Water Data to Answer Urgent Water Policy Questions:

Monitoring design, available data, and filling data gaps for determining the effectiveness of agricultural management practices for reducing tributary nutrient loads to Lake Erie

The first in a series of three reports focused on water data needed to address water policy issues. Future reports will focus on shale gas development in the Susquehanna River Basin and an overview of existing water-quality data across the Northeast-Midwest region.

A report published by
The Northeast-Midwest Institute in collaboration with the U.S. Geological Survey

NORTHEAST-MIDWEST INSTITUTE

USGS
science for a changing world

For More Information

For more information about the Northeast-Midwest Institute please see www.nemw.org. Additional information about this report, including an Addendum update, and associated companion reports is available at www.nemw.org.

Citation

Betanzo, E.A., Choquette, A.F., Reckhow, K.H., Hayes, L., Hagen, E.R., Argue, D.M., and Cangelosi, A.A., 2015, Water data to answer urgent water policy questions: Monitoring design, available data and filling data gaps for determining the effectiveness of agricultural management practices for reducing tributary nutrient loads to Lake Erie, Northeast-Midwest Institute Report, 169 p., http://www.nemw.org/.

Funding

This work was made possible by a grant from the U. S. Geological Survey.

ISBN: 978-0-9864448-0-7

Cover photos: USGS (tractor and streamgage photos); NASA (Lake Erie satellite image);
Ryan Hodnett via Wiki Commons (lake background photo)

Project Team

Elin A. Betanzo, Northeast-Midwest Institute, Lead researcher and project manager
Anne F. Choquette, U.S. Geological Survey, Hydrologic data analyst
Kenneth H. Reckhow, Professor Emeritus at Duke University, Statistical analyst
Laura Hayes, U.S. Geological Survey, GIS specialist
Erik R. Hagen, Northeast-Midwest Institute, Water policy researcher
Denise M. Argue, U.S. Geological Survey, Hydrologic data analyst
Allegra A. Cangelosi, Northeast-Midwest Institute, Senior Policy Analyst, Environmental Projects

Technical Advisory Committee

Rajesh Bejankiwarr, International Joint Commission
William Brown, Pennsylvania Department of Environmental Protection
Anne Choquette, U.S. Geological Survey
Gabrielle Ferguson, Ontario Ministry of Agriculture and Food and Ministry of Rural Affairs
R. Peter Richards (retired), Heidelberg University
Dale Robertson, U.S. Geological Survey
Paul Stacey, Great Bay National Estuarine Research Reserve, New Hampshire
Mark Tomer, Agricultural Research Service, United States Department of Agriculture
Elizabeth Toot-Levy, Northeast Ohio Regional Sewer District

Technical Reviewers

Jeffrey Deacon, U.S. Geological Survey
Jeffrey Frey, U.S. Geological Survey
James Gerhart (retired), U.S. Geological Survey
Robert Hirsch, U.S. Geological Survey
Gary Rowe, U.S. Geological Survey
Michael Woodside, U.S. Geological Survey

Blue Ribbon Project Steering Committee

Glenn Benoy, International Joint Commission
Paul A. Biedrzycki, City of Milwaukee Health Department
Suzanne Bricker, National Oceanic and Atmospheric Administration
Allegra Cangelosi, Northeast-Midwest Institute
Julius Ciaccia, Northeast Ohio Regional Sewer District
Thomas Davenport, Environmental Protection Agency, Region 5
Joseph Depinto, Limnotech
Blayne Diacont, Range Resources
Timothy Eder, Great Lakes Commission
Norman Grannemann, U.S. Geological Survey
Carlton Haywood, Interstate Commission on the Potomac River Basin
Jonathan Higgins, The Nature Conservancy
Jennifer Hoffman, Chesapeake Energy
Susy King, New England Interstate Water Pollution Control Commission
James Miller, Organic Valley
Jeffrey Myers, New York Department of Environmental Conservation
Laura Rubin, Huron River Watershed Council
Robert Tudor (retired), Delaware River Basin Commission
Mark Walbridge, United States Department of Agriculture, Agricultural Research Service
Susan Weaver, Pennsylvania Department of Environmental Protection

The Northeast-Midwest Institute President and CEO Foreword

No resource is more vital to the future of the Northeast-Midwest region—its population, industry, food production, and quality of life—than its naturally abundant fresh water. Our capacity to monitor the state of this natural asset and changes to its quality over time should reflect its value to the region. Evaluating the extent to which there is sufficient water monitoring capacity and information is especially critical at this time as large-scale development activities such as shale gas extraction and agriculture dramatically expand freshwater use.

The Northeast-Midwest Institute, a Washington, D.C.-based nonprofit and nonpartisan research and policy organization dedicated to economic vitality, environmental quality, and regional equity for Northeast and Midwest states, has examined the question of whether sufficient water-quality data are available, with the case study presented here focusing on the Lake Erie drainage basin. The economic, human health and environmental benefits of ensuring quality fresh water are of particular importance for the Northeast-Midwest Institute and are major drivers for undertaking this study. Policy makers in particular need a source of objective, scientific, and timely information on water quality as they fulfill their responsibility to ensure the quality of fresh water both today and in the future. In its efforts to meet this critical need, the Northeast-Midwest Institute teamed with the U.S. Geological Survey to investigate the capacity of the Northeast-Midwest region's water monitoring programs to provide monitoring information to support informed policy decisions.

The Northeast-Midwest Institute and the U.S. Geological Survey undertook two case studies to illustrate the types and amounts of data needed to answer urgent water policy questions and to determine how much of those water-quality data are currently available. This report summarizes one case study: our investigation of water-quality data relating to nutrient enrichment in the Lake Erie drainage basin. A companion case-study investigating the availability of water-quality data relating to shale gas development in the Susquehanna River Basin will be presented separately. The results of these case studies will inform a qualitative review of water data availability across the Northeast-Midwest region in an upcoming State of the Region Report.

Policy-making by elected officials usually involves value judgments about acceptable levels of risk, and acceptable ways to assure they are not exceeded over time, for a given issue or decision. Objective information, ideally an outcome of responsible and unbiased research, is critical to assessing risk and the potential of human activities to alter it one way or the other. While this case study does not make value judgments as to acceptable levels of risk, it researches the minimum quantity of water-quality data required to detect a statistically significant change in nutrient concentrations and loads, and analyzes whether those data are currently being generated.

Among the significant findings of this case study are the following:

- Although some of the water data needed to quantify the effectiveness of agricultural management practices in the Lake Erie drainage basin are available, additional water-quality data are needed, particularly for small watersheds. Key steps to generating the needed information include strategically selecting watersheds for monitoring, maximizing management practice impact in monitored watersheds, and collecting ancillary data necessary for water-quality data analysis.

- The most effective data collection and analysis can be achieved through increased coordination between producers, conservation staff, and water monitoring agencies, as well as improved data sharing among monitoring agencies.

- Finally, this report indicates the specific additional data that are needed and asserts the urgency to begin collecting those data.

Ensuring quality fresh water, now and in the future, is a priority of the highest order for the Northeast-Midwest region. The research presented in this report documents the significant need for additional information on water quality to control harmful algal blooms in the Lake Erie drainage basin. Surely collecting and using the needed information to identify effective policy solutions is a first step to achieving this critical environmental and economic goal for the Northeast-Midwest region in the years ahead.

Michael J. Goff, Ph.D.
President and CEO
Northeast-Midwest Institute

Table of Contents

List of Figures

List of Tables

Acronyms

ArcGIS: Arc Geographic Information System

ARS: Agricultural Research Service

BACI: Before-After-Control-Impact

BMP: Best Management Practice

CEAP: Conservation Effects Assessment Project

DIP: Dissolved inorganic phosphorus

DRP: Dissolved Reactive Phosphorus

FRP: Filterable reactive phosphorus

FY: Fiscal year

GAGES: Geospatial Attributes of Gages for Evaluating Streamflow

GLOS: Great Lakes Observing System

GLAHF: Great Lakes Aquatic Habitat Framework

GLEAM: Great Lakes Environmental Assessment and Mapping

GLRI: Great Lakes Restoration Initiative

HAB: Harmful Algal Bloom

HUC: Hydrologic Unit Code

MTA: Metric tonnes per year

NAWQA: National Water Quality Assessment

NEMI: National Environmental Methods Index

NEMW: Northeast Midwest

NEMWI: Northeast Midwest Institute

NERRS: National Estuarine Research Reserve System

NHDPlus: National Hydrography Dataset Plus

NOAA: National Oceanic and Atmospheric Administration

NPDES: National Pollution Discharge Elimination System

NWIS: National Water Information System

ODNR: Ohio Department of Natural Resources

OH EPA: Ohio Environmental Protection Agency

QA/QC: Quality Assurance/Quality Control

RSE: Residual standard error

SOP:	Standard Operating Procedures
SRP:	Soluble Reactive Phosphorus
SPARROW:	SPAtially Referenced Regressions On Watershed attributes
SSURGO:	Soil Survey Geographic Database
STEWARDS:	Sustaining The Earth's Watersheds - Agricultural Research Database System
STORET:	STOrage and RETreival
SWAT:	Soil and Water Assessment Tool
TAC:	Technical Advisory Committee
TDP:	Total dissolved phosphorus
TKN:	Total Kjeldahl nitrogen
TP:	Total Phosphorus
USDA:	U.S. Department of Agriculture
USEPA:	U.S. Environmental Protection Agency
USGS:	U.S. Geological Survey
WQX:	Water Quality Exchange
WRTDS:	Weighted regressions on time, discharge, and season

Glossary

Anoxic: Water that is depleted of dissolved oxygen.

Best Management Practice (management practice): A practice or system of practices designed to prevent or mitigate damage or adverse effects caused by farming, construction, manufacturing, or other anthropogenic activities (U.S. Department of Agriculture, 2014a)

Conservation practice: A method that reduces soil erosion and retains soil moisture (U.S. Department of Agriculture, 2014a)

Concentration: Mass per unit volume. TP and DRP concentrations are typically measured as milligrams per liter and are generally positively correlated with streamflow, which can vary significantly from year to year.

Flow-adjusted concentration: A measure of concentration for which the influence of streamflow and seasonal patterns has been reduced or removed (Helsel and Hirsch, 2002), usually by modeling.

Load (or Flux): Total mass of a constituent delivered to some location in a specific period of time. Annual TP and DRP loads to Lake Erie are typically estimated from concentration and continuous records of streamflow and are typically expressed as metric tons per year.

Hypoxic: Water that has low dissolved oxygen concentrations.

Management Practice: Methods that have been referred to as conservation practices, best management practices, and other methods designed to prevent or mitigate the release of nutrients from nonpoint sources to receiving waters.

Parameter: In this document, a generic term referring to water quality measurements, including field measurements and water-quality constituents analyzed in a laboratory.

Streamgage: An active, continuously-functioning measuring device in the field for which a daily mean streamflow is computed or estimated and quality assured for at least 355 days of a water year (U.S. Geological Survey, 2014a)

Tile drainage: Series of underground tiles that are installed on poorly drained soils to improve soil quality and water infiltration, reduce compaction, improve crop yields, and potentially provide a system to control the amount of surface runoff.

Type I Error: Detecting an effect that is not present (incorrectly rejecting the null hypothesis).

Type II Error: Failing to detect an effect that is present (incorrectly failing to reject the null hypothesis).

Yield: Mass per time per unit area, calculated as stream load divided by contributing drainage area. Used to provide a standardized metric for comparison of nutrient export among watersheds of differing sizes or across a broad region.

Conversion Factors

Inch/Pound to SI

Multiply	By	To obtain
Length		
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
Area		
square mile (mi^2)	259.0	hectare (ha)
square mile (mi^2)	2.590	square kilometer (km^2)
Volume		
cubic foot (ft^3)	0.02832	cubic meter (m^3)
Flow rate		
foot per second (ft/s)	0.3048	meter per second (m/s)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
cubic foot per second per square mile [$(ft^3/s)/mi^2$]	0.01093	cubic meter per second per square kilometer [$(m^3/s)/km^2$]

SI to Inch/Pound

Multiply	By	To obtain
Length		
centimeter (cm)	0.3937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
Area		
square meter (m^2)	0.0002471	acre
hectare (ha)	2.471	acre
square kilometer (km^2)	247.1	acre
square meter (m^2)	10.76	square foot (ft^2)
hectare (ha)	0.003861	square mile (mi^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
Volume		
liter (L)	33.82	ounce, fluid (fl. oz)
liter (L)	0.2642	gallon (gal)
liter (L)	61.02	cubic inch (in^3)
Mass		
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound avoirdupois (lb)
metric ton per day	1.102	ton per day (ton/d)
metric ton per year	1.102	ton per year (ton/yr)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

°F=(1.8×°C)+32

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

°C=(°F-32)/1.8

Vertical coordinate information is referenced to the “North American Vertical Datum of 1988 (NAVD 88)”

Horizontal coordinate information is referenced to the “North American Datum of 1983 (NAD 83)”

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25°C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (µg/L).

1. Executive Summary

Throughout its history, the United States has made major investments in assessing natural resources, such as soils, timber, oil and gas, and water. These investments allow policy makers, the private sector and the American public to make informed decisions about cultivating, harvesting or conserving these resources to maximize their value for public welfare, environmental conservation and the economy. As policy issues evolve, new priorities and challenges arise for natural resource assessment, and new approaches to monitoring are needed. For example, informed conservation and use of the nation's finite fresh water resources in the context of increasingly intensive land development is a priority for today's policy decision-makers. There is a need to evaluate whether today's water monitoring programs are generating the information needed to answer questions surrounding these new policy priorities.

The Northeast-Midwest Institute (NEMWI), in cooperation with the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program, initiated this project to explore the types and amounts of water data needed to address water-quality related policy questions of critical concern to today's policy makers. The collaborating entities identified two urgent water policy questions and conducted case studies in the Northeast-Midwest region to determine the water data needed, water data available, and the best ways to fill the data gaps relative to those questions. This report details the output from one case study and focuses on the Lake Erie drainage basin, a data-rich area expected to be a best-case scenario in terms of water data availability.

1.1 Case Study Question

The policy question that frames this case study evaluating water monitoring capacity is: How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale in the Lake Erie drainage basin? This question is of urgent importance to Great Lakes decision-makers. A harmful algal bloom (HAB) in Toledo, Ohio, contaminated the city's drinking water supply in August 2014, forcing local governments to prohibit home and commercial water use for 3 days. Phosphorus is a primary nutrient that influences HAB growth, and nonpoint-source runoff in the Lake Erie drainage basin contributes 61 percent of the total phosphorus (TP) load to the lake (Ohio Lake Erie Phosphorus Task Force, 2010). Tributaries in the western Lake Erie drainage basin, which is dominated by agricultural land use, contribute the greatest external nutrient loadings to Lake Erie[1]. Any approach to control HABs must include a means of reducing nonpoint-source contributions of phosphorus and other nutrients from agricultural lands.

Understanding the effectiveness of agricultural management practices is also a priority for the Northeast-Midwest region, generally, as evidenced in an informal project survey of Northeast-Midwest Congressional Coalition member offices where many asked whether implementation of agricultural management

[1] The Detroit River and Canadian tributaries were not evaluated in this case study; the watersheds draining to the Detroit River do not meet the large watershed and agricultural land-use criteria established for this case study, and data from Canadian tributaries were not compiled.

practices could reduce nonpoint-source nutrient loads that contribute to HABs. Moreover it is clear that quantifying the effectiveness of agricultural management practices for reducing nutrients from nonpoint sources is an enduring national concern. Nutrient loads are known to cause dead zones in the Gulf of Mexico, Lake Erie, and the Chesapeake Bay (Committee on Environment and Natural Resources, 2010).

Despite this ongoing interest, the question of whether management practices produce desired nutrient reductions remains largely unresolved at the watershed scale, but not for lack of trying. Over the last 25 years, several water-quality and conservation research programs were initiated to quantify relationships between agricultural activities and water quality at the watershed scale, including the U.S. Department of Agriculture (USDA) Water Quality Initiative (Womach J., 2005), USDA Conservation Effects Assessment Project (CEAP) (Mausbach and Dedrick, 2004), and most recently, in 2012, the USDA Natural Resources Conservation Service National Water Quality Initiative (Larsen, 2014). In summary, this case study topic represents an urgent environmental and public health priority for the Great Lakes region, a national agricultural policy priority, and an issue for which a comprehensive and analytical look at water-quality data needs and data availability has the potential to create the data needed to inform new policy solutions.

1.2 Case Study Approach

The case-study approach consisted of three main tasks:

1. Describe the types and amounts of water data needed to answer the policy question,
2. Assess the extent to which those data are available and usable, and
3. Identify the additional data that would be needed to more effectively answer the policy question and estimate the level of effort to collect the additional data.

For the first task, the study team and technical advisory committee (TAC) identified the data types that were most critical for measuring the effectiveness of management practices at the watershed scale and analyzed available data to determine the quantity of data needed to detect statistically significant changes in water quality. The study team collected data available from agencies and organizations that monitor tributary water quality and USGS streamgages within the Lake Erie drainage basin, evaluated those data against the data needed, and reviewed the usability of those data to inform the policy question. Once data availability and usability were determined, the study team assessed data gaps and made recommendations for meeting data needs to inform the policy question. This report identifies the minimum amount of water data needed to detect statistically significant change in water quality due to agricultural management practice implementation within a 10-year time frame, given certain study design assumptions; identifies available water data; and makes recommendations for filling the information gaps.

1.3 Case Study Findings

The case study findings relative to water data needed, water data available and usable, and approaches to filling the data gaps are summarized below.

1.3.1 Water data needed to answer the policy question

- **Water data needs for addressing the case-study policy question are highly dependent on study design.**

It is not possible to identify water data needs for answering the case-study policy question without an initial discussion of an appropriate study design (Figure ES-1). For example, the selection of appropriate monitoring sites for answering the case-study policy question is critical. Monitoring sites must be located in watersheds dominated by agricultural land use, and where management practices can, and will, be widely implemented in optimal places for reducing nutrient loss. Monitoring sites in these types of watersheds allow for the detection of water-quality changes that these practices can generate. Further, tributary water-quality and streamflow data at these monitoring sites must be available to evaluate trends in concentration and load in these watersheds over time. Finally, data on management practice implementation and other changes in land use and nutrient sources throughout the watershed must be available to correlate water-quality change with alterations on the land. Without this information, the relationship between management practices and water quality cannot be evaluated, even if management practices are delivering detectable reductions in nutrient loads.

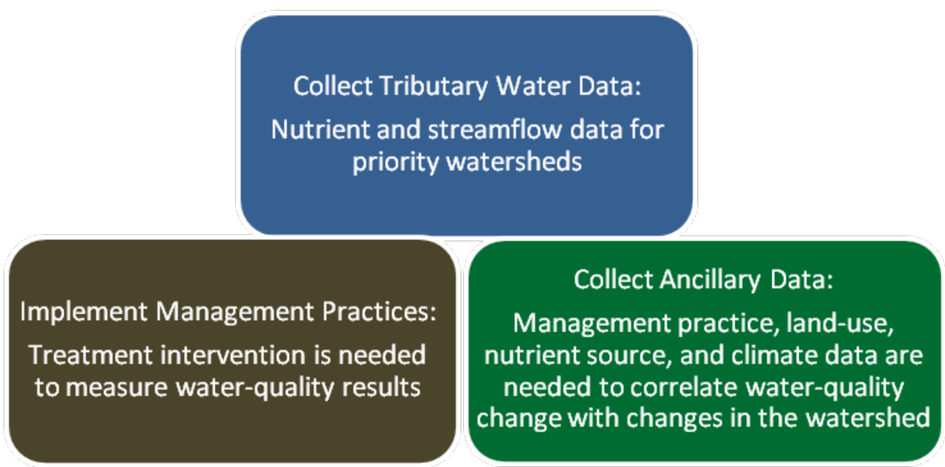


Figure ES-1. Study design needed to answer "How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?"

- **Water data are needed from two watershed scales to answer the case-study policy question.**

The time and effort required to establish the necessary study conditions described in Figure ES-1 will be most readily achievable in small (less than 50 square miles) agricultural watersheds. At the same time, large (more than 1,000 square miles) agricultural watersheds that drain directly to Lake Erie are largely responsible for the nutrient loads that lead to HABs; complex nutrient fate and transport mechanisms in these large tributaries preclude scaling results from small watersheds. Therefore, water data from large watersheds, specifically the Maumee River, Sandusky River, and the River Raisin in the Lake Erie drainage basin, are also critical to addressing the case-study policy question.

- **The sampling frequency and duration of monitoring must meet minimum requirements to adequately characterize and detect changes in nutrient concentrations and loads to be used to answer the case-study policy question.**

Assuming the necessary management practice intensity and ancillary data (Figure ES-1) are available, tributary water-quality and streamflow data requirements can be characterized as shown in Table ES-1. A minimum of six monitoring sites in six small watersheds are needed to provide spatial representation of the Lake Erie drainage basin, and one monitoring site is needed in each of the three large watersheds that meet the identified criteria.

The most critical parameters for assessing the effectiveness of management practices in the Lake Erie drainage basin are total phosphorus (TP), dissolved reactive phosphorus (DRP), and streamflow. The sampling frequency at each monitoring site must capture the full range of hydrological conditions within the watershed annually and over time. Several sampling frequency options are presented in Table ES-1. The increase in sampling frequency across these options reflects improved ability to characterize the relationship between streamflow and concentration.

Monitoring duration must be sufficient to detect the effects of new changes to the landscape and distinguish them from historical land management practices, climate effects, and other factors. If appropriate agricultural management practices are implemented and consistently maintained throughout a watershed such that annual TP loads are reduced by 40 percent, a load reduction goal recommended by the International Joint Commission (2014), a monthly sampling program would be able to detect that change for both small and large watersheds with statistical significance within 10 years. However, current or moderately increased rates of management practice implementation are expected to generate reductions in TP loads that are closer to 10 percent, particularly in large watersheds, according to available models (Bosch et al., 2013; Lund et al., 2011). This case study found that more than 40 years of monthly TP data would be needed to detect a 10-percent change at a given monitoring site with statistical significance because the natural variation that occurs in streamflow and water quality from year to year obscures this small magnitude of change.

Table ES-1. Summary of water data needed to detect water-quality change resulting from agricultural management practices in the Lake Erie drainage basin. Analysis supporting these findings is presented in Chapter 5.

[Abbreviations: %, percent; TP, Total Phosphorus; DRP, Dissolved Reactive Phosphorus]

	Small Watersheds	Large Watersheds
Monitoring sites located in watersheds with these characteristics	• Less than or equal to 50 square miles, • Greater than or equal to 40% of row-crop coverage, • High phosphorus yield and high soil vulnerability	• Greater than or equal to 1,000 square miles • Drains directly to Lake Erie • Greater than or equal to 40% of row-crop coverage
Number of watersheds and monitoring sites	Minimum of 6 active monitoring sites (1 per watershed) representing a variety of watershed characteristics and spatially distributed across the agricultural areas of the basin	3 active monitoring sites (1 per watershed)
Monitoring parameters	• TP, DRP, streamgage • Suite of parameters from Table 3	• TP, DRP, streamgage • Suite of parameters from Table 3
Sampling frequency options	• Monthly plus supplemental sampling (24/year) • Two-year intensive monitoring followed by adaptive management to modify sampling plan (100 per year then 24 per year) • Daily plus storm sampling (approx. 500 per year), or • Monthly plus continuous monitoring (turbidity and/or dissolved phosphorus)	• Monthly plus supplemental sampling (24/year) • Two-year intensive monitoring followed by adaptive management to modify sampling plan (100 per year then 24 per year) • Daily plus storm sampling (approx. 500 per year), or • Monthly plus continuous monitoring (turbidity and/or dissolved phosphorus)
Minimum duration of monitoring to detect change	>8 years[1] (assumes 40% reduction in TP and DRP over 20 years)[2]	>20 years (assumes 20% reduction in TP and DRP over 20 years)[2]

[1]If less than 10, review monitored years to verify a range of climatic conditions
[2]See Table 16 and Table 17

1.3.2 Availability and usability of existing water data to answer the policy question

This investigation found more than 300,000 TP and DRP records collected at nearly 2,000 monitoring sites over the last 70 years in the Lake Erie drainage basin (Figure ES-2). However, as indicated in Figure ES-3 only six of those monitoring sites use a sampling plan that meets the specifications summarized in Table ES-1 for addressing the case-study policy question. This study found the following specific results regarding currently available data:

- **The small watershed data needed for answering the case-study policy question are not currently being collected.**

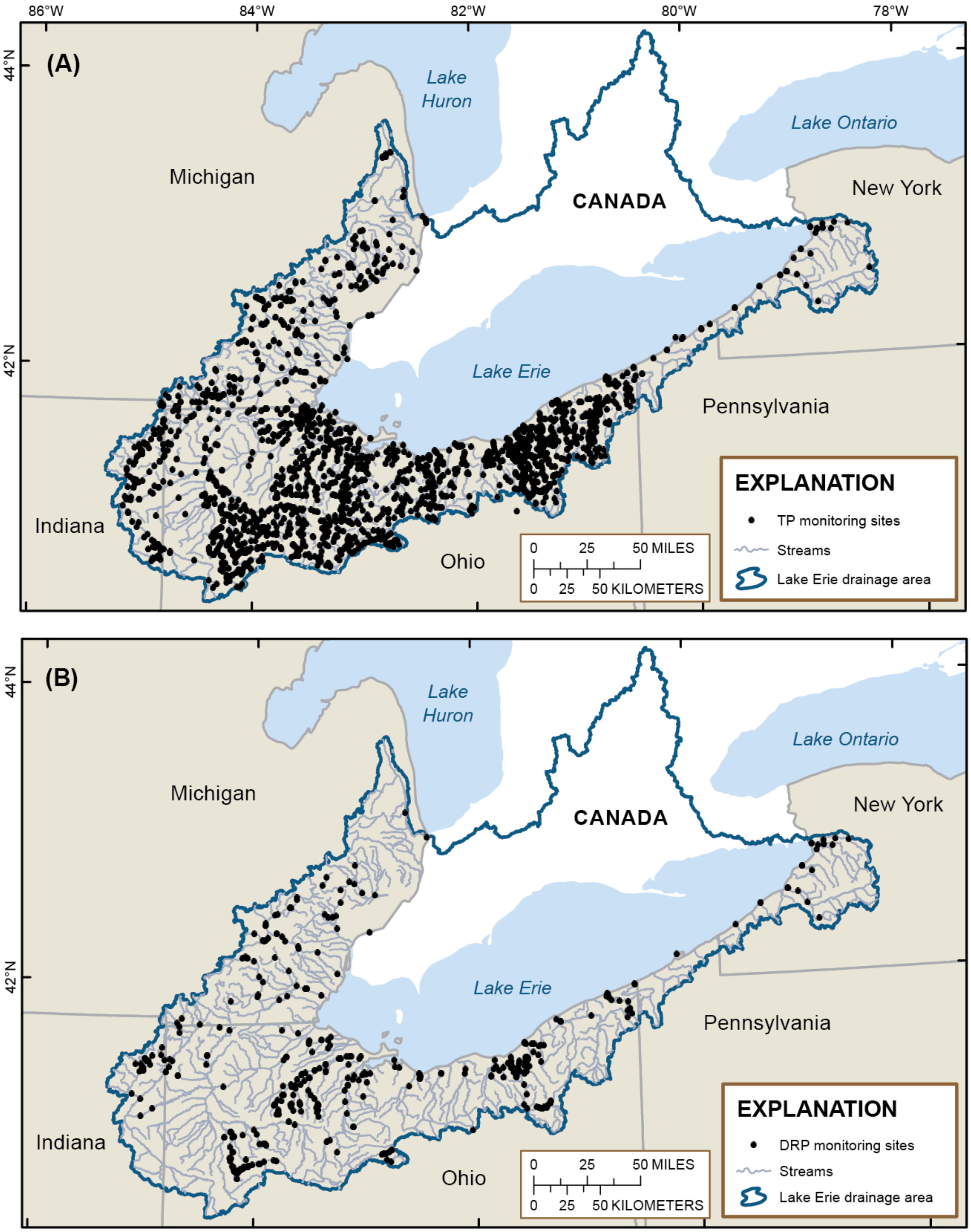
The water data collected at only two existing small watershed monitoring sites meet the requirements described in Table ES-1. The two sites (site E, unnamed Tributary to Lost Creek, and site F, Rock Creek, in Figure ES-3) are monitored by Heidelberg University at USGS streamgages. Water data collected at two other sites maintained by the USDA and the National Oceanic and Atmospheric Administration (NOAA) National Estuarine Research Reserve System (NERRS) could, with increased sampling frequency, provide the needed data for two additional monitoring sites. Nonetheless, a minimum of two entirely new small watershed monitoring sites also would be needed to meet the data needs for answering the case-study policy question.

- **The needed water-quality data are being collected for the three large watersheds; data collection at these sites should continue uninterrupted into the future to be useful in answering the policy question.**

Water-quality and streamflow data are being collected by Heidelberg University and the USGS for the three large agricultural watersheds that drain directly to Lake Erie, at monitoring sites on the Raisin, Maumee, and Sandusky Rivers where they discharge into the western basin of Lake Erie. These monitoring sites (A-D in Figure ES-3) measure the needed parameters with daily or continuous sampling frequency, and over 30 years of data records are available for these monitoring sites. Data collection should continue at these sites to measure changes in nutrient concentrations and loadings to Lake Erie over time as management practices continue to be implemented throughout the watersheds.

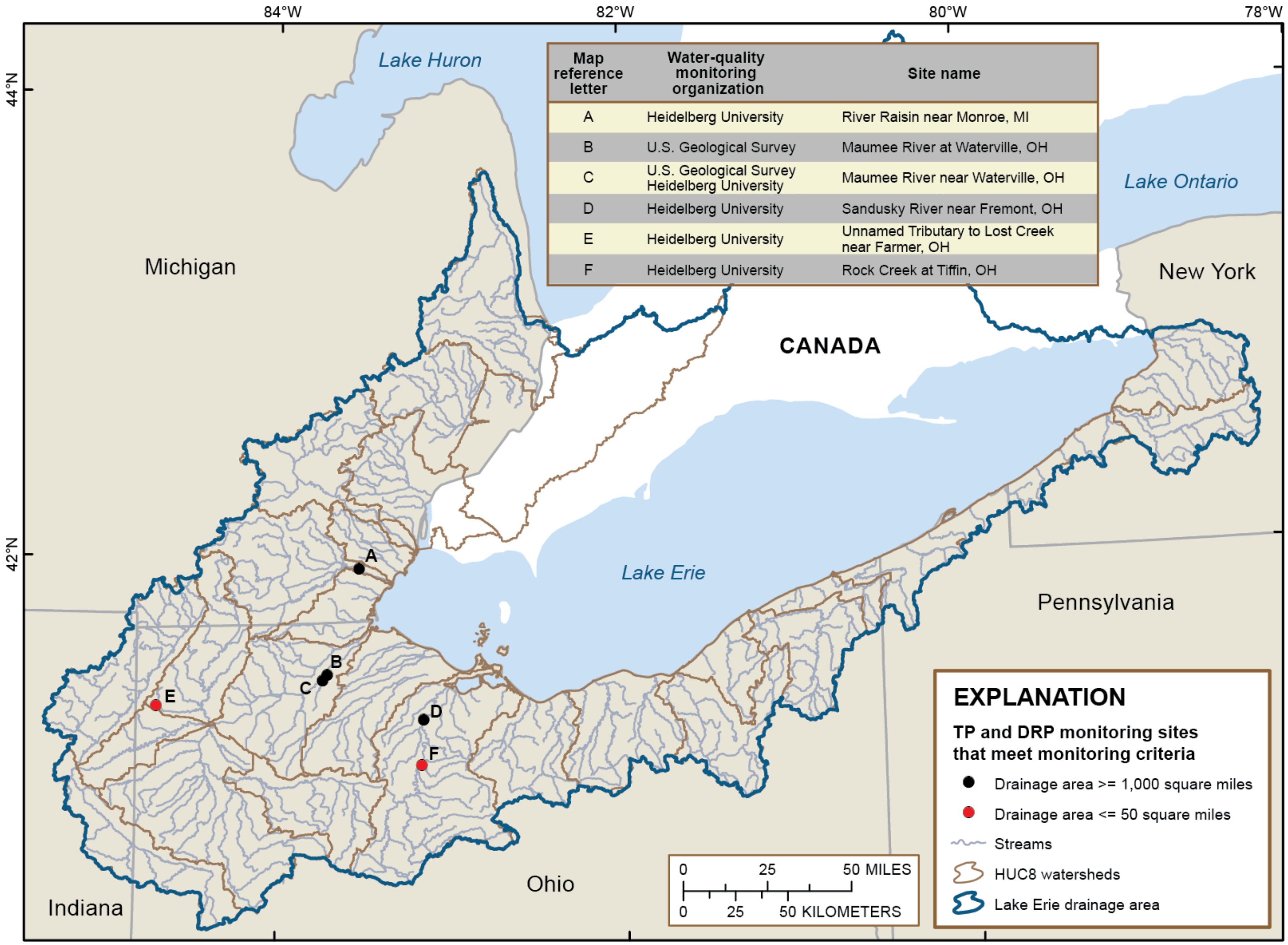
- **Current water data usability for answering the policy question is limited by insufficient or inconsistent data documentation and sharing.**

Water-quality monitoring programs are usually designed to meet a stated objective or follow a historical precedent. Data collected for one monitoring objective may not be directly applicable to another objective, due to the location of monitoring sites, frequency of monitoring, parameters measured, and analytical methods used. The water-quality records identified through this case study were generated by 17 organizations that collect nutrient-related data in the Lake Erie drainage basin. Insufficient and inconsistent documentation of available data limited the utility of these existing data sets. Substantial project time and effort over the course of this multi-year project were required to locate, obtain, and consistently format data.



State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Streams from U.S. Geological Survey, 2012, 1:1,000,000-scale digital data
Albers projection, NAD 1983

Figure ES-2. Monitoring sites in the Lake Erie drainage basin with one or more water-quality records for (A) total phosphorus (TP) or (B) dissolved reactive phosphorus (DRP).



Map reference letter	Water-quality monitoring organization	Site name
A	Heidelberg University	River Raisin near Monroe, MI
B	U.S. Geological Survey	Maumee River at Waterville, OH
C	U.S. Geological Survey Heidelberg University	Maumee River near Waterville, OH
D	Heidelberg University	Sandusky River near Fremont, OH
E	Heidelberg University	Unnamed Tributary to Lost Creek near Farmer, OH
F	Heidelberg University	Rock Creek at Tiffin, OH

State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Streams from U.S. Geological Survey, 2012, 1:1,000,000-scale digital data
Albers projection, NAD 1983

HUC8 watersheds from U.S. Department of Agriculture, February 2012, ftp://ftp.ftw.nrcs.usda.gov/wbd/

Figure ES-3. Active water-monitoring sites in the Lake Erie drainage basin in priority watersheds that collect water data needed for detecting changes in water quality resulting from agricultural management practices. These sites collect TP, DRP, and streamflow daily or have continuous monitors, and have 5 to over 30 years of data records. The U.S. Geological Survey operates the streamgages at each of these monitoring sites. [**Abbreviations**: TP, total phosphorus; DRP, dissolved reactive phosphorus; HUC, hydrologic unit code]

Data sharing and data accessibility were also limiting factors in data availability in this case study. It is possible that despite the work completed for this case study, additional relevant data that are not being shared or are not available in electronic format may exist. The Water Quality Portal (National Water Quality Monitoring Council, 2014a), a cooperative service that provides publicly available water-quality data from federal databases, including data collected by more than 400 state, federal, tribal, and local organizations, was established to facilitate water data sharing. Yet data collected at only 26 percent of the monitoring sites identified through this case study in the Lake Erie drainage basin were available through the Water Quality Portal, and only 8 percent of the water-quality data records are available through the Portal.

Finally, for water data to be useful for addressing the policy question, they must be compatible in terms of sampling plans and protocols, analysis, and interpretation. Several agencies and organizations collect small watershed data that meet or nearly meet the data needs identified in section 1.3.1, including USGS, Heidelberg University, USDA, NOAA NERRS, and additional agencies described in the addendum to this report (Betanzo et al., 2015). However, these agencies all use different sampling plans that limit the ability to compare trends in concentration and load over time at these monitoring sites.

1.3.3 Approaches for filling data gaps to answer the policy question

This section presents the study findings regarding approaches for filling the data gaps to address the case-study policy question in the Lake Erie drainage basin.

- **Add at least two new small watershed monitoring sites in watersheds with priority characteristics.**

As noted in Table ES-1, six monitoring sites for each of six small watersheds are needed to address the policy question. Given data available in the Lake Erie drainage basin, two additional monitoring sites, one per each of two small watersheds, will be needed. Table ES-2 presents a strategy for identifying and prioritizing candidate small watersheds for additional monitoring to answer the case-study policy question. This report identifies several small watersheds with both high phosphorus yield and high vulnerability to soil loss and no current monitoring sites. At least one of the two new monitoring sites should be located in this area. The second site should be a watershed with high phosphorus yield or high vulnerability to soil loss but also provide spatial representation of the drainage basin. New monitoring sites may require new streamgages in addition to new water-quality data. New water monitoring sites and management practice incentive programs are already under development in the Lake Erie drainage basin and are described in an addendum to this report (Betanzo et al., 2015). The recommendations of this report, site selection process (Table ES-2), and data needed (Table ES-1) should be considered and incorporated as plans for new small watershed monitoring sites are finalized if the new sites are to be instrumental in answering the case-study policy question.

- **Increase sampling frequency at two existing small watershed monitoring sites.**

Increased and consistent sampling frequency of both TP and DRP at two existing monitoring sites maintained by USDA and NOAA NERRS would qualify these sites to become a part of the set of six monitoring sites needed to address the case-study policy question. Specifically, monthly year-round data collection, in addition to the current sampling frequency, would be necessary to meet the monitoring needs identified in Table ES-1.

Table ES-2. Process for identifying the most effective small watershed monitoring sites.

Process for identifying new small watershed monitoring sites
1. Identify candidate watersheds: • Watersheds with priority characteristics (high phosphorus yield and/or high vulnerability to soil loss) • Watersheds with existing streamgages and/or water-quality data.
2. Examine location of candidate watersheds relative to other monitoring sites that might allow for nested monitoring designs, such as edge-of-field, mid-size, and large watershed monitoring sites.
3. Examine candidate watersheds on a case-by-case basis for local information: • Representation of tile drainage in the watershed, • Untreated areas with potential for high implementation rates of new management practices • Willingness of agricultural community to implement and maintain new management practices throughout candidate watershed, and • Willingness of agricultural community to share management practice and land-management data with monitoring agency.
4. There may be situations in which monitoring sites in watersheds without priority characteristics are the most feasible study locations.

- **Maintain water-quality and streamflow monitoring at the two small watershed sites monitored by Heidelberg University and the USGS.**

The remaining two small watershed monitoring sites needed to compose the set of six sites are currently in place (sites E and F, Figure ES-3). However, monitoring would need to continue unchanged over time at these sites as new agricultural management practices are implemented within these watersheds.

- **Maintain data collection and analysis at all small watershed monitoring sites for a minimum of 10 years during implementation of new management practices.**

Water-quality and streamflow data should be collected in the six small watersheds for at least 10 years after new practices are implemented. New monitoring and new management practice implementation should begin as soon as possible to minimize the time to detect water-quality change and produce policy-relevant information regarding the case-study policy question. The new data should be evaluated and loads calculated annually so the sampling plans can be adjusted as necessary to adapt to an evolving understanding of management practice effectiveness and water quality.

- **Maintain monitoring at large watershed monitoring sites.**

Data collection at the large watershed monitoring sites (A-D in Figure ES-3) should continue to capture changes in TP and DRP concentrations and loadings to Lake Erie. In addition to supporting evaluation of agricultural management practices, monitoring these large watersheds provides critical information to estimate the total nutrient loads to the western basin of Lake Erie; to measure long-term water-quality change that may result from agriculture, urban development, or climate change; and to support additional river, lake, and ecosystem research and resource management applications.

Filling the data gaps to answer the case-study policy question:

- A minimum of two additional small watershed monitoring sites are needed. Effective monitoring sites should be identified using the process described in Table ES-2. New small watershed monitoring sites should collect the water data identified in Table ES-1.
- Increased sampling frequency at the USDA and NERRS small watershed monitoring sites to include monthly year-round data collection, in addition to the current sampling frequency, is needed for these monitoring sites to fill the need for two small watershed monitoring sites. Both TP and DRP should be sampled at the same frequency at these sites.
- Continued water-quality and streamflow monitoring are needed at the two small watershed sites monitored by Heidelberg University and the USGS who collect the needed water data.
- All small watershed monitoring sites need a minimum of 10 years of monitoring during implementation of new management practices and sharing of management practice implementation data.
- Continued long-term water-quality and streamflow monitoring are needed at the Raisin, Maumee, and Sandusky River sites monitored by Heidelberg University and the USGS.

- **Improve water data coordination and sharing across monitoring agencies and organizations in the Lake Erie drainage basin.**

For water data to be used to answer the case-study policy question, monitoring agencies and organizations should coordinate sampling plans among new and existing monitoring sites so data collection, analysis, and interpretation can be compatible and comparable. Data coordination across agencies can be achieved through a coordinating entity that facilitates collaboration on sampling plans, data sharing, and data analysis in the Lake Erie drainage basin. Improved data documentation and data sharing will facilitate the use of water data for answering the case-study policy question. Tools such as the Water Quality Exchange (WQX) (U.S. Environmental Protection Agency, 2015) and the Water Quality Portal provide the infrastructure for organizations to format and share their data, but greater participation is needed. Consistent, thorough data documentation and wider availability of data sources through services such as the Water Quality Portal will increase the value of water-quality data from all monitoring agencies and reduce the amount of time needed to access and prepare data for new applications. A continued commitment to water-quality data-sharing systems is essential for maximizing use of existing water-quality data.

Improve water data usability:

- Establish a coordinating entity for ensuring compatible data collection, sharing, and analysis across the Lake Erie drainage basin.
- Adopt common data-management standards, data-entry protocols, and consistent naming and coding conventions across monitoring agencies.
- Additional monitoring agencies should submit data annually to the U.S. Environmental Protection Agency (USEPA) Storage and Retrieval (STORET) Data Warehouse and additional partners should participate in the Water Quality Portal.

- **Maximize management practice impact in monitored watersheds.**

As noted in section 1.3.1, appropriate agricultural management practices must reduce TP loads in a watershed by 40 percent for a monthly sampling program to detect that change with statistical significance within 10 years. To achieve this goal, appropriate management practices should be strategically and extensively installed in areas most likely to result in nutrient reductions. Due to complexities of nutrient transport, agricultural specialists are in the best position to identify the most effective agricultural management practices for specific applications. Substantial treatment intervention will be needed to produce a 40-percent reduction in TP load. Generating this coverage in large watersheds almost certainly would require policy interventions, such as incentive programs. Smaller watersheds, though more practical to work with due to their size and the smaller number of producers, may also require incentives within specified watersheds.

- **Collect consistent, detailed data on implementation of agricultural management practices and other changes to the land and other nutrient sources within monitored watersheds.**

As noted earlier, in addition to water-quality data, consistent, detailed documentation of changes on the land and other nutrient sources within a watershed are needed to interpret water-quality data to answer the case-study policy question. Agricultural management practice implementation data are generally not available due to data sharing restrictions and lack of documentation at the level of detail needed for water-quality analysis (Jackson-Smith et al., 2010). Moreover, Section 1619 of the Farm Bill[2] restricts access to conservation practice data that have been provided to the USDA; water-quality researchers must depend on farmers' willingness to share their land management data. Protected data collection and data sharing systems for these types of ancillary data are needed to efficiently collect, store, and share these data at the level of detail needed for water-quality data analysis. Annual land management data are needed to

[2] Section 1619 7 U.S.C. § 8791

correlate water-quality change with annual changes on the land. The detailed ancillary data needed to interpret the water-quality data are either unavailable or difficult to obtain.

1.4 Conclusion

Additional water data are needed at both small and large watershed scales to answer the case-study policy question in the Lake Erie drainage basin. The recommendations in this report present the additional water data that will allow the question to be answered in a policy-relevant time frame. Key steps to generating the needed information include strategically selecting watersheds for monitoring, maximizing management practice impact in monitored watersheds, and collecting ancillary data necessary for water-quality data analysis. However, with cooperation and coordination of producers, local conservation staff, and water monitoring agencies, data collection and analysis can answer this long-standing policy question of critical importance to the Northeast-Midwest region. The sooner the region gets started, the better.

Summary of information needs to answer "How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?"

Collect tributary water data

- Increase small watershed monitoring capacity: Additional monitoring sites, additional sampling frequency at existing sites, and continued monitoring at selected small watershed monitoring sites are needed.
- Continue to invest in large watershed monitoring: Continued long-term water-quality and streamflow monitoring are needed at monitoring sites on the Raisin, Maumee, and Sandusky Rivers where they drain to Lake Erie.
- Improve usability of new and existing water data: Establish an entity for coordinating water monitoring and management practice implementation; encourage use of data-management standards and data-entry protocols, and increase participation in data sharing programs.

Implement agricultural management practices

- Maximize management practice impact in monitored watersheds to reduce time to detect changes in water quality.

Collect ancillary data

- Collect detailed management practice implementation and other ancillary data in both large and small monitored watersheds and make available for water-quality data analysis.

2. Case-Study Introduction and Background

2.1 Purpose of the case study

Throughout its history, the United States has made major investments in assessing natural resources, such as soils, timber, oil and gas, and water, as they support the health, education, employment and economic status of the nation. These investments allow policy makers to make informed decisions about cultivating, harvesting or conserving these resources to maximize their value for the nation's public and environmental health, and economy. As policy issues evolve, new priorities and challenges arise for natural resource assessment, and new approaches to monitoring are needed. For example, informed conservation and use of the nation's finite fresh water resources in the context of increasingly intensive land development is a priority for today's policy decision-makers. There is a need to evaluate whether today's water monitoring programs are generating the information needed to answer questions surrounding these new policy priorities.

The Northeast-Midwest Institute (NEMWI), in cooperation with the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program, initiated this project to explore the types and amounts of water data needed to address water quality related policy questions of critical concern to today's policy makers. The collaborating entities identified two urgent water policy questions and conducted two case studies in the Northeast-Midwest region to determine the water data needed, water data available, and the best ways to fill the data gaps relative to those questions. This report details the output from one case study and focuses on the Lake Erie drainage basin, a data-rich area expected to be a best-case scenario in terms of water data availability.

The policy question that framed this case study is: ***How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?*** Other water-quality data gaps that have been identified in Lake Erie and the Lake Erie drainage basin, but are not addressed in this report, include the Detroit River, open-lake, and near-shore monitoring (International Joint Commission, 2014; Ohio Lake Erie Phosphorus Task Force, 2013).

The question of whether water-quality data can be applied to issues beyond the purpose for which they were originally collected was considered for both technical and practical purposes. Water-quality monitoring programs are usually designed to meet a stated objective or follow a historical precedent. Data collected for one monitoring objective may not be directly applicable to another objective, due to the location of monitoring sites, frequency of monitoring, parameters measured, and methods used to meet that objective. The technical requirements of a new policy question must be considered when determining if water-quality data can be used for a purpose other than their original objective. In addition, this case study considers the practical considerations of whether water-quality data collected by different monitoring agencies in different jurisdictions can be taken together to support regional decision-making.

2.2 Study approach and organization of report

Nutrient enrichment was selected by the project team as the case-study subject due to the importance of the issue to policy makers and the NEMW region. NEMWI interviewed Congressional and Senate staff and decision-makers participating in the Blue Ribbon Project Steering Committee to identify their most urgent water-quality questions related to nutrient enrichment to help design a relevant case study. Twelve Congressional offices and 18 Steering Committee members, including representatives from the federal government, states, cities, industry, and environmental groups, participated in these discussions and their responses were distilled into priority policy questions.

With respect to nutrient enrichment, questions around the effectiveness of conservation and management practices for reducing nutrient loadings from nonpoint sources were raised most frequently. The term 'management practice' is used in this report to encompass methods that have been referred to as (1) conservation practices, methods that reduce soil erosion and retain soil moisture (U.S. Department of Agriculture, 2014a); (2) Best Management Practices (BMPs), a practice or system of practices designed to prevent or mitigate damage or adverse effects caused by farming, construction, manufacturing, or other anthropogenic activities (U.S. Department of Agriculture, 2014a); and (3) other methods for reducing nutrient loadings from nonpoint sources to receiving waters. The term 'agricultural management practices' is used to designate management practices designed for use on agricultural lands.

Decision-makers wanted more information about the effectiveness of management practices to better characterize environmental impacts of nutrient runoff, identify strategies for reducing algal blooms and related ecosystem impacts, and assist design of cost-effective incentives to balance agricultural and environmental interests. The following case-study policy question was selected by the Blue Ribbon Project Steering Committee as the foundation of this nutrient-enrichment case study:

> **"How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?"**

This question makes a connection between a Congressional, regional-level policy issue—whether nutrient-management practices are effective tools for reducing nutrient loads from nonpoint sources for the purpose of controlling harmful algal blooms (HABs)—and the need for water data to quantify the nutrient-load reductions realized at the watershed scale as a result of using these practices. The Lake Erie drainage basin was selected as the geographic location for applying the nutrient case-study policy question to determine the availability of water-quality data. The algal bloom in Toledo, Ohio, that contaminated the city's drinking water supply in August 2014 speaks to the ongoing relevance of this topic to the NEMW region. Nutrient loads are known to cause dead zones in the Gulf of Mexico, Lake Erie, and the Chesapeake Bay (Committee on Environment and Natural Resources, 2010).

Despite this ongoing interest, the question of whether management practices produce desired nutrient reductions remains largely unresolved at the watershed scale, but not for lack of trying. The Water Quality Initiative was developed in 1990 by the U.S. Department of Agriculture (USDA) to determine relationships between agricultural activities and water quality (Womach J., 2005). The Conservation Effects Assessment Project (CEAP) was initiated in 2002 with the goal of establishing the scientific understanding of the effects of conservation practices at the watershed scale (Mausbach and Dedrick, 2004). The CEAP program analysis yielded a variety of lessons learned for improving water-quality studies of conservation practices (Tomer at Locke, 2011). Most recently, the National Water Quality Initiative was established through the USDA Natural Resources Conservation Service in 2012 to improve water quality in small watersheds impaired by agricultural use, with associated monitoring to assess the water quality impacts of agricultural conservation practices (Larsen, 2014). In summary, this case-study topic represents an urgent environmental and public health priority for the Great Lakes region, a national agricultural policy priority, and an issue for which a comprehensive and analytical examination of water-quality data needs and data availability has the potential to create the data needed to inform new policy solutions.

