Geospatial Information Handbook for Water Resources and Watershed Management

Advanced Applications and Case Studies

Edited by
John G. Lyon
Lynn Lyon

CRC Press
Taylor & Francis Group

Geospatial Information Handbook for Water Resources and Watershed Management, Volume III

Volume III of *Geospatial Information Handbook for Water Resources and Watershed Management* discusses water and watershed issues such as water quality, evapotranspiration, water resource management, and ecological services.

Featured is a two-stage ditch and river geomorphology case study section with related water geospatial applications, including historical image analyses of floodplains and channels and resulting change in river geomorphology through erosion and transport and influence on dependent vegetation communities.

- Captures advanced Geospatial Technologies (GT) and their applications to address a wide spectrum of water issues
- Provides real-world two-stage ditch and river geomorphology case studies using river, stream and channel measures and change models, and bankfull discharge modeling
- Global in coverage with applications demonstrated by more than 170 experts in water sciences and two-stage ditch and river geomorphology

This handbook is a wide-ranging and contemporary reference of advanced geospatial techniques used in numerous practical applications at the local and regional scale and is an in-depth resource for professionals and the water research community worldwide.

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Volume III

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CRC Press
Taylor & Francis Group
Boca Raton London New York

CRC Press is an imprint of the
Taylor & Francis Group, an **informa** business

First edition published 2023
by CRC Press
6000 Broken Sound Parkway NW, Suite 300, Boca Raton, FL 33487-2742

and by CRC Press
4 Park Square, Milton Park, Abingdon, Oxon, OX14 4RN

CRC Press is an imprint of Taylor & Francis Group, LLC

ISBN: 978-1-032-00655-0 (hbk)
ISBN: 978-1-032-00659-8 (pbk)
ISBN: 978-1-003-17509-4 (ebk)

DOI: 10.1201/9781003175094

Typeset in Palatino
by codeMantra

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Editors

Over the years, John G. and Lynn Lyon have worked on water issues with great interest and heart. A hallmark of their scholarship has been helping others with scientific authorship. Books are a great way to capture thoughts and methods and make them accessible to others worldwide. In that vein, here they have advanced a whole cadre of thought leaders with a broad focus on water.

Whether on the local, regional, continental, or global scale, water issues unite passions. Part of the challenge is the breadth and complexity of water. If many people take a piece, the whole society can fashion sustainable outcomes. This handbook is focused on presenting a number of ways to facilitate these thoughtful contributions.

John earned a PhD from the University of Michigan, and Lynn earned a master's degree from Ohio State University.

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Introduction to Volume III

John G. Lyon and Lynn Lyon

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Introduction to the Water Geospatial Handbook

Many current capabilities of geospatial analyses, modeling, and remote sensor measurements are presented. Together these are known as geospatial technologies (GT). Once the domain of geospatial experts with expensive tools, modern technologies are now widely available and accessible to a broad audience seeking to better understand a range of large-scale issues.

This volume looks at the use of GT in applications focused on water resources, land use land cover analyses (LULC), water quality, and landscape ecology arenas, among others.

History

The use of geospatial technologies has allowed more complex and larger projects to be conceived and executed. Researchers typically have a given study location such as Hubbard Brook Ecosystem Study site, Agriculture Research Service sites, Forest and Range Experimental Stations, or UN Heritage Ramsar and World Heritage Convention sites, where they develop

DOI: 10.1201/9781003175094-1

and test new techniques or propose how to measure a construct with new tools. These techniques and measurements are frequently used in conjunction with legacy techniques, tools, and multiple ways of measuring to gain a broader understanding of the study area.

Traditional Approaches Augmented

In 2000, the book *GIS for Watershed and Water Resource Management* consolidated the results of some of the large GT water projects under way at the time. Many journal articles and books have been published before and since as a lot has happened in 20+ years.

Modeling, in particular, has grown exponentially. Traditional measurements coupled with GIS and remote sensing–derived data are now the standard foundation for many environmental and ecological models.

Take as an example water flows. Traditional measurements often come from gauges that provide continuous or interval-based data. Data from gauges is preferred but limited to locations where gauges already exist. New gauges are expensive to install and operate, and new or temporary sites do not have the history of old. Historic water flow data for multiple years is available from sources such as the US Geological Survey's (USGS) National Water-Quality Assessment (NAWQA) program and their projects. Detailed coverage – but still local or regional at best.

With the addition of geospatial technologies, researchers can take traditional water flow and quality measurements, and add data from remote sensors. Remote measures can provide large area coverage information. Scientists can interpolate on a scientific basis between and across the data fields. This work can be done in such a manner that facilitates scaling or interpolations based on known characteristics in the remote sensor–generated database.

Another example where geospatial information augments traditional data is in measuring land use. Land use is difficult to measure but can be inferred by using a surrogate or indicator measurement. Remote sensors can show land cover data that can be measured and quantified. Land use then can be inferred by land cover. Land cover serves as a surrogate or indicator for land use, and correlations can be made between land cover and land use (LULC). Formulas and models based on these data can be developed. Coupling processes and variable behaviors associated with land cover allows for integration across the landscape.

In large-area projects, researchers may want to look at water quality and land use, so they would use a combination of traditional water flow and quality measurements and geospatial data. Gauges provide measurements of water, and satellites provide land cover data. An entire watershed can be mapped and analyzed with this combination of traditional and geospatial data. Researchers can use these same methods with other watersheds and then compare relative water quality of one watershed to another.

Adding geospatial technologies and approaches has reduced the burdensome issue of limited data allowing researchers to get a big-picture view of larger sections of the landscape.

Building Spatial Datasets

A great advantage of remotely sensed data is the collection of uniform samples over large areas. Data is collected in a grid cell or matrix format yielding a searchable and analyzable spatial dataset. The resulting dataset can be projected as an image in two dimensions, or as a matrix cube in three dimensions. In three plus dimensions, the data can be stored as an n-dimensional hypervolume of variables or spectral values or principal component axes. The layers stacking in n-dimensions can be the multispectral or hyperspectral wavelength values, or ratio indices, or even land cover identification numbers aka categorical variables.

Mega datasets can be fashioned with layers of variables creating a cube of data, with x and y coordinates being the grid cells navigated into a map projection and the z value being the brightness number or radiance or irradiance.

The matrix of grids cells allows one to image the materials and to operate on the image database as layers of content and foster visualization. For human interpretation, a variety of images can allow evaluation of phenomena. For machine learning or AI efforts, the variety is endless and need only be guided by theory and hypotheses. Certain procedures are valuable for data visualization and analysis, including image processing techniques, simulation modeling, virtual reality, and artificial reality.

Spatial Positioning

The advent of global positioning system (GPS) or three-dimensional satellite positioning in X, Y, and Z dimensions, and subsequent navigation is a real advantage of GT. These GT technologies can include positioning via GPS, the Russian GLONASS, or the European Space Agency's Galileo global navigation satellite system or a combination of them.

Digital elevation models (DEMs) can provide pixel point elevations or Z values in grid cells. The resulting products can be processed for details like slope and aspect or be used to identify low and high points. They can be used to model and route water on its pathways through gullies, streams, and rivers.

DEMs can be obtained at various mapping scales from government or commercial sources. One can make their own if elevation or Z data are available and navigated into X, Y coordinate systems. GPS, photogrammetry, or surveying can supply these details for DEM creation.

All these mapping projects can be navigated into map projections such as Universal Transverse Mercator (UTM) or latitude and longitude values.

Spectral Resolution

In the current world of remote sensor imaging, a variety of spatial and spectral resolutions are available. Tools like Google Maps can help, and Google Earth Engine (GEE) and other software can ingest the likes of Landsat 8, the Indian IRS, and European Sentinel satellite sensors, as well as other systems processing small satellites as well as aircraft and drone-borne sensor data. Along with big data computer processing and analysis, much can be done. One still makes use of spectroscopy, spectral signature analyses, photogrammetry, surveying, geodesy, and light-ranging LiDAR and microwave sensors or Radar in these endeavors.

Advanced sensors can also measure spectral differences in materials by parsing wavelengths of light into digital range bins. This allows discrimination of the spectral characteristics of materials in relative terms and/or in absolute light energy terms upon "cleanup" of random source contributions to the spectra.

This brings detailed sensing at a distance from the sensor. Remote sensing is like laboratory spectroscopy that has been used up close in identification of chemicals and their concentrations via laboratory instruments. This is now the use of light spectra from the earth's surface to characterize those mixtures within the imaged grid cell or pixel. The mix of materials within the pixel reflects spectrally and creates a pattern. Called a spectral signature, it acts like a "fingerprint" of the pattern of materials imaged and/or their mixtures and can be subject to detailed analyses.

Many methods of spectral analysis are available to help pull detail out of the measured parameter. They can include wavelength or wave band selection (e.g., principal component analysis, band to band correlation, stepwise discriminant analysis); use of spectral indices (e.g., band ratios or normalized difference indices); linear multivariate statistics and models (e.g., multivariate regression, partial least squares regression, principal component regression); and nonlinear methods (e.g., spectral angle mapper, wavelets). These approaches are detailed in the chapters ahead.

With all these bands, it is also possible to winnow their number to only those that supply actual information for the task at hand. Feature selection is the action and methods that can reduce the array of bands to meaningful numbers while reducing the dimensionality of the dataset and "noise" associated with the multiple bands. This is particularly true of hyperspectral sensors where optimization of band selection is necessary to avoid too much duplicative or intercorrelated data.

Scaling and Modeling

Spatial computing power is used fully by models, and hence more detailed and spatially pertinent results are produced. A strength of GT is that it is possible to process the datasets using many types of numerical analysis procedures. One can process thematic and cartographically true matrix computations via linear algebra, and/or simpler methods.

The capability to store and quantify data on a spatial basis is an inherent characteristic of geospatial technologies or GT. Modeling with GT facilitates the approximation of processes to understand results and forcing function variables, all the while stored within a georeferenced database.

The use of statistical analyses has proven of great value in water studies over large areas.

Statistical approaches evaluate variables or phenomena as to the variability of their behavior. One can measure model accuracy and precision by traditional statistical measures such as probability and significance levels, goodness of fit via coefficients of determination, correlations, and analysis of variance and regressions with parametric or non-parametric assumptions as to distribution.

The goals are to test hypotheses and to develop relational models of empirical origin. If these models are robust, the relationships can often be applied to other systems or locations and to different times of the year or season.

Deterministic Modeling

Models often take the deterministic form, where the phenomena being studied are mathematically modeled and simulated. Using numerical descriptions for the physical, chemical, or biological processes of interest results in complex models composed of submodels addressing each phenomenon or process with weighted contributions to model results.

Both statistical and deterministic models often consist of several submodel units with "fitting" coefficients. Coefficients used with this approach reflect the characteristics of natural processes, and they will adjust the contribution of variables or submodels to the overall model parameters to mimic the modeled processes.

A more "natural" coefficient better defines the behavior of model variables in mimicking processes. It also allows the modeler to achieve high fidelity between natural systems and their model simulations. To optimize the simulations of natural phenomenon and processes, the coefficients need to reflect the reality of the situation. As a given model begins to approximate nature, its further development often takes the path of improving the quality of coefficients.

Frequently, several experiments will be executed to better measure the level of a coefficient and its effect on the whole model. This to better have it mimic nature and help supply better model predictions.

Experiments using statistical or deterministic models can be greatly facilitated by the analysis of these individual model coefficients. These analyses are driven to find the sensitivity of the overall model results or simulations to a given variable.

The characterization of the behavior of a given variable and its influence on the results of model simulations is called a sensitivity analysis. Sensitivity analyses are part of good modeling strategy because it is very desirable to understand the individual contributions of model coefficients to the overall results, and to ensure that each variable and/or submodel contribution is appropriate to or like nature.

Verification

The success of a model is commonly evaluated using verification incorporating independent data sets. Verifications or accuracy assessments are part of a good modeling strategy because it is important to demonstrate precision and accuracy independent of the data sets used to "train" the model.

Remote sensor data sets are often used to fulfill the requirement for an independent data set to check model results. This can be done *a priori* by subsampling the original data set and retaining the independent subset to use at the end, or another dataset can be obtained and used at the end. The validation can be subjected to accuracy assessment testing for correct identification of say land cover and identifying error of commission and omission. By storing the results in a two-by-two "confusion" matrix, one can judge overall accuracy and class accuracies along the matrix diagonal, and the identity of errors or confusion found in the off-diagonal cells. Further analyses with Kappa statistics can help show goodness of the effort and utility.

Applications

The thoughtful management of resources can lead to the betterment of water, soils, plants, animals, other biota, and humans. Efforts over the last 100 years internationally and domestically have included forest management for watersheds and drinking water production, conservation, fire management, and land and water management. Current approaches include sustainability, landscape ecology, ecological services, and generally the thoughtful use of renewable resources. Climate solutions are being studied and implemented.

To understand the natural variability of earth processes and the impact of humans, research is necessary. Once characteristics and processes are known, one can implement approaches that parse and reduce use of resources such as water.

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Introduction to Volume III and Two-Stage Ditch and River Geomorphology Case Study

Jonathan D. Witter, Jessica D'Ambrosio, John G. Lyon, and Lynn Lyon

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The last decades have seen increasing interest in protecting, enhancing, and restoring the ecology of creeks and ditches, streams, rivers, and their watersheds. Achieving these goals requires sound fundamental and applied knowledge, as well as close interaction between ecological scientists and engineers. Systems approaches, as well as a good understanding of processes at a variety of spatial and temporal scales, are needed.

The team has addressed the importance of an active floodplain in wadable two-stage streams, where it plays a role in sustaining or establishing dynamic equilibrium. Focus is placed on the size and geometry of the active floodplain relative to the size of a main channel that is shaped by channel-forming discharges. Consideration has also been given to floodplains for modified streams and constructed channels such as agricultural ditches. All aid in understanding the hydrology, hydraulics, and fluvial geomorphology of these systems and facilitate use of this knowledge to protect or size a self-sustaining two-stage channel system (Volume III, Chapters 3–6).

The section also focuses on measurements and geospatial technologies or GT to render an understanding of stream and river suspended sediments

DOI: 10.1201/9781003175094-2

loads and bed loads and their transport and fate over time. Many questions exist about channelization and increased flows, and peak flows resulting from weather and climate. Studies on the Minnesota River looked at changes since 1936 and potential sources of erosion, sediment transport and fate, and ecosystem services provided by river waters and riparian vegetation.

Here several methods are described to study change and to model these changes over time. All are good approaches for readers to make their own measurements and models to understand local and regional trends and the influence of climate. The goal of these contributions is to aid the reader in understanding the hydrology, hydraulics, and geomorphology of these systems by implementing these GT tools and approaches at a variety of scales in their own efforts.

The Two-Stage Ditch and River Geomorphology Case Study was co-edited and co-authored by Jonathan Witter and Jessica D'Ambrosio in addition to John Lyon and Lynn Lyon. This Case Study is dedicated to Professor Andy D. Ward of Ohio State University. He was a true friend, mentor, and thoughtful leader who loved us as family. We all worked together for years enjoying challenging scientific and engineering endeavours while sharing life at its best. He is missed but never far from our hearts living on in cherished memories.

Volume III, Chapter 3, History of Ditches and Drainage

In "Water Resources and Agricultural Ditch Management," Jessica D'Ambrosio, Jonathan Witter, and John Lyon share a brief history of improved drainage in the Midwestern USA including a summary of recent advancements in agricultural ditch designs that are more consistent with fluvial form and processes (Volume III, Chapter 3).

Two-stage ditches can be a stable, viable option for drainage ditch management if designed and installed properly on the landscape. The approach may also enhance ecological services while meeting drainage needs essential for agricultural production.

Volume III, Chapter 4, Active Floodplain Characteristics and Discharges

In "Active Floodplain Requirements for Sustaining Two-Stage Channel Geometry," A.D. Jayakaran, Dan E. Mecklenburg, Jonathan Witter, G.E.

Powell, and A.D. Ward address active floodplains in two-stage stream systems where the active floodplain plays an important role in sustaining or establishing dynamic equilibrium. The focus is placed on the size and geometry of the active floodplain (Stage 2) relative to the size of the main channel (Stage 1) that is shaped by channel-forming discharges (Volume III, Chapter 4).

A floodplain ratio (FPR) was used to evaluate different active floodplains. FPR is the ratio floodplain width, when the channel-forming discharge is exceeded, divided by the top width of the channel when this discharge occurs. Ideally, active floodplains should have FPRs greater than five though smaller floodplains will have some beneficial influences on the sustainability of channel systems. The approaches described here are relatively straightforward and represent a minimum level of analysis that should be performed if modifications or protection strategies are proposed for a stream system.

The goal is to provide the reader an understanding of the hydrology, hydraulics, and geomorphology of these systems and to use this knowledge to size a self-sustaining two-stage channel system.

Volume III, Chapter 5, Design Tools for Two-Stage Ditches

In "Enhanced Channel Design v2.6: A Design Tool for Two-Stage Ditches and Self-Forming Channels," Jonathan Witter and Dan Mecklenburg summarize the capabilities of a freely available spreadsheet tool developed to facilitate design of two-stage ditch and self-forming channel approaches for surface drainage in low-gradient landscapes (Volume III, Chapter 5).

Drainage channel designs require input and reduction of survey data; analysis of channel morphology, hydrology, and hydraulics; estimates of earthwork volumes; and production of construction plans. The spreadsheet tools facilitate rapid evaluation of multiple design alternatives and assessment of costs and benefits to facilitate informed decision-making.

Volume III, Chapter 6, Ditches and Stability over Time

In "Evaluating Geomorphic Change in Constructed Two-Stage Ditches," Jessica D'Ambrosio and others studied geomorphic changes in channels, benches, and banks of two-stage ditches in the US Midwest (Volume III, Chapter 6).

Three to ten years after construction, inset channel changes reflected natural adjustments, but not all ditches had reached their quasi-equilibrium state.

Ditches had experienced both degradation and aggradation on the benches at a rate of 0.5–13 mm/year. Localized scour was observed on the banks at some sites, but for all but one site, these changes were not statistically significant. Except for the removal of woody vegetation, none of the ditches required routine maintenance since construction.

This is important as ecological and socioeconomic impacts of drainage ditch maintenance activities can be significant, leading to destruction of microhabitat, and downstream harmful algal blooms and increased sedimentation. The approach has potential to enhance ecological services that ditches provide while meeting natural drainage needs essential for agricultural production and sustainability.

Volume III, Chapter 7, Remote Measures of River Change

In “Air-Photo-Based Change in Channel Width in the Minnesota River Basin: Modes of Adjustment and Implications for Sediment Budget,” Chris Lenhart and team used aerial photograph-based measurements of channel width for the 1938–2015 period (Volume II, Chapter 7). Measurements were taken at 16 multibend subreaches by digitizing the area between vegetation lines and dividing by centerline length.

The Minnesota River and major tributaries have experienced large increases in discharge over the past century. Width change results for a 146.5 km reach of the lower Minnesota River between 1938 and 2008 showed considerable increases in width for the main stem (0.62 ± 0.10%/year) and major tributaries (0.31 ± 0.08%/year) but were inconclusive for smaller channels (width < 25 m).

Analyses of DEMs and regional hydraulic geometry showed that the main stem and larger tributaries accounted for the vast majority (~85%) of bankfull channel volume. High-order channels were thus disproportionately responsible for sediment production through cross-section enlargement, although floodplains or off-channel water bodies adjacent to these channels likely represent important sediment sinks.

Widening was associated with lateral centerline movement and temporal change in at-a-station hydraulic geometry for water surface width, indicating that widening was associated with cross-sectional change and not simply upward movement of the vegetation line.

Because channel enlargement can play an important role in sediment production, it should be considered in sediment reduction strategies in the Minnesota River basin and carefully evaluated in other watersheds undergoing long-term increases in discharge.

Volume III, Chapter 8, Fluvial Geomorphology and River Widening

In "The Role of Hydrologic Alteration and Riparian Vegetation Dynamics in Channel Evolution along the Lower Minnesota River," Chris Lenhart and team investigate bank retreat in the lower Minnesota River since 1938. Specifically, how have changes to river form influenced sediment transport and deposition in the lower Minnesota River and how hydrological and ecological processes affect channel change (Volume III, Chapter 8).

To quantify channel sediment and phosphorus-loading rates in the lower Minnesota River, they analyzed historic aerial photos for evidence of channel change. They performed long-term monitoring of erosion and deposition rates within the river corridor, and calculated channel sediment transport rates (Volume III, Chapter 7).

It was hypothesized that channel straightening, reduction in floodplain access, and streamflow increases contributed to increased channel-derived sediment load and decreased point bar deposition. Secondly that hydrologic changes have reduced woody riparian vegetation on sandbars, further promoting channel widening.

Sediment deposition rates in the floodplain have increased since European settlement by an order of magnitude. Results showed that the river has widened by 52% between Mankato and St. Paul since 1938, on average contributing 280,000 Mg of sediment per year and 153 Mg total phosphorus. The river also was shortened by 7% since 1938, likely increasing bankfull shear stress and stream power.

Ecohydrological studies also showed that establishment of woody riparian plants has been inhibited on sandbars by prolonged summer flow duration and scour at high flow. This created the potential reduction in point bar growth and strata for colonization by woody plants.

Volume III, Chapter 9, Prioritizing Remediation of Lakes

In "A Comparison of Methods for Prioritizing Lakes in Minnesota," Paul Ramonski and Kristin Carlson prioritize mitigation of water quality and make recommendations on how to combine prioritization approaches along with a peer review process to produce lake priority lists that were both defensible and practical, and realistic in terms of budget (Volume II, Chapter 10).

Identifying lakes in which to invest water quality conservation efforts called for comparisons of different approaches to prioritization. Many

Minnesota lakes are candidates for water quality protection or restoration. Lakes were objectively ranked using a multi-criteria values-based model that included phosphorus-loading resilience, level of watershed degradation, and feasibility of water quality protection or restoration.

Ramonski and Carlson explored how the list of priority lakes for remediation of water quality might change when incorporating benefit:cost ratios. The ratios were used with a hedonic model to predict land value increases with total phosphorus-loading reductions. In addition, they examined the influence of including data on lakes with unique or high-quality biological communities. The multi-criteria values-based model was moderately correlated with the benefit:cost ratio approach.

However, the exclusion of benefits and cost in the prioritization would likely result in the loss of a modest amount of potential benefit (~20%). A focus on impaired waters would likely result in considerable forgone benefit (~80%) and substantially higher costs.

All provide great inputs to the public and state decision makers, and to facilitate annual decisions that do the best possible good with funds and resources available.

Volume III, Chapter 10, Ecological Services, Water, and Climate

In "Water Yield Ecosystem Services Assessment in Periyar Tiger Reserve, Southern Western Ghats of India," Shiju Chacko and team address protected area water resources, climate, and ecosystem services (Volume III, Chapter 10). Water yield calculations and mapping are of great importance to conservation planning and resource management.

A water yield model based on InVEST was employed to estimate water resources in the Periyar Tiger Reserve (PTR). The analysis included land use/land cover (LULC), average annual precipitation and potential evapotranspiration, soil depth at various elevations, and plant available water content. The estimated water yield of PTR was 2.33E+09 m^3/year with 67% from Periyar river watershed and 33% from Pamba river watershed.

A comparison of water-yielding capacity of forest ecosystems with different vegetation types indicated that there was a decreasing trend from evergreen forest to semi-evergreen forest, to grassland-savanna, to moist deciduous forest, and on to plantations. It was estimated that 37% of evergreen and 24% of semi-evergreen forests share the major portion of the total water yield or a total of 63.5% of the Periyar Tiger Reserve, revealing the significance of ecosystem services of the protected area and the importance to conserve and require efficient management systems.

Volume III, Chapter 11, Ecological Services and Water Quality

In "Water Quality Parameters as Related to Small Watershed Land Cover," Shyamal Karmakar, Enamul Hoque, and team found different land covers that affect water quality through a variety of interactions directly or indirectly. At root microsites during nutrient uptake, the litter layer and its decomposition, microbial interactions, and physio-mechanical characteristics all play roles, and vegetation, soils, and water all influence the draining water quality (Volume III, Chapter 11).

Water samples from creeks and seepages were collected from mountainous catchments of varied land cover in southeastern Bangladesh. To further understand land cover effects water chemistry was compared using paired watersheds. Sample analysis for common anions and cations revealed shifting cultivated catchments contained higher SO_4^{2-} and NO_3^- and lower HCO_3^-, Na^+, K^+, and TDS compared to natural vegetation areas. The cations showed a regular trend with land cover change more so than anions. Other relationships were found demonstrating how chemical changes in watersheds could be linked with LULC.

Volume III, Chapter 12, Capturing Soil Moisture Variations

In "Soil Moisture Estimation Using a SAR Water Cloud Model for an Improved Anchor Pixel Selection Process in SEBAL," M. M. Prakash Mohan, K. Rajitha, and Murari R. R. Varma address the sensible heat flux component calculation in the Surface Energy Balance Algorithm for Land (SEBAL). They attempted to reduce the gap between the ground realities and simulated results by adding soil moisture as a supplementary parameter.

The conventional methods for anchor pixel selection have limitations in capturing the soil moisture variations beneath the canopy as they utilize only the thermal remote sensing images and ancillary datasets like land cover and crop type maps.

The prospects of the semi-empirical Water Cloud Model (WCM) to estimate the soil moisture content was tested in a fragmented agricultural region for different time periods. The concurrent satellite data obtained from Sentinel 1A, and Landsat 8 supplied inputs for inversion modeling based on the Levenberg-Marquardt approach. The present research introduced the "Virtual Normalized Difference Vegetation Index" concept to refine the WCM and yielded reliable soil moisture output to supplement the anchor pixel selection process.

The robustness of the approach was justified by considering the available energy at anchor pixel locations. The research outcomes revealed that the anchor pixel selection “with” and “without” soil moisture criterion significantly impacts actual evapotranspiration estimation. The research also explores the scope of the synergetic use of optical and Synthetic Aperture Radar (SAR) inputs in SEBAL.

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Water Resources and Agricultural Ditch Management

Jessica D'Ambrosio, Jonathan D. Witter, and John G. Lyon

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DOI: 10.1201/9781003175094-3

Background

Rivers, lakes, and streams and their floodplains have supported human civilizations for millennia. Settlements were established next to a river or stream where there would be ready supplies of good-quality water and fertile land to support food production. The primary functions of rivers and streams in early civilization included providing a potable water supply, efficient navigation, and sustaining agricultural crop growth irrigation or draining water from the land. Water management has rendered improvements to natural drainage conditions one of the most influential engineering technologies in human history.

The role of streams and rivers, including their floodplains, has changed concurrently with changes in the role of agriculture in society. With the societal benefits provided by natural, modified, and constructed lotic systems have also come great economic and environmental costs that only in recent history have garnered sufficient attention by the scientific, engineering, and regulating communities. Agricultural ditching and drainage are very much needed today, but in some locations classic drainage system designs have resulted in over drainage of the land, severe water quality problems, and an overall loss of watershed ecosystem function (Urban and Rhoads, 2003).

Drainage and ditching technologies developed out of necessity resulting from population growth and land development pressure. Channelization and land drainage were applied for two prevailing reasons: (1) reclamation of "swamp" land for agricultural use and referred to as horizontal expansion, and (2) improved drainage on existing agricultural land and referred to as vertical expansion. Over the past 150 years, more than 200,000 miles of waterways have been modified and 80% of some watersheds have subsurface or surface drainage in the United States (Schoof, 1980; Blann et al., 2009). The United States Department of Agriculture (USDA) Economic Research Service (Pavelis, 1987) estimated that 110 million acres of agricultural land, nearly 70% of crops, in the USA have benefited from artificial drainage. The total US investment in drainage since 1855 has been estimated to be $56 billion (Pavelis, 1987).

In the 21st century, we find that along with widely recognized ecological and economic costs came recognition that agricultural drainage system designs may need to be modified to serve wider purposes beyond agriculture (Jayakaran et al., 2005; Shedekar et al., 2020). In some cases, this might simply be a non-agricultural application, but in other cases this may reflect a fully new approach to drainage design and maintenance projects that focuses on stream and watershed system function. Recognizing the need to maintain a viable agronomic economy, recent efforts have focused on returning floodplains to modified channels and maximizing ecosystem services rather than restoration to pre-settlement, pre-agricultural conditions. Innovative alternative ditch designs are emerging as in-stream agricultural best management practices that, when coupled with landscape best management practices, have potential for meeting multiple management goals in low gradient, agriculture-dominated watersheds.

Here is a review of the function and modification of natural channels and agricultural drainage as agronomic, scientific, sociopolitical, and environmental cultures have evolved throughout history. We recognize the impacts of major water works projects for irrigation and navigation in the USA and throughout the world; however, this focuses specifically on development and management of agricultural drainage ditches that occurred in the Midwest USA, with special attention on activities that occurred in or affected the state of Ohio.

Numerous review papers have documented the impacts of agricultural drainage in the USA (Pavelis, 1987; Skaggs et al., 1994; Fausey et al., 1995; Zucker and Brown, 1998; Needleman et al., 2007; Blann et al., 2009 among others), but few have focused primarily on the role of drainage ditches, specifically two-stage ditches, as an agricultural best management practice. First, we review a brief history of water management and drainage from early civilization to the 19th century. Then, we discuss the modification of headwater and mid-order stream systems (0.5–10 mi^2) for agricultural land drainage. We highlight the technological, sociopolitical, and scientific advances that shaped stream and ditch management in the 19th and 20th centuries. Next, we discuss the scientific and cultural shift in how agricultural drainage ditches are viewed and managed at the local, state, and national level. Finally, we discuss an alternative design, the two-stage ditch, developed to return hydrologic function and geomorphic stability to agricultural ditches undergoing frequent maintenance activities and current research on the two-stage ditch design to be considered an agricultural best management practice and water quality conservation practice. We primarily focus on agricultural land drainage in the Midwestern USA because of its history of extensive land drainage activities, its economically important agricultural industry, its recent history of pioneering scientific and engineering work in aquatic biological indicators and alternative drainage channel designs, and its location as the headwaters for ecologically and economically important water resources such as the Mississippi River, Gulf of Mexico, and the Great Lakes.

Agricultural Drainage in Europe and Early America

The Greeks and Romans used clay tiles for supplying urban areas with potable water as early as the 4th century B.C. (Beauchamp, 1987). The Hohokam Indians are credited with developing the first irrigation system in North America (Beauchamp, 1987). The French are credited with being the first to use clay roofing tile for farm drainage purposes as early as the 14th century.

Around 1500 A.D., the first large-scale design for drainage was developed in England to prevent flooding and reclaim marshes and peat lands. A system of cylindrical clay pipe drains was discovered in France in a garden that dates back to 1620 (Beauchamp, 1987). In most countries of the temperate zone, 20%–35% of agricultural land had been developed with the use of drainage by the 18th century (Shirmohammadi et al., 1995). European settlers brought surface drainage technology with them to America, including the use of small open ditches to drain wet spots in fields and the cleaning out of small streams.

While sufficient to cultivate the land, these small-scale surface drainage systems did not lower the water table fast enough to drain the soil profile, and crop yields generally were low. Flooding after large rain events was common in areas with heavy, poorly drained soils. American farmers turned to subsurface-covered drains that emptied into open ditches as a drainage solution for small, individual farm plots. Farmers quickly realized how important the outlets to the open ditches were to effectively drain the land.

Many larger receiving ditches were dug at the edges of properties and existing streams were widened and deepened to accommodate subsurface tile flows. Early subsurface drains and open ditches were dug with shovels, followed by a combination of plowing and hand digging. Subsurface drains typically were placed 24–36 inches deep and two furrows wide (Skaggs et al., 1994). Open ditches were constructed, and natural streams were cleared and straightened, but very little was documented on the extent of these activities throughout early America or whether a common design was developed.

Early America's interest in drainage was not only for agricultural purposes but also to eradicate water-borne human diseases such as malaria and spotted fever. The first large-scale American drainage effort was documented in 1754 when the colony of South Carolina passed an act to drain the Cacaw Swamp. The federal government transferred land authority to the states. Subsequent state drainage laws established drainage districts that could create large drainage outlets beyond individual farm boundaries. At this time, water policy generally fell under the Common Enemy Doctrine. Surface water was regarded as a common enemy, which each property owner could fight off or control by any means without regard to rights or well-being of their neighbor (Callahan, 1979).

Very little was documented as it relates to natural stream and river channel modifications in Europe and early America because of land drainage, but forest clearing likely was extensive and had wide-ranging impacts to stream networks. It is widely accepted that most modern streams in the USA have little in common with those found prior to European settlement. Headwater stream channels in low gradient landscapes prior to settlement likely were either prairie wet meadows or were heavily shaded by riparian vegetation and contained large amounts of fallen wood (Petersen, 1992). Much of the land from western Ohio to the Mississippi River Valley was covered in dense beech, ash, and elm forests, prairie, and extensive tracts of swamp land

(Petersen, 1992). Stream waters were characteristically described as being exceptionally clear and free of silts and pollutants, often serving as a source of potable water for the early settlers. Additionally, beaver activity prior to the mid-1800s may have resulted in extensive impoundments throughout the stream network that would have made headwater streams wider, slower, and more mid-order in character (Minshall et al., 1985). As populations expanded and moved west during the 1800s, vegetative cover and beaver populations across the USA were dramatically reduced, which likely resulted in the earliest impacts on stream systems. Drainage laws, population pressure forcing settlers to move west in search of more agricultural land, and advances in technology in the 19th century resulted in a cascade of activities that would forever change stream networks in the North American landscape.

American Settlement of the Midwest

Technological advances in the 1800s resulted in rapid changes to farming practices of American settlers (Turner and Rabalais, 2003). John Johnston was the first person to lay drain tile in the USA on his farm in New York in 1835 (Klippart, 1861). Population pressure, a need for more suitable farmland, and completion of the Erie Canal and Ohio & Erie Canal pushed more settlements into the Ohio and Mississippi River valleys by the 1830s. The transportation industry, including the railroads and canals, facilitated westward expansion and land clearing (Turner and Rabalais, 2003). To enhance both commerce and national defense, Congress passed the first federal acts involving interstate commerce in 1824 that granted the Corps of Engineers authority to survey canal routes and "improve" the navigation of the Ohio and Mississippi rivers by removing sandbars, snags, and other obstacles (http://www.usace.army.mil/About.aspx, last checked December 1, 2021).

Passage of the Swamp Land Acts in 1849 and 1850 further encouraged settlers to move west. The Black Swamp, a forested wetland mainly located in the northwestern corner of Ohio and estimated to have been 120 miles long and 40 miles wide, covering nearly 11 counties, was a major barrier to travel and settlement within the Midwest (Brown and Stearns, 1991; Dahl and Allord, 1999). A series of public ditch laws were passed including the Ohio Ditch Law in 1859 that facilitated land clearing and drainage of swamp land to develop productive agricultural lands. The new ditch laws granted local officials the authority to design and construct drainage projects and assess local landowners for the cost of the projects. In Ohio, these laws are now known as the County Petition Ditch Law that is still in use today (Brown and Stearns, 1991).

The Black Swamp is the most extreme example of the impact of ditch laws on Ohio and the Midwest landscapes. Supported by drainage and levee districts at the local and state level to help cover the immense task of land reclamation, draining of the Black Swamp had begun in 1859 and been completely drained by 1885 (Fausey et al., 1995; Dahl and Allord, 1999; Turner and Rabalais, 2003). By 1884, Ohio had 20,000 miles of public ditches designed to drain 11 million acres of land.

As drainage laws evolved, so did national water authority throughout the 1800s. In direct opposition to the Common Enemy Doctrine, the rule of water drainage law, which later became known as Civil Law Rule, was increasingly recognized by state and federal courts. Civil Law Rule mandated that a downstream landowner must accept the surface water that naturally drained onto their land, and that the upstream landowner had no right to change the natural system of drainage to increase the burden on the landowner downstream (Callahan, 1979). The concept of riparian rights also was adopted by some states at this time. In Ohio, riparian rights mandated that a landowner had a right to use the water that passed over his land if it was transmitted by its natural channel to the downstream landowner (Callahan, 1979).

By the mid-19th century, land management practices shifted from land clearing to more intensive agricultural use that spurred entirely new industries and technological advances in agricultural production. The first patent for a chemical fertilizer was issued in 1849 in Baltimore, and phosphate fertilizer production began in South Carolina in the late 1800s (Turner and Rabalais, 2003). Clay tile manufacturing for subsurface drainage boomed, and by 1867 Ohio was leading the way producing 2,000 miles of drain tile per year (Skaggs et al., 1994; Fausey et al., 1995). Also, in Ohio, the steam-powered Buckeye Trencher No. 88 was built in 1892 that would become the model from which subsurface ditch trenching machines still are based on today (Beauchamp, 1987). The original company that patented the Buckeye Trencher was founded in Bowling Green in 1893, and grew to become the largest tile ditching and construction trenching company in the world (Yannopoulos et al., 2020).

Subsurface drainage technology was widely credited for land reclamation of the American Midwest; however, improved methods for constructing large-scale outlet ditches beyond the farm field for surface drainage probably were the most important factors in how quickly and efficiently subsurface drainage spread in the Midwest. Ditches and channels were first built, and stream channels were first excavated by hand using tools such as the ditching spade, round point shovel, and a wheel barrow. Horse horse-drawn plows and slip scrapes soon followed, but these small shallow ditches did not often provide adequate drainage needed in the heavy clay soils and swamp lands of the Midwest. Engineering designs for open ditches were not well documented at that time and it seemed that their sizing was left up to the discretion of the farmer or drainage district manager. Early on, drainage ditch sizing and design (Yannopoulos et al., 2020) were a function of available money and equipment to build the ditch. A report from Johnstone in 1834 indicated open ditch recommendations having a trapezoidal shape with a top width three times wider than the bottom width, side slopes stable enough to prevent falling in, and at a slope sufficient for the water to move obstructions but not injure the bottom. In a report to the Ohio Legislature, Klippart in 1861 suggested that open ditches serving as main drains be nearly seven times wider at the top than at the bottom, 4–6 ft deep, and have a side slope ratio of 1.5:1. He also suggested that the bottom of the ditch should be 12–18 inches below the outlet of any smaller drains. Another account from a drainage test farm in Illinois in 1908 indicated that their open ditches were 3–7 ft deep, 3 ft wide on the bottom, and had 1.5:1 ratio for side slopes.

By the late 1880s, a steam-powered floated dredge was invented that revolutionized open drainage ditch construction methods. Floating and land dredges became the most economical way to construct ditches, but the dragline excavator in the early 1900s proved to be the most universal allowing for various ditch sizes and wide berms along the ditch. By the 1930s, the crawler was commonly used because it could level ditch spoil banks and could construct v-shaped, w-shaped, and wide bottom flat ditches (Fausey et al., 1995).

The result of nearly a century of engineering and agricultural innovation was productive, well-drained, and extensively ditched agricultural landscapes that converted the swamp lands of the American Midwest into one of the most productive agricultural regions of the world. Yet, becoming the "breadbasket" of the world was not without consequences. The impact of settlement and land reclamation on the natural wetlands of the USA and the Midwest was immense. The impacts on natural streams and rivers, although less well documented, likely were equally far-reaching. Land drainage and cultivation was preceded by tree cutting and burning. Greeley in 1925 documented that the virgin forests of 1850 in the USA were largely remnants by 1920. Ohio's forests, for example, declined from 54% in 1853 to 18% in 1883 and were used mainly as fuel for the railroads (Steyaert and Knox, 2008).

Massive forest clearing on essentially all North American rivers in the 1800s and early 1900s, together with large-scale channelization efforts to facilitate agricultural drainage, likely had severely altered the pattern and complexity of the natural stream network (Minshall et al., 1985). Mid-sized streams that were once heavily braided or meandering systems became single, relatively straight channels. Channelization also altered the function of the riparian zones adjacent to streams. The forested or prairie wet meadow riparian zone probably produced a very different channel than that found in the agricultural landscape today. Braided channels, backwaters, and side-channel streams probably caused many mid-sized rivers to behave more like headwaters (Minshall et al., 1985). Beaver activity may have helped ameliorate the initial impacts of channelization, but expansion of American settlement and the fur trade resulted in near extirpation of beaver from the Midwest landscape.

While not well documented in the literature until the 20th century, we can surmise deleterious impacts to water quality and aquatic biota during this time. Depletion of the riparian canopy removed shading benefits to in-stream organisms and raised water temperatures. Populations of sensitive aquatic species likely were influenced by siltation and fine sediment delivery to streams. Loss of riparian vegetation along with channelization also made stream flows more variable and flashier as more water reached channels faster (Armitage et al., 2002). In one account from Champaign County, Illinois, which was the headwater region for six different rivers, an average of 95% of first-order streams had been channelized since the 1850s (Mattingly et al., 1993).

Technological Advances and Legislation Affecting Agricultural Drainage Systems in the 20th Century

The earliest reported signs of trouble on the landscape because of extensive drainage works occurred in the early 20th century. Naturalists and fishermen noted the decline of sport fishes and migratory hunting birds, which led to the first legislation aimed at restoring wetlands, the Migratory Hunting Bird Stamp Act of 1934. Farmers observed extensive soil loss from the landscape. Publications began to surface documenting widespread soil losses in the Midwest and the Dustbowl of 1933–1934 led to the formation of the USDA Soil Erosion Service, which was renamed the Soil Conservation Service (SCS) in 1935. Many instances of channelized streams widening and deepening that resulted in sediments burying tile drains and filling in ditches downstream were reported. Recognition of the threat of soil erosion to farm management practices was widely acknowledged by the 1930s.

Local water laws had evolved yet again because of increasing landowner conflicts stemming from flooding and erosion control measures on the landscape. The rule of "reasonable use" was developed as an alternative between the Civil Law Rule and the Common Enemy Doctrine. The Reasonable Use Rule stipulated that a landowner can legally make reasonable use of his land, even if it alters the flow of water and causes harm to others. However, if the alteration is unreasonable, a landowner is liable for damages caused as a result (Callahan, 1979). Many states and local jurisdictions passed laws to try to prevent devastating floods. In Ohio, the Conservancy Act was passed in 1914 after the largest flood to date occurred on the Muskingum River. The Conservancy Act enables landowners or communities to establish conservancy districts to solve water management issues, including flood reduction and protection, and provide other services such as conserving and developing water supplies (http://www.foresthistory.org, last checked December 1, 2021) and treating wastewater and providing recreational opportunities.

Under Ohio Drainage Law, conservancy districts had special powers to regulate use of water in streams to the extent that the flow was increased by improvements made by the district (Sections 6101.24 and 6119.06; Callahan, 1979). In 1933, Ohio established the Muskingum Conservancy District to assist the Army Corps of Engineers with a large-scale flood control and water conservation project that resulted in construction of 14 reservoir and dam systems along the Muskingum River. Technological advances and new legislation in the 1940s brought a resurgence of drainage and flood control channel construction to the Midwest by local drainage districts, the SCS, and the Army Corps of Engineers. In 1941, drainage and irrigation work was approved as a conservation practice by the USDA. The Flood Control Act of 1944 together with the authorization of USDA to plan and construct drainage outlet channels in cooperation with local and state governments in 1954 led to a new way to manage drainage ditches. In Ohio, county commissioners and county engineers noticed that drainage ditches were being constructed multiple times to remove sediment or other debris that had accumulated over time resulting high costs to landowners and local governments. To address this, county commissioners were granted authority to establish a fund for the county that would be used to "maintain" ditches and tiles installed through the petition ditch laws. At first, maintenance programs were optional. In 1957, ditch maintenance programs were mandated by the law (Brown and Stearns, 1991).

Technological advances expanded the efficiency of drainage, particularly in the Midwest, and led to the "Green Revolution" in the 1950s and 1960s (Armitage et al., 2002). By 1960, corrugated plastic tubing was on the market and quickly replacing clay and cement as a cheaper and longer-lasting drain tile. The first laser grade control systems that enabled the precise depth and grade of subsurface drains by regulating trenching and plow-type drainage machines were demonstrated in 1968 at the Ohio State Farm Science Review (Fouss and Fausey, 2007). Also, at this time, the USDA released the first federal guidance on the design of open channels, Technical Release-25, requiring open channels to be trapezoidal and designed to convey water discharges of all magnitudes from base flow through flood flow without significant damage to the channel or to fish habitat (USDA-NRCS, 2009).

Concurrent to federal legislation that promoted drainage and ditching improvements, the Surgeon General warned of widespread threats to drinking water and public health after it was reported that over 3,500 communities pumped 2.5 billion tons of raw sewage into streams, lakes, and coastal waters every day in the USA. Historically, water regulation was left up to the states; however, Congress passed the Federal Water Pollution Control Act in 1948 in direct response to the Surgeon General's warning. The Act preserved states' control of their waterways by only regulating interstate waters, established federal technical services and grants to state and interstate government bodies and, ultimately, did very little to limit water pollution. Significant federal legislation protecting US waters did not arrive for another 24 years.

Scientific and Cultural Shifts in the View of Stream Systems and Land Drainage

The early to mid-20th century might be considered a period of enlightenment for stream and river research. A better understanding of stream and river systems by the scientific and regulating communities, combined with a largely grass-roots-led effort against air and water pollution, resulted in significant changes in cultural views on agricultural drainage by the end of the 20th century.

Stream and River Morphology Research

Many geologists had accepted for some time the idea that climate and geology were the ultimate determinants of river morphology through their effect on discharge and sediment load. As early as 1902, Davis defined a "graded" stream as the condition of balance between erosion and deposition attained by mature rivers. Lindley, in 1919, defined the regime concept as the dimensions, width, depth, and gradient, of a channel to carry a given supply (of water) loaded with a given silt that were all fixed by nature. It was postulated that the floodplain of the highest quality streams experienced frequent overflow with a main goal of absorbing and dissipating energy from high flows. The main channel, in turn, adjusted its dimensions and pattern to efficiently move sediment and water during low flows. Working together, the channel and its floodplain balanced sediment and water transport, storage, and supply, a process that became known as the dynamic equilibrium concept (Leopold and Maddock, 1953; Lane, 1955a, b; Wolman and Miller, 1960; Langbein, 1964; Leopold, 1964).

In natural systems, dimensions of the main channel were associated with a range of flows referred to as the bankfull, effective, or dominant discharges. Bankfull discharge originally was described as the amount of water that filled up the main channel to a depth just before it spilled out onto the floodplain (Leopold and Wolman, 1956; Wolman and Miller, 1960). Later, it was described as a range of flows that was most effective in forming a channel, floodplains, banks, and bars. Effective discharge, in turn, was based on sediment transport concepts and was described as the flow that transports the largest cumulative sediment load over time (Wolman and Miller, 1960). Benson and Thomas (1966) were the first to define dominant discharge as the discharge that over a long time period transports the most sediment; and further describing that it was much less than bankfull-stage discharge, somewhere between the mean annual flow and the effective discharge.

Collectively, bankfull and effective discharges were considered channel-forming discharges (Leopold et al., 1964). Hydraulic geometry relationships between channel width, mean depth, mean velocity, slope, and friction were developed that enabled researchers to predict characteristics of stream channels and classify them based on form (Leopold, 1953; Leopold and Langbein, 1962; Scheidegger, 1968; Hey, 1978; Ferguson, 1981; among others).

Regional relationships for bankfull stream characteristics that were based on drainage area, referred to as regional curves, also were developed to verify field determinations of bankfull discharge and measured stream channel characteristics. Regional curves expressed the mathematical relationships between contributing drainage area and channel dimensions corresponding to the bankfull discharge (Sherwood and Huitger, 2005; Witter, 2011). Work referenced above, and by others during this time, served to make a seemingly wild, unpredictable river system predictable and, hence, manageable.

Effects of Channelization on Hydrology and Hydraulics

By the mid-1970s, it was widely accepted that fluvial systems were connected and that changes to one part of the system would have widespread effects throughout the entire system (Brookes, 1985; Sherwood and Huitger, 2005). While it had long been known that trapezoidal-shaped channels successfully drained the soil profile and efficiently moved water downstream, they were often constructed larger than what would be formed by natural fluvial processes and disconnected the channel from its floodplain. The primary reason that drainage ditches were constructed so large was to accommodate the depth of subsurface tile drains. Subsurface tile drains typically were installed 24–36 inches below the surface. To provide sufficient freeboard for drainage, ditches often were dug an additional 12–24 inches below the tile outlets resulting in drainage channels that were a minimum of 5–6 ft deep.

From a geomorphic perspective, trapezoidal channels were too large to transport small flows and provided no floodplain to dissipate the energy of large flows and, therefore, an imbalance was created. In response to this imbalance, natural fluvial processes worked to create a small main channel by building a floodplain or bench within the confines of the ditch. However, this increased friction along with sometimes buried drain tiles, and increased hydraulic residence time, which backed up water into drain pipes further saturating the soil profile and limiting crop yield (Blann et al., 2009). County ditch maintenance programs would clean out vegetation and sediment deposits to restore a trapezoidal shape and maintain drainage capacity. Channel maintenance, which often included straightening, smoothing, and deepening stream channels led to an immediate increase in bed slope and carrying capacity of the channels. Smoothing, or removal of woody vegetation on channel banks and sediments from channel beds, reduced friction that increased flow velocities and improved drainage capacity.

Unfortunately, in many cases, higher flow velocities cause instability on channel beds and on steep and un-vegetated banks causing channels to further deepen and widen (Emerson, 1971; Nunnally, 1978; Simon and Rinaldi, 2000). The newly formed and still-oversized channel failed to transport sediment, which resulted in aggradation that flattened the channel slope and increased channel hydraulic residence; the fluvial cycle of imbalance began again. Deviation from natural fluvial processes in drainage ditches drove the need for constant and frequent "improvement" to maintain drainage capacity.

Routine maintenance activities typically occur every 1–20 years. The ecological and socioeconomic impact of drainage ditch clean-out is significant. In Minnesota, $12 million per year were spent on drainage ditch maintenance, and in Ohio, it was estimated that an average of $450/mile was spent annually on open-channel ditch maintenance (Hansen et al., 2006). Ohio has nearly 4,000 miles of open channels that are routinely maintained for agricultural drainage.

Stream Ecology and Water Quality Research

A holistic view of streams as ecosystems did not begin until the late 1950s. Most research at this time focused on fish and macroinvertebrates in forested streams. Ross, in 1963, was among the first to note similarities among stream communities over broad geographic areas and was the first to recognize the importance of the riparian-channel interaction. The importance of streams as ecosystems appeared in a seminal paper positing that stream systems should be part of the study of landscape ecology (Hynes, 1975). Work on nutrient transport in streams was occurring at this time that, collectively, became known as the nutrient spiraling concept (Newbold et al., 1981, 1982a, b). Recognition that the action of flowing water, bed form and stability, and organic matter storage and transport varied with climate has begun to emerge in ecological literature that spurred a re-evaluation of the idea that streams are more than conduits that simply transported materials and, hence, to a greater appreciation of the metabolic and retention role of stream systems (Hynes, 1975; Minshall et al., 1985). By the early 1980s, the prevalent scientific view of stream ecosystems was that of the River Continuum Concept (Vannote, 1985). Pioneering work in this area emphasized conditions in relatively undisturbed streams draining forests, but the broader implication of these studies, which was well supported in the literature, indicated that the floodplain was crucial to the function of stream ecosystem processes

(Cummins et al., 1984; among others). In Ohio, monitoring and research on fish, macroinvertebrates, in-stream habitat as indicators of stream impairments based on deviations from natural conditions led to the development of bioassessment indices (i.e., Index of Biotic Integrity, Invertebrate Community Index, and Qualitative Habitat Evaluation Index) and threshold criteria for stream health (Karr and Schlosser, 1978). Ohio became the first state to incorporate bioassessment criteria into state Water Quality Standards and served as a model for adaptation and adoption of bioassessment indices by other states. Until this time, stream health was determined from sampling water chemistry constituents.

Effects of Channelization on Stream Biota and Water Quality

Blann et al. (2009) provide a comprehensive review on the impacts of drainage on aquatic organisms. Channelization resulting in a trapezoidal-shaped channel had an immediate effect on aquatic biota by producing a channel devoid of habitat complexity. Straightening channels resulted in loss of channel length and, therefore, in-stream habitat. For example, Hansen, in 1971, reported a 54% total reduction in the length of a lowland reach of the Little Sioux River in Iowa. Increased flow velocities and the removal of vegetation that de-stabilized the channel bed and banks resulted in elevated concentrations of suspended sediments in the water column and increased sediment loads to rivers, which buried substrates that were key spawning area for fish (Keller, 1976; Karr and Schlosser, 1978). Increases in flow may also have had direct ecological implications since many aquatic organisms had specific water velocity requirements (Brooker, 1985).

Maintenance activities to remove benches or bars that had formed also removed pool-riffle sequences that were critical feeding and breeding areas (Brooker, 1985; Jenkins and Keeley, 2010). Channelization also altered energy dynamics in channels and trophic interactions. Removal of shading vegetation from banks and changes in water depth affected in-stream temperatures. Since most streams receive their primary source of energy from allochthonous organic matter, loss of bank-side vegetation may have also substantially reduced energy flow in the aquatic system (Cummins, 1974, 1984).

Skaggs et al. (1994) provided a review of the impact of drainage on water quality that began to appear in the literature in the 1970s and found conflicting results on the impacts of land clearing, channelization of streams, and subsurface drainage improvements on peak flows, runoff, sedimentation, and nitrogen and phosphorus levels in stream systems. Impacts of agricultural drainage were as much related to design and implementation as they were to cultural practices and "good-housekeeping." Early studies stressed the importance of factors such as soils, climate, drainage design, and the location of drainage improvements in the effects of land drainage and effects could be minimized if appropriate measures were taken upon land development such as reseeding drainage ditches immediately after excavation.

Concurrently, by the late 1960s, the channelization work of the SCS and the Army Corps of Engineers became so controversial that Congress commissioned an independent national survey of the environmental effects of the channelization projects (Schoof, 1980). The impact of land drainage and ditching activities through sedimentation effects on aquatic biodiversity did not go unnoticed at the state and federal level, causing an added impetus for protection of agricultural soils. As a result, the implementation of various conservation tillage practices (e.g., no-till, mulch-till) began in the 1970s and expanded so rapidly that about 40% of cropped land in the USA now is farmed with some type of conservation tillage. The benefits of conservation tillage were substantial in reducing soil erosion and there were some positive biological responses to this effort.

Federal Legislation and the Culture of Agricultural Drainage

More than 34,000 miles of waterways were modified by the Army Corps of Engineers and the SCS from 1940 to 1970 for drainage and flood protection. Most of the work occurred in five Midwestern states: Illinois, Indiana, North Dakota, Ohio, and Kansas. At the time, channelization was considered "channel improvement" or "watershed management" with clearly demonstrated societal benefits (Schoof, 1980). The primary objections to stream channelization from the environmental community were the reduction in fishery resources, the destruction of wildlife habitat due to timber removal, reduced aesthetics, and increased flooding and sedimentation downstream. Lack of federal regulation and poor state and local regulations resulted in continued water pollution throughout the USA. Two independent events occurred that finally focused the attention of the federal government on the widespread problems of environmental degradation. The first was the publication of Rachel Carson's Silent Spring in 1962, which brought national awareness to the effects of pesticide use on birds and on the environment. The second was the burning of the heavily polluted Cuyahoga River in Cleveland, Ohio, in 1969. Reporters from *Time* magazine witnessed it and reported in their national publication that the river "oozed rather than flowed." Growing national environmental concerns catalyzed by these events led to the federal government forming the Environmental Protection Agency in 1970 and passing the National Environmental Policy Act in 1969 and the Federal Clean Water Act in 1972.

Currently, the USA was facing a crop surplus; air quality and water quality were now as important as or more so than agricultural quality. For example, wetlands were no longer considered inaccessible, unprofitable "wetlands" as much as they were unique and valuable ecosystems. The Food Security Act of 1985 exemplified a national shift toward environmentalism by eliminating indirect federal incentives to convert wetlands into cropland under the Swampbuster provision and by creating conservation programs such as the Conservation Reserve Program. Shrimohammadi reported that most states in the continental USA prepared agricultural water quality plans in the 1970s that recommended the evaluation of certain best management practices (BMPs) in agricultural watersheds with respect to their impact on water quality. Agricultural drainage was not considered a BMP in any of these plans. Also, by this time the effects of stream channelization on water quality, benthic invertebrate communities, fisheries resources, and the recreational value of streams were well documented (Karr and Schlosser, 1978). Publications and technical guidance from SCS were (http://www.ohiohistorycentral.org, last checked December 1, 2021) updated with input from environmental interests. Channel improvements that were funded or supported by federal dollars now required environmental impact statements. Changes in the way humans related to the local and regional ecosystem combined with the cultural legacy of agricultural drainage led to many conflicts between agriculture and environmental groups throughout the 20th century.

The image of drainage had changed dramatically over the second half of the 20th century, but the integration of drainage into American culture was clear. Urban and Rhoads (2003) provided an account of impacts of drainage on American culture, especially in the Midwest, which is summarized. Natural streams that had been straightened, deepened, and widened over the last 100 years were now often identified on topographic maps as ditches. Constructed ditches were no longer considered channels that had been dredged into existence but now were included in consideration with the channelized natural streams. Most channel modifications occurred on or adjacent to farms, implicating the agricultural community in effecting changes in local stream systems (Urban and Rhoads, 2003). Increased knowledge and understanding of hydrologic and hydraulic processes in stream and river systems made it apparent that agricultural drainage not only affected regional ecosystems but also the underlying physical environment and the geomorphic processes shaping it over time. By the mid-1980s there was deep concern about the legacy effects of disruptions to the biological communities of channelized stream systems and unabated water pollution. Restoring wetlands, stream and river management, and protection of coastal waters became a national cultural theme. What was a theory of stream restoration less than a decade earlier surged to become a billion-dollar industry by the end of the 20th century to deal with these impacts.

Restoration, Rehabilitation, and Naturalization of Drainage Ditches

Persistent Nutrient Enrichment Becomes a National Priority

Increased nitrate and phosphorus levels were detected in the Gulf of Mexico in the 1970s and in Lake Erie as early as the 1960s (Beeton, 1961; Turner and Rabalais, 2003). Algal blooms because of eutrophication were causing major ecological problems and dead zones in these economically and ecologically valuable water bodies (Mitsch et al., 2001; Rabalais et al., 2010; Turner and Rabalais, 2003). The USA and Canada signed the Great Lakes Water Quality Agreement in 1972 that resulted in a 60% reduction in phosphorus loading to Lake Erie and no reports of algal blooms by the early 1980s.

