathymetric survey data and a 1957 U.S. Army Corps of Engineers topographic map
Negative numbers indicate loss of material during the 50-year period; positive numbers indicate accumulated material (siltation).
Image courtesy of Kansas Biological Survey

Acoustic Estimation. In the acoustic approach, high-frequency and low-frequency transducers (200 kHz and 38 kHz) are operated simultaneously during a lake survey. Differencing acoustic returns from high and low frequencies (reflecting off the current reservoir bottom and the pre-impoundment bottom, respectively) have shown considerable promise for successful sediment thickness mapping in inland reservoirs (Figure 6; Dunbar et al., 2000).

> Our results indicate that mapping the base of sediment acoustically works best in reservoirs that are dominated by fine-grained deposition (clay and silt, rather than silt and sand). Reservoirs with fined-grained-deposition fill from the dam towards the backwater and no delta forms at the tributary inlet. As long as the water depth is greater than the sediment thickness, the base of sediment can be mapped without interference from the water-bottom multiple reflection, and the entire reservoir can be surveyed from a boat. Coarse-grained dominated reservoirs fill from the backwater towards the dam and form deltas in the backwater. In the time [sic] the backwater region cannot be surveyed, because it is dry land. In these cases, the only option is differing the bathymetry. (John Dunbar, personal communication, 2007)

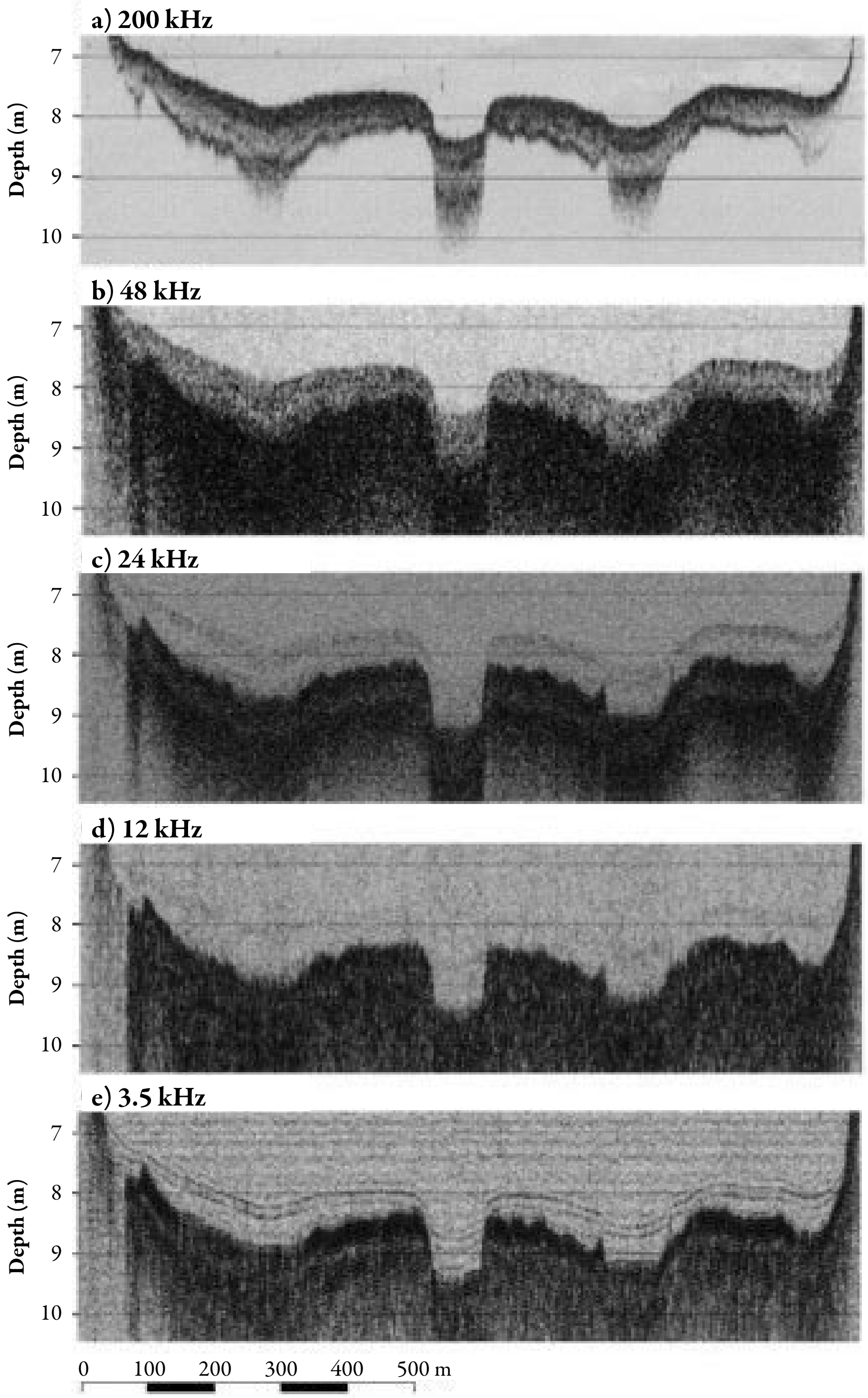

Figure 6. Echograms of acoustic reflectance at multiple frequencies for reservoir sediments: a) High frequency, showing strong discrimination of sediment-water interface; b through e) Increasing penetration of post-impoundment sediments and increasing return from pre-impoundment substrate with progressively lower frequencies.
Figure reprinted from Dunbar et al. (2000) with permission

Information Needs

Information needs related to lake bathymetry and reservoir assessment can be divided into two broad categories: 1) reservoir-scale information needs, which can be satisfied by applying bathymetric technology in an integrated reservoir assessment program, and 2) technology-specific information needs that explore strengths and limitations of bathymetric technology. Crucial information is lacking, and many questions remain.

Reservoir-Scale Information Needs

What is the current depth and volume of the state's reservoirs?

Bathymetric data are not available for a majority of the more than 5,000 regulated reservoirs in Kansas. A review of 18 federal reservoirs in Kansas showed that average time since last bathymetric survey was 15 years (USGS, 2008), but an increasing number of federal reservoirs have been mapped using acoustic echosounding to produce whole-lake maps of water depth. Bathymetric data for state and local lakes in Kansas are even more rare, collected on an as-needed or ad-hoc basis, and often incomplete.

How much and where has sediment accumulated in a given reservoir? Like bathymetric data, sediment thickness information is limited in Kansas. Federal reservoirs have the most complete data sets, state and local reservoirs have the poorest. Even in the best cases, sediment thickness and distribution data likely are limited to a few point samples or transects and thus provide a very limited representation of actual sediment accumulation patterns and rates.

What is the rate of sedimentation, and is sedimentation continuous or episodic? Repeated bathymetric surveys provide significant insight into the nature of sedimentation in a reservoir. Changes in reservoir bottom topography can be monitored over time, allowing an overall estimate of the rate of sediment accumulation and a spatially explicit representation of sediment accumulation and movement across a reservoir. Bathymetric surveys before and after major rain events can provide information on whether significant sediment movement occurred.

Technology-Specific Information Needs

To better understand data produced by bathymetric surveying, research should be conducted to explore strengths and limitations of this technology. Answering the following questions can help improve speed, accuracy, and precision of data acquisition, which is necessary for making informed decisions about reservoir management and renovation.

Topographic and acoustic sediment thickness estimation techniques

- What are the possible sources of error of this approach?
- What are the effects of sediment composition on estimating sediment thickness?
- What are the limitations to identifying the pre-impoundment bottom contour in acoustic data?
- What are the effects of scale (horizontal and vertical resolution) on accuracy?

- What spatial error results from differences between pre-impoundment published topographic data and "as-built" topographic conditions?

Processing acoustic data for bathymetric and sediment surveying

- What are the optimal interpolation algorithms, in terms of speed, accuracy, and precision, for bathymetry and sediment thickness estimation?
- Can advanced signal processing of acoustic echosounder data accurately identify pre-impoundment lake bottom traces?
- Can advanced signal processing of acoustic echosounder data coupled with an "acoustic library" of Kansas reservoir substrate signatures improve bottom type classification?

Mapping and Assessment Program

A long-range bathymetric mapping and reservoir assessment program for Kansas will have numerous benefits. Decision makers will be able to easily assess current conditions of a given reservoir and identify and prioritize reservoirs based on sediment load and need for renovation. Enhanced knowledge of sediment deposition in reservoirs will help determine effectiveness of watershed protection practices. When dredging appears to be the best alternative to extend the life of a reservoir, sediment deposition data will indicate how much sediment needs to be removed and can help determine how much was removed by the dredger. Such a program should contain the following elements:

Sustained Reservoir-Mapping Program

These surveys will provide a set of baseline bathymetric elevations and sediment data. One advantage is that water quality and bathymetric data can be measured simultaneously from the same boat. Also, because surveys will be conducted with the same equipment and methods, it will be possible to compare results among reservoirs and from the same reservoir over time.

Change Detection Studies

These studies would involve revisiting previously mapped reservoirs, re-mapping the bathymetry and cores, and comparing past and present maps to identify sedimentation locations and rates. This element likely will not occur during the first few years of the program but eventually could grow into a major focus as baseline bathymetric and sediment data are accumulated for comparison.

Before/After Mapping, Coring, and Sediment Estimation

Comparing high-resolution contours of bottom topography with pre-impoundment topography and selected sediment coring to verify thickness in certain locations will enable stakeholders to develop well-defined project goals and work plans. Dredging contractors can receive an accurate representation of reservoir bottom contours to be reconfigured. This will minimize unknown factors and encourage preparation of the most accurate and cost-effective bids and the most mutually acceptable work plan.

Ad-hoc Mapping of Small Reservoirs

In this capacity, the program can provide timely, unbiased, impartial bathymetric data and sediment estimates to help local stakeholders make management decisions relating to water quality, watershed management, and reservoir renovation.

Large-Scale Mapping and Sediment Studies

Because of the intensive effort required and large amount of data generated, we envision this program mapping four to six federal-size reservoirs per year.

Reservoir Information System

Multiple constituencies in Kansas need or desire information on water depth, sediment type, sediment accumulation, and related conditions affecting reservoirs. However, data and information are of little use unless readily and easily accessible to a wide variety of users.

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Concept for a Long-Range Bathymetric Mapping and Reservoir Assessment Program

- Sustained reservoir-mapping program that includes a number (≈10 to 20) of bathymetric and coring surveys per year
- Change detection studies to estimate rates and locations of sediment accumulation
- Before/after bathymetric mapping, coring, and sediment volume estimation for reservoir dredging projects
- Ad-hoc bathymetric mapping of small reservoirs for state, local, and private entities
- Large-scale federal reservoir bathymetric mapping and sediment studies
- Development of a reservoir information system

Effects of Sedimentation on Biological Resources

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Summary

Sedimentation is a natural process, but too much sediment in aquatic ecosystems can cause loss or impairment of fish, macroinvertebrates, and other aquatic organisms. Our current ability to quantify relationships among aquatic sediment variables and aquatic biota in the Central Plains is limited by available data and the complexity of direct and indirect linkages between resource components. At present, turbidity appears to be the best indicator of suspended sediment for defining biological impairment in flowing water systems. Better coordination of existing and new research, use and analysis of well-selected indicators of suspended and deposited sediment and ecosystem function, and advanced statistical analyses will allow us to more accurately identify and quantify effects of sediment on aquatic ecosystems in Kansas.

Introduction

Water from streams and rivers is used for drinking, irrigation, waste dilution, power generation, transportation, and recreation and provides habitat for fish and other aquatic organisms (Allan, 1995). This water also contains sediment (e.g., eroded soil particles), which can be either suspended in the water or deposited on the bottom. Sedimentation is the process by which sediment is transported and deposited in aquatic ecosystems.

In-stream sediments come from two sources: runoff from surrounding areas and erosion from both the sides and bed of the channel. The complex interaction of streams and the surrounding landscape can be characterized to a large extent by describing sediment movements. Erosion and sediment deposition affect many stream characteristics including channel depth, channel shape, substrate, flow patterns, dissolved oxygen concentrations, adjacent vegetation, and aquatic communities (Leopold et al., 1964; ASCE, 1992; OMNR, 1994; Rosgen, 2006).

Sedimentation is a natural process that occurs in most aquatic ecosystems, and sediment-borne organic materials provide the primary food source for a number of filtering macroinvertebrates (Waters, 1995; Wood and Armitage, 1997). However, human activities such as urbanization, agriculture, and alteration of riparian habitat and flow regimes have increased the concentrations and rates at which sediment enters streams and rivers (Wood and Armitage, 1997; USEPA, 2000; Zweig and Rabeni, 2001; Angelo et al., 2002); and losses of habitat, biota, and ecosystem services due to sediment have caused severe socioeconomic impacts (Duda, 1985). As a result, sedimentation is listed as one of the most common stream impairments in the United States (USEPA, 2000, 2004), occurring in almost one-third of the river and stream miles recently assessed by the U.S. Environmental Protection Agency (USEPA; 2004).

Increased sedimentation and sediment loading are also threatening the ecological integrity of other aquatic systems. For example, sedimentation at higher than normal rates can reduce or impair habitat and primary production in wetlands (Gleason and Euliss Jr., 1998; USEPA, 2002; Gleason et al., 2003). Similar habitat reduction has been observed in lakes; several Kansas reservoirs are experiencing 10% to 40% decreases in conservation-pool water-storage capacity. If sedimentation continues at current rates, sediment pools of these reservoirs will be filled by the 2020s (Juracek, 2006). In other reservoirs (e.g., Perry, Tuttle Creek), increased sedimentation is occurring primarily in the riverine upper reaches, reducing both quality and quantity of habitat.

Both “clean” and “dirty” sediment directly and indirectly affect the structure and function of all aquatic ecosystems (Figure 1). Clean sediment is free from additional contaminants (e.g., volatile organics, metals, or other toxic compounds), and dirty sediment harbors these materials. Effects of dirty sediment are due to the nature and concentration of both sediment and contaminants, whereas effects of clean sediment are due to the nature and concentration of sediment particles alone. Duration of exposure is also important. In the environment, clean and dirty sediments constantly interact as contaminants are added, broken down, and removed. Because both sediment types occur simultaneously, clean and dirty sediment effects are difficult to separate. To begin understanding sediment interactions, this white paper focuses on effects of clean sediment.

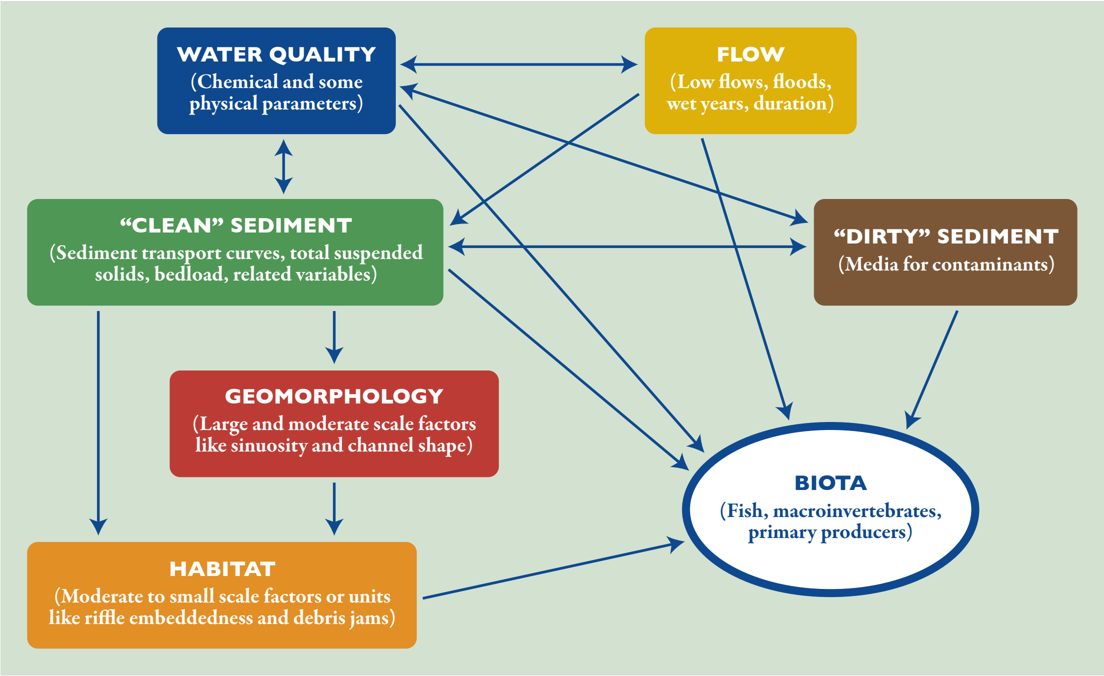

Figure 1. Conceptual framework showing interactions of sediment in aquatic ecosystems
Boxes illustrate important ecosystem units with examples, and arrows represent functional directions and links between those units. Ultimate goals are to understand the links and quantify sediment effects on biota.

Although effects of sedimentation are widespread, a comprehensive theory of these effects on benthic communities does not currently exist (Zweig and Rabeni, 2001). Appropriate management of aquatic ecosystems in Kansas requires improving our ability both to more accurately quantify relationships among aquatic sediment variables and aquatic biota and to distinguish between natural and anthropogenic sediment loading in this region. As a first step in that process, this white paper summarizes current knowledge and provides recommendations for future research.

State of the Art: Review of Science to Date

Brief Literature Review

Most sedimentation research focuses on cold water systems. Representative works include basic research studies (Luedtke and Brusven, 1976; Erman and Ligon, 1988; Lisle and Lewis, 1992; Goodin et al., 1993; Maund et al., 1997; Simon et al., 2003; Dodds and Whiles, 2004), literature reviews (Cordone and Kelley, 1961; Foess, 1972; Newcombe and MacDonald, 1991; Doisy and Rabeni, 2004), and books (Ford et al., 1990; Waters, 1995). Previous studies report both direct and indirect effects of sedimentation. Direct physical effects include light interruption; smothering of organisms; and coverage of sites used for germination, feeding, spawning, and other activities. Biotic effects include direct mortality; reduced fecundity; reduced disease resistance; and inhibited feeding, growth, and reproduction. Reviews by Newcombe and MacDonald (1991) and Doisy and Rabeni (2004) have also grouped direct biotic effects into three categories:

- Lethal effects, which cause direct mortality of organisms, reduce populations, or damage ecosystem capacity for production
- Sublethal effects, which injure organism tissues or cause physiological stress, both without causing mortality
- Behavioral effects, which alter the activity of affected organisms

Both suspended and deposited sediment particles can affect aquatic ecosystems (Waters, 1995; Zweig and Rabeni, 2001; Richardson and Jowett, 2002). For example, increased suspended solid concentrations can reduce primary production (Van Nieuwenhuyse and LaPerriere, 1986), disrupt feeding and respiration rates of macroinvertebrates (Lemly, 1982), and reduce growth and feeding rates of many stream fish (Wood and Armitage, 1997). Both intensity (concentration) and duration (time of exposure) of suspended sediment loading contribute to biological impairment, and models that consider both are better predictors of impairment than models that use either intensity or duration alone (Newcombe and MacDonald, 1991). Increased sediment deposition can reduce the complexity of stream habitat (Allan, 1995) and smother aquatic organisms including macroinvertebrates, fish, and macrophytes (Waters, 1995; Wood and Armitage, 1997).

In addition to the abundance of studies on cold water systems, the majority of stream sediment research has been conducted in systems with either a naturally high gradient (i.e., steep downhill slope) or naturally low turbidity (Dodds et al. 2004). However, aquatic systems in the Central Plains—especially those in agriculturally dominated areas like the Central Great Plains, Western Corn Belt Plains, and,

to a lesser extent, the Central Irregular Plains—generally are characterized as warm water, low-gradient (i.e., mild downhill slope), high-turbidity systems, though early reports suggest that many Central Plains streams that have been turbid for the past 100 years might have been clear prior to widespread plowing in the region (Matthews, 1988). Compounding the issue, many systems in the Central Plains have sand as the natural substrate (Angelo et al., 2002). In biological terms, sand-bottom streams are different than streams with either large substrates (e.g., bedrock, cobble, gravel) or fine substrates (e.g., silt, mud, muck) because they provide different structural and chemical characteristics that affect aquatic life. Sand-bottom systems can have significant movement of sand in the channel bed (i.e., high bedload) under natural conditions. However, induced loading of silt or mud can still impair sand-bottom streams (Angelo et al., 2002). Distinguishing between natural sediment loading and induced sediment loading in these systems can be very difficult; significant regional testing is required to understand how anthropogenically altered sediment loading can affect aquatic ecosystems.

Because of the need for regional testing and current lack of a comprehensive theory of sediment effects, a sediment workgroup sponsored by USEPA Region VII developed a conceptual framework for interactions of sediment in lotic (i.e., flowing water) ecosystems (Figure 1). This framework provides hypothesized direct and indirect linkages among both clean and dirty sediments, geomorphology, flow regimes, chemical and physical water quality parameters, habitat effects, and biotic components including primary producers, macroinvertebrates, and fish.

Recent Regional Findings

To analyze complex systems, it often is necessary to construct linked individual relationships to depict indirect effects. Statistically significant relationships between indicators (i.e., representative, measurable components of the ecosystem) form the links. For example, effects of clean sediment (i.e., sediment only, without associated nutrient or chemical loading considerations) on biology can be modeled by relating a sediment loading indicator (e.g., total suspended solids) to a water quality indicator (e.g., turbidity) then relating that water quality indicator to a biological one (e.g., number of fish species) (Figure 2). Additional indirect effects are modeled in a similar fashion.

A variety of potential sediment and erosion indicators exist. USEPA uses water column indicators (e.g., suspended sediment, bedload sediment, and turbidity), streambed indicators (e.g., streambed particle size and embeddedness), and riparian indicators (e.g., buffer size and vegetation community composition) to set criteria for allowable loading of induced sediment (i.e., Total Maximum Daily Loads for sediment; USEPA, 1998). Several biological indicator groups such as macroinvertebrates and fish also respond to sediment-related effects (Luedtke and Brusven, 1976; Culp et al., 1986; Richards and Bacon, 1994; Rier and King, 1996; Birtwell, 1999). However, except for a study by Whiles and Dodds (2002), linkages between sediment indicators and biological indicators both within and between streams in the Central Plains remain largely undocumented.

Sediment–Water Quality Links.

Using data from more than 500 samples in 16 small watersheds throughout the West-

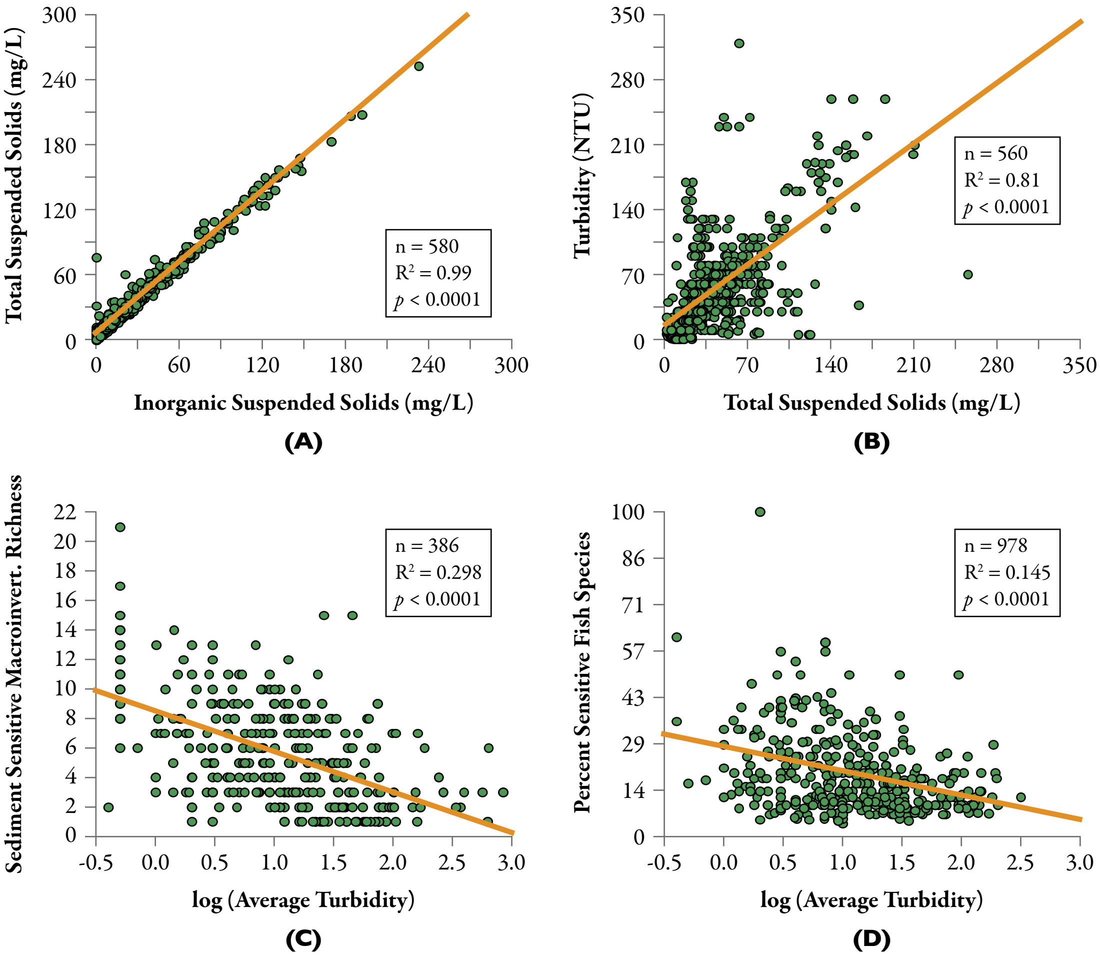

Figure 2. An example of relating sediment effects to biological responses using indirect effects
(A) Inorganic suspended solids, a measurement of the amount of mineral sediment particles floating in the water column, is related to total suspended solids, a measure of the amount of all particles (both inorganic and organic) floating in the water column. (B) Total suspended solids is related to turbidity, and turbidity is related both (C) to the number of macroinvertebrate taxa that are known to be sensitive to sediment and (D) to the percentage of fish species that are known to be sensitive to sediment. In this example, we relate "clean" sediment (e.g., inorganic suspended solids and total suspended solids) to biological responses (e.g., sediment sensitive macroinvertebrate richness and percentage of sensitive fish species) via the indirect effects of water quality (e.g., turbidity). Additional and more complicated analyses follow this general concept.

ern Corn Belt Plains, the Central Plains Center for BioAssessment (CPCB; 1994) found that inorganic suspended solids (ISS) explained 99% of the variation in total suspended solids (TSS) and that TSS explained 81% of the variation in turbidity. The USEPA Region VII Regional Technical Assistance Group (RTAG) found that TSS explained 98% of the variation in turbidity for more than 13,800 sites throughout the Central Plains and across ecoregions (RTAG, 2006); and Dodds and Whiles (2004), using nationwide data, found that TSS explained 89% of the variation in turbidity. Because turbidity is highly correlated with TSS and, by extension, ISS, turbidity measurements can be used as a surrogate indicator for suspended clean sediment in streams in the Central Plains.

Sediment/Water Quality–Biota Links. In most biological systems, greater diversity of organisms implies better or "healthier" environmental conditions. Models developed using RTAG (2006) data suggest that macroinvertebrate richness (i.e., number of unique taxonomic groups of macroinvertebrates) significantly declines with increasing turbidity. Such declines usually are associated with impairment or decreasing environmental quality. However, statistical analysis and modeling determined a "threshold range" of turbidity levels between 10 and 25 NTU above which macroinvertebrate richness drops very little. Even though turbidity can, and often does, increase significantly beyond this threshold range (the average turbidity level of 125 Central Plains streams is 42 NTU), relatively few taxa are lost, presumably because some ecological limit of turbidity impairment has already been reached. In other words, increased turbidity has changed ecosystem function or structure (or both) such that more turbidity does not elicit a biological response. Lack of response could be because the sensitive species are gone because of death or emigration or because the ecology has been altered to a new state that cannot be further degraded by turbidity. As a corollary, reduction of turbidity might not result in a significant increase of taxa unless turbidity is reduced below the threshold range. Such threshold ranges often are used as the basis for developing benchmarks and criteria for other types of impairments (e.g., nutrient loading).

Regional RTAG (2006) data also revealed that the taxa richness of three typically habitat-sensitive orders of aquatic insects (i.e., Ephemeroptera, Plecoptera, and Trichoptera [EPT]) and the percentage of sediment-sensitive fish also declined with increasing turbidity. Data collected during the National Wadeable Streams Assessment (USEPA, 2004) from 125 sites in Kansas, Nebraska, Iowa, Missouri, and Oklahoma showed similar trends. Total macroinvertebrate and EPT taxa richness both decreased with increasing TSS. Richness of EPT taxa and macroinvertebrate scrapers (i.e., macroinvertebrates that scrape their food off substrates) decreased as the percentage of fine substrates (i.e., silt or mud but not sand) increased, but taxa richness of macroinvertebrate shredders (i.e., macroinvertebrates that shred larger particles for food) and macroinvertebrate predators (i.e., macroinvertebrates that eat other macroinvertebrates) were generally unaffected by changes in the percentage of fine substrates. Three things are important to note about these relationships. First, evidence for impairment is consistent across many ecological and taxonomic groups because increasing sediment loading correlates with decreasing diversity. Second, though the relationships are significant, the amount of variance in biological indicators explained by changes in sediment indicators alone is relatively low (10% to 30%). Advanced statistical techniques might allow us to better understand the complexity of these relationships. Third, some groups (e.g., macroinvertebrates as a whole, EPT taxa, scrapers) are more impaired by increased sediment loads than others (e.g., shredders and predators); this is consistent with a priori expectations based on known ecology of the organisms.

Habitat–Sediment/Biota Links. Current data and quantification of interactions between small-scale habitat indicators and both sediment and biology are limited. One commonly measured habitat indicator,

"percent embeddedness," is the degree to which sediments fill spaces around rocks, gravel, and other substrates at the bottom of water bodies. When these spaces fill with sediment, they can no longer provide habitat or shelter for fish and macroinvertebrates. Data from the National Wadeable Streams Assessment (USEPA, 2004) revealed that percent embeddedness explained about 12% of the variation in turbidity and 26% of the variation in percentage of fine substrates present. As expected, total macroinvertebrate richness and EPT taxa richness declined as percent embeddedness increased (USEPA, 2004), but the amount of explained variation in richness was limited (13% and 10%, respectively).

Geomorphology–Sediment/Habitat/Biota Links. Geomorphology is the measure of the physical structure and geometry of streams and rivers. Geomorphic variables include reach-scale indicators (e.g., reach length, number and length of riffles, sinuosity or "curviness") and channel-scale indicators (e.g., channel depth, channel width, cross-sectional area). Differences in scale make it difficult to relate some ecosystem units (e.g., geomorphology) to others (e.g., habitat, biota) (see Fausch et al., 2002, for a general overview). Although geomorphology can be important for describing particular aspects of streams and rivers, more research is required to relate these aspects to smaller-scale indicators of sediment, habitat, and biota in the Central Plains. For example, though Dauwalter et al. (2007) found that substrate type and geomorphology were related to increased smallmouth bass density, the streams they examined were cold water, high-gradient, low-turbidity streams in the Boston Mountain, Ouachita Mountain, and Ozark Highland areas of eastern Oklahoma, which are not representative of the majority of streams in the Central Plains. Analysis of 53 geomorphic variables from 16 stream reaches throughout Kansas showed no statistical correlation with any indicators of sediment, habitat, or biota. Better understanding of scale (i.e., reach-scale vs. channel-scale vs. site-scale measurements) and advanced statistical techniques (e.g., principal components analysis, regression trees) are required for regional explanations of sediment links with geomorphology, habitat, and biology.

Conclusions

Effects of sedimentation in low-gradient aquatic systems are complex and difficult to measure directly. Often, surrogate variables are required to relate different ecosystem components such as habitat, biota, water quality, clean and dirty sediment, geomorphology, and flow. Based on Kansas and regional data, turbidity appears to be a reliable and easily measurable indicator for clean sediment in lotic systems throughout the Central Plains.

