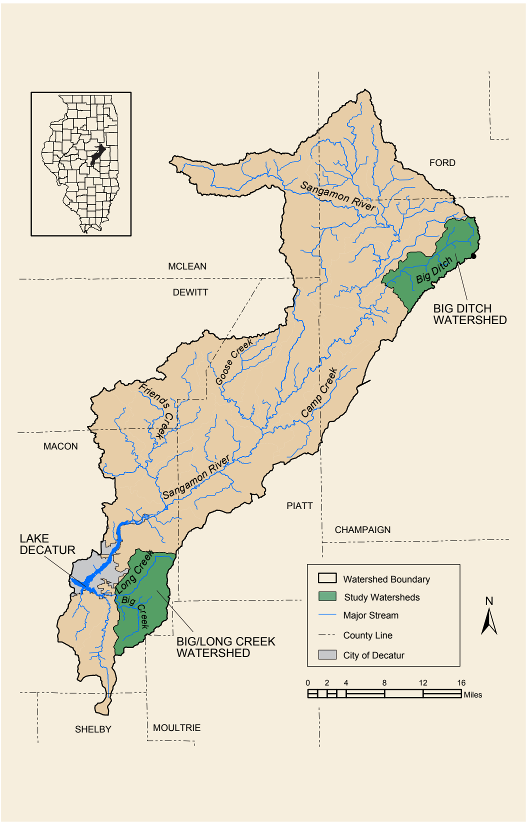
ILLINOIS STATE WATER SURVEY
PRAIRIE RESEARCH INSTITUTE

Contract Report 2014-02

Decision Support Model for Generating Optimal Alternative Scenarios of Watershed Best Management Practices

Elias G. Bekele, Laura Keefer, and Sangeetha Chandrasekaran



ILLINOIS

Decision Support Model for Generating Optimal Alternative Scenarios of Watershed Best Management Practices

by

Elias G. Bekele, Laura Keefer, and Sangeetha Chandrasekaran

Illinois State Water Survey
Prairie Research Institute
University of Illinois at Urbana-Champaign

Prepared for
Illinois Environmental Protection Agency

May 2014

Table of Contents

Page

1. Introduction and Background 1
2. Watershed Modeling 4
  2.1 Study Watersheds 4
  2.2 Soil and Water Assessment Tool (SWAT) 5
    Sediment Yield and Routing 5
    Nutrient Transformation and Movement 6
    Plant Growth Simulation 7
    Field Management Operations 7
  2.3 Spatial Sensitivity Analysis 8
    Sensitivity of Watershed Responses to Spatial Scale 11
      Streamflows and Sediment Loads 11
      Nutrient Loads 12
    Recommended Subbasin Delineations 14
  2.4 Big Ditch and Big/Long Creek Watershed Models 15
    Model Calibration and Validation 15
      Incorporating Field Management Operations 15
      Model Performance Metrics 17
      Calibration and Validation Results 18
3. Selected Best Management Practices and Suitability Criteria 24
  3.1 Nutrient Management 24
  3.2 Cover Crops 25
  3.3 Perennial Crops 25
  3.4 Constructed Wetlands 26
  3.5 Drainage Water Management 27
  3.6 Bioreactors 29
  3.7 Saturated Buffers 29
  3.8 Filter Strips 30
  3.9. Suitability of HRUs for BMP Implementation 30

Table of Contents (cont'd)

Page

4. Decision Support Models 33
4.1 Multi-Objective Optimization 33
4.2 AMGA2: Archived-Based Micro-Genetic Algorithm 34
4.3 Coupled AMGA2-SWAT: Decision Support Model 35
4.4 Application of the Decision Support Model 39
Nutrient Management Scenarios 42
Cover Crop Scenarios 44
Perennial Crop Scenarios 46
Constructed Wetland Scenarios 49
Drainage Water Management Scenarios 52
Bioreactor Scenarios 54
Saturated Buffer Scenarios 56
Filter Strip Scenarios 58
4.5 Optimal BMP Implementation Scenarios 60
4.6 Impact of Perennial and Cover Crops on Water Yield 62
5. Summary, Conclusions, Limitations, and Recommendations 65
5.1 Summary 65
5.2 Conclusions 67
5.3 Limitations of the Study 68
5.4 Recommendations for Future Work 68
References 70
Appendix A. Suitable HRUs in Big Ditch and Big/Long Creek Watersheds for Implementation of Selected BMPs. 74
Appendix B. Optimal Placements of BMPs for Cost-Effective Reduction of NPS Pollutants in Big Ditch Watershed 85
Appendix C. Optimal Placements of BMPs for Cost-Effective Reduction of NPS Pollutants in Big/Long Creek Watershed 95

List of Tables

Page

Table 2.1 Watershed subdivision levels..9
Table 2.2 Appropriate watershed subdivisions for streamflow, sediment, and nutrient simulations 14
Table 2.3 Data availability for model calibration and validation 15
Table 2.4 Typical field management operations...17
Table 2.5 Calibrated parameters for flow, sediment, nitrate, and total phosphorus 19
Table 2.6 Performance statistics for Big Ditch and Big/Long Creek watershed simulations........20
Table 2.7 Annual observed and simulated watershed responses 20
Table 3.1 Costs and amounts of cover crops 25
Table 3.2 Field operations for alfalfa...26
Table 3.3 Cost of field operations..27
Table 4.1 Land management operations simulated in the baseline scenario 40
Table 4.2 Optimal tradeoff solutions for NM in Big Ditch watershed 43
Table 4.3 Optimal tradeoff solutions for NM in Big/Long Creek watershed...............................43
Table 4.4 Pollutant reduction for the best tradeoff optimal placements of BMPs in Big Ditch watershed 60
Table 4.5 Pollutant reduction for the best tradeoff optimal placements of BMPs in Big/Long Creek watershed 61
Table 4.6 Equal annual cost (EAC) for the best tradeoff optimal placement of BMPs in Big Ditch..62
Table 4.7 Equal annual cost (EAC) for the best tradeoff optimal placement of BMPs in Big/Long Creek..62

List of Figures

Page

Figure 1 Location map of Big Ditch and Big/Long Creek watersheds..2
Figure 2.2 Coarse, intermediate and fine delineations for Big Ditch watershed 9
Figure 2.3 Effect of watershed subdivision levels on subbasin average overland slope length 10
Figure 2.4 Effect of watershed subdivision levels on subbasin average channel slope and drainage density 10
Figure 2.5 Effect of watershed subdivision levels on average streamflows and sediment loads ..11
Figure 2.6 Effect of watershed subdivision levels on average nitrate and organic N loads 13
Figure 2.7 Effect of watershed subdivision levels on mineral P and organic P loads 13
Figure 2.8 Flow diagram used in preparation of land management files for the watershed models..16
Figure 2.9 Observed and simulated monthly flow for Big Ditch watershed 20
Figure 2.10 Observed and simulated monthly sediment for Big Ditch watershed 21
Figure 2.11 Observed and simulated monthly nitrate for Big Ditch watershed 21
Figure 2.12 Observed and simulated monthly total phosphorus for Big Ditch watershed 22
Figure 2.13 Observed and simulated monthly sediment for Big/Long Creek watershed 22
Figure 2.14 Observed and simulated monthly nitrate for Big/Long Creek watershed 23
Figure 3.1 Big Ditch HRUs suitable for saturated buffers...31
Figure 3.2 Big/Long Creek HRUs suitable for saturated buffers 32
Figure 4.1 Solution framework of the Decision Support Model (DSM) 36
Figure 4.2 Optimization string representing BMP implementation scenario 38
Figure 4.3 NM scenarios evolving to Pareto optimal front 41
Figure 4.4 NM decision variables for Pareto optimal front and all model evaluations 42
Figure 4.5 NM: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds 43
Figure 4.6 Cover crops: optimal tradeoffs and treatment areas for Big Ditch watershed 44
Figure 4.7 Cover crops: optimal tradeoffs and treatment areas for Big/Long Creek watershed ...45
Figure 4.8 Optimal placement of cereal rye in Big Ditch watershed 46
Figure 4.9 Perennial-Alfalfa: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds ..47
Figure 4.10 Optimal placement of alfalfa in Big/Long Creek watershed 48
Figure 4.11 Constructed wetlands: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds...50
Figure 4.12 Optimal placements of constructed wetlands in Big Ditch watershed 51

List of Figures (cont'd)

Page

Figure 4.13 Drainage water management: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds 52
Figure 4.14 Optimal placement of drainage water management in Big/Long Creek watershed ... 53
Figure 4.15 Bioreactors: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds 54
Figure 4.16 Optimal placement of bioreactors in Big Ditch watershed 55
Figure 4.17 Saturated buffers: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds 56
Figure 4.18 Optimal placement of saturated buffers in Big/Long Creek watershed 57
Figure 4.19 Filter strips: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds 58
Figure 4.20 Optimal placement of filter strips in Big Ditch watershed 59
Figure 4.21 Impacts of cereal rye, ryegrass, crimson clover, and alfalfa on Big Ditch water yield 63
Figure 4.21 Impacts of cereal rye, ryegrass, crimson clover, and alfalfa on Big/Long Creek water yield 64

Acknowledgments and Disclaimer

This report is based on research funded by the Agricultural Watershed Institute (AWI) through a TMDL implementation planning grant from the State of Illinois Environmental Protection Agency (IEPA). The support of Steve John, executive director of AWI, and Trevor Sample, project manager for IEPA, in defining the project scope and reviewing the report is highly appreciated. Santhosh Gajendran performed extensive data processing in the preparation of land management operations for the decision support models. Drs. Momcilo Markus and Gregory McIsaac reviewed the report and provided valuable comments. Lisa Sheppard edited the report and Sara Olson designed the cover. Any opinions, findings, and conclusions or recommendations expressed in this publication are those of the authors and do not necessarily reflect the views of the Illinois State Water Survey.

1. Introduction and Background

Surface and subsurface agricultural runoff has been the main cause of water quality problems in Lake Decatur, which is the major source of public water supply for the City of Decatur and the Village of Mt. Zion, serving a total population of more than 80,000. The lake has a watershed area of 925 square miles and was created by building a dam on the Sangamon River in 1922 with a modification in 1956 to increase its capacity. Extensive siltation is another critical issue, causing loss of significant storage volume. Nearly 90 percent of the Lake Decatur watershed is cropland, of which corn and soybean account for 44 and 39 percent, respectively. The watershed is extensively tile-drained to lower the water table, creating favorable conditions for agricultural production. Hydrologic and water quality monitoring has been conducted from 1993 to 2008 by the Illinois State Water Survey (ISWS) with support from the City of Decatur in an effort to alleviate the water quality problem in Lake Decatur through watershed management alternatives. Additional watershed monitoring was carried out from 2005 to 2008 by ISWS for a United States Environmental Protection Agency (USEPA) targeted watershed study with the goal of addressing economic and environmental aspects of nutrient management in the Upper Sangamon River watershed.

The Illinois Environmental Protection Agency (IEPA) added Lake Decatur to the Illinois 2004 Section 303(d) list as impaired for nitrogen-nitrate and total phosphorus (IEPA, 2004). Consequently, a Total Maximum Daily Load (TMDL) assessment was completed for the Sangamon River/Lake Decatur watershed in 2007 and was approved by the USEPA. The TMDL study provided an overview of implementation alternatives that reduce nitrate and phosphorous loads, including nutrient management, conservation tillage, conservation buffers, and restriction of livestock. In addition, practices that limit losses from private sewage discharges and sedimentation were also proposed to reduce phosphorus loading (IEPA, 2007). Most cropland in the Lake Decatur watershed has been extensively tile-drained and therefore, the effectiveness of surface water-based best management practices (BMPs) for reducing nitrate may be limited. Specific placement areas for implementation of these alternatives have not been identified, which is the focus of this study.

Two tributary watersheds of Lake Decatur were identified for developing alternative implementation scenarios of selected BMPs that are designed to reduce nonpoint source pollutants (NPS) from agricultural sources. The watersheds are Big/Long Creek and Big Ditch watersheds, as illustrated in Figure 1. The Big/Long Creek watershed is located in the downstream portion of the Lake Decatur watershed, draining directly into the lake. In contrast, the Big Ditch watershed is located about 50 miles from the lake in the northeastern edge of the Lake Decatur watershed. Both are agriculturally dominated watersheds and their areas considered in this study correspond to the drainage areas of ISWS monitoring stations, which are close to the respective watershed outlets.

The objective of this research was to evaluate the water quality benefits of selected BMPs at a watershed scale, generating alternative scenarios for implementation in Big Ditch and Big/Long Creek watersheds. This was accomplished through the development of decision support models (DSMs) for each watershed. The DSMs were developed based on an integrated modeling approach, coupling a watershed simulation model known as the Soil and Water Assessment Tool (SWAT) with an Archived-Based Micro-Genetic Algorithm 2 (AMGA2) - a multi-objective optimization algorithm. Such integrated modeling approach, which involves interfacing a simulation model with an optimization algorithm, has been extensively applied to solve complex problems in watershed management (Bekele et al., 2013; Bekele et al., 2011), reservoir operations (Nicklow and Mays, 2000), groundwater monitoring design (Reed and Minsker, 2004), and others. The DSM was designed to generate cost-effective implementation scenarios of selected conventional and newly emerging BMPs that include nutrient management, cover crops, perennial crops, constructed wetlands, drainage water management, bioreactors, saturated buffers, and filter strips. It is capable of providing optimal BMP placement scenarios that result in maximum reduction of NPS pollutants for a prescribed level of BMP implementation. BMP scenarios that strike a balance between NPS reduction and total cost of implementation are identified as best tradeoff solutions and are recommended for preparation of watershed implementation plans.

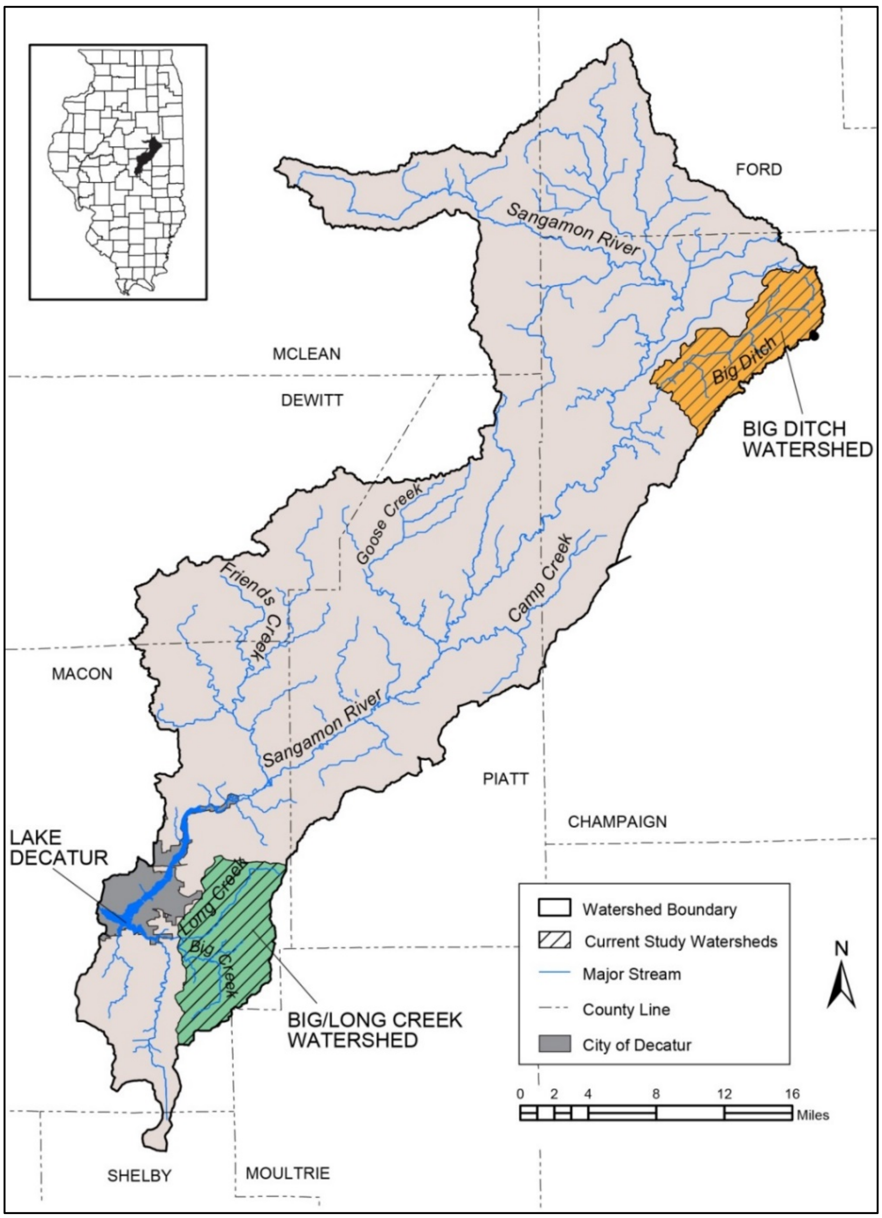


Figure 1. Location map of Big Ditch and Big/Long Creek watersheds

This report discusses the generation of cost-effective alternative BMP scenarios for implementation in the two study watersheds, which required the development and application of watershed and decision support models, representation of selected BMPs, and optimization of BMP placements in the watersheds for maximum possible reduction of NPS pollutants. In Section 2, a series of tasks that were accomplished in developing the watershed models for Big Ditch and Big/Long Creek watersheds are presented. These include characterization of the study watersheds for model development, identifying appropriate discretization levels, and model calibration and validation. BMPs selected for evaluation in this study are described in Section 3. In addition, the specific suitability criteria for implementation of these BMPs are discussed in this section. Section 4 discusses the development and application of the DSMs. In this section, the watershed model and the multi-objective optimization algorithm used in the development of the DSMs are explained, and the operation of the DSMs including application results are also discussed. Finally, a summary and conclusions of this study are provided in Section 5. Limitations of the study and recommendations for future work are also discussed in this section. Further information regarding suitability maps for BMPs and optimal placements of BMPs in the Big Ditch and Big/Long Creek watersheds are provided in Appendices A, B, and C, respectively.

2. Watershed Modeling

In the course of developing the watershed models, a sequence of tasks has been accomplished, and it includes characterizing the study watersheds, identifying appropriate discretization levels, determining sensitive model parameters, and calibration and validation. The watershed characterization is focused on data collected from various sources, including watershed topography, land uses and management practices, soils, weather, stream flows, and water quality. Appropriate discretization levels for the study watersheds were determined through spatial sensitivity analyses conducted with respect to hydrologic and water quality simulations. Next, model parameters that are sensitive to watershed hydrology and water quality processes were identified and calibrated. Validation of the watershed models was conducted in cases where there are available data. This section provides details on the study watersheds and the process of modeling them.

2.1 Study Watersheds

The study watersheds are Big Ditch and Big/Long Creek, which are tributaries of the Upper Sangamon River watershed and eventually drain into Lake Decatur. The water quality of Lake Decatur has been affected by agricultural activities in its upstream watersheds that include Big Ditch and Big/Long Creek. The Big/Long Creek watershed has a drainage area of 46.2 square miles and is located in the downstream portion of the Lake Decatur watershed in Macon County, draining directly into Lake Decatur. In contrast, the Big Ditch watershed is located about 50 miles from the lake in the northeastern edge of the Sangamon River watershed and has a drainage area of 38.2 square miles. The watershed areas correspond to the drainage areas of ISWS monitoring stations on Big Ditch (Station 106) and Long Creek (Station 101) that are very close to the watershed outlets.

To develop a watershed model, specific information about watershed topography, land use, management practices, soil, and weather are required. Hydrologic and water quality data are required for model calibration and validation. For watershed delineation, topographic information was extracted from the National Elevation Data (NED) and National Hydrographic Data (NHD), both of which are obtained from EPA's BASINS website, available at http://www.epa.gov/waterscience /ftp/basins/gis_data/huc/. Land use information used in the model development includes the National Land Cover Database (NLCD2001) obtained from the Multi-Resolution Land Characteristics Consortium (MRLC) project website (http://www.mrlc.gov/nlcd2001.php) and Crop Data Layers (CDLs) for years 1999–2010 from the National Agricultural Statistics Service (http://nassgeodata.gmu.edu-/CropScape/). Although the NLCD was revised in 2006, major updates were made to coastal mapping zones. In addition, since a classification lookup table for NLCD2001 is readily available in the ArcGIS interface of the watershed model used, NLCD2001 is used for general classification of land use in the study watershed. Tillage data from 1995 to 2010 were obtained for Champaign and Macon Counties from the Illinois Department of Agriculture. For accurate representation of historical crop rotations in the study watersheds, tillage data and land use information extracted from NLCD and CDL were utilized. Historical planting, harvesting, and field work dates were obtained from regional Crop Progress Reports. Fertilizer type, application rate, and timing were determined using inputs from stakeholders and experts invited to project meetings. Soil characteristics of the study watersheds were extracted from SURRGO maps for Champaign and Macon Counties (http://websoilsurvey.sc.egov.usda.gov). Weather information, including precipitation, temperature, relative humidity, solar radiation, and wind speed, were obtained for Decatur and Rantoul from Illinois Climate Network (http://www.isws.illinois.edu/warm/) and the Midwestern Regional Climate Center (http://mrcc.isws.illinois.edu/). Hydrologic and water quality data were obtained from the Illinois State Water Survey (ISWS). With support from the city of Decatur, ISWS has conducted 15 years of hydrologic and water quality monitoring throughout the Lake Decatur watershed from 1993 to 2008 (Keefer et al., 2010). Additional watershed monitoring from 2005 to 2008 was conducted by ISWS as part of a USEPA targeted watershed study (Keefer and Bauer, 2011). Both datasets were used in the watershed model development for the study watersheds.

More than 90 percent of the Big Ditch watershed is used for agricultural row crops, particularly corn and soybean. About two-thirds of the watershed has a slope of less than 2 percent. In contrast, agricultural row crops account for about 80 percent of the Big/Long Creek watershed, and 80 percent of the watershed has slopes of less than 2 percent. Almost all soils of the Big/Long Creek watershed and about three-fourths of Big Ditch watershed soil belong to hydrologic soil group B, exhibiting moderate infiltration capacity. The average annual total precipitation in Big Ditch and Big/Long Creek watersheds is 999 and 1,015 millimeters, respectively.

2.2 Soil and Water Assessment Tool (SWAT)

SWAT is one of the most widely used, semi-distributed hydrologic models in the U.S. and elsewhere. It is designed to predict the long-term impacts of land management practices on water, sediment, and agricultural chemical yields in large complex watersheds with varying soils, land use, and management conditions (Nietsch et al., 2011). A suite of algorithms is incorporated into SWAT to simulate hydrologic and water quality processes such as surface and subsurface flows, sediment transport, nutrient transport and cycling, and plant growth. For simulating a watershed, the model requires specific information about its topography, land uses, soils, weather, and land management conditions. SWAT has an ArcGIS extension (ArcSWAT) that simplifies spatial data processing, watershed delineation, writing model input files, and visualization of model simulation outputs. The minimum data required for watershed simulation are predominantly available from governmental agencies (Nietsch et al., 2011).

In modeling with SWAT, a watershed is divided into subbasins and in each subbasin, hydrologic response units (HRUs) are defined based on a unique combination of land use, soil, and slope categories. In an HRU, the water balance is represented by storage volumes for snow, soil profile (less than 2 meters below the surface), shallow aquifer (2-10 meters), and deep aquifer (greater than 20 meters). Flow, sediment, and nutrient and pesticide loadings generated in the HRUs are added and routed through channel networks, reservoirs, ponds, and/or wetlands to the watershed outlet. In this study, the Natural Resources Conservation Service's Curve Number method (SCS, 1972) and the variable storage method (Williams, 1969) were used for surface runoff generation and channel routing, respectively.

Sediment Yield and Routing

SWAT estimates erosion and sediment yield caused by a runoff using the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1995) that is given as:

$$Sed_{yield} = 11.8 \times \left(Q_{surv} \times q_{peak} \times A_{hru}\right)^{0.56} \times K \times C \times P \times (LS) \times F_c$$

where Sed_{yield} is the sediment yield on a given day (metric tons), Q_{surv} is the surface runoff volume ($mm\ H_2O/ha$), q_{peak} is the peak runoff rate (m^3/s), A_{hru} is the area of the HRU (ha), K is the Universal Soil Loss Equation (USLE) soil erodibility factor, C is the USLE cover and management factor, P is the USLE support practice factor, LS is the USLE topographic factor, and F_c is the coarse fragment factor. MUSLE uses the runoff energy factor in both detachment and transportation of sediments. The sediment routing model in SWAT is controlled by deposition and degradation processes in the channel. In this study, a simplified version of the Bagnold (1977) stream power equation, which calculates the maximum sediment transport as a function of peak channel velocity, is used. In this method, erosion is limited only by the transport capacity, resulting in unlimited sediment supply from channel erosion. At each time step, the maximum sediment concentration is compared with sediment concentration in the channel to determine the amount of sediment deposited and/or re-entrained. Once deposition and degradation amounts are determined, the final amount of sediment in the reach is determined by (in metric tons)

$$Sed_{ch} = Sed_{ch,i} - Sed_{dep} + Sed_{\deg}$$

where Sed_{ch} is the amount of suspended sediment in the reach, $Sed_{ch,i}$ is the amount of suspended sediment in the reach at the beginning of the time period, Sed_{dep} is the amount of sediment deposited in the reach segment, and Sed_{deg} is the amount of sediment re-entrained in the reach segment. The amount of sediment transported out of the reach is computed as

$$Sed_{out} = Sed_{ch} \times \frac{V_{out}}{V_{ch}}$$

where Sed_{out} is the amount of sediment transported out of the reach, V_{out} is the volume of outflow during the time step, and V_{ch} is the volume of water in the reach segment.

Nutrient Transformation and Movement

The transformation and movement of nutrients within an HRU is based on nitrogen (N) and phosphorus (P) cycles. SWAT tracks five different N pools in the soil, two of which are inorganic (mineral) forms including ammonium and nitrate. The remaining three are organic forms of nitrogen and they include fresh organic nitrogen associated with crop residue and microbial biomass and organic nitrogen associated with humus, which has active and stable forms based on their availability to mineralization. Six different pools of P forms are simulated in SWAT, and three of them are inorganic P forms that include solution and active and stable pools. Similar to N pool, the remaining three are organic P forms associated with crop residue, microbial biomass, and soil humus.

Inorganic and organic forms of N and P are commonly introduced to the soil system through fertilizer and/or livestock manure applications. Plant residue is also one of the sources of organic N and P in the soil. Nitrogen and phosphorus losses from the soil system occur as a result of plant uptake and surface runoffs in a solution form or nutrients bound to eroded sediments. Losses of N can also take place with the percolation of water below the root zone (leaching), with lateral subsurface flow including tile drainage, and by volatilization to the atmosphere. Nitrate is prone to leaching as it is not sorbed by soil particles. In modeling with SWAT, the N and P plant uptakes are calculated using a supply and demand approach. The movement of nitrate in surface runoff, lateral subsurface flow, and percolation is simulated as the product of the average nitrate concentration in the soil layer and associated volume of water in the flow pathway. Simulation of soluble P loss via surface runoff is determined as a function of the solution P concentration in the top 10 millimeters of the soil, the surface runoff volume, and a partitioning factor for the P concentration. Because of its low mobility, leaching of soluble P is allowed only from the top 10 millimeters of the soil. Movement of organic N or organic P and inorganic P bound to eroded sediment is estimated with a loading function initially derived by McElroy et al. (1976) and later modified for individual runoff events by Williams and Hann (1978). Daily N and P losses are computed as a function of the nutrient concentration in the topsoil layer, the sediment yield, and an enrichment ratio, which is a ratio of N or P concentration transported with the sediment to N or P concentration in the soil layer.

Plant Growth Simulation

SWAT simulates plant growth based on the concept of heat unit theory, which states that plants have a quantifiable heat requirement related to their time of maturity. Temperature is one of the most important factors governing plant growth, and the minimum temperature required for growth is the plant's base temperature. A heat unit is calculated as the difference between mean daily temperature and the plant's base temperature, contributing to the plant development. The total number of heat units required to bring a plant to maturity can be computed once the mean daily temperatures, base temperature, and planting and maturity dates are known. Modeling of the plant growth includes simulations of leaf area development, light interception, and conversion of intercepted light into biomass using plant-specific radiation use efficiency (i.e., the amount of plant biomass produced per unit of intercepted solar radiation). A portion of plant biomass accumulated above the ground on the day of the harvest is calculated as yield. Harvest efficiency can be specified to identify the biomass yield removed during harvesting. SWAT also simulates the reduction of plant growth as a result of extreme temperatures and inadequate water and nutrient availability.

Field Management Operations

Since SWAT is developed to predict the impact of land management operations on watershed hydrology and water quality, it allows a detailed simulation of agricultural management operations, water management, and some urban processes. Accurate representation of the agricultural and water management operations are central to this study, and a brief description of these management operations, which include *plant*, *harvest*, *harvest and kill, tillage*, *fertilization,* and tile drainage are provided here. A *plant* operation is used to designate the initiation of plant growth in a hydrologic response unit (HRU). Planting dates and the total number of heat units are the only inputs required. Without killing the plant, removal of plant biomass from an HRU can be done using a *harvest* operation, and the harvesting date is the only input required for this operation. Harvest efficiency determines the fraction of plant biomass removed from an HRU, and the remaining portion will be converted to residue. Plant residue plays an important role in reducing soil erosion. Plant growth in an HRU can be terminated by either a *kill* or a *harvest* and *kill* operation. A *kill* operation stops the plant growth and converts all plant biomass into a residue, whereas a *harvest* and *kill* operation removes only a portion of the plant biomass from an HRU based on the harvest efficiency, and only the remaining portion is converted to a residue. Tillage plays an important role in redistribution of residue, nutrients, and pesticides in the soil profile and can be simulated if the type of tillage and the time of operation are known. A *fertilization* operation can be used to simulate a fertilizer application. Date, type, and rate of fertilizer application are required inputs and if these inputs are not available, SWAT's auto-fertilization can be invoked.

Simulation of tile drainage can be done using SWAT by specifying the depth of tile drains from the soil surface, the time required to drain the soil to field capacity, and the lag time between water entering and leaving tile drains. Tile drainage occurs when the water table rises above the depth where the tile drains are located. It is calculated as:

$$tile_{flow} = \frac{h_{wi} - h_{di}}{h_{wi}} \times \left(W_{sp} - W_{fc}\right) \times \left(1 - \exp\left[\frac{-24}{t_{drain}}\right]\right)$$

where $tile_{flow}$ is the amount of water removed from the soil layer by tile drainage (mm/day), h_{wi} is water table height above the impervious zone (mm), h_{di} is tile drain height above the impervious zone (mm), W_{sp} is the water content of the soil profile (mm/day), W_{fc} is the water content of the soil profile at field capacity (mm/day), and t_{drain} is the time required to drain the soil to field capacity using tile drains (hrs).