In contrast, algal blooms in the Gulf of Mexico had resulted in an extensive hypoxic zone related to fertilizer application on agricultural fields in the states within the Mississippi River by the mid-1980s (Turner and Rabalais, 2003). The primary focus of drainage management that continued into the 21st century was on nutrient and sediment export from agricultural fields (Turner and Rabalais, 2003). Nutrient enrichment and the resulting hypoxia in the Gulf of Mexico and the Great Lakes appeared to be on the decline through the latter half of the 1980s and into the early 1990s; however, harmful algal blooms increased again toward the end of the 1990s. Harmful algal blooms caused an estimated cost of $2.2 billion in losses to public health, commercial fishing, tourism, property values, and management in the USA (Dodds et al., 2009). In response, the federal government passed the Harmful Algal Bloom and Hypoxia Research and Control Act of 1998, which mandated the National Oceanic and Atmospheric Association to advance the scientific understanding and to develop programs for research into methods of prevention, control, and mitigation of harmful algal blooms. In response to the 1992 National Water Quality Inventory, which found that 56% of the stream miles surveyed were not meeting their designated use, and 44% of the remaining stream miles surveyed had sediment and nutrient impairments, 15 federal agencies collaborated to publish the *Federal Stream Corridor Restoration Handbook* (NEH-653).

Despite extensive research efforts and the implementation of landscape BMPs to minimize nutrient losses from agricultural lands, water quality problems associated with anthropogenic eutrophication persisted (Carpenter et al., 1998). In 2003, toxic Microcystis was detected in algal blooms in western Lake Erie that forced beach closures and impacted fishing and recreation in both Ohio and Michigan. The Hypoxia Research and Control Act of 1998 was amended in 2004 and harmful algal blooms became a high priority national issue.

Managing Agricultural Drainage Ditches to Improve Water Quality

It was estimated that up to 80% or more of the entire stream network in some Midwest states consisted of streams and drainage ditches channelized and modified to a trapezoidal geometry for agricultural purposes (Blann et al., 2009). Considerable research documented the role of drainage ditches as conduits of field pollutants and the effects of routine ditch maintenance practices such as dredging in disrupting the natural buffering ability of ditches (Pappas and Smith, 2007; Smith and Pappas, 2007). A growing body of research suggests the potential for using vegetated open ditches as the best management practices in mitigating potential agricultural contaminants. Strock et al. (2007) identified several landscape and in-stream practices to reduce the off-site transport of pollutants in drainage water. There had been considerable debate on whether anthropogenic eutrophication problems could best be resolved with landscape BMPs, in-stream BMPs, or some combination in a systems approach. Some had called for a moratorium on all drainage works. The debate very much reflected cultural divisions in the Midwest that may prove to be critical to the future of drainage in the Midwest as water quality problems persist and as management solutions are developed.

Herein, we turn our attention to current research trends in agricultural drainage ditches in the Midwest that have led to the design and management of open ditches as in-stream best management practices.

Alternative Agricultural Drainage Ditch Approaches and Design

Geomorphology and Fluvial Processes in Agricultural Drainage Ditches

Significant research on geomorphology concepts was abundant, but attempts to apply these concepts to engineering design only occurred in earnest within the last three decades. Hydraulic geometry relationships relied on channel-forming discharges that were not easily measurable and, therefore, were based on calculating bankfull discharges from stream geomorphology measurements or calculating effective discharges from measured or estimated sediment and stream flow data (Ward et al., 2008). Established methods on field measurement techniques of stream geomorphology were widely available (i.e., Harrelson et al., 1994). However, estimates of bankfull discharges relied on accurately identifying and measuring bankfull dimensions in the field. Most published studies on bankfull or effective discharges had been based on natural systems in the western USA that had steeper gradients, were not underlain by subsurface drainage, and were in less modified landscapes than systems in the Midwest (Powell et al., 2007a). Johnson and Heil (1996) questioned the use of geomorphic bankfull relationships on unstable channels like agricultural ditches; however, a few studies had shown that, if left unmaintained, disturbed headwater systems developed stable geomorphic features, such as an inset channel, bars, and vegetated benches (Rhoads and Monahan, 1997; Kuhnle et al., 1999; Rhoads et al., 1999; Frothingham et al., 2002; Landwehr and Rhoads, 2003; Jayakaran et al., 2005; Ward et al., 2015).

Jayakaran et al. (2005), using a logistic regression model, found that stable bench formation could be predicted by the width of the ditch and drainage area in northwest Ohio ditches. Drainage area was a key factor in determining the dimensions a stream will create for itself (Leopold et al., 2005), while ditch width was an anthropogenic constraint imposed on the channel. In a ditch that was too wide, the stream adjusted to a narrower width by building benches (Landwehr and Rhoads, 2003). Stability of a bench could be modeled as a function of drainage area and ditch width and could be sized by traditional engineering designs to accommodate extreme events (Jayakaran et al., 2005). Channel-forming discharge concepts were suitable for engineering applications in large rivers in the Midwest, but the recurrence intervals of channel-forming discharges were less in modified low gradient watersheds than the often reported 1.5- to 2.0-year published values used for engineering design (Simon et al., 2004; Jayakaran et al., 2005; Leopold et al. 2005; Powell et al., 2007a).

Andrew Ward (personal communication) speculated that, in ditch systems having subsurface drainage, the inset channel was formed by channel-forming discharges associated with high tile flows, which was the reason benches formed at the bottom of ditches rather than near the top. In an unpublished study, he analyzed this concept by considering typical design standards for subsurface drainage (ASABE, 2008) and fitting regional curves to inset channel data from 18 sites located in tributaries to the Portage River, OH, measured by Jayakaran et al. (2005). Regional curves for the Portage River watershed were taken from Powell et al. (2007a), who interpreted measured data for one site differently than Jayakaran et al. (2005). The weak correlation between drainage area and measured bankfull dimensions was a result of some sites having unstable benches or no dominant fluvial features (Jayakaran et al., 2005). The analysis also included regional curves for the St Joseph River watershed, located in northwest Ohio, Indiana, and Michigan (Powell et al., 2007b) and the USGS Region A curves for natural streams in Ohio (Sherwood and Huitger, 2005).

The following two theoretical tile-drained scenarios were considered: Channel A that drains 320 acres of cropland in a 640-acre watershed and has a subsurface drainage coefficient of 0.5 inches per day that produces a mean daily discharge of 6.7 cubic feet per second (cfs) at the tile outlet; and Channel B that drains 480 acres of cropland in a 640-acre watershed, and has a subsurface drainage coefficient of 0.75 inches per day that produces a discharge of 15.1 cfs at the tile outlet. To represent a larger drainage system, discharges at the outlet were multiplied by ten resulting in 67 cfs in Channel A and 151 cfs in Channel B. Using Manning's equation and the continuity equation to calculate bankfull discharge associated with channel dimensions, Ward found, at the lower threshold in the Portage River and St. Joseph River watersheds, the bankfull discharge for the inset channel in an agricultural ditch was like the design discharge from subsurface drainage systems. In contrast, the bankfull discharge estimated by the USGS regional curves for natural streams (Sherwood and Huitger, 2005) was many times larger than the bankfull discharge for the inset channel. Results of this analysis confirmed that the inset channel was formed by high tile flows that conformed to channel-forming processes and that the dimensions of the inset channel could be predicted using appropriate regional curve relationships.

Jayakaran et al. (2005) provided a comprehensive review of research on fluvial processes and management of agricultural ditches, and we summarize some of the key findings pertaining to Midwest agricultural channels and the development of the two-stage ditch concept. Jayakaran et al. (2005) reported that naturally formed benches in agricultural ditches in northwest Ohio evolved by vertical accretion and in similar ways to how floodplains form in natural systems. Results were consistent with research findings on ditches in Illinois (Landwehr and Rhoads, 2003). A hydrologic study of agricultural ditches containing low benches indicated that the benches were flooded between 10 and 60 days annually, and flooding events were associated with discharges equivalent to 30% of the 2-year discharge (Kallio, 2010). Additionally, simulation studies suggested that nitrate-N removal may be as great as 20% if the floodplain area is equivalent to at least 1% of the watershed area (Kallio, 2010). Fry et al. (2012) found that overbank flow was 1%–4% of the total volume of flow in a small agricultural watershed with attached floodplains. Rhoads and Massey reported that lateral migration of the inset channel was minimal in an Illinois ditch that had naturally formed with well-vegetated benches.

Aquatic Diversity and Water Quality in Agricultural Drainage Ditches

Heavily managed agricultural watersheds typically had been viewed as devoid of viable populations of aquatic life and supported little, if any, ecological function (Crail et al., 2011). Ditches largely were ignored as contributors to aquatic biodiversity. Recent work has resulted in increasing awareness that highly modified agricultural watersheds could support diverse populations of aquatic biota. As early as the mid-1970s, Keller (1976) postulated that establishing a pilot channel in trapezoidal channels and building riffle-pool sequences in the channel could vastly improve aquatic communities. Schlosser in 1995 found that sensitive macroinvertebrate communities in Michigan ditches were most affected by substrate quality and composition. Evaluating fish communities in Illinois streams and developed a framework for rehabilitating impaired channels to achieve best potential ecological function. Ditches were considered a high-quality habitat for certain frog species in habitat-limited agricultural watersheds in central Iowa (Rustigian et al., 2003). Work by Lyons (2000) and Rhoads et al. established a new way of thinking about riparian vegetation and the contribution of grasses to habitat heterogeneity in modified headwater streams. Studies in Ohio, Indiana, and Michigan concluded that although not as complex as natural stream systems, in-stream habitat was an important determinant of fish and macroinvertebrate community structure in modified agricultural streams (Smiley et al., 2008; D'Ambrosio et al., 2009; Crail et al., 2011). Rhoads and Massey suggested that leaving grassed benches in ditches could provide improved habitat for aquatic organisms. Leslie et al. (2012) have suggested that recent interest in the management of drainage ditches to improve water quality may provide the potential to improve habitat for aquatic biota; and might consider tradeoffs between the benefits of ditches as a source of biodiversity and as a tool for improving water quality. Impacts of agricultural drainage on water quality in the 21st century currently revolve around two main activities: drainage improvements on land already used for agriculture and conversion of undrained lands to agriculture; the former being the most critical impact to hydrology and water quality (Skaggs et al., 1994; Blann et al., 2009). Subsurface drainage systems have a useful life of 20–40 years (Skaggs et al., 1994). To sustain productivity of drained lands, major renovation or replacement will be necessary that may lead to greater intensity of subsurface drainage (Skaggs et al., 1994). Studies on a wide range of soils, crops, and site conditions have shown that increasing subsurface drainage intensity on agricultural lands may have both positive and negative impacts on hydrology and water quality. For example, conversion of undrained land to subsurface drainage generally results in increased peak flows and flashy hydrographs (Skaggs et al., 1994). Conversely, the improvement of subsurface drainage on land that was already drained could reduce runoff and peak flow rates (Skaggs et al., 1994). Strategies such as controlled drainage and sub-irrigation are examples of water table management practices that have the potential to both substantially improve agricultural productivity and reduce environmental impacts (Fouss and Fausey, 2007; Strock et al., 2007). More extensive reviews on improved management of subsurface drainage are provided by Skaggs et al. (1994) and Blann et al. (2009).

Ecological and socioeconomic costs attributed to routine ditch maintenance in agricultural watersheds has led researchers to question the cultural practice of maintenance on ditches and contributed to a collective rethinking of ditch management strategies in the Midwest. Traditional ditch design and maintenance short-circuited hydrologic and nutrient processing functions of stream channels. Ditches, once viewed as primary nutrient and sediment conduits, now play a key role as buffers between the landscape and valuable downstream receiving systems. Agricultural streams transport most nitrates during high flow events. Retaining vegetated benches in ditches increases the surface area of ditches and retention time during high flows. In Ohio and Indiana, denitrification rates were greater in sediments on naturally formed benches in ditches than in sediments from side slopes of trapezoidal ditches and having benches in ditches did not reduce in-stream denitrification rates (Roley and Tank, 2012). Managing flow regimes in ditches can reduce nitrogen and moderate downstream phosphorus losses through sorption capacities (Needelman et al., 2007; Strock et al., 2007). Managing floodplains of ditch systems, either in channel (Powell et al., 2007a, b) or adjacent to the channel can be effective in reducing sediment export losses from agricultural watersheds.

Managing Floodplains in Agricultural Ditches: The Two-Stage Ditch Approach

The two-stage ditch is a floodplain establishment design that results in a more self-sustaining agricultural drainage system based on the principles of fluvial geomorphology that will reduce or eliminate the need for traditional ditch clean-out activities (Jayakaran et al., 2005; Powell et al., 2007a, b). The approach for designing two-stage systems consisted of: (1) an inset channel to convey the bankfull discharge, (2) a floodplain for the inset channel, and (3) sufficient capacity above the benches to reduce the likelihood that flow will overtop the ditch banks and flood surrounding crop land (Powell et al., 2007a, b; Kallio, 2010). In theory, the result is an inset channel sized by channel-forming processes that are in a stable, quasi-equilibrium state. Two-stage channels are considered stable if they are neither aggrading nor degrading based on geomorphic theory (Lane, 1955a, b; Leopold, 1994) and should require little or no maintenance to maintain conveyance capacity and drainage function. The two-stage ditch approach does not construct or alter the existing inset channel. Instead, the benches are "pulled back" to a width that is a multiple of the inset channel bankfull width (or wide enough to accommodate small machinery) and at an elevation that corresponds to the regional curve predicted inset channel bankfull depth. In some cases, the design predicted from regional curves is adjusted to reflect the existing geometry of the inset channel and benches that nature has already formed. The ditch banks are sloped to a stable angle, often at a 2:1 or 3:1 horizontal to vertical angle. If present, vegetation growing along the edge of the channel is left intact during construction. Major benefits of this approach include maximizing stability until vegetation can become established on the newly constructed bench surface and preserving local ecology that may be present.

The first two-stage ditch designed using engineering principles and channel-forming concepts was constructed in Wood County, Ohio, in 2002 (Powell et al., 2007a, b). The primary goal of the project was to increase the capacity of the under-sized ditch and reduce flooding of adjacent fields after rain events. The project was considered a prototype for future two-stage channels. It led to development of a nine-step procedure outlining the design of two-stage ditches (Powell et al., 2007a). Powell et al. (2007b) provided a case study review of the first 8 two-stage ditch projects in Ohio, Indiana, and Michigan designed or constructed using the nine-step procedure. Ward et al. (2008) investigated floodplain ratios in constructed two-stage channels that would maximize bench stability, provide floodplain benefits, and not promote lateral migration of the inset channel within the ditch. Results from that study suggested design guidance that benches be constructed at a width of three to five times the bankfull inset channel width.

The two-stage ditch approach originally developed in Ohio as an alternative to traditional ditch maintenance for the purpose of increasing ditch stability, reducing bank erosion, and reducing flooding into adjacent fields. Work by Kallio (2010) and D'Ambrosio et al. (2012) evaluated the evolution of two-stage channels in Ohio, Indiana, and Michigan designed and constructed using the nine-step procedure outlined in Powell et al. (2007a) to determine where geomorphic changes were occurring over time (i.e., scour on ditch side slopes, scour or deposition on constructed benches, or aggradation or degradation of the inset channel), if they were maintaining drainage capacity, and what maintenance had been needed since construction. Preliminary findings of a weight-of-evidence approach presented by D'Ambrosio et al. (2012) suggest that two-stage ditches 3–11 years after construction have experienced small adjustments to their dimensions over time, but have remained stable and maintained both overall ditch capacity and a stable inset channel (i.e., have not aggraded or degraded). None of the ditches have required traditional maintenance since construction. Additionally, all the ditches constructed with the goal to improve bank stability and/or reduce flooding into adjacent fields have successfully achieved these goals.

Ongoing research on constructed two-stage ditches in the tri-state region indicates that improved soil-water-vegetation interactions on the benches may have implications for water quality and ecological benefits (Roley and Tank, 2012). To our knowledge, the majority of water quality research in constructed two-stage ditches has been led by Dr. Jennifer Tank and her laboratory at the University of Notre Dame. The findings of their work are summarized as follows. Multiple years of monitoring at 9 two-stage ditches suggest that the two-stage ditch reduces stream water turbidity, and that turbidity is related to total suspended sediment. Therefore, the two-stage ditch has reduced total suspended solids presumably by stabilizing the ditch if it was previously unstable, slowing water velocities, and promoting deposition on the benches. This effect seemed to persist in ditches up to at least 10 years old.

Inorganic nutrients (dissolved nitrate (NO_3^-), ammonium (NH_4^+), and soluble and particulate phosphorus (P) appear to be reduced in the two-stage ditch relative to upstream treatment controls that were evaluated using a before-after-control-impact (BACI) study design. Unfortunately for dissolved nitrate, the loads were so high (up to >5 mg/l) and the treatment reaches were so small relative to the contributing watershed that a significant load reduction was not measured despite a significant increase in denitrification potential along the two-stage reach relative to the upstream conventional control ditch. Increased denitrification is due to the presence of the floodplain benches. Furthermore, older floodplains tended to have higher denitrification rates as the floodplains aged and increased in organic matter. Despite increased denitrification, either the two-stage ditch practice would need to be implemented on a broader scale or dissolved nitrate loading would have to decrease to realized significant improvements in downstream water quality.

There was a significant decrease in soluble reactive phosphorus at two of four streams that were studied. One stream that exhibited an increase in soluble reactive phosphorus or SRP levels was attributed to substantial manure application on adjacent fields during the monitoring period which likely masked the effect, if any, of the two-stage ditch. A relationship between turbidity and SRP has been established which suggests that if turbidity is reduced, so too will SRP, and it may provide a means to inexpensively monitor SRP levels in the future.

In-stream habitat appeared to improve from altered channel hydraulics due to two-stage implementation which concentrated low flows and allowed the energy of larger flood flows to be reduced as they spread across the floodplain benches. Stream bottom sediments appeared to become coarser as fine silts were flushed and likely deposited on benches whereas coarser materials were exposed and maintained on the channel bed. Aquatic macroinvertebrates, mussels, and fishes tend to benefit from coarser substrates.

Finally, the two-stage ditch practice appears to be an economical practice to reduce nitrate-nitrogen pollution (Roley and Tank, 2012). In an analysis of two-stage ditch effectiveness relative to wetlands and cover crops across a range of time horizons and interest rates the two-stage ditch (over a time horizon of 50 years) reduced nitrate-nitrogen at a cost of \$1.07–\$1.44 per kg N. Compared to the estimated cost of excess N to society, \$63–\$66 (Dodds et al., 2009; Birch et al., 2011; Compton et al., 2011), the practice appears to be viable pollution abatement technology.

Conclusions

Successful implementation of the two-stage ditch concept in Ohio, Indiana, and Michigan led to its incorporation in Part 654 of the *Stream Restoration Design National Engineering Handbook* (USDA-NRCS, 2009). Additionally, the two-stage ditch is an approved best management practice for Indiana and Ohio's Environmental Quality Incentive Program and has seen application in other upper Midwest and Southern states with success (Magner et al., 2003; Bruce Wilson, personal communication). Like any practice designed for highly managed landscape, implementation of the two-stage ditch has tradeoffs that have resulted in barriers to adoption at the local, state, and regional level. Discussed further in Witter et al. (2011), tradeoffs include: land must be taken out of production to accommodate the wider benches of the two-stage design: existing federal cost-share practices on the landscape (i.e., grass buffers) might be impacted, which may have implications to the cost-share agreement; and the cost of two-stage ditch construction might be higher than traditional ditch maintenance practices. Other barriers to adoption that warrant further analysis and education include visual perception of a ditch that does not have the traditional trapezoidal shape, the ditch filling in, concerns about ditch bank stability during high flow events, and increased capacity causing downstream flooding.

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4

Active Floodplain Requirements for Sustaining Two-Stage Channel Geometry

Anand D. Jayakaran, Daniel E. Mecklenburg, Jonathan D. Witter, G. E. Powell, and Andrew D. Ward

CONTENTS

Introduction

The last few decades have seen increasing interest in enhancing, restoring, and protecting the ecology of wetlands, streams, and watersheds. Achieving these goals requires sound fundamental and applied knowledge, close interaction between scientists and engineers, a systems approach, and a good understanding of spatial and temporal scales. Here we address the role and importance of an active floodplain in wadeable two-stage stream systems where the active floodplain plays an essential role in sustaining or establishing dynamic equilibrium. Specifically, the focus is placed on the size and geometry of the active floodplain (Stage 2) relative to the size of the main channel (Stage 1) that is shaped by channel-forming discharges (Figure 4.1). Consideration is also given to incorporating floodplains into channelized streams and ditches. The goal is to provide an understanding of the hydrology, hydraulics, and geomorphology of these systems and to use this knowledge to protect or size a self-sustaining two-stage channel system.

DOI: 10.1201/9781003175094-4

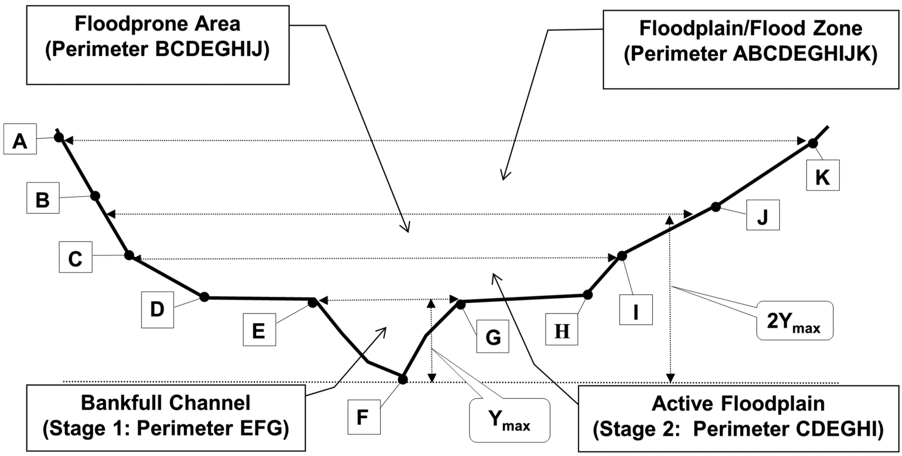

FIGURE 4.1
llustration of differences between a floodplain, flood-prone area, and active floodplain.

The term channel-forming discharge describes both the bankfull and effective discharge (Powell et al., 2006a). Wolman and Miller (1960) defined the bankfull discharge as the streamflow that fills the main channel and begins to spill onto the active floodplain, while the effective discharge is the discharge that transports the most sediment over time. Many studies have been conducted on channel-forming discharges and recurrence intervals associated with these flows (Biedenharn et al., 2000; Emmett and Wolman, 2001; Simon et al., 2004; Powell et al., 2006a). Most studies have suggested that the recurrence interval ranges from 1 year to more than 5 years (Nash, 1994; Whiting et al., 1999; Emmett and Wolman, 2001; Petit and Pauquet, 1997; Simon et al., 2004). However, in two studies on some large rivers (Powell et al., 2006a) and agricultural channels (Jayakaran et al., 2005) in Ohio, the authors suggest that flows larger than the channel-forming discharges occur many times annually, and the recurrence interval of channel-forming discharges is often <1 year when analyzed using a partial duration series.

The terms floodplain, flood zone, flood-prone area, active floodplain, and riparian zone are often used synonymously, causing much confusion as each of these terms can describe different locations on the landscape (Figure 4.1). Only occasionally do these locations coincide. A floodplain or flood zone could be associated with any point on the landscape, though most often associated with anthropocentric concerns with flood damage. Additionally, seldom are these areas flat, as the word "plain" implies. When considering dynamic equilibrium, rarely are we concerned with locations only inundated by infrequent events (recurrence interval of many hundreds of years). In the Rosgen (1994) stream classification system, the width of the flood-prone area is measured at an elevation above the thalweg that is twice the maximum bankfull depth. It is one of several factors used in this classification system and is not explicitly related to a specific recurrence interval flow. The term active floodplain is associated with any flows that exceed the channel-forming discharges. However, if these flows do not immediately spill out onto the active floodplain, then the main channel is described as incised or entrenched. A problem with the term active floodplain is that there is no upper limit to the depth or size of the area described by this term. Finally, the word riparian means riverbank, so a riparian zone is a piece of land located on the banks of a channel. In the United States, the term riparian zone is generally used to describe areas (such as forests, grasslands, and wetlands) beneficial to the riparian ecosystem and functioning as natural biological filters (Ward et al., 2015).

Processes in Channel Systems

Dynamic Equilibrium

Flowing water exerts a force on the bed and banks of a channel. If this force exceeds the resistance of the bed and banks to this force, then geomorphic work occurs, and there is a change in the channel geometry or bed slope. A channel system is said to be in dynamic equilibrium when it maintains its shape and character over time, and sediment inputs to the system are equal to its outputs (Heede, 1986). Lane (1955) stated that dynamic equilibrium exists between stream power and the discharge of bed material sediments:

$$Q_s d \propto QS \tag{4.1}$$

where Q_s is the sediment discharge, d is the median sediment size, Q is the discharge, and S is the bed slope. In the context of equation 4.1, if Q exceeds the channel-forming discharge, then equilibrium is achieved if much of the excess stream power (QS) is dissipated across an active floodplain. Sediment transported in a channel might consist of suspended load and bedload. In low gradient channels, the suspended load might be 95% or more of the total sediment load, while in steep upland channels, more than half of the sediment load might be bedload. Sediment movement can be related to the critical shear stress at which particles begin to move; while total sediment transport rate is related to discharge or stream power, and numerous bedload transport functions. The average shear stress, or tractive force, on the bed of a straight reach, can be estimated by (Newbury and Gaboury, 1993):

$$T = 1{,}000YS \tag{4.2}$$

where T is the tractive force (kg force/m^2), Y is the flow depth (m), and S is the bed slope (m/m). Lane (1955) found that a tractive force of 1 kg force/m^2 would move bed material with a mean particle size of about 10 mm. The average shear stresses on the banks of a straight channel can be approximated to be about 80% of the bed shear stresses (Lane and Carlson, 1953). Equation 4.2 is based on several simplifications and is not always consistent with observations. Many scientists have studied and proposed enhancements to tractive force concepts (Darby and Van de Wiel, 2003).

If the stream power is not dissipated across an active floodplain for discharges larger than the bankfull discharge, the bankfull channel will downcut and/or widen. This creates a domino effect as larger and larger flows are confined within the channel leading to more instability; the channel system is then said to be degraded or failing. At some point, a potential might occur for aggradation and/or the building of a new active floodplain; at this point, there is potential for channel recovery. Channel evelolution models (e.g., Simon, 1989) describe the process and sequences that lead to eventual restabilization or the attainment of equilibrium.

The classic work of Trimble (1999) describes how sediment budgets and channel geometry changed for Coon Creek, Wisconsin, over more than 130 years. The work shows why it is necessary to consider the interaction between landscape processes and in-stream processes when evaluating a channel system. Channel-forming discharges and the geometry of the bankfull channel are a function of many factors, including the drainage area, land use, watershed topography, sediment supply, in-stream sediment transport, riparian vegetation, the resistance of the bed and banks materials to shear, bed slope, and attributes of the active floodplain such as its geometry and resistance to flow. Spatial and temporal variations of these factors confound the assessment of their effects on the stream system. Montgomery and MacDonald (2002) outline a diagnostic approach to making stream assessments. Powell et al. (2007) describe a weight-of-evidence approach for sizing a two-stage channel system in agricultural ditches in the Midwest Region.

Channel-Forming Discharges

Substantially lengthy records of sediment transport and discharge are rarely available to calculate effective discharge. Therefore, they are estimated by either: calculating the bankfull discharge based on geomorphological measurements of the stream or calculating the effective discharge based on measured or estimated sediment and discharge data. Bankfull discharge is calculated based on measuring bankfull features and then using a resistance equation such as Manning's equation or the Darcy-Weisbach equation to calculate the mean flow velocity and discharge (Ward et al., 2015). In the USA, survey procedures similar to those described by Harrelson (1994) are commonly used to measure stream geomorphology. Typically, stream reaches that are at least 20 times the bankfull width are surveyed. Elevations of the channel bed, water surface, and bankfull features are recorded at points of noticeable change in the geomorphic character of the stream. Additionally, cross-section geometry and bed materials size are measured at regular intervals along the surveyed reach.

Regional curves that relate bankfull attributes and drainage areas are often used to assess stream morphology. Doll et al. (2002) stated that regional curves were especially useful to stream restoration projects, where identifying bankfull features is critical to designing a stable system. Jayakaran and Ward (2007) showed that bankfull dimensions of stable inset channels in agricultural streams were consistent with a regional curve. Powell et al. (2007) used regional curves as factors in their weight-of-evidence approach to size stable channels. However, regional curves should be used with caution and only provide a general indication of the bankfull geometry. The variability that might occur is illustrated in Figure 4.2, where bankfull width measurements by increasing drainage areas within the Olentangy and Upper Scioto River watersheds are reported.

The effective discharge is determined by an analysis of suspended or bedload obtained from long-term records or predicted using sediment transport equations. Approaches for calculating the effective discharge from observed data are widely published (Biedenharn et al., 2000; Powell

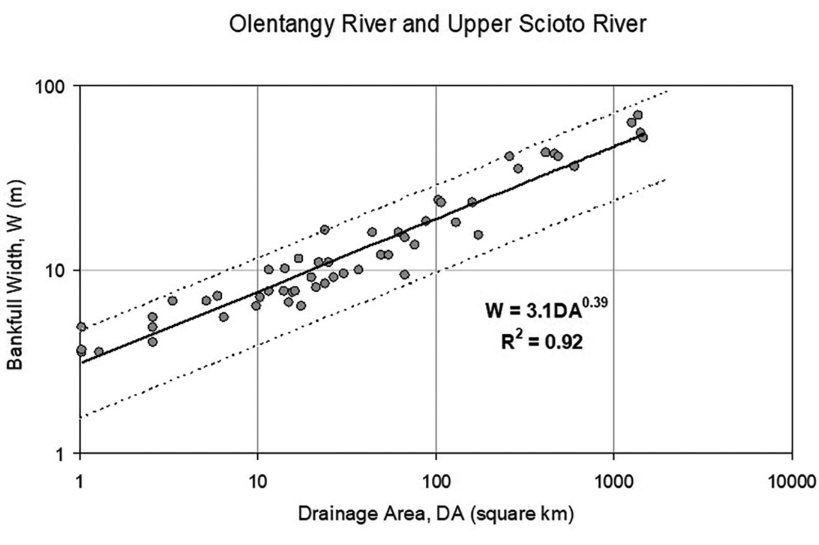

FIGURE 4.2
Bankfull width versus drainage area for the Olentangy and Upper Scioto Rivers, OH. Dashed lines are for values ±50% of the regression values.

et al., 2006a). The most common approaches use the Wolman-Miller model (Andrews and Nankervis, 1995). In addition, an approach for estimating the effective discharge based on the Meyer-Peter-Muller bedload transport equation is incorporated in the Spreadsheet Tools for River Evaluation, Assessment, and Monitoring (STREAM) modules (Powell et al., 2006b; Witter and Mecklenburg, 2022).

Active Floodplain Requirements

Channel Hydraulics and Characteristics

Turbulence, vertical vorticity, secondary flows, reverse and lateral mixing of flows on the floodplain and within the main channel, together with the associated sediment transport and shear stress differences that occur particularly within a meandering channel, make the analysis of the system complex. Practical approaches are presented in Shiono et al. (1999), Patra and Kar (2000), and Ward et al. (2015). Applying resistance (roughness) equations, such as Manning's equation, to compound straight two-stage channels presents challenges, particularly with shallow over-bank flows. As the stage initially rises above bankfull, there will be an increase in the wetted perimeter, little change in cross-sectional area, and a consequent decrease in hydraulic radius. The decrease in hydraulic radius results in a discontinuity in velocities estimated by Manning's equation. Another challenge is accounting for the interaction between the slower over-bank flows and the faster main channel flow.

Posey (1967) evaluated several commonly used resistance equation methods. They concluded that dividing the main section and two over-bank sections by vertical lines worked well when the over-bank flow was shallow. However, dividing into subsections is not necessary when the over-bank flow is at least half as deep as the bankfull channel depth. The method incorporated in the STREAM tools gives similar results to the methods suggested by Posey (1967) in the desired respective ranges. The problem of momentum transfer between sections is managed by dividing the sections with "virtual" banks that are perpendicular to lines of equal shear. Also, the hydraulic radius is based only on each section's physical boundaries and is weighted by the area of each section. Flow velocity is estimated using Manning's equation with different roughness factors assigned to the main channel and the flood plain. This method is an inexact approximation of the complex hydraulics of a two-stage meandering channel. It is only intended to provide estimates of relative values, not to predict actual bedload transport. Readers are directed to a modifed method to predict boundary shear distributions in two-stage channels as proposed by Khatua et al. (2012).

Active Floodplain Minimum Size Requirements

Recently in the USA, many local, county, and state organizations have expressed an interest in establishing streamway setbacks to help protect stream systems, particularly in urbanizing watersheds. However, these efforts are hampered by a lack of published information on how to size active floodplains and streamway setbacks to sustain dynamic equilibrium.

Williams (1986) proposed that meander beltwidth (B, m) and the bankfull width (W, m) to be estimated as follows:

$$B = 4.3W^{1.12} \tag{4.3}$$

where beltwidth is the lateral extent of the meander pattern embodying the long-term down-valley migration of the meander pattern. To account for meander migration over time, we recommend that natural streams be provided a streamway width calculated by increasing the coefficient of 4.3 to at least 6.5. This recommendation is based on stream surveys and the use of orthophotos to evaluate meander patterns and meander migrations for several streams in Ohio. For most unmodified natural streams, this approach will estimate a streamway width that is 7–12 times the bankfull width. For agricultural ditches typical of the Midwest Region, the authors have found that stable fluvial features form and maintain themselves within streamways as narrow as three times the bankfull width (Ward et al., 2015). These systems are relatively low gradient, transport little bedload sediment, and have robust herbaceous vegetation, mostly dense grass that grows on the side slopes, benches, and the banks.

A more process-based approach to estimating effective discharge considers sediment transport, the shear stresses on the bottom and sides of each stage, and mean velocities in the two-stage system. The following example illustrates how changes in the active floodplain geometry influence these factors. An analysis was performed on an incised trapezoidal channel that is typical of many large agricultural ditches in the Midwest Region of the USA (Figure 4.3a and b). An analysis was performed for a hypothetical 10 km^2 drainage area and a channel with a bedslope of 0.5%. The bed width was 4 m, the side slopes of the main channel were 2:1, the depth of the incised trapezoidal channel was 3 m, and the initial top width of the second stage was 16 m. For simplicity, it was assumed that the bankfull channel had vertical side slopes. Discharge versus recurrence interval relationships were estimated using an empirical procedure based on streamflow data developed by Sherwood (1993) for urban areas in Ohio. The basin-development factor, a measure of urbanization in the equation, was set to zero to represent rural conditions. The mean particle size of the bed material was set to 40 mm. An evaluation was made with Manning's n values of 0.03, 0.045, and 0.06 for the active floodplain.

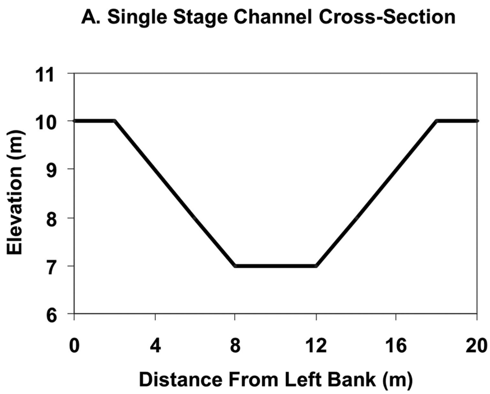

FIGURE 4.3A
Single-stage channel geometry used in the process analysis example.

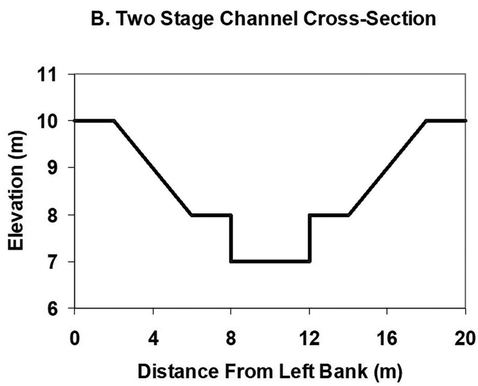

FIGURE 4.3B
Two-stage channel geometry used in the process analysis example.

Shear stresses were calculated using equation 4.2. Bedload transport was determined using the method in the STREAM tools (Witter and Mecklenburg, 2022). The two-stage system was proportioned, between the floodplains and the main channel, using the approach described by Posey (1967). Flow velocities and flow depths were determined using Manning's equation. A floodplain ratio (FPR) was used to evaluate different active floodplain widths. The FPR was defined as the ratio of the floodplain width at the bottom of the second stage, to the bankfull width, which is the channel width at the top of the first stage. The depth of the bankfull channel was 1 m, and depositional benches formed to give an FPR of 2 (Figure 4.3b). FPR values >2 required widening the second stage of the channel. Shear stresses, total annual bedload transport, and hydraulic properties were related to the bankfull discharge of 7.2 m^3/s, or the 25-year recurrence interval flow of 41 m^3/s.

TABLE 4.1

Summary Results for Channel System Analysis

System Attributes	Units	Floodplain $n=0.03$		Floodplain $n=0.045$		Floodplain $n=0.06$	
		Channel	Active FP	Channel	Active FP	Channel	Active FP
Bankfull Channel							
Mean velocity	m/s	1.8					
Max depth of flow	m	1					
Shear stress	kg/m^2	5					
Bedload transport[a]	m^3/year	35					
FPR = 1							
Mean velocity	m/s	2.7					
Max depth of flow	m	1.9					
Shear stress	kg/m^2	9.7					
Bedload transport	m^3/year	42					
FPR = 2							
Mean velocity	m/s	2.5	2.5	2.6	1.8	2.7	1.4
Max depth of flow	m	2.2	1.2	2.4	1.4	2.6	1.6
Shear stress	kg/m^2	11.0	6.0	12.2	7.2	13.1	8.1
Bedload transport	m^3/year	75		84		90	
FPR = 3							
Mean velocity	m/s	2.3	2.3	2.5	1.7	2.5	1.4
Max depth of flow	m	2.0	1.0	2.2	1.2	2.4	1.4
Shear stress	kg/m^2	9.9	4.9	11.0	6.0	11.8	6.8
Bedload transport	m^3/year	67		75		81	

(Continued)

TABLE 4.1 (*Continued*)

Summary Results for Channel System Analysis

		Floodplain $n = 0.03$		Floodplain $n = 0.045$		Floodplain $n = 0.06$	
System Attributes	**Units**	**Channel**	**Active FP**	**Channel**	**Active FP**	**Channel**	**Active FP**
FPR = 5							
Mean velocity	m/s	2.2	2.0	2.3	1.5	2.4	1.2
Max depth of flow	m	1.7	0.7	1.9	0.9	2.1	1.1
Shear stress	kg/m^2	8.7	3.7	9.6	4.6	10.3	5.3
Bedload transport	m^3/year	59		65		70	
FPR = 9							
Mean velocity	m/s	2.1	1.6	2.2	1.2	2.2	1.0
Max depth of flow	m	1.5	0.5	1.7	0.7	1.8	0.8
Shear stress	kg/m^2	7.7	2.7	8.4	3.4	8.9	3.9
Bedload transport	m^3/year	51		56		60	
FPR = 13							
Mean velocity	m/s	2.1	1.4	2.1	1.1	2.2	0.9
Max depth of flow	m	1.4	0.4	1.5	0.5	1.6	0.6
Shear stress	kg/m^2	7.2	2.2	7.7	2.7	8.2	3.2
Bedload transport	m^3/year	48		52		55	

[a] Annual bedload transport is only in the main channel that extends up into the second stage based on the Posey (1967) partitioning approach.

A summary of the results is presented in Table 4.1. For the scenario where small depositional benches (FPR=2) have developed, there is a 20% increase in the shear stresses in the main channel and an 80% increase in bedload transport compared to the single-stage channel (FPR=1). This trend is consistent with our observation in agricultural ditches. The formation of benches and an inset channel (the bankfull channel) results in coarser substrate and a self-flushing system with reduced or non-existent aggradation.

Notably, flows on the floodplain cause shear stresses on the banks of the second stage that are much less than with the single-stage scenario with no floodplain. Even with high roughness on the floodplain (n of 0.06), the mean bank shear stresses in the second stage are lower than they were for the single-stage trapezoidal channel (FPR of 1).

For an FPR of 3, the flow depth for the 25-year recurrence-interval discharge is at twice the maximum bankfull depth and corresponds to the flood-prone area described by the Rosgen classification system (Rosgen, 1994). However, the floodplain is undersized to provide the meander pattern that might be expected. An FPR of 5 reduces the shear stresses and velocities of flow in the main channel to values that are similar or less than those in the single-stage trapezoidal channel (FPR of 1). An FPR of 9, and high floodplain roughness, provides lower values than the single-stage channel for all attributes except bedload transport. As expected, further increases in the FPR will continue to provide gradual improvements in the system attributes. However, based on these illustrations and other studies by the authors, it appears an FPR between 5 and 10 is needed to obtain a self-sustaining system. This is consistent with the empirical approach presented earlier.

Discussion

Applying these approaches reduces the amount of initial engineering of the system and focuses on utilizing natural processes to develop a system that sustains typical physical, chemical, and biological process of fluvial systems (Beechie et al., 2010). An important consideration in most stream projects is the availability of an active floodplain with adequate width to dissipate the energy of high flows and a properly sized bankfull channel to sustain continuity in sediment transport. Where channelization is extensive, an active floodplain width that is three times or greater multiple of the bankfull channel typically results in a stable geometry and enhances ecosystem services associated with floodplains (e.g., Roley et al., 2012; Vastila et al., 2021). Simple and complex tools are available—STREAM tools (Powell et al., 2006b; Witter and Mecklenburg, 2022) or HEC-RAS (2022)—to evaluate two-stage systems on a case-by-case basis. Shields et al. (2003) provide useful additional discussion on this topic. Procedures helpful in stream channel design are present in an Natural Resources Conservation Service handbook (NRCS, 2007). It is recommended that a bank stability analysis be performed as many channel systems fail due to mass wasting. The freely available USDA-ARS bank stability model (Simon and Langendoen, 2007) is a valuable tool.

The approaches described in this chapter are relatively simple and represent a minimum level of analysis that should be performed if modifications or protection strategies are proposed for a stream system. Krider et al. (2017) demonstrated that a trapezoidal ditch converted to a two-stage design with FPR 3 exhibited increased stability and distinct fluvial features like gravel riffles and deeper pools. Ideally, active floodplains should have FPRs >5 though smaller floodplains will have some beneficial influences on the sustainability of channel systems. Constructed two-stage ditches have been shown to be stable and require little maintenance even 10 years after construction (D'Ambrosio et al., 2015); however, proper design and installation are critical.

This chapter has not addressed ecological issues or the debate on designing bankfull channels. Where possible, we recommend an ecological engineering or naturalization approach. Useful discussions on these topics are provided by Palmer et al. (2005), Herricks and Suen (2006), and Roley et al. (2012).

Acknowledgments

Part of this contribution is based on "Ward, A. et al. 2007. Requirements for Two-Stage Channel Geometry on Active Floodplains. In Trimble, S., Ed, Encyclopedia of Water Science, 2nd edition, CRC Press, Boco Raton, FL." Copyright 2007 and reproduced by permission from the publisher Taylor and Francis Group, LLC, a division of Informa PLC.

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5

Enhanced Channel Design v2.6: A Design Tool for Two-Stage Ditches and Self-Forming Channels

Jonathan D. Witter and Daniel E. Mecklenburg

CONTENTS

Introduction

Spreadsheets were developed to facilitate design of two-stage ditch and self-forming channel approaches for surface drainage in low-gradient landscapes. Drainage channel designs require input and reduction of survey data; analysis of channel morphology, hydrology, and hydraulics; estimates of earthwork volumes; and production of construction plans. The spreadsheet facilitates rapid evaluation of multiple design alternatives and assessment of costs/benefits to facilitate informed decision-making.

Enhanced Channel Design v2.6 Software

The Enhanced Channel Design v2.6 is a Microsoft Excel spreadsheet program included in the STREAM Modules (**S**preadsheet **T**ools for **R**iver **E**valuation, **A**ssessment and **M**onitoring: Mecklenburg and Ward, 2004). The primary application of the program is specific to drainage channels constructed to enhance surface drainage or provide outlet for subsurface drainage systems (i.e., tile drainage). The spreadsheet includes five visible worksheets including "Instructions," "Start," "Survey," "Design," "Hydraulics," and "Drawings."

DOI: 10.1201/9781003175094-5

The "Instructions" worksheet provides a general overview of the module and will include updates and document major revisions to future releases of the software.

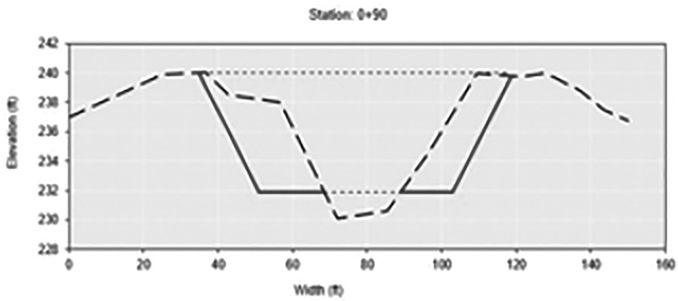
The "Start" worksheet allows for the selection of measurement units (English/US Customary or Metric) and documentation of general project information (e.g., watershed drainage area, channel name, watershed name, geographic coordinates, and narrative description of the project site). The user also can specify how the survey data was collected including options for: (1) a profile survey completed with a laser level utilizing distances and elevations from an upstream starting point, or (2) a GPS survey utilizing an appropriate State Plane Coordinate System.

The user also has the option to specify the type of channel that will be designed with the following options: (1) Bench (i.e., two-stage channel design), (2) Self-Forming, or (3) Bench and Channel. The "Bench" option (Figure 5.1a) would simply widen the channel at the elevation of an inset bankfull channel, if one has formed within the larger channel, or created a floodplain at the bankfull elevation predicted by a regional hydraulic geometry relationship. The "Self-Forming" design option (Figure 5.1b) would result in a flat or nearly flat surface at the channel bed elevation and widen the existing channel to a user selected multiple of the estimated bankfull channel width. The floodplain width on a two-stage channel or the bottom width of a self-forming channel is typically three to five times wider than the predicted bankfull channel estimated by the regional hydraulic geometry relationship. The final design type would be the "Bench and Channel" construction (Figure 5.1c) where both the floodplain benches and inset channel are constructed. A typical example of the "Bench and Channel Design" approach would occur when a channel is being relocated to an area where a channel did not previously exist and a channel with a two-stage geometry is the preferred design alternative.

The user is also able to specify an appropriate regional hydraulic geometry relationship, which relates channel bankfull dimensions to drainage area, as a starting point for sizing the inset channel and floodplain benches. There are two options to choose a regional hydraulic geometry relationship including selection from a pull-down menu populated with pre-loaded values determined from other research studies and a second option that allows the user to enter defined relationships developed from field measurements for a specific watershed, reach, or project.

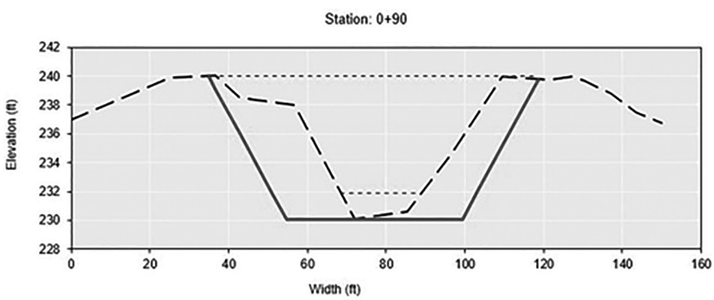
To assess conveyance capacity of the existing channel and various design alternatives using at-a-station open channel hydraulics, the user has the option to input hydrologic estimates (2-, 5-, 10-, 25-, 50-, and 100-year recurrence interval discharges) using locally accepted peak discharge equations or outputs from a model of application (e.g., USGS StreamStats, https://streamstats.usgs.gov, last checked February 6, 2022).

The "Survey" worksheet allows input of survey data for 12 cross sections for the study reach. The Survey worksheet can be copied up to three times if additional cross sections are needed. Copies of the Survey worksheet are numbered sequentially (Survey 1–4) and automatically integrated and linked to the previous Survey worksheets. The spreadsheet plots the longitudinal profile and calculates reach slope by connecting longitudinal stations and bed elevation data derived from consecutive cross sections along the channel course. Figures 5.2 and 5.3 present an example of multiple cross sections and the longitudinal profile, respectively. The user can also specify the "Bench" (i.e., bankfull) stage and the "Ditch Full Stage" where the channel begins to flood the adjacent land.



Station: 0+90	Ditch Full	Bench Stage
Existing depth (ft)	9.9	1.8
Width (ft)	117.1	19.7
X-Section Area (ft.sq.)	441.0	26.7

FIGURE 5.1A
Example of a two-stage ditch design where the channel is widened at the inset channel bankfull elevation.



Station: 0+90	Ditch Full	Bench Stage
Existing depth (ft)	9.9	1.8
Width (ft)	117.1	19.7
X-Section Area (ft.sq.)	441.0	26.7

FIGURE 5.1B
Example of a self-forming design where the channel bottom is widened at the channel bed or design elevation.

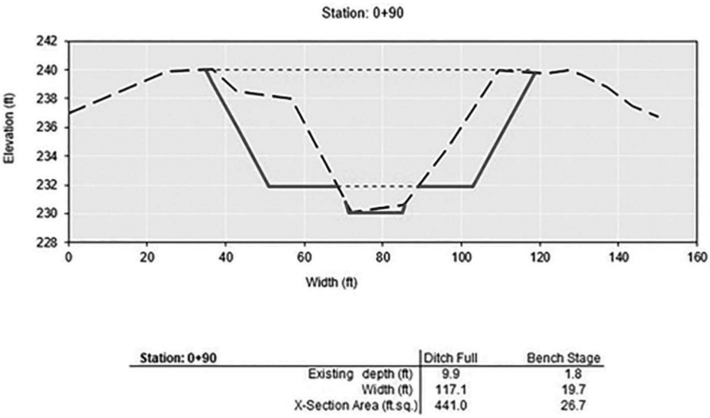


FIGURE 5.1C
Example of a channel and bed design where the channel and floodplain bench is designed and constructed.

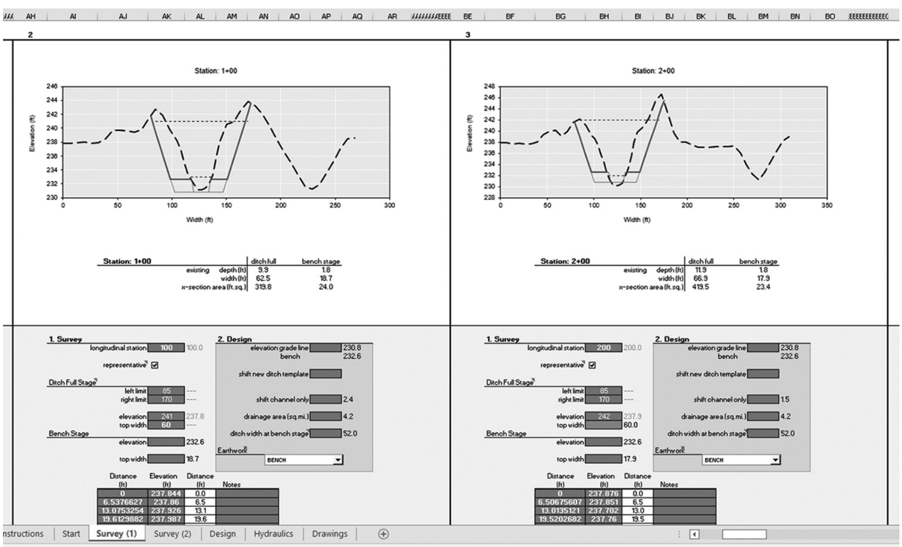


FIGURE 5.2
Survey data entry and cross-section design tab in the Enhanced Channel Design v2.6 STREAM Module.

Design choices for individual cross sections can be made on the "Survey" worksheets. The spreadsheet estimates the designed channel dimensions based on the hydraulic geometry relationships specified earlier in the process and attempts to fit the design to the existing cross-section dimensions. However, the user has the option to shift the designed channel vertically or horizontally to best fit the existing channel and achieve any specific design goals (e.g., channel capacity, threshold flow velocities, and minimize earthwork volume).

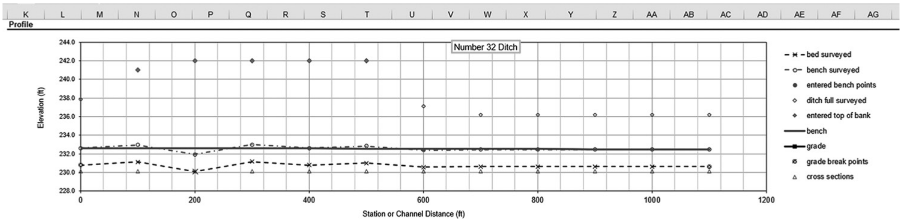


FIGURE 5.3
Longitudinal profile construction from cross-section survey data in the Enhanced Channel Design v2.6 STREAM Module.

The "Design" worksheet allows the user to change various aspects of the designed channel (e.g., bench width ratio, ditch and inset channel side slopes, and channel bankfull dimensions) for all cross sections and quickly determine the impact on earthwork volume and overall footprint of the designed channel.

The "Hydraulics" worksheet facilitates calculation of various channel hydraulics variables (e.g., discharge rate, water depth, mean velocity, shear stress, and recurrence interval) at multiple stages including the bankfull or bench stage, the ditch full stage, and the 2- and 10-year discharge recurrence intervals. The user must specify the vegetative cover or materials that constitute the channel bed, floodplain benches, and channel side slopes, which correspond to published roughness values (i.e., Manning's *n*-values) used to estimate hydraulics properties of the proposed designs. The user can quickly evaluate multiple conditions including: (1) the newly constructed channel with bare soils to determine need for erosion control measures and potential for failure if design discharges are exceeded, or (2) the "aged" or vegetated condition to determine best- and worst-case scenarios for maintained and unmaintained vegetation and the effects on channel stability and capacity.

The final worksheet "Drawings" provides construction plans and quantities for the designer to communicate with the contractor to use in bidding and implementation. Example outputs are provided in Figure 5.4.

Due to the changing nature of the STREAM worksheets, it is advisable to contact either Jonathan Witter or Daniel Mecklenburg for current materials and guidance.

Literature for Further Background Reading

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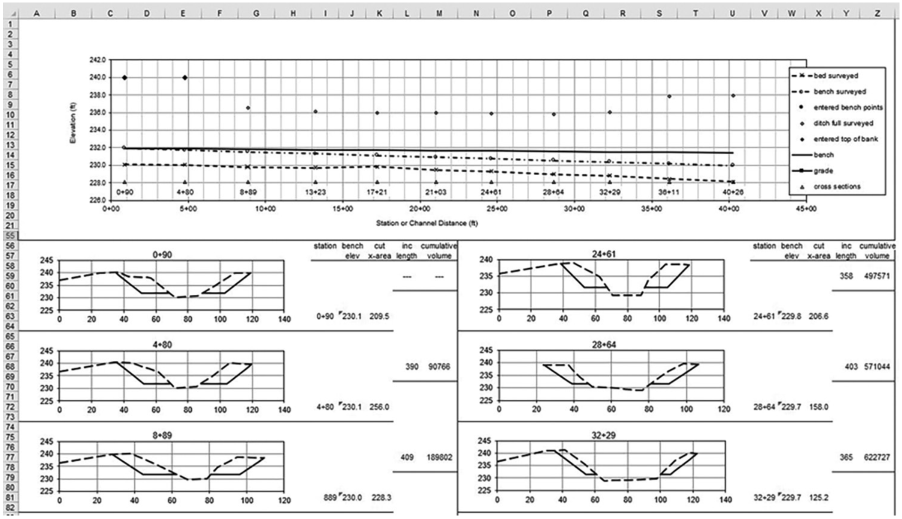

FIGURE 5.4
Construction drawings and earthwork volumes generated by the spreadsheet.

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6

Evaluating Geomorphic Change in Constructed Two-Stage Ditches

Jessica L. D'Ambrioso, Jonathan D. Witter, and Andrew Ward

CONTENTS

Introduction

In the Midwestern USA, agricultural drainage channels dominate the headwaters of drainage networks. More than 120,000 km of drainage channels, often called ditches, was constructed in Ohio, Michigan, Indiana, and Minnesota by 1930 (USDOC, 1930). These ditches, which may form up to 80% or more of the entire drainage network length (Blann et al., 2009), have poorly drained soils and hillslope gradients that are often less than one percent. Straight, trapezoidal-shaped drainage ditches, built wide (10–20 m) and deep (1.5–4 m) to serve as the collection system for subsurface tile drains that are commonly located 0.8–1.0 m below the ground surface and are spaced 9–25 m apart under agricultural fields, have been the design standard for 50 years (USDA, 1964).

DOI: 10.1201/9781003175094-6

Surface drainage ditches are large enough to convey the 10–100-year recurrence interval storm events and have little or no connection to an active floodplain. They efficiently drain the soil profile, but their deviations from natural fluvial conditions drive the need for frequent maintenance. In many cases, these low-gradient ditches fail to transport sediment which results in aggradation that flattens the hydraulic gradient and increases channel hydraulic residence (Magner et al., 2012). Increased hydraulic residence increases the depth of flow, which then impedes drainage and causes wet conditions in the fields and reduced crop yields (Blann et al., 2009). Maintenance activities that might occur every 1–20 years to restore hydraulic capacity can be costly. Hansen et al. (2006) reported that in Minnesota, $12 million per year were being spent on drainage ditch maintenance. Ohio has more than 6,400 km of open channels that are maintained for agricultural drainage (ODNR, 2009). In Ohio, an average of $720/km is spent annually on open-channel ditch maintenance (Hansen et al., 2006).