Although data indicate that increased sediment has a negative effect on many biological variables, regional data are limited, direct relationships are statistically weak, and indirect relationships are difficult to quantify (Figure 3). In addition, factors other than sediment might contribute to these relationships. Therefore, depiction of direct and indirect sediment effects via a hypothetical framework coupled with advanced statistical analyses such as multiple linear regression, principal components analysis, regression trees, and quartile regression (Koenker, 1995, 2005; Cade

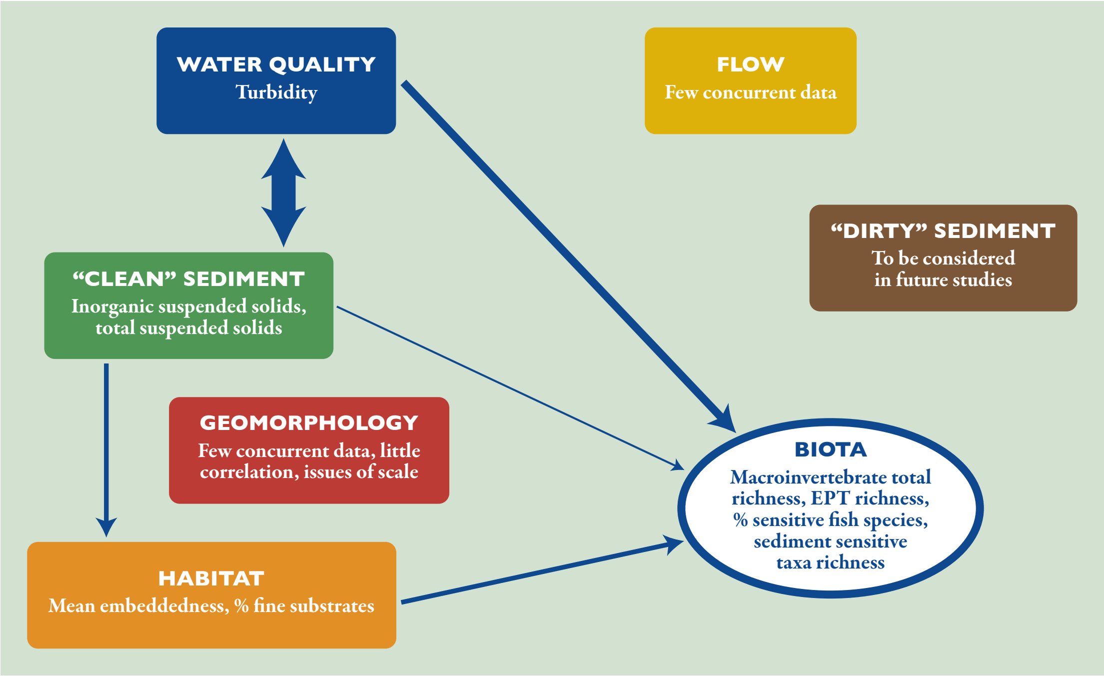

Figure 3. Conceptual framework showing observed effects of sediment on aquatic biota
Boxes illustrate important ecosystem units, and arrows represent functional directions and links observed in this study. Specific indicators used for each ecosystem unit are listed. Relative weights of the arrows indicate relative strengths of relationships observed in this study.

and Noon, 2003) might lead to a better understanding and quantification of complex sediment-biota relationships. Better understanding leads to better management, including more effective interventions and better estimates of socioeconomic losses associated with sediment impairment of aquatic ecosystems.

Acknowledgments

This white paper is based on Report no. 146 of the Kansas Biological Survey, "Effects of Sedimentation on Biological Resources" (Huggins et al., in press), which contains detailed descriptions of recent regional findings including additional figures, tables, and analyses. Kansas Biological Survey reports and other technical publications are available at: http://www.kbs.ku.edu/larc/tech/html/default.htm.

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Recommendations for Future Research

We offer the following recommendations, based on reviews of available literature and recent research results, to guide future research on the effects of sedimentation on biological resources:

- ***Adopt a multidisciplinary approach.*** The complex nature of sedimentation spans topics including hydrology, geomorphology, aquatic ecology, water chemistry, soil and sediment chemistry, and landscape-level phenomena (e.g., urban development and agriculture). Usually, sediment studies are approached from only one or two of these points of view.
- ***Observe both sediment loading and biological response.*** Surprisingly, little overlap exists between datasets on sediment loading and biological indicators. Future studies should emphasize concurrent collection of physical, chemical, geomorphic, and biological data to gain a more comprehensive understanding of complex and integrated relationships.
- ***Begin with gaged locations.*** Often, sediment loading rates are the limiting factor in a multidisciplinary suite of sediment data. To better estimate effects of sediment on biological resources, those resources should be evaluated at locations where sediment loading data is available. Typically, stream gaging stations provide available loading data or opportunities to calculate sediment loads.
- ***Determine reference conditions for sedimentation.*** To evaluate the extent of sedimentation effects on biological resources (i.e., how "good" or "bad" a site is), a condition of high quality must be established for comparison. Currently, there is little agreement among hydrologic, geomorphic, and biological definitions of this reference condition, making assessment of sediment-biological quality interactions problematic.
- ***Consider the regional context.*** In many cases, the full range of geomorphic, hydrologic, and biological characteristics of certain aquatic systems are not present in Kansas. However, such a range might be observable at a regional or multi-state scale. Study of related systems in other states is appropriate.
- ***Record both intensity and duration of sedimentation events.*** Research shows that an ecotoxicological model (i.e., one that considers both amount of sediment and length of sediment exposure) better predicts effects of sedimentation. However, most

current studies report only sediment concentration (intensity). Temporal cycling of sediment could be important for biological systems.

- ***Distinguish between natural and induced sedimentation.*** Some low-gradient, high turbidity systems in the Central Plains have elevated natural sediment loads as an ambient condition. Discerning impairment in these systems could require significant study.
- ***Use advanced statistical techniques.*** Interactions between response and predictor variables in ecological systems are complex. Statistical procedures used to analyze response data must be robust to account for variation, and techniques such as multiple linear regression, principal components analysis, regression trees, analysis of covariance, quantile regression, and structural equation modeling might be appropriate.

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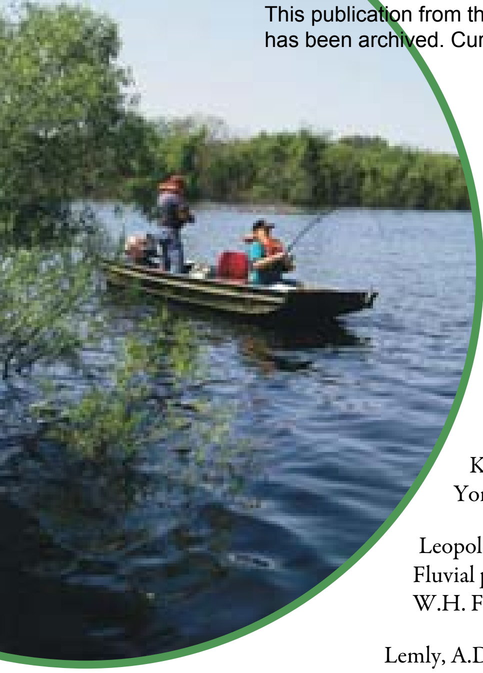

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Management Practices to Control Sediment Loading From Agricultural Landscapes in Kansas

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Introduction

Suspended solids are the largest category of water pollutants in Kansas (Devlin and Powell, 1996). Almost all Kansas lakes and streams contain undesirable levels of suspended solids in the water and sediment deposits in lake beds and stream channels. Suspended solids typically consist of solid organic or mineral particles in water suspension that have been detached and transported (eroded) from their original site. Sedimentation occurs when water slows enough to allow particles to settle out. The terms "suspended solids" and "sediment" (or sedimentation) often are used interchangeably. However, "sediment" actually refers to particles that have settled out of suspension to the bottom of streams, rivers, or lakes.

Major sources of suspended solids in agricultural landscapes include cropland, grazing lands, livestock confinement operations, forest lands, roads and ditches, rural homesites, and unstable stream beds and channels. Several types of erosion occur from these sites:

- Sheet erosion: a relatively uniform thin layer of soil is removed by rainfall and largely unchanneled surface runoff
- Rill erosion: numerous and randomly occurring small channels only a few inches deep with steep sides form on sloping fields
- Ephemeral erosion: small channels eroded by concentrated flow that can be filled easily by normal tillage re-form in the same location during subsequent runoff events
- Gully erosion: accumulated water repeatedly fills narrow channels and, over short periods, removes soil from this narrow area to considerable depths resulting in channels that are too deep to correct easily with farm tillage machinery

Implementing best management practices (BMPs) to minimize erosion can improve water quality. However, some studies showed that despite implementation of conservation practices, sediment yield in many of our nation's streams and lakes remained constant for several decades (Trimble, 1999). In many cases, this continuing sediment yield comes from additional erosion that occurs in streams and lakes as channels and banks are eroded by varying velocities of flowing water (Simon and Rinaldi, 2006). This form of erosion can be accelerated by channelization or modification of stream banks.

Sediment Sources from Agricultural Lands

In Kansas, main sources of sediment from agricultural landscapes are cropland fields, grazing lands, streambeds and streambanks. Runoff also occurs from livestock confinement operations, roads and roadway ditches, forest lands, and rural homesites. Previous research includes field measurements and modeled estimates of erosion from crop fields, but few studies discuss other sediment sources (Schnepf and Cox, 2006a, 2006b). Data from the National Resources Inventory (NRI), a nationwide survey conducted by the Natural Resources Conservation Service (NRCS, 2007), indicate that erosion from cropland and pasturelands in Kansas declined over the last 20 years (Table 1).

Table 1. Estimated erosion on nonfederal lands in Kansas, 1982 to 2003[a]

Year	Land Use		
	Cultivated Cropland	Pasture-land	CRP Land[b]
	tons/acre per year		
1982	2.7	0.8	---
1987	2.6	0.8	2.3
1992	2.3	0.7	0.4
1997	2.2	0.7	0.3
2003	2.1	---	---

[a] Data source: NRCS (2007)
[b] CRP = Conservation Reserve Program

Another NRCS (1992) study in northeast Kansas quantified sediment yields from different sources for two watersheds (Figure 1). In both the Missouri River and Kansas River basins, unprotected croplands contributed the majority of sediment loads, more than 20 tons/acre per year in the Kansas River basin. The second-largest contributor was unprotected pasture, with values near 5 tons/acre per year. Sheet and rill erosion contributed more than 60% of sediment loads, and ephemeral and classic gullies each contributed around 10% to 20% (Figure 2).

Limitations in Determining Agricultural Contributions to Reservoir Sedimentation

Erosion by water from croplands and grazing lands is estimated with the Revised Universal Soil Loss Equation (RUSLE), which estimates sheet and rill erosion occurring in an individual field, but the model does not estimate ephemeral gully erosion or amount of sediment leaving the field. Further, few studies have examined relationships between in-field or edge-of-field sediment losses and actual sediment delivery into Kansas rivers and lakes.

Two terms, "sediment delivery ratio" and "sediment yield," are important for determining the effect of erosion on sedimentation of Kansas lakes. Sediment delivery ratio is defined as the extent to which eroded soil (sediment) is delivered from the erosion source to the watershed outlet and accounts for sediment deposition along the path from source to outlet. Deposition areas include buffers, waterways, ponds, road ditches, fence rows, edges of fields, and terraces. Sediment delivery ratio is calculated using the following equation:

$$\text{SDR} = \frac{\text{sediment yield at outlet}}{\text{total erosion}}$$

The larger the watershed, the smaller the sediment delivery ratio (Figure 3). For example, a watershed with a drainage area of 1 square mile has a predicted delivery ratio of 37%, but a watershed with a drainage area of 100 square miles has a predicted delivery ratio of 11%. A large reservoir with a large drainage area, such as the Tuttle Creek Reservoir, might have a delivery ratio as low as 3% or 4%.

Practices to Reduce Erosion and Sedimentation

Erosion Process

The first step in developing an erosion management strategy is to understand the three-stage erosion process. Implementing BMPs can reduce erosion and sediment yield at any or all of these stages:

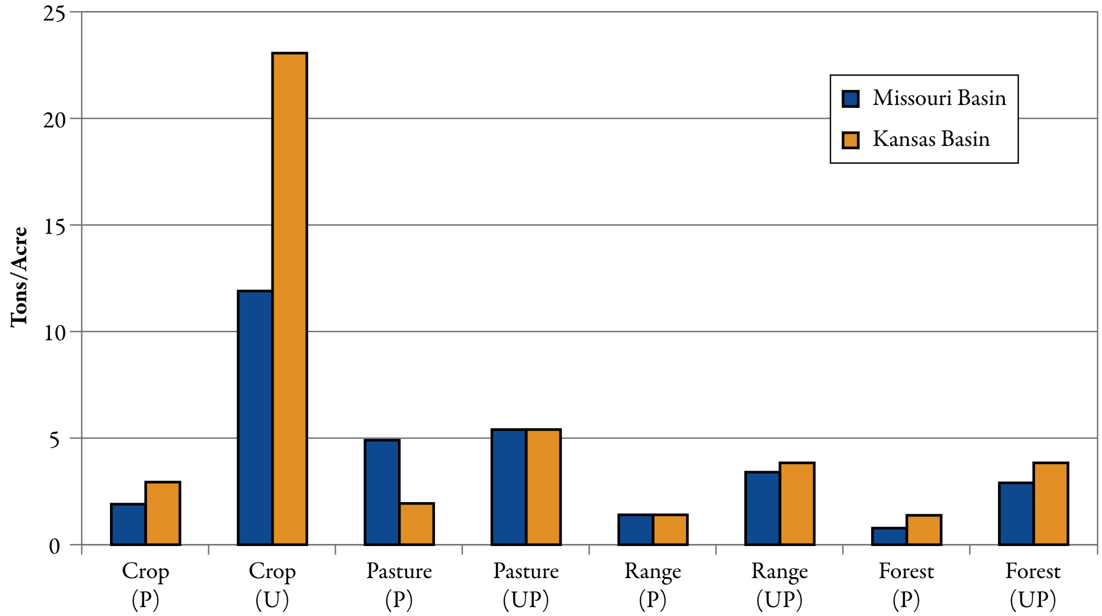

Figure 1. Sediment load contribution from various sources in two northeast Kansas watersheds
P = protected, UP = unprotected
Data source: NRCS (1992)
Figure adapted from McVay et al. (2005) with permission

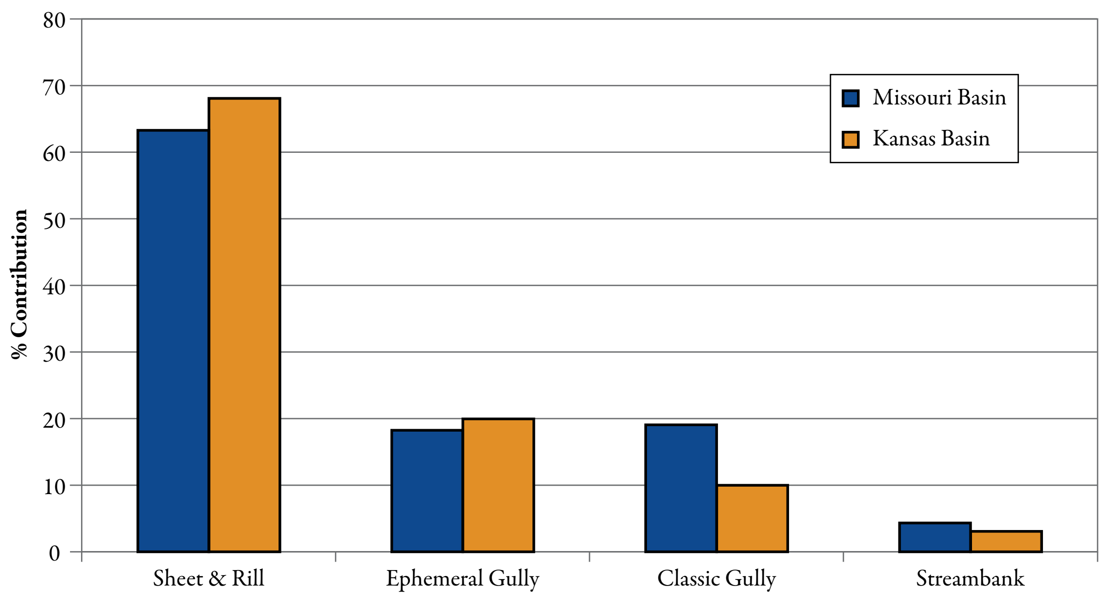

Figure 2. Sediment load contribution from various types of erosion in two northeast Kansas watersheds
Data source: NRCS (1992)
Figure adapted from McVay et al. (2005) with permission

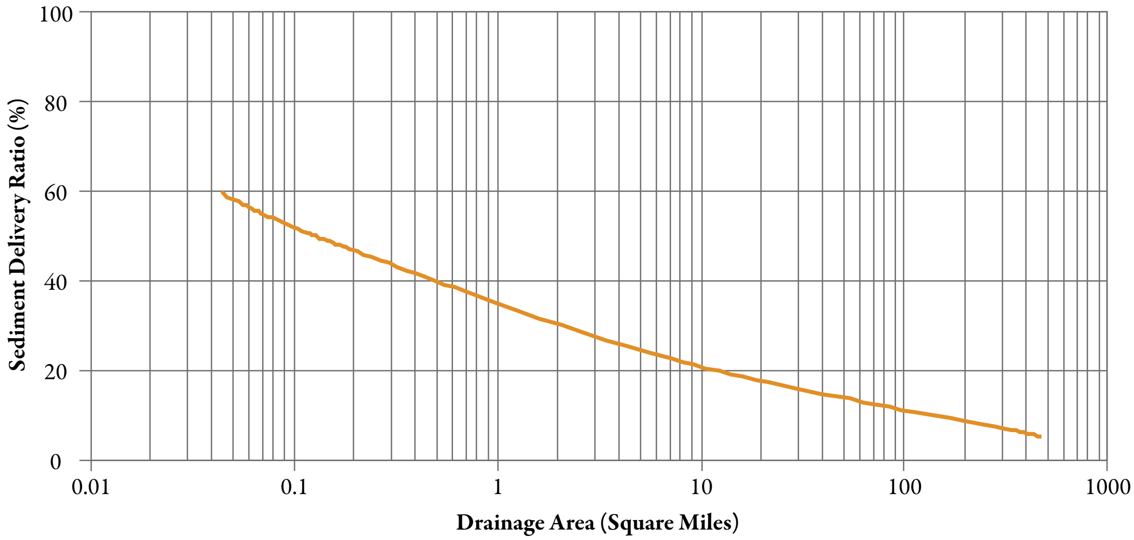

Figure 3. Relationship between sediment delivery ratio and drainage area
Data source: NRCS (1992)

Detachment. Erosion starts with the impact of a raindrop. Raindrops' collisions with the soil break the soil aggregate into its component parts of sand, silt, and clay. Moving water picks up smaller particles of silt and clay. The force of flowing water also detaches soil particles.

Transport. As initial water movement into the soil slows, smaller particles settle out, and finer particles start to plug pores at the soil surface. Runoff occurs if rain continues to fall at rates greater than what the soil can absorb, and soil particles move with runoff water.

Deposition. Soil particles are deposited when water velocity slows enough that it can no longer support them.

Best Management Practices

Although any soil surface left unprotected is vulnerable to erosion, this paper focuses on BMPs that reduce erosion from crop fields and grazing lands, major sources of sediment in Kansas. Strategies for reducing erosion and sedimentation can be divided into two categories: conservation structures and management practices.

Conservation Structures. Conservation structures typically include an engineering design and often have been cost-shared through conservation districts and the NRCS using state and federal funds. Generally, these long-term practices have an expected useful life span of at least 15 to 20 years. Examples of conservation structures include:

- Terraces—gradient, level, tile outlet
- Grassed waterways
- Vegetative and riparian buffers/filters
- Grade stabilization structures
- Water and sediment control structures

Management Practices. These practices generally are related to agronomic practices and typically do not require an engineering design. Examples of management practices include:

- No-till
- Reduced or minimum tillage
- Contour farming
- Crop rotations

Some strategies reduce soil erosion; others trap sediment in the field. A system that combines conservation structures and management practices will be most effective at reducing soil erosion and sediment yield. Several methods including in-field and edge-of-field measurements, in-field models, and watershed models have been used to evaluate effectiveness of conservation structures and management practices.

Effectiveness of Selected Practices

Terraces (gradient, level, or tile outlet) are the backbone of conservation practices in many Kansas fields. Although terraces can be somewhat expensive, with onetime installation costs of $30/acre to $40/acre plus an annual cost of $13.60/acre (Devlin et al., 2003), they also can be quite effective. Terraces reduce erosion by breaking slopes into segments, which reduces the speed of runoff and amount of soil and adsorbed pollutants that can be transported, and reduce ephemeral and gully erosion by safely transporting surface runoff to a stable outlet. Some sediment deposition will occur in the terrace channel. Kent McVay (personal communication, September 1, 2005) used the RUSLE equation to estimate soil erosion losses from four different soil series in central Kansas. In a field with a 6% slope and 150-foot slope length, soil erosion would be reduced approximately 54% by using one terrace and approximately 90% by installing two terraces (Table 2). Field studies in Kansas showed that terraces with tile outlets or those draining into grassed waterways reduced soil erosion approximately 30% (Devlin et al., 2003; Tables 3 and 4). McVay et al. (2005) used the Soil and Water Assessment Tool (SWAT), a watershed model, to examine the effect of terraces and other management practices in the Little Blue River watershed located in Kansas and Nebraska and estimated that terraces would reduce sediment loss by 89% and 98% on conventional and no-till fields, respectively (Table 5).

Table 2. Predicted terrace effectiveness for reducing soil loss from four soil types in a field with 6% slope and 150-foot slope length in central Kansas[a]

Terraces	Silty clay loam	Clay loam	Loam	Sandy clay loam
	tons/acre per year			
None	16.8	13.9	14.4	14.3
1	7.75	6.69	6.62	6.65
2	1.75	1.52	1.36	1.42

[a] Data source: Kent McVay, personal communication, 2005

Table 3. Effectiveness of BMPs for reducing edge-of-field soil losses in conventionally tilled fields[a]

Best Management Practice	Reduction in runoff (%)
Crop rotations	25
Establish vegetative buffer strips	50
Conservation tillage (>30% residue cover following planting)	30
No-till farming	75
Contour farming (without terraces)	35
Terraces with tile outlets	30
Terraces with grass waterways (with contour farming)	30

[a] Data source: Devlin et al. (2003)

Table 4. Effectiveness of BMPs for reducing soil losses in no-till fields[a]

Best Management Practice	Reduction in runoff (%)
Crop rotations	25
Establish vegetative buffer strips	50
Contour farming (without terraces)	20
Terraces with tile outlets	30
Terraces with grass waterways (with contour farming)	30

[a] Data source: Devlin et al. (2003)

Grassed waterways are also used widely in Kansas. They serve as an outlet for excess field runoff water and sediment, reducing the potential for gully erosion and excessive sedimentation. Grassed waterways often are used as outlets for water from gradient terraces or diversions. Vegetative cover in the grassed waterways slows runoff water and allows for sediment deposition before runoff water leaves the field. Grassed waterways can reduce sediment loss from crop fields by 15% to 35% (Devlin et al., 2003). Placing vegetative buffers on the downhill slopes of crop fields or riparian buffers next to streams are recommended practices for removing sediment from runoff water prior to the runoff water leaving the crop field. Well-designed buffers can reduce sediment loss by 50% (Tables 3 and 4), and the SWAT model predicted that installing a 20-meter buffer on the downhill side of every field in the Little Blue River watershed would reduce sediment loss by 89% to 97% (Table 5).

No-till and minimum/reduced tillage farming practices are being adopted rapidly in Kansas (Figure 4) and, when adopted, will significantly reduce sediment loss from crop fields. In a continuous corn field in Brown County, Kansas, converting from conventional tillage with <10% residue to no-till or minimum/reduced tillage reduced sediment loss from 10.5 tons/acre per year to 0.20 tons/acre per year and 0.53 tons/acre per year, respectively (B. Marsh, personal communication, November 23, 1992). In a Franklin County, Kansas, field with a grain sorghum/soybean rotation, adopting no-till reduced sediment loss from 0.85 tons/acre per year to 0.23 tons/acre per year (K. Janssen, personal communication, May 22, 2000). Devlin et al. (2003) reported that adopting reduced/minimum tillage (>30% residue cover following planting) and no-till reduced erosion by 30% and 75%, respectively (Table 3). The SWAT model predicted that adopting no-till on all crop fields in the Little Blue River watershed would reduce sediment loss from crop fields by 77% (Table 5).

Table 5. Estimated reductions in water flow and sediment loss due to BMP implementation compared with a conventional tillage scenario[ab]

Tillage System	Best Management Practice	Water Discharge	Sediment Loss
		% Reduction	
Conventional	None	0.0	.0
	10-m buffer	0.0	72
	20-m buffer	0.0	89
	contour	0.9	50
	effective terraces	0.9	89.4
	10-m buffer + contour	0.8	86
	10-m buffer + contour + terraces	0.8	97
Conservation tillage with 20% residue	None	0.4	47
Conservation tillage with 50% residue	None	0.8	63
No-till	None	13	77
	10-m buffer	13	93
	20-m buffer	13	97
	contour	20	90
	effective terraces	20	98
	10-m buffer + contour	20	97
	10-m buffer + contour + terraces	20	99
Mixed grass prairie/range	None	42	99

[a] Table adapted from McVay et al. (2005) with permission
[b] Based on SWAT model results in the Little Blue River basin of Kansas and Nebraska averaged over 22 years

Tilling and planting along the contour of field slopes can reduce soil erosion. Contour farming is a recommended practice for all sloping, erosive fields and is the only acceptable method of farming in terraced fields. Conducting tillage and planting operations on the contour without terraces reduced soil erosion by 35% (Devlin et al., 2003; Table 3), and computer modeling with SWAT in the Little Blue River watershed predicted contour farming could reduce sediment loss by 50% to 90% depending on tillage system (Table 5).

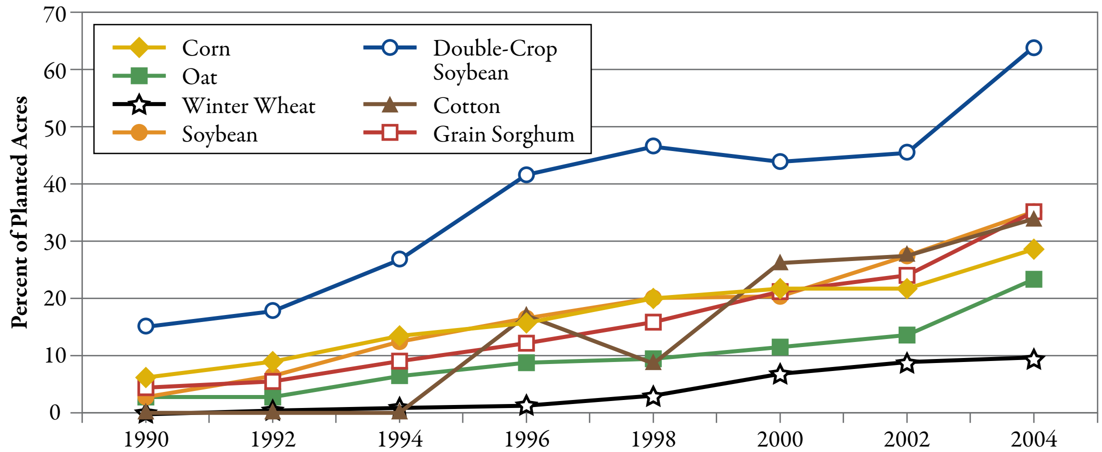

Figure 4. Rate of no-till adoption for various crops in Kansas
Data source: CTIC (2005)
Figure adapted from McVay et al. (2005) with permission

Recommendations

Because erosion is a natural process, there always will be some suspended solids in streams. However, excessive soil erosion and sedimentation are negatively affecting nearly all lakes and streams in Kansas. It is difficult to determine to what extent measured and predicted erosion actually affects sedimentation in rivers and lakes, but minimizing erosion can improve water quality. A variety of BMPs, used separately or together, can reduce erosion and sediment loss from agricultural lands, and the most effective erosion reduction strategies include a combination of conservation structures and management practices.

We offer the following recommendations:

- Determine sediment sources and methods of sediment delivery from agricultural lands to surface waters
- Determine effectiveness of cropland and grazing land erosion control practices at both the field and watershed scale
- Develop an understanding of the lag period between implementation of erosion control practices and sediment delivery to surface water bodies
- Develop methods that link changes in land cover with hydrologic response, sediment-load response, and stream channel changes
- Determine the sediment load contributions of roads, urban sprawl, and other nonagricultural activities in the rural landscape

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Can Reservoir Management Reduce Sediment Deposition?

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Introduction

This white paper describes why sedimentation in public water supply reservoirs is cause for concern about both water quantity and quality, proposes management options for reducing sedimentation and strategies for implementing these options, identifies needed resources, and recommends assessment methods. We focus on federal reservoirs because of their large storage capacity, but similar problems and solutions also apply to smaller, locally owned reservoirs, commonly referred to as lakes.

Thirteen federal reservoirs in Kansas are in the state public water supply marketing and assurance programs, both managed by the Kansas Water Office (KWO). The marketing program allows public water suppliers to purchase and withdraw state-owned water stored in federal reservoirs. The assurance program allows public water suppliers who draw water from streams downstream from reservoirs to purchase stored water that can be released during low flow conditions to supplement natural flows. About 50 smaller, mostly city-owned reservoirs provide additional localized public water supplies to municipalities and industries. All these reservoirs are vital resources because they provide regional sources of stored, untreated water to public water suppliers in Kansas for use in surrounding communities and industries. Many Kansans rely on the long-term availability of water supplies. However, long-term planning to ensure sustained water availability is lacking. Plans similar to those created for public infrastructures, such as roads and bridges, are needed for long-term protection and maintenance of reservoirs.

Reservoir Construction

Natural lakes form through geologic processes, such as glacial melting or scouring. Although often referred to as lakes, reservoirs do not occur naturally; they are created by transforming part of a flowing water body (lentic) into a still water body (lotic) by building a dam to hold back the water. Sediment transport occurs naturally through streamflow, and reservoirs act as settling basins for soil, clay, and smaller rock particles deposited through sedimentation. Eventually, sediment will fill all reservoirs unless removed or properly redistributed. Reservoirs commonly are constructed in areas with few natural lakes, particularly locations south of the glaciated area in middle latitudes. Soils in these areas are inherently very erodible and can be disturbed further by human activities. In addition to receiving sediment deposited through streamflow, most federal reservoirs in Kansas experience shoreline erosion, due to erodible soils, and constant maintenance is required to stabilize shorelines.

Federal reservoirs were built and are owned by either the U.S. Army Corps of Engineers (USACE) or the Bureau of Reclamation and were constructed at the direction and authorization of Congress. Authorizations for use vary, but most reservoirs were constructed primarily for flood control, irrigation, water supply storage, aquatic habitat, low flow supplementation, and water quality maintenance. The USACE also uses some reservoirs to maintain flows

for navigation in the Missouri River. Congress has the final authority to authorize reservoir use or propose operational changes.

Many reservoirs were designed and built based on a 100-year planning framework. During this time, reservoirs were expected to remain fully functional and satisfy the needs and purposes for which they were constructed. Now, nearly half that time has passed. Federal reservoirs in Kansas were built between 1946 and 1976. The oldest, Kanopolis, is 60 years old, the youngest, Hillsdale, is 30 years old, and the average age is about 44 years. Reservoir designers assumed future stakeholders would be better equipped, with new technologies and enhanced perspectives, to deal with future water resource problems. But, management strategies to ensure longer reservoir lifespans have not been adequately considered.

Sedimentation in Reservoirs

Erosion

Reservoir sedimentation begins with runoff and soil erosion. Various types of land cover produce different runoff characteristics that, in turn, transform hydrology of streams, and watersheds across the country are dynamically adapting to continuous land use changes. Original land cover in the uplands of most watersheds draining into federal reservoirs in Kansas was grassland, the native, pre-settlement condition. Riparian forests occurred along most streams and rivers, especially in floodplains. Prior to human settlement, Kansas forests covered about 8% of the landscape; today they cover about 4% (Bob Atchison, Kansas Forest Service, personal communication, 2004). Historically, the prairie ecosystem produced steady, prolonged runoff from storms. Over hundreds of thousands of years and long periods of climate change, stream channels formed to accommodate runoff during normal and extreme rainfall events.

Watersheds gradually transformed from expansive tallgrass prairie to a checkerboard of rural communities, cropland, urban land, and managed pastureland. Native prairie and riparian woodlands were removed for crop cultivation and building materials, resulting in increased soil erosion from the disturbed landscape. Land gradually became less fertile as soil was lost, and maintaining desired production levels required applying additional nutrients. As erosion continued on steeper slopes, added nutrients washed into tributaries and rivers.

Runoff from cultivated and grazed land occurs quickly, frequently, and intensely, accelerating natural bank and channel erosion. Other alterations, such as levees that contain flood waters and prevent sediment from settling out on the floodplain, constrictions due to road crossings, and hundreds of dams forming small ponds, affect stream channel stability and will take many years to fully manifest. Urban development results in increased impervious cover, further exacerbating erosion and decreasing stability of watersheds and stream systems.