2.3 Spatial Sensitivity Analysis

Identifying the optimal watershed subdivision levels is central to adequately representing the heterogeneity in watershed characteristics and thereby efficiently simulating watershed processes. While developing watershed models, it is commonplace to divide a watershed into smaller subbasins that are representative of homogenous conditions. Coarser watershed delineations result in a small number of subbasins, aggregating areas with variable conditions. As the delineation gets finer, the number of subbasins increases, capturing watershed variability. This in turn increases model input data preparation and associated computational demand. Different watershed subdivision levels affect watershed simulations as result of changes in geomorphic properties, including channel network, topography, land use, and soils. Weather inputs are homogenous at the subbasin level. Several studies showed that streamflow simulation is relatively insensitive to subbasin subdivision levels, whereas sediment simulations are found to be sensitive (Binger et al., 1997; FitzHugh and MackKay, 2000; Jha et al., 2004). Jha et al., (2004) reported that nitrate losses in their study watersheds showed clear sensitivity to subwatershed sizes and the trends in predicted nitrate losses reflect the complexity of SWAT nutrient routing algorithms in terms of simulating nutrient losses and transformations. The two watersheds in this study have drainage areas about 5 percent as large as the smallest watershed considered in the study of Jha et al., (2004). Sensitivities of hydrologic and water quality responses to spatial scale are watershed-specific and therefore, a spatial sensitivity analysis is conducted to identify the threshold subdivision levels for Big Ditch and Big/Long Creek watersheds. The threshold critical source area (CSA), which is the minimum drainage area required to define the detail of watershed stream network, is used to set up the watershed subdivision levels. Nine different watershed subdivision levels were generated for each watershed using CSAs ranging from 0.5 to 11.3 percent of the watershed area as presented in Table 2.1. Therefore, a total of 18 watershed models has been developed and evaluated to analyze the sensitivity of hydrologic and water quality simulations to spatial scale.

Table 2.1 Watershed subdivision levels

Big Ditch watershed (A = 10,639 ha)			Big/Long Creek watershed (A = 12,336 ha)		
Minimum drainage area (ha / %)	Number of subbasins	Average subbasin area (ha)	Minimum drainage area (ha / %)	Number of subbasins	Average subbasin area (ha)
1,200/11.3	6	1,773	1,200/9.7	5	2,467
600/5.6	16	665	600/4.9	15	822
360/3.4	22	484	360/2.9	19	649
300/2.8	26	409	240/1.9	27	457
210/2.0	38	280	210/1.7	35	352
150/1.4	48	222	150/1.2	49	252
120/1.1	56	190	120/1.0	59	209
90/0.8	62	172	90/0.7	71	174
65/0.6	96	111	60/0.5	95	130

For illustration purposes, watershed subdivisions resulting from CSAs of 1200, 210, and 65 hectares are shown in Figure 2.2, representing a range of delineations from coarsest to finest. The different subdivision levels impacted model topographic attributes, including subbasin average overland slope, slope length, channel slope, and channel length, as illustrated in Figures 2.3 and 2.4. As the delineation gets finer, the average overland slope length showed a decreasing trend, whereas the average channel slope and drainage density (i.e., the total channel length divided by total drainage area) increased because of an increased representation of stream networks with channels instead of simplified overland flow components.

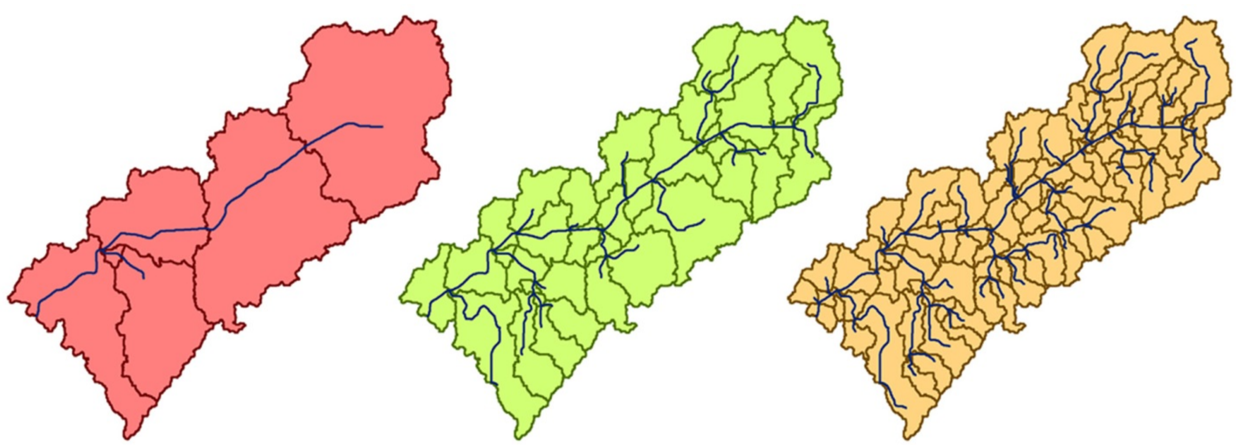


Figure 2.2 Coarse, intermediate, and fine delineations for Big Ditch watershed

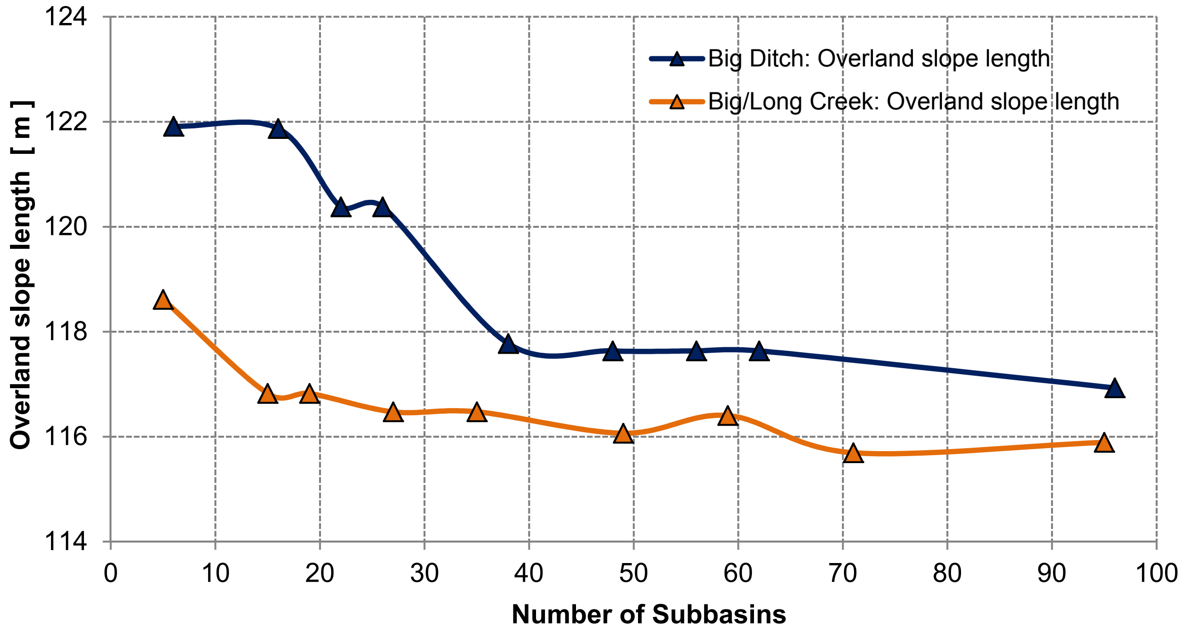


Figure 2.3 Effect of watershed subdivision levels on subbasin average overland slope length

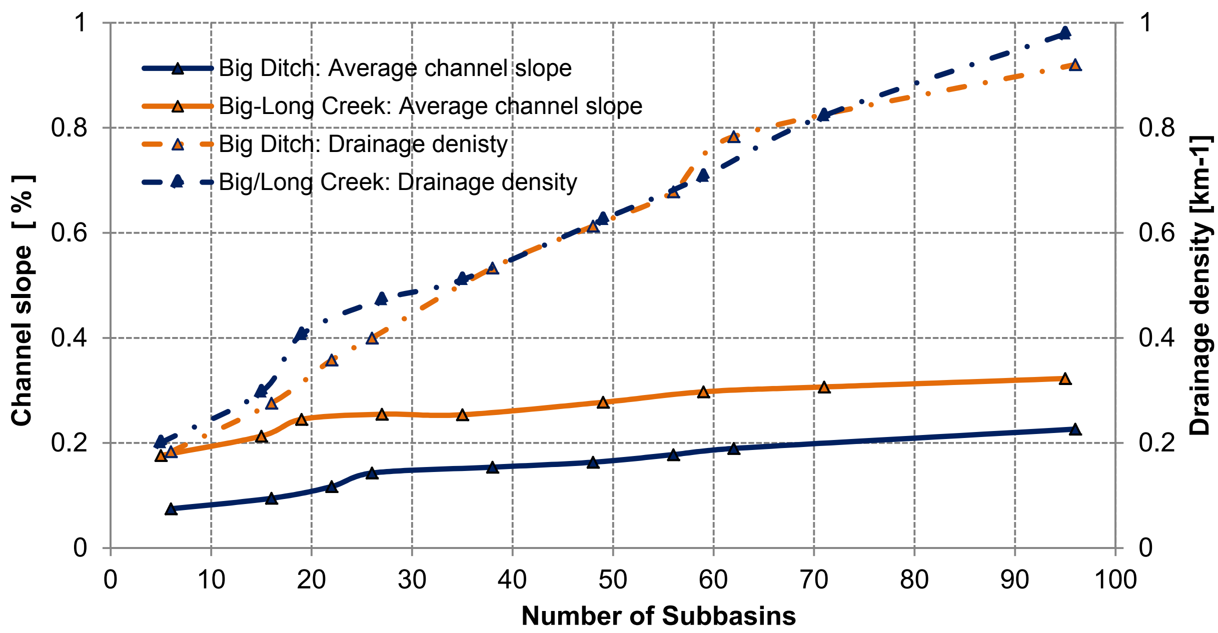


Figure 2.4 Effect of watershed subdivision levels on subbasin average channel slope and drainage density

Sensitivity of Watershed Responses to Spatial Scale

Long-term watershed simulations were conducted to evaluate the sensitivity of watershed responses to different watershed subdivision levels. Each of the 18 watershed models was run for a simulation period of 25 years from 1986 to 2010, and average annual watershed responses were compared between the different delineations as a percentage of watershed responses of the coarsest delineations. The watershed responses considered include average annual streamflows at the watershed outlets, and average annual sediment, nitrate N, organic N, mineral P, and organic P loads transported out of the watershed outlets.

Streamflows and Sediment Loads

In Figure 2.5, percentage changes in simulated streamflows and sediment loads for different watershed subdivision levels are plotted for Big Ditch and Big/Long Creek watersheds as a function of watershed responses for their respective coarsest delineations. The maximum difference in streamflow simulations between the delineations is less than 1 percent for both watersheds, indicating that streamflow simulated by SWAT is relatively insensitive to changes in watershed subdivision levels. This is consistent with other studies reported by Muleta et al. (2007), Jha et al. (2004), and FitzHugh and Mackay (2000). It should be noted that HRU definitions were kept identical in all of the watershed models because any alterations will significantly affect streamflow simulations. Streamflow results imply that CN-based runoff generation simulated at an HRU level is not significantly affected by the size of subbasins. In addition, in SWAT model, groundwater and lateral flow components are assumed to reach to subbasin outlets for further routing downstream and thus are not affected by subbasin size.

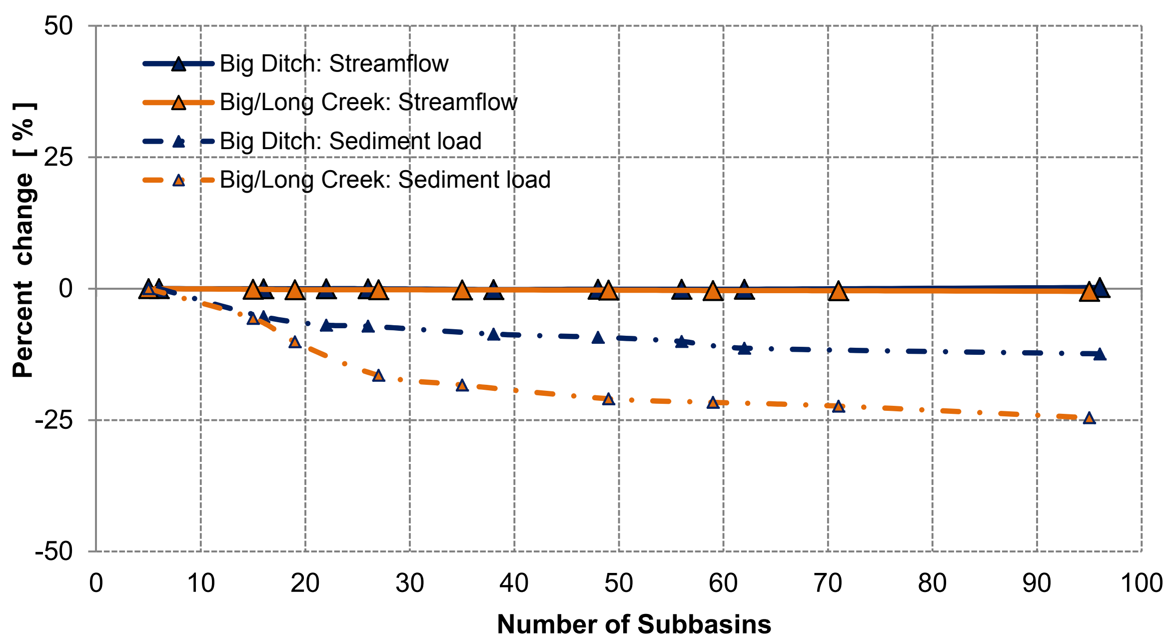


Figure 2.5 Effect of watershed subdivision levels on average streamflows and sediment loads at the watershed outlets

Unlike streamflows, sediment load simulations in both watersheds are found to be sensitive to the number of subbasins and show a decreasing trend with an increase in the number of subbasins. Between the coarsest and finest delineations, sediment loads vary by 13 and 25 percent for Big Ditch and Big/Long Creek watersheds, respectively. Watershed subdivision levels affect sediment load-generating processes and sediment routing because of resulting variations in subbasin sizes and channel lengths. Subbasin sediment loadings are estimated using MUSLE, and MUSLE's runoff factor is a function of subbasin area and its topographic factor. The topographic factor is a function of overland slope length, which is affected by subdivision levels for both watersheds (see Figure 2.3). As the delineation gets finer, the sediment load simulations generally showed a decreasing trend similar to the overland slope lengths. The sediment routing model in SWAT makes use of peak channel velocity to control channel deposition and degradation processes and the channel velocity is affected by channel dimensions that can vary with different delineations. Drainage density and channel slope affect the deposition of sediment caused by settling and degradation processes in the channel, impacting sediment loads at the watershed outlets. For the Big Ditch watershed, the sediment load decreased at a higher rate as the number of subbasins increased from 6 to 22. Subdivisions resulting in more than 22 subbasins provide no appreciable change in sediment load values. However, fewer than 22 subbasins could result in unstable sediment simulations. Similarly, the decreasing rate in sediment loads was higher for the Big/Long Creek watershed as the number of subbasins increased from 5 to 27 and further subdivision levels have not changed simulation results significantly. Therefore, the appropriate threshold levels for sediment load simulations in Big Ditch and Big/Long Creek watersheds are 22 and 27 subbasins, respectively.

Nutrient Loads

The effect of watershed subdivision levels on average annual nutrient loads including nitrate, mineral (soluble) phosphorus, organic nitrogen (N), and phosphorus (P) are presented for Big Ditch and Big/Long Creek watersheds in Figures 2.6 and 2.7. Similar to streamflows, nitrate loads for both watersheds were found to be relatively insensitive to subbasin sizes. Between the different delineations, the maximum fluctuation in nitrate loads was 3.7 percent. In contrast, mineral P loads show clear sensitivity to watershed subdivision levels, particularly for Big/Long Creek watershed as the number of subbasins rises from 5 to 27. This is due to varying in-stream nutrient processes in the two watersheds resulting from topographic differences such as channel slopes and lengths between the different delineations. For simulation of mineral P losses, threshold levels of 38 and 27 subbasins appear to be appropriate for Big Ditch and Big/Long Creek watersheds, respectively.

Simulated organic N and P loads are found to be sensitive to subbasin sizes and showed a decreasing trend as the delineation gets finer, similar to sediment loads because the sediment routing tracks nutrients adsorbed to the sediment. For simulation of organic N and P in the Big Ditch watershed, a threshold level of 22 subbasins is adequate. For the Big/Long Creek watershed, the thresholds were determined to be 35 and 27 subbasins for organic N and P load simulations, respectively.

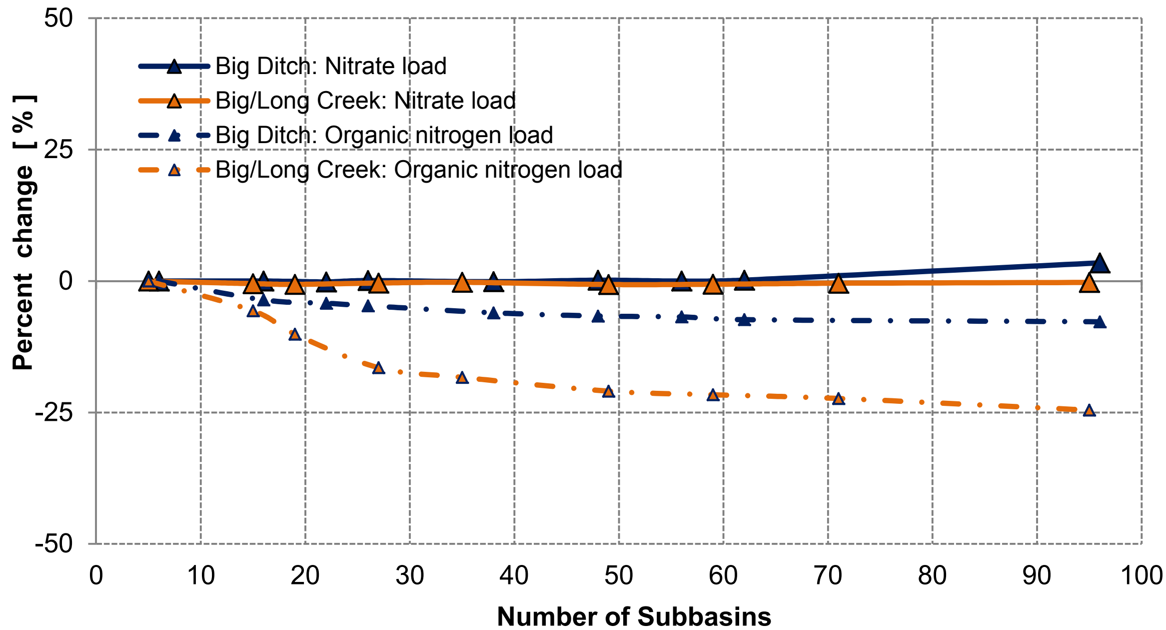


Figure 2.6 Effect of watershed subdivision levels on average nitrate and organic N loads

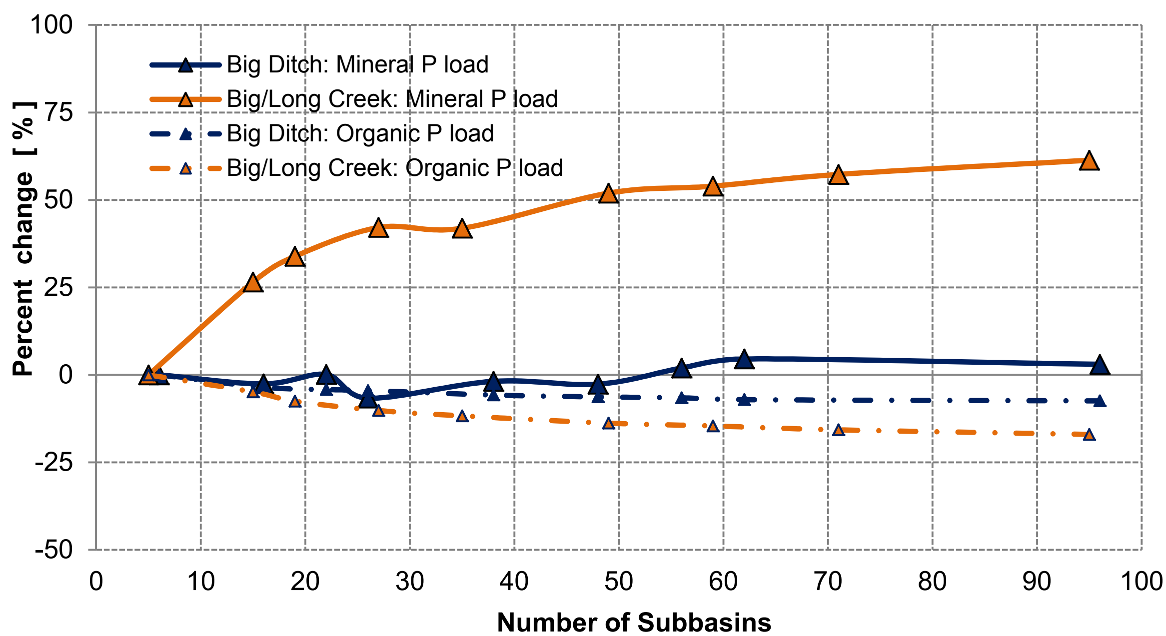


Figure 2.7 Effect of watershed subdivision levels on mineral P and organic P loads

Recommended Subbasin Delineations

The spatial sensitivity analysis showed that there is a threshold watershed subdivision level beyond which no significant change in simulated watershed responses is exhibited. Table 2.2 lists appropriate watershed subdivisions for simulating flow, sediment, and nutrient loads in Big Ditch and Big/Long Creek watersheds. Since streamflow and nitrate were found to be insensitive to different subbasin delineations of the study watersheds, all resulting watershed subdivision levels would be appropriate for streamflow and nitrate simulations. For sediment simulation, the respective threshold drainage areas were found to be 3.4 percent of Big Ditch and 2.9 percent for Big/Long Creek watershed areas. This result is consistent with the studies of Jha et al. (2004) in which a threshold area of 3 percent is recommended for adequate and efficient simulation of sediment yield. Threshold drainage areas that are 2 and 2.9 percent of the total watershed area are recommended for simulating mineral P losses in Big Ditch and Big/Long Creek watersheds, respectively. For both organic N and P simulations, a threshold drainage area of 3.4 percent is recommended for the Big Ditch watershed. In contrast, appropriate threshold drainage areas for the Big/Long Creek watershed were found to be 1.9 and 2.9 percent for simulating organic N and organic P losses, respectively.

Table 2.2 Appropriate watershed subdivisions for streamflow, sediment, and nutrient simulations

Simulated responses at watershed outlets	Big Ditch watershed			Big/Long Creek watershed		
	Number of subbasins	Average subbasin area (ha)	Critical source area (%)	Number of subbasins	Average subbasin area (ha)	Critical source area (%)
Flow	6 - 96	111 - 1773	0.6 - 11.3	5 - 95	130 - 2467	0.5 - 9.7
Sediment	22	484	3.4	27	457	2.9
Nitrate	6 - 96	111 - 1773	0.6 - 11.3	5 - 95	130 - 2467	0.5 - 9.7
Organic N	22	484	3.4	35	352	1.9
Mineral P	38	280	2	27	457	2.9
Organic P	22	484	3.4	27	457	2.9

The result of the sensitivity analysis underscores the fact that required watershed subdivision levels vary based on the watershed response of interest to be simulated. Since the watershed models for Big Ditch and Big/Long Creek watersheds are developed to evaluate the water quality benefits of best management practices, accurate simulations of watershed hydrology and water quality, including sediment and nutrient loads, is important. Therefore, the threshold drainage area close to 2 percent for which sediment and nutrient load simulations stabilized were selected for further model development, resulting in subdivision of the Big Ditch and Big/Long Creek watersheds into 38 and 35 subbasins, respectively. The selected subdivision levels for both watersheds are highlighted in Table 2.2, showing their critical source areas, number of subasins, and average subbasin areas.

2.4 Big Ditch and Big/Long Creek Watershed Models

The total number of subbasins in the Big Ditch and Big/Long Creek watershed models is set to 38 and 35, respectively, which is further subdivided into hydrologic response units. The HRU is the smallest modeling unit with a unique combination of land use, soil, and slope information. In this study, three slope categories are considered for the HRU definition (i.e., slopes less than 1 percent, between 1 and 2 percent, and greater than 2 percent). For both watersheds, the threshold area for the HRU definition was set at 5 hectares for land use, soil, and slope categories. This means that land use, soil, and slope categories that cover less than 5 hectares are not considered in generating HRUs but their areas are proportionally accounted for by other HRUs in a subbasin. SWAT allows the simulation of detailed land management operations at the HRU level.

Model Calibration and Validation

The watershed model for Big Ditch is calibrated for streamflows, sediment, nitrate, and total phosphorus loads, whereas the Big/Long Creek watershed model is calibrated only for streamflow and nitrate due to a lack of sediment and phosphorus data. Table 2.3 lists data used in model calibration and validation for both watersheds. Sensitive model parameters were first identified for all watershed responses of interest and automatic model calibrations were then performed using a multi-objective optimization algorithm coupled with the SWAT model. Flow and nitrate calibrations were done using observed data from 1994 to 2000, and the remaining data were used for model validation. Due to short periods of records, all the sediment and phosphorus data were used for calibration.

Table 2.3 Data availability for model calibration and validation

Watershed	Station number and location	Watershed Response	Data period
Big Ditch	Station 106: Big Ditch at	Flow	01/1994 - 06/2003
	Champaign county road	Sediment	01/2000 - 06/2003
		Nitrate	01/1994 - 06/2003
	Station 223: Big Ditch at CR 3100N	Total Phosphorus	09/2005 - 12/2008
Big/Long Creek	Station 101: Long Creek at	Flow	01/1994 - 12/2007
	Twin bridge road	Nitrate	01/1994 - 12/2007

Incorporating Field Management Operations

A computer code is written to prepare historical land management inputs for all HRUs in the study watersheds. Figure 2.8 shows a flow diagram for preparing land management files for the watershed models. For each watershed, land use information extracted from crop data layers and additional corn-soybean ratios was used to determine the crop rotations for each HRU. Annual tillage data for Champaign and Macon Counties, which include types of tillage and percent coverage, were applied to Big Ditch and Big/Long Creek watersheds, respectively, depending on HRU land use information. Typical field operations for a corn-soybean rotation were prepared using information collected from stakeholders and experts invited to project meetings. Table 2.4 provides an example of field operations for a corn-soybean rotation with conventional, mulch, reduced, or no till systems. Dates of field operations were estimated using information obtained from historical crop progress reports for east and central Illinois.

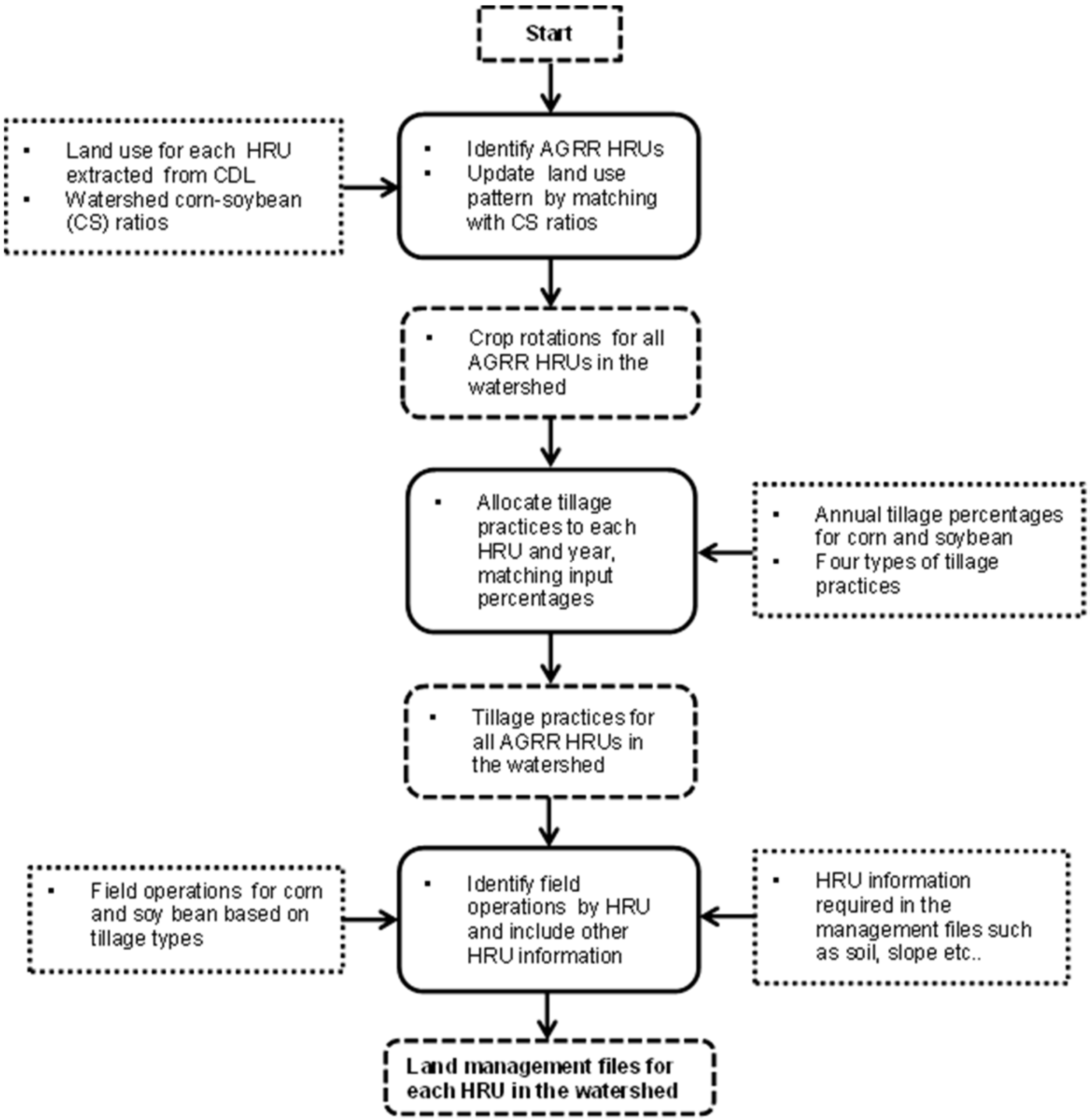


Figure 2.8 Flow diagram used in preparation of land management files for the watershed models

Table 2.4 Typical field management operations

Date of operation	List of land management operations	Operations by tillage types			
		Conventional	Reduced	Mulch	No-till
<u>Year Zero</u>					
October 21	Fertilizer application: DAP (18 lb N/acre; 20.2 lb P/acre)	x	x	x	x
October 22	Tillage implement: Tandem disc	x			
October 22	Tillage implement: Disc ripper		x		
October 22	Tillage implement: Vertical tillage			x	
October 23	Tillage implement: Disc ripper	x			
November 02	Fertilizer application: Anhydrous Ammonia (120 N lb/acre - 60% of total)	x	x	x	
<u>Year One</u>					
February 02	Fertilizer application: UAN-28 (40 lb N/acre - 20% of total)				x
April 21	Fertilizer application: UAN-28 (40 lb N/acre - 20% of total)	x	x	x	
April 21	Fertilizer application: Anhydrous Ammonia (120 N lb/acre - 60% of total)				x
May 05	Tillage implement: Field Cultivator	x		x	
May 05	Tillage implement: Vertical tillage		x		
May 07	Plant corn	x	x	x	x
June 01	Fertilizer application: UAN-28 (40 lb N/acre - 20% of total)	x	x	x	x
October 07	Harvest corn	x	x	x	x
October 21	Fertilizer application: DAP (18 lb N/acre; 20.2 lb P/acre)	x	x	x	x
October 22	Tillage implement: Disc ripper	x			
October 22	Tillage implement: Tandem disc	x	x		
<u>Year Two</u>					
May 20	Tillage implement: Soil finisher	x		x	
May 20	Tillage implement: Field Cultivator		x		
May 22	Tillage implement: No-till drill				x
May 22	Plant soybean	x	x	x	x
September 14	Harvest soybean	x	x	x	x

Model Performance Metrics

Model performance is evaluated using three quantitative statistics, including Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and ratio of the root mean square error to the standard deviation of measured data (RSR). The performance metrics were recommended by Moriasi et al. (2007) after reviews of model applications and evaluation methods. RSR incorporates the benefits of error index statistics and is calculated as:

$$RSR = \sqrt{\frac{\sum_{j=1}^{N}(O_j - S_j)^2}{\sum_{j=1}^{N}(O_j - \overline{O})^2}}$$

where O_j and S_j are the j^{th} observed and simulated data points, respectively, $\overline{O}$ is the mean of observed data, and N is the total number of data used during the simulation period. The value of RSR ranges from an optimal value of zero for a perfect model to a very large positive number for a poor model. The lower the value of RSR, the better the model performance. During automatic model calibration, the RSR is used as an objective function to be minimized in a search for optimal model parameters. The second evaluation metric is NSE, which is a normalized statistic quantifying the relative magnitude of the residual variance compared to the variance of the measured data (Nash and Sutcliffe, 1970). The NSE model efficiency shows how well a plot of observed and simulated data fits the 1:1 line and is given as:

$$NSE = 1 - \left[\frac{\sum_{j=1}^{N} \left(O_j - S_j\right)^2}{\sum_{j=1}^{N} \left(O_j - \overline{O}\right)^2} \right]$$

where O_j, S_j, and $\overline{O}$ are as defined earlier. NSE values range from an optimal value of 1.0 to minus infinity, and a value greater than zero indicates a minimally acceptable performance (Gupta et al., 1999). If NSE is less than or equal to zero, the mean of the observed data is a better predictor than the model. The third static used for model evaluation is the percent bias (PBIAS), which measures the average deviation of simulated values from their observed counterparts (Gupta et al., 1999). Lower PBIAS values generally signify accurate model simulations, and exact simulation of observed values provides a PBIAS value of zero. PBIAS is calculated as:

$$PBIAS = 100 \times \left[\frac{\sum_{j=1}^{N} \left(O_j - S_j\right)}{\sum O_j} \right]$$

Moriasi et al. (2007) recommended performance ratings of RSR, NSE, and PBIAS for watershed simulations at a monthly time step. According to their recommendations, model simulations can generally be judged as satisfactory if $NSE > 0.50$ and $RSR < 0.70$, and if PBIAS is within ±25 percent for streamflow, ±55 percent for sediment, and ±70 percent for N and P simulations.