The case-study approach consisted of three main tasks:

1. Describe the types and amounts of water-quality data needed to answer the policy question,
2. Assess the extent to which those data are available and usable, and
3. Identify the additional data that would be needed to more effectively answer the policy question and estimate the level of effort to collect the additional data.

As described throughout this report, there is a wide range of data needs to answer the case-study policy question. Data characterizing nutrient status, ecosystem response, and details of management practice implementation are necessary to analyze the effectiveness of these practices. This report focuses on quantifying and determining the availability of stream-nutrient water-quality data in the Lake Erie drainage basin. Although multiple nutrient sources from two countries contribute to Lake Erie, the data compilation used in this study focuses on United States water-quality data in tributaries (see Chapter 4). Canadian, in-lake, and management practice implementation data were not part of the data set used for this case study. These data are important to the case-study policy question, but they are not the focus of this study. This case study examines the monitoring needed to evaluate suites of agricultural management practices implemented throughout watersheds, rather than evaluating single practices.

To identify the water data available to answer the case-study policy question, the USGS assembled a broad, multi-agency compilation of water data called the National Data Aggregation with special focus on compiling a nutrient data set for the Lake Erie drainage basin (Chapter 4). This nutrient data set served as the primary basis for assessing existing water-quality monitoring in the Lake Erie drainage basin.

A conceptual model (Figure 1) was developed to identify the types of data needed to answer the case-study policy question. A hypothesis was proposed at two different scales to guide development of the monitoring design needed to answer the case-study policy question, and a power analysis was completed to illustrate the quantity of data needed to detect statistically significant changes in concentration and load for the two most critical water-quality parameters (Chapter 5).

Monitoring sites from the nutrient data set were mapped and analyzed to identify the number of sites that meet the sampling requirements (Chapter 6). Data gaps were identified, monitoring recommendations were made for filling the gaps, and a range of costs for new monitoring was estimated (Chapter 7).

2.3 Background

The Lake Erie drainage basin was selected for this case study for several reasons. Out of the regions considered for this project in the NEMW region, the Lake Erie drainage basin:

- Encompasses multiple states and the receiving water body is located within the region, making it a good test for a regional multi-agency study.
- Nutrients in the basin come from a mix of agricultural and urban sources, making the basin representative of nutrient issues being faced across the NEMW region.
- There are several agencies and universities that have long histories of monitoring water quality in the basin, making for a rich data set for evaluation.
- A recent focus on Lake Erie due to HABs allows this study to build on other studies evaluating the science of Lake Erie.

The Lake Erie drainage basin (Figure 2) encompasses parts of Canada, but Canadian water-quality data were not available for analysis through this study.

2.3.1 Description of Lake Erie and the drainage basin

The Lake Erie drainage basin includes parts of the States of Indiana, Michigan, New York, Ohio, Pennsylvania, and the province of Ontario, Canada (Figure 2). The total surface area of the Lake is about 9,900 square miles, and the land basin covers about 22,700 square miles (International Joint Commission, 2014). The land use is dominated by row crops, but there are also intensively industrialized and highly urbanized areas (Fry et al., 2011). About one third of the total population of the Great Lakes Basin resides within the Lake Erie drainage basin, including 10 million U.S. and 1.6 million Canadian citizens (Lake Erie Lakewide Management Plan Work Group, 2011). Lake Erie provides important natural, economic, and recreational value, and supplies drinking water to 11 million people. Agriculture, industry, and urbanization are major stresses on the lake water quality. Compared to the other Great Lakes, Lake Erie receives the most effluent from sewage-treatment plants and the greatest sediment loads from geologic, agricultural, and urban sources (Lake Erie Lakewide Management Plan Work Group, 2011).

Lake Erie is particularly susceptible to these stresses because it is the smallest of the Great Lakes by volume and also the shallowest. The lake warms quickly in the spring and summer and cools quickly in the fall. The shallowness of the basin and the warmer temperatures make it the most biologically productive of the Great Lakes (Lake Erie Lakewide Management Plan Work Group, 2011). Lake Erie receives the most phosphorus of all of the Great Lakes (from U.S. source areas), including the highest loads of agricultural fertilizers (Robertson and Saad, 2011; USDA, 2011). These characteristics also make the lake particularly vulnerable to algal blooms. Additional information on the ecology and economy of the Lake Erie drainage basin and the background of algal blooms and phosphorus loads to the lake can be found in reports by the Lake Erie Lakewide Management Plan Work Group (2011), the International Joint Commission (2014), the Ohio Lake Erie Phosphorus Task Force (2010 and 2013), and by Koslow et al. (2013).

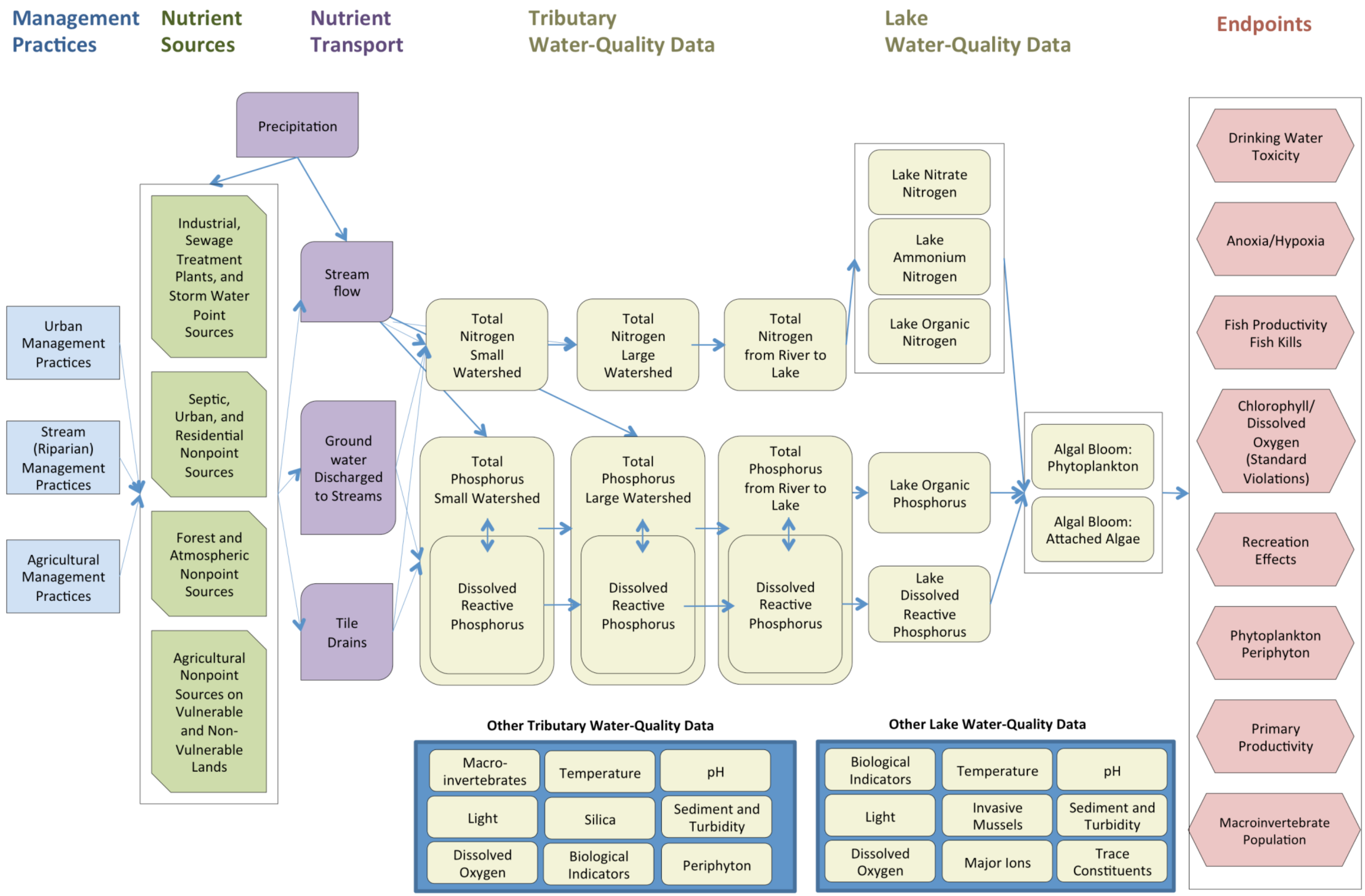


Figure 1. Conceptual model that identifies water-quality and ancillary data needed to determine the effectiveness of management practices at reducing nutrients from nonpoint sources at the watershed scale.

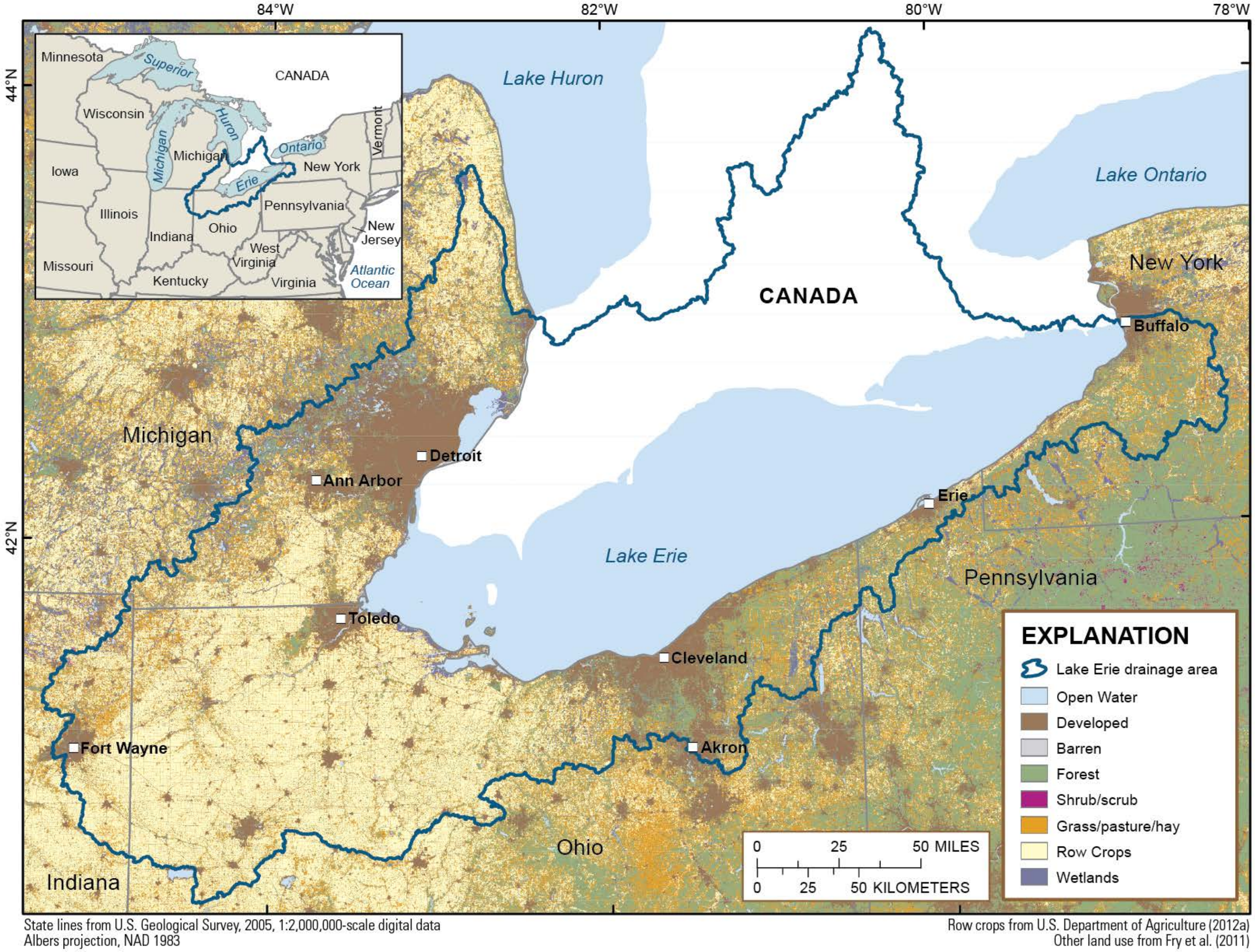


Figure 2. Location and land cover within the Lake Erie drainage basin study area.

2.3.2 Nutrient enrichment in Lake Erie

Elevated nutrient loads from the western Lake Erie drainage basin have resulted in HABs in Lake Erie (Figure 3; Reutter et al., 2011; Michalak et al., 2013a) with a record-setting bloom in 2011 and another significant bloom in 2013. Another HAB in 2014 resulted in the issuance of a drinking-water advisory for the city of Toledo, Ohio, for 3 days, which included prohibitions for most home and commercial water uses (Henry, 2014). Phytoplankton (free floating algae) and periphyton (attached algae and aquatic plants) growth in Lake Erie is driven primarily by the nutrients nitrogen and phosphorus. In Lake Erie, this algal growth is considered phosphorus-limited (Ohio Lake Erie Task Force, 2010; Reutter et al., 2011; Kane et al., 2014), in part because cyanobacteria present in the lake can convert atmospheric nitrogen into a form of nitrogen that it is biologically available for algal growth. The availability of nitrogen may influence the particular species that dominate an HAB in Lake Erie and the toxicity of the bloom (Michalak et al., 2013a; Dolman et al., 2012; Reutter et al., 2011; Pennuto et al., 2014), and more research is needed to fully assess the role of nitrogen in Lake Erie algal blooms (Reutter et al., 2011; Chaffin et al., 2013; Pennuto et al., 2014). Microcystis sp. and Anabaena sp. are often dominant species in these HABs, and they can release potent toxins resulting in increased drinking-water-treatment costs, and recommendations to avoid water contact (International Joint Commission, 2014; Ohio Bureau of Environmental Health, 2012).

> **Phosphorus Cycling**
>
> "There are three types of internal phosphorus cycling in Lake Erie. First, much of the phosphorus that is loaded to the Lake is delivered to the western basin, where it undergoes cycling and transport eastward. Second, there are various biological transformations and food chain transfers of phosphorus that recycle the highly bioavailable DRP to organic phosphorus in various trophic levels and the water column, and to organic phosphorus in the sediment. The third type of internal lake phosphorus cycling is regeneration of sediment phosphorus and its transport back into the water column. About half of the phosphorus that is deposited as particulate organic phosphorus on the sediment surface is regenerated and returned to the water column as phosphate (inorganic phosphorus), especially under anoxic conditions."
>
> - Ohio Lake Erie Phosphorus Task Force 2010

The increased algal biomass of a bloom typically forms in the western basin of Lake Erie and flows to the central basin where it decomposes, consumes oxygen, and forms hypoxic or anoxic zones resulting in an extensive "dead zone" (Reutter et al., 2011; Ohio Lake Erie Phosphorus Task Force, 2013). This anoxic zone kills fish, reducing recreational and commercial fisheries. Tourism and recreation on the lake are affected during an algal bloom even if it is not producing toxins; attached algae can cause difficulty for swimming and boating, not to mention a foul odor when it begins to decompose. The HABs have resulted in multiple beach closures and drinking-water advisories. Because there is limited monitoring to confirm toxic forms of algae, health departments recommend avoiding the water altogether if a toxic bloom is suspected (Ohio Bureau of Environmental Health, 2012).

In addition to these policy-level concerns with direct human impact, the excessive algal growth also can affect the foundation of the lake ecosystem, disrupting the normal distribution of phytoplankton and periphyton. This disruption changes the equilibrium of primary productivity that supports the aquatic ecosystem, and affects the macroinvertebrate population, which in turn affects fisheries. Phytoplankton, primary productivity, and macroinvertebrates are all indicators of lake health that can serve as warnings of water-quality decline, even when the more visible blooms are not present. These nutrient-enrichment endpoints of concern are depicted in the conceptual model (Figure 1). Information and data on current and historic algal blooms, Lake Erie hypoxia, and their ecosystem effects can be found in Michalak et al. (2013a), Reutter et al. (2011), Stumpf et al. (2012), and the Ohio Lake Erie Phosphorus Task Force (2010 and 2013).

Figure 3. Photograph of harmful algal bloom, 2013, in Lake Erie.
[Photo by: Jeff Reutter, Ohio Sea Grant and Stone Laboratory]

Two primary measurements of phosphorus are used to measure the overall phosphorus contributions to Lake Erie and its tributaries from nonpoint land-based sources, point sources, and atmospheric sources. Total phosphorus (TP) refers to all forms of phosphorus and includes both particulate and dissolved forms. The abbreviation, TP, will be used to reflect total phosphorus in the remainder of this document. Although TP is not completely bioavailable to stimulate algal growth, much of it could become available in the future as a result of biological and chemical processes. The dissolved, reactive portion of TP that directly stimulates algal growth is often referred to as bioavailable phosphorus, dissolved reactive phosphorus (DRP) or soluble reactive phosphorus (SRP) (International Joint Commission, 2014). The abbreviation, DRP, will be used to reflect this portion of phosphorus in the remainder of this document. Researchers have identified a relationship between DRP and cyanobacterial biomass in lakes including Lake Erie (Reutter et al., 2011; Ohio Lake Erie Phosphorus Task Force, 2013). TP and DRP were selected for detailed analysis in this case study and are discussed further in sections 3.2 and 5.1.

Over the past 20 years, TP loadings to Lake Erie have remained relatively stable, but the DRP fraction of TP loads in measured watersheds has increased more than 100 percent during this time period while the fraction of particulate phosphorus has decreased (Figure 4) (International Joint Commission, 2014; Ohio Lake Erie Phosphorus Task Force, 2013). These recent increases in DRP have been attributed to more frequent high-magnitude storm events, changes in fertilizer application timing and rate, increasing extent and intensity of artificial drainage, and changes in agricultural management practices (such as increasing extent of conservation tillage) that increase phosphorus accumulation at the soil surface (Daloğlu et al., 2012; Michalak et al., 2013a; Ohio Lake Erie Phosphorus Task Force, 2013; Baker et al. 2014). Figure 4 shows this trend in four different Lake Erie watersheds from 1975 through 2012.

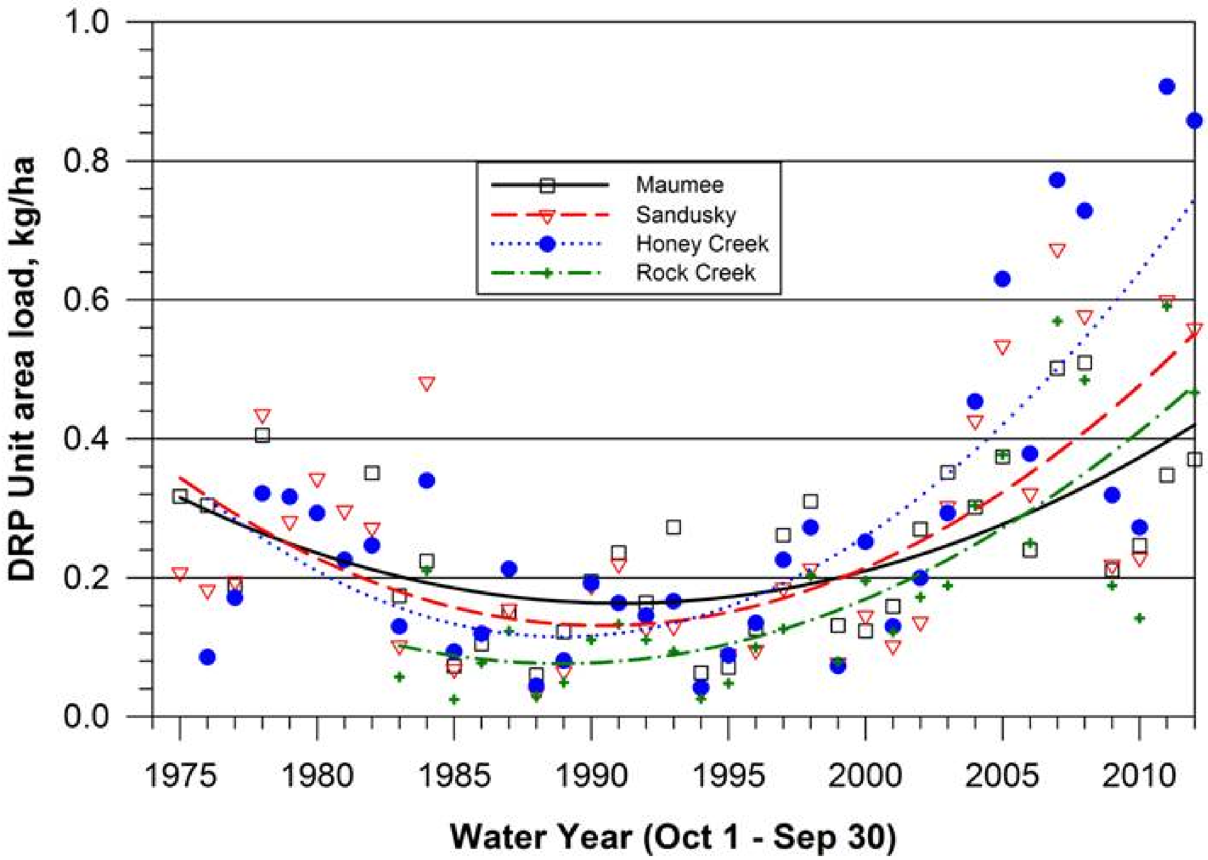


Figure 4. Annual unit area loads of dissolved reactive phosphorus (DRP) in four streams draining into western Lake Erie, 1975 through 2012 (National Center for Water Quality Research, Heidelberg University, unpublished data). [Abbreviations: kg/ha, kilograms per hectare]

These changes, combined with extensive agricultural lands in the primary source areas delivering phosphorus into Lake Erie, have increased attention on the need for management practices to control agricultural nonpoint-source loadings of phosphorus to Lake Erie.

2.3.3 Agricultural management practices in the Lake Erie drainage basin

Tributaries draining into the western Lake Erie drainage basin contribute the greatest external nutrient loadings to the lake, and the Maumee River is the largest single source of DRP to Lake Erie (International Joint Commission, 2014). The western basin of the lake received approximately 64 percent of average-annual phosphorus loadings to Lake Erie during 2003-2011 (International Joint Commission, 2014), compared to 26 percent to the central basin and 11 percent to the eastern basin. Nonpoint sources accounted for 61 percent of the total phosphorus load to the entire lake, but 71 percent of the load to the western basin (Figure 5; Ohio Lake Erie Phosphorus Task Force, 2010). Estimates indicate between 33 and 44 percent of the Lake Erie nonpoint-source total phosphorus load comes from agricultural sources (Robertson and Saad, 2011). Row crops are the dominant land use of watersheds draining to the western basin of Lake Erie, accounting for over 71 percent of the land area (Figure 6; Ohio Lake Erie Phosphorus Task Force, 2010). Extensive poorly drained soils in the Lake Erie drainage basin have resulted in use of tile drainage systems throughout the basin, which are necessary to sustain agricultural production in this region (Reutter et al., 2011).

Given the significance of agricultural nonpoint source contributions in the Lake Erie drainage basin, and the western Lake Erie drainage basin in particular, any solution for reducing HABs and hypoxia in Lake Erie must include a means for addressing agricultural nonpoint-source contributions. Agricultural phosphorus loadings to Lake Erie come primarily from fertilizer application and manure transported by runoff water during spring snowmelt and heavy rainstorms (International Joint Commission, 2014).

External Phosphorus Source	Connecting Channel MTA	Western Basin MTA	Central/Eastern Basin MTA	Total Loads MTA	Percent of Total
Nonpoint	522	3,987	1,094	5,604	60.8%
Point	1,051	388	469	1,908	20.7%
Upper Lakes	1,080	0	0	1,080	11.7%
Atmospheric		80	548	628	6.8%
Total	2,653	4,455	2,111	9,220	100%
Percent of total	**29%**	**48%**	**23%**	**100%**	

Figure 5. Approximate distribution of phosphorus entering each component of the Lake Erie system from various external phosphorus sources, 1998 - 2005 (from Ohio Lake Erie Phosphorus Task Force, 2010; reprinted with permission).
[Abbreviations: MTA, Metric tonnes/year; %, percent]

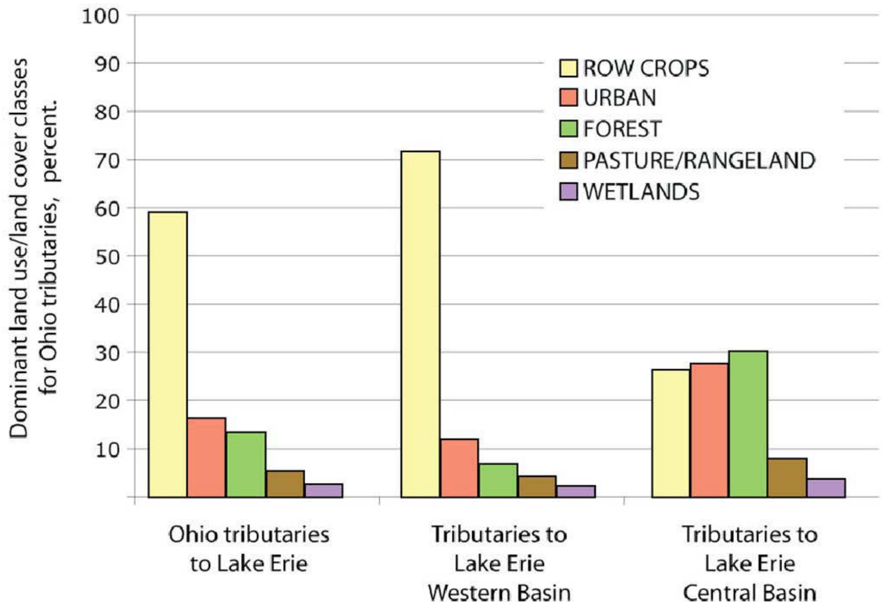


Figure 6. Dominant land use/land cover classes for groups of tributaries in the Lake Erie basin (from Ohio Lake Erie Phosphorus Task Force, 2010, prepared by Dan Button; reprinted with permission).

Management practices can be applied to achieve different goals. For example, agricultural management practices have traditionally focused on preventing erosion and maintaining soil on the land; examples include conservation tillage, cover crops, and strip cropping. Conservation tillage is the practice of leaving residue from the prior year’s crop at the soil surface to reduce erosion, and strip cropping is growing crops in a narrow band of tilled soil that leaves the soil between bands protected by the previous year’s crop residue as barriers to wind and water erosion. In this case study, the focus is on agricultural management practices for the purpose of controlling phosphorus loads, which is generally accomplished by reducing both TP and DRP transport from fields and/or controlling stormwater onsite. Practices for the purpose of controlling phosphorus loads include agricultural management practices that reduce the amount of phosphorus applied to fields, slow the movement of water to the field drainage system, and detain flows at field drainage outlets (International Joint Commission, 2014).

Agricultural management practices that are designed for sediment control may not have the same effectiveness for controlling DRP as they do for sediment-bound TP, and their success in DRP control has been mixed. For example, no-till and conservation tillage can result in increased accumulation of phosphorus at the soil surface, an increase in subsurface transport of DRP via preferential flow through soil macropores to tile drains, and increased DRP released to tributaries (International Joint Commission, 2014; Ohio Lake Erie Phosphorus Task Force, 2013; Daloğlu et al., 2012). The extensive use of tile drainage in the Lake Erie drainage basin may facilitate the transport of DRP to streams (Ohio Lake Erie Phosphorus Task Force, 2013). Tile drainage is discussed in more detail in section 2.4.3.

The International Joint Commission (2014) has identified several agricultural management practices that are known or anticipated to be effective for controlling TP and/or DRP: managing phosphorus inputs in agricultural operations, manure treatment, conservation tillage, cover crops, wetlands and wetland restoration, and drainage-water management. Some of these practices are known to decrease TP but their impact on DRP is less well understood (International Joint Commission, 2014). While information available to date supports implementation and continued use of these practices, there are few studies that have quantified the resulting phosphorus load reduction within the Lake Erie drainage basin, and a survey of available literature shows a wide variation in reductions achieved (International Joint Commission, 2014). Management practices produce the greatest reduction in nutrients when sited on land that is highly vulnerable to nutrient and sediment loss (Tomer and Locke, 2011; Bosch et al., 2013). Management practices targeted to high-source areas, near-stream areas, and hydrologically active areas have been shown to result in significant phosphorus load reductions in modeling and field-scale monitoring programs (Bosch et al., 2013; Rao et al., 2009). Table 1 summarizes agricultural management practices that are expected to be effective for reducing TP and/or DRP, practices with mixed results (reductions and increases), and practices that may increase DRP loss from the land and soil. These are practices for which it would be useful to have existing water data to evaluate their effectiveness at the watershed scale.

Models have been used increasingly to estimate management practice efficiencies, especially where water-quality data are unavailable to directly measure the impact of these interventions. However, the National Research Council (2011) has cautioned that excessive reliance on models in the absence of monitoring data can increase rather than reduce uncertainties. They also emphasize the need for better integration of monitoring and modeling activities to support effective adaptive management.

Many task forces and workgroups have identified a need for data collection and analysis to determine the effectiveness of management practices in the Lake Erie drainage basin, including the International Joint Commission (2014), Koslow et al. (2013), the Ohio Lake Erie Phosphorus Task Force (2010 and 2013), and Reutter et al. (2011). Reutter et al. (2011) suggest that low land slopes, poorly drained soils and intensive agricultural drainage in the western Lake Erie drainage basin create unique conditions requiring localized research. In addition, the International Joint Commission (2014) found that most agricultural management practice studies have focused on sediment and TP reductions, and do not typically include results for DRP. Watershed-scale water-quality monitoring studies are needed because, as the U.S. Environmental Protection Agency (USEPA) points out, “few, if any, data suggest actual watershed-wide implementation efficiencies as high as those in the research literature. Several recent small watershed studies have indicated considerably lower reductions when groups of practices are applied in the watershed than would have been expected according to current efficiencies” (U.S. Environmental Protection Agency, 2010).

Table 1. Agricultural management practices for reducing total phosphorus and dissolved reactive phosphorus. [**Abbreviations:** TP, total phosphorus; DRP, dissolved reactive phosphorus]

Practice	Description of Practice	Explanation of Nutrient Transport Results
Practices expected to reduce TP and/or DRP transport		
Managing Phosphorus Inputs in Agricultural Operations (4R)	Right fertilizer source, right rate, right time, right place. Includes soil testing, modifying application rate, timing application based on meteorological conditions, and different application methods. Mineral phosphorus is best applied by banding below the surface at seeding. Manure applications are best incorporated with minimal-disturbance methods.	Phosphorus should only be added to soils with less than sufficient soil-phosphorus levels. Using fertilizer application based on scientific principles, including site-specific considerations and adaptive management, will result in improved sustainability.[1]
Manure Treatment	Physical, chemical, and biological treatment of manure.	Reduces solubility of phosphorus.[1]
Wetlands Protection and Restoration	Wetland restoration involves converting cropland and other drained areas into wetlands.	Wetlands act as a buffer by providing the benefits of flow attenuation, reduced runoff, filtering of nutrients.[3, 8]
Drainage Water Management	Practices that slow down or retain water in tile-drain systems.	Reduces the overall volume of water released from the tile drainage system over time.[3, 8]
Cover Crops	Any crop grown to provide soil cover, regardless of whether it is later incorporated into the soil.	Cover crops prevent nutrients from leaching or leaving in runoff waters, reduce discharge volume, improve soil tilth and quality, and reduce erosion.[3, 8]
Crop Rotation	Different crops are planted in consecutive growing seasons to maintain soil fertility and minimize crop disease/pest issues.	Use of legumes in rotation can increase phosphorus uptake. Improved soils exhibit less runoff volume and increased organic-matter contents.[8]
Contour Strip Cropping	Integrates wheat or hay with row crops in alternating strips planted on the topographic contour.[8]	Disperses phosphorus application, diversifies land cover, and slows runoff velocities.[8]
Grassed Waterways	Grassed waterways are natural/constructed vegetated channels near cropland where water concentrates and flows off fields.	Grassed waterways help prevent gully formation and erosion and may also trap sediment in surface runoff and absorb chemicals and nutrients.[3,4,5]
Two-Staged Ditch	Drainage ditches that have been modified by adding benches that serve as floodplains within the overall channel.[6]	Creates an in-ditch bench that facilitates denitrification and nutrient uptake while enhancing the stability of the channel and reducing sediment movement.[6]
Phosphorus Reactors	A structure installed to capture nutrients leaving the field in tile flow or sheet flow and reduce the concentration of phosphorus released to the stream.[7]	Intercepts flow, treats and removes phosphorus from the flow through physiochemical adsorption.[7]

Practice	Description of Practice	Explanation of Nutrient Transport Results
Filter Strips	A filter strip consists of vegetation installed along channel-segment edges to reduce sediment, nutrients, pesticides, and bacteria in surface runoff as it passes through.	Filter-strip effectiveness relies on sheet flow, whereas in practice buffers often are bypassed by field furrows and concentrated-flow outlets.[1, 3]
	Practices with mixed results for reducing TP and/or DRP transport	
Conservation Tillage	Conservation tillage involves management practices that leave at least 30 percent of the soil surface covered with crop residue following tillage and planting to reduce soil erosion.[1]	Has been shown to reduce TP associated with soil, but can lead to soil stratification.[1, 8] May increase or decrease DRP loss from the land.[2, 8] Enhanced flow through soil matrix and reduced preferential flow lead to better contact with P adsorption sites, thus reducing loss from the soil; improves water-retention capacity of the soil.[2] Zonal or strip tillage may allow incorporation of phosphorus with limited surface disruption to maintain a residue cover that reduces runoff.
Tile Drainage	A series of underground tiles that are installed on poorly-drained soils to improve soil quality and water infiltration, reduce compaction, improve crop yields, and potentially provide a system to control the amount of surface runoff.	Tile drainage with a healthy soil matrix can lead to better contact with phosphorus-adsorption sites, decreasing DRP loss from the soil. Inadequate tile drainage could lead to reducing conditions in the soil that could increase DRP losses. The combination of other management practices with tile drainage will influence whether tile drains reduce or increase phosphorus transport. DRP released from soils may be transported to streams through tile drains. A poor soil matrix (often associated with no-till) can increase TP losses through soil macropores that provide direct conduits from the surface to the tiles.[1,2]
	Practices that may increase DRP loss	
Fall Fertilizer Application	Consolidation of farms leads to need for greater efficiency; promotes surface application of fertilizers in the fall.	Longer exposure to precipitation increases opportunities for loss.[2]
Fertilizer Broadcast on the Surface (Not Injected or Incorporated)	Applies to no-till, fall fertilizer application, and fertilizer application to frozen ground.	More direct exposure to precipitation, lack of binding to soil particles.[2]

Practice	Description of Practice	Explanation of Nutrient Transport Results
No-Till	Maintaining sediment on the soil surface by not loosening soil through tillage practices.	Leads to stratification of P in soil. Soil cracks can create preferential-flow paths to tile drains where P is transported directly to the stream.[1, 8]
Excessive Fertilizer Sales	Applying phosphorus fertilizer without soil-test results and making application decisions based on historical farm practices rather than current information that tracks soil-fertility management needs.	Risk of phosphorus accumulation in soil leading to increased export.[2]
Animal Numbers	Increased livestock management and concentration of livestock into large farms.	Discourages the use of phosphorus-based manure application rates in watersheds with livestock facilities, leading to increased phosphorus transport from manure.[2]

[1]International Joint Commission, 2014
[2]Michalak et al., 2013b Supporting Information
[3]Ohio Lake Erie Phosphorus Task Force, 2010
[4]Baker et al., 2007
[5]Scavia et al., 2014
[6] Ward and Mecklenburg, [n.d.]
[7]Ohio Lake Erie Phosphorus Task Force, 2013
[8]Crumrine, 2011

Nutrient reductions from agricultural management practices at the field scale have been quantified in the available literature (Meals et al., 2010; Reutter et al. 2011; Crumrine, 2011; Iowa Department of Agriculture and Land Stewardship, 2013; U.S. Environmental Protection Agency, 2010) but are more challenging to document at the watershed scale (Sharpley et al., 2009). The effectiveness of agricultural management practices can vary substantially within and among watersheds, and the cumulative effects of combinations of practices can produce results that are different than the sum of their individual reductions (Sharpley et al., 2009; Francesconi et al., 2014). Factors complicating watershed-scale assessments of management practice effectiveness include:

- It takes time for management practices to be implemented at the watershed scale with a density that results in water-quality change
- Land-use and land-management practices are constantly changing
- Legacy phosphorus already in soil and sediment can continue to be released after conservation practices have been implemented (Jarvie et al. 2013 a and 2013b; Sharpley et al., 2013)
- Precipitation and streamflow vary from year to year, which can affect the length of time required to measure water-quality change
- There is a lack of long-term monitoring (Meals et al., 2010)
- It is challenging to maintain an adequate and appropriate monitoring program to document results (Tomer and Locke, 2011)

- Data on management practice implementation and maintenance, and other land-use records are not available or are difficult to obtain.

Watershed-scale studies documenting effective management practice results are rare due to a lack of intensity of management practice application, lack of data regarding the use of land-management practices, and short water-quality monitoring records.

These factors result in a time lag affecting when the results of management practices can be measured at the watershed scale, or an inability to correlate any water-quality change with changes in management practices. The lack of specific data regarding land management practices leads to unverifiable generalizations regarding cropping and livestock systems' nutrient contributions to Lake Erie, and may undermine efforts to identify specific practices that improve water quality. Documentation of land-management practices and access to this documentation are needed to explain differences in water quality among streams draining agricultural areas. Watershed-scale studies documenting effective management practice results are rare due to a lack of intensity of management practice application, lack of data regarding the use of land-management practices, and short water-quality monitoring records.

The effectiveness of management practices in agricultural operations is likely to be challenged by a changing climate. For example, recent data indicate that large phosphorus loads – including DRP – are exported to Lake Erie during major storms. Climate-change models suggest storms will become more frequent and more intense (Koslow et al., 2013; International Joint Commission 2014; Michalak et al., 2013a; Melillo et al., 2014), and increased loads of TP and suspended sediment will be transported to the western basin of Lake Erie (U.S. Environmental Protection Agency, 2013). Currently, the vast majority of TP and DRP nutrient loads occur during major storm events (Reutter et al., 2011). Studies have shown that 80 percent of annual phosphorus loadings can be produced by just one or two storms (Richards and Holloway, 1987). Monitoring during storm flows is needed to quantify performance of agricultural management practices, particularly during these intense storm events.

These factors indicate a need for water-quality data to measure the effectiveness of agricultural management practices at the watershed scale in the Lake Erie drainage basin, and they indicate there is value in quantifying the water-quality data that are needed and available:

- the increasing temporal trend in DRP loads in streams,
- the finding that management practices effective for controlling TP are not always effective for reducing DRP loads to streams,
- the intensive tile drainage in the Lake Erie drainage basin,
- time lags in measurement of management practice effects at the watershed scale,
- lack of existing studies at the watershed scale in the Lake Erie drainage basin, and
- increasing frequency of intense storms.

The appropriate water-quality data, and the information produced from their analysis, are critical for policy makers to determine whether agricultural management practices can be effective tools for controlling HABs in the lake now and into the future.

2.4 Types of data needed to answer the case-study policy question

The first case-study task was to describe, and quantify where possible, the types and amounts of water-quality data that are needed to answer **"How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?"** in the Lake Erie drainage basin.

A conceptual model (Figure 1) was developed to delineate the multiple types of data necessary for answering the case-study policy question, both water-quality data and other types of data. The conceptual model presents relationships between management practices, nutrient sources, water data, and the endpoints of concern to decision-makers. For the data types shown in Figure 1, this section describes the type of data and why they are needed, potential data sources, and whether the data type was quantified as part of this case study. The specific data types that were evaluated in this case study directly or were examined as ancillary data are shown in Table 2.

2.4.1 Management practices

The conceptual model (Figure 1) shows management practices that can be used in three different sectors to address nonpoint sources of nutrients. Urban nutrient-management practices are practices geared toward urban settings, typically focused on managing stormwater flows, such as filter strips, porous pavements, green roofs, and wetlands. Their presence within a watershed affects total nutrient loadings, but urban management practices were not the focus of this study. The distribution of land cover within a given watershed between agricultural and urban uses is a major factor in identifying the source of nutrient loadings to a stream and the distribution of urban and agricultural practices that would be needed to reduce those loadings. Management practices that address agricultural nonpoint sources include agricultural practices, such as those described in section 2.3.3, practices used to address livestock operations, and stream or riparian practices that address runoff as it flows to streams, such as wetlands and riparian buffer strips.

In order to analyze water-quality data to quantify management practice effectiveness, considerable information regarding the implementation of management practices across each monitored watershed is necessary. These implementation data include management practice installation dates, management practice design and maintenance information, soil phosphorus data, and farm-management data. In addition to implementation of management practices, farming and land management continuously change; crops change from year to year and fields are taken in and out of production, also affecting nutrient runoff. Areas with land and soil vulnerability to nutrient and sediment loss have been associated with greater load reductions from management practices compared to average soils (Tomer and Locke, 2011; Bosch et al., 2013); information on site vulnerability to nutrient and sediment loss is necessary to extrapolate measured nutrient-removal efficiency to the same soil types in other watersheds.

Table 2. Water-quality and ancillary data types necessary to answer the case-study policy question, identifying the data types evaluated in this report[1].

Data type	Evaluated in this report
Water-quality data	
Tributary Water-Quality Data	Quantified
In-Lake Water-Quality Data	Not evaluated
Ancillary data	
Agricultural Management Practice Implementation Data	Not evaluated
Nutrient-Source Data	Not evaluated
Land Use	Supporting Analysis
Soil-Vulnerability Data	Supporting Analysis
"Spatially Referenced Regressions on Watershed Attributes" (SPARROW) and Soil and Water Assessment Tool (SWAT) model data	Supporting Analysis
Tile-Drainage data	Supporting Analysis
Streamflow	Quantified
Weather variables (precipitation and temperature)	Not evaluated
Endpoints of Concern	Not evaluated

[1] U.S. tributaries only; phosphorus data associated with the inflow of water from Lake Huron and Lake St. Clair to western Lake Erie from the Detroit River also were not evaluated because the drainage area did not meet the case-study criterion of 40-percent row crop cover.

All these types of data are needed in order to associate a water-quality change with management practice activities within a watershed. Any new water-monitoring study evaluating effectiveness of management practices should develop an inventory of practices already in use, and document ongoing maintenance of those practices and/or associated structures, so that the incremental water-quality improvement related to additional practices can be identified. It is important for these data to be documented at a level of detail that will support analysis of water-quality and streamflow data to identify causality of water-quality change, with an appropriate data system for storage and retrieval.

There are significant logistical, institutional, and legal barriers to assembling and sharing management practice data, including the confidentiality restrictions outlined in Section 1619 of the Farm Bill and other access limitations (Weller et al., 2010). As a consequence, water-quality researchers must depend on farmers' willingness to share their land management data. Agricultural management practice implementation data, when they are available, are further complicated by a lack of documentation at the level of detail needed for water-quality analysis (Jackson-Smith et al., 2010). Grady et al. (2013) evaluated three different methods for obtaining geospatial information for management practice implementation and found that using only one method for obtaining management practice data can result in incomplete information. Grady et al. (2013) recommended using a variety of methods for identifying the extent of management practice implementation in a watershed, including government records, producer interviews, and remote-sensing aerial-photo interpretation. The National Agriculture Statistics Service, Ohio Department of Natural Resources (ODNR), Soil and Water Conservation Districts, and Western Lake Erie Basin Partnership are currently involved in collecting these types of data but there is room to improve data collection and coordination (Ohio Lake Erie Phosphorus Task Force, 2013) and sharing of these data for water-quality evaluations. Agricultural management practice implementation and maintenance data were not compiled and analyzed for this study.

2.4.2 Nutrient sources

The conceptual model (Figure 1) identifies four categories of nutrient sources that contribute to watershed nutrient loadings, both point and nonpoint sources. Nutrient loads are calculated at a watershed outlet because it is not practical to monitor nonpoint-source nutrient loadings at their source. In order to attribute a portion of nutrient loads to agricultural sources, the other point and nonpoint sources within the watershed must be monitored or estimated. Point-source contributions, including wastewater-treatment plants and combined sewer overflows, are the most straightforward to assess; monitoring of these nutrient sources is usually required as part of a National Pollution Discharge Elimination System (NPDES) permit. These data are usually tracked as part of the Permit Compliance System or Integrated Compliance Information System and are not stored in the same data system as ambient water-quality data. In general, point-source data are available, but difficulties exist in compiling these data for load calculations supporting national and regional evaluations (Maupin and Ivahnenko, 2011). These data were not quantified or compiled for this project. The history of major sources of TP loading to Lake Erie since the late 1960s (R. Peter Richards, Heidelberg University, written commun., 2014) is shown in Figure 7.

In tributaries, once point sources are accounted for, the majority of the remaining nutrient loads can be attributed to nonpoint sources (Robertson and Saad, 2011). Contributions from these sources are often estimated as a percentage of overall load based on impervious area or the extent of each land use within a watershed.

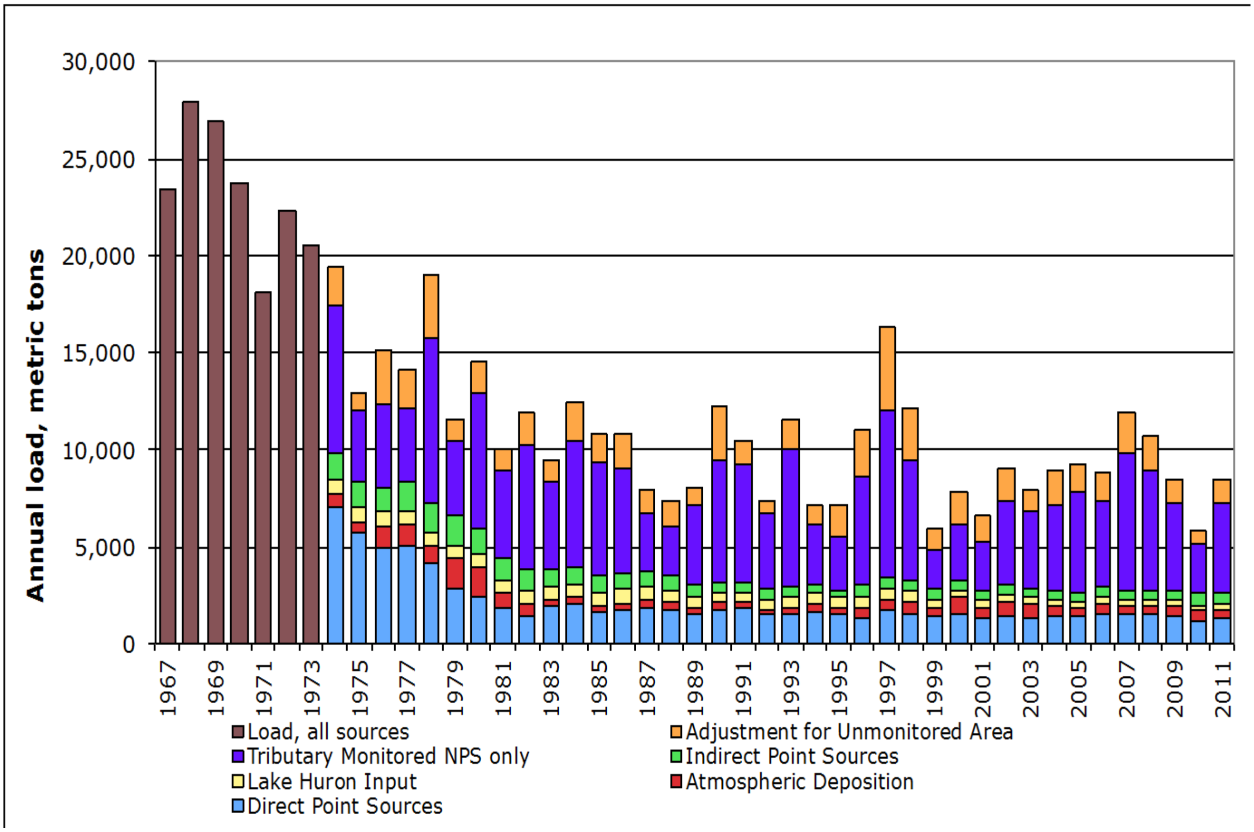


Figure 7. Annual Lake Erie total phosphorus loading by source, 1967-2011. Data provided by Dr. David Dolan of the University of Wisconsin Green Bay (May 2013). (Prepared by Heidelberg University National Center for Water Quality Research staff, from International Joint Commission, 2014; reprinted with permission).
[Abbreviations: NPS, non-point sources]

2.4.3 Nutrient transport

The conceptual model (Figure 1) depicts streamflow, driven by precipitation, as one of the primary transporters of nutrients from nonpoint sources to tributaries, which can be measured as tributary water-quality data. Continuous streamgages or stage monitors measure the variation in flow throughout the year, and these year-round flow measurements are needed to calculate annual load estimates; flow must be measured for each individual water-quality monitoring site to support load calculations. Because nutrient concentrations in tributaries typically are closely correlated with streamflow, streamflow data are also critical information needed for any assessment of stream-nutrient-concentration changes over time.

USGS is the primary provider of streamflow data across the United States. Although some other agencies collect streamflow data for specific purposes, such data are often not readily available. The national data aggregation did not include continuous-record streamflow data from other agencies, so the USGS National Water Information System (NWIS) database (U.S. Geological Survey, 2014b and 2014c) was the source of streamflow data for this study.

The most critical nonpoint nutrient loadings occur during storms, and recent climate models predict increasing frequency of intense storms (see section 2.3.3). Precipitation and flow data are needed on an ongoing basis to track storm intensity and duration, and identify long-term climatic trends. Weather and climate affect many stressors other than TP and DRP and cause variations that must be quantified to identify the role of management practices in a changing ecosystem. Temperature and precipitation records should be maintained so that weather variability, including drought and climate trends, can be identified and analyzed. Temperature and precipitation data were not compiled and analyzed for this study, however, long-term precipitation and climate data sets are collected by cooperative observers and several agencies, including National Oceanic and Atmospheric Administration (NOAA) research laboratories and the National Weather Service. Long-term data sets for the Lake Erie drainage basin are available from NOAA (2014).