Sedimentation

Soil particles eroded from land surfaces (e.g., uplands, prairie, pasture, agricultural fields, and streambanks) by wind and rain become suspended in water that flows into streams. Streamflow slows as it enters a reservoir, and suspended particles begin to settle out. Eventually, all sediment will settle to the bottom of a still pool of water, but heavier sediment particles are deposited first. Sedimentation does not occur uniformly; it is affected by many factors including flow and volume of water produced by the incoming stream and size and weight of sediment particles.

Regardless of rate or location, sediment accumulation reduces reservoirs' water storage capacity. Therefore, reservoirs were built larger than necessary for their intended purposes, allowing sediment to be collected behind the dam. Most Kansas reservoirs were designed with a reserve sediment storage pool that was projected to fill with sediment over a 100-year period. Designers expected that although sediment would continue to accumulate beyond this point, occupying space previously available for water storage, aquatic habitat, recreation, or other uses, reservoirs would maintain flood control capacity for an extended period of time and wetland and marsh habitats would slowly form. Designers expected other forms of recreation, such as waterfowl hunting and bird watching, to develop but did not anticipate how much communities and the state would rely on reservoirs for public water supplies. Although water supply is an authorized reservoir use, designers gave the effects of reduced water storage capacity little consideration and did not anticipate a need for operating procedures or funding to address this situation.

Recent USGS studies indicate that decreases in conservation-pool water-storage capacity of federal reservoirs due to sedimentation range from less than 10% for Cheney Reservoir (south-central Kansas), Hillsdale Lake (northeast Kansas), and Webster Reservoir (north-central Kansas), to about 25% to 40% for Perry Lake and Tuttle Creek Lake (northeast Kansas). If sedimentation continues at historical rates, designed sediment pools in Perry Lake and Tuttle Creek Lake will be filled by 2021 and 2023, respectively, and further sediment accumulation will encroach on water supply storage intended for other purposes (Juracek, 2006). Yet, KWO population projections indicate that many communities that currently use these reservoirs continue to grow and have increased demand for municipal and industrial water supplies. When reserve pools of existing reservoirs fill with sediment, few viable options will exist for maintaining resources provided by reservoirs.

Eutrophication

Although this paper focuses primarily on preservation of public water supply storage capacity, a related and more imminent concern is the effect of sedimentation on water quality. The main water quality issue is eutrophication, the process that both natural lakes and constructed reservoirs undergo as they age. Eutrophic conditions occur as sediment and nutrients attached to sediment or suspended in water gradually accumulate, leading to excessive aquatic plant growth, especially algae. Most federal reservoirs in Kansas already are in some stage of eutrophication, and some are in advanced stages.

Originally, most sediment was expected to accumulate at the bottom of reservoirs near dams. However, large quantities of sediment are settling out in upper arms of reservoirs, creating shallow flats and deltas. Because inflow waters typically are nutrient rich, these shallow areas of still water provide ideal conditions for algal growth and dense growth of rooted aquatic plants. When nutrient ratios are favorable, much of the algal biomass is composed of blue-green algae that produce geosmin. This compound causes taste and odor problems in drinking water, issues that can be difficult and expensive to treat. Additionally, aquatic plant growth produces living and decaying biomass that restricts boat access and adds to taste and odor problems.

Managing Sedimentation

Current state and federal management efforts focus on reducing sediment inputs from the watershed landscape, including streambanks and streambeds. However, it also is necessary to manage sediment already deposited in reservoirs. Reducing sedimentation will extend the useful life of reservoirs, particularly for water storage capacity, and reduce the amount of nutrients entering reservoirs, slowing the eutrophication process and reducing taste and odor problems and associated treatment expenses.

Reservoir sedimentation management strategies can include one or more of the following techniques (Palmieri et al., 2003; WOTS, 2004):

- Reducing sediment inflows
- Managing sediment in the reservoir
- Removing sediment from the reservoir
- Replacing lost storage
- De-commissioning the reservoir

Reducing Sediment Inflows

Techniques applied to the watershed system before water enters the reservoir include watershed management; upstream debris dams, sediment basins, and wetlands; reservoir bypass, and off-channel storage.

Watershed Management. Erosion is a natural, geologic process but can accelerate when the soil surface is exposed. Implementing best management practices (BMPs) that minimize soil exposure and soil particle detachment can reduce soil loss. Watershed BMPs are categorized as rural/agricultural or urban/suburban practices. Agricultural BMPs include conservation crop rotations, cover crops, and conservation tillage. These BMPs improve soil structure and increase soil organic matter content and surface roughness. Other beneficial agricultural practices include terraces that trap sediments and pond and infiltrate water, waterways and filter strips that slow water flow and trap sediment, and buffers along streambanks that reduce streambank erosion and trap nutrients and sediment. Each farm enterprise is unique, and encouraging widespread adoption of appropriate BMPs requires one-on-one interaction with producers (Birr and Mulla, 2006).

Urban/suburban BMPs include settling basins, construction erosion control, construction timing, buffers, and on-site detention. These practices focus on preventing soils from leaving construction sites. Other practices include swales, open space detention, streambank protection, and on-site infiltration. These are applicable in developed areas and focus on preventing increased runoff and flows; many are included as recommended practices in Phase 1 and 2 stormwater permits.

Agricultural BMPs are designed to be effective for 24-hour storm events ranging from 10- to 25- year recurrence intervals,

but available data indicate that the greatest sediment loads occur during storm events that exceed these parameters. During flows of this magnitude, watershed BMPs have reduced effectiveness for containing sediment. However, stable streambanks are beneficial during flood events. During the 1993 flood in northeast Kansas, forested riparian buffers along the main stem of the Kansas River were considerably less impacted (amount of bank loss) than areas with streamside land cover of cropland or grass (Geyer et al., 2003). Additionally, streambank stabilization projects with associated riparian buffer establishment are effective at minimizing streambank erosion and reducing sediment loads.

Where properly designed, installed, and maintained, watershed landscape-level BMPs (e.g., terraces, waterways, and filter strips) are effective at reducing soil erosion on specific sites, which, theoretically, should reduce reservoir sedimentation rates. Unfortunately, research does not support this. Studies estimate that reducing sediment yields by 10% to 20% might require intensive conservation efforts spanning several decades. Furthermore, conservation measures often are considered ineffective, from a reservoir sedimentation management point of view, because of the large time lag between when erosion control measures are implemented and when their effects on sediment reduction are realized (Birr and Mulla, 2006).

A paired watershed study on effects of agricultural BMPs revealed that many factors, including lag time, rate of BMP adoption, mechanisms of nutrient transport, and climatic variability, influence BMP evaluation at the watershed scale (Birr and Mulla, 2006). Current knowledge of watershed processes and sediment transport in rivers and how these relate does not allow modeling to consistently or adequately predict effects of watershed management techniques on sediment discharge in rivers. No simple solution exists, and assessing potential effects of optional watershed management approaches requires detailed study of watersheds under consideration and thorough analysis of available data and local knowledge.

Currently, the USGS is investigating sediment sources in the Perry Reservoir and Lake Wabaunsee watersheds in Kansas. The objective of this study is to determine, by comparing composition of reservoir bottom sediments and sources materials, whether the majority of deposited sediment originated from surface soil erosion or streambeds and streambanks. Understanding sediment sources will help target future management efforts to achieve meaningful reductions in sediment yield. Preliminary results indicate that sediment source is watershed specific and cannot be generalized.

Good upland watershed management can reduce sediment yields and produce many associated benefits for agriculture, rural and urban environments, food production, forestry, and water availability. But during the next 25 to 50 years, upland BMPs likely will not have a large effect on reservoir sedimentation.

Upstream Debris Dams, Sediment Basins, and Wetlands. Debris dams are used for streams in steep watersheds and those with coarse-grained sediments. Debris dams usually are located on one or more tributaries upstream from the reservoir, and sediments are removed periodically from behind the dam. Ease of access to remove sediment from the debris dams and potential to reuse sediment make debris dams a feasible option. Sediment basins and constructed or enhanced wetlands in upstream tributaries use the same concepts.

Reservoir Bypass. Installing conveyance structures (i.e., closing gates) upstream from the reservoir diverts sediment-laden flows around the reservoir, carrying away large volumes of sediment that otherwise would accumulate in the reservoir. This technique limits flood control capability of the reservoir, and floodplain development might already have occurred downstream. Often, bypassing is feasible only when favorable hydrological and morphological conditions exist. Operating costs of conveyance structures and benefits lost by not capturing flood flows must be considered, and this technique might require a change in congressional authorization.

Off-Channel Storage. Off-channel storage reservoirs are built adjacent to the main river channel (e.g., a small tributary or on the floodplain). Water from the main river is routed into the reservoir when sediment concentrations are low. This option does not manage sediment in existing reservoirs but could be considered for new projects. A variation of this technique that can be applied in existing reservoirs is construction of settling basins in tributaries; sediment can be periodically removed from these basins.

Managing Sediment in the Reservoir

Techniques for preventing sediment from settling once water enters the reservoir include multilevel selective withdrawal, changes in lake level management plans (LLMPs), inflow routing, sluicing, and density current venting. These techniques are most applicable in reservoirs that stratify thermally. Because of their large size and prevailing windy conditions that promote mixing, most federal reservoirs in Kansas do not thermally stratify for long periods of time during the summer, which limits the applicability of these methods.

Multilevel Selective Withdrawal. Operating a multilevel withdrawal structure to manage sedimentation requires considering numerous conditions and constraints, most important of which is thermal stratification. Selective withdrawal, the capability to identify the vertical distribution of withdrawal from a stratified reservoir and use that capability to selectively release the desired quality of water, can be used to determine the appropriate or best available operation of a release structure, design multilevel withdrawal structures, or modify existing projects.

Lake Level Management Plans. Many reservoir projects operate under some type of LLMP that incorporates seasonal changes in elevation. Modifying LLMPs can enhance water quality and reduce sedimentation. By changing the hydraulic residence time of the reservoir, inflows can be retained or routed quickly through the reservoir. This allows the reservoir to retain high quality water for later release or retain poor quality water for treatment by in-reservoir processes.

The KWO, in consultation with the Kansas Department of Wildlife and Parks, the USACE, and the Bureau of Reclamation, is responsible for annual development and oversight of LLMPs. Planned variations in operating level of each reservoir are published each year and cover the period from October through September of the following year. Stakeholders can submit concerns through a public input process.

Depending on project authorization, the management plan might include large, pool-level fluctuations on an annual basis. For example, operation of a flood control project usually involves drawing down the reservoir level during fall and winter and filling during spring and early summer, resulting in a stable pool through summer and early fall. Water quality might be a concern when summer pool elevation is kept relatively stable. Water quality components that could be affected include inflow with undesirable qualities, nutrient loading of the reservoir (and associated effects on algal growth and fisheries), turbidity, and sedimentation. For example, inflows with a high sediment load might be delayed in the upper reaches of a reservoir and settle out before reaching the outlet works. Additionally, shoreline erosion can accelerate when high elevations are maintained for long periods of time. This results in sediment accumulation around the edges of the reservoir and contributes to formation of flats and deltas, promoting unwanted algal growth.

Inflow Routing. Poor inflow water quality (e.g., high concentrations of nutrients, suspended solids, or other undesirable constituents) can result in poor reservoir water quality. Depending on volume of inflow and retention time of the reservoir, inflow constituents can settle and become trapped in the reservoir, contributing to eutrophication and increasing sediment accumulation. If inflow quality is a concern, it might be possible to route inflow through the reservoir for downstream release without significantly affecting reservoir water quality. Because inflow will seek its layer of neutral density in a thermally or density-stratified reservoir, a density current will develop and proceed through the reservoir. Using the existing release structure and operating within the existing water control plan, undesirable water is routed through the pool as quickly as possible.

Sluicing. Sluicing is an operational technique in which a substantial portion of the incoming sediment load moves through the reservoir and dam before sediment particles can settle, reducing the reservoir's trap efficiency. In most cases, sluicing is accomplished by operating the reservoir at a lower level during the flood season to maintain higher flow velocity and sufficient sediment transport capacity of water flowing through the reservoir. Increased sediment transport capacity reduces the volume of deposited sediment. After flood season, the pool level in the reservoir is raised to store relatively clear water. Effectiveness of sluicing operations depends on availability of excess runoff, grain size of sediment, and reservoir morphology. In many cases, sluicing and flushing are used in combination. If flood control is an authorized purpose of a reservoir, use of sluicing might require a change in congressional authorization.

Density Current Venting. Density currents occur because the density of sediment-carrying water flowing into a reservoir is greater than the density of clearer water impounded in a reservoir.

The increased density, increased viscosity, and concomitant reduction in turbulence intensity result in a uniform current with high sediment concentration that dives underneath the clear water and moves toward the dam. In reservoirs with known density currents, installation and operation of low-level gates allows sediment currents to pass through the dam for downstream discharge. Density current venting is an attractive option because, unlike flushing operations, it does not require lowering the reservoir level. However, this approach results in increased downstream sediment loads that can degrade stream habitats. In addition, dispersing sediment across large areas makes it more difficult to eventually remove sediment from the watershed system.

Removing Sediment from the Reservoir

Techniques for removing accumulated sediment include flushing, aeration, and mechanical removal (e.g., dredging, dry excavation, and hydro-suction).

Flushing. Flushing increases flow velocities in a reservoir to the extent that deposited sediments are resuspended and transported through low-level outlets in the dam. Flushing occurs in two ways: complete draw-down flushing and partial draw-down flushing. Complete draw-down flushing occurs when the reservoir is emptied during flood season; this creates river-like flow conditions in the reservoir. Deposited sediment is remobilized and transported through low-level gates to the river reach downstream from the dam. Low-level gates are closed toward the end of flood season to capture clearer water for use during the dry season.

Partial draw-down flushing occurs when the reservoir level is partially reduced. Sediment transport capacity in the reservoir increases only enough to allow sediment from upstream locations to move farther downstream, closer to the dam. Partial draw-down flushing can clear more water storage space upstream and transport sediment to a more favorable location for future complete draw-down flushing.

Artificial Lake Stirring and Aeration. Dissolved oxygen (DO) is an important component of reservoir water quality and can affect drinking water taste and odor, especially during lake turnover events or when a fish die-off occurs because of eutrophication and low DO levels. In surface waters, DO occurs naturally through two primary sources: surface exchange with atmospheric oxygen (adsorption) and algae that create oxygen as a by-product of photosynthesis; DO is removed from the water column by fish respiration and decomposition of organic matter by aerobic bacteria. An oxygen shortage or depletion can occur when photosynthesis ceases or is substantially reduced during snow and ice cover or when excessive biomass is present (e.g., in eutrophic lakes).

Several methods can supply DO artificially: mechanical stirring by fountains, pumps, and electric or wind-driven stirrers; compressed air supplied by oilless compressors or blowers and diffusers; and chemical oxidizers such as potassium permanganate. Each method varies in effectiveness. Compressed air likely provides the best long-term treatment, and potassium permanganate provides the most immediate results. The compressed air method also promotes long-term organic matter decomposition and nutrient recycling that can help correct eutrophication-related taste and odor problems.

Compressed air is more efficient than mechanical stirring or a water fountain system at moving the oxygen-deprived water column located in the bottom, or hypolimnion, region of a lake (i.e., lake turnover). Stirrers usually cannot reach the lake bottom, and fountain intakes can plug if mounted too close to bottom sediments. Chemical oxidizers might be the best solution for providing DO immediately, but this is a short-lived, temporary solution. In addition to providing supplemental oxygen, an aeration system can:

- Suspend organic matter in the water column, allowing better oxidation and fertilization of phytoplankton and zooplankton in the epilimnion, the upper portion of water in the reservoir
- Resuspend and transfer sediment during water exchange events
- Partially restore lake depth and volume

One type of aeration system, the pipe venturi, can perform a specific type of dredging called "air dredging" (Haag, 2006) and has been used in wastewater lagoon systems to move and aerate sludge. The pipe venturi is approximately 3 feet in length and 6 inches in diameter with holes cut in the bottom and sides. It is placed vertically on the bottom of the reservoir in loosely compacted sediments. Air is released at the bottom of the pipe, and bubbles rise through the pipe creating a suction, or venturi, effect. The venturi disturbs the loosely compacted, unconsolidated sediment, which becomes resuspended in the epilimnion and then moves out of the lake through the spillway by flushing. The pipe venturi might be useful for dredging and redistributing organic matter, clay particles, and nutrients in eutrophic lakes. However, type and quality of sediment must be quantified prior to using this technique. Some sediment might contain contaminants or unusually high levels of nutrients that could have deleterious effects on water in the reservoir and receiving stream.

Mechanical Removal. Mechanical removal of deposited sediment occurs through conventional dredging techniques, dry excavation, and hydro-suction.

The process of excavating deposited sediments from under water is termed dredging. This is a highly specialized activity used mostly used for clearing navigation channels in ports, rivers, and estuaries. Dredging also can be used to reclaim reservoir storage capacity lost to sediment deposition. However, conventional hydraulic dredging often is much more expensive than the cost of storage replacement and generally is not economically viable or necessary. Excavating sediment from existing reservoirs will require moving 20 to 50, even up to 100, times more material than originally moved to construct the dam. Costs of dredging larger reservoirs in Kansas in their latter stages of filling could be more than 100 times the original construction cost, billions of dollars in some cases. In addition, it will be necessary to find a disposal location for the excavated material, preferably close to the reservoir to reduce transportation costs. Spreading sediment uniformly over one square mile in a one-foot-thick layer would dispose of 640 acre-feet of sediment, but sediment accumulation in a reservoir can total tens of thousands of acre-feet.

Disposing dredged material can cause environmental problems, and solutions, which can be quite expensive, have to be developed on a case-by-case basis. Discharging high sediment concentrations generally associated with dredging directly downstream from the dam can be environmentally unacceptable. However, it might be possible to reduce the sediment concentration of water flowing in the river by concurrently releasing clean water and dredged material from the reservoir. If dredged material is not deposited downstream, large expanses of landfill might be required. Although dredged material can be a liability, it also can be seen as an asset (WOTS, 2004). Uses for dredged sediment include habitat development, soil improvement for agriculture and forestry, and construction (e.g., brick making).

Most Kansas reservoirs are about the same age and fill at about the same rate. Excavating sediment-filled basins and finding a place for the excavated material from all these reservoirs is currently beyond our means. Fortunately, we likely will not need to dredge entire basins of most reservoirs. Tactical dredging of upper arms of the reservoir removes sediment from where it is accumulating most rapidly. Excavating upper basins deeper than their original contour creates settling basins that serve as sediment traps. Preserving an infrastructure that allows access to these settling basins will allow convenient redredging every 20 to 30 years, a possible long-term management strategy.

Dry excavation (i.e., trucking) requires lowering the reservoir during the dry season, when reduced river flows can be adequately controlled without interfering with excavation work. Sediment is excavated and transported using traditional earth-moving equipment. Excavation and disposal costs are high; therefore, this technique generally is used in relatively small reservoirs. Sediment from some reservoirs excavated using this method has been used as engineered landfill in hills adjacent to the reservoirs. It can be difficult to dry the bottom of the reservoir thoroughly enough for heavy excavating equipment to operate on it.

A hydro-suction removal system is a variation of traditional dredging. Traditional dredging uses pumps powered by electricity or diesel. Hydro-suction uses energy from the hydraulic head available at the dam. Where sufficient head is available, operating costs for hydro-suction are substantially lower than for traditional dredging.

Replacing Lost Storage

Lost storage can be replaced by constructing a new dam (upstream, downstream, or on another river) or raising the existing dam, but neither option can be accomplished easily. Raising the dam requires conducting reallocation studies and mitigation of additional flooded land and recreational structures. A single reallocation study can cost more than $1 million. Few good locations for large-scale new dam development remain, and construction costs can be prohibitive. Urban and rural development steadily surrounded reservoirs, limiting our ability to raise dam heights and flood additional land. Also, environmental effects of large-scale dam projects would be difficult to overcome because of enhanced environmental protection regulations.

Decommissioning the Reservoir

Decommissioning should be regarded as the last possible option. There are no reported cases of decommissioning dams higher than 40 meters. Decommissioning large dams is problematic and needs careful consideration, particularly when the reservoir behind the dam is full of sediment. Other options to explore before decommissioning include maintaining the dam at a lower level or using the silted reservoir for ecological enhancement (e.g., wetland habitat, farming, or recreation). However, these options are site specific and still result in lost water storage capacity.

Implementing Sediment Management Strategies

Implementing sediment management strategies requires collecting various data, securing financial resources, and developing a comprehensive plan with input from all stakeholders.

Information

Sedimentation rate is the fundamental problem in all reservoirs; all other reservoir problems are linked, by various degrees, to this issue. Therefore, determining a reservoir's sedimentation rate is necessary for developing both short- and long-term management strategies and will help determine the flow regimes under which most sediment is delivered and deposited. This, in turn, can guide design and placement of appropriate BMPs.

To determine appropriate sediment management strategies for Kansas reservoirs, the following questions should be answered:

- What are the sediment delivery ratios for all watersheds above federal reservoirs?
- Does most sediment delivery occur during high-flow events, or are cumulative low-flow delivery ratios more important?
- What are the stratification characteristics of the reservoir?
- How will dredging affect the usefulness of the reservoir for water supply during the dredging process? Would alternative water supplies be necessary for a period of time? Does the infrastructure exist to manage this?
- Which reservoirs have multilevel release structures? Which could be modified to incorporate these structures?
- What are the effects of discontinuing flood control benefits in favor of water supply capacity preservation? What is the economic effect on properties that would no longer be protected from flooding? What is the cost to buy land for floodplain preservation? What other options for flood control are available?
- What is the sediment quality?
- What are the costs and benefits of various management techniques or combinations of techniques?
- What is the true cost of providing water via a reservoir if costs for long-term storage capacity maintenance are included?
- Which reservoirs should be decommissioned first if this becomes necessary? If a reservoir is decommissioned, how will water storage, flood control, and other needs be satisfied?

Resources

At typical dredging costs of $5,000/acre-foot to $6,000/acre-foot ($3/cubic yard to $5/cubic yard) (deNoyelles et al, 2004) and assuming a constant rate of sediment deposition, removing the annual sediment load deposited in Kansas reservoirs is cost prohibitive. Estimated annual costs for four reservoirs are (KWO, 2005):

- Clinton: $1.6 million
- John Redmond: $4.5 million
- Perry: $5.6 million
- Tuttle: $22.4 million

These costs alone are more than two times the annual State Water Plan Fund. Although BMPs can reduce sedimentation and, ultimately, dredging costs, implementing these practices can be cost prohibitive. To help defray costs of BMP implementation, landowners can participate in government cost-share programs such as the Natural Resources Conservation Service (NRCS) Environmental Quality Incentive Program. A recent report evaluating natural resource cost-share programs summarized expenditures during recent years for the two voluntary government cost-share programs with greatest participation. The NRCS cost shared $89,450,451 to private landowners in Kansas from 2003 through 2005 for BMPs including terraces, livestock production improvements, and wetlands. The State Conservation Commission cost shared $12,972,721 from 2004 to 2006 for similar practices.

Planning

The original "design life" approach to reservoir construction did not consider what would happen to dams at the end of their lifespan or how benefits could be replaced. Future generations were left to deal with substantial environmental, social, economic, and safety issues. The conventional wisdom is to assume dams have a finite ability to store water and accept that this ability will diminish gradually because of sedimentation. An alternative approach is to view dams and reservoirs as sustainable structures.

Sustainability requires that resources be developed and used in a way that accounts for interests of all stakeholders, including future generations. For infrastructure projects, this means that future generations should not be burdened with emergency maintenance or decommissioning of assets built to benefit their predecessors. The ultimate goal of developing sustainable reservoirs is to maintain the major functions of the dam through appropriate management and maintenance in perpetuity. When this is not possible, decommissioning can be used as a last resort, provided that this action is funded by an accumulating dam retirement fund. Sustainable reservoir management will ensure that current and future generations enjoy the benefits of the facility and spread ownership, operation, and maintenance costs over many generations.

With reservoir capacity and water quality concerns looming, it is time to change the paradigm of viewing reservoirs as projects with a defined life span. We must develop and implement sustainable strategies to maintain and extend reservoirs' useful life, beginning by taking action to preserve reservoir water supply infrastructures with a phased plan that first address the most crucial problems in the most important reservoirs.

Developing a Comprehensive Plan

We offer the following suggestions as a guide for developing comprehensive sediment management plans for Kansas reservoirs:

- Establish goals or targets for reductions in sediment deposition for each reservoir.
- Stabilize watersheds. Continue Watershed Restoration and Protection Strategy efforts and other watershed conservation work with focus on public water supply (PWS) reservoirs and erosion control.
- Increase acres of cropland managed with no-till. Prioritize streams for buffer installation and streambank stabilization. Provide cost-share payments or cover the entire cost to ensure implementation and maintenance. Develop means of monitoring and enforcing BMP implementation.
- Prioritize reservoirs for maintenance and infrastructure development and upgrades. Determine which developments will have the greatest effect on water supply infrastructure considering population served, water quality problems, demand, and location of alternative supplies.
- Develop a dedicated state dredging/infrastructure upgrade and maintenance fund.
- Complete a reservoir dredging pilot study. During the pilot study, excavate deeper sedimentation basins in reservoir arms or tributaries and establish maintenance infrastructures.
- Consider establishing critical water quality management areas above PWS reservoirs, and direct funds to establish widespread erosion control practices and streambank stabilization.
- Determine viable management options for in-reservoir water manipulation.
- Establish maintenance infrastructures in priority reservoirs.
- Identify sediment disposal areas.
- Establish wetlands and riparian areas for use in conjunction with sediment disposal sites.
- Increase number and size of wetlands to store flood waters and filter sediment.
- Protect sites that have potential for new dam construction.

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Economic Issues of Watershed Protection and Reservoir Rehabilitation

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Summary

Comprehensively analyzing economic issues of watershed protection and reservoir rehabilitation projects is a somewhat daunting task because of the extensive effects of the costs and benefits of alternative management strategies. In addition, data on economics of sediment control at a watershed scale are lacking, and previous studies have not evaluated whether dredging sediment or preventing sedimentation is more economical. Many questions remain unaddressed and unanswered. Nevertheless, the economics of watershed protection and reservoir rehabilitation is an important topic. Although this white paper is not a complete, comprehensive analysis, it provides valuable insight into potential watershed/reservoir management strategies, the magnitude of costs, preferred analysis approaches, and research needs.

In this white paper, we provide an overview of costs of soil erosion and sedimentation based on existing literature and review features and costs of common soil erosion and sediment control methods. Appropriate in-field soil erosion management practices and their costs vary by site. Thus, we limit our analysis to estimating potential savings from implementing individual, in-field erosion control methods in a watershed to reduce future costs of dredging sediment from a reservoir. Then, we compare potential savings with the cost of management practices.

Our brief analysis indicates that in situations where the amount of accumulated sediment has not reduced a reservoir's usefulness, it could be more economical for the government to fund expenditures for management practices that reduce further erosion and sedimentation in a watershed than to rely on dredging in the future. However, our economic analysis is not complete because critical data are not available. We do not know, among other things, the source of sediment, how suitable management practices are for various locations in a watershed, or the number of acres that, from a technical and economic perspective, actually need these practices applied. We also do not know the benefits of reduced sedimentation. Evaluating the best approach for reducing additional sedimentation in watersheds with reservoirs for which dredging is being considered is a demanding task. A team approach that integrates expertise from various disciplines is essential for analyzing sediment prevention, erosion management, and reservoir rehabilitation. Ultimately, a variety of models that incorporate both physical and economic watershed characteristics are needed.

Soil Erosion and Sedimentation Costs

Erosion of cropland and streambanks of Kansas rivers increases sediment loads deposited in downstream reservoirs, alters fish and wildlife habits, and can cause significant damage to fields bordering streams, resulting in direct economic losses for landowners and Kansas citizens. Soil erosion causes loss of cropland, particularly along streambanks and riverbanks, and loss of soil productivity in crop fields and pastures. Cropland erosion from high-flow events in floodplains can disrupt farming operations. Soil erosion also has several off-site consequences (Figure 1); it can reduce reservoirs' water storage capacity, which affects public water supplies, flood control capability, and water availability for downstream navigation.

Suspended soil particles can affect viability of aquatic life, reduce recreational value of lakes and waterways, and increase operational costs for power plants, city water supplies, and navigation. Deposited sediment causes extensive damage to aquatic life, shortens reservoirs' useful life, and clogs navigation channels (Clark et al., 1985). Sediment can fill drainage channels, such as ditches and culverts, causing localized flooding if not removed. Sediment also includes nutrients such as nitrogen and phosphorus that alter water quality and affect aquatic life. Other potential contaminants in sediment include agricultural pesticides and dissolved solids such as calcium, sodium, magnesium bicarbonate, and chloride ions (Clark et al., 1985).

Sedimentation caused by soil erosion can create significant societal costs (Mooney and Williams, 2007). For example, sedimentation can affect the enjoyment people derive from using waterways for recreation. MacGregor (1988) estimated that increased sedimentation at Ohio state park lakes reduces the economic benefit of recreation to out-of-state boaters by an average of \$0.49/ton of sediment. Bejranonda et al. (1999) examined effects of sedimentation on lakeside property values at 15 Ohio state park lakes and found that homeowners are willing to pay more for properties on lakes with less sedimentation. A broader, national estimate of the value of recreation loss attributable to sedimentation is between \$612 million and \$3.6 billion (2006\$[1]) (Tegtmeier and Duffy, 2004).

1 Amounts reported in 2006 dollars (2006\$) or 2005 dollars (2005\$) as indicated.

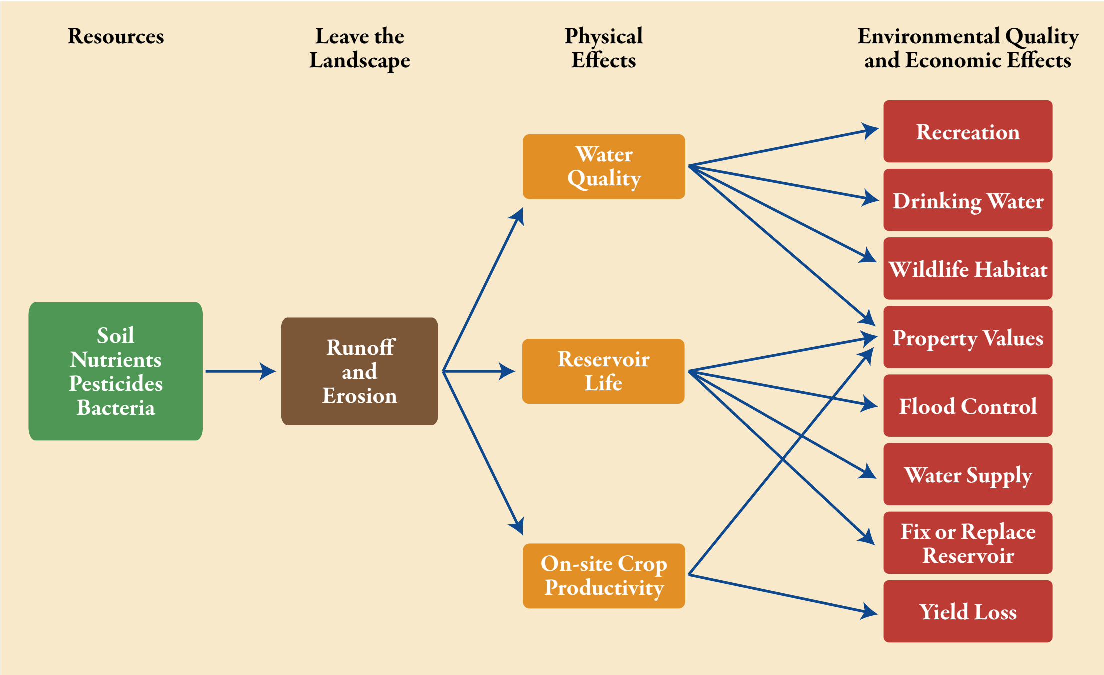
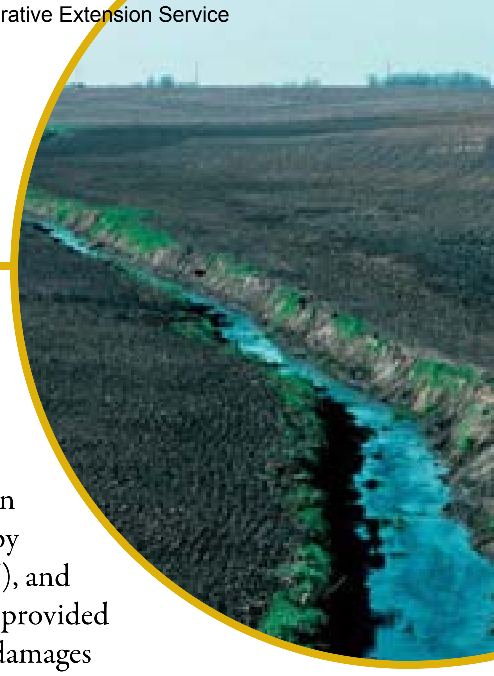


Figure 1. General effects of soil erosion

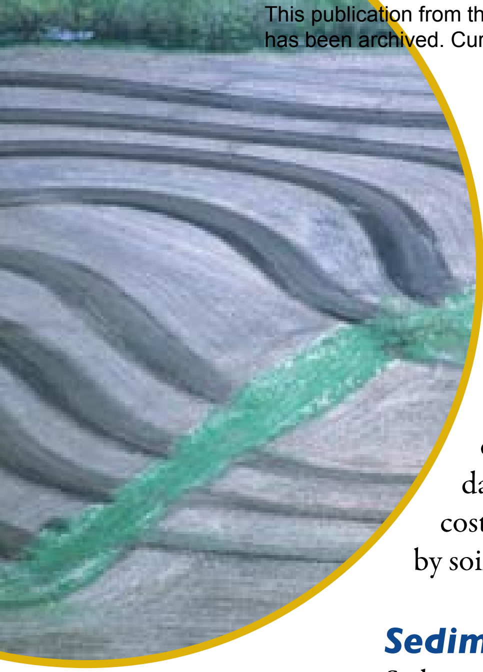


Hansen et al. (2002) examined costs of soil erosion and sedimentation to downstream navigation across different areas of the United States. Their results suggest that eroded soil poses no costs to navigation in areas with no downstream shipping channels or harbors but can create costs up to $5.67/ton of soil erosion (2006$) in areas where navigation is affected (Hansen et al., 2002). Tegtmeier and Duffy (2004) estimated that costs of sedimentation to shipping range from $345 million to $383 million (2006$). Moore and McCarl (1987) estimated effects of sediment on municipal water treatment, road drainage system maintenance, navigation channel maintenance, reservoir capacity deterioration, and hydroelectric power plant costs in Oregon. In this state alone, the total average cost of erosion for navigation channel maintenance, municipal water treatment, and country road and state highway maintenance is approximately $5.5 million annually (Moore and McCarl, 1987). Tegtmeier and Duffy (2004) updated figures from Ribaudo (1986); they estimated that annual costs of sediment removal from roadside ditches and irrigation channels throughout the United States range from $304 million to $895 million (2006$) and annual costs of flood damage attributable to erosion range from $215 million and $622 million (2006$). Nationally, estimates of annual costs of dredging inland waterways range from $282 million to $291 million, and reservoir siltation costs range from $274 million to $851 million (Clark et al., 1985; Ribaudo, 1986; Hansen et al., 2002; Tegtmeier and Duffy, 2004).