Calibration and Validation Results

Model calibrations were done for monthly flow, sediment, nitrate, and total phosphorus at the outlets of the study watersheds. Calibrated parameter values for both watersheds are given in Table 2.5. The resulting evaluation statistics for calibration and validation are provided in Table 2.6, showing that model simulations are generally more than satisfactory in most cases except sediment and nitrate load simulations for the Big Ditch watershed. Insufficient sediment data and a shorter period of nitrate data for validation resulted in unsatisfactory model outputs at the monthly time step. However, a comparison of average annual observed and simulated watershed responses during the entire simulation period showed more accurate estimation of long-term streamflow, sediment, and nutrient simulations as presented in Table 2.7. Figures 2.9-2.14 show graphical comparisons of monthly observed and simulated watershed responses, indicating models' good performance in simulating flow, sediment, nitrate, and total phosphorus simulations.

Table 2.5 Calibrated parameters for flow, sediment, nitrate, and total phosphorus

Calibration parameters	Description of parameters	Calibrated values for	
		Big Ditch	Big/Long Creek
Flow			
CN2*	Initial SCS curve number for moisture condition II	-0.043	-4.146
GWQMN	Threshold depth of water in the shallow aquifer for return flow to occur (mm)	305.512	300.000
REVAPMN	Threshold depth of water in the shallow aquifer for "REVAP" percolation to deep aquifer to occur (mm)	400.000	326.315
GW_DELAY	Groundwater delay time (days)	22.496	21.678
ESCO	Soil evaporation compensation factor	0.759	0.746
SOL_AWC*	Available water capacity of the soil layer (mm)	-0.013	-5.272
DDRAIN	Depth to the sub-surface drain (mm)	1015.457	1000.000
TDRAIN	Time to drain soil to field capacity (hours)	23.476	24.000
GDRAIN	Drain tile lag timed (hours)	14.004	24.000
SMFMN	Melt factor for snow on December 21 (mm/^{o}C-day)	6.253	8.031
SMFMX	Melt factor for snow on June 21 (mm/^{o}C-day)	1.383	1.000
Sediment			
CH_COV1	Channel erodibility factor	0.146	
CH_COV2	Channel cover factor	0.000	
SPCON	Linear parameter for calculating maximum sediment that can be re-entrained during channel sediment routing	0.000	
SPEXP	Exponent parameter for calculating sediment re-entrained in channel sediment routing	1.395	
USLE_C*	Minimum value of USLE C factor for water erosion	-0.750	-
USLE_P	USLE equation support practice factor	0.100	
HRU_SLP*	Average slope steepness (m/m)	-0.200	
SLSUBBSN*	Average slope length (m)	0.250	
PRF*	Peak rate adjustment factor for sediment routing in the main channel	-0.500	
ADJ_PKR*	Peak rate adjustment factor for sediment routing in the subbasin (tributary channels)	0.377	
Nitrate			
SOL_NO3	Initial NO_3 concentration in the soil layer (ppm)	4.671	3.922
SOL_ORGN	Initial organic N concentration in the soil layer (ppm)	929.180	200.000
NPERCO	Nitrate percolation coefficient	0.319	0.351
CMN	Rate factor for humus mineralization of active organic N	0.001	0.001
CDN	Denitrification exponential rate coefficient	0.001	0.001
SDNCO	Denitrification threshold water content	0.085	0.010
RSDCO	Residue decomposition coefficient	0.023	0.100
BIOMIX	Biological mixing efficiency	0.575	0.900
Total Phosphorus			
SOL_SOLP	Initial soluble P concentration in soil layer (ppm)	2.548	
SOL_ORGP	Initial organic P concentration in soil layer (ppm)	249.690	-
PHOSKD	Phosphorus soil partitioning coefficient (m^3/Mg)	200.000	
PPERCO_SUB	Phosphorus percolation coefficient in soil layer (10 m^3/Mg)	12.173	

*Percent change from original value

Table 2.6 Performance statistics for Big Ditch and Big/Long Creek watershed simulations

Performance evaluation statistic	Calibration				Validation	
	Flow	Sediment	Nitrate	Phosphorus	Flow	Nitrate
<u>Big Ditch Watershed</u>						
RSR	0.366	0.736	0.549	0.656	0.559	0.756
NSE	0.866	0.458	0.699	0.570	0.688	0.428
PBIAS [%]	0.887	6.105	2.313	18.867	6.235	22.423
<u>Big/Long Creek Watershed</u>						
RSR	0.648		0.500		0.500	0.492
NSE	0.580		0.750		0.750	0.758
PBIAS [%]	5.006		6.393		-13.387	-9.148

Table 2.7 Annual observed and simulated watershed responses

Watershed responses	Big Ditch Watershed		Big/Long Creek Watershed	
	Observed	Simulated	Observed	Simulated
Flow [mm]	240.1	235.5	245.8	262.8
Sediment [t/ha]	0.22	0.22		
Nitrate [kg/ha]	31.0	29.0	33.8	31.5
Total Phosphrous [kg/ha]	0.52	0.42		

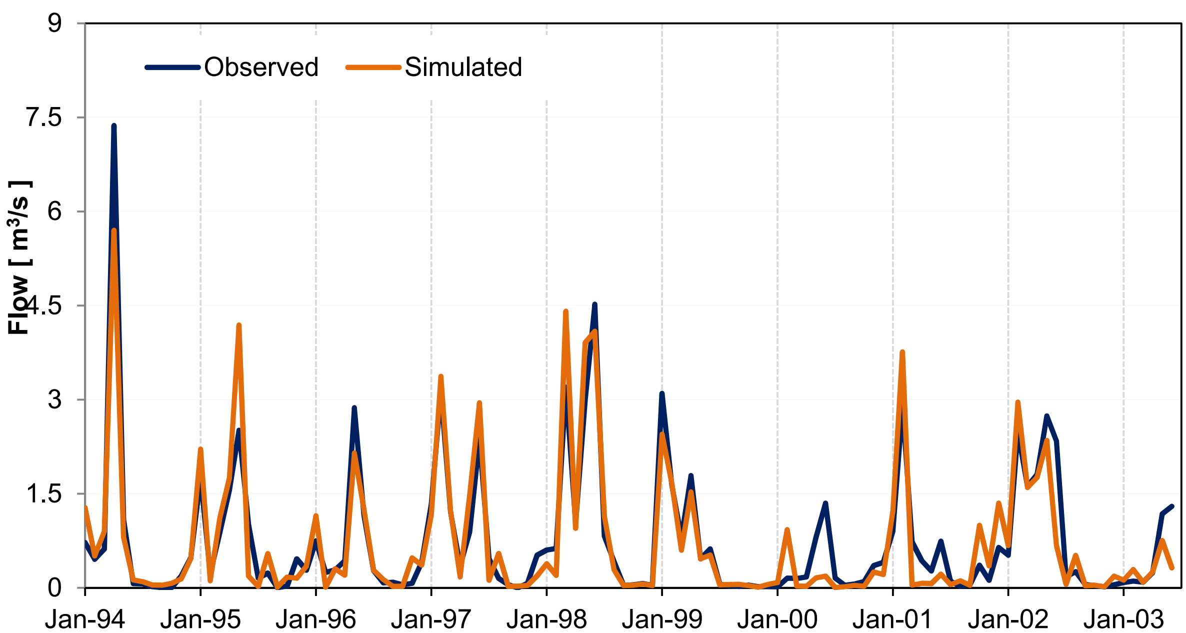


Figure 2.9 Observed and simulated monthly flow for Big Ditch watershed

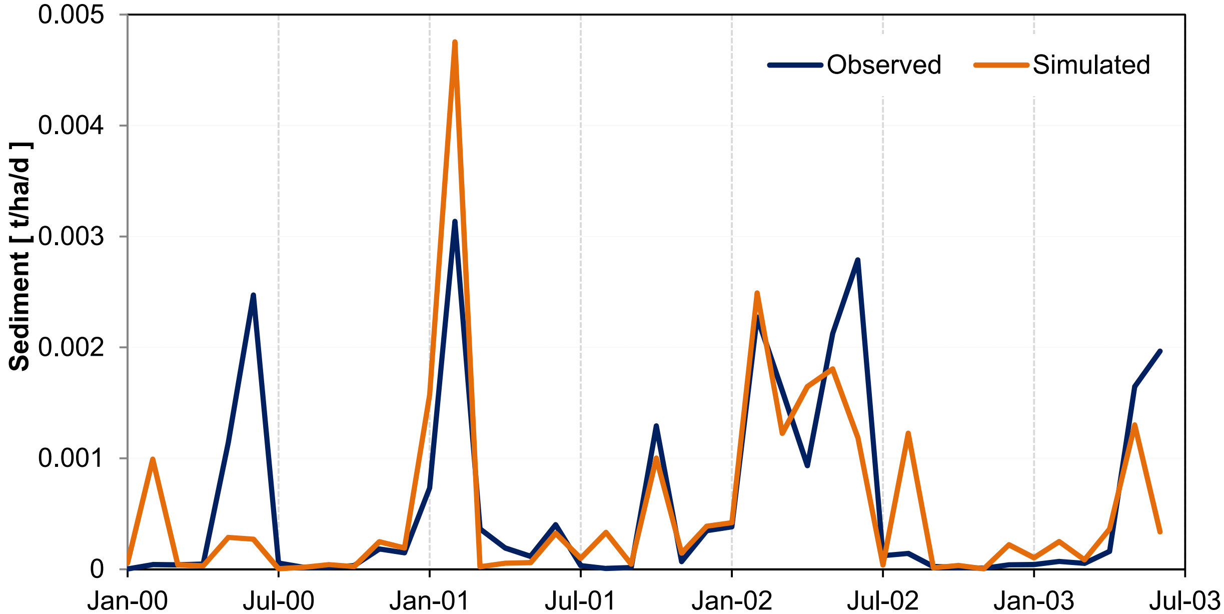


Figure 2.10 Observed and simulated monthly sediment for Big Ditch watershed

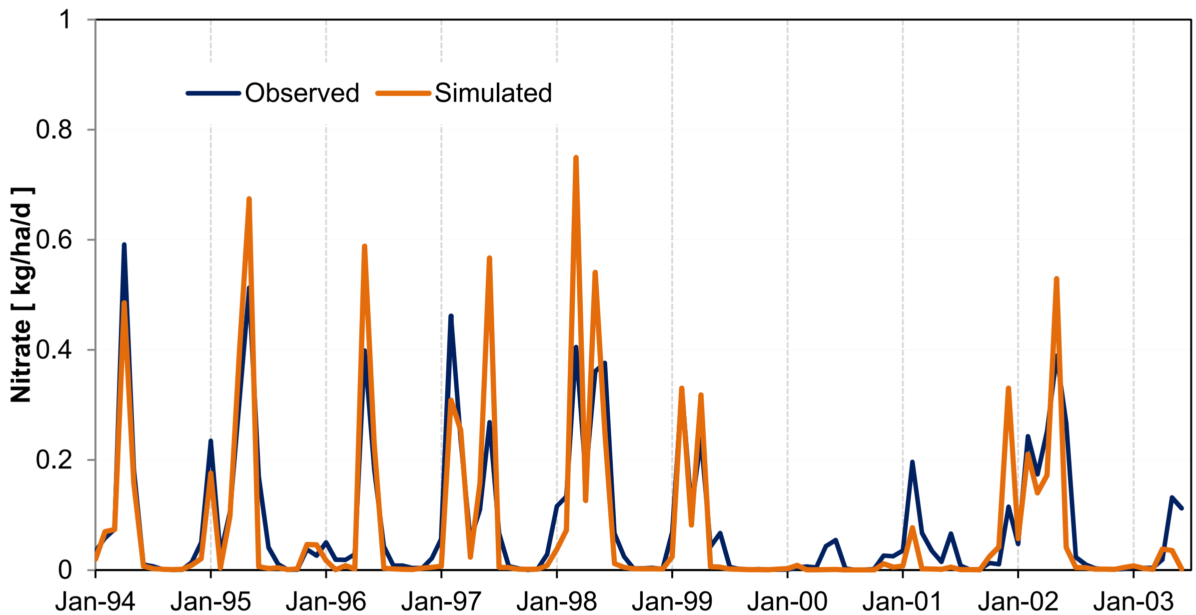


Figure 2.11 Observed and simulated monthly nitrate for Big Ditch watershed

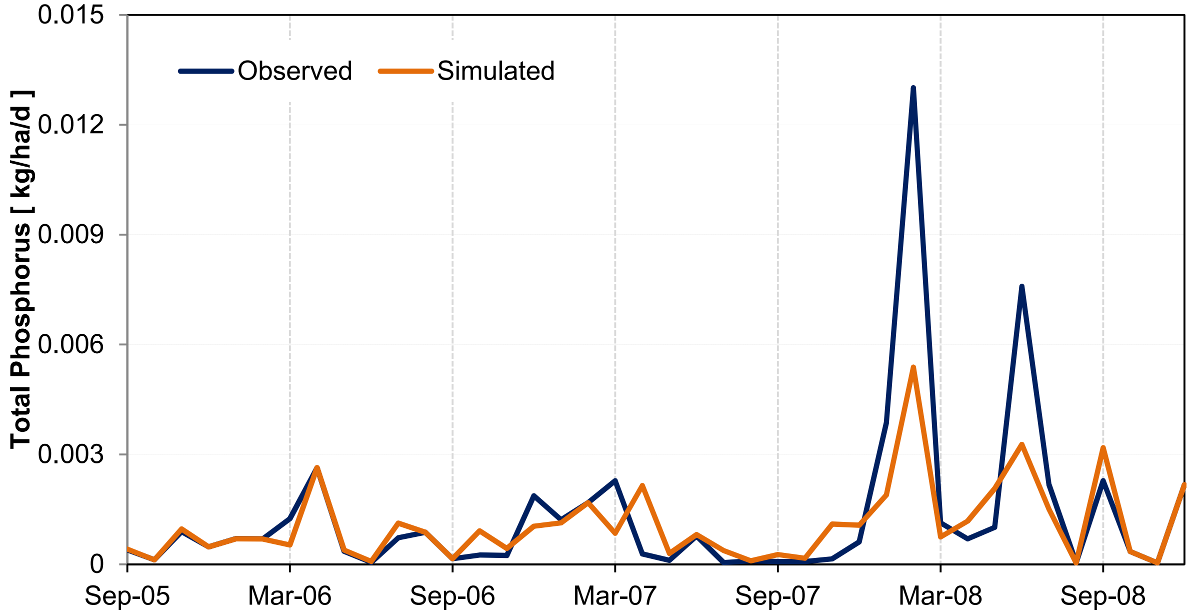


Figure 2.12 Observed and simulated monthly total phosphorus for Big Ditch watershed

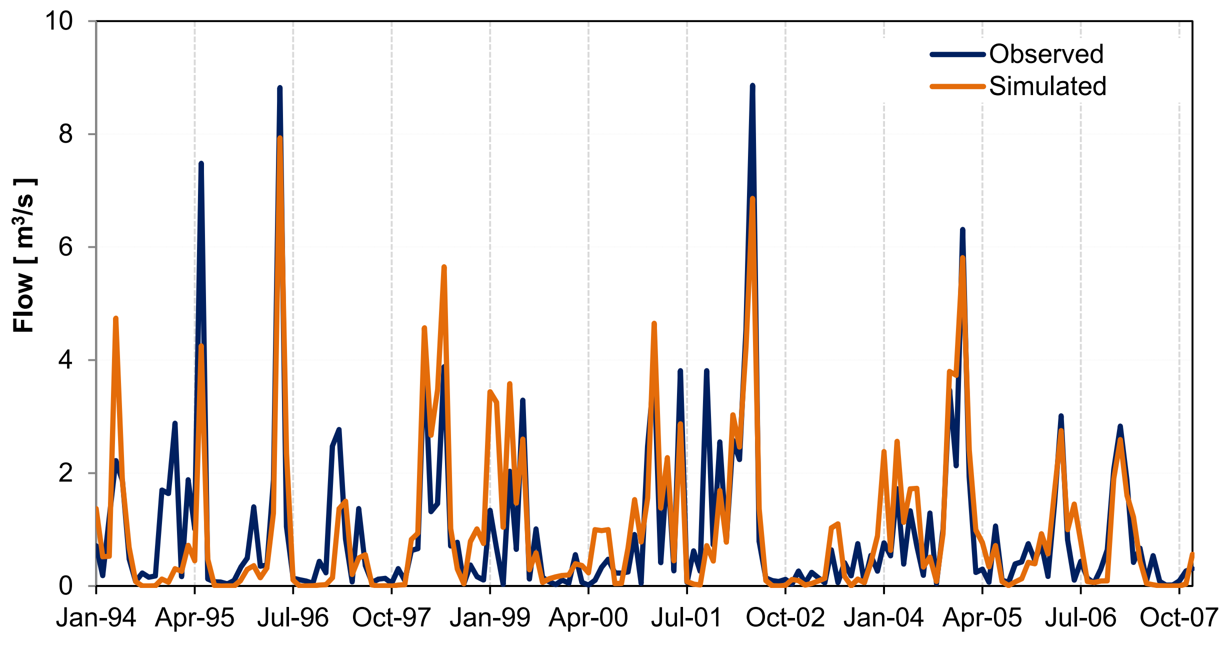


Figure 2.13 Observed and simulated monthly sediment for Big/Long Creek watershed

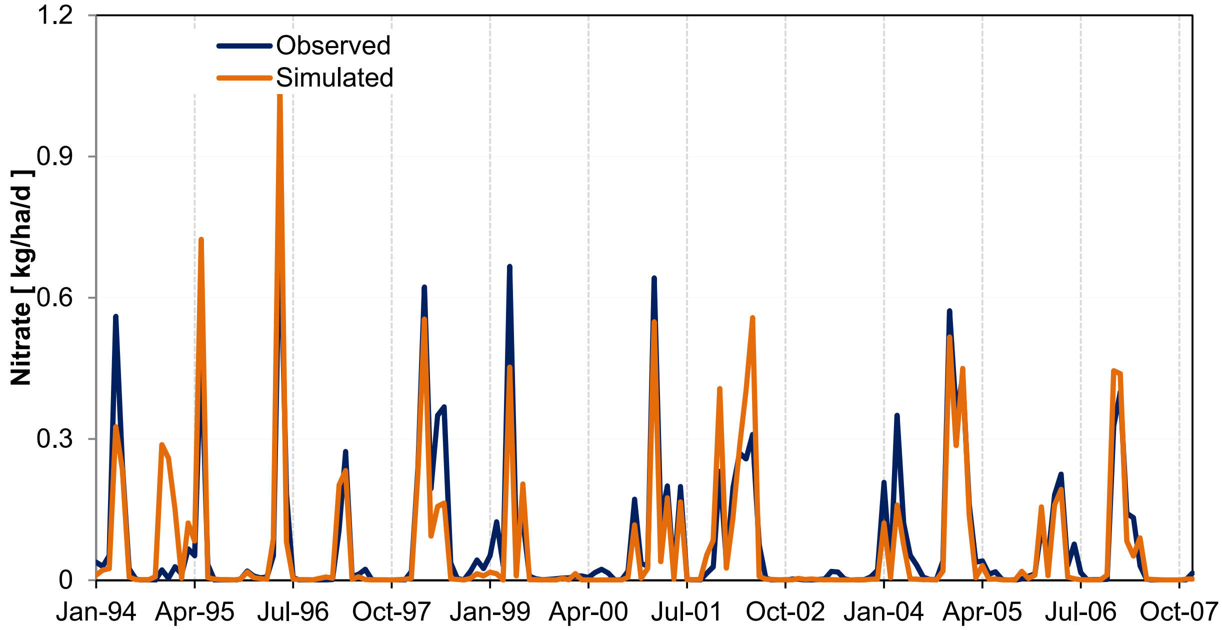


Figure 2.14 Observed and simulated monthly nitrate for Big/Long Creek watershed

3. Selected Best Management Practices and Suitability Criteria

Agricultural best management practices (BMPs) are structural or non-structural control measures that are implemented to reduce the movement of non-point source (NPS) pollutants such as sediment and nutrients from land to water resources. Structural BMPs are conservation controls (e.g., constructed wetlands) built to remove pollutants after they leave their sources. In contrast, non-structural BMPs are conservation practices that are employed to limit the transport of pollutants from their sources (e.g., nutrient management using optimal fertilizer application rate and timing). In this study, a total of nine agricultural BMPs that include both conventional and new emerging practices are considered in an effort to evaluate their implementation and quantify their water quality benefits. The BMPs selected for evaluation include nutrient management (i.e., fertilizer application rate and timing), cover crops, perennial crops, constructed wetlands, drainage water management or controlled drainage, bioreactors, saturated buffer, and filter strips. Brief description of each BMP type, their water quality benefits, and representation in the watershed model are provided in the remainder of this section.

3.1 Nutrient Management

Nutrient management is the practice of using nutrients essential for plant growth such as nitrogen fertilizers in proper quantities and at appropriate times for optimal economic and environmental benefits (USEPA, 2013). For example, fall nitrogen application for the growing season is commonplace because it is more economical to farmers and the fertilizer industry (Fernandez et al., 2010). A significant portion of fall N application is, however, prone to be lost before crop uptake, thereby resulting in water quality problems downstream.

The nutrient management BMP considered in this study is a nitrogen fertilizer application rate and its temporal distribution (i.e., percent application of fall/winter, spring (pre-plant), and side-dressing). The Maximum Return to N (MRTN) approach, which is a cooperative effort among Midwestern universities, is used to limit the lower and upper bound of nitrogen application rate for use in scenario simulations of nutrient management. The MRTN calculation for Illinois makes use of data generated from 400 trials in Illinois since the 1990s that are separated based on regions and type of crop rotations (Fernandez et al., 2010). The profitable N rate range was between 157 lb N/acre and 186 lb N/acre for central Illinois and a corn-soybean rotation, and it was calculated in April 2014 using a corn price of $5 per bushels and an N fertilizer (i.e., anhydrous ammonia) price of $690 per ton.

The types of fertilizers, their application rate, and temporal distribution between fall/winter before and after planting were determined for the study watersheds after consultation with producers and stakeholders. The common fertilizer types used in the watersheds include di-ammonium phosphate (DAP; 18-46-0) with 18 percent N, anhydrous ammonia with 82 percent N, and urea-ammonium nitrate with 28 percent N (UAN-28). Based on information obtained during stakeholders meetings, many producers do not account for the N in DAP as part of their overall N application rate since its application is primarily as a P source. However, all N applications are accounted for during scenario simulations using the watershed models. Anhydrous ammonia is the preferred fertilizer type for fall application because it is inexpensive as compared to others and nitrifies more slowly than other forms (Fernandez et al., 2010). UAN-28 is used for spring pre-planting and side-dressing after planting applications. The average N application rate for the baseline is set at 220 pounds per acre, which also includes the N in DAP. The percent distribution between fall, pre-planting, and after planting applications is 60, 20, and

20, respectively. During BMP scenario simulations, the percentage of N rate for pre-planting and after planting applications are allowed to vary between 20 and 50 percent, and the remaining portion is applied in fall if the HRU is under conventional, reduced, or mulch tillage, or it is applied in winter if the HRU is under no till. The unit cost of the fertilizers used in the scenario simulations is extracted from a bi-weekly Illinois Production Cost Report for December 19, 2013, published by USDA- Illinois Department of Agriculture Market News and it was $498 per ton for DAP, $655 per ton for anhydrous ammonia, and $314 per ton for UAN-28.

3.2 Cover Crops

Cover crops are planted during or after the corn and soybean growing season with the primary goal of improving or maintaining ecosystem quality (Midwest Cover Crops Council (MCCC), 2014). They provide land cover that improves infiltration, reduces soil erosion by wind and water, and decreases nutrient leaching. In drainage waters, cover crops improve water quality by scavenging residual soil nitrate and ammonia. They help increase the quality of the soil by building soil organic matter and sequestering carbon. Additional benefits include provision of winter food and cover for wildlife, enhancing biodiversity (MCCC, 2014).

In this study, cover crops are primarily used as a BMP for improving water quality downstream and therefore, those cover crops that effectively scavenge nitrogen and prevent soil erosion are of particular interest. Taking into account stakeholder inputs and some experiences in the study watersheds, cereal rye, annual ryegrass, and crimson clover were selected as cover crop BMPs for evaluating their water quality benefits. Cover crops are planted between a corn-soybean rotation after the harvest of corn or soybean and are killed before planting of soybean or corn. The planting and killing dates are set to be September 15th and April 10th, respectively. The corn harvesting date is changed from October 7th in the baseline scenario to September 14th during implementation of cover crops in the HRUs. In addition, fall tillage is removed from the field operations in the HRUs whenever the cover crop BMP is implemented. Table 3.1 lists the total establishment cost per acre for cereal rye, annual ryegrass, and crimson clover, which includes the cost of drill planting ($16.4 per acre), in addition to the seed cost. Since all farmers apply herbicide in the spring, no additional herbicide cost will be incurred as a result of cover crop implementation.

Table 3.1 Costs of cover crop implementation

Type of cover crop	Amount [lb/acre]	Unit cost [$/lb]	Total seed cost [$/acre]	Establishment cost [$/acre]
Cereal rye	70	0.37	25.9	42.3
Annual ryegrass	15	1.3	19.5	35.9
Crimson clover	16	2.5	40	56.4

3.3 Perennial Crops

Perennial cover to erodible agricultural land can greatly reduce non-point source pollutants by avoiding and/or reducing fertilizer application and soil erosion. Alfalfa is the only perennial crop considered for evaluation of its water quality benefits. For maximum productivity, alfalfa requires well drained soils. Since its implementation in this study is to maximize its water quality benefits, all agricultural HRUs are deemed to be suitable for alfalfa. Information about the types, costs, and dates of field operations associated with alfalfa implementation such as mowing, conditioning, and baling are obtained from University of Illinois Extension (Doug Gucker, pers. comm.). In Table 4.2, all required field operations are listed and are used to prepare land management files for scenario simulations using the watershed models. The costs of field operations are provided in Table 4.3.