Tile drains represent another nutrient-transport pathway in the Lake Erie drainage basin. Water discharged from tile drains typically consists of both surface water and groundwater and can contain nutrients derived from recently applied agricultural fertilizers on the surface and from phosphorus sources in soil or geologic materials within the subsurface. The use of tile drainage is more extensive in the Lake Erie drainage basin than in any other agricultural region in the United States due to the presence of extensive, poorly drained soils (Reutter et al., 2011). The contribution of tile drainage to nutrient loadings and to assimilating nutrients from surrounding farmland is poorly understood and water discharged from tile drains is often poorly managed (International Joint Commission, 2014). The increase in extent and intensity of tile drainage has been identified as a potential cause contributing to recent increases in DRP loading in western Lake Erie tributaries (Ohio Lake Erie Phosphorus Task Force, 2013), and additional research is needed to fully understand and quantify the tile-drainage contribution to phosphorus loading to Lake Erie tributaries, particularly during storms. Once tile-drainage loadings to tributaries are better understood, new management practices or treatment options may need to be identified to address this transport pathway for DRP. Although this case study focuses on phosphorus, tile drains are a primary pathway for nitrogen, specifically nitrate, to streams, and to understand the effectiveness of agricultural management practices, both nitrogen and phosphorus need to be measured.

Legacy sources of phosphorus and nitrogen, in soils, groundwater, sediments, wetlands, and other areas, can create lags in transport and complicate assessment of nutrient reductions from management practice implementation (Baker et al., 2007; Jarvie et al. 2013 a and 2013b; Sharpley et al., 2013). For example, groundwater discharged to tributaries is a transport pathway for nutrient loadings that is not easily measured. Groundwater can be a source of legacy nitrate resulting in a lag in nitrogen reductions relative to implementation of new management practices (Baker et al., 2007). Transport lag times can interfere with load estimation and prediction, especially when assessing results of management practices that may lag implementation by many years as nutrients are flushed from groundwater over time.

2.4.4 Tributary water-quality data

The conceptual model (Figure 1) includes the primary forms of phosphorus and nitrogen species measured in water as it moves from the headwaters of small watersheds, to large watersheds, to the main stem, and eventually to the lake itself. Both phosphorus and nitrogen must be present and bioavailable to trigger algal growth in streams and Lake Erie. The chemical form and bioavailability of each of these nutrients change as they move from headwaters to the lake. Other trace elements, such as silica, which is used by diatoms, may also play an important role in algal growth, indicating that data for additional parameters identified in the conceptual model (Figure 1) may be required to understand these biological systems and conditions triggering algal blooms (DeBruyn et al., 2004).

The biochemical processes that affect the phosphorus and nitrogen cycles are influenced by flow, sediment loads, aquatic biota, and other water parameters such as temperature, pH and dissolved oxygen. Silica can also be a limiting nutrient in certain environments (Rabalais, 2002). These parameters are reflected in the “Other Tributary Water-Quality Data” box of Figure 1. A number of biologists and modelers are examining how these parameters affect nutrient bioavailability and drive algal growth in Lake Erie (e.g., Scavia et al. 2014; Stumpf et al., 2012). The suite of tributary-monitoring parameters identified in the conceptual model as necessary to answer the policy question for this case study are summarized in Table 3.

Table 3. Suite of tributary water-quality monitoring parameters identified in the conceptual model as necessary to answer the case-study policy question “How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?”

Parameter	Laboratory measurement	Field measurement	Bioindicators
Total phosphorus	x		
Dissolved reactive phosphorus	x		
Total nitrogen	x		
Silica	x		
Suspended sediment	x		
Water temperature		x	
Air temperature		x	
pH		x	
Dissolved oxygen		x	
Streamflow		x	
Macroinvertebrates		x	x
Periphyton (attached algae)		x	x

2.4.5 Lake water-quality data

Bioavailable forms of nitrogen and phosphorus in Lake Erie drive algal biomass in Lake Erie, shown under the heading “Lake Water-Quality Data” in the conceptual model (Figure 1). Loadings come from both external (e.g., tributary) and internal (e.g., sediments, biota) sources, and concentrations vary by depth and location. Lake water-quality monitoring, for both nutrients and algal biomass, is very different from tributary monitoring. The USEPA Great Lakes National Program Office, the Lake Erie Index Station

Monitoring program, the United States National Coastal Assessment, and other programs focus on collecting water-quality data in the open waters of Lake Erie (International Joint Commission, 2014). Lake water-quality data are the data that link management practice effectiveness to the policy endpoints of concern in Lake Erie, but the tributary water-quality data are the most direct measures of management practice effectiveness. There is a variety of recent and ongoing research attempting to standardize Lake Erie water-quality data collection and to model the relationship between nutrient loadings, nutrient availability, and algal growth within the lake (Qian et al., 2013; Scavia et al., 2014). Coordinated international monitoring, research, and response are needed for lake water-quality monitoring programs to be effective.

Changes in temperature, light penetration, wind, and biological communities over time due to climate change and invasive species may affect the frequency and intensity of algal blooms and could mask the effects of reduced nutrients from management practices (Koslow et al, 2013; Reutter et al., 2011). Invasive zebra and quagga mussels are known to alter the phosphorus cycle and influence algal growth by recycling phosphorus in near-shore environments, improving water clarity, and providing substrate for filament attachment (Reutter et al., 2011). Consequently, parameters such as temperature, light (as measured by secchi depth), and species surveys are needed to characterize these phenomena. These parameters are reflected under the heading “Other Lake Water-Quality Data” in the conceptual model (Figure 1). Open-lake water-quality data were not compiled and analyzed for this study.

2.4.6 Endpoints of concern to policy makers

The endpoints of concern to policy makers were discussed in section 2.3.2. It is important to monitor the endpoints in addition to changes in nutrient concentrations and loads because a lack of response in these endpoints could indicate other ecosystem characteristics are changing the ecosystem response to nutrient loadings. Additionally, the use of one parameter alone to define results may lead to a misunderstanding of nutrient conditions in the streams and lake. For example, algal blooms will consume nutrients in the water, which decrease nutrient concentrations. If only nutrient data are collected, it would suggest water-quality improvements. However, if chlorophyll-*a* concentrations are collected, it would reflect increased algal biomass and account for at least some of the decreased nutrient concentrations in the water. Availability of data on the endpoints of concern, in-lake, and tributary water-quality data are extremely important to be able to reconstruct and understand the ecosystem response. Endpoint data are collected in Lake Erie by many of the same organizations collecting open-lake data, such as state environmental protection agencies (e.g., Ohio Environmental Protection Agency (OH EPA)) and academic research programs. Endpoint data were not compiled and analyzed for this case study.

3. Most Critical Data Types Needed to Answer the Case-Study Policy Question

The conceptual model makes clear that many data types are needed to quantify the effectiveness of management practices at reducing nonpoint nutrient sources at the watershed scale. The Technical Advisory Committee (TAC) further refined the scope of this case study to more clearly identify and prioritize the quantities of data needed to answer the policy question. This section describes the most important data types selected by the TAC for measuring the effectiveness of management practices at the watershed scale; the data types are summarized in Table 4.

Table 4. Most important watershed and monitoring data types for detecting water-quality change resulting from management practices in the Lake Erie drainage basin, selected by the Technical Advisory Committee.

Small watersheds	Large watersheds
Less than or equal to 50 square miles	Greater than or equal to 1,000 square miles
--	Drains directly to Lake Erie
Greater than or equal to 40-percent row-crop coverage	Greater than or equal to 40-percent row-crop coverage
High phosphorus yield and high soil vulnerability	--
TP, DRP, and streamgage	TP, DRP, and streamgage
Active as of 2014	Active as of 2014

3.1 Study design needed to answer the case-study policy question

The conceptual model (Figure 1) can be distilled into three critical components that must be present in a monitored watershed to answer the case-study policy question, as shown in Figure 8. For example, the selection of appropriate monitoring sites is critical for answering the case-study policy question. Monitoring sites must be located in watersheds dominated by agricultural land use, and where management practices can, and will, be widely implemented in optimal places for reducing nutrient loss. Monitoring sites in these types of watersheds allow for the detection of water-quality change that these practices can generate. Further, tributary water-quality and streamflow data at these monitoring sites must be available to evaluate trends in concentration and load in these watersheds over time. Finally, data on management practice implementation and other changes in land use and nutrient sources throughout the watershed must be available to correlate water-quality change with alterations to the land. Without this information, the relationship between management practices and water quality cannot be evaluated, even if management practices are delivering detectable reductions in nutrient loads.

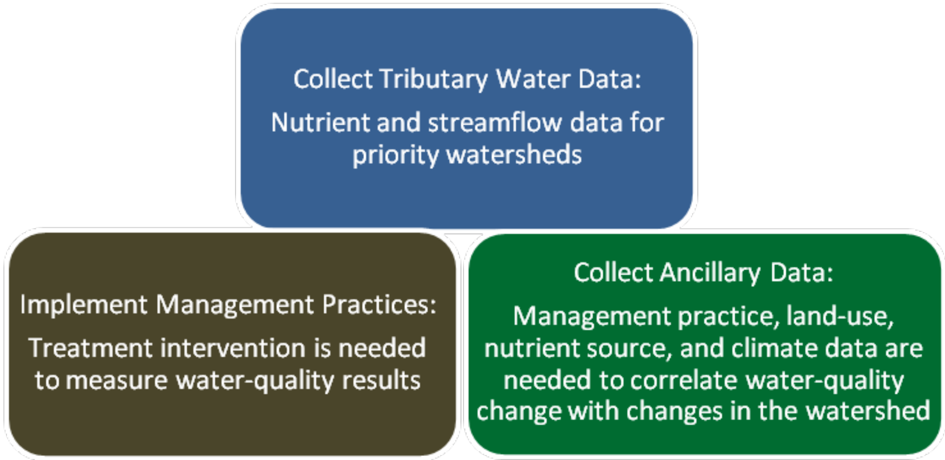


Figure 8. Study design needed to answer "How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?"

3.2 Focus on tributary TP, DRP, and streamflow data

Phosphorus is considered the nutrient limiting algal growth in Lake Erie (Reutter et al., 2011). As discussed in section 2.3.2, both TP and DRP are important measures of phosphorus when considering the effectiveness of agricultural management practices, although all the parameters in Table 3 are needed to provide the most informed answer to the case-study policy question. A streamflow measurement must be available for every TP or DRP sample in order to calculate load, and a continuous record of streamflow throughout the year is needed to support calculations of annual TP and DRP loads.

3.3 Focus on agricultural management practices.

The focus on agricultural management practices was selected because tributaries in the western drainage basin contribute the greatest external nutrient loadings to the lake, and the majority of the western drainage basin is dominated by agricultural land use (section 2.3.3). Watersheds with greater than or equal to 40-percent row-crop cover were indentified in the Lake Erie drainage basin, and the majority of these watersheds are located in the western Lake Erie drainage basin.

Due to the dominance of agricultural nutrient sources in the Lake Erie drainage basin, effective management practices for controlling agricultural nonpoint sources of nutrients will play a critical role in reducing external nutrient loadings to Lake Erie to control HABs and hypoxia.

3.4 Focus on monitoring small agricultural watersheds with modeled high phosphorus yield and high soil loss vulnerability

Small agricultural watersheds, defined here as those smaller than about 50 square miles and with greater than or equal to 40-percent row-crop cover, were selected for focus because this is the most practical scale over which agricultural management practices can be implemented and maintained across the majority of the watershed. Not included in this definition of small watersheds was edge-of-field scale monitoring, which typically covers areas measured in acres and focuses on individual management practices. Field-scale monitoring is the most appropriate scale for evaluating the performance of individual management practices. Field-scale monitoring guidance and examples of monitoring installations are available from the U.S. Department of Agriculture (2012b) and Stuntebeck et al. (2008).

The 50 square miles represents an area small enough that can be reasonably managed, and it coincides approximately with the drainage areas of Hydrologic Unit Code (HUC)-12 watershed delineations (U.S. Geological Survey, 2014d) in the Lake Erie drainage basin. The larger the area where appropriate management practices are used within a watershed, the greater the expected change in water quality, and the greater the likelihood that change will be detected through water-quality monitoring. Small watersheds, due to their size, are likely to have more homogeneous land use and fewer potential confounding factors than large watersheds, which would make it difficult to discern the influence of management practices on water quality, such as multiple large point sources.

Ancillary data were not quantified in this case study, but they are critical for answering the case-study policy question. To determine whether water-quality trends result from the implementation of agricultural management practices or other drivers, the following data are needed for monitored watersheds to interpret observed trends:

- Annual data documenting installation of agricultural management practices and their maintenance,
- Data on overall changes in land use and market driven changes in agriculture,
- Data on changes to urban and point source nutrient loadings,
- Data on climate conditions, and
- Data on changes in land drainage technology and hydrologic response.

Data collected at small watershed monitoring sites measure transport from cumulative fields to the receiving stream. Monitoring at small watershed sites in headwater catchments allows an analyst to verify reductions at the watershed scale that have been measured at the edge-of-field scale, and to measure the cumulative effects of multiple management practices across a range of varying soil, drainage, slope, and cropping patterns that occur in the watershed. As watershed size increases, it is more difficult to achieve a density of management practice implementation that would produce a magnitude of change that is easily detected at the watershed outlet. If water-quality changes resulting from agricultural management practices are not detectable in small watersheds, there is little chance that agricultural management practice effects will be detectable in larger watersheds, because the larger watershed loads are the sum of the loads from small watersheds.

Although medium and large watershed monitoring (watersheds greater than 50 square miles) is informative for many purposes, monitoring in these size watersheds was not selected as a priority for answering this case-study policy question.

The results of agricultural management practices and the magnitude of change in TP and DRP loadings will vary across different watersheds in the Lake Erie Basin. Six watershed characteristics were evaluated to prioritize watersheds where water-quality monitoring is most likely to detect water-quality change as a result of agricultural management practices: soil-runoff vulnerability, soil-leaching vulnerability, watersheds with modeled estimates of high TP yield from fertilizer and manure, and watersheds with modeled estimates of high TP and DRP yield. Monitoring sites in priority watersheds with these characteristics are likely to detect greater management practice effectiveness, and statistically significant results may be achievable earlier compared to monitoring in average watersheds.

Agricultural management practices are expected to be most effective in areas with inherent vulnerability to runoff and nutrient loss (Bosch et al., 2013; Lund et al., 2011). Areas vulnerable to surface runoff, such as those with steeper slopes and lower-permeability soils, are expected to exhibit greater phosphorus reductions from BMP implementation compared to average areas (Lund et al., 2011). Watersheds identified as having high phosphorus yields are also priority areas for water-quality monitoring to determine the effectiveness of agricultural management practices. Bosch et al. (2013) found the greatest reduction in nutrient yields occurred when BMPs were placed in small watersheds that delivered the largest nutrient inputs from land to streams based on model results using the Soil and Water Assessment Tool (SWAT). SWAT is a process-based model that predicts changes in water quality in response to changes in practices on the land (Arnold et al., 2012). For example, up to a 9-percent TP reduction was predicted when BMPs were placed in those watersheds contributing high TP yields, but only a 2-percent reduction in watersheds with a random distribution of BMPs (Bosch et al., 2013).

Spatial data sets of soil vulnerability and modeled watershed phosphorus yields were used to assess the availability of phosphorus monitoring sites within the priority areas and to identify priority watersheds for new monitoring (section 4.5).

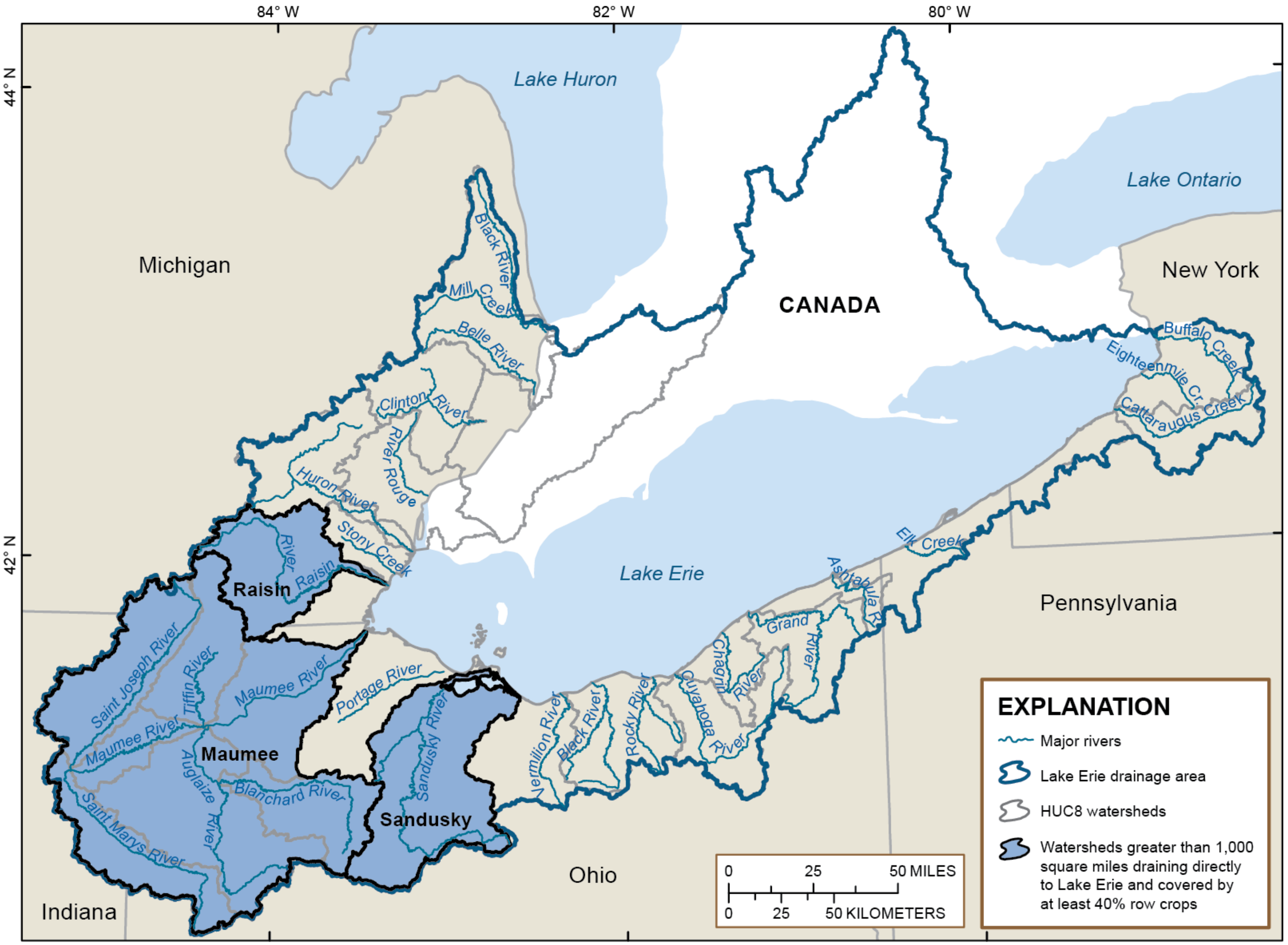
3.5 Focus on monitoring large agricultural watersheds that drain directly to Lake Erie

Because nutrient loading to western Lake Erie is an important factor that drives HABs and episodes of hypoxia in the lake, management practices need to be capable of reducing nonpoint source nutrient loads to mitigate these concerns. Priority large watersheds were defined for this study as watersheds encompassing major tributaries that drain directly to Lake Erie, and have a minimum of 1,000 square miles and greater than 40-percent agricultural land use. The watersheds of the Maumee River, Sandusky River, and River Raisin meet these criteria and contribute about 68 percent of the average annual total phosphorus load to the western Lake Erie Basin (Scavia et al., 2014), and are highlighted in Figure 9. These three agricultural watersheds plus the Detroit River are estimated to account for 97 percent of the total phosphorus loadings to western Lake Erie (Scavia et al., 2014). The Detroit River was not examined through this case study because the watersheds draining to the Detroit River do not meet the 40-percent row-crop criterion. Data for the three large watersheds are needed to determine whether agricultural management practices result in phosphorus load reductions to the lake at a scale that will reduce HABs and hypoxia.

Results from small watershed water-quality monitoring cannot be extrapolated directly to large watersheds to predict the magnitude or timing of cumulative management practice results at the mouths of much larger watersheds. A number of factors influence the fate and transport of nutrients as they move from small to large tributaries. Nutrient loadings at the outlet to Lake Erie are influenced by nutrient inputs along the entire length of the tributary from cumulative smaller watersheds, point sources, and nonpoint sources both with and without management practice interventions; in-stream chemical transformations; and varying rates of nutrient transport and delivery due to factors including storage (such as in sediment or groundwater) and episodic events (storm events). A benefit of focusing on the watershed scale (both small and large) is that it allows assessment of cumulative effects of multiple management practices integrated over space and time.

3.6 Focus on active, long-term monitoring sites

As described in section 2.3.3, conditions in Lake Erie and management practices in the drainage basin have changed substantially over recent years and continue to change due to invasive species and climate change. Due to these recent changes in the Lake Erie drainage basin, water-quality data are most informative for answering the case-study policy question if they were collected recently. Active monitoring sites that have been sampled consistently over long time periods are critical for evaluating long-term trends in water quality. Long records are necessary to be able to distinguish water-quality conditions caused by variations in weather patterns from those that are reflective of changes brought about by changes in management practices. Active monitoring sites with more than 5, 10, or 30 or more years of data are uncommon, but the information they provide to identify potential causes of water-quality change cannot be obtained from new and short-term (less than 5 years) monitoring sites.



State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Selected streams from U.S. Geological Survey, 2012, 1:1,000,000-scale digital data
Albers projection, NAD 1983

HUC8 watersheds from U.S. Department of Agriculture, February 2012, ftp://ftp.ftw.nrcs.usda.gov/wbd/

Figure 9. Watersheds of 1,000 square miles or greater that drain directly to Lake Erie and with more than 40-percent row-crop area. [Abbreviations: HUC, hydrologic unit code; %, percent]

4. Compilation and Screening of Water Data

Water-quality data are collected by at least 17 different organizations in the Lake Erie drainage basin, and there is no single data repository utilized by all of these organizations. In 2011 the USGS began assembling a national-scale multi-agency compilation of water-quality data with consistent data format to assist in conducting research on regional, multi-state, and national-scale water-resources issues called the National Data Aggregation (Argue et al. 2014). A subset of the National Data Aggregation covering the United States area of the Lake Erie drainage basin ("nutrient data set" in this report) served as the primary data set for assessing existing water-quality monitoring data in this case study. The nutrient data set also includes subsequent updates and additions made specifically for this project, resulting in part from suggestions by TAC members familiar with local monitoring programs. A complete multi-agency water-monitoring data set was a critical component for completing this case study, and obtaining and processing this data set required expertise, time, and resources. The nutrient data set was screened using a list of selected water-quality parameters identified by the TAC to provide an overview of available water-quality data relevant to management practice evaluation.

4.1 Water data types and sources

The nutrient data set consists of water-quality and associated hydrologic data collected by federal, state, and regional governmental agencies and non-governmental organizations. The compilation provides a "snapshot" of historic and current monitoring data available at the time of the project. The largest online data sources for the nutrient data set were the USEPA Storage and Retrieval (STORET) Data Warehouse (U.S. Environmental Protection Agency, 2014) and the USGS NWIS (U.S. Geological Survey, 2002 and U.S. Geological Survey, 2014b). STORET is a repository for water quality, biological, and physical data of known quality, submitted by state agencies, USEPA and federal agencies, universities, and others. Submission of data to STORET is voluntary, but STORET is the preferred mechanism for states to submit data to the USEPA, which allows states to remain eligible for Section 106 state assistance grants under the Clean Water Act (U.S. Environmental Protection Agency, 2003). The NWIS database contains data collected by USGS. Additional substantial effort was made to obtain monitoring records that were in electronic format but were not in NWIS or STORET by contacting individual agencies and organizations as well as locating data available in published reports.

The types of data targeted in the nutrient data set included water-quality records for surface water (streams and rivers), groundwater (wells), and aquatic biology (macroinvertebrates). Water-quality records from the sampling sites for surface water and groundwater consisted of results for inorganic, organic, and physical parameters, while the bioassessments included information on presence and diversity of macroinvertebrates at stream sites. Associated data important for interpreting water quality, or "metadata," were also included, such as site location, sampling protocols, laboratory methods, units of measurement, and data qualifiers, to the extent possible. The National Data Aggregation for stream macroinvertebrate communities is the first assemblage of such data which, for most state agencies, does not reside in the USEPA Modernized STORET database. Macroinvertebrate-sampling sites and years of record provided by state-government monitoring agencies for the nutrient data set appear in Table 5.

Although water-quality data for groundwater, which can be a long-term source of nutrients to streams were compiled, they were not evaluated further in this case study.

Table 5. Summary of macroinvertebrate (community structure) monitoring records collected by state-government agencies in the nutrient data set. Data records span the time period 1974 to 2011.

Organization name	Number of sites	Mean number of unique years sampled per site	Maximum number of unique years sampled per site
Ohio Environmental Protection Agency	2,142	1.5	13
Michigan Dept. of Environmental Quality	788	1.3	5
New York Dept. of Environment and Conservation	83	1.8	8
Indiana Dept. of Environment and Management	66	1.2	2
Pennsylvania Dept. of Environmental Protection	4	6.3	13
Total	3,083		
Overall Mean		1.5	8

Information was also compiled for continuous streamflow and groundwater-level monitoring sites, which provided the available period of record for the continuous data. Two primary types of ancillary spatial information were used. Spatial data that delineated hydrologic boundaries were used to compute the drainage areas for surface-water sites, and descriptive spatial data were used to delineate agricultural land use within the basin.

The time lags between submission of a data request and receipt of the data varied significantly, ranging from immediate data transfer to 6 months or more due to time required for laboratory analysis, quality-assurance checking and screening, and data entry. The most recent updates for data sets included in the nutrient data set range from late 2009 to early 2013.

After the data were received, they were processed, re-coded, and formatted to generate a combined data set in a uniform format that allowed the data to be queried, compared, and summarized. The data-processing tasks included consistent naming of the parameters and chemical species measured; uniform coding of data qualifiers and units of measurement (e.g. less-than values, mg/L); and documentation of the organization's field and laboratory comment codes. The compiled data were reviewed to identify and remove, as much as possible, duplicate records resulting from some agencies' data records residing in more than one database.

4.2 Tributary water-quality monitoring programs in the Lake Erie drainage basin

Monitoring programs in the Lake Erie drainage basin that are focused on assessing nonpoint agricultural sources and transport of nutrients in streams include governmental, academic, and non-governmental programs, which have been described and summarized elsewhere (Myers et al., 2000; Western Lake Erie Basin Partnership, 2009a, b; Lake Erie Improvement Association, 2012; Ohio Lake Erie Phosphorus Task Force, 2013; International Joint Commission, 2014).

The Heidelberg Tributary Loading Program operated by the National Center for Water Quality Research at Heidelberg University (National Center for Water Quality Research, 2013a), was initiated in 1974 and is an important historical and ongoing stream-monitoring program for nutrients and other water-quality constituents in the Lake Erie drainage basin. The Heidelberg Tributary Loading Program, conducted in partnership with other organizations, has collected daily nutrient-monitoring data for 20 or more years at many streamgages and represents an important program regionally for documenting long-term trends and daily fluctuations in concentrations and loads of nutrients in streams.

In 2003 the USDA Agricultural Research Service (ARS) selected the St. Joseph River watershed in Indiana, Michigan, and Ohio, and a major tributary to the Maumee River, as an ARS Benchmark Research Watershed. This detailed monitoring through the USDA's Upper Cedar Creek Conservation Effects Assessment Project (CEAP) includes daily monitoring of climate, water quality, and streamflow since 2004 at 16 to 20 sites ranging from field scale to watershed scale in row-crop areas (U.S. Department of Agriculture, 2014b and 2014c).

At least four new nutrient-monitoring programs were initiated between 2011 and 2013 in Lake Erie streams, including the Great Lakes Restoration Initiative (GLRI) tributary monitoring and edge-of-field studies (White House Council on Environmental Quality, 2010; U.S. Geological Survey, 2014e), an ODNR-USGS Ohio Water Science Center cooperative program focused on tributaries in the Maumee River Basin, and USDA research studies. The GLRI is a broad-scoped, federal- and state-partnered program that includes the Great Lakes region, and has delineated several Lake Erie tributary basins as priority areas for nutrient monitoring (White House Council on Environmental Quality, 2010). Because of their recent establishment, some of the data from these studies were not included in the nutrient data set compiled for this case study; some of these programs and monitoring-site locations were considered in identifying data gaps and future monitoring needs (Chapter 7). Additional monitoring sites are currently under development by OH EPA and USGS, and a description of these new monitoring sites can be found in the addendum to this report (Betanzo et al., 2015).

4.3 Screening of available monitoring data

The TAC prepared a screening list of water-quality parameters (Table 6) relevant to nutrient assessment of tributaries to Lake Erie, and the list is derived in part from the recommended nutrient-monitoring requirements (Caffrey et al., 2007) for the National Monitoring Network for U.S. Coastal Waters and Tributaries (National Water Quality Monitoring Council, 2006). This screening list was used to identify available monitoring data and to assess the completeness of metadata in the nutrient data set. Seventeen organizations provided data for one or more of these screening parameters in tribtuary-water samples in the Lake Erie drainage basin (Table 7), including federal, state, academic, and non-profit organizations. A summary of records in the nutrient data set, by parameter, is shown in Table 8.

The data-processing steps provided insight into several challenges in building a multi-agency data set, and the screening process revealed challenges in interpreting records, including the existence of duplicate records (both within and among agencies) and missing metadata elements. For many parameters, including nutrients, there was considerable variation and inconsistency in naming conventions, both within a single agency's data set and between agencies' data sets. In the source-data records, fraction analyzed, chemical species, and units of concentration might be listed only in the parameter name, or might be separated and listed in up to four separate data columns (name, fraction, species, or units). Sometimes metadata essential to accurately identify a chemical parameter were not listed at all. For example, 11,460 nutrient records had unspecified sample fraction (Table 8), which is required in order to identify specific nutrient parameters, including TP and DRP, and to perform quantitative interpretive analyses such as evaluating trends.

Table 6. Screening parameters selected by the Technical Advisory Committee for querying the nutrient data set for the Lake Erie drainage basin.
[Abbreviations: NA, not applicable; mg/L, milligrams per liter; as N, as nitrogen; mg/m^2, milligrams per square meter; ft^3/sec, cubic feet per second; as P, as phosphorus; #cells/mL, number of cells per milliliter; uS/cm, microsiemens per centimeter; ft, feet; NTU, Nephelometric Turbidity Units; ft/sec, feet per second; deg C, degrees Celsius]

Parameter name	Fraction measured	Typical units of measurement
Ammonia	dissolved	mg/L, as N
Ammonia	total	mg/L, as N
Chlorophyll a	NA	mg/L; mg/m^2
Discharge (streamflow)	NA	ft^3/sec
Dissolved oxygen	NA	mg/L
Inorganic carbon	dissolved	mg/L
Nitrate	dissolved	mg/L, as N
Nitrate	total	mg/L, as N
Nitrite	dissolved	mg/L, as N
Nitrite	total	mg/L, as N
Nitrite plus nitrate	dissolved	mg/L, as N
Nitrite plus nitrate	total	mg/L, as N
Organic carbon	dissolved	mg/L
Organic carbon	total	mg/L
Orthophosphate (DRP)	dissolved	mg/L, as P

Parameter name	Fraction measured	Typical units of measurement
Orthophosphate	total	mg/L, as P
Particulate carbon	NA	mg/L
Particulate nitrogen	NA	mg/L
Periphyton, biomass	NA	g/m^2
pH	NA	standard units (SU)
Phytoplankton	total	#cells/mL
Silica	dissolved	mg/L
Specific conductance	NA	uS/cm
Stage (stream, water level)	NA	ft above sea level
Suspended sediment (concentration)	NA	mg/L
Total Kjeldahl nitrogen (TKN)	dissolved	mg/L, as N
Total Kjeldahl nitrogen (TKN)	total	mg/L, as N
Total dissolved nitrogen	dissolved	mg/L, as N
Total nitrogen	total	mg/L, as N
Total phosphorus	dissolved	mg/L, as P
Total phosphorus (TP)	total	mg/L, as P
Turbidity	NA	NTU
Velocity (stream)	NA	ft/sec
Water temperature	NA	deg C

Table 7. Sources of surface water quality data in the nutrient data set for the screening parameters identified by the Technical Advisory Committee, ranked by the number of monitoring sites. Data records span the period 1943 through 2013, with date ranges varying by organization, site, and parameter; record end dates range from 2009 to 2013 among the primary organizations actively collecting data.

Organization name	Organization type	Number of monitoring sites	Number of water- quality records	Number of parameters measured
Ohio Environmental Protection Agency	State	1,296	53,586	12
U.S. Geological Survey, National Water Information System	Federal	873	167,419	25
Indiana Department of Environmental Management	State	183	26,516	13
Michigan Department of Environmental Quality	State	175	15,745	14
Pennsylvania Fish and Boat Commission	State	137	425	4
Northeast Ohio Regional Sewer District	State	121	31,167	15
Saint Joseph River Watershed Initiative	Regional	48	48,270	8
U.S. National Park Service	Federal	31	14,617	9
Hoosier Riverwatch	Volunteer	29	845	8
U.S. Department of Energy	Federal	23	92	4
University of Michigan-Ann Arbor (Dr. Nathan Bosch)	Academia	22	3,173	13
New York Department of Environmental Conservation	State	17	4,939	15
Heidelberg University, National Center for Water Quality Research	Academia	12	630,938	7
U.S. Environmental Protection Agency	Federal	11	78	8
Pennsylvania Department of Environmental Protection	State	6	2,337	14
National Oceanic and Atmospheric Administration, National Estuarine Research Reserve System	Federal	5	14,247	6
U.S. Department of Agriculture, Agricultural Research Service	Federal	16	176,448	8

Table 8. Summary of records in the nutrient data set by the screening parameters selected by the Technical Advisory Committee, and ranked by number of sampling sites. Data records span the period 1943 through 2013, with date ranges varying by organization, site, and parameter; record end dates range from 2009 to 2013 among the primary organizations actively collecting data.
[**Abbreviations:** NA, not applicable]

Parameter name	Fraction analyzed	Number of data sources	Number of sites	Number of data records
Parameters, specified fraction				
Specific conductance	NA	14	2,748	89,536
Nitrite plus nitrate	Total	7	1,891	22,040
Total phosphorus (TP)	Total	11	1,890	168,247
Ammonia	Total	9	1,858	23,450
Total Kjeldahl nitrogen (TKN)	Total	8	1,765	135,452
Nitrite	Total	6	1,662	16,700
pH	NA	15	1,609	62,958
Water temperature	NA	13	1,501	37,268
Dissolved oxygen	NA	11	931	25,880
Streamflow	NA	8	813	131,258
Turbidity	NA	9	767	24,563
Nitrate	Dissolved	4	417	9,260
Dissolved Reactive Phosphorus (DRP)	Dissolved	9	411	137,677
Organic carbon	Total	7	386	6,739
Nitrate	Total	5	381	6,672
Silica	Dissolved	3	354	46,622
Orthophosphate	Total	5	195	1,942
Ammonia	Dissolved	5	172	25,284
Nitrite plus nitrate	Dissolved	4	164	147,502
Total nitrogen	Total	4	134	4,691
Nitrite	Dissolved	2	132	4,506
Organic carbon	Dissolved	6	129	904

Parameter name	Fraction analyzed	Number of data sources	Number of sites	Number of data records
Stage	NA	2	123	3,203
Total phosphorus	Dissolved	4	118	1,953
Suspended sediment concentration	NA	1	106	23,256
Chlorophyll a	NA	6	91	3,136
Total Kjeldahl nitrogen (TKN)	Dissolved	1	74	20,876
Particulate nitrogen	NA	3	57	420
Total nitrogen	Dissolved	2	36	445
Inorganic carbon	Dissolved	1	16	21,340
Periphyton	NA	1	13	236
Phytoplankton	Total	1	10	376
Velocity	NA	1	9	78
Particulate carbon	NA	1	1	6
				1,204,476
Parameters, unspecified fraction:				
Nitrate	not listed	2	41	588
Nitrite	not listed	2	36	509
Orthophosphate	not listed	3	33	2,312
Ammonia	not listed	3	20	2,623
Nitrite plus nitrate	not listed	2	18	2,708
Total Kjeldahl nitrogen (TKN)	not listed	2	18	2,720
Total:				**11,460**

TP and DRP records were assessed in further detail. The available TP and DRP data in the nutrient data set were provided by the 14 monitoring organizations shown in Table 9. Figure 10 summarizes the number of sites in the Lake Erie drainage basin, where TP and DRP are being monitored, by monitoring organization.

Table 9. Organizations included in the nutrient data set that collected data for total phosphorus (TP) and dissolved reactive phosphorus (DRP).

Organization name	Organization abbreviation	Provided TP or DRP data records
Heidelberg University, National Center for Water Quality Research	OH HDLBG	TP, DRP
Hoosier Riverwatch	IN HRW	TP
Indiana Department of Environmental Management	IN DEM	TP
Michigan Department of Environmental Quality	MI DEQ	TP
National Oceanic and Atmospheric Administration, National Estuarine Research Reserve System	NOAA NERRS	DRP
New York Department of Environmental Conservation	NY DEC	TP, DRP
Northeast Ohio Regional Sewer District	OH NEORSD	TP, DRP
Ohio Environmental Protection Agency	OH EPA	TP, DRP
Pennsylvania Department of Environmental Protection	PA DEP	TP, DRP
Saint Joseph River Watershed Initiative	SJRWI	TP
University of Michigan-Ann Arbor	UNIV_MI	TP, DRP
U.S. Department of Agriculture, Agricultural Research Service	USDA ARS	TP, DRP
U.S. Geological Survey, National Water Information System	USGS-NWIS	TP, DRP
U.S. National Park Service	US NPS	TP

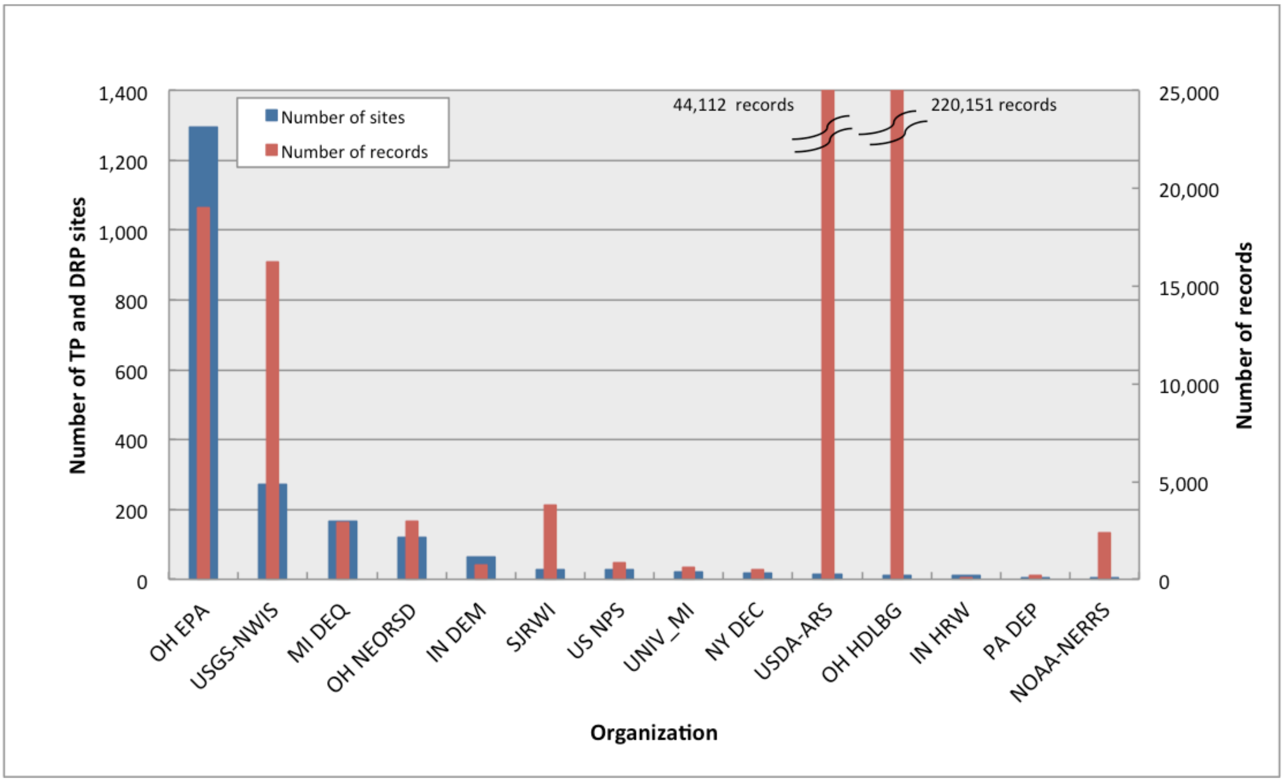


Figure 10. Number of sites in the nutrient data set monitoring total phosphorus (TP) and dissolved reactive phosphorus (DRP), in the Lake Erie drainage basin. Monitoring organization abbreviations are defined in Table 9.

Figure 11 shows all sites within the Lake Erie drainage basin that have either TP or DRP data available in the nutrient data set. TP data were available for more than five times the number of sites (n=1,890) for which DRP data were available (n=411). The majority of sites where DRP is monitored also are being monitored for TP.

Phosphorus is a complex element and the chemical properties of phosphorus, type of species of interest, naming conventions, sampling methods, and analytical techniques for phosphorus add to the complexity; these complexities add documentation requirements for phosphorus data if those data are to be shared, combined, analyzed, and compared across different organizations. In addition to the more common terms for DRP mentioned in section 2.3.2, DRP is also equivalent, or nearly equivalent, to dissolved orthophosphate, soluble reactive phosphorus (SRP), dissolved inorganic phosphorus (DIP), filterable reactive phosphorus (FRP), total dissolved phosphorus (TDP) and reactive phosphorus for a filtered sample (Jarvie et al., 2002). The term "total" for describing phosphorus water-quality analyses has been used to refer both to the sum of the analyzed chemical forms (or species) of phosphorus (e.g. inorganic plus hydrolysable forms) and to the sum of sample fractions analyzed (e.g. the sum of dissolved phosphorus in a filtered water sample plus particulate phosphorus). The potential double meaning of the term 'total' results in the need for additional descriptive information in the water-quality record in order to correctly interpret the sample analysis, including clear identification of the chemical forms analyzed and clear identification of the sample fraction analyzed. These details were not always available in the nutrient data sets compiled for this project.

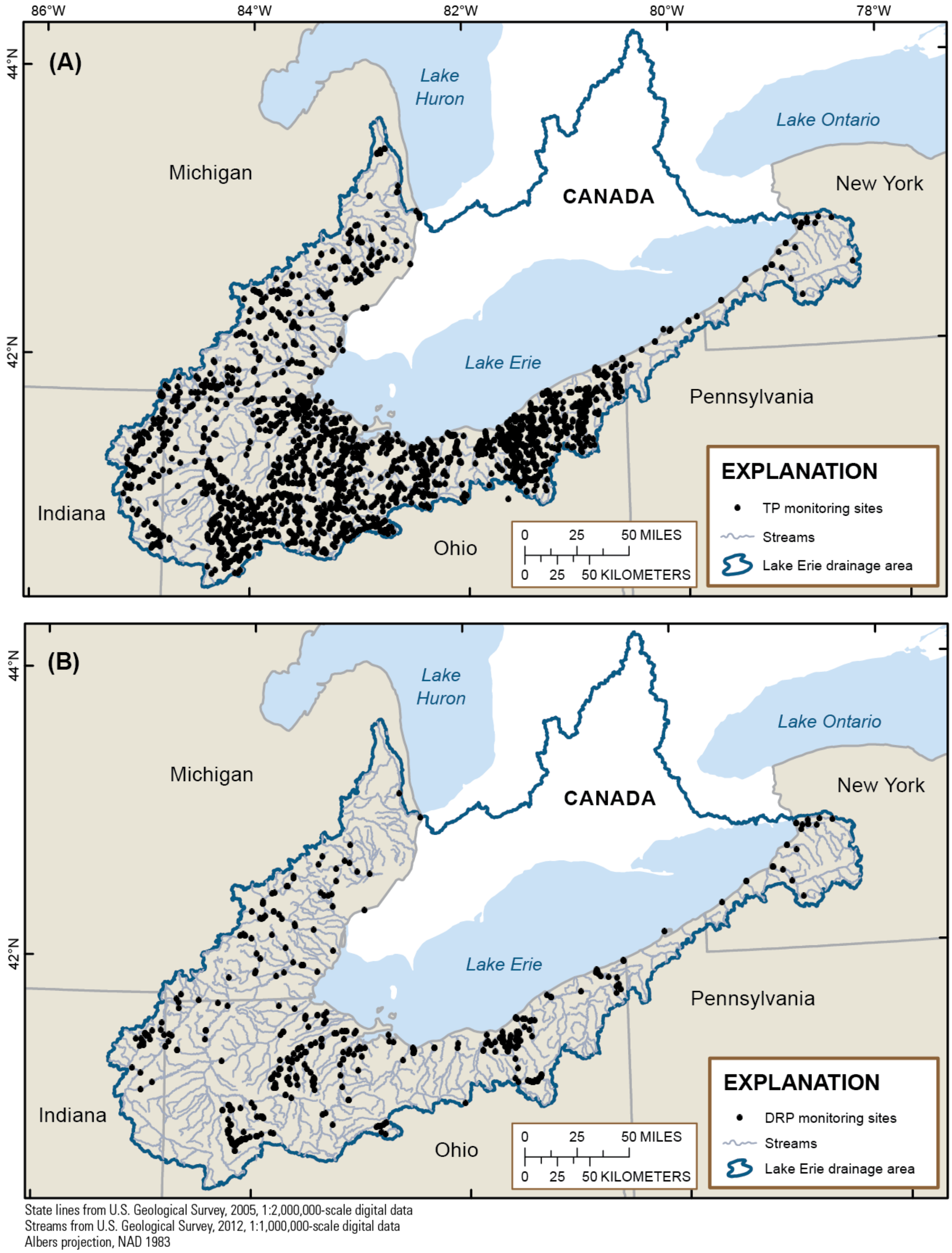


Figure 11. Monitoring sites in the Lake Erie drainage basin with water-quality records for (A) total phosphorus (TP) or (B) dissolved reactive phosphorus (DRP).

Regarding sampling methods, the choice of sampling method can affect the concentration measured in a sample. Dissolved and adsorbed phosphorus are often not in equilibrium in runoff and stream waters, so the sampling method plays a major role in determining what occurs in the sample bottle. This also means the distribution between the two phases can change in the sample bottle during the time between bottle filling and its retrieval, transport, and laboratory submission. Further, automated methods that are used to take water samples from stormflows are usually negatively biased—underestimate the true concentration— in terms of sediment concentrations (Roseen et al., 2011). Technologies exist to overcome or account for these issues but they are costly to implement. As a result, it is critical for monitoring organizations to document all details of sample collection and handling to determine whether the sampled records collected by different organizations or by different methods are comparable between sites and over time. The National Field Manual for the Collection of Water-Quality Data (U.S. Geological Survey, 2006) provides a peer-reviewed and up-to-date summary of sampling methods.

The lack of metadata further limited the ability to understand phosphorus records. Any records for which the specific parameter could be discerned as TP or DRP were analyzed in this case study. There was only about 1 DRP monitoring site for every 5 TP monitoring sites in the nutrient data set (see Figure 11). Water-quality records with unspecified fraction further limited the ability to identify DRP data. For example, the sites with the unspecified fraction, orthophosphate, which were lacking metadata necessary for classifying as either DRP or total orthophosphate, represented 5 percent of the combined DRP and total orthophosphate monitoring sites, or 8 percent of all DRP monitoring sites.

Missing metadata for records used in the case-study analysis included fraction of water sample analyzed (filtered versus whole water samples), laboratory method, species reporting units (as P, or as PO_4), units of measurement (mg/L, ug/L), and for samples with no detectable concentration (less-than values), the specification of a laboratory reporting limit or detection limit. Detection limits can change over time, and by sample batch, so specification is important for analyzing data within the same monitoring program in addition to comparing across different programs. With the exception of the power analysis discussed in section 5.4, these additional metadata were not used as a filter for including data records in the case-study evaluation, as it was assumed further communication with the source agencies might produce these needed metadata. However, if not available, samples lacking this critical information are of limited use for many types of interpretive analyses such as comparing concentrations among sites, evaluating changes over time (trend analysis), and determining sampling requirements to assess change. For example, if metadata information is missing on laboratory units used for reporting chemical concentrations (e.g. as N, as NO_3, as P, as PO_4) it is not possible to accurately quantify the measured concentrations.

4.4 Matching water-quality sites with streamgages

As discussed in section 2.4.3, streamflow measurements are needed to support nutrient load calculations. To examine availability of streamflow data, water-quality sites were matched with nearby USGS streamgages using Arc Geographic Information System (ArcGIS) software and National Hydrography Dataset (NHD) Plus data (U.S. Environmental Protection Agency and U.S. Geological Survey, 2014; and McKay et al., 2012). The term streamgage refers to an active, continuously-functioning measuring device in the field for which a mean daily streamflow is computed or estimated and quality assured for at least 355 days of a water year (U.S. Geological Survey, 2014a). The term “water year” is defined as the 12-month period beginning October 1 for any given year through September 30, of the following year (U.S. Geological

Survey, 2015). The streamgage matching procedure involved comparing differences between the drainage areas of the water-quality site and nearby streamgage(s). The drainage areas for both the water-quality sites and the streamgages were estimated by associating each point with the nearest NHDPlus "flowline" and assigning it the drainage area of that stream segment. If a water-quality site could not be assigned to an NHDPlus flowline, it was not possible to match the site with a streamgage. This could result from inaccurate site-location coordinates provided from the source agency, or from there being no mapped NHDPlus flowline in the vicinity of the water-quality site (for example, for water-quality sites on very small streams).