National Estimates of Total Damage from Water Erosion

Mooney and Williams (2007) reviewed literature on water erosion damages. They summarized work by Clark et al. (1985), Ribaudo (1986), and Tegtmeier and Duffy (2004), who provided national estimates of total annual damages attributable to water-based soil erosion ranging from $2 billion to $31 billion (2006$). These analyses included costs related to recreation, navigation, water storage facilities, municipal and industrial water users, water conveyance systems, and flooding. Estimated damages did not include all sectors of the economy or all possible activities and thus represent a partial estimate of the value of reduced soil erosion.

No comprehensive studies examining damages attributable to water-based soil erosion have been conducted since 1986 when Ribaudo calculated the value to society of reducing soil erosion by one ton based on potential annual damages from erosion in 1983. Per-ton values ranged from a low of $0.98/ton in the northern Plains to $11.29/ton in the Northeast (2006$). However, erosion of cropland and other agricultural soil declined considerably during the past 20 years (NRCS, 2007b), partly because improved farming practices and government programs such as the Conservation Reserve Program (CRP) retired highly erodible lands. According to the 2003 National Resources Inventory (NRCS, 2007b), the average rate for sheet and rill erosion on cropland declined from 4.0 tons/acre per year in 1982 to 2.6 tons/acre per year in 2003. Wind erosion rates dropped from 3.3 tons/acre per year in 1982 to 2.1 tons/acre per year in 2003.



Damages created by soil erosion probably changed over time, and previous estimates likely are inaccurate now. Today, actual per-ton damages depend on the value and cost of downstream activities affected by soil erosion.

Sediment Control Costs

Sediment source as well as type and effectiveness of erosion management strategies affect costs of reducing sedimentation. Sediment sources include upland landscape areas (e.g., cultivated fields, poorly maintained pastures, construction sites, and streambanks), sediment previously deposited in floodplains, and in-stream sources. Knowing the distribution of sediment sources can help determine what management strategies to use. For example, if sediment occurs mainly from high-flow events causing streambank erosion, it might be economically efficient to target streambanks rather than cultivated fields. However, if most erosion and sedimentation occurs during infrequent but heavy precipitation and water flow events, targeting erosion control strategies to these events rather than to average rainfall events might be useful. Presence of contaminants, such as phosphorous or other chemicals, in sediment also could influence source targeting decisions.

Stakeholders must consider several questions regarding effectiveness of soil erosion management strategies. How effective are the variety of in-field management strategies at reducing soil erosion? If we implement practices to reduce field erosion, what is the effect on reservoir sedimentation? Will sediment already accumulated in streambeds or low-lying areas continue negatively affecting water bodies even if sediment loads from new sources decline? It is likely that reducing sedimentation will require a multi-strategy approach.

Strategies and Costs for Reducing Soil Erosion and Controlling Sediment

Sediment reduction strategies include keeping sediment in place on upland landscapes, flood plain management, upstream sediment traps, and dredging reservoirs once sedimentation occurs. Landscape management encompasses a variety of soil conservation measures. For example, no-till systems leave crop residue on fields to reduce soil disturbance and erosion during wind and rainfall events. Alternatives include cropping rotations that increase crop and residue cover, vegetative buffers and CRP land, contour farming, and terraces.

According to the 2003 National Resources Inventory (NRCS, 2007b), soil erosion in Kansas declined from 1987 to 1997 (Figure 2). Cultivated cropland remains the largest erosion source; but, like other land use categories, erosion from this source is declining. This decline likely is due to increased use of conservation practices and enrollment of land in the CRP. Still, we must ask: Has reduced soil erosion decreased reservoir sedimentation?

Putnam and Pope (2003) reported results of time-trend tests from 1970 to 2002 for sediment sampling sites in Kansas. Most of the 14 sampling sites, including five of the six sites upstream from reservoirs, exhibited decreasing suspended sediment concentrations, but only two sites had trends that were statistically significant.

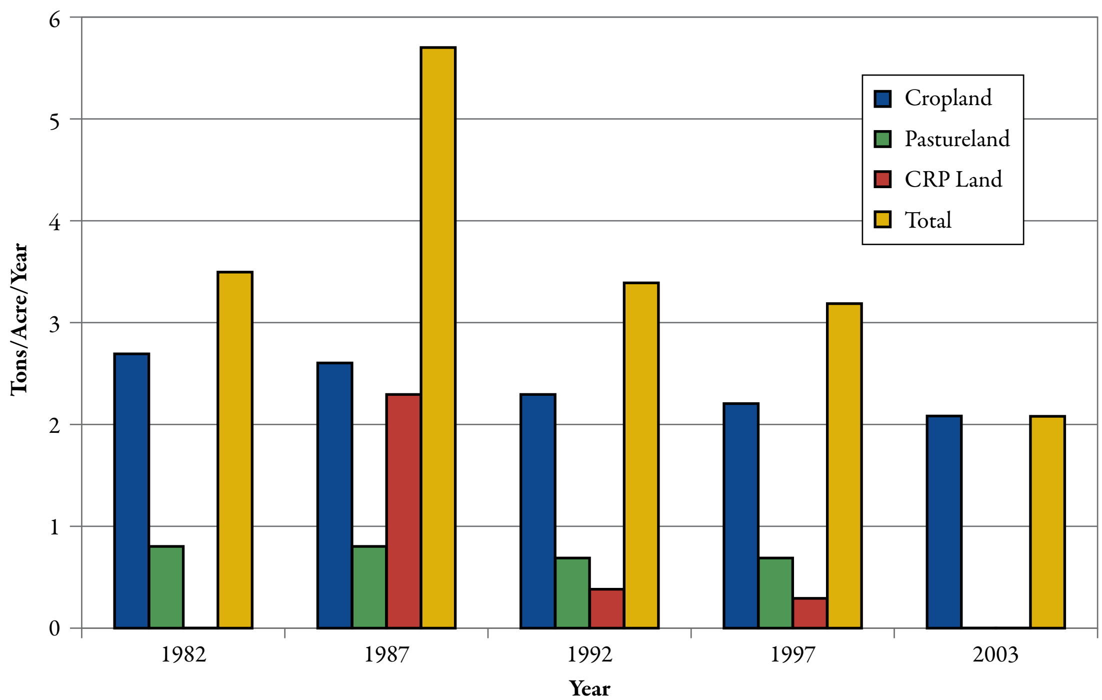


Figure 2. Soil erosion trends in Kansas
Data source: NRCS (2007b). Erosion rates for pastureland and CRP land were not reported in 2003.

Increasing suspended sediment concentrations occurred at three sites but were not statistically significant. Both sites that exhibited statistically significant decreasing suspended sediment concentrations have a large number of watershed impoundments (i.e., structures designed to trap sediment) in their respective drainage basins. The relationship between percentage of the watershed affected by impoundments and suspended sediment concentration for 11 sites indicated that suspended sediment concentration decreases as number of watershed impoundments increases. Implementing other conservation practices, such as terracing and contour farming, could further reduce suspended sediment concentrations.

The following sections contain rough cost estimates of various erosion management strategies; applicability and economic viability of these strategies vary greatly by location. The ultimate decision to adopt a specific management strategy depends on several factors including physical, biological, and economic characteristics of a site and land managers' capabilities and risk preferences.

In-Field Strategies

Altering Residue Cover with Crop Rotations. Plant residue management is one strategy used to reduce soil erosion. Crop rotations and tillage methods affect the amount of plant residue left on the soil surface. Al-Kaisi (2000) provided a relative ranking of erosion from selected cropping systems in Iowa. Fallow ground had the highest rate of erosion followed by a corn-soybean rotation. Increasing the amount of corn relative to beans in the cropping rotation reduced erosion as did adding a permanent cover crop or small grain crop, such as oat. Devlin et al. (2003) reported that the cost of using soil-conserving crop rotations is $0.00/acre (Table 1). However, actual costs of crop rotations vary by site, and changes in commodity prices cause the relative profitability of rotations to change from year to year.

Contour Farming. Contour farming uses field operations that follow the contour of the field rather than move straight up and down field slopes. This method requires additional labor and time and

Table 1. Cost for implementing sediment management practices

Practice	Cost
Altered crop rotations	$0.00/acre/year[b]
Contour farming without terraces	$6.80/acre/year[b]
Land retirement	$30 to more than $82/acre/year[f]
No-till	$0.00/acre[b] -$37.00 to more than $37.00/acre[g] -$3.81 to $1.51/acre[d]
Riparian forest buffer	$585/acre establishment plus annual loss of annual income (land rent)[h]
Sediment trap	$1,300 establishment cost per acre of construction area drainage to trap[a]
Streambank stabilization	$5,381 establishment cost per acre of CRP ($12.31/linear foot)[g] $3,252/acre of CRP after accounting for preserved land value and income[h]
Terraces	$0.66 to $3.30/linear foot depending upon slope[e]
Terraces with tile outlet	$40 establishment cost per acre plus annual cost of $13.60/acre[b] $1.05/linear foot[e]
Terraces with grassed waterways and contour	$30 establishment per acre plus annual cost of $13.60/acre in field plus loss of annual income (land rent)[b]
Vegetative buffers	$100 establishment cost per acre plus annual loss of land rent[b] $73 establishment cost per acre plus loss of annual income (land rent) – Annualized cost $63.29/acre[c] $104 establishment cost per acre plus annual loss of annual income (land rent)[j]
Waterways (grass) including topsoiling	$870 establishment cost per acre plus loss of annual income (land rent)[e]

[a] California Stormwater Quality Association (2003)
[b] Devlin et al. (2003)
[c] KSU-VegetativeBuffer estimate; Smith and Williams (2007)
[d] Langemeier and Nelson (2006)
[e] NRCS (2007a)
[f] Taylor et al. (2004)
[g] Williams et al. (2007)
[h] Williams et al. (2004)

might necessitate using equipment with reduced widths, which reduces efficiency. Clark et al. (1985) reported that contour farming practices reduced suspended residue by 25% to 50%. Contour farming costs a few dollars per acre on fields with consistent contours and is significantly more expensive on farms with variable field topography; Devlin et al. (2003) reported contour farming costs approximately $6.80/acre (Table 1).

Land Retirement. The CRP and other programs that retire land from crop production and require establishment of permanent cover can reduce soil erosion. The major cost of the CRP to land managers is the value of lost production. Therefore, the CRP pays an annual rental rate to those who enroll land. Kansas CRP payments range from $30/acre to $82/acre (Table 1; Taylor et al., 2004). Land not currently enrolled in CRP likely is more profitable in crop production and will require a larger incentive payment than land currently enrolled. Cash rental payments landlords received for renting nonirrigated land in northeast Kansas in 2006 averaged $69/acre. The statewide average is $39/acre and ranges from $26/acre to $69/acre (Dhuyvetter and Kastens, 2006).

No-Till. No-till is a form of conservation tillage in which chemicals are used in place of tillage for weed control and seedbed preparation. In a 100% no-till system, the soil surface is never disturbed except for planting or drilling operations. Two other forms of tillage, reduced tillage and rotational no-till, involve light to moderate use of tillage equipment. These methods also control erosion and nutrient runoff but are not as effective as 100% no-till.

Previous research shows that profitability of no-till systems varies. Dhuyvetter et al. (1996) reviewed nine studies including 23 comparisons between no-till and other tillage systems for various crops and rotations in the Great Plains and found that no-till had higher returns than conventional tillage in eight comparisons. Most of these comparisons were either wheat-fallow or wheat-sorghum-fallow rotations. Williams et al. (1990) found that net returns from no-till for continuous corn and corn-soybean rotations were higher than from conventional tillage in several government commodity program designs. In contrast, no-till had lower average net returns than conventional tillage for continuous soybean. Similarly, average net returns from no-till were less than from conventional tillage for wheat and grain sorghum in the central Plains (Williams et al., 2004). In Texas, no-till had higher net returns than conventional tillage for three cropping rotations: sorghum-wheat-soybean, wheat-soybean, and continuous wheat, but conventional tillage had higher net returns for continuous sorghum and soybean (Ribera et al., 2004).

Pendell et al. (2005) reported that net returns for no-till were higher than for conventional tillage for continuous corn in northeast Kansas. A recent study (Williams et al., 2007) of five cropping systems in northeast Kansas showed that returns from no-till compared with conventional tillage ranged from a negative $37.25/acre to a positive $37.12/acre; compared with reduced tillage, returns from no-till ranged from a negative $40.10/acre to a positive $21.14/acre depending on crop rotation (Table 1). Langemeier and Nelson (2006) reported that reducing tillage had a small effect on production costs in northeast

Kansas but significantly reduced soil erosion. Changing from conservation tillage to no-till for a corn-soybean rotation could reduce production costs by $3.81/acre, assuming no yield change, and soil erosion by 15% to 42% depending on soil type; changing to no-till in a sorghum-soybean-wheat rotation could increase production costs by $1.51/acre and reduce soil erosion by 2.1% to 26.9% (Langemeier and Nelson, 2006).

Nationally, the percentage of planted cropland in a no-till system increased from 6% in 1990 to 22% in 2004 (CTIC, 2005). However, these data are just a snapshot of the total no-till acres in a given year. The number of continuous no-till acres, which are important for soil erosion control, is less than the total. Dhuyvetter and Kastens (2005) summarized no-till adoption data for Midwest crop production. In 2004, no-till use was highest in soybean at 36.9%, up from 26% in 1994, and next highest in grain sorghum at 33.2%, up from 13.6%. No-till use in corn and fall small grains was 17.8% and 18.5%, respectively. Overall adoption of no-till in Kansas (21.2%) was slightly less than in the Midwest region (24.8%). In central and eastern Kansas, no-till use is increasing primarily because of lower costs, but higher yields and the associated revenue provide incentives for no-till adoption in western Kansas (Dhuyvetter and Kastens, 2005).

Large-scale adoption of no-till is relatively slow, indicating many farm managers regard it as unprofitable or that changing tillage practices has high transaction costs. We need to learn more about the types and magnitude of incentives that could encourage farm managers not already using no-till to adopt this practice.

Riparian Forest Buffers. Riparian forest buffers are areas of forested land adjacent to streams, rivers, or other water bodies. Shrubs and grasses often are located upslope from the trees to reduce nutrient and sediment losses from agricultural fields, improve runoff water quality, and provide wildlife habitat (Goard, 2006). Because of these societal benefits, several federal and state programs encourage installation and maintenance of riparian forest buffers. Establishment costs for a riparian forest buffers with trees, shrubs, and grass range from $243/acre to $970/acre with an average of $585/acre (Table 1; Williams et al., 2004). Annual income is lost from land in the buffer that can no longer be cropped, but the cash rental rate for land in the area is approximately the same as the amount of lost income.

Sediment Traps. Sediment traps, or basins, are excavated areas designed to temporarily impound runoff water long enough for suspended sediment to settle out. Typically, these structures are temporary and used to control erosion from construction sites. Sediment traps can be constructed along a watercourse or between a field and an outlet to a stream by excavating or forming an earthen embankment across a drainage area or waterway. Limited information is available on cost and effectiveness of these structures in agricultural watersheds; however, the California Stormwater Quality Association (2003) reported costs of $1,300/acre of drainage (Table 1). Costs for sediment traps or impoundments for agricultural erosion control vary by site because of differences in area and characteristics of land from which runoff is collected; removal and disposal of accumulated sediment add to the cost.

Terraces. Terraces are embankments constructed perpendicular to the slope of a field; they reduce the length of a field slope, catch water flowing off the slope, reduce the rate of runoff, and allow soil particles to settle out. Terraces can increase the time needed complete field operations because farm managers are unable to use larger or wider equipment, and loss of some farmable land might occur depending on slope and whether part of the terrace must be seeded with grass. Terrace construction costs include earth work for regrading land and can be several hundred dollars per acre. Carman (2006) reported construction costs ranging from $1/linear foot to $6/linear foot, and the NRCS (2007a) reported costs of $.66/linear foot to $3.30/linear foot depending on slope (Table 1).

Terraces with Grassed Waterways and Contour Farming. Grassed waterways are used to prevent erosion and gully formation and also function as outlets for water from terraces. Vegetative cover slows water flow and minimizes channel surface erosion (Green and Haney, 2006). Grassed waterways might require removing land from production, which reduces potential income, and also affect efficiency of field of equipment, which increases time and cost of field operations. Maintenance includes harvesting and marketing forage, repairing rills and gullies, and removing accumulated sediment. Establishment costs include field grading and vegetation establishment and usually are less than costs for terraces. Devlin et al. (2003) estimated that grassed waterway costs include a onetime $30/acre cost, $13.60/acre for all acres in the field, and the annual loss of income from land in the waterway (Table 1).

Vegetative Buffers. Vegetative buffers are land areas maintained in permanent vegetation that reduce nutrient and sediment losses from agricultural fields, improve runoff water quality, and provide wildlife habitat. Several federal and state programs encourage installation and maintenance of vegetative buffers. Establishment costs for vegetative buffers average $104/acre (Table 1; Williams et al., 2004) plus annual loss of income from land in the buffer that can no longer be cropped. We used the K-State Vegetative Buffer Decision-Making Tool (Smith and Williams, 2007) to estimate annualized costs. Establishment costs for a buffer using native grass are $73.12/acre without any cost-share or incentive payments, and annualized costs including loss of income are $63.29/acre (Table 1).

In-Stream Strategies

Streambank Stabilization and Bendway Weirs. Various strategies are used to reduce streambank degradation, and several can be combined for effective streambank stabilization. Willow posts can be planted on the outer bend of smaller streams to create a natural riparian zone that slows water flow and dissipates energy that will erode the bank toe. Posts usually are 3 to 4 inches in diameter and 10 to 14 feet tall. Depending on bank height, three to five rows are planted 4 feet apart. Other strategies include bendway weirs, stone toes, pools and riffles, and stream barbs. These methods use rock structures to slow or divert water flow and protect the bank toe. Bendway weirs are jetties constructed at an upstream angle of 10 to 25 degrees that directs water toward the center of the stream. Typically, they are less than half the stream width in length and slightly higher than low water. Sediment tends to collect on the downstream side of weirs. Bendway weirs generally are suitable for larger watercourses, such as sections of the Little Blue River in Washington County, Kansas. A stone toe is a row of large rocks along the toe of the outside bank of a bend. Perpendicular "keys" are constructed at intervals along the bank to protect against undercutting. In the pool and riffle method, a series of riffles (i.e., rock and sand bars) are constructed across a stream. These slow stream flow by breaking a steeply sloping stretch of streambank into a series of gently sloping sections (Johnson, 2003). Other methods include placing tree revetments or riprap on the outer bank of a bend. Tree revetment involves securing cut trees, often cedars, at the bank toe. The trees slow stream flow and trap sediment from the eroding bank. Eventually, willow and cottonwood seedlings will sprout and grow, re-establishing a riparian zone. Riprap is large rocks placed on the face of a bank to resist scouring from water flow.

Williams et al. (2004) analyzed costs of streambank stabilization based on actual construction and establishment costs at 13 sites on a 35-mile stretch of the Little Blue River in Washington County, Kansas (Table 2). Each site required establishing a 125-foot-wide riparian buffer consisting of trees, shrubs, and grass; 100 feet of the width was CRP land (Figure 3). Land area enrolled in the CRP ranged from .8 acres to 4.6 acres with an average size of 3 acres.

Table 2. Construction and establishment costs of streambank stabilization by site[a]

Site	Length in feet	CRP Acres	Total Costs	Percent Equipment & Labor[b]	Percent Materials[c]	Cost per Acre[d]	Cost per Linear foot
1	1,250	2.9	$18,261	32.5%	67.5%	$6,364	$14.61
2	925	2.1	16,524	32.0%	68.0%	7,781	17.86
3	1,140	2.6	23,988	20.3%	79.7%	9,166	21.04
4	882	2.0	17,410	29.5%	70.5%	8,598	19.74
5	2,016	4.6	12,552	19.3%	80.7%	2,712	6.23
6	336	0.8	3,019	82.9%	17.1%	3,914	8.99
7	1,049	2.4	8,392	72.3%	27.7%	3,485	8.00
8	1,255	2.9	20,814	65.3%	34.7%	7,225	16.58
9	1,188	2.7	14,488	61.5%	38.5%	5,312	12.20
10	1,150	2.6	10,395	61.6%	38.4%	3,937	9.04
11	1,983	4.6	35,073	23.0%	77.0%	7,704	17.69
12	1,980	4.5	16,937	75.6%	24.4%	3,726	8.55
13	1,888	4.3	12,008	54.2%	45.8%	2,771	6.36
Average	1,311	3.0	$16,143	42.2%	57.8%	$5,592	$12.31

[a] Table adapted from Williams et al. (2004) with permission

[b] Percentage of total costs for engineering and design, equipment, and labor to prepare the site including planting grass seed

[c] Percentage of total costs for material including rock, trees, shrubs, grass seed, tree shelters, and chemicals

[d] Cost per acre in the CRP stabilization area

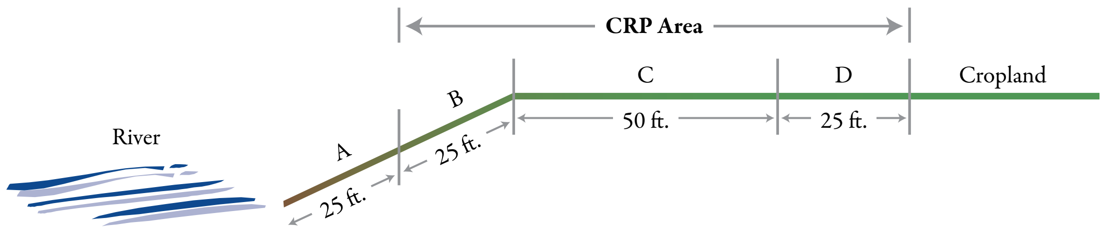


Figure 3. Streambank stabilization profile
A. Willow (4-foot spacing) and cottonwood (6-foot spacing)
B. Cottonwood, silver maple, and/or sycamore trees (8-foot spacing)
C. One row each of green ash, black walnut, and burr oak trees (10-foot by 12-foot spacing) and one row each of choke cherry, fragrant sumac, and American plum shrubs (6-foot by 6-foot spacing)
D. Native grass mixtures of big bluestem, Indiangrass, switchgrass, sideoats grama, and western wheatgrass
Figure adapted from Williams et al. (2004) with permission

Construction equipment reshaped the streambank so it rose 1 foot from stream level to field level for every 3 feet of distance from stream to field. Bank width from stream level to field level was approximately 55 feet. Approximately every 175 feet, bendway weirs were constructed from rock boulders one-eighth ton to 1 ton in size (Oertal, 2002). A typical weir was one-third the width of the stream at low water and about 2 feet high, 18 feet wide at the base, and 10 feet wide on top.

Per-site construction and establishment costs ranged from $3,019 to $35,073 with an average of $16,143. Cost per linear foot of streambank ranged from $6.23 to $21.04 with an average of $12.31. Cost per acre, based on acres in the 100-foot-wide stabilization portion, ranged from $2,712 to $9,166 with an average of $5,592 (Table 2). The annualized cost, using a 15-year period and 6% interest rate, ranged from $279/acre per year to $944/acre per year with an average of $576/acre per year. Landowners also incurred some annual maintenance costs.

Landowners receive several benefits from streambank stabilization including income or rental payments from preserved cropland and market value of land saved from erosion. Streambank stabilization projects typically require that cropland be taken out of production to install the vegetative buffer but result in long-term net savings of land. If the project can be enrolled in the CRP, landowners will receive annual rental payments. Cost-share payments, subsidies, or other incentives also might be provided. For purpose of cost calculations, the value of CRP, cost-share, and other incentive payments are not included in our analysis. The present value of CRP payments over 15 years ranges from $724 to $3,602 with an average of $1,907. Present value of rental income not lost because of erosion over 15 years ranges from negative $1,211 to $4,805 with an average of $474. Present market value of cropland preserved after 15 years ranges from $990 to $16,985 with an average of $6,121 (Table 3).

In-Reservoir Management

Dredging. Dredging is the removal of accumulated sediment from bottoms of lakes, reservoirs, or other water bodies by mechanical, hydraulic, or pneumatic means (Hudson, 1998). Sediments often are removed from rivers and ports for navigation and boating purposes. Less frequently, dredging is used in lakes and reservoirs to reclaim water storage capacity. Dredging

Table 3. Present value and annualized present value of streambank stabilization costs

	Average Site	$/acre[a]
Costs without potential landowner benefits:		
– Total construction and establishment cost	–$16,143	–$5,381
– PV of annual maintenance costs over 15 years	–$206	–$69
= Present value of costs of project to land owner	–$16,349	–$5,450
Annualized present value of costs[b]	–$1,670	–$557
Costs with potential landowner benefits:		
– Total construction and establishment cost	–$16,143	–$5,381
– PV of annual maintenance costs over 15 years	–$206	–$69
+ PV of rental income effect over 15 years	+$474	+$158
+ PV of net land conserved in the terminal year (year 15)	+$6,121	+$2,040
= NPV of costs of project to land owner	–$9,754	–$3,252
Annualized present value of costs	–$997	–$332

[a] Cost per acre in the CRP stabilization area
[b] Annualized using a 5.81% discount rate

Table 4. Dredging costs[a]

Data Source	Cost per cubic yard	Dredging and disposal cost per acre-foot	Tuttle Creek sediment deposited as of 2005 (acre-feet)	Cost to remove sediment deposited by 2005
Corps of Engineers[b]	$3.75	$6,049.99	165,000	$998,247,938
Iowa Minimum[c]	$2.55	$4,115.58	165,000	$679,070,469
Iowa Maximum[c]	$5.31	$8,574.48	165,000	$1,414,789,093
Kansas Minimum[d]	$3.00	$4,839.99	165,000	$798,598,350
Kansas Maximum[d]	$8.67	$13,983.22	165,000	$2,307,230,768

[a] Costs provided in 2005 dollars
[b] USACE (2005)
[c] Iowa Department of Natural Resources (K. Jackson, personal communication, Nov. 1, 2006)
[d] Kansas Water Office (2004)

costs provided by the Iowa Department of Natural Resources (K. Jackson, personal communication, Nov. 1, 2006), U.S. Army Corps of Engineers (USACE; 2005), and Kansas Water Office (2004) range from $2.55/cubic yard to $8.67/cubic yard (Table 4) and include dredging, equipment mobilization, and sediment disposal. Mobilization costs represent a higher percentage of overall costs in smaller projects. Other organizations and companies report hydraulic dredging costs ranging from $4.00/cubic yard to $14/cubic yard (Illinois Environmental Protection Agency, 2005; Kansas Biological Survey, 2005; Dredging Specialists, 2007).

Other In-Reservoir. Information on options for and economics of in-reservoir sediment management strategies other

than dredging is limited. Baker (2007) reviewed some options for managing sediment within a reservoir. However, in-reservoir management strategies have limited effectiveness for reducing sediment loads introduced from the watershed and transported via major tributaries. In some areas, shoreline erosion control techniques reduce sediment loading in reservoirs, but sedimentation from shoreline erosion likely is quite small compared with sedimentation from off-site erosion.

One dredging alternative does not involve sediment management; rather, it raises the legal level of water storage elevation in the lake. This approach is a temporary measure. Costs for this strategy are very site specific and can include purchasing additional land, loss of income from cropland or other uses, habitat loss on environmentally sensitive sites, and relocation of roads, walks, ramps, or other structures.

Huffaker and Hotchkiss (2006) grouped sediment control strategies into three broad approaches:

1. Reducing inflow by controlling erosion in the catchment area (i.e., watershed)
2. Diverting sediment by routing it to off-stream reservoirs or sluicing it through a dam before it can settle
3. Removing accumulated sediment by hydraulic flushing, hydraulic dredging, or dry excavation.

They also examined another sediment removal strategy called hydro suction dredging, but their research was limited to developing a theoretical economic model of this process and determining the optimal volume of reservoir water to allocate to this sediment removal strategy. (Huffaker and Hotchkiss, 2006). In hydro suction dredging, one end of a pipeline is located at the reservoir bottom upstream from a dam. The pipeline extends through the dam to a downstream discharge point and draws sediment-entrained water into the pipe for transport downstream. This sluicing process relies on the availability of "surplus" water that drives suspended sediment beyond the reservoir. The "surplus" is annual inflow beyond storage capacity that is involuntarily spilled.

Targeting Best Management Practice Implementation

Identifying land in a watershed that should be treated is crucial for reducing sedimentation. Because limited resources are available to fund implementation of best management practices (BMPs), it is most cost-effective to target areas that contribute most to sedimentation. Targeting attempts to identify BMPs and land that have the greatest sediment reduction benefits relative to cost.

Khanna et al. (2003) developed a framework that includes a hydrological model that uses geographic information system (GIS) data and an economic model to determine cost-effectiveness of retiring land using the Conservation Reserve Enhancement Program (CREP) to reduce sediment. They applied the model to a 61,717-acre Illinois watershed and assumed that sloping cropland adjacent to a stream or riparian buffers within 900 feet of a stream were eligible for land retirement using the CREP. Model results revealed that 8,172 acres could be targeted. To achieve 20% sediment reduction for a 5-year storm, 11%

of the acres in this area needed to be in the CREP; this had an average cost of $31/ton. With a 30% reduction goal, average cost was $47/ton. Marginal costs rose from $29/ton at a 10% reduction goal to $117/ton at a 30% reduction goal. To achieve a 20% sediment reduction goal, the cost-effective land rental payment is $54/ton times the tonnage of reduction per acre. Results also showed that most land selected for enrollment was from highly sloping and highly erodible areas rather than less erodible flat floodplains.