Table 3.2 Field operations for alfalfa

Year	Month	Day	Field Operation	Year	Month	Day	Field Operation
Year 1	4	1	Drill Alfalfa	Year 6	6	25	Mow/Condition
Year 1	8	15	Mow/Condition		6	28	Bale
	8	18	Bale	Year 6	7	30	Mow/Condition
Year 2	5	20	Mow/Condition		8	2	Bale
	5	23	Bale	Year 6	9	1	Mow/Condition
Year 2	6	25	Mow/Condition		9	4	Bale
	6	28	Bale	Year 7	5	20	Mow/Condition
Year 2	7	30	Mow/Condition		5	23	Bale
	8	2	Bale	Year 7	6	25	Mow/Condition
Year 2	9	1	Mow/Condition		6	28	Bale
	9	4	Bale	Year 7	7	30	Mow/Condition
Year 3	5	20	Mow/Condition		8	2	Bale
	5	23	Bale	Year 7	9	1	Mow/Condition
Year 3	6	25	Mow/Condition		9	4	Bale
	6	28	Bale	Year 8	5	20	Mow/Condition
Year 3	7	30	Mow/Condition		5	23	Bale
	8	2	Bale	Year 8	6	25	Mow/Condition
Year 3	9	1	Mow/Condition		6	28	Bale
	9	4	Bale	Year 8	7	30	Mow/Condition
Year 4	5	20	Mow/Condition		8	2	Bale
	5	23	Bale	Year 8	9	1	Mow/Condition
Year 4	6	25	Mow/Condition		9	4	Bale
	6	28	Bale	Year 9	5	20	Mow/Condition
Year 4	7	30	Mow/Condition		5	23	Bale
	8	2	Bale	Year 9	6	25	Mow/Condition
Year 4	9	1	Mow/Condition		6	28	Bale
	9	4	Bale	Year 9	7	30	Mow/Condition
Year 4	9	15	Termination Herbicide Application		8	2	Bale
	10	1	Drill Wheat	Year 9	9	1	Mow/Condition
Year 5	7	1	Harvest Wheat		9	4	Bale
	8	15	Plant Alfalfa	Year 9	9	15	Termination Herbicide Application
					10	1	Drill Wheat
				Year 10	7	1	Harvest Wheat
					8	15	Drill Alfalfa
				Year 11			Repeat cycles for year 6-10

Table 3.3 Cost of field operations

Field Operation	Total cost of operation ($/acre)
No-Till Drill	16.4
Rotary Mower	15
Mower/Conditioner	20.7
Small Sq. Baler	28.9
Sprayer	4.3
Combine	28.3
Herbicide Cost	20
Alfalfa Seed	100
Wheat Seed	46

Source: University of Illinois Extension

3.4 Constructed Wetlands

Constructed wetlands are artificial wetlands that are designed to emulate natural wetland functions including removal of pollutants from surface runoff and subsurface drainages. In wetlands, the water movement is slowed down, enabling settlement of sediments and thereby providing water purification functions. Nutrients such as nitrogen and phosphorus are taken up by wetland plants and microorganisms. Microbes convert organic nitrogen into inorganic forms that are essential for plant growth and also into nitrogen gas, releasing it to the atmosphere. Through sediment deposition and plant and microbial activities, constructed wetlands can provide water quality benefits by removing sediment and nutrients from agricultural runoff.
In simulating the water quality benefits using the watershed models, wetland model parameterization was done based on information obtained from the Franklin Farm demonstration project, which was a collaborative effort between The Nature Conservancy, University of Illinois, McLean County Soil and Water Conservation District, and Illinois Department of Natural Resources. The ultimate goal of the demonstration project was to address issues of nutrient loading from tile-drained agricultural systems in Illinois and determine the wetland to watershed ratio required for effective nutrient removal. Experimental results from the first 2 year of the study from 2007 to 2008 showed that a wetland to watershed ratio of 0.03, 0.06, and 0.09 removed a total of 18, 34, and 43 percent of nitrate, respectively, and 43, 55, and 56 percent of orthophosphate, respectively (Lemke and Kovacic, 2008).

Constructed wetlands are modeled as a water body in an HRU and its drainage area can be varied as a function of the HRU area. The wetland drainage area is set to 50 percent of the HRU area with a minimum area of at least 5 hectares for effective removal of NPS pollutants. The wetland surface area is calculated using a wetland to watershed area ratio of 0.05 (i.e., 2.5 percent of qualifying HRU area). Modifications were made to the wetland routines of the watershed model, routing tile flows and its constituents through the wetlands. Wetland outflows are released when the normal storage volume is exceeded. The maximum and normal storage volumes were determined using wetland depths of 1.0 and 1.25 meters, respectively. The transport of sediment in and out of a wetland is simulated using a simple mass balance routine in the watershed model. The removal of total suspended sediment is computed assuming that half of the sediment in the wetland remains suspended in impoundment after settling for one day. Nutrient transformations in the wetland are not simulated by the model. The wetland routine in the watershed model makes use of empirical methods for nutrient removal that are based on apparent settling velocity, accounting for nutrient processes in a wetland. The cost of constructed wetlands used is $2,700 per acre of wetland surface area (Bekele et al., 2011). An annual maintenance cost of $0.11 per acre of wetland treatment area is considered for spot mowing of 10 percent of the wetland buffer, which is based on a wetland buffer to wetland treatment area ratio of 0.035 (Christianson et al., 2013).

3.5 Drainage Water Management

The study watersheds are extensively tile-drained and the subsurface drainage has been used to enhance crop yields but at the expense of increased nutrient losses downstream. Controlling the drainage can provide water quality benefits. Drainage water management (DWM), also known as controlled drainage, is the practice of managing water table depths in such a way that nutrient transport from agricultural tile drains is reduced during the fallow season and plant water availability is maintained during the growing season. DWM practice can also help improve crop production and reduce oxidation of organic matter in the soil. It has additional benefits of decreasing wind erosion and providing seasonal habitat for wildlife (Natural Resources Conservation Service (NRCS), 2013).

DWM requires water control structures that are installed to raise or lower the effective height of the water table in the fields. Farmers need to pay attention to monitoring of these control structures for effective water management. New technologies such as satellite-based water control structures allow monitoring water tables from computers connected to the internet. Implementation of DWM can include new structures, main and lateral drains, or existing subsurface drainage outlets can be retrofitted. In this study, the second option is considered in computing the total cost of DWM implementation, which includes structure and its transport, design, and contractor fees. The cost of DWM implementation used in this study is $161.6 per acre of treatment area with an annual maintenance cost of $1.2 per acre of treatment area. For new drainage systems, the total cost of DWM ranges from a minimum of $59.4 to a maximum of $138.9 per acre (Christianson et al., 2013). New systems cost at least 15 percent less because of the fact that lateral and main drains can be designed to accommodate DWM by following contours, allowing management of large areas with a single water control structure (Skaggs et al., 2012).

Depending on the drainage system design, location, soil, and site conditions, the water quality benefits of DWM may vary. Studies in the Midwest showed that DWM reduced nitrogen loss to surface waters by 18 to more than 75 percent (Skaggs et al., 2012). Cooke and Verma (2012) conducted a paired field study to evaluate implementation of DWM for nitrate load reduction without affecting crop yields. Their study indicated that the annual nitrate reduction as a result of DWM ranged from 37 to 79 percent, with an average of 61 percent. However, no consistent relationship was found between yields and DWM as yields increased in some of the managed fields and decreased in others. Data reported in their study have been used to derive relationships between nitrate loads from managed and conventional free drainage systems. This relationship is incorporated to the watershed model at the HRU level to evaluate the water quality benefits of DWM.

3.6 Bioreactors

A bioreactor is an emerging conservation practice used to remove nitrate from agricultural tile drain discharge. It is an edge-of-field practice consisting of a buried pit filled with wood chips as a carbon source for denitrifying microorganisms so as to convert nitrate to atmospheric nitrogen gas. Enhanced denitrification occurs as a result of carbon source availability and maintenance of anaerobic conditions in the bioreactors. Retention time and the type of carbon source are important design considerations for effective removal of nitrate. The selection of carbon fill material should be based on cost, porosity, C: N ratio, and durability, and generally, woody media are the preferred choice (Robertson et al., 2005; Schipper et al., 2010). As a by-product of the denitrification process in bioreactors, nitrous oxide - a greenhouse gas - may be released to the atmosphere. However, some environmental conditions such as highly dissolved oxygen resulting from fluctuating flow rates and depths in bioreactors dictate the amount of nitrous oxide released. Investigation of nitrous oxide emission during denitrification in laboratory settings showed that its overall release was not greater than 1 percent (Christainson et al., 2013, Greenan et al., 2009).

The flow rate and hydraulic retention time in bioreactors control the performance efficiency in nitrate removal. In this study, a non-linear regression equation derived from a laboratory study by Greenan et al., (2009) was used to simulate the water quality benefits of bioreactors. According to this study, complete nitrate removal could be obtained up to a flow rate of 4.3 meters per day. For greater flow rates, the efficiency decreases as the hydraulic retention time decreases. The regression equation is integrated into the watershed model at the HRU level. Experimental field studies in Illinois showed that bioreactors were able to reduce annual nitrate load by up to 98 percent (Verma et al., 2010). The cost of bioreactors and associated maintenance vary from a maximum of $82 to $184/acre, and from $0.5 to $1.5 per acre of treatment area, respectively. While simulating bioreactor scenarios, the average establishment and maintenance costs of $133 and $1.0 per acre, respectively, were used.

3.7 Saturated Buffers

A saturated buffer is one of the newly emerging BMPs in which drainage water is diverted as shallow groundwater flow through a riparian buffer for nitrate removal. A saturated buffer system consists of a control structure for diversion of drainage water from the outlet to a lateral distribution line that runs parallel to the buffer. The redistribution of drainage water through the lateral lines into a riparian buffer enhances denitrification and plant uptake of water and nutrients. For effective removal of nitrate, drainage water should be directed as interflow close to the soil surface due to the high prevalence of organic matter near the surface.

The Agricultural Drainage Management Coalition (ADMC), in cooperation with USDA and several universities, has been conducting field studies at nine demonstration sites in Iowa, Illinois, Indiana, and Minnesota to evaluate saturated buffers for removal of nitrates and phosphorus from surface and subsurface drainage systems. In research conducted by the Leopold Center for Sustainable Agriculture in Iowa, a saturated buffer was implemented in Bear Creek, where 55 percent of the tile flow was diverted to be distributed to the riparian buffer. The saturated buffer system removed 100 percent of the nitrate, showing very promising results and the corresponding cost of implementation was $140.2 per acre of treatment area (Jaynes and Isenhart, 2014). In this study, the same performance efficiency is incorporated to the SWAT model at the HRU level to simulate the water quality benefit of a saturated buffer at the watershed scale and similar cost of establishment was used. In addition, an annual maintenance cost of $1.2 per acre of treatment area was assumed.

3.8 Filter Strips

A filter strip is a vegetative grass cover placed at the edge of the field to reduce erosion and pollutant loading from surface runoff. As runoff passes through a filter strip, its velocity is reduced, facilitating the trapping of sediments, nutrients, pesticides, and bacteria. The performance of filter strips in nitrate removal is minimal in tile-drained watersheds since they are designed to intercept surface runoffs. The performance efficiency of filter strips depends on their size and placement location. Generally, the drainage area to filter strip area ratio varies between 40 and 300 (Arnold et al., 2011). In this study, this ratio is fixed at 125 for all filter strip scenario simulations, and $500 per acre of filter strip is used as the implementation cost. The average areas of agricultural HRUs in Big Ditch and Big/Long Creek watersheds are 65.3 and 86.2 acres, respectively, and a ratio of 125 provides an average filter strip area of 0.52 acres for Big Ditch and 0.69 acres for Big/Long Creek HRUs.

3.9 Suitability of HRUs for BMP Implementation

A modeling criterion common to all BMPs is that only agricultural HRUs are considered for implementation scenario simulations. Some of the BMPs, however, require additional criteria such as specific topographic features, drainage, and soil characteristics. Thus, the suitability of HRUs varies by BMP type and only HRUs that satisfy the modeling criteria are included in the search for optimal placements of BMPs. For nutrient management, cover crops, perennial crops and filter strips, all agricultural HRUs with row crops (AGRR) are deemed to be suitable candidates, accounting for 91 and 82 percent of Big Ditch and Big/Long Creek watershed areas, respectively. Constructed wetlands can be implemented in HRUs with hydric soils categorized as somewhat to very poorly drained. For cost-effective removal of pollutants, the drainage area of constructed wetland needs to be at least five hectares. Based on these criteria, HRUs that are suitable for placement of wetlands in the study watersheds are identified, making up 71 and 70.2 percent of the Big Ditch and Big/Long Creek watershed areas, respectively. In selecting HRUs suitable for drainage water management, bioreactors, and saturated buffers, the existence of tile drainage in the HRUs and topographic features are taken into account. HRUs must have tile drainage to be selected for implementation of any of these three BMPs. For DWM and saturated buffer, there are additional slope requirements. Only HRUs with tile drains and slopes less than or equal to 1 percent are selected as suitable candidates for DWM, whereas bioreactors can be implemented in all tile-drained HRUs. For saturated buffers, HRUs are required to have slopes greater than 2 percent and the difference between slopes of HRU and the nearby stream has to be at least 0.5 percent. Based on these criteria, the percentage areas of Big Ditch watershed that are suitable for DWM, bioreactors, and saturated buffers are 34.2, 81.8, and 28.1, respectively. Similarly, 58.7, 79.5, and 8.5 percent of Big/Long Creek watershed area is suitable for implementation of DWM, bioreactors, and saturated buffers, respectively. Suitability maps showing potential implementation areas for saturated buffers are provided in Figures 3.1 and 3.2 for Big Ditch and Big/Long Creek watersheds, respectively. For all remaining BMPs, the suitability maps are given in Appendix A.

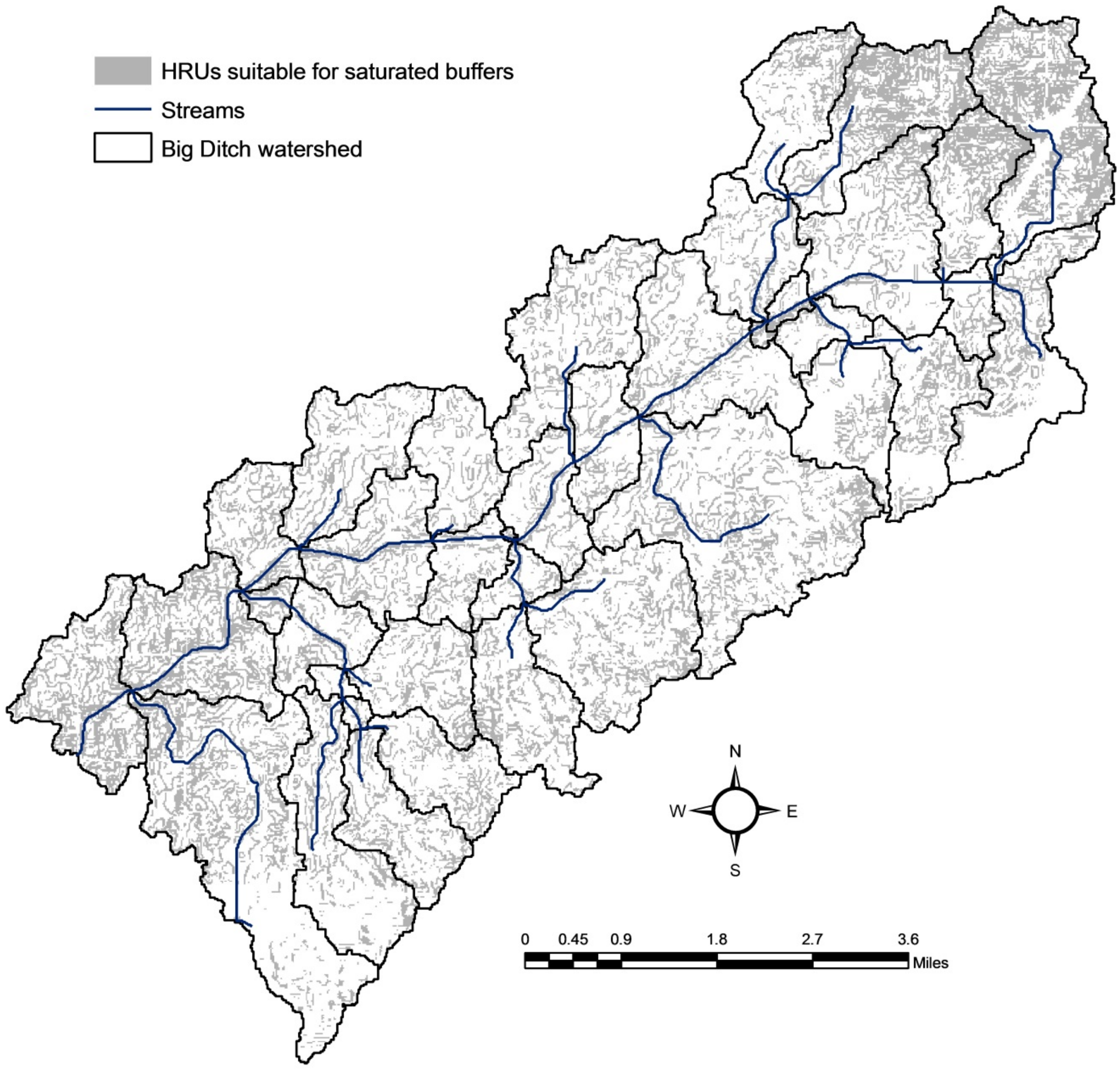


Figure 3.1 Big Ditch HRUs suitable for saturated buffers

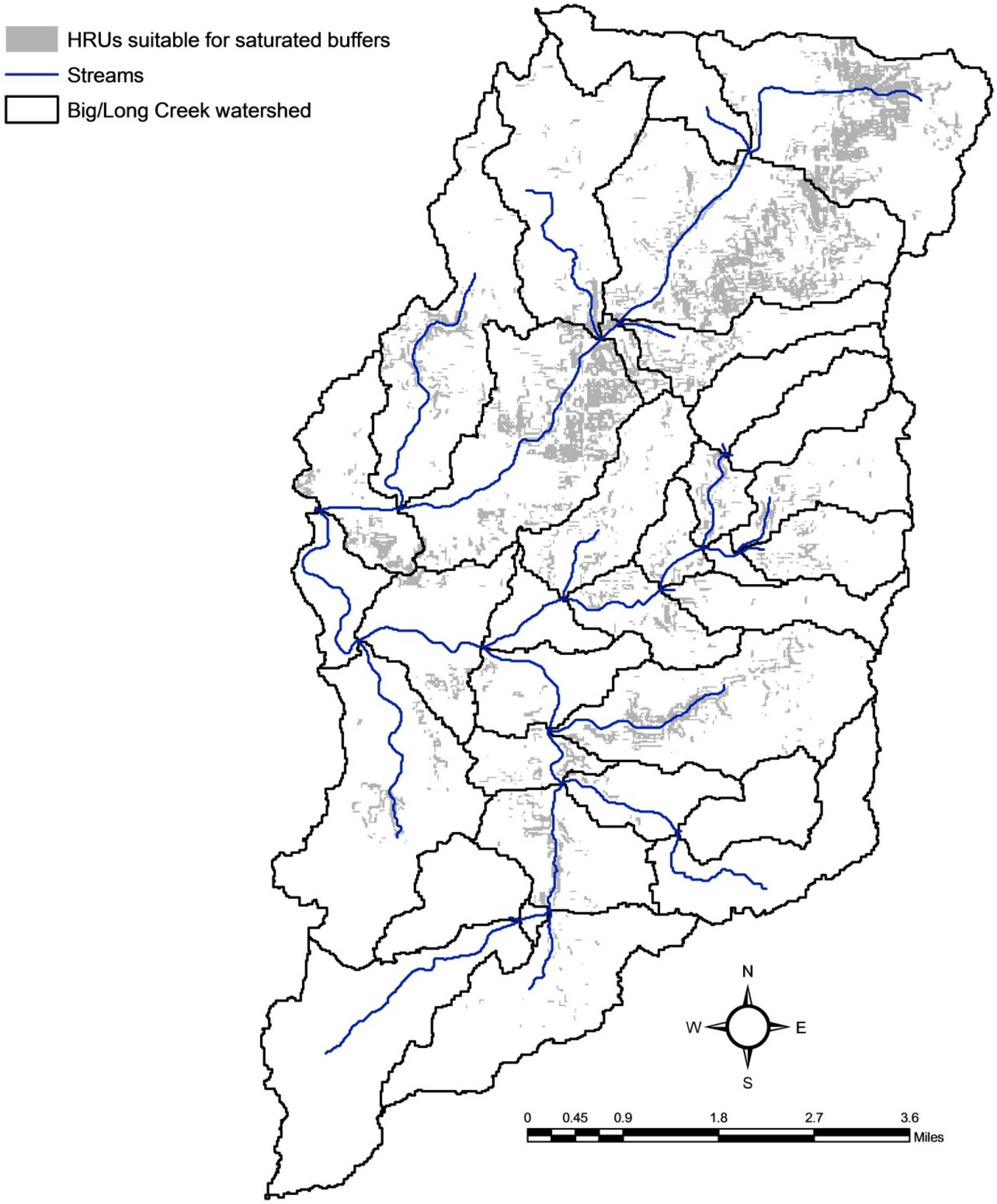


Figure 3.2 Big/Long Creek HRUs suitable for saturated buffers

4. Decision Support Models

The water quality benefits of implementing best management practices depend on their type and appropriate placements in the watershed. Experimental field and watershed-scale studies have been common approaches to evaluate the effectiveness of BMPs in pollutant removal. However, it is cost-prohibitive to experiment with a number of BMP scenarios at those scales. In contrast, watershed modeling is the most cost-effective way to evaluate the water quality impact of best management practices before implementation. However, it requires models that are capable of simulating watershed hydrology and water quality. Furthermore, accurate representation of BMPs in the watershed model is an important factor to simulate the resulting water quality benefits. As described in Section 2, SWAT is used to develop, calibrate, and validate hydrologic and water quality models for Big Ditch and Big/Long Creek watersheds. Modifications to the watershed model were made to simulate new and emerging BMPs such as drainage water management, bioreactors, and saturated buffers, as indicated in Section 3. All of the BMPs selected for scenario evaluations in this study incur the cost of implementation with the exception of nutrient management (i.e., fertilizer application rate and timing), which provides cost savings. When financial resources are limited, implementation of BMPs should focus on those critical areas that produce much of the pollutants. For cost-effective reduction of non-point source pollutants, it is crucial to locate optimal BMP placement areas. In this study, a decision support model (DSM) is developed using an integrated modeling approach that involves coupling simulation models with a multi-objective search algorithm. In the DSM, SWAT and the BMP cost functions are the simulation models, and the search algorithm is a multi-objective evolutionary algorithm known as Archived-Based Micro-Genetic algorithm 2 (AMGA2, Tiwari et al., 2011). The description of the SWAT model has already been provided in Section 2, and the multi-objective optimization algorithm and decision support model are explained in this section.

4.1 Multi-Objective Optimization

Optimization problems involving multiple competing objectives produce a set of optimal tradeoff solutions, instead of a single optimal solution. While conducting optimization, it is a common practice to aggregate multiple objectives into one using weighting factors or handling all objectives but one as constraints. Doing so, however, results in a partial evaluation of the solution space, thereby losing significant information about the tradeoff characteristics (Singh et al., 2004). In addition, assigning weights to objectives or handling objectives as constraints cannot be done without giving preference to some objectives over others, which is highly subjective.

With the emergence of multi-objective genetic algorithms (GA), the direct evaluation of problems involving multiple objectives has become possible. These algorithms employ population-based approaches and the concept of Pareto dominance and optimality to identify a set of tradeoff solutions also known as Pareto optimal solutions (Deb, 2001). GA's population-based approach allows it to avoid premature convergence to local optima, making it ideal for handling optimization problems involving highly nonlinear and multi-modal functions (Tiwari et al., 2011).

A multi-objective optimization problem can generally be expressed as (Zitzler and Thiele, 1999):

$$\boldsymbol{Minimize}\ \boldsymbol{f(\bar{d}) = [f_1(\bar{d}), f_2(\bar{d}), \ldots, f_n(\bar{d})]}$$

$$\boldsymbol{Subject\ to}\qquad \boldsymbol{\bar{d} = (d_1, d_2, \ldots, d_m) \in D}; \qquad \boldsymbol{\bar{f} = (f_1, f_2, \ldots, f_n) \in F}$$

where $\boldsymbol{s}$ is a vector-valued function that maps a set of $\boldsymbol{m}$ decision variables $\boldsymbol{\bar{d}}$ (e.g. BMP implementation in HRUs) to a set of $\boldsymbol{n}$ state variables or objectives $\boldsymbol{\bar{f}}$ (e.g., nitrate load, cost of BMP implementation)**;** $\boldsymbol{D}$ and $\boldsymbol{F}$ are the decision variable space and state variable space, respectively. Optimization problems involving maximization of objective functions can be formulated as minimization problems by negating the objective functions to be maximized. For two solutions or BMP implementation scenarios, $\boldsymbol{\bar{d}_A}$ and $\boldsymbol{\bar{d}_B}$, $\boldsymbol{\bar{d}_A}$ is said to dominate $\boldsymbol{\bar{d}_B}$:

$$\boldsymbol{iff}\ \forall \boldsymbol{i} \in \{\boldsymbol{1, 2, \ldots, n}\}\colon\ \boldsymbol{f(\bar{d}_A) \le f(\bar{d}_B)}; \qquad \exists \boldsymbol{i} \in \{\boldsymbol{1, 2, \ldots, n}\}\colon\ \boldsymbol{f(\bar{d}_A) < f(\bar{d}_B)}$$

Alternatively stated, solution $\boldsymbol{\bar{d}_A}$ dominates solution $\boldsymbol{\bar{d}_B}$, only if solution $\boldsymbol{\bar{d}_A}$ performs no worse than solution $\boldsymbol{\bar{d}_B}$ in all $\boldsymbol{n}$ objectives and is strictly better than solution $\boldsymbol{\bar{d}_B}$ in at least one of the objectives. Solution $\boldsymbol{\bar{d}_A}$ is said to be Pareto optimal only if it is not dominated by any solution in the solution space. A set of non-dominated solutions forms the Pareto optimal front.

4.2 AMGA2: Archived-Based Micro-Genetic Algorithm

AMGA2, like any genetic algorithm, is a heuristic search technique that is inspired by biological evolution such as natural selection, inheritance, crossover, and mutation to solve combinatorial optimization problems through iterative progress in a solution space. AMGA2, which is an improved version of the original AMGA (Tiwari et al., 2008), is considered as a steady-state GA because it makes use of a very small working population of potential solutions at any given iteration. It maintains a large external archive of good solutions found at all iterations. This external archive stores a large number of solutions and thereby enables generating more Pareto optimal solutions. In addition, it provides useful information regarding the search space. Genetic operators such as crossover and mutation are used to generate a small set of new potential solutions in every iteration. These new solutions are used to update the archive in every iteration and the process continues until a user-defined number of evaluations is exhausted. AMGA2 uses two fitness assignment mechanisms to discriminate between good and bad solutions in updating the archive and creating parent solutions. The primary fitness metric is the domination level or rank of a solution in the population obtained using the Pareto optimality and dominance criteria described earlier. The other fitness measure is the diversity metric. While updating the archive at any given iteration, the nearest neighbor search method is used to prune crowded solutions. For creating parent population, a numerical value of diversity is calculated using crowding distance metric (Deb et al., 2002). In AMGA2, a mating pool is formed using solutions from the parent population (i.e., primary parents) and the archive (i.e., auxiliary parents). An offspring population is then generated from the mating pool using crossover and mutation operations. At the early stages of optimization, the archive is largely populated with dominated solutions and thus only fewer non-dominated solutions are included in the parent population. In contrast, the majority of the solutions in the archive are non-dominated in the later stages of the search, and the parent population is designed to include less crowded or more diverse solutions at this stage. This strategy makes the algorithm efficient, reducing the number of function evaluations required for good approximation of the Pareto optimal front. Tiwari et al. (2011) provides a detailed description of the AMGA2 algorithm. The conceptual steps of AMGA2 can generally be outlined as follows:

Step 1: Generate initial population of potential solutions
Step 2: Evaluate initial population
Step 3: Update the archive using the initial population
Step 4: Create parent population from solutions in the archive
Step 5: Create mating pool from solutions in the parent population
Step 6: Create offspring population from solutions in the mating pool using crossover and mutation operations
Step 7: Evaluate offspring population of solutions
Step 8: Update archive with offspring population
Step 9: Repeat Steps 5 to 8 and terminate if user-defined number of evaluations is reached
Step 10: Extract desired number of optimal solutions from the archive

4.3 Coupled AMGA2-SWAT: The Decision Support Model

The Decision Support Model (DSM) is a coupled AMGA2 -SWAT model that is designed to explore the role of basin-wide implementation of best management practices for evaluating their water quality benefits. Figure 4.1 illustrates the solution framework of the DSM. The solution methodology dictates that the simulation model evaluates watershed responses resulting from implementation of BMPs and associated cost each time the search algorithm requires that information. The search algorithm is tasked with identifying optimal or near-optimal BMP implementation scenarios that are cost-effective to achieve prescribed goals (i.e., reduction of nonpoint source pollutants).

In developing the DSM, the watershed-scale implementation of BMPs is formulated as a multi-objective optimization problem. The objectives are reductions of nonpoint source pollutants (e.g., nitrate loads at the watershed outlet) at minimum possible costs of BMP (e.g., bioreactors) implementations in the watershed. This can be mathematically expressed as:

$$\boldsymbol{Minimize}\ \boldsymbol{f}(\overline{\boldsymbol{x}}) = [\boldsymbol{f_1}(\overline{\boldsymbol{x}}), \boldsymbol{f_2}(\overline{\boldsymbol{x}})]$$

$$\boldsymbol{Subject\ to} \quad \overline{\boldsymbol{x}} = (\boldsymbol{x_1}, \boldsymbol{x_2}, \dots, \boldsymbol{x_M}) \in \boldsymbol{X}; \qquad \overline{\boldsymbol{f}} = (\boldsymbol{f_1}, \boldsymbol{f_2}) \in \boldsymbol{F}$$

$$\boldsymbol{where} \quad \boldsymbol{f_1}(\overline{\boldsymbol{x}}) = \frac{\sum_{t=1}^{T} \boldsymbol{P_t}}{\boldsymbol{T}}; \quad \boldsymbol{f_2}(\overline{\boldsymbol{x}}) = \sum_{j=1}^{M} \boldsymbol{C_j}$$

$$\boldsymbol{Subject\ to}\ \boldsymbol{P_t} = \emptyset(\overline{\boldsymbol{x}});\ \boldsymbol{C_j} = \emptyset(\overline{\boldsymbol{x}});$$

where $\boldsymbol{F}(\bar{\boldsymbol{x}})$ is a vector-valued objective function to be minimized; $\bar{\boldsymbol{x}}$ represents a BMP implementation scenario in the decision solution space $\boldsymbol{X}$; $\boldsymbol{f_1}(\bar{\boldsymbol{x}})$ is the average annual pollutant load (e.g., nitrate) during the simulation period, $\boldsymbol{T}$; $\boldsymbol{f_2}(\bar{\boldsymbol{x}})$ is the total cost of BMP implementation in the watershed; $\boldsymbol{P_t}$ is the average annual pollutant load at the watershed outlet during year $\boldsymbol{t}$; $\boldsymbol{C_j}$ is the cost of BMP implementation in the $\boldsymbol{j}^{\text{th}}$ HRU; $\boldsymbol{M}$ is the total number of HRUs that are suitable for a particular BMP type; $\emptyset(:)$ is a generic function that represents all constraints that affect hydrologic and water quality simulations, cost and BMP placement criteria.