Maintenance activities cause hydraulic disruptions throughout the watershed that disturb existing ecology, adversely impact water quality, and expose the channel bed and banks to erosion by removing stabilizing vegetation. Ecological and socioeconomic impacts of drainage ditch maintenance activities can be significant. Nutrient and sediment export from farm fields and through surface ditches causes hypoxic zones in receiving water systems from downstream river reaches and reservoirs to the Gulf of Mexico and the Great Lakes. Excessive nutrients can result in algal blooms that threaten human health and the economy and have resulted in an estimated $2.2 billion cost in losses to public health, commercial fishing, tourism, and management in the USA (Dodds et al., 2009). If left unmaintained, a return of natural fluvial processes and forms has been observed in some agricultural drainage channels in the upper Midwest. Deposition of fine material leads to the formation of bars, benches covered with dense vegetation, and a small meandering inset channel (Landwehr and Rhoads, 2003; Jayakaran et al., 2005; Jayakaran and Ward, 2007; Rhoads and Massey, 2011). Rarely in agricultural ditches is there sufficient space or energy for nature to reestablish all the ecosystem services of a high-quality stream system that existed prior to disturbance if a stream channel existed there at all. Robust ditch systems, which often are extensions of the drainage network due to land use changes, can provide desirable ecological functions while still providing adequate drainage (Rhoads and Herricks, 1996; Rhoads et al., 1999; Frothingham et al., 2002; Jayakaran et al., 2010). These findings have led researchers to develop alternative channel designs and management strategies that are more consistent with fluvial form and process. The approaches enhance the provision of ecological services while meeting drainage needs essential for agricultural production. In this chapter, we focus on an alternative floodplain establishment design, called the two-stage agricultural ditch, that we developed 12 years

FIGURE 6.1
Study site Shatto Ditch before two-stage ditch construction in 2007 (a) and one year after two-stage ditch construction (b).

ago (Figure 6.1). We hypothesized that reconfiguration of a trapezoidal ditch to a two-stage geometry would result in a self-sustaining agricultural drainage channel system based on the principles of fluvial geomorphology and would require less frequent and less substantial maintenance (Ward et al., 2004; Powell et al., 2007b; Rhoads and Massey, 2011). The two-stage design retains a first stage (lower stage) that is the existing inset channel associated with channel forming discharges. In areas with poorly draining soils, this may be the subsurface tile flow rather than the natural hydrology. In the tri-state region of this study bankfull conditions in ditches and many large rivers often occur more frequently than a one-year recurrence interval (Jayakaran et al., 2005; Powell et al., 2006, 2007b; Gorney et al., 2011; Fry et al., 2012). In two separate studies on Ohio streams, Fry et al. (2012) and Powell et al. (2006) determined that bankfull discharges occurred 1–24 and 1–18 times annually, respectively. The second stage is the construction of a floodplain (called a bench) related to a minimum size to provide bank stability, the position of subsurface drainage outlets, and a desired conveyance capacity to minimize frequent flooding of adjacent fields (Ward et al., 2008). The approach we recommend does not engineer the inset channel (Powell et al., 2007b). The benches are excavated to the designed width at the bankfull elevation, and the expectation of the two-stage ditch is a channel system in a stable, quasi-equilibrium state (Schumm et al., 1984; Simon and Hupp, 1986; Schumm, 1993). Two-stage channels are considered stable if they are neither aggrading nor degrading based on geomorphic theory (Lane, 1955; Leopold, 1994). If the self-sustaining hypothesis is correct, then the design should require no maintenance to maintain sediment transport or the geometry of the inset channel, benches, and ditch side slopes. Successful implementation of the two-stage ditch practice has led to its incorporation in Part 654 Stream Restoration Design in the National Engineering Handbook (USDA-NRCS, 2007), and it has seen application in the upper Midwest region of the USA with great success (Magner et al., 2012). More than 50 agricultural ditches, most of them in Indiana, have been modified to a two-stage design; yet there has been little monitoring or research done on these systems. Furthermore, the significance of their contribution to the larger watershed is unknown. Of particular importance is whether the systems are: (1) self-sustaining, (2) meeting the drainage needs of farmers, (3) reducing the amount of nitrate nitrogen, phosphorus, and sediments reaching downstream receiving systems, and (4) providing other ecological services such as improving in-stream biota. We present the results of a geomorphic study on trapezoidal ditches that were modified to two-stage geometries.

The main question addressed was had there been significant geomorphic changes in the inset channel, the benches, and the banks of the second stage? We also addressed the question of whether these two-stage ditches are maintaining their drainage capacity over time without the need for routine maintenance.

Methods

Study Site Description

We selected seven two-stage ditch study sites located in Indiana, Michigan, and Ohio that were constructed between 2002 and 2009 (Figure 6.2). Study sites are in low-gradient watersheds with fine-grained, poorly drained soils (Table 6.1). Row crop agriculture is the primary land use in each of the watersheds. Five sites drain to Lake Erie, and two sites drain into the Mississippi River. Study sites were sized and constructed with the nine-step procedure developed by Powell et al. (2007a, b). The recommended design target was a flooded bench width that was two to four times the inset channel width (Ward et al., 2008).

Data Collection

We collected pre-construction and post-construction geomorphic data at each study site. We measured cross-section data every 30.5 or 71 m using standard laser leveling techniques (Harrelson et al., 1994; Powell et al., 2007b). We used measured bed elevations and water depths at each cross section to construct the ditch profile and to ascertain channel bed slope. We entered data into the STREAM Reference Reach Spreadsheet tool for analysis (RRSS, Witter and Mecklenburg, 2022). The RRSS calculated the minimum, maximum, and average cross-sectional area (m^2), width (m), mean depth (m), and maximum depth (m) for each datum and each year used in the analysis. At each site, we assumed a Manning's n value of 0.03 for the inset channel and a value of

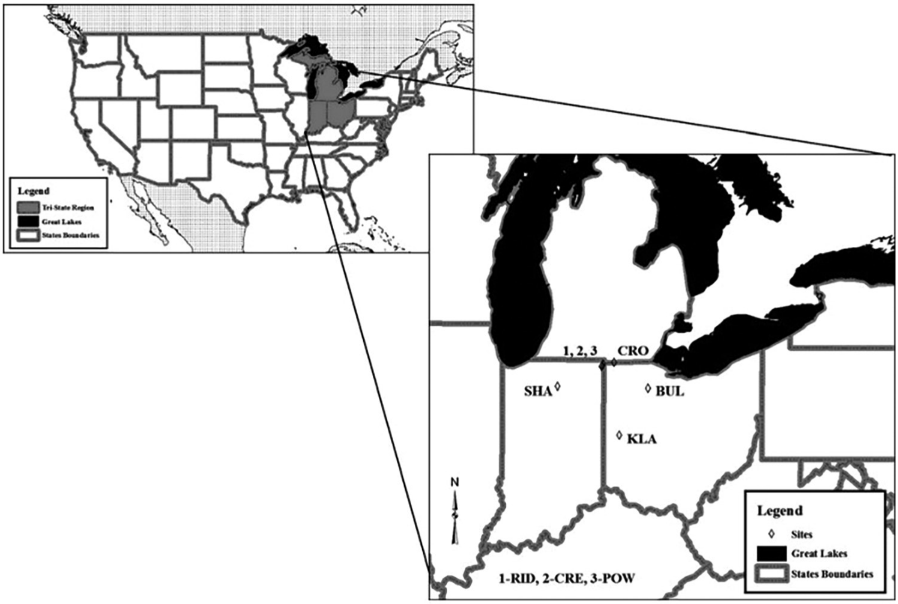

FIGURE 6.2
(a) Regional map of locations. (b) Location of the seven constructed Two-Stage ditch study sites in the tri-state region of Michigan, Ohio, and Indiana: Shatto Ditch (SHA), Klase Ditch (KLA), Bull Creek Tributary (BUL), Crommer Ditch (CRO), Ridenour Ditch (1-RID), Creel Ditch (2-CRE), and Powers Ditch (3-POW), and (c) The legend.

0.06 for the grass-covered benches and banks of the second stage (Ward et al., 2015). Precipitation data were downloaded from the National Oceanic and Atmospheric Administration's National Climatic Data Center to quantify the frequency and magnitude of high flow events affecting the sites. For each site, daily precipitation data were obtained for the period of the first year after construction to the last year of data collection. The gages were located 10–15 km from each study site. Data were sorted into the number of daily precipitation events at each site that were between 10 and 30 mm and >30 mm. Daily precipitation events of 10–30 mm result in subsurface tile drainage flow depending on the time of year and antecedent soil water conditions. Based on observations, daily precipitation larger than 30 mm usually causes bankfull flows in the ditches in the tri-state region.

Determinations of Geomorphic Change

There was a lack of fixed vertical and horizontal datum points at each of the study sites. The main reasons for this were (1) that the ditches were

TABLE 6.1

Attributes and Ditchfull Geomorphic Dimensions for Each of the Study Sites Prior to Construction and the First Survey after Construction

Site	Latitude	Longitude	Watershed Area (km²)	Row Crop (%)	Length (m)	Bed Slope (%)	Survey Year	X-Area (m²)	W (m)	D_{mean} (m)
Bull Creek Tributary	41.1785	83.5909	9.3	77	640	0.2	2001	4.44 (0.50)	6.92	0.69 (0.17)
							2004	5.09 (0.77)	8.48	0.61 (0.09)
Crommer Ditch	41.7080	84.5159	10.3	64	610	0.3	2003	9.62 (1.24) 8.87 1.09 (0.10)		
							2004	12.61 (2.21)	13.05	0.97 (0.20)
Klase Ditch	40.2216	84.3630	6.7	70	975	0.2	2004	8.51 (1.13)	8.16	1.11 (0.07)
							2009	17.00 (3.00)	13.39	1.28 (0.23)
Shatto Ditch	41.2218	86.0440	11.1	90	640	0.3	2007	1.36 (0.54)	3.32	0.42 (0.04)
							2009	6.67 (2.31)	11.66	0.58 (0.20)
Creel Ditch	41.6206	84.8535	8.6	64	1,189	0.4	2007	10.68 (1.88)	8.74	1.33 (0.17)
							2009	4.10 (2.71)	18.03	1.34 (0.11)
Powers Ditch	41.6394	84.8329	25.6	64	427	0.3	2009	10.46 (1.80)	8.65	1.17 (0.13)
							2010	13.21 (4.16)	12.28	1.07 (0.37)
Ridenour Ditch	41.6699	84.8106	6.0	64	640	0.2	2009	2.85 (0.42)	5.08	0.62 (0.07)
							2010	5.42 (1.16)	8.39	0.65 (0.14)

constructed at different times and with varying, non-contiguous funding sources that provided for different levels of monitoring; (2) at the request of the farmers, cross sections were not monumented in the field; and (3) each ditch contained inherent variability in channel and ditch dimensions along the constructed reach. Lack of monumented sites prevented the use of paired statistical approaches and determination of temporal estimates of aggradation or degradation at each site. The approach we selected to reduce variation as much as possible over time when analyzing changes in ditch dimensions, therefore, was like soil sampling whereby a composite set of data points were collected in a systematic fashion (i.e., along a cross section every 60 m during late summer), and all of the measurements at a site were treated as a composite sample for that site. Just as in soil testing, the analytical results from composite sampling provided average values for the sampled area. The number of samples measured depended on ditch length. For each site, we then developed an approach to evaluate: (1) aggradation or degradation of the non-constructed inset channel, herein called bankfull; (2) scour or deposition below a fixed lower datum located just above the constructed benches; and (3) scour or deposition below a fixed upper datum located near the top of each ditch (Figure 6.3). We adjusted the elevation line in the RRSS, which is used to determine flow and geometry values, to different heights to establish inset channel, lower, and upper fixed datums for the measured cross sections at each site and for each year (Figure 6.3). For the inset channel bankfull datum, changes in the area, maximum depth and mean depth would be a function of aggradation or degradation in the inset channel. For the lower datum, changes in the cross-sectional area, maximum depth and mean depth would be a function of aggradation or degradation in the inset channel and on the benches. For the upper datum, changes in the cross-sectional area, maximum depth and mean depth would be a function of aggradation or degradation in the inset channel, the benches, and the banks of the second stage. Statistical analyses compared the first geomorphic survey after construction to the most recent geomorphic survey at each site (i.e., 2004 and 2012). The RRSS plots a best-fit regression line on the ditch profile based on the field measured bankfull elevation, which often is identified by a grade break, scour line, or change in vegetation (Ward et al., 2015; Witter and Mecklenburg, 2022). The elevation of the regression line is automatically plotted onto each cross section in the RRSS. To ensure we were comparing similarly defined inset channel areas across sites, we determined that the top of the inset channel (the bankfull datum) was the highest measured point in the inset channel that did not include any bench area (point G or I in Figure 6.3). In the RRSS, we adjusted the elevation of this line up or down accordingly at each cross section making each bankfull datum cross section specific within and across study sites. In cases where the inset channel could not clearly be identified without including bench area, we removed that cross section from the analysis. The lower datum at each site was based on the measured point (width FJ in Figure 6.3) that was just above the bench for all

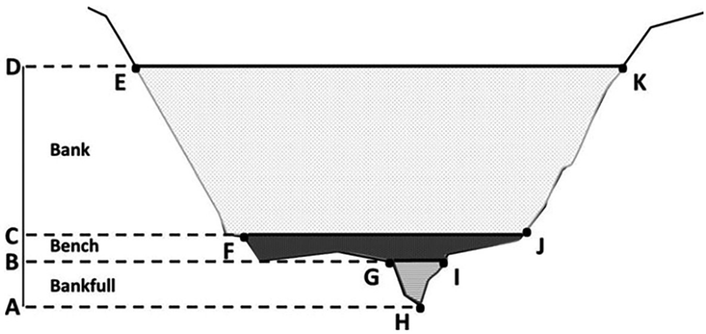

FIGURE 6.3
A surveyed cross section of Creel Ditch illustrating statistical analyses of bankfull, bench, and bank geomorphic change over time. Data for the bankfull area (GHI) are presented in Table 6.3. Data for the lower fixed width area (FGHIJ) are presented in Table 6.4. Data for the upper fixed width area (EFGHIJK) are presented in Table 6.6. Calculated percent changes for the banks (CD) and benches (BC) are presented in Table 6.5.

the cross sections. The cross section that had the smallest width at this elevation was used as the fixed width value. We then used the goal-seek function in the RRSS to adjust the elevation of the line at each cross section to create a common fixed width across the data sets for each site. We used a similar approach to establish the upper datum at each site except that the fixed common width was determined based on the highest measured point that was located within the ditch (width EK in Figure 6.3).

Data for the bankfull area (GHI) are presented in Table 6.3. Data for the lower fixed width area (FGHIJ) are presented in Table 6.4. Data for the upper fixed width area (EFGHIJK) are presented in Table 6.6. Calculated percent changes for the banks (CD) and benches (BC) are presented in Table 6.5.

For the inset channel, we determined the average cross-sectional area below GI (area, m^2), average width at GI (width, m), average maximum depth AB (D_{max}, m), and average mean depth (D_{mean}, m), and standard deviations at each site and for each year of data. The D_{mean} at each cross section is the cross-sectional area divided by the top width. For the lower and upper datums, we determined the average cross-sectional area below FJ and EK (area, m^2), average maximum depth AC and AD (D_{max}, m), and the average mean depth (D_{mean}, m), and standard deviations, respectively, at each site and for each year of data. To partition out changes in dimension for just the benches (BC), we subtracted the cross-sectional area of the inset channel (GHI) from the cross-sectional area of the lower datum (FGHIJ) determined by the fixed width analysis for each year. We then calculated the difference between the first year after construction and the most recent survey year to determine an overall percent change in bench area at a site over time. An indicator of the depth of scour or deposition is the mean depth of the cross section below the lower datum. We calculated a depth of aggradation or degradation of the benches by dividing bench area change by the average width of the lower datum (FJ) and converting to millimeters. We determined a rate of aggradation or degradation by dividing the calculated depth by time between surveys in months. To partition out changes in dimension for just the banks (CD), we subtracted the cross-sectional area of the lower datum (FGHIJ) from the cross-sectional area of the upper datum (EFGHIJK) determined by the fixed width analysis for each year. We then calculated the difference between the first year after construction and the most recent survey year to determine an overall percent change in bank area at a site over time. We statistically compared average width, average mean depth, average maximum depth, and average cross-sectional area for the inset channel, lower, and upper dimensions for each site for the first year after two-stage construction and the last year of survey data that we had obtained using t-tests assuming unequal variances ($a=0.05$). The two-tailed p-value was reported because changes in data reflected both increases and decreases in values.

Regional Hydraulic Geometry Relationships

For each ditch, a site-specific regional curve was used to size the benches (Powell et al., 2006, 2007a, b; D'Ambrosio, 2013). However, for illustration purposes, we used a composite set of regional curves that we developed for use in the tri-state region of Indiana, Ohio, and Michigan (Figure 6.4; Ward et al., 2013). We estimated drainage areas with the U.S. Geological Survey's StreamStats tool (Koltun et al., 2006; Rao, 2006).

Results

Precipitation Events Precipitation data periods of record ranged from 34 to 118 months (Table 6.2). From 2010 to 2012, the study sites experienced both extreme high and low rainfall conditions. The wettest year on record occurred in 2011. The hottest year on record for many locations within the tri-state region occurred in 2012. On average since construction, each site potentially experienced 31–36 daily events annually with 10 mm or more of precipitation (Table 6.2) and at least five daily events annually that exceeded 30 mm. Debris and flattening of grass on the benches were observed at each site as well as signs of both scour and deposition. Roley et al. (2014), Mahl et al. (2015), and Davis et al. (2015) report that benches were inundated 13–116 days/year on average between 2009 and 2010 for the two-stage ditch sites in this study

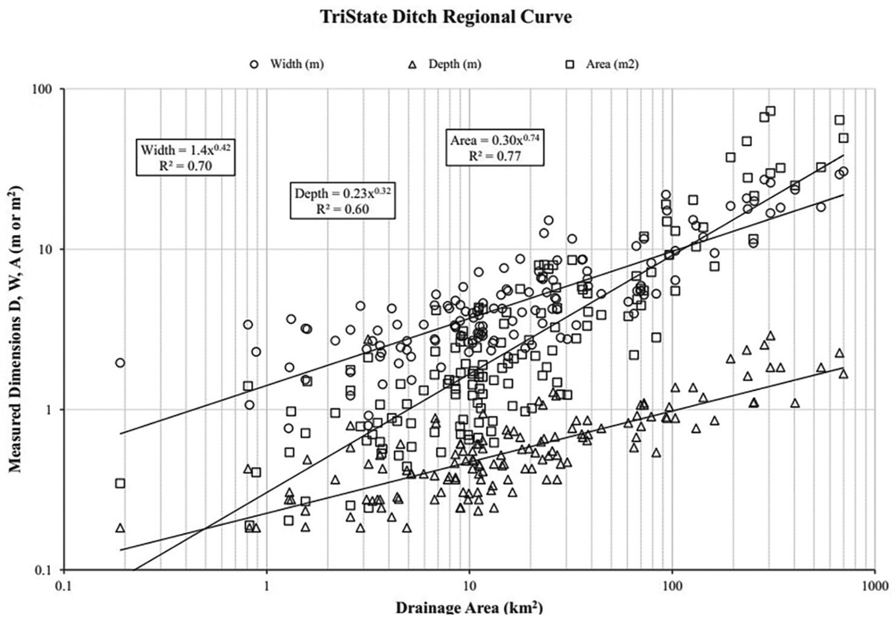

FIGURE 6.4
Regional hydraulic geometry relationships for agricultural ditches in Michigan, Indiana, and Ohio (Ward et al., 2013) that were used to size and evaluate geomorphic change in the two-stage ditch study sites.

TABLE 6.2

Summary of Daily and Annual Precipitation Events from Weather Stations within 15 km of Each Study Site That Occurred Since Two-Stage Ditch Construction

Site	Weather Station	Months	Daily 10–30 (mm)	Daily 30+ (mm)	Total	Annually 30+ (mm)	Largest Daily (mm)
Bull Creek Tributary	Findlay, OH	118	272	63	335	6.4	121
Crommer Ditch	Montpelier, OH	83	188	58	246	8.4	91
Klase Ditch	Versailles, OH	82	212	39	251	5.7	246
Shatto Ditch	Warsaw, IN	58	120	32	152	6.6	70
Creel Ditch	Angola, IN	49	120	26	146	6.4	66
Powers Ditch	Angola, IN	34	82	15	97	5.3	66
Ridenour Ditch	Angola, IN	34	82	15	97	5.3	66

depending on bench height and drainage area of the site. Mahl et al. (2015) report that wetter conditions resulted in twice as many flood days in 2011, and the lowest floodplains (i.e., Klase Ditch, Bull Creek, and Shatto Ditch) were inundated for more than 120 days. The largest precipitation events at each site ranged from 66 to 246 mm. Rainfall of 60–70 mm would approach a 5-year recurrence interval discharge event. Bull Creek tributary experienced an event in the 50–100-year range.

Geomorphic Change in the Inset Channel

Since construction, there has been some widening of most of the inset channels (Table 6.3 and Figure 6.5). Changes in the mean and maximum depths have been very small except for Powers Ditch, which has downcut slightly rather than widened. Bull Creek tributary, Klase Ditch, and Powers Ditch had inset channels that, at the time of modification to two-stage geometry, were narrower than regional curve predictions (Figure 6.5). This might be due to inherent uncertainty in the regional curve relationships. Bull Creek tributary and Shatto Ditch were divided into two separate reaches for statistical analysis. Each ditch had a major tributary entering about half way through the reach that resulted in a significant change to the ditch dimensions downstream. Percent changes in dimensions of the inset channel varied from 19% at the upstream reach of Shatto Ditch to 49% in the upstream reach at Bull Creek tributary (Table 6.3). The only inset channel changes that were statistically significant were increases in cross-sectional area, width, and D_{max} at the upstream reach of Bull Creek tributary and an increase in D_{max} at Crommer Ditch. The inset channel changes that were not statistically significant reflect natural adjustments to dimensions over time. Prior to constructing the benches, the inset channels were constrained by the trapezoidal ditch geometry and lack of attachment to wide benches. Changes in the dimensions of each inset channel were expected as they were not modified during construction.

Geomorphic Change below the Lower Datum and on the Benches

Each site experienced some aggradation or degradation on the constructed benches and inset channel (Table 6.4). The percent changes in dimensions varied from 27% in the upstream reach of Shatto Ditch to 19% in the upstream reach of Bull Creek tributary (Table 6.4). Except for these two reaches, only

TABLE 6.3

Bankfull (Area GHI, Figure 6.3) Geomorphic Data for the First Survey Post-Construction and the Most Recent Survey Post-Construction (Year); Number of Observations in the Sample (n), and Average Dimensions of Cross-Sectional Area (Area), Width, Maximum Depth (D_{max}), and Mean Depth (D_{mean})

Site	Year	n	Area (m²)	Width (m)	D_{max} (m)	D_{mean} (m)
Bull Tributary downstream	2004	10	0.92 (0.37)	2.16 (0.88)	0.45 (0.21)	0.32 (0.15)
	2012	10	0.78 (0.31)	2.72 (0.65)	0.48 (0.11)	0.29 (0.09)
		% Change	−15	26	7	−9
Bull Tributary upstream	2009	11	0.60 (0.22)	2.41 (0.41)	0.41 (0.10)	0.25 (0.06)
	2012	11	0.89 (0.20)	3.12 (0.70)	0.52 (0.09)	0.30 (0.08)
		% Change	49	30	27	17
Crommer Ditch	2004	19	1.45 (0.45)	3.69 (0.72)	0.62 (0.11)	0.39 (0.07)
	2010	20	1.65 (0.43)	3.93 (0.85)	0.69 (0.10)	0.42 (0.06)
		% Change	14	7	12	10
Klase Ditch	2009	24	0.71 (0.40)	2.79 (1.23)	0.43 (0.09)	0.25 (0.07)
	2012	29	0.76 (0.33)	2.91 (0.24)	0.45 (0.04)	0.27 (0.03)
		% Change	7	4	6	5
Shatto Ditch	2009	13	1.04 (0.44)	3.24 (0.74)	0.51 (0.16)	0.32 (0.11)
	2012	20	0.84 (0.41)	2.94 (0.63)	0.46 (0.14)	0.28 (0.10)
		% Change	−19	−9	−10	−13
Creel Ditch	2009	20	0.98 (0.55)	2.81 (0.99)	0.60 (0.18)	0.34 (0.12)
	2012	35	1.14 (0.62)	3.07 (0.91)	0.58 (0.17)	0.37 (0.13)
		% Change	16	9	−3	7
Powers Ditch	2010	13	2.20 (0.93)	4.25 (1.42)	0.75 (0.20)	0.52 (0.11)
	2012	14	2.38 (0.90)	4.25 (1.30)	0.86 (0.15)	0.56 (0.11)
		% Change	9	0.1	13	9
Ridenour Ditch	2010	20	1.24 (0.22)	3.23 (0.38)	0.60 (0.09)	0.39 (0.06)
	2012	21	1.32 (0.20)	3.31 (0.25)	0.64 (0.08)	0.40 (0.06)
		% Change	7	3	6	4

Note: Standard deviations are provided in parentheses.

the downstream reach of Shatto Ditch had experienced changes that were larger than 5%. None of the changes across sites were statistically significant.

When we partitioned out just the bench area (FGIJ), the largest changes were a 32% reduction in the upstream reach of Shatto Ditch and an 8% increase in the downstream reach of Bull Creek tributary (Table 6.5). A reduction is primarily associated with aggradation on the benches and an increase is primarily associated with degradation on the benches. There could be changes in the small bank surfaces (FG and IJ) in this zone. Of more meaning than percent changes in these areas are the depths and rates of aggradation and degradation. Dividing the change in average area (as a fraction) by the average width gave the depth of aggradation or degradation, which was divided

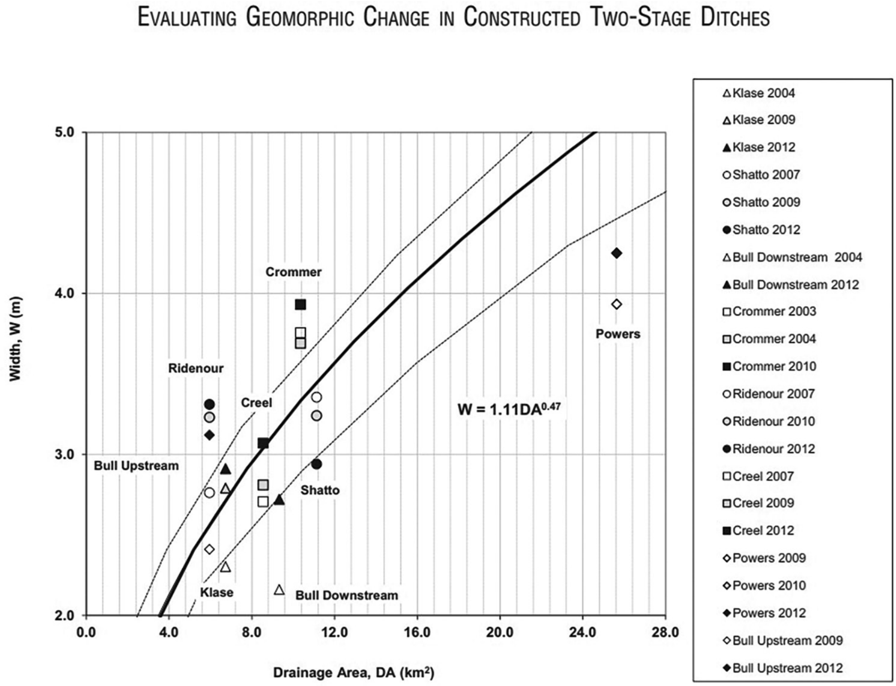

FIGURE 6.5
Inset channel width vs. drainage area for pre-construction survey and post-construction surveys* together with a common regional curve (solid line) and 95% confidence intervals (dashed line). *Bull Creek upstream did not have a pre-construction survey.

by the time period between the first and last year surveyed after construction to estimate an average annual rate of change that ranged from 13 mm/year of aggradation in the upstream reach of Shatto Ditch to 8.9 mm/year of degradation in Powers Ditch.

Geomorphic Changes below the Upper Datum and in the Banks

Results showed that 3–10 years after construction the constructed two-stage ditch sites had experienced small adjustments in their total cross-sectional area (Tables 6.5 and 6.6). The percent change in dimension varied from –8% at Klase Ditch to 18% in the upstream reach of Bull Creek tributary (Table 6.6). Shatto Ditch upstream section was not included in the bank analysis because the location of fixed depth line for this section did not provide enough bank area to evaluate. Statistically significant changes occurred in the upstream reach of Bull Creek tributary.

TABLE 6.4

Lower Data Fixed Width (Area FGHIJ, Figure 6.3) Analysis Showing the First Survey Post-Construction and the Most Recent Survey Post-Construction (Year); Number of Observations in the Sample (*n*), and Average Dimensions of Cross-Sectional Area (Area), Width, Maximum Depth (D_{max}), Mean Depth (D_{mean}), and Percent Change (%)

Site	Year	*n*	Area (m²)	Width (m)	D_{max} (m)	D_{mean} (m)
Bull tributary downstream	2004	10	2.08 (0.33)	6.14	0.78 (0.13)	0.34 (0.05)
	2012	10	2.00 (0.43)	6.14	0.77 (0.13)	0.33 (0.07)
		% Change	–4	0	–0.6	–4
Bull tributary upstream	2009	10	2.23 (0.39)	6.46	0.75 (0.13)	0.35 (0.06)
	2012	10	2.65 (0.61)	6.46	0.86 (0.15)	0.41 (0.10)
		% Change	19	0	15	19
Crommer Ditch	2004	19	3.41 (0.68)	9.74	0.89 (0.12)	0.35 (0.07)
	2010	20	3.29 (0.82)	9.74	0.91 (0.09)	0.34 (0.08)
		% Change	–4	0	2	–3
Klase Ditch	2009	27	4.26 (1.73)	9.11	0.91 (0.26)	0.47 (0.19)
	2012	29	4.04 (1.71)	9.11	0.90 (0.23)	0.45 (0.19)
		% Change	5	0	1	5
Shatto Ditch upstream	2009	12	2.28 (1.03)	7.73	0.55 (0.19)	0.30 (0.13)
	2012	13	1.68 (0.85)	7.73	0.57 (0.17)	0.22 (0.11)
		% Change	–27	0	3	-27
Shatto Ditch downstream	2009	8	2.11 (0.71)	7.27	0.76 (0.17)	0.29 (0.10)
	2012	8	2.16 (0.94)	7.27	0.76 (0.15)	0.30 (0.13)
		% Change	2	0	1	2
Creel Ditch	2009	19	8.44 (1.87)	13.12	1.30 (0.29)	0.65 (0.14)
	2012	39	8.73 (2.44)	13.12	1.33 (0.30)	0.67 (0.19)
		% Change	4	0	2	4
Powers Ditch	2010	14	8.97 (3.19)	10.23	1.63 (0.54)	0.88 (0.31)
	2012	14	8.77 (3.24)	10.23	1.69 (0.54)	0.87 (0.32)
		% Change	–2	0	4	2
Ridenour Ditch	2010	20	3.42 (1.00)	6.99	1.01 (0.18)	0.49 (0.14)
	2012	21	3.41 (0.97)	6.99	1.02 (0.18)	0.49 (0.14)
		% Change	–0.2	0	1	–0.2

Note: Standard deviations are provided in parentheses.

All the other sites had changes of 6% or less in any dimension, which were not statistically different ($p < 0.01$). The increase in dimensions at the upstream reach of Bull Creek tributary is partially due to degradation (scour) on the benches (Table 6.5). This reach is just downstream of a culvert so that might be influencing change. When we partition out just the bank area between the upper and lower datum, the changes range from a 17% increase in the upstream reach of Bull Creek tributary to an 8.8% decrease in Klase Ditch.

TABLE 6.5

Bank and Bench Area Differences and Percent Change (CD and BC on Figure 6.3, Respectively)

Site	Year	Months	Area (m^2)	Differences (m^2)	% Change	Area (m^2)	Differences (m^2)	% Change	Depth (mm)	Rate (mm/year)
Bull tributary downstream	2004	97	3.29			1.16				
	2012		3.09	–0.20	–6.1	1.22	0.06	5.2	9.8	–1.2
Bull tributary upstream	2009	38	1.79			1.65				
	2012		2.10	0.31	–17	1.78	0.13	8.0	20	–6.4
Crommer Ditch	2004	64	9.20			1.96				
	2010		8.94	–0.26	2.8	1.64	–0.32	–16	–33	10
Klase Ditch	2009	39	12.74			3.57				
	2012		11.64	–1.10	8.8	3.30	–0.27	–7.6	–30	5.6
Shatto Ditch upstream	2009	47	—	—	—	1.24				
	2012		—	—	—	0.84	–0.40	–32	–52	13
Shatto Ditch downstream	2009	36	4.56			1.11				
	2012		4.36	–0.20	4.4	1.00	–0.11	–9.7	–15	5.0
Creel Ditch	2009	36	15.63			6.24				
	2012		15.00	–0.63	4.0	6.40	0.11	1.8	–8.4	2.8
Powers Ditch	2010	37	4.24			7.73				
	2012		4.22	–0.02	0.5	7.45	0.28	3.6	27	-8.9
Ridenour Ditch	2010	37	2.00			3.42				
	2012		2.11	0.11	–5.5	3.41	0.01	0.3	1.4	0.5

Depth of aggradation or degradation, and rate are provided for the bench area (BC).

TABLE 6.6

Upper Datum Geomorphic Data (Area EFGHIJK, Figure 6.3) Showing the First Survey Post-Construction and the Most Recent Survey Post-Construction (Year); Number of Observations in the Sample (*n*), and Average Dimensions of Cross-Sectional Area (Area), Width, Maximum Depth (D_{max}), Mean Depth (D_{mean}), and Percent Change (%)

Site	Year	n	Area (m²)	Width (m)	D_{max} (m)	D_{mean} (m)
Bull tributary downstream	2004	10	5.37 (0.60)	8.48	1.22 (0.15)	0.64 (0.07)
	2012	10	5.09 (0.77)	8.48	1.19 (0.17)	0.61 (0.09)
		% Change	–5	0	–3	–5
Bull tributary upstream	2009	10	4.02 (0.56)	7.86	1.00 (0.16)	0.52 (0.07)
	2012	10	4.75 (0.60)	7.86	1.16 (0.15)	0.61 (0.08)
		% Change	18	0	16	18
Crommer Ditch	2004	19	12.61 (2.51)	13.05	1.70 (0.23)	0.97 (0.20)
	2010	20	12.23 (2.22)	13.05	1.69 (0.21)	0.95 (0.17)
		% Change	–3	0	–0.3	–2
Klase Ditch	2009	27	17.00 (3.00)	13.39	2.05 (0.34)	1.28 (0.23)
	2012	29	15.66 (2.61)	13.39	1.94 (0.25)	1.18 (0.20)
		% Change	-8	0	-6	-8
Shatto Ditch upstream	2009	—	—	—	—	—
	2012	—	—	—	—	—
		% Change	—	—	—	—
Shatto Ditch downstream	2009	8	6.67 (2.31)	11.66	1.23 (0.35)	0.58 (0.20)
	2012	7	6.52 (2.28)	11.66	1.21 (0.27)	0.56 (0.20)
		% Change	–2	0	–2	–2
Creel Ditch	2009	19	24.07 (2.74)	18.03	2.31 (0.31)	1.35 (0.15)
	2012	39	23.73 (2.57)	18.03	2.30 (0.25)	1.33 (0.14)
		% Change	–1	0	–0.5	–1
Powers Ditch	2010	14	13.21 (4.16)	12.28	2.02 (0.59)	1.09 (0.34)
	2012	14	12.99 (4.50)	12.28	2.07 (0.61)	1.07 (0.37)
		% Change	–2	0	3	–2
Ridenour Ditch	2010	20	5.42 (1.16)	8.39	1.27 (0.19)	0.65 (0.14)
	2012	21	5.52 (1.39)	8.39	1.30 (0.22)	0.66 (0.17)
		% Change	2	0	2	2

Note: Standard deviations are provided in parentheses.

Discussion

Geomorphic theory and recent research on bench formation in ditches would suggest narrowing and deepening of the inset channel after two-stage ditch construction due to frequent overbank flows depositing sediments on the benches and the establishment of vegetation on the benches (Jayakaran et al., 2005; Jayakaran and Ward, 2007; Powell et al., 2007b). Channel widening and aggradation could indicate recovery processes that occur in straight reaches consistent with Stage IV of the channel evolution model (Schumm et al., 1984; Simon and Hupp, 1986; Schumm, 1993) and might indicate a slightly different evolutionary starting point than was anticipated at the time of two-stage design and construction. Regional hydraulic geometry relationships are part of the design approach used to determine the elevation for constructed benches and can provide a useful guide in sizing the flooded width. In many places in the upper Midwest region, the inset channel dimensions for agricultural drainage ditches will be smaller than those of natural streams (Ward et al., 2013). However, in parts of Minnesota, the relationships developed for agricultural drainage ditches are similar to relationships developed for natural streams (Magner et al., 2012). Developing a watershed-specific regional set of regional curves for agricultural drainage ditches is key to a successful geomorphic two-stage design. Nearly all of the sites fell within the expected range for the inset channel width despite adjustments to the inset channel over time. It is quite possible, with half of the sites <4 years old at the time of this study, that thus far in the evolution of the two-stage ditches we are seeing periodic adjustments that do not reflect the final quasi-equilibrium geometry of the inset channel.

Crommer Ditch, Creel Ditch, and Klase Ditch each had total bench widths that were within two to three times the quasi-equilibrium bankfull width predicted from regional curves. All the other ditches had undersized constructed benches that were less than two times the quasi-equilibrium bankfull width predicted from regional curves.

There was evidence that the benches were built lower than designed at upper reach of Bull Creek and at Shatto Ditch or that one bench was built higher or lower than the other making determinations of the bankfull elevation and bench scour or deposition difficult. Frequent flooding on the benches can lead to vertical accretion (aggradation), the mechanism by which benches form (Landwehr and Rhoads, 2003; Jayakaran and Ward, 2007). Bench formation often is viewed negatively as the ditch "filling up" and threatening tile drain outlets. We did observe evidence of aggradation occurring on the benches at all of the sites except Bull Creek tributary and Powers Ditch. The highest rates occurred in the upstream reach of Shatto Ditch and at Crommer Ditch. At both sites, large amounts of fine sediment were observed in the ditches prior to construction. One of the motivating factors to construct a two-stage channel at Crommer Ditch was that the farmer was cleaning fine sediment deposits out of the ditch every 3–5 years. At Shatto Ditch, which is shallow and undersized for its drainage area, a large slug of fine sediments that deposited at the beginning of two-stage reach was observed the first year after construction. Two years of subsequent surveys showed that the ditch was able to transport much of those fine sediments; however, the upstream section of Shatto is still in the process of establishing its final equilibrium state. Further investigation into sediment transport processes was outside the scope of this study but is warranted in two-stage ditches especially as they evolve toward equilibrium. It is unlikely that deposition on the benches would threaten tile outlets at the rate they are occurring (0.5–13 mm/year) or that that vertical accretion will threaten tile outlets because the higher the benches become, the less frequently they will be inundated, and the rate of sediment deposition will slow as the ditches age. It is also important to note that these are average rates of change and are occurring along two-stage ditches that are 400–1,200 m long.

Benches that are constructed or formed lower than expected can be a design option of two-stage ditches and can facilitate sediment retention and denitrification (Roley et al., 2012a, b). Frequent inundation of the benches and soil-water-vegetation interactions that are vital to nutrient processing, coupled with landscape best management practices, has shown promise for reducing nitrate and phosphorus in constructed two-stage ditches (Roley et al., 2012a, b). Ongoing studies may further elucidate the benefits of two-stage ditches in reducing phosphorus, sediment, and returning ecological services and function to modified agricultural watersheds. We have observed, however, places where the depth of excavation resulted in the benches being constructed in the C horizon. At these sites, vegetation has been slow to establish and infiltration into the benches is slow.

Primary barriers to adoption of two-stage ditch designs for landowners and ditch managers have included: long-standing cultural expectations and a design standard of a drainage ditch as being trapezoidal, concerns about ditch stability and downstream flooding, and additional upfront costs for construction, among others. Lateral migration of the inset channel that could impact ditch banks was discovered to be minimal in a naturalized Illinois ditch but is still viewed as a threat to ditch stability by ditch managers (Rhoads and Massey, 2011). We did not see evidence of much lateral migration of the inset channel or impacts to ditch banks. We did observe scour on the side slopes in localized areas at Klase Ditch, Powers Ditch, and Creel Ditch. Average velocities were highest in these ditches, but the scour problems also were associated with poorly established vegetation, one-sided bench construction, and/or high flows from subsurface mains redirecting the flow of water from the main channel out on to the opposite bench. One-sided bench construction was performed in parts of three study sites due to various project constraints. In sections of Klase Ditch and in Powers Ditch, there was considerable scour or deposition where benches were constructed only on one side. There has been some success with one-sided construction in Indiana in places where establishment of vegetation after construction was rapid, but this modification to the two-stage design has not yet been proven to be a reliable alternative in terms of geomorphic stability. We would recommend that one-sided construction is avoided in future applications.

At all but one study site, the landowner has allowed a small amount of woody vegetation to establish in the two-stage ditch after construction. The use of woody or grass vegetation in riparian areas of two-stage ditches is under debate. Woody vegetation with larger roots tends to result in channels that are wider and shallower than streams bordered by grasses or prairie types of vegetation (Zimnierman et al., 1967; Davies-Colley, 1997; Lyons et al., 2000; Hession et al., 2003). The establishment of grasses on the newly constructed benches is a key component to their stability by increasing roughness on the benches and promoting channel narrowing (Friedman et al., 1996; Landwehr and Rhoads, 2003). Allmendinger et al. (2005) found that dense grasses on active floodplains created high sediment accumulation rates that counteracted the effect of high bank erodibility rates. This is an especially important design consideration when two-stage ditches are constructed at sites that have less cohesive soils or where sand lenses are prevalent. Successful stable bench construction has been achieved at sites with less cohesive soils, due in large part to particular attention paid to the rapid and dense establishment of grasses on the benches (Kent Wamsley, personal communication). Chemical or mechanical removal of woody and broadleaf vegetation has occurred in some of the constructed two-stage ditches. It is the only routine maintenance activity that has been performed since construction.

Conclusions

The two-stage ditch concept represents a potential paradigm shift in the way that agricultural drainage ditches and channelized headwater streams have been and could be managed with focus on naturalization, returning ecosystem function where it has been lost, and protecting valuable downstream water resources while sustaining a viable and productive agricultural economy. Results of this study provided no indication that the two-stage ditch design was not a viable management practice for maintaining or improving drainage capacity and reducing the need for frequent ditch clean-out. The results of this study and our observations of many naturalized and constructed two-stage ditches in the upper Midwest region suggest the two-stage ditch approach, where implemented at appropriate locations in the landscape, could result in stable, low-gradient ditches that are able to provide drainage function and require little to no maintenance. There was no conclusive evidence, however, that all seven of the two-stage ditches in this study were self-sustaining and had reached their quasi-equilibrium state thus far in their geomorphic evolution.

Acknowledgments

This study was funded by the U.S. Department of Agriculture's Natural Resources Conservation Service through the National Institute of Food and

Agriculture (NIFA) grant program, #2008-51130-04766. We thank Dr. Anand Jayakaran and Dr. G. Erick Powell for designing, implementing, and monitoring the Bull Creek tributary, Crommer Ditch, and the Klase Ditch two-stage projects and laying the foundation for understanding bench development in agricultural drainage ditches. We thank Rebecca Kallio, Sara Kallio, Matt MacFarland, Jamie Johnson, Ellie Kern, Marybeth Mayo, Natalie Struble, and Rob Davis for their assistance in data collection. We thank The Nature Conservancy in Indiana for their education and outreach efforts in Indiana and Michigan to implement the two-stage ditch projects. We especially thank the landowners who asked for and/or agreed to the implementation of the two-stage ditch concept on their properties.

Part of this contribution is based on "D'Ambrosio, J. et al. 2015. Evaluating Geomorphic Change in Constructed Two-Stage Ditches, Paper No. JAWRA-14–0046-P of the Journal of the American Water Resources Association" and used here by permission of the publisher American Water Resources Association.

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7

Air-Photo-Based Change in Channel Width Sedimentation in the Minnesota River Basin

J. Wesley Lauer, Christian F. Lenhart, Caitlyn Echterling, Patrick Belmont, and Rachel Rausch

CONTENTS

Part of this contribution is based on "Lenhart, C. F., M. L. Titov, J. S. Ulrich, J. L. Nieber, and B. J. Suppes. 2013. The role of hydrologic alteration and riparian vegetation dynamics in channel evolution along the lower Minnesota River. Transactions of the ASABE 56(2): 549–561" by permission from the publisher the American Society of Agricultural and Biological Engineers.

DOI: 10.1201/9781003175094-7

Introduction

The geometry of alluvial river channels depends strongly on water discharge and sediment loads delivered from upstream (ASCE Task Committee, 1998a, b; Savenije, 2003; Eaton et al., 2004; Parker et al., 2007; Kleinhans, 2010; Li et al., 2015; Call et al., 2017). Changes in channel width can thus be indicative of changes in discharge (Schumm and Lichty, 1963; Schumm, 1968), sediment supply (Trimble, 2009), or possibly changes in vegetation dynamics and/or bank cohesion (Tal and Paola, 2010; Dean and Schmidt, 2011; Eke et al., 2014). While extreme events sometimes cause reach-wide changes in channel geometry that may persist for years (Harvey, 2007; Konrad et al., 2011), the widespread acceptance of downstream hydraulic geometry relationships (e.g., Leopold and Maddock, 1953; Wilkerson and Parker, 2011; Knighton and Wharton, 2014, and references therein) implies that despite short-term variability, cross-sectional geometry depends on long-term conditions in regionally consistent ways. This raises obvious questions regarding the implications of potential future land use and climate change on the geometry of alluvial rivers such as: How well might existing hydraulic geometry relationships characterize geometry under new discharge regimes? Will the adjustment play out quickly, continuously, and uniformly? Will there be lags between perturbation and response?

In meandering rivers, changes in channel width occur because of complex interactions between vegetation, discharge, bed elevation, bank erosion, and sediment deposition. Vegetation strongly influences floodplain development and channel width by redirecting flow, trapping fine sediment, and strengthening sedimentary deposits (Hickin, 1984; Millar and Quick, 1993; Millar, 2000; Gran and Paola, 2001; Braudrick et al., 2009; Tal and Paola, 2010; Corenblit et al., 2014; Gurnell et al., 2016; Kleinhans et al., 2015). Rivers subjected to reductions in sediment supply or increases in sediment transport capacity often respond by incising, widening, and eventually creating an inset floodplain (Schumm et al., 1984; Simon, 1989). Conversely, channels subjected to overloading by bed material sediment can respond through bed aggradation and bar formation. These responses can increase bank erosion rates and/or flood flow rates on the floodplain, thereby stripping out vegetation and floodplain material (Burroughs et al., 2009; Madej et al., 2009; Zheng et al., 2014; East et al., 2015; Call et al., 2017). In either case, the finer fractions of eroded material are exported downstream, a process that makes channel widening a potentially large contributor to a river's sediment budget and an important water quality management consideration (Simon, 1989; Belmont et al., 2011; Schottler et al., 2014). By extension, systematic narrowing or aggradation of channels or storage of sediment in floodplains throughout a river basin may represent a significant sediment sink.

Identifying and Measuring Bankfull Width

When changing environmental conditions force change in channel geometry, the bankfull discharge (Leopold et al., 1964) also presumably changes (Wilkerson and Parker, 2011). Detecting such a change, however, is challenging because most definitions of bankfull discharge require field identification of banks or other indicators of recent channel occupation (Knighton and Wharton, 2014). Furthermore, the definition of a streambank can be ambiguous (Navratil et al., 2006), and even using a single definition, the top-of-bank elevation can be subject to significant spatial variability (Lauer and Parker, 2008). In many cases, particularly where a river is adjusting to new conditions, certain channel boundaries (e.g., the top of cutbanks along an eroding terrace) may depend more upon historic conditions rather than on contemporary river processes. Consequently, making a retrospective assessment of bankfull discharge is difficult, even in well-gaged rivers with historically surveyed cross sections.

In many instances, the characteristic width of a river channel is more readily apparent from above than from the ground. Aerial photographs of rivers that are not flowing *full* generally contain easily recognizable areas of bare sediment. The discharge that just begins to submerge all the unvegetated sediment in the active channel is similar conceptually to the traditionally defined bankfull discharge. In other words, the lowest discharge that fills the entire width of the channel, which we refer to here as the width-filling discharge (as opposed to the more traditional definition, which requires that the flow to vertically fill the channel up to a predetermined bankfull elevation) is a characteristic discharge that, once exceeded, can mobilize much or most of the sediment along a channel's boundary. Focusing on channel width has the advantages of: (1) straightforward measurements (i.e., viewing a reach from above and determining the width of the unvegetated zone is relatively simple) and (2) removing ambiguity introduced through field identification of banks. Drawbacks include the subjective definition of what constitutes bare sediment, as opposed to a surface colonized by perennial vegetation, and regional variation in the ability of vegetation (particularly fast-growing annual species) to colonize fresh sediment. In any case, temporal changes in driving variables such as discharge or climate are likely to shift the balance between processes of erosion, sediment deposition, and vegetation colonization that shape the boundary between channel and floodplain. Such changes should manifest themselves as measurable changes in the average width of a channel's unvegetated zone.

Mechanisms of Channel Width Change

For systems undergoing net channel widening, the rate at which widening supplies sediment to the downstream system depends on the overall bank-top widening rate, the depth of the channel, and whether adjustment occurs over the entire cross section. Figure 7.1 illustrates several possible geometric responses to a long-term increase in water discharge along a simple, single-thread river (see also ASCE Task Committee, 1998a, b). The simplest geometric response occurs when the channel boundary remains fixed while vegetation shifts vertically because of changes in flood frequency and duration that make near-channel areas more (or less) suitable for the hardiest species. Such a response, which we refer to here as mechanism a, could occur in relatively stable channels with boundaries consisting of immobile bedrock or lag deposits or in low gradient systems with relatively low coarse sediment supply and highly cohesive beds and banks. In the case of a long-term increase in discharge (Figure 7.1a), upward movement of the lower limit of vegetation would lead to an increase in the width of the unvegetated zone. In this case, the magnitude of width change would depend on the shape of the fixed channel boundary. While upward movement of the vegetation line itself does not change sediment flux, it does leave the sediment that is no longer protected by vegetation more susceptible to mobilization.

A second potential response to increased discharge would be for erosion to occur on both banks because of increased shear stress (Figure 7.1b). This mechanism (*mechanism b*) is most plausible along straight reaches with relatively stable centerline positions or curved reaches in which the outside bank is particularly resistant to erosion such as along valley walls or bluffs. As with *mechanism a*, this would lead to an increase in width of the unvegetated zone; however, in this case, the increase would be associated with a net transfer of bank sediment into the channel zone. A third mechanism (*mechanism c*) applies to actively migrating rivers that undergo significant centerline change while width adjustment occurs. In mechanism c (Figure 7.1c), cutbank erosion and deposition on the opposite bank occur in tandem, and the long-term difference between these leads to widening. As with mechanism b, width change by mechanism c is associated with net transfer of sediment to the channel. The size distribution of any sediment exported downstream would depend on the size distributions of sediment eroded from cutbanks as well as that deposited in point bars. In real river systems, channel width probably adjusts through all three mechanisms, but we maintain that mechanism c best represents the dominant driving force in the widening of meandering rivers. Mechanism a may act as an important feedback process that contributes to the channel widening process in mechanism c.

In this chapter, we estimate bankfull width change for extended reaches of the Minnesota River, MN, and several of its major tributaries using aerial

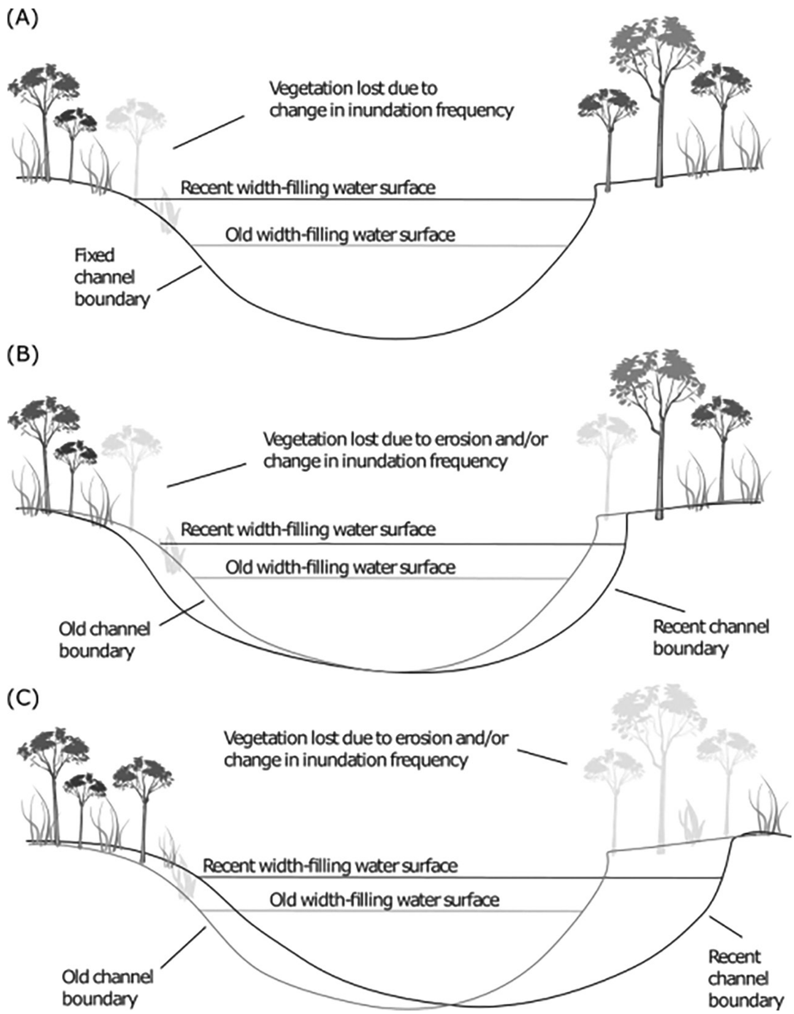

FIGURE 7.1
Conceptual representation of several modes of width change. Mechanisms include (a) change in channel bankfull geometry depends only on increased water level and change in vegetation line, (b) change in bankfull geometry depends on erosion of both banks, and (c) change in bankfull geometry occurs because of a change in overall cross-sectional geometry consistent with a migrating river.

photographs spanning 1938–2013. Even cursory review of aerial photographs shows that channel width has increased significantly in the lower Minnesota River valley in response to large increases in discharge that has occurred over the past century. For example, aerial photographs from 1940 and 2008 along the lower Minnesota River ~57 river km upstream from the confluence

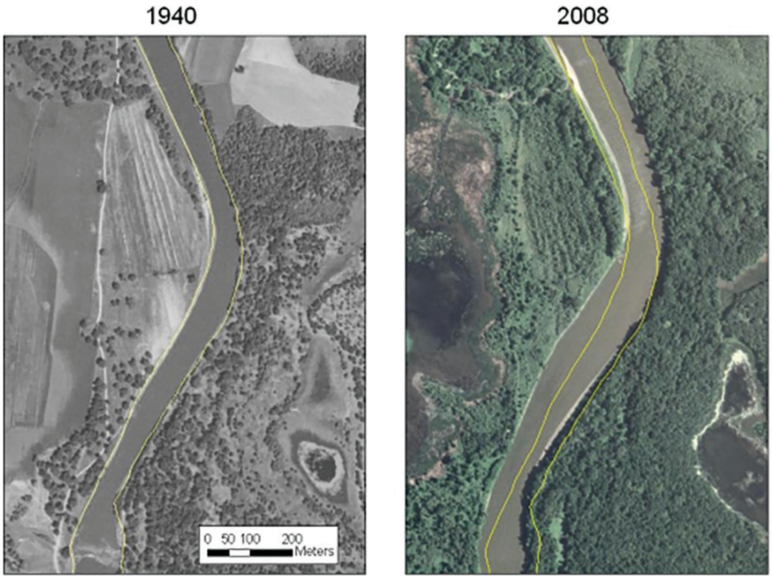

FIGURE 7.2
Historic and recent channel width for a low amplitude meander bend on the lower Minnesota River. Apex of bend is located at 44°43′58″N, 93°37′42″W. Also note the afforestation and apparent increase in overbank inundation in the 2008 image.

with the Mississippi River (Figure 7.2) illustrate that the channel has shifted laterally and widened in tandem, consistent with mechanism c. However, some reaches of the Minnesota River have experienced erosion on both banks (as in Figure 7.1b) and increases in discharge may have affected the elevation at which vegetation is able to colonize point bars (Lenhart et al., 2013). Consequently, the extent to which width change influences the sediment budget for the lower Minnesota River has not yet been determined. The objectives of this study include (1) estimating bankfull width change throughout the Minnesota River basin, (2) identifying which of the processes presented in Figure 7.1 best represents this change, and (3) quantifying and discussing the implications of width change for the sediment budget of the Minnesota River.