The drainage-area difference ratio, in percent, was calculated as the absolute value of the difference between the respective drainage areas of the water-quality site and the streamgage divided by the drainage area of the water-quality site. When multiple streamgages were located in the vicinity of a water-quality monitoring site, the streamgage associated with the smallest drainage area difference ratio was selected as the best match for the water-quality site. All water quality sites that were considered to be co-located with a streamgage had a percent drainage area difference of less than or equal to 5 percent; however, the vast majority of water-quality monitoring sites with multi-year records were located at streamgages, and all of the sites that were analyzed in the power analysis (section 5.4) were located at or in close proximity to streamgages.

4.5 Determination of watershed characteristics and priority watersheds

The case-study analysis required information on watershed drainage area (size) and watershed land use upstream of the monitoring sites in the nutrient data set. These watershed characteristics were not requested from the data providers as part of the metadata in the nutrient data set.

For USGS monitoring sites, watershed drainage area was obtained from the NWIS online Web portal (U.S. Geological Survey, 2014c). For sites monitored by Heidelberg University and the Northeast Ohio Regional Sewer District, drainage area was supplied by the data provider. For other monitoring sites, drainage area was determined using ArcGIS V10 software (Esri, Inc., 2012) and the NHDPlus V2 data set (U.S. Environmental Protection Agency and U.S. Geological Survey, 2014; McKay et al., 2012).

Land use, as percentage of watershed in row crops, was determined using the database Geospatial Attributes of Gages for Evaluating Streamflow (GAGES) (Falcone et al., 2010; and James A. Falcone, U.S. Geological Survey, written commun., 2013), which was available for some of the USGS sites in the nutrient data set, and was also applied to non-USGS sites co-located with these USGS gages. Land use in row crops for the HUC-8 and HUC-12 delineated basins used in the spatial assessment was determined using ArcGIS and the National Agricultural Statistics Service Cropland Data Layer (U.S. Department of Agriculture, 2012a).

Data from the USGS model "SPAtially-Referenced Regressions On Watershed Attributes" (SPARROW) and applications of the USDA SWAT model were used to identify TP and DRP yields in small watersheds in the Lake Erie drainage basin (Robertson and Saad, 2011; U.S. Geological Survey, 2013; Scavia et al., 2014). Both SPARROW and SWAT are widely used models for estimating nutrient transport and for evaluating land-use effects on stream-nutrient loading.

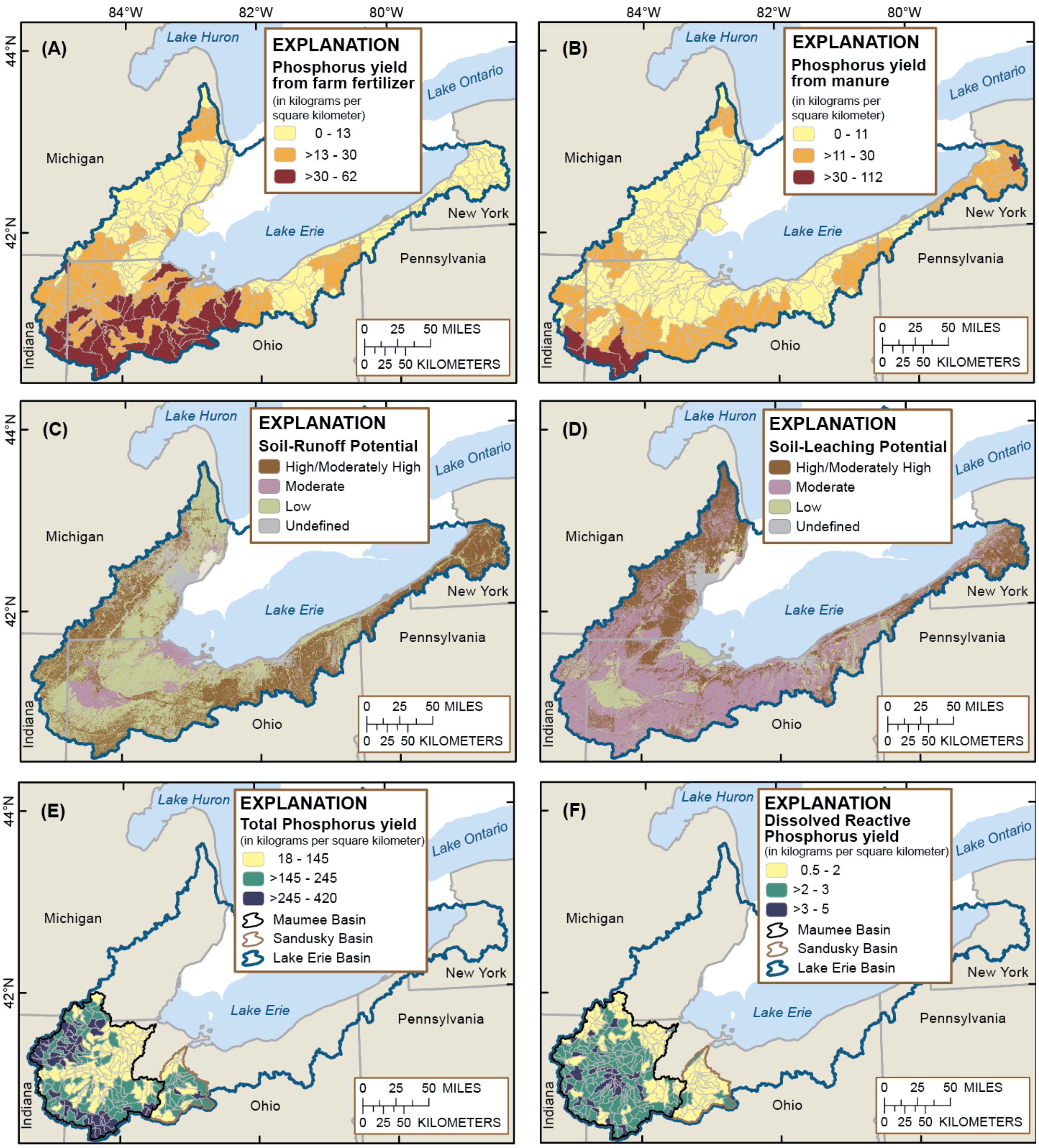
Figure 12 (A) and (B) respectively show SPARROW TP yield from fertilizer and manure by watershed. The manure TP yields include both confined and unconfined manure. The SPARROW model links a water-quality model to a network of monitoring stations and models the sources, transport, and fate of nutrients in watersheds based on characteristics of the sources, land, and stream channels (Preston et al., 2009). SPARROW associates watershed TP yield with source categories of fertilizer, manure, nonpoint sources, atmospheric, forested land, and urban and open areas. DRP assessments are not yet available for the Lake Erie drainage basin through SPARROW.

The USDA CEAP data analyzing soil-runoff potential and soil-leaching potential in the Lake Erie drainage basin were used to identify watersheds vulnerable to nutrient and sediment loss as shown in Figure 12 (C) and (D) (Dean Oman, U.S. Department of Agriculture, written commun., 2014). The methods used for developing these data sets using Soil Survey Geographic database (SSURGO) data are described in Lund et al. (2011). The High and Moderately High areas identified in Lund et al. (2011) were used to identify priority areas for small watershed monitoring as shown in Figure 13.

Scavia et al. (2014) presented TP and DRP yield data from calibrated SWAT models described in Bosch et al. (2013) and Bosch et al. (2011) to show average-annual TP and DRP yields at the outlets of subwatersheds in the Maumee, Sandusky, and Cuyahoga watersheds. The SWAT model results (Figure 12 (E) and (F)) show TP and DRP yields, by subwatershed, including all phosphorus sources within the watershed, including point sources. SWAT is a process-based model that predicts changes in water quality in response to changes in practices on the land (Arnold et al., 2012). The SWAT results presented here do not break out TP or DRP yields into individual source categories.

The ArcGIS (Esri, Inc., 2012) natural breaks tool was used to divide both the SPARROW and SWAT watershed yields into three categories. The highest category from each of Figure 12 (A) through (F) appears in Figure 13 to identify priority areas for small watershed monitoring. Figure 13 highlights the areas where these high characteristics overlap for further prioritization. The results of these assessments are presented in Chapters 6 and 7.

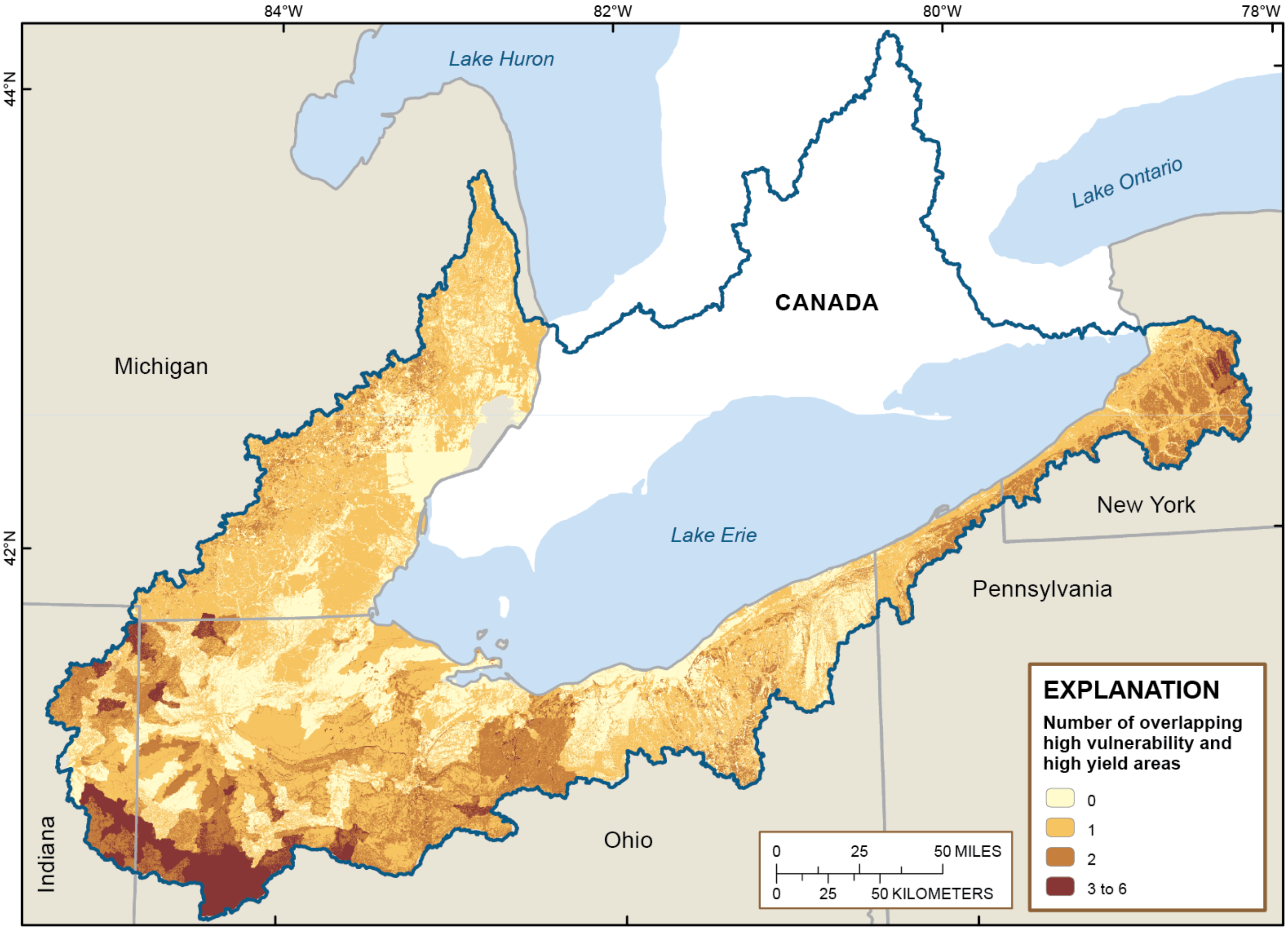
Two different data sets were used to examine the spatial coverage of tile drains in the Lake Erie drainage basin. The tile-drainage area in Figure 14(A) is based on data from a 1992 USDA county-level survey on the use of tile drains, converted to spatial data by USGS, and represents actual tile drainage in place more than 20 years ago. The county totals were apportioned to a grid of 30-meter cells in agricultural areas, summed for each NHDPlus (v1) catchment (Wieczorek and LaMotte, 2010) and then normalized by the area of the catchment. Figure 14(B) is derived from SSURGO soil data (Dean Oman, Natural Resources Conservation Service, written commun., 2014) combined with data from the Cropland Data Layer (U.S. Department of Agriculture, 2012a) using soil types to identify areas requiring artificial drainage to remove excess water. SSURGO's hydrologic soil groups B/D, C/D, and D have moderate to slow infiltration rates, and tile drains are likely in use if row crops are growing in these soil types (Mark Tomer, U.S. Department of Agriculture, written commun., 2014; Jaynes and James, 2007; Sugg, 2007). Figure 14(B) essentially shows locations where tile drains might be needed on the basis of soil type and areas of row-crop land use. Neither of these portrayals of tile drainage has been field-verified to confirm their accuracy, but they provide regional-scale estimates of tile drainage in the Lake Erie drainage basin. More accurate, up-to-date, and field-verified identification of artificial-drainage extent and density would be valuable in the Lake Erie drainage basin.



State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Albers projection, NAD 1983

(A, B) Phosphorus yield from Saad, D.A., et al. (2011)
(C, D) Soil vulnerability from Dean Oman, U.S. Department of Agriculture, written commun., 2014 [Original source: Lund, et al. (2011)]
(E, F) Phosphorus yield from Scavia, D., et al. (2014)

Figure 12. Watershed characteristics for prioritizing water-quality monitoring sites to detect the effectiveness of agricultural management practices for reducing phosphorus transport to streams: (A) SPARROW total phosphorus yield from farm fertilizer, (B) SPARROW total phosphorus yield from manure, (C) USDA soil-runoff potential, (D) USDA soil-leaching potential, (E) SWAT total phosphorus yield, and (F) SWAT dissolved reactive phosphorus yield.
[Abbreviations: kg/km2, kilograms per square kilometer; SPARROW = SPAtially-Referenced Regressions On Watershed Attributes; SWAT= Soil and Water Assessment Tool]



State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Albers projection, NAD 1983

Soil vulnerability from Dean Oman, U.S. Department of Agriculture, written commun., 2014 [Original source: Lund, et al. (2011)]
Phosphorus yields from Saad, D.A., et al. (2011); and from Scavia, D., et al. (2014)

Figure 13. Overlapping high soil vulnerability and high phosphorus yield priority areas for small watershed monitoring to measure effectiveness of agricultural management practices.

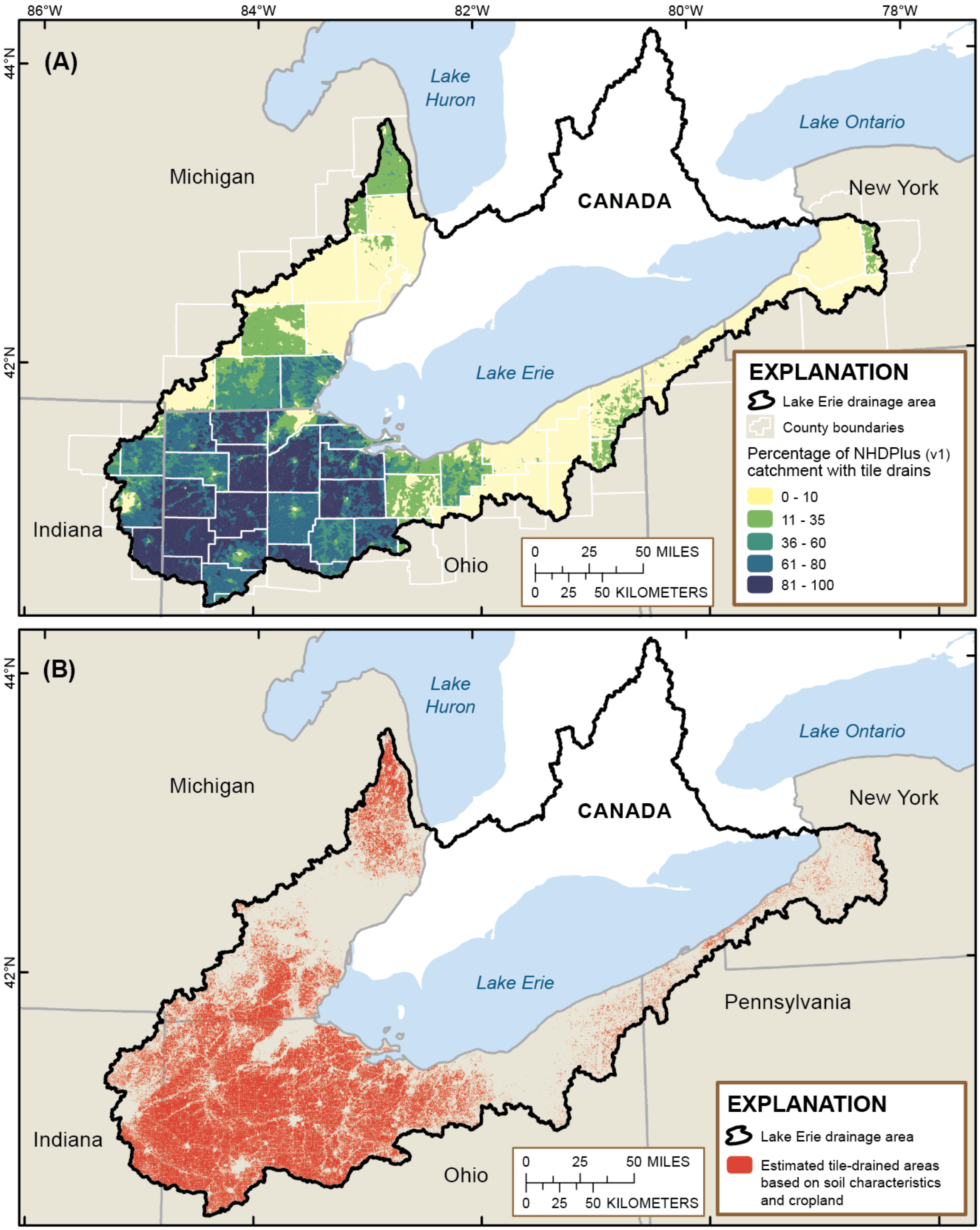


Figure 14. Tile-drain estimates using (A) USGS artificial drainage and (B) USDA SSURGO hydrologic soil groups. [Abbreviations: NHDPlus v1, National Hydrography Dataset Plus version 1; SSURGO, Soil Survey Geographic Database]

5. Quantities of Water Data Needed to Answer the Case-Study Policy Question

This chapter defines the quantities of water data needed to answer “How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?” given the most important data types discussed in Chapter 3 and summarized in Table 4. The overarching objectives for answering this question are 1) to determine how much phosphorus loadings are changing over time (identify temporal trends), and 2) to determine whether those changes resulted from agricultural management practices or other factors (identify cause of change). This chapter quantifies the water-quality data needed to achieve the first objective. Once the needed water-quality data are available to identify change, additional data and analysis are needed to achieve the second objective correlating water-quality change with agricultural management practices or other factors. These two objectives, plus two testable hypotheses, were used to develop a sampling plan that defines the types of water data, statistical methods to interpret monitoring results, frequency and duration of monitoring, location criteria (watershed characteristics), and number of monitoring sites needed to answer the case-study policy question.

5.1 Hypotheses

Clear identification of monitoring objectives and the hypotheses to be tested is critical in determining a successful water-quality monitoring design (Sanders and others, 1983; Gilbert, 1987; U.S. Environmental Protection Agency, 2002 and 2006). TAC guidance and published literature were used to develop two hypotheses for this case study and to identify the targeted watershed population in the Lake Erie drainage basin. An appropriate monitoring design was identified for testing the hypotheses to which the existing monitoring data could be compared. One hypothesis was developed for the large watershed scale and one for the small watershed scale:

- Implementation and maintenance of appropriate agricultural management practices throughout large agricultural watersheds (>=1,000 square miles) that drain directly to Lake Erie will reduce agricultural nonpoint annual loadings of TP and/or DRP in tributaries draining the watershed.
- Implementation and maintenance of appropriate agricultural management practices throughout small agricultural watersheds (<= 50 square miles) will reduce agricultural nonpoint annual loadings of TP and/or DRP in streams draining the watershed.

These hypotheses are subsequently referred to as the ‘large watershed hypothesis’ and the ‘small watershed hypothesis’, respectively.

The following core set of water data is needed to achieve the first overarching objective for answering the case-study policy question, identify temporal trends in water quality, for each size watershed:

- TP and DRP data collected with sufficient frequency and duration to calculate and detect trends in both flow-adjusted concentrations and load, and

- An active streamgage near the water-quality monitoring site, to calculate loads and flow-adjusted concentrations.

Constraints on the types of monitored watersheds and approach to implementation are expected to maximize the ability to statistically discern such trends:

- Locate monitoring sites in agriculture-dominated high-phosphorus yield watersheds and watersheds vulnerable to soil leaching and soil runoff,
- Implement agricultural management practices at appropriate scales and locations throughout monitored watersheds, and
- Maintain consistency in sampling design between watersheds to allow comparison among watersheds.

To determine whether observed trends in phosphorus concentrations and loads result from the implementation of agricultural management practices (the second objective), additional data and analysis are needed for monitored watersheds to determine which factors are playing a role in observed concentration or load trends (see sections 2.4.1 and 2.4.2):

- Annual data documenting where, when, and what type of agricultural management practices have been installed in the watershed and how they have been maintained,
- Data on overall changes in land use and market-driven changes in agriculture (e.g., choice of crops, cropping methods, and fertilizer application),
- Data on changes to urban and point source nutrient loadings including pollution management actions,
- Data on climate conditions including precipitation and streamflow, and
- Data on changes in land drainage technology and hydrologic response.

The monitoring plan design (sampling frequency, duration, and location and number of monitoring sites) for testing each hypothesis must be appropriate for the statistical approach for analyzing the data, for the watershed scale being evaluated, and adequate for calculating and estimating trends in load. Studies have shown that sampling frequency, sampling strategy, load-calculation method, watershed size, and the behavior of chemical species affect the accuracy, bias, and precision of loading estimates (Richards and Holloway, 1987). Attempts to derive quantitative relationships to predict sampling frequency and pattern required to attain a specified level of precision in load calculations have not been successful (Richards and Holloway, 1987). In the absence of quantitative approaches for defining the monitoring plan, this case study relied on the TAC recommendations, available literature, and an illustrative power analysis to design the monitoring plan for testing the two hypotheses.

Any analysis for trends in load must be built on information about concentrations and streamflow because load is the product of concentration and streamflow. Analysis of loading trends is accomplished through the use of any one of a number of statistical methods that use a combination of concentration and streamflow data. These methods use a variety of techniques (which include flow-adjustment or flow-normalization) to increase the ability to detect trends by removing the variability of water quality that is due to variation in streamflow and season.

Concentrations, streamflow, and the relation between concentration and streamflow can vary seasonally due to both climatic and land-use patterns. Estimation of annual loads and water-quality statistics requires sampling throughout the year to capture seasonal variability, and to determine accurate annual statistics and loading estimates. Trend analyses should focus on flow and seasonally-adjusted concentrations or loads, to remove the extraneous variations in concentration caused by changing streamflow and seasonal effects, in order to more clearly and accurately identify water-quality trends and their causes.

Flow-adjusted concentrations and load are related, but trends in load cannot be directly inferred from trends in flow-adjusted concentration. Large changes in concentrations that take place at infrequent high streamflow conditions can be important contributors to trends in loads because a large part of the annual load is transported during these events, but such changes will have a limited effect on estimated trends in flow-adjusted concentrations. For a quantitative example of this concept, the reader is referred to Figures 40 and 41 in Hirsch and De Cicco (2015), which demonstrates how concentration-flow relationships can change over time (Figure 40 is shown here as Figure 15). These graphs show minimal difference between the DRP concentrations during low to medium flows (10 to 50 m^3/s), in two examined years, but the DRP concentrations in 2010 are approximately double the 1988 concentration during high flows (200 to 250 m^3/s) for the Maumee River at Waterville, Ohio, during May and August. These differences would result in a higher calculated annual load for 2010 compared to 1988, assuming consistency in this pattern during other months of the year. Trends in concentration and load can differ both in magnitude and sign (increasing/decreasing) for the same set of data. This disparity between potential trends in load and flow-adjusted concentration was an important consideration in the development of sampling-frequency strategies recommended in this report (section 5.3). This disparity

Concentration: Mass per unit volume. TP and DRP concentrations are typically measured as mg/L and are generally positively correlated with streamflow, which can vary significantly from year to year.

Flow-adjusted (and seasonally adjusted) concentration: A measure of concentration for which the influence of streamflow and seasonal patterns has been reduced or removed (Helsel and Hirsch, 2002), usually by modeling.

Load (or Flux): Total mass of a constituent delivered to some location in a specific period of time. Annual TP and DRP loads to Lake Erie are typically estimated using periodic measurements of concentration and continuous records of streamflow and are typically expressed as metric tons/year.

Yield: Mass per time per unit area, calculated as stream load divided by contributing drainage area. Used to provide a standardized metric for comparing nutrient export among watersheds of differing sizes or across a broad region.

between concentration trends and load trends is discussed in detail by Moyer et al. (2012) and Hirsch and De Cicco (2015).

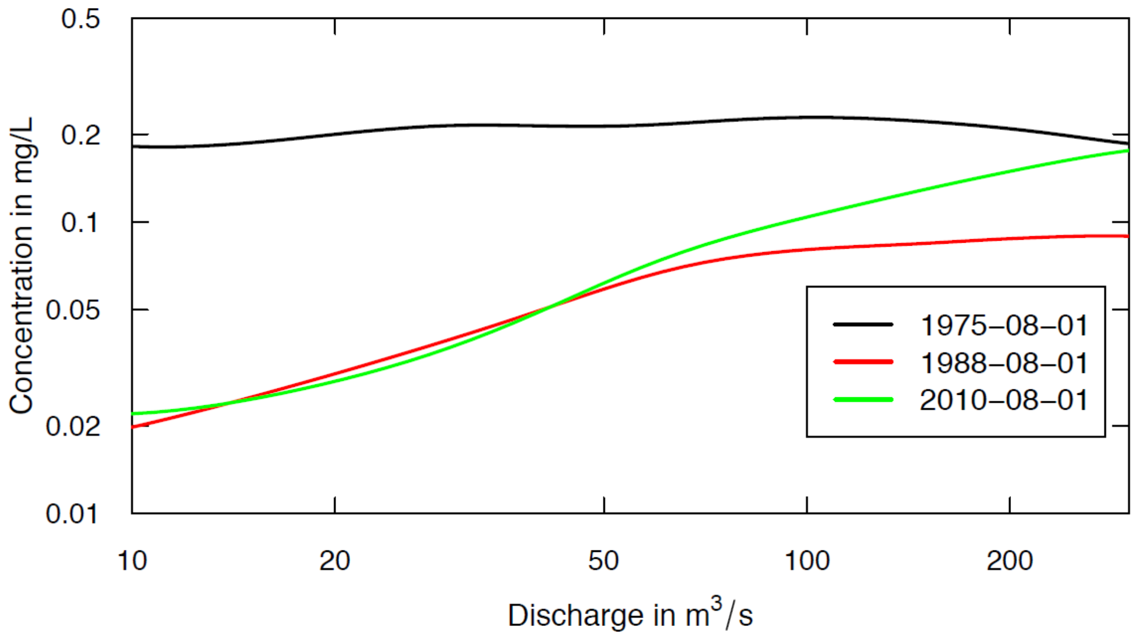


Figure 15. Plot produced by the plotConcQSmooth function for the relation between concentration of dissolved reactive phosphorus and discharge, centered on August 1 for three different years for the Maumee River at Waterville, Ohio (Figure 40 from Hirsch and De Cicco (2015)).

TP and DRP concentration and load trends during recent decades at two long-term monitoring sites in the Lake Erie drainage basin (Table 10) provide examples of change that are plausible for detection through future monitoring. These load estimates and flow-normalized concentration trends were determined using weighted least squares regression (Hirsch and De Cicco, 2015), water-quality data provided by Heidelberg University (National Center for Water Quality Research, 2013a), and streamflow data provided by the USGS (U.S. Geological Survey, 2014b). Table 10 summarizes the average trends for the Maumee River and Rock Creek for approximately 10- and 20-year intervals. TP concentrations decreased in the Maumee River by 33 percent and loads decreased by 21 percent between 1977 and 2000, but both concentrations and loads decreased less than 1 percent between 2000 and 2012. In Rock Creek, TP concentrations increased by 12 percent from 1984 to 2005, and TP loads increased by 45 percent over the same time period. Substantial DRP increases were measured in both load and concentration for both the Maumee River and Rock Creek, with DRP loads increasing in Rock Creek by 159 percent between 1992 and 2012. Future decreases in TP and DRP resulting from agricultural management practices will have to reverse these increases and may yield decreasing trends with small overall changes that are difficult to detect. The magnitude of anticipated change is discussed in section 5.4.

Table 10. Recent flow-normalized concentration and load trends for total phosphorus (TP) and dissolved reactive phosphorus (DRP) for the Maumee River at Waterville, Ohio (6,330 square miles), and Rock Creek at Tiffin, Ohio (34.5 square miles). The trends were determined using weighted least squares regression (Hirsch and De Cicco, 2015) water-quality data provided by Heidelberg University (National Center for Water Quality Research, 2013a), and streamflow data provided by U.S. Geological Survey (U.S. Geological Survey, 2014b).

Site name and number	Monitoring period	Number of years	Total change in concentration over monitoring period (percent)	Total change in load over monitoring period (percent)
		Total phosphorus		
Rock Creek (ohUSGS:04197170)	1984-2005	21	+12[1]	+45
	2005-2012	7	+1.2[1]	+0.6[1]
Maumee River (ohUSGS:04193500)	1977-2000	23	-33	-21
	2000-2012	12	-0.6[1]	-0.8[1]
		Dissolved reactive phosphorus		
Rock Creek (ohUSGS:04197170)	1992-2012	20	+74	+159
Maumee River (ohUSGS:04193500)	1992-2012	20	+100	+118

[1]Power-analysis results for these sites (see section 5.4 and chapter 11) indicate that monitoring records of 40 years or longer would be required to detect trends of this magnitude with statistical significance (p=0.20).

A sampling plan that is appropriate for quantifying phosphorus concentrations, loads, and trends is important for evaluating the effects of agricultural management practices. Concentration statistics (e.g. means, medians) and temporal trends are useful for determining whether water quality is improving or degrading at a given monitoring location (Moyer et al., 2012) and for comparisons among different watersheds. Load, or flux, measures the total mass of nutrients transported and is important for assessing impacts to downstream water bodies (Moyer et al., 2012). If changes in concentration due to agricultural management practices occur mainly during medium and low flows, which can be the case for certain management practices, such as filter strips (Table 1), these changes may not be detectable in annual loads, due to the far greater influence of high flows on annual loads. Due to the longer duration of low to medium flows during the year, such changes would likely be more discernible in monthly, seasonal, or annual concentration statistics (Moyer et al., 2012), or in monthly or seasonal load estimates.

5.2 Statistical designs to test the hypotheses

To test the hypotheses, sufficient data need to be collected using an appropriate sampling design such that phosphorus loads can be estimated with reasonable accuracy, unbiased concentration and load statistics can be calculated, and statistically significant trends can be detected through statistical analysis. Table 11 summarizes statistical-design options for testing the hypotheses. Monotonic trend and weighted-regressions trend analysis focus on trend detection but do not identify causality. Step-trend, paired watershed, and before and after statistical designs evaluate water quality relative to a treatment intervention. Additional statistical designs, such as multiple watershed and gradient designs, are applicable only to small watersheds.

Monotonic trend analysis (defined here to include both parametric and nonparametric trend methods) has been widely used as a statistical tool for assessing water-quality trends, and assumes that the trend is consistently increasing or decreasing over time. Monotonic trend analysis can be used to characterize trends in both flow-adjusted concentration and load. Methods for calculating confidence intervals and statistical significance of trends using monotonic trend analysis are available. The weighted-regression trend analysis method, Weighted Regression on Time, Discharge, and Season (WRTDS; Hirsch et al., 2010; Hirsch and De Cicco, 2015), is a relatively new tool for estimating loads and for evaluating monotonic and non-linear trends in concentrations and loads over time at sites that have relatively long monitoring records (10 years or longer). Methods for confidence intervals and statistical significance of trends within WRTDS have been developed and have undergone extensive testing, but documentation and release of the software has not yet taken place. Both monotonic and weighted-regression trend analysis were identified as the most feasible statistical designs for detecting temporal trends for this case study because management practice implementation data are not needed to detect water quality change. Step-trend and before and after statistical designs require information on the timing of management practice implementation to analyze water-quality data with respect to that intervention, and paired watershed design requires extensive watershed-specific data to identify experimental and control watersheds.

Table 11. Statistical designs for testing the case-study hypotheses.

Statistical Design	Description	Applicability for testing the case-study hypotheses	References
Statistical Designs for Large and Small Watersheds			
Monotonic trend	Monotonic/linear trend is a consistent increase or decrease in a water-quality variable over time.	Most feasible method for detecting water-quality change that may result from agricultural management practices.	Helsel and Hirsch 2002; Hirsch et al. 2010; Tetra Tech 2011a; 2011b
Weighted-regressions trend	Weighted regressions on time, discharge, and season; exploratory method capable of identifying non-linear trends in concentration and flux.	Most feasible method for detecting water quality change that may result from agricultural management practices, for sites with adequate sampling and long records (> about 10 years).	Hirsch et al. 2010; Hirsch and De Cicco, 2015.
Step-trend	Step-trend analysis can be used when the water-quality record includes two distinct time periods separated by a gap in monitoring, or when a known event has occurred that is likely to have affected water quality.	A potential fit if management practice implementation data are available. Gap in monitoring is not ideal and leaves a period of unknown changes in water quality.	Hirsch 1988; Helsel and Hirsch 2002
Paired watershed	Control and experimental watersheds are used to measure nutrient loadings and control for seasonal climate variations.	Not a good fit due to lack of implementation data in the drainage basin, difficulty in identifying paired watersheds, inconsistent monitoring designs by different organizations monitoring nearby watersheds, and unknown management practices may be used in control watersheds leading to a lack of reference conditions.	USDA, 2003; Weller et al., 2010
Before and after	Same as monotonic trend, but timing of management practice implementation is known and distinct.	A potential fit if management practice implementation data are available.	USDA, 2003 Weller et al., 2010
Statistical Designs for Small Watersheds			
Multiple watershed	Compares loads from a group of watersheds with treatments already in place in the region of interest. Storm sampling should be available and annual loads must be calculated for all the watersheds included in the study.	May be a good fit if sufficient data are available; many watersheds can be evaluated and compared, and results are transferable to the region because true variability among watersheds is included in the variance for each treatment.	USDA, 2003; Weller et al., 2010
Gradient design	Gradient design measures how land use or land cover has changed over time and relates water-quality changes to different intensities of management practice implementation through the use of multiple watershed monitoring sites.	Not a good fit for evaluating existing data, but may be an appropriate study design for a significant new monitoring program.	McMahon and Cuffney, 2000

Monotonic and weighted-regression trend analyses are appropriate tools for evaluating temporal trends in water quality related to agricultural management practices. Using these approaches requires that there are no large gaps in the data set, no changes in water-quality sampling or laboratory methods over time (unless the effect of method changes can be characterized statistically), and streamflow data are available for the duration of the study. Both approaches require long-term monitoring, usually greater than 5-10 years (U.S. Department of Agriculture, 2003). Monotonic trend analysis has been described as an effective approach to document response to treatment in situations where treatment (agricultural management practices in this case) is widespread, gradual, or where water-quality data are collected only at a single watershed outlet station (Tetra Tech, 2011a). Both trend analysis methods can be used to identify water quality change, even where treatment is inadequately documented. Once statistically significant water quality trends are detected, additional analysis and data are needed to identify whether agricultural management practices are the cause of that change.

Monotonic and weighted-regression trend analyses are particularly appropriate for this case study and the Lake Erie drainage basin where multiple organizations collect long-term water quality data. When multiple agencies collect data over the long term at each agency's own monitoring sites with similar sampling strategies, the data record at each site can be analyzed to detect trends and the trends can be compared across different sites due to long-term consistency at each monitoring site. Multiple-watershed or gradient statistical designs might be more effective for identifying change and causality in small watersheds, but there are not enough existing monitoring data for small watersheds to support those approaches (section 6.2).

5.3 Sampling frequency

The sampling frequency for testing the hypotheses must support estimation of annual, and potentially seasonal, load and concentration statistics, and evaluation of trend in flow-adjusted concentrations and loads. The TAC selected monthly water-quality samples as a starting point for considering the monitoring frequency needed to capture seasonal variation throughout the year.

Samples collected during each month of the year are critical for calculating accurate loads, but it is not uncommon for monitoring programs in the Lake Erie drainage basin to stop collecting samples during winter. Seasonal analysis of long-term data records for sites in the western Lake Erie drainage basin indicates that the winter TP and DRP loads (transported during December, January, and February) are larger than other seasons and show recent increases. Richards' (2012) analysis of Maumee River records shows winter TP and DRP loads from 1975 to 2012 to be consistently higher than the loads during other seasons, and the winter DRP loads in recent years are increasing more rapidly than in other seasons. Estimates of the 1995 to 2012 DRP load increase for the Maumee River indicate a 142-percent increase for winter compared to 113 percent for the full year (Robert Hirsch, U.S. Geological Survey, written commun., 2014). Average winter transport of DRP loads in the Maumee River and Rock Creek represents 36 percent and 38 percent, respectively, of the average total annual load; the winter loading would be 25 percent if the winter transport rates were equal to the annual average rates (Robert Hirsch, U.S. Geological Survey, written commun., 2014).

Additional samples targeting specific flows and seasons are needed to obtain more accurate load estimates so that trends in load can be adequately examined. Figure 16 shows 3 years of daily and monthly TP and DRP data at the Maumee River (large watershed) and Rock Creek (small watershed) monitoring sites. This figure emphasizes that significant variation in TP and DRP concentration is not captured by monthly monitoring, especially for the small watershed. Monthly sampling is an approximation of the full record and misses some high-flow, high-concentration storm events in both large and small watersheds (Figure 17). However, due to high variability in smaller watersheds, more of these events are missed by monthly monitoring in small watersheds.

Concentration of TP and DRP plotted against streamflow for both the Maumee River and Rock Creek for water year 2011 (October 2010 through September 2011) are shown in Figure 17. The hollow circles show daily sampling, while the colored circles show monthly sampling. For the Maumee River, only two monthly samples for TP and DRP were taken at streamflow greater than 10,000 cubic feet per second (ft^3/s), although visual inspection indicates about a third of daily samples were taken at streamflow between 10,000 and 100,000 ft^3/s. For Rock Creek, only one sample was taken at streamflow greater than 30 ft^3/s, although more than one third of the daily samples were taken at streamflow between 30 and 1,000 ft^3/s. These graphs demonstrate that monthly samples do not adequately represent concentrations at high flows, which are needed for accurate load calculations.

Collection of supplemental samples characterizing the range of concentrations over the hydrograph is valuable for the purpose of calculating more accurate and precise loads (Richards and Holloway, 1987; Robertson, 2003; Richards, 1998; Gilroy et al., 1990), although the specific sampling design for collecting additional samples varies and depends on the specific monitoring objectives. The choice of sampling design is influenced in part by the interpretive methods used, and the purpose for which trends or loads are being evaluated.

The TAC identified four sampling strategies that better support load and trend analyses compared to monthly sampling for the purpose of answering the case-study policy question, which are presented in Table 12. These four recognized strategies have the potential to support adequate load and trend calculations, but analyzing the costs and benefits of these options was beyond the scope of this study. The table summarizes the annual number of samples for each of these options.

Once an option is selected, the sampling plan must be designed appropriately for the studied watersheds. A variety of factors (Table 13) should be considered when designing a sampling plan to measure agricultural management practice effectiveness in a specific watershed. Data collected using these sampling strategies, depending on which option is selected, can be used to support analysis of time-based (e.g. monthly, seasonal, annual) or flow-based (e.g. low, medium, high flow conditions) phosphorus loads and concentration statistics (e.g. means or medians) in addition to annual concentration and load trend evaluations.

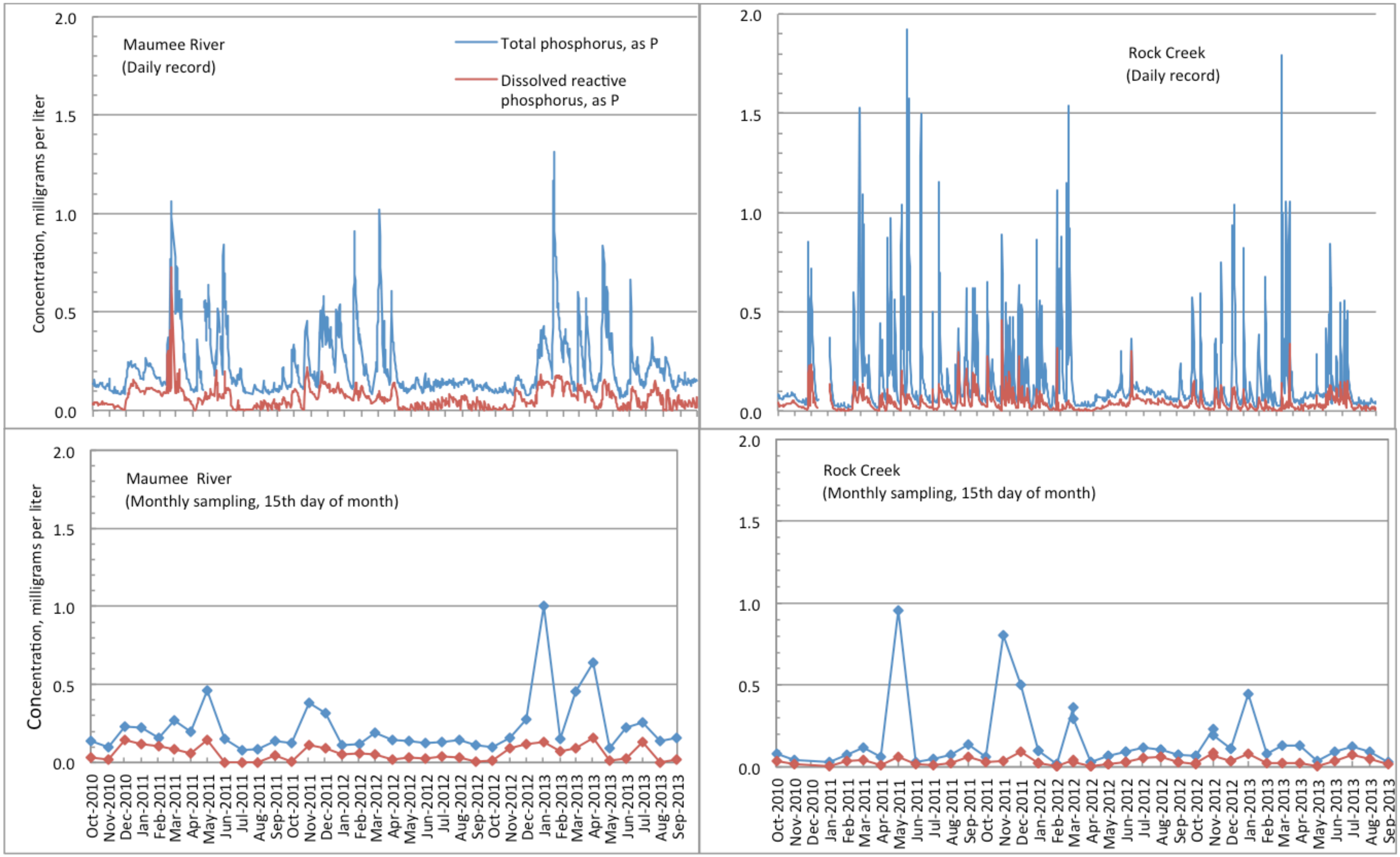


Figure 16. Time series of daily and monthly sampling of total phosphorus and dissolved reactive phosphorus for the Maumee River at Waterville, Ohio (6,330 square miles), and Rock Creek at Tiffin, Ohio (34.6 square miles), during water years 2011 through 2013. Data provided by Heidelberg University (National Center for water Quality Research, 2013a).

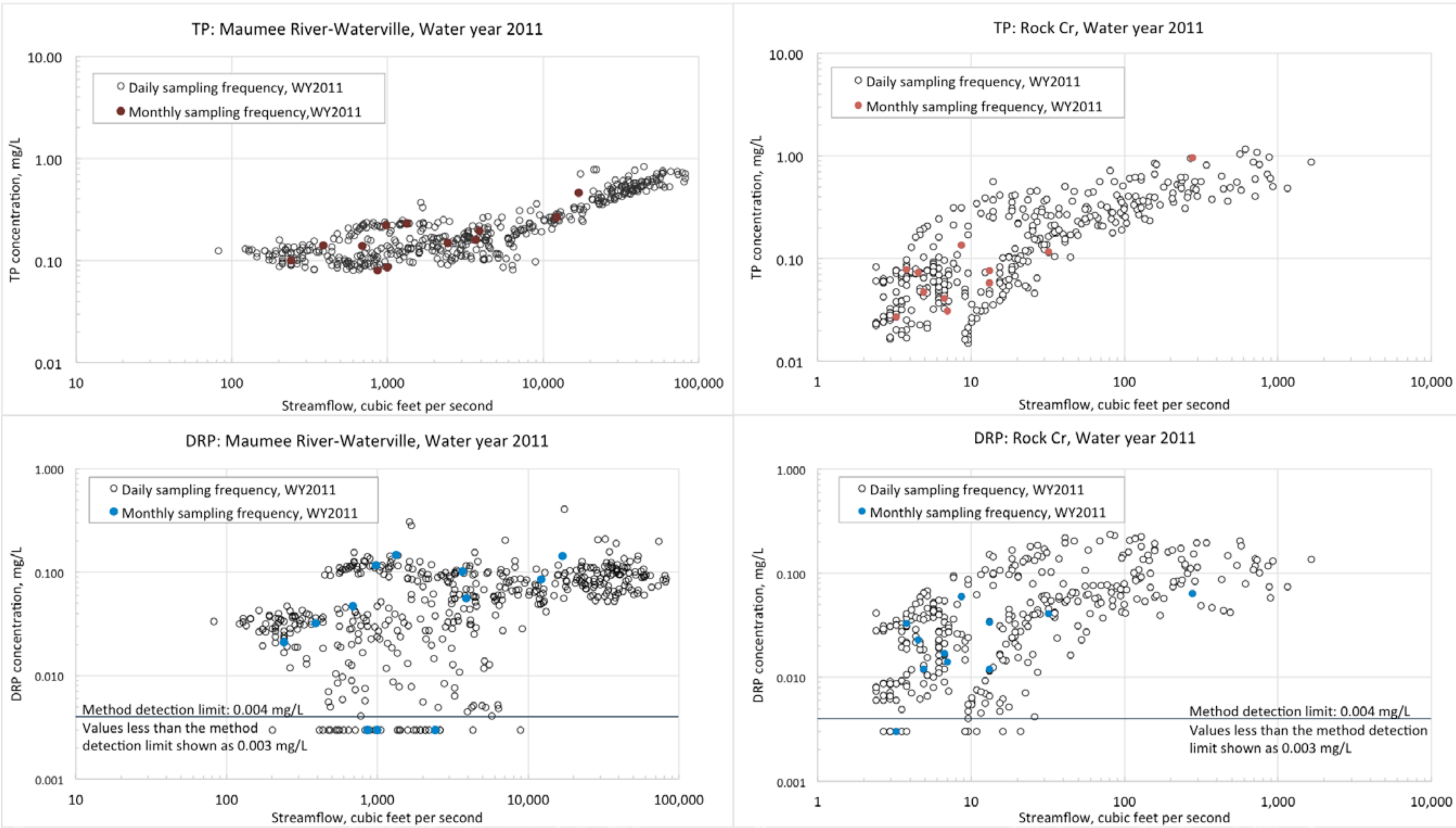


Figure 17. Concentration versus streamflow showing daily and monthly sampling frequencies for TP (total phosphorus) and DRP (dissolved reactive phosphorus) for the Maumee River at Waterville, Ohio (6,330 square miles), and Rock Creek at Tiffin, Ohio (34.6 square miles), during water year 2011. Values of DRP below the 0.004 method detection limit would be expected to have a greater associated uncertainty than values above this level. Water-quality data provided by Heidelberg University (National Center for Water Quality Research, 2013a) and streamflow data provided by the U.S. Geological Survey (2014b).

Table 12. Sampling frequency options for testing the case-study hypotheses.

Sampling frequency option	Description	Number of discrete samples per year	Total number of samples over 10 years	References
1	Monthly plus 12 additional samples targeting a range of high- and low-flows on the hydrograph	24	240	NAWQA Sampling Design (Gary Rowe, U.S. Geological Survey, written commun., 2014)
2	Two years of monthly sampling plus sampling during all storm events (average 100 samples/year (Richards and Holloway, 1987)) followed by adaptive management to identify long-term sampling plan.	100	392[1]	Richards and Holloway, 1987
3	Daily plus up to 3x daily during storms (average 500 samples per year (Baker, 2009))	500	5000	Baker, 2009
4	Continuous monitoring plus supplemental discrete samples. Two years at 2 samples/month, and 1 sample/month thereafter. (turbidity and/or dissolved phosphorus)	24/12	144[2]	Casey Lee, U.S. Geological Survey, written commun., 2014

[1]Assumed 24 samples in years 3-10
[2]24 samples in years 1-2, monthly in years 3-10

Table 13. Factors for consideration when designing a sampling plan to monitor effectiveness of agricultural management practices.

	Factors for Consideration in Sampling-Plan Design for Measuring Agricultural Management Practice Effectiveness
Small Watersheds	• For TP and DRP, target high-flow periods to capture maximum concentrations • For TP and DRP, target times of greatest transport from field to stream: high flows, rain on frozen ground, and during fertilizer application • For DRP, target medium- to low-flow periods to assess changes due to transport via tile drains and groundwater.
Large Watersheds	• Target total annual TP and DRP input to the lake (total nutrients available to feed algal growth) • Target spring-summer input that drives algal growth during bloom season.
Small and Large Watersheds	• Consider the specific management practices planned/in place in the watershed and monitor flows targeted by the management practices planned or in place • Supplemental sampling should be designed in coordination with all agencies monitoring small watersheds so that trend analyses can be completed both within watersheds and compared between watersheds.