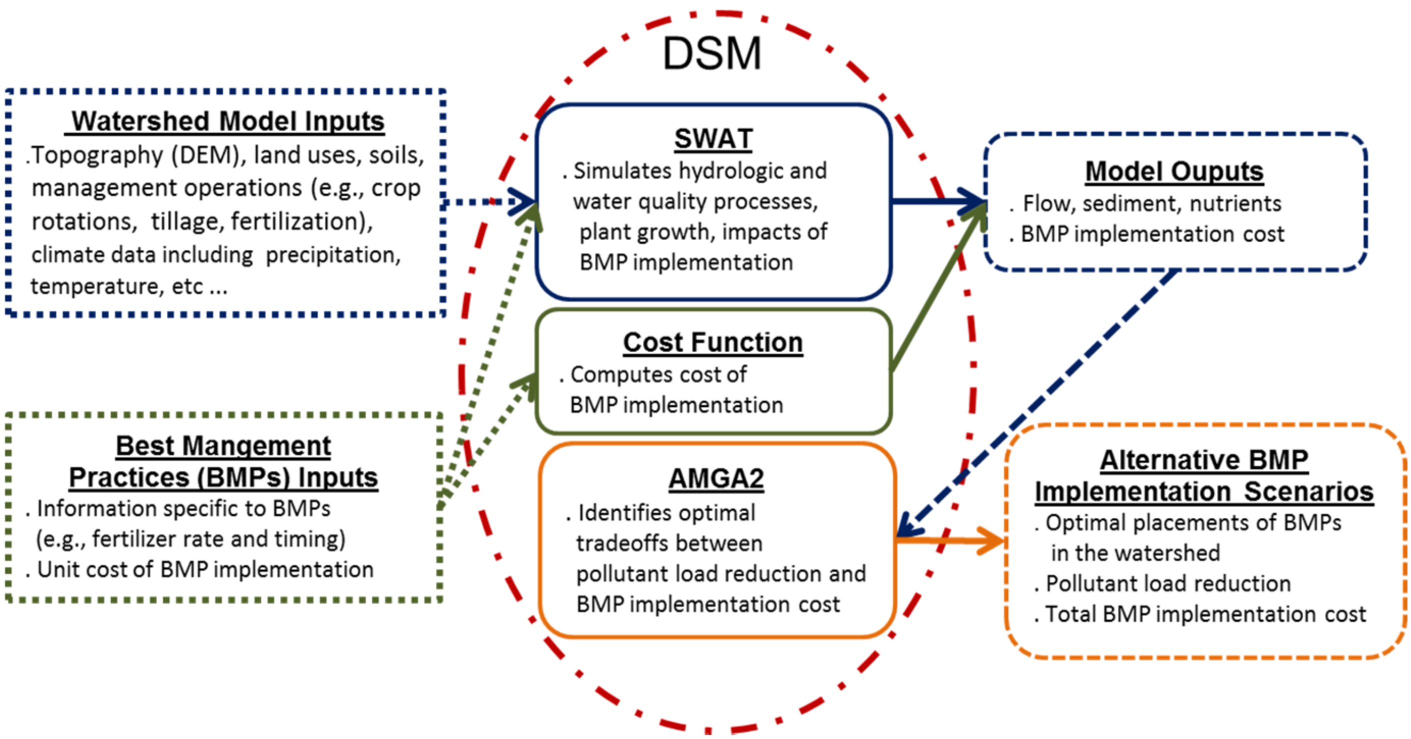


Figure 4.1 Solution framework of the Decision Support Model (DSM)

In the DSM, the placement of BMPs is simulated at hydrologic response unit level, which is assumed to be equivalent to a field. The variation between HRUs and a field is to be expected since HRUs are merely patches of land with a unique combination of land use, soil, and slope. As a result of a particular BMP implementation, watershed responses including flow, sediment, nitrate, and total phosphorus loads are evaluated at the watershed outlet. The DSM can be run for different decision horizons and, in this study, the decision horizon is set at 20 years to account for seasonal and annual variations, taking into account the life span of most of the BMPs selected for simulation. The decision horizon of 20 years is also used to assess the cost of BMP implementation, allowing a comparison between different BMPs. Cost assessments are based on information obtained from University of Illinois Extension (pers. Comm. with Doug Gucker) and published articles (Christianson et al., 2013). Total present value cost for each BMP type was assessed in year 2013 using a standard cost model (Klemperer, 1996) using a discount rate of 3.5 percent (i.e., 2014 rate for federal water projects):

$$TPVC_{BMP} = C_{est} + C_{main}$$

where $TPVC_{BMP}$ is the total present value cost of a BMP; C_{est} is the BMP establishment cost, and C_{main} is the BMP maintenance cost incurred in the decision period. To allow direct comparison of $TPVC_{BMP}$ between different BMPs, equal annual costs (EAC_{BMP}) are calculated over the decision period. The EAC_{BMP} is the equal annual payment that would be made at the end of each year in present value terms (Christianson et al., 2013). The conversion of $TPVC_{BMP}$ **into** EAC_{BMP} is done using a capital recovery factor (Gumaa et al., 1998) as follows:

$$EAC_{BMP} = CRF \times TPVC_{BMP}$$

$$and \qquad CRF = \frac{i(1+i)^n}{(1+i)^n - 1}$$

where CRF is the capital recovery factor; i is the annual real discount rate, and n is the number of years in the evaluation (i.e., the decision period).

Implementation of BMPs such as constructed wetlands and filter strips requires fertile land to be taken out of production. In addition, conversion of agricultural land from a corn-soybean rotation to perennial alfalfa could result in a significant reduction of revenues. To account for such revenue losses associated with implementation of these BMPs, the county average cash rents compiled by the National Agricultural Statistical Service of USDA were used. The 2013 average cash rents for Champaign and Macon Counties, which were \$255 and \$309 per acre of land, respectively, were used to calculate revenue losses resulting from BMP implementations in Big Ditch and Big/Long Creek watersheds. These revenue losses are included in assessing the total BMP implementation cost. In the case of perennial alfalfa implementation, the cash rent values were adjusted to reflect the reduced revenue losses as compared to constructed wetlands or filter strips. Assuming that the average cash rents for Champaign and Macon Counties (i.e., \$255 and \$309 per acre, respectively) are representative average crop revenues for a corn-soybean rotation in the study watersheds, the adjustments were made based on revenue ratio of alfalfa to average corn-soybean yields for the year 2013. The corn and soybean yield revenues for Central Illinois were obtained from crop costs publication of UIUC Department of Agricultural and Consumer Economics (ACE) and they were projected to be \$792/acre and \$646 for 2013, respectively (ACE, 2013). The 2013 alfalfa yield revenues were calculated to be \$477 per acre for Champaign and \$630 per acre for Macon Counties. The revenue computation was done using alfalfa yield estimates for Champaign County and Central Illinois, and alfalfa price for Illinois, which were obtained from Illinois Alfalfa Estimate and Illinois Agricultural Prices released by National Agricultural Statistics Service (NASS) in April of 2014. The revenue ratios calculated for Champaign and Macon counties were 0.66 and 0.88, respectively, resulting in adjusted average cash rent values of \$169.2 per acre for Big Ditch and \$270.7 per acre for Big/Long Creek watersheds.

For each BMP, the BMP scenarios that provide the maximum possible reduction of nonpoint source pollutants under a prescribed level of implementation make up the Pareto optimal or near-optimal front. The DSM outputs the water quality benefits of each BMP implementation scenario including sediment, nitrate, and total phosphorus reductions, their spatial allocation in the watershed, percentage of BMP treatment area, and corresponding implementation costs. In the DSM, the AMGA2 is invoked to optimize the placement of BMPs in the watershed, which is essentially a BMP implementation plan. This plan or scenario corresponds to an optimization string ($\boldsymbol{\beta}$) with a length equal to the total number of qualifying HRUs for a given BMP type. Figure 4.2 illustrates the optimization string for placement of a BMP, and the decision variables or genes are represented by BMP identifying numbers ($\boldsymbol{iD}$) if a BMP is to be implemented or a zero for no implementation. The length $\boldsymbol{l}$ of a string corresponds to the total number of genes or decision variables for a BMP (i.e., the total number of qualifying HRUs). The number of model evaluations, which dictates the number of iterations or generations in AMGA2, has been determined based on the number of decision variables, the desired size of optimal solutions, and several model testing simulations. The BMP scenarios in the $\boldsymbol{k^{th}}$ generation ($\boldsymbol{G^k}$) with a parent population of $\boldsymbol{np}$ can be expressed as:

$$\boldsymbol{G^k} = \begin{bmatrix} \boldsymbol{\beta_1^k} \\ \vdots \\ \boldsymbol{\beta_j^k} \\ \vdots \\ \boldsymbol{\beta_{np}^k} \end{bmatrix} = \begin{bmatrix} \boldsymbol{\beta_{1,1}^k} & \cdots & \boldsymbol{\beta_{1,l}^k} \\ \vdots & \ddots & \vdots \\ \boldsymbol{\beta_{i,j}^k} & \cdots & \boldsymbol{\beta_{i,l}^k} \\ \vdots & \ddots & \vdots \\ \boldsymbol{\beta_{np,1}^k} & \cdots & \boldsymbol{\beta_{np,l}^k} \end{bmatrix}$$

$$\forall \boldsymbol{i} \in \{\mathbf{1}, \mathbf{2}, \ldots, \boldsymbol{np}\} \; \boldsymbol{and} \; \forall \boldsymbol{j} \in \{\mathbf{1}, \mathbf{2}, \ldots, \boldsymbol{l}\}: \quad \boldsymbol{\beta_{i,j}^k} = \boldsymbol{iD}/\mathbf{0}$$

where $\boldsymbol{\beta_i^k}$ denotes the $\boldsymbol{i^{th}}$ BMP implementation scenario in$\boldsymbol{G^k}$; $\boldsymbol{\beta_{i,j}^k}$ is the $\boldsymbol{j^{th}}$ gene in $\boldsymbol{\beta_i^k}$ (i.e., a BMP placement decision at a given HRU); and $\boldsymbol{iD}$ is a BMP identifying number in the DSM.

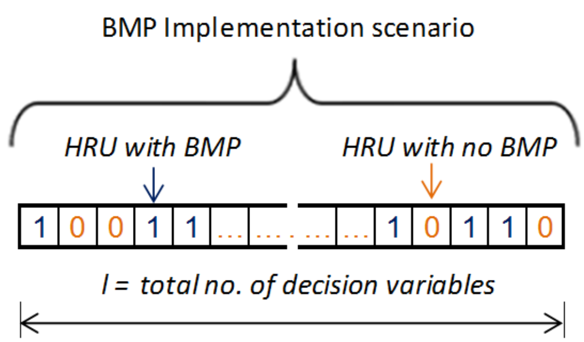


Figure 4.2 Optimization string representing BMP implementation scenario

In general, the operation of the DSM starts with the generation initial population of potential solutions. Each solution is a string of integers for BMP or no BMP implementation populating all qualifying HRUs. SWAT and a cost model are used to simulate the watershed responses and BMP implementation cost for each individual solution. Next, the optimization algorithm is invoked. In the first iteration, all solutions are kept in an archive for good solutions. A parent population of 20 good solutions is extracted from the archive, which is designated as the working population for the remainder of the iterations. Although the number of working population could be even smaller, it is set as 20 to take advantage of available computational resources through implementation of parallel coding. AMGA2 operators are then used to create a mating pool of good solutions from the parent and archive populations. Using crossover operations, solutions in the mating pool are used to create offspring solutions that are expected to perform better than their parents. Based on several model test runs, the maximum number of evaluations or iterations beyond which no improvement in the tradeoff solutions is considered as convergence criterion. This maximum number of evaluations is allowed to vary based on the length of the decision variables and, as a rule of thumb, it is calculated to be at least 100 times the total number of decision variables. To avoid convergence to local optima, mutation is then introduced to some of the solutions. Both crossover and mutation operation are also intended to widen the optimization search space. In all iterations, bad solutions in the archive are replaced by good ones, evolving to cost-effective BMP scenarios. In generating the Pareto optimal front, nitrate loading at the watershed outlet is primarily used as the water quality objective in all optimization model simulations.

4.4 Application of the Decision Support Model

A total of eight different types of BMPs were simulated and these include nutrient management (fertilizer application rate and timing), cover crops (cereal rye, annual rye grass, and crimson clover), perennial crop (alfalfa), constructed wetlands, drainage water management, bioreactors, saturated buffers, and filter strips. Some of these BMPs including nutrient management, bioreactors, drainage water management, and saturated buffers are considered strictly for nitrate load reduction, whereas the remaining BMPs are evaluated for phosphorus and sediment load reductions, in addition to nitrate. The water quality impact of each BMP is evaluated with respect to a baseline condition that is prepared using representative land use, land management, and climate conditions derived from historical data and several stakeholders' meetings in the study watersheds. Under the baseline scenario, all hydrologic response units (HRUs) or farms with agricultural row crops have a corn-soybean rotation with land management conditions that are common in the study watersheds. Historical transect survey data of tillage practices were obtained for Champaign and Macon counties and were used to determine the average percent coverage of conventional, reduced, mulch, and no-till systems in Big Ditch and Big/Long Creek watersheds, respectively. Information on dates of field operation, types of fertilizers used, application rate, and timing were collected from producers and representatives of fertilizer dealerships and Soil and Water Conservation Districts (SWCDs) during stakeholder meetings. Table 4.1 lists a sequence of field operations that were simulated in the baseline scenario for Big Ditch and Big/Long Creek watersheds. Based on HRU's land use and tillage type, corresponding field operations are simulated.

Table 4.1 Land management operations simulated in the baseline scenario

Date of operation	List of land management operations	Operations by tillage types			
		Conventional	Reduced	Mulch	No-till
Year Zero					
October 21	Fertilizer application: DAP (18 lb N/acre; 20.2 lb P/acre)	x	x	x	x
October 22	Tillage implement: Tandem disc	x			
October 22	Tillage implement: Disc ripper		x		
October 22	Tillage implement: Vertical tillage			x	
October 23	Tillage implement: Disc ripper	x			
November 02	Fertilizer application: Anhydrous Ammonia (120 N lb/acre - 60% of total)	x	x	x	
Year One					
February 02	Fertilizer application: UAN-28 (40 lb N/acre - 20% of total)				x
April 21	Fertilizer application: UAN-28 (40 lb N/acre - 20% of total)	x	x	x	
April 21	Fertilizer application: Anhydrous Ammonia (120 N lb/acre - 60% of total)				x
May 05	Tillage implement: Field Cultivator	x		x	
May 05	Tillage implement: Vertical tillage		x		
May 07	Plant corn	x	x	x	x
June 01	Fertilizer application: UAN-28 (40 lb N/acre - 20% of total)	x	x	x	x
October 07	Harvest corn	x	x	x	x
October 21	Fertilizer application: DAP (18 lb N/acre; 20.2 lb P/acre)	x	x	x	x
October 22	Tillage implement: Disc ripper	x			
October 22	Tillage implement: Tandem disc	x	x		
Year Two					
May 20	Tillage implement: Soil finisher	x		x	
May 20	Tillage implement: Field Cultivator		x		
May 22	Tillage implement: No-till drill				x
May 22	Plant soybean	x	x	x	x
September 14	Harvest soybean	x	x	x	x

To illustrate the application of the DSM, the multi-objective optimization of nutrient management (NM) scenarios (i.e., fertilizer rate and timing) for the Big Ditch watershed are used as an example. The Big Ditch watershed has 450 HRUs, out of which 368 HRUs are agricultural row crops and all are considered for NM implementation. The optimization decision variables are fertilizer application rate, its distribution between pre-planting and after planting applications. The fall or winter application is calculated as a residual percentage based on the tillage used in the HRU. The optimization was carried out with a parent population of 20 solutions, which are allowed to evolve through generations or iterations. Unlike other population-based genetic algorithms, the algorithm makes use of a smaller parent population. In this particular case, the parent population is set at 20 to make use of the available computer resources through implementation of parallelized computer code. All model evaluations have been completed using a Dell Workstation with Intel Xeon E5-2670 CPU, 32 GB RAM, 20 cores and 2.5GHz speed. The implementation of parallel code made it possible to complete 1500 model executions per hour on average, resulting in an overall reduction of the computational demand by 95 percent. The DSM has taken approximately 2 hours to identify the Pareto optimal front for NM implementation in the Big Ditch watershed. The Pareto optimal front is approximated by the optimal or near-optimal tradeoff solutions between nitrate load reductions and cost (see Figure 4.3). This figure also shows how the parent solutions evolved to the Pareto optimal front through a number of model evaluations, maximizing the cost-effectiveness of NM scenarios. In this particular case, the negative cost of implementation indicated the cost savings as compared to the baseline condition. In Figure 4.4, the decision variables (i.e., fertilizer application rate, percentages of pre-planting and after planting fertilizer applications) are shown for NM scenarios corresponding to the Pareto optimal front and all simulations performed. The optimal fertilizer application rate was found to be 155 lb N/acre, which is the lower bound of the MRTN (i.e., maximum return to N) value used during the optimization process. For this optimal application rate, a maximum nitrate reduction of 5.79 kg N/ha/year is obtained when the percentage of spring pre-plant and after planting application is 50 and 45.7, respectively (see Table 4.2). The optimal tradeoff solutions also include other combinations of split applications that resulted in nitrate reduction greater than 5 kg N/ha/year. However, in almost all cases, minimal fall/winter application is favored for higher nitrate load reduction. The DSM application results indicate that the AMGA2 algorithm is effective in identifying the Pareto optimal front, providing tradeoffs between water quality benefits and cost of implementation. The DSM was used to identify cost-effective NPS pollutant reduction strategies for Big Ditch and Big/Long Creek watersheds through evaluation of a suite of BMPs.

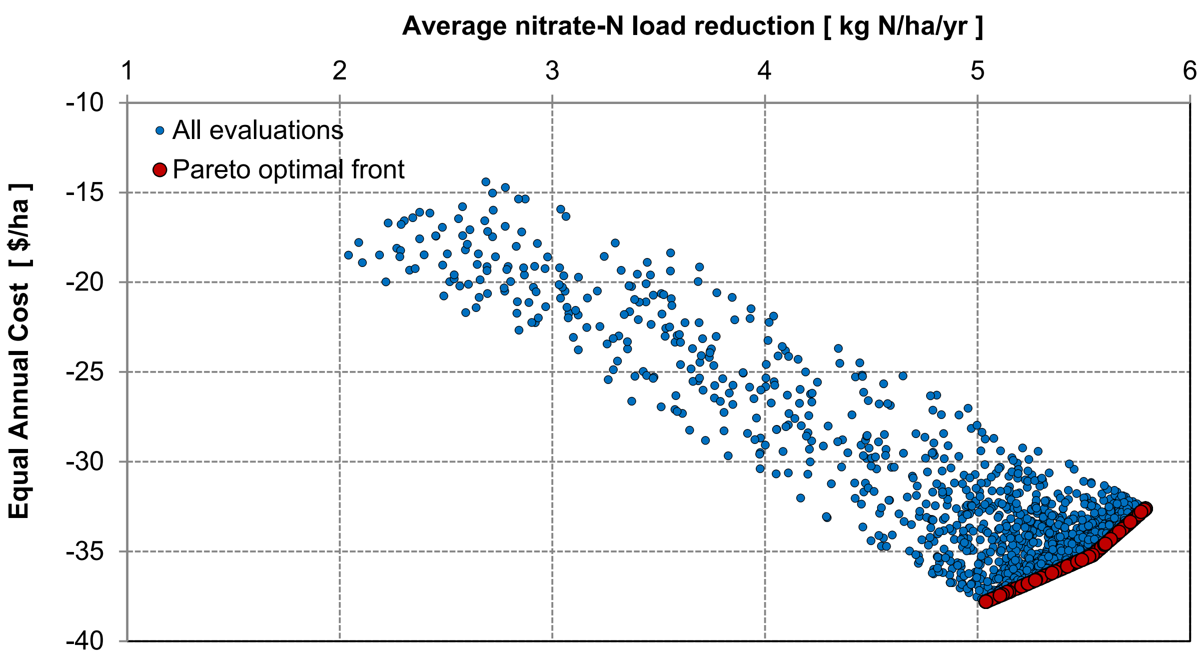


Figure 4.3 NM scenarios evolving to Pareto optimal front

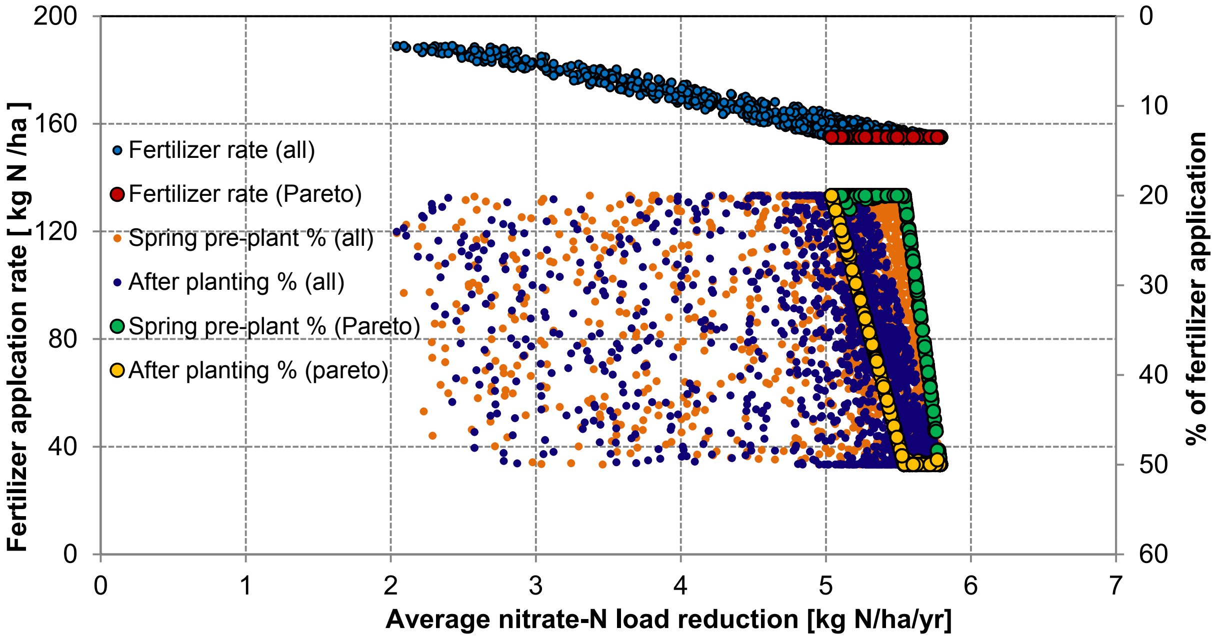


Figure 4.4 NM decision variables for Pareto optimal front and all model evaluations

Nutrient Management Scenarios

Two trials were tested for simulating the implementation of nutrient management scenarios. First, the optimization of fertilizer rate and timing were considered as decision variables, assuming that all producers will sign up for implementation of nutrient management. In this first trial, the nitrogen fertilizer application rate and timing were allowed to vary from the baseline scenario in all of the HRUs with agricultural row crops. This scenario produced an optimal fertilizer application rate of 155 lb N/acre, which is equal to the lower bound of the MRTN rate used in the optimization model evaluations (see Tables 4.2 and 4.3). The best tradeoff solutions for Big Ditch and Big/Long Creek watersheds provide an optimal fall application that is equal to 30 percent of the annual N application or 46.5 lb N/acre. The after planting application has increased by more than two folds from the baseline. For both watersheds, maximum nitrate load reductions are obtained when distribution between pre-planting and after planting fertilizer application is at approximately the 50 percent level (i.e., no or minimal fall application) but with a small decrease in cost savings attributed to greater fertilizer costs in the spring.

In the second trial, in addition to the fertilizer rate and timing, its location of application was considered as a decision variable during optimization, assuming that not all producers may sign up for implementation of a nutrient management plan. The resulting optimal tradeoff solutions obtained for both watersheds show that all HRUs with row crops should adopt the nutrient management plan (see Figures 4.5) and the corresponding fertilizer rates and application percentages are similar to that of the first trial. The best tradeoff solutions provide average nitrate reductions of 15.8 and 14 percent for Big Ditch and Big/Long Creek watersheds, respectively, and the average annual cost savings per nitrate reduction is at least \$6.42 /kg N/ha or \$7.2/lb N/acre.

Table 4.2 Optimal tradeoff solutions for NM in Big Ditch watershed

NM Scenario	Fertilizer application: Rate [lb N/acre]	Timing [%] Fall/Winter	Spring pre-plant	After planting	Load Reduction Nitrate [kg N/ha/yr]	[lb N/acre/yr]	[%]	Equal Annual Cost (EAC) [$/yr]
Baseline	**220**	**60.0**	**20.0**	**20.0**				
1	155	51.8	20.0	28.2	5.039	4.496	14.4	-367,410
2	155	33.1	20.0	47.0	5.118	4.566	14.6	-363,552
3	155	46.8	20.0	33.2	5.288	4.718	15.1	-355,050
4	155	30.7	20.0	49.3	5.457	4.869	15.6	-346,457
5	155	34.5	20.0	45.5	5.518	4.923	15.8	-343,373
6	155	44.9	30.6	24.5	5.629	5.022	16.1	-333,339
7	155	4.3	50.0	45.7	5.790	5.166	16.6	-317,057

Table 4.3 Optimal tradeoff solutions for NM in Big/Long Creek watershed

NM Scenario	Fertilizer application: Rate [lb N/acre]	Timing [%] Fall/Winter	Spring pre-plant	After planting	Load Reduction Nitrate [kg N/ha/yr]	[lb N/acre/yr]	[%]	Equal Annual Cost (EAC) [$/yr]
Baseline	**220**	**60.0**	**20.0**	**20.0**				
1	155	51.1	20.0	28.9	4.308	3.843	12.6	-373,998
2	155	35.4	20.0	44.6	4.404	3.930	12.8	-369,292
3	155	47.4	20.0	32.6	4.561	4.069	13.3	-361,669
4	155	42.3	20.0	37.7	4.700	4.194	13.7	-354,898
5	155	30.5	20.0	49.5	4.798	4.281	14.0	-350,106
6	155	11.6	38.4	50.0	4.996	4.457	14.6	-334,185
7	155	0.0	50.0	50.0	5.110	4.559	14.9	-324,686

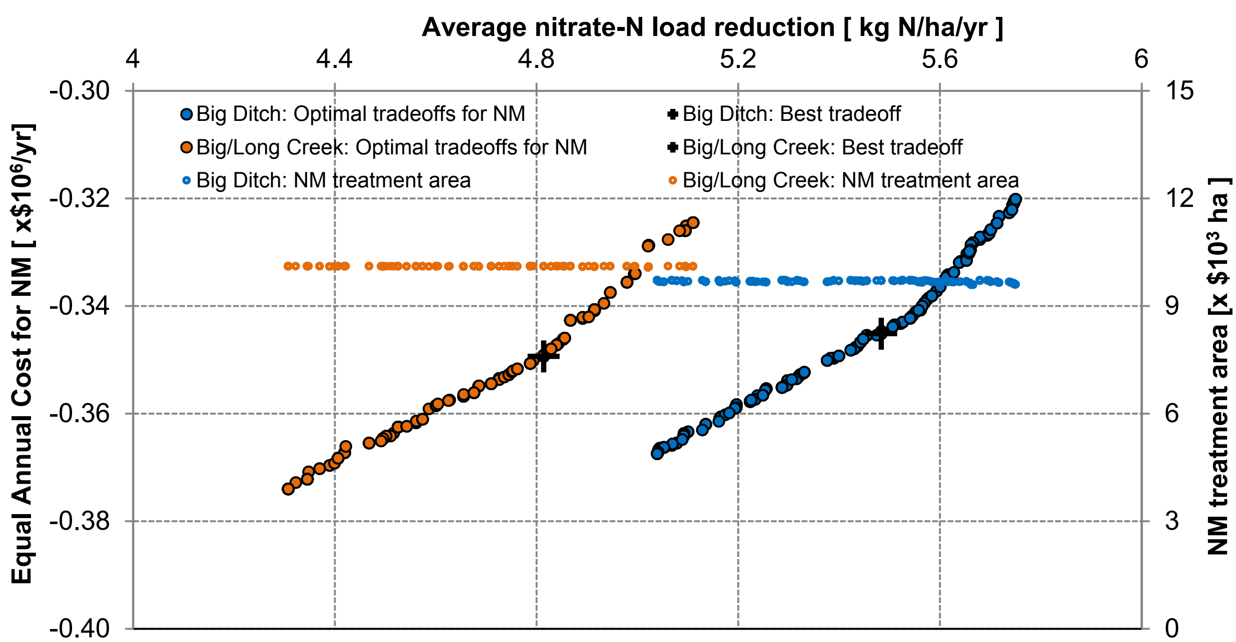


Figure 4.5 NM: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds

Cover Crop Scenarios

Cereal rye (CR), annual rye grass (RG), and crimson clover (CC) are the three cover crops simulated as BMPs for non-point source pollution reduction of nitrate, sediment, and total phosphorus. In the DSM, all HRUs that are designated as agricultural row crops in the study watersheds qualify for cover crop implementation, and a given implementation scenario constitutes the placement of the cover crops in any number of these HRUs. Therefore, a decision vector in the optimization process includes all qualifying HRUs. The implementation cost of cover crops includes seeds and planting. Additional herbicide and spraying costs are not included for cover crop implementation as all farmers apply herbicides in the spring. The DSM was executed to identify Pareto optimal fronts for each cover crop. Optimal tradeoffs and corresponding treatment areas for cereal rye, annual ryegrass, and crimson clover in Big Ditch and Big/Long Creek watersheds are presented in Figures 4.6 and 4.7, respectively. Each solution in the Pareto optimal front represents the best possible implementation scenario under that level of cover crop treatment area. While analyzing all DSM results, the best tradeoff solution is defined as the solution with a minimum distance from the maximum pollutant load reduction and minimum cost of BMP implementation, providing the cost-effective scenario. For the Big Ditch watershed, the best tradeoff scenarios resulted in nitrate, total phosphorus, and sediment load reductions of at least 10.6, 7.8, and 8.4 percent, respectively, requiring an average treatment area equal to 45 percent of the watershed. The percentage load reductions estimated for the Big/Long Creek watershed are 7.8, 9.7, and 4.4 percent for nitrate, phosphorus, and sediment, respectively, with an average treatment area of 39.4 percent for the best tradeoff solutions. The cost per nitrate reduction is the least for cereal rye and the highest for crimson clover.