Regional Setting

The Minnesota River drains a 45,000 km^2 watershed that includes parts of Minnesota, Iowa, and South Dakota (Figure 7.3). It meanders across a 3–4 km wide valley that was carved during the Pleistocene by Glacial River Warren,

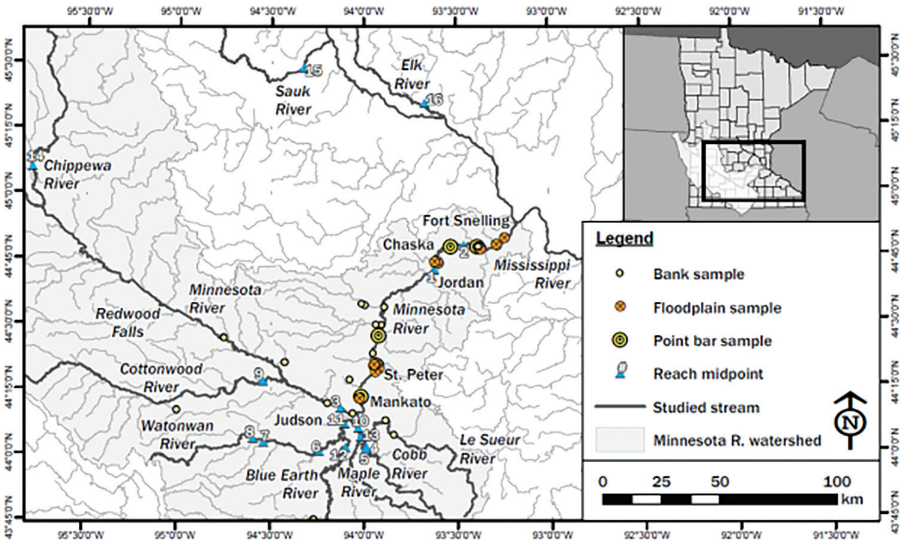

FIGURE 7.3
Minnesota River watershed and locations of width measurement subreaches.

an outlet of Glacial Lake Agassiz (Clayton and Moran, 1982; Thorleifson, 1996). Because the valley bottom is topographically much lower than the surrounding landscape, the lower portions of tributaries have incised through the substrate of consolidated glacial sediments, forming steep knickzones that mark the extent of incision since glacial times (Belmont, 2011; Gran et al., 2013). Bluffs along the lower reaches of major tributaries continue to provide a significant amount of sediment to the channel system (Thoma et al., 2005; Belmont et al., 2011; Day et al., 2013; Lenhart et al., 2013).

The Minnesota River has undergone large increases in water discharge over the past century and thus represents an interesting site to study the implications of long-term hydrologic change on channel geometry. It is also listed as impaired by high turbidity under section 303d of the U.S. Clean Water Act, meaning that the sediment budget implications of the geomorphic adjustment have important management implications (MPCA, 2009; Wilcock, 2009; Gran et al., 2011; Lenhart et al., 2013; Belmont and Foufoula-Georgiou, 2017). Belmont et al. (2011) conducted geochemical fingerprinting analyses on sediment cores from Lake Pepin, a naturally dammed lake on the Mississippi River downstream of the confluence with the Minnesota River. Results indicated dominance of near-channel sediment sources (banks and bluffs) ca. 500 YBP, when this landscape was mostly covered by tall-grass prairie and wetland. In addition, fingerprinting analyses documented a pulse of agricultural erosion in the mid-20th century followed by a more recent shift back toward near-channel sources in the past few decades. The same study compiled a sediment budget for the Le Sueur River, a major tributary of the Minnesota River, which corroborated the dominance of near-channel sediment sources during 2000–2010.

Prior to agricultural development, much of the Minnesota River watershed was grassland and contained large areas of poorly drained prairie (MCBS, 2007; Musser et al., 2009). Today, most of the watershed is intensively cultivated and includes extensive surface and subsurface drainage systems (Musser et al., 2009). Ditches now connect most of the closed basins to the river network, and subsurface tile drains allow for relatively rapid removal of moisture from the upper meter of the soil column. Average annual precipitation has increased slightly over the past few decades, but Schottler et al. (2014) and Kelly et al. (2017) showed that monthly precipitation amounts have not changed significantly in the May to June time period. This suggests that artificial drainage practices are driving the observed increases in flow during the sensitive time of year when soil pore pressures are high and bank cohesion is relatively low.

Regardless of mechanism, discharge in the Minnesota River basin has clearly increased significantly over the historical period (Novotny and Stefan, 2007; Lenhart et al., 2011, 2013; Schottler et al., 2014). For example, the USGS has maintained a stream gage on the Minnesota River at Mankato that represents one of the longest gage records available in the USA. While data were not typically recorded in winter prior to 1930, daily discharge is available for April through November beginning in 1903. At this site, mean April–November discharge for the second half of the 1903–2015 time period is just over twice the mean April–November discharge for the first half. Similarly, mean annual discharge after 1975 is essentially twice the mean annual discharge from 1930 to 1974.

Increases in flow are significant across the entire flow duration distribution, including high discharges responsible for doing geomorphic work and during spring and early summer, when vegetation typically colonizes point bars (Lenhart et al., 2013). Figure 7.4a shows the time series for the 1% exceedance probability discharge, as interpolated from all daily data in a given calendar year (starting 1930) or from April through November daily discharges (starting 1904). Exponential curves fit to the 1% exceedance data show similar trends for either data set, indicating year-on-year increases of $1.25 \pm 0.48\%$/year for the April–November data or $1.63 \pm 0.65\%$/year for the annual data, with error representing 95% confidence limits. Figure 7.4b shows a flow duration curve based on daily discharge for the 1930 through 1974 period and the 1975 through 2015 period. At essentially any exceedance probably, the ratio between these two distributions (Figure 7.4c) shows an increase of a factor of ~2–3, even in the gynomorphically important top 15%–20% of the distribution. Lenhart et al. (2013) and Foufoula-Georgiou et al. (2015) provided additional discussion of hydrologic changes affecting this site.

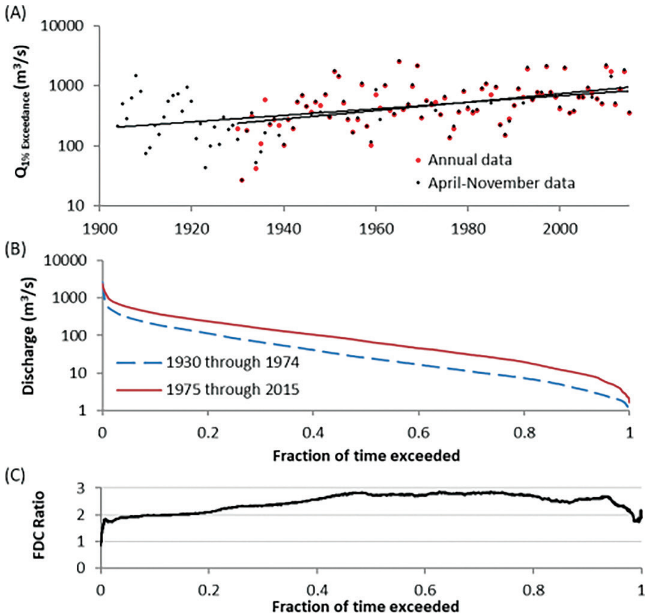

FIGURE 7.4
Changes in discharge at USGS 05325000 (Minnesota River at Mankato) over the period of record. (a) Timeseries of 1% exceedance probability daily discharge for entire calendar year or April–November period (April–November data extend further back in time), with best-fit exponential trend lines for each. (b) Flow duration distributions based on all daily discharge observed for 1930–1974 or 1975–2012. (c) Ratio of discharge at a given exceedance probability for the 1975–2012 period relative to the 1930–1974 period.

Methods

Measurements of Channel Width

Historical river channel width can be measured using either ground-based data or by aerial photograph analysis. When historic cross sections have been surveyed, the ground-based approach allows for the reconstruction of detailed sediment budgets (Trimble, 1997, 2009) and has the advantage of characterizing geometric change across the entire cross section, not just near the top of bank. However, ground-based methods are only feasible in cases where carefully surveyed historic cross sections are available and would not provide representative coverage without an extremely large number of sites.

Where ground-based surveys are not available, aerial photograph analysis has been used to document geomorphic changes in channel bankfull width and associated sediment fluxes. Buckingham and Whitney (2007) developed a sediment budget for a reach of the Las Vegas Wash, Nevada, using historic photographs to estimate channel volumes for three historic periods. Galster et al. (2008) used aerial photographs to analyze changes in width for two streams in the Lehigh Valley, Pennsylvania. Based on two photographic periods, 1946/1947 and 1999, they concluded that width change is discernable for streams ranging from 6 to 15 m wide given a sufficiently large sample size. Many other studies have used sequential aerial imagery to document channel change or estimate alluvial sediment loads on larger rivers (e.g., Martin and Ham, 2005; Aalto et al., 2008; Lauer and Parker, 2008; Belmont et al., 2011; Cadol et al., 2011; Konrad et al., 2011).

For the present study, channel width changes were assessed using aerial photographs spanning 1937 and 2015. Some of the aerial photograph-derived estimates of width change presented here were originally presented in Schottler et al. (2014). However, we have expanded the area under analysis and extended the length of the record to include 2015 measurements. In addition, we extend and improve upon the Schottler et al. (2014) results by using width change estimates, bathymetric data, and grain size measurement to develop first-order estimates of the widening-induced supply of sediment to the channel of the lower Minnesota River, while accounting for channel migration that occurs along this reach. We also extend beyond previous work by using regional hydraulic geometry relationships to determine which parts of the channel network (i.e., small tributaries vs. large mainstem channels) likely play the largest role in supplying sediment through the widening process.

Historic photographs were obtained from the Minnesota Department of Natural Resources and the John R. Borchert map library at the University of Minnesota. In most cases, the scale information for the original image had been stripped from the digital images. However, in general, these aerial surveys produced standard 9-inch by 9-inch (22.9-cm by 22.9-cm) panels with ground resolutions on the order of 1:20,000. Pixel resolutions are on the order of 2 m for the lowest resolution images and better than 1 m for most images, including most of those from the 1930s. While we recognize that scale-related effects may reduce precision in measurements based on the earlier images, we have no reason to believe such effects would bias measurements.

Photographs were georeferenced to the year 2009 National Agricultural Imagery Program (NAIP) aerial photographs using a minimum of ten ground control points and a second-order polynomial transformation (see Hughes et al., 2006). Note that any alignment error between historic and modern images would be unlikely to influence scale-related measurements such as width or channel planform area.

While automated procedures are available for the delineation of water/sediment boundaries (e.g., Merwade, 2007), light and dark regions associated with shadows and variable amounts of glare on both water and bare sediment prevented the bank delineation from being automated (Güneralp et al., 2013, 2014). Consequently, banks were manually delineated along each study reach. In general, the bank position was relatively obvious on the outside of meander bends, particularly where lighting conditions were good. However, some judgment was required when delineating banks located on freshly deposited point bars or where shadows or overhanging trees obscured cut banks. Where shadows or overhanging vegetation were an issue, the operator attempted to find points where vegetation clearly abutted water or unvegetated bar sediment. The bank line was then interpolated across the intervening visual obstructions. Operators attempted to place vertices in the bank lines at intervals of ~50% of the bankfull channel width. Operators were encouraged to regularly zoom in and out on the photograph in order to identify the best position for each point, but in general, points were digitized with no more than a single meander bend visible on the screen. To minimize systematic errors caused by subjectivity in bank delineation, we used the same GIS technician for as many points on a given study reach as possible.

Banks were defined through this process on 16 separate river reaches, each of which was ~10 meander bends in length. Sites were selected based on good visibility of the channel boundaries and representativeness of distinct process domains, such as above and within knickzones on major tributaries. We found the ten meander bend length scale to be sufficiently long to average out subreach variability, yet sufficiently short to minimize systematic increases in width in the downstream direction. Average width for each reach was computed by dividing total bank-to-bank planform area by the reach's centerline length.

For all but one reach, the oldest photograph was taken during the late 1930s. Because the relative importance of error in identifying bank position increases as overall channel width decreases, we focused primarily on the largest tributaries of the Minnesota River and did not consider channels narrower than about 10 m. Field observations suggest that many of these smaller channels have widened less, possibly because of strong root cohesion associated with mature woody vegetation exposed in both banks and because of relatively low energy gradients (average slope ca. 0.0005 m/m) above the knickzones on tributaries.

Average rates of increase in width were derived for each of the 16 study reaches by computing the slope of a simple linear regression line placed through the raw width vs. time data for each reach. Comparison between reaches was facilitated by also computing width change rates relative to the average width during the 2000–2009 period. Spatial variability outside of the 16 sampled subreaches was characterized by measuring channel width at evenly spaced intervals of 40 m (approximately ½ channel width) along the majority of the lower Minnesota River below Mankato for 1938 and for 2008 (the uppermost 20 river km, based on the 2008 channel centerline, were not available at the time of analysis). Work was performed using the Planform Statistics Toolbox (Lauer and Parker, 2008, available at www.NCED.umn.edu, last checked February 28, 2022), which develops centerline points at evenly spaced intervals between the manually digitized bank lines. Width was estimated at each centerline point by measuring the distance to the nearest part of each bank polyline. Width change between the 1938 and 2008 photo sets was assessed at each point by tracing a trajectory from the 1938 centerline to the 2008 centerline and then subtracting the 1938 width from the 2008 width at corresponding points. While the approach does not account for the total change in channel bankfull area because of channel sinuosity change between the two dates associated with several engineered bend cutoffs and increased rates of natural cutoffs (Lenhart et al., 2013), it provides a simple mechanism for characterizing spatial variability in width change and for relating this to overall lateral channel activity.

We quantified the area reworked via channel migration as the summation of lateral offsets between the 2008 and 1938 centerlines multiplied by centerline point spacing (40 m). However, some of the centerline offset occurred near 11 bend cutoffs that formed between 1938 and 2008 and thus does not actually represent area reworked by channel change. To develop a better estimate of the area reworked due to lateral change; we recomputed the offset while excluding the bends affected by cutoff.

Because it is theoretically possible for bankfull width to change even without real geomorphic change (e.g., mechanism a in Figure 7.1), we tested whether change occurred at stages below bankfull by using the historic aerial photographs to estimate average water surface width for all dates when a daily stream discharge was available from a nearby stream gage. The analysis was performed at three locations on the mainstem Minnesota River: Judson (upstream from Mankato), Jordan, and Chaska. In some cases, the photographs covering the respective reaches were taken several days or weeks apart. In these cases, a representative discharge was computed by weighting the daily discharges on the dates of each photograph by the relative length of the reach included in the photograph. Discharge data from the Jordan gage (USGS 05330000) were used for the Jordan and Chaska study reaches. Discharge at the Judson study reach, which is located several kilometers upstream from the confluence of the Blue Earth and Minnesota Rivers, was estimated by subtracting the sum of the gaged discharge on the Blue Earth River near Rapidan (USGS 05320000) and the Le Sueur River near Rapidan (USGS 05320500) from the daily discharge on the Minnesota River at Mankato (USGS 05325000). Discharge was not available for the Blue Earth or Le Sueur Rivers in 1938, so the discharge associated with the 1938 photograph of the Judson study reach was estimated by regression between discharge of the Minnesota River at Mankato and the estimated discharge of the Minnesota River at Judson for all dates after gaging began on the Blue Earth and Le Sueur rivers.

In each set of historic aerial photographs for which a discharge could be estimated, the average water surface width in the study reaches was determined by visually delineating the edge of water in the photograph, which was usually easily identifiable, particularly at low flow stage, and then dividing the wetted planform area by the overall reach length.

Hydraulic Geometry/Channel Cross-Sectional Area

We validated our in-channel water surface width estimates by developing a power-function relationship between water surface width on the date of a given photograph and discharge. We compared historical measurements of this relationship (prior to 1975) to recent measurements (2000–2009) to see whether change occurred on average at below the bankfull level.

We also used flow measurements at the Jordan gage and width measurements at the Jordan and Chaska study reaches to develop estimates of the bankfull cross-sectional area representative of the 2000–2009 period. While 2010–2015 data were available, high flow events in 2010 and 2011 widened the channel considerably. The process involved (1) using flow measurements at the gage to derive a power-function relationship between discharge and cross-sectional area for all discharges between 75 and 750 m^3/s and (2) independently relating the aerial photograph-based water surface width in our study reaches to discharge for all 2003–2009 photographs and then evaluating at the average 2003–2009 total bankfull width. The resulting width-filling discharge can be interpreted as the discharge that just submerged the unvegetated zone during the 2003–2009 period. We then evaluated the relationship developed between discharge and cross-sectional area developed in (1) with the width-filling discharge estimate made in (2), thereby providing the cross-sectional area at the Jordan gage for flow conditions similar to those that just fill banks in our study reaches.

We compared the photo-/gage-based cross-sectional area estimates to field-measured areas derived from a combination of LiDAR and multi-beam River Ray Acoustic Doppler Current Profiler (Teledyne RD Instruments, Poway, CA) bathymetric measurements. The depth below bankfull surface was constructed at a point density of ~3 points/100 m^2 along 33 km of the mainstem near the confluence with the Blue Earth River, ~16 km upstream and ~17 km downstream from the confluence. Bankfull cross-sectional area was computed by dividing the volume below the bankfull datum by the centerline length of the reach.

We extrapolated bankfull cross-sectional areas to the entire watershed-wide channel network by combining our mainstem Minnesota River cross-sectional area estimates with several other existing data sets to generate a functional relationship between channel cross-sectional area and drainage area. Regional hydraulic geometry data surveyed throughout Central Minnesota by Magner and Brooks (2007) were supplemented by the reach-average cross-sectional areas described above along with 54 cross sections that were measured in the field over a total of 160 km on the Maple and Le Sueur rivers in summer 2008 using an Impulse laser rangefinder and stadia rod (Belmont, 2011). Bank position was identified based on breaks in slope at the geomorphic transition between channel and floodplain. Data from Magner and Brooks (2007) are also based on field-surveyed cross sections in riffles near USGS gage sites, with bankfull defined using field indicators.

Flow lines from the National Hydrographic Dataset (NHD) were rasterized and burned into a Shuttle Radar Topography Mission (SRTM)-based DEM (CGIAR-CSI, 2008) of the watershed with cell size of 80 m. Flow direction and flow accumulation were computed in ArcGIS and were used to estimate the drainage area at the upstream end of each link in the NHD flow line network. The associated cross-sectional area for each link was then computed using a regional hydraulic geometry equation derived from the channel cross-sectional area data sources described above.

Sediment Size

Particle size information was collected as part of several separate studies. Data describing banks along the Minnesota River were obtained from samples collected at six separate sites evenly distributed downstream from Redwood Falls. Four samples were collected at each site, three spaced evenly along the face of the bank and one from the top of the bank (Hansen et al., 2010). Floodplain particle size samples were collected beyond the streambanks within the interior of the floodplain from the surface down to a depth of 2 m (Lenhart et al., 2013). Point-bar samples were collected at five separate sites on the mainstem Minnesota River downstream from Mankato. At each site, a soil auger was used to collect soil samples at depths of 0–25 and 25–50 cm. Sample pits were spaced evenly across the point bar in a transect perpendicular to the river from the water line to the mature tree line. The number of samples collected at each site ranged from seven to nine depending on the width of the bar, resulting in a total of 41 sample pits. Particularly near the tree line, some of the point-bar sites were in the process of being colonized by woody vegetation such as sandbar willow (Salix interior) and may thus include higher fractions of silt and clay than are characteristic of fresh point-bar deposits. Grain size distributions were obtained by sieving and hydrometer analysis, respectively, for point-bar and floodplain samples. Sediment sample sites are shown in Figure 7.3.

Results

Width Change Measurements

Average linear trends in channel width (m/y) for each of the 16 study reaches were calculated. These temporal trends, which in many cases are not statistically significant at the 95% confidence level, are nevertheless all positive except for reaches with average 2000–2009 widths below 25 m. Data were collapsed by normalizing the width measured from each photograph in a given reach by the average of all widths for that reach measured between years 2000 and 2009. These normalized data show that for (1) all reaches on the mainstem Minnesota River, (2) all tributaries with 2000–2009 width > 25 m, and (3) all tributaries with 2000–2009 width < 25 m. Linear and exponential curves were fit to the relative change data and provide results that are sufficiently similar that only the linear fits are shown in Figure 7.5. Results for both regressions (i.e., annual rates of change relative to the average 2003–2009 width from the linear fit, or annual rates of change computed over the entire study period from the exponential fit) show that width increase is apparently greater for the mainstem Minnesota River and larger tributaries than for the smaller tributaries. At least for the reaches located on the mainstem Minnesota River below Mankato, there do not appear to be any periods when width increased at a rate that was noticeably higher than the long-term average.

Sources of error in width estimation include photographic misinterpretation caused by shadows and poor lighting conditions as well as natural seasonal and year-to-year variability in the vegetation that was used as the indicator of bankfull position. We assessed error in our estimates by computing the RMSE between the average widths for a given reach estimated during the 2003–2009 period. We excluded post-2009 data because 2010 was a particularly wet year, with the second-largest mean annual discharge in the Mankato flow record. While the RMSE for 2003–2009 width also includes what may be real width changes, it provides a reasonable upper limit on the precision of our methods. For the 12 reaches where at least four width estimates were available between 2003 and 2009, the RMSE with respect to the mean 2003–2009 width for the respective reach is 2.3 m ($n = 66$). Particularly for the larger channels in the data set, the magnitude of change between the late 1930s and 2008 is well above this error estimate.

Widening and overall centerline position change for the downstream-most 146.5 km of the mainstem Minnesota River between 1938 and 2008 are shown in Figure 7.2. In general, width change is highly variable, although very little of the reach (<1% of total channel length) experienced a net decrease in width between the two dates. The most significant decrease occurred 105 river km upstream from the Mississippi River in an area that was shortened significantly by a natural bend cutoff that occurred between 1998 and 2003. The overall average width for the entire reach increased from 69.9 to 101.1 m between 1938 and 2008. This represents an average annual rate of 0.44% of the recent (2008) width per year (0.53%/year if compounded annually), which is similar to the rate obtained from the subreach-based analysis along all mainstem Minnesota River study reaches (0.52% of mean 2003–2009 width/year or 0.62%/year if compounded annually). Furthermore, for this 146.5 km reach, the overall amount of widening closely follows the overall rate of lateral offset (excluding cutoffs), implying that widening and lateral change from meander migration occur in tandem, consistent with mechanism c in Figure 7.1.

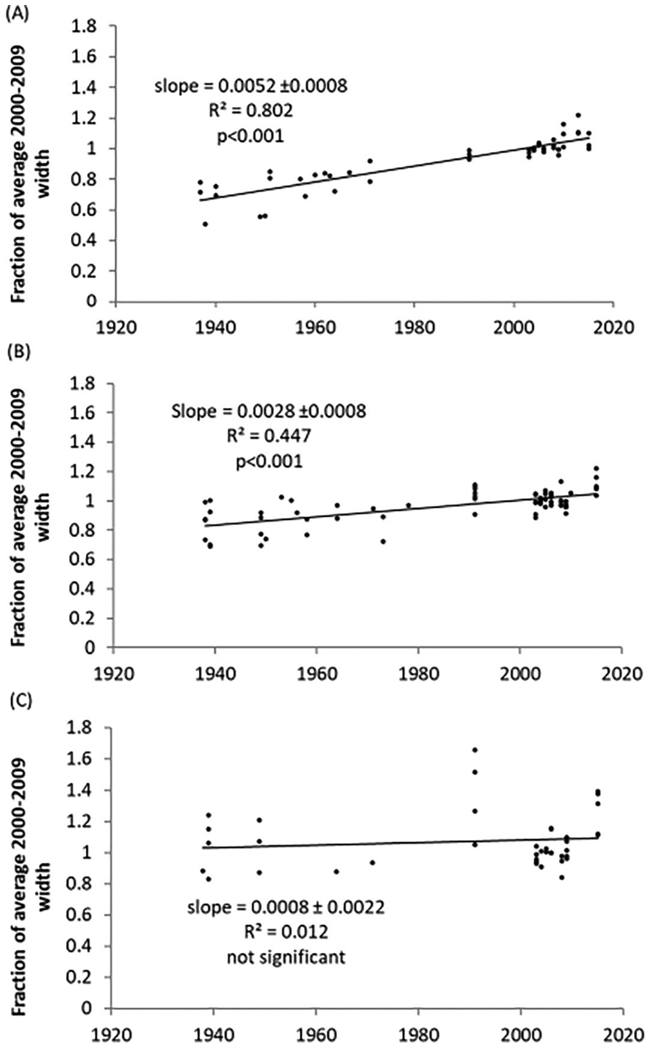

FIGURE 7.5
Trends in normalized width over time. Data are plotted separately for (a) mainstem Minnesota River reaches; (b) tributary reaches with mean 2000–2009 width > 25 m; and (c) reaches with mean 2000–2009 width < 25 m.

Temporal change illustrates that widening has occurred consistently throughout the watershed and not just in isolated reaches. Despite some bend-to-bend variability, the increase is distributed relatively uniformly along the entire lower Minnesota River. The results show that at least for the lower Minnesota River, widening appears to occur most rapidly upstream from Jordan, where the total along-channel change in centerline position is greater than the total along-channel change in width. In other words, above

Jordan, widening appears to be associated with a process more similar to that illustrated in Figure 7.1c—where the primary mode of channel change is meander migration, and widening occurs simply because of a disparity in erosion and deposition rates.

The widening process is less clear downstream of Jordan, where, except for a few bends within five km of the confluence with the Mississippi River that were cut off artificially in the 1960s for the purpose of improving commercial barge navigation, the overall areal change since 1938 due to widening has been roughly equal to the overall along-channel accumulated change in centerline position. Here, lateral centerline change is not sufficiently high for widening to have occurred exclusively along actively migrating meander bends. In other words, mechanisms a or b of Figure 7.1, neither of which results in net centerline position change, could be partly responsible for the widening.

Change in Hydraulic Geometry

Estimating the net export of sediment associated with widening requires some assumptions regarding the mode of change illustrated in Figure 7.1. If the observed increases in width of the unvegetated zone are primarily caused by changes in the vertical position of vegetation resulting from increased flood frequency or changes in flood timing (e.g., mechanism a), width increase may not result in much sediment export. However, if the geometry of the entire section changes (mechanisms b and/or c), then width change should be associated with an overall increase in bank-to-bank volume and thus should influence the sediment budget for the reach. Because repeat cross section or bathymetric surveys are generally unavailable except for recent periods or at gage locations and because the three USGS gages along the lower Minnesota River are built near relatively stable cross sections, the at-a-station hydraulic geometry for water surface width is one of the few available approaches for evaluating whether change occurred below the bankfull level during this period.

For at-a-station analysis for the pre-1975 and post-1975 periods, along with power-function regressions representing relationships between average water surface width and discharge for each reach were demonstrated as results. Note that except for a single data point from 1991, all post-1975 data are for photographs taken after the year 2003 and are thus representative of relatively recent conditions. In all cases, the power-function representing width vs. discharge for the later period plots was well above the power function for the pre-1975 period, indicating that the water surface width was typically much greater for a given discharge during the more recent period. Determining whether either of these results are statistically significant is complicated by the lack of available data for the pre-1975 period, the larger discharges during the post-1975 period, and because change in the cross section probably occurred continuously throughout each period. However, the figure supports the idea that water surface widths have increased through time even at relatively low discharges, implying that cross-sectional enlargement like that shown in Figure 7.1b has occurred throughout the lower Minnesota River valley. Because the width change is greatest at Jordan and Judson, in reaches where the channel migrates regularly, an imbalance between cut bank erosion and point-bar deposition appears to be the most likely mechanism for the enlargement.

Cross-Sectional Area Estimates

Estimates of width-filling discharge were calculated for each of the three mainstem Minnesota River study reaches as well as the corresponding cross-sectional areas for the Jordan and Chaska reaches. Bathymetric survey-based estimates of bankfull cross-sectional area were also presented for 16 km of the mainstem Minnesota River immediately upstream from the confluence with the Blue Earth River (i.e., roughly the Judson study reach), and 17 km immediately downstream from the Blue Earth confluence, near Mankato. We note that each of the cross-sectional area estimates derived from the width-filling discharge depends on the discharge-area function at the Jordan gage site, which is located in a relatively stable location that could result in smaller cross-sectional areas than are typical for the reach. In any case, either approach (width-filling discharge-based estimates or reach-scale bathymetric surveying) results in cross-sectional areas that are somewhat larger than cross-section-based estimates of average cross-sectional area used by Lenhart et al. (2013).

Size Distribution Data

Grain size fractions for material sampled from eroding cutbanks and point bars demonstrated that the lowest sand fractions were observed in floodplain samples collected along the main Minnesota River valley downstream from Mankato. Samples collected from cut banks in this part of the valley contained similar sand fractions but were somewhat less clay-rich than the floodplain surface deposits. In both cases (cutbanks and floodplains), average sand fractions were between 40% and 50%. Cutbanks on tributaries were sandier, with average sand fractions exceeding 60%. Point bars on the lower river were the sandiest of all observed deposits, with average sand fractions exceeding 80%.

Bankfull Volume Estimates

Our regional compilation of cross-sectional area and drainage area shows while there is a lot of scatter, perhaps because of differences in climate through the watershed or differences in geomorphic processes occurring above and within knickzones in tributaries, a single power function represents (equation 7.1),

$$A_{xs} = 0.164 A_D^{0.82} \tag{7.1}$$

where A_{xs} represents bankfull cross-sectional area (m^2) and A_D represents drainage area (km^2), appears to hold reasonably well across a wide range of scales, from drainage areas as small as 1 km^2 up to the entire basin area. However, note that primarily because of the inclusion of the Maple River data set (Belmont, 2011), which was characterized by particularly large cross-sectional areas, the regression results in an apparent positive bias for the largest drainage areas.

Maps were made of the results of cross-sectional area analyses across the stream network, using three separate classes of drainage area: below 640 km^2, between 640 and 10,000 km^2, and above 10,000 km^2. The lower threshold was selected because our width estimates in the Le Sueur River basin show that 640 km^2 is about the threshold at which the width of the active channel falls below 25 m. As shown in Figure 7.5, at these widths, our data do not show any measurable widening, although we are unsure whether the lack of a significant trend for these small channels is from lack of resolution in our methods. The threshold of 10,000 km^2 conveniently delineates most of the mainstem Minnesota River. Channels with drainage areas between the thresholds typically are large, named tributaries.

The potential for widening-related sediment supply presumably correlates with the parts of the channel network that contain the most bankfull volume. We estimate bankfull volume by multiplying the streamwise length of each link by the cross-sectional area estimated using equation 7.2.

Well over half of the total of 0.7 km^3 bankfull volume in the basin was represented by the mainstem Minnesota River below the Pomme de Terre River. Another roughly 23% is represented by the main trunks of major tributaries. Small channels narrower than 25 m account for only around 15% of the overall channel volume in the basin. Note that this result is similar even if we exclude Maple and Le Sueur river data from the regression, which would eliminate the apparent positive bias in estimated cross-sectional area for the largest drainage areas. This more conservative estimate reduces the overall volume for the basin by roughly a factor of two, but the mainstem still represents the majority of the bankfull volume. For this reason, Figure 7.5 should be interpreted as providing a general description of the relative position of channel volume within the watershed, but it should not be used as a definitive estimate of the watershed-wide total. Additional bathymetric surveys along the mainstem Minnesota River upstream from Mankato would be necessary to increase confidence in the overall volume estimate.

Discussion

Temporal Change in Hydraulic Geometry

The well-documented, large, and relatively continuous change in discharge in the Minnesota River basin over the 20th and early 21st centuries makes this system a useful test case for river response to environmental change. If there are no significant lags between adjustment in driving variable (discharge Q) and response variable (width W), the standard power-function form for hydraulic geometry for width (equation 7.2),

$$W = aQ^b \tag{7.2}$$

implies that the ratio of bankfull widths measured at two separate times, W_2/W_1, can be found from the ratios of driving discharges at the corresponding times:

$$\frac{W_2}{W_1} = \left(\frac{Q_2}{Q_1}\right)^b \tag{7.3}$$

Equation 7.3 makes the simplest possible assumption, that coefficient a and exponent b remain constant throughout the adjustment process. Church (1995) showed that equation 7.3 can represent geomorphic change over decadal timescales, particularly on systems that experience changes in formative discharge large enough to mobilize bed material during the adjustment period. The changes on the Minnesota River documented here represent one of the longest timescale tests we are aware of for evaluating the fluvial response to a long-term increase in discharge.

As discussed above, perhaps the simplest way to characterize long-term change in formative discharge along the lower Minnesota River is to use simple exponential curves. While this clearly neglects the important physics behind watershed-scale change, it provides a growth rate that can be used to place our width change estimates in an appropriate context. In the case of the Minnesota River at Mankato, the annual growth rate in the 1% exceedance daily discharge for the 1930–2015 period gives $Q_2/Q_1 = 1.0163 \pm 0.0065$. Similar annual growth rates result when using other potential representations of formative discharge, such as mean annual discharge or discharge at other exceedance probabilities.

Using standard error propagation (95% confidence limits) and the frequently cited value of 0.5 for the hydraulic geometry exponent b (Knighton and Wharton, 2014), the error in the annual increase in width should be half the error in the annual increase in discharge. This implies that width should have increased over this period at an annual rate of $W_2/W_1 = (1.0163 \pm 0.0065)^{-0.5} = 1.0081 \pm 0.0033$. This derived annual rate of increase in width, $0.81 \pm 0.33\%$/year, is within the error of the observed annual change in width for our mainstem Minnesota River study reaches, which was computed to be $0.62 \pm 0.10\%$/year based on a semi-log regression of the normalized data. The similarity between the rates predicted by hydraulic geometry and measured from historical images suggests that the lower mainstem of the Minnesota River has remained near an equilibrium between width and discharge during much of the past century. Thus, continued widening over the past decades may be primarily due to recent increases in discharge and not to long-term lags in geomorphic response. However, the lower Minnesota River is not presently capable of transporting all sediment supplied to it from upstream (MPCA, 2009), implying that it may be undergoing long-term depth adjustment. Some of the adjustment depends at least partly on storage of sediment on the floodplain, as described by Wilcock (2009) and Lenhart et al. (2013) and elaborated upon below. This raises the possibility that geometric adjustment of channel depth could lag well behind adjustment of width, particularly in aggrading systems.

Basin-Wide Implications

Near-bank sediment sources have become an increasingly important component of the sediment load of the Minnesota River in recent decades (Belmont et al., 2011). Some of this sediment originates in bluffs that are present along tributaries, particularly in the Blue Earth and Le Sueur basins (Sekely et al., 2002; Thoma et al., 2005; Day et al., 2013; Belmont et al., 2014; Schaffrath et al., 2015). Our results indicate that in addition to these localized sources, channel widening is probably occurring consistently at many locations in the basin. Furthermore, the ata-station hydraulic geometry analysis supports the idea that widening is associated with net geomorphic change. An obvious question, then, is whether basin-wide changes in channel geometry could be responsible for significant amounts of sediment production, and if so, where in the basin the effect would be strongest.

Our observed increases in channel width are greatest for the widest channels. Furthermore, our hydraulic geometry analysis shows that overall cross-sectional area and thus overall bankfull channel volume are strongly weighted toward the highest order channels. Consequently, any sediment production caused by widening would appear to be most important on the mainstem Minnesota River and its main tributaries. This is likely the case even if the resolution limitations of our analysis caused us to miss widening along channels below 25 m in width. Small, low-order channels simply do not account for enough cross-sectional area for widening there to outweigh sediment production on larger, higher-order channels. However, this should not be taken to mean the overall sediment contribution from low-order streams is universally small. On the contrary, particularly in systems undergoing rapid hydrologic change, net incision, and/or net erosion into terraces along even small channels may represent very large components of sediment budgets (Simon, 1989; Trimble, 2009; Stout et al., 2014). Our analysis does not consider these other potential, non-widening-related sediment inputs, although previous studies have shown that channel erosion high in the network is probably not a major sediment source (Gran et al., 2011; Belmont et al., 2014).

Despite the relative paucity of bankfull volume along the lowest-order segments of the stream network, observations indicated the channel network may contain as much as 0.5 km^3 of channel volume upstream from Mankato. Most of this volume occurs in larger tributaries and the mainstem Minnesota River itself. Accepting this volume as plausible, and assuming it is enlarging at the average widening rate for all our sub reaches, 0.36%/year, then the overall annual volume of sediment supplied to the channel network would be on the order of $0.5\ km^3 \times 0.0036 = 1.8 \times 10^6\ m^3$/year. This calculation assumes that widening is occurring because of enlargement of the cross section rather than vertical change in the vegetation line, and it also makes the relatively conservative assumption that depth has not increased along these channels through bed incision. Presumably, a large influx of sand from widening would lead to bed aggradation, not degradation, although we are not aware of any analysis in this part of the basin that has quantified long-term bed elevation change. If incision is occurring instead, our volume estimate would necessarily be larger. In any case, at a bulk density of 1.35 Mg/m^3, our estimated volume represents around 2.4×106 Mg/year of sediment produced simply from widening. While this number admittedly depends on the regional hydraulic geometry and could be improved significantly with additional cross-section surveys on the mainstem Minnesota River upstream from Mankato, its magnitude is clearly large relative to observed loads. Ellison et al. (2014) estimated an annual silt/clay load at Mankato of 1.16×106 Mg/year based on suspended sediment concentration measurements. Even if only 40% of our widening-related sediment supply is finer than sand and thus travels as washload, judged a reasonable estimate from data, widening probably represents a significant near-channel source of sediment. Furthermore, taken together with other well-documented sediment sources such as bluffs, surface erosion, ravines, etc., the results imply that there is probably a relatively large sediment sink distributed throughout the system (c.f. Beach, 1994), even upstream of Mankato. This sink is probably strongest for sand-size and larger sediment, but even silt/clay loads could be influenced by net storage along channels and in floodplains.

Significance of Widening-Related Sediment for Lower Minnesota River

The consistently large increase in width along the mainstem Minnesota River, together with its relatively important contribution to overall bankfull volume in the basin, indicates that widening here has probably represented an important transfer of sediment to the channel during much of the 20th century. Here we consider the nature of the sediment transfer and the implications it may have on the sediment budget for the reach.

Lenhart et al. (2013) estimated a gross widening-based sediment supply for the lower Minnesota River of 280,000 ± 56,000 Mg/year. This estimate is based on the overall change in channel planform area between 1938 and 2009 and a bank height of 3.2 m that was derived from surveyed bank profiles at seven cross sections. However, this gross sediment production estimate may be somewhat low because the cross-sectional areas we estimate here are larger than those available to Lenhart et al. (2013) and because our widening rates, which are based on a more extensive dataset, are also somewhat larger. Assuming for the time being that average channel depth remains constant, consistent with Lenhart et al. (2013), using the average of the three cross-sectional area estimates, 522 m^2, and using the average widening rate for the mainstem Minnesota River as 0.52%/year (relative to average 2003–2009 width), the overall volumetric sediment supply from the 166.4 km lower Minnesota River downstream of Mankato is ~450,000 m^3/year. At the average bulk density from the cut bank samples of 1.35 g/cm^3, this is equivalent to 610,000 Mg/year, or roughly twice the estimate of Lenhart et al. (2013). Placing error bars on this estimate is challenging because we do not have sufficient data to place confidence limits on our average cross-sectional area estimate, but using the confidence limits on the slope of the widening rates implies that 95% confidence limits are at least ±90,000 Mg/year.

A more precise accounting for sediment eroded by channel widening would consider the possibility of storage within the reach. Because bed material is mostly sand, but banks contain a significant fraction of silt/clay size sediment, this requires a size-specific calculation. Based on sediment sampling data, sand probably represents between 40% and 50% of the material eroded from banks or deposited on floodplains along the mainstem Minnesota River and perhaps 60% of the material eroded from the banks of tributaries. If widening typically occurs into cutbanks (Figure 7.1c) that have a sand fraction of

46%, widening could be responsible for a gross supply of sand to the channel of the mainstem Minnesota River downstream from Mankato of roughly 610 ,000 ± 90,000 × 0.46 = 280,000 ± 41,000 Mg/year. Incidentally, this is over 50% of the suspended sand load at Mankato reported by Ellison et al. (2014).

Sand also enters and leaves the channel through regular meander migration, even if the channel is not widening. A long-term estimate of this flux can be developed using the overall area of lateral offset without cutoffs, which we refer to here as A_o. If widening occurs exclusively through erosion of a single bank (mechanism c in Figure 7.1), the resulting centerline offset would be half of the overall widening. Assuming that this is the primary process responsible for widening, and neglecting major changes in sinuosity, migration should have reworked an area roughly equal to Ao minus half of the cumulative widening (equation 7.4):

$$A_m = A_o - 0.5A_w \tag{7.4}$$

where A_m is the total area reworked by migration, and A_w is the total increase in channel planform area for the reach.

For the 146.5 km considered, the cumulative lateral offset from 1938 to 2008 (neglecting cutoffs) represents an area of 6.17 km^2, while the cumulative widening represents 4.53 km^2. This implies an area reworked by lateral migration between 1938 and 2008 of 6.17 − 4.53/2 = 3.91 km^2, or, if divided across the 146.5 km length for which the figure applies, a lateral migration rate of 0.38 m/year simply due to progressive bend migration. Lateral migration at such a rate would presumably have occurred even in the absence of any widening. If migration erodes sediment over a bank thickness of 5 m (roughly the average bankfull depth at Jordan), a migration rate of 0.38 m/year would produce a volumetric flux of roughly 166.4 km × 0.38 m/year × 5 m = 316,000 m^3/year for the lower Minnesota River downstream from Mankato. We note that cumulative lateral offset is weighted toward the upstream end of the reach, so most of this erosion probably occurs upstream from Jordan. If banks and bars had identical elevations and consisted entirely of sand, migration would simply be associated with the exchange of sand from one bank to another, with little net impact on the reach-scale budget. However, because our data show that cutbanks contain less sand than do point bars (46% vs. 81% sand, respectively) migration probably represents a net sink for sand. Furthermore, bulk density for sandy point-bar deposits may be higher than the value of 1.35 Mg/m^3 we estimate from our relatively silt-rich cutbank samples. For natural quartz sand deposits with typical porosities of around 40% (Fraser, 1935; Román-Sierra et al., 2014), bulk density should be about 1.59 Mg/m^3. Multiplying the 316,000 m^3/year exchange flux by the appropriate sand fractions and bulk densities results in sand supply at cut banks of 316,000 m^3/year × 0.46 × 1.35 Mg/m^3 = 196,000 Mg/year and deposition in point bars of 316,000 m^3/year × 0.81 × 1.59 Mg/m^3 = 407,000 Mg/year, for a net difference (i.e., net migration-related sand storage) of 211,000 Mg/year.

It is thus plausible, at least within the error of our flux estimates, that most of the sand produced by widening (280,000 ± 41,000 Mg/year) is sequestered in nearby point bars. Dividing the remaining sand across the channel area of the lower Minnesota River would result in an aggradation rate of under 3 mm/year, so small changes in average bed elevation or natural deposition in oxbow lakes could easily be sequestering the rest.

Another implication of our findings relates to the supply of fine (silt/clay size) sediment associated with eroding banks. Unless most widening occurs on point-bar banks, which is not consistent with Figure 7.2, or into cutbanks that are significantly sandier than we observed, roughly 50%–60% of material supplied to the channel through widening and channel migration consists of silt and clay. Using our overall estimate of volumetric widening from the 166.4-km lower Minnesota River of ~450,000 m^3/year and a silt + clay fraction of 54% (streambanks below Mankato), this implies a flux of 450,000 $m^3/year \times 0.54 \times 1.35$ Mg/m^3 = 330,000 Mg/year of silt/clay transferred to the channel exclusively from widening. Net erosion of cut banks by regular meander migration presumably transfers another 330,000 $m^3/year \times 0.54 \times 1.35$ Mg/m^3 = 239,000 Mg/year of silt/clay to the channel. Assuming that none of this silt/clay is deposited in point bars, the total production of silt/clay from meander migration and widening together comes to ~569,000 Mg/year. Unlike sand, silt/clay size material is unlikely to be stored in a channel deposit, so it probably does increase the net down-channel load unless it is deposited on the floodplain.

While fully characterizing the silt/clay budget for the lower Minnesota River valley is beyond the scope of our study, our results indicate that the trap efficiency of floodplains and channel cutoffs in this reach could be much higher than previously recognized. Wilcock (2009) used total suspended solids (TSS) loads gaged at several points along the lower Minnesota River to estimate a net TSS sink (presumably mostly silt/clay) of 350,000 Mg/year. Even without considering the supply of silt/clay associated with widening of the mainstem Minnesota River, Wilcock (2009) estimated that 25%–50% of washload is stored downstream from Mankato. If the 569,000 Mg/year silt/clay we estimate as being transferred to the channel by widening is assumed stored within the reach, which is necessary for the Wilcock (2009) budget to close, the overall sequestration of fine sediment within the reach becomes considerably larger. This is generally consistent with Groten et al. (2016), whose SSC load estimates for the 2011–2014 period show a large increase in suspended sediment load between Mankato and Jordan and a large decrease, to levels somewhat below those at Mankato, between Jordan and Fort Snelling. The increase occurs within the reach where we show the greatest long-term widening and the most active channel migration, and the decrease occurs in a reach with much lower overall migration rates and extensive off-channel water bodies. In fact, adding ~569,000 Mg/year silt/clay to the Wilcock (2009) storage estimate implies that sediment storage within the reach could have a magnitude similar to upstream sediment supply at

Mankato, which has been estimated at 797,000 Mg/year (2000–2008 TSS load, MPCA, 2009) or 1.6×106 Mg/year with 28% coarser than 62.5 μm (2007–2011 SSC-based estimate; Ellison et al., 2014).

Other Geomorphic Adjustments

An increase in formative discharge should eventually lead to increases in depth as well as in width. In principle, depth adjustment can occur through changes to either average bed or banktop elevation. Because widening provides bed material stored in the floodplain to the channel, sufficiently large rates of widening could theoretically mobilize so much sediment that it could overwhelm the down-channel transport capacity of the system, causing bed aggradation and thus, decreasing depth. However, in the present study, we calculate relatively high storage of sand in bars and thus minimal to modest net storage of sand on the bed. Furthermore, cross-section surveys within our lower-most study reach at Chaska show minimal overall bed elevation change between 1948 and 2000 (Lenhart et al., 2013). In-channel dredging, which has been used to maintain navigability along the downstream-most 23 km of the Minnesota River since the 1960s (Lenhart et al., 2013), may also locally play some role in preventing bed aggradation. However, the effect is probably limited because dredging occurs well downstream from Chaska.

Whether or not bed elevation has remained relatively stable, increased discharge has presumably led to an increase in flood duration and frequency, providing ample opportunities for sediment accumulation at the top of bank. This is consistent with the apparent storage of large quantities of fine sediment within the valley of the lower Minnesota River (Wilcock, 2009; Lenhart et al., 2013; Groten et al., 2016). However, dividing the sum of the Wilcock (2009) storage estimate and our 569,000 Mg/year of widening-related silt/clay production across a ~130-km-long, ~1-km-wide lower Minnesota River floodplain produces average deposition rates on the order of just 5 mm/year. So, to the extent that the Minnesota River cannot incise to gain depth, many years of overbank sediment accumulation would be required to build banks to a level sufficiently high to regain equilibrium. This is particularly true if net sand storage is causing moderate amounts of bed aggradation.

Finally, meander bend cutoffs along the lower Minnesota River may serve yet another important sediment sink. Although cutoffs did not occur within our two lower Minnesota River study reaches, bend cutoffs (natural and engineered) and local straightening at bridges reduced the overall sinuosity of the uppermost 82 km of the lower Minnesota River by about 18% between 1938 and 2009 (Lenhart et al., 2013). The largest cutoff occurred in 2001, causing the abandonment of ~4 km of channel ~105 river km upstream from the confluence with the Mississippi River. Oxbow lakes associated with these cutoffs were usually partially filled with sediment several years after formation. Even where lakes have persisted, they are generally connected to the channel during floods and continue to experience sedimentation. The cutoffs may also have had important implications for widening. Channel adjustment near bend cutoffs can occur through upstream incision and/or bed coarsening, downstream aggradation, and the growth of new bends that increase sinuosity and thus eventually allow slope to relax back toward the pre-cutoff value. However, because meander regrowth can occur slowly relative to bed adjustment (Talbot and LaPointe, 2002), the slope increase associated with cutoff can persist for some time. To the extent that channel width is set by a formative bankfull Shield's stress (e.g., Parker et al., 2007; Wilkerson and Parker, 2011), the increased slope would lead to an increase in width even if formative discharge remained unchanged. Data clearly shows a larger increase in width on the lower Minnesota River upstream from Jordan than farther downstream where fewer cutoffs occurred. It thus appears plausible that width increase may have been exacerbated by slope changes near cutoffs. However, the increase in width even for the lower section of the lower Minnesota River, downstream from the cutoffs, as well as the observation that width has continued to increase relatively consistently in all mainstem study reaches, indicate that the observed increase in discharge is the primary driver.

Management Implications

These findings have important implications for sediment management. Specifically, sediment reduction strategies should consider the large and dynamic sources and sinks for sediment that exists within a relatively narrow near-channel corridor that comprises the channel and its gynomorphically active floodplain. This corridor represents a small fraction of the total landscape but plays a disproportionately important role in the sediment budget of large, transport-limited alluvial rivers like the Minnesota River. Furthermore, because this corridor tends to become exponentially larger and more dynamic in the downstream direction, stream stabilization on relatively small first- and second-order channels may not provide the overall sediment supply reduction benefits that are sometimes assumed. Additional analysis focused on the role of these low-order channels is probably warranted, but such studies should recognize that the majority of bankfull volume—and thus most of the potential for volumetric adjustment to influence the sediment budget—is located farther downstream. On the other hand, while relatively dynamic high-order channels do apparently provide an important sediment reduction target, the highly distributed nature of the widening and the fact that widening often occurs in tandem with natural meander migration mean that simply stabilizing the most rapidly eroding banks along the Minnesota River mainstem would probably not be effective at reducing overall widening-related sediment supply. Furthermore, interrupting natural bend growth and cutoff processes through bank stabilization could influence the sand budget on the mainstem Minnesota River and may have important ecological consequences. Addressing the increases in discharge that has been caused by increased precipitation and agricultural drainage may represent a more effective management strategy.

Conclusions

Air photograph analysis of the unvegetated zone of the Minnesota River and major tributary channels shows that long-term widening has occurred at a relatively consistent rate since at least 1937, a period characterized by a large increase in overall water discharge. The overall widening rate computed using all our data is on the order of 0.36%/year, with rates as large as 0.62%/year for the largest channels in the watershed. Error on these estimates is on the order of ±0.1%/year. On the mainstem Minnesota River below Mankato, widening occurred on most meander bends and was somewhat higher in areas that experienced large amounts of centerline change. Furthermore, changes in the width of the water surface during low discharge periods along three reaches of the Minnesota River indicate that widening has occurred throughout the cross section and is not exclusively associated with changes in the elevation and horizontal position of top-of-bank vegetation. A regional relationship between channel cross-sectional area and drainage area was used to develop estimates of overall bankfull volume in the basin. Most of the bankfull volume occurs within the Minnesota River itself, and major tributaries (width > 25 m) account for most of the remainder. Extrapolation of our 1937–2015 widening rates to all these channels indicates that widening could conceivably produce more sediment than has been observed at gages monitored for TSS and SSC on the lower.

Minnesota River over the Past Several Decades

While we did not observe measurable width change in the smallest channels we studied (those with widths between roughly 10 and 25 m), this may have resulted from an inability to clearly delineate banks from aerial imagery on relatively small channels that often have dense riparian vegetation. Furthermore, we did not attempt to measure width change in channels narrower than 10 m. Larger relative changes in width along small channels would increase the importance of widening as a sediment source. Our results also imply that significant floodplain-related sinks for sand and easily suspended silt/clay size sediment are present along the mainstem Minnesota River and its major tributaries. The widening-related source and the floodplain sinks represent very important and underappreciated parts of the overall sediment budget for the watershed. Managing widening-related sediment production is complicated by the fact that the source is highly distributed. Because widening appears to be driven primarily by increases in flow caused by increases in precipitation and agricultural drainage, management strategies that focus on reducing high flows may be the most prudent and sustainable mechanism to reduce the associated sediment loads.

Acknowledgments

We would like to thank many colleagues and students who collected data or provided ideas in support of this study. The work benefited from feedback provided by Karen Gran, Peter Wilcock, Gary Parker, Enrica Viparelli, Carrie Jennings, Hervé Piégay (who suggested the use of a photographically based measure of bankfull width), Utah State University undergraduate researcher Tim Beach and graduate student Sara Kelley, and Seattle University undergraduate researchers Jenny Graves and Matt Tiedeman. We also appreciate the thoughtful review comments provided by Richard Marston and two anonymous reviewers—they improved the chapter significantly. Work was supported by the following organizations: The Minnesota Pollution Control Agency; the National Center for Earth Surface Dynamics, a U.S. National Science Foundation-sponsored Science and Technology Center (EAR-0120914) and NSF CBET1209445; Seattle University; and the Minnesota Corn Growers Association, which supported the initial 1938–2008 channel widening study on the lower Minnesota River and for which John Nieber, David Zumr, Jason Ulrich, and Mikhail Titov all provided assistance.

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8

Hydrologic Alteration Drives Channel Widening and Alters Sandbar Vegetation Dynamics on a Large, Alluvial River in Minnesota, USA

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DOI: 10.1201/9781003175094-8

Introduction

Many upper Midwestern US rivers in the upper Mississippi River basin have experienced unusually large streamflow increases in the past three decades (Zhang and Schilling, 2006; Lenhart et al., 2011a; Kelly et al., 2017) contributing to widespread channel disequilibrium and subsequent channel adjustment. In graded or stable streams there is balance between erosion on outer bends and deposition on point bars and floodplains over time-scales of decades or more (Leopold, 1994; Lauer and Parker, 2008). When erosion and deposition are out of balance in an unstable stream, sediment load can increase downstream (Simon, 1989a). One indicator of stream disequilibrium is channel widening which typically occurs when erosion of the outer bends exceeds deposition on the point bars. Many streams in the upper Midwest are thought to be out of equilibrium and increasing in width (Odgaard, 1987; Simon and Rinaldi, 2002), due to stream flow increases caused by increased precipitation (climate change), land cover change, and increases in subsurface and surface drainage as well as direct channel modifications (Lenhart et al., 2011b).

Increased rainfall (Zandlo, 2008) has likely contributed to more streamflow in some parts of the Upper Midwest. Land-use change, which in this region is manifested primarily as increasing row crop coverage and expanded subsurface tile drainage, also impacts streamflow levels. For example, Zhang and Schilling (2006) found increasing flows in the Mississippi River since 1940, primarily due to agricultural land-use change and subsurface drainage increases. Often increased flows, particularly frequently occurring floods, will lead to channel evolution, either downcutting or widening of channels. Simon (1989b) described the Channel Evolution Model, a typical sequence of channel adjustment due to increased flow or channelization where downcutting of the stream is followed by widening, aggradation, and finally a return to an equilibrium. Although not all streams follow this exact sequence, it is a useful reference for characterization of channel evolution stages.

Direct channel modification and increased flows have led to widespread loss of sinuosity in many upper Midwestern streams (apart from agricultural drainage ditches). Verry (2000) found that logging impacts and increased flow from agricultural land uses led to sinuosity that was 6%–45% lower than pre-impact conditions. Lenhart et al. (2011b) found that a loss of sinuosity of about 15% in Elm Creek located in Minnesota resulted from channel straightening at road crossings and increased rates of channel cutoff from greater streamflow. Decreased sinuosity increases slope and tends to promote greater shear force and more efficient transport of sediment in streams.

Riparian vegetation also plays an important role in channel evolution. Corenblit et al. (2009) showed there are inter-relationships between vegetation and sediment scour or deposition on sandbars that affect the timescale and progression of channel evolution, particularly the later stages of evolution that involve narrowing the river by revegetation of the pointbar tending back toward channel equilibrium. Dixon et al. (2002) showed how hydrologic alteration of streamflow patterns from dams may influence riparian vegetation establishment and growth on the Wisconsin River. Rood et al. showed how hydrologic alteration from dams can create unfavorable conditions for the establishment and survival of cottonwood trees along rivers in Alberta and introduced the idea of the recruitment box. The recruitment box model frames the period under which appropriate flow conditions overlap with seed dispersal and exposed sandbars in summer allowing for establishment and survival of tree seedlings. Later high flows, ice-scouring, and drought can all kill recently established seedlings, reducing the zone of woody plant recruitment (NRC, 2002).

Changes to the above hydrologic, geomorphic, and ecological factors have contributed to increased sediment load in many streams in the United States, particularly in low-gradient rivers of the northern Midwest in Minnesota, Iowa, and the eastern Dakotas. At the same time, the Minnesota Pollution Control Agency (MPCA) has been developing turbidity and phosphorus Total Maximum Daily Loads for the Minnesota River and its tributaries over the past 15 years. These studies require data on the sources of sediment loading from near-channel sources of sediment vs. upland sources. Sediment dynamics are also important to manage natural areas along the river such as the Minnesota River National Wildlife Refuge, which manages floodplain wetlands for biodiversity and wildlife. Studies were undertaken by the University of Minnesota and collaborators to evaluate the relative quantities of sediment from in-stream and near-channel (bluff and ravine) sediment in the Minnesota River Basin (MRB) and drivers of the increased rates of channel erosion observed in recent decades (Gran et al., 2009; Nieber et al., 2010; Day et al., 2013). The study described here focuses on changes in stream bank retreat rates, streamflow changes, and interactions between hydrology and riparian vegetation. Although many efforts have examined bluff erosion in the Minnesota River Basin (Thoma et al., 2005; Gran et al., 2009; Day et al., 2013), the role of hydrology-sediment-vegetation interactions on sandbars has not been examined in detail in the MRB, at least since widespread streamflow increases have occurred in the region starting in 1980 or 1990. Noble (1979) did investigate riparian vegetation along the Minnesota River prior to the major flow increases.

Research Site Location and History

The MRB drains over 43,000 km^2 with agricultural land use covering more than 80% of the total area, mostly in corn and soybeans. The geologic setting of the Minnesota River watershed consists of flat to rolling glacial till plains and former lakebeds along with steep valley walls, particularly along the Blue Earth and LeSueur rivers south of Mankato (Ojakangas and Matsch, 1982). The Minnesota River itself runs through an alluvial valley comprised of silty to sandy loam while its tributaries are mostly in finer-textured glacial till soils.

Bluffs, which are defined as valley wall boundaries, supply large sediment loads to the Minnesota River along the main channel and larger tributaries, such as the LeSueur and Blue Earth (Sekely et al., 2002; Thoma et al., 2005; Gran et al., 2009). The valley boundaries are well defined in the MRB with the valley floor about 70 meters lower than the surrounding landscape. The valley is unusually deep for a post-glacial landscape in this region due to the rapid downcutting of the Glacial River Warren that occurred during the beginning of the Holocene (10,000 years ago) as a result of the catastrophic drainage of Glacial Lake Agassiz. This event created steep valley walls that are susceptible to erosion (Gran et al., 2009) and a river valley 800–2000 m wide in the lower river, south of Mankato, Minnesota. Ravines are small intermittent tributary channels formed by cutting through the steep valley walls of larger rivers such as the Minnesota or Mississippi Rivers. They are less extensive than streambanks or bluffs in total length and surface area but may be sources of much local sediment.