Each of the sampling-frequency options from Table 12 can be used for both small and large watersheds. Option 1 (Table 12) represents a strategy that has been evaluated using existing data (Charles Crawford, U.S. Geological Survey, written commun., 2014) and is currently (2014) used by the USGS NAWQA Program (Gary Rowe, U.S. Geological Survey, written commun., 2014). Option 1 includes a total of 24 samples per year, with 12 samples collected monthly at fixed intervals, and at least 12 supplemental, seasonally-weighted fixed-interval samples collected during seasons of high water-quality-concentration variability and(or) seasons of fertilizer and pesticide applications. For testing the case-study hypotheses, the 12 supplemental samples should be taken during times of anticipated phosphorus reduction from agricultural management practices (Table 13).

Option 2 (Table 12) reflects work completed by Richards and Holloway, 1987, where they recommended frequent initial watershed sampling for about 2 years followed by application of sampling-theory calculations to determine whether sampling can be reduced as a form of adaptive management. Option 3 (Table 12) describes the sampling strategy used by Heidelberg University (Baker, 2009) and by the USDA (U.S. Department of Agriculture, 2014b and 2014c) at water-quality monitoring sites in the Lake Erie drainage basin. This strategy features collection of a least one sample per day with multiple samples collected per day during storm events.

Continuous water-quality monitoring, as presented in option 4 (Table 12), is an option transitioning from an experimental monitoring strategy to being adopted for widespread routine applications. Continuous turbidity monitors are used in combination with discrete samples to correlate turbidity with total suspended solids and TP. This relationship can be used to provide high frequency estimates of TP concentrations (Spackman et al., 2011). Discrete samples are necessary for developing and updating the turbidity/TP relationship over the period of record. Twenty-four discrete samples for the first 2 years of monitoring to develop the correlation, and 12 discrete samples per year thereafter to maintain and update the correlation are used to illustrate this monitoring approach in Table 12. Continuous monitors for dissolved phosphorus are also under development (Texas Commission on Environmental Quality, 2008). Continuous monitors often measure a group of parameters that can include turbidity, nitrate, water temperature, dissolved oxygen, pH, and specific conductance. These properties are all part of the suite of parameters needed to answer the case-study policy question identified in Table 3. Continuous monitoring technology is further discussed in sections 7.3 and 8.2.

There are some differences between sampling in small watersheds compared to large watersheds in targeting high flows as recommended in Table 13 using a strategy like option 2 (Table 12). Rainfall events produce a larger relative change in streamflow over a shorter time period in smaller rivers compared to larger rivers (Figure 16). Smaller watersheds are more volatile in storm-loading response compared to larger watersheds, and hourly samples may be required to adequately capture the range of flows during storm events in small watersheds. Therefore, a higher number of samples per storm per day are necessary to capture the variation in high flow concentrations of TP and DRP in small watersheds (Robertson, 2003). However, high flows related to storms will last for more days in large watersheds compared to small watersheds (Baker and Richards, 2000), so the total number of samples needed to cover the range of flows may be similar. Sampling frequencies selected for new and existing monitoring sites should be coordinated across monitoring agencies so that data collection, analysis, and interpretation can be planned for compatibility and comparable analysis.

5.4 Duration of monitoring to detect change

Substantial post-implementation data collection—most likely 10 years of data or more—is needed at dedicated monitoring sites to quantify the effectiveness of management practices at reducing nutrients, to assess temporal trends, and to determine whether there is a related ecosystem response (Meals et al., 2010). This long interval of data collection is due to a variety of factors:

- Water-quality time series less than 10 years tend to be highly influenced and can be biased by short-term weather patterns that differ from long-term patterns (Burt et al., 2014).
- Climatic effects (such as floods and droughts) and changes in agricultural production and crop type (which are driven by market conditions or crop rotations) can affect agricultural management practice effectiveness as well as spatial and temporal fluctuation of both nutrient loading and ecosystem response (Michalak et al., 2013a).
- A fraction of existing (legacy) soil and sediment phosphorus will likely continue to be released from the system as "internal" phosphorus sources via surface and subsurface hydrologic pathways to streams, tile drains, and shallow groundwater even after agricultural applications of phosphorus and stormwater runoff are reduced (Jarvie et al. 2013a and 2013b; Sharpley et al., 2013).

The duration of sampling needed to statistically discern water-quality trends is influenced by a number of factors, principally the sampling frequency, the inherent variability of the system (i.e. concentrations and streamflow), the ability of the trend method to remove extraneous variability other than land-use effects, and the rate and magnitude of water-quality change in the watershed. The approach selected by the TAC to estimate the duration of sampling to detect phosphorus trends used existing TP and DRP data for streams in the study area (the nutrient data set), a 5-parameter log-linear-regression flow-adjusted trend-analysis model, and a "power analysis". The power analysis estimates the duration of monitoring needed to identify statistically significant trends, using a specific sampling frequency, at specified levels of significance and power.

Power, in the statistical context, relates to the probability of not committing a Type II error (Somers, 1997). Power is the probability that one will reject the null hypothesis of no trend when there is, in fact, a trend of some specific magnitude. In other words, it is the ability of a statistical test to detect a water quality trend if the trend exists. In watersheds exhibiting a weak water-quality signal (small rate of change) and characterized by highly variable water-quality and/or streamflow systems, long-term monitoring will be required to discern temporal trends. Figure 18 shows an example of variability over time at a monitoring site, using monthly TP and DRP concentrations from the Maumee River at Waterville.

Type I Error: Detecting an effect that is not present (incorrectly rejecting the null hypothesis).

Type II Error: Failing to detect an effect that is present (incorrectly failing to reject the null hypothesis).

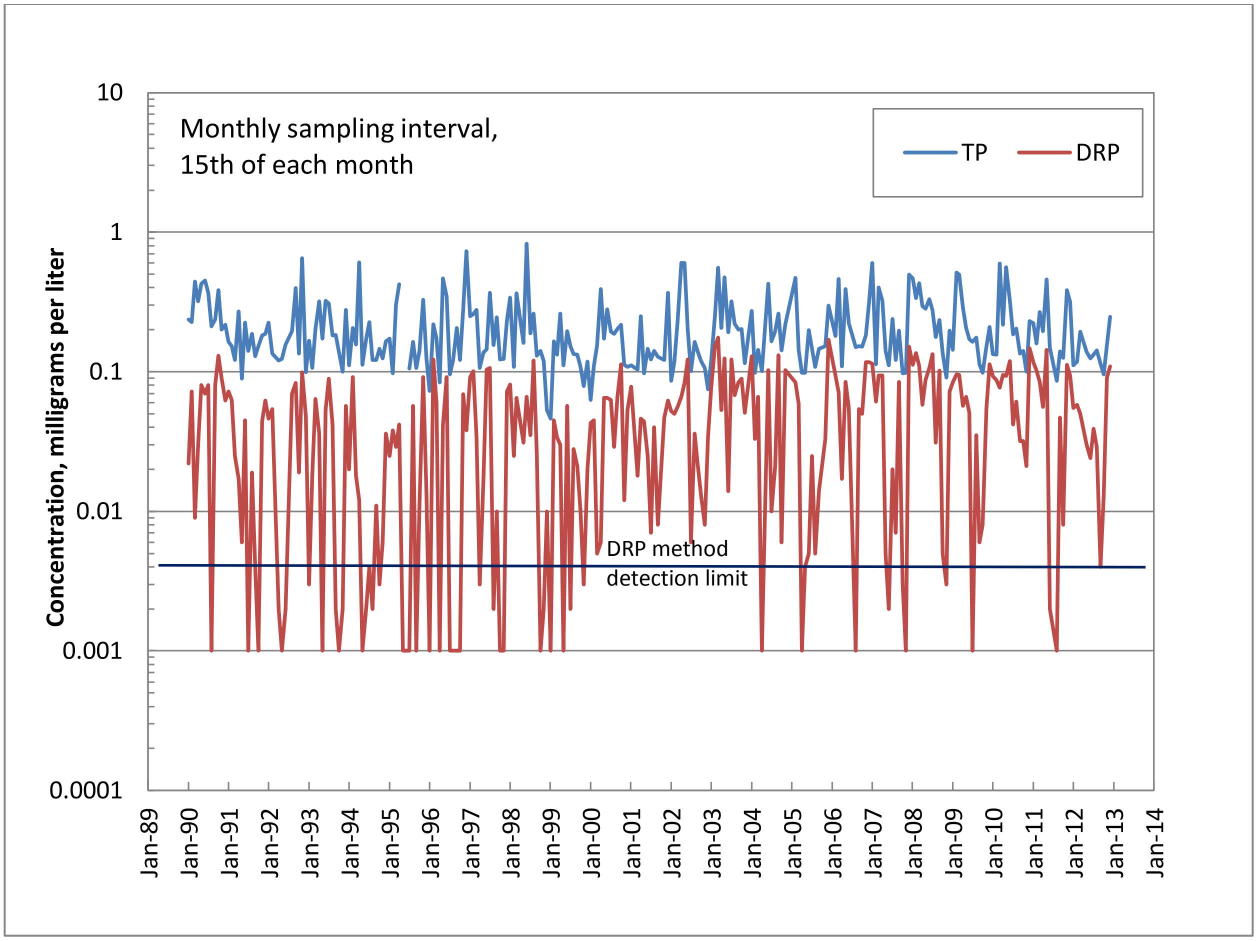


Figure 18. Time series of monthly samples of total phosphorus (TP) and dissolved reactive phosphorus (DRP) during January 1990 to December 2012 at the stream-monitoring site Maumee River at Waterville, OH; data from Heidelberg University. Values of DRP below the 0.004 method detection limit would be expected to have a greater associated uncertainty than values above this level.

The power analysis method is described in detail in the Appendix (Chapter 11). Fixed interval monthly samples of TP and DRP from existing "data-rich" water-quality monitoring sites in the nutrient data set were used in a log-linear regression analysis using flow- and seasonally-adjusted TP and DRP concentrations to estimate background variability. Monthly sampling was selected as the input sampling frequency for the power analysis because it was a simple, consistent interval that could be extracted from all the data records available for the analysis. The data-rich water-quality monitoring sites selected for the power analysis (Table 14) were located at or near a streamgage, were in watersheds covered by more than 40-percent row crops, and most of these sites were sampled at least monthly for 4 or more years. Due to limitations on the number of available data-rich sites, watersheds of all sizes were considered for this analysis, and results are provided for a range of watershed sizes. In locations where multiple nearby candidate sites were available, sites with the longest and (or) most recent record were selected for power analysis.

The regression analysis calculated a residual standard error (RSE) for each monitoring site, a measure of background variability, which was used to estimate the duration of monitoring needed to identify trends of different magnitude as described in the Appendix (Chapter 11) and generating results as shown in Figure 19. The RSE is the same for both concentration and load for a given data set using the log-linear model. As a result, the duration of monitoring to detect a specific magnitude of change applies for detecting trends in both concentration and load. However, for a given set of data the magnitude (and possibly direction) of trend for flow-adjusted concentration will likely be different from the magnitude of trend for load as discussed in section 5.1.

An example of the power analysis results (Figure 19), determined using the TP record for the Maumee River (site ohUSGS:04193500), shows the estimated years of monthly sampling needed to detect trends in TP (percent change) ranging from 10 to 45 percent over a 20-year monitoring period. The three error curves in Figure 19 each represent both statistical significance (α) and power (β), which were set as equal and analyzed at levels of 10-, 20-, and 30-percent. In this example, the 20-percent error curve indicates that 13 years of monthly monitoring would be required to detect a trend in median TP given a rate of change of 20-percent. This means if the true trend was 20 percent there would be a 20-percent chance ($\beta=0.2$) that the test would fail to identify that trend, or an 80-percent chance of correctly identifying the trend; alternatively, if there was no true underlying trend, there would be a 20-percent chance ($\alpha=0.2$) that the test would incorrectly identify a trend (due to random chance), or an 80-percent chance of correctly identifying the lack of a trend. If variability increases in the future or is greater than identified in the available records, a longer duration of monitoring would be needed. If variability decreases in the future or is less than identified in the records analyzed, a statistically significant trend may be identified sooner than estimated here.

The duration of sampling to detect change is a function of the anticipated magnitude of change and the desired error level (significance and power). More samples are required to detect change for lower error levels and smaller percentages of change. Deciding what error level is appropriate for supporting decision-making is a policy choice. The policy choice of what error level to choose is a decision for managers who must balance the resources available for further investigation against the number of sites that are monitored and resources for management practice implementation. The choice of error level can be iteratively and adaptively modified as ongoing investigations reveal water-quality changes (or lack thereof) in a watershed.

Table 14. Data-rich monitoring sites in the nutrient data set that were evaluated using power analysis to estimate sample-size requirements for detecting temporal trends in total phosphorus (TP) and dissolved reactive phosphorus (DRP).
[Abbreviations: mi^2, square miles; %, percent]

Monitoring site	Station name	Basin area (mi^2)	Row crop area (% of basin area)	Water quality parameter
ohUSGS:04193500	Maumee River at Waterville, OH[1]	6,330	73%	TP and DRP
03231500	Scioto River at Chillicothe OH[2]	3,849	80%	TP
ohUSGS:04198000	Sandusky River near Fremont OH	1,252	78%	TP and DRP
inLES060-0005; Flow site 04182000	Saint Mary's River St. Mary's River near Fort Wayne, IN	716	81%	TP
04178000	St. Joseph River near Newville, IN	618	45%	TP and DRP
ohUSGS:04189000	Blanchard River near Findlay, OH	351	77%	TP
04186500	Auglaize River near Fort Jennings OH	331	76%	TP
ohUSGS:04199500	Vermilion R Near Vermillion Ohio	260	55%	TP and DRP
ohUSGS:04197100	Honey Creek at Melmore, OH	150	81%	TP
ohUSGS:04197170	Rock Creek at Tiffin OH	34.6	73%	TP and DRP
ohBR; Flow site 04199155	Old Woman Creek at Berlin Road near Huron, OH	22.1	70%	DRP
402913084285400	Chickasaw Creek at St. Marys, OH[1]	16.4	79%	TP
ohUSGS:04185440	Unnamed Tributary to Lost Creek near Farmer, OH	4.23	78%	TP and DRP

[1]Heidelberg University water-quality monitoring occurs at USGS site 04193490; USGS monitors flow at USGS site 04193500.
[2]Not in the Lake Erie Basin.

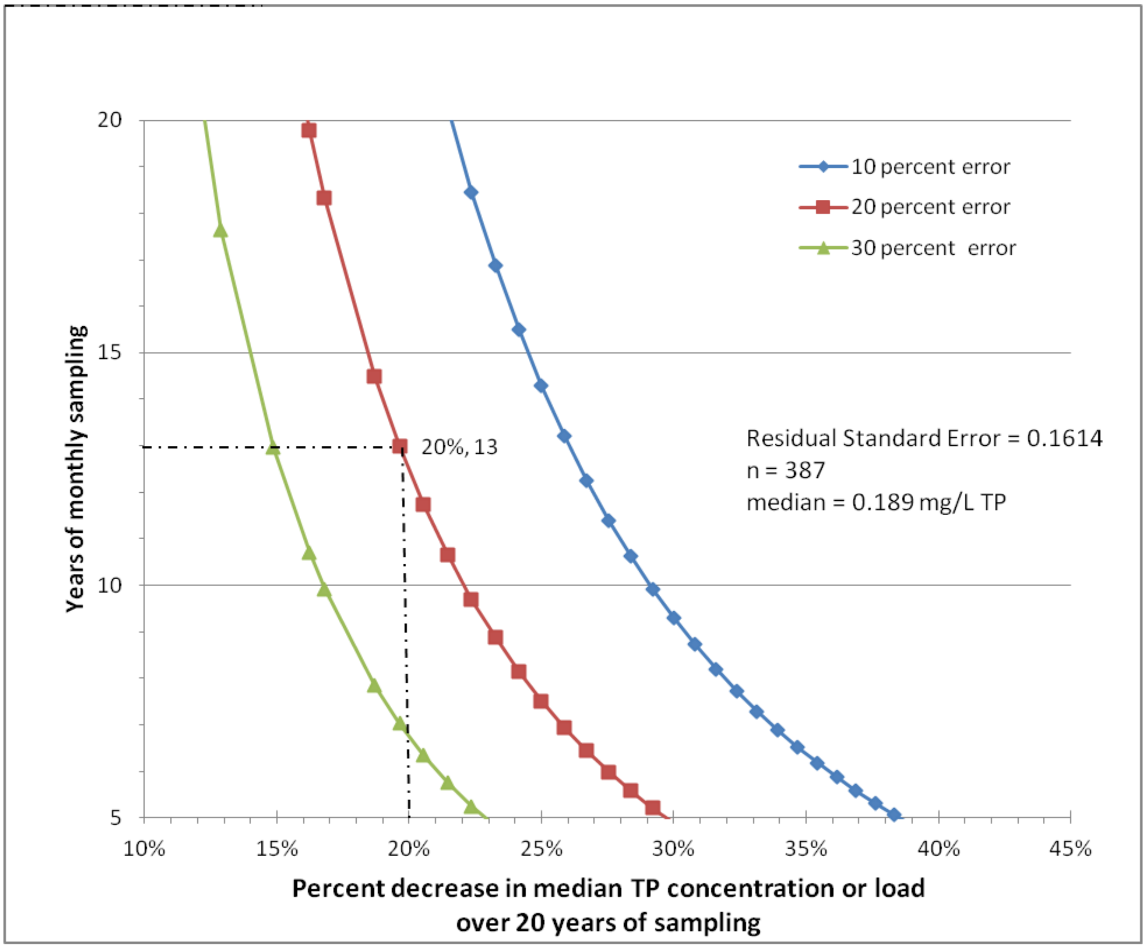


Figure 19. Power analysis estimates of the number of years of monthly sampling needed to detect decreasing trends in median total phosphorus (TP) concentration or load, for different error levels, over 20 years of sampling at Maumee River at Waterville, OH (ohUSGS:04193500 from Table 14). The power analysis method is described in Chapter 11. [Abbreviations: mg/L, milligrams per liter]

As indicated in the power-analysis results (Figure 19), the anticipated magnitude of change is a critical factor influencing the duration of sampling needed to statistically discern phosphorus trends. Literature was reviewed to identify measured and modeled ranges of TP and DRP changes associated with agricultural management practices in the Lake Erie drainage basin. Bosch et al. (2013) used the SWAT model to simulate a suite of BMPs considered by agricultural-conservation staff to be "feasible" through voluntary implementation, in addition to practices currently in place in large watersheds in the Lake Erie drainage basin. According to the authors, "With implementation affecting only about 25% of cropland area, it appears that sediment and nutrient yields will decrease by about 10% at most" from a baseline of current watershed conditions simulated in Bosch et al. (2011). According to Scavia et al. (2014), "a 20% reduction in TP or DRP load requires implementing the BMPs on more than 50% of the agricultural land."

A USDA CEAP study (Lund et al, 2011) evaluating conservation practices in the Great Lakes Basin found that implementation of additional agricultural management practices in priority cropland areas in the Lake Erie drainage basin that have already implemented some agricultural management practices would be expected to reduce average annual phosphorus loading to large watershed outlets by 4 percent, and that treatment of all the “under-treated” cropland areas in the Lake Erie drainage basin should reduce the phosphorus loading by 32 percent. The simulation models used in this study did not account for “legacy phosphorus” adsorbed to soil particles as a result of prior farming activities, which may be an important contributor to current levels of in-stream phosphorus loads, and its continued release to tile drains and shallow groundwater may result in a lag in detecting the effects of agricultural management practices—especially in large watersheds. Release of legacy phosphorus would result in smaller observed phosphorus reductions in the short term compared to the CEAP study results. These estimates of agricultural management practice effectiveness and associated percentage reductions in phosphorus yields and loads are summarized in Table 15.

Small watershed evaluations of agricultural management practice effectiveness primarily rely on monitoring studies in watersheds with concentrated use of agricultural management practices. For example, Corsi et al. (2005) estimated annual reductions in rainfall storm loads for the non-vegetative season at 48 percent for TP in a 9.5-square-mile watershed in Wisconsin, although other USGS studies in Wisconsin found a range of reductions in TP, including some studies that documented no change (Corsi et al., 2013). Zollweg and Makarewicz (2009) determined that a suite of agricultural management practices implemented in a 38-hectare (0.15-square-mile) watershed in New York, including reduction of fertilization rates and elimination of fall and winter spreading of manure, resulted in an 80-percent reduction in TP and 57-percent reduction in SRP (DRP) during storm events over the course of 5 years. Bishop et al. (2005), found a 43-percent event-load reduction in TP and a 29-percent reduction in soluble reactive phosphorus (SRP or DRP) on a 160-hectare (0.6-square-mile) farm watershed in New York. Additional small watershed studies evaluating phosphorus-targeted agricultural management practices found TP and DRP reductions of between 50 and 75 percent through long-term water-monitoring programs (Spooner et al., 2012).

Table 15. Estimates of agricultural management practice effectiveness in the Lake Erie drainage basin. [Abbreviations: %, percent]

Description of Management Practice Implementation	Estimated Percent Reduction	Reference
Large Watershed "Feasible" scenarios with voluntary management practice implementation		
Random application of no-till, cover crop, and filter strips to 20-25% of cropland in the Maumee, Sandusky, and Raisin River Basins.	-10% from baseline TP and DRP average annual yield	Bosch et al. (2013)
Management practice application to priority cropland in Lake Erie drainage basin.	-4% TP annual load	Lund et al. (2011)
Large Watershed High-Implementation Scenarios with mandated management practice implementation requirements		
No-till, cover crop, and filter strips implemented together applied to 100% of cropland in the Maumee River Basin.	-30% from baseline TP average-annual yield -26% from baseline DRP average-annual yield	Bosch et al. (2013)
Management practice application to 50% of cropland in Maumee River Basin.	-20% TP or DRP annual load	Scavia et al. (2014)
Management practice application to all "under-treated" cropland areas in the Lake Erie drainage basin.	-32% TP annual load	Lund et al. (2011)

Agricultural management practices are just one watershed factor that can produce changes in phosphorus concentrations and loads, as discussed in sections 2.3 and 2.4. Once statistically significant changes in TP or DRP concentrations or loads are identified, additional analysis would be needed to identify whether agricultural management practices produced that change, using the variety of data types shown in Figure 1. Other factors of change, such as changing patterns of precipitation and streamflow, may be increasing TP and DRP loads while agricultural management practices are reducing them, resulting in a small net change that is difficult to detect. The recent increasing trends in DRP presented in Table 10 will likely complicate, reduce, or delay the water quality signal from DRP loading reductions as management practices are implemented. If the extent of agricultural management practices and the magnitude of their results are small in relationship to other factors of change, it will be difficult to discern the effects of management practices. Consequently, to get the greatest benefit from monitoring to answer the case-study policy question, appropriate interventions to increase the signal from agricultural management practices are needed in monitored watersheds.

Given the literature cited above, 10 percent represents a reduction in TP or DRP annual load that might be reasonably expected in watersheds as a result of current or slightly increased implementation of agricultural management practices in the most vulnerable areas and additional row-crop areas. A larger reduction in TP or DRP, such as 20 percent, could be considered for scenarios involving mandatory use of appropriate agricultural management practices resulting in high-density implementation.

To get the greatest benefit from monitoring to answer the case-study policy question, appropriate interventions to increase the signal from agricultural management practices are needed in monitored watersheds.

The International Joint Commission (2014) states that March to June DRP load reductions of 37-41 percent and average-annual TP load reductions of 38-39 percent would be required to achieve a goal of no or mild algal blooms in Lake Erie. Based on the large watershed modeling results presented, a combination of agricultural management practices and other nutrient source reductions would be needed to achieve these load-reduction goals. Small watershed monitoring studies indicate that intensive and consistent implementation of agricultural management practices throughout small watersheds may be capable of generating a 40-percent reduction within individual small watersheds. The estimated years of monthly sampling needed to detect these three scenarios of 10-, 20-, and 40-percent reduction as identified through the power analysis at the 20-percent error level are summarized in Table 16 and Table 17.

The power-analysis estimates for several medium and large watersheds indicated that a 40-percent reduction in TP concentration or load could be detectable in less than 5 years of monthly monitoring (*Appendix*). However, a minimum of 5 to 10 years of sampling is required to capture an adequate range of meteorological conditions representative of long-term variability to confirm that observed water-quality changes are not an artifact of short-term weather patterns and to confirm the effectiveness of management practices under varying climatic conditions. In practice, the specific years of data collection should be reviewed to confirm that the range of weather conditions during the monitored period is representative of the range of conditions observed over longer time periods. As discussed in the beginning of this section, the need and value of long-term monitoring for management practice evaluation studies have been recognized by a number of researchers. Long-term studies also provide the ability to evaluate maintenance requirements to assure consistent performance of agricultural management practices.

Table 16. Power analysis estimates of the number of years of monthly sampling needed to detect reductions in median total phosphorus (TP) concentration or load, at the 20-percent error level, for watersheds of different sizes in the Lake Erie drainage basin. The power analysis method is described in the *Appendix*.
[Abbreviations: %, percent; >, greater than]

	Years of monthly sampling to detect reductions in median TP concentration or load at the 20-percent error level		
Watershed size	**10% reduction** Estimated decrease in TP load with current or slightly increased rates of management practice implementation	**20% reduction** Estimated decrease in TP load with mandatory appropriate high-density management practice implementation	**40% reduction** International Joint Commission (2014) TP load reduction goal; likely requires multiple TP source reductions
Large: >1,000 square miles	>40	13 to 26	5-10[1]
Medium: > 50 to 1,000 square miles	>40	9 to >40	5-11[1]
Small: 50 square miles and smaller	>40	27 to >40	5-10[1]

[1]If less than 10 years, verify the climatic conditions span a range representative of long-term climatic variability

Table 17. Power analysis estimates of the number of years of monthly sampling needed to detect reductions in median dissolved reactive phosphorus (DRP) concentration or load, at the 20-percent error level, for watersheds of different sizes in the Lake Erie drainage basin. The power analysis method is described in the *Appendix*.
[Abbreviations: %, percent; >, greater than]

	Years of monthly sampling to detect reductions in median DRP concentration or load at the 20-percent error level		
Watershed size	**10% reduction** Estimated decrease in DRP load with current or slightly increased rates of management practice implementation	**20% reduction** Estimated decrease in DRP load with mandatory appropriate high-density management practice implementation	**40% reduction** International Joint Commission (2014) DRP load reduction goal; likely requires multiple DRP source reductions
Large: >1,000 square miles	>40	>40	24-35
Medium: > 50 to 1,000 square miles	>40	>40	8-15[1]
Small: 50 square miles and smaller	>40	>40	8-23[1]

[1]If less than 10 years, verify the climatic conditions span a range representative of long-term climatic variability

TP and DRP load reductions resulting from agricultural management practices take place over time as more and more management practices are implemented within a watershed and the cumulative new practices result in reduced phosphorus transport to streams and major tributaries. To simplify the concept of reductions over time, consider the changes of 10, 20, or 40 percent for TP (shown in Table 16) taking place over a 20-year study period. If a reduction in TP concentration or load in a small watershed is occurring at a rate of 40-percent over 20 years, one would be able to detect that trend within 10 years of monthly monitoring at the 20-percent error level. For DRP, the estimates of monthly sampling needed to detect a 40-percent reduction in small watersheds at the 20-percent error level ranged from 8 to 23 years (Table 17). On the other hand, if minimal interventions are taken to reduce nutrient transport, for example resulting in a 10-percent decrease in nutrient concentrations or load over a 20-year study period, neither TP nor DRP trends would be detected with statistical significance and power in small or large watersheds.

The power analysis of existing TP and DRP records for some sites estimated a need for monthly sampling longer than 40 years for some of the modeled phosphorus-reduction scenarios (10- and 20-percent reductions) in Table 16 and Table 17. Given the uncertainty associated with such long-term estimates of nutrient reductions; the multiple influences that can affect observed trends; and the limitation of monthly sampling frequency in the power analyses, which is less frequent sampling than recommended here for answering the case-study policy question; a cutoff of greater than 40 years was used to illustrate duration of monitoring needed for potential scenarios.

5.5 Locations and number of monitoring sites

Quantifying adequate spatial representation through a monitoring program across an area as large as the Lake Erie drainage basin is challenging. Optimization approaches exist for choosing a monitoring network design that gives an acceptable degree of certainty in estimates of the state of water quality, subject to the constraint of cost to acquire the information (e.g., Speight et al., 2009). The disadvantage of applying such a method to this case study is that it would require extensive modeling and statistical analysis, and significant time and money; the advantage is that the degree of certainty (i.e., confidence) in estimates of water quality could be quantified. Instead, a qualitative assessment of watershed characteristics and TAC judgment was used in this study to identify a minimum number of sampling sites needed to test the case-study hypotheses. The disadvantage of this qualitative approach is that conclusions about the degree of certainty in estimates of water quality cannot be quantified, which is frequently the case with water monitoring programs due to budget limitations.

Figure 9 identifies the 3 large watersheds that meet the spatial criteria for testing the large watershed hypothesis. Monitoring sites should be located in each of these watersheds near their drainage point to Lake Erie to answer the case-study policy question.

As discussed in Chapter 3 and Table 4, small watershed monitoring sites will be most effective if located in agricultural areas of high phosphorus yield and high soil vulnerability. The TAC identified six watershed characteristics (Figure 12) in the Lake Erie drainage basin that are considered to influence the effectiveness of agricultural management practices in reducing phosphorus inputs to streams. Watersheds exhibiting these characteristics are likely to experience larger nutrient reductions in response to agricultural management practices, so statistically significant results may be available earlier in comparison to less vulnerable or low-yield watersheds. The TAC determined there should be at least one small watershed monitoring site for each of these priority watershed characteristics, for a minimum of 6 monitoring sites in the Lake Erie drainage basin. The 6 priority watershed characteristics are:

- Soil-runoff vulnerability,
- Soil-leaching vulnerability,
- High TP yield from fertilizer, estimated using SPARROW,
- High TP yield from manure, estimated using SPARROW,
- High TP yield, estimated using SWAT, and
- High DRP yield, estimated using SWAT.

Monitored watersheds should maximize coverage and spatial representation of these priority characteristics across the Lake Erie drainage basin. Figure 12 (A) through (F) shows the coverage of these characteristics across the Lake Erie drainage basin, and Figure 13 shows where the highest values overlap.

Additional monitoring sites will improve the ability to measure the effectiveness of a variety of agricultural management practices in a variety of conditions; a large number of small watershed monitoring sites could support the multiple watershed or gradient statistical designs described in Table 11. One water-quality study using gradient design used 30 monitoring sites to study the water-quality impacts of urban density in each of three study areas (McMahon and Cuffney, 2000).

This report does not include recommendations for monitoring of “reference” or “control” watersheds, defined as watersheds where land use and streamflow conditions are minimally affected by human activities, which could represent baseline nutrient loading and/or streamflow conditions. Information from control watersheds could be helpful for evaluating effects of management practices, climate variability, and streamflow alterations on nutrient loading. In the Lake Erie drainage basin, there are no currently active USGS streamgages (as of December 2014) in watersheds having less than 20 percent row crops that might serve as potential control sites for evaluating agricultural impacts on water quality, on the basis of information in the GAGES-II dataset (Falcone et al., 2010; Falcone, 2011) and the National Water Quality Monitoring Council’s National Network of Reference Watersheds (Wilber and Deacon, 2012; Michael McHale, National Water Quality Monitoring Council and U.S. Geological Survey, written commun., 2014). In the western Lake Erie drainage basin, the extensive agricultural land use (Figure 2) and drainage alterations (Figure 14) limit the availability of watersheds that are minimally affected by agricultural and drainage management practices.

Table 18 summarizes the water data needed to answer the case-study policy question; these are the criteria that must be met for water data to be used to test the case-study hypotheses. In addition to TP and DRP, each of the parameters identified in Table 3 is needed to fully address the case-study policy question.

Table 18. Summary of water monitoring data needed to detect water-quality change resulting from agricultural management practices in the Lake Erie drainage basin.
[Abbreviations: %, percent; TP, Total Phosphorus; DRP, Dissolved Reactive Phosphorus]

	Small Watersheds	Large Watersheds
Monitoring sites located in watersheds with these characteristics	• Less than or equal to 50 square miles, • Greater than or equal to 40% of row-crop coverage, • High phosphorus yield and high soil vulnerability	• Greater than or equal to 1,000 square miles • Drains directly to Lake Erie • Greater than or equal to 40% of row-crop coverage
Monitoring parameters	• TP, DRP, streamgage • Suite of parameters from Table 3	• TP, DRP, streamgage • Suite of parameters from Table 3
Sampling frequency **(pick one)**	• Monthly plus supplemental sampling (24/year) • Two-year intensive monitoring followed by adaptive management to modify sampling plan (100 per year then 24 per year), or • Daily plus storm sampling (approx. 500 per year) • Monthly plus continuous monitoring (turbidity and/or dissolved phosphorus)	• Monthly plus supplemental sampling (24/year) • Two-year intensive monitoring followed by adaptive management to modify sampling plan (100 per year then 24 per year), or • Daily plus storm sampling (approx. 500 per year) • Monthly plus continuous monitoring (turbidity and/or dissolved phosphorus)
Minimum duration of monitoring to detect change	>8 years[1] (assumes 40% reduction in TP and DRP over 20 years)[2]	>20 years (assumes 20% reduction in TP and DRP over 20 years)[2]
Number of monitoring sites	Minimum of 6 active monitoring sites (1 per watershed) representing a variety of watershed characteristics and spatially distributed across the agricultural areas of the basin	3 active monitoring sites (1 per watershed)

[1]If less than 10, review monitored years to verify a range of climatic conditions
[2]See Table 16 and Table 17

6. Water-Data Availability

This chapter explores the extent to which data are available that meet the monitoring criteria identified in Table 18. Table 19 summarizes all the monitoring sites discussed in this chapter.

6.1 Large watersheds

The three large watersheds in the Lake Erie drainage basin that meet the monitoring criteria for testing the large watershed hypothesis each have at least one active monitoring site, as shown in Figure 20. Monitoring sites that measure load to the lake are commonly found upstream from the river mouth; to avoid “backwater effects” from Lake Erie, these sites are typically located in the final reach of consistent unidirectional flow in the tributary. Sites A, C, and D are monitored daily by Heidelberg University, with more frequent samples measured during storm flows, and the USGS maintains streamgages at or near all of these sites. These sites meet all the monitoring criteria identified in Table 18, including daily plus storm sampling frequency. Site A in the River Raisin also has a USGS-operated multi-parameter continuous monitor that includes turbidity. At site B (Maumee River, 04193500), the USGS maintains a streamgage and monitors periodic water-quality (monthly plus supplemental sampling). At site C (Maumee River, 04193490) the USGS operates a multi-parameter continuous monitor including turbidity and two types of continuous-nitrate monitors. The monitoring conducted by the USGS and Heidelberg University at sites B and C on the Maumee River provides data needed for accurate determination of nutrient loads in the Maumee River, consistency of long-term historical records and sampling methods to assess nutrient trends, and information to evaluate the performance and reliability of continuous-nitrate monitors for potential widespread use. As long as monitoring continues for these three large watersheds, there are no water-quality data gaps for testing the large watershed case-study hypothesis.

6.2 Small watersheds

Figure 21 shows active small watershed monitoring sites that meet, or nearly meet, the monitoring criteria identified in Table 18 that were available through the nutrient data set. Sites E, F, and G from Figure 21 are active monitoring sites that meet all the parameter, streamgage, and sampling-frequency criteria. Sites E and F are monitored by Heidelberg University using a daily plus storm sampling frequency; Site E (Unnamed Tributary to Lost Creek) was initiated recently and has less than 10 years of data, while site F (Rock Creek) has been monitored for close to 30 years. Site G is a new site monitored by the USGS using a monthly plus supplemental sampling strategy in addition to continuous turbidity monitoring. This site was initiated together with two upstream edge-of-field monitoring sites using a nested monitoring design. Sites E-G monitor three small watersheds. The Lake Erie drainage basin includes 384 HUC-12 watersheds that are 50 square miles or less with greater than 40-percent row-crop cover. These three sites do not offer sufficient spatial representation of the basin.

Table 19. Water-monitoring sites in the nutrient data set active as of 2014 that meet, or nearly meet, all monitoring criteria for detecting water-quality change resulting from agricultural management practices. Map reference letter refers to sites shown in figures 19-22.
[**Abbreviations:** mi^2, square miles; %, percent; TP, total phosphorus; DRP, dissolved reactive phosphorus; NERRS, National Estuarine Research Reserve System; USDA-ARS, US Department of Agriculture-Agricultural Research Service; USGS, US Geological Survey]

Map reference letter	Monitoring site identification number	Monitoring organization	Site name	Basin area (mi^2)	Row crop area (% of basin)	Water-quality parameter	Years of record	Sampling frequency
			Meets all monitoring criteria for testing hypothesis					
A	miUSGS:04176500 04176500[1]	Heidelberg University [Flow: USGS]	River Raisin near Monroe, MI	1043	49%	TP, DRP, streamgage	>30	Daily plus storm; USGS monitors continuous turbidity
B	ohUSGS:04193500 04193500[1]	USGS	Maumee River at Waterville, OH	6330	73%	TP, DRP, streamgage	>20[2]	Monthly plus supplemental
C	ohUSGS:04193490 04193500[1]	Heidelberg University, USGS	Maumee River near Waterville, OH	6330[3]	73%	TP, DRP, streamgage	>30	Daily plus storm; USGS monitors continuous turbidity
D	ohUSGS:04198000 04198000[1]	Heidelberg University [Flow: USGS]	Sandusky River near Fremont, OH	1252	78%	TP, DRP, streamgage	>30	Daily plus storm
E	ohUSGS:04185440 04185440[1]	Heidelberg University [Flow: USGS]	Unnamed Tributary to Lost Creek near Farmer, OH	4.23	78%	TP, DRP, streamgage	5-10	Daily plus storm
F	ohUSGS:04197170 04197170[1]	Heidelberg University [Flow: USGS]	Rock Creek at Tiffin, OH	34.6	73%	TP, DRP, streamgage	>20	Daily plus storm

Map reference letter	Monitoring site identification number	Monitoring organization	Site name	Basin area (mi^2)	Row crop area (% of basin)	Water-quality parameter	Years of record	Sampling frequency
			Lacks duration of monitoring					
G	04188496	USGS	Eagle Creek above Findlay, OH	51.0	79%	TP, DRP, streamgage	<5	Monthly plus supplemental; continuous turbidity
			Lacks sampling frequency					
H	ohBR 04199155[1]	NERRS	Old Woman Creek at Berlin Road near Huron, OH	22.1	70%	TP[4], DRP, streamgage	5-10	Monthly plus supplemental March-Dec
I	inINSJ-INSJBLG	USDA-ARS	Tile-fed drainage ditch 2.5 mi SSE of Waterloo, IN	5.47	83%	TP and DRP, streamflow	5-10	Daily plus storm Apr-Nov[5]
J	inINSJ-INSJCLG	USDA-ARS	Tile-fed drainage ditch 1.4 mi SE of Waterloo, IN	5.33	73%	TP and DRP, streamflow	5-10	Daily plus storm Apr-Nov[5]
K	inINSJ-INSJAXL	USDA-ARS	Tile-fed drainage ditch 1.4 mi SE of Waterloo, IN	16.6	78%	TP and DRP, streamflow	5-10	Daily plus storm Apr-Nov[5]
L	inINSJ-INSJALG	USDA-ARS	Tile-fed drainage ditch 3.5 mi ENE of Waterloo, IN	7.47	77%	TP and DRP, streamflow	5-10	Daily plus storm Apr-Nov[5]
M	inINSJ-INSJAME	USDA-ARS	Tile-fed drainage ditch 6.3 mi ENE of Waterloo, IN	1.15	79%	TP and DRP, streamflow	5-10	Daily plus storm Apr-Nov[5]

[1] USGS streamgage identification number
[2] DRP collected <10 years
[3] Drainage area for USGS streamgage 04193500; the drainage area at the water quality monitoring site is 6,313 sq. mi.
[4] TP sampling by OH EPA and Heidelberg University has started and stopped multiple times over the data record at this site.
[5] Primary sampling months. Samples were collected intermittently between December and March during some years.

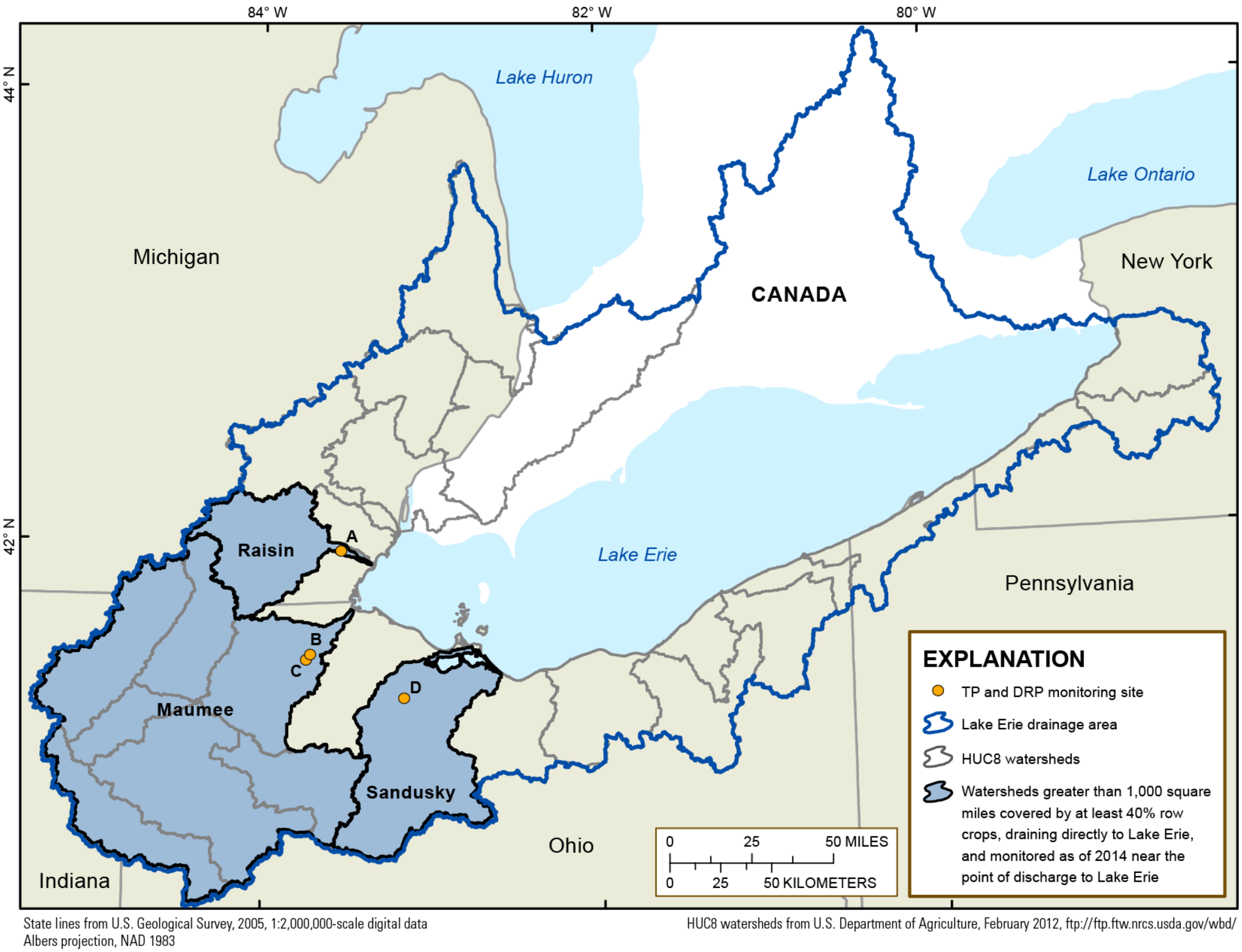


Figure 20. Water-monitoring sites in watersheds of 1,000 square miles or greater that drain directly to Lake Erie with more than 40-percent row-crop area that are monitored as of 2014 near the point of discharge to Lake Erie for total phosphorus (TP) and dissolved reactive phosphorus (DRP) with active streamgages. Map labels are defined in Table 19.
[Abbreviations: HUC, hydrologic unit code; %, percent]

Sites H-M (Figure 21) have been sampled infrequently during the winter months over several years of record. Site H (Old Woman Creek) is maintained by the NOAA NERRS Program, with about 10 years of DRP data using a monthly plus supplemental sampling strategy that does not include winter months; TP monitoring at this site has started and stopped multiple times over the available data record. A group of 5 sites maintained by the USDA in the St. Joseph watershed are represented by sites I-M on Figure 21. These sites have nearly 10 years of record using a daily plus storm-sampling approach during the growing season but inconsistent sampling of winter months, and USDA has been collecting agricultural management practice data for these sites. Two monitoring sites from this program were recently moved, so they do not meet the currently active criterion and data were not available for the new locations. This USDA monitoring program also includes 8 additional edge-of-field monitoring sites nearby.

Figure 22 superimposes the highest contributing areas for each of the six priority watershed characteristics described in section 4.5 on one map with the locations of the monitoring sites from Figure 21. The map in Figure 22 shows sites E and H in areas with overlapping priority characteristics. Site E monitors a watershed with high TP and DRP yield as modeled by SWAT, as well as high vulnerability to soil runoff. Site H monitors a watershed with high TP yield from fertilizer as modeled by SPARROW and high vulnerability to soil leaching. Sites F and I-M monitor watersheds in which much of their upstream areas have high vulnerability to soil leaching or runoff. Site G does not have significant coverage of any of these priority characteristics upstream.

Sites E and G-M have monitoring records shorter than 10 years, but they are all active monitoring sites with the ability to reach that milestone in the near future. The minimum length of record needed to detect change at a monitoring site depends on the magnitude of change expected and the background variability of the parameter of interest. If appropriate agricultural management practices are implemented consistently throughout a small watershed, a 40-percent reduction in TP might be expected and a monitoring record between 5 and 10 years is needed to detect that change with power and significance at the 20-percent error level (Table 16). Site F (Rock Creek) has a data record exceeding 20 years, but the analysis completed for this case study did not identify a statistically significant trend at this site. Management practice implementation data are needed for the Rock Creek watershed over the entire monitoring period to make any observations about agricultural management practice effectiveness from these results. A data record longer than 10 years is needed to detect a change in TP that is smaller than 40 percent, so all the available sites need a longer data record to detect any statistically significant change.