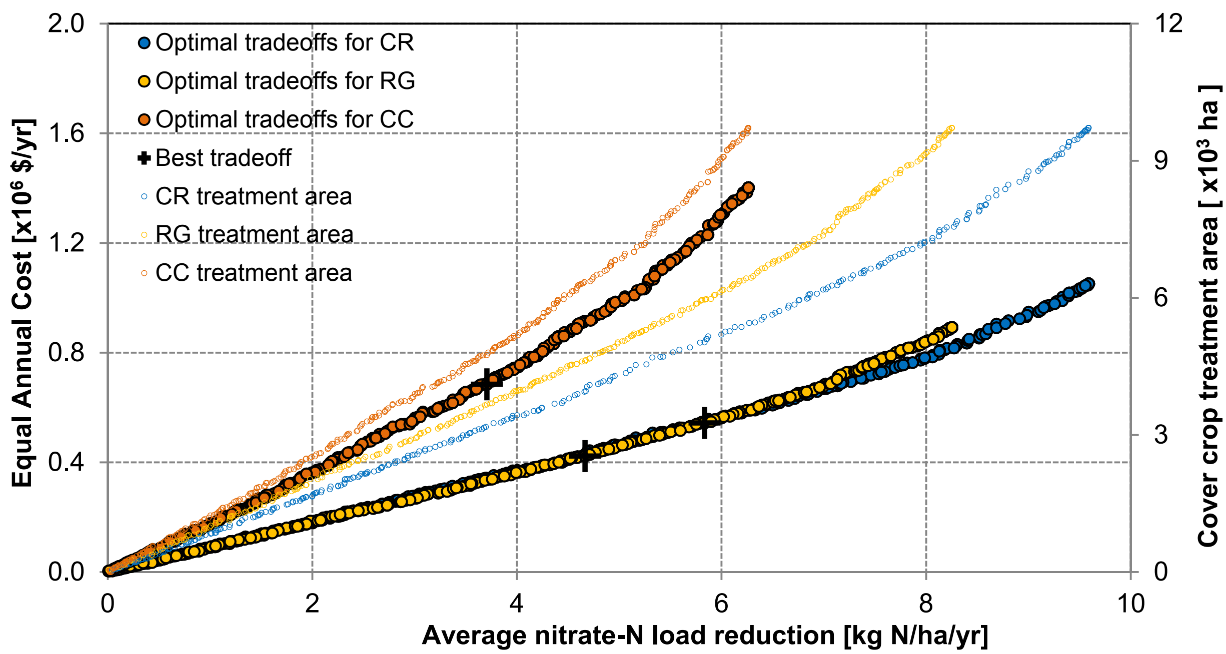


Figure 4.6 Cover crops: optimal tradeoffs and treatment areas for Big Ditch watershed

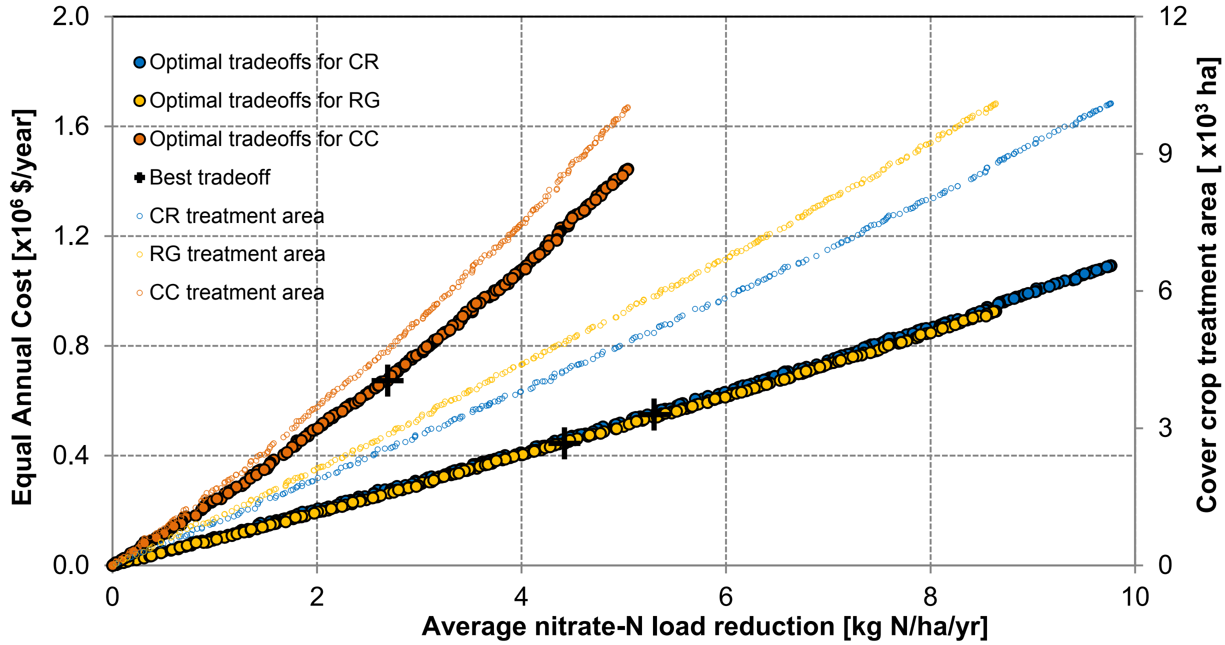


Figure 4.7 Cover crops: optimal tradeoffs for Big/Long Creek watershed

The optimal placement of cereal rye in the Big Ditch watershed is shown in Figure 4.8, covering 47.3 percent of the watershed area. In order to identify priority areas for BMP implementation, each HRU with BMP allocation (i.e., HRU with cereal rye in this particular case) has been independently evaluated using the DSM for its impact on nitrate load reduction at the watershed outlet. The optimality of HRU for BMP implementation depends on its characteristics such as area, land use, soil, land management condition, slope, proximity to a stream, and the watershed outlet. The simulated nitrate load reductions are then normalized to values between 1 and 3, which are later used as priority ranks. HRUs with a priority ranking ranging between 1 and 1.5 are designated as priority area 1 for BMP implementation and are color-coded as red in the BMP placement maps. Those HRUs with priority rankings between 1.5 and 2.5, and greater than 2.5 are categorized as priority area 2 (blue) and 3 (green), respectively. For the Big Ditch watershed, HRUs belonging to the priority area 1 for cereal rye cover 4.5 percent of the watershed area (see Figure 4.8). All BMP allocation maps showing cover crops and their corresponding priority placement areas are provided in Appendices B and C for both watersheds. In addition, all maps show HRUs with no BMP, which represent agricultural HRUs suitable for that particular BMP but are not part of the optimal solution.

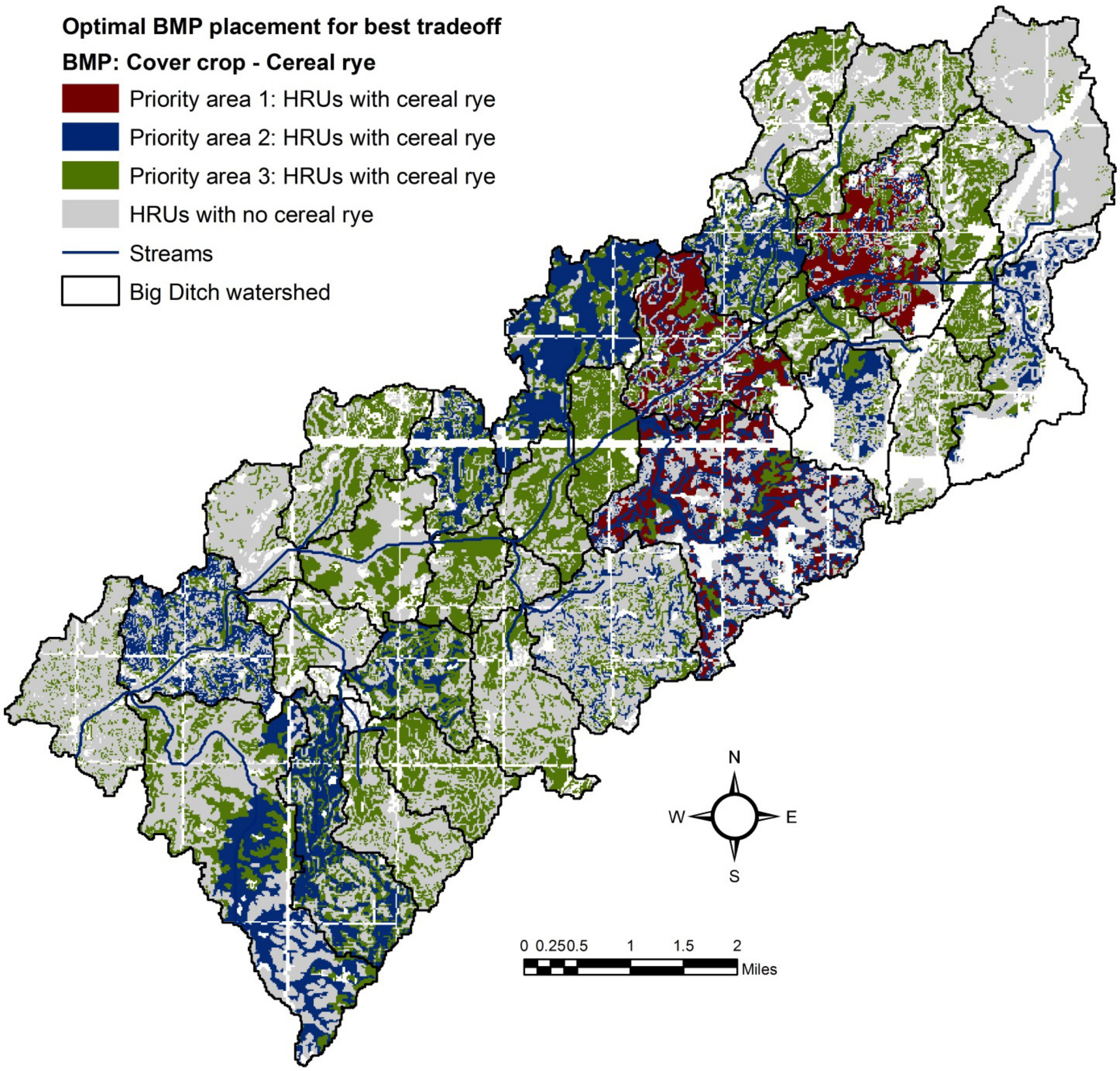


Figure 4.8 Optimal placement of cereal rye in Big Ditch watershed

Perennial Crop Scenarios

Similar to cover crops, all agricultural HRUs are suitable for the implementation of a perennial crop. The perennial crop simulated in this study is alfalfa and its implementation has water quality benefits by reducing soil erosion and nutrient loss. The DSM is executed to identify optimal placement of alfalfa (AA) for a cost-effective reduction of nitrate, sediment, and phosphorus loads. Implementation costs include seeds, planting, mowing and bailing, herbicides, and spraying costs. The costs of seeds and herbicides are calculated for each year of planting period and the cost of mowing and bailing are based on three cuttings per year. In addition, revenue losses as a result of converting corn-soybean rotation to alfalfa are included in the computation of the total cost of implementation.

The optimal tradeoff solutions obtained for alfalfa placements in Big Ditch and Big/Long Creek watersheds are presented in Figure 4.9. The slopes of the optimal tradeoff curves imply that implementation of alfalfa provides more nitrate load reduction per dollar for Big Ditch watershed ($53/kg N/ha/year), as compared to the Big/Long Creek watershed ($87/kg N/ha/year). This variation is largely attributed to high cash rents for Macon County, which is about 60 percent more. The best tradeoff solution for the Big Ditch watershed requires implementation of alfalfa in 45.4 percent of the watershed area, and it would result in nitrate, phosphorus, and sediment load reductions of 46.8, 20.6, and 15.4 percent, respectively. Similarly, for the Big/Long Creek watershed, the cost-effective implementation scenario requires an alfalfa treatment area of 38.9 percent of the watershed and provides estimated reductions of 37.4, 11.3, and 5.6 percent in nitrate, phosphorus, and sediment loads at the watershed outlet, respectively. Figure 4.10 shows the optimal placements of alfalfa in the Big/Long Creek watershed for the best tradeoff solution, and the most critical treatment area designated as priority area 1 covers 11.3 percent of the watershed area. For both watersheds, the cost-effective alfalfa placements and corresponding pollutant reductions are provided in Appendices B and C.

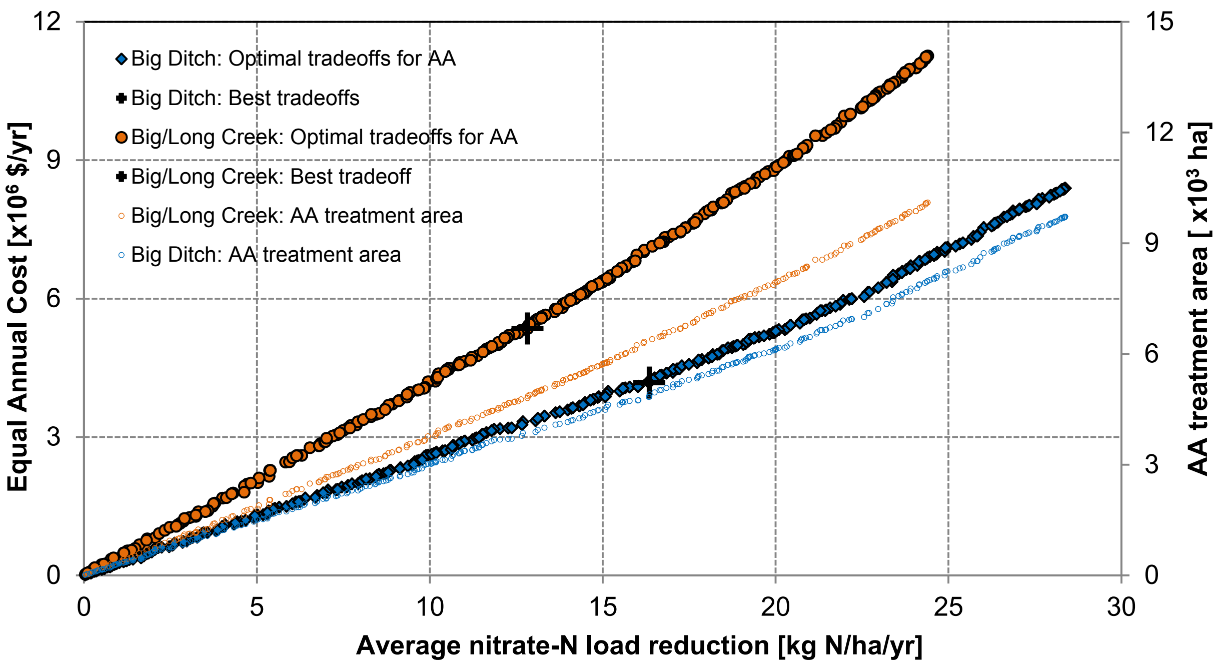


Figure 4.9 Perennial-Alfalfa: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds

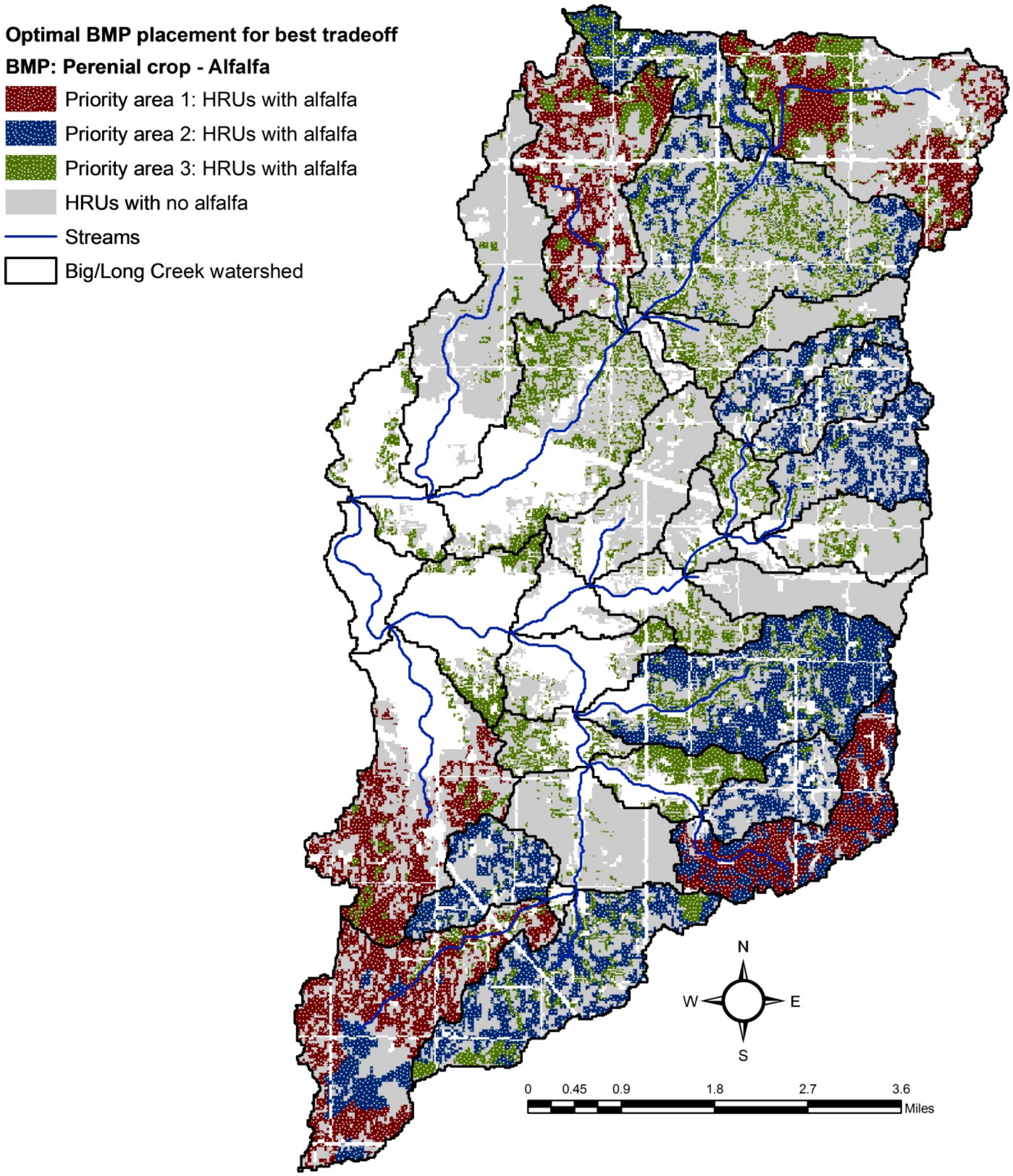


Figure 4.10 Optimal placement of alfalfa in Big/Long Creek watershed

Constructed Wetland Scenarios

Not all agricultural HRUs in the study watersheds are suitable for implementation of constructed wetlands. In the Big Ditch and Big/Long Creek watersheds, qualifying HRUs make up 71 and 70.2 percent of their respective watershed areas. The total number of suitable HRUs for implementation of constructed wetlands, which determines the length of the decision vector in the optimization process, is 225 for Big Ditch and 160 for the Big/Long Creek watershed. The treatment area, which is the same as the drainage area of the wetland, is modeled as 50 percent of the HRU areas in both watersheds. Consequently, the maximum treatment area is 35.5 percent for Big Ditch and 35.1 percent for the Big/Long Creek watershed. The DSM model identified the Pareto optimal solutions for both watersheds, providing tradeoffs between implementation cost and pollutant reduction (see Figure 4.11). Also shown in the figure is a tradeoff between nitrate load reduction and size of constructed wetlands. As indicated earlier, the best tradeoff corresponds to the most cost-effective implementation scenario, but all other solutions also provide optimal placement of constructed wetlands at their level of watershed treatment area. For Big Ditch watershed, the cost-effective implementation scenario provides nitrate, phosphorus, and sediment load reductions of 17.1, 7.2, and 5.0 percent, respectively, requiring 99 hectares of constructed wetlands with a treatment area equal to 18.6 percent of the watershed. The best tradeoff for the Big/Long Creek watershed would require implementation of 94.2 hectares of constructed wetlands, providing reductions of nitrate, phosphorus, and sediment loads by 14.3, 3.3, and 1.4 percent, respectively. The corresponding wetland treatment area would cover 15.3 percent of the total watershed area. For both watersheds, DSM results indicate that constructed wetlands perform better in nitrate load reduction and the average reduction was estimated to be 5.4 kg N/ha/year with an average cost efficiency of $10.9/kg N/ha/year. Based on required areas for constructed wetlands, revenue losses were estimated using cash rents for Champaign and Macon counties where the study watersheds are located. This additional cost is included in the total BMP cost of implementation. The optimal placement of constructed wetlands in the Big Ditch watershed is illustrated in Figure 4.12, and 2.2 percent of the watershed designated as priority area 1 is identified as the most critical area for implementation of constructed wetlands. It must be noted that only 50 percent of HRU areas shown in the figure are considered as actual treatment areas for implementation of constructed wetlands. The optimal placements of constructed wetlands and the resulting NPS reductions for both watersheds are included in Appendices B and C.

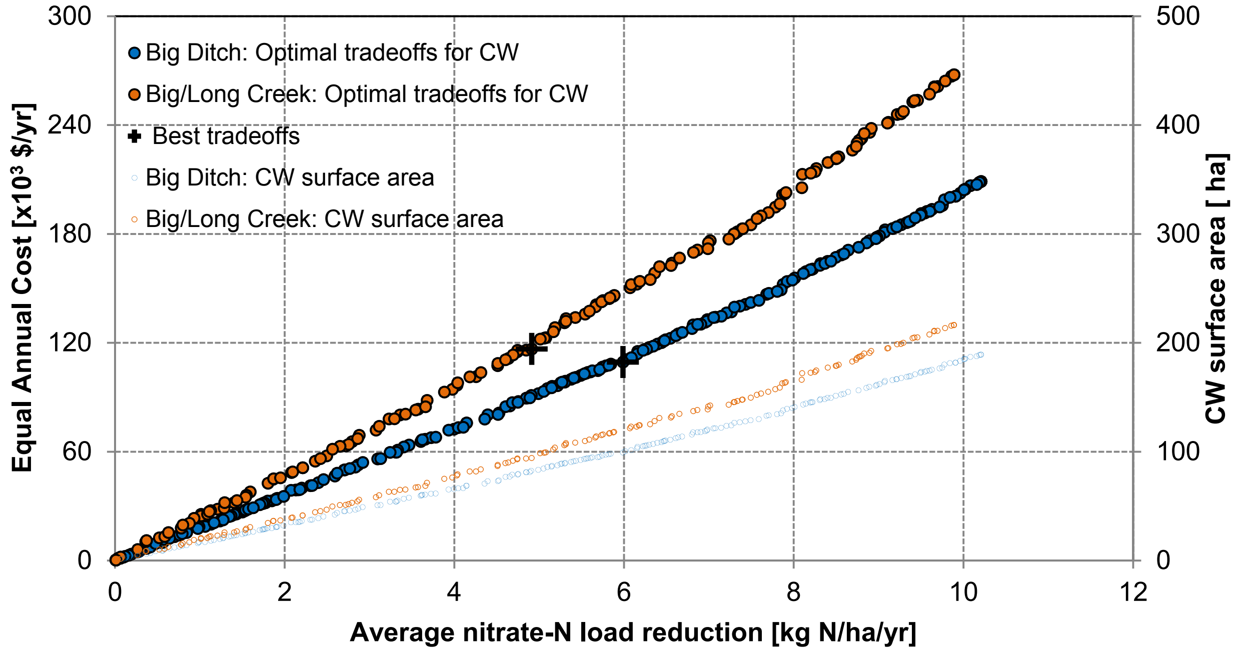


Figure 4.11 Constructed wetlands: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds

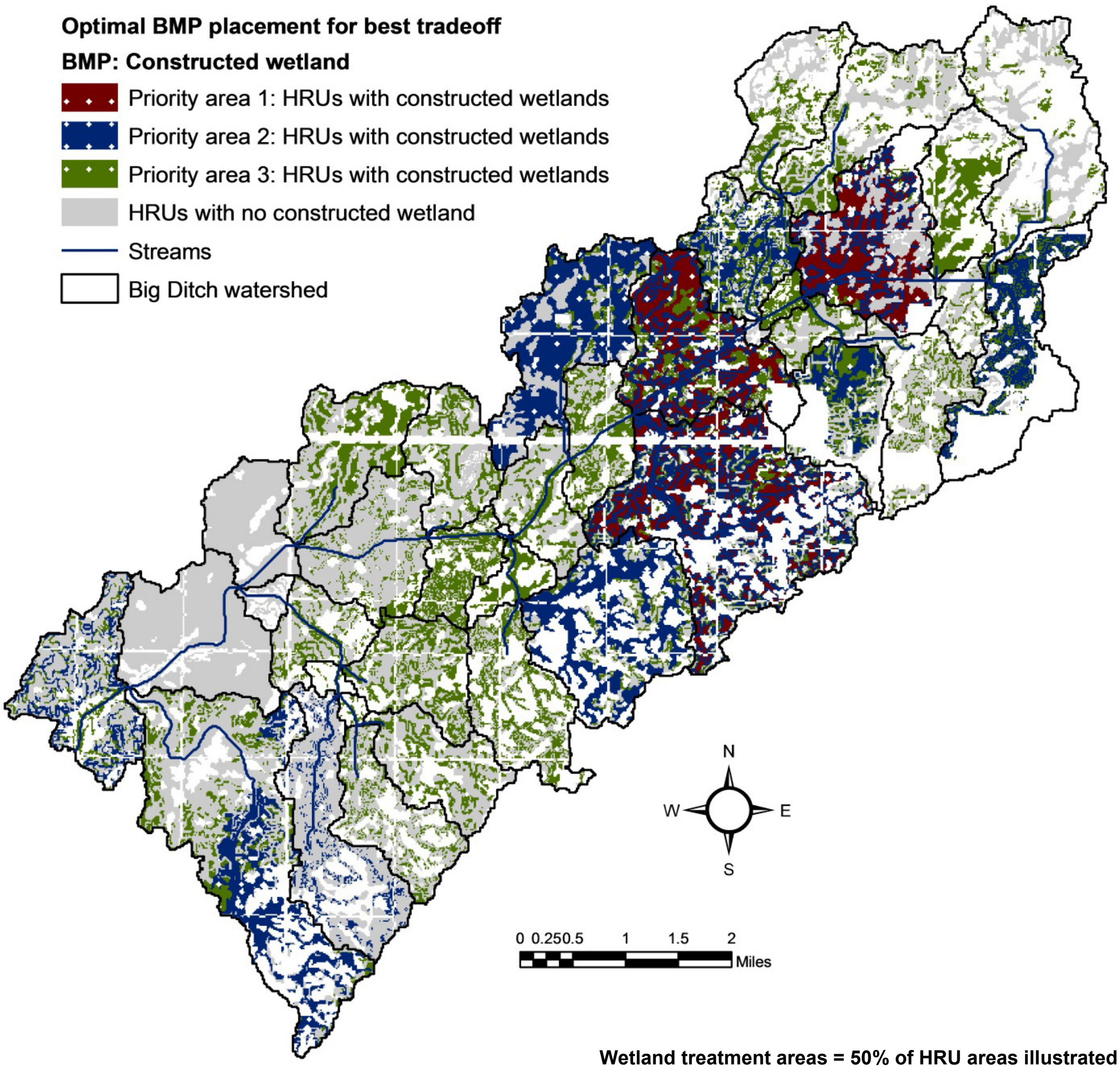


Figure 4.12 Optimal placement of constructed wetlands in Big Ditch watershed

Drainage Water Management Scenarios

The implementation of drainage water management (DWM), which is the practice of managing water table depths to reduce nitrate transport from subsurface drainage, is considered in those HRUs that have tile drains and meet slope requirements of less than 1 percent. Based on the selection criteria, a total of 116 HRUs in Big Ditch and 126 in Big/long Creek watersheds are suitable for implementation of DWM. Because of uncertainties associated with lumped physical characteristics at the HRU level, only 50 percent of the HRU areas are considered as DWM treatment areas with a total area of 17.1 percent in Big Ditch and 29.3 percent in the Big/Long Creek watershed. Application of the DSM identified the Pareto optimal fronts and corresponding placement areas as shown in Figure 4.13 for both watersheds. The results indicate that 17 percent nitrate removal could be achieved with the DWM treatment area of 17 percent in Big Ditch watershed. A maximum treatment area of 29.3 percent for the Big/Long Creek watershed can reduce nitrate load by 31 percent. For every 4 to 8 ha of the treatment areas, a DWM structure is required (Christianson et al., 2013). Assuming a structure for every 8 ha of maximum DWM treatment areas in the study watersheds, 226 DWM structures will be required for Big Ditch and 264 for Big Long Creek watershed. Nitrate load reductions for the cost-effective solutions are found to be 3.2 kg N/ha/year for Big Ditch and 5.6 kg N/ha/year for Big/Long Creek watershed, having an average annual cost per reduction of $7.6/kg N /ha. DWM implementation areas corresponding to the best tradeoff are presented in Figure 4.14 for Big/Long Creek watershed and in Appendix B for Big Ditch watershed. For Big/Long Creek, the critical treatment area belonging to priority area 1 covers 5.6 percent of the watershed area. Similar to constructed wetlands, only 50 percent of HRU areas shown in the figure are considered as DWM treatment areas.