Today the Minnesota River carries the largest load of sediment to the Mississippi River north of Illinois with a mean suspended sediment load of 816,000 Mg per year. While agricultural field erosion was thought to be the main source of sediment for most of the 20th century, erosion of near-channel sources of sediment is now considered to be the dominant sediment source sediment contributor to the river mouth (Thoma et al., 2005; Hart, 2008; Engstrom et al., 2009; Belmont et al., 2011). Recent studies have been conducted to determine the relative proportion of sediment coming from different channel and near-channel sources, primarily ravines and bluffs. Many of these studies indicate that bluffs, which are defined as valley wall boundaries (Wilcock et al., 2009), are the major net input of sediment from channel or near-channel sources. Much of the near-channel sediment is thought to come from bluffs in steep zones of the Blue Earth River, the largest tributary to the Minnesota River in terms of flow and sediment (Wilcock et al., 2009; Day et al., 2013). Much of the sediment load derived from near-channel sources in the Blue Earth River is deposited in the lower Minnesota River floodplain or on pointbars where the river is a low-gradient (0.004%), meandering river in a broad valley, 1–3 km wide. Much of the fine sediment (silt and clay-size particles) is transported out of the basin and into the Mississippi River.

Hydrologic and Geomorphic Change on the Lower Minnesota River

Since 1938 (the date of the earliest aerial photos for the region), the percentage of land in agriculture has not changed substantially, but there have been large shifts in cropping practices impacting streamflow, including conversion of grasslands, pasture, and small grains to row-crops as well as expansion and improvement of the surface and subsurface drainage network (Zhang and Schilling, 2006). In addition, there has been about a 10% increase in annual average precipitation in southern Minnesota over the past 30 years (Zandlo, 2008). All of these factors have purportedly contributed to streamflow increases within the region over time (Lenhart et al., 2011a; Kelly et al., 2017). The flow increases have occurred across the whole range of flows, from baseflow to floods, with the greatest changes percentage-wise occurring at the low to median flows. Large flood magnitude (100-year recurrence interval) has not changed significantly in most of the watersheds assessed in Lenhart et al. (2011a), including the Minnesota River.

The causes of increased streamflow which have led to stream disequilibrium and increased channel erosion rates have been much debated. Some researchers have found that the streamflow increases are primarily caused by land-use alterations (including increased use of artificial drainage practices) and vegetative cover changes, although increased precipitation has contributed as well (Novotny and Stefan, 2007). While there is disagreement over the relative contributions of land-use versus climate change to increased streamflow, it is clear that increased stream flow results in greater sediment and nutrient loading to downstream waterbodies, creating an increased need for watershed and stream management.

Flow increases lead to geomorphic adjustment, so many of the streams in the region are in a state of disequilibrium due to changes in flow, direct channel modifications, and altered sediment load (Lenhart et al., 2011b, c). The river was moved to make way for levees and bridges in towns along the river, while greater streamflow in the past three decades likely contributed to a higher rate of natural channel cutoffs. Some of the larger tributaries in the MRB as well as the Minnesota River have been shown to have undergone widening (Simon, 1989b; Kessler et al., 2012; Lenhart et al., 2012). The lower Minnesota River is mostly in Stages IV and V of Simon's (1989b) Channel Evolution Model. It appears that the lower Minnesota River did not undergo downcutting as described in that model but mostly widened in recent decades. The steep tributaries such as Seven Mile Creek or Rush River, which drop down to the lower valley of the Minnesota River could be considered to be in Stage III (which is defined by streambed degradation), as there has been gradual incision progressing over many centuries (Gran et al., 2009). Some tributary streams above the steep drop down to the Minnesota River valley have undergone widening in the past century (Lenhart et al., 2011b), demonstrating that even streams in relatively flat landscapes have undergone dramatic channel change recently, an observation supported by the research of Yan et al. (2010) in the nearby upper Iowa River basin.

Research Questions

At the time of the study, there had been little research documenting long-term rates of streambank retreat in the MRB, so it is lateral erosion rates were not well known for the region. Key questions remain concerning the timescale for channel adjustment to increased flow and return to an equilibrium state (Davenport et al., 2010), particularly in the latter stages of channel evolution. The process of plant establishment on pointbars and increased deposition of sediment is responsible for channel narrowing, helping to maintain average river width by countering the erosion occurring on the outer bends. The importance of ecohydrological interactions, specifically the role of altered hydrology on riparian vegetation and feedbacks with sediment load, has not been examined in detail in the MRB.

This study is unique because it addresses the role of streambank erosion as a sediment input to the lower Minnesota River from about 1940 to 2010 and examines hydrologic and ecological interactions contributing to that widening. We also describe how altered vegetation-hydrology-sediment interactions (ecohydrological interactions) may contribute to increased sediment load within large alluvial rivers in this geographic region. Increased duration of average to high flows and prolonged submergence of sandbars can promote channel width increases along with the fluvial erosion caused by the high flows which transport the most sediment and shape the channel over time (Leopold, 1994).

We addressed the following questions in this study: How have human alterations to the lower Minnesota channel-floodplain system affected sediment and phosphorus load? Specifically, how have changes to river width and sinuosity changed sediment transport capacity and deposition in the lower Minnesota River? What are the physical, hydrological, and ecological mechanisms causing channel widening? In relation to the latter, how have altered hydrologic patterns influenced the colonization of woody riparian species which, in turn affects river width and sediment load?

Methods

Investigations of historic channel change in GIS, field data collection on erosion and deposition rates, calculations of changes to bankfull shear stress, streamflow trend analysis, and vegetation-hydrology interaction studies were conducted in order to quantify changes to channel form and channel sediment loading rates in the Minnesota River Basin. The following methods were performed.

Historic Aerial Photo and GIS Investigations

Historic investigations using aerial photos from 1938, 1991, and 2009 were utilized to document long-term rates of bank retreat and changes to sinuosity and width in the lower Minnesota River. The 1938 aerial photos were ortho-rectified in GIS. Measurements of length were taken in 1938 and 2009 and divided into three geomorphic reaches based on river slope and maintenance for barge traffic (Table 8.1). Measurements of river width change were taken by digitizing the entire river corridor at the bankfull elevation (the top of bank visible from aerial photos) in 1938 and again in 2009 for comparison. This provided an areal estimate of channel change over the 120 km corridor between Mankato and St. Paul. The total eroded surface area was divided by the channel length to obtain an average width change which was integrated over this corridor for the period between 1938 and 2009.

Cross-sectional data from 1948 through 2000 were compared to see if there had been river bed aggradation or degradation (see the "Calculations of Change to Shear Stress in the Lower Minnesota River" section).

Field Data Collection of Stream Bank Erosion, Sediment and Nutrient Properties, and Floodplain Deposition

Field data collection entailed measurements of streambank height, soil particles size, and soil bulk density (Das, 2009) were taken to calculate the bank erosion hazard index score and for use in calculations of sediment loading. Samples were collected over the period of 2008–2011 at 5 mainstem and 11 tributary sites. River area was delineated in GIS by tracing the boundaries of the channel between Mankato and St. Paul using both 1938 and 2009 aerial photos. The error in width was estimated at about ±15 feet (3%–6%) while length estimates were more accurate with an estimated error of ±1%–3%. Volumetric estimates were multiplied by river length to calculate the total net mass of sediment eroded by channel widening in metric tons (Mg). The total mass of sediment eroded from widening was then divided by the time interval of 1938–2009 (71 years) to obtain an annual estimate of sediment load from streambank retreat in the lower Minnesota River.

Phosphorus samples were taken from streambanks across the MRB and analyzed for total phosphorus (TP) at 0–12″ depth at 41 locations, 23 on the main Minnesota River, and 18 on tributary sites. A subset of 16 samples was analyzed for plant-available phosphorus using the Olsen test (Olsen et al., 1954). Lab analysis was performed at the Research Analytical Lab of the University of Minnesota. TP loads from bank retreat on the Minnesota River were estimated from these samples by using average TP concentrations in mg/kg and multiplying by the total sediment mass to obtain a total load of phosphorus.

Physical properties including soil particle size (Das, 2009), cohesive strength, and shear strength (Zreik et al., 1998) were measured for evaluation of bank susceptibility to erosion and for use in future modeling of bank erosion processes with the Bank Stability and Toe Erosion Model. The soils' mechanical or cohesive strength was measured with the Borehole Shear Tester (BST), which was placed on the top of the soil profile down a hole of 0.3–1 m. The soil's resistance to fluvial erosion was estimated with an instrument called the Cohesive Strength Meter (CSM), which uses a small water jet to dislodge soil particles and measure turbidity increase at the threshold of soil dislodgement (Vardy et al., 2007). The CSM provides a measure critical shear stress (τcr) required to dislodge a soil, but does not provide any estimate of the erodibility coefficient. CSM and BST data were collected on 5 Minnesota River sites and 10 tributary sites. In total, 45 CSM tests were done at 14 different streambank sites and 15 BST tests at 10 different locations.

Recent bank retreat was measured using bank pins for the time period 2007 to 2011 following methods described by Hooke (1979). Sixteen (16) bank pin sites were measured six times over the study period. Five (5) sites were on the main stem of the MN River, and 11 on tributaries. A subset of sites had two bank pin sets located on the outer bend and on a straighter reach. To facilitate comparison of erosion rates on the Minnesota River to other rivers, erosion rate was expressed as the percentage of bankfull channel width change.

Floodplain deposition estimates from soil borings across the lower Minnesota River valley were made to better understand channel evolution, sediment exchange, and deposition within the channel and floodplain (Figure 8.1). Soil borings were also done at two recent channel cutoffs near Mankato (1993) and Minneapolis (1965) to determine historic channel width and soil deposition rates. Increased bank height from deposition alongside the channel may lead to decreased floodplain connectivity and water storage, leading to greater sediment transport downstream (Yan et al., 2010). Post-European settlement (circa 1850 for central Minnesota) alluvial (PSA) sediments were distinguished from underlying deposits by chemical, textural, and structural characteristics described by Beach (1994) and Yan et al. (2010). Calcium carbonate, organic matter, soil texture, and structure were used to differentiate recently deposited soils (in the past 160 years) from older soils. Calcium carbonate is high in recently deposited sediment and gets leached out to low levels in older soils (>100–1,000 years) (Birkeland, 1999).

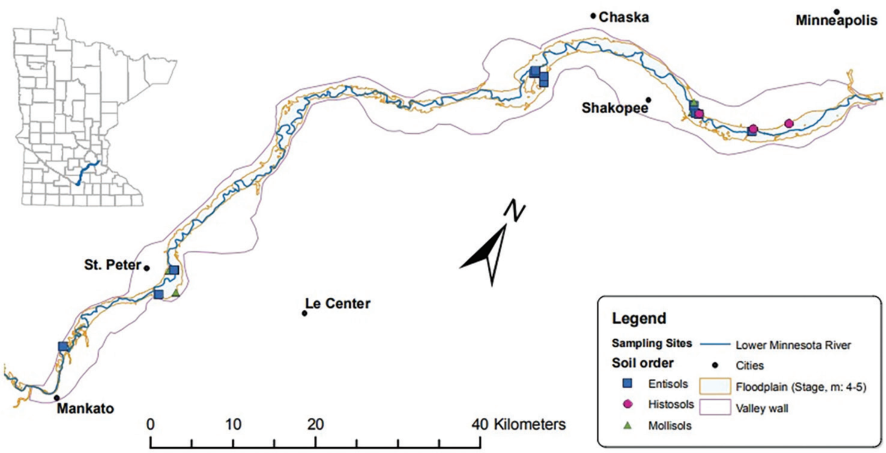

FIGURE 8.1
Location of study area within the state of Minnesota (upper left). Measurement locations for floodplain deposition within the valley of the lower segment of Minnesota River are shown above. The valley boundary is shown in pink while the active floodplain area is contained within the smaller corridor in brown. The floodplain area was delineated using a digital elevation mod and flow accumulation algorithm. Sediment sampling locations are shown by squares, circles, and triangles.

In contrast, organic matter is low in recently deposited sediment and is typically higher in the pre-European settlement surface soils (Bridge, 2003). In sandy levee deposits, where the above methods did not work, the rate of soil deposition was calculated by excavating soil adjacent to trees down to the root collar and aging the trees using dendrochronology (tree rings). In total, 40 soil borings were taken at seven research sites spaced across the Minnesota River valley including floodplain, terrace, and recent channel cutoffs near St. Peter, Ottawa, Jordan, Mankato, and the three sites in Bloomington, a suburb of St. Paul.

Sediment deposits were grouped by landform, soil order, and by distance from channel boundary. The major land forms encountered in the floodplain include recent sand deposits, most commonly found on the levee and classified by soil order as entisols; backwater open marshes with organic soils and low rates of alluvial deposition, classified by soil order as histosols; and floodplain flats, with intermediate levels of organic matter located between the levee and the valley wall and classified by soil order as mollisols. Estimates of cumulative floodplain deposition rates were then made by multiplying average PSA depth over the entire floodplain area. Sediment mass estimates were made by multiplying soil deposition volume by soil bulk density obtained from local soil samples.

Floodplain delineation was done to determine the extent of flood-prone area in the Minnesota River valley in GIS. Using a digital elevation model and flow accumulation algorithm, flood-prone areas were delineated that correspond to roughly a 5-year recurrence interval flow. The GIS data layer was edited to exclude areas protected by levees, railroad tracks, and other structures that blocked off portions of the floodplain.

Calculations of Change to Shear Stress in the Lower Minnesota River

Quantification of sediment transport capacity consisted of evaluating changes in bankfull shear stress (defined by the relationship: $\tau=\gamma RS$, where γ=specific weight of water, R=hydraulic radius, and S=channel slope), stream power (defined by the relationship: $\Omega=\gamma QS$, where Ω=stream power, Q=stream discharge), and unit stream power (Ω/channel width) using width, slope, and mean discharge measurements from 1938 to 2009. Calculations of shear stress and stream power assumed the width and slope were the only channel attributes that varied over the study period and that the channel could be represented by a trapezoidal channel. Thus, it was also assumed that the average bankfull depth and top to bottom width ratio remained relatively constant over time. The assumption of non-changing average bankfull depth is an important one, as a widened channel under this assumption will result in an increase in hydraulic radius, thereby also increasing bankfull shear stress. Evidence supporting this assumption was obtained from two sources: (1) surveyed cross sections from 1948 for the mouth of the Minnesota River upstream to the City of Chaska and (2) a US Army Corp of Engineers (USACE) HEC-RAS model constructed in 2010 by USACE St. Paul District for investigation of Minnesota River flood inundation. This hydraulic model used cross sections surveyed in 2000 for the lower one-third of the study area (Buesing, personal communication, 2011). The two data sources collectively only provided evidence for areas in the downstream one-third of the study area; however, it was deemed reasonable that these observations were roughly applicable to reaches in the upstream two-thirds of the study area. A subset of 15 surveyed cross sections from 1948 and 2000 were analyzed for the study. They were located near Chaska, MN (the furthest upstream point where both 1948 and 2000 cross sections were available) downstream to near Shakopee, MN, a distance of ~6 km along the river.

Streamflow Trends

Streamflow trends were assessed using the Indicators of Hydrologic Alteration (IHA) (Richter et al., 1996) to identify changes to the magnitude, frequency, timing, and duration of flow in the Minnesota River at Mankato, site of a long-term USGS stream gauge # 5325000. IHA uses daily mean flow values to calculate a suite of hydrologic statistics. While the magnitude of the high flows is considered important for erosion and sediment transport, the timing and duration of stream flow were examined as well because they may have significant ecological impacts, particularly during the establishment period of riparian vegetation. Data was collected during a previous study of stream flow trends (see Lenhart et al., 2011a). The flow data from the Minnesota River was utilized in this analysis to assess the ecological and hydrological drivers of channel widening by looking at the change in duration and magnitude of summer flows in particular. This assisted in gaining an understanding of the physical, hydrologic, and ecological factors and their interactions relative to channel evolution. To calculate significance in stream flow differences between the time periods of 1940–1979 and 1980–2009, the software program randomly shuffles all years of input data and recalculates (fabricated) pre- and post-impact medians 1,000 times, similar to the boot-strapping approach. Though non-parametric, the significance level can be interpreted similarly to a p-value in parametric statistics.

Ecohydrological Investigations of Sandbar Vegetation

Ecohydrological investigations were undertaken to better understand the potential role of sandbar vegetation in channel widening. We hypothesized that flow changes during the spring and summer months of May–August would have the greatest impact on channel width as these are the months that plants disperse seeds, initiate growth, and establish new seedlings under appropriate conditions. Flow trends over the period of 1940–2009 were examined in more detail to assess the impact of hydrologic alteration on sandbar vegetation establishment and survival. Plant vegetation establishment dates were taken from Dixon et al. (2002) in the Wisconsin River and compared to values listed for Minnesota by Smith (2008). Sandbar slope and the elevation of woody riparian seedlings of cottonwood and willows were measured where previous research had been done by Noble (1979) in the Minnesota River to see how the elevation of recently established riparian plants compared to the 1970s data. Increased vegetation elevation was taken as an indicator of less successful vegetation colonization and survival due to hydrologic alteration. Later studies done by Triplett (2015) built on this work by quantifying plant composition and coverage and rates of soil deposition on seven sandbars on the lower Minnesota River.

Results and Discussion

Field Study of Streambank Properties and Erosion Rates

The alluvial soils that comprise the Minnesota River valley were found to be primarily silt to sandy loam in texture with the exception of pockets of heavier silt and clay loam that occur in the floodplain of the river valley further from the channel edge (Schulzetendberg, 1989). Streambank height ranged from 3 to 4 m. Stream bank soils from the MRB had cohesive strength values that averaged 19 kPa ($n=15$ at ten different sites) and a friction angle of 44° as measured by the BST. They had shear strength that ranged from 0 to 30 Pa, with an average 2.2 Pa ($n=45$ at 14 different sites) as measured with a portable CSM tester. These numbers are similar to the range of values found by Hanson and Simon (2001) in the highly erodible loess soils of the Midwest and Mississippi River valley. While the cohesive soils provide some resistance to mass-wasting, the shear strength values indicate that they are very susceptible to fluvial erosion from increased streamflow. Observations of erosion processes during the bank erosion monitoring period from 2007 to 2011 showed that both fluvial erosion and mass-wasting contribute to bank retreat on the lower Minnesota River. Fluvial erosion at the streambank toe occurred after every storm event causing a hydrograph rise sufficient to submerge at least the lowest bank pins at the times we visited the sites. Mass-wasting occurred after large flow events that occurred in each of the monitoring years as evidenced by the slumps of vegetated bank material that fell into the river or dropped onto bank pins lower on the stream bank.

Rates of Bank Retreat in the Minnesota River Basin Since 1991

Bank retreat rates vary considerably across the Minnesota River Basin, by stream order, geologic material, and stream flow change (Lenhart et al., 2011c). When normalized by dividing erosion rate by river width, the steep tributaries that descend the 50 m drop down to the Minnesota River valley (such as Seven Mile Creek, High Island Creek, and Rush River sites) had the highest rates of bank retreat followed by the lower main channel of the Minnesota River. The lower Minnesota River had the highest absolute rates of bank retreat, with a maximum of 4 m per year in some locations. While lateral erosion may be balanced by point bar deposition in a graded (equilibrium) stream, many MRB streams have been undergoing widening at least since the 1938 aerial photos and increasingly since 1980, indicating that erosion exceeded deposition within the active channel area (Christner, 2009; Lenhart et al., 2011b). In the lower Minnesota River, four sites monitored in the field over 3 years had an average erosion rate of 0.96% of total river width from 2008 to 2010. The rate was greater (1.46%) over a 20-year (1991–2010) time period based on aerial photo measurements and greater than the longer-term 1938–2009 rate. Elevated erosion rates are likely caused by flow increases and localized channelization at road crossings or natural cutoffs that increase channel evolution in that area. (See Lauer et al., 2017 and the corresponding chapter in this book.)

Channel Change in the Lower Minnesota River 1938–2009

In the past 70 years, the Minnesota River between Mankato and St. Paul has widened considerably and shortened from straightening at bridges and increased rates of natural cutoff, shortening its length by 7% (12% shorter since the 1854 early land survey maps). The majority of length reduction occurred in three reaches, between Mankato and St. Peter, LeSueur to Henderson, and Shakopee to St. Paul. The lower 20 km from St. Paul to Savage was straightened for commercial barge shipping in the 1960s (Table 8.1).

Widening in the lower Minnesota varied considerably by river reach, but on average, it increased about 36 m, in total about a 52% width increase for the Mankato to St. Paul corridor. This is equivalent to 0.5 meters per year or 0.74% per year (Table 8.2). Over the 70-year time period an estimated 20 million tons of sediment was eroded (280,000 Mg per year) from the lower Minnesota River between Mankato to St. Paul (Table 8.2).

Analysis of historic cross sections for change in width and bed elevation showed two important trends. First, many of the 1948 cross sections were re-surveyed in 1966, and in 85% of the cases, cross-sectional profile remained relatively constant between 1948 and 1966, suggesting most widening observed in this reach occurred later than 1966. Second, 1948 cross sections were

TABLE 8.1

Change in Channel Length in the Lower Minnesota River, 1938–2009

	Geomorphic Reaches of Lower Minnesota River		
Geomorphic unit, slope, and shipping usage	*Mankato-Belle Plaine* Steeper slope (0.02%), not maintained for barges	*Belle Plaine-Savage* Low slope (0.003%), but not dredged	*Savage-St. Paul* Very low slope (<0.004%) and dredged for barge traffic
Reach length in 2010	82 km	56 km	23 km
Change in river length since 1938 and causes	–18% from channel cutoffs and straightening at bridges in cities	–5% primarily from natural cutoffs during large floods, e.g., 1993	–10% for channelization to maintain nine-foot deep barge channel built in the 1960s

Change in length in the lower Minnesota River by geomorphic reaches, based on slope and channel maintenance practices.

TABLE 8.2

Sediment and Nutrient Loading from Channel Widening

Metric	Values	Method
Bank height average (m) – present	3.2±1 m	Field measurement
Change in bed elevation (m)	0±1 m	Historical cross-section comparison (1948) from HEC-RAS models
Bulk density (g/cm^3) – present time	1.35 g/cm^3, range (1.08–1.72)	Field measurement
Change in river width 1938–2009 (m), % change	36 m, 52% ±5 m width	Overlay of digitized channel corridors, 1938 and 2009
Widening % per year (1938–2009)	0.74%	Overlay of digitized channel corridors
Tons (MegaG) sediment/year due to channel widening	280,000±56,000	Width change from aerial photos*bank height * soil bulk density
Tons (MegaG)/phosphorus/year	153±30 Mg	Width change from aerial photos *bank height * soil bulk density* mean phosphorus concentration

Metrics of channel change, results, and methods used in calculating total load of sediment from widening of lower Minnesota River between Mankato and St. Paul, 1938–2009.

compared to applicable HEC-RAS 2000 cross sections, showing that in 85% of the cases, both average and maximum bankfull depths remained relatively constant over the period while widths increased an average of 30% (Figure 8.2).

Phosphorus loading from streambanks along the lower Minnesota Rivers was estimated at 153 tons/year between Mankato and St. Paul, based on an average total phosphorus (TP) concentration of 538 mg/kg (range of 300–800 mg/kg) in 20 stream bank samples taken along the Minnesota River (Table 8.2). This represents about 17% of the annual TP load of the Minnesota River estimated at 908 tons by the MPCA (2009). Olsen phosphorus, a measure of plant-available phosphorus, averaged 13.6 mg/kg in stream bank samples. TP content was found to decline with increasing sand content, which is typically high in alluvial channels. Therefore, backwater swamps with higher clay and silt content could be expected to have more TP than sandy levee soils. Olsen phosphorus levels were similar to agricultural soils of the Midwestern region, with an optimal level for farm soils being 8–14 mg/kg. The 13.6 mg/kg average in the streambanks was higher than may expected for a soil in natural plant cover

Calculations of Changes to Bankfull Shear Stress

Changes to width and sinuosity shown in Table 8.1 affected sediment transport capacity independent of the flow increases as the 7% loss of river length

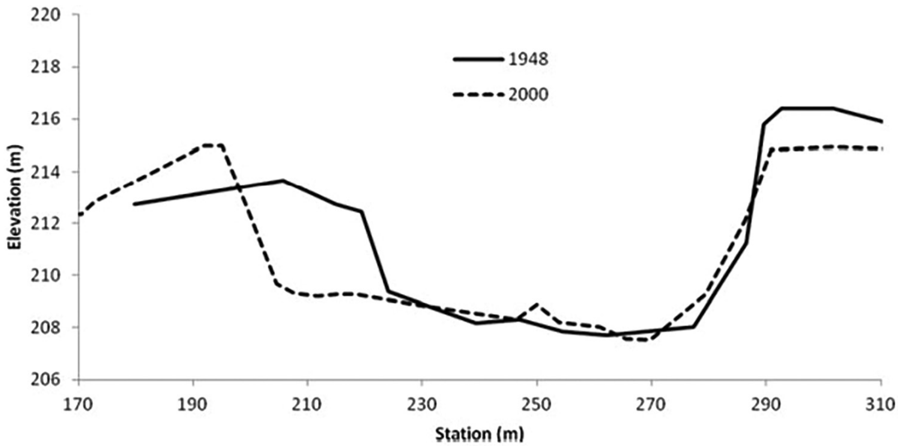

FIGURE 8.2
Change in Minnesota River cross-sectional profile near Chaska, MN, from 1948 to 2009, illustrating change in width and depth. The average and maximum depth remained about the same over this time period at the cross sections assessed width at this site over the period increased ~30%, less than the average for the river corridor.

increased channel slope, increasing bankfull shear stress. Our calculations show that for most stream segments the bankfull shear stress increased (Figure 8.3). The maximum increase was 144%. Four percent of the reaches outlined in Figure 8.2 had decreased bankfull shear stress due to channel re-meandering and related slope reduction.

While shear stress provides an indicator of localized river erosive force, stream power is a measure of the energy available in a river to transport materials. Stream power increased at the Minnesota River near Mankato from ~350 to 1,000 W due to a 25% increase in slope in that reach along with a 75% increase in mean annual flow. Over most of the lower Minnesota River, the stream power change was not as great because the change in slope was less. Due to increased river width, unit stream power (watts/m^2) increased about 30%.

Depositional Rates

The lower Minnesota River valley is in a depositional setting that tends to make it a sink for sediment. However, the depth of PSA is highly heterogeneous (Figure 8.4) varying by landform, soil type, and distance from the main channel (Birkeland, 1999; Bridge, 2003). In samples taken near the channel margin (the natural levee, entisol soil order) the mean deposition rate

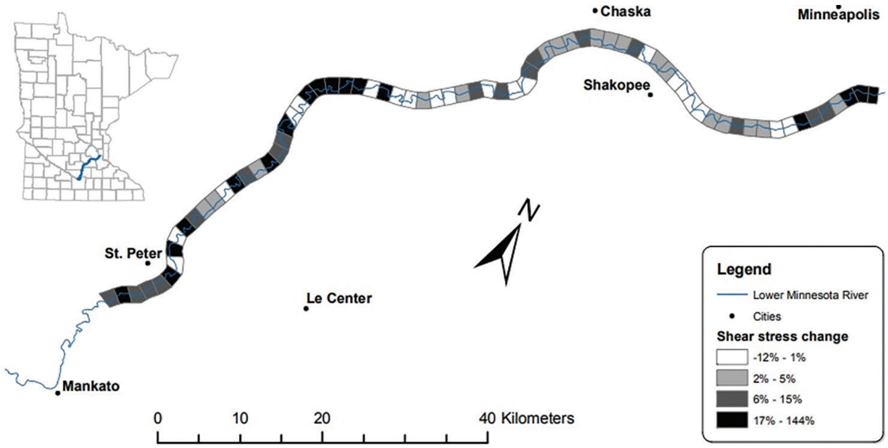

FIGURE 8.3
Change in bankfull shear stress calculated for the lower Minnesota River between Mankato and St. Paul based on changes to slope and width since 1938. Depth was assumed not to have changed based on review of historic cross-sectional data in two locations. 25% of segments had large increases in bankfull shear stress (17%–144%) and 25% decreased slightly due to channel widening and decreased slope from re-meandering. 50% of the segments had a slight increase (2%–15%).

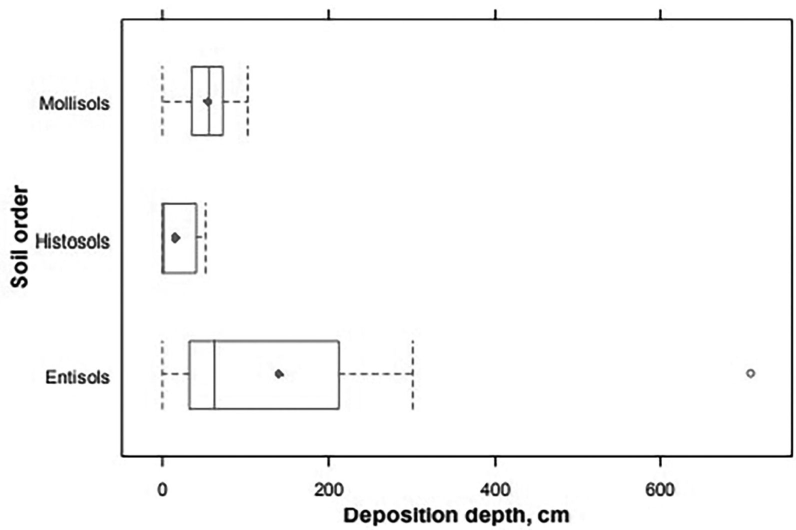

FIGURE 8.4
Depth of PSA by soil order (ordinate) and related landform. Entisols (bottom graphic) are recent sand deposits typically found on the levee next to the river where the highest rates of floodplain deposition occur (deposition depth in cm ranging from 0 to 400 cm on abscissa). Histosols (middle graphic) are organic soils typical of backwater swamps with much lower sediment deposition rates as they were often distant from the stream bank. Mollisols (top graphic) are loamy soils typically found in between the levee and valley wall.

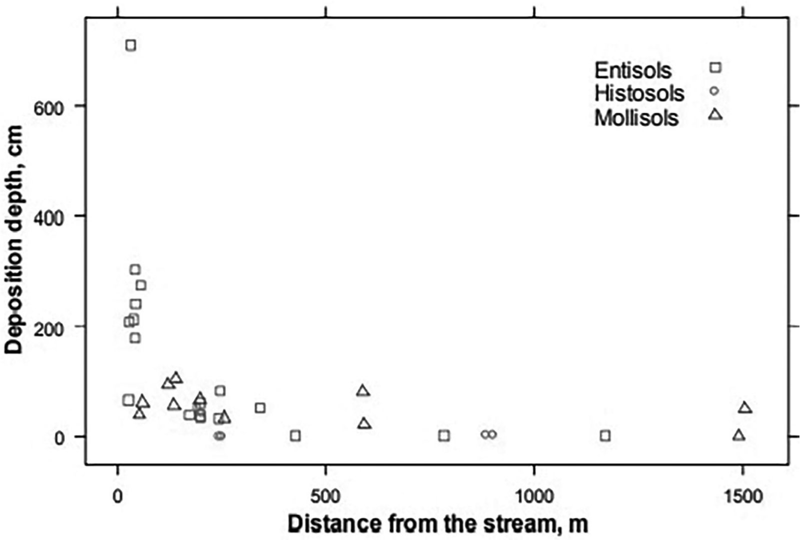

FIGURE 8.5
Depth of PSA by distance from channel edge. Sediment deposit depth decreases from the channel boundary. The Minnesota River valley bottom is approximately 1,000–2,000 m, so the greatest distance from channel was about 1,500 m. The outlying point on the upper left of the graph (>600 cm) was a channel cutoff since 1960.

was estimated to be 0.80 cm/year. The mean rate was calculated at 0.17 cm/year in organic histosols (muck soils) which are typically located far from the channel margin (Figure 8.4). There was an inverse relationship between distance from the channel boundary and depth of PSA (Figure 8.5), as researchers found in northern Iowa (Yan et al. 2010). The natural levee adjacent to the channel was significantly higher than the floodplain. Increased elevation of the channel boundary tends to reduce floodplain access in these areas. However, calculations of flood-prone area show that much of the lower Minnesota River valley is regularly inundated, although parts of it are entrenched and less accessible by high flows. The lower 30 km of the river, nearest to the Mississippi River is regularly inundated.

Sandbar deposition rates were measured in a later study by Triplett (2015) on seven sandbars using tree-ring dating. The rates of deposition were estimated to range from 1.8 to 14.8 cm/year and average 7.1 cm/year in the upper portion of the sandbar vegetated with willow trees or shrubs. This is approximately nine times the rate near the stream banks or sand levees and 40 times the rate on the organic-soil wetlands further away from the channel boundary. Deposition is not even over time, but rather occurs primarily in large flood events where tens of centimeters may be deposited in 1 year based on post-flood observations, followed by years with no deposition. In total, floodplain deposition rates in the lower Minnesota River valley were estimated to range from 90,000 to 450,000

Mg/year over the post-European settlement time period. The total suspended sediment load at Mankato ranged from 270,000 to 1.6 million Mg/year (average 816,000 tons/year), greater than the total load at St. Paul at 180,000 to 1.3 million Mg/year between 2000 and 2003 (Lenhart et al., 2011c). The difference between the upstream and downstream gauges suggests that about 105 tons are deposited each year in the lower Minnesota River valley (Wilcock et al., 2009).

Post-European settlement rates of deposition on the floodplain have increased. Based on soil survey data on floodplain soil depth, the rate of soil accumulation over the past 10,000 years since the downcutting of River Warren was estimated at 0.008–0.014 cm/year. Although this is a coarse estimate since soil survey data is less accurate below the A horizon, it does provide a relative comparison of floodplain deposition rates showing that the post-European settlement rate is roughly ten times the pre-European settlement rate. Compared to other studies done on floodplains in the region, in Wisconsin and along the Mississippi River, the Minnesota River floodplain and backchannel cutoffs have much higher rates of deposition (Knox, 1987, 2006).

Although deposition increased on the floodplain since 1850 sandbars may be trapping less sediment. Increased flow tends to mobilize more unvegetated sediment on the sandbars. In a negative-feedback loop, reduced vegetation colonization prevents velocity reduction and settling of sediment. Infrastructure has also restricted floodplain access and deposition in areas where human developments impinge on the floodplain. Examples of this include roads built in the river valley and/or levees to protect towns, particularly in Mankato and stretches along State Highway 169 north toward the Twin Cities. In the lower 20 km of the river in the Twin Cities, water control structures also shunt water out of wetlands at smaller flood levels in the Minnesota River valley National Wildlife Refuge to protect water quality in wetlands for waterfowl. The above factors have tended to make the lower Minnesota River valley a less efficient sink for sediment compared to the pre-hydrologic and geomorphic alteration state.

Certain terraces and elevated zones of the floodplain are too high for substantial sediment deposition due to geologic history. Natural terraces that lie 10–20 m above the active floodplain are no longer flooded near Jordan and Ottawa. Tributary alluvial fans may be 5–10 m above low floodplain locations (Hudak and Hajic, 1999).

Riparian Vegetation and Hydrology Interactions Effect on Channel Evolution

Ecohydrological interactions of stream flow, sediment transport, and vegetation may influence the rate of in-channel deposition. Changes to the timing and duration of stream flows in the Minnesota River (Tables 8.3 and 8.4) can

TABLE 8.3

Streamflow Alteration in the Minnesota River at Mankato, 1980–2009 vs. 1940–1979 Annual Statistics

Time Frame	Mean Annual Flow	Coef. var. (cv)	1.5 Years Flood Peak	2 Years Flood Peak	2 Years Flood Dur.	10 Years Flood Peak	10 Years Flood Dur.	1-Day Min	30-Day Min	90-Day Min
1940–1979	89.1	1.76	520	633	3.1	2013	2.8	6.0	7.7	11.5
1980–2009	155.9	1.36	626	784	4.1	2127	6.0	14.5	19.7	37.3
IHA	–	–	–	**0.09**	**0.05**	0.44	**0.02**	**0.00**	**0.00**	**0.00**

Using the IHA statistical analysis tool, changes to the hydrologic regime were characterized, including metrics of duration, low flow, and timing. Stream flow is in cubic meters/second; duration is in days.
*Significance is at the alpha=0.10 level as indicated by bold font; Coef. Var. is the coefficient of variation, a measure of stream flow variability relating daily flow to mean annual flow; flood dur=r flood duration in days.

TABLE 8.4

Hydrologic Change during Plant Dispersal and Establishment Period along the Minnesota River, Monthly Statistics

Month	Stream Flow Metric	Time Period 1940–1979	Time Period 1980–2009	Significance Level	Significant Difference between Time Periods?
June	Median monthly	128.1	213.6	0.001	Yes
	Low flow[a]	52.8	90.5	0.095	Yes
July	Median monthly	82.7	143.7	0.023	Yes
	Low flow	52.1	70.5	0.072	Yes
August	Median monthly	33.6	48.1	0.096	Yes
	Low flow	24.3	40.6	0.023	Yes

Flow is in cubic meters/second in summer months comparing 1940–1979 to 1980–2009. Significance was tested using a non-parametric metric in the IHA program at the 0.1 level.
[a] Low flow is defined in IHA as the dominant flow condition in most rivers in terms of duration. It is the flow level after storm event runoff has passed and the river has subsided to its base- or low-flow level. It is not the lowest flow that occurs and is higher than the annual 1-, 30-, or 90-day minimums shown in Table 8.3.

strongly influence channel width by preventing or reducing establishment of plants on point bars. Channel widening may be increased by lack of plant colonization on sandbars, the normal mechanism for narrowing or maintaining river width as a counterbalance to erosion on outer bends. Vegetation stabilizes sandbar sediment allowing progressive growth of the point bar toward the river. In contrast, sandbars with no vegetation or sparse annual plants are more easily mobilized at high flows (Triplett, 2015). Batts (2017) confirmed this finding with experimental work done in a flume to simulate

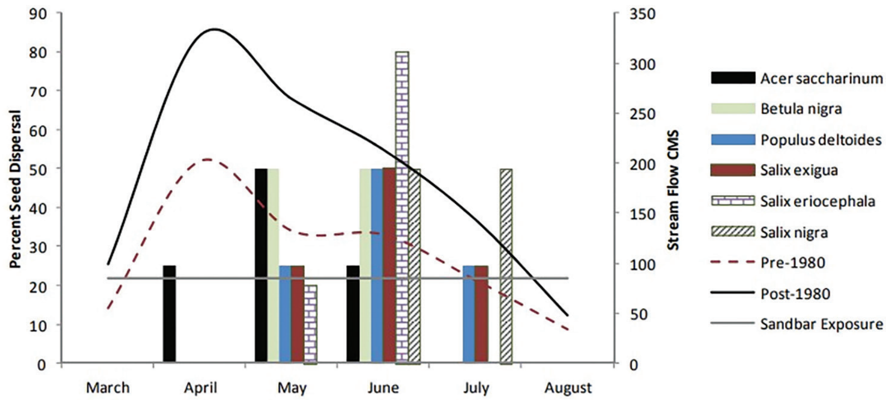

FIGURE 8.6
Timing of tree seed dispersal in the upper Midwest showing % of seeds released by month for six woody species silver maple (*Acer saccharinum*), river birch (*Betula nigra*), eastern cottonwood (*Populus deltoides*), sandbar willow (*Salix interior*), peachleaf willow (*Salix eriocephala*), and black willow (*Salix nigra*) (based on Dixon et al., 2002). Riparian tree seeds that land on exposed sandbars as the flow level declines from May were successfully established. Prolonged high summer water levels >85 cms (post-1980 hydrograph) delay sandbar exposure reducing the establishment and survival of riparian woody species. Only the higher sandbar zones have been colonized in recent years, primarily by *Salix interior* and *Populus deltoides*, the two dominant Minnesota River species (Noble, 1979).

the effects of woody plant stems on flow velocity, sediment deposition, and channel form.

Increased flow in the spring and summer may impede seedling establishment on sandbars, by submerging soil surface when seeds are dispersed by trees. Riparian trees and shrubs such as silver maple (*Acer saccharinum*), river birch (*Betula nigra*), cottonwood (*Populus deltoides*), sandbar willow (*Salix interior*), black willow (*Salix nigra*), and peachleaf willow (*Salix eriocephala*) disperse seeds between May to August, requiring exposed, moist sand to establish and survive (Dixon et al., 2002) (Figure 8.6). Increases to the low and median flows in the summer result in shorter duration of sandbar exposure and a delay in the timing of seed establishment to later in the summer. June and July median flow increased from 128 to 213 cms and from 82 to 143 cms, respectively (Table 8.4). The low flows are higher than they were pre-1980, with the 30-day more than doubling and the 90-day minimum flows more than tripling (Table 8.3). Triplett (2015) found that the average period of sandbar submergence during the growing season (April 15 to September 20) at seven sites increased from ~29% to 47%. The increased duration of sandbar-submerging flows shifts the recruitment box to the right on Figure 8.6 for most of the past three decades. The period of appropriate water level for seedling establishment has often occurred later in the growing season in August or September in recent decades, well past the time of peak seed dispersal for most riparian species. As a result, patches of recently

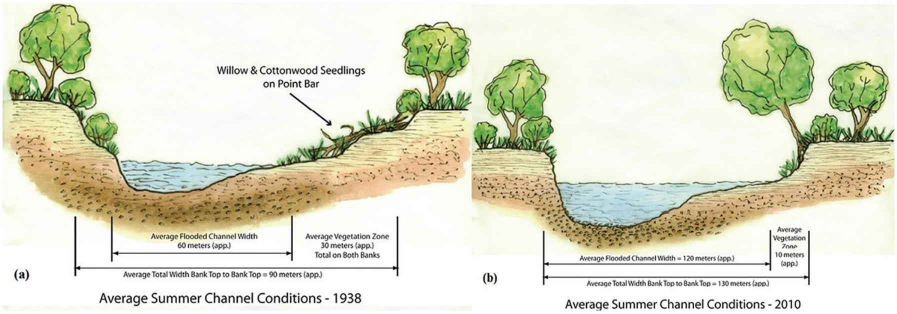

FIGURE 8.7
In addition to increased flow, channel widening has been impacted by reduced colonization of woody plants on pointbars due to prolonged high river stages in summer. Plant establishment on sandbars facilitates the narrowing of channels (a) Depiction of typical sandbar prior to hydrologic alteration (increased river flows) in 1938. During this time, the June–July water level was much lower on average allowing for more time and surface area for woody riparian plant establishment. (b) Post-hydrologic alteration (after 1980) where sandbars are exposed less frequently and for shorter duration during the growing season, reducing woody plant establishment, survival, and consequently reduced or slower point bar growth. Other factors contribute to reduced plant survival including increased shear forces causing vegetation scour, ice scour, and drought. Reduced vegetative cover in turn allows for greater mobilization of unvegetated sandbar sediment.

established riparian species are limited to a narrow band high up on the channel margin (Figure 8.7).

Even if seed establishment is successful, later submergence from high water levels may kill seedlings from prolonged flooding or directly by excess shear force, scouring by ice blocks, or by drought. Although data was not collected on the occurrence of dead seedlings, several patches of 2–3-year-old willows that were recently killed were observed during the course of our study. Scouring or seedling flooding was also observed on the Wisconsin River by Dixon et al. (2002) and found to be a major cause of seedling mortality. Triplett (2015) documented the lack of trees larger than seedlings (>2–3 years old) except for sandbar willow.

Re-survey of riparian vegetation colonization at the same locations surveyed by Noble (1979) near Mankato and LeSueur shows that increased summer stream flow has raised the elevation at which woody vegetation has colonized on sandbars compared to what Noble found in the 1970s. Plant elevation data showed that riparian woody species (willow (*Salix* spp.) and cottonwood (*Populus deltoides*) that were established in the past decade were about 2.5 m (±0.5 m) higher than found in Noble's survey. Given an average sandbar slope of 10%, this translates to 25 m of unvegetated sandbar that would have been more likely to have been vegetated prior to flow increases. 25 m is roughly half of the average channel widening observed in the Minnesota River between 1938 to 2009.

Later studies by Triplet showed that sandbar willow occurred most frequently and had the greatest coverage on Minnesota River sandbars

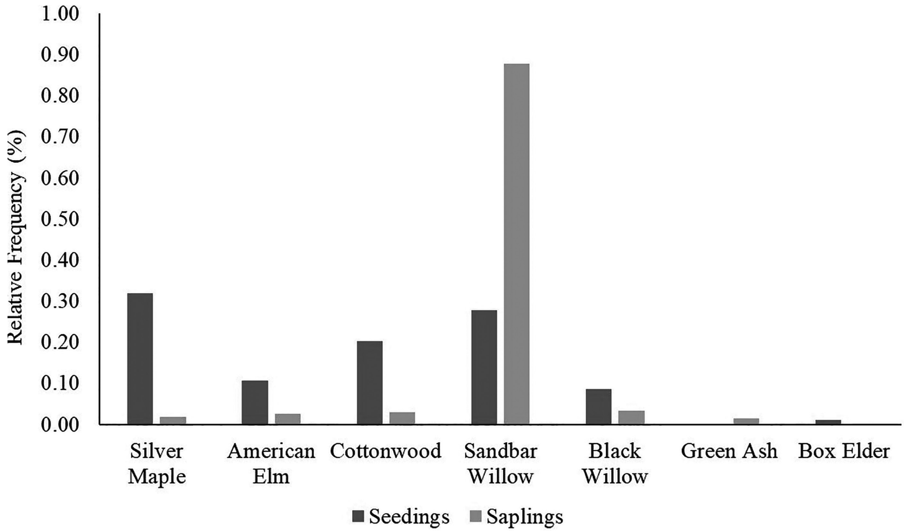

FIGURE 8.8
Relative frequency of woody species surveyed on seven sandbars of the lower Minnesota River, 2013.

(Triplett, 2015) (Figure 8.8). Sandbar willow (*Salix interior*) spreads by clonal growth and is capable of adventitious rooting, adaptations that favor frequent flooding and high sediment deposition rats. Most woody plants other than sandbar willow were found mostly in seedling form (Figure 8.9), less than a few years old suggesting that they were not surviving later, flood, scour, or drought. The seed dispersal window for sandbar willow also extends into August (Figure 8.6) giving it a competitive advantage in rivers with prolonged summer high flow levels. Triplett (2015) found the average date of sandbar submergence across all sampling sites was July 27, indicating that sandbar willow is often the only woody species with seed dispersal continuing into periods when sandbars are exposed at low water levels.

Management Implications

Given the large mass of sediment originating from channel widening, many management questions arise concerning the best course of action for addressing this anthropogenically increased sediment load. One alternative is to simply let the channel re-stabilize on its own over a course of years to decades while pursuing watershed management to reduce stream flow. Yet it unclear how long the Minnesota River may take to evolve to an equilibrium state or whether it wise to just wait. Meanwhile, management agencies such

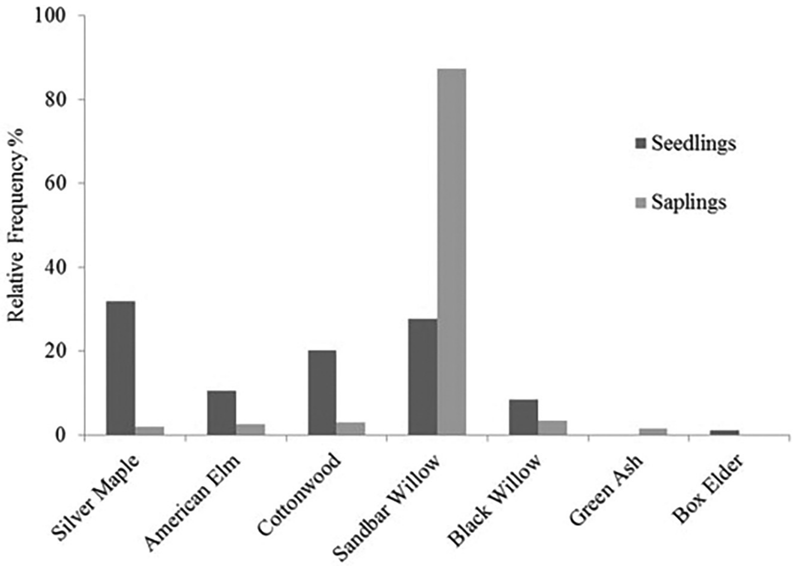

FIGURE 8.9
Relative frequency of seedling and saplings at the seven sandbar sampling sites on the Minnesota River. Relative frequency is obtained by dividing the individual species' frequency by the sum of all the plants' frequencies, multiplied by 100.

as the Minnesota Pollution Control Agency are required to act to improve water quality to meet Total Maximum Daily Load regulations. Observations on the channel evolution process within the region suggest that the transition back to an equilibrium condition takes much longer (decades to centuries) than widening caused by outer bend erosion, which can progress rapidly during large flow events such as the flood of 1993 (Rosgen and Silvey, 1996). Re-stabilization takes longer because increased flows in the Minnesota River have altered riparian vegetation establishment and survival in a variety of ways. Riparian vegetation establishment is highly impacted by increased summer flows (Noble, 1979; Dixon et al., 2002; Triplett, 2015) and now there are suitable plant establishment conditions less often.

Reduction of channel-derived sediment could potentially be accomplished through watershed management to reduce stream flow and by targeted riparian corridor management practices to reduce erosion. Watershed management may be the most sustainable solution in the long term, but is difficult to accomplish in watershed that is predominately privately owned farmland at a time when corn and soybean prices are at an all-time high (Coiner et al., 2001; Santelmann et al., 2004; Nassauer et al., 2011). In contrast, stream restoration actions have the advantage of immediately reducing sediment loading in reaches that deposit sediment directly to rivers. However restoration of streams or stabilization of eroding bluffs is often very expensive, in the range of \$150–\$1,000/linear foot or \$10,000–\$100,000 as per project and up to millions of dollars (Moerke and Lamberti, 2004; Lenhart et al., 2018). Certain broad-scale policies and restoration strategies may have greater benefit-cost ratios than others, such as protecting the valley walls from bluff erosion and restoring sinuosity lost by channelization (Lenhart et al., 2018).

There are aquatic ecological concerns as well. Floodplain management is a challenge in the Minnesota River National Wildlife Refuge (located in the lower reach between Chaska and St. Paul (see Figure 8.1)) where there is a need to balance sediment removal vs. aquatic life needs (aquatic vegetation, fish nurseries, and waterfowl habitat) in backwater wetlands. Sediment removal in floodplains is valuable for downstream water quality but increased sedimentation rates in floodplain lakes and wetlands are detrimental to aquatic vegetation and waterfowl (Sparks et al., 1990; Lemke et al., 2017). One approach toward addressing these conflicting goals that our research suggests would be to develop management units by hydrogeomorphic categories based on soil types and landforms. Under this approach, areas near the channel boundary and inlets to backchannel cutoffs would serve as sediment "forebays", while more diverse peatland wetlands would be protected from sediment deposition. Development of sediment forebays or water control structures could route floodwaters into the near channel boundary and/or recently cutoff oxbows with very high sedimentation rates. This would allow for removal of most of the sediment prior to entering marshes and oxbow lakes where high-quality aquatic vegetation and waterfowl habitat are the management goals.

Conclusions

The lower Minnesota River has widened substantially since 1938 and lost 7% of its length from increased flow and direct channel modifications. Its bankfull shear stress increased in about 50% of river reaches modeled between 1938 and 2009, while stream power increased across the entire lower Minnesota River. This has contributed substantially to the excess sediment problem in the lower Minnesota River and the filling of Lake Pepin in the Mississippi River.

While much sediment is deposited in the Minnesota River valley, it is now less of a sink than it was a century ago. While soil borings show that the floodplain and backchannel cutoffs have high rates of deposition, increased flow and altered riparian vegetation dynamics have possibly reduced deposition on point bars. Evidence is provided that altered vegetation-sandbar interactions contributed partially to the river widening or interacted with it by reducing vegetative establishment, survival, and growth on point bars. This was confirmed by Triplett's (2015) study of Minnesota River sandbars. Though there are complex interactions, she found that hydrologic changes reduced the area and time available for riparian plant seedling establishment and on average, pushed it back later in the growing season. The main species colonizing sandbars was sandbar willow, a shrub that can out-compete others by clonal growth. The ecohydrological mechanism described here likely applies in similar settings, though it should be examined in other geomorphic settings to assess how it applies in different environmental situations.

To expedite a return to channel equilibrium and reduce sediment load in the lower Minnesota River for water quality benefits, there is a need to reduce flow in the Minnesota River and its tributaries. Given the large scale of the watershed and high cost of agricultural land currently, that is a difficult task. Nonetheless, there have been numerous policy initiatives in Minnesota to reduce streamflow by providing greater retention or water storage in the watershed (Lewandowski et al., 2016). There is also a need to develop strategies to target channel stabilization or restoration projects that make sense economically and have ecological benefits for riparian vegetation and aquatic life. Policies could also be pursued such as restoration of sinuosity and more vigorous management of the vegetation in the riparian corridor where high rates of lateral erosion occur. More research is needed on the prioritization of management actions, especially given the high value of agricultural land that currently makes it challenging to retire large areas of land in conservation programs for watershed management efforts (Bangsund et al., 2011; Lenhart et al., 2018). Though there have been many improvements in water quality in the Minnesota River in recent years, there is still a long way to go to achieve state nutrient reduction goals for nutrients and to improve recreational opportunities and aesthetic qualities of the river (Nelson, 2019).

Acknowledgments

The studies were funded by the Minnesota Pollution Control Agency (MPCA) from 2007 to 2010 and the Minnesota Corn Growers Association from 2010 to 2012. Data collection and analysis assistance was provided Brad Hansen, Britta Suppes, Caleb Arika, Ben Underhill, Geoff Kramer, Nick Moore, Michael Talbot, David Zumr, and Zuzana Kunesova. Steve Roos from Minnesota Department of Agriculture provided the channel-vegetation evolution drawings in Figure 8.6. Materials from the Lenhart et al. 2013 article "The Role of Hydrologic Alteration and Riparian Vegetation Dynamics in Channel Evolution along the Lower Minnesota River" in Transactions of the ASABE 56(2): 549-561, doi: 10.13031/2013.42686, were used with permission of the American Society of Agricultural and Biological Engineers (ASABE).

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9

A Comparison of Methods for Prioritizing Lakes in Minnesota

Paul J. Radomski and Kristin Carlson

CONTENTS

Introduction

Identifying lakes in which to invest water quality conservation efforts can help more effectively target efforts and more efficiently utilize limited resources. The objective of this study was to compare different approaches to prioritize Minnesota lakes primarily for water quality protection or restoration. Lakes were objectively ranked using a multi-criteria values-based model that included phosphorus-loading resilience, level of watershed degradation, and feasibility of water quality protection or restoration. We explored

DOI: 10.1201/9781003175094-9

how the list of priority lakes might change when incorporating benefit to cost or "benefit:cost" ratios that used a hedonic model to predict land value increases with total phosphorus-loading reductions. In addition, we examined the influence of including data on lakes with unique or high-quality biological communities. The multi-criteria values-based model was moderately correlated with the benefit:cost ratio approach; however, the exclusion of benefits and cost in the prioritization would likely result in the loss of a modest amount of potential benefit (~20%). A focus on impaired waters would likely result in considerable forgone benefit (~80%) and substantially higher costs. We provide recommendations on how to combine prioritization approaches along with a peer review process to produce lake priority lists that are both defensible and practical.

Minnesota's state government agencies work closely with local governments to manage watersheds and improve water quality through land use controls and water management projects. The state agencies lead the efforts to collect and analyze hydrology, geomorphology, and geospatial data. This information provides guidance for many local and statewide decisions, as well as informs Minnesota's two watershed planning efforts: Watershed Restoration and Protection Strategies and the One Watershed, One Plan.

First, we describe simple standard deliverables that the state provides to local governments for lake hydrologic assessment and prioritization of conservation efforts. These geospatial products assist in determining the relative scale of effort necessary for meaningful lake conservation and can be used in conjunction with other tools (e.g., limnological models, stormwater models, land surface hydrologic models) to evaluate and design various watershed- or site-specific water quality improvement projects. Three Geographic Information System (GIS)-based metrics are provided for about 2,900 lakes: groundwater dominance classification, lake watershed transport capacity classification, and a lake watershed health index.

Groundwater contributions to lakes have a profound influence on nutrient loading and water level stability. Topography, geology, and surface-water drainage area determine the degree to which a lake is dominated by groundwater contributions. Information on a lake's dependency on groundwater is valuable to managers tasked with protection of water quality.

A lake's watershed is the surface area of land that drains into the lake. The ratio of the watershed area to lake area (W:L ratio) is a useful indicator of the relative importance of surface water and groundwater to a lake's water budget (Wetzel 2001). The W:L ratio is strongly correlated with lake water residence time and stream inflow (Gianfagna et al. 2015, Soranno et al. 2015). Minnesota Department of Natural Resources (DNR) has defined watersheds for lakes greater than 10 acres (DNR Watersheds - DNR Level 09). In addition, the hydraulic conductivity of soils and sediment surrounding a lake influence the groundwater importance on the lake's water budget.

Water flow into a lake is a key process that influences the overall mean time that water spends in the lake (water residence time) as well as the magnitude of seasonal and annual water level fluctuations. Changes in the hydrologic input to a lake can have significant impacts on water levels, water quality, and biological communities. Water flow into a lake can also affect water quality, as nutrient loading to lakes depends not only on the availability of nutrients in the watershed but on their potential for movement to a lake. Information on a watershed's ability to transport nutrients will be valuable to managers tasked with protection of water quality and other lake resources.

Assessing a lake's watershed transport capacity (WTC) provides one method to characterize the relative potential hydrologic input to a lake. WTC is a composite index of several critical hydrologic and geologic variables (Fraterrigo and Downing 2008). In lakes with low watershed transport capacity, shoreland development and alteration are often highly impactful and in-lake processes more important, whereas in lakes with high watershed transport capacity, the cumulative consequences of land use across the entire watershed are the dominant cause of declining water quality.

A healthy watershed is able to support functional aquatic ecosystems. The ability to assess the general risk to a lake's watershed may help focus conservation efforts. A simple lake watershed health index can be described as a function of the lake watershed transport capacity and the extent of human disturbances within the watershed.