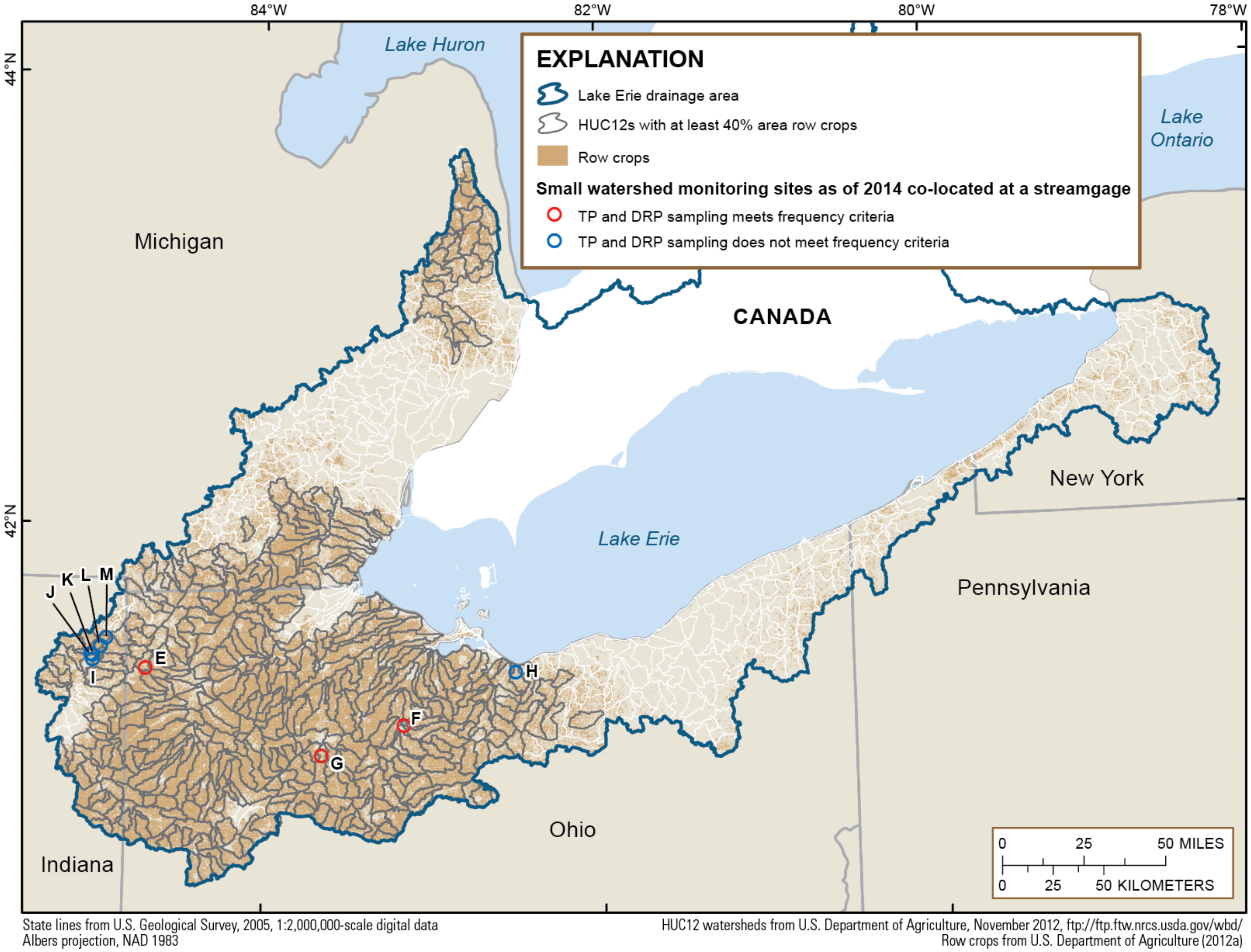
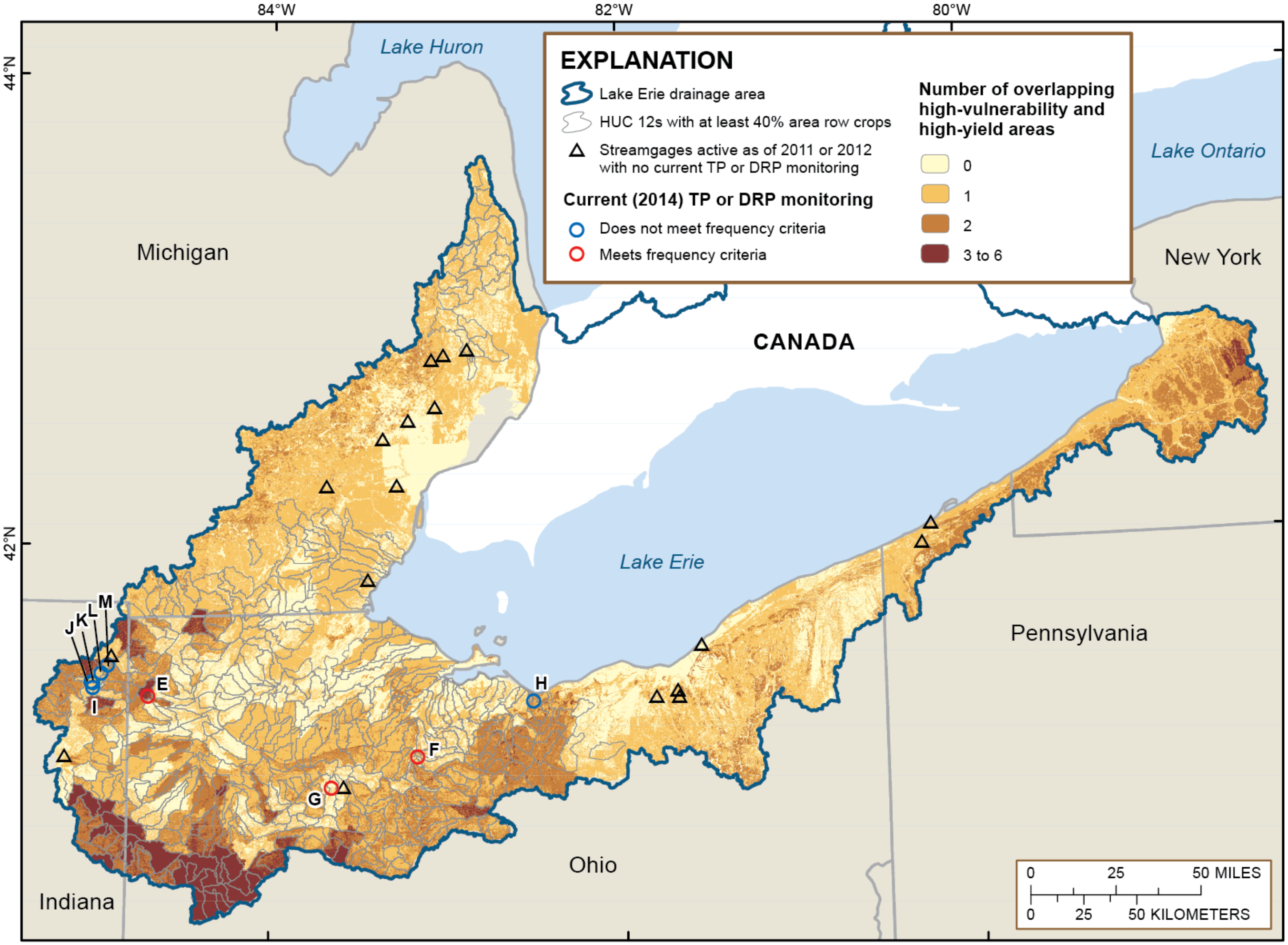


Figure 21. Water-monitoring sites in small watersheds of approximately 50 square miles and less with streamflow data in the Lake Erie drainage basin that were monitored as of 2014 for total phosphorus (TP) and dissolved reactive phosphorus (DRP) in areas with 40-percent row-crop coverage. Map labels are defined in Table 19.
[**Abbreviations:** HUC, hydrologic unit code; %, percent]



State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Albers projection, NAD 1983

Soil vulnerability from Dean Oman, U.S. Department of Agriculture, written commun., 2014 [Original source: Lund, et al. (2011)]
Phosphorus yields from Saad, D.A., et al. (2011); and from Scavia, D., et al. (2014)
HUC12 watersheds from U.S. Department of Agriculture, November 2012, ftp://ftp.ftw.nrcs.usda.gov/wbd/

Figure 22. Spatial distribution of small agricultural watersheds and watershed characteristics related to phosphorus sources and transport to streams in the Lake Erie drainage basin, showing active (2014) total phosphorus (TP) and dissolved reactive phosphorus (DRP) monitoring sites and streamgages in small watersheds of approximately 50 square miles or less. Map labels are defined in Table 19.
[Abbreviations: HUC, hydrologic unit code]

Sites E and F meet all the monitoring and site location criteria for the case study; water-quality data should continue to be collected at these sites to answer the case-study policy question, and agricultural management practice implementation data should be collected for these areas. Sites H-M do not meet the sampling-frequency requirements for year-round sampling described in section 5.3, but they monitor watersheds with priority characteristics. Sites I-M are clustered close together and do not offer significant spatial representation among this group of monitoring sites. Sampling should be increased to include monthly year-round samples in addition to the current sampling frequency at site H and one of sites I-M, with both TP and DRP sampled consistently going forward; collection of agricultural management practice implementation data should be initiated and continued as appropriate for the selected sites. These sites cover four out of the minimum six small watershed monitoring sites needed to test the small watershed hypothesis.

Site G meets all the sampling criteria, but it does not monitor a watershed with priority characteristics. Monitoring should be prioritized at this location if the local agricultural community is actively involved and participating in management practice implementation within this watershed. However, because this site does not monitor a watershed with significant coverage of priority characteristics, at least two additional small watershed monitoring sites are recommended to achieve the minimum of six monitoring sites. Table 20 summarizes the existing small watershed monitoring sites and recommendations for continued monitoring.

6.3 Data consistency and quality assurance

This case study considered whether water-quality data collected by different monitoring agencies in different jurisdictions can be taken together to support regional decision-making given the variety of objectives, processes, and procedures used in different jurisdictions. Only two organizations, Heidelberg University and the USGS, are collecting water-quality data that meet all the monitoring criteria for testing the hypotheses. These organizations are coordinating their ongoing work in the Lake Erie drainage basin. Table 19 and Figure 23 show all the monitoring sites that were active as of 2014 and that meet all the monitoring criteria identified in Table 18. Data quality assurance procedures are discussed below for each agency.

Heidelberg University follows documented data quality assurance procedures (Baker, 2009). Samples are tested by the National Center for Water Quality Research laboratory, which meets the requirements for level three credible data (highest level) by the OH EPA. These monitoring sites are all associated with USGS streamgages. Samples are collected through the use of refrigerated automatic samplers three times daily at 8-hour intervals, and 21 samples plus three quality-control samples are collected each week. Method performance evaluation criteria are specified in the standard operating procedures (SOPs), and replicates, blanks, and spikes are included in every sample batch. Heidelberg University collects data at sites A, C, D, E, and F.

The USGS follows a set of documented methods, data analysis, and data quality control assurance procedures for the collection of streamflow and water-quality data. Overall, the USGS stresses objective and replicable data collection procedures that are transparent and peer reviewed. Data that are collected for publication must be documented to describe the methods or techniques used to collect, process, and analyze the data. For an overview of methods used to collect, analyze, and quality assure streamflow data see Rantz et al. (1982); for detailed descriptions of updated and new methods for collecting streamflow data, see individual chapters of USGS Techniques of Water Resources Investigations Report Book 3-Application of Hydraulics, Section 3-Surface Water Techniques available at *http://pubs.usgs.gov/twri/*. For information on methods used to collect and quality assure water-quality samples see the USGS National Field Manual available at *http://pubs.water.usgs.gov/twri9A*. For documentation of specific laboratory or field analytical methods and related quality assurance/quality control procedures see *http://water.usgs.gov/owq/methods.html*. The USGS collects data at sites B and G and maintains streamgages at A-H.

If monitoring is modified at sites H-M to meet the criteria for testing the small watershed hypothesis, data-quality, sampling, and data-sharing procedures should be reviewed for these sites. USDA flow monitoring at sites I-M should be reviewed for comparability to USGS streamflow data. More timely access to USDA data is necessary if these data are to be used to answer the case-study policy question on a policy-relevant timeline. The most current data that could be obtained from the USDA for this case study was from 2009, compared to 2013 for Heidelberg University and USGS.

Table 20. Summary and recommendations for small watershed monitoring sites. Map reference letters refer to sites shown in figures 21-23.
[Abbreviations: SWAT, Soil and Water Assessment Tool; SPARROW, SPAtially Referenced Regressions On Watershed attributes]

Map reference letter	Monitoring criteria	Priority watershed characteristics	Monitoring recommendation
E	Meets all monitoring criteria	• High TP yield as modeled by SWAT • High DRP yield as modeled by SWAT • High vulnerability to runoff	• Continue monitoring • Collect management practice implementation data
F	Meets all monitoring criteria	• High vulnerability to soil runoff	• Continue monitoring • Collect management practice implementation data
G	Meets all monitoring criteria	• Watershed does not have substantial priority characteristics	• Continue monitoring[1] • Collect management practice implementation data
H	Does not meet frequency requirements: lacks winter monthly monitoring	• High TP fertilizer yield as modeled by SPARROW • High vulnerability to soil leaching or soil runoff	• Increase monitoring to include monthly year-round sampling in addition to current sampling frequency • Collect both TP and DRP at the same sampling frequency • Collect management practice implementation data
I - K	Do not meet frequency requirements: lack consistent winter monthly monitoring	• High vulnerability to soil runoff	• Increase monitoring to include monthly year-round sampling at one of these sites in addition to current sampling frequency • Continue to collect management practice implementation data at these sites
L-M	Do not meet frequency requirements: lack consistent winter monthly monitoring	• High vulnerability to soil runoff • High vulnerability to soil leaching	• Increase monitoring to include monthly year-round sampling at one of these sites in addition to current sampling frequency • Continue to collect management practice implementation data at these sites

[1]Monitoring at this location is a priority if the local agricultural community is actively involved and participating in agricultural management practice implementation in this watershed.

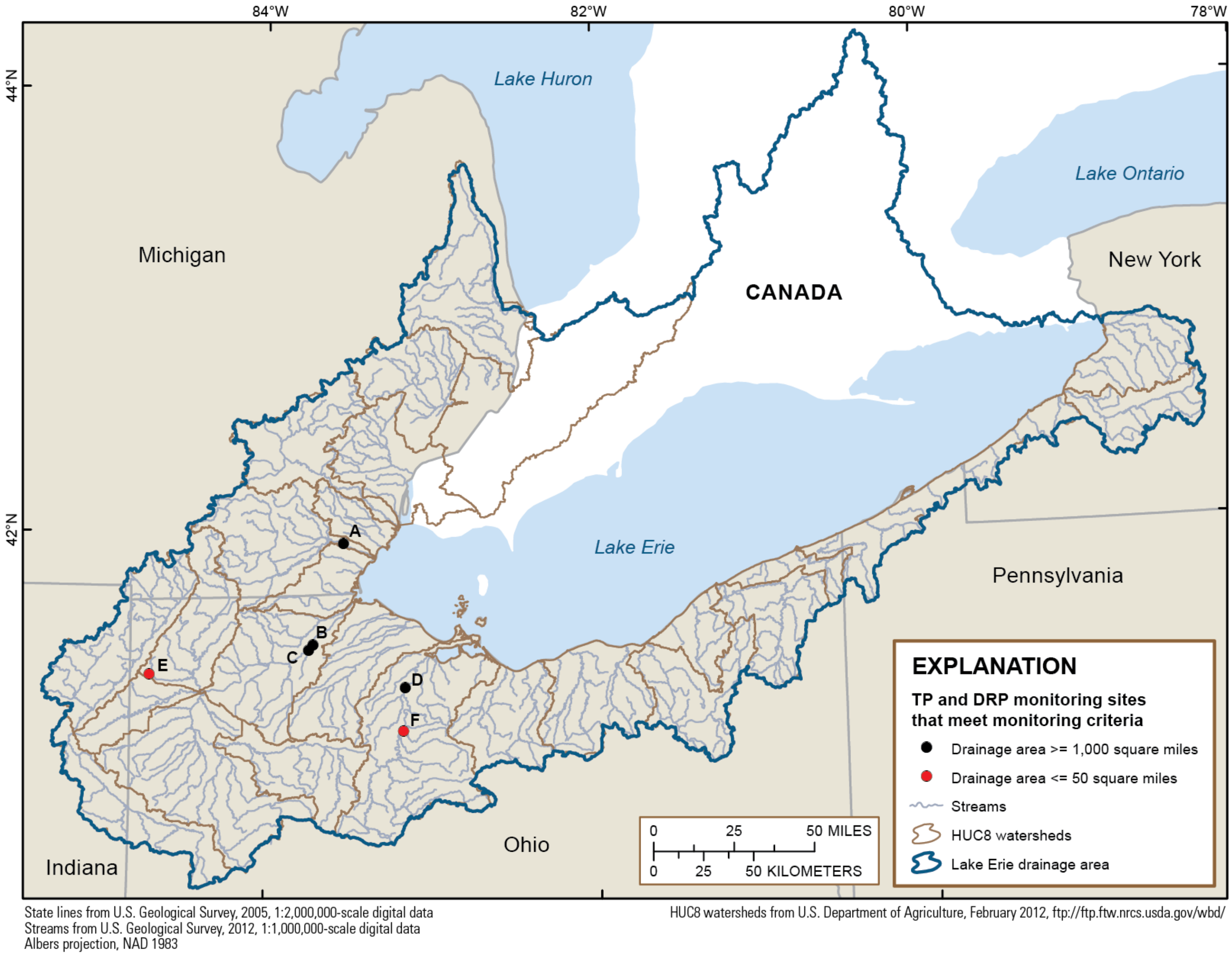


Figure 23. Water-monitoring sites in the Lake Erie drainage basin in priority watersheds that meet all case-study criteria for detecting water-quality change resulting from agricultural management practices. The monitoring criteria are identified in Table 18 and map labels are defined in Table 19. [Abbreviations: TP, total phosphorus; DRP, dissolved reactive phosphorus; HUC, hydrologic unit code]

6.4 Water data usability

Water-quality monitoring programs are usually designed to meet a stated objective or follow a historical precedent. Data collected for one monitoring objective may not be directly applicable to another objective, due to the location of monitoring sites, frequency of monitoring, parameters measured, and analytical methods used. The water-quality records identified through this case study, summarized in Table 7, were generated by 17 organizations that collected nutrient-related data in the Lake Erie drainage basin. Insufficient and inconsistent documentation of available data limited the utility of these existing data sets. As discussed in section 4.3, significant time and effort were required to locate, obtain, and format water-quality records from multiple organizations that use different sampling plans and data documentation practices. Data sharing and data accessibility were also limiting factors in data availability in this case study. It is possible that despite the work completed for this case study, additional relevant data may exist that were not compiled for this study. The Water Quality Portal (National Water Quality Monitoring Council, 2014a) is a cooperative service that integrates publicly available water-quality data from NWIS, STORET, and the USDA ARS Sustaining The Earth's Watersheds - Agricultural Research Database System (STEWARDS), and includes data collected by more than 400 state, tribal, and local organizations. The Water Quality Portal provides an interface that allows a user to download water-quality data from these systems in a consistent format. Data collected at only 26 percent of the monitoring sites identified through this case study in the Lake Erie drainage basin were available through the Water Quality Portal, and only 8 percent of the water-quality data records are available through the Portal. This limited availability of data is because the Lake Erie data set is heavily biased by the data from the Heidelberg Tributary Loading Program which is not in the Portal but is available for download on a Heidelberg University website. In fact Heidelberg University is responsible for generating nearly 53 percent of the approximately 1.2 million surface-water quality records compiled for this case study (Table 7), yet less than 1 percent of the monitoring sites.

Finally, for water data to be useful for addressing the policy question, they must be compatible in terms of sampling plans and protocols, analysis, and interpretation. Several agencies and organizations collect small watershed data that meet or nearly meet the data needs identified in section 6.2, including USGS, Heidelberg University, USDA, NOAA NERRS, and additional agencies described in the addendum to this report (Betanzo et al., 2015). However, these agencies all use different sampling plans that limit the ability to compare trends in concentration and load over time at these monitoring sites.

7. New Water-Quality Data Needed to Answer the Case-Study Policy Question and Associated Costs

While the data needed to test the large watershed hypothesis are currently being collected, there is limited, consistent, long-term monitoring of small watersheds in the Lake Erie drainage basin. This chapter identifies the water data needed to fill the data gap for small watershed monitoring in the Lake Erie drainage basin and provides general cost estimates for collecting the data.

As indicated in section 6.2 and Figure 22, a combination of continued and increased monitoring at existing monitoring sites and the addition of new monitoring sites would be needed to achieve the minimum of six monitoring sites for testing the small watershed hypothesis (Table 18). This chapter does not prescribe specific monitoring sites but identifies a process for identifying new sites that are good candidates for testing the small watershed hypothesis.

7.1 Water-data needs

Section 6.2 and Figure 22 identified two monitoring sites that should continue monitoring and two monitoring sites that should increase monitoring throughout the year to meet the criteria for testing the small watershed hypothesis. At least two new sites are needed to meet the minimum of six monitoring sites. Additional sites would help improve spatial representation of the Lake Erie drainage basin and may support multiple watershed or gradient statistical designs (Table 11).

Substantial planning and coordination are needed to identify appropriate water-quality monitoring sites and agricultural communities willing to participate in the use of new and extensive agricultural management practices. A process for identifying new water-quality monitoring sites is presented here and is summarized in Table 21. The first step for identifying new monitoring sites is to consider watersheds exhibiting priority characteristics as identified in Figure 22. Areas where priority characteristics overlap are the highest-priority areas for new monitoring, but all the highlighted areas should be considered as candidates. Priority areas without existing monitoring sites can be identified using Figure 12 and Figure 22. Areas with significant coverage of priority watershed characteristics but no existing monitoring sites include the following, which are shown in Figure 12:

- The southwest portion of the drainage basin. This is the only part of the basin with high TP yield from manure, as modeled by SPARROW. This area also represents high TP yield from fertilizer, high TP yield as modeled by SWAT, and high vulnerability to soil runoff.
- The area with high TP yield from fertilizer and high vulnerability to soil runoff just south of site H.
- The high DRP yield areas just east and west of the Ohio/Indiana border.
- The areas with high TP yield from fertilizer both north and south of sites F and G.

Table 21. Process for identifying new monitoring sites for testing the small watershed hypothesis.

Process for identifying new monitoring sites for testing the small watershed hypothesis.
1. Identify candidate watersheds: • Watersheds with priority characteristics (high phosphorus yield and/or high vulnerability to soil loss) • Watersheds with existing streamgages and/or water-quality data.
2. Examine location of candidate watersheds relative to other monitoring sites that might allow for nested monitoring designs, such as edge-of-field, mid-size, and large watershed monitoring sites.
3. Examine candidate watersheds on a case-by-case basis for local information: • Representation of tile drainage in the watershed, • Untreated areas with potential for high implementation rates of new management practices • Willingness of agricultural community to implement and maintain new management practices throughout candidate watershed, and • Willingness of agricultural community to share management practice and land-management data with monitoring agency.
4. There may be situations in which monitoring sites in watersheds without priority characteristics are the most feasible study locations.

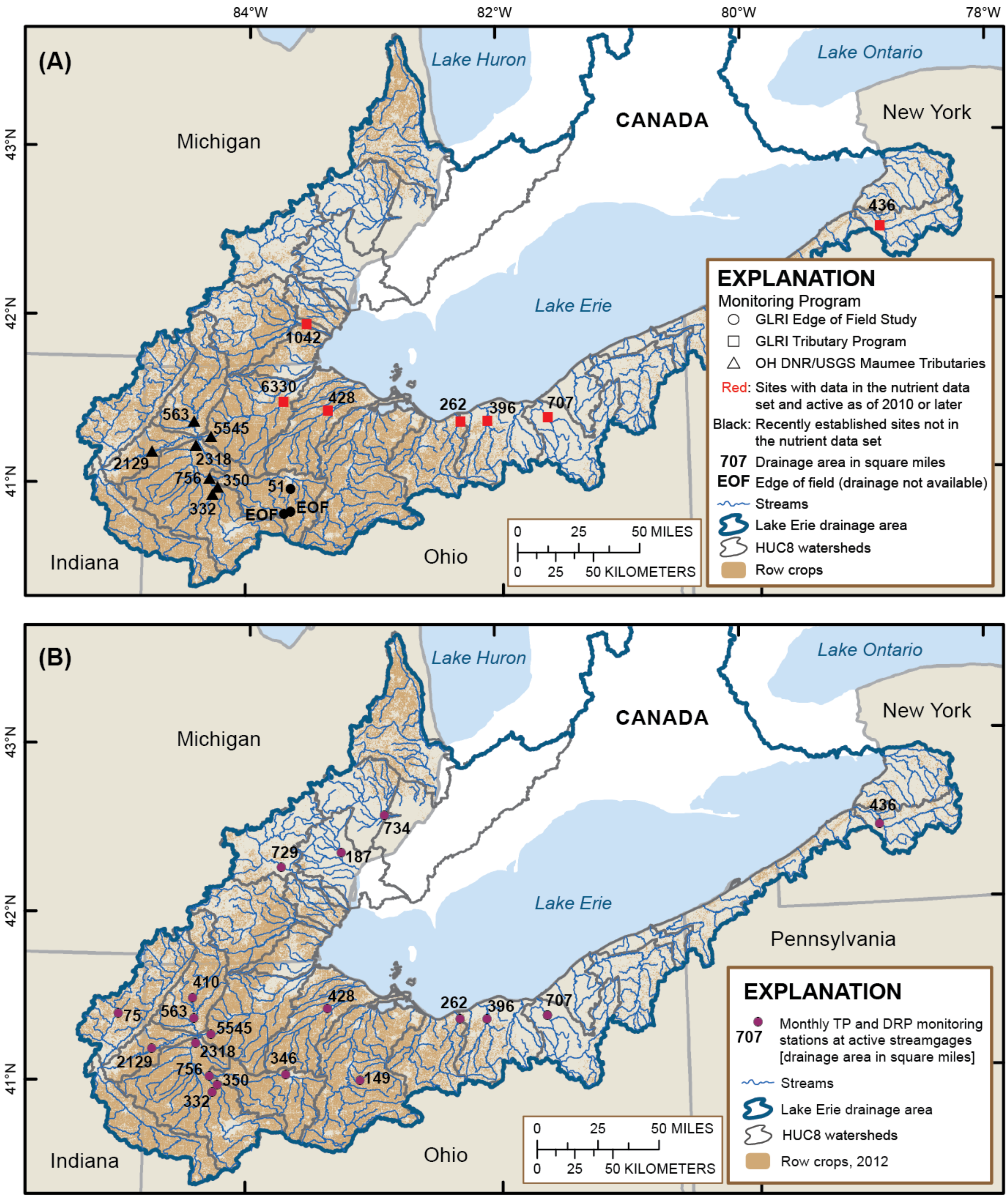
Next, it is most informative, as well as cost effective, to use monitoring sites that have existing water-quality data and streamgages. USGS streamgages were the only source of streamflow information available to complete this part of the analysis. Figure 22 identifies active USGS streamgages in the Lake Erie drainage basin in watersheds approximately 50 square miles or less that are not currently collecting TP and DRP data. The majority of the existing streamgages are in urban areas. Only one current streamgage in southeast Michigan is in an area with a priority watershed characteristic, which is high vulnerability to soil leaching. There are no existing USGS streamgages in the specific priority areas identified above that lack current water-quality monitoring sites.

There are additional active and discontinued small watershed TP and DRP monitoring sites where samples were collected less frequently than monthly, during only part of the year, and/or lacked a streamgage that were not fully evaluated as candidates for testing the small watershed hypothesis. Some of these monitoring sites, such as those maintained by the St. Joseph River Watershed Initiative that are sampled weekly during selected months, may be candidates for increased monitoring to fill the small watershed data gaps; some sites may lack monthly data due to intermittent or frozen streams. Site-by-site analysis is required to identify whether there are sufficient existing water-quality or streamflow data at sites in identified watersheds to make them good candidates for increased monitoring to answer the case-study policy question.

Small watershed monitoring sites can provide valuable information when they are nested with other active monitoring sites. This means that large monitored watersheds should encompass small watersheds with new monitoring sites so sequential water-quality changes can be detected. This allows results from small watershed agricultural management practices to be observed and “scaled up” as nutrients transported from multiple small watersheds come together in larger watersheds. In turn, small watersheds that include edge-of-field monitoring sites would also be of high value for monitoring because they would allow information collected at the field scale to be evaluated and compared at the watershed scale. As a second step, candidate watershed monitoring sites should be prioritized when they allow for nesting with edge-of-field and larger watershed monitoring sites. Figure 24 (A) and (B) show new and long-term monitoring sites in the Lake Erie drainage basin that include edge-of-field, medium, and large inland watersheds that can be used as part of a nested monitoring design when selecting new small watershed monitoring sites.

The third step is to evaluate candidate watersheds on a case-by-case basis to determine local characteristics. As discussed in section 2.4.3, tile drainage is a pervasive feature in the western Lake Erie drainage basin. Monitoring sites should ideally represent a range of tile-drain density to better characterize the tile-drain contribution to nutrient transport at the watershed scale, but it may not be possible to identify appropriate monitoring sites representing a range of tile-drain density. Figure 14 shows regional-scale estimates for tile-drain density in the Lake Erie drainage basin, but these regional-scale data are not adequate for assessing tile drainage locally. Candidate watersheds should be reviewed at the local level to confirm the density of tile drainage and locations of outfalls relative to potential monitoring sites to determine whether water-quality data collected there would be highly influenced by the presence of tile drains. If feasible, new small watershed monitoring sites should reflect watersheds with a range in density of tile drainage.

Another important consideration in the selection of new small watershed monitoring sites is the ability to achieve a high density of new and appropriate agricultural management practices to effect change in nutrient transport. If appropriate agricultural management practices are already in place throughout the watershed (Table 1), there will be limited opportunity to create measurable reductions in phosphorus transport through new practices. Due to the complexities of nutrient transport and management practices described in Table 1, agricultural specialists would be needed to identify the most appropriate agricultural management practices to best target high phosphorus source areas and transport pathways.



State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Streams from U.S. Geological Survey, 2012, 1:1,000,000-scale digital data
Albers projection, NAD 1983

Row crops from U.S. Department of Agriculture (2012a)
HUC8 watersheds from U.S. Department of Agriculture, February 2012, ftp://ftp.ftw.nrcs.usda.gov/wbd/

Figure 24. Water monitoring sites in the Lake Erie drainage basin with monthly or more frequent total phosphorus (TP) and dissolved reactive phosphorus (DRP) sampling located at streamgages active as of 2014 that may be candidates for spatially nested monitoring designs. (A) Locations of monitoring sites recently established or re-activated during 2011 through 2013, including Great Lakes Restoration Initiative (GLRI) and Ohio Department of Natural Resources/ U.S. Geological Survey (OH DNR/USGS) monitoring sites. (B) Water monitoring sites with drainage areas larger than 50 square miles, excluding the large watershed sites at locations near outlet to Lake Erie. Map labels provide drainage area in square miles.
[Abbreviations: HUC, hydrologic unit code]

Finally, the willingness of farmers, land owners, and the agricultural community at large to participate in implementing and sharing agricultural management practice data is critical to the success of any monitoring sites to help answer the case-study policy question. It is possible to identify the perfect monitoring site in a priority watershed, perhaps with an existing streamgage, but if the local agricultural community does not want to participate in implementing agricultural management practices and sharing their implementation data, the water-quality monitoring will be of little value. There may be situations in which water-quality monitoring sites in non-priority watersheds are the most feasible locations for this type of evaluation, which may be the case for site G. Site G may produce important information if the agricultural community in the watershed has already agreed to participate and share agricultural management practice data. At the same time, there are likely other small watersheds that may be more effective candidates for new water-quality monitoring to detect the effectiveness of agricultural management practices.

To answer the case-study policy question, the process outlined in Table 21 should be used to identify at least two new monitoring sites. Table 22 summarizes the data that should be collected at the new monitoring sites. If funding is available for more sites, additional sites should be added to improve spatial representation. A new suite of small watershed water-quality monitoring sites, such as those described in the addendum to this report (Betanzo et al., 2015), may provide the opportunity to use the multiple-watershed or gradient statistical designs discussed in section 5.2. Care should be taken to coordinate sampling plans among new and existing monitoring sites and monitoring agencies so that data collection, analysis, and interpretation allows for comparison of results across the small watershed monitoring sites. Finally, adaptive management approaches should be used for all new monitoring sites. Data collected at all monitoring sites should be analyzed on an annual basis to determine whether monitoring plans should be adapted to better characterize concentration changes over the stream hydrographs and to identify opportunities to use monitoring resources more efficiently. Data analysis must be incorporated into the study design to ensure this will happen.

In addition to new water data, improvements in water-data usability are also needed. For water data to be used to answer the case-study policy question, monitoring agencies and organizations should coordinate sampling plans among new and existing monitoring sites so data collection, analysis, and interpretation can be compatible and comparable. This effort can be achieved through a coordinating entity that facilitates collaboration on sampling plans, data sharing, and data analysis in the Lake Erie drainage basin. Improved data documentation and data sharing will facilitate the use of water data for answering the case-study policy question. Tools such as Water Quality Exchange (WQX) (U.S. Environmental Protection Agency, 2015) and the Water Quality Portal provide the infrastructure for organizations to format and share their data, but greater participation in these types of services is needed to get the greatest value out of these services. Consistent, thorough data documentation and wider availability of data sources through services like the Water Quality Portal will increase the value of water-quality data from all monitoring agencies and reduce the amount of time needed to access and prepare data for new applications. A continued commitment to water-quality data-sharing systems is essential for maximizing use of existing water-quality data.

Table 22. Data collection needed for new small watershed monitoring sites, for a minimum planned monitoring period of 10 years.

Parameter	Sampling frequency: (select one where multiple options are presented)
Laboratory measurements[1]: Total Phosphorus Dissolved Reactive Phosphorus Total Nitrogen Silica Suspended Sediment	**Monthly plus supplemental** 12 monthly + 12 targeted samples **Two-year intensive monitoring followed by adaptive management:** 100 samples/year for two years + approx 24/year thereafter **Daily plus storms** approx. 500 samples/year **Monthly plus continuous monitoring** 24 samples/year for two years + 12 samples/year thereafter
Field measurements[1]: Water Temperature Air Temperature pH Dissolved oxygen	During sample visits
Streamflow	Continuous via USGS streamgage or equivalent
Biodindicators[1]: Macroinvertebrates Periphyton (attached algae)	Annually
Land-management data for the monitored watershed	Annually
Management practice implementation data for the monitored watershed	Annually

[1]From Table 3

7.2 Streamflow monitoring

In order to calculate load, a streamflow measurement is required for each water-quality sample. Streamgages are ideal for this purpose as they collect data continuously. Sites A-H in Table 19 are each associated with a USGS streamgage. Sites I-M use streamflow monitoring conducted by USDA. Establishing new streamgages for new water-quality monitoring sites will be a substantial cost associated with filling the data gaps identified in this report. The number of streamgage sites that monitor small watersheds in the Lake Erie drainage basin has decreased over the past 20 years as shown in Figure 25, although the most recent years show modest increases. As discussed in section 7.1, only one active streamgage site is located in a watershed with priority characteristics. New streamgages will be needed to support new water-quality monitoring sites for testing the small watershed hypothesis.

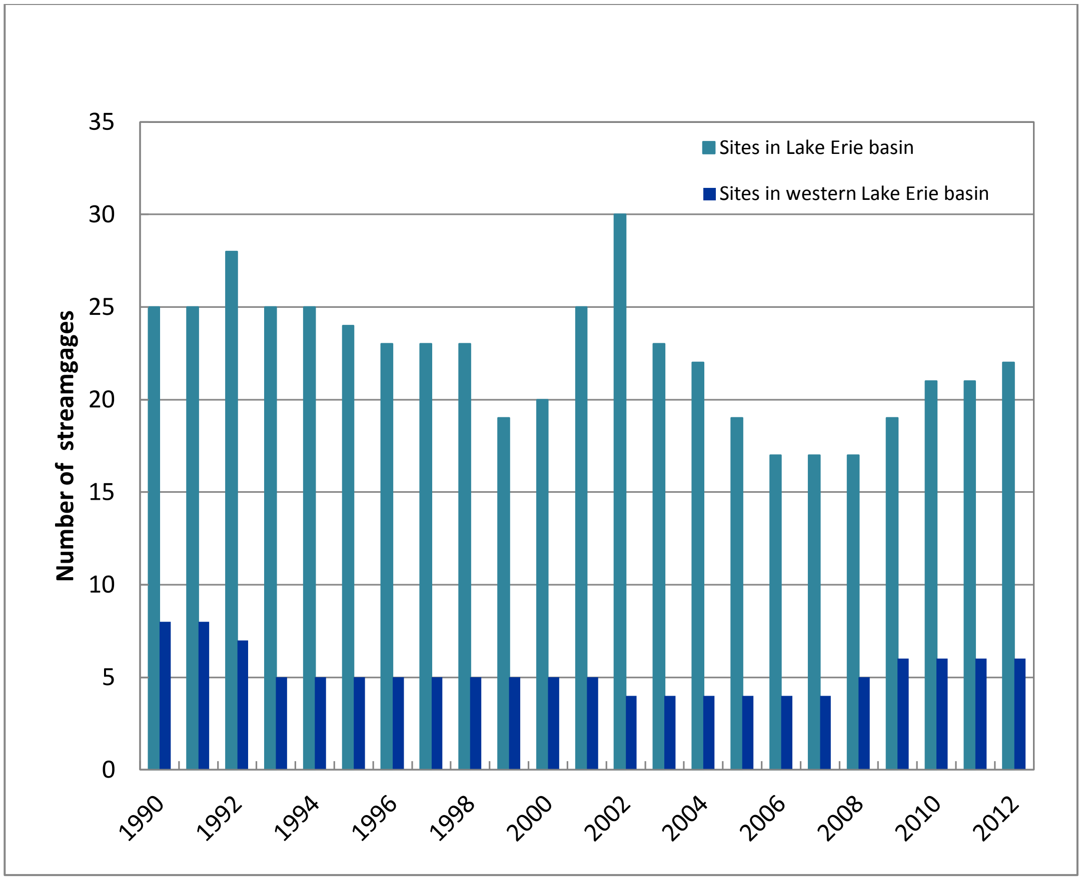


Figure 25. Active U.S. Geological Survey streamgages in watersheds of 50 square miles or smaller in the Lake Erie drainage basin by year, 1990-2012. The area defined as the western Lake Erie drainage basin extended from the Ottawa-Stony River HUC8 in the north through the Huron-Vermilion HUC8 in the south.
[Abbreviations: HUC, hydrologic unit code]

7.3 Cost of monitoring

The cost of water monitoring to answer the case-study policy question is substantial. However, these monitoring costs should be considered in the context of the economic value of Lake Erie and the economic impact of HABs and hypoxia in the lake. As discussed in 2.3.3, effective agricultural management practices are a key part of the solution to HABs in Lake Erie due to the magnitude of nonpoint-source contributions to the lake compared to other nutrient sources. A lack of new monitoring to verify agricultural management practice effectiveness and to answer the case-study policy question necessarily means delays in understanding and achieving improvements in the health of Lake Erie. According to the Lake Erie Improvement Association (2012), Lake Erie supports an annual $11.5 billion tourism industry. The International Joint Commission (2014) estimated the impact of the 2011 HAB on recreational fishing and beach recreation at $2.4 million and $1.3 million, respectively. Media reports from the August 2014 “do not drink” advisory in Toledo indicate city operational costs of $130,000 over 3 days, not including losses to local businesses in restaurants that were forced to close during the advisory (Henry, 2014). The annual cost of new monitoring, as estimated in the following paragraphs, is small in comparison to the past and potential future economic impacts of HABs in Lake Erie.

The cost of new monitoring presented here relies on data provided by the USGS and Heidelberg University. Table 23 shows average surface-water-related sampling costs for stream and river sites currently being monitored (2014) as part of the USGS NAWQA Program based on fiscal year (FY) 2012 data; the table also includes estimates of what these costs would have been in December of 2013 to account for inflation. At these sites, water-quality samples are collected between 6 and 26 times per year to meet specific monitoring objectives (median = 18). Average costs were calculated on a per sample basis for each site. Costs include salary, travel, supplies, equipment, miscellaneous, laboratory analysis, and administrative services. Note that salary costs include field labor, office labor including sample preparation, clean-up, sample shipping, data entry, and quality control and quality assurance management costs. The average FY 2012 cost per sample was about $4,200 and the cost range per sample was $2,900 to $5,900. Adjusting these costs by a factor of 4.7 percent (U.S. Department of Labor, 2014) to account for inflation results in an average cost of nearly $4,400 per sample as of December, 2013, and the range of cost per sample across all sites is $3,000 to $6,200.

Table 23. Estimated cost per nutrient sampling event based on cost of operating a typical U.S. Geological Survey stream water-quality monitoring site, exclusive of streamgages. Average costs are presented from 43 stream and river monitoring sites, adjusted for inflation (4.7 percent) as of December, 2013.
[Abbreviations: FY, fiscal year; QA/QC, Quality Assurance/Quality Control]

Cost category	Average cost per sample, FY 12	Average cost per sample adjusted for inflation as of December, 2013	Percent of total cost by category
Salary (2 person crew)	$2,188	$2,291	52%
Vehicle	$178	$186	4%
Travel	$38	$40	1%
Supply	$114	$119	3%
Equipment	$214	$224	5%
Laboratory Analysis[1]	$86[2]	$86[2]	2%
Total plus administrative services	**$4,189**	**$4,382**	**100%**

[1]Includes the following parameters: Nitrogen (ammonia + organic nitrogen), Nitrogen (nitrite + nitrate), Total phosphorus, Orthophosphate (DRP), Silica, and Suspended sediment concentration.
[2]$75 plus 15% for QA/QC samples. Costs current as of fiscal year 2014.

The range in costs reflects factors such as differences in distances required to travel between an office and monitoring sites as well as differences in stream size and the requisite sampling time, equipment, and sampling methods. In addition, these costs reflect the data-quality requirements of the USGS NAWQA Program. Relating to sampling techniques and costs, the USGS has found that grab samples may be non-representative for load estimates if the stream is not well mixed. The USGS NAWQA data-quality requirements specify use of equal-width increment or multiple-verticals sampling techniques to ensure the collection of representative samples. These more labor-intensive techniques result in USGS NAWQA sampling costs that are higher than programs that use grab or auto-sampler techniques.

The laboratory analysis line of Table 23 shows the total cost of analysis for the suite of nutrient-related parameters needed to answer the case-study policy question (identified in Table 3) based on laboratory costs that would be incurred by use of the USGS National Water Quality Laboratory in Denver, Colorado, using FY 2014 prices. The total cost of analyzing these parameters is $75. For the constituent group "nutrients," blank and replicate samples are the usual quality-assurance/quality-control (QA/QC) samples collected to assess data quality; for the NAWQA Program, the number of QA/QC samples collected for nutrient analysis constitutes about 15 percent of the total cost and number of nutrient samples collected and analyzed by the program (Gary Rowe, U.S. Geological Survey, written commun., 2014). Adding in a factor of 15 percent ($11) to account for the cost of quality-control samples yields a total net cost of $86 for the laboratory analytical costs and costs of quality-control samples.

The annual cost for the minimum sampling frequency of 24 samples per year as identified in Table 12, excluding the cost of a streamgage, ranges from $105,200 to $137,200 due to inflation over the course of 10 years, with an average of $120,600. As discussed in the previous section, streamflow is required for calculating load. A continuous streamgage would provide the flow data needed to support load calculations. Cost information provided by the USGS indicates that operation and maintenance of a standard streamgage is currently $14,000 per year (Greg Koltun, U.S. Geological Survey, written commun., 2014). These costs do not include initial installation and equipment costs, which can be as low as $15,000 for a simple installation to $20,000-22,000 for a walk-in shelter (Greg Koltun, U.S. Geological Survey, written commun., 2014). To estimate costs for this case study, $17,500 was used as the average cost to represent one-time startup costs for a new streamgage.

As a point of comparison, costs were provided by the Heidelberg Tributary Loading Program at Heidelberg University, which operates automated samplers three times daily with site visits weekly to collect and exchange sample bottles. Under stable flow conditions, only one of the three samples is analyzed; the other two samples are analyzed if they were collected during a flow event. Their water-quality laboratory charges $60 for analysis of their nutrient and sediment package (The National Center for Water Quality Research, 2013b), and they typically analyze 500-600 samples per site per year (Baker, 2009). According to Peter Richards (Heidelberg University, written commun., 2013), the average cost per site is $35,000 to $40,000 per year, or a high estimate of $80 per sample. This estimate does not include startup costs associated with obtaining and setting up equipment, and it presumes that there is already a working analytical laboratory. A significant part of the costs of operation is largely fixed-staff salaries, equipment, maintenance fees, and purchases. The cost per site will vary based on the number of sites maintained from year to year and the number of staff required to collect and analyze the samples associated with that number of sites.

Table 24 presents the estimated cost per monitoring site for new small watershed monitoring using the monitoring frequency options presented in Table 12, using a generic 3-percent annual inflation rate to estimate future costs. Ten years of data collection should be planned to detect a statistically significant decrease in TP, or to confirm that no statistically significant decrease occurred during the study period per the results in Table 16. Table 24 presents four costs related to the options presented in Table 12, plus costs for streamgaging. An estimated cost for the minimum sampling frequency of 24 samples per year (Table 12 option 1) is presented both with and without a new streamgage. Cost estimates are also presented for Table 12 options 2 and 4 reflecting 392 samples per 10 years, and monthly plus continuous monitoring (water temperature, specific conductance, dissolved oxygen, turbidity, and nitrate) for 10 years, respectively. Cost information provided by the USGS indicates that up-front capital costs for continuous-monitoring equipment was about $33,800 and annual maintenance $44,000 as of 2011 (Casey Lee, U.S. Geological Survey, written commun., 2014). About 24 annual up-front discrete water-quality samples are needed during the first 2 years to develop site-specific relationships between sampled and laboratory-analyzed parameters and optically-monitored indicators, and 12 annual samples would be collected thereafter to maintain and update those relationships as necessary. The average annual cost per new monitoring site, including a new streamgage, ranges from $138,000 to $206,000. For monitoring sites recommended for increased sampling frequency, an additional 4 monthly samples are needed during winter months for an additional annual average cost of $20,000 per monitoring site.

The cost for daily monitoring (Table 12 option 3) is not included in Table 24 because average USGS monitoring costs do not reflect the economies of scale that would be achieved through such a high-frequency monitoring program that uses auto-samplers that are serviced weekly. Heidelberg University costs are likely more representative of the cost to operate a high-frequency monitoring program which would cost nearly $3.8 million over 10 years for 6 monitoring sites not including streamgages, startup costs, or a working laboratory, for an average of nearly $382,000 per year. Multiple sites must be maintained to support this monitoring approach to support the independent laboratory; this cost does not apply for individual monitoring sites.

The monitoring cost estimated here does not include the cost of agricultural management practice implementation. Substantial implementation of agricultural management practices throughout the monitored watersheds is essential for water-quality data collected at new monitoring sites to detect any resulting changes in nutrient concentrations and loads. Relating change in water quality to these practices requires documentation of the agricultural management practices, their maintenance, and other potential source of nutrient change within the watershed (Figure 1). Documentation and a data management system for maintaining management practice implementation data would also result in an additional cost.

These generalized cost estimates are based on historical, national average monitoring costs for the USGS NAWQA Program and those provided by Heidelberg University. A more rigorous estimate would include multiple estimates of cost from additional monitoring programs, and could be further refined by more rigorously estimating the travel and other costs as specific monitoring sites are selected to fulfill the monitoring program recommended here.

Table 24. Estimated water-quality and streamflow-monitoring costs per small watershed monitoring site sampled for 10 years, assuming 3-percent annual inflation.

Monitoring design	A. 2014 cost per water-quality sample	B. Total samples over 10 years	C. 2014 annual streamgage-operation cost	D. Streamgage-installation cost	E. Ten-year total cost assuming 3% annual inflation[2]	F. Average annual cost per monitoring site
Streamgage only	NA[1]	NA[1]	$14,000	$17,500	$177,994	$17,799
Option 1: 24 annual samples (existing streamgage)	$4,382	240	NA[1]	NA[1]	$1,205,633	$120,563
Option 1: 24 annual samples (new streamgage)	$4,382	240	$14,000	$17,500	$1,383,628	$138,363
Option 2: Two-year intensive monitoring followed by adaptive management (new streamgage)	$4,382	392	$14,000	$17,500	$2,059,683	$205,968
Option 4: Monthly plus continuous monitoring[3] (new streamgage)	$4,382	144	$14,000	$17,500	$1,474,243	$147,424

[1]Not applicable
[2]Does not include bioindicator monitoring
[3]Water temperature, specific conductance, dissolved oxygen, turbidity, and nitrate

7.4 Cost of data analysis

Water-quality monitoring data on their own do not provide information to answer policy questions; data analysis is the critical step of converting data to information. The cost of data analysis must be factored into the cost of new monitoring because data without analysis provide minimal value. Data analysis plays a critical role in quality control and is essential for adaptive management. The USGS NAWQA Program, based on more than two decades of executing multiple data collection activities and analyzing, modeling, and reporting on those data, has found that the total cost of data collection is a reasonable starting point for budgeting the cost of data analysis (Gary Rowe, U.S. Geological Survey, written commun., 2014). New data collected through the program described here should be analyzed annually to understand ongoing trends in the data and to support efficient use of monitoring resources through adaptive management. The total dollar value obtained by doubling the estimates presented in Table 24 would cover data collection, data management, and data analysis. The cost of monitoring plus data analysis results in an average annual cost of $277,000-412,000 per monitoring site.

8. Recent Developments

A surge in collaborative planning and actions between federal, state, academic, and private organizations has occurred in response to the recent HABs in Lake Erie and other lakes in the region (Ohio Lake Erie Phosphorus Task Force, 2013). These have resulted in increased support for programs to improve, implement, and evaluate agricultural management practices for nutrient reduction. In addition, some new or re-activated nutrient and streamflow monitoring sites have been added in some of the watersheds draining to Lake Erie. Some of the data from these new sites were too recent to have been retrieved as part of this study. Several of these new and re-activated sites were considered in the spatial evaluation of tributary monitoring needs for the Lake Erie drainage basin, but the most recent planned monitoring sites are described in the addendum to this report (Betanzo et al., 2015).

8.1 Recent programs focused on agricultural management practices and tributary monitoring

The Natural Resources Conservation Service has committed to a second, more extensive CEAP study in the western basin of Lake Erie, as a result of the first Great Lakes CEAP study findings (Lund et al., 2011) and the emergence of the Lake Erie algal issues. This special study will include detailed tracking of management practice implementation data (Ohio Lake Erie Phosphorus Task Force, 2013). Also, the CEAP model will be modified to better account for DRP transport through tile drainage. Another effort working with the SWAT model is underway at Purdue University with the goal of modeling the cumulative impacts of management practices in the Maumee River Basin (Chaubey et al. 2014). Modeling studies at the University of Notre Dame are evaluating newly-identified agricultural management practices, such as the two-stage ditch and tile-drain management practices (Erickson, 2013).

Recently initiated (2011-2013) nutrient- and streamflow-monitoring programs in the Lake Erie drainage basin (Figure 24) include the GLRI tributary monitoring networks, GLRI and USEPA's Priority Watersheds edge-of-field monitoring studies, and an ODNR and USGS study that includes monitoring tributaries of the Maumee River watershed (Ohio Lake Erie Phosphorus Task Force, 2013). These programs include monthly and storm-event sampling for nutrients (to the extent possible during winter) and streamgages.

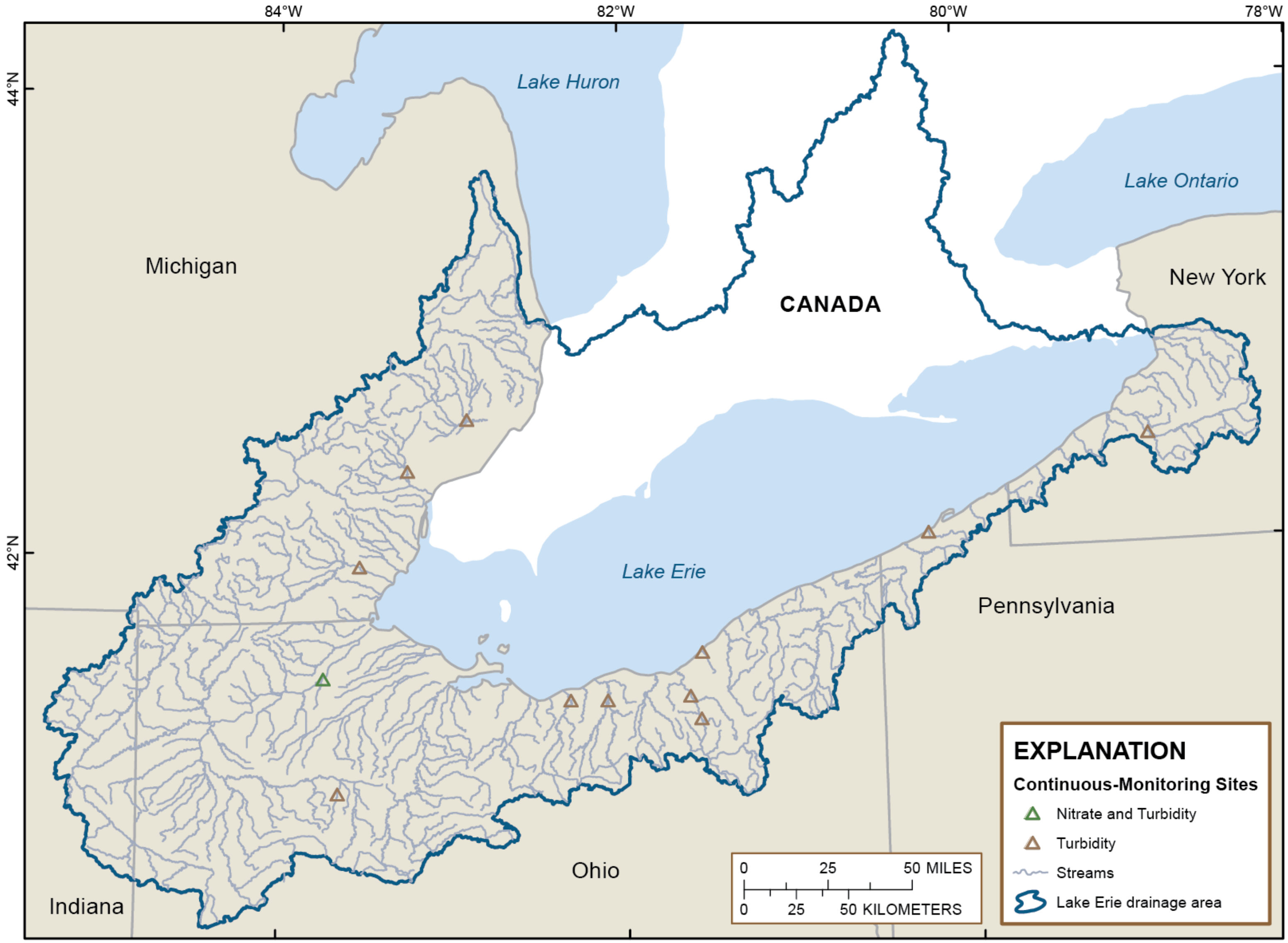
The GLRI programs in the Lake Erie drainage basin include a tributary monitoring network and edge-of-field field studies, initiated in 2011 and 2012, respectively (Figure 24). Seven GLRI tributary-network sites were identified in the nutrient data set as having active monthly monitoring (2010 or later). Data for the GLRI edge-of-field study sites in the Eagle Creek watershed (Figure 24) were not included in the nutrient data set. This study includes two edge-of-field sampling sites on drains or ditches, and one stream-sampling site with a streamgage (site G from Figure 21).