> …it is preferable to capture the sediment at the end of the flow channel by retiring parcels adjacent to the water body rather than reduce sediment generated by retiring upslope parcels that are farther from the water body. Those parcels with high on-site erosion and high sediment trapping effectiveness are also given priority. (Khanna et al., 2003, p. 547-548)

Yang et al. (2003) used a similar approach to examine CREP use across 12 contiguous Illinois watersheds to reduce sedimentation in the Illinois River by 20%. For a 5-year storm event, a 20% reduction goal translated into a 32,000-ton sediment reduction in the 617,763-acre region. However, only a 900-foot-wide area along all streams and tributaries was eligible for CREP establishment. To achieve 20% reduction at the aggregate level (i.e., total watershed) at least cost, sediment reduction in the 12 watersheds ranged from 4.1% to 33.3%. A uniform standard applied to all 12 watersheds was more costly and required that more cropland be placed in CREP.

> Regardless of the variability across watersheds, cropland selected for retirement in all watersheds is closer to water bodies, more sloping, more erosive, and more likely to receive larger volumes of upland sediment flows than the cropland not selected for retirement. (Yang et al., 2003, p. 261)

This study focused on one management practice and did not consider sediment reduction benefits, optimal amount of sediment reduction, or dredging costs.

To extend research by Khanna et al. (2003) and Yang et al. (2003), Yang and Weersink (2004) examined cost-effectiveness of targeting riparian buffers in a 36,077-acre agricultural watershed in Ontario by combining economic and hydrologic models with GIS data. Their model minimizes the loss of economic return from crop production, which varies by watershed location subject to fixed levels of sediment reduction goals for a variety of buffer strip widths in each sub-catchment of the watershed. The model selects appropriate buffer strip widths and locations for five separate sediment reduction goals but does not estimate benefits of sediment reduction, determine the optimal level of sediment to control, suggest an optimal combination of management practices other than buffer strips, or consider dredging. Results indicated that cost-effective targeting results in buffer strip locations that vary across the watershed and are not necessarily located on sites adjacent to streams or having the greatest slopes. Marginal costs of sediment control increase as amount of land used for buffer strips and desired sediment reduction increase.

Yang et al. (2005) used a similar approach to examine spatial targeting of no-till to improve water quality and carbon retention benefits in a 90,470-acre agricultural watershed in Ontario. They reported that 9.7%, 16.5%, 25.2%, and 39.6% of the land must be in no-till to achieve 20%, 25%, 30%, and 35% sediment reduction, respectively. Their study assumed a corn-soybean-wheat cropping system. Average costs ranged from $10.89/acre per year to $12.26/acre per year. As the sediment reduction goal increased, additional no-till acreage needed and costs also increased. As expected, the percentage of land in no-till varied across subwatersheds because of the cost relative to amount of reduced sedimentation. Subwatersheds that required a higher percentage of no-till cropland to achieve the least-cost solution generally had higher land slope and more erosive soils; costs for using no-till were relatively lower in these areas.

Within a watershed, the number of subwatersheds that might need to be modeled affects the number of regions that can be used to determine a cost-effective targeting approach for sediment reduction. Using data from Soil and Water Assessment Tool (SWAT) models of four Iowa watersheds, Jha et al. (2004) developed preliminary guidelines for subdividing a watershed into subwatersheds for modeling purposes. They found that predicted sediment yields are related to subwatershed size and delineation and suggested modeling studies should include sensitivity analyses with various subwatershed delineations to determine the appropriate level for actual analysis.

These spatial targeting studies reveal an important policy issue: How should BMPs for sediment control be spatially distributed to achieve optimal results (i.e., additional costs of sediment control equal the additional benefits)?

Dredging Versus Best Management Practices

We use Tuttle Creek Lake and its watershed as a preliminary case study to examine the economics of watershed protection and reservoir rehabilitation. Tuttle Creek Lake is a 14,000-acre impoundment in northeast Kansas at the lower end of the Big Blue River. The 9,628-square-mile watershed supplying the lake is largely agricultural. The majority of the watershed extends north into Nebraska, and the lower quarter is in Kansas. The USACE built Tuttle Creek Lake in 1962 for flood control, irrigation, water supply, recreation, fish and wildlife, low-flow augmentation, and navigation-flow supplementation for Missouri River barge traffic. The lake provides up to 50% of the Kansas River flow; this river is a public water source for Topeka, Lawrence, and Kansas City. Table 5 provides additional information about the lake and watershed.

As of 2005, which was 43 years since the reservoir was completed, Tuttle Creek Lake contained 266,199,450 cubic yards of sediment. Although Table 4 shows a range of dredging costs, we assume the cost of dredging is $5.00/cubic yard for the remainder of our discussion. Total estimated cost of removing sediment from Tuttle Creek Lake at $5.00/cubic yard is $1,330,997,250 (which translates to $301/watershed-acre). Calculating the annual payment on a loan for the total dredging cost provides perspective; assuming a 7% interest rate and 43-year period, the loan payment is $98,541,573/year or $22.28/acre of cropland in the watershed. Clearly, dredging is an expensive option.

Table 5. Tuttle Creek Lake and watershed characteristics, dredging costs, and equivalent per-acre expenditures

Characteristics	
Original conservation storage pool (acre-feet)	425,000
Sediment deposited as of 2005 (acre-feet)	165,000
Sediment deposited as of 2005 (cubic yards)[a]	266,199,450
Drainage area (square miles)	9,600
Drainage area (acres)	6,144,000
Pastureland (%)	16%
Pastureland (acres)	983,040
Cropland (%)	72%
Cropland (acres)	4,423,680
Other (%)	12%
Other (acres)	737,280
Dredging Cost in 2005	
Cost per cubic yard	$5.00
Dredging and disposal cost per acre foot	$8,066.65
Sediment deposited as of 2005 (acre-feet)[a]	165,000
Cost to remove sediment deposited until 2005	$1,330,997,250
Onetime Equivalent Costs	
Cost per acre of cropland	$301

[a] Projected based on average annual sedimentation rate to 1999

Although dredging is effective at removing sediment, it does not prevent sedimentation. If accumulated sediment has not significantly reduced reservoir functions and benefits, it might be reasonable to forgo dredging and instead implement management practices that significantly reduce the need for future dredging. This decision will depend on sediment source, sedimentation rate with and without management practices, effectiveness and cost of management practices, dredging cost inflation, the planning horizon, and the discount rate used to calculate present values.

A detailed analysis of this decision for Tuttle Creek Lake or other reservoirs is beyond the scope of this paper. A complete study should consider costs, benefits, and the optimal level of sediment control, and determining the best approach to manage erosion and sediment requires a more detailed economic analysis of the number of acres within a watershed that can be treated with a variety of practices. The following analysis does not consider all costs and benefits. However, given a number of assumptions, we estimate how many acres of land can be treated with four individual management practices: vegetative buffers, no-till, terraces, and streambank stabilization.

Our analysis examines how many acres a management practice can be applied to if savings generated from reduced dredging finance the management practice (Figure 4). Estimated future savings from dredging costs avoided because of implementing sediment reduction management practices are a key component of this analysis. To determine these values, we estimate the reservoir sedimentation rate with and without management practices over a 20-year planning period. We also estimate the cost of dredging 20 years in the future based on the current rate of sedimentation versus a reduced rate of sedimentation that will result from implementing management practices. Because we do not know the specific effectiveness, cost, or combination of management practices that will be appropriate for each area in the watershed, we apply general assumptions to the entire watershed. Our analysis is limited to costs; therefore, we do not consider any benefits resulting from reduced erosion and sedimentation (Smith et al., 2007).

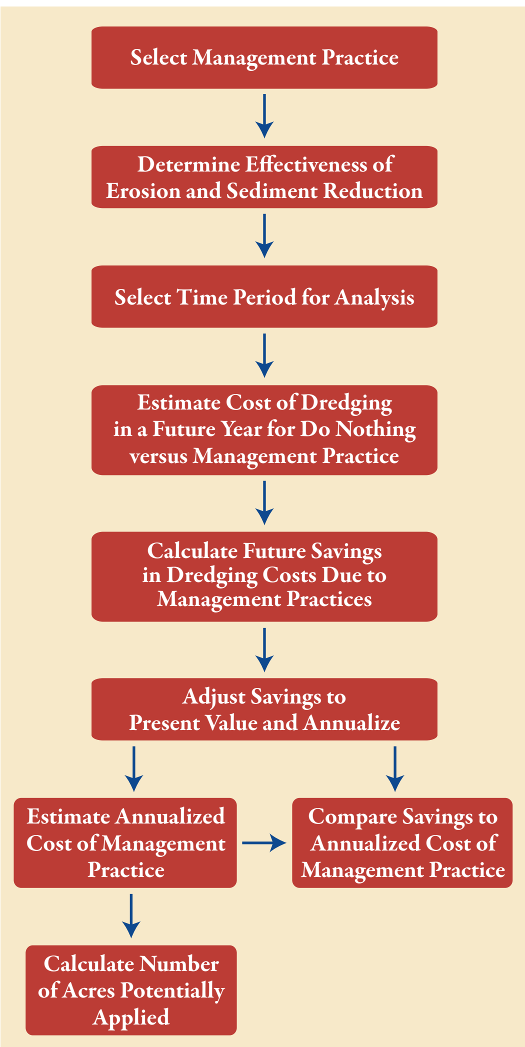


Figure 4. Method for comparing dredging with management practices

In the following scenarios, we use characteristics of the Tuttle Creek Watershed listed in Table 5. We examine four management strategies (i.e., vegetative buffers, no-till, terraces, and streambank stabilization) to determine how many acres they can potentially be applied to based on cost savings from reduced dredging. Cost savings for all strategies, summarized in Table 6, are based on a 2005 dredging cost of \$5.00/cubic yard inflated at a 5.81% annual inflation rate over the next 20 years and a 7% annual discount rate for the present value calculations (Appendix A). Dredging cost inflation rate is based on historical data (Figure 5).

In the first scenario, we assume vegetative buffers are 50% effective at reducing erosion and the sedimentation rate (Devlin et al., 2003). Compared with applying no management practices, installing vegetative buffers will result in 61,906,849 fewer cubic yards to dredge in 20 years. Estimated dredging cost savings are \$247,495,574 (2005\$), equivalent to \$55.95/cropland acre. Average annual savings over the 20-year period are \$5.28/cropland acre. The interpretation of this value is that spending \$5.28/acre per year on every acre of cropland in the watershed over the next

Table 6. Estimation of dredging cost savings from erosion reduction

	Management Practice				
	Do Nothing	Vegetative Buffer[a]	No-Till	Terrace	Streambank Stabilization[ab]
Dredging inflation	5.81%	5.81%	5.81%	5.81%	5.81%
Initial dredge cost ($/cubic yard)	$5.00	$5.00	$5.00	$5.00	$5.00
Sediment rate based on 1962 to 2005 (cubic yards per year)	6,190,685	6,190,685	6,190,685	6,190,685	6,190,685
Reduction in sediment rate due to management practice	0.00%	50.00%	75.00%	30.00%	90.00%
New sediment rate (cubic yards per year)	6,190,685	3,095,342	1,547,671	4,333,479	619,068
Number of years at this new sediment rate	20	20	20	20	20
Accumulated sediment over 20-year period (cubic yards)	123,813,698	61,906,849	30,953,424	86,669,588	12,381,370
Reduction in sediment to dredge compared with do nothing		61,906,849	92,860,273	37,144,109	111,432,328
Future dredging cost ($/cubic yard)	$15.47	$15.47	$15.47	$15.47	$15.47
Future dredge cost of 20-year accumulation	$1,915,459,554	$957,729,777	$478,864,888	$1,340,821,688	$191,545,955
Future savings in dredging cost		$957,729,777	$1,436,594,665	$574,637,866	$1,723,913,598
Discount rate for present value calculations	7.00%	7.00%	7.00%	7.00%	7.00%
Present value of future dredging cost[c]	$494,991,148	$247,495,574	$123,747,787	$346,493,803	$49,499,115
Present value of dredging savings compared with do nothing[c]		$247,495,574	$371,243,361	$148,497,344	$445,492,033
Annualized savings		$23,361,831	$35,042,747	$14,017,099	$42,051,296
Onetime savings per acre					
Savings per acre of cropland		$55.95	$83.92	$33.57	$100.71
Savings per acre of 50% of cropland		$111.90	$167.84	$67.14	$201.41
Annualized savings per acre over selected year period					
Savings per acre of cropland		$5.28	$7.92	$3.17	$9.51
Savings per acre of 50% of cropland		$10.56	$15.84	$6.34	$19.01
Management practice cost $/acre/year		$65.00	$10.00	$20.00	$332.04
Potential crop acres applied		8,985,320	3,504,275	700,855	3,166,132
Percentage of cropland in watershed		203.12%	79.22%	15.84%	71.57%
Management practice cost $/streambank mile/year					$4,024.73
Potential streambank miles					10,448

[a] We assume 1 acre of buffer treats 25 aces of cropland
[b] Cost for streambank stabilization includes an adjustment for landowner benefits (Refer to Table 3)
[c] Amounts reported in 2005 dollars

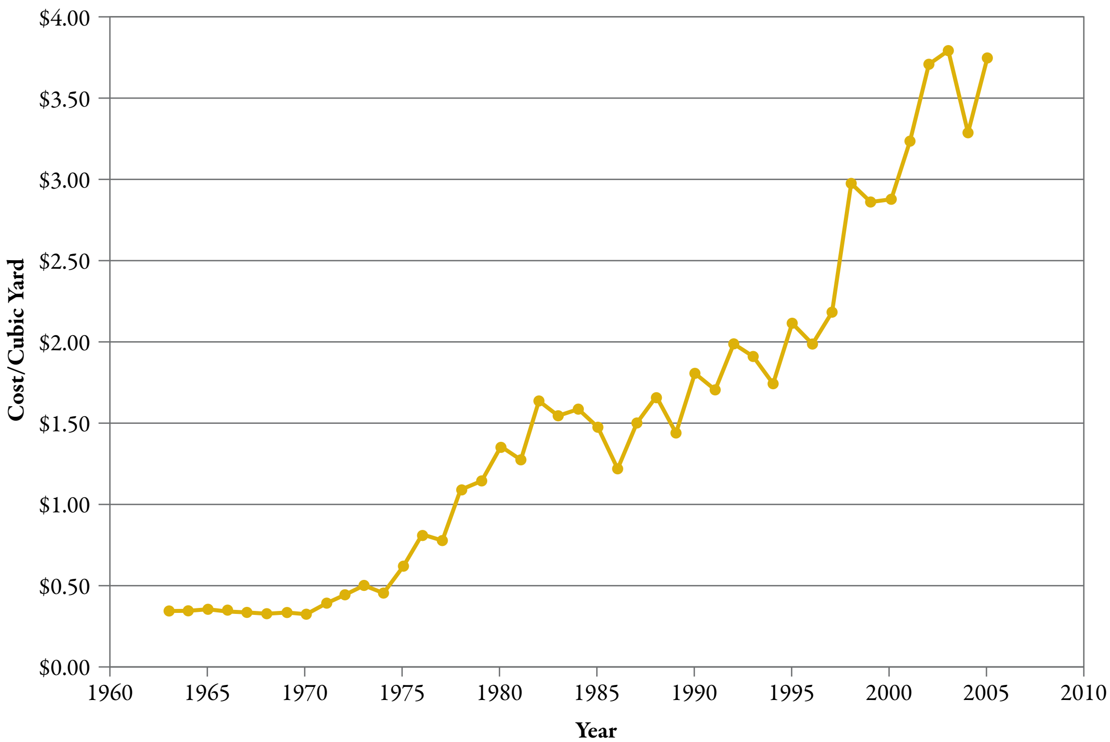


Figure 5. Historical dredging costs in nominal dollars
Data source: USACE (2005). Average annual increase is 5.8%.

20 years will equal savings from reduced dredging due to implementing buffer strips that are considered 50% effective. If only 50% of crop acres actually need management practices, $10.56/acre can be spent each year. If we assume the annualized cost of a vegetative buffer is $65/acre per year and each acre of vegetative buffer treats 25 acres (Smith, 2004), savings will cover costs when buffers are applied on up to 8,985,320 acres. This is more than twice (203%) the number of cropland acres in the watershed. Under these assumptions, using vegetative buffers appears more economical than dredging.

The second scenario examines no-till, which is considered 75% effective at reducing erosion and the sedimentation rate (Devlin et al., 2003). Compared with applying no management practices, using no-till will result in 92,860,273 fewer cubic yards to dredge in 20 years. Estimated dredging cost savings are $371,243,361 (2005$), equivalent to $83.92/cropland acre. Average annual savings over the 20-year period are $7.92/cropland acre. If only 50% of crop acres actually need management practices, $15.84/acre can be spent each year. At $10/acre per year, no-till can be used on 3,504,275 acres (79% of crop acres) in addition to any acres already under no-till. If no-till needs to be applied to less than 79% of cropland acres, savings from reduced dredging will pay for the cost of a program that provides $10/acre per year to use no-till.

The third scenario examines farmable terraces, which are considered 30% effective at reducing erosion and the sedimentation rate (Devlin et al., 2003). Compared with applying no management practices, installing farmable terraces will result in

37,144,109 fewer cubic yards to dredge in 20 years. Estimated dredging cost savings are $148,497,344 (2005$), equivalent to $33.57/cropland acre. Average annual savings over the 20-year period are $3.17/cropland acre. If only 50% of crop acres actually need management practices, $6.34/acre can be spent each year. We assume 223 linear feet of terrace are applied to each acre requiring terracing. Using the lowest establishment cost reported in Table 1 ($.66/linear foot), terrace establishment costs $147.18/acre. Annualizing this cost using a 10-year life and 6% interest rate results in a $20/acre per year cost. At this cost, terraces can be used on 700,855 acres (15.8% of crop acres) in addition to any acres that already have terraces. Fully evaluating whether establishing terraces is more economical than dredging requires determining how many acres of terraces are needed to reduce sedimentation.

In the final scenario, we assume streambanks are stabilized as described previously (Williams et al., 2004) and include buffer strips that are 90% effective at reducing erosion and the sedimentation rate. Compared with applying no management practices, streambank stabilization will result in 111,432,328 fewer cubic yards to dredge in 20 years. Estimated dredging cost savings are $445,492,033 (2005$), equivalent to $100.71/cropland acre. Average annual savings over the 20-year period are $9.51/cropland acre. If only 50% of crop acres actually need management practices, $19.01/acre can be spent each year. We assume that 100 linear feet of buffer width are required for each linear foot of streambank requiring stabilization. Therefore, 1 acre of land is required for every 436 feet of streambank stabilized. Costs for streambank stabilization are very site specific, but given the average costs reported in Table 3, annualized cost for a 15-year period is $332/acre excluding any cost-share and annual incentive payments but including benefits to the landowner in the form of annual income from and asset value of preserved land (Williams et al, 2004). At this cost, streambank stabilization can be used on 10,448 streambank miles. Assuming each acre of vegetative buffer in the stabilization area treats 25 acres (Smith, 2004), the stabilization project can treat 3,166,132 acres (71.6% of crop acres). If each stabilization acre treated only 10 acres, dredging cost savings could be used to treat 1,266,453 acres (29.6% of crop acres). Fully evaluating whether streambank stabilization is more economical than dredging requires determining how many streambank miles need to be stabilized to reduce sedimentation.

Our calculations do not reflect the optimum number of acres to which management practices should be applied. We do not know, from a technical and economic perspective, how suitable no-till, terraces, vegetative buffers, or streambank stabilization might be for various locations in the watershed or the number of acres or miles that actually need these practices applied. Numbers presented represent only the potential area to which these management practices can be applied based solely on cost savings from reduced dredging. The more expensive or less effective the practice, the fewer acres to which it can be applied. Estimates provide some perspective on dredging versus use of soil erosion management practices. Our brief analysis indicates that in situations where the amount of accumulated sediment has not reduced a reservoir's usefulness, it could be more economical for the government to fund expenditures for management practices that reduce further erosion and sedimentation in a watershed than to rely on dredging in the future.

From an economic perspective, the optimal level of sedimentation control is when the marginal (additional) benefits of control practices equal the marginal costs of implementing those practices. Because data on costs and benefits of sediment management practices are significantly limited, this paper provides only a rough analysis and does not consider benefits of BMPs versus dredging. Further, none of the articles we reviewed presented a model for or attempted a comprehensive analysis. In previous literature, alternative levels of sedimentation reduction simply are assumed. Although we do not attempt to quantify benefits of BMPs or dredging, we recognize that in-field BMPs might provide benefits, in addition to reduced sediment loads, that need to be accounted for in a comprehensive in-field versus in-reservoir management strategy comparison. For example, if an in-field strategy keeps more soil on a field, productivity of that field will be greater over time; this benefit is not realized as much through in-reservoir strategies. Other benefits associated with in-field strategies include reduced nutrient and pesticide loads entering the reservoir and increased wildlife habitat.

Sensitivity Analysis

Although many variables in our analysis are unknown, we perform sensitivity analyses on three variables that can affect the analysis: sedimentation rate, dredging cost, and discount rate. Because BMPs such as CRP land, terraces, and no-till have been established over time in the watershed, simply averaging sedimentation rates over the time period since the impoundment was constructed might overstate the future sedimentation rate. Therefore, we use alternative sedimentation rates (110%, 100%, 90%, 80%, and 70% of the original) in a sensitivity analysis (Figure 6). The

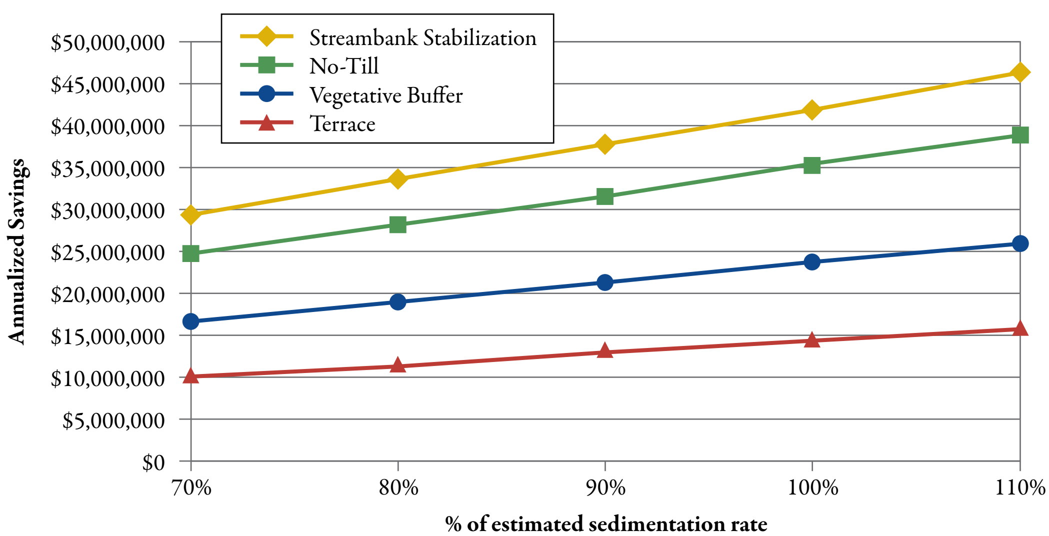


Figure 6. Annualized savings as a function of sedimentation rate

Annualized savings are avoided dredging costs that can be spent on BMPs. Annualized cost savings from reduced dredging at the original sedimentation rate for each management practice are reported in Table 7 and range from $14 million for terraces to $42 million for streambank stabilization. These values are represented at the 100% level in this figure.

original rate is 6,190,685 cubic yards/year. If the expected sedimentation rate without management practices is lower than the original rate, annualized savings decline. A lower sedimentation rate means fewer dredging costs are avoided by using BMPs. As a result, savings from reduced dredging costs cover application of BMPs on fewer acres or stream miles (Table 7).

Original dredging cost is $5.00/cubic yard (2005$). Future dredging costs are unknown and could be lower because of technological improvements. Therefore, we performed a sensitivity analysis using various dredging costs inflated at an annual rate of 5.81% (Figure 7). If dredging costs less in the future, annualized savings from reduced dredging decline and will cover application of BMPs on fewer acres or stream miles (Table 7). Alternatively, if dredging costs are higher than expected, annualized savings increase.

Table 7. Sensitivity of percentage of acres or potential streambank miles to which BMPs could be applied based on dredging savings for various sedimentation rates, dredging costs, and discount rates

Sedimentation rate (% of original)	70%	80%	90%	100%	110%
Percentage of cropland in watershed					
Vegetative buffer	142.2%	162.5%	182.8%	203.1%	223.4%
No-till	55.4%	63.4%	71.3%	79.2%	87.1%
Terrace	11.1%	12.7%	14.3%	15.8%	17.4%
Potential streambank miles					
Streambank stabilization	7,314	8,359	9,403	10,448	11,493
Dredging cost	$3.00	$4.00	$5.00	$6.00	$7.00
Percentage of cropland in watershed					
Vegetative buffer	121.9%	162.5%	203.1%	243.7%	284.4%
No-till	47.5%	63.4%	79.2%	95.1%	110.9%
Terrace	9.5%	12.7%	15.8%	19.0%	22.2%
Potential streambank miles					
Streambank stabilization	6,269	8,359	10,448	12,538	14,628
Discount rate	4%	5%	6%	7%	8%
Percentage of cropland in watershed					
Vegetative buffer	279.6%	251.8%	226.4%	203.1%	182.0%
No-Till	109.1%	98.2%	88.3%	79.2%	71.0%
Terrace	21.8%	19.6%	17.7%	15.8%	14.2%
Potential streambank miles					
Streambank stabilization	14,384	12,954	11,644	10,448	9,360

Discount rate also affects the analysis. The original discount rate is 7% per year. A lower discount rate discounts future values less. Therefore, future values discounted to the present are worth more (in 2005$) or are larger (Figure 8). The lower the discount rate, the larger the annualized savings from reduced future dredging. These larger savings can be used to implement BMPs on more acres or streambank miles (Table 7).

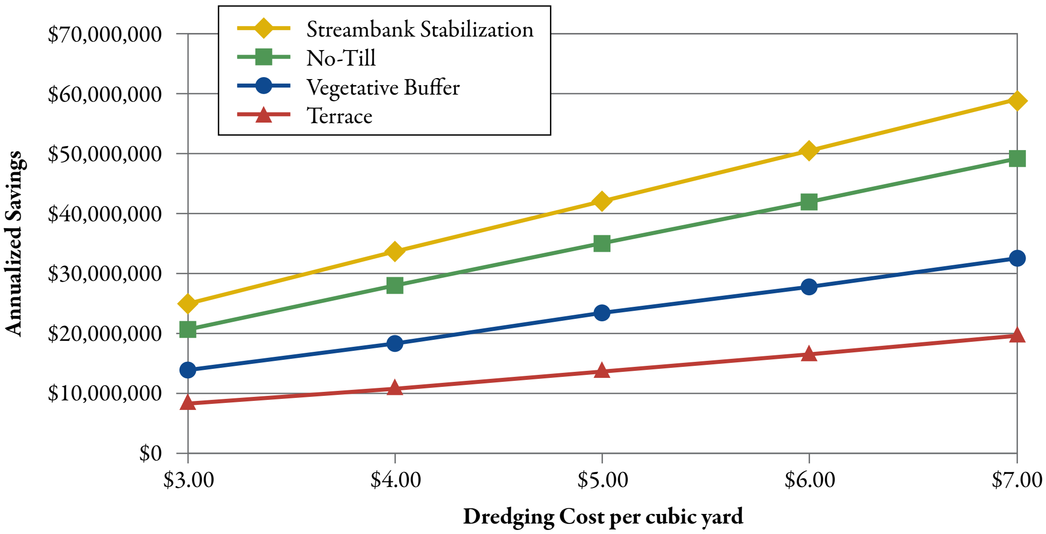


Figure 7. Annualized savings as a function of dredging cost
Annualized savings are avoided dredging costs that can be spent on BMPs. Annualized cost savings from reduced dredging at the original dredging cost for each management practice are reported in Table 7 and correspond to the $5.00 level in this figure.

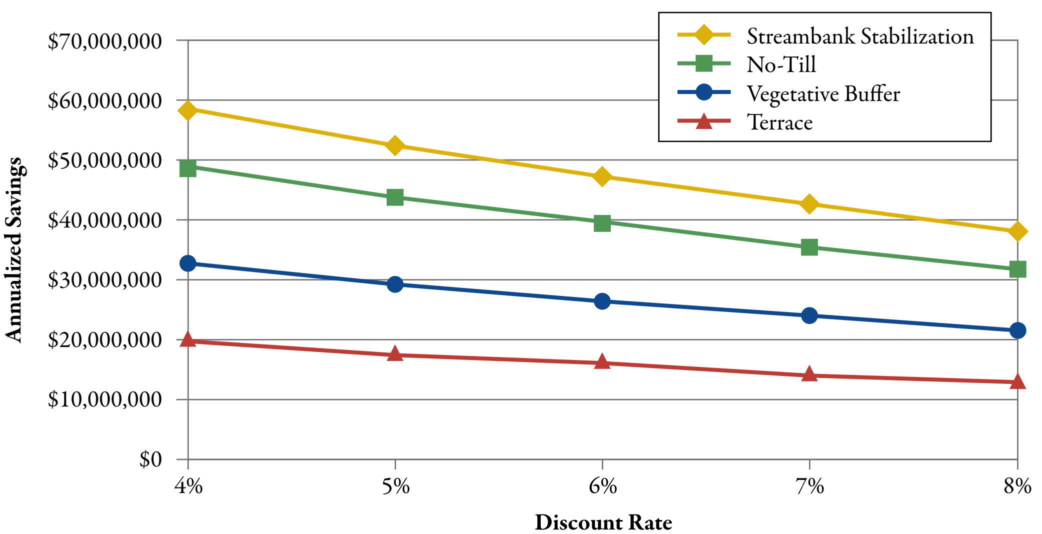


Figure 8. Annualized savings as a function of discount rate
Annualized savings are avoided dredging costs that can be spent on BMPs.

A Potential Process for Evaluating the Best Approach for Sediment Reduction and Reservoir Rehabilitation

Evaluating the best approach for reducing additional sedimentation in watersheds with reservoirs for which dredging is being considered is a large and demanding task that is beyond the scope of this paper. Many issues require discussion and analysis: determining important sediment sources, effectiveness of management practices for these sources, effectiveness of management practices under heavy rainfall and high stream flow events, and location and amount of and appropriate management practices for acres needing treatment. Many questions remain: Is there an acceptable level of sedimentation? What levels of sedimentation are acceptable before dredging is the only option? What is the appropriate combination of management practices and dredging? What are the environmental, flood control, irrigation, water supply, recreation, fish and wildlife, low-flow augmentation, and navigation-flow supplementation costs and benefits of alternative management approaches? What time period should be considered? What is the quality of sediment and is any of it marketable?

We need to know more about management practice costs, which vary by site and with commodity price changes. The following outline provides a general research approach for a more detailed analysis of these issues and questions; it is not inclusive of the entire decision-making process. Figure 9 provides an overview of this approach.

1. Sediment Source Identification
2. Data Collection
   - Watershed characteristics
   - Rates of sedimentation and erosion
   - Extent and types of management practices currently in place
   - Potential management practices
3. Modeling
   - Develop baseline watershed model
   - Evaluate erosion and sedimentation changes under alternative management scenarios
4. Economic Analysis
   - Evaluate effects of erosion and sedimentation on stakeholders
   - Determine costs of alternative practices modeled at field and watershed scale
   - Evaluate sediment reduction cost-effectiveness using spatial targeting approaches
   - Estimate benefits of alternative scenarios for land managers, producers, and watershed users

Available Tools and Tool Development

Livestock and cropland BMPs can benefit society as a whole, but it also is important to consider how these BPMs affect producers and land managers who decide whether to adopt the practices and are responsible for implementing them. To facilitate this analysis, several spreadsheet-based decision-assistance tools are currently under development in the Department of Agricultural Economics at Kansas State University. These tools are designed to analyze BMPs based on economic benefits and costs at the individual field or farm level (societal benefits and costs are not included in the

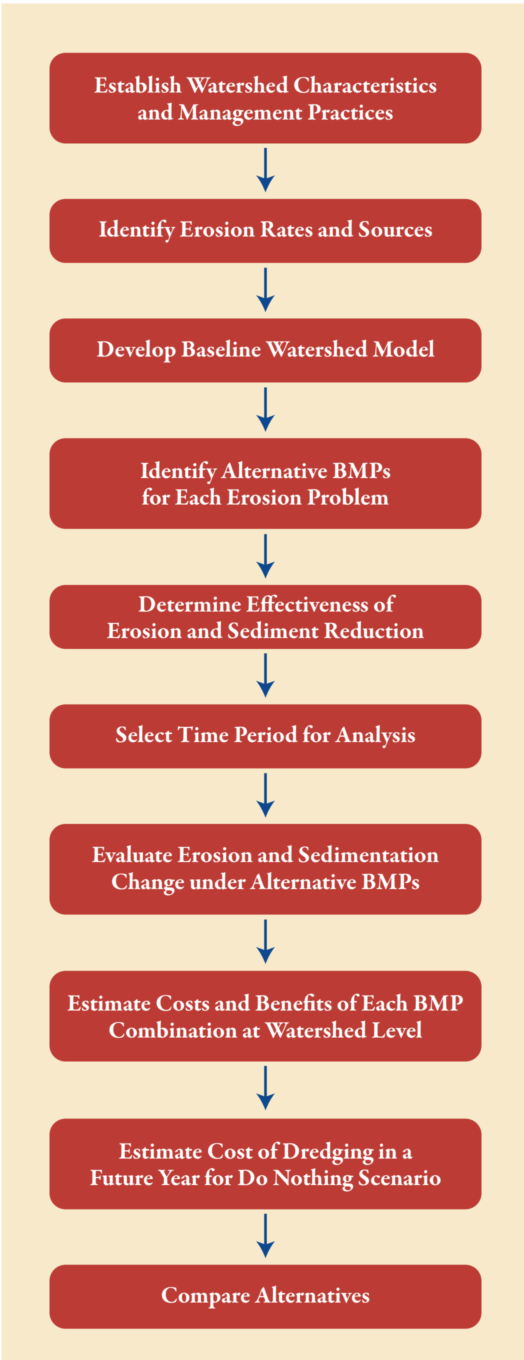


Figure 9. Preferred approach for comparing dredging with sedimentation BMPs

analysis) and will allow producers or land managers to identify sediment- and nutrient-reducing BMPs, determine costs and benefits of BMPs for their operations, and identify available cost-share funding.