Second, as threats to Minnesota's lakes continue to mount, it is becoming increasingly important to prioritize where limited conservation funds could best be directed. Within the state, about $34 million/year has been spent on water quality monitoring and impaired waters assessment research and programs, under the requirements of the United States Clean Water Act for state agencies to identify impaired lakes and to study the pollution loads for those waters. Appropriations from Minnesota's Clean Water Fund, which funds a substantial portion of lake and water quality restoration and protection within the state, total about $110 million/year. From 2009 to 2017, 80% of this fund has been spent on restoration projects for impaired waters.

Determining how and where to allocate those funds are critical questions. On which lakes should the state invest its Clean Water Fund? How much funding should go to implement lake protection efforts on unimpaired waters versus restoration efforts on impaired waters? There are many opportunities for lake protection or restoration beyond existing regulatory controls (Radomski and Van Assche 2014). Identifying on which lakes to invest some of these water quality conservation dollars can help more effectively target efforts and more efficiently utilize limited resources. A number of information tools are available for prioritizing and targeting conservation efforts. A systematic approach seems critical in any prioritization (Game et al. 2013). Two of the most common approaches to conservation prioritization are values-based models and benefit:cost ratios.

Values-based models, or value models, use a compilation of individual criteria (valuable features) and aggregated criteria with an objective function to prioritize places on the landscape for conservation (Moilanen et al.

2009). The use of an additive or multiplicative benefits objective function in a value model allows for the retention of as many conservation features as possible. This approach allows the investigator to recognize that attempts to solve clean water needs are not separate from our other conservation needs; some places could provide multiple conservation benefits. Value models provide a reasonable approach when costs are unknown or have high uncertainty; however, they do not provide good guidance on the most cost-effective places to implement different types of projects.

Ranking benefit:cost ratios assesses benefits and costs of projects while explicitly acknowledging that there is a budget constraint on conservation. Several economic studies using hedonic models have shown a relationship between lake water quality and lakeshore property prices (Maine Department of Environmental Protection 1996, Michael et al. 1996, Krysel et al. 2003), so the monetary benefits of water quality protection or restoration efforts may be assessed. Costs to protect or improve lake water quality can often be estimated based on recent conservation efforts. However, environmental problems are often system problems, and the funds available will likely only deal with part of the system or a portion of the problem. The higher the benefit:cost ratio generally means the better the conservation investment, but there are several shortcomings to this approach. First, it does not address non-monetary benefits, costs, or consequences. These non-monetary items may be valuable to a community, and the difficulty in adequately assessing worth of intangibles is often non-trivial. Second, the discount rates, risk premiums, or project risks and feasibilities used in the analysis have a large influence on the calculated result. The strength of this approach is that it attempts to include a measure of the benefits of any projects that might be implemented to protect or restore a conservation feature or asset (Joseph et al. 2009, Pannell et al. 2012, Beher et al. 2016).

Lake restoration and protection prioritizations are readily available. Several biodiversity-based lake prioritizations have been developed (e.g., Duker and Borre 2001, Minnesota Department of Natural Resources 2015), and Heiskary (1997) proposed a lake prioritization approach emphasizing protection of Minnesota's unimpaired waters ranked by lake size. Jacobson et al. (2016) developed and implemented a framework to prioritize Minnesota watersheds to protect lake fish habitat. However, there are few studies that compare different conservation prioritization approaches (Joseph et al. 2009, Pannell and Gibson 2014) and few assessments on lake restoration and protection benefit:cost ratios.

Here, we compare different approaches to prioritize Minnesota lakes primarily for water quality protection or restoration. Lakes at greatest risk of becoming degraded or further degraded were identified and objectively ranked based on their phosphorus-loading resilience, level of watershed degradation, and feasibility of water quality protection or restoration. We explored how the list of priority lakes might change when incorporating benefit:cost ratios. In addition, we examined the influence of including data on unique or high-quality biological communities associated with a subset of these lakes.

The resulting information can be used to identify lakes that may benefit from well-designed phosphorus-loading reduction projects in their watersheds.

Lastly, we review how this information was used by a local government to prioritize lake conservation projects within a watershed.

Material and Methods

Study Lakes and Environmental Data

A subset of Minnesota lakes was used in the analysis (Figure 9.1). For the GIS-based metrics and values-based model, lakes were selected based on the availability of water chemistry data, watershed data, and lake morphological information (n=2,732). For the benefit:cost analysis, lakes were selected based on the availability of land value data (n=1,127). Study lake distribution roughly corresponded with the natural distribution of lakes in Minnesota. Using the Level II Ecoregion Classification (Omernik 1987), 50% of the study lakes were located in the Northern Lakes and Forests ecoregion, 40% in the North Central Hardwood Forests ecoregion, and 6% in the Western Corn Belt Plains ecoregion. The median lake surface area was 79 ha (195 ac). Most lakes (80%) were between 20 and 406 ha (49 and 1,002 ac). Most of the lakes (69%) were deeper water lakes, where the maximum depth was greater than or equal to 15 ft.

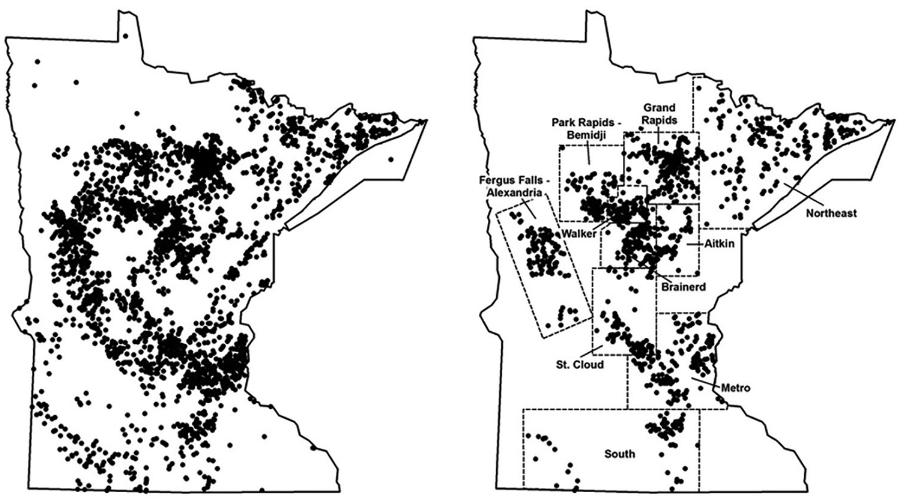

FIGURE 9.1
Locations of lakes used in the lake phosphorus sensitivity significance (LPSS) values-based model and benefit:cost ratio (BCR) analysis. Dotted lines on BCR map represent approximate regional real estate markets. Walker=Walker – Hackensack.

Water chemistry focused on summer conditions, which generally included the period from mid-June to mid-September. In-lake total phosphorus (TP) summer mean concentrations were averaged across all available years for each lake. Water clarity as measured by Secchi disk transparency depth (SDT) was also used. Regression analysis was used to generate an equation of log-transformed mean SDT and average summer mean TP. To determine SDT trends, a seasonal Kendall statistical test was used to determine whether the data for each lake exhibited any trend. Years with four or more SDT readings were used, and only lakes with at least 8 years of data were analyzed. A total of 128 lakes had a decreasing water clarity trend, and 222 had an increasing trend. For use in the value model, a trend score (T) was assigned to each lake based on the significance of the test and the number of measurement years. Lakes with evidence of a negative trend were given one of four scores (0.25, 0.5, 0.75, 1) with the highest value assigned to lakes with strongest evidence of declining water clarity. Lakes with no evidence of a negative trend were given a score of 0.

Physical attributes of the lakes included mean depth, maximum depth, lake volume, hydraulic inflow rate, and a disturbance index of the lake's watershed. Lake volume was available for 1,821 lakes, and for the rest lake volume was estimated by multiplying lake area by maximum depth and 0.464 (Wetzel 2001). Lake hydraulic inflow was estimated using equation 1 in Wilson and Walker (1989), which is a mass-balance equation that uses the lake's watershed size and ecoregion-specific runoff coefficients. Land disturbance within the watershed was estimated by summing the area of land in cultivated and developed land use classes (2011 National Land Cover Data) within the lake's immediate catchment divided by the total land area in the catchment. The mean proportion of watershed disturbance was 0.05 (standard deviation = 0.05) for the lakes in the Northern Lakes and Forests ecoregion, 0.42 (standard deviation = 0.22) for lakes in the North Central Hardwood Forests ecoregion, and 0.69 (standard deviation = 0.17) for lakes in the Western Corn Belt Plains.

The Minnesota Pollution Control Agency impaired lake classification was also used within these analyses. Recreational use impairment designation is a weight of evidence decision based on review of a lake's water quality (TP, chlorophyll a, and SDT data) compared to the regulatory ecoregion-specific thresholds for impairment. The regulatory ecoregion-specific recreational use eutrophication impairment thresholds are documented in Heiskary and Wilson (2005).

Groundwater Dominance, Lake Watershed Transport Capacity, and Lake Watershed Health Index

Lakes with a large W:L ratio (>10) usually have a large surface-water inflow. Lakes with smaller W:L ratios usually have less surface-water inflow, indicating that groundwater is a larger component of the water budget. Petersen and Solstad (2007) identified three basic lake types with regard to connection to groundwater: lakes dominated by surface-water inflow and outflow resulting from a large W:L ratio; lakes dominated by groundwater, occurring in a landlocked basin typically having a small W:L ratio; and lakes intermediate to the above. In addition, the hydraulic conductivity of soils and sediment surrounding a lake influence the groundwater importance on the lake's water budget.

To create a watershed transport capacity classification, we applied the methods used by Fraterrigo and Downing (2008). They used a non-metric multidimensional scaling ordination to stratify Iowa lake watersheds by transport capacity. Variables included watershed area, water residence time, watershed stream length, W:L ratio, percentage of watershed with tile drainage, and the ratio of lake area to lake volume. The top 10% of watersheds were identified as having high transport capacity, and the bottom 10% were classified as having low transport capacity. These classifications were generally associated with high and low values of watershed size and sum of stream length. Our analysis differed from that of Fraterrigo and Downing in two ways. First, the percentage of tile drainage was unavailable, so watershed wetland area and estimated lake inflow were added. Second, rather than stream length, watercourse length was used, which was the sum of the total length of streams and ditches within the lake's watershed.

A lake watershed health index was computed by multiplying the lake watershed transport capacity class times the percentage of watershed disturbed. The results were normalized to produce values between 0 and 100, with 100 representing the best health. Watershed disturbance was estimated by summing the area of land in cultivated and developed land use classes within the lake's watershed divided by the total land area in the watershed.

Values-Based Model Prioritization

A values-based model was formulated to represent the objective of "focusing on high quality lakes at greatest risk of becoming impaired or further degraded." The values-based model is based on both a multiplicative and an additive benefit function, and all lakes were ranked on the resulting priority score. The model had three components. First, the model included a measure of TP loading sensitivity. Several statistical models were developed to predict annual TP loading using the dataset from Brett and Benjamin (2008), which included 305 temperate lakes from North America and Europe. The influence of TP concentration, hydraulic inflow rate, lake volume, lake depth, and flushing rate were analyzed as fixed effects. The best model in the suite of models developed was then used to predict annual TP loading for the 2732 Minnesota lakes. A lake's TP loading sensitivity index (S) was estimated using the mass-balance limnological equation from Cheng et al. (2010; equation 6), which predicts in-lake TP as a function of annual TP loading and the lake's mean hydraulic retention time. To determine the sensitivity of each lake to additional loading, increasing TP loads were entered into the mass-balance equation and predicted SDT depths were made for each increasing load. The TP loading sensitivity was then expressed as the loss of SDT in inches per 45.36 kg (100 pounds) of TP added.

The second component was an index, denoted as TP loading sensitivity significance (*SS*), which was computed using the TP loading sensitivity index (*S*) times three multipliers. For lakes where the ratio of the predicted TP load (L_p) to the TP load threshold (L_t) was 1 or less, a lake's TP loading sensitivity significance was calculated by:

$$SS = S \times A / TP \times L_p / L_t \times D$$

where the multipliers were the ratio of lake surface area (*A*, acres) to the in-lake summer mean TP concentration (*TP*), the ratio of predicted TP load (L_p) to the TP load threshold (L_t), and the proportion of the lake's watershed that was disturbed (*D*; proportion developed plus the proportion cultivated). These lakes could be generally considered assets to protect.

For lakes where the predicted TP load to TP load threshold ratio was greater than one, TP loading sensitivity significance index (*SS*) was calculated by:

$$SS = S \times A / TP \times [1 - (L_p / L_t - \min(L_p / L_t)) / (\max(L_p / L_t) - \min(L_p / L_t))] \times D_n$$

where the predicted *TP* load to *TP* load threshold ratio was normalized to between 0 and 1 and the proportion of the lake's watershed that was disturbed was multiplied by a normal probability density function with a mean of 0.4 and a standard deviation of 0.2 and then normalized between 0 and 1 (D_n). The latter multiplier placed more significance on lakes with moderate watershed disturbance, as the results of Cross and Jacobson (2013) showed a critical threshold of anthropogenic land use disturbance at 40% that once exceeded could significantly alter in-lake TP concentrations. These lakes could be generally considered assets to restore.

The final component of the values-based model aggregated priority by summing two attributes:

$$\text{Priority score} = 0.5 * T + SSn$$

where the lake phosphorus sensitivity significance (LPSS) priority score for the lake for protection or restoration is equal to one-half of the decreasing water clarity trend score *T* plus the normalized TP loading sensitivity significance index *SSn*. The water clarity trend score, *T*, is based on the *P* value of a seasonal Kendall test applied to June through September transparency data for the lake, values range from 0 to 1 with 0 showing no trend and 1 having strong evidence for a negative trend (Heiskary and Egge 2016). The priority score was then normalized between 0 and 100. All normalization, or rescaling, followed the general formula:

$$X' = \left(X - \min(X)\right) / \left(\max(X) - \min(X)\right) \times \left((U - L) + L\right)$$

where X is the original value, X' is the normalized value, U is the upper scale range, and L is the lower scale range.

Benefit:Cost Ratio Analysis

Benefit:cost ratios were developed using information from 1,127 lakes. The lakes used in these analyses were a subset of those used in the values-based model ($n=2{,}732$ lakes), and they were generally representative of the full dataset (Table 9.1). As with the values-based model, lakes were ranked or prioritized; in addition, for the benefit:cost ratio analysis each lake also had explicit project activities assigned. County parcel data available to the MDNR were used to calculate the mean shoreline parcel value ($) and mean shoreline parcel length (m) for each lake, as well as total lake shoreline value. Parcel value was defined as estimated land value, in order to eliminate the effect of wide differences in building values and to include undeveloped parcels. In some cases, because of gaps in the parcel data, land values were not available for the entire shoreline of a lake; lakes with parcel data for <50% of the shoreline were excluded from further analysis. Based on location, lakes were assigned to one of ten regional real estate markets.

Economic models were developed to predict land values across a range of Minnesota lakes. Hedonic linear regression models were used employing a generalized least squares approach; this approach extends regression by modeling the heterogeneity with covariates. The model development

TABLE 9.1

Attributes of the Two Lake Datasets Used to Develop the Values-Based Model and the Benefit:Cost Ratios (BCR)

Attribute	Number of Lakes Used for Values-Based Model	Number of Lakes in the BCR Subset
Managed fish lakes	2355 (86%)	1127 (100%)
Recreational use impaired lakes	522 (20%)	208 (18%)
Large lakes (>500 ha)	583 (21%)	133 (12%)
Lakes of Biological Significance	779 (29%)	418 (37%)

strategy followed the suggestions of Zuur et al. (2009), with mean land value per shoreline frontage ($/shoreline m) as the response variable. The influence of lake size (m^2), maximum lake depth (m), lake mean summer TP concentration (μg/L), proportion of frontage in private ownership, ecoregion, mean shoreline parcel length (m), and real estate market were analyzed as fixed effects. After initial testing to determine significant fixed effects, candidate models were developed that addressed variance covariate structure. The changes in the AIC score were used to select a preferred model (Burnham and Anderson 2002). Statistical analyses were conducted using R (R Development Core Team 2017).

The benefits of water quality protection or restoration activities were calculated for each lake using the preferred economic model to estimate increases in land value ($/shoreline m) with management activities that were assumed to reduce TP loading by 5%. For unimpaired lakes, a 5% load reduction goal is currently being recommended by Minnesota agencies as a reduction in the amount of pollution entering a lake that watershed partners can reasonably strive for in guiding local stewardship practices.

The costs of water quality protection or restoration activities assigned to each lake depended on the lake and its watershed characteristics. For minimally disturbed lakes in forest watersheds (watersheds with no land in cultivation and less than 10% in developed land use and little or no shoreland development), protection costs were based on the amount of riparian land necessary to maintain a portion of shoreland in forest based on differences between TP loading from woods and developed lakeshores to achieve the 5% TP load reduction and the typical Minnesota conservation easement cost for state agencies. The estimated TP load difference was 0.3 kg/ha/year [0.46 kg/ha/year in residential development – 0.14 kg/ha/year in forest (0.3 pounds/ac/year)] (Graczky et al. 2003, Radomski and Van Assche 2014). The Minnesota cost of acquiring and enforcing conservation easements for each lake was estimated by multiplying the riparian area (61 m or 200 ft landward) needed to achieve the 5% reduction for the lake by 60% of the observed mean land value for the lake per hectare (Minnesota Office of the Legislative Auditor 2013). For lakes with disturbed watersheds, a lake's restoration cost was the cost of agricultural and stormwater best management practices (BMPs) per kilogram TP removed, assessed proportionally based on agricultural and developed land use within the lake's watershed, multiplied by the 5% load reduction goal (kg/year) for the lake. For agricultural BMPs, the cost per kilogram was assessed at $39/kg ($18/pound) (Johansson et al. 2004), which is within the range of projected costs for a variety of BMPs appropriate for Minnesota (Lazarus et al. 2015). For stormwater BMPs, the cost per kilogram was assessed at $46,298/kg ($21,000/pound) (Hunt et al. 2012, Houle et al. 2013).

Consistent with methodology outlined by Pannell (2015), the benefit:cost ratio included multipliers for probabilities of a lake's protection or restoration activities being successful. These multipliers adjusted raw benefit:cost ratios based on the likelihood of project success (not all lake protection and restoration projects succeed). These multipliers were impartially, but subjectively, ascribed. The benefit:cost ratio (BCR) was calculated as follows:

$$\mathrm{BCR} = \left(\left(\mathrm{LV}_{\mathrm{p5\%}} - \mathrm{LV}_{\mathrm{pe}}\right) \times \mathrm{SM} / C\right) \times P_t \times P_{sp}$$

where for each lake the $\mathrm{LV}_{\mathrm{p5\%}}$ is the predicted mean land value per shoreline frontage with a 5% reduction in TP loading ($/shoreline m), $\mathrm{LV}_{\mathrm{pe}}$ is the predicted mean land value per m shoreline frontage for existing conditions, SM is the shoreline length (m), C is the cost of water quality protection or restoration activities, P_t is the probability of technical feasibility, and P_{sp} is the probability of social and political willingness to act and fund the lake's protection or restoration. The probability of technical feasibility ranged from 0.4 to 0.9, and it decreased log-linearly based on the amount of disturbed land in the lake's watershed (i.e., lakes with large, disturbed watersheds were assumed to be more technically challenging to successfully identify and target agricultural and stormwater BMPs to achieve the 5% TP load reduction). The probability of social and political willingness ranged from 0.1 to 0.9 and it increased log-linearly based on the total riparian land value (i.e., social capacity and political willingness increased as the wealth of the lake community increased). Statistical differences between mean BCR by different classes of lakes were tested with the Mann-Whitney U test (SAS 2017).

Lake Biological Community Prioritization

As an example of how non-monetized benefits may influence prioritization, we included important biological community lakes within the analysis. The Minnesota Department of Natural Resources (MDNR 2015) created a list of high-quality lakes based on dedicated biological sampling for the stated purpose of focusing protection efforts (Lakes of Biological Significance; $n = 1{,}449$ lakes). Lakes were rated and grouped for each of the following communities: aquatic plants, fish, birds, and amphibians. Lakes were assigned one of three biological significance classes (outstanding, high, or moderate). The goal of this list was to identify lakes that exhibit the highest quality features within any of the four assessed biological communities (as opposed to identification of lakes that exhibit diversity across communities). Therefore, a lake needed to meet criteria for only one of the community types (aquatic plants, fish, birds, amphibians) to be identified as a Lake of Biological Significance. Occurrences of high-quality features within the community types determined the biological significance class. About half the lakes on this list were also used in the values-based model.

Comparison of Prioritizations

Values-based model and benefit:cost ratio absolute outputs as well as their associated lake rankings were used to generate lists of high priority lakes. Simple comparisons were then made using the computed scores or ranks. Additional lake priority lists were also developed using two or more scores where the distance from the origin in either two or three dimensions determined the priority rank. This is a simplified multi-criteria decision analysis method, where the highest priority lake was selected based on the longest geometric distance from the worst solution, i.e., the origin, which has a zero score for all criteria. Distance from the origin was calculated using the Pythagorean distance formula, and dimensions included the normalized LPSS priority score from the values-based model, normalized score of BCR, and classes of Lakes of Biological Significance (0 for not designated, 0.33 for moderate, 0.67 for high, and 1 for outstanding lakes).

Results

Groundwater Dominance, Lake Watershed Transport Capacity, and Lake Watershed Health Index

Based on the watershed to lake area ratio and local geology, DNR hydrologists proposed a classification for whether a lake was groundwater dominated (Table 9.2). W:L ratio, watershed watercourse length, and water residence time were important variables in the watershed transport capacity ordination, and the latter two variables were important in distinguishing transport capacity between watersheds (Figure 9.2). A lake WTC class was assigned to each lake based on these two variables. Seven classes were created, with class 1 having the lowest WTC and class 7 having the highest WTC (Table 9.3). Most lakes with available data had large watersheds with considerable surface-water drainage. Lakes with high WTC classification often had a low lake watershed health index (Figure 9.3). This index can be used to compare risk across a group of lakes within a local jurisdiction or watershed.

Values-Based Model Prioritization

Lake TP concentrations were generally lowest in the Northern Lakes and Forests ecoregion and higher in the Western Corn Belt Plains ecoregion (Figure 9.4). The best model to predict TP loading was a linear log-log regression model, with in-lake TP concentration, lake volume, and hydraulic inflow

TABLE 9.2

Lake Groundwater Dominance Classification

Local Geology	Watershed/Lake Area Ratio	Groundwater Dominated
All textures	>10	Not likely
Sandy	5–10	Likely
Sandy	<5	Very likely
Loam or clay	<5	Possible

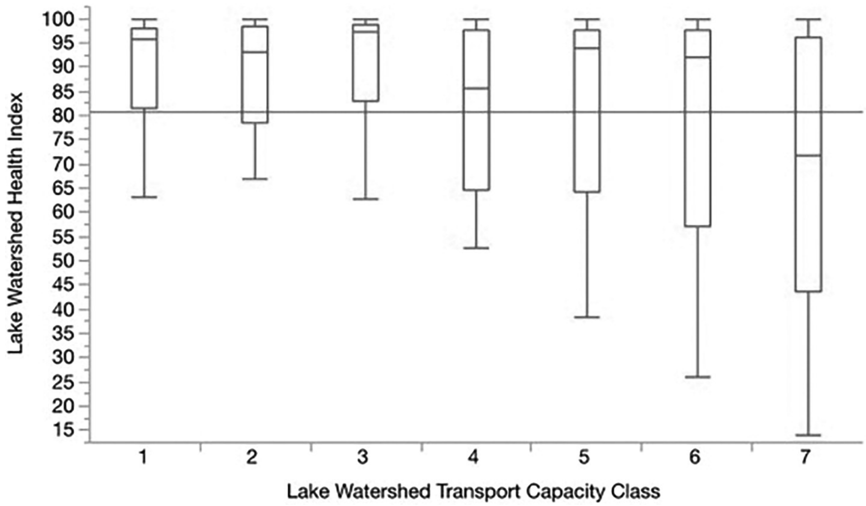

FIGURE 9.2
Non-metric multidimensional scaling ordination to classify lake watersheds by transport capacity. Contour lines of W:L ratio (log-transformed; in red) and watercourse length (log-transformed, in blue) overlay the ordination.

TABLE 9.3

Lake Watershed Transport Capacity Classes (WTC) Based on W:L Ratio and Watercourse Length

W:L Ratio	Watercourse Length	WTC	% of Lakes
>10	>75th percentile	7	24
>10	Median to 75th percentile	6	19
>10	<Median	5	16
5–10	≥Median	4	6
5–10	<Median	3	16
<5	≥Median	2	2
<5	<Median	1	18

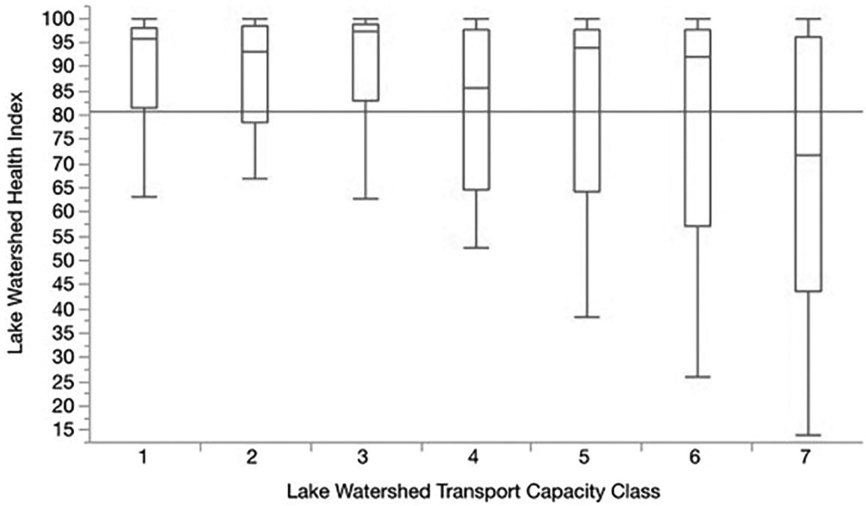

FIGURE 9.3
Box plots of lake watershed health index by lake watershed transport capacity class. The box is the interquartile range. The vertical endpoints are not longer than 1.5 times the interquartile range, and the line within the box in the median. The horizontal line is the mean for all lakes.

TABLE 9.4

A Summary of the Linear Regression Model for TP Loading (the Log-Transformed Response Variable)

Source of Variation	Coefficient	SE	*T*	*P*
Intercept	0.3349	0.0585	5.7221	<0.0001
logTP_lake	1.0470	0.0332	31.5394	<0.0001
logQ	0.8169	0.0150	54.5380	<0.0001
logV	0.2986	0.0268	11.1305	<0.0001
logTP_lake*logV	–0.9450	0.0163	–5.7980	<0.0001

The explanatory variables included log-transformed TP concentration (logTP_lake), log-transformed hydraulic inflow rate (logQ), log-transformed lake volume (logV), and one interaction term (*).

rate as input variables (multiple R^2=0.9689; adjusted R^2=0.9685; Table 9.4). The fitted values showed no bias about the observed values, and the average absolute percent difference between the observed and fitted values was 44% (standard deviation=38%). The prediction intervals for the 2,732 Minnesota lakes were wide (80% prediction intervals were –48% to +95%).

The TP loading sensitivity index was generally highest for oligotrophic lakes in the Northern Lakes and Forests ecoregion and lower for eutrophic lakes in the Western Corn Belt Plains ecoregion (Figure 9.4). Many lakes with the top LPSS priority scores were located in the ecological transition zone from Detroit Lakes southeast to Minneapolis and in northcentral Minnesota.

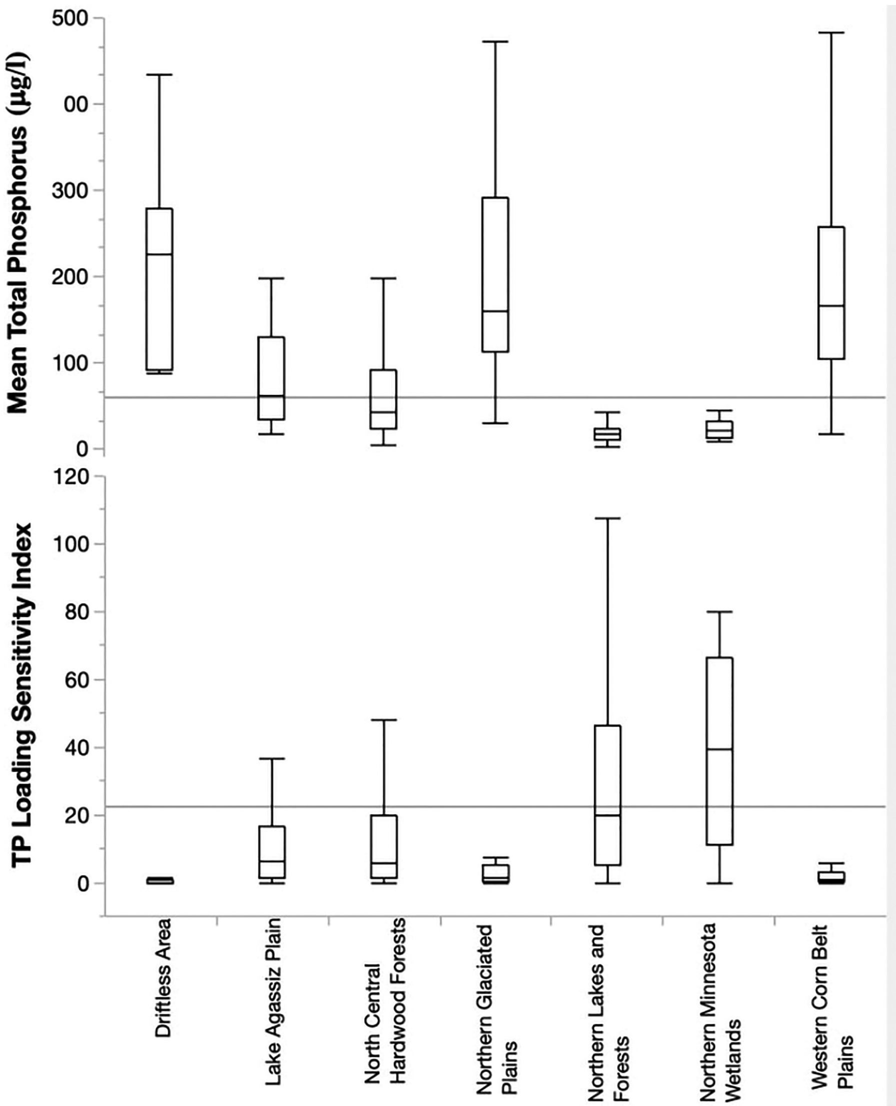

FIGURE 9.4
Box plots of lake summer mean total phosphorus (TP) concentrations and TP loading sensitivity index for lakes grouped by ecoregion. The box is the interquartile range. The vertical endpoints are not longer than 1.5 times the interquartile range, and the line within the box in the median. The horizontal line is the mean for all lakes.

Lake watershed size was an important factor in this index; lakes with large watersheds were less likely to have high indices. As intended, the LPSS priority score generally produced high values for oligotrophic lakes that were vulnerable to phosphorus loading and near their estimated loading threshold, and low values for small, hypereutrophic lakes with high estimated phosphorus loading and watershed disturbance.

Benefit:Cost Ratio Prioritization

Lakes in the Brainerd or Metro real estate markets had the highest land value ($/shoreline m; Figure 9.5). These markets also had higher land value variability. The Grand Rapids, Northeast, and South real estate markets had the lowest variability in lakeshore land value. The average lake mean land value was $1,750/shoreline m (standard deviation=2,101), and the maximum was $19,224/shoreline m (Lake Minnetonka).

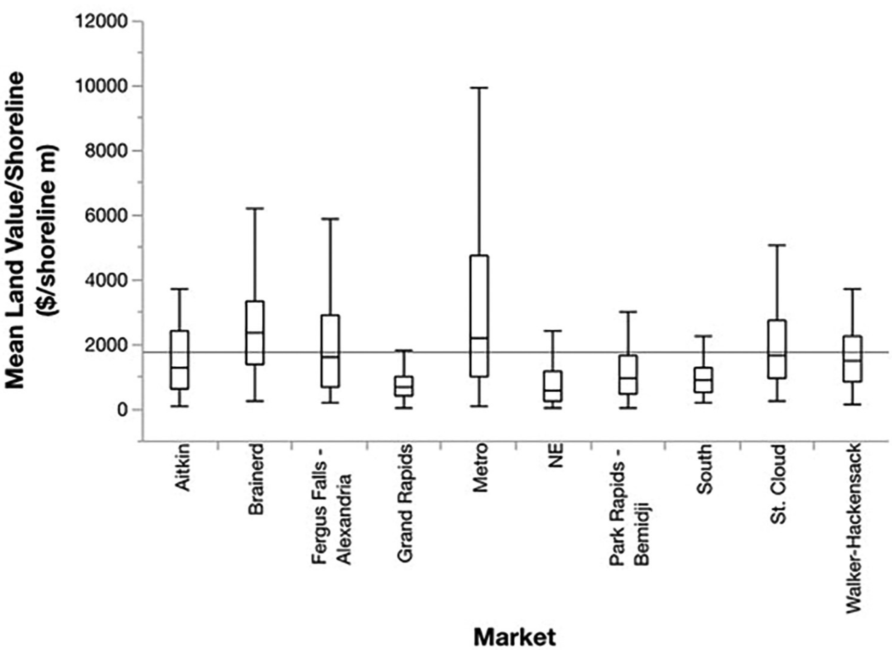

FIGURE 9.5
Box plots of mean land value per shoreline distance (m) for lakes grouped by real estate market. The box is the interquartile range. The vertical endpoints are not longer than 1.5 times the interquartile range, and the line within the box in the median. The horizontal line is the mean for all lakes. NE = Northeast.

The preferred hedonic model that predicted land values ($/shoreline m) included lake size, maximum lake depth, lake mean summer TP concentration, mean shoreline parcel length, real estate market, several interactions as fixed effects, and an exponential function of the variance covariate for the mean shoreline parcel length (this variance structure allowed for an increase in the residual variance for this fixed effect) (Table 9.5). Mean land value decreased with increasing lake mean total phosphorus concentration, and it increased with lake size and maximum depth (Figure 9.6; predictions for Brainerd real estate market).

TABLE 9.5

A Summary of the Preferred Economic Hedonic Linear Regression Model Using a Generalized Least Squares Approach to Predict Land Value (the ln-Transformed Response Variable)

Source of Variation	Coefficient	SE	*T*	*P*
Intercept	1.2522	1.4830	0.8444	0.3986
Market – Aitkin	0			
Market – Brainerd	0.8287	0.4979	1.6642	0.0963
Market – Fergus Falls-Alexandria	–0.6568	0.4536	–1.4479	0.1479
Market – Grand Rapids	–0.9450	0.5026	–1.8802	0.0603
Market – Metro	0.2916	0.4334	0.6727	0.5013
Market – Northeast	–1.7968	0.4395	–4.0885	<0.0001
Market – Park Rapids-Bemidji	–0.8703	0.5271	–1.6512	0.0990
Market – South	–1.0411	0.5213	–1.9972	0.0460
Market – St. Cloud	–0.3904	0.4958	–0.7876	0.4311
Market – Walker-Hackensack	–0.9033	0.5958	–1.5160	0.1298
LN_LAKEAREA	0.7163	0.1012	7.0795	<0.0001
LN_MEAN_FF	0.0184	0.2711	0.0679	0.9459
LN_TP	1.0823	0.2727	3.9687	0.0001
LN_MAXDEPTH	0.2141	0.0266	8.0467	<0.0001
LN_LAKEAREA*LN_MEAN_FF	–0.0699	0.0180	–3.8865	0.0001
LN_LAKEAREA*LN_TP	–0.0893	0.0193	–4.6397	<0.0001
Market – Brainerd*LN_MEAN_FF	–0.0926	0.1184	–0.7827	0.4340
Market – Fergus Falls-Alexandria*LN_MEAN_FF	0.2290	0.1058	2.1636	0.0307
Market – Grand Rapids*LN_MEAN_FF	0.2078	0.1108	1.8760	0.0609
Market – Metro*LN_MEAN_FF	0.1687	0.1020	1.6535	0.0985
Market – Northeast*LN_MEAN_FF	0.3934	0.0990	3.9726	0.0001
Market – Park Rapids-Bemidji*LN_MEAN_FF	0.2182	0.1183	1.8452	0.0653
Market – South*LN_MEAN_FF	0.3986	0.1164	3.4244	0.0006
Market – St. Cloud*LN_MEAN_FF	0.1905	0.1182	1.6119	0.1073
Market – Walker-Hackensack*LN_MEAN_FF	0.2806	0.1329	2.1114	0.0350

Variables include the various real estate markets and several ln-transformed variables: lake size (LN_LAKEAREA), mean shoreline parcel length (LN_MEAN_FF), lake mean summer TP concentration (LN_TP), and maximum lake depth (LN_MAXDEPTH).

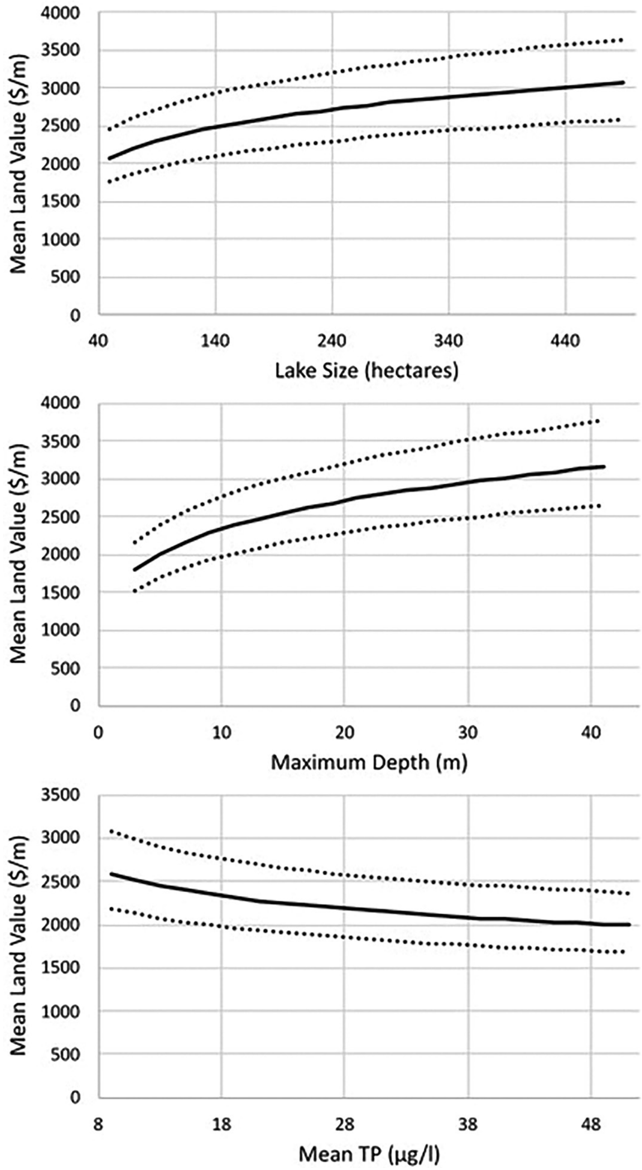

FIGURE 9.6
Mean land value ($/shoreline m) as a function of lake size, maximum depth and summer mean total phosphorus (TP) concentration for lakes in the Brainerd real estate market. Upper panel predictions varied lake size with summer mean TP set at 18 μg/L, maximum depth at 10 m, and mean shoreline parcel length at 61 m. Middle panel predictions varied maximum depth with lake size set at 100 ha, summer mean TP set at 18 μg/L, and mean shoreline parcel length at 61 m. Lower panel predictions varied summer mean TP with lake size set at 100 ha, maximum depth at 10 m, and mean shoreline parcel length at 61 m. Dotted lines represent 90% prediction intervals.

The median estimated lake protection or restoration costs for management activities that assumed a TP loading reduction of 5% for a set of 1,127 Minnesota lakes was $243,000 (Table 9.6; $32/shoreline m, range: $0.2–$48,000/shoreline m). For most lakes, the cost was below $1 million; several lakes had exorbitant costs and these lakes were impoundments on large rivers or floodplain lakes of large rivers where their watersheds and hydraulic loading volumes were large. Small lakes, in forested watersheds or in watersheds dominated by agriculture, had the lowest cost for protection or restoration. For minimally disturbed lakes in forested watersheds, the median cost of using conservation easements for TP loading protection was $15,626/kg ($7,088/pound), or $66,000/lake ($n$=200 lakes; $9/shoreline m), where the median conservation easement size was 12 lakeshore ha (28 ac). The median cost for agricultural-dominated watersheds (>50% of the watershed disturbed and >75% of the disturbance was due to cultivated crops) was $245,000 ($n$=93 lakes; $30/shoreline m), and the median cost for urban-dominated watersheds (>10% of watershed disturbed and >75% of the disturbance was due to developed land classes) was $422,000 ($n$=92 lakes; $54/shoreline m).

The median benefit, measured as the total land value increase for a lake assuming a successful 5% reduction in TP loading resulting in improved water quality, was $58,000 (Table 9.6; n=1,127; $8/shoreline m; range $0.3–$350/shoreline m). Benefit was correlated to lake surface area (r^2=0.67, power-law function). Large lakes or lakes in the Metro real estate market were estimated to have benefits of TP load reduction near or over $1 million (e.g., Minnetonka, Leech, Vermilion, Gull, Otter Tail, Pelican). Small lakes (<100 ha) generally had the lowest benefits. There was a poor relationship between cost and benefits (r^2=0.22).

The median BCR by real estate market were highest in the Fergus Falls – Alexandria market, followed by the St. Cloud, Metro, and Brainerd markets

TABLE 9.6

Quantiles and Summary Statistics of Benefits (Total Land Value Increase in $), Lake Protection, or Restoration Costs for Management Activities That Were Assumed to Reduce TP Loading by 5%, and Benefit:Cost Ratio (BCR) for 1127 Minnesota Lakes

Variable	Benefit	Cost	BCR
Maximum	64,349,000	645,799,000	8.84
75th quartile	177,000	968,000	0.24
Median	58,000	243,000	0.07
25th quartile	22,000	82,000	0.02
Minimum	2,000	1,000	0.00
Mean	360,000	2,322,000	0.27
Standard deviation	2,297,000	20,666,000	0.62

Values rounded to the nearest thousand.

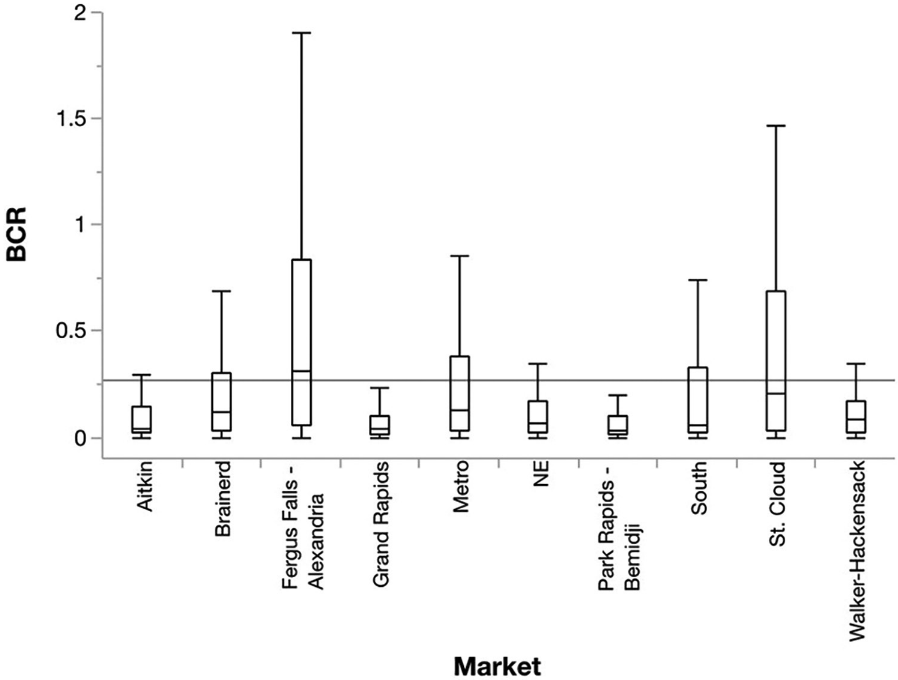

FIGURE 9.7
Box plots of benefit:cost ratio (BCR) for lakes grouped by real estate market. The box is the interquartile range. The vertical endpoints are not longer than 1.5 times the interquartile range, and the line within the box in the median. The horizontal line is the mean for all lakes. NE, Northeast.

(Figure 9.7, Table 9.7). Lakes in the Fergus Falls – Alexandria market generally had lower costs given their watersheds had a greater proportion of land in agricultural use, and they had a substantial number of lakes estimated to be responsive to a 5% TP reduction. Lakes in the St. Cloud, Metro, and Brainerd markets, with their higher land value, generally had high benefits as measured in increased land value with TP reduction. The top BCR lakes were clustered around Fergus Falls, west and south of Minneapolis, in north-central Minnesota, and scattered throughout the northeast.

Some priority lakes based on this analysis that were not a high priority based on LPSS include Lake Minnetonka (Metro market), Black Duck Lake (Northeast market), and Washington Lake (South market). Lake Minnetonka is a large, high land value lake on the outskirts of Minneapolis; it was predicted that restoration efforts that reduced TP loading by 5% may increase total land value by $64 million. Black Duck Lake is a minimally developed lake in the northern part of the state. While the benefits of protecting Black Duck Lake were modest ($250,000), the cost of protection was low ($160,000). Protecting Black Duck Lake's shorelands, 39% of which were in private

TABLE 9.7

For the Top 20 BCR Lakes by Location, Total Benefit, Total Cost, and Median BCR for Management Activities That Assumed a TP Loading Reduction of 5%

Location	Total Benefit ($)	Total Cost ($)	Median BCR (Range)
Ecoregion			
Northern Lakes and Forests	27,111,000	10,506,000	0.96 (0.69–5.23)
North Central Hardwood Forests	81,920,000	11,790,000	3.26 (2.32–8.84)
Western Corn Belt Plains	2,155,000	8,515,000	0.03 (0.01–0.46)
Real Estate Market			
Aitkin	6,589,000	14,116,000	0.18 (0.09–1.09)
Brainerd	27,394,000	11,989,000	0.69 (0.50–5.23)
Fergus Falls – Alexandria	22,666,000	4,322,000	2.30 (1.20–8.84)
Grand Rapids	12,327,000	10,759,000	0.24 (0.17–0.78)
Metro	86,481,000	15,833,000	1.57 (0.85–5.35)
Northeast	7,112,000	6,517,000	0.48 (0.32–0.91)
Park Rapids – Bemidji	8,081,000	13,101,000	0.17 (0.11–3.10)
South	4,127,000	2,892,000	0.44 (0.24–2.49)
St. Cloud	8,766,000	2,444,000	1.18 (06.5–5.56)
Walker – Hackensack	36,913,000	85,828,103	0.25 (0.16–0.69)

ownership, via conservation easements may provide a good return on investment. Lastly, Washington Lake is a highly developed, southern lake near the city of Mankato; it is a popular lake for water recreation. Washington Lake's watershed is dominated by cultivated agricultural land use. If low-cost agricultural practices could be effectively implemented to reduce TP load to the lake, then those efforts may produce sufficient water quality benefits to increase total shoreland value.

Several types of lakes had high mean BCR. Large lakes (>500 ha) had significantly higher mean BCR than small lakes (Mann-Whitney U test, $P<0.0001$). High land value lakes, where the total shoreland value was >$48 million (the 90th percentile, $n=113$ lakes) had significantly higher mean BCR than lakes with lower total value shorelands (Mann-Whitney U test, $P<0.0001$). Lakes of Biological Significance (LOBS) had significantly higher mean BCR than lakes that were not (Mann-Whitney U test, $P=0.0431$). Lakes that were estimated to be highly vulnerable to additional TP loading (lakes with a TP loading sensitivity index, S, greater than the 90th percentile, $n=113$ lakes) had significantly higher mean BCR than those that were less sensitive (Mann-Whitney U test, $P<0.0001$).

Two classes of lakes had low mean BCR. First, lakes with high predicted TP load to TP load threshold ratios (ratios between 0.75 and 1.0, $n=131$ lakes) had lower mean BCR than other lakes (Mann-Whitney U test, $P<0.0066$). These lakes, which may have higher probability of tipping into recreational use impairment, generally had higher protection or restoration costs. Second, and related, the mean BCR for lakes listed as recreational use impaired was significantly lower than unimpaired lakes (Mann-Whitney U test, $P<0.0001$). The mean BCR for impaired lakes was 0.14 ($n=208$ lakes) compared to 0.30 for unimpaired lakes ($n=919$ lakes). If restoration efforts focused on impaired lakes, prioritizing those lakes ranked by BCR, then the restoration costs for the top 100 ranked lakes would have a cumulative cost of $80 million and a cumulative benefit of $34 million in total land value increase. For the same $80 million, selecting any high BCR-ranked lake in the state without regard to impairment status, protection, and restoration activities could be conducted on 198 lakes (versus 100) and the benefit would be $209 million (versus $34 million). Prioritizing for impaired lakes resulted in a 49% reduction in the number of lakes and 84% loss in benefits. Prioritizing without regard to impairment status was predicted to have a six times greater return on investment than focusing on impaired lakes with high BCR. Only a few impaired lakes would be targeted for restoration with the any-lake BCR prioritization approach, i.e., only the highest BCR impaired lakes would be included for restoration with such a prioritization.

Comparison of Prioritizations

The values-based model (LPSS) and BCR prioritizations shared many lakes in their respective top 100 ranked lakes, and these two prioritizations were moderately correlated (Figure 9.8; for values $r^2=0.45$; for their ranks $r^2=0.43$). Sixty-nine lakes scored in the top 100 ranked lakes for both LPSS and BCR, and most of these lakes are in the ecological transition zone from Detroit Lakes southeast to Minneapolis (Figure 9.9). Sixty-one lakes were in the top-ranked lakes for the LPSS prioritization that were not in the top-ranked lakes for BCR (61/130=47%; the total lakes summed to more than 100 in the LPSS priority score due to ties). Thirty-one lakes were in the top-ranked lakes for the BCR prioritization that were not in the top-ranked lakes for LPSS (31%; no ties in the BCR). Notably, several of these lakes were minimally disturbed lakes in forested watersheds, and these lakes had very low LPSS priority scores and ranks because their risk of becoming impaired was low (those lakes are visible in Figure 9.8, lower panel, as a line of points in the upper left).

LPSS and BCR priorities also differed regarding high land value lakes—high land value lakes generally had lower LPSS priorities than comparable BCR-ranked lakes. If protection and restoration efforts focused on the top 100 LPSS lakes, the cumulative cost was $30 million and the cumulative benefit was $124 million. For the same $30 million cost, selecting lakes just by BCR would get 77 lakes with a cumulative benefit of $143 million (a 15% increase in benefits). If protection and restoration just focused on the top 100 BCR lakes, the cumulative cost was $36 million and the cumulative benefit was

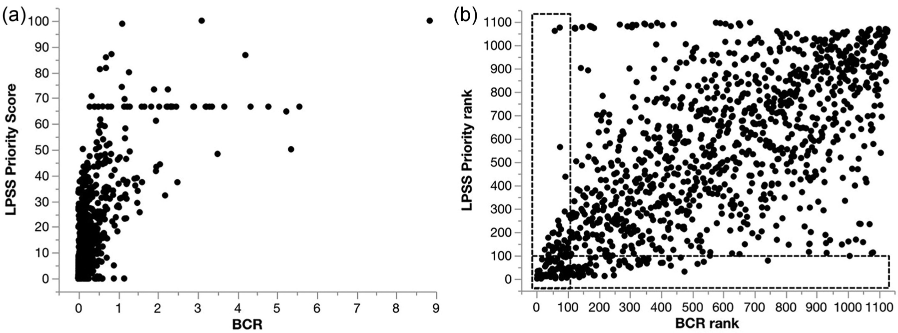

FIGURE 9.8
(a) The benefit:cost ratio (BCR) for lakes plotted against the lake phosphorus sensitivity significance (LPSS) priority score (upper panel; $r2 = 0.45$). (b) The BCR rank plotted against the LPSS priority rank (lower panel; $r2 = 0.43$). The dash boxes show the top 100 ranked lakes for BCR and LPSS.

$154 million (compared to the LPSS prioritization, a 20% increase in cost with a 24% increase in benefits).

From a statewide perspective, lake protection and restoration priorities vary spatially based on the different prioritization approaches (Figure 9.10). LPSS and BCR priorities both focus on lakes located in the ecological transition zone from Detroit Lakes southeast to Minneapolis and in northcentral Minnesota. The top BCR priorities include more lakes in northeastern Minnesota. The two-dimensional BCR-LPSS prioritization produces a spatial distribution that blends the two, while the top three-dimensional BCR-LPSS-LOBS priority lakes tend to be in northcentral and northeast Minnesota, where a large proportion of LOBS lakes exist.

Lake Conservation Case History

In Minnesota, county Soil and Water Conservation Districts (SWCDs) assist private property owners in the conservation of soil, water, and related natural resources on their lands. For many lake-rich Minnesota counties, the need for conservation exceeds local government capacity to deliver effective and meaningful solutions. For example, Crow Wing County has 159 lakes >50 ha (124 ac), and many of these lakes' watersheds are dominated by private property. There is a great interest in protection of lake water quality across the county, and the Crow Wing SWCD must prioritize, assess, and administer construction projects to meet reasonable and attainable conservation goals.

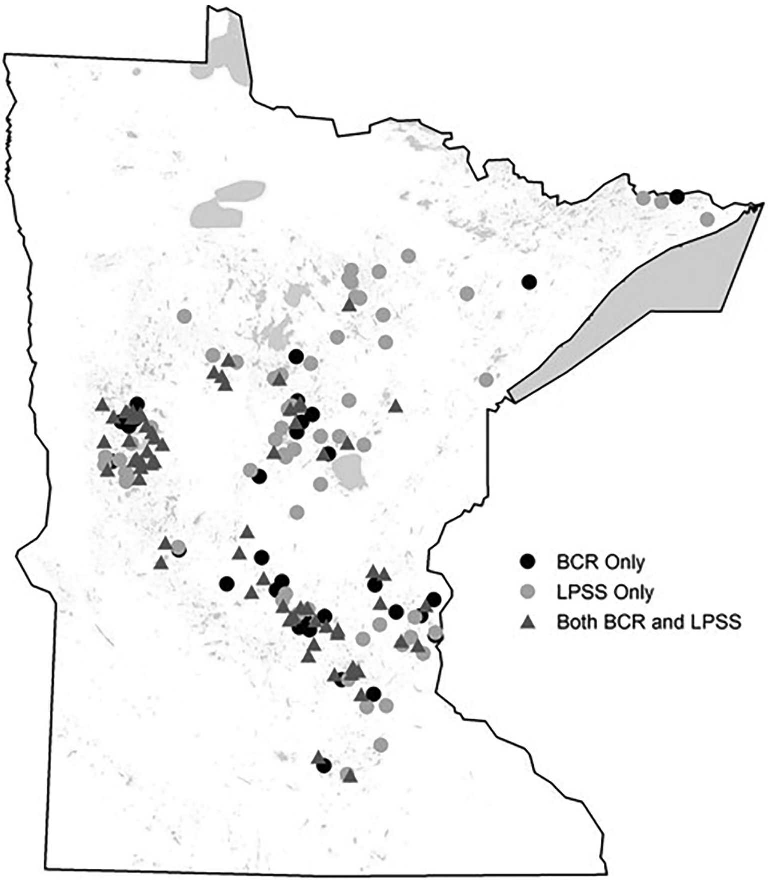

FIGURE 9.9
The top 100 ranked lakes by benefit:cost ratio (BCR) and lake phosphorus sensitivity significance (LPSS). Multiple lakes scored in the top 100 for both BCR and LPSS (triangles), $N=69$ lakes.

The prioritization tools reviewed here are used by SWCDs to assist them in their work. In part, the tools were developed based on discussions with the SWCDs regarding their needs and suggestions for improvement in work processes. After the tools were designed, we provided summarized data, documentation, training, and technical guidance during watershed planning efforts. We conducted training on how to prioritize lakes and projects using a simple risk assessment approach that included: reviewing summarized lake data (e.g., W:L ratio, phosphorus sensitivity, water quality trends, LPSS), understanding phosphorus load reduction goals for each lake, incorporating socioeconomic factors (e.g., BCR, political will, collaborators), and consideration of various best management practices for each lake.

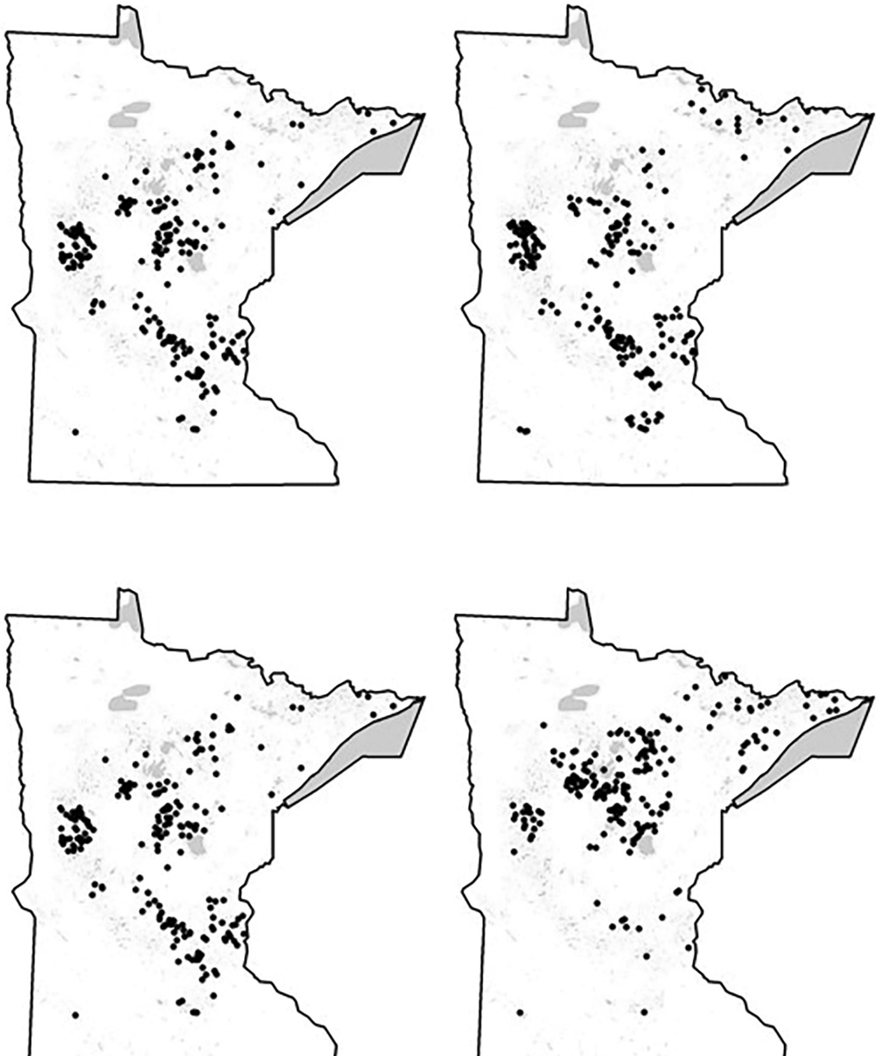

FIGURE 9.10
The top 200 ranked lakes by lake phosphorus sensitivity significance (LPSS) priority score, benefit:cost ratio (BCR), two-dimensional BCR-LPSS feature prioritization, and three-dimensional BCR-LPSS-Lakes of Biological Significance (LOBS) feature prioritization.