At USDA, the ARS Soil Drainage Research Unit has initiated edge-of-field research sites in the western Lake Erie drainage basin to quantify the hydrologic and water-quality impacts of various conservation and crop-management practices. Each research site includes paired facilities to implement before-after-control-impact (BACI) research studies to allow evaluation of individual practices. These sites are located in Crawford, Defiance, Henry, Paulding, Seneca, and Wood Counties in Ohio and the monitored fields drain to the Maumee, Portage, and Sandusky Rivers.

The ODNR-USGS Maumee River tributary network, initiated in October 2013, includes seven sites in watersheds ranging in size from 332 to 5,545 mi^2 (Figure 24). Four of these sites include new streamgage installations (Ottawa River near Kalida, OH; Tiffin River near Evansport, OH; Blanchard River near Dupont, OH; and Maumee River at Antwerp, OH). Although six of these seven sites had historical water-quality samples in the nutrient data set, none of these recent ODNR-USGS sites were identified as having active TP or DRP sampling data for 2010 or later. Two of the sites had 1 or more years of monthly sampling for TP or DRP: Auglaize River near Ft. Jennings, OH, had 3 years of monthly TP and DRP data between 1997 and 2007, and Maumee River at Antwerp, OH, had 3 years of monthly TP data from 1970 to 1972. Both the ODNR and GLRI programs have additional monitoring sites under development, but specific information on those sites is not yet available.

8.2 Future considerations for stream monitoring and network design

Water-quality samples taken over the range of flows throughout the year are ideal for estimating nutrient loads. Twenty-four discrete water-quality samples per year as the recommendation for minimum monitoring frequency in this report is partly a function of currently available sampling technology; for many water monitoring agencies more frequent collection of discrete samples is cost prohibitive. Most of the sampling frequency options presented in this report rely on grab sampling or the use of automated samplers that can only hold a limited number of samples that must be refrigerated onsite and transported to a laboratory on a regular basis for analysis. These approaches limit the frequency with which discrete water-quality samples can be taken, particularly during high-flow events. Sampling approaches are changing as continuous monitors become a feasible option for high-frequency monitoring, as discussed in section 5.3 and 7.3, especially for event-based monitoring. New turbidity sensors are being developed using optical backscatter techniques for continuous-record measurements that are used to estimate suspended sediment from turbidity readings (Spackman et al., 2011). Techniques associating TP concentrations with suspended sediment allow this technology to provide essentially continuous estimates for TP concentration with some limitations at high suspended sediment conditions where the correlation between turbidity measurements and suspended sediment concentrations is degraded. While analytical methods for quantifying nutrient concentrations are not practical for automated samplers in the field, surrogate measures like turbidity, which can be observed with high frequency in field installations, have potential for generating high-frequency estimates of total suspended solids and TP concentrations (Spackman et al., 2011). These continuous monitors can potentially be equipped with nitrate sensors for measuring dissolved nitrogen and specific conductance sensors for measuring dissolved solids and chloride. Figure 26 shows locations in the Lake Erie drainage basin that are using this technology.



State lines from U.S. Geological Survey, 2005, 1:2,000,000-scale digital data
Streams from U.S. Geological Survey, 2012, 1:1,000,000-scale digital data
Albers projection, NAD 1983

Figure 26. Continuous record turbidity and nitrate monitoring sites in the Lake Erie drainage basin active as of 2014.

This technology may make it possible to accurately characterize constituent concentrations and loads, generating temporally dense data sets (i.e. sensors take readings every 15 minutes or every hour). These data would greatly improve characterizations of constituent concentrations and loads over the entire hydrograph, with some caveats for the accuracy of the sensors under certain flow conditions. These strategies can overcome the costs and management challenges associated with collecting individual storm samples that improve load estimates. Costs for continuous monitoring (water temperature, specific conductance, dissolved oxygen, turbidity, and nitrate) were estimated in section 7.3, with the cost of 10 years of continuous monitoring at a single site only $100,000 more than 10 years of 24 discrete water-quality samples per year at a single site (total approximately $1.5 million from Table 24). Regular maintenance of continuous monitors is a requirement to ensure the sensors are performing as expected, resulting in a cost of continuous monitoring that is higher than the cost of the equipment and installation (Rasmussen et al., 2009; Pellerin et al., 2013; Pellerin et al., 2012), and is factored into the cost estimate presented in Table 24.

Hybrid spatial-statistical models such as SPARROW can be used to evaluate the effectiveness of current monitoring networks and have been used to identify which existing monitoring sites are critical for model accuracy. SPARROW models can be used to identify where additional monitoring sites could improve model accuracy, and similar techniques can inform the spatial design aspects of monitoring programs. SPARROW modeling could be used as a tool to support development of new monitoring sites such as those recommended in section 7.1. It should be noted that SPARROW does not currently include DRP as a modeled nutrient and does not incorporate the implementation of management practices at the watershed scale. Modifications to the SPARROW model are ongoing, and DRP modeling capabilities are being added. Additional future work may increase the level of detail in these models such that they can be used to consider management practice effectiveness at the watershed scale.

The USGS and others continue to develop and improve the methodology for calculating loads from available water-quality data. Efforts are underway to better define data requirements and the conditions under which current load-estimation methods are less effective and develop new techniques to overcome those limitations (Gary Rowe, U.S. Geological Survey, written commun., 2014).

8.3 Data sharing and data standards

Data-sharing efforts are underway to bring water-quality databases from multiple agencies together at the regional to national scale and to standardize reporting and metadata. The STORET Data Warehouse is USEPA's repository of water-quality monitoring data (U.S. Environmental Protection Agency, 2014). Data in the STORET Data Warehouse can come from states, tribes, watershed groups, federal agencies, volunteer groups, and universities. These organizations can upload data to STORET using the Water Quality Exchange (WQX) framework, which defines a standard set of data elements and internet protocols for submitting data to the USEPA. The USGS NWIS database (U.S. Geological Survey, 2002 and U.S. Geological Survey, 2014b) stores water-quality data collected by the USGS.

Use of STORET, the WQX framework, and the Water Quality Portal should continue to be supported and enhanced. There are also ongoing projects to aid in comparison of analytical methods used by different organizations; the National Environmental Methods Index (NEMI) website offers a searchable database that allows scientists and managers to find and compare analytical and field methods for all phases of environmental monitoring (National Water Quality Monitoring Council, 2014b).

There are challenges and costs for developing and maintaining integrated data systems, as well as costs for the monitoring organizations that choose to participate in these systems. Individual monitoring agencies will need to define, adopt, and maintain data-management standards and move legacy data systems to information management systems that are compatible with the WQX framework in order to participate.

The Lake Erie drainage basin presents additional challenges for data coordination since Lake Erie is an international water body. Data coordination efforts that are currently underway include the Great Lakes Observing System (GLOS), Great Lakes Aquatic Habitat Framework (GLAHF), and Great Lakes Environmental Assessment and Mapping (GLEAM).

9. Findings and Recommendations

This case study explored data collected by monitoring agencies throughout the Lake Erie drainage basin to determine whether water-quality data are available to answer the case-study policy question “How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?” Overall, this case study found that the types of water-quality data that are needed to answer the case-study policy question are being collected at the large watershed scale but are less available in smaller watersheds where the effects of management practices may be easier to detect. Although more than 300,000 TP and DRP records were collected at nearly 2,000 monitoring sites over the last 70 years in the Lake Erie drainage basin, data collected at only six monitoring sites meet the case-study criteria for monitoring parameters, streamflow, site location, and sampling frequency. For the sites that do collect the needed water data, this information is only useful for answering the question if appropriate management practices have been implemented at appropriate scales and locations throughout the monitored watershed and ancillary data on changes in the watershed over time are available to correlate water-quality change with alterations on the land.

The case-study findings relative to water data needed, water data available and usable, and approaches for filling the data gaps are summarized below.

9.1 Water data needed to answer the policy question

- **Water data needs for addressing the case-study policy question are highly dependent on study design.**

It is not possible to identify water data needs for answering the case-study policy question without an initial discussion of an appropriate study design (Figure 8). For example, the selection of appropriate monitoring sites for answering the case-study policy question is critical. Monitoring sites must be located in watersheds dominated by agricultural land use, and where management practices can, and will, be widely implemented in optimal places for reducing nutrient loss. Monitoring sites in these types of watersheds allow for the detection of water-quality changes that these practices can generate. Further, tributary water-quality and streamflow data at these monitoring sites must be available to evaluate trends in concentration and load in these watersheds over time. Finally, data on management practice implementation and other changes in land use and nutrient sources throughout the watershed must be available to correlate water-quality change with alterations on the land. Without this information, the relationship between management practices and water quality cannot be evaluated, even if management practices are delivering detectable reductions in nutrient loads.

- **Water data are needed from two watershed scales to answer the case-study policy question.**

The time and effort required to establish the necessary study conditions described in Figure 8 will be most readily achievable in small (less than 50 square miles) agricultural watersheds. At the same time, large (more than 1,000 square miles) agricultural watersheds that drain directly to Lake Erie are largely responsible for the nutrient loads that lead to HABs; complex nutrient fate and transport mechanisms in these large tributaries preclude scaling results from small watersheds. Therefore, water data from large watersheds, specifically the Maumee River, Sandusky River, and the River Raisin in the Lake Erie drainage basin, are also critical to addressing the case-study policy question.

- **The sampling frequency and duration of monitoring must meet minimum requirements to adequately characterize and detect changes in nutrient concentrations and loads to be used to answer the case-study policy question.**

Assuming the necessary management practice intensity and ancillary data (Figure 8) are available, tributary water-quality and streamflow data requirements can be characterized as shown in Table 18. A minimum of six monitoring sites in six small watersheds are needed to provide spatial representation of the Lake Erie drainage basin, and one monitoring site is needed in each of the three large watersheds that meet the identified criteria.

The most critical parameters for assessing the effectiveness of management practices in the Lake Erie drainage basin are total phosphorus (TP), dissolved reactive phosphorus (DRP), and streamflow. The sampling frequency at each monitoring site must capture the full range of hydrological conditions within the watershed annually and over time. Several sampling frequency options are presented in Table 18. The increase in sampling frequency across these options reflects improved ability to characterize the relationship between streamflow and concentration.

Monitoring duration must be sufficient to detect the effects of new changes to the landscape and distinguish them from historical land management practices, climate effects, and other factors. If appropriate agricultural management practices are implemented and consistently maintained throughout a watershed such that annual TP loads are reduced by 40 percent, a load reduction goal recommended by the International Joint Commission (2014), a monthly sampling program would be able to detect that change for both small and large watersheds with statistical significance within 10 years. However, current or moderately increased rates of management practice implementation are expected to generate reductions in TP loads that are closer to 10 percent, particularly in large watersheds, according to available models (Bosch et al., 2013; Lund et al., 2011). This case study found that more than 40 years of monthly TP data would be needed to detect a 10-percent change at a given monitoring site with statistical significance because the natural variation that occurs in streamflow and water quality from year to year obscures this small magnitude of change.

9.2 Availability and usability of existing water data to answer the policy question

This investigation found more than 300,000 TP and DRP records collected at nearly 2,000 monitoring sites over the last 70 years in the Lake Erie drainage basin (Figure 11). However, as indicated in Figure 23, only six of those monitoring sites use a sampling plan that meets the criteria summarized in Table 18 for addressing the case study's policy question. This study found the following specific results regarding currently available data:

- **The small watershed data needed for answering the case-study policy question are not currently being collected.**

The water data collected at only two existing small watershed monitoring sites meet the requirements described in Table 18. The two sites (sites E and F in Figure 23) are monitored by Heidelberg University at USGS streamgages. Water data collected at two other sites maintained by the USDA and the National Oceanic and Atmospheric Administration (NOAA) National Estuarine Research Reserve System (NERRS) could, with increased sampling frequency, provide the needed data for two additional monitoring sites. Nonetheless, a minimum of two entirely new small watershed monitoring sites also would be needed to meet the data needs for answering the case-study policy question.

- **The needed water-quality data are being collected for the three large watersheds; data collection at these sites should continue uninterrupted into the future to be useful in answering the policy question.**

Water-quality and streamflow data are being collected by Heidelberg University and the USGS for the three large agricultural watersheds that drain directly to Lake Erie, at monitoring sites on the Raisin, Maumee, and Sandusky Rivers where they discharge into the western basin of Lake Erie. These monitoring sites (A-D in Figure 23) measure the needed parameters with daily or continuous sampling frequency, and over 30 years of data records are available for these monitoring sites. Data collection must continue at these sites to measure changes in nutrient concentrations and loadings to Lake Erie over time as management practices continue to be implemented throughout the watersheds.

- **Current water data usability for answering the policy question is limited by insufficient or inconsistent data documentation and sharing.**

The water-quality records identified through this case study were generated by 17 organizations that collected nutrient-related data in the Lake Erie drainage basin. Insufficient and inconsistent documentation of available data limited the utility of these existing data sets. Substantial project time and effort over the course of this multi-year project were required to locate, obtain, and consistently format data. Data sharing and data accessibility were also limiting factors in data availability in this case study. It is possible that despite the work completed for this case study, additional relevant data may exist that were not compiled for this investigation. The Water Quality Portal (National Water Quality Monitoring Council, 2014a), a cooperative service that provides publicly available water-quality data from federal databases that include data from state, federal, tribal, and local organizations, was established to facilitate water data sharing. Yet data collected at only 26 percent of the monitoring sites identified through this case study in the Lake Erie drainage basin are available through the Water Quality Portal, and only 8 percent of the water-quality data records are available through the Portal.

Finally, for water data to be useful for addressing the case-study policy question, they must be compatible in terms of sampling plans and protocols, analysis, and interpretation. Several agencies and organizations collect small watershed data that meet or nearly meet the data needs identified in Chapter 5, including USGS, Heidelberg University, USDA, NOAA NERRS, and additional agencies described in the addendum to this report (Betanzo et al., 2015). However, these agencies all use different sampling plans that limit the ability to compare trends in concentration and load over time at these monitoring sites.

9.3 Approaches for filling data gaps to answer the policy question

This section presents the study findings regarding approaches for filling the data gaps to address the case-study policy question in the Lake Erie drainage basin.

- **Add at least two new small watershed monitoring sites in watersheds with priority characteristics.**

As noted in Table 18, six monitoring sites for each of six small watersheds are needed to address the policy question. Given data available in the Lake Erie drainage basin, two additional monitoring sites, one per each of two small watersheds, will be needed. Table 21 presents a strategy for identifying and prioritizing candidate small watersheds for additional monitoring to answer the case-study policy question. This report identifies several small watersheds with both high phosphorus yield and high vulnerability to soil loss and no current monitoring sites. At least one of the two new monitoring sites should be located in this area. The second site should be a watershed with high phosphorus yield or high vulnerability to soil loss but also provide spatial representation of the drainage basin. New monitoring sites may require new streamgages in addition to new water-quality data. New water monitoring sites and management practice incentive programs are already under development in the Lake Erie drainage basin and are described in an addendum to this report (Betanzo et al., 2015). The recommendations of this report, site selection process (Table 21), and data needed (Table 18), should be considered and incorporated as plans for new small watershed monitoring sites are finalized if the new sites are to be instrumental in answering the case-study policy question.

- **Increase sampling frequency at two existing small watershed monitoring sites.**

Increased and consistent sampling frequency of both TP and DRP at two existing monitoring sites maintained by USDA and NOAA NERRS would qualify these sites to become a part of the set of six monitoring sites needed to address the case-study policy question. Specifically, monthly year-round data collection, in addition to the current sampling frequency, would be necessary to meet the monitoring needs identified in Table 18.

- **Maintain water-quality and streamflow monitoring at the two small watershed sites monitored by Heidelberg University and the USGS.**

The remaining two small watershed monitoring sites needed to compose the set of six sites are currently in place (sites E and F, Figure 23). However, monitoring would need to continue unchanged over time at these sites as new agricultural management practices are implemented within these watersheds.

- **Maintain data collection and analysis at all small watershed monitoring sites for a minimum of 10 years during implementation of new management practices.**

Water-quality and streamflow data should be collected in the six small watersheds for at least 10 years after new practices are implemented. New monitoring and new management practice implementation should begin as soon as possible to minimize the time to detect water-quality change and produce policy-relevant information regarding the case-study policy question. The new data should be evaluated and loads calculated annually so the sampling plans can be adjusted as necessary to adapt to an evolving understanding of management practice effectiveness and water quality.

- **Maintain monitoring at large watershed monitoring sites.**

Data collection at the large watershed monitoring sites (A-D in Figure 23) should continue to capture changes in TP and DRP concentrations and loadings to Lake Erie. In addition to supporting evaluation of agricultural management practices, monitoring these large watersheds provides critical information to estimate the total nutrient loads to the western basin of Lake Erie; to measure long-term water-quality change that may result from agriculture, urban development, or climate change; and to support additional river, lake, and ecosystem research and resource management applications.

Filling the water data gaps to answer the case-study policy question:

- A minimum of two additional small watershed monitoring sites are needed. Effective monitoring sites should be identified using the process described in Table 21. New small watershed monitoring sites should collect the water data identified in Table 18.
- Increased sampling frequency at the USDA and NERRS small watershed monitoring sites to include monthly year-round data collection, in addition to the current sampling frequency, is needed for these monitoring sites to fill the need for two small watershed monitoring sites. Both TP and DRP should be sampled at the same frequency at these sites.
- Continued water-quality and streamflow monitoring are needed at the two small watershed sites monitored by Heidelberg University and the USGS who collect the needed water data.
- All small watershed monitoring sites need a minimum of 10 years of monitoring during implementation of new management practices and sharing of management practice implementation data.
- Continued long-term water-quality and streamflow monitoring are needed at the Raisin, Maumee, and Sandusky River sites monitored by Heidelberg University and USGS.

- **Improve water data coordination and sharing across monitoring agencies and organizations in the Lake Erie drainage basin.**

For water data to be used to answer the case-study policy question, monitoring agencies and organizations should coordinate sampling plans among new and existing monitoring sites so data collection, analysis, and interpretation can be compatible and comparable. Data coordination across agencies can be achieved through a coordinating entity that facilitates collaboration on sampling plans, data sharing, and data analysis in the Lake Erie drainage basin. Improved data documentation and data sharing will facilitate the use of water data for answering the case-study policy question. Tools such as the Water Quality Exchange (WQX) (U.S. Environmental Protection Agency, 2015) and the Water Quality Portal provide the infrastructure for organizations to format and share their data, but greater participation is needed. Consistent, thorough data documentation and wider availability of data sources through services such as the Water Quality Portal will increase the value of water-quality data from all monitoring agencies and reduce the amount of time needed to access and prepare data for new applications. A continued commitment to water-quality data-sharing systems is essential for maximizing use of existing water-quality data.

Improve water data usability:

- Establish a coordinating entity for ensuring compatible data collection, sharing, and analysis across the Lake Erie drainage basin.
- Adopt common data-management standards, data-entry protocols, and consistent naming and coding conventions across monitoring agencies.
- Additional monitoring agencies should submit data annually to the USEPA STORET Data Warehouse and additional partners should participate in the Water Quality Portal.

- **Maximize management practice impact in monitored watersheds.**

As noted in section 5.4, appropriate agricultural management practices must reduce TP loads in a watershed by 40 percent for a monthly sampling program to detect that change with statistical significance within 10 years. To achieve this goal, appropriate management practices should be strategically and extensively installed in areas most likely to result in nutrient reductions. Due to complexities of nutrient transport, agricultural specialists are in the best position to identify the most effective agricultural management practices for specific applications. Substantial treatment intervention will be needed to produce a 40-percent reduction in TP load. Generating this coverage in large watersheds almost certainly would require policy interventions, such as incentive programs. Smaller watersheds, though more practical to work with due to their size and the smaller number of producers, may also require incentives within specified watersheds.

- **Collect consistent, detailed data on implementation of agricultural management practices and other changes to the land and other nutrient sources within monitored watersheds.**

As noted earlier, in addition to water-quality data, consistent, detailed documentation of changes on the land and other nutrient sources within a watershed are needed to interpret water-quality data to answer the case-study policy question. Agricultural management practice implementation data are generally not available due to data sharing restrictions and lack of documentation at the level of detail needed for water-quality analysis (Jackson-Smith et al., 2010). Moreover, Section 1619 of the Farm Bill[3] restricts access to conservation practice data that have been provided to the USDA; water-quality researchers must depend on farmers' willingness to share their land management data. Protected data collection and data sharing systems for these types of ancillary data are needed to efficiently collect, store, and share these data at the level of detail needed for water-quality data analysis. Annual land management data are needed to correlate water-quality change with annual changes on the land. The detailed ancillary data needed to interpret the water-quality data are either unavailable or difficult to obtain.

9.4 Conclusion

Additional water data are needed at both small and large watershed scales to answer the case-study policy question in the Lake Erie drainage basin. The recommendations in this report present the additional water data that will allow the question to be answered in a policy-relevant time frame. Key steps to generating the needed information include strategically selecting watersheds for monitoring, maximizing management practice impact in monitored watersheds, and collecting ancillary data necessary for water-quality data analysis. However, with cooperation and coordination of producers, local conservation staff, and water monitoring agencies, data collection and analysis can answer this long-standing policy question of critical importance to the Northeast-Midwest region. The sooner the region gets started, the better.

Summary of information needs to answer "How effective are management practices at reducing nutrients from nonpoint sources at the watershed scale?"

Collect tributary water data

- Increase small watershed monitoring capacity: Additional monitoring sites, additional sampling frequency at existing sites, and continued monitoring at selected small watershed monitoring sites are needed.
- Continue to invest in large watershed monitoring: Continued long-term water-quality and streamflow monitoring are needed at monitoring sites on the Raisin, Maumee, and Sandusky Rivers where they drain to Lake Erie.
- Improve usability of new and existing water data: Establish an entity for coordinating water monitoring and management practice implementation; encourage use of data-management standards and data-entry protocols, and increase participation in data sharing programs.

Implement agricultural management practices

- Maximize management practice impact in monitored watersheds to reduce time to detect changes in water quality.

Collect ancillary data

- Collect detailed management practice implementation and other ancillary data in both large and small monitored watersheds and make available for water-quality data analysis.

[3] Section 1619 7 U.S.C. § 8791

10. References

Argue, D.M., Deacon, J., and Gilliom, R., 2014, A national compilation of water-quality monitoring data to support local, regional, and national scale water quality assessments: National Water Quality Monitoring Council, 9th National Monitoring Conference, April 28-May 2, 2014, Cincinnati, Ohio, Abstracts, p.2, *https://custom.cvent.com/FC469F3A209E4BC3BDE91EEC849E5474/files/a793bc9d7d2a44f88e6463ba165ca150.pdf*, or *http://acwi.gov/monitoring/conference/2014/*.

Arnold, J.G., Moriasi, D. N., Gassman, P.W., Abbaspour, K.C., White, M.J., Srinivasan, R., Santhi, C., Harmel, R.D., Griensven, A. van, Van Liew, M.W., Kannan, N., and Jha, M.K., 2012, SWAT: Model use, calibration, and validation: transactions of the ASABE (Amer. Soc. of Agricultural and Biological Engineers), v. 55, no. 4, pp. 1491-1508.

Baker, D.B., 2009, Quality assurance project plan for the Honey Creek targeted watershed project: Tiffin, Ohio, National Center for Water Quality Research, Heidelberg College, 238 p., *http://www.heidelberg.edu/sites/default/files/dsmith/pdf/NCWQR%20-%20HC%20TWG%20QAPP.pdf.*

Baker, D.B., and Richards, R.P., 2000, Effects of watershed scale on agrochemical concentration patterns in midwestern streams *in* chap. 3 of Agrochemical Fate and Movement: American Chemical Society, ACS Symposium Series, v. 751, p. 46-64.

Baker, N.T., Stone, W.W., Frey, J.W., and Wilson, J.T., 2007, Water and agricultural- chemical transport in a midwestern, tile-drained watershed: Implications for conservation practices, U.S. Geological Survey Fact Sheet 2007-3084, 6 p., *http://pubs.usgs.gov/fs/2007/3084/pdf/fs2007-3084_web.pdf*.

Betanzo, E., Choquette, A.F., and Hayes, L., 2015, Water data to answer urgent water policy questions: Monitoring design, available data, and filling data gaps for determining the effectiveness of agricultural management practices for reducing tributary nutrient loads to Lake Erie, Addendum describing new, expanded, and planned monitoring sites, Northeast-Midwest Institute Report, 27 p., *http://www.nemw.org*.

Bishop, P.L., Hively, W.D., Stedinger, J.R., Rafferty, M.R., Lojpersberger, J.L., and Bloomfield, J.A., 2005, Multivariate analysis of paired watershed data to evaluate agricultural best management practices effects on stream water phosphorus: Journal of Environmental Quality v. 34, p. 1087–1101.

Bosch, N.S., Allan, J.D., Dolan, D.M., Han, H., and Richards, R.P., 2011, Application of the soil and water assessment tool for six watersheds of Lake Erie: Model parameterization and calibration: Journal of Great Lakes Resources, v. 37, p. 263–271.

Bosch, N.S., Allan, J.D., Selegean, J.P., and Scavia, D., 2013, Scenario-testing of agricultural best management practices in Lake Erie watersheds: Journal of Great Lakes Research, v. 39, Issue 3, p. 429-436, *http://dx.doi.org/10.1016/j.jglr.2013.06.004*.

Burt, T.P., Howden, N.J.K., and Worrall, F., 2014, On the importance of very long-term water quality records: WIREs (Wiley Interdisciplinary Reviews) Water 2014, v. 1. p. 41–48., *http://onlinelibrary.wiley.com/doi/10.1002/wat2.1001/pdf.*

Caffrey, J., Younos, T., Connor, M., Kohlhepp, G., Robertson, D.M., Sharp, J., and Whitall, D., 2007, Nutrient requirements for the National Water Quality Monitoring Network for U.S. coastal waters and their tributaries: National Water Quality Monitoring Council, Nutrient Workgroup, 6 p., *http://www.lisser.us/council-priorities/water-quality/nutrients.pdf*.

Chaffin, J.D., Bridgeman, T.B., and Bade, D.L., 2013, Nitrogen Constrains the Growth of Late Summer Cyanobacterial Blooms in Lake Erie, Advances in Microbiology v. 3, pp. 16-26, http://dx.doi.org/10.4236/aim.2013.36A003.

Chaubey, I., Engel, B., Frankenberger, J., and Merwade, V., 2014, Cumulative impacts of BMP implementation in the Maumee River basin: Purdue University, GLRI Project Summary, *https://engineering.purdue.edu/ecohydrology/proj/EPA%20GLRI%202010-13.pdf*.

Committee on Environment and Natural Resources, 2010, Scientific Assessment of Hypoxia in U.S. Coastal Waters: Interagency Working Group on Harmful Algal Blooms, Hypoxia, and Human Health of the Joint Subcommittee on Ocean Science and Technology, 166 p., Washington, DC, *https://www.whitehouse.gov/sites/default/files/microsites/ostp/hypoxia-report.pdf*.

Corsi, S.R., Walker, J.F., Wang, L., Horwatich, J.A., and Bannerman, R.T., 2005, Effects of best-management practices in Otter Creek in the Sheboygan River priority watershed, Wisconsin, 1990-2002: U.S. Geological Survey Scientific Investigations Report 2005-5009, 34 p., *http://pubs.er.usgs.gov/publication/sir20055009.*

Corsi, S.R., Horwatich, J.A., Rutter, T.D., and Bannerman, R.T., 2013, Effects of best-management practices in Bower Creek in the East River priority watershed, Wisconsin, 1991-2009, U.S. Geological Survey Scientific Investigations Report 2012-5217, 34 p., *http://pubs.er.usgs.gov/publication/sir20125217.*

Crumrine, J.P., 2011, A BMP toolbox for reducing dissolved phosphorus runoff from cropland to Lake Erie: National Center for Water Quality Research, Heidelberg University, *https://www.heidelberg.edu/sites/default/files/jfuller/images/8%20Toolbox%2011-21-2011draft.pdf*.

Daloğlu, I., Cho, K.H., and Scavia, D., 2012, Evaluating causes of trends in long-term dissolved reactive phosphorus loads to Lake Erie: Environmental Science Technology, v. 46, p. 10660−10666.

DeBruyn, J. M., Leigh-Bell, J.A., Mckay, R.M.L., Bourbonniere, R.A., and Wilhelm, S.W., 2004, Microbial distributions and the impact of phosphorous on bacterial activity in Lake Erie: Journal of Great Lakes Research, v. 30, p. 166-183.

Dolman, A.M., Rücker, J., Pick, F.R., Fastner, J., Rohrlack, T., Mischke, U., and Wiedner, C., 2012, Cyanobacteria and cyanotoxins: The influence of nitrogen versus phosphorus, PLoS ONE (Public Library of Science) v. 7 no. 6: e38757, *http://doi:10.1371/journal.pone.0038757*.

Erickson, J., 2013, Notre Dame researcher awarded grant to study improving water quality in Lake Erie: Notre Dame, IN, University of Notre Dame, Notre Dame News, *http://news.nd.edu/news/42605-sheila-christopher-wins-155-000-research-grant-from-university-of-michigan-water-center/*.

Esri, Inc., 2012, ArcGIS for Desktop, V. 10: Redlands, California, Esri, Inc., *http://www.esri.com/*.

Falcone, J.A., Carlisle, D.M., Wolock, D.M, Meador, M.R., 2010, GAGES: A stream gage database for evaluating natural and altered flow conditions in the conterminous United States: Ecology, v. 91, no. 2, Ecological Archives E091-045, p. 621, *http://esapubs.org/Archive/ecol/E091/045/metadata.htm*.

Falcone, J.A., 2011, GAGES–II, Geospatial Attributes of Gages for Evaluating Streamflow [digital spatial dataset], *http://water.usgs.gov/GIS/metadata/usgswrd/XML/ gagesII_Sept2011.xml*.

Francesconi, W., Smith, D., Flanagan, D., Huang, C., and Wang, X., 2014, Modeling conservation practices in APEX: from the field to the watershed: Soil and Water Conservation Society, 69th International Annual Conference, July 27-30, 2014, Lombard, Illinois, Abstract.

Fry, J., Xian, G., Jin, S., Dewitz, J., Homer, C., Yang, L., Barnes, C., Herold, N., and Wickham, J., 2011, Completion of the 2006 National Land Cover Database for the conterminous United States: PE&RS (Photogrammetric Engineering and Remote Sensing), v. 77, no. 9, p. 858-864, *http://digital.ipcprintservices.com/publication/?i=78634*.

Gilbert, R.O., 1987, Statistical methods for environmental pollution monitoring: Van Nostrand Reinhold, New York, 320 p.

Gilroy, E.J., Hirsch, R.M., and Cohn, T., 1990, Mean square error of regression-based constituent transport estimates: Water Resources Research, v. 276, no. 9, p. 2069-2077.

Grady, C.A., Reimer, A.P., Frankenberger, J., and Prokopy, L.S., 2013, Locating existing best management practices within a watershed: The value of multiple methods: Journal of the American Water Resources Association, v. 49, no.4, p. 883-895.

Helsel, D.R. and Hirsch, R. M., 2002, Statistical methods in water resources techniques of water resources investigations, Book 4, chapter A3: U.S. Geological Survey, 522 p., *http://water.usgs.gov/pubs/twri/twri4a3/*.

Henry, T., 2014 "Toledo seeks return to normalcy after do not drink water advisory lifted", Toledo Blade August 5, 2014, *http://www.toledoblade.com/local/2014/08/05/Toledo-seeks-return-to-normalcy-after-do-not-drink-water-advisory-lifted.html*

Hirsch, R.M., 1988, Statistical Methods and Sampling Design for Estimating Step Trends in Surface Water Quality, Water Resources Research, v. 24, p. 493-503.

Hirsch, R. M., Moyer, D.L., and Archfield, S.A., 2010, Weighted regressions on time, discharge, and season (WRTDS), with an application to Chesapeake Bay river inputs: Journal of the American Water Resources Association, v. 46, no. 5, p. 857-880.

Hirsch, R.M., and De Cicco, L., 2015, User guide to exploration and graphics for RivEr Trends (EGRET) and dataRetrieval: R packages for hydrologic data: U.S. Geological Survey Techniques and Methods book 4, chap. A10, 94 p., *http://dx.doi.org/10.3133/tm4A10*.

International Joint Commission, 2014, A balanced diet for Lake Erie: reducing phosphorus loadings and harmful algal blooms: Report of the Lake Erie Ecosystem Priority, Washington, D.C., and Ottawa, Ont., 100 p., *http://www.ijc.org/files/publications/2014%20IJC%20LEEP%20REPORT.pdf.*

Iowa Department of Agriculture and Land Stewardship, 2013, Iowa nutrient reduction strategy, 204, p. *http://www.nutrientstrategy.iastate.edu/sites/default/files/documents/NRSfull-130529.pdf*.

Jackson-Smith, D.B., Halling, M., de la Hoz, E., McEvoy, J.P., and Horsburgh, J.S., 2010, Measuring conservation program best management practice implementation and maintenance at the watershed scale, Journal of Soil and Water Conservation, v. 65, no. 6, p. 413-423, http://doi:10.2489/jswc.65.6.413.

Jarvie, H. P., Withers, J.A., and Neal, C., 2002, Review of robust measurement of phosphorus in river water: Sampling, storage, fractionation and sensitivity: Hydrology and Earth System Sciences, v. 6, no. 1, p. 113-132.

Jarvie, H.P., Sharpley, A.N., Spears, B., Buda, A.R., May, L., and Kleinman, P.J.A., 2013a, Water quality remediation faces unprecedented challenges from “legacy phosphorus”: Environmental Science and Technology v. 47, p. 8997-8998.

Jarvie, H.P., Sharpley, A.N., Withers, P.J.A., Scott, J. T., Haggard, B. E., and Neal, C., 2013b, Phosphorus mitigation to control river eutrophication: Murky waters, inconvenient truths, and “postnormal” science: Journal of Environmental Quality, v. 42, p. 295-304.

Jaynes, D.B., and James, D.E., 2007, The extent of farm drainage in the United States, presented at the annual meeting of the Soil and Water Conservation Society, Tampa FL, July 21, 2007, *http://www.ars.usda.gov/SP2UserFiles/Place/36251500/TheExtentofFarmDrainageintheUnitedStates.pdf*

Kane, D.D., Conroy, J.D., Richards, R.P., Baker, D.B., and Culver, D.A., 2014, Re-eutrophication of Lake Erie: Correlations between tributary nutrient loads and phytoplankton biomass: Journal of Great Lakes Research, v.40, p. 496-501.

Koslow, M., Lillard, E., and Benka, V., 2013, Taken by storm: How heavy rain is worsening algal blooms in Lake Erie with a focus on the Maumee River in Ohio: National Wildlife Federation, 23 p., *http://www.nwf.org/~/media/PDFs/Water/Taken_By_Storm_NWF_2013.ashx*.

Lake Erie Improvement Association, 2012, Strategic plan for Lake Erie Partners: Sustaining healthy waters for Lake Erie’s economy: Lake Erie Improvement Association, Oregon, Ohio, 57 p. *http://www.lakeerieimprovement.org/wp-content/uploads/2012/02/leia-strategic-plan-final-12-17-2012.pdf*.

Lake Erie Lakewide Management Plan Work Group, 2011, Lake Erie binational nutrient management strategy: Protecting Lake Erie by managing phosphorus: Lake Erie LaMP Work Group Nutrient Management Task Group, 21 p., *http://www.epa.gov/lakeerie/binational_nutrient_management.pdf*.

Larsen, E., 2014, The National Water Quality Initiative’s Monitoring Framework, : National Water Quality Monitoring Council, 9th National Monitoring Conference, April 28-May 2, 2014, Cincinnati, Ohio, *http://acwi.gov/monitoring/conference/2014/2ConcurrentSessions/M5/M5Larsen.pdf*.

Lund, D., Atwood, J., Bagdon, J.K., Benson, J., Goebel, J., Ingram, K., Kellogg, R.L., Lemunyon, J., and Norfleet, L., 2011, Assessment of the effects of conservation practices on cultivated cropland in the Great Lakes region: United States Department of Agriculture, Natural Resources Conservation Service, Conservation Effects Assessment Project (CEAP), 174 p., *http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1045480.pdf.*

Maupin, M.A. and Ivahnenko, T., 2011, Nutrient loadings to streams of the continental United States from municipal and industrial effluent: Journal of the American Water Resources Association, v. 47, no. 5, p. 950-964.

Mausbach, M.J., and Dedrick, A.R., 2004, The length we go: Measuring environmental benefits of conservation practices, Journal of Soil and Water Conservation, v. 59, no. 5, p. 96A-103A.

McKay, L., Bondelid, T., Dewald, T., Rea, A., Johnston, C.M., and Moore, R.B., 2012, NHDPlus Version 2—User guide: Horizon Systems, Inc.,*ftp://ftp.horizon-systems.com/NHDPlus/NHDPlusV21/Documentation/NHDPlusV2_User_Guide.pdf*.

McMahon, G., and Cuffney, T.F., 2000, Quantifying urban intensity in drainage basins for assessing stream ecological conditions: Journal of the American Water Resources Association, v. 36, no. 6, p. 1247-1261, *http://nc.water.usgs.gov/reports/abstracts/UrbIntensity.pdf.*

Meals, D.W., Dressing, S.A., and Davenport, T.E., 2010, Lag time in water quality response to best management practices: A Review: Journal of Environmental Quality v. 39, p. 85-96.

Melillo, J.M., Richmond, T.C., and Yohe, G.W., Eds., 2014, Climate change impacts in the United States: The Third National Climate Assessment: U.S. Global Change Research Program, 841 pp., *http://nca2014.globalchange.gov*.

Michalak, A.M., Anderson, E.J., Beletsky, D., Boland, S., Bosch, N.S., Bridgeman, T.B., Chaffin, J.D., Cho, K., Confessor, R., and Daloğlu, I., 2013a, Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions: Proceedings of the National Academy of Sciences, v. 110, no. 16, p. 6448–6452, *http://www.pnas.org/cgi/doi/10.1073/pnas.1216006110*.

Michalak, A.M., Anderson, E.J., Beletsky, D., Boland, S., Bosch, N.S., Bridgeman, T.B., Chaffin, J.D., Cho, K., Confessor, R., and Daloğlu, I., 2013b, Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions, Supporting information: Proceedings of the National Academy of Sciences, *http://www.pnas.org/content/suppl/2013/03/29/1216006110.DCSupplemental/pnas.201216006SI.pdf#nameddest=STXT.*

Moyer, D.L., Hirsch, R.M., and Hyer, K.E., 2012, Comparison of two regression-based approaches for determining nutrient and sediment fluxes and trends in the Chesapeake Bay watershed: U.S. Geological Survey Scientific Investigations Report 2012-5244, 118 p., *http://pubs.usgs.gov/sir/2012/5244/*.

Myers, D.N., Thomas, M.A., Frey, J.W., Rheaume, S.J., and Button, D.T., 2000, Water quality in the Lake Erie-Lake Saint Clair drainages, Michigan, Ohio, Indiana, New York, and Pennsylvania, 1996-98: U.S. Geological Survey Circular 1203, 37 p., *http://pubs.usgs.gov/circ/circ1203/#pdf.*

National Center for Water Quality Research, 2013a, The Heidelberg Tributary Loading Program: Tiffin, Ohio, Heidelberg University, *http://www.heidelberg.edu/academiclife/distinctive/ncwqr/research/tribloading*.

National Center for Water Quality Research, 2013b, Surface water testing: Tiffin, Ohio, Heidelberg University, accessed April 11, 2014, *http://www.heidelberg.edu/academiclife/distinctive/ncwqr/water/surface*.

National Oceanic and Atmospheric Administration, 2014, Great Lakes states monthly precipitation data: Great Lakes Environmental Research Laboratory, Ann Arbor, MI, accessed 5/13/2014, *http://www.glerl.noaa.gov/data/precip/precip.html*.

National Research Council, 2011, Achieving nutrient and sediment reduction goals in the Chesapeake Bay: An evaluation of program strategies and implementation: The National Academies Press, Washington, DC, 258 p.

National Water Quality Monitoring Council, 2006, A national water quality monitoring network for U.S. coastal waters and their tributaries: The Advisory Committee on Water Information, 99 p., *http://acwi.gov/monitoring/network/design/Entire_Report_v18_060506.pdf.*

National Water Quality Monitoring Council, 2014a, Water quality portal, accessed April 18, 2014, *http://www.waterqualitydata.us/index.jsp*.

National Water Quality Monitoring Council, 2014b, National environmental methods index, accessed May 16, 2014, *https://www.nemi.gov/home/*.

Ohio Bureau of Environmental Health, 2012, Blue-green algae/cyanobacteria harmful algal bloom (HABs): Ohio Bureau of Environmental Health, *http://epa.ohio.gov/Portals/28/documents/HABs/Publications/cynobacteriahabs.pdf*.

Ohio Lake Erie Phosphorus Task Force, 2010, Lake Erie Phosphorus Task Force final report: Ohio Dept. of Agriculture, Ohio Dept. of Natural Resources, Ohio Environmental Protection Agency, and Lake Erie Commission, 97p., *http://www.epa.ohio.gov/portals/35/lakeerie/ptaskforce/Task_Force_Final_Report_April_2010.pdf.*

Ohio Lake Erie Phosphorus Task Force, 2013, Ohio Lake Erie Phosphorus Task Force II final report: Ohio Dept. of Agriculture, Ohio Dept. of Natural Resources, Ohio Environmental Protection Agency, and Lake Erie Commission, 96 p., *http://lakeerie.ohio.gov/Portals/0/Reports/Task_Force_Report_October_2013.pdf.*

Pellerin, B.A., Bergamaschi, B.A. and Horsburgh, J.S., 2012, In situ optical water-quality sensor networks—Workshop summary report: U.S. Geological Survey Open-File Report 2012–1044, 13 p., *http://pubs.usgs.gov/of/2012/1044/*.

Pellerin, B.A., Bergamaschi, B.A., Downing, B.D., Saraceno, J.F., Garrett, J.A., and Olsen, L.D., 2013, Optical techniques for the determination of nitrate in environmental waters: Guidelines for instrument selection, operation, deployment, maintenance, quality assurance, and data reporting: U.S. Geological Survey Techniques and Methods 1–D5, 37 p.

Pennuto, C.M., Dayton, L., Kane, D.D., and Bridgeman, T.B., 2014, Lake Erie nutrients: From watersheds to open water, Journal of Great Lakes Research, v. 40, pp. 469-472: http://dx.doi.org/10.1016/j.jglr.2014.07.002.

Preston, S.D., Alexander, R.B., Woodside, M.D., and Hamilton, P.A., 2009, SPARROW MODELING—Enhancing understanding of the Nation's water quality: U.S. Geological Survey Fact Sheet 2009–3019, 6 p., *http://pubs.usgs.gov/fs/2009/3019/pdf/fs_2009_3019.pdf.*

Qian, S., Bridgeman, T., Chaffin, J., Evans, M. A., Johengen, T., Klei, A. J., Kreiger, K., and Weimer, E., 2013, A Bayesian hierarchical modeling approach for comparing water quality measurements from different sources: University of Michigan Water Center, 1 p., *http://graham.umich.edu/media/files/watercenter-tier1-qian.pdf.*

Rabalais, Nancy N., 2002, Nitrogen in aquatic ecosystems: Ambio v. 31, no. 2, p.102-112.

Rantz, S.E., and others, 1982, Measurement and computation of streamflow: Volume 1. Measurement of stage and discharge, U.S. Geological Survey Water-Supply Paper 2175, *http://pubs.usgs.gov/wsp/wsp2175/pdf/WSP2175_vol1a.pdf*.

Rao, N.S., Easton, Z.M., Schneiderman, E.M., Zion, M.S., Lee, D.R., and Steenhuis, T.S., 2009, Modeling watershed-scale effectiveness of agricultural best management practices to reduce phosphorus loading: Journal of Environmental Management v. 90, p.1385–1395, *http://www.sciencedirect.com/science/article/pii/S0301479708002429*.

Rasmussen, P.P., Gray, J.R., Glysson, G.D., and Ziegler, A.C., 2009, Guidelines and procedures for computing time-series suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data: U.S. Geological Survey Techniques and Methods book 3, chap. C4, 53 p.

Reutter, J., Cibrowski, J., DePinto, J., Bade, D., Baker, D., Bridgeman, T., Culver, D., Davis, S., Dayton, E., Kane, D., Mullen, R., Pennuto, C., 2011, Lake Erie nutrient loading and harmful algal blooms: Research findings and management implications: Final report of the Lake Erie Millennium Network Synthesis Team: Ohio Sea Grant College Program, The Ohio State University, Technical Summary Number OHSU-TS-060, 17 p., *http://www.ohioseagrant.osu.edu/_documents/publications/TS/TS-060%2020June2011LakeErieNutrientLoadingAndHABSfinal.pdf*

Richards, R.P., 1998, Estimation of pollutant loads in rivers and streams: A guidance document for NPS programs: Tiffin, OH, National Water Quality Laboratory, Heidelberg University, U.S. EPA Region VIII Grant X998397-01-0, 123 p., *http://141.139.110.110/sites/default/files/jfuller/images/Load_Est1.pdf*.

Richards, R.P., 2012, Phosphorus loading to Lake Erie: an update (mostly Maumee): Ohio Lake Erie Phosphorus Task Force-Phase II, Meeting, October 3, 2012, slide presentation accessed 11/6/2014, *http://www.epa.state.oh.us/Portals/35/lakeerie/ptaskforce2/Richards.pdf*.

Richards, R. P., and Holloway, J., 1987, Monte Carlo studies of sampling strategies for estimating tributary loads, Water Resources Research, v. 23, issue 10, p. 1939–1948.

Robertson, D. M., 2003, Influence of different temporal sampling strategies on estimating total phosphorus and suspended sediment concentration and transport in small streams: Journal of the American Water Resources Association, v. 39, p. 1281–1308.

Robertson, D.M., and Saad, D.A., 2011, Nutrient inputs to the Laurentian Great lakes by source and watershed estimated using SPARROW watershed models: Journal of the American Water Resources Association, v. 47, no. 5, pp. 1011-1033.

Roseen, R.M, Ballestero, T.P., Fowler, G.D., Guo, Q., and Houle, J., 2011, Sediment Monitoring Bias by Automatic Sampler in Comparison with Large Volume Sampling for Parking Lot Runoff, Journal of Irrigation and Drainage Engineering, v. 137, issue 4, pp. 251-257.

Saad, D.A., Schwarz, G.E., Robertson, D.M., and Booth, N.L., 2011, A multi-agency nutrient dataset used to estimate loads, improve monitoring design, and calibrate regional nutrient SPARROW models: Journal of the American Water Resources Association v. 47, no. 5, p. 933-949.

Sanders, T.G., Ward, R.C., Loftis, J.C., Steele, T.D., Adrian, D.D., and Yevjevich, V., 1983, Design of networks for monitoring water quality: Water Resources Publications, Littleton, Colorado, 328 p.

Scavia, D., Allan, J.D., Arend, K.K., Bartell, S., Beletsky, D., Bosch, N.S., Brandt, S.B., Briland, R.D., Daloğlu, I., DePinto, J.V., Dolan, D.M., Evans, M.A., Farmer, T. M., Goto, D., Han, H., Hook, T.O., Knight, R., Ludisn, S.A., Mason, D., Michalak, A.M., Richards, R.P., Roberts, J.J., Rucinski, D.K., Rutherford, E., Schwab, D.J., Sesterhenn, T.M., Zhang, H., and Zhou, Y., 2014, Assessing and addressing the re-eutrophication of Lake Erie: Central basin hypoxia: Journal of Great Lakes Research, v. 40, issue 2, p. 226-246, *http://dx.doi.org/10.1016/j.jglr.2014.02.004*.

Sharpley, A.N., Kleinman, P.J.A., Hordan, P., Bergstrom, L., and Allen, A.L., 2009, Evaluating the success of phosphorus management from field to watershed: Journal of Environmental Quality, v. 38, pp. 1981-1988.

Sharpley, A., Jarvie, H., Buda, A., May, L., Spears, B., and Kleinman, P., 2013, Phosphorus legacy: Overcoming the effects of past management practices to mitigate future water quality impairment: Journal of Environmental Quality, v. 42, p. 1308-1326.

Somers, Keith M., 1997, Power analysis: A statistical tool for assessing the utility of a study: Ontario Ministry of the Environment Technical Report, 44p.

Spackman, A.J., Stevens, D.K., Horsburgh, J.S., and Mesner, N.O., 2011, Surrogate measures for providing high frequency estimates of total suspended solids and total phosphorus concentrations: Journal of the American Water Resources Association v. 47, no. 2, p. 239-253.