The K-State Vegetative Buffer Decision-Making Tool (Smith and Williams, 2007) is designed to answer the following questions: What are the benefits and costs of vegetative buffers, and does it make sense to install a buffer on my operation? This spreadsheet tool provides information, specific to vegetative buffers, about three factors: economic benefits, costs, and available financial programs and incentives. The tool also compares net benefits of buffers with net benefits from cropping. Other decision-making tools currently under consideration will focus on economics of alternative tillage (e.g., reduced tillage or no-till), riparian forest buffers, streambank stabilization projects, and various livestock and rangeland management strategies.

Research Issues and Opportunities

Predicting effects of management practices on erosion and sedimentation requires development of detailed watershed models. These models must include information about sediment source because source location will influence the type of management practices used to reduce the sedimentation rate. Type of management practices selected influences the cost of sediment reduction and overall costs and benefits of sediment reduction versus dredging. Even if watershed models can be designed to determine the technically best sediment reduction management practices, site-specific costs will influence selection

of the most economical management practices. Currently, we can assign only a general cost to each management practice even though different managers can have different costs for implementing the same practice. Two possible approaches for gaining additional information about costs of implementing management practices are Water Quality Trading (WQT) programs and BMP auctions.

Water Quality Trading programs create a market for "water quality credits." Farmers generate income by selling these credits and then are obligated to implement certain BMPs on their farms. A recent report (Breetz et al., 2004) identified more than 70 WQT programs operating in the United States, and additional WQT programs are being adopted rapidly to manage a variety of water quality problems. Most existing WQT programs aim to reduce nutrient concentrations, primarily nitrogen and phosphorous, in streams and lakes. In a typical program, point source polluters (mainly municipal wastewater treatment plants) buy water quality credits from nonpoint source polluters (farmers). Point sources polluters use credits to offset some of their current nutrient discharges to meet regulatory discharge limits. In essence, they purchase nutrient reductions from farmers instead of installing potentially costly treatment technologies.

Economists often favor market approaches to environmental management because these approaches ensure cost-effectiveness. If certain conditions are met, active markets ensure the environmental quality target is met at the lowest possible cost for the watershed as a whole (Atkinson and Tietenberg, 1991). For example, in the active sulfur dioxide market for air emissions, the emissions target was achieved for less than half its originally estimated cost (NCEE, 2001).

Although current programs target nutrient reduction, WQT programs for erosion control can be structured similarly. Eligible BMPs in nutrient trading programs also reduce sedimentation because a large portion of discharged nutrients are dissolved in soil particles. In an erosion control WQT program, each credit represents a specified amount of erosion reduction (e.g., one ton of soil loss). A schedule delineating the number of credits generated by each BMP will need to be developed from watershed modeling simulations, which already are being developed for various watersheds in Kansas (Mankin, 2005). Credits generated by a particular BMP can vary across different subwatersheds depending on soil and topographic features. Landowners across the watershed will be eligible to sell credits, and likely buyers include state agencies, local municipalities, recreation entities, and concerned environmental groups. Buyers will set prices based on financial gain from reduced dredging, improved recreation opportunities, and other benefits. Essentially, sellers will be providing cost estimates for controlling erosion and sediment to various degrees, allowing more accurate identification of erosion and sediment reduction costs.

A BMP auction is another market-based program with potential to accurately identify sediment reduction costs. In a BMP auction, agricultural producers compete by submitting bids to supply the buyer (i.e., project sponsor) with water quality improvements through BMP implementation. Bids are ranked by amount of water quality improvement generated per dollar. Buyers contract first with the producer who can offer water quality improvement at the lowest price. This process is repeated until a predetermined point is reached (e.g., funds are exhausted or bids no longer meet a certain water quality improvement/price ratio target). These auctions allow buyers to identify and purchase the most cost-effective water quality improvements with a specified budget.

A unique characteristic of BMP auctions is that if existing incentives (e.g., cost-share or incentive payments) are insufficient to induce cooperation for high-priority, high-impact improvements, a producer can "reveal" the price required to undertake the desired action. In the marketplace, numerous producers provide such information, and project sponsors can select among competing bids to purchase the most cost-effective bundle of pollution reduction investments. Further, the totality of information provides valuable insight into the incentive levels required to induce producers to adopt various desirable practices.

Research Needs

Additional research should be conducted to determine:

1. Sources of sediment and rates of sedimentation
2. Effectiveness of sediment-reducing BMPs in high- and low-runoff events
3. Costs and returns of alternative BMPs
4. Future dredging costs
5. Environmental and economic effects of alternative watershed protection and reservoir rehabilitation strategies

Appendix A: Selecting a Discount Rate

Selecting an appropriate discount rate reflects the time value of money. A dollar received today is valued more than a guarantee today of a dollar to be received in the future because the future payment implies forgone consumption or investment opportunities today. Many agree that a discount rate should be based on the minimum acceptable rate of return or the opportunity cost of dollars invested in private investments. Raising taxes to pay for public projects removes dollars from private investments that earn a rate of return for investors. Selecting a discount rate can be difficult and controversial because higher discount rates place less emphasis on future benefits and costs. Choice of discount rate is further complicated if both private and public funds are involved in projects because of the nature of who pays and benefits from the projects over time.

Discounting is necessary for economic analysis of projects that have benefits and costs over many years. Discounting benefits, savings, and costs transforms dollar flows occurring in different time periods to a common measure of time value for analysis and comparison, but discount rate affects results. In this study, present values are calculated in 2005 dollars.

One alternative is to calculate a discount rate according to the following formula:

$$i = (r + 1)(1 + f) - 1$$

where: i = nominal discount rate or minimum acceptable rate of return on the investment of dollars, r = real rate of return or discount rate, and f = inflation rate.

A reasonable real rate of return for a risk-free investment is 2.0% to 3.5% (AAEA, 2000). Inflation rate can be measured by the average rate of change in the Personal Consumption Expenditure Index. The long-run average annual rate from 1960 to 2007 was approximately 3.6% (Bureau of Economic Analysis, 2007). Thus, the real rate of return (r) at the midpoint of the suggested range is 2.75%, and an inflation rate (f) of 3.6% gives a resulting discount rate (i) of 6.45%.

The Office of Management and Budget suggests using a 7% rate for benefit-cost analysis of projects (National Center for Environmental Decision-Making Research, 2007), but those who place more emphasis on the future and are in favor of larger government investments in public projects will argue for a lower discount rate. Those who favor less government expenditures and place more emphasis on the present will favor a higher discount rate. The USACE (2002) recently used 6.875% and 3.5% discount rates. The NRCS provides discount rates used for projects since 1957 at: http://www.economics.nrcs.usda.gov/cost/discountrates.html. Since 1990, rates ranged from 4.875% to 8.875%. We used a 7% rate to calculate present values and amortized or annualized present values.

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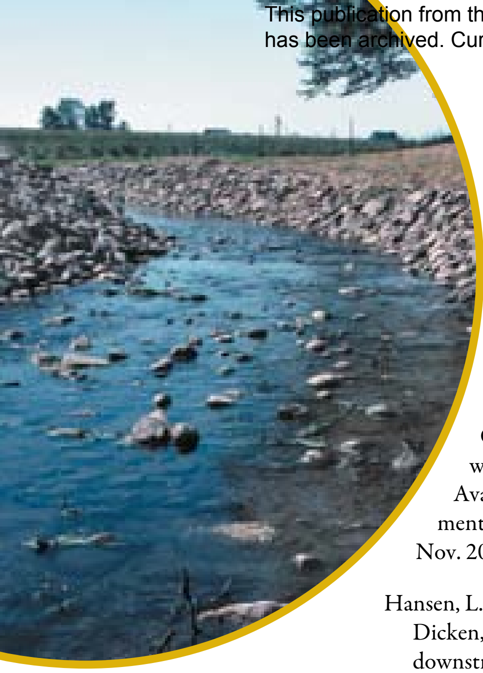

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Reusing Dredged Sediment: Geochemical and Ecological Considerations

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Introduction

Options for rehabilitating federal reservoirs and water-supply lakes filling because of excess sedimentation include controlling pollutant and sediment inputs, decommissioning dams, renovating dams or building them higher, and dredging (Peterson, 1982; Caldwell, 2007). In Kansas, dredging sediments is one of several methods being considered for reservoir restoration. However, many issues need to be addressed prior to pursuing this option: dredging cost, finding sites on which to apply dredged sediments, transporting sediments to those sites, effects on aquatic biota due to sediment resuspension, effects on reservoir water quality due to release of trace elements from sediment during the dredging process, and effects on aquatic and land biota due to chemical changes in land-applied sediment. In this white paper, we focus on the last issue and provide an overview of possible sediment chemistry effects associated with dredging Kansas reservoirs.

Chemistry of reservoir sediment is of concern because of 1) effects on reservoir water quality during the dredging process, particularly if the reservoir is a primary drinking-water source, and 2) effects of remobilized metals, trace elements, or nutrients on aquatic and land biota and water resources. If large quantities of chemical constituents of concern are present, potential chemical effects of land-applied sediment can be important for elements such as lead, zinc, chromium, cadmium, arsenic, or selenium and nutrients such as nitrogen and phosphorus.

Estimates of chemicals deposited in sediments from eight Kansas reservoirs range from approximately 9,720 to 3 million lb/year of phosphorus; 19,000 to 7.6 million lb/year of nitrogen; 96 to 2,700 lb/year of selenium; 620 to 58,000 lb/year of arsenic; 330 to 85,000 lb/year of lead; 1,400 to 366,000 lb/year of zinc; and 340 to 100,000 lb/year of copper (Appendix A-1). These trace elements and nutrients may be mobile or immobile when sediment is moved from reservoirs to land. However, these estimates imply that large quantities of trace elements and nutrients are associated with reservoir sediment and may cause problems if land application is used as a sediment disposal method (see Appendix A-2 for estimates of total sediment and chemical loads in eight reservoirs).

Overview of Dredging and Sediment Disposal Options

Dredging is used to remove sediments from lakes or reservoirs to restore storage area, reduce eutrophication, remove aquatic plants and algae buildup along lake edges, and improve water quality by reducing nutrient sources, especially phosphorus (Ryding, 1982; Darmody et al., 2004; Sigua, 2005). Most literature on dredging deals with harbor or canal remediation in areas of high shipping traffic, such as the Great Lakes or oceanic ports, and many studies involve toxic chemical concentrations higher than those found in Kansas environments. The keys to removing polluted sediments are minimizing disruption of the sediment-water interface, containing polluted, dredged materials until they are remediated, and treating water accompanying the sediment (Peterson, 1982). Prior to dredging, sediment should be characterized for deposited thickness, particle size, bulk density, organic and nutrient content, and potential contaminant concentrations.

Typically, dredged sediment is disposed of by either placement in a confined disposal facility (CDF) or application to agricultural land, grasslands, brownfields, strip mines, highway borders, and other areas, provided that the chemistry of the sediment will not harm the soil or environment (Skogerboe et al., 1987; Darmody et al, 2004; Kelly et al., 2007). CDFs are used for contaminated sediments, those with metal concentrations above environmental toxicity levels recommended by the USEPA (1997), and in areas with river or lake contamination from mining or industrial waste (Darmody et al., 2004).

Confined Disposal Facilities (CDFs)

CDFs are diked enclosures in which sediment or sediment slurries are deposited to permit water mixed with the sediment to leach or evaporate. Disposing dredged sediment in a CDF can be problematic if the facility is too small. Factors such as sediment volume, water content, and underlying soil type need to be considered in the CDF design phase (Myers, 1996). Freshwater sediments have a high water-to-sediment ratio and are slow to settle. Slow settling can cause diked areas to fill faster because of the volume of water accumulating with the sediment; usable retention area decreases until water has either evaporated or drained from the sediment (Peterson, 1982). Settling of fine-grained sediments can seal the diked area, preventing draining and delaying remediation (Dunst, 1987). Lack of storage space can cause flooding overflow from the CDF, and excess nutrients, sediment, and other contaminants could return to the lake if the CDF is located nearby.

Land Application

Issues associated with land application of dredged sediment include compatibility of sediment with the host soil with respect to leaching, stability of the land for use of heavy machinery, and overland flow of sediments and contaminants back to the lake (Cooke et al., 1986). Land application of dredged sediments is frequently used when chemistry of the sediments is not potentially toxic to aquatic or plant life and areas are located away from the dredged lake. Dredged material that has low metal or contaminant toxicities generally is suitable for various land uses including strip mine reclamation, brownfields, construction areas, agriculture, forestry, wetland areas, parks, beaches, and landscaping (Darmody and Marlin, 2002; Machesky et al., 2005).

Work by Kelly et al. (2007) showed that mixing dredged sediment from the Illinois River with wastewater biosolids increased organic and nutrient content of the sediments, had a positive effect on microbial biomass of the soil, and resulted in usable farmland. In another study, using manure as an amendment to dredged sediments helped retain metals in the sediment and enhanced removal of metals by plants (i.e., phytoremediation; Skogerboe et al., 1987).

Work by Sigua et al. (2004a, 2004b) showed that sediments from Lake Panasoffkee, FL, had 82% calcium carbonate content. When combined with agricultural soil, these sediments had the same favorable effects as liming, including enhanced phosphorus and micronutrient availability, nitrification, nitrogen fixation, and soil physical conditions (Nelson, 1981). All trace element contents of reservoir sediments were below USEPA sediment toxicity values (USEPA, 1997); therefore, agricultural or livestock industries could use the sediments to produce forages. In addition, sediment properties improved the level of local soil compaction and structure. One Sigua et al. (2004b) study showed that grass yield from amended fields was greater than that from control sites, and forage from amended fields had increased crude protein content. These studies show that freshwater sediments with low levels of trace elements can be used by agricultural industries with no obvious side effects.

A study by Darmody and Marlin (2002) indicated that dredged, fine-grained lake sediment is suitable for agriculture if allowed to drain sufficiently to support heavy machinery. In that study, dredged sediment was applied to nearby agricultural land. Heavy metal composition of the soil was below USEPA toxicity values (USEPA, 1997), and nutrient values were sufficient to encourage plant growth and survival. Bramley and Rimmer (1988) showed that with proper remediation by drainage, mixing with manure or biosolids, and use of phytoremediation and other methods, contaminated Rhine River sediments were usable for landscaping, agriculture, and other material-fill situations.

Economics of dredged sediment transportation and availability of sufficient land for disposal must be addressed before land application can be used in Kansas. Estimated volumes of sediment and trace elements currently stored in and annually moving into Kansas lakes are large (Appendices A-1, A-2). Chemistry, soil physical properties, and potential contaminant hazards of dredged material must be evaluated prior to land disposal (Cooke et al., 1986; USACE, 1987).

Environmental Effects of Dredging Sediment

Much of the sediment in freshwater lakes is fine grained, generally silt and/or clay. Dredging this type of material, regardless of the depth of sediment removed, results in resuspension of some sediment. Resuspended sediment can interfere with light and food needs of benthic communities and is a major concern associated with reservoir dredging.

Resuspension also can cause water-quality problems, particularly in Kansas where many rivers, streams, and lakes have Total Maximum Daily Loads (TMDLs) of suspended sediments and nutrients above the recommended limits adopted by the Kansas Department of Health and Environment (2008). Chemical effects associated with fine-grained sediment include: 1) adsorption of phosphorus species to fine sediment particles and subsequent transport into lakes, 2) recycling of phosphorus in water with potential increased eutrophication if sediment is disturbed, 3) increased total inorganic nitrogen concentrations in water from conversion of ammonium-nitrogen in the sediment to nitrate-nitrogen in water, and 4) release of adsorbed trace elements because of an environmental change from an anaerobic (low oxygen) to an aerobic (oxygenated) environment (Barnard, 1978; Cooke et al., 1986).

Physical Effects of Reusing Dredged Sediment on Land

Soil physical properties determine how liquids, gases, and heat move through a soil profile, thereby affecting internal drainage and aeration of the soil profile. However, sediment that is dredged from reservoirs is not soil and lacks many of the properties present in natural soils (SSSA, 2001). Vermuelen et al. (2003) estimated that 70% (by volume) of recently dredged sediment is water. To use dredged sediment on land for growing vegetation, several physical properties must be modified. This section provides a brief review of selected physical properties that should be considered prior to land application of dredged, dewatered sediments.

Particle-Size Distribution

Particle-size distribution, also referred to as texture, influences sediment use. For example, sandy sediments can be used in beach construction, whereas clayey material might make a liner material for ponds or lagoons. Sediment with a loamy texture often is the best choice for supporting vegetation, whether in cropland, residential areas, or reclamation of degraded land. Reservoir sediment reflects local geography and soils but generally is composed of finer-grained sizes such as silt and clay (Darmody and Marlin, 2002).

Organic matter is important for supporting vegetation; it provides nutrient storage and cycling (cation exchange capacity), water-holding capacity (important in coarser sediments), and increased aggregation (ability for particles to combine; Tisdall and Oades, 1982). Sediment with minimal organic matter content has reduced usefulness. However, organic matter content can be improved by adding amendments such as compost, waste-treatment biosolids, or manure.

Applied sediment should have a texture similar to that of the original underlying material; a layered system of contrasting textures is undesirable. The different textures affect sediment-soil permeability and movement of water vertically and horizontally in both unsaturated and saturated conditions (Hillel, 1998). Incorporating organic amendments such as compost decreases textural differences and improves overall permeability (Burden and Sims, 1999).

Soil Strength

Soil strength is the measure of the capacity of a soil mass to withstand stresses. Soil strength is most affected by changes in soil-water content and bulk density, although other factors including texture, mineralogy, cementation, cation composition, and organic matter content also affect soil strength (SSSA, 2001). In agricultural settings, increases in strength and bulk density usually result in decreased plant emergence, decreased soil aeration, and increased compaction (Unger and Kaspar, 1994). In Florida pastures, incorporating carbonate-rich, dredged sediment increased overall soil permeability and reduced soil compaction (Sigua et al., 2006). Freshly deposited, dredged sediments usually have low soil strength and need modification by amendments, plants, and/or drainage to facilitate their future use (Darmody and Marlin, 2002). At sites in Illinois, sediment soil strength increased after addition of amendments, use of water-loving plants, and drainage of excess water (Darmody and Marlin, 2002).

Structural Properties

Structure is the arrangement of individual soil particles into aggregates, groups of primary soil particles that cohere more strongly to each other than to other surrounding particles (SSSA, 2001). Soil structure can be difficult to assess and quantify. However, soil structural characteristics are important because they control movement of gases and solutes as well as a variety of other biological, chemical, and physical processes (for specific examples, see Diaz-Zorita et al., 2002, p. 5). Immediately after dewatering, dredged sediment contains no structure or aggregation.

Soil aggregation is a function of organic matter content, clay mineralogy, concentration and ratio of ions, vegetation type and abundance, and soil biology (Bronick and Lal, 2005). Soil aggregates develop as a function of five soil-forming factors: climate, organisms, parent material, relief, and time (Jenny, 1941). Aggregate stability is the ability of an aggregate to retain its shape when wetted. The degree of both structure and aggregate development affect entry and movement of water and air through soil. Plants cause changes in soil structure through penetration of roots, modification of the soil-water regime, enmeshment of soil particles and micro-aggregates by roots, and deposition of carbon below ground; microbes alter soil structure by increasing soil stability (Angers and Caron, 1998). Soil aggregates and soil structure develop with time, vegetative growth, and wetting-drying and freeze-thaw cycles. Presence of water-stable aggregates decreases soil erodibility (Tisdall and Oades, 1982).

Initial development of internal drainage is referred to as conditioning or ripening. Vermuelen et al. (2003) categorized the development of sediments into soil into three processes: physical, biological, and chemical. The physical process of forming structure occurs through desiccation and the resulting formation of cracks. During this process, sediment bulk density decreases and void space increases, thereby increasing internal drainage of the sediments. However, physical formation of structure occurs only in sediments with clay content greater than 8% and/or organic matter content greater than 3% (Vermuelen et al., 2003).

Growing aquatic plants in draining sediment aids development of physical properties by creating root cavities. These cavities allow oxygen to penetrate soil, leading to microbe growth and increased aggregation and permeability (Loser and Zehnsdorf, 2002). Terrestrial plants are introduced naturally from seeds transported by wind and birds (Vermuelen et al., 2003). Soil fauna such as bacteria, fungi, and earthworms decompose fresh organic matter and produce humus, a more stable form of organic matter that increases binding of soil particles into aggregates.

Surface soils containing stable aggregates resist formation of a surface soil crust and allow water to enter (infiltration) and move through (permeability) the soil profile. Darmody and Marlin (2002) showed that the rate of aggregate formation in dredged sediments used for agriculture became similar to that of native soils over a 10-year period.

Exposure of dredged sediments to air is termed chemical ripening. Exposure to oxygen results in oxidation of metals occurring as reduced species such as iron or selenium and either decreased or increased mobility of these elements. Also, mineral weathering of primary soil minerals can increase secondary (clay) minerals and change the cation exchange capacity, thereby affecting soil solution concentrations (Vermuelen et al., 2003). Chemical changes affecting selected trace elements and nutrients are described in more detail in the remainder of this paper.

Sediment Chemistry Studies in Kansas Reservoirs

Studies of sediment chemistry in Kansas reservoirs are mainly limited to studies performed by the USGS (2008a). These studies assessed a variety of nutrients and trace elements to determine which reservoirs have potential contamination problems. Information obtained from these studies provides a background database that can be used for future comparison of trace elements if reservoirs are dredged and sediment is disposed of by land application.

The USGS sediment data presented in this review are from eight of 24 federal reservoirs and 14 of the many freshwater lakes in Kansas. The studies provide information on measured concentrations, potential nutrient sources, trace elements, and pesticides and the volume of sediment, trace elements, and nutrients deposited at selected lakes (USGS, 2008a). Results are summarized in Appendices A-1, A-2, A-3, B-1, and B-2 and cited throughout this paper.

Sediment quality guidelines adopted by the USEPA allow assessment of reservoir sediment with respect to level-of-concern concentrations of various trace elements and organochlorine compounds, including polychlorinated biphenyls and several pesticides (Smith et al., 1996; USEPA, 1997; USEPA, 2004). Two such level-of-concern concentrations are the threshold-effects level (TEL) and the probable-effects level (PEL). The TEL represents the concentration below which toxic biological effects rarely occur. In the range between the TEL and PEL, toxic effects occasionally occur. The PEL represents the concentration above which toxic effects usually or frequently occur. These guidelines are used by the USEPA as screening tools and are not enforceable (Sigua et al., 2004a; USEPA, 2004).

As of 2006, the USGS used a combination of the USEPA level-of-concern concentrations and the consensus-based sediment quality guidelines developed by MacDonald et al. (2000), which consist of a threshold-effect concentration and a probable-effect concentration. Much of the USGS sediment work prior to 2006 reported level-of-concern concentrations using TELs and PELs; thus, those levels are reported in this paper to provide a level of comparison.

Of the trace elements and pesticides evaluated in Kansas lakes, six contaminants typically exceeded TELs: arsenic, chromium, copper, lead, nickel, and zinc (Table 1). No TEL is established for selenium. In addition, DDE, a daughter product of the pesticide DDT, was measured in a number of the tested reservoirs and lakes. Typically, pesticides concentrations are less than the TELs (USGS, 2008a). Most national studies had similar results, with variation occurring because of different metal sources and varying depths of collected samples.

Christensen and Juracek (2001) observed an increase in arsenic, selenium, and stron-

Table 1. Kansas reservoirs with trace elements above USEPA TELs[a]

Reservoir	Trace Elements and TELs					
	Arsenic (7.24 mg/kg)	Chromium (52.3 mg/kg)	Copper (18.7 mg/kg)	Lead (30.2 mg/kg)	Nickel (15.9 mg/kg)	Zinc (124 mg/kg)
Swanson	x		x		x	
Harlan County	x		x		x	
Milford	x		x		x	
Kirwin	x		x			
Webster	x		x			
Waconda	x		x			
Tuttle Creek	x	x	x		x	x
Perry	x	x	x		o	
Centralia	x	x	x		x	
Mission	x	x	x		x	x
Pony Creek	x	x	x		x	
Cheney	x	x	x		x	
Lake Afton	x	x	x	x	x	x
Hillsdale	x	x	x	x	x	x
Cedar Lake	x	x	x	x	x	x
Lake Olathe	x	x	x	x	x	x
Gardner	x	x	o	x	o	x
Bronson	x	x	o	x	x	x
Crystal	x	x	o	x	x	x
Otis Creek	x	x	x		x	

[a] Data source: USGS (2008a)
o = Value above USEPA PEL
TEL = threshold-effects level
PEL = probable-effects level

tium in several reservoirs in the Republican and Solomon River basins. The increase might be related to increased irrigation throughout the two basins. Arsenic and copper values often exceeded TELs, but overall, other trace elements (i.e., cadmium, nickel, lead, zinc, and chromium) tested in the basins did not.

Empire Lake in Cherokee County in southeastern Kansas is the most contaminated lake examined by the USGS in Kansas (Juracek, 2006). This lake is affected by lead and zinc mining that occurred in the tri-state area of Missouri and Kansas beginning in 1870 (Brosius and Sawin, 2001). Concentrations of lead, zinc, and cadmium, an element that occurs with lead and zinc, are above PELs (Appendix B-1). Concentrations decreased over time, but present surface sediment concentrations are still above PELs of concern for aquatic life.

Other Kansas reservoirs have various metals above TELs but not PELs. In most Kansas reservoirs and lakes studied by the

USGS, arsenic, copper, and nickel were measured above TELs; at a few of the lakes and reservoirs, chromium, lead, and zinc were measured above TELs; and at all the reservoirs and lakes, cadmium and mercury were below TELs (USGS, 2008a). Use of total concentrations implies the amount of constituent measured in the sediment. It does not imply that this total quantity is available for mobility or use by plants if sediment is dredged. However, total quantity does report the amount of the constituent that is stored and could be mobilized under certain chemical conditions or change when sediment is dredged and removed from the lake environment.

Chemical Changes in Dredged Sediment

Much literature on dredged-sediment disposal on upland areas describes chemical changes that occur when sediment from the bottom of a lake is brought into an oxidizing situation. Many studies focused on one or more trace elements, nutrients, and organic matter. This section of the paper focuses on trace elements typically found in Kansas reservoirs above TELs: arsenic, chromium, copper, lead, nickel, and zinc (Table 1). Potential sources and effects of mercury, methylmercury, and selenium are also included because of possible biogeochemical effects of these compounds on fish and other aquatic life.

Work by Delfino et al. (1969) and Nrigau (1968) showed a strong relationship between water depth and increased concentrations of nitrogen, phosphorus, iron, and total- and sulfide-sulfur in Lake Mendota, WI. This trend was mirrored in observations made by Iskandar and Keeney (1974), who also found that post-cultural sediment (from 1818-1970) showed increased levels of chromium, copper (related to the use of copper sulfate for algal control in the lakes), lead, and cadmium in the more recent sediments (1970s) of five hard-water and five soft-water Wisconsin lakes. Sources for these metals were sewage effluent, chemical treatment for algae, and vehicular traffic; trace-element concentrations increased overall because of human activities.

Chemical results from selected USGS studies conducted in Kansas (USGS, 2008a) are summarized in Appendices A-1 and A-2. Appendix A-3 shows the mercury concentration in the few lakes where it was detected, and ranges of concentrations observed in cores from selected lakes are presented in Appendices B-1 and B-2.

Reduction-Oxidation Chemistry

Reduction-oxidation (redox) potential describes the chemical reactions in sedimentary environments that occur with changes in dissolved oxygen levels. When dissolved oxygen in reservoir sediments decreases to a very small amount, redox potential decreases and the system is described as anoxic or anaerobic. Some elements such as arsenic, iron, manganese, and phosphorus are more mobile in an anaerobic environment and can move with pore water. If sediments become exposed to oxygen, these elements can become oxidized and coprecipitate with other elements, forming compounds such as iron- or manganese-hydroxides or oxides (Forstner, 1977). Coprecipitated oxides and hydroxides also can serve as adsorptive surfaces, thereby increasing adsorption of other potential contaminants (e.g., phosphorus).

Increased redox potential (i.e., more oxygen is available) can result in decomposition of organic material and transformation of redox-sensitive elements, such as copper, zinc, cadmium, and nickel, to oxidized states. These dissolved metals can coprecipitate as oxides or oxyhydroxides (DeLaune and Smith, 1985; Brannon et al., 1994; Rognerud and Fjeld, 2001; Davidson et al., 2005). Increased aeration of silt/clay sediment in a Mississippi reservoir resulted in decreased concentrations of copper, zinc, cadmium, and nickel in water and increased coprecipitation of these elements with iron, forming solid amorphous oxides (Davidson et al., 2005). Oxidizing conditions generally favor metal insolubility, and reducing conditions favor metal solubility or mobility (Miao et al., 2006).

Organic Carbon Cycling

Rate and extent of organic matter cycling in a lake help determine oxygen levels and redox-potential levels in the water column and sediments (Avnimelech et al., 1984). Lake sediments generally contain greater concentrations of organic matter and nutrients than the overlying water. Organic matter decomposition also contributes to nutrient recycling in sediment and water. During dredging, microbial transformations of nutrients to more mobile forms and trace elements to less mobile forms occurs if sufficient organic carbon is present and the sediment environment changes from anaerobic to aerobic.

In cores, organic carbon often decreases with increasing depth, indicating organic matter degradation and changes in sediment chemistry with depth. However, the organic-carbon concentration in several Kansas lakes is uniform throughout the profile, suggesting rapid sedimentation with few chemical or biological changes with depth (Callender, 2000; Mahler et al., 2006; Juracek, personal communication, 2007).

Estimated total organic carbon loads from the Kansas lake studies included in this literature survey range from 19,300 to 928,000 tons (Appendix A-2). Total organic carbon measured from specific cores ranged from 0.7 to 3.9 mg/kg in many of the lakes; Webster, Kirwin, and Waconda had unusually high values ranging from 3,440 to 16,200 mg/kg (Appendix B-1). This large volume of organic carbon suggests that microbial transformations of trace elements are likely if dredging is used as a remediation method in Kansas. Because availability of organic carbon and oxygen affects mobility of trace elements and nutrients, potential changes that could occur in Kansas require further study.

Contaminants of Interest

Final use of land where dredged sediment is applied depends on the amount of contamination in the sediment. When contaminants are present at high levels, vegetative growth on the deposited sediment can be harmed or completely restricted. Sediment in a number of Kansas reservoirs is contaminated with trace elements and nutrients, but contaminant levels are relatively low compared with other parts of the country.

Lead, zinc, copper, cadmium, arsenic, selenium, nickel, dissolved salts, DDE, and nutrients are the contaminants of most interest in Kansas lake sediments. Lead, zinc, copper, cadmium, arsenic, and nickel are found above TELs for sediment quality in Kansas (USGS, 2008a). These trace elements, DDE, and the nutrient content of sediment are of concern because of resuspension of sediment (Peterson, 1982), effects on drinking water quality, and ecological effects if sediment is applied on agricultural land (Skogerboe et al., 1987; Darmody et al., 2004; Kelly et al., 2007). Mercury is also of concern because it has potentially deleterious biological effects if resuspended with sediment or dissolved in lake water during dredging.

Mercury. Atmospheric deposition, agricultural chemicals, power-plant and waste-incineration emissions, and decomposition of terrestrial litter are potential sources of mercury in Kansas. Forest fires as well as industrial sources such as mining or coal-fired power plants can also add mercury to the environment (Wiedinmyer and Friedli, 2007).