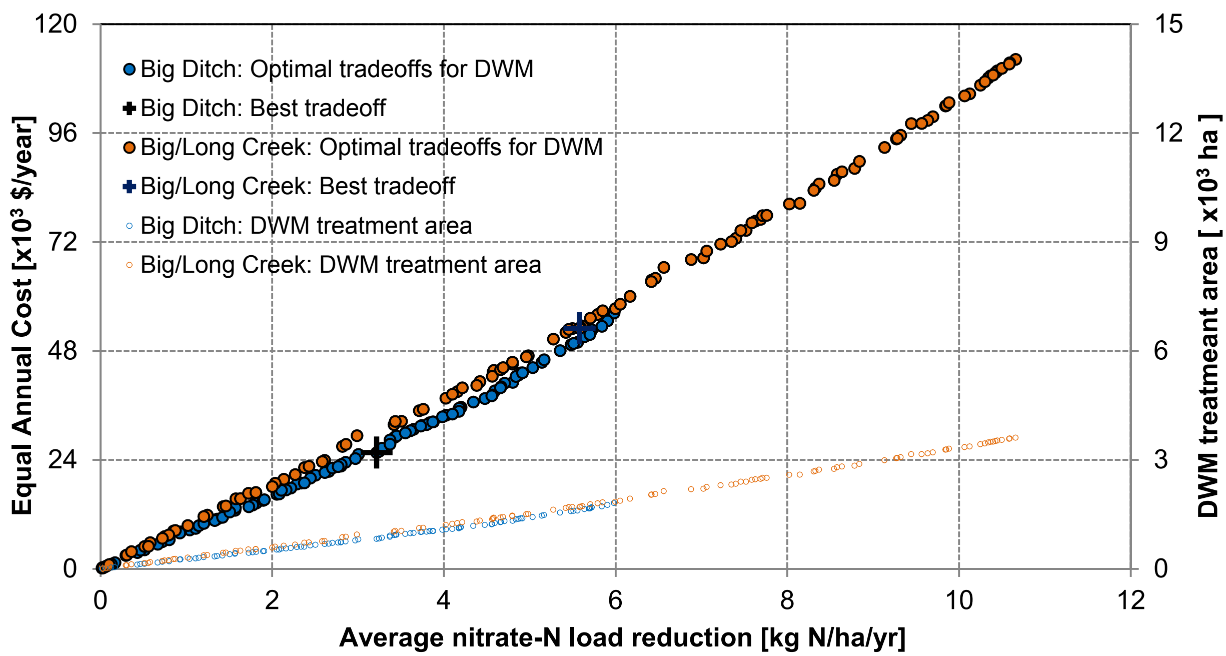


Figure 4.13 Drainage water management: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds

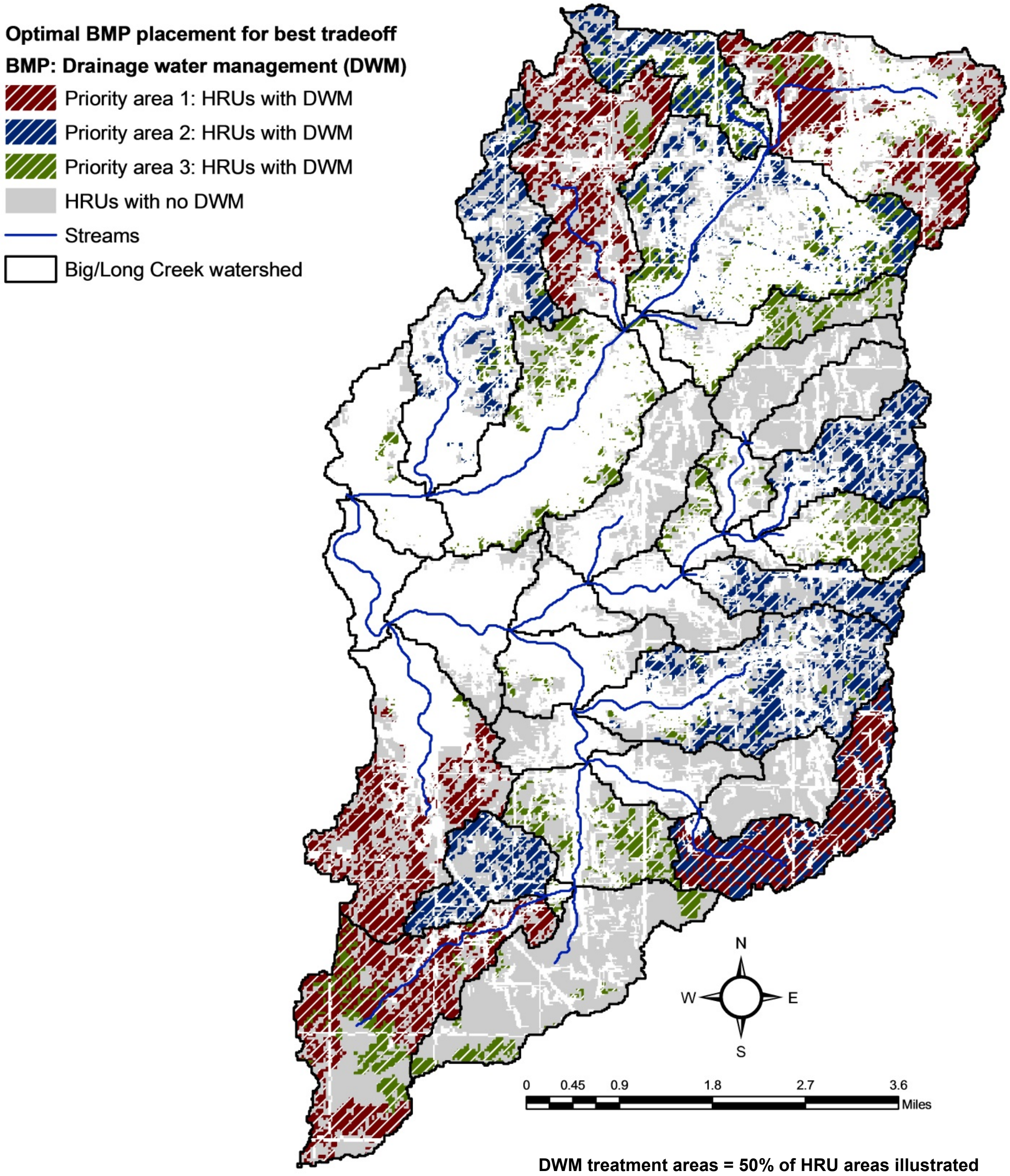


Figure 4.14 Optimal placement of drainage water management in Big/Long Creek watershed

Bioreactor Scenarios

All agricultural HRUs with tile drains are considered to be suitable for implementation of bioreactors in the study watersheds. In Big Ditch and Big/Long Creek watersheds, there are 319 and 267 qualifying HRUs, respectively, with corresponding treatment areas of 81.8 and 79.5 percent of their respective watershed areas. The DSM evaluation of bioreactors for both watersheds provided Pareto optimal fronts consisting of tradeoff solutions with treatment levels ranging from a few percentages of the watershed area to the maximum possible. Figure 4.15 shows optimal tradeoff solutions and the corresponding percentage of treatment areas for Big Ditch and Big/Long Creek watersheds. As illustrated in the figure, the maximum nitrate reduction is evidently obtained for the maximum treatment level. The best tradeoff solutions would result in a nitrate reduction of 41.8 percent in Big Ditch and 43.2 percent in Big/Long Creek watersheds with an average annual cost of $1.74/kg N/ha for both watersheds. If bioreactors were to be implemented at the maximum treatment level, a nitrate reduction of 66.8 percent would be achieved in Big Ditch and 80.5 percent in Big/Long Creek watersheds. For every 20.2 ha of treatment area, 0.1 ha bioreactor is required (Christianson et al., 2013). Implementation of the best tradeoff solutions would require 221 and 265 bioreactors in Big Ditch and Big Long Creek watersheds, respectively. The optimal placement of bioreactors in Big Ditch watershed is illustrated in Figure 4.16, and the critical treatment areas cover 3.6 percent of the watershed area shown as priority area 1 in the figure. The cost-effective bioreactor placements for both watersheds with corresponding priority areas and nitrate load reductions are provided in Appendices B and C.

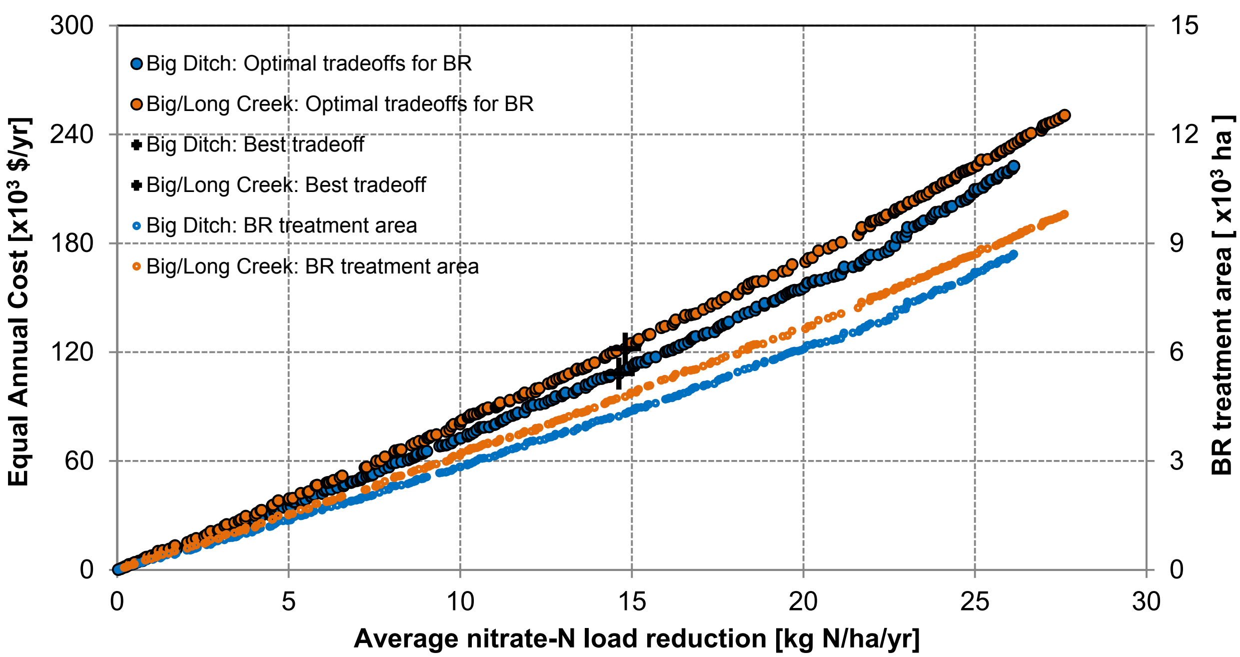


Figure 4.15 Bioreactors: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds

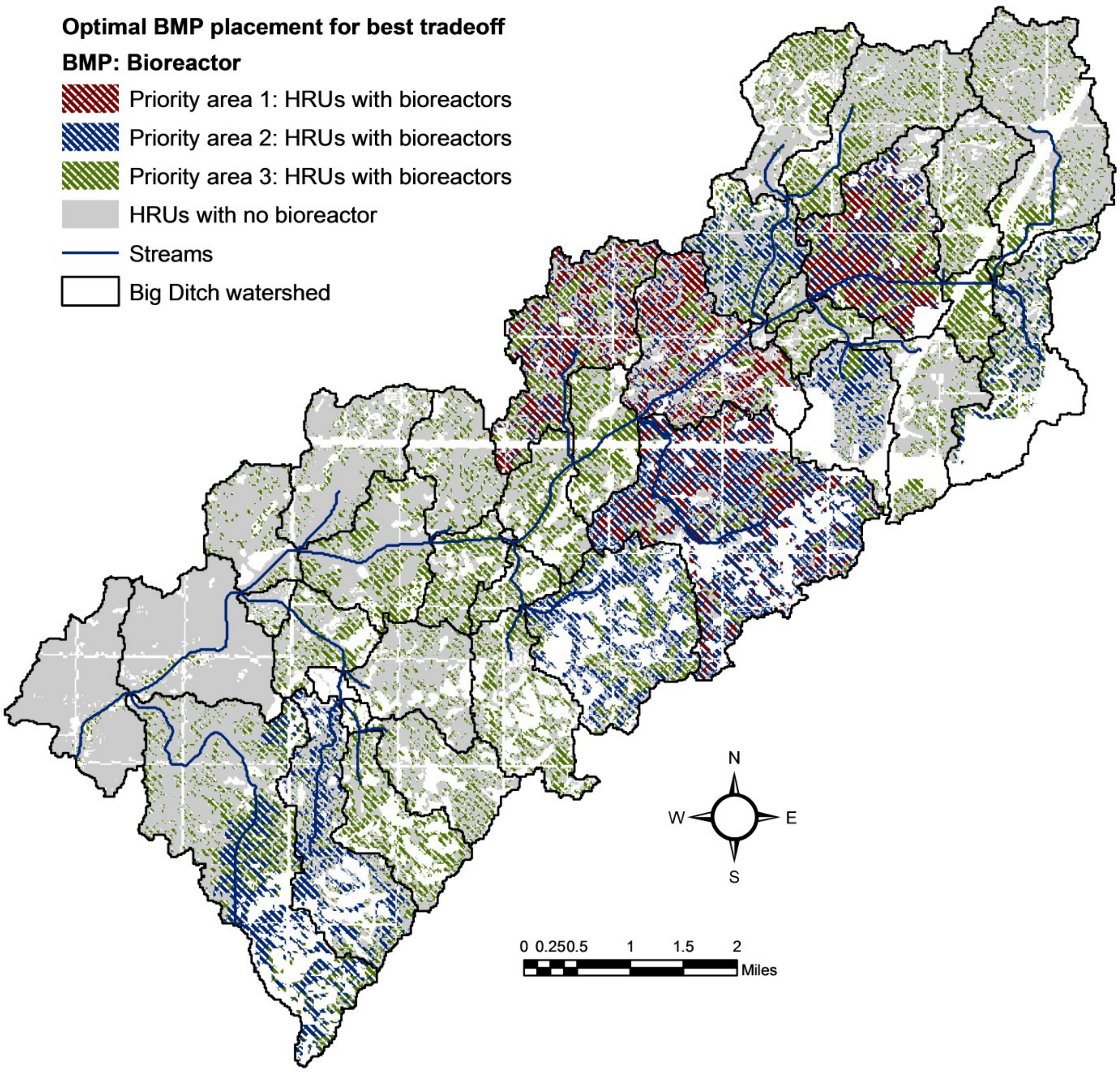


Figure 4.16 Optimal placement of bioreactors in Big Ditch watershed

Saturated Buffer Scenarios

Similar to drainage water management and bioreactors, the presence of tile drains is considered as a prerequisite for implementation of saturated buffers. Additional requirements include HRUs that have slopes greater than 2 percent and nearby ditches with slopes of at least 0.5 percent less than that of the HRUs. Since most of the HRUs in the study watersheds have slopes less than 2 percent, the strict slope requirements excluded large portions of both watersheds and suitable HRUs for saturated buffer implementation cover only 28.1 percent of the Big Ditch and 8.5 percent of the Big/Long Creek watershed area. The decision vector in the optimization algorithm included only those suitable HRUs (i.e., 115 for Big Ditch and 69 for Big/Long Creek watersheds). In this application, only 50 percent of the HRU drainage areas are considered as treatment areas for saturated buffers because of uncertainties associated with lumped physical characteristics at the HRU level. The DSM is applied to identify optimal placement of saturated buffers in the study watersheds and the Pareto optimal solutions are presented in Figure 4.17. The maximum nitrate load reductions obtained are 11.8 percent for Big Ditch watershed with a treatment area of 14.1 percent and 4 percent for Big/Long Creek with a treatment area of 4.2 percent. A control structure for every 10.1 ha of treatment area may be required for effective nitrate removal using saturated buffers, costing $3,500 per structure including labor and additional distribution tile (Jaynes and Isenhart, 2014). To achieve the maximum nitrate removal using saturated buffers, the total number of structures required will be 149 for Big Ditch and 51 for Big/Long Creek watersheds. The nitrate load reductions corresponding to the best tradeoff solutions are 2.1 kg N/ha/year for Big Ditch and 0.72 kg N/ha/year for Big/Long Creek watersheds with an estimated average annual cost of $11.74/kg N/ha and $34.4/kg N/ha, respectively. Figure 4.18 illustrates the placement of saturated buffers in the Big/Long Creek watershed for the best tradeoff solutions and, for Big Ditch watershed, it is included in Appendix B. Ranking of priority areas for implementing saturated buffers are also illustrated in the figures.

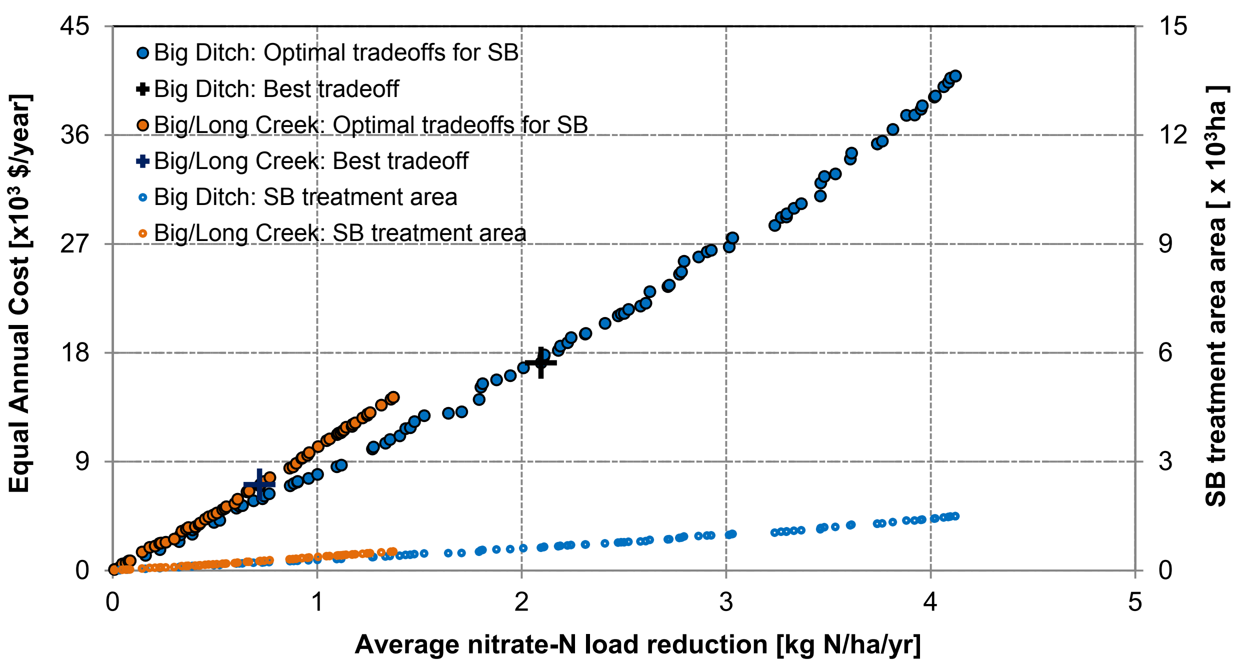


Figure 4.17 Saturated buffers: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds

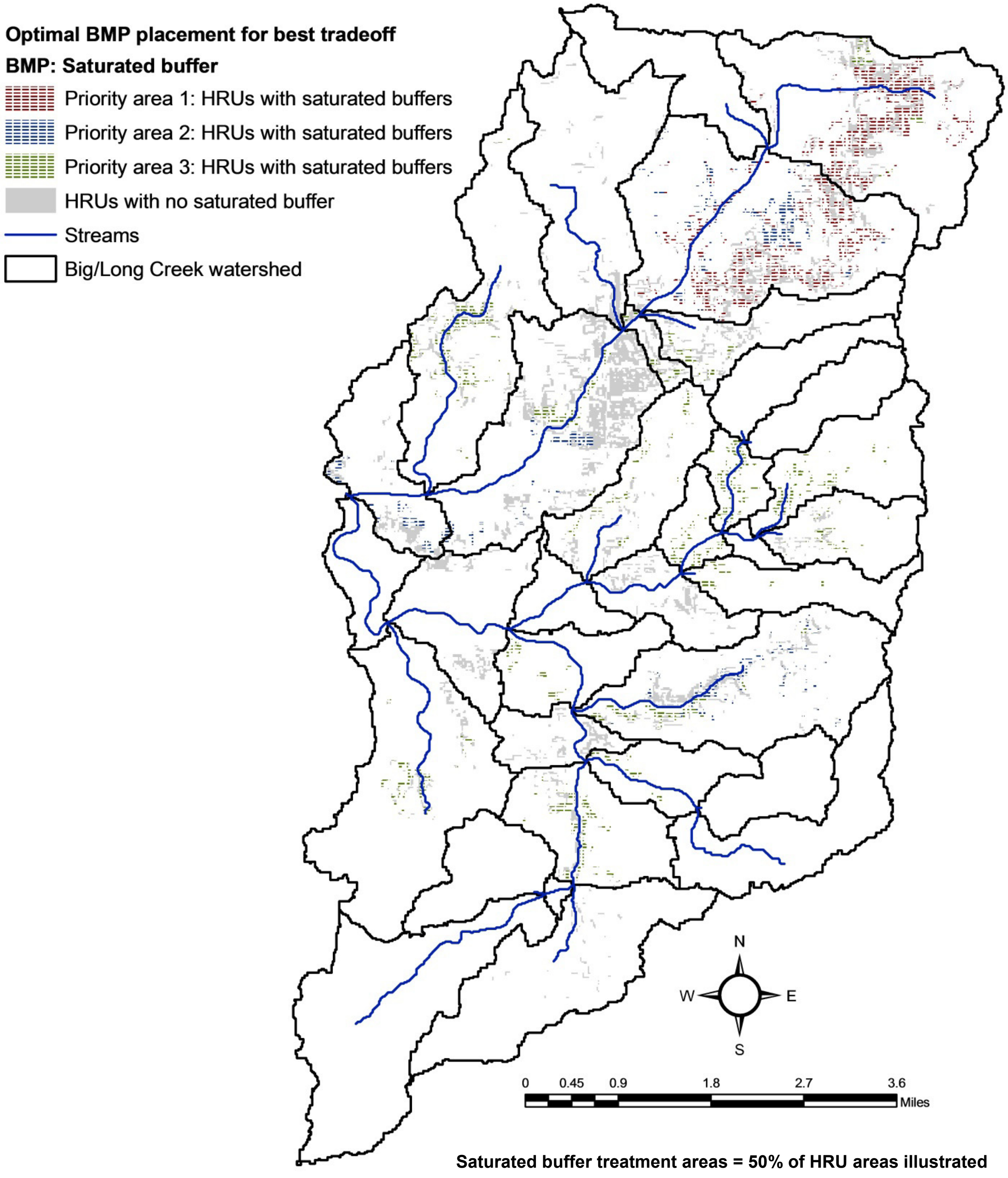


Figure 4.18 Optimal placement of saturated buffers in Big/Long Creek watershed

Filter Strip Scenarios

All agricultural HRUs are considered to be suitable for implementation of filter strips, similar to perennial and cover crop scenarios. While running the DSM for filter strip scenarios, the ratio of HRU area to filter strip area is set at 125, resulting in an average filter strip area of 0.52 acres for Big Ditch and 0.69 acres for Big/Long Creek watersheds. Figure 4.19 displays the Pareto optimal front, showing tradeoffs between nitrate load reduction and implementation cost, and corresponding treatment areas for both watersheds. The implementation costs of filter strips include estimated revenue losses due to removal of land out of production. The best tradeoff solutions require implementation of filter strips in about 40 percent of both watersheds. The resulting nitrate, phosphorus, and sediment load reductions are 9.9, 25.8, and 21.5 percent for the Big Ditch watershed and 6.4, 13.2, and 8.2 percent for the Big/Long Creek watershed, respectively. The result indicates that implementation of filter strips is more effective in reducing phosphorus and sediment loads in both watersheds as compared to nitrate load reductions. In addition, it shows that filter strips implementation is more cost-effective in the Big Ditch watershed with an estimated average annual cost of $1.65/ kg N/ha. A cost-effective placement of filter strips in the Big Ditch watershed is shown in Figure 4.20, including priority ranking of treatment areas. For the cost-effective solutions, the NPS pollutant reductions and placements of filter strips in both watersheds are presented in Appendices B and C.

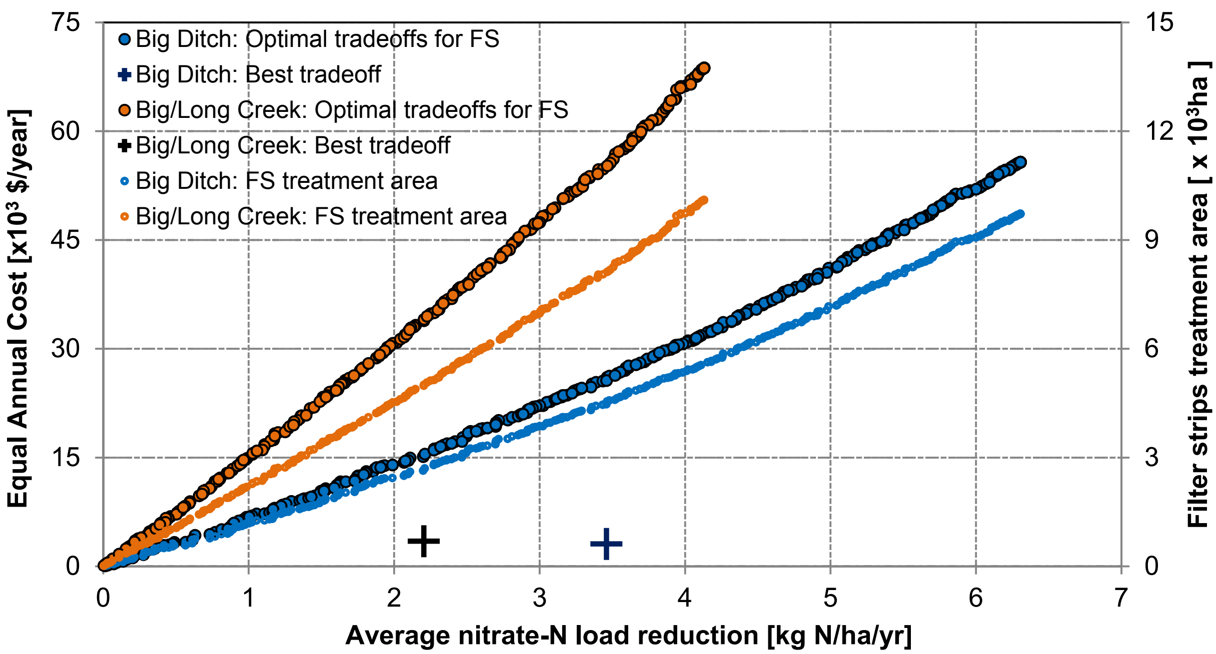


Figure 4.19 Filter strips: optimal tradeoffs for Big Ditch and Big/Long Creek watersheds

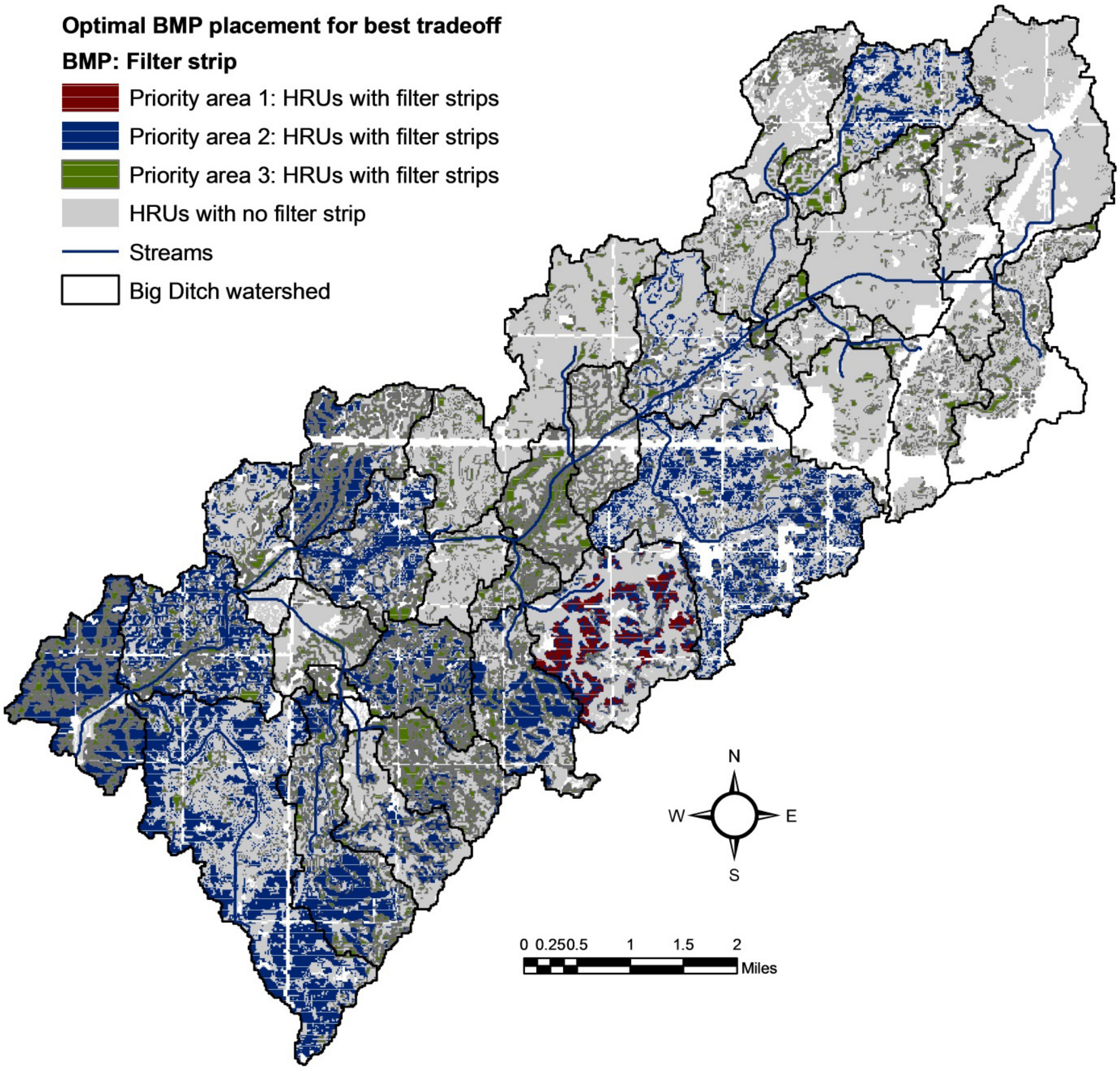


Figure 4.20 Optimal placement of filter strips in Big Ditch watershed

4.5 Optimal BMP Implementation Scenarios

The BMPs evaluated in this study have varying performance efficiencies in pollutant removal and implementation costs. NPS reductions obtained for the best tradeoff optimal placements of BMPs in Big Ditch and Big/Long Creek watersheds were summarized in Table 4.4 and 4.5. BMPs such as nutrient management, drainage water management, bioreactors, and saturated buffers were evaluated only for their impact on nitrate-N load reduction. In contrast, cover crops, constructed wetlands, filter strips, and perennial crops were considered not only for nitrate-N reduction but also for sediment and phosphorus load reductions. The size of the BMP treatment area is limited by the suitability of HRUs in the study watersheds on which a given BMP will be placed. In addition, only a fraction of the HRU area is considered as a treatment area for some of the BMPs including constructed wetland, saturated buffers, and drainage water management. For example, implementation of a saturated buffer requires strict topographic criteria, which include prescribed HRU slope and slope differences between HRU and a nearby ditch. In the DSM for Big Ditch and Big/Long Creek, the smallest modeling units are HRUs, but most fields are smaller, requiring aggregation of physical characteristics. In order to account for uncertainties arising from lumped representation of fields in the HRUs, only 50 percent of the HRU area is considered as a treatment area for a saturated buffer, drainage water management, and constructed wetlands. For the Big Ditch watershed, the treatment area for the best tradeoff optimal placement varies from 631 hectares for saturated buffers to 9,763 hectares (i.e., all agricultural HRUs) for nutrient management. Similarly, the treatment area varies from 257 hectares for a saturated buffer to 10,113 hectares for nutrient management implementations in Big/Long Creek watersheds.

The nitrate-N load reduction obtained was higher for the implementation of perennial alfalfa and bioreactors in both watersheds as compared to other BMPs. In contrast, it was the lowest for saturated buffers because suitable treatment areas were smaller, covering between 2 to 6 percent of the study watersheds. In addition to nitrate load reduction, perennial alfalfa resulted in 11 to 21 percent phosphorus load and 5 to 15 percent sediment load reductions at the watershed outlets. Cover crops also exhibited on average 7 percent sediment and phosphorus load reductions in both watersheds. Implementation of filter strips provided the maximum sediment and phosphorus load reductions, ranging from 8 to 22 and 13 to 26 percent, respectively.