Speight, V., Uber, J., Grayman, W., Martel, K., Friedman, M., Singer, P., and DiGiano, F., 2009, Probabilistic modeling framework for assessing water quality sampling programs: Water Research Foundation Report, 186 p., *http://www.waterrf.org/PublicReportLibrary/3017.pdf*.

Spooner, J., Line, D., Osmond, D., Meals, D., and Dressing, S., 2012, National nonpoint source monitoring program (NNPSMP): Long-term monitoring projects to document water quality improvements from BMPs: U.S. Environmental Protection Agency, National Nonpoint Source Monitoring Workshop, Tulsa, Oklahoma.

Stumpf R.P., Wynne T.T., Baker D.B., and Fahnenstiel G.L., 2012, Interannual variability of cyanobacterial blooms in Lake Erie: PLoS ONE (Public Library of Science) v. 7, issue 8, 11 p., *http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0042444*

Stuntebeck, T.D., Komiskey, M.J., Owens, D.W., and Hall, D.W., 2008, Methods of data collection, sample processing, and data analysis for edge-of-field, streamgaging, subsurface-tile, and meteorological stations at Discovery Farms and Pioneer Farm in Wisconsin, 2001–7: U.S. Geological Survey Open-File Report 2008–1015, 51 p., *http://pubs.usgs.gov/of/2008/1015/pdf/ofr2008-1015.pdf.*

Sugg, Z., 2007, Assessing U.S. farm drainage: Can GIS lead to better estimates of subsurface drainage extent?, World Resources Institute, *http://pdf.wri.org/assessing_farm_drainage.pdf*.

Tetra Tech, 2011a, Statistical analysis for monotonic trends, Tech Notes #6: Fairfax, Virginia, Developed for U.S. Environmental Protection Agency, 23 p., *http://www.bae.ncsu.edu/programs/extension/wqg/319monitoring/TechNotes/technote6_trend_analysis.pdf*

Tetra Tech, 2011b, Minimum detectable change analysis, Tech Notes #7: Fairfax, Virginia, Developed for U.S. Environmental Protection Agency, 21 p., *http://www.bae.ncsu.edu/programs/extension/wqg/319monitoring/TechNotes/technote7_MDC.pdf*.

Texas Commission on Environmental Quality, 2008, Analysis of total phosphorus (TP) and total reactive phosphorus (TRP) in ambient surface water using an Aqualab Greenspan Continuous AutoAnalyzer TP1119, 14 p., *https://www.tceq.texas.gov/assets/public/compliance/monops/water/wqm/sops/ampm_020.pdf*.

Tomer, M.D. and Locke, M.A., 2011, The challenge of documenting water quality benefits of conservation practices: Water Science & Technology v. 64, no. 1, p. 300–310.

U.S. Department of Agriculture Natural Resources Conservation Service, 2003, National water quality handbook: Washington, D.C., U.S. Department of Agriculture, 450-VI-NWQH, *http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf.*

U.S. Department of Agriculture, 2012a, National Agricultural Statistics Service, Cropland data layer, published crop-specific data layer [Online]: U.S. Department of Agriculture, National Agricultural Statistics Service, Washington, D.C., accessed December 12, 2013, *http://nassgeodata.gmu.edu/CropScape/*.

U.S. Department of Agriculture, 2012b, Edge-of-field water quality monitoring data collection and evaluation conservation activity: Natural Resource Conservation Service, 14 p., *http://directives.sc.egov.usda.gov/OpenNonWebContent.aspx?content=33916.wba.*

U.S. Department of Agriculture, 2014a, Agricultural thesaurus: U.S. Dept. of Agriculture, National agricultural library online document, *http://agclass.nal.usda.gov/glossary.shtml.*

U.S. Department of Agriculture, 2014b, STEWARDS data base: National Water Quality Monitoring Council, 2014, Water Quality Data Portal, accessed August 18, 2014, *http://www.waterqualitydata.us/index.jsp* .

U.S. Department of Agriculture, 2014c, St. Joseph River/Upper Cedar Creek Conservation Effects Assessment Project (CEAP), study description: Agricultural Research Service, *http://amarillo.nserl.purdue.edu/ceap/index.php*.

U.S. Department of Labor, 2014, Bureau of Labor Statistics, Employer costs for employee compensation, historical listing, Table 5, State and local government workers, by major occupational group, accessed on April 20, 2014, *http://www.bls.gov/ncs/ect/sp/ececqrtn.txt*.

U.S. Environmental Protection Agency, 2002, Guidance on choosing a sampling design for environmental data collection (EPA QA/G-5S): EPA Office of Environmental Information, Report EPA/240/R-02/005, 166 p., *http://www.epa.gov/QUALITY/qs-docs/g5s-final.pdf*.

U.S. Environmental Protection Agency, 2003, Elements of a state water monitoring and assessment program (EPA 841-B-03-003): EPA Office of Wetlands, Oceans and Watersheds, 14 p., *http://www.epa.gov/owow/monitoring/elements/elements03_14_03.pdf*.

U.S. Environmental Protection Agency, 2006, Guidance on systematic planning using the data quality objectives process (EPA QA/G4): EPA Office of Environmental Information, Report EPA/240/B-06/001, 111 p. *http://www.epa.gov/QUALITY/qs-docs/g4-final.pdf*.

U.S. Environmental Protection Agency, 2010, Chesapeake Bay phase 5.3 community watershed model, Section 6, *http://www.chesapeakebay.net/about/programs/modeling/53*.

U.S. Environmental Protection Agency, 2013, Watershed modeling to assess the sensitivity of streamflow, nutrient, and sediment loads to potential climate change and urban development in 20 U.S. watersheds, National Center for Environmental Assessment, Washington, DC: EPA/600/R-12/058F, available at *http://www.epa.gov/ncea*.

U.S. Environmental Protection Agency, 2014, STORET (STOrage and RETrieval) data warehouse, accessed on April 18, 2014 at *http://www.epa.gov/storet/*.

U.S. Environmental Protection Agency, 2015, STORET/WQX: What is WQX?, accessed on January 7, 2015 at *http://www.epa.gov/storet/wqx/*.

U.S. Environmental Protection Agency and U.S. Geological Survey, 2014, National Hydrography Dataset Plus (NHDPlus) Version 2 user guide (Data model version 2.1), accessed February 24, 2014 *ftp://ftp.horizon-systems.com/nhdplus/NHDPlusV21/Documentation/NHDPlusV2_User_Guide.pdf*.

U.S. Geological Survey, 2002, NWISWeb: New site for the Nation's water data: U.S. Geological Survey Fact Sheet 128-02, 2 p., *http://pubs.usgs.gov/fs/fs-128-02/pdf/fs-128-02.pdf*.

U.S. Geological Survey, 2006, National field manual for the collection of water-quality data, 231 p., *http://water.usgs.gov/owq/FieldManual/chapter4/pdf/Chap4_v2.pdf.*

U.S. Geological Survey, 2013, Regional SPARROW model assessments of streams and rivers, accessed December 12, 2013, *http://water.usgs.gov/nawqa/sparrow/mrb/3.html*.

U.S. Geological Survey, 2014a, USGS definition of "streamgage", accessed May 16, 2014, *http://water.usgs.gov/nsip/definition9.html*.

U.S. Geological Survey, 2014b, Water data for the Nation—National Water Information System (NWIS), *http://waterdata.usgs.gov/nwis*.

U.S. Geological Survey, 2014c, National Water Information System (NWIS) water-quality web services, accessed March 13, 2014, *http://qwwebservices.usgs.gov/*.

U.S. Geological Survey, 2014d, Hydrologic unit maps, accessed April 23, 2014, *http://water.usgs.gov/GIS/huc.html*.

U.S. Geological Survey, 2014e, web page 'Forecast/nowcast Great Lakes nutrient and sediment loadings, *http://cida.usgs.gov/glri/projects/nearshore_health/forecast_loadings.html*.

U.S. Geological Survey, 2014f, U.S. Geological Survey information quality guidelines, *http://www.usgs.gov/info_qual/#guidelines*.

U.S. Geological Survey, 2014g, Fundamental science practices: Planning and conducting data collection and research: Office of Science Quality and Integrity, accessed March 3, 2014, *http://www.usgs.gov/usgs-manual/500/502-2.html*.

U.S. Geological Survey, 2014h, Fundamental science practices: Review, approval, and release of information products: Office of Science Quality and Integrity, accessed March 3, 2014, *http://www.usgs.gov/usgs-manual/500/502-4.html*.

U.S. Geological Survey, 2015, USGS definition of "water year", accessed April 1, 2015, *http://water.usgs.gov/nwc/explain_data.html*.

Ward, A., and Mecklenburg, D., [n.d.],Two-stage ditch design, accessed September 2, 2014, *http://projects.glc.org/basin//pubs/projects/wi_WtSedCoBs_pub1.pdf*.

Weller, D.E., Jordan, T.E., Sellner, K.G., Foreman, K.L., Shenk, K.E., Tango, P.J., Phillips, S.W., and Dubin, M.P., 2010, Small watershed monitoring designs: A report prepared for the Chesapeake Bay Program Scientific and Technical Advisory Committee (STAC): STAC Publication no. #10-004, Annapolis, MD, 18 p., *http://www.chesapeake.org/stac/Pubs/swmreport.pdf.*

Western Lake Erie Basin Partnership, 2009a, Western Lake Erie Basin: State of the basin: U.S. Army Corps of Engineers, Buffalo, NY, and other agencies, 84 p., *http://wleb.org/wordpress/wp-content/uploads/2011/12/WLEB-State-of-the-Basin-Final-091509.pdf*.

Western Lake Erie Basin Partnership, 2009b, Watersheds (Historical assessments of streamflow and water-quality activities by watershed, for the 8 major watersheds in the western Lake Erie basin): web-based documents, *http://wleb.org/wordpress/?page_id=24*.

White House Council on Environmental Quality, U.S. Department of Agriculture, U.S. Department of Commerce, U.S. Department of Health and Human Services, U.S. Department of Homeland Security, U.S. Department of Housing and Urban Development, U.S. Department of State, U.S. Department of the Army, U.S. Department of the Interior, U.S. Department of Transportation, U.S. Environmental Protection Agency, 2010, FY2010-FY2014 Great Lakes Restoration Initiative—Action plan: White House Council on Environmental Quality, 41 p., *http://greatlakesrestoration.us/pdfs/glri_actionplan.pdf*.

Wieczorek, M.E., and LaMotte, A.E., 2010, Attributes for NHDPlus catchments (Version 1.1) in the conterminous United States: Artificial drainage (1992) and irrigation types (1997): U.S. Geological Survey Digital Data Series DS-490-01, accessed March 4, 2014, *http://water.usgs.gov/GIS/metadata/usgswrd/XML/nhd_adrain.xml*.

Wilber, W., and Deacon, J., 2012, Establishing a Collaborative and Multipurpose National Network of Reference Watersheds and Monitoring Sites for Freshwater Streams in the United States: National Water Quality Monitoring Council Report, 2 p., *http://acwi.gov/monitoring/workgroups/wis/National_Reference_Network_for_Streams_rev2.pdf.*

Womach, J., 2005. Agriculture: A Glossary of Terms, Programs, and Laws, 2005 Edition, 279 p., http://cnie.org/NLE/CRSreports/05jun/97-905.pdf.

Zollweg, J., and Makarewicz, J.C., 2009, Detecting effects of best management practices on rain events generating nonpoint source pollution in agricultural watersheds using a physically-based stratagem: Journal of Great Lakes Research, v. 35, supplement 1, p. 37-42, *http://dx.doi.org/10.1016/j.jglr.2008.10.005*.

11. Appendix: Water Quality Sampling Requirements to Assess Change: Statistical Power Analysis

By K.H. Reckhow

List of Figures

List of Tables

11.1 Introduction

The Northeast-Midwest Institute (NEMWI), in cooperation with the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program, investigated the ability of the Northeast-Midwest (NEMW) region's water-monitoring programs to inform policy decisions through two case studies, one exploring the effectiveness of management practices for reducing nonpoint sources of nutrients in the Lake Erie drainage basin and one exploring shale gas development in the Susquehanna River Basin. Both case studies used a statistical power analysis to help quantify the water-quality data that are needed to answer each case-study policy question.

The goal of the statistical power analysis was to determine how many water-quality samples are needed to detect a "signal" (change) in the midst of the noise term, at specified statistical significance and power levels. To do this, deterministic features were modeled in the historic data; these features were expected to be trends, seasonal patterns, and/or a flow effect for each sampling station. These modeled features were then subtracted from the raw data, resulting in the residual standard error, which is the background noise term used in the power analysis. To conduct this analysis, water quality and flow data were acquired from a number of monitoring stations on rivers flowing into western Lake Erie and in the Susquehanna River Basin. The power analysis method described here was used to estimate the number of water-quality samples needed to detect a change in water quality for both the nutrient case study and the shale gas development case study.

11.2 Method

A time series of historic data for each water quality parameter and monitoring station of interest is needed for the statistical analysis. It is best if the sampling frequency is at least monthly, and several years of data are needed. These data are necessary to estimate the historic natural variability, which serves as an estimate of the natural variability that may be found in the future data to be collected for assessment of a water-quality change. In essence, an estimate of the background "noise" (natural variability) is needed to estimate how many samples (at a specified sampling frequency) are necessary to detect a "signal" (the change) in the midst of that noise. Once a suitable data set has been acquired, the following factors may need to be assessed in the analysis of the existing data:

- even spacing of observations/missing data
- censored data (i.e., data below detection limits)
- relationship between concentration and flow or stage
- seasonality
- trend
- autocorrelation
- normality

The purpose of the analysis of historic data that precedes the monitoring design (sample size) calculations is the elimination of all components of variability so that the remaining residual data series is stationary (i.e., the remaining variability is background white noise with possible autocorrelation).

Approximately 3 years (or more) of monthly (or more frequent) water quality data are needed to begin the historic data analysis. The choice of 3 years of data is based on a general approximation of how much data are needed and on expert judgment. The 3 year minimum can be considered a rough approximation and an initial starting point for monitoring. If there are fewer data, or if the sampling is less frequent than monthly, it is more likely that seasonal patterns, historic trends, and background variability will be mis-estimated. This analysis emphasizes monthly data, defined as having a frequency of 25-35 days; anything outside that frequency might be considered either too frequent (therefore possibly a replicate), or a missing value.

To begin the analysis, one comprehensive regression analysis was conducted on the monthly historic data for each monitoring site to model three deterministic features affecting the water quality parameter. If the water quality variable had an asymmetric distribution, the data were first logarithmically transformed to diminish the impact of data points having leverage and influence and to achieve approximate normality of regression residuals. All of the nutrient and streamflow data for the nutrient case study were logarithmically transformed, but most of the specific conductance and barium data for the shale case study were not transformed because these data generally appeared normally-distributed without a transformation. Once the appropriate data were logarithmically transformed, most of the power analysis method was the same for both case studies. The three deterministic features are:

1. **Relationship between concentration and flow** For rivers and streams, it is common to observe that flow and concentration are correlated, often due to either a high flow dilution effect of wastewater discharges, or high rainfall-runoff from nonpoint sources. If there is a deterministic relationship, then a concentration-flow regression model term should be part of the analysis. For the current analysis, a concentration-flow model term was used.
2. **Trend in water quality over time** With a relatively short period of record for the historic data, a trend may not be likely. Any trend should be modeled, perhaps using either a linear or polynomial regression model (with time) term, or perhaps using differencing (differencing is based on the changes between time periods). For the current analysis, a linear regression model with time was used.
3. **Seasonal pattern** Modeling can be done using either a deseasonalization technique or more structured sinusoidal regression terms. For this analysis, a structured sinusoidal regression was used per equation 1.

$$Asin\left(\frac{2\pi}{12}\right)t + Bcos\left(\frac{2\pi}{12}\right) \qquad (1)$$

where A and B are regression coefficients (to be estimated) and t is the monthly time index.

In the nutrient case study censored (e.g., below detection limits) or missing data were found to be small in number, so this issue was ignored here. These sample dates were deleted for most analyses and the impact was ignored.

One comprehensive regression analysis was applied for the three deterministic features listed above, based on a linear model assumption (log-linear when the log transform was applied to variables). The residuals that result from this regression for the water quality variable were used for the remaining analysis.

The residuals from the historic data regression analysis were determined to be white noise (i.e., no autocorrelation); therefore, the residuals provide an estimate of the background variance (independent of the sampling frequency). If this were not the case, autocorrelation must be estimated in the residuals. Then, using only positive lag autocorrelations that are significantly different from zero at α= 0.05 (or if the lag-2 autocorrelation, r^2, is significant, but lag-1 is not, then use both) estimate the background variance for monthly sampling as (Gilbert, 1987, pg. 43):

$$\sigma_\epsilon{}^2 = s^2/[1 - \left(\frac{2}{n(n-1)}\right)\sum(n-k)r_k] \qquad (2)$$

where s^2 is the residual variance, $\sigma_\epsilon{}^2$ is the autocorrelation-corrected variance, *n* is the number of samples, k is the autocorrelation lag number (Gilbert, 1987, p. 43), and r_k is the lag-k autocorrelation. In most cases, 0-2 lags will be sufficient. This slight correction to the sample background variance (the "noise" term) is necessary when autocorrelated samples are used to estimate the residual variance. For the analyses conducted for the two case studies, autocorrelation corrections were not necessary as the residuals did not appear to demonstrate much autocorrelation, so equation 2 was not applied.

Finally, sample size (number, n) was estimated using the power functions in Berryman et al. (1988) (equations 3 and 4) for the t-test where α is the significance level (α = type I error probability) and 1-β is the power (β = type II error probability):

$$(1-\beta) = F_g(N_t - Z_{(1-\alpha/2)}) \qquad (3)$$

In equation 3, F_g is the cumulative distribution function for the standard t-distribution with n-2 degrees of freedom, $Z_{(1-\alpha/2)}$ is the quantile of the standard normal distribution at probability (1-α/2), and N_t is a dimensionless trend statistic given by, for a linear trend/change:

$$N_t = \frac{|\Delta\mu|\sqrt{n}}{\sigma_\epsilon\sqrt{12}} \qquad (4)$$

In equation 4, $\Delta\mu$ is the total change in mean level over the length of the series, n is the number of observations, and σ_ϵ is the standard error of the noise term with mean zero. Solving for n yields the number of samples required.

Equation 5 can be applied for a step trend/change:

$$N_t = \frac{|\mu_1 - \mu_2|\sqrt{n}}{2\sigma_\epsilon} \quad (5)$$

where μ_1 and μ_2 in equation 5 are the means before and after the step change. Equation 5 was not applied for the current analysis.

The seasonal Kendall Tau test (Hirsch et al., 1982; Hirsch and Slack, 1984) for water quality trend detection may be used in place of equations 4 or 5; tests based on other parametric and nonparametric statistics could also be used. Lettenmaier (1976) has evaluated the adequacy of equations 4 and 5 for design and analysis using the nonparametric Mann-Whitney and Spearman tests. Adequate information on the power function for the seasonal Kendall test is not available, but Berryman et al. (1988) note that equations 4 or 5 provide a reasonable approximation for the seasonal Kendall test (sample size may be slightly underestimated). Thus the sample size can be estimated using the background variance estimate and the power expressions in either equation 4 or 5. This yields the number of independent samples, or the "effective sample size" (n_e) of a larger number (n) of correlated samples. To calculate the actual number of samples when samples are autocorrelated, the equation below can be used (Yevjevich, 1972, p. 188):

$$n \approx n_e(1 + 2\sum r_k) \quad (6)$$

where the approximation is good when n>k. In this case, only positive autocorrelations that are significant at the 0.05 level were used. Since monthly samples were used, it was assumed that autocorrelation was not a factor and equation (3) was applied in the analysis. Testing of the sample residuals confirmed that autocorrelation was not a significant factor.

A two-tailed test was applied at this point, with type I and type II errors set equal to each other for simplicity. Contours for 0.10, 0.20, and 0.30 error levels were plotted on a trend magnitude versus sample number plot, with the magnitude of change expressed as percent change for log-transformed data or absolute concentration values for untransformed data.

No suspected outliers were removed in either case study. Robust estimators for deterministic patterns and trends, for autocorrelation and for background variance, could be applied in other applications of this analysis approach.

Monthly samples were required for the power analysis; more-frequent-than-monthly samples were removed from high sampling-frequency data sets to ensure that there would be no autocorrelation in the residuals. Autocorrelation would violate the assumptions used to conduct the power analysis. The sample that fell closest to the 15th of the month was selected, and, if there were multiple samples on that day, the sample that was closest to noon was selected to generate the monthly samples data set.

11.3 An Example: Phosphorus in the Maumee River

Heidelberg University has collected nearly 40 years of daily phosphorus water-quality data on the Maumee River. Due to expected autocorrelation in water-quality data collected more frequently than monthly, a monthly subsample of the Maumee data was used, collected on the 15th of each month. The method described in this Appendix was performed for both total phosphorus (TP) and dissolved reactive phosphorus (DRP).

Table 1 contains the results of a regression analysis for the monthly TP data for the Maumee River. Note that the regression parameters for each of the deterministic terms are significant at the 0.05 level. The residual standard error is 0.161385; this is the background noise term that is used in the power analysis. Figure 1 shows the relationship between sample size and linear (continuous) change in TP concentration in the Maumee River, as a function of the Type I and Type II error levels (significance and power) set equal to each other for simplicity of presentation. In this analysis, the samples were assumed to be independent; if autocorrelation is expected, the required sample size will be larger. It is clear from this analysis that a large number of samples is necessary to detect the signal (change in TP) from the noise. Note that the "fractional decrease" in TP concentration has a different scale on the x-axis than does the "fractional increase;" this occurs due to the asymmetry of the TP residuals in the concentration metric. (The concentration metric is the original metric that has not been log transformed.) The remaining analyses for the nutrient case study addressed only fractional decrease, since decrease is expected with management practice implementation.

A similar analysis was undertaken for dissolved reactive phosphorus (DRP) for the Maumee River; Figure 2 shows the results of the power analysis for DRP in the Maumee. In comparing Figure 1 and Figure 2, note that substantially more samples are required to detect a change in DRP than required for detecting the same change in TP. The number of samples required to detect a change in DRP is greater because of the higher variability detected in background noise for log-transformed DRP, in comparison to log-transformed TP. Figure 3 shows that, under the log-transformation, DRP regression residuals are far more variable than are TP residuals.

In this study censored (e.g., below detection limits) or missing data were deleted for most analyses and the impact was ignored. However, Heidelberg University maintains zeros and small negative numbers in their data, and they consider zeros to represent what was measured. They do not censor their data at the detection limit (Peter Richards, Heidelberg University, written commun., 2014). To test what effect these values have on the regression analysis results, one regression analysis was completed for the nutrient case study as a sensitivity test, where 0.001 mg/L was substituted for 44 censored dissolved reactive phosphorus samples in the Sandusky River. The results of this sensitivity analysis indicated that including these censored values increased the background variability and consequently increased the residual standard error for this monitoring site. The number of years of sampling required to detect a 10-percent change in median value is on the order of hundreds of years, and tens of years for detecting a 40-percent change for both analyses. Including the censored values for other monitoring sites would not significantly affect the conclusions regarding the amount of DRP data needed to test the case-study hypotheses.

Table 1. Regression analysis for deterministic features: Maumee Total Phosphorus data.

ANOVA

	Df	*SS*	*MS*	*F*	*Significance F*
Regression	4	9.098658	2.274665	87.3355	1.14E-52
Residual	383	9.975286	0.026045		
Total	387	19.07394			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	-1.59609	0.056336	-28.3319	5.14E-96	-1.70686	-1.48533	-1.70686	-1.48533
time	-0.00042	7.33E-05	-5.66565	2.88E-08	-0.00056	-0.00027	-0.00056	-0.00027
sine(m)	-0.08682	0.013875	-6.25736	1.05E-09	-0.1141	-0.05954	-0.1141	-0.05954
cos(m)	-0.03375	0.011635	-2.90054	0.00394	-0.05662	-0.01087	-0.05662	-0.01087
logFlow	0.290123	0.016562	17.51703	7.02E-51	0.257558	0.322687	0.257558	0.322687

Regression Statistics	
Multiple R	0.690667
R Square	0.47702
Adjusted R Square	0.471558
Standard Error	0.161385
Observations	388

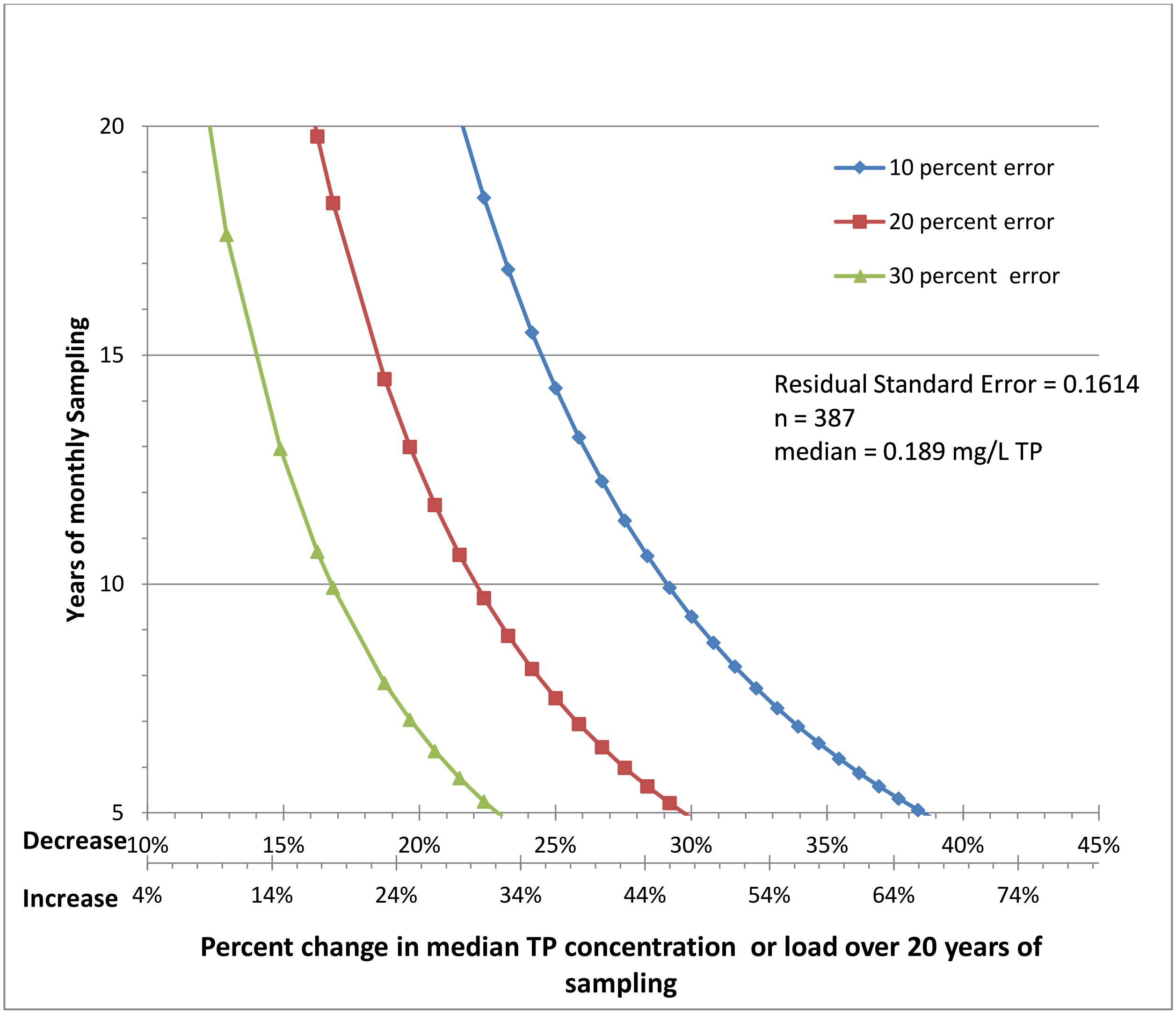


Figure 1. Power analysis estimates of the number of years of monthly sampling needed to detect trends in median total phosphorus (TP) concentration or load, for different error levels, over 20 years of monitoring at Maumee River at Waterville, OH. The different scales for the percent decrease and increase on the horizontal axis occurs because the log transform introduces an asymmetric distribution in the original concentration metric.
[**Abbreviations:** mg/L, milligrams per liter]

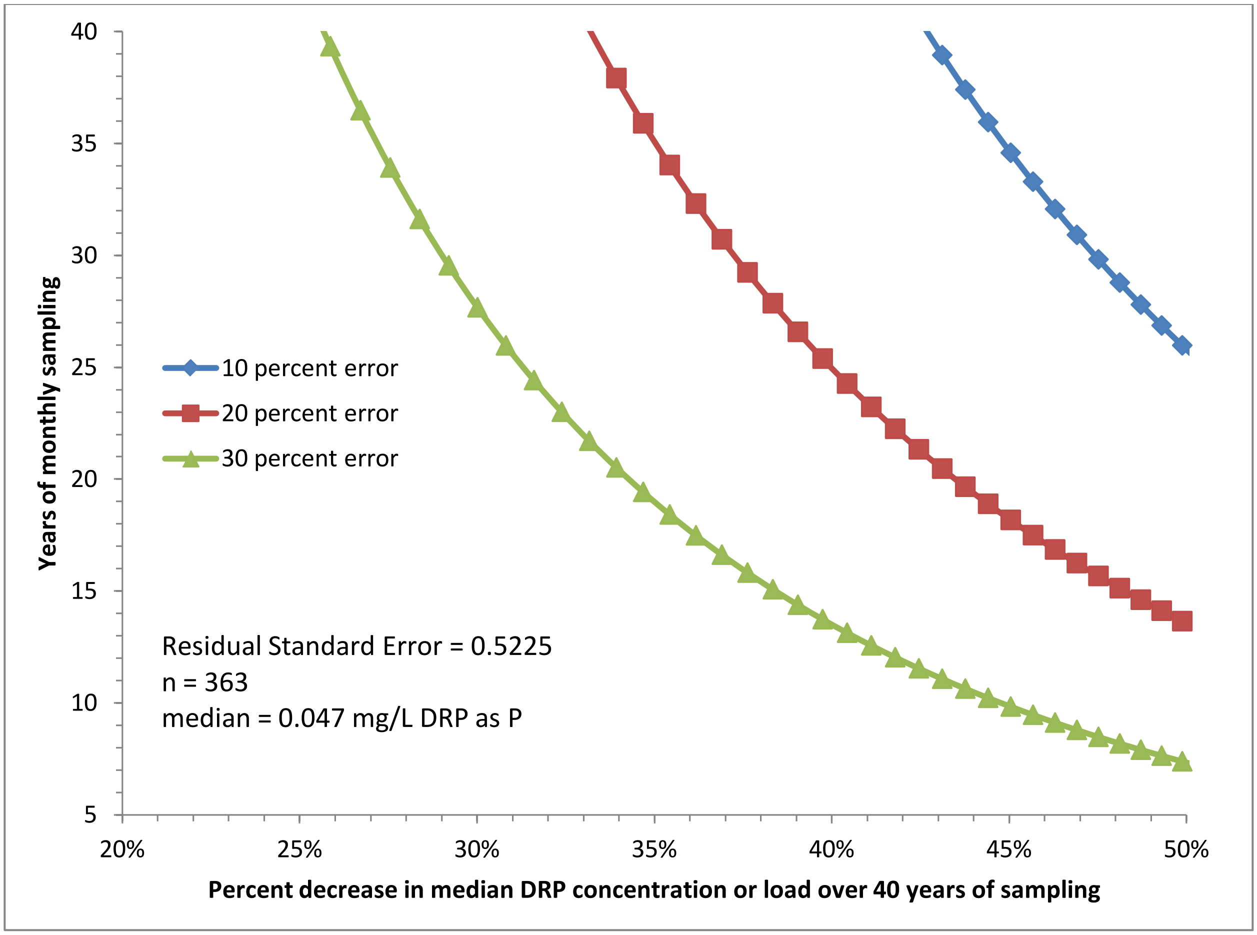


Figure 2. Power analysis estimates of the number of years of monthly sampling needed to detect trends in median dissolved reactive phosphorus (DRP) concentration or load, for different error levels, over 40 years of monitoring at Maumee River at Waterville, OH.

[**Abbreviations** mg/L, milligrams per liter]

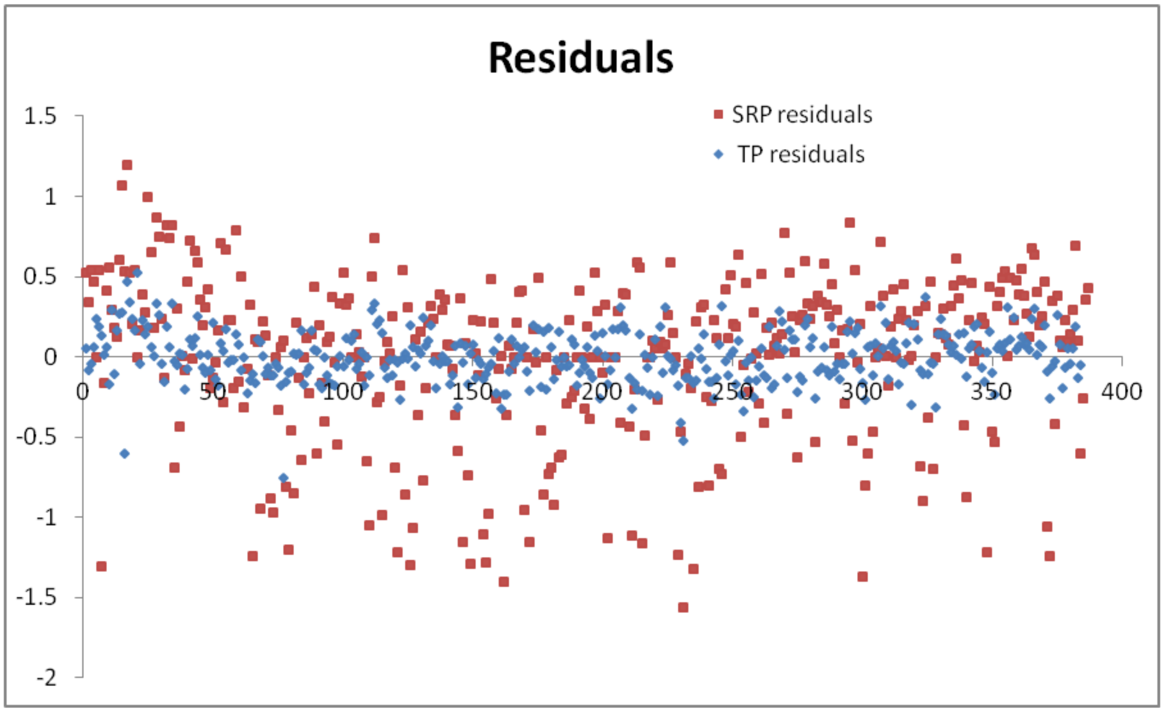


Figure 3. Comparison of residuals from total phosphorus (TP) and dissolved reactive phosphorus (DRP or SRP) regression models.

Hirsch et al. (2010) stated that pollutant flux may be a better measure than pollutant concentration when looking for changes in nonpoint source pollutant loads; this conclusion is relevant when the daily concentration and flux estimates are not both based on a log-linear model for concentration and flow. Since the log-linear flow-concentration model was used here, concentration and flux changes were indistinguishable from each other.

The sample numbers versus change graphs all reflect a linear change. This is reasonable for assessing the impact of management practice implementation, as it may be expected that the water quality effects of management practice implementation will be gradual. However, if water-quality monitoring to detect change is discontinued at the start of management practice implementation and is not re-established until sufficient time has elapsed to allow for a water-quality change to be observed, the change could be considered a step function from the perspective of the monitoring results. To illustrate, assume that management practice implementation is completed in 2014, and downstream water-quality monitoring is temporarily terminated at that time. Further, assume that 6 years are required (based on model results and a confidence interval assessment) for the water-quality impact to be measured at a downstream monitoring station. We can then consider the pre-2014 monitoring data to reflect “before” conditions and the post-2020 monitoring data to reflect “after” conditions; this is effectively a step-change assessment. This approach would require only 1/3 the number of independent samples (taken after 2020) as needed for the linear change assessment. In this case, equation 5 would be applied to determine the number of samples required to detect change at a given power and significance.

11.4 Results

Table 2, Table 3 and Table 4 summarize the results of the power analyses completed for the Nutrient Case Study. Table 5, Table 6, and Table 7 summarize the results of the power analyses completed for the Shale Gas Development Case Study.

Table 2. Summary of total phosphorus and dissolved reactive phosphorus regression analysis results.
[**Abbreviations**: TP, total phosphorus; DRP, dissolved reactive phosphorus; sq.miles, square miles; mg/L, milligrams per liter; P, phosphorus]

Monitoring site	Station Name	TP or DRP	Basin Area (sq. miles)	Residual Standard Error[1]	Number of samples used in regression analysis[2]	Median Concentration (mg/L as P)
ohUSGS:04193500	Maumee River at Waterville, OH	TP	6013	0.1614	385	0.189
03231500	Scioto River at Chillicothe, OH	TP	3849	0.1607	137	0.279
ohUSGS:04198000	Sandusky River near Fremont, OH	TP	1252	0.2079	398	0.122
inLES060-0005 Flow site 04182000	Saint Mary's River St. Mary's River near Fort Wayne, IN	TP	716	0.1303	50	0.265
04178000	St. Joseph River near Newville, IN	TP	618	0.1908	66	0.130
ohUSGS:04189000	Blanchard River near Findlay, OH	TP	351	0.2344	62	0.215
04186500	Auglaize River near Fort Jennings, OH	TP	331	0.2278	120	0.174
ohUSGS:04199500	Vermilion R Near Vermillion, OH	TP	260	0.3431	118	0.069
ohUSGS:04197100	Honey Creek at Melmore, OH	TP	150	0.2483	409	0.121
ohUSGS:04197170	Rock Creek at Tiffin, OH	TP	34.6	0.3098	339	0.074
402913084285400	Chickasaw Creek at St. Marys, OH	TP	16.4	0.273	45	0.248
ohUSGS:04185440	Unnamed Tributary to Lost Creek near Farmer, OH	TP	4.23	0.2325	50	0.078
ohUSGS:04198000	Sandusky River near Fremont, OH	DRP[3]	1252	0.6137	398	0.030
ohUSGS:04193500	Maumee River at Waterville, OH	DRP	6336	0.5225	363	0.047
ohUSGS:04198000	Sandusky River near Fremont, OH	DRP	1252	0.5124	354	0.035
04178000	St. Joseph River near Newville, IN	DRP	618	0.2855	69	0.037
ohUSGS:4199500	Vermilion R Near Vermillion, OH	DRP	260	0.4063	72	0.019
ohBR	Old Woman Creek @ Berlin Road	DRP	22.1	0.2894	91	0.017
ohUSGS:04197170	Rock Creek at Tiffin, OH	DRP	34.6	0.4931	307	0.020
ohUSGS:04185440	Unnamed Tributary to Lost Creek near Farmer, OH	DRP	4.23	0.3744	51	0.030

[1] Log base 10-transformed concentration units
[2]The number of observations in the raw data minus observations deleted from regression due to "missingness" (missing data).
[3]Sensitivity analysis

Table 3. Estimated years of monthly sampling required to detect a decrease in median concentration of total phosphorus (TP) of 10 percent and 40 percent for analyzed watersheds at the 20 percent error level.
[**Abbreviations**: mg/L, milligrams per liter; P, phosphorus; %, percent]

Station name	**Residual standard error** (log-transformed concentration units)	**Median concentration (mg/L as P)**	**Years of monthly TP sampling required to detect 10% change from median value at the 20% error level**	**Years of monthly TP sampling required to detect 40% change from median value at the 20% error level**
Watersheds larger than 1,000 square miles				
Maumee River at Waterville, OH	0.1614	0.189	55	2
Scioto River at Chillicothe OH[1]	0.1607	0.279	55	2
Sandusky River near Fremont, OH	0.2079	0.122	92	4
Watersheds between 50 and 1,000 square miles				
Saint Mary's River near Ft. Wayne, IN	0.1303	0.265	36	2
St. Joseph River near Newville, IN	0.1908	0.130	77	3
Blanchard River near Findlay, OH	0.2344	0.215	117	5
Auglaize River near Fort Jennings, OH	0.2278	0.174	110	5
Vermilion River Near Vermillion, OH	0.3431	0.069	250	11
Honey Creek at Melmore, OH	0.2483	0.121	131	6
Watersheds less than 50 square miles				
Rock Creek at Tiffin, OH	0.3098	0.074	204	9
Chickasaw Creek at St. Mary's,OH[1]	0.273	0.248	159	7
Unnamed Tributary to Lost Creek near Farmer, OH	0.2325	0.078	115	5

[1]Not in the Lake Erie Basin

Table 4. Estimated years of monthly sampling required to detect a decrease in concentrations of dissolved reactive phosphorus (DRP) of 10 percent and 40 percent for analyzed watersheds at the 20 percent error level.
[**Abbreviations**: mg/L, milligrams per liter; P, phosphorus; %, percent]

Station name	Residual standard error (log-transformed concentration units)	Median concentration (mg/L as P)	Years of monthly DRP sampling required to detect 10% change from median value at the 20% error level	Years of monthly DRP sampling required to detect 40% change from median value at the 20% error level
Watersheds larger than 600 square miles				
Maumee River at Waterville, OH	0.5225	0.047	581	25
Sandusky River near Fremont, OH	0.5124	0.035	559	24
St. Joseph River near Newville, IN	0.2855	0.037	173	8
Watersheds between 50 and 600 square miles				
Vermilion River Near Vermillion, OH	0.4063	0.019	351	15
Watersheds less than 50 square miles				
Old Woman Creek at Berlin near Huron, OH	0.2894	0.017	178	8
Rock Creek at Tiffin, OH	0.4931	0.020	517	23
Unnamed Tributary to Lost Creek near Farmer, OH	0.3744	0.030	298	13

Table 5. Summary of barium and specific conductance regression analysis results.
[Abbreviations: mi2, square miles; µg/l, micrograms per liter; µS/cm, microSiemens per centimeter]

Monitoring site	Station Name	Parameter	Basin Area (mi^2)	Residual Standard Error[1]	Number of samples used in regression analysis[2]	Median Concentration (Barium: µg/l; Specific Conductance: µS/cm)
1538709	West Branch Fishing Creek near Elk Grove, PA	Barium	20.2	0.7504	67	11.6
1545600	Young Womans Creek near Renovo, PA	Barium (diss.)	46.2	4.09	52	29
1548476	Cedar Run above Mine Hole Run near Cedar Run, PA	Barium	26.3	1.508	62	22.4
1533610	Unnamed Tributary to Tunkhannock Creek at Gelatt, PA	Barium	9.02	3.278	33	21.1
1557990	Sinking Run near Spruce Creek, PA	Barium	28.3	5.566	67	26.7
155979602	Bobs Creek below Wallacks Branch at Pavia, PA	Barium	22.1	3.608	74	45.6
1569195	Conodoguinet Creek above Reservoir near Roxbury, PA	Barium	27.2	2.998	68	28
1571820	Swatara Creek at Ravine, PA	Barium (diss.)	43.3	2.337	46	20
1571820	Swatara Creek at Ravine, PA	Barium	43.3	74.76	46	23
1538709	West Branch Fishing Creek near Elk Grove, PA	Spec. Cond.	20.2	2.919	64	27
1545600	Young Womans Creek near Renovo, PA	Spec. Cond.	46.2	4.601	252	39
1548423	Wilson Creek at Morris, PA	Spec. Cond.	22.8	87.29	44	412
1548476	Cedar Run above Mine Hole Run near Cedar Run, PA	Spec. Cond.	26.3	0.08681[3]	59	46
1508800	Factory Brook at Homer, NY	Spec. Cond.	15.8	44.03	59	321
1509150	Gridley Creek above East Virgil, NY	Spec. Cond.	10.4	24.91	71	195
1528000	Five mile Creek near Kanona, NY	Spec. Cond.	66.8	31.32	113	258
BNTY000.9-4276	Bentley Creek at Wellsburg, NY	Spec. Cond.	54.2	27.19	49	193
1557990	Sinking Run near Spruce Creek, PA	Spec. Cond.	28.3	32.77	64	186
155979602	Bobs Creek below Wallacks Branch at Pavia, PA	Spec. Cond.	22.1	9.464	72	76
1569195	Conodoguinet Creek above Reservoir near Roxbury, PA	Spec. Cond.	27.2	13.23	67	65
1571820	Swatara Creek at Ravine, PA	Spec. Cond.	43.3	27.93	101	203
CCPASEC_1216	Lick Run at Howard, PA	Spec. Cond.	11.4	98.12	37	438

[1] Barium: µg/l; Specific Conductance: µS/cm
[2] The number of observations in the raw data minus observations deleted from regression due to missingness
[3] Log transformed; Log base 10-transformed concentration units

Table 6. Estimated number of samples needed to detect a twenty percent reduction from median values for barium by ecoregion for watersheds of less than 71 square miles.

[**Abbreviations**: mi^2, square miles; µg/L, micrograms per liter; %, percent; ecoregion, U.S. Environmental Protection Agency Level III Ecoregion]

Station number	Station name	Drainage area (mi^2)	Parameter fraction	Minimum concentration (µg/L)[1]	Median concentration (µg/L)[1]	Maximum concentration (µg/L)[1]	Number of barium samples required to detect a 20% change from median values	
							20 % error	10 % error
				Northern Allegheny Plateau ecoregion				
01533610	Unnamed Tributary to Tunkhannock Creek at Gelatt, PA	9.02	Total	16.6	21.1	37.9	33	62
				North Central Appalachians ecoregion—total barium				
01538709	West Branch Fishing Creek near Elk Grove, PA	20.2	Total	9.5	11.6	14.6	6	11
01548476	Cedar Run above Mine Hole Run near Cedar Run, PA	26.3	Total	17.2	22.4	31.8	6	12
				North Central Appalachians ecoregion				
01545600	Young Womans Creek near Renovo, PA	46.2	Dissolved	20.0	29.0	37.0	27	51
				Ridge and Valley ecoregion				
01557990	Sinking Run near Spruce Creek, PA	28.3	Total	17.9	27.0	55.6	59	112
0155979602	Bobs Creek below Wallacks Branch at Pavia, PA	22.1	Total	34.5	45.6	59.4	8	16
01569195	Conodoguinet Creek above Reservoir near Roxbury, PA	27.2	Total	22.8	28.0	40.6	15	29
01571820	Swatara Creek at Ravine, PA	43.3	Total	18.0	23.0	520	14,000[2]	27,000[2]
01571820	Swatara Creek at Ravine, PA	43.3	Dissolved	16.0	20.0	29.0	18	35

[1]Minimum, maximum and median values calculated based on the monthly data set.

[2]Monthly dataset includes one potential storm sample resulting in increased background variability.

Table 7. Estimated number of samples needed to detect a twenty percent reduction from median values for specific conductance levels by ecoregion for watersheds of less than 71 square miles.

[Abbreviations: Map ID, site number shown in shale case study figure 20; mi2, square miles; µS/cm, microSiemens per centimeter; %, percent; ecoregion, U.S. Environmental Protection Agency Level III Ecoregion]

Station number	Station name	Drainage area (mi^2)	Minimum specific conductance (µS/cm)	Median specific conductance (µS/cm)	Maximum specific conductance (µS/cm)	Number of specific conductance samples required to detect a 20% change from median values	
						20% error	10% error
	Northern Allegheny Plateau ecoregion						
01508800	Factory Brook at Homer, NY	15.8	96	321	420	25	48
01509150	Gridley Creek above East Virgil, NY	10.4	75	195	335	22	42
01528000	Five mile Creek near Kanona, NY	66.8	78	258	450	20	38
BNTY000.9-4276	Bentley Creek at Wellsburg, NY	54.2	114	193	320	27	51
	North Central Appalachians ecoregion						
01538709	West Branch Fishing Creek near Elk Grove, PA	20.2	21	27	36	16	30
01545600	Young Womans Creek near Renovo, PA	46.2	25	39	80	19	36
01548423	Wilson Creek at Morris, PA	22.8	130	412	975	61	115
01548476	Cedar Run above Mine Hole Run near Cedar Run, PA	26.3	21	46	176	45	86
	Ridge and Valley ecoregion						
01557990	Sinking Run near Spruce Creek, PA	28.3	103	186	266	42	80
0155979602	Bobs Creek below Wallacks Branch at Pavia, PA	22.1	63	76	115	21	40
01569195	Conodoguinet Creek above Reservoir near Roxbury, PA	27.2	37	65	135	56	106

Station number	Station name	Drainage area (mi^2)	Minimum specific conductance (µS/cm)	Median specific conductance (µS/cm)	Maximum specific conductance (µS/cm)	Number of specific conductance samples required to detect a 20% change from median values	
						20% error	10% error
01571820	Swatara Creek at Ravine, PA	43.3	117	203	346	26	50
CCPASEC_1216	Lick Run at Howard, PA	11.4	139	438	635	68	129

[1]Minimum, maximum, and median values are calculated based on the monthly data set.

11.5 References

Berryman, D., Bobee, B., Cluis, D., and Haemmerli, J., 1988, Nonparametric tests for trend detection in water quality time series, Water Resources Bulletin, vol. 24, issue 3, p. 545-56.

Gilbert, R.O., 1987, Statistical Methods for Environmental Pollution Monitoring, Van Nostrand Reinhold, New York, pp. 336.

Hirsch, R.M., Slack, J.R., and Smith, R.A., 1982, Techniques of trend analysis for monthly water quality data, Water Resources Research, vol.18, no. 1, p. 107-21.

Hirsch, R.M., and Slack, J. R., 1984, A nonparametric trend test for seasonal data with serial dependence, Water Resources Research, vol. 20, issue 6, p. 727-32.

Hirsch, R.M., Moyer, D.L., and Archfield, S.A., 2010, Weighted Regressions on Time, Discharge, and Season (WRTDS), with an Application to Chesapeake Bay River Inputs: Journal of the American Water Resources Association, vol. 46, no. 5, p. 857-880.

Lettenmaier, D.P., 1976, Detection of trends in water quality data from records with dependent observations, Water Resources Research, vol. 12, issue 5, p. 1037-46.

Yevjevich, V., 1972, Probability and Statistics in Hydrology. Water Resources Publications, LLC, Fort Collins, Colorado, pp. 312.