Using these tools, the Crow Wing SWCD prioritized Big Trout Lake, and developed a stormwater treatment project to address the lake's needs. Big Trout Lake has a moderate W:L ratio (6), a high sensitivity to phosphorus loading, a high BCR, and it is a Lake of Biological Significance. Crow Wing SWCD targeted runoff from a 49 ha (121 ac) catchment that drained to the lake via a network of ditches and culverts. The project was designed to reduce phosphorus loading to the lake by 18 kg/year (40 pounds/year), which was consistent with the calculated 5% load reduction goal for the lake. This Crow Wing SWCD project was a collaborative project with the lake association, the city of Manhattan Beach, and the county highway department. The success of this project led to similar lake protection efforts for a nearby lake and community.

Discussion

Lakes with a low W:L ratio may provide more straightforward opportunities to successfully address land use activities to decrease erosion and rainwater runoff. Lakes with a high W:L ratio often demand greater effort to address those land use practices with negative consequences. The W:L ratio can also be used to plan for potential climate change influences on lake hydrology. Landlocked lakes that are dominated by groundwater fluctuate with groundwater levels. As climate change produces wetter conditions, these lakes could rise more than lakes that are dominated by surface-water flow. Identifying these lakes and planning for potential water level changes will be important. For lakes with a high watershed transport capacity classification, land use across the entire watershed is likely important in explaining lake hydrology and lake water quality. Lakes with a low classification are more likely to be driven by shoreland use, in-lake processes, and groundwater. The lake watershed health index provides a generalized risk assessment of a lake's watershed. Watersheds with a high health index may benefit from protection-based efforts, while those watersheds with a low health index may be at risk or suffering degradation.

There are both benefits and shortcomings associated with the standard deliverables. These simple metrics were easy to craft, and they are useful when prioritizing a large number of lakes based on the relative scale of effort necessary for meaningful lake conservation. However, there are many factors that influence the contribution of groundwater to lakes, and variability in lake water levels and water quality are influenced by a range of factors that are not easily assessed with readily available GIS information, including stream inflows, presence of outlets, outlet height, outlet alteration, water level management, and lake bathymetry. The lake watershed transport capacity classification did not include stream volumes or the importance of the relative position of the lake in the watershed; these variables are likely important in accurately assessing surface-water movement within a lake's watershed. The lake watershed health index does not factor in pollutant volumes, or the risk of the pollutant loads on a lake. In many cases those risks are not yet determined, so would need to be assessed and added by the local governments when and if available.

The multi-criteria values-based model identified a list of priority lakes based on the objective of identifying high-quality lakes at greatest risk of becoming degraded or further degraded. The results of this approach were moderately correlated with the results of the BCR approach. Our analyses indicate that the exclusion of benefits and cost in prioritization would likely result in a modest amount of potential benefit forgone (~20%). In other comparisons, values-based models had higher forgone benefits and higher costs than those based on benefit:cost analyses (Joseph et al. 2009, Pannell and Gibson 2014).

Protection of lakes with mostly undisturbed forested watersheds was estimated to be a cost-effective use of resources. Of the lakes studied, many were sensitive to TP loading and the cost of protection via conservation easement was often lower than the cost of restoration; on average the conservation easement cost was a third of that for restoration activities related to stormwater management. While we used conservation easements only for minimally impacted lakes, a protection approach applied more broadly, where feasible, would likely have high BCRs. For example, medium to large parcels on vulnerable lakes would be good opportunities for investment with willing landowners. This approach may also produce other environmental benefits (e.g., fish and wildlife habitat, aesthetic).

A focus just on impaired waters would likely result in considerable forgone benefit (our results suggest a potential benefit forgone at ~80%). There are benefits of restoring degraded lakes, but there are also shortcomings associated with a dominant focus on this subset of lakes. First, in many impaired waters, TP loading is from non-point sources that have non-regulatory and more challenging source reduction strategies than control of discharges from end-of-pipe (Carpenter et al. 1998). Second, these lakes are often difficult to restore (Carpenter 2005, Cook et al. 2005), and they often require TP load reductions greater than the 5% reduction used in this analysis. Restorations may be hindered by internal TP cycling or the ability to scale non-point pollution controls (Huser et al. 2016). Higher returns of conservation investments may be achieved with a greater share of resources dedicated to protecting and restoring lakes with high resiliency and high benefit:cost ratios.

There are shortcomings to using benefit:cost ratios only. Ackerman and Heinzerling (2002) and Ackerman (2008) made a compelling case that benefit:cost analysis should not be the central method for decision-making; that is, taking action on environmental protection should not be dependent on such analysis. While our results show that the use of BCR would improve return on investment, it is still useful to consider those concerns that are relevant to prioritizing lakes for protection and restoration. First, whereas some costs are often well-defined, benefits are hard to define well. For example, we did not include many benefits, such as the presence of unique lake characteristics (cultural, biological, etc.) or the value of recreational activities; these benefits have clear value but were not monetized. Riparian land value was an important factor in the calculation of BCR, and when prioritizing by location, lakes in high-value real estate markets were prioritized over other high-quality lakes in less valuable markets or those more distant from population centers (Aitkin, Grand Rapids, Northeast, and Walker – Hackensack). The list of priority lakes changed substantially when we included high-quality biological lakes within the prioritization (i.e., more high-quality lakes in less valuable or more distant real estate markets were included). Second, benefit:cost analyses can lead to troubling tradeoffs that are not addressed. The use of the estimated costs for agricultural and stormwater BMPs assumed no explicit impact to society and the benefits accrued only to those with shoreland property. Third, the technicalities of benefit:cost analysis may lead to biases in the promotion of policies or in the interpretation of the results. For example, many minimally disturbed lakes in forested watersheds had high BCR priorities; however, while some of these lakes would likely benefit with proactive protection via conservation easement, others would not (e.g., they are located in watersheds predominantly in public ownership). In addition, only a few management options were used and their costs had considerable uncertainties and variabilities. Thus, even in efforts to prioritize lakes for protection and restoration, it is important that BCR analysis is not the main method for deciding on which lakes to invest greater resources. Incorporation of additional information through a peer review process may help mitigate some of the shortcomings associated with BCR analysis and lead to a better priority list (Armsworth et al. 2017).

Peer review is an important process to include in any prioritization. BCR analysis is constrained, and it is necessary to include information not expressed in monetary terms. A deliberative process is necessary when adding expert judgments (Martin et al., 2012). Adding expert judgment is important not because funding decisions are inherently subjective, but because there is good information that is not incorporated into even the most thorough and complex values-based model or BCR analysis. Silver (2012) noted that predictions are often improved when the models were supplemented with human judgments that incorporate information not used. In prioritization of lakes, the same can be said – the incorporation of human judgments to alter the prioritization based on information not used in a multi-criteria values-based model or a BCR analysis can produce a better priority list. However, the judgments expressed should be transparent and contestable (Game et al., 2013).

We pose the following recommendations based on our comparisons and understanding of the benefits and shortcomings of different prioritization approaches. First, define a clear objective. A reasonable one for Minnesota might be "focus on high-quality, high-value lakes that provide the greatest return on investment." The objective should include the diverse aspects of lakes. High-quality would refer primarily to water quality but include biological character, cultural importance, and other attributes that are measured or subjectively assessed. High-value would relate to economic factors including shoreland property value, recreational use values, and other attributes that might be priceless. Greatest return on investment would relate to phosphorus-loading resilience and economic considerations. For example, Keeler et al. (2015) found that lake recreational value, as assessed by lake visitation, was a function of lake size, water clarity, boat access, and near-lake human population size. They also determined that people traveled further to recreate clearer lakes.

Second, develop one or more multi-criteria values-based models that reasonably capture all or a portion of the defined objective function. An alternative values-based model, with an objective function of "focusing on high-quality, high-value lakes that likely provide the greatest return on investment," is:

$$\text{Priority score}_2 = S^a \times A^b \times (D + 0.01)^c$$

where the multipliers are the TP loading sensitivity index (*S*), lake surface area (*A*, acres), and the proportion of the lake's immediate catchment in disturbed land cover (*D*), and *a*, *b*, and *c* are multiplier weights. This values-based model is better correlated with the BCR ($r^2 = 0.71$, power-law function, versus $r^2 = 0.45$) than the original values-based model which had the objective of "focusing on high quality lakes at greatest risk of becoming impaired or further degraded."

Third, use benefit:cost analysis to re-sort an initial priority list of lakes from the values-based models. For example, take the top lakes from the values-based model and re-prioritize based on benefit:cost analysis. What lakes are likely to provide higher benefit per cost of investment? Essentially, benefit:cost analysis becomes more of a cost-effectiveness analysis. In the absence of a formal cost-effectiveness analysis the following guidelines could be used: (1) give higher priority to large lakes; (2) give higher priority to lakes that are sensitive to changes in TP loading; (3) give higher priority to lakes that can be protected with cost-effective strategies, e.g., forested watershed lakes that can be protected with proactive shoreland conservation easements; and (4) give higher priority to developed lakes in or close to large cities as they have high social values. Finally, make the draft priority list available for peer review. Reviewers will bring information and insights not included in the analyses and help make the priority list even more defensible and practical.

Acknowledgments

We thank Pam Anderson, Jeff Hrubes, Steve Heiskary, and Michael Duval for advice and guidance in the development of the values-based model. We thank Todd Petersen and Glen Champion for guidance on lake hydrology.

We thank Dave Wright, Jeff Hrubes, and Dan Steward for reviewing earlier drafts of the manuscript. Finally, we thank the Crow Wing SWCD, Melissa Barrick, her staff, and the board members for their insight into information that would be helpful in their valuable work.

Parts of this contribution made use of materials from a previous publication "Radomski, P., and Carlson, K. 2018. Prioritization approaches along with a peer review process to produce lake priority lists, Lake and Reservoir Management, 34(4), 401–416" with permission of the North American Lake Management Society and by publisher Taylor & Francis Group.

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10

Water Yield Ecosystem Services Assessment in Periyar Tiger Reserve

Shiju Chacko, Jikku Kurian, C. Ravichandran, S. M. Vairavel, and Krishan Kumar

CONTENTS

Introduction

Ecosystem services are dominantly used to assess the potential impacts of global change in societal and economic terms and to provide a rationale for environmental management (Tallis, Kareiva, Marvier, and Chang, 2008). To conserve or manage the environment, there is a requirement for accurate mapping and measurement of ecosystem services (Malinga, Gordon, Jewitt, and Lindborg, 2015). Mapping approaches are therefore useful to recognize and implement ecosystem

DOI: 10.1201/9781003175094-10

services in decision-making and land management strategies (Maes, Crossman, and Burkhard, 2016). Compared with complex models, mapping approaches that use to link vegetation classification and landscape properties with ecosystem services are relatively efficient and easier to ensure timely update of management strategies and policies (Malherbe, Pauleit, and Lorz, 2019).

Conventional management practices of Protected Areas (PAs) across the globe are primarily based on the concept of habitat preservation with the least external interference for nature to sustain itself (Terborgh, 2000; Karanth, 2007). The establishment of tiger reserves in India is meant to protect landscape-level features and all life forms for biodiversity conservation with genetic, species, and ecosystem diversity (Jhala, Qureshi, and Gopal, 2010). These PAs conserve a wide range of ecosystem services, and they provide social, economic, and cultural benefits. Ecosystem service estimation from a tiger reserve can help in making conservation more significant, and thereby it can be evident to the decision-makers and national policy builders (Verma et al., 2017).

Water services are one of the prime ecosystem services (ES) benefits of forests, and that could be seen as more important for their impacts on local communities, as vital resources for command area users, and even as large-scale regulators of regional climates through transpiration and cloud formation (Ellison et al., 2017; Netzer et al., 2019). Water yield ecosystem services have a key role in agriculture, aquaculture, industry, and energy generation, healthy life for humans and ecological balance (Burkhard et al., 2012). Quantity and quality of water access is a basis for the sustainable development and is critical for socioeconomic development, healthy ecosystems, and human survival itself (Yu et al., 2015). Conserving natural forests and developing capacity to measure and monitor biodiversity and ecosystems for their provisioning services is thus an essential step toward better management of our natural capital (Daily, 1997). Water yield valuation is increasingly being used as a tool to communicate the values emanating from natural ecosystems to the policymakers and thus help in prioritizing conservation and proclaim the free service of the nature to the society (TEEB, 2010b).

The Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model has been developed to enable decision-makers to assess tradeoffs among ecosystem services and to estimate changes in biodiversity under different demographic, land use, and climate scenarios (Tallis et al., 2010; Sharp et al., 2014). InVEST was developed as part of the Natural Capital Project (www.naturalcapitalproject.org, last checked December 15, 2021), the aim being an alignment of economic forces with conservation objectives and mainstreaming the approaches. Evaluation through InVEST water yield model estimates the relative contributions of water produce from different parts of the PA, offering insight into how different vegetations affect annual surface water yield and its spatial distribution (Sharp et al., 2014).

In the present study the model has been used to assess the water provisioning service being rendered by the watersheds of the Periyar Tiger Reserve (PTR). The focus on water services implies a focus on decisions related to PA management, thus requiring spatially explicit descriptions of the landscape and associated hydrologic parameters (Guswa et al., 2014; Meisch, Schirpke, Huber, Rüdisser, and Tappeiner, 2019).

The national and international scenario requires the quantification of ecosystem services, so they can be included in policy- and decision-making regarding forest conservation and management (Olander et al., 2018). However, due to the lack of environmental information available in the Southern Western Ghats area where there have been limited or no studies conducted, greater precision is still needed to understand the water benefits generated by the tiger reserve ecosystem. It is, therefore, necessary to obtain a more robust spatial distribution and its contribution by different vegetation communities for the water in the PTR watershed using an advanced modeling study for understanding ecosystem services.

The aim was to assess annual water yield ecosystem services for the different land-use patterns in the PTR. The analysis was based on forest classification, soil properties, and meteorological and biophysical data from the InVEST water yield model. The specific objectives were to: (1) quantify surface annual water yield provision of different vegetations in PTR and (2) develop a classification framework among these land-use patterns and their spatial distribution.

Study Area

Western Ghats (WG) form a mountain range that extends along the coast of the Arabian Sea from south of the Tapti River and ending at Kanyakumari at the southern tip of India. This area is categorized and considered as one of the world's richest biodiversity hotspots. The area selected for the study was the PTR in the Periyar-Agasthyamalai landscape. The catchment of the Periyar and Pamba rivers provides livelihoods for many rural and urban communities. The protected area is listed in the UNESCO World Natural Heritage Sites in India (Myers et al., 2000).

PTR is situated in the Western Ghat Hills and specifically Cardamom and Pandalam Hills of the Southern Western Ghats between latitudes 9°17′56.04″ and 9°37′10.2″ N and longitudes 76°56′12.12″ and 77°25′5.52″ E (Figure 10.1). PTR covers 925 km^2 of highly undulating terrain, and its elevation ranges from 81 m above MSL (lowest) to 2016 m (highest) with an average elevation of 1,200 m. The climate is cool and humid with an average annual rainfall of 2,500 mm. The temperature ranges from 15°C to 31°C, the hottest months are April and May, and the coolest months are December and January. The forest vegetation ranges from montane evergreen forests at higher elevations to all other major tropical vegetation types along the lower reaches (Satis, 1991).

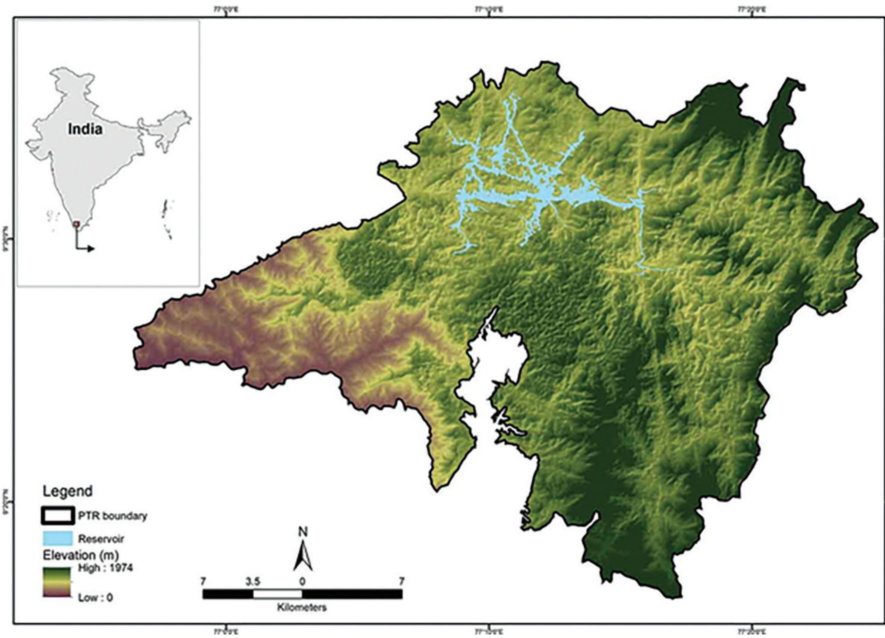

FIGURE 10.1
Location map of PTR. (a) Drawing of India with site noted. Figure. 10.1b Watershed with terrain highlighted.

Materials and Methodology

In this study, InVEST (version 3.3.3) was used with input data which included gridded maps of vegetation, soil properties, climate, and some biophysical coefficients. The following sections provide the description of the model and the input data preparation.

The InVEST water yield model is defined as the amount of water that runs off the catchment, and it calculates the sum and averages of the water yield based on the principle of water balance at the sub-watershed level. The water yield model is based on the Budyko curve (Budyko, 1974) and the annual average precipitation. The model estimates the total annual water yield (Y) for each grid square (x) of the study catchment as follows:

$$Y(x) = \left(1 - \mathrm{AET}(x) / P(x)\right) \cdot P(x) \tag{10.1}$$

where AET(x) is the annual actual evapotranspiration for pixel x and $P(x)$ is the annual precipitation on pixel x.

The model assumes that, on an annual time step, all water falling as rainfall over a catchment area, minus that which is evapotranspired, leaves the catchment. No distinction is made between the surface and sub-surface water flows.

Practically, the estimation of the catchment and scale measurement of annual actual evapotranspiration is extremely difficult. Even plot scale evaluation requires highly sophisticated equipment, and plot and field scale methods to determine actual evapotranspiration are challenging to apply at the landscape scale (Evans et al., 2012).

The evapotranspiration portion of the water balance, AET(x)/P(x) for vegetated land use is calculated (Zhang et al., 2004) in a spatially explicit way on pixel x by

$$\mathrm{AET}(x)/P(x) = 1 + \mathrm{PET}(x)/P(x) - \left[1 + \left(\mathrm{PET}(x)/P(x)\right)^{\omega}\right]^{1/\omega} \tag{10.2}$$

Potential evapotranspiration PET(x) is defined as

$$\mathrm{PET}(x) = K_c(l_X) \cdot \mathrm{ET}_0(x) \tag{10.3}$$

where $ET_0(x)$ is the reference evapotranspiration from pixel x and $K_c(l_x)$ is the plant (vegetation) evapotranspiration coefficient associated with the land use l_x on pixel x. $\mathrm{ET}_0(x)$ reflects local climatic conditions, based on the evapotranspiration of a reference vegetation at that location. $\omega(x)$ is related to the plant available water content (AWC), precipitation, and the constant Z which captures the local precipitation pattern and additional hydrogeological characteristics (equation 10.4) (Donohue et al., 2012; Sharp et al., 2014).

$$\omega(x) = Z\frac{\mathrm{AWC}(x)}{P(x)} + 1.25 \tag{10.4}$$

AWC(x) defines the soil texture and effective rooting depth, which establishes the amount of water that can be held and released in the soil for use by a plant. Z is an empirical constant, which captures the local precipitation pattern and additional hydrogeological characteristics. It is positively correlated with N, the number of rain events per year. The constant, 1.25, in equation 10.4 is the minimum value of $\omega(x)$ corresponding to bare soil, following Donohue et al. (2012). The water yield model generates the outputs as the total and average water yield at a sub-watershed level.

Data Preparation

InVEST Water yield model requires average annual reference evapotranspiration, plant available water content, soil depth, land use/land cover, precipitation, watersheds, and sub-watersheds as well as a biophysical table containing data on biophysical coefficients. All the input data were resampled at a spatial resolution of 30 m and projected using the World Geodetic System 84 (WGS84).

Average Annual Reference Evapotranspiration

PTR has ten weather stations with daily mean, maximum, and minimum temperature data from the 2007 to 2010 period that was collected and used for the analysis. Annual reference evapotranspiration was obtained using the Hargreaves equation (Hargreaves and Samani, 1985). The raster format of average annual reference evapotranspiration was generated through Kriging interpolation (Figure 10.2a). Northern regions experienced the highest average annual reference evapotranspiration.

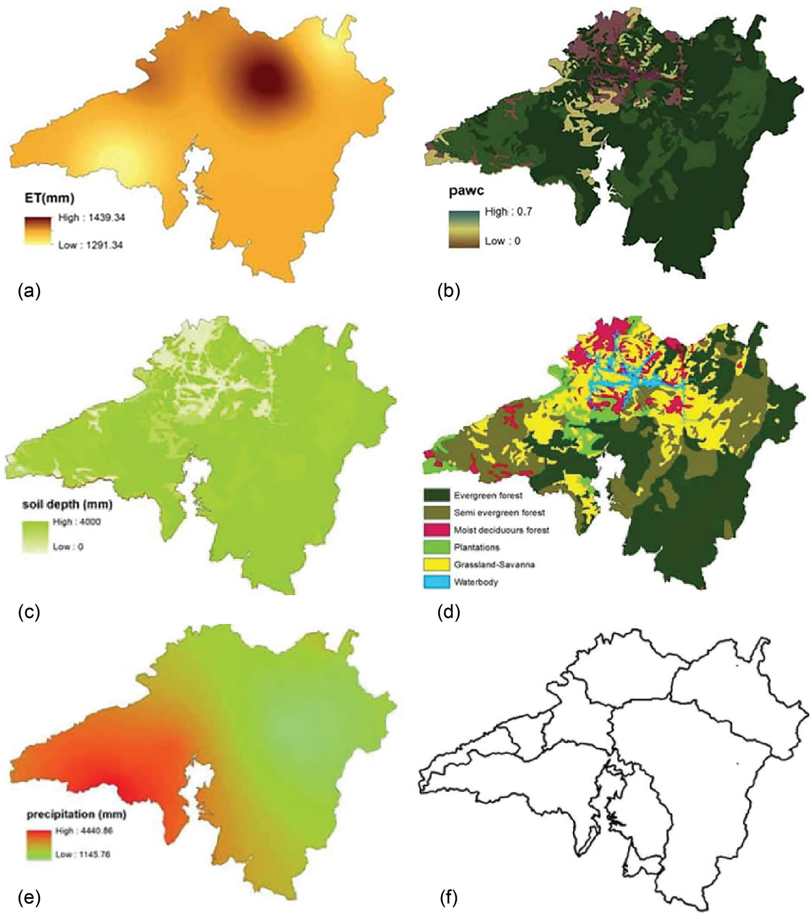

FIGURE 10.2
(a) Average annual potential evapotranspiration; (b) plant available water content; (c) soil depth; (d) land use and land cover; (e) average annual precipitation; and (f) watershed and sub-watersheds.

Plant Available Water Content

Plant available water content (PAWC) is defined as the fraction of water that can be stored in the soil profile that is available for plants' use (Zhou, Liu, Pan, and Feng, 2005). PAWC was calculated using data from the Soil Survey of India and SPAW Hydrology and Water Budgeting software downloaded from the United States Department of Agriculture (USDA) and shown in Figure 10.2b.

Soil Depth

A GIS raster dataset with an average soil depth value for each cell was generated based on the Soil Survey of India datasets. The main soil types are laterite and alluvial soil and depth values varying from 1 to 5 m (Sekhar et al., 2009) and given in Figure 10.2c.

Land Use Land Cover

Land use land cover statistics were derived from the forest maps of South India on the scale of 1:250,000 with vegetation classifications published by the French Institute of Pondicherry (FIP, 1992). The whole park had been under a strict conservation regime during the map generation and the duration of 20–25 years can be considered a short period of time for the natural systems to change. Based on the knowledge of the land cover and ground truth information, six different land cover classes were identified (Figure 10.2d) and described in Table 10.1.

Precipitation

The annual precipitation data from 2007 to 2010 of ten weather stations located in PTR were used for the analysis. The annual mean precipitation raster value in millimeters was generated using the Kriging interpolation method (Figure 10.2e). The average annual precipitation was 2,777 mm for the period of 2007–2010. The higher rainfall was experienced in the northeast part of the study area and lower rainfall in the southwest part.

TABLE 10.1

Land Cover Statistics and Their Classification in PTR

Sl. No.	Vegetation Class/Land Cover	Area km^2	Area %
1	Evergreen Forest	342.12	36.99
2	Semi-evergreen Forest	223.2	24.13
3	Moist Deciduous Forest	79.48	8.59
4	Plantations	55	5.95
5	Grassland-Savanna	199.2	21.53
6	Waterbody	26	2.81
	Total	925	100

Watersheds and Sub-Watersheds

Based on the digital elevation model (DEM), the watersheds and sub-watersheds were generated using ArcGIS. Each watershed and sub-watershed was given a unique identification number which is named ws_id and subws_id. There are nine watersheds and 20 sub-watersheds in PTR. The watersheds layer is presented in Figure 10.2f.

Biophysical Table

Biophysical table labels of land use/land cover classes contained data on biophysical coefficients. These data were attributes of each land use/land cover classes, and these parameters were determined with reference to FAO (Allen et al., 1998) as shown in Table 10.2.

Seasonality Factor (*Z*)

Z is an empirical constant that captures the local precipitation pattern and hydrogeological characteristics (Donohue et al., 2012). It was estimated as 0.2*N, where N is the average number of rain days (>1 mm) per year over the study period (Hamel and Guswa, 2015). Thus, the *Z* value of PTR was calculated as 30.

TABLE 10.2
Biophysical Details for PTR

LULC Description	LULC Code	LULC Vegetation	Root Depth (mm)	Kc
Evergreen Forest	1	1	2000	0.95
Semi-evergreen Forest	2	1	2000	0.92
Moist Deciduous Forest	3	1	2500	0.9
Plantations	4	1	1500	0.9
Grassland-Savanna	5	1	500	0.7
Water spread area	6	0	200	0.5

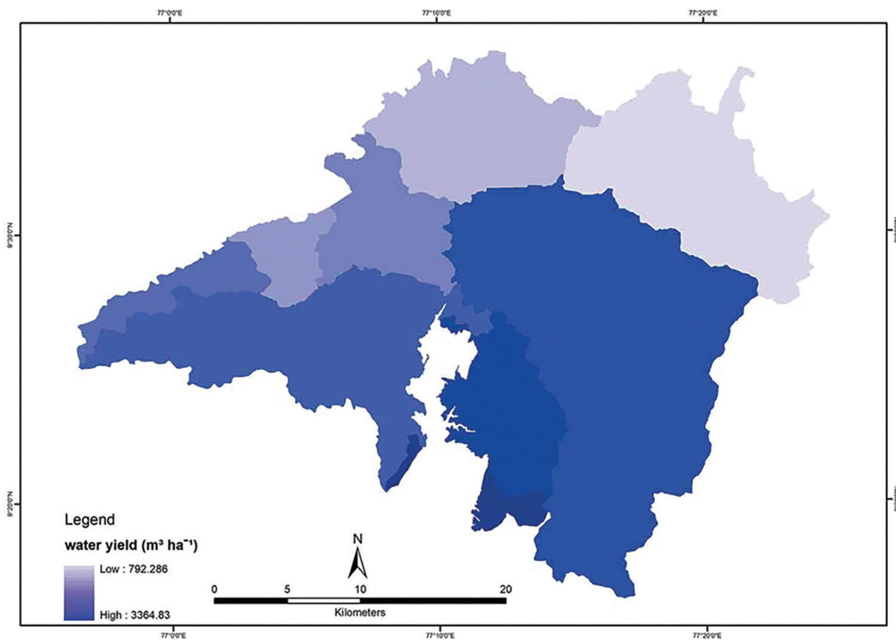

FIGURE 10.3
Water yield map of PTR.

Result

The estimated water yield of PTR was ~2.33E+09 m^3/year, and its spatial distribution map is given in Figure 10.3. Two major rivers in Kerala, namely Periyar and Pamba, originate from the PTR landscape and their watersheds share 67% and 33% of average annual water yield, respectively. Water yield ratio of different forest ecosystems in PTR is depicted in Figure 10.4.

The evergreen and semi-evergreen forest ecosystems contribute 9.29E+08 m^3/year (40%) and 5.46E+08 m^3/year (23.5%), respectively. Thirty-seven percent

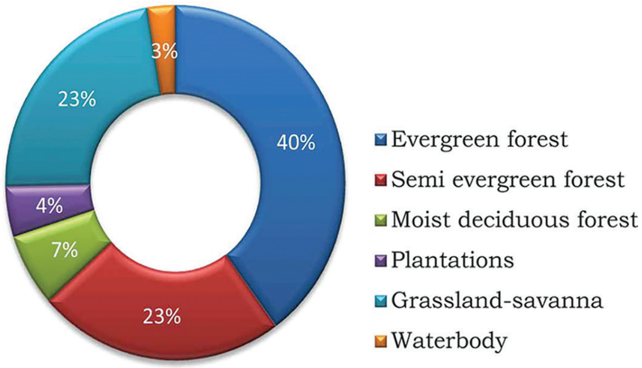

FIGURE 10.4
Water yield ratio of different forest ecosystems in PTR.

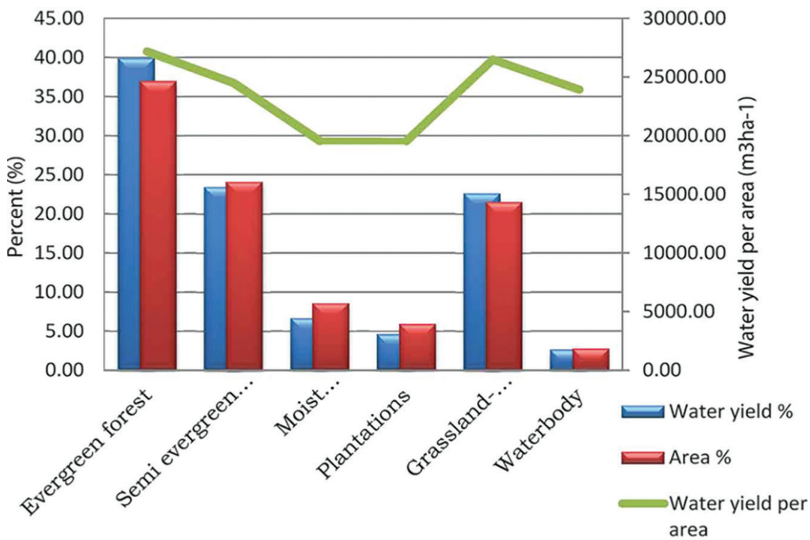

FIGURE 10.5
Water yield of different forest ecosystems in PTR.

of evergreen and 24% of semi-evergreen share the major portion of the total water yield, 63.5% of the PA. Other ecosystems of the moist deciduous forest have 1.55E+08 m^3/year (6.5%), grassland-Savanna 5.27E+08 m^3/year (22.7%), plantations 1.07E+08 m^3/year (4.6%), and waterbody 6.2E+07 m^3/year (2.7%) for the total water yield of the study area. The water yield contribution of these different forest ecosystems is shown in Figure 10.5. Area and water yield percentage of different ecosystems in PTR are depicted in Table 10.3.

Periyar river watershed contributes 1.56E+09 m^3/year and that of Pamba 7.67E+08m^3/year for the average annual water yield of PTR. Out of the nine

TABLE 10.3

Area and Water Yield Percentage of Different Ecosystems in PTR

PTR		
Land Cover/Vegetation Class	**Area (sq.km)**	**Water Yield %**
Evergreen Forest	342.12	40
Semi-evergreen Forest	223.2	23.5
Moist Deciduous Forest	79.48	6.5
Plantations	55	4.6
Grassland-Savanna	199.2	22.7
Waterbody	26	2.7
Total	925	100

TABLE 10.4

Periyar and Pamba Rivers' Ratio of Different Ecosystems

Land Cover/Vegetation Class	Area (km^2)	Water Yield (%)
Evergreen Forest	342.12	40
Semi-evergreen forest	223.2	23.5
Moist Deciduous Forest	79.48	6.5
Plantations	55	4.6
Grassland-Savanna	199.2	22.7
Waterbody	26	2.7
Total	925	100

major watersheds of the study area, Periyar and Pamba rivers consist of 4 and 5 numbers, respectively. Periyar and Pamba rivers' ratios of different ecosystems are listed in Table 10.4. Periyar has a reservoir that consists of 26 km^2 area and stores the water yield of the Periyar river watershed area.

Discussion

This study revealed that the water yield ecosystem services of PTR, a natural forest, using the InVEST model and mapped its spatial distribution patterns. The accurate assessments of water yield are important for examining the current distribution and impact of climate change on forest and associated ecosystem services including, water quality in terms of nutrients and sediment, drinking water, as well as crop production and hydropower generation (Redhead et al., 2016). With this model we are identifying areas of high water yield in PTR, and concentrated efforts can be made by the park management in sustaining the areas with high water yield and also improve upon areas facing degradation.

These studies are useful in terms of their ability to respond to global patterns of precipitation and provide the information necessary for users to assess the water yield at a national or regional scale (Martínez-Harms and Balvanera, 2012). This modeling gives the appropriate scale at which the majority of ecosystem service mapping exercises are performed, and the most strategic water resource planning takes place (Watts et al., 2015). InVEST model-based studies have had the primary aim of quantifying spatial variation and predicted changes in water yield and found to be a good predictor of measured water yield (Bai et al., 2013; Boithias et al., 2014; Terrado et al., 2014; Xiao et al., 2015).

Conclusions

Water yield calculation and mapping are of great importance to PA management and conservation planning. The provision of freshwater ecosystem services that contribute to the welfare of society is necessary for the survival of both flora and fauna. Ecosystem service models such as InVEST are the potential tool to provide a crucial underpinning to decision and policy making for national parks. This chapter assessed annual water yield ecosystem services for the different land-use patterns in the PTR and quantified the surface annual water yield also their spatial distribution.

The study provides a range of possibilities to inform the general public and the policymakers, a major piece of evidence for the positive impacts of PAs on human well-being. The result shows that the evergreen forest with 40% water yield has a major contribution over PTR annual water yield and then semi-evergreen with 23.5%. It shows the significance of conserving the forest to retain its evergreen nature by its very specialized plant community structure in the Southern Western Ghats region for the water yield ecosystem services. Water yield spatial distribution map of PTR provides an effective tool for conservation prioritization of land use and also a framework for the sub-watershed-level appropriate conservation measures. Biodiversity and ecosystem services in the tiger reserves are natural assets with a key role to play in future sustainable development strategies seeking to combat climate change and find prosperity in India. This study provides a useful initial exploration into the research of ecosystem services at regional scales in a mountainous landscape.

Acknowledgments

The authors sincerely thank officers and staff of Periyar Tiger Reserve, Periyar Tiger Conservation Foundation, and Kerala Forests and Wildlife Department for their encouragement and support.

Parts of this contribution made use of materials from a previous publication "Chacko, S., Kurian, J., Ravichandran, C., Vairavel, S.M., and Kumar, K. 2021. An assessment of water yield ecosystem services in Periyar Tiger Reserve, Southern Western Ghats of India, Geology, Ecology, and Landscapes, 5, 32–39" with permission by publisher Taylor & Francis Group.

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11

Water Quality Parameters as Related to Small Watershed Land Cover

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CONTENTS

DOI: 10.1201/9781003175094-11

Introduction

Natural forests and well-managed plantations usually have lower input of nutrients, pesticides, and other chemicals than agriculture and other intensive land use (IUFRO, 2007). Eventually, this can lead to lower nutrient transport allowing for the riparian forest to reduce nitrates, phosphorus, and sediment loading to the stream and lake (Lindell, Åström, and Öberg, 2010; Kändler et al., 2017). To predict the response of watershed ecosystem services to land use and land cover change and climate variability, as well as develop indicators of the effects of human activities on stream conditions and effective management plans (Hunsaker and Levine, 1995), study at a suitable scale is crucial. Different environmental and human-induced alterations are likely set into the different spatial and temporal scale to wield their influence on water quality in a catchment. Recognizing scale of domination for existing natural variation in ecological patterns and processes, it is often difficult to determine the best scale of the study (Levin et al., 1992) to evaluate the impact of land cover or land use change on the stream or receiving waters. Hence, expanding research efforts to establish the effects of land cover change at multiple spatial scales on neighboring waters is critical (Bruijnzeel, 2004). Moreover, riparian vegetation near the stream or creek is always regarded as the most significant component in sediment load reduction and water quality treatment in a catchment. However, Vigiak et al. (2007) reported that in reducing sediment pollution to streams, bamboo as a riparian strip is not regarded as effective whether it was naturally occurring or planted. Hence, ambiguity over the form and extent of vegetation influences stream water quality in a mountainous watershed cannot always be understood from a case study of a region.

Shifting cultivation and secondary forest are dominant land use types in the mountainous watershed of Chittagong Hill Tracts (CHTs), Bangladesh. This watershed comprises 13,180 km^2 to the southeastern region, with a semi-consolidated to consolidated rocks formation possessing a steep slope in most part (>75%) of the area. This watershed is the home of at least 12 ethnic communities and has been regarded as in a degradation state following rapid population increase since the settlement of people from overpopulated plains land as well as from high growth rate of both ethnic and migrant communities. Moreover, other major interventions such as the impoundment of a hydroelectric dam in 1965, commercial forestation, and agroforestry program are just a few to be mentioned that triggered the rapid change in land use and land cover of this region. The population ratio of indigenous communities to the plain land migrant community has significantly changed since 1965, with an increase in total population as well. Consequently, the loss of arable land forced people to move to further uphill areas. The exodus and increased population were seen to be fueling the expansion of shifting cultivation land and other agricultural practices (Tripura and Harun, 2003) onto steep and moderate slopes. The increasing trend of destruction of forest cover with a radically reduced farming-fallow cycle (Gafur et al., 2003) is evident everywhere to meet the increasing need for food and forest products for timber, fuelwood, and fodder. Shifting cultivation has become unsustainable because of reducing its cultivation cycle to 2–3 years, which can hardly sustain productivity to jhum crops for about 3–4 months due to loss of topsoil and development of aluminum, manganese, and iron toxicities in soil (Aray, 1999).

A land cover survey a priori to the water sampling revealed that natural regrowth of herbs, shrubs, and trees recolonized lands, which was previously cultivated in the following rainy season. It was also observed that on many hills, people raised fruit trees. Moreover, vegetation types were increasingly subject to fragmentation by rotational shifting cultivation as well as regrowth of natural vegetation following monsoon rain. The hill tracts are thus a mosaic of vegetation cover interspersed with deforested areas (Haque, Karmakar, and Hossain, 2010; Karmakar et al., 2012). However, shifting cultivation, with a low nutrient practice and higher physicochemical impact on topsoil (Haque, Gupta, and Miah, 2014), water quality may not be dependent only on temporal variation. A micro-catchment is defined by a single and first-order creek only contributing to the discharge point a monotonic watershed divide exists between neighboring catchments. Hence, a micro-catchment scale study in a paired watershed will be composed of single or two land use/land cover (LULC) in its sides of the catchment, composing full or half of the area.

Hill people of Bangladesh still now use small dug wells by the side of the hill for drinking water and creek water for bathing and other household purposes. In some places, they also store water for daily uses by digging small ponds or keeping the surroundings of a natural reservoir undisturbed from grazing and frequent walking. Even then, they get short of daily water during the dry season and are forced to travel to distant sources from the residences to collect water. Haque, Karmakar, and Hossain (2010) and Karmakar et al. (2012) have studied land use alteration effect on soil, water, and microbial diversity in this hilly region. With the existence of scattered natural forests, presently shifting cultivation is being practiced all over. Land use or land cover (LULC) alteration could be detected without a clear understanding of the effect of individual practice on the repository water bodies in this catchment following the very dynamic nature of the vegetation in this region. This nature of replenishment of land cover is widespread over the topics. Instantly, micro-catchment-based water quality parameter has never been considered to predict the land cover change in the catchment.

Many studies have reported the strongest association of agriculture and urban lands with water quality variables (e.g. Datta et al. 2022) and sediments in rainy seasons . Though a handful of studies have been conducted over decades, further study on the complex association should be considered as much as possible, particularly on how the association of land use and water quality varies with region and with different topographical characteristics of land use. Dominant factors of the temporal scale, such as precipitation, temperature, and agricultural activities, vary among seasons and given their role on flow convergence processes and contaminant inputs into water bodies. Researchers have suggested that it is imperative to consider seasonal variation when studying the impact of land use on river water quality (Ahearn et al., 2005). Ye et al. (2014) indicated that while non-point source pollution was predominant in rainy seasons, agricultural and forested land showed a stronger association with water chemistry. Under the circumstances, realizing the importance of water in the life of hill people, presently under study was conducted to investigate the impact of existing land covers on the water quality of creeks and seepage. Moreover, the land use and land cover change studies reported earlier in many catchments until now based on the relative percentage of land cover (e.g., Uriarte et al., 2011; Shi et al., 2017) do not explain a definite land cover contribution on the surrounding water quality. Zorzal-Almeida et al. (2018) and Vrebos, Beauchard, and Meire (2017) have emphasized and expected a spatial influence on the water quality of river or repository water bodies to explain the variation along with the land use effect. A paired and micro-catchment-based study (less than ten hectares) instead of a catchment or sub-catchment scale of a river or lake to discern the effect of mixing from the different land covers on water drainage is relatively new compared with results of similar studies except for monsoon runoff sediment estimation. Since individual land use or land cover contribution is not enumerated in any studies due to a complex association of different land use, the effect of the riparian zone, trapping and mixing of the nutrient element through its paths, this study was targeted to lessen the ambiguity of the cause-effect relationship (Figure 11.1).

Materials and Methods

Water Sample Collection and Land Use Survey

Water sampling was conducted from flowing creeks of shifting cultivated, plantation, and comparable adjacent natural vegetated watersheds as well as from seepages. Water samples were collected from creeks and seepages of three areas, viz. Kutukchari, Kichingmontala, and Ghilachari in a sampling campaign during September (a month after the rainy season) in 19 different sites. Three fields replicated grab samples from each creek were collected and analyzed separately for each watershed. We had selected small creek for water sampling, which had a width ranging from 1.5 to 10 cm and a depth

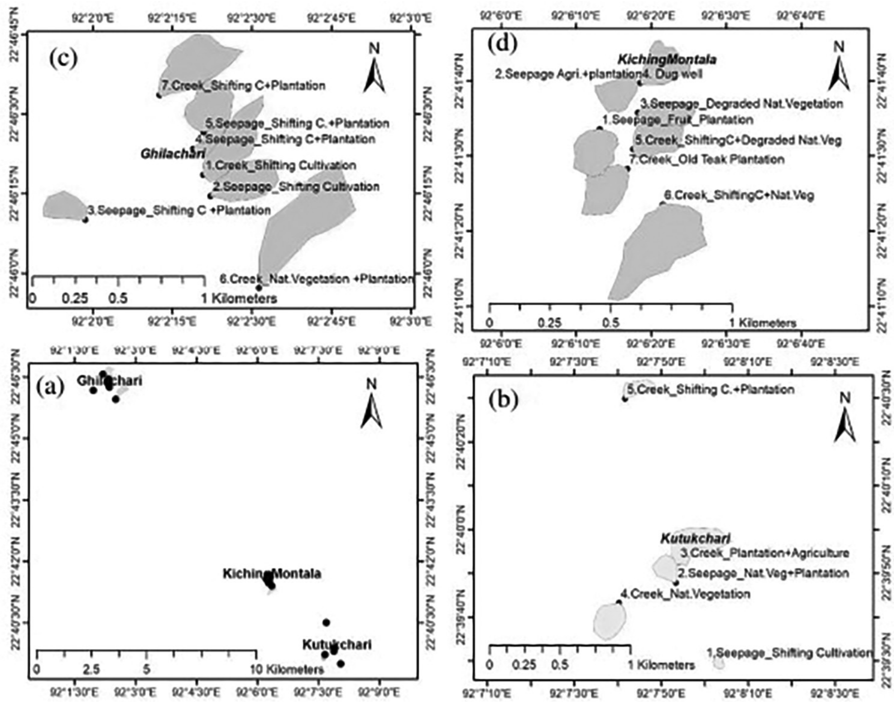

FIGURE 11.1
(a) Relative position of the water sampling site. Three clusters of water samples collected from Ghilachari, Kichingmontala, and Kutukchari areas (from top to bottom). Water sampling point and catchment area of seepage and creek water sample at (b) Kutukchari area; (c) at Kichingmontala area; and (d) at the Ghilachari area.

ranging from 1 to 2 cm. Seepage water was collected by digging a small hole on shale or clay layers at the base of a hill in triplicate, which were 100–130 cm apart from each other, but above the creek water level to avoid mixing of creek water. The dug holes were allowed to fill with water afresh twice before collecting water samples in a clean sample bottle. In selecting creeks or seepage dug-holes for water sampling, the structure and composition of vegetation in the catchment were documented, and the catchment was devoid of human settlement so that any discharge from human habitations could be avoided. Each of the creeks or seepage waters drained from small watershed ranged from 0.5 to 10 ha having parent materials mainly of shale and Orthi-Feric Alfisols as reported in the UNDP-FAO (1988) soil survey. Twelve representative parameters were measured, including pH, dissolved oxygen, electrical conductivity (EC), total dissolved solids (TDS), chloride (Cl^-), sulfate (SO_4^{2-}), nitrate nitrogen (NO_3-N), orthophosphate (PO_4^{3-}), Na^+, K^+, Ca^{2+}, and Mg^{2+}. The pH, EC, and TDS values were measured in situ using a multiparameter probe. Specific descriptions of the sites at each water sampling point are given below.

Kutukchari

At Kutukchari, three flowing creeks, two seepages, and one irrigation drain water sample were collected, and existing land cover was sampled. Seepage water from slashed and burned shifting cultivated (SC): Water sampling was done from a dug-hole near Kutukchari Range office under Unclassed State Forest Division, 10 km away from Manikchari toward Khagrachari. The hill above the dug-hole was slashed and cleared for shifting cultivation except few stands of shrish, and banana on southern aspect of 30% slope.

Seepage water from mature mixed plantation: Seepage water sampling was done at Kutukchari under Sukurchari Union, situated 15 km away from Manikchari toward Khagrachari. Hill above the dug-hole was covered with mature trees, mainly of teak (*Tectona grandis*), gamar (*Gmelina arborea*), shrish (*Albizia procera*), shimul (*Bombyx ceiba*), and bamboo sporadically with 90% undergrowth on the southern aspect of 35% slope. Tree canopy coverage was about 85% and undergrowth consisted of assamlata 80%, bhat 10%, and climbers and sun grass together 2%.

Drainage between mature mixed plantation and agricultural land: Water sample was collected from artificially created drain between hill slope covered with above-mentioned mixed plantation and fallow agricultural land after harvesting paddy situated on a valley.

Flowing creek from natural vegetation: Water samples were collected from a flowing creek by the side 55% hill slope on the western aspect. Both sides of this creek were covered with good natural vegetation, composed of bamboo, a few trees and shrubs, herbs, and climber on both hill slope and creek bank, with canopy coverage of about 85%.

Flowing creek from shifting cultivated (SC) area at Kutukchari: Water samples were collected from flowing creek by the side of 45% hill slope. On both sides of this creek were 1-year-old cultivated lands. Major crops were zinger and sesame along the slope with shrubs and climbers on creek bank only.

Kichingmontala

At this area, the water sampling campaign was carried out from three flowing creeks and four seepages, and existing land cover was sampled.

Seepage water from mature fruit trees and young plantation: Terrain above the dug-hole was covered with mature fruit trees and pole stage timber species along with dense undergrowth in patches on southern aspect of hill on 35% hill slope. Major fruit trees were *Syzigium cumini* (jam), *Litchi chinensis* (litchi), *Arthocarpus heterophylus* (jackfruit), and *Musa sapientum* (banana). Timber species were *Gmelina arborea* (gamar), *Tectona grandis* (teak), and *Albizia procera* (shrish). A dense undergrowth vegetation was observed in the watershed area that consisted of creepers mainly *Mikania cordata* (assamlata), climbers, and sun grass made up 80%, and 2% was *Clerodendrum inerme* (bhat).

Seepage water from shifting cultivation area (pineapple and plantation): The hill above the dug-hole was covered with current year shifting and young plantation on western aspect of 45% slope. A cultivated valley was present below this dug-hole. Major crops were turmeric (*Curcuma longa*), ginger (*Zingiber officinale*), and pineapple (*Ananas sativus*) on one part of the hill slope and the other part covered with young teak plantation. In this area, shifting cultivation was done many times, and pineapple grown across the slope with the mulching managed nicely.

Seepage water from dense natural vegetation and partly disturbed area: Part of the hill above the dug-hole was covered with dense natural vegetation, and other part was disturbed due to a road through which people used to walk frequently situated on the eastern aspect of the 55% hill slope. The hill above the dug-hole point was covered with dense climbers and undergrowth of 1.5–2.0 m in height.

Natural drinking water wells: Indigenous people use water from this kind of natural well for drinking, cooking, and other domestic purposes, commonly occurring in the valley community in this region. This type of natural well is the only water source for domestic uses for the indigenous people in the Chittagong Hill Tracts. Part of the hill slope above this well was covered with natural bushy vegetation and partly by disturbed roadbed through which people used to frequently walk and situated on the eastern aspect of 35% hill slope.

Flowing creek, with one side of shifting cultivation and the other side of natural vegetation: Water samples were collected from a flowing creek by the side of a 45% hill slope on an eastern aspect. One side of this creek was a current year shifting cultivated area of crops such as banana and bamboo (*Bambusa vulgaris* and *B. tulda*), and the other side of the creek had natural vegetation consisting of shrubs and herbs on both hill slopes and creek banks.

Flowing creek, one side of shifting cultivation and the other side of natural vegetation: Water samples were collected from flowing creeks by the side of a 40% hill slope on an eastern aspect. One side of this creek was a current year shifting cultivation area in patches of crops such as banana and bamboo, and the other side of the creek had natural vegetation consisting of shrubs, herbs, and climbers on both hill slope and creek bank.

Flowing creek from 8-year-old teak plantation: Water samples were collected from the flowing creek by the side of a 40% hill slope on a western aspect. Both sides of the creek contained 8-year-old teak plantations without any undergrowth.

Ghilachari

In this area, water samples from four flowing creeks and three seepages were collected, and existing land cover types were sampled.

Flowing creek from an SC cultivated area: Water sampling from this flowing creek was done near Ghilachari Bazar, 25 km away from

Manikchari toward Khagrachari. Both sides of this creek had current year shifting cultivated areas with crops of paddy, 2-year-old turmeric, and banana along with dense undergrowth on an eastern aspect of a 70% hill slope.

Seepage water from a slashed and burned SC area: Water sampling from this flowing creek was done near Ghilachari Bazar. The catchment was composed of shifting cultivated areas with crops of paddy, 2-year-old ginger, and banana along with undergrowth on a southern aspect of the hill (45% hill slope).

Seepage water from cultivated areas (hillslope agriculture and plantation): Seepage water sampling was done near Ghilachari. Hill above the dug-hole was covered with sloping land agriculture mainly ginger and turmeric in one part and another part covered with mature trees, mainly of gamar (*Gmelina arborea*), shrish (*Albizia procera*), shimul (*Bombyx ceiba*), and bamboo sporadically with 60% undergrowth on a southern aspect of 35% slope. Tree canopy coverage was about 85%, and the undergrowth consisted of assamlata at 60%, bhat at 25%, and climbers and sun grass together at 5%.

Seepage water from shifting cultivation and plantation areas: Seepage water samples were collected from dug-hole by the side of a 30% hill slope on an eastern aspect. One side of this catchment was current year shifting cultivated areas in patches of crops such as banana, bamboo, and the other side of the area was composed of plantations consisting of shrubs, herbs, and climbers on both hill slopes.

A flowing creek from shifting cultivation and plantation areas: Water samples were collected from a flowing creek at a catchment with 40% hill slopes on an eastern aspect. One side of this catchment was current year shifting cultivated areas in patches of crops such as paddy, banana, and other crops, and the other side of the area was composed of timber plantations where undergrowth consisted of shrubs, herbs, and climbers on both hill slopes.

A flowing creek from natural vegetation and partly disturbed areas: Water samples were collected from a flowing creek at Ghilachari with a 55% hill slope catchment on a western aspect. Both sides of this creek were covered with partly disturbed natural vegetation, composed of bamboo, a few trees and shrubs, herbs, and climbers on both hill slopes and creek banks, with a canopy coverage of about 60%.

A flowing creek with one side of shifting cultivation and the other side of natural vegetation: Creek water samples were collected from a 45% hill slope catchment on a southern aspect near Ghilachari. One side of this catchment was current year shifting cultivated areas in patches of crops such as paddy and banana, and the other side of the area was composed of disturbed 3-year-old gamar (*Gmelina arborea*) plantation areas. The area comprised 30% undergrowth cover consisting of shrubs, herbs, and climbers.

Water Sample Preservation and Analyses

Water sample was collected from creeks and seepages in triplicate in 200 mL polyethylene bottles filled with no headspace, caped well to prevent leakage, and preserved at 4°C in an icebox. The sample box was transferred immediately to the laboratory, and the samples were filtered using Whitman paper No. 42. Water pH, conductivity, and TDS were measured in the field using a multiprobe meter. Water samples were then preserved in a refrigerator at 4°C for subsequent analyses. Base cations such as Ca^{2+} and Mg^{2+} were determined using Perkin-Elmer AAS in the regional laboratory of the Soil Resource Development Institute (Petersen, 2002). Dissolved phosphate was analyzed by stannous chloride and ammonium molybdate method (APHA, 1998) and SO_4^{2-} by $BaCl_2$ method, NO_3-N by salicylation-sodium hydroxide method, and K^+ and Na^+ by flame photometer (Immamulhuq and Didar-ul-Alam, 2005). The result is reported in concentration owing to limitation with stage-discharge data from very small creeks where creek discharges are <100 l/day.

Results

Water quality parameters of different sites of the study area were not compared with land cover randomly; rather, the comparison and sampling were done using paired catchments (Gafur et al., 2003), so that other influencing factors such as spatial, physiographic, rainfall, and dry deposition remained independent to the water quality parameter between catchments except for land cover. The components of the composition are given by the concentrations of Na^+, K^+, Ca^{2+}, Mg^{2+}, HCO_3^-, CO_3^{2-}, SO_4^{2-}, conductivity, and TDS. Even if the number and type of variables were not exhaustive for understanding the effects on the abundance of natural and anthropic phenomena, some preliminary indications of the data structure could be inferred.

Considering the whole dataset without a spatial difference, we will not find any significant change in water quality parameters (Table 11.1) from different land uses. Hence, it is impossible to define the land use-water quality relationship while considering very different location catchments in the same analysis. The water quality results are discussed hereafter in terms of two different sources: creek water and seepage water.

Land Use and Creek Water Quality

Two creeks from one side of shifting cultivation areas and the other side natural vegetation showed lower conductivity, HCO_3^-, SO_4^{2-}, Na^+, K^+, Mg^{2+}, and TDS compared to the paired catchment of the 8-year-old timber plantation area at Kichingmontala. The water quality parameters determined for drinking water wells, traditionally used by the inhabitants, were within the safe limit of drinking water standards set by the Department of Environment (ECR 1997). The flowing creek at Kutukchari from the shifting cultivation area contained higher SO_4^{2-}, NO_3^-, Mg^{2+}, and PO_4^{3-} and lower Ca^{2+}, Na^+, K^+, and TDS than adjacent natural vegetation areas. In flowing water from a shifting cultivated area Na^+, Ca^{2+}, Mg^{2+}, and TDS contents were 10.17, 4.76, 2.57, and 30 mg/L and in natural vegetated area 11.33, 12.57, 2.25, and 80 mg/L, respectively. All these parameters, except HCO_3^- and pH, for flowing waters from shifting cultivation areas at Ghilachari, situated 10 km away from Kutukchari, showed similar differences compared to natural vegetation area. The content of HCO_3^- was similar and pH higher in shifting cultivation area at Ghilachari than natural vegetation areas. The highest total phosphorous (TP) levels were observed in creeks draining watersheds of agriculture, followed by those dominated by young forests and shifting cultivation that led to higher instream TP concentrations.