Total mercury was measured in only a few of the Kansas lakes and reservoirs evaluated by the USGS (Appendix A-3). All mean and median values were below the USEPA TEL of 0.13 mg/kg, except Empire Lake in southeast Kansas, which had one core with values above the TEL (Juracek, 2003, 2004). Several lakes in northeast Kansas had mean annual net loads for mercury ranging from 0.39 to 317 lb/year (Juracek, 2003, 2004; Juracek and Mau, 2002). Although the majority of lakes studied had mercury values below USEPA TELs, because of mercury's potential ecological effects and the presence of measurable mercury in some lakes, sediment should be tested for mercury prior to dredging.

Methylmercury. Methylmercury is a neurotoxin that is harmful to both aquatic and terrestrial biota. This compound is toxic to fish; it also bioaccumulates, which can affect human health.

Methylmercury is formed by sulfate-reducing bacteria in anaerobic environments, particularly lake sediments and wetlands. Bacteria metabolize mercury into methylmercury. Sources of methylmercury include mercury sources mentioned previously as well as terrestrial runoff and direct atmospheric deposition onto a lake surface (Rudd, 1995).

Organic carbon has a strong connection to presence of mercury or methylmercury in the environment. Several studies show that water movement through wetlands and peat bogs, which have relatively high dissolved organic carbon concentrations, increases methylmercury formation and transport (Jackson, 1989; Kelly et al., 1995; Krabbenhoft et al, 1995; Rudd, 1995). Methylation increases when sulfate and salinity levels are low and concentrations of organic fermentation products are high (Kongchum et al., 2006). Jackson (1989) demonstrated that quantity and type of clay minerals, oxides, and humic matter also affects methylmercury production in sediments.

Land uses such as agriculture, forestry, or mining also affect occurrence of methylmercury in surface water, sediments, and fish (Brumbaugh et al., 2001). The increased quantities of plant matter, organic carbon, and sediment transported to rivers or lakes during storms enhance the potential formation of methylmercury within a lake or river system (Rudd, 1995). Most sampled lakes in Kansas had mercury values below the TEL, but methylmercury sediment-core pore waters were not evaluated. Because a large volume of organic

carbon and sediment enters Kansas lakes and reservoirs, it is likely that methylmercury could form in the sediment or mercury could dissolve in lake water if sediment is resuspended during dredging. This issue warrants further research.

Lead. Lead concentrations in lake and river sediment cores are directly related to exhaust from vehicles that use leaded gasoline (Callender and Van Metre, 1997; Machesky et al., 2005). A study of 10 small lakes in Kansas (Juracek and Ziegler, 2006) showed strong relationships between observed lead concentration and traffic volume, reservoir size, and basin size. Lead profiles showed an increasing concentration trend related to leaded gasoline use from 1940 to 1970 and a decreasing trend after lead was removed from gasoline in 1972. Over time, lead concentrations in sediment might return to baseline conditions. However, the buried, high lead concentrations (often above both TELs and PELs) could cause future concerns if reservoirs are dredged, dams are removed, or dams fail.

The estimated mean annual net lead load of lead for Empire Lake is 6,500 lb/year, and approximately 650,000 lb of lead have been deposited in the lake since the dam was closed (Appendices A-1, A-2). Lead concentrations in younger sediments have decreased over time, but the present surface sediment concentrations are still above PELs of concern for aquatic and plant life (Appendix B-2). Lead concentrations in lake sediments can remobilize if pH and oxygen levels change (Telmer et al., 2006). If Empire Lake is dredged, lead in sediments could affect the environment because of chemical changes in the sediment caused by exposure to oxygen and drainage. Lead needs to be evaluated prior to dredging because it has potential environmental effects and occurs, sometimes at high levels, in most lakes studied in Kansas.

Zinc. Zinc is present at levels between TELs and PELs in Kansas lakes (Table 1, Appendices A-1, A-2, B-2). At high concentrations, zinc causes a range of biological and toxic responses in a variety of aquatic organisms (Mullis et al., 1996; Lefcort et al., 1998). Atmospheric deposition of zinc occurs from metal production, waste-incineration and fossil fuel emissions, phosphate fertilizer use, and cement production. Water-contamination sources include deicing salts; automotive exhaust; and wear and tear of rubber tires, brake linings, and galvanized metal parts (Councell et al., 2004). There is a strong relationship between traffic density and zinc concentrations in sediment cores in Georgia and Florida lakes (Callender and Rice, 2000).

Estimated annual zinc loads for the Kansas lakes studied range from 1,363 to 366,000 lb/year, and estimated total chemical loads of zinc range from 84,506 to 11.7 million lb (Appendices A-1, A-2). Ecological effects of and phytoremediation possibilities for zinc should be evaluated prior to dredging.

Arsenic. Arsenic is of environmental concern in freshwater lakes, surface water, and groundwater (Huang et al., 1982). Potential sources of arsenic include past use of arsenical pesticides (banned in the 1970s), smelters, coal-fired power plants, erosion caused by intensive land use, leaching from lumber pressure treated with chromated copper arsenate, mineral weathering, high evaporation rates in arid environments, and irrigation-return flows (Huang et al., 1982; Welch et al., 2000; Rice et al., 2002; Smedley and Kinniburgh, 2002). Arsenic is strongly sorbed to surfaces of aluminum and iron oxides and edges of clay minerals under oxidizing conditions (Kneebone and Hering, 2000).

Arsenic in solution exists as either arsenite (As^{+3}, toxic form) or arsenate (As^{+5}, non-toxic form). In a laboratory experiment conducted by Oscarson et al. (1980), manganese and iron compounds of clay particle size (0.002 mm) in sediment affected oxidation of arsenite (As^{+3}) to arsenate (As^{+5}). This suggests that presence of manganese and iron compounds in freshwater sediments could help detoxify arsenic concentrations.

Arsenic in lake-sediment pore water is generally present as arsenite, which is toxic in anaerobic conditions. Dredging permits mixing of arsenite in the water column until oxidation or sorption remove it or render it nontoxic (Brannon and Patrick, 1987). Disposal of arsenic-rich sediment in a CDF could result in pulses of toxic (arsenite) and nontoxic (arsenate) forms of arsenic in leachate depending on whether oxidizing or anaerobic conditions are present. Precipitation reactions occurring at the site could result in pH changes at both the surface and at depth, which could affect the form of arsenic present in sediment (Brannon and Patrick, 1987; Kneebone and Hering, 2000).

Arsenic is an issue in Kansas lakes where it is present above TELs (Table 1, Appendix B-2). In lakes where arsenic was measured, annual chemical loads ranged from 619 to 57,800 lb/year with total chemical loads ranging from 24,760 to 1.8 million lb (Appendices A-1, A-2). Arsenic should be evaluated prior to using land application of dredged material for remediation.

Selenium. Selenium is derived primarily from weathering of rocks. In the northern Great Plains, Cretaceous-aged Pierre Shale is a primary source of seleniferous soils. Other sources include volcanic activity and fossil fuel combustion. Selenium often occurs in colloid- and sulfide-rich lake and river sediments and in organic- and iron-rich soil layers. Selenium is more likely to leach in sediments with low organic matter and clay content and an alkaline pH (above 7) and in calcareous soils (Sarma and Singh, 1983). Processes that affect selenium transport include soil leaching, groundwater transport, metabolic uptake and release by plants and animals, sorption and desorption, chemical or bacterial reduction and oxidation, and mineral formation (Juracek and Ziegler, 1998).

Selenium is generally inert under reducing conditions. In an oxidized environment, several forms of selenium exist: selenate (Se^{+6}), selenite (Se^{+4}), and an organic form of selenium (Se^{-2}) (USEPA, 1996). Because selenium can coexist in several forms, the USEPA set the toxicity limit for total selenium and not separate forms of the element. Selenium concentrations equal to or greater than 4.0 mg/kg in sediment are of concern for fish and wildlife because of food-chain bioaccumulation (Lemly and Smith, 1987).

Selenium is of particular concern in rivers and lakes in western Kansas because of geologic sources from the Pierre Shale and other Upper Cretaceous strata and from evapoconcentration of irrigation water. Estimated mean annual net chemical loads deposited in the few Kansas lakes where selenium was measured range from 96 to 2,730 lb/year with estimated total loads in bottom sediments ranging from 3,856 to 87,360 lb (Appendices A-1, A-2, B-2).

If selenium-containing dredged soils are land applied, selenium likely will be immobilized in soils amended with manure or compost because it binds well with organic matter and clays (Geering et al., 1968). The potential for selenium to become remobilized in lake water during the dredging process is of more concern. If dredging occurs in lakes used for public drinking-water supply, the USEPA drinking-water limit of 0.05 mg/L selenium will need to be carefully monitored.

Copper. Copper is a micronutrient and toxin. It strongly adsorbs to organic matter, carbonates, and clay, which reduces its bioavailability. Copper is highly toxic in aquatic environments and affects fish, invertebrates, and amphibians; all three groups are equally sensitive to chronic toxicity (USEPA, 2007a). Cu^{+2} is the oxidation state of copper generally encountered in water. When Cu^{+2} occurs in the environment, the ion typically binds to inorganic and organic materials contained in water, soil, and sediments (ATSDR, 2004).

The most obvious copper source, particularly in Kansas lakes, is use of copper sulfate to control algal blooms. The compound is used in smaller ponds and lakes, where algae problems are most severe (Peterson and Lee, 2005; D.E. Peterson, KSU Agronomy Dept., personal communication, 2007). Copper in lake sediments also comes from leaching from animal waste and pressure-treated lumber, atmospheric deposition by precipitation, and road wear of brake linings (Rice et al., 2002).

In a number of Kansas lakes, copper values exceeded the TEL; in a few lakes, cores showed copper values above the PEL (Table 1, Appendix B-2). Estimated mean annual net loads of copper in Kansas lakes range from 336 to 100,000 lb/year; estimated total loads range from 19,845 to 3.2 million lb (Appendices A-1, A-2, B). Copper can pose disposal problems because changes from an anaerobic to aerobic environment during dredging or when sediment is deposited on land can permit remobilization (Baccini and Joller, 1981).

Cadmium. Sources of cadmium are generally associated with industrial processes, wastewater, or emissions. Atmospheric sources include waste incineration, fossil fuel combustion, mining, and smelting of zinc ore for galvanized roofing (Mahler et al., 2006). Additional sources include application of phosphate fertilizers, biosolids, or manure on fields; weathering of rocks and soils; and leaching from landfills. Major factors governing cadmium speciation, adsorption, and distribution in soils are pH, soluble organic matter content, hydrous metal oxide content, clay content and type, presence of organic and inorganic ligands, and competition from other metal ions (UN, 2006).

Relative to more industrialized areas of the country, Kansas lakes have minimal cadmium concentrations that generally are below the TEL (Appendix B-2). Estimated mean annual net loads of cadmium in selected Kansas reservoirs range from 3.2 to 1,520 lb/year, and stored quantity estimates range from 171 to 48,640 lb (Appendices A-1, A-2). However, remobilization of cadmium from sediments in lakes with higher concentrations, such as Empire Lake in southeastern Kansas, can affect

aquatic life and plants (USEPA, 2001).

Chromium. Chromium is found in rocks, animals, plants, soil, and volcanic dust and gases. The concentration of naturally occurring chromium in U.S. soils ranges from 1 ppm to 2,000 ppm (USEPA, 2007b). Chromium is present in the environment in several different forms. Its valence states range from $^{+}2$ to $^{+}6$, but in natural environments, it is generally found as trivalent chromium (Cr^{+3}) or hexavalent chromium (Cr^{+6}). Trivalent chromium occurs naturally in many fresh vegetables, fruits, meat, grains, and yeast and is added to vitamins as a dietary supplement. However, release of Cr^{+3} to the environment can be toxic because of conversion to $Cr^{+6.}$ Hexavalent chromium is most often produced from industrial sources such as coal-fired power plants, steel making, leather tanning, chrome plating, dyes and pigments, and wood preservation and can indicate environmental contamination. This form of chromium exists in oxidizing conditions and can move through soil to underlying groundwater. Hexavalent chromium is the more toxic form and is a threat to aquatic life and human health if ingested (ATSDR, 2001).

Chromium enters air, water, and soil mostly in the trivalent (Cr^{+3}) and hexavalent (Cr^{+6}) forms. In air, chromium compounds are present mostly as fine dust particles that eventually settle over land and water. Chromium concentrations are generally low, but the metal often concentrates in hydrous manganese and iron oxides or adsorbs to clay-size particles in sediment. The clay-size fraction is the particle size most likely to be transported to lakes, particularly during floods or in areas where streambank erosion is high (Whittemore and Switek, 1977; ATSDR, 2001).

Chromium was detected at concentrations exceeding the TEL at some Kansas reservoirs (Table 1; Appendix B-2). Estimated mean annual net loads of chromium for selected Kansas lakes range from 864 to 302,000 lb/year; estimated total chemical loads range from 52,740 to 9.6 million lb (Appendices A-1, A-2). Because of the potential for environmental and human harm, chromium in lake sediment needs to be evaluated prior to dredging and land application.

Nickel. Some anthropogenic sources of nickel include oil combustion, oil-burning and coal-fired power plants, trash incinerators, treated wastewater, car exhaust, abrasion of nickel-containing automobile parts, and animal waste (Lagerwerff and Specht, 1970; ATSDR, 2005). Because of the density of agricultural land use near lakes and reservoirs, the most likely anthropogenic sources of nickel for Kansas lakes are wastewater and animal waste.

Nickel strongly adsorbs to soil or sediment containing iron or manganese. However, nickel becomes more mobile if water or sediment pH becomes more acidic, such as in an anaerobic environment. Nickel does not appear to accumulate in fish or other animals used as food but is a known carcinogen and toxic to humans and animals when maximum tolerable amounts are exceeded (ATSDR, 2005; LENNTECH, 2008).

Nickel occurs at levels between TELs and PELs in Kansas lakes (Table 1; Appendices A-1, A-2, B-2). Estimated mean annual net loads range from 355 to 152,000 lb/year;

estimated total net loads range from 22,000 to 4.8 million lb. Phytoremediation possibilities for nickel should be evaluated prior to dredging.

DDE and DDD. DDE and DDD are degradation products from the pesticide DDT, which was used extensively in agriculture during the 1950s and 1960s until it was banned in 1972 (USGS, 2008a). Sources of DDT, DDE, and DDD include runoff from agricultural fields and deposition on lakes, streams, and land of soil particles carried by wind (Rapaport et al., 1985).

Detection of DDE and DDD in recently deposited sediments of eight Kansas reservoirs (Appendix B-1) indicates that DDT use was widespread in eastern Kansas (Juracek, 2004). DDT, with a half-life of 2 to 15 years, lasts for years in soil (ATSDR, 2002). Detection of the daughter products DDD and DDE in upper parts of cores indicates that DDT breakdown products are continuing to enter Kansas lakes, probably from eroding soils in upstream watersheds.

Salinity. Salinity affects both mobilization and biological availability of metal contaminants. Lakes in semi-arid or arid environments can have increased salinity in both water and sediments because of evaporation or inflow of soils affected by evapotranspiration upgradient. Disposing of saline sediments in upland, unconfined areas could result in increased remobilization of many metals; subsequently, these metals could be taken up by plants, leach into groundwater, or run off to surface water (Francingues et al., 1985). Additional information on salinity is presented in the next section.

Nutrient Transformations and Salinity Effects

Land application of dredged sediments has potential benefits. Even if minimally contaminated with trace elements or toxic organic compounds, lake sediments can support crop growth and even improve agricultural soils. Woodard (1999) found that amending soil with dredged sediment had minimal effects on growth of soybean, corn, and sunflower but increased growth and nutrient concentrations in big bluestem. Sigua et al. (2004b) reported increased growth and nutrient uptake of bahiagrass forage when grown on soils amended with dredged lake sediment.

Dredged lake sediments can have higher nutrient concentrations than the originating soil. This phenomenon, known as sediment enrichment, is caused by selective transport and deposition of fine particles (silt and clay) to which nutrients are attached (Sigua, 2005). Studies by Mau (2001), Pope et al. (2002), and Juracek and Ziegler (2007) showed that sediments from several Kansas lakes have higher total phosphorus concentrations than soils in contributing watersheds (Table 2). Furthermore, sediment transport from an anaerobic lake bottom to aerobic surface soils can result in nutrient transformations associated with changes in redox potential. Therefore, sediment properties and nutrient transformations should be considered when evaluating agronomic and environmental sustainability of land application of dredged sediments.

Nitrogen

Total nitrogen concentrations of lake sediments are generally similar to those of contributing watershed soils (Juracek and Ziegler, 2007). Although the majority of nitrogen in lake sediments likely is organic-nitrogen (non-plant available), various forms of nitrogen in sediment samples are rarely determined. Nitrogen transformations in soils and sediments are dynamic and highly influenced by oxygen supply (Figure 1). These transformations influence plant availability, transport, and potential environmental effects of nitrogen.

Organic-nitrogen forms must be mineralized before they are available to plants. Nitrogen mineralization occurs in anaerobic sediments, but the rate nearly doubles when sediments are subjected to an aerobic environment (Moore et al., 1992). Nitrification, conversion of nitrogen from ammonium (NH_4^+) to nitrate (NO_3^-), is strictly an aerobic process (Figure 1). Nitrification is particularly important from an environmental standpoint because nitrate is subject to leaching loss. Moore et al. (1992) found that pore-water nitrate-nitrogen concentrations increased to greater than 30 mg/L of nitrogen during the first 15 days in an aerobic environment. This indicates that there can be a sudden flush of nitrate following land application of lake sediments; however, nitrate leaching from dredged sediments has not been studied. Aerobic conditions followed by anaerobic conditions could convert much of the nitrate to nitrogen gas through denitrification (Figure 1). Soil-water content, water flux, and plant uptake influence nitrate losses. Proper management could minimize nitrate losses, but additional research is needed to confirm this supposition.

Measured nitrogen in sediment cores ranges from 30 to 5,200 mg/kg depending on the lake (Appendix B-1). This variation in nitrogen availability reflects different source areas and land uses near the sampled lakes. Estimated mean annual net loads

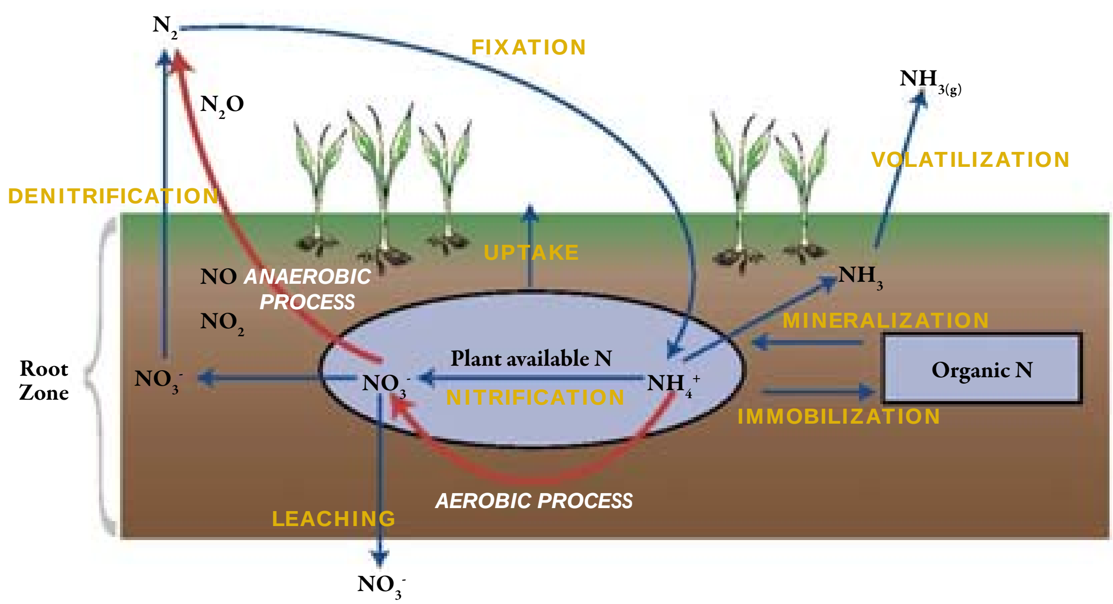


Figure 1. General nitrogen cycle in soils
Red lines emphasize aerobic or anaerobic processes

of nitrogen available in sediments from selected Kansas lakes range from 19,200 to 7.6 million lb/year; estimated stored chemical loads range from 1.2 million to 243 million lb (Appendices A-1, A-2). In Kansas lakes, the volume of nitrogen available for transformation when sediment redox conditions change is large and needs to be considered when selecting a disposal area and plants for use in phytoremediation.

Phosphorus

Total phosphorus concentrations in lake sediment generally are higher than average phosphorus concentrations in soils in contributing watersheds (Table 2). Although elevated phosphorus concentrations in sediment are not a concern for crop production, high phosphorus concentrations are an environmental concern. Increases in lake sediment phosphorus content are correlated to increased phosphorus loss through erosion and runoff (Sharpley, 1995). Implementing best management practices to control erosion and capture eroded sediment before it reaches surface water bodies is an important component of plans for land application of dredged sediments.

Loss of dissolved phosphorus can have environmental effects even when erosion is controlled. Phosphorus cycling in soils is generally governed by inorganic phosphorus reactions, such as adsorption and precipitation. Phosphorus desorption and dissolution release sediment-bound phosphorus into soil solution or runoff water (Figure 2). Changes in redox status of the soil, which occur during dredging and land application of sediment, affect adsorption/desorption and precipitation/dissolution reactions (Sharpley, 1995; Miao et al., 2006).

The majority of research on redox effects on phosphorus sorption reactions has been conducted on either agricultural soils that are flooded or in situ lake-bottom sediments. Under reduced conditions,

Table 2. Total phosphorus concentrations in lake sediments and corresponding watershed soils for several Kansas lakes

Lake/Watershed	Upstream sediments	Downstream sediments	Watershed soils
	Mean Total Phosphorus Concentration (mg/kg)		
Atchison County Lake[a]	800	1000	520
Banner Creek Reservoir[a]	770	860	740
Mission Lake[a]	670	1100	620
Perry Lake[a]	810	930	610
Lake Wabaunsee[a]	620	870	550
	Out-of-channel	**In-channel**	**Nonagricultural (cemetery)**
Cheney Lake[bc]	430	630	245

[a] Juracek and Ziegler (2007)
[b] Pope et al. (2002)
[c] Mau (2001)

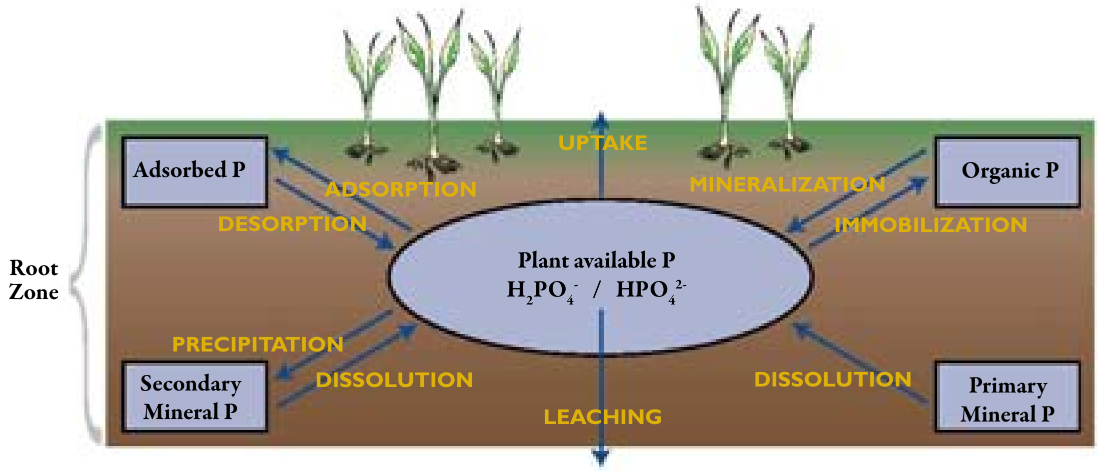


Figure 2. Phosphorus transformations and reactions in soil environments

phosphorus sorption is greater at high phosphorus concentrations and lower at low phosphorus concentrations compared with oxidized soils (Khalid et al., 1977; Vadas and Sims, 1998). Therefore, predicted phosphorus release patterns resulting from oxidizing reduced sediments depend on the phosphorus status of soils.

Oxidizing sediments with low phosphorus concentrations would decrease soluble phosphorus, but oxidizing sediments with high phosphorus concentrations could increase soluble phosphorus release. Reactions can be further complicated if soils undergo repeated cycles of oxidation and reduction. A general increase in phosphorus release has been observed in soils that are reduced, oxidized, and reduced again (Young and Ross, 2001; Shenker et al., 2005). Chen et al. (2003) found that pore-water phosphate concentrations did not increase following initial application of dredged sediment to soil surface but increased 10-fold over the control following the first drying and wetting cycle.

Measured phosphorus in sediment cores ranges from 422 to 1,300 mg/kg depending on the lake (Appendix B-1). This variation in phosphorus availability reflects different source areas and land uses near the sampled lakes. Estimated mean annual net loads of phosphorus available in sediments from selected Kansas lakes range from 9,720 to 3.4 million lb/year; estimated stored chemical loads range from 437,400 to 109 million lb (Appendices A-1, A-2). In Kansas lakes, the volume of phosphorus available for transformation when sediment redox conditions change is large (Appendix B-1) and needs to be considered when selecting a disposal area. Because of the complex redox effects on phosphorus sorption and lack of phosphorus-loss data from soils amended with dredged sediments, additional research is needed to fully characterize phosphorus loss risks from land application of dredged sediments.

Salinity

Salinity of land-applied dredged sediments is a concern (USACE, 1987); however, most salinity problems cited are due to sediments taken from brackish or saltwater waterways (Winger et al., 2000; Novak and Trapp, 2005). Data on potential salinity issues in land-applied sediments dredged from freshwater Kansas reservoirs are lacking; therefore, the following information is based on reservoir water analyses.

Salinity of Kansas reservoirs varies because of the precipitation gradient and variety of drainage-basin geology found across the state (Table 3). In general, reservoir salinity increases from east to west in the state, as indicated by increases in electrical conductivity of the water. Soil salinity is determined by measuring electrical conductivity of a saturated paste extract (EC_{se}); EC_{se} is an approximation or relative index of the electrical conductivity of soil water (Zhang et al., 2005). Assuming that sediment pore water is in equilibrium with reservoir water, sediment EC_{se} can be approximated by determining electrical conductivity of reservoir water.

Soils with EC_{se} greater than 1 dS/m are referred to as having high salts, and soils with EC_{se} greater than 4 dS/m are classified as "saline," or severely limited because of salts. Effects of salinity on crop growth are evaluated by comparing EC_{se} with crop-specific threshold values. Yield loss or plant growth problems occur when EC_{se} exceeds threshold values. Most major Kansas crops will tolerate low levels of salinity (Table 4).

Salinity is not an issue for land application of sediment dredged from the majority of Kansas lakes. Some lakes in western Kansas, such as Wilson Lake, probably have sediment with a high dissolved salt content, but the predicted EC_{se} of sediments from these lakes is less than threshold values for sorghum, soybean, and wheat. Corn production, however, could be limited on dredged sediments from Wilson Lake. Correcting soil-salinity problems can be costly. Therefore, salinity of material dredged from lakes with electrical conductivities greater than 1 dS/m should be confirmed with sediment analysis before land-application. Field or greenhouse research trials can help quantify effects of saline sediment applications on crop growth.

Table 3. Electrical conductivity of lake water in Kansas reservoirs[a]

Reservoir[b]	Mean electrical conductivity (dS/cm)	Standard deviation	Number of samples	First sampling date	Last sampling date	Longitude
Olathe Lake	0.539	0.105	103	6/21/00	9/30/05	94°50′ W
Perry Lake	0.308	0.034	15	5/1/92	8/30/93	95°27′ W
Tuttle Creek Lake	0.348	0.131	9	5/5/92	9/1/93	96°38′ W
Milford Lake	0.518	0.086	9	5/5/92	9/1/93	96°55′ W
Cheney Reservoir	0.835	0.056	148	10/7/70	6/13/07	97°50′ W
Kanopolis Lake	1.097	0.339	39	12/28/49	9/3/93	98°00′ W
Waconda Lake	0.766	0.104	8	5/6/92	9/2/93	98°21′ W
Wilson Lake	2.649	0.428	258	8/16/66	9/2/93	98°33′ W
Cedar Bluff Reservoir	1.317	0.255	27	10/23/75	8/9/82	99°47′ W

[a] Data source: USGS (2008b)

[b] Reservoirs sorted from east to west

Table 4. Salt Tolerance Ratings for Various Field and Forage Crops[a]

Sensitive (0-4 dS/m)[b]	Moderately Tolerant (4-6 dS/m)	Tolerant (6-8 dS/m)	Highly Tolerant (8-12 dS/m)
Field beans (dry)	Corn	Wheat	Barley
Red clover	Grain sorghum	Oat	Rye
Ladino clover	Soybean	Triticale	Bermudagrass
Alsike clover	Bromegrass	Sunflower	Crested Wheatgrass
	Sudangrass	Alfalfa	
	Sorghum-Sudans	Tall fescue	
		Sweet clovers	

[a] Table adapted from Lamond and Whitney (1992) with permission
[b] dS/m = deciSiemens per meter, a measure of electrical conductivity of a soil solution. Soils with electrical conductivity of 4 dS/m or greater are considered saline.

Phytoremediation Processes and Methods

Contamination in land-applied dredged sediment is a concern. However, some contaminants can be contained, degraded, or removed through phytoremediation, a process in which plants remediate contamination by taking up contaminated water (USGS, 2008c). Phytoremediation is useful because plants survive higher concentrations of hazardous wastes than most microorganisms used for bioremediation. Phytoremediation works best when soil contaminants are less than 5 meters deep (Schnoor et al., 1995).

Types of phytoremediation include phytoextraction, phytostabilization, rhizofiltration, phytodegradation, phytovolatilization, rhizosphere degradation, and phytorestoration (Salt et al., 1998; Peer et al., 2005). In Kansas, phytostabilization and phytoextraction are most applicable because of generally low trace-element concentrations in lake sediments, the likelihood of dredged sediments being land applied, and the variety of plant species (many with rooting depths less than the 5 meters recommended by Schnoor et al., 1995) available in the state for treatment of different contaminants.

Metal Uptake

One part of the phytoremediation process is uptake of metals through plant roots. Heavy metals can be taken up without negatively affecting plant growth, but the quantity taken up depends on several factors including soil characteristics and plant species. Only some forms of metals in the soil are available for uptake, and availability of different metals is affected by soil pH, organic matter, soil water content, and presence of other metals (Madejon and Lepp, 2006). As water drains out of soil, oxidation can lower soil pH. Lower pH levels increase availability, solubility, and mobility of most metals, which increases their availability to plants (Borgegard and Rydin, 1989; Turner and Dickinson, 1993). Arsenic is an exception; its mobility is lower at lower pH levels (Madejon and Lepp, 2006). Plant-available arsenic could decrease as soil acidity increases with plant growth. This requires further research, especially for Kansas lake sediments with arsenic concentrations that exceed the TELs.

Usually, dredged material is anaerobic, has a pH around 7.0, and has a moisture content greater than 40%. In these conditions, heavy metals are tightly bound to sediment and not available to vegetation (Skogerboe et al., 1987). As dredged material dries and is oxidized, heavy metals become more soluble and are of concern to the environment because they are more mobile in surface runoff and more available to vegetation (Skogerboe et al., 1987).

Contaminated soil becomes less so as plant roots take up metals and translocate them to other parts of the plant. After uptake, many metals are immobilized in roots and not translocated to other plant parts (Cunningham and Lee, 1995). Metals that immobilize in tree roots include chromium, mercury, lead, aluminum, tin, and vanadium. Metals that are translocated include boron, cadmium, cobalt, copper, molybdenum, nickel, selenium, arsenic, manganese, and zinc (Pulford and Dickinson, 2005).