Land Use and Seepage Water Quality

Seepage water pH, SO_4^{2-}, NO_3^-, TP, and Ca^{2+} were higher and conductivity, HCO_3^-, Na^+, K^+, Mg^{2+}, and TDS were lower from shifting cultivated areas than either from dense natural vegetation with partially disturbed areas or from mature fruit orchard trees and banana plantations and timber wood plantations area at Kichingmontala. At this location, seepage water SO_4^{2-}, Ca^{2+}, and conductivity draining from shifting cultivated areas were 1.19 mg/L, 11.16 mg/L, and 122.67 μS/cm; in mature fruit tree and young plantation area 0.83 mg/L, 6.40 mg/L, and 185.33 μS/cm; and in dense natural vegetation with partially disturbed area 0.79 mg/L, 5.79 mg/L, and 167 μS/cm, respectively. At Kutukchari, seepage water from shifting cultivated areas, contained higher conductivity, HCO_3^-, NO_3^-, Na^+, K^+ and TDS, and lower SO_4^{2-} and Mg^{2+} than matured mixed plantation areas mainly of teak (*Tectona grandis*), gamar (*Gmelina arborea*), shrish (*Albizia procera*), shimul (*Bombax ceiba*), and bamboo sporadically. The major shifting cultivation crops were ginger and sesame along the slope with shrubs and climbers on the creek bank. Contents of SO_4^{2-}, Na^+, and TDS in seepage water from slashed and burned area were 0.99, 18.50, and 160 mg/l, and from mature mixed plantation 2.23, 10.75, and 27.50 mg/L, respectively.

At this location, seepage water from mature mixed plantation showed higher pH, SO_4^-, total phosphorous (TP), and Mg^{2+}, and lower NO_3^-, Ca^{2+}, HCO_3^-, Na^+, K^+, TDS, and conductivity in water than adjacent artificially drained areas developed between mature mixed plantation on the hill slope and agricultural land on the valley (Figure 11.2). The farmer used this drain to remove excess water from the agricultural field that received as runoff and seepage from adjacent hill slopes or to irrigate crops whenever required. The water sample was collected from this artificial drain to see any unusual composition

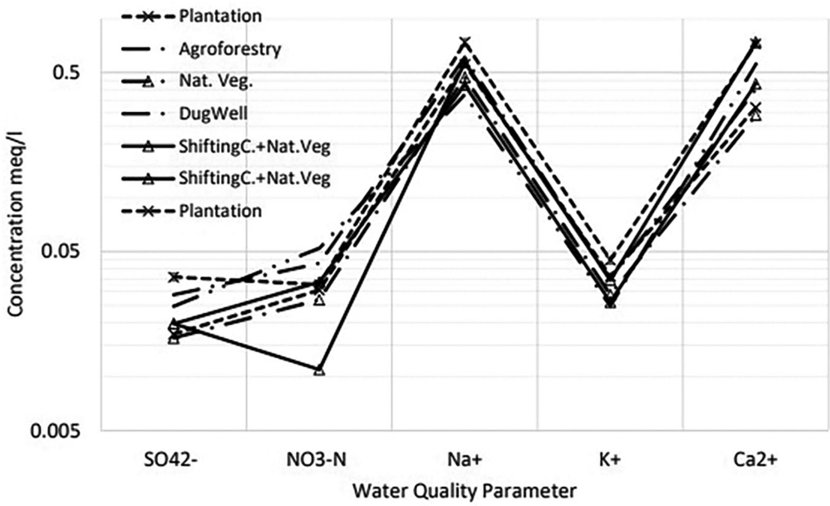

FIGURE 11.2
Concentration variation in water quality parameter for different land use/land cover in three different sites: Kichingmontala area, Kutukchari area and Ghilachari area.

of water compared to either creek or seepage water draining from matured mixed plantation areas because this drain received water from managed agricultural land in many ways. The seepage water and creek water at three sampling locations (Kichingmontala, Kutukchari, and Ghilachari areas) from shifting cultivated catchments showed higher SO_4^{2-}, NO_3^-, and TP and lower HCO_3^-, Na^+, K^+, and TDS compared to natural vegetation area (Figure 11.2).

Plantation age also influenced total N concentrations in the creek and seepage water, with lower nitrate concentrations from watersheds draining young, forested catchments than a mature forest. In contrast, when evaluated relative to a natural forest baseline, phosphorus concentrations increased for all land use change categories.

Discussion

Water quality parameters comparing different land use types (LULC) described hereafter were based on paired watersheds both for seepage water and creek water chemistry. It is well documented that the soil parameters, slope, and aspect of the hill can affect stream water chemistry along with land use. In a catchment, untangle land use effects from other potential factors, one needs to analyze and evaluate other factors carefully. Moreover, two sources of water, here creeks or seepages, can produce slightly varied characteristics that originated from the same land use.

Effect of Land Use/Land Cover of Water Quality Parameters

Higher concentration of base cations in seepage water from matured fruit trees of *Syzigium cuminii*, *Litchi chinensis*, *Arthocarpus heterophyllous*, banana, and young timber plantations of *Gmelina arborea*, *Tectona grandis*, and *Albizia procera* were associated with higher organic acid accumulation from partial decomposition of litter. Higher organic acid from partial decomposition on the young plantation and forest floor helps desorption of cations and metal ions from soil microsites (Osman, 2013). Two flowing creeks, with one side of current year shifting cultivation areas of crops such as banana and bamboo (*Bambusa vulgaris* and *B. tulda*) and the other side of natural vegetation consisting of shrubs and herbs on both hill slope and creek bank, showed a lower concentration of base cations in creek water, which might be due to higher uptake by shifting cultivation crops and natural vegetation of varied species. Higher base cations in creek water could result from cations leaching from the teak plantation where undergrowth and other vegetation are absent. Seepage water from mixed plantations indicated more retention of soil nutrients compared to shifting cultivation.

A lower concentration of nutrient (NO^3-N and PO_4^{3-}) in seepage water from mature plantations compared to small irrigation drains beside agricultural lands also indicates no fertilizer application in the plantation and shifting cultivation catchments, which agrees with the traditional concept of shifting cultivation (Brand and Pfund, 1998). Lower TDS in creek and seepage waters from shifting cultivated areas revealed low cation status or more cation absorption on soil sorption sites or soil that we believed was depleted of cations, which might be due to more runoff losses through soil erosion during the first year of shifting cultivation as the erosion trend reported in Gafur et al. (2003). Seepage water from slash and burn areas showed higher nitrate, TDS, and Ca^{2+} concentrations than comparable nearby catchments. It revealed that slash and burn areas had augmented nutrient leaching from the soils. Levin et al. (1992) observed the leaching of Mg^{2+} and NO_3^- from soils after clearing and burning of tropical rainforests. Hence, stream water and seepage from shifting cultivated watersheds should show similar trends in our study areas, whereas we observed a different trend while comparing paired watershed. This observation is not surprising as studied shifting cultivation sites of 1–2 years in age exhibit depletion or leaching that would be substantial during the first flood flashes during monsoon rains and subsequent years. The shifting cultivation sites would experience degradation or lack of nutrient elements in the soil as Haque et al. (2014) observed in many parts of this region. de Mello et al. (2018) reported an increased concentration of $NO_{3\text{-}}$N in stream water in the watershed with higher intensities of agricultural activities. A similar NO_3-N concentration trend was revealed in the studied paired catchment of agroforestry than comparable plantation or natural watershed seepage waters at Kichingmontala. The trend does not clearly describe the effect because fertilization was never practiced during shifting cultivation or other intensive land use practices compared to the intensive land use in other regions of the world in mountainous catchments (Allan, 2004; Bu et al., 2014; Yu et al., 2016). From this study, we have discerned this effect was attributable to shifting cultivation or young plantations. Natural vegetation retains more cations or nutrient element eventually, less loss of these nutrients to the seepage water.

Correlation of Spatial Variation, Slope, and Aspect of the Catchments on Water Quality Parameters

Depending on local conditions (e.g., terrain slope, geology, riparian vegetation types), grassy areas in riparian zones can reduce sediment load in runoff water. However, natural vegetation catchment with trees is necessary for riparian zones not only for bank stabilization and reduction of erosive processes but also to reduce allochthonous nutrient inputs (Monteiro et al., 2016; Zorzal-Almeida et al., 2018). The results found by Zorzal-Almeida et al. (2018) illustrating gross negative and significant correlations between natural areas, at the buffer-zone scale, and total phosphorous, conductivity, and pH strongly suggest the need to maintain the riparian vegetation (natural areas with woody plants). Their study has taken spatial autocorrelation into account while testing the effects of land use both on sediment and water quality in reservoirs (Vrebos et al., 2017). They have made a significant effort to report the ("pure") effect of land use variables on sediment and water quality.

Table 11.1 Pearson Correlation matrix for the catchment factors that may have an effect on the creek or seepage water quality parameter values. Significant correlations ($p < 0.05$) are indicated by asterisks

The correlation coefficient to the slope variation and three locations of aspect of the catchment shows no significant ($p < 0.05$) relationship with most of the studied water quality parameters (Table 11.1). However, the location has a significant influence on common cations such as Na^+, K^+, and Mg^{2+} (Table 11.1). The local geology has an influence on these parameters, although the studied area belongs to a single geological region (UNDP-FAO, 1988). The slope of the catchment in the studied areas was moderate to very steep slope classes. Though the elevation of the hills ranges from 100 to 400 m, the slope is mostly unsuitable for any sedentary agriculture and seasonal vegetation. However, shifting cultivation is often practiced on moderate to steep slopes. Due to the moderate and steep slope topography of shifting cultivation land, it loses part of the topsoil during the first year of shifting cultivation. This study did not conduct the runoff water from monsoons due to the challenging accessibility and maintenance of water sample collection during that season. However, the slope can be an indicative factor of land use that similarly affects water

TABLE 11.1

Pearson Correlation matrix for the catchment factors that may have an effect on creek or seepage water quality parameter values

Parameters	Slope	Land Use	Source	Location
pH	–0.03	–0.01	–0.15	–0.08
Conductivity	0.01	–0.37	–0.01	0.07
HCO_3^{2-}	–0.12	–0.18	0.14	–0.08
SO_4^{2-}	–0.25	–0.019	–0.27	–0.14
NO_3-N	–0.33	0.13	–0.30	–0.21
Total-P	–0.14	–0.24	–0.12	–0.03
Na^+	0.02	0.27	–0.10	0.72[a]
K^+	0.03	0.29	0.07	0.63[a]
Ca^{2+}	–0.08	–0.35	–0.13	0.02
Mg^{2+}	0.27	–0.46	0.08	0.49[b]
TDS	–0.06	–0.22	–0.21	0.02

quality parameters. Land uses in steeper slopes generally had a stronger influence on stream water quality than those in flatter ones. A steep slope basin creek has lower pH and higher TDS values than a relatively flatter basin. Land use types near streams were generally a better indicator of the effectiveness of water quality (Yu et al., 2015).

Compositional analysis for land use change effect using paired watersheds (Aitchison, 1986) realized that data size is irrelevant for compositional data since the interest is in the relative proportions of the measured components. He introduced two transformations based on ratios, the additive log-ratio transformation (alr), and the centered log-ratio transformation (clr). Classical statistical analysis methodologies can be applied to the transformed observations, taking care to use 'alr' for modeling and 'clr' for techniques based on a metric. This is because 'alr' transformation does not preserve distances while 'clr' preserves them but leads to a singular covariance matrix.

Higher variability matrix for 11 Eigenvector compositions was deduced. The covariance matrix of paired watersheds compared with natural vegetation or old plantation watersheds revealed that variation due to land use and water quality could be detected. Compositional package (van den Boogaart and Tolosana-Delgado, 2013) in "R" has been used to prepare the principal components biplot. The first two components comprising 70% of the variation among land uses and water quality revealed the dynamics of cations and anions and other parameters in the sampled water. From the principal components biplot, it revealed that the variation of nitrate and total phosphorous were independent of the variation of conductivity and calcium concentration. The total phosphorous, sulfate, nitrate concentrations, and pH values were correlated, and nitrate and Ca^{2+} values represent higher variations compared to other parameters.

Water Quality Parameters and Plantation Watershed

Plantation watersheds in the three-study area have a varied composition of tree species and age of plantations in different sampling sites. We were envisioning that old (>8 years) mixed plantations without or minimal forest management practices would act the same function as natural vegetation. Instantly there was a plantation site with a mixed vegetation composition. Water quality parameters Mg^{2+}, TDS, and nitrate showed more significant variation in plantation catchments with shifting cultivation catchment waters. The seepage in shifting cultivation (SC) would be more depleted due to higher drainage or erosional loss of surface soil and soluble cations. Hence, soil solution in SC would be depleted in contents of exchangeable cations and anions. A similar interaction was also observed in other catchments. Lower content of PO_4^{3-} in plantations than in shifting cultivation areas would be associated with uptake of this compound by trees and/or immobilization of phosphate compounds in the soil from tree ashes during the slash and burn phase of shifting cultivation. Again, during shifting cultivation, immobile plant and soil phosphate compounds would get released, which can add additional phosphate in the creek and seepage water due to leaching during the early years of shifting cultivation. Karmakar et al. (2012) and Haque et al. (2010) also reported a similar phosphate trend from shifting cultivation and agricultural or plantation catchments compared to forest catchments. Natural vegetation watersheds usually composed of a riparian vegetation strip along the creeks would retain more allochthonous material (Zorzal-Almeida et al., 2018), eventually with the least loss of these contents in seepage and creek water compared to other land use watersheds. Centered-log-transformed ratios (clr) of the water quality parameter reveal that among the allochthonous material, total-P Ca^{2+} and Mg^{2+} and TDS values would be highly indicative of understanding the impact of plantation land use in the watershed. Many studies (Morgan et al., 2006; Zampella et al., 2007; Faithful et al., 2012) also reported that a higher concentration of Ca^{2+} and Mg^{2+} in stream water in altered watersheds than forested ones.

Water Quality Parameters and Shifting Cultivation Watershed

Conductivity, a composite indicator of ions in water samples, reveals links to the shifting cultivation land use type. The "clr" analysis of the data shows that shifting cultivation and agricultural practice on the slopes reveal different variations. Though the practice of shifting cultivation is linked to the agricultural practice, the agricultural practice is a more detrimental type of land use that is linked to higher erosion in this area. It follows moderate to deep plowing compared to zero tillage in shifting cultivation practices (Gafur et al., 2003). Hence, runoff loss would deplete more exchange cations in agricultural soils than shifting cultivation soils. Hence, a higher variation of cations compared to natural vegetation is expected and revealed for total-P, Ca^{2+}, Mg^{+}, and TDS.

Hence higher variation of these ions is essentially linked to agricultural practices and shifting cultivation. Moreover, the same trend is prominent in the mixed catchments of shifting cultivation and plantation land use. Total-P, conductivity, and cations can indicate the change of land use from natural vegetation. The agricultural practices in the studied catchments found less or no fertilization based on deep plowing on the mild hill slopes. However, the variation of total-P shows a relationship to agriculture. Lower concentrations in Na^+, K^+, and Mg^{2+}in seepage water in agricultural and shifting cultivated areas appeared due to possibly of soils subjected to conservation practices through mulching and alley cropping across the slope by farmers. This form of agricultural land use practice on a mild slope was not very common in this study area as observed during water sampling.

Water Quality Parameter in Plantation to Shifting Cultivation Watershed

We have observed that the mixed old plantation site has attained natural vegetation as its vegetation composition includes mixed storied vegetation such as herbs, shrubs, and trees. Hence, the conversion of mixed plantation or natural vegetation watersheds to shifting cultivation would strongly respond to the water quality. The comparison of shifting cultivation watersheds to the plantation reveals higher variation in anions and cations except Na^+ and K^+ concentrations. Thus, the conversion of the plantation to shifting cultivation will experience almost similar changes in water quality parameters in this region. It should be noted that the natural vegetation in this region is mostly secondary regenerated vegetation that grows on the shifting cultivated land or the plantations. Hence, pure natural vegetation for a more extended period does not exist in the region due to rotational shifting cultivation over the centuries. As a result, vegetation characteristics of natural vegetation and mixed plantations do not vary greatly except for pure monoculture plantation of exotic tree species such as *Tectona grandis* or *Acacia* spp.

Schilling and Spooner (2006) and Zampella et al. (2007) emphasized that grasslands or other perennial vegetation in agricultural settings should be part of a long-term solution to water quality problems associated with basin-wide upland land uses. Shifting cultivation land use practices were found more frequently in the sampled catchments, where it mainly composed half or less of the parts of catchment areas. The slope of shifting cultivated lands is relatively less steep compared to permanent vegetation such as fruit trees or timber plantation areas, or natural vegetation. Gafur et al. (2003) also observed similar characteristics of shifting cultivation lands in this region, which was probably due to higher accessibility and thicker soil layers at the mild slope land. Though it is out of the scope of this study, we observed a lower soil depth in shifting cultivation land compared to the plantation and other land use types. Haque and his group (Biswas et al., 2012) also reported a similar trend in this region in their watershed study. Though the discharge rate from the catchment was not studied due to the lack of continuous monitoring facilities, more reliable discharges in the creeks were experienced and expected from natural vegetation catchments by the local community, which is frequently articulated by the inhabitants. They also believe that good taste and clearer water with the more frequent presence of aquatic fauna were associated with the creek of natural vegetation catchment compared to the catchment of other land uses.

Limitation of Micro-catchment Approach in Land Use-Water Quality Study

The correlation study has shown that watershed-specific characteristics, rather than land use, influenced controlled stream NO_3-N, SO_4^{2-}total-P and Ca^{2+}, Mg^{2+}, and Na^{+} concentrations to some extent. This micro-catchment-based paired watershed approach could fail to correlate the land use with stream water quality parameters such as nitrate, Ca^{2+}, or other nutrient element availability since the pure form effect of land use cannot be achieved under field conditions if the spatial variation is not teased out from water quality parameter variations. The major limitation of this study is that the conclusion drawn was based on water samples collected during the lean season of flows of creeks rather than both in lean and monsoon, which would be highly indicative of the land use change.

Urban land had a higher positive relationship with degraded water quality at small scales, whereas agricultural land displayed the opposite scale effects (e.g. Datta et al 2022; Shi et al., 2017). However, in this study, catchment with human habitation was consciously avoided from sampling. The sampling campaign of this study could not be conducted multiple times because many of the study site land use and land cover (LULC) had been altered while the second sampling campaign was planned. Hence, field scale studies of this kind will have limited data available for paired catchments to compare and a specific land use type throughout the catchment. Few water quality parameters such as DO, NH_4^{+}, total Fe, and SiO_2 were not reported in this study though we believe these parameters could be interesting to study to characterize land use for certain conditions (Zorzal-Almeida et al., 2018). Dissolved oxygen (DO) in creek water is also assumed to be responded strongly to gradations in land cover due to eutrophication attributed to urban catchments (Morgan et al., 2006; Rodrigues et al., 2018).

Conclusions

Thus, findings of the present study indicated that shifting cultivation, plantation, and agricultural land covers had an impact on water quality of creek and subsurface seepage waters, which ultimately affected stream water quality. Stream creek and seepage water NO_3-N, SO_4^{2-}, total-P as well as Ca^{2+}, Mg^{2+}, and Na^+ concentrations monitoring would be effective to understand land use and land cover (LULC) dynamics in upstream headwater watersheds. Hence, to stop further deterioration in water quality due to shifting cultivation and monoculture plantations in CHTs, one needs to replace potentially abusive land use – shifting cultivation and agriculture on steep slopes, by a mixed type of permanent vegetation through the protection of existing natural vegetation and planting of seedlings artificially. However, the cumulative effect from different sub-catchments to a higher order confluence would be interesting to understand the accumulation and mixing processes from highly variable land cover catchments. It is important to note that remotely sensed data with high resolution would reduce ambiguity on watershed delineation issue for micro-catchments. Secondly, a watershed-scale micro-catchment study would help to calibrate water quality influenced by land cover change as well as nutrient or contaminant transport/accumulation in the catchments.

Acknowledgments

Thanks to the reviews for the suggestions and comments, which have improved the readability of the manuscript. The authors highly appreciate USDA grant BG-ARS-123 for funding this study and Regional Soil Resources Development Institute in Comilla for providing facilities for analyzing some water quality parameters.

Part of this contribution is based on "Karmakar, S. et al. 2019. Water quality parameter as a predictor of small watershed land cover. Ecological Indicators 106, 105462" by permission from the publisher Elsevier.

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12

Soil Moisture Estimation Using a SAR Water Cloud Model for an Improved Anchor Pixel Selection Process in SEBAL

M. M. Prakash Mohan, K. Rajitha, and Murari R. R. Varma

CONTENTS

Introduction

Evapotranspiration (ET) is a proxy parameter of the hydrological cycle, which defines the pathways toward the solutions for various problems related to agriculture and climate change. The discrete mode of measurement of ET fails to provide synoptic variability of ET from a spatial and temporal perspective. The lacuna of traditional point measurements of ET has been minimized by remote sensing–based models (Moran et al. 1994, Norman, Kustas and Humes 1995, Roerink, Su and Menenti 2000, Su 2002, Allen et al. 2011, Senay et al. 2013) which use multi-spectral datasets as inputs.

Surface Energy Balance Algorithm for Land (SEBAL) (Bastiaanssen et al. 1998) is one of the most popular satellite remote sensing models which uses a hybrid approach combining both empirical and physical parameterization

DOI: 10.1201/9781003175094-12

schemes. For the last two decades, SEBAL has proven its application in diverse crops with different growth stages and various agricultural practices in more than 30 countries under different climatic conditions (Lee et al. 2016, Rawat et al. 2017, Grosso et al. 2018, Kamali and Nazari 2018). The less ground data requirement, simplicity, and robustness with a better performance matrix for diverse field conditions popularized SEBAL compared to other ET models.

SEBAL applies surface energy balance theory to estimate ET as a residual of the energy balance equation. In SEBAL, sensible heat flux (*H*) is the most sensitive and computationally complex component, which is highly susceptible to heterogeneous conditions of agricultural land (Marx et al. 2008). SEBAL calculates *H* based on the assumption that the near-surface air temperature gradient (dT) and the land surface temperature (T_s) at two extreme conditions in the field are linearly related (Bastiaanssen et al. 1998). The linear relationship is expressed as follows:

$$dT = aT_s + b \tag{12.1}$$

Calibration coefficients *a* and *b* of equation 12.1 were determined by a hot extreme and a cold extreme selected by the operator from the T_s image by correlating with Normalized Difference Vegetation Index (NDVI) and land-cover maps. The basic philosophy of hot and cold pixels is that they represent two extreme energy consumption conditions for ET. The selection process follows a subjective judgment using knowledge about the study area. Cold pixels represent well-irrigated, actively transpiring vegetation, whereas hot pixels are bare open fields with very high radiometric heating. The two anchor pixels are employed to calculate dT, and thereby *H* was calculated using the modified bulk transfer equation 12.2:

$$H = \frac{\rho C_p{}^{dT}}{r_{ah}} \tag{12.2}$$

where r_{ah} is the aerodynamic resistance, C_p is the specific heat of air at constant pressure (J/kg K), and ρ is the mean air density (kg/m^3). The SEBAL adopted an iterative procedure to calculate *H* by correcting the value of r_{ah} by using inputs from anchor pixels. This iterative autocalibration procedure could even mitigate the errors in T_s measurements.

The research conducted in the domain of anchor pixel selection showed that the process was too sensitive, and it was reported that the deviations of ±2 K in T_s values could result in the variation of estimates of *H* in the order of 10%–15% (Long, Singh and Li 2011). Thus, the selection process of the anchor pixels corresponding to the extremities is critical, which control the further parameterization for *H* computations and hence the accuracy of ET estimation in SEBAL. The current research is moving toward automating the selection process to make it more robust and numerically stable. Different approaches for the automation of the anchor pixel selection process were developed using optical images as input (Allen et al. 2013, Bhattarai et al. 2017a). Allen et al. (2013) proposed a semi-automatic procedure for selecting the anchor pixels, which followed a statistical approach considering T_s and NDVI histograms. Later Bhattarai et al. (2017a, b) proposed a fully automated procedure using Exhaustive Search Algorithm (ESA), assuming the selection procedure as an optimization problem. It was identified that the input of NDVI and T_s alone would result in similar anchor pixels and suggested incorporating additional inputs to refine the selection process.

One of the main avenues for widening the scope of SEBAL applications for various surface heterogeneity conditions is to identify the proper anchor pixels by capturing more biophysical properties of the field. Incorporating biophysical parameters like NDVI, land-cover type, soil moisture content, etc., in the anchor pixel selection process limits the user intervention and makes it more versatile (Allen et al. 2011).

Soil moisture content is a critical parameter that facilitates the appropriate identification of well-irrigated vegetation and hot bare open fields. Optical images could not accurately retrieve the soil moisture variations under dense canopy conditions. It was assumed that the use of both NDVI and T_s indirectly accounted for the influence of soil moisture variations in locating anchor pixels. However, the use of T_s at water-stressed conditions changes roughness lengths of momentum and heat, leading to changes in kB^{-1} parameter. This parameter directly influences the extra resistance term added to the numerator of the bulk transfer equation while using aerodynamic T_s as the surrogate to aerodynamic temperature.

In water-stressed semi-arid conditions, when water availability becomes the limiting factor for ET, Surface Energy Balance (SEB) models were found to perform better for a range of crops and land covers by integrating soil moisture information (Gokmen et al. 2012). It was reported that in both dry and wet conditions of soil, SEBAL performed better compared to other popular models like METRIC (Mapping ET with Internalized Calibration), SEBS (Surface Energy Balance System), and SSEBop (operational Simplified Surface Energy Balance) (Bhattarai et al. 2017b). Similarly, improved performance of the SEBAL was reported by incorporating soil moisture or plant water stress component in the case of a specific agricultural crop (high biomass sorghum) (Wagle et al. 2017). These significant outcomes affirm the prospect of the SEBAL model to integrate with soil moisture in this study.

The onset of Synthetic Aperture Radar (SAR) technology with different polarimetric modes facilitates soil moisture retrieval under various canopy conditions. Since the SAR images are sensitive to the changes in dielectric properties of the soil, they are more suitable to understand the soil moisture conditions beneath the crop canopies. Numerous studies and experiments for the last three decades with SAR data and its behavior with different soil wetness conditions improved our understanding of SAR backscatter to soil moisture relations. This led to the development of various soil moisture estimation models (physical models, semi-empirical, and empirical models). The semi-empirical Water Cloud Model (WCM), known for its efficiency in agricultural regions with varying crop cover, was used to estimate soil moisture information in this study.

The existing ambiguities related to the anchor pixel selection process for fragmented agriculture regions and its highlight in the SEBAL are the major impetus of the present research. This study has been aligned with the potential of Sentinel 1A and Landsat 8 data for soil moisture retrieval through WCM to demarcate the probable zones of hot and cold pixel locations in an agricultural region of diverse canopy architecture.

Study Area and Datasets

The location selected for the study is an agricultural village located 45 km north of Hyderabad city, India (Figure 12.1). The study area is located within latitudes 17°40′49.59″N and 17°48′15.14″N and longitudes 78°28′37.6″E and 78°40′27.025″E, covering an area of ~136 km^2. The region has tropical wet and dry climate with a mean annual temperature of 25°C. The predominant soil texture of the region is loam. The rainy season of the location is restricted to a 3-month period which spans between July and September with an average annual rainfall of 900 mm. The study area consists mainly of fragmented agricultural plots with relatively flat terrain, dominated by mango, corn, cotton, paddy, and grapes. The major source of irrigation in the area is

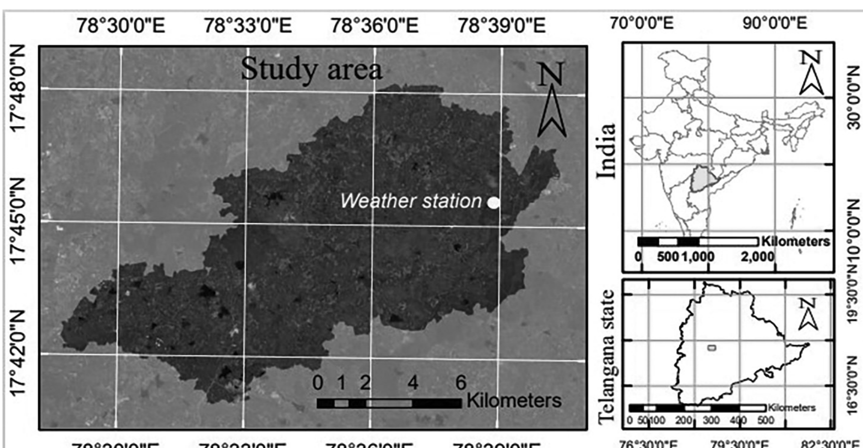

FIGURE 12.1
Location map showing the study area using a standard false color composite of Landsat 8 image.

precipitation and groundwater tapped through bore wells. The area comes under the proposed Kaleswaram Irrigation project.

Ground Data

The field surveys were carried out concurrently with SAR and optical RS data acquisitions. In situ datasets used for the analysis were the meteorological data obtained from the weather station located within the study area, soil moisture content, and crop type information from various agricultural plots. For moisture content estimation, soil samples were collected from various plots with different crop types across the study area. The samples were collected during morning hours, just after the SAR data acquisition. The depth of soil sample collection was restricted within 5 –10 cm for all the image acquisition dates, assuming that the WCM could model the soil moisture from the top layers of the soil column. The total number of soil moisture values for calibrating the samples was 70, and out of that, 11 samples were discarded due to a higher amount of crop residues. Each soil moisture value is the average of five to six values taken from selected plots. The geographic coordinates of the sampling points were recorded using a handheld Garmin GPS. The samples were oven-dried at 105°C for 24 hours to estimate the gravimetric soil moisture content and multiplied by bulk density to convert it into volumetric soil moisture content.

Satellite Data Acquisition and Pre-Processing

SAR images of Sentinel 1A and optical remote sensing images of Landsat 8 were used in this study. The dates of image acquisitions were chosen so that both the satellites pass over the study area on the same day. Since the temporal resolution of Landsat 8 is 16 days and Sentinel 1A is 12 days, concurrent acquisition happens only once every 48 days. The images used in this research were acquired between February 2017 and April 2018. The details of the images are shown in Table 12.1.

Sentinel 1A satellite has a C band Synthetic Aperture Radar instrument with an operating frequency of 5.405 GHz with dual-pol data in various imaging modes. Ground Range Detected data in Interferometric Wide (IW) swath mode was used in this study with a spatial resolution of 10 m. Five images were collected (Source: European Space Agency, Sentinel datahub) for the analysis in ascending mode with incidence angles ranging from 31.07° to 46.10° with VV and VH polarization modes. The study site was imaged with

TABLE 12.1
Sentinel 1A and Landsat 8 Datasets

Data Acquisitions Dates
3 February 2017
18 November 2017
5 January 2018
22 February 2018
11 April 2018

an average incidence angle of 38°. The level-1 Ground Range Detected data were radiometrically corrected to obtain backscattering coefficient (σ^0 images in decibels (dB)), and then Refined Lee filter (Lee et al. 1994) was applied to eliminate the speckle noise. Further, the images were geometrically corrected, and Universal Transverse Mercator coordinate system was assigned. The datum selected was World Geodetic System 1984.

Landsat 8, level-1 data were downloaded concurrent with the date of acquisition of Sentinel 1A data sets from the United States Geological Survey data distribution platform. Level-1 data (geometrically corrected with Universal Transverse Mercator coordinate system and World Geodetic System 1984 datum) include acquisitions by both Operational Land Imager (OLI) and Thermal Infrared Sensor. Operational Land Image instrument collects data in nine spectral bands with 15–30 m spatial resolution, whereas Thermal Infrared Sensor records thermal radiance at 100 m spatial resolution using two thermal infrared spectral bands. The images acquired were radiometrically corrected using the information provided in the product metadata.

Water Cloud Model

WCM (Attema and Ulaby 1978) is a canopy backscattering model used to extract soil moisture and canopy biophysical parameters by using simple inversion schemes. WCM is popular for modeling the SAR backscattered signal using soil moisture values from the field and vegetation parameters from satellite images. The model splits the total backscattering coefficient of a vegetated area into the contribution from vegetation scattering and the soil surface scattering for a given SAR incidence angle (θ_i). Figure 12.2 shows the graphical representation of WCM illustrating the scattering contributions of soil and vegetation. The following equations represent WCM (equations 12.3–12.6):

$$\sigma_{\text{copol}} = \sigma^0_{\text{veg}} + \sigma^0_{\text{veg+soil}} + T^2\sigma^0_{\text{soil}} \tag{12.3}$$

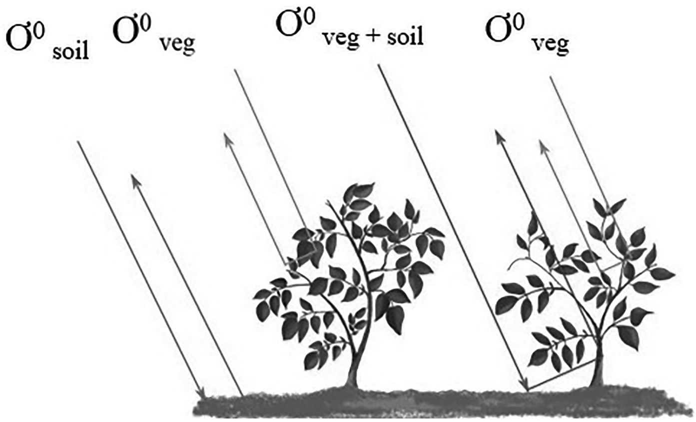

FIGURE 12.2
Graphical representation of SAR Water Cloud Model. Light rays from left to right are Ơ0 soil, Ơ0 veg, Ơ0 veg+soil, Ơ0 veg.

$$\sigma^0_{\text{veg}} = AV_1 \cos(\theta_i)\left(1 - T^2\right) \tag{12.4}$$

$$T^2 = \exp\left(-2BV_2 \sec(\theta_i)\right) \tag{12.5}$$

$$\sigma^0_{\text{soil}} = C + DM_v \tag{12.6}$$

where T^2 is the two-way attenuation factor through the canopy layer; V_1 and V_2 are vegetation descriptors (vegetation water content or leaf area index or NDVI); M_v is the volumetric soil moisture content; A is a parameter that depends upon the backscatter from an optically thick canopy at the full cover; B is a canopy attenuation coefficient; C is the radar backscatter from a completely dry soil; and D relates to the sensitivity of the radar to variations in volumetric soil moisture content. Since $\sigma^0_{\text{veg+soil}}$ is negligible for co-polarized channels (VV or HH), it was excluded from further calculations.

These empirical parameters (A, B, C, and D) are generally estimated using a multiple regression approach, and the earlier researchers showed that the physical interpretations of the parameters were difficult to explain using regression fitting (Graham and Harris 2002, Bériaux et al. 2015).

Various researchers proposed different parameterization schemes due to the lack of a standard procedure for WCM (Ulaby et al. 1984, Bouman and Goudriaan 1989, Prevot, Champion and Guyot 1993, Xu, Steven and Jaggard 1996, Shoshany et al. 2000, Champion, Prevot and Guyot 2000, Svoray and Shoshany 2002, Graham and Harris 2003, Kumar, Prasad and Arora 2012). Since agricultural crops dominate the area, the NDVI values were low for most regions, and the errors due to NDVI saturation were not expected. However, the less penetration capability of the SAR C band resulted in less agreement between simulated and measured backscattering coefficient.

Hence a modified procedure was adopted in this research, which involves estimating empirical parameters using the Levenberg-Marquardt (Marquardt 1963) based optimization approach modified by a "virtual NDVI" correction method. The correction procedure using the "virtual NDVI" concept developed in this study is explained in detail as follows.

Inversion of Soil Moisture Content

The estimation of soil moisture content using WCM was based on inversion modeling followed by the model calibration. The calibration process resulted in a system of nonlinear homogenous equations with four unknown parameters, *A*, *B*, *C*, and *D*. The nonlinear nature of these unknown parameters and the requirement of an optimal non-trivial solution became challenging during model calibration. The convergence toward the global minimum from a parameter space that contained multiple local minima decided the reliability of the inversion model for the accurate estimation of soil moisture content. The widely used optimization approach based on the Levenberg-Marquardt was applied in this study to obtain the parameters *A*, *B*, *C*, and *D*. The minimization of the difference between the calculated and measured σ^0 value was considered as the cost function of the optimization, which considered *A*, *B*, *C*, and *D* as decision variables.

The model calibration in the present study required the synergetic use of inputs from Landsat 8 and Sentinel 1A data. The Landsat 8-derived NDVI was used as vegetation descriptors (V_1 and V_2) in equations 12.5 and 12.6. The Sentinel 1A derived σ^0 values corresponding to the ground measured points of soil moisture content were used for calibrating the model. The model became more complicated under multiple layers of canopy conditions due to the higher-order scattering components. This might be due to the assumption of homogeneous vegetation for the implementation of WCM. In this study, NDVI is the only vegetation descriptor adopted, which was not appropriate to capture the intricacies of varying canopy structures. This leads to inconsistent performance of the model for varying vegetation conditions. Without addressing these inconsistencies, the model converged with a biased output where the lower NDVI areas showed less soil moisture values and the higher NDVI areas showed higher soil moisture values. To overcome this, the model required a wide range of NDVI values and corresponding ranges of in situ soil moisture measured values to represent the heterogeneous agricultural field conditions in the calibration data sets. The demand for numerous calibration datasets was not feasible in larger areas with heterogeneous field conditions. The lack of representative field samples leads to erroneous under or overestimations of resultant soil moisture values. To account for these ambiguities, this study adopted a virtual NDVI-based correction of model parameters on a local scale. The corrected or the virtual NDVI values were further used for soil moisture inversion for the entire image.

Correction Factor Applied to Estimate the Moisture Content in WCM

The results obtained using the Levenberg-Marquardt approach were refined in this study by employing a new approach of error factor estimation by simultaneously utilizing the inputs from Sentinel 1A and Landsat 8. Sentinel 1A-derived backscatter coefficients obtained after radiometric calibration and NDVI values extracted for the same locations were used as inputs for the correction procedure.

The Levenberg-Marquardt approach terminated with a set of *A*, *B*, *C*, and *D* values which were not satisfied for all calibration points, leaving a significant magnitude of error that is of concern for further applications like anchor pixel selections. Hence a new approach of error reduction (estimated vs. measured soil moisture) was applied in the current research, which was based on the estimation of virtual NDVI. The virtual NDVI in this context is the NDVI for which the σ^0 "observed" and "calculated" is equal for a given set of *A*, *B*, *C*, *D* and soil moisture values. Then the NDVI factor (*f*) of deviation of virtual NDVI from the original NDVI was calculated by taking the ratio between the virtual NDVI and the original NDVI. The value of the NDVI factor was unity for sampling locations where the estimated and the measured soil moisture were equal. In all other cases, the *f* was above or less than unity. In order to apply the NDVI factor for the entire image, an empirical relationship was established between *f* and $\sigma^0_{VV} / AV\cos(\theta)$. The linear variation of *f* with respect to the $\sigma^0_{VV} / AV\cos(\theta)$ facilitated the retrieval of virtual NDVI for the entire image. This correction method was a local method that was applied to every pixel of the image.

Anchor Pixel Selection Method

This study modified the existing manual method for selecting anchor pixels in SEBAL. It was observed that the manual approach is based on the behaviors of NDVI albedo and T_s exhibited at hot and cold pixel locations. The primary step performed for the anchor pixel selection process was to identify the potential candidates of pixels for a further selection of hot and cold pixels. To achieve these, the pixels with NDVI values greater than zero were retained in the image, and then it was further partitioned to soil and vegetation areas by considering an NDVI threshold of 0.3. The selected threshold was an approximated value obtained by analyzing NDVI values against various crop covers in the study area. Since the NDVI values of cold and hot pixels usually lie much away from the threshold value, a slight change in the threshold value had no impact on the anchor pixel selection. The main objective of partitioning the NDVI image was to obtain approximate colder and hotter regions in the study area. We tested the homogeneity of NDVI by adopting the methodology using albedo (Allen et al. 2013).

Generally, the hot pixels were relatively dry, bare soil regions with minimum NDVI and maximum T_s values clustered together and geographically nearer to the weather station. The intersections of regions of minimum soil moisture content combined with less NDVI and maximum T_s became the prospective candidates for the hot pixels. Similarly, in the case of cold pixels, the regions with relatively higher NDVI and higher soil moisture content combined with lower T_s were identified as candidate cold pixels.

The knowledge about the relative variations of soil moisture in the region was more significant than its absolute values to locate the cold and hot pixels. Therefore, to understand the relative variation between moisture content within the scene, normalization of the WCM-derived soil moisture content (SM_n) was applied to a scale of 0 to 1 for all the dates of acquisitions.

For selecting the cold pixels, 20% pixels with higher NDVI values were extracted from the NDVI-partitioned image corresponding to vegetated areas. The corresponding pixels from T_s image were selected for further processing. The resultant pixels were further constrained to the proximity of a 5 km radius from the observatory located within the study area. The landuse details of the site collected through a ground survey facilitated the filtering of prospective pixels by eliminating the region corresponding to urban forest and buildings. From the subset of pixels obtained, those pixels with higher NDVI, less T_s, minimum distance from the observatory, and higher SM_n were selected. The final selection was subjected to manual judgment by giving more preference to higher NDVI, lower T_s, and higher SM_n values. Since all the selected pixels lie within the 5 km radius of the observatory, minor variations in distance values were not considered for conflicting cases. Similarly, hot pixels were selected by changing the criteria of higher NDVI to lower NDVI and lower T_s values to higher values and higher SM_n values to lower values. The flow chart, which shows the method for selecting cold and hot pixels, is given in Figure 12.3. To justify the appropriateness of the method, available energy ($R_n - G$) was calculated for all the resultant hot and cold pixels, which were obtained by considering "with" and "without" soil moisture conditions. The methods for estimating R_n and G are explained in detail by Allen et al. (2007).

The available energy criterion was applied based on the concept that the available energy is completely converted to sensible heat flux at hot pixel locations, whereas in the case of cold pixel locations, it is converted into latent heat flux. Based on this concept, selected anchor pixels should have higher available energy than other candidates.

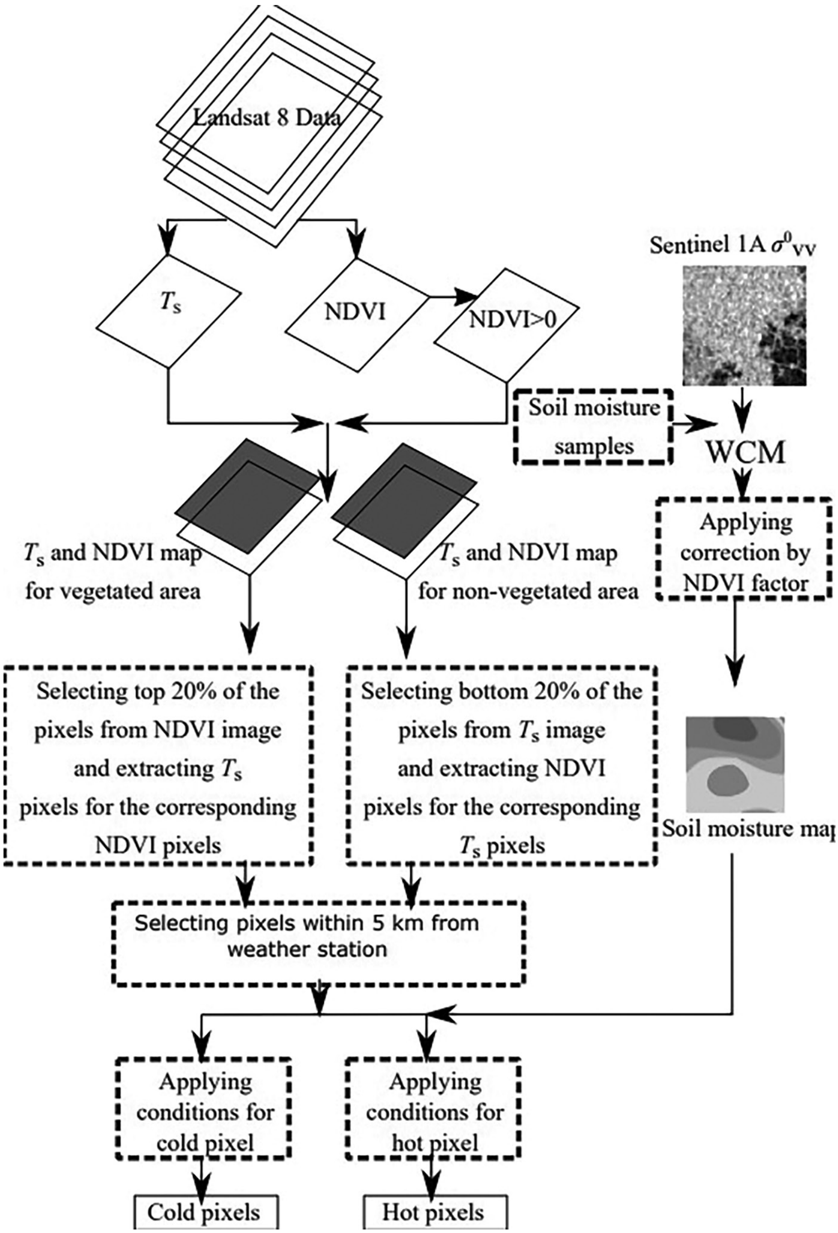

FIGURE 12.3
Flowchart for locating cold and hot pixels.

Results and Discussions

The calibration of the WCM was the prime step executed for selecting anchor pixels for all the dates of acquisitions (Table 12.1). In situ volumetric moisture content and Landsat 8-derived NDVI values and σ^0 values derived from Sentinel 1A were used to calibrate the WCM. The values of *A*, *B*, *C*, and *D* were obtained for each day of satellite image acquisition by executing the Levenberg-Marquardt algorithm. Virtual NDVI and *f* were calculated for all the calibration dataset locations by following the procedure explained below. A relationship between $\sigma_{VV}^0 / AV\cos(\theta)$ and *f* was established for every date of acquisition.

It was observed that there existed a linear relationship between the parameters for all the dates with a high coefficient of determination (R^2) value (>0.85). The slope indirectly indicated the crop cover status of the study area for the day under consideration. The maximum slope was observed for 11 April 2018, compared to other dates and, it was found that the area was progressing toward dry weather conditions with less green vegetation. The data of 18 November 2017 showed a mild slope with comparatively healthy crop cover for the region. The average NDVI of the land area was also compared, and it was found that 11 April 2018 had the least value of 0.24, whereas 18 November 2017 showed the highest value of 0.45.

Using the linear equations, factor *f* was estimated for the entire image for various dates, thereby creating a virtual NDVI image.

Using the virtual NDVI as vegetation descriptors along with *A*, *B*, *C*, and *D* parameters, σ_{VV}^0 was calculated using equation 12.2. The calculated and observed σ_{VV}^0 values were compared, and a significant correlation was observed. The details of R^2 and Root Mean Square Error (RMSE) values are shown in Figure 12.4.

Soil moisture values were calculated for the entire study area for various acquisition dates. The calculated and measured values of soil moisture showed good agreement for all the dates ($R^2 > 0.7$ and RMSE < 2%). The soil moisture maps for the study area for the selected dates indicate the extremely dry conditions prevailing during April in this region.

The selection process discussed in Figure 12.3 was applied to locate the anchor pixels for the entire set of images. The process was executed "with" and "without" soil moisture values to identify its significance as an auxiliary parameter.

The parameters considered in the process of anchor pixel selection were NDVI, T_s, relative variation of soil moisture content, and pixel proximity to the weather station. The distance criterion was applied only to restrict the anchor pixels within a 5 km radius of the observatory, and hence minute variations in the distances were not given much importance.

The results (Table 12.2) showed that the anchor pixels selected "with" and "without" considering the soil moisture differed for most cases.

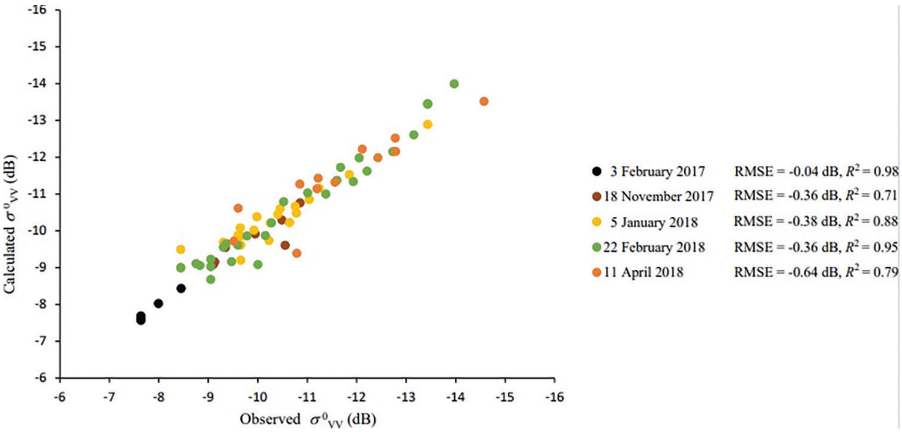

FIGURE 12.4
Scatter plot between calculated and measured σ^0 values.

For 3 February 2017, hot pixel locations were the same for both approaches. Similarly, for 22 February 2018 and 11 April 2018, cold pixel locations were the same. In all other cases, both approaches delivered different anchor pixel locations. The geographical locations of anchor pixels obtained for both approaches are shown in Figure 12.5.

To justify the appropriateness of the selection method, the available energy of the anchor pixels obtained "with" and "without" soil moisture was compared. It was found that the anchor pixels obtained by considering soil moisture as the auxiliary parameter exhibited higher available energy compared to that of the "without soil moisture" criterion.

A comprehensive representation of the distribution of parameters for resultant cold and hot pixels was obtained and it was observed that the available energy for all cold pixels was higher than that of the hot pixels. It was justified by the fact that there was a phase change of water at cold pixel locations, whereas, at hot pixel locations, energy was utilized entirely to increase the surrounding air's temperature without any phase change. Therefore, among the candidate cold and hot pixels, the suitable anchor pixels should possess higher available energy than other pixels. This was also justified by the fact that at cold pixels, the soil heat flux (G) is low due to good vegetation cover and the net radiation (R_n) is high because of low albedo, which maintained the value of available energy term ($R_n - G$) higher compared to bare soil conditions (Van der Kwast et al. 2009).

In the case of the soil moisture-based approach, all the selected anchor pixels had either higher or equal available energy compared to the approach where soil moisture was not considered as an auxiliary parameter. Available energy-based comparison was used in this study to verify the robustness of the approach where soil moisture was incorporated.

TABLE 12.2

Hot (H) and Cold (C) Pixels Were Obtained for Various Dates, Considering Surface Moisture Variations Calculated by WCM

		Considering Surface Soil Moisture					Without Considering Surface Soil Moisture				
Date	**C/H**	**Longitude (°)**	**Latitude (°)**	**NDVI**	**Ts (K)**	**SM_n**	**Longitude (°)**	**Latitude (°)**	**NDVI**	**Ts (K)**	**SM_n**
3 February 2017	C	78.6449	17.7690	0.75	303.58	0.53	78.6457	17.7679	0.76	302.96	0.48
	H	78.6852	17.8095	0.10	310.63	0.12	78.6852	17.8095	0.10	310.63	0.12
18 November 2017	C	78.6472	17.7666	0.79	300.73	0.84	78.6483	17.7663	0.80	301.28	0.80
	H	78.6828	17.7979	0.12	306.90	0.10	78.6836	17.7973	0.12	306.81	0.10
5 January 2018	C	78.6445	17.7268	0.67	299.38	0.42	78.6469	17.7677	0.69	300.58	0.23
	H	78.6273	17.7236	0.15	306.90	0.15	78.6262	17.7230	0.15	307.40	0.16
22 February 2018	C	78.6283	17.7343	0.54	300.96	0.93	78.6283	17.7343	0.54	300.96	0.93
	H	78.6713	17.7718	0.07	309.40	0.15	78.6719	17.7729	0.07	311.95	0.25
11 April 2018	C	78.6263	17.7815	0.65	303.22	0.95	78.6263	17.7815	0.65	303.22	0.95
	H	78.6804	17.7791	0.18	310.14	0.16	78.6292	17.8179	0.18	310.39	0.36

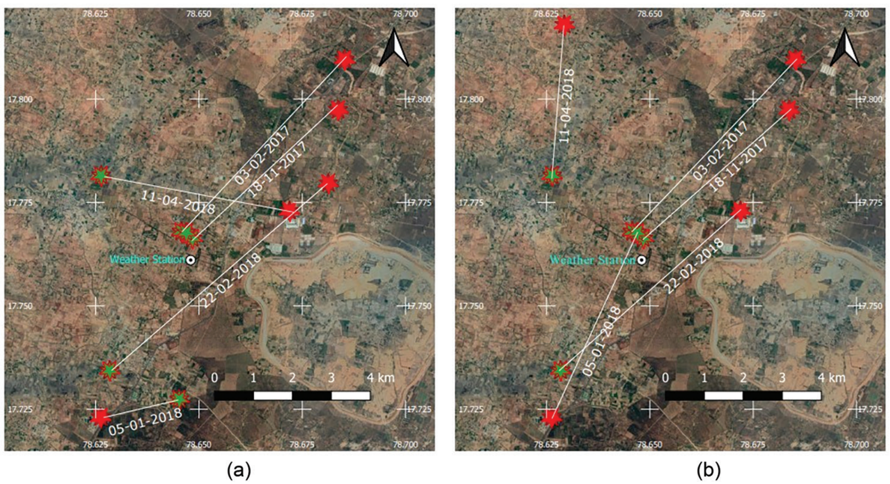

FIGURE 12.5
Locations of cold and hot pixels for various dates with and without considering soil moisture. Red (dark grey) stars and green (medium grey) stars, respectively, are the locations of hot and cold pixels, which are connected by a straight line for a specific date. (a) Considering soil moisture and (b) without considering soil moisture.

The impact of changes in combinations of anchor pixels with actual ET values was analyzed. It was observed that there was an offset between ET values derived using anchor pixels selected based on "with" and "without" considering soil moisture as an auxiliary parameter. The absolute difference between the difference of T_s of anchor pixels for both the approaches ($\Delta T_{s,\ \mathrm{abs}} = |(T_{s,\ \mathrm{hot}} - T_{s,\ \mathrm{cold}})_{\mathrm{with\ sm}} - (T_{s,\ \mathrm{hot}} - T_{s,\ \mathrm{cold}})_{\mathrm{without\ sm}}|$) was related to the deviation between ET values. This relationship was exhibited by all the datasets, and for 22 February 2018, maximum deviation between the ET values was observed, which had maximum $\Delta T_{s,\ \mathrm{abs}}$ (2.5 K). The $\Delta T_{s,\ \mathrm{abs}}$ values for various dates were calculated from Table 12.2. The lowest value of $\Delta T_{s,\ \mathrm{abs}}$ (0.25 K) was observed for 11 April 2018, where the deviation between the ET values was minimal. The combined influence of hot and cold pixels was observed rather than their individual effects on the resultant ET. This was justified based on the results obtained for the datasets of 3 February 2017, 22 February 2018, and 11 April 2018. For all these datasets, either cold or hot pixel locations were the same, and they exhibited different ET histograms. The Chi-square (X^2) distance (Yang et al. 2015) was calculated to compare the histograms of actual ET, obtained with and without considering soil moisture for various dates. The X^2-distance was relatively high for ET histograms of 5 January 2018 and 22 February 2018. A similar trend was also exhibited in the case of $\Delta T_{s,\ \mathrm{abs}}$ values calculated for anchor pixels in all the datasets. Plots between X^2 and $\Delta T_{s,\ \mathrm{abs}}$ revealed that there existed a proportionate variation between X^2 and $\Delta T_{s,\ \mathrm{abs}}$ for all the dates. These outputs indicate that a small variation in the anchor pixel locations can significantly affect the final ET calculated. The results obtained in this study showed the utility of soil moisture for a better anchor pixel selection process in SEBAL. The limitation of the methodology adopted in this research was the requirement of in situ soil moisture content measured at the time of satellite data acquisitions for the calibration of WCM.

Conclusions

The prospects of incorporating soil moisture content as an auxiliary parameter for choosing the anchor pixel selection in SEBAL were unveiled in this study. The synergy of Sentinel 1A and Landsat 8 satellite data and ground observations within the framework of WCM was found to be effective in understanding the relative soil moisture variations of the region of interest. The estimation of soil moisture by WCM was refined using a new approach of virtual NDVI, and the proposed method was found to be effective. The influence of soil moisture content was visible in the anchor pixel selection by comparing the cases "with" and "without" considering the soil moisture content. This study considered the soil moisture parameter as a conflict resolving criterion where many candidate pixels were present. It gave a strong theoretical backup to the cold and the hot pixel selection concept where NDVI and T_s alone could not satisfy all the criteria for the selection process. The available energy, calculated at anchor pixel locations, determined by "with" and "without" soil moisture criterion, justified the robustness of the soil moisture-driven approach. The proper selection of anchor pixels is critical, which affects the calculation of actual ET values and hence the cost and efficiency of further water management practices for an agricultural region with diverse crop types. The process of anchor pixel selection demands a less subjective and robust approach to identify the appropriate pixels from a large set of candidate pixels. This study bridges this gap to an extent by considering soil moisture content as an additional parameter.

The present study did not conclusively demonstrate the potential of spatially explicit soil moisture information for ET modeling, and it focused only on integrating soil moisture as an auxiliary parameter for the anchor pixel selection process in SEBAL. Future research in the domain of SEBAL can be carried out by integrating the SAR-derived soil moisture within the main framework of the SEBAL, which require extensive modification of the present model. The future missions in SAR remote sensing together with advances in data assimilation techniques will improve the constraints to an extent by refining the approaches of integration of soil moisture.

Acknowledgments

The authors would like to thank European Space Agency and United States Geological Survey for providing Sentinel 1A data and Landsat 8 satellite images. Thanks to Nagarjuna Innovation Centre, Wargal, India, for providing meteorological data for this study. We also express our gratitude to the reviewers for critically reading the manuscript and suggesting substantial improvements.

Parts of this contribution made use of materials from a previous publication "Mohan, M. M., K. Rajitha, and M. R. R. Varma, 2020, Integration of soil moisture as a supporting parameter for the anchor pixel selection process in SEBAL using Landsat 8 and Sentinel 1A images, International Journal of Remote Sensing 41(3): 1214–1231" with permission by publisher Taylor & Francis Group and the SMT protocols.

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