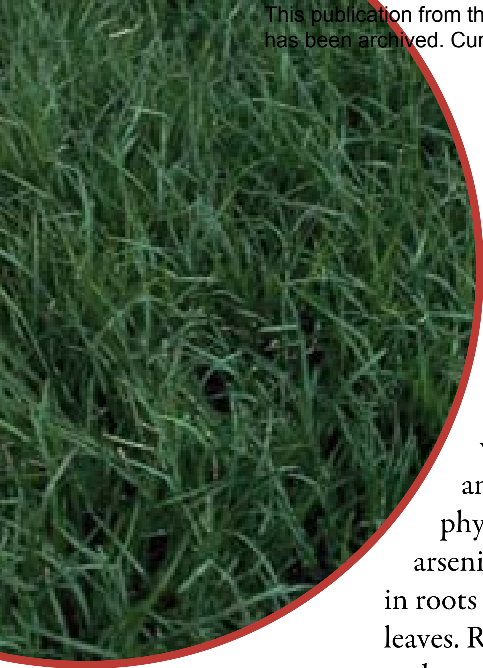
Not every heavy metal is mobilized the same way. Uptake and retention of many heavy metals available to trees follows the pattern of roots > leaves > bark > wood (Pulford and Dickinson, 2005). Copper uptake by weeping willow (*Salix spp.*) occurred in the roots > wood > new stems > leaves (Punshon and Dickinson 1997). Sycamore maple (*Acer pseudoplatanus* L.) concentrated lead and zinc in roots, then lead translocated to stems, and zinc moved to leaves (Turner and Dickinson, 1993). Only cadmium and zinc accumulate in above-ground tree tissues at concentrations sufficiently high enough for phytoremediation to be useful (Pulford and Dickinson, 2005). Other commonly studied metals (e.g., chromium, copper, nickel, and lead) are either poorly bioavailable in soil or not translocated out of roots.

Phytostabilization

According to Raskin and Ensley (2000), the purposes of phytostabilization are to: 1) stabilize waste so no wind or water erosion occurs, 2) stop leaching of contaminants to groundwater, and 3) immobilize contaminants both physically and chemically by making them bind to roots and organic matter. Phytostabilization is best used on soils with low contaminant levels (Raskin and Ensley, 2000; McCutcheon and Schnoor, 2003).

Vegetation reduces wind and water erosion in several ways. Accumulation of leaf litter forms a barrier over the surface of contaminated soil, which provides physical stabilization by reducing splash erosion. Roots of grasses, shrubs, and trees bind and stabilize soil as water runs over it. The litter layer and binding of soil by roots also help reduce wind erosion. Reducing wind erosion with vegetation also lowers human exposure because of reduced potential for inhalation of contaminated soil and ingestion of contaminated foods (Schnoor et al., 1995).

Vegetation also takes up large amounts of water that are lost from leaf surfaces through transpiration. Tree species such as poplar, willow, and cottonwood are good at taking up water from the top 2 to 3 meters of soil (Raskin and Ensley, 2000). This large amount of water moving through the plant from the soil to the atmosphere can decrease the potential for metals to leach from the soil (Schnoor et al., 1995).



Madejon and Lepp (2006) studied three contaminated sites that were naturally revegetated with mosses, ferns, and herbaceous and woody plant species suitable for phytostabilization. They found that arsenic was taken up and immobilized in roots with little transfer to stems and leaves. Root uptake of arsenic in herbaceous and woody plants ranged from 0.66 to 18.3 mg/kg and 1.37 to 5.54 mg/kg, respectively. Amount of arsenic translocated to stems and leaves in herbaceous and woody species was less than 2 mg/kg. These data show that the species tested would work well for phytostabilization because little arsenic was translocated out of roots.

Another study in the southern United States noted successful use of Bermudagrass for phytostabilization on dredged material containing low amounts of zinc (Best et al., 2003). Bermudagrass responded to an increase in zinc levels by increasing bioaccumulation of zinc until a decrease of plant mass occurred at a zinc phytotoxicity level of 324 mg/kg. Red fescue (*Festuca rubra*) also is metal tolerant and has been used in grazing-management strategies on highly contaminated soils (Cunningham and Lee, 1995).

Species native to a particular area are best for phytostabilization because they are most likely adapted to the climate, insects, and diseases present (Peer et al., 2005). The U.S. Army Corps of Engineers (1987) published a list of vegetation that can be used on dredged material; Appendix C contains a condensed version that lists 161 species found in the mid-Plains.

Phytoextraction

Phytoextraction removes contaminants from the system by immobilizing them in soil or biomass (Peer et al., 2005; Pulford and Dickinson, 2005). Vegetation such as grasses and trees can be harvested, which helps permanently remove contaminants from soil. Dried, ashed, and composted plant material can be isolated as hazardous waste or recycled as a metal ore (Kumar Nanda et al., 1995).

Hyperaccumulator plants are good for phytoextraction of metals and include herbs, shrubs, and trees. To be labeled a hyperaccumulator, a plant must take up more than 10,000 µg/g (ppm) of zinc and 1,000 µg/g of copper, nickel, chromium, and lead (Baker and Brooks, 1989). More than 300 of the approximately 400 species of hyperaccumulators accumulate nickel (Brown, 1995). Table 5 compares amounts of selected metals taken up by hyperaccumulators with amounts normally found in plant leaves and soil.

Metals present in Kansas reservoir sediments at concentrations greater than TELs include arsenic, chromium, copper, lead, nickel, selenium, and zinc (Appendix B-2). Examples of hyperaccumulator species that take up these metals are shown in Table 6. Most plants that accumulate metals are slow-growing, small, weedy plants that have a low biomass, but some herbaceous hyperaccumulators such as *Brassica juncea* have a high biomass (Kumar Nanda et al., 1995). Plants with higher biomass can remove greater amounts of contamination and have more uses when harvested (Pulford and Dickinson, 2005). For a more complete list of hyperaccumulators, see Baker and Brooks (1989) and McCutcheon and Schnoor (2003).

Compared with herbaceous hyperaccumulators, trees are advantageous for several reasons (Pulford and Dickinson, 2005):

- Certain tree species have high-yielding biomass that would only need to take up moderate amounts of metal to be effective.
- Trees have more uses when harvested than most hyperaccumulators.
- A greater genetic diversity of fast growing, short-rotational trees such as *Salix spp.* and *Populus spp.* are available. This allows for selection of traits for resistance to high metal concentrations as well as genotypes that have high or low metal uptake.
- Woody plant biomass use is well established.
- Short-rotation, fast growing coppice trees have a high economic value.
- Trees on highly contaminated land are visually aesthetic.
- Trees protect soil surfaces from wind and water erosion because their roots stabilize substrate and their leaves produce organic matter when they drop.
- Uptake of water and transpiration through leaves helps limit leaching of heavy metals from soils and protects groundwater and surface waters from contamination.

Sites most favorable for timber growth include marginal land or abandoned disposal sites on which dredged, dewatered material has been deposited (Best et al., 2003). An additional benefit of using trees is that dredged materials can be applied in thicker layers because tree roots descend farther than herbaceous plant roots. This allows marginal land to be made more productive. And, trees can be used to produce a variety of beneficial products. Tree species suitable for use on dredged material include eastern cottonwood, American sycamore, eucalyptus, green ash, water oak, and sweet gum on periodically flooded sites; and longleaf pine, slash pine, loblolly pine, black walnut, white ash, pecan, and several oak and hickory species on upland sites (Best et

Table 5. Comparison of amounts of heavy metals normally found in plant leaves and soil with the minimum amounts required for plants to be considered hyperaccumulators

Metal	Normal Range of Element Concentrations in Dried Plant Leaves[a]	Normal Range of Metal Concentration in Soil (United States)[b]	Minimum Amount of Metal Taken up to be a Hyperaccumulator[c]
		(mg/kg)	
Arsenic	---	3.6-8.8	---
Chromium	0.2-5	14-29	1,000
Copper	5-25	20-85	1,000
Nickel	1-10	13-30	1,000
Lead	0.1-5	17-26	1,000
Selenium	0.05-1	---	---
Zinc	20-400	34-84	10,000

[a] Raskin and Ensley (2000)
[b] Sandia National Laboratory (2007)
[c] Baker and Brooks (1989)

Table 6. Hyperaccumulator species used to remove metals present in Kansas reservoir sediments

Metal	Scientific Name	Common Name	Family	Location
Arsenic	*Pteris vittata*[a]	Ladder brake	Pteridadeae	USA
	Pteris cretica[a]	Cretan brake	Pteridadeae	USA
	Pteris longifolia[b]	Long-Leaved brake	Pteridadeae	USA
	Pteris umbrosa[b]	Jungle brake	Pteridadeae	USA
	Holcus lanatus[b]	Common velvetgrass	Poaceae	USA*
Copper	*Salix spp.*[b]	Willow species	Salicaceae	USA*
	Brassica juncea[ac]	Indian mustard	Brassicaceae	USA*
	Helianthus annuus[d]	Common sunflower	Asteraceae	USA*
	Avena sativa[e]	Oat	Poaceae	USA*
	Hordeum vulgare[e]	Barley	Poaceae	USA*
Chromium	*Salix spp.*[a]	Willow species	Salicaceae	USA*
	Betula spp.[a]	Birch species	Betulaceae	USA*
	Salsola kali[a]	Russian thistle	Chenopodiaceae	USA
	Brassica juncea[c]	Indian mustard	Brassicaceae	USA*
Nickel	*Brassica juncea*[cd]	Indian mustard	Brassicaceae	USA*
	Helianthus annuus[e]	Common sunflower	Asteraceae	USA*
Lead	*Brassica juncea*[acd]	Indian mustard	Brassicaceae	USA*
	Brassica spp. (others)[a]	---	Brassicaceae	USA
	Helianthus annuus[d]	Common sunflower	Asteraceae	USA*
Selenium	*Astragalus bisulcatus*[a]	Two-grooved milk vetch	Fabaceae	USA*
	Brassica juncea[a]	Indian mustard	Brassicaceae	USA*
Zinc	*Thlaspi caerulescens*[ad]	Alpine pennycress	Brassicaceae	USA
	Brassica juncea[acd]	Indian mustard	Brassicaceae	USA*
	Helianthus annuus[d]	Common sunflower	Asteraceae	USA*
	Avena sativa[e]	Oat	Poaceae	USA*
	Hordeum vulgare[e]	Barley	Poaceae	USA*

[a] Peer et al. (2005)
[b] Baker and Brooks (1989)
[c] Kumar Nanda et al. (1995)
[d] McCutcheon and Schnoor (2003)
[e] Ebbs and Kochian (1998)
*Natural or introduced to Kansas

al., 2003). However, not all are suitable for Kansas.

Using trees for phytoextraction also has disadvantages. Because trees take longer to mature, the time period between sediment disposals is longer than if herbaceous plants are used (Best et al., 2003). Disposing additional dredged sediment around planted trees limits the quantity of oxygen available to the roots and can result in death of the trees. Trees also tend to acidify soil, which could cause increased bioavailability of metals.

Phytoextraction and phytostablization have method-specific advantages and disadvantages (PRC, 1997). Phytoextraction by trees has high biomass production but is disadvantageous because of the potential for off-site migration and transportation of metals to the leaf surface. Phytoextration by grasses has high accumulation but low biomass production and a slow growth rate. A disadvantage for both methods is that metals concentrated in plant biomass must eventually be disposed of. Phytostabilization does not require disposal of contaminated biomass but does necessitate long-term maintenance.

The main disadvantage of all phytoremediation methods is that they are cyclical and occur only during the growing season. Some sites might require additional soil amendments such as manure, sawdust, and lime or the addition of chemicals to increase solubility of metals (Brown, 1995; Murray, 2003). An additional concern is whether plants take up enough contaminants to make a difference or if it will take thousands of years to clean soil to acceptable contaminant levels (Pulford and Dickinson, 2005).

Summary

Dredging and land-applying sediments pose a variety of concerns including costs, locations of disposal sites, economics, and transportation. From a chemical standpoint, dredging and subsequent disposal of sediments as a means to renovate Kansas reservoirs appears viable. Contaminants of concern in most Kansas reservoir sediments evaluated in USGS studies are not analytically detectable or are present at concentrations below USEPA TELs.

A few metals (e.g., lead, arsenic, selenium, copper, nickel, chromium, and zinc) will be issues in some parts of the state. Whether lead will create problems depends on depth of sediment dredged, lake location, and source of lead. Sediments from Empire Lake in southeastern Kansas are most likely to have disposal-related problems because of high concentrations of cadmium, lead, and zinc. Other reservoirs have higher concentrations of lead in deeper sediments because of past use of leaded gasoline and will need to be evaluated if dredging is considered as a remediation option.

Phytoremediation is a natural process that can be beneficial, especially on sediments with low contamination levels, such as those found in many Kansas reservoirs. Many vegetation species, including some native to Kansas, are appropriate for phytoremediation. Although phytoremediation requires further research, it is viable for Kansas and should be considered along with land application of dredged sediment as part of an overall reservoir dredging evaluation.

Questions and Research Needs

Chemical Issues

Although contaminants in Kansas reservoir sediments generally are below TELs, changes that could occur when sediment is removed and placed on land have not been evaluated. Research should examine:

- Redox potential and pH of sediment
- Potential changes in concentrations of metals and nutrients due to land disposal
- Leaching of lake sediment in combination with disposal site soil: Are contaminants of interest mobile or retained in the soil matrix?
- Retention of trace elements in soil: Do amendments (biosolids and liming) prevent trace-element mobility and/or uptake by plants?
- Quantity of manganese- and iron-oxides and hydroxides and content and type of clay present in combined soils at disposal sites: What combination will optimize retention of metals?
- Leaching of nitrate and other contaminants from land-applied dredged sediments
- Management of dredged sediment to minimize nitrate losses
- Phosphorus loss risks from land-applied dredged sediments due to complex redox effects on phosphorus sorption
- Sediment from lakes with high electrical conductivity (field or greenhouse research trials) prior to dredging; correcting soil-salinity problems can be costly
- Methylmercury content in sediment pore-water

Phytoremediation

Successful establishment of vegetation for phytoremediation depends on physical qualities of dredged materials, contaminants present, soil water, soil structure, and salinity. Questions to ask prior to using phytoremediation on dredged sediments include:

- What are the risks of metal uptake by plants and potential transmission up the food chain?
- What can or should be done with material after it accumulates in plant tissue? Can woody material be used if it contains heavy metals? What other disposal possibilities exist?

- What volume of metals is removed by phytoremediation, and what is the rate of uptake?
- What are the economic effects of phytoremediation compared with other remediation methods?
- Will species currently used for phytostabilization and phytoextraction be successful in Kansas? Many species used for phytoremediation are short-rotation, hybrid, fast-growing woody species; are they feasible here?
- Should controlled, pre-trial experiments be conducted to determine likelihood of success before various plant species are used on dredged materials?

Appendix A-1. Estimated annual sediment deposition and chemical loads for selected Kansas reservoirs

		Reservoir (year completed)							
		Perry Lake (1969)[a]	Hillsdale Lake (1981)[bc]	Tuttle Creek Lake (1962)[d]	Cheney Reservoir (1964)[e]	Webster Reservoir (1956)[fc]	Empire Lake (1906)[g]	Lake Olathe (1956)[h]	Cedar Lake (1938)[h]
Reservoir characteristics	Drainage area[i], mi^2	1,117	144	9,628	933	1,150	2,500	16.9*	16.9*
	Years since dam closure[i]	32	15	37	33	40	100	45	62
	Total deposition, acre-ft	56,700	2,100	142,000	7,100	1,267	1,000	317	338
	Mean Annual Sediment Deposition, lb/yr	3,040 million	265 million	1,633 million	453 million	7.8 million	24 million	12.6 million	9.6 million
Mean net load (lb/year) based on values reported for period of study	Phosphorus	3.4 million	154,000	2.5 million	226,000	29,400	---	9,720	14,700
	Nitrogen (Total organic-N + Ammonia-N)	7.6 million	---	672,000	840,000	129,000	---	29,610	19,200
	Total organic carbon	58 million	---	33 million	---	966,000	---	---	---
	Selenium	2,730	---	1,324	190	96	---	---	---
	Arsenic	57,800	---	22,861	15,800	619	---	---	---
	Lead	85,100	---	40,824	8,640	---	6,500	416	326
	Zinc	366,000	---	196,000	37,500	---	120,000	1,966	1,363
	Copper	100,000	---	55,000	7,940	---	830	441	336
	Chromium	302,000	---	132,450	31,740	---	1,591	1,172	864
	Cadmium	1,520	---	717	146	---	780	3.8	3.2
	Nickel	152,000	---	62,143	13,830	---	840	441	355

[a] Juracek (2003)
[b] Juracek (1997)
[c] Mau and Christensen (2000)
[d] Juracek and Mau (2002)
[e] Mau (2001)
[f] Christensen (1999)
[g] Juracek (2006)
[h] Mau (2002)
[i] USGS (2008a)
*Watershed includes both Lake Olathe and Cedar Lake

Appendix A-2. Estimated total sediment deposition and chemical loads for selected Kansas reservoirs

		Reservoir (year completed)							
		Perry Lake (1969)[a]	Hillsdale Lake (1981)[bc]	Tuttle Creek Lake (1962)[d]	Cheney Reservoir (1964)[e]	Webster Reservoir (1956)[fc]	Empire Lake (1906)[g]	Lake Olathe (1956)[h]	Cedar Lake (1938)[h]
Reservoir characteristics	**Drainage area[i], mi^2**	1,117	144	9,628	933	1,150	2,500	16.9*	16.9*
	Years since dam closure[i]	32	15	37	33	40	100	45	62
	Total deposition, acre-ft	56,700	2,100	142,000	7,100	1,267	1,000	317	338
	Total estimated sediment deposition since dam closure, tons	48.6 million	1.9 million	30.2 million	7.5 million	156,000	1.2 million	283,500	297,500
Total estimated load (lb) based on values reported for period of study	**Phosphorus**	109 million	2.3 million	93 million	7.5 million	1.1 million	---	437,400	911,400
	Nitrogen (Total organic-N + Ammonia-N)	243 million	---	24.8 million	27.7 million	5 million	---	1.3 million	1.2 million
	Total organic carbon	928,000	---	610,000	---	19,300	---	---	---
	Selenium	87,360	---	48,988	6,270	3,856	---	---	---
	Arsenic	1.8 million	---	845,868	521,400	24,760	---	---	---
	Lead	2.7 million	---	1.5 million	285,120	---	650,000	18,720	20,212
	Zinc	11.7 million	---	7.25 million	1.2 million	---	12 million	88,470	84,506
	Copper	3.2 million	---	825,000	262,020	---	83,040	19,845	20,832
	Chromium	9.6 million	---	4.9 million	1 million	---	159,110	52,740	53,568
	Cadmium	48,640	---	26,517	4,818	---	78,000	171	198
	Nickel	4.8 million	---	2.3 million	456,390	---	84,000	19,845	22,010

[a] Juracek (2003)
[b] Juracek (1997)
[c] Mau and Christensen (2000)
[d] Juracek and Mau (2002)
[e] Mau (2001)
[f] Christensen (1999)
[g] Juracek (2006)
[h] Mau (2002)
[i] USGS (2008a)
*Watershed includes both Lake Olathe and Cedar Lake

Appendix A-3. Mercury values from sediment cores at selected Kansas reservoirs and estimated mean annual net loads

	Reservoir										
	Lake Afton[a]	Gardner City Lake[a]	Mission Lake[a]	Perry Lake[b]	Tuttle Creek[c]	Cedar Lake[d]	Hiawatha Lake[a]	Lake Olathe[d]	Empire Lake (one core)[e]		
									Top	Middle	Bottom
Number of samples	---	---	---	19/19	41/41	---	---	---	---	---	---
Measured value	---	---	---	---	---	---	---	---	0.12	.029	.022
Minimum, mg/kg	0.03	0.05	0.02	0.01	<0.02	0.05	0.1	---	---	---	---
Mean, mg/kg	0.04	0.06	0.02	---	---	0.07	0.04	0.06	---	---	---
Median, mg/kg	0.04	0.06	0.04	0.05	0.04	---	---	---	---	---	---
Maximum, mg/kg	0.04	0.07	0.04	0.07	1.4	0.14	0.06	---	---	---	---
TEL	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0.13
PEL	0.696	0.696	0.696	0.696	0.696	0.696	0.696	0.696	0.696	0.696	0.696
Mean annual net load, kg/yr[f]	0.46	0.39	0.93	69	144	---	---	---	---	---	---
Mean annual net load, lb/yr[f]	0.35	0.33	0.93	152	317	---	---	---	---	---	---

TEL = threshold-effects level
PEL = probable-effects level; values given for comparison
[a] Juracek (2004)
[b] Juracek (2003)
[c] Juracek and Mau (2002)
[d] Mau (2002)
[e] Juracek (2006)
[f] Values calculated using median concentration, bulk densities, and annual sediment loads reported in studies cited

Appendix B-1. Nutrient and pesticide concentrations in sediment cores from selected Kansas reservoirs

		Nutrients				Pesticides		
		Total nitrogen mg/kg	Total phosphorus mg/kg	Organic carbon (TOC) mg/kg	Total carbon %	DDD µg/kg	DDE µg/kg	DDT µg/kg
US EPA TELs and PELs[a]								
	TEL	---	---	---	---	1.22	2.07	1.19
	PEL	---	---	---	---	7.81	374	4.77
Reservoir		**Values from Sampled Sediment Cores**						
Webster[b]	Range	30-1,910	251-692	10,600-16,200	---	<0.2	<0.2	<0.2
Kirwin[b]	Range	1,200-1,980	422-795	8,310-13,600	---	<0.2	<0.2	<0.2
Waconda Lake[b]	Range	704-3,210	281-904	3,440-19,900	---	---	---	---
Tuttle Creek[c]	Range	600-5,200	198-952	0.84-2	0.93-2.2	<0.50	<0.2	<0.50
Cedar Lake[d]	Range	2,000-2,700	1,370-1,810	---	---	---	---	---
	Median	2,350	1,540	---	1,540	<0.2	<0.2	<0.2
Lake Olathe[d]	Range	1,300-2,700	588-1,030	---	---	---	---	---
	Median	2,000	774	---	---	<0.50	0.2	<0.50
Pony Creek[e]	Range	3,000- 3,400	1,100-1,200	2.6-2.8	3.2-3.5	---	---	---
Otis Creek[e]	Range	2,200-2,400	600-640	2-2.3	3.5-3.7	---	---	---
Mission Lake[e]	Range	1,900-2,400	750-1,200	2.1-2.6	1.9-2.3	<0.50	1.86	<0.50
Lake Afton[e]	Range	2,200-2,600	740-840	1.9-2.4	2-2.5	<1.25	0.22	<1.25
Hiawatha[e]	Range	1,000-1,700	400-680	1.1-2.4	1.1-2.6	1.19	1.99	<0.50
Gardner Lake[e]	Range	1,600-3,100	1,100-1,300	3.2-3.4	3-3.9	0.46	<0.50	---
Edgerton City[e]	Range	1,000-2,200	480-610	0.7-2.1	0.6-2	<0.50	0.22	<0.50
Crystal Lake[e]	Range	2,600-4,300	690-1,300	2.6-3.9	2.7-6.1	<0.50	4.76	<0.50
Centralia[e]	Median	2,400	1,300	2.7	2.7	<0.50	0.27	<0.50
Bronson[e]	Median	3,700	1,100	3.6	3.9	<0.50	0.52	<0.50
Perry Lake[f]	Range	1,300-2,800	630-1,300	1-2.1	1-2.3	<0.5-0.55	<0.2-0.8	<0.50
Empire Lake[g]	Mean	1,250	610	1.35	1.7	---	---	---
Cheney Lake[h]	Range	1,400-2,400	495-647	---	---	---	---	---
Hillsdale Lake[i]	Range	---	410-810	---	---	---	---	---
Milford Lake[j]	Range	---	---	---	---	---	---	---

TEL = threshold-effects level
PEL = probable-effects level
[a] USEPA (2004)
[b] Christensen (1999)
[c] Juracek and Mau (2002)
[d] Mau (2002)
[e] Juracek (2004)
[f] Juracek (2003)
[g] Juracek (2006)
[h] Mau (2001)
[i] Juracek (1997)
[j] Christensen and Juracek (2001)

Appendix B-2. Trace element concentrations in sediment cores from selected Kansas reservoirs

		Arsenic mg/kg	Cadmium mg/kg	Chromium mg/kg	Copper mg/kg	Lead mg/kg	Nickel mg/kg	Selenium mg/kg	Zinc mg/kg
		Trace Elements							
USEPA TELs and PELs[a]									
	TEL	7.24	0.676	52.3	18.7	30.2	15.9	---	124
	PEL	41.6	4.21	160	108	112	42.8	---	271
Reservoir		**Values from Sampled Sediment Cores**							
Webster[b]	Range	8-15	<3.0	<6-26	19-29	16-32	<12-30	0.5-2.7	---
Kirwin[b]	Range	4.6-10	<2.7-3.7	9-33	17-28	14-26	<11-24	<0.5-2.2	---
Waconda Lake[b]	Range	5.4-13.1	<5.1	<10 - 17	7-27	<14-25	<21	<0.6 - 3.6	35-137
Tuttle Creek[c]	Range	6.9-18	0.26-0.6	48-120	20-44	16-160	19-77	0.34-1.5	65-150
Cedar Lake[d]	Range	13-16	0.23-0.36	90-98	32-45	32-35	32-39	0.86-1.1	150-170
	Median	14	0.31	90	32	33	34	0.99	150
Lake Olathe[d]	Range	15-18	0.27-0.41	88-94	32-38	28-40	35-39	0.95-1.2	140-150
	Median	16	0.33	89	36	36	37	0.99	140
Pony Creek[e]	Range	13-17	0.4-0.5	70-77	26-29	24-25	35-38	0.8-0.9	170-200
Otis Creek[e]	Range	10-13	0.4-0.6	76-82	21-22	23-24	35-39	1-1.2	66-74
Mission Lake[e]	Range	12-16	0.3	74-84	28-35	24-31	37-41	0.8-0.9	120-140
Lake Afton[e]	Range	9.9-15	0.6-0.8	69-74	32-35	34-54	37-40	0.7	120-140
Hiawatha[e]	Range	8.2-12	0.5-0.9	43-54	14-20	21-58	21-26	0.4-0.6	58-250
Gardner Lake[e]	Range	7.3-15	0.6-1	83-100	39-210	1,300-1,600	1.3-1.6	130-150	<.50
Edgerton City[e]	Range	7.3-15	0.1-0.4	48-61	14-19	21-24	19-24	0.7-1.1	58-76
Crystal Lake[e]	Range	14-21	0.5-1.4	67-93	25-1600	28-65	32-43	0.8-1.2	130-250
Centralia[e]	Median	18	0.8	77	30	19	40	0.8	110
Bronson[e]	Median	15	0.7	74	200	34	37	0.9	150
Perry Lake[f]	Range	8-25	0.2-0.6	59-100	18-35	18-30	22-54	0.4-1	58-140
Empire Lake[g]	Mean	4.6	29	66.3	34.6	270	25	0.8	4,900
Cheney Lake[h]	Range	5.9-10	0.24-0.40	55-100	12-24	14-24	18-43	0.31-0.52	58-120
Hillsdale Lake[i]	Range	---	---	---	---	---	---	---	---
Milford Lake[j]	Range	6.1-9.9	≈0.9-1.5	34-46	21-30	<33-53	29-38	0.2-2.2	29-38

TEL = threshold-effects level
PEL = probable-effects level
[a] USEPA (2004)
[b] Christensen (1999)
[c] Juracek and Mau (2002)
[d] Mau (2002)
[e] Juracek (2004)
[f] Juracek (2003)
[g] Juracek (2006)
[h] Mau (2001)
[i] Juracek (1997)
[j] Christensen and Juracek (2001)

Appendix C. Vegetation used to remediate dredged materials[a]

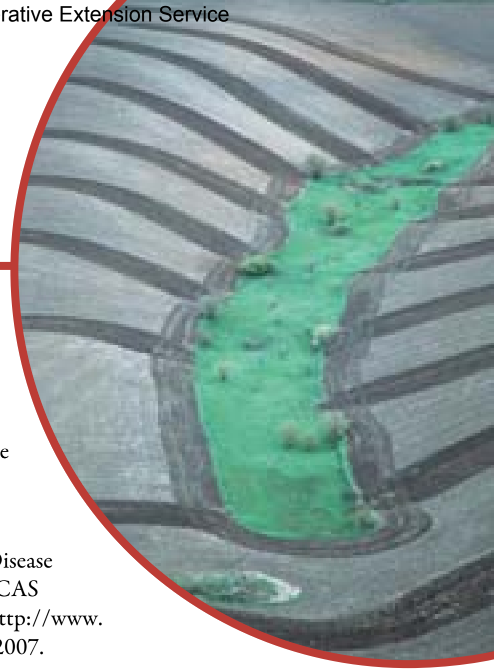
Plant Name	Natural	Planted
Grasses		
Barley	x	x
Barnyard grass	x	
Beaked panic grass	x	
Big bluestem	x	
Bromegrass	x	
Bromesedge	x	
Corn	x	x
Deertongue	x	
Fall panic grass	x	
Foxtail millet		
Goose grass	x	
Green bristlegrass	x	
Johnson grass	x	
Jungle rice	x	
Large crabgrass	x	
Oat		
Orchardgrass	x	
Panic grass	x	
Prairie cordgrass	x	x
Quackgrass	x	
Red fescue	x	
Redtop	x	
Red canary grass	x	
Rice cutgrass	x	
Rye		
Sand dropseed	x	
Sixweeks fescue		
Smooth crabgrass	x	
Sorghum		
Sudan grass		
Switchgrass	x	
Timothy	x	x
Wheat		
Wild rye	x	
Yellow bristlegrass		

Plant Name	Natural	Planted
Herbs		
Alfalfa		
Alsike clover		
Arrow-leafed tearthumb	x	
Black medic	x	
Black nightshade	x	
Blackseed plantain	x	
Bracted plantain	x	
Broadleaf plantain	x	
Chufa	x	x
Common chickweed	x	
Common lambsquarters	x	
Common mullein	x	
Common purslane	x	
Common ragweed	x	
Common spikerush	x	
Common threesquare	x	
Cow pea	x	
Crimson clover		
Curly dock	x	
Dwarf spikerush	x	
Flowering spurge	x	
Giant ragweed	x	
Goosefoot	x	
Hairy vetch		
Hardstem bulrush	x	x
Hop clover		
Horseweed	x	
Japanese clover		
Jerusalem artichoke		
Korean clover	x	
Ladino clover		
Ladysthumb	x	
Lespedeza		
Malta starthistle	x	
Mapleleaf goosefoot	x	
Marsh pea	x	

Plant Name	Natural	Planted
Marsh pepper	x	
Maximillian's sunflower		
Mexican tea	x	
Narrowleaf vetch	x	
Nodding smartweed	x	
Nutsedge	x	
Pennsylvania smartweed	x	
Pokeberry	x	
Prostrate knotweed	x	
Prostrate spurge		
Purple nutsedge	x	
Purple vetch	x	
Red clover	x	
Redroot pigweed	x	
Schweinitz's nutsedge	x	
Sea blite	x	
Seaside dock	x	
Sericea lespedeza		
Sheep sorrel	x	
Soybean	x	x
Spotted burclover		
Spotted spurge	x	
Squarestem spikerush	x	
Sunflower	x	
Tansy mustard		
Tumbleweed	x	
Virginia pepperweed	x	
Western ragweed	x	
White clover	x	x
Wild bean	x	
Wild buckwheat	x	
Wild sensitive pea		
Wild strawberry		
Wooly croton	x	
Wooly indianwheat	x	

Plant Name	Natural	Planted
Vines		
Common greenbrier	x	
Fringed catbrier	x	
Japanese honeysuckle	x	
Kudzu		
Muscadine grape	x	
Peppervine	x	
Virginia creeper	x	

Plant Name	Natural	Planted
Shrubs and small trees		
American elderberry	x	
American hornbeam		
American plum	x	
Black raspberry	x	
Carolina ash		
Carolina rose	x	
Eastern hophornbeam	x	
Gray dogwood	x	
Halberd-leaved willow	x	
Multiflora rose	x	
Poison ivy	x	
Possumhaw	x	
Rough-leafed dogwood	x	
Russian olive	x	x
Sandbar willow	x	
Shining sumac	x	
Silky dogwood	x	
Smooth sumac	x	
Squaw huckleberry		
Staghorn sumac	x	
Tartarian honeysuckle	x	
Thorny eleagnus	x	
Wild apple	x	
Witch hazel		
Shining sumac	x	



Plant Name	Natural	Planted
Large Trees		
American sycamore	x	
Black cherry	x	
Black gum	x	
Black locust	x	
Black walnut[b]	x	
Black willow[b]	x	
Eastern redcedar[b]	x	
Green ash	x	
Hackberry	x	
Honeylocust	x	
Mockernut hickory		
Peachleaf willow	x	
Pecan		
Persimmon	x	
Pignut hickory		
Red maple	x	
Red mulberry	x	
River birch	x	
Sassafras	x	
Sugarberry	x	
Sugar Maple	x	
White ash	x	
White oak		
White poplar		

[a] Data source: USACE (1987)
[b] Not listed as in mid-Plains, but is present in Kansas

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