Table 4.4 Pollutant reduction for the best tradeoff optimal placements of BMPs in Big Ditch watershed

Best management practices (BMPs)	BMP treatment area		Pollutant load reduction at the watershed outlet					
	as % of		Nitrate-N		Total phosphorus		Sediment	
	watershed area	[ha]	[kg N/ha/yr]	[%]	[kg P/ha/yr]	[%]	[t/ha/yr]	[%]
Nutrient management (NM)	91.4	9,763	5.518	15.8	n/a	n/a	n/a	n/a
Cover crop: Cereal rye (CR)	47.3	5,050	5.834	16.7	0.050	8.4	0.025	9.1
Cover crop: Annual ryegrass (RG)	43.3	4,630	4.666	13.3	0.047	7.8	0.023	8.4
Cover crop: Crimson clover (CC)	44.6	4,768	3.707	10.6	0.052	8.7	0.023	8.4
Perenial crop: Alfalfa (AA)	45.4	4,856	16.349	46.8	0.124	20.6	0.042	15.4
Constructed wetlands (CW)	18.6	1,990	5.987	17.1	0.043	7.2	0.014	5.0
Drainage water management (DWM)	7.8	829	3.216	9.2	n/a	n/a	n/a	n/a
Bioreactors (BR)	39.8	4,250	14.623	41.8	n/a	n/a	n/a	n/a
Saturated buffers (SB)	5.9	631	2.095	6.0	n/a	n/a	n/a	n/a
Filter strips (FS)	42.0	4,487	3.460	9.9	0.155	25.8	0.059	21.5

Table 4.5 Pollutant reduction for the best tradeoff optimal placements of BMPs in Big/Long Creek watershed

Best management practices (BMPs)	BMP treatment area		Pollutant load reduction at the watershed outlet					
	as % of		Nitrate-N		Total phosphorus		Sediment	
	watershed area	[ha]	[kg N/ha/yr]	[%]	[kg P/ha/yr]	[%]	[t/ha/yr]	[%]
Nutrient management (NM)	82.0	10,113	4.798	14.0	n/a	n/a	n/a	n/a
Cover crop: Cereal rye (CR)	41.2	5,083	5.295	15.4	0.073	6.9	0.039	4.6
Cover crop: Annual ryegrass (RG)	39.3	4,842	4.419	12.9	0.064	6.1	0.037	4.4
Cover crop: Crimson clover (CC)	37.8	4,666	2.691	7.8	0.060	5.7	0.039	4.6
Perenial crop: Alfalfa (AA)	38.9	4,801	12.826	37.4	0.119	11.3	0.048	5.6
Constructed wetlands (CW)	15.3	1,883	4.911	14.3	0.035	3.3	0.012	1.4
Drainage water management (DWM)	13.8	1,705	5.579	16.3	n/a	n/a	n/a	n/a
Bioreactors (BR)	38.7	4,770	14.806	43.2	n/a	n/a	n/a	n/a
Saturated buffers (SB)	2.1	257	0.718	2.1	n/a	n/a	n/a	n/a
Filter strips (FS)	40.4	4,978	2.204	6.4	0.138	13.2	0.070	8.2

A summary of average annual cost efficiencies for optimal placement of BMPs in Big Ditch and Big/Long Creek watersheds are presented in Tables 4.6 and 4.7, respectively. The BMPs are listed in order of decreasing cost efficiencies calculated for the best tradeoff solutions. For both watersheds, nutrient management is the best alternative with an average annual cost savings of \$6.4/kg N/ha. In both cases, the treatment areas for nutrient management include all agricultural HRUs, resulting in an average nitrate reduction of 14.9 percent at the watershed outlets. All the remaining BMPs require implementation costs with filter strips and bioreactors being the most cost efficient alternatives in Big Ditch and Big/Long Creek watersheds, respectively. The cost efficiency for implementation of filter strips in Big/Long Creek watershed is about 70 percent less than that of Big Ditch watershed, largely because of higher revenue losses built into the total implementation costs. Cash rent for land areas in Macon County, where Big/Long Creek watershed is located, is 60 percent higher. With an average cost efficiency of \$1.74/kg N/ha, the bioreactor is the next best alternative, ranking higher than all the remaining BMPs considered. Bioreactors have no added benefit to producers, and large-scale implementation is unlikely without providing incentives to the producers. Constructed wetlands have an average cost efficiency of \$10.89/kg N/ha including estimated revenue losses as a result of removing fertile land out of production to the total implementation cost. The difference between cost efficiencies of constructed wetlands in the study watersheds is mainly attributed to varying land values. Similar to bioreactors, its adoption is dependent on the provision of incentives to the land owners. Drainage water management is found to be more cost-effective in the Big/Long Creek watershed because of the larger optimal treatment area. It must be noted that the cost efficiencies are representative of the entire watershed area since the pollutant reductions were simulated for the watershed outlets. In the case of drainage water management implementation, there may be additional benefits in the form of yield increases, which could make this BMP more appealing to farmers. Accounting for revenues associated with yield increases could improve DWM's cost-effectiveness. In this study, revenue increases are not included in the computation of total cost for DWM scenarios. Saturated buffers are more cost-effective than cover crops or perennial crops in the Big Ditch watershed. In contrast, they are less cost-effective than cereal rye and annual ryegrass implementation scenarios for Big/Long Creek watershed but more cost-effective than crimson clover and perennial alfalfa implementation scenarios. The implementation of saturated buffers is limited to a small percentage of the watershed areas due to strict suitability criteria. Cover crops are generally among the least cost-effective alternatives as compared to other BMPs evaluated in this study. In quantifying the cost of implementation, revenues and other benefits from cover crop production, or yield decreases associated with it, are not considered. The success of cover crops is dependent upon timely harvest and adequate soil moisture, although they can be planted and harvested during the fallow period between corn-soybean rotations. Perennial alfalfa is the least cost-effective scenario for both watersheds. Although all agricultural HRUs are suitable for its implementation, its adoption to a large portion of the study watersheds is very unlikely. Alfalfa implementation should focus on critical areas that are designated as priority area 1 and 2 in the best tradeoff optimal placement maps (see Figures B.4 and C.4 in the Appendices).

Table 4.6 Equal annual cost (EAC) for the best tradeoff optimal placements of BMPs in Big Ditch

Best management practices (BMPs)	EAC per pollutant load Reduction					
	Nitrate		Total phosphorus		Sediment	
	[$/kg N/ha]	[$/lb N/acre]	[$/kg P/ha]	[$/lb P/acre]	[$/t/ha]	[$/lb /acre]
Nutrient management (NM)	-6.37	-7.14	n/a	n/a	n/a	n/a
Filter strips (FS)	1.65	1.85	36.88	41.33	97.29	0.11
Bioreactors (BR)	1.74	1.95	n/a	n/a	n/a	n/a
Constructed wetlands (CW)	9.18	10.29	1,279	1,433	3,997	4.48
Drainage water management (DWM)	9.62	10.78	n/a	n/a	n/a	n/a
Saturated buffers (SB)	13.00	14.57	n/a	n/a	n/a	n/a
Cover crop: Cereal rye (CR)	17.30	19.39	2,000	2,242	4,078	4.57
Cover crop: Annual ryegrass (RG)	19.59	21.96	1,956	2,192	3,982	4.46
Cover crop: Crimson clover (CC)	38.74	43.42	2,739	3,070	6,252	7.01
Perenial crop: Alfalfa (AA)	52.61	58.96	6,943	7,782	20,430	22.90

Table 4.7 Equal annual cost (EAC) for the best tradeoff optimal placements of BMPs in Big/Long Creek watershed

Best management practices (BMPs)	EAC per pollutant load Reduction					
	Nitrate		Total phosphorus		Sediment	
	[$/kg N/ha]	[$/lb N/acre]	[$/kg P/ha]	[$/lb P/acre]	[$/t/ha]	[$/lb /acre]
Nutrient management (NM)	-6.47	-7.26	n/a	n/a	n/a	n/a
Bioreactors (BR)	1.73	1.94	n/a	n/a	n/a	n/a
Filter strips (FS)	3.09	3.46	49.25	55.21	96.97	0.11
Drainage water management (DWM)	5.57	6.24	n/a	n/a	n/a	n/a
Constructed wetlands (CW)	12.61	14.13	1,785	2,001	5,316	5.96
Cover crop: Cereal rye (CR)	20.43	22.90	1,491	1,672	2,749	3.08
Cover crop: Annual ryegrass (RG)	20.78	23.29	1,436	1,610	2,465	2.76
Saturated buffers (SB)	38.09	42.69	n/a	n/a	n/a	n/a
Cover crop: Crimson clover (CC)	53.61	60.08	2,402	2,693	3,690	4.14
Perenial crop: Alfalfa (AA)	86.93	97.43	9,405	10,541	23,326	26.15

4.6 Impact of Perennial and Cover Crops on Water Yield

The implementation of perennial and cover crops would lengthen the growing period in a given year, thereby providing extended soil cover, increasing infiltration, and reducing soil erosion, runoff, and nutrient and sediment losses to downstream water bodies. In addition, it extends the period of plant water uptake, reducing the volume of drainage water and nutrient leaching. Although nutrient removal is the desired effect in the study watersheds, reduction of water yield as a result of implementing perennial and cover crops could negatively impact the amount of flows downstream during low flow periods. Therefore, water yield impacts were evaluated for the best tradeoff optimal placements of perennial and cover crops in Big Ditch and Big/Long

Creek watersheds and are illustrated in Figures 4.21 and 4.22, respectively. The figures show long-term monthly average water yields (1991-2010) for the baseline and the implementation of cereal rye, annual ryegrass, crimson clover, and alfalfa.

Simulation results indicate that all three cover crops caused the reduction of water yields in both watersheds for most of their growing period (i.e., between September 15th and April 10th) as expected. A greater reduction was particularly exhibited in October and November, ranging from 9 to 16 percent for cereal rye and annual ryegrass, but the water yield reduction was less than 5 percent for crimson clover. Annual evapotranspiration increased by less than 1.5 percent in both watersheds, but in the month of October, the increase in evapotranspiration was higher, ranging between 38 to 43 percent. For both watersheds, the impact on annual water yields was found to be minimal, which was less than a 2.5 percent reduction. In the case of cereal rye and annual ryegrass implementation, the water yields increased during the corn-soybean growing season, particularly in the months of April and May due to increased tile flows in those months. Annual tile flows account for 27.5 and 30.4 percent of annual water yields in Big Ditch and Big/Long Creek watersheds, respectively.

For implementation scenarios of perennial alfalfa, the reduction of average water yields was evident in all months with the exception of July and August. The average annual water yields decreased between 4 and 7 percent from the baseline, but a higher water yield reduction was exhibited in October and November, ranging between 14 and 24 percent. Higher precipitation in the months of July and August during the simulation period and field operations considered for alfalfa implementation, which includes several cuttings per year and minimal residue left on the ground, may have contributed to increased water yields in those months.

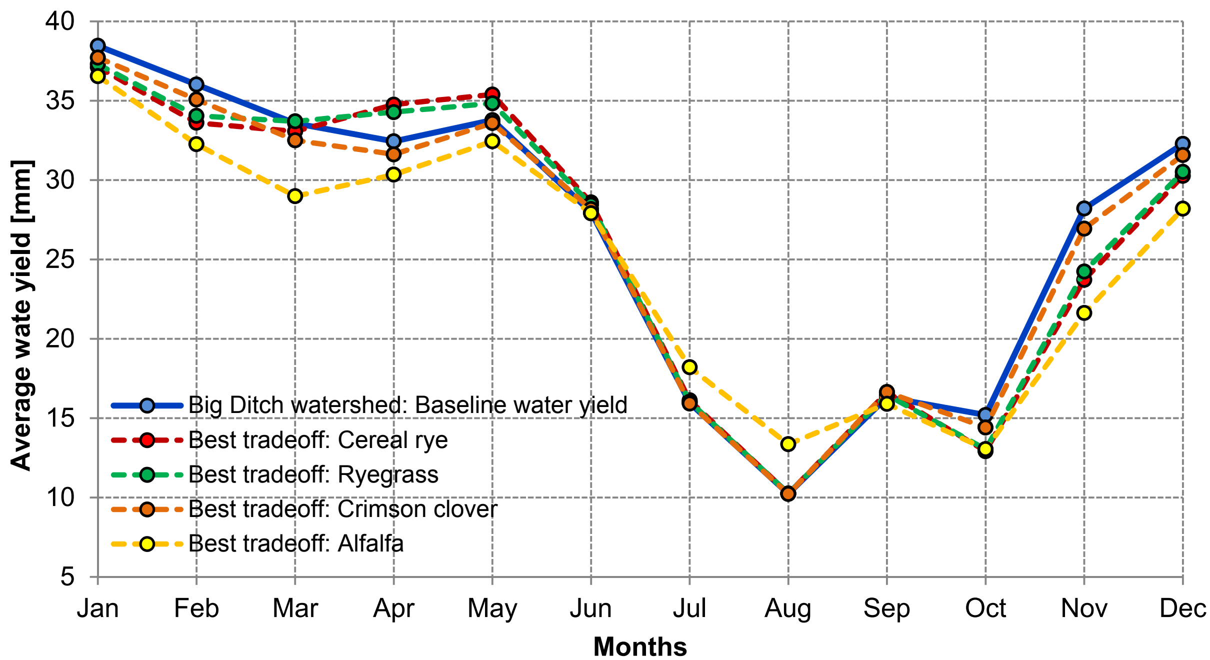


Figure 4.21 Impacts of cereal rye, ryegrass, crimson clover, and alfalfa on Big Ditch water yield

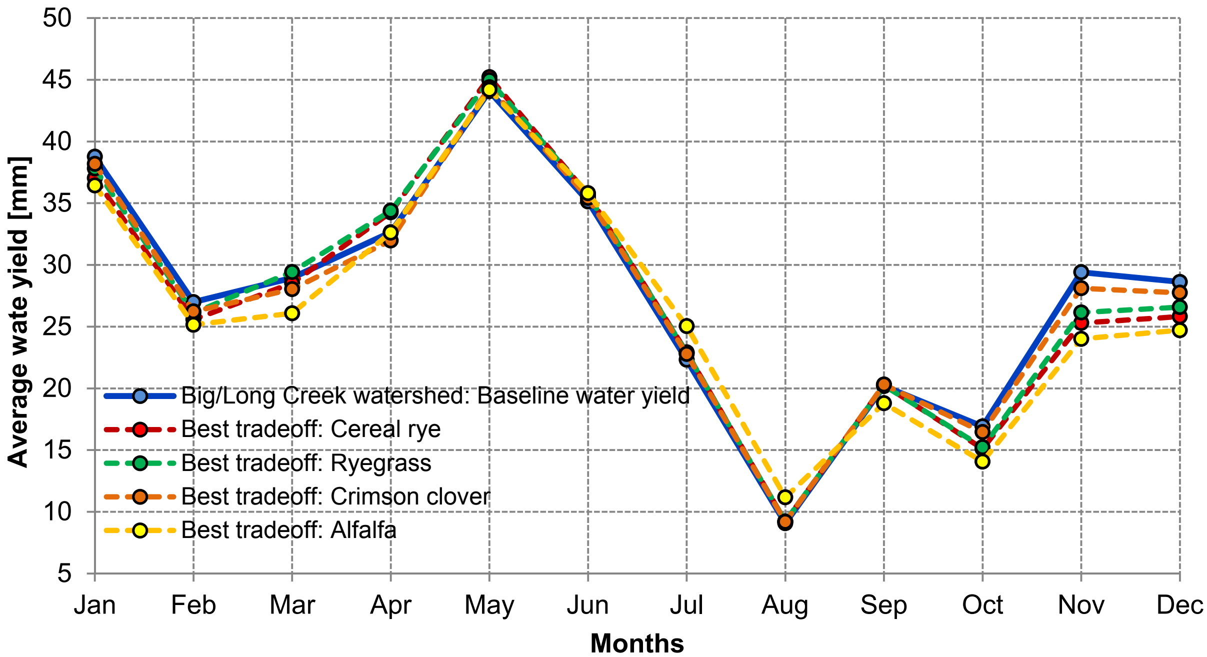


Figure 4.22 Impacts of cereal rye, ryegrass, crimson clover, and alfalfa on Big/Long Creek water yield

5. Summary, Conclusions, Limitations, and Recommendations

5.1 Summary

The objective of this research is to generate alternative BMP implementation scenarios for Big Ditch and Big/Long Creek watersheds by evaluating their water quality impacts at a watershed scale. This is accomplished through the development and application of decision support models (DSMs), which are coupled optimization-watershed models. In the DSM, the watershed model-SWAT simulates the hydrology, water quality, and impact of BMPs on water quality, whereas the optimization algorithm, AMGA2, is tasked with identifying the optimal placement of BMPs that provides tradeoffs between NPS reduction and the cost of BMP implementation. The DSMs were applied to evaluate the water quality benefits of conventional and emerging BMPs that include nutrient management (fertilizer rate and timing), cover crops (cereal rye, annual ryegrass, and crimson clover), a perennial crop (alfalfa), constructed wetlands, drainage water management, bioreactors, saturated buffers, and filter strips.

A spatial sensitivity analysis was conducted to identify the threshold subdivision levels for Big Ditch and Big/Long Creek watershed models. Nine different watershed subdivision levels were generated for each watershed using critical source areas ranging from 0.5 to 11.3 percent of the watershed area, resulting in a total of 18 watershed models for evaluating the sensitivity of watershed hydrologic and water quality responses to the spatial scale. A threshold drainage area close to 2 percent for which sediment and nutrient load simulations stabilized were selected to determine the number of subbasins for the watershed models, dividing Big Ditch and Big/Long Creek watersheds into 38 and 35 subbasins, respectively. For accurate representation of watershed characteristics, further subdivision of Big Ditch and Big/Long Creek watersheds into 450 and 467 HRUs were done based on a threshold area of 5 hectares for land use, soil, and slope categories in the subbasins.

Detailed land management operations that include crop rotations, fertilizer application, and tillage practices were prepared for agricultural HRUs in the watershed. Big Ditch and Big/Long Creek watershed models were then calibrated and validated for monthly streamflows and nitrate load simulations. Sediment and total phosphorus calibrations were done only for Big Ditch watershed and calibrated parameters were then transferred to the Big/Long Creek watershed model for use in sediment and phosphorus simulations.

The suitability of agricultural HRU areas for BMP implementation was identified using topographic features, drainage, and soil characteristics. Suitable HRU areas vary by BMP type, and they define the search solution space for optimal BMP placement in the study watersheds. All agricultural HRUs were deemed to be suitable candidates for nutrient management, cover crops, perennial crops, and filter strips, accounting for 91 and 82 percent of Big Ditch and Big/Long Creek watershed areas, respectively. HRUs that are potentially suitable for placement of constructed wetlands make up 71 and 70.2 percent of the Big Ditch and Big/Long Creek watershed areas, respectively. In selecting HRUs suitable for drainage water management, bioreactors, and saturated buffers, the presence of tile drainage in the HRUs and topographic features are taken into account. The percentage areas of Big Ditch watershed that are suitable for drainage water management, bioreactors, and saturated buffers are 34.2, 81.8, and 28.1, respectively, whereas for Big/Long Creek, 58.7, 79.5, and 8.5 percent of the watershed area are suitable for implementation of drainage water management, bioreactors, and saturated buffers, respectively.

Using the DSM models, the water quality benefits of the BMPs were evaluated with respect to baseline scenarios developed for Big Ditch and Big/Long Creek watersheds. The baseline scenarios were derived from representative land uses, land management, and climate conditions for the study watersheds. The DSM evaluation of nutrient management scenarios identified an optimal fertilizer application rate of 155 lb N/acre for both Big Ditch and Big/Long Creek watersheds with 30 percent fall, 20 percent spring pre-plant, and 50 percent after planting fertilizer applications as the best tradeoffs. In both watersheds, the maximum nitrate load reduction is obtained when there is no fall application and when the distribution between pre-planting and after planting fertilizer application is at approximately the 50 percent level. The optimal placements for nutrient management covers all agricultural HRUs in the study watersheds, providing average nitrate load reductions of 14.9 percent with an average annual cost savings of at least $6.42 /kg N/ha or $7.2/lb N/acre.

Three cover crops, including cereal rye, annual rye grass, and crimson clover, were simulated using the DSM for reduction of nitrate-N, sediment, and phosphorus. The best tradeoff scenarios for all cover crops resulted in average nitrate-N and total phosphorus load reductions of at least 7.8 and a sediment load reduction of 4.4 percent, requiring 4600 to 5050 hectares of treatment area. The least average cost per nitrate reduction is obtained for cereal rye ($18.9/kg N/ha) and the highest is for crimson clover ($46.1/kg N/ha), partly because the seed cost is more expensive.

Perennial alfalfa is simulated for its water quality benefits, resulting in reductions of soil erosion and fertilizer application, and the DSM identified optimal placements of alfalfa in Big Ditch and Big/Long Creek watersheds for cost-effective reduction of nitrate, sediment, and phosphorus loads. The resulting optimal tradeoff solutions indicate that the implementation of alfalfa provides more nitrate load reduction per dollar for the Big Ditch watershed (i.e., $52.6/kg N/ha/year), as compared to the Big/Long Creek watershed (i.e., $86.9/kg N/ha/year). Lower cost efficiency for the Big/Long Creek watershed is mainly attributed to higher revenue losses associated with converting corn-soybean rotation into perennial alfalfa. On average, the best tradeoff solutions provide nitrate, phosphorus, and sediment load reductions of at least 37.4, 11.3, and 5.6 percent, respectively, in both watersheds. The most critical treatment areas for alfalfa implementation were identified in the best tradeoff optimal placements for Big Ditch and Big/Long Creek watersheds and cover 4.5 and 11.3 percent of its respective watershed areas.

Constructed wetlands were modeled to drain only 50 percent of the HRU in which they are located because of uncertainties associated with lumped physical characteristics at the HRU level. The cost-effective implementation scenarios provide nitrate, phosphorus, and sediment load reductions of at least 14.3, 3.3, and 1.4 percent, respectively, requiring an average wetland area of 240 acres in each watershed. Results show that constructed wetlands performed better in nitrate load reduction for both watersheds, providing an average reduction of 5.4 kg N/ha/year. Accounting for the cost of constructed wetlands and associated revenue losses, its average cost efficiency was calculated to be $10.9/kg N/ha/year. The most critical treatment areas for implementation of constructed wetlands were identified within the best tradeoff placements in both watersheds and are 2.2 and 4.6 percent of Big Ditch and Big/Long Creek watershed areas, respectively.

Nitrate load reductions for cost-effective DWM scenarios were found to be 3.2 kg N/ha/year for Big Ditch and 5.6 kg N/ha/year for Big/Long Creek watersheds, having an average annual cost per reduction of $7.6/kg N/ha. The corresponding optimal treatment areas cover 829 and 1,705 hectares in Big Ditch and in Big/Long Creek watersheds, respectively. DWM may affect flow, sediment, and phosphorus transport. In this study, the impact of DWM on sediment and phosphorus reduction was not evaluated due to lack of data.

Through implementation of bioreactors in the study watersheds, maximum nitrate load reductions ranging from 66 to 81 percent could be achieved. Higher load reduction would be possible because of extensive tile drainage in both watersheds, covering about 80 percent of the watershed areas. The best tradeoff solutions would provide an average nitrate reduction of 42 percent with an average annual cost of $1.74/kg N/ha.

Due to strict topographic requirements for implementation, the maximum nitrate load reduction obtained for saturated buffers was 11.8 for the Big Ditch watershed and 4 percent for the Big/Long Creek watershed. The nitrate load reductions corresponding to the best tradeoff scenarios are 2.1 kg N/ha/year for Big Ditch and 0.72 kg N/ha/year for Big/Long Creek watersheds with estimated average annual costs of $11.74/kg N/ha and $34.4/kg N/ha, respectively. The results indicate that saturated buffers are more cost-effective in the Big Ditch watershed, partly due to availability of larger treatment areas suitable for saturated buffers.

Filter strip areas in both watersheds were simulated as 0.8 percent of the HRU where they are placed, resulting in an average filter strip area of 0.6 acres. Assuming all agricultural HRUs are suitable, the best tradeoff solutions require implementation of filter strips in approximately 40 percent of both watersheds. However, the resulting nitrate, phosphorus, and sediment load reductions were higher for Big Ditch watershed. Filter strips were found to be more cost-effective in the Big Ditch watershed with an estimated average annual cost of $1.65/ kg N/ha as compared to $3.09/ kg N/ha. High revenue losses estimated for filter strip implementation in the Big/Long Creek watershed contributed to lesser cost efficiency. The DSM result indicates that filter strips are more suitable for phosphorus and sediment load reduction than nitrate load reduction.

5.2 Conclusions

For both watersheds, nutrient management is found to be the best alternative with an average annual cost savings of $6.42/kg N/ha, resulting in an average nitrate reduction of 14.9 percent at the watershed outlets. Filter strips and bioreactors are more cost-effective as compared to all BMPs evaluated with the exception of nutrient management. In contrast, perennial alfalfa is the least cost-effective with an average annual cost per reduction of $69.8/kg N/ha, providing an average nitrate load reduction of 42.1 percent. Successful adoption of BMPs such as bioreactors and constructed wetlands would require provision of incentives in the form of cost-sharing because they provide no added value to the producers. Implementation of saturated buffers is possible only to a smaller percentage of the study watersheds because of strict topographic requirements. It is found to be more cost-effective than perennial and cover crops for Big Ditch watershed even with its strict suitability criteria for implementation. It is, however, less cost-effective than cereal rye and annual ryegrass implementation scenarios for the Big/Long Creek watershed. Cover crops in general are found to be the least cost-effective as compared to other BMPs but their implementation could be appealing to the land owners since they can be grown during the fallow period between corn-soybean rotations. Their implementation caused monthly water yield reductions in most of their growing periods. Higher reductions were obtained for the months of October and November with a maximum of 16 percent. Reduction in annual water yield is less than 2.5 percent for both watersheds. Due consideration should be given to other impacts of cover crop production such as yield increases or decreases while calculating its cost-effectiveness. The least cost-effective BMP simulated for both watersheds is perennial alfalfa with an average treatment area of 4,829 hectares. Since its adoption to such a large portion of the watershed is unlikely, its implementation should focus on the most critical areas. The optimal implementation scenario for alfalfa has caused reduction of water yield in most of the months with a higher reduction in October and November. The implementation of perennial and cover crops for water quality benefits should consider their impact on water yields of the study watersheds, particularly in periods of low flows.

5.3 Limitations of the Study

The implementation of multiple BMPs could help achieve greater pollutant load reduction. For example, nutrient management can be coupled with any of the BMPs evaluated in this study. However, in such cases, other BMPs may not be as effective. One of the limitations of this study is that interactions between the implementation of multiple BMPs were not evaluated and the water quality impact of each BMP was compared with a baseline scenario that is believed to be representative of current practices in the study watersheds, including a higher fertilizer input and fall N application. All model evaluations performed in this study were based on a time series of climate data that are identical to the last two decades, which may or may not reflect future weather conditions. The emergence of new practices in the future such as new crops or fertilizer formulations may alter the land use practices, thereby affecting the hydrology and water quality. Optimal placements of BMPs for the study watersheds were provided for HRUs, not fields. HRUs are discontinuous land areas with homogeneous land use, soil and slopes spatially located in a subbasin but their responses are not specific to any particular field, rather to the patches of land areas designated as HRUs. Therefore, implementation of BMPs requires further identification of suitable areas through mapping of HRUs into actual fields. In addition, the estimated sediment and phosphorus load reductions for Big/Long Creek watersheds as result of BMP implementation were based on un-calibrated model since no sediment and phosphorus data is available.

5.4 Recommendations for Future Work

Optimizing the placement of multiple BMPs in the study watersheds could be a daunting task because of huge solution search space resulting from a large number of BMP combinations for each HRU. It is recommendable to evaluate the water quality benefits of multiple BMPs by developing implementation scenarios that make use of the best tradeoff solutions identified in this study. With additional modeling effort, the current decision support model can be extended to evaluate the interaction of multiple BMPs. A tool that maps DSM outputs to field level results will be needed to identify actual fields for BMP implementation. Hydrologic and water quality monitoring are required to calibrate Big/Long Creek watershed for sediment and phosphorus, thereby improving accuracy of DSM outputs. Future climate projections can be incorporated into the current modeling framework to investigate the impact of climate change on the performance efficiencies of the BMPs.

Optimal BMP scenarios presented in this study will provide greater water quality benefits at the outlets of the study watersheds. However, the study watersheds cover only about 9 percent of the Lake Decatur drainage area. In order to have a full picture of achievable sediment and nutrient load reductions through implementation of selected BMPs, the decision support system should include the remaining larger area of the Lake Decatur watershed. Development of a decision support model and TMDL implementation plan for the entire Lake Decatur watershed will provide optimal placement of BMPs in the context of the entire watershed and identify priority sub-watersheds for BMP implementation. Such models can also be used to identify optimal implementation scenarios of selected BMPs to meet nitrate and phosphorus TMDL goals for Lake Decatur.

References

ACE. 2013. Revenue and costs for corn, soybeans, wheat, and double-crop soybeans, actual for 2007 through 2013, projected for 2013 and 2014. Department of Agricultural and Consumer Economics, University of Illinois.

Arnold, J., J. R. Kiniry, R. Srinivasan, J. R. Williams, E. B. Haney, and S. L. Neitsch. 2011. *Soil and water assessment tool: Input/output file documentation*. TR-365. Grassland, Soil and Water Research Laboratory, Agricultural Research Service, College Station, TX.

Bagnold, R. A. 1977. Bedload transport in natural rivers. *Water Resources Res*. 13(2): 303–312.

Bekele, E. G., M. Demissie, and Y. Lian. 2011. Optimizing the placement of best management practices (BMPs) in agriculturally-dominated watersheds in Illinois. *World Environmental and Water Resources Congress,* 22-27 May, Palm Springs, CA.

Bekele, E. G., C. Lant, S. Soman, and G. Misgna. 2013. The evolution and empirical estimation of ecological-economic production possibilities. *Ecological Economics* 90: 1-9. doi: 10.1016/ - j.ecolecon.2013.02.012.

Binger, R. L., J. Garbrecht, J. G. Arnold, and R. Srinivasan. 1997. Effect of watershed subdivision on simulation runoff and fine sediment yield. *Transaction of the ASAE* 40(5): 1329–1335.

Christainson, L., J. Tyndall, and M. Helmers. 2013. Financial comparison of seven nitrate reduction strategies for Midwestern agricultural drainage. *Water Resour. Econ.* 2(3): 30–56.

Cooke, R. and S. Verma. 2012. Performance of drainage water management systems in Illinois, United States. *J. Soil and Water Cons.* 6:453–464.

Deb, K. 2001. *Multiobjective optimization using evolutionary algorithms*. John Wiley and Sons, Chichester, UK.

Deb, K., A. Pratap, S. Agrawal, and T. Meyarivan. 2002. A fast and elitist multiobjective genetic algorithm: NSGA-II. *IEEE Trans. On Evolutionary Computation* 6(2):182–197.

Fernandez, F. G, S. A. Ebelhar, E. D. Nafziger, and R. G. Hoeft. 2010. Managing nitrogen. In *Illinois agronomy handbook*. p. 113–132.

FitzHugh, T. W. and D. S. Mackay. 2000. Impacts of input parameter spatial aggregation an agricultural nonpoint source pollution model. *J. Hydrol.* 236(2): 35–53.

Greenan, C. M., T. B. Moorman, T. B. Parkin, T. C. Kaspar, and D. B. Jaynes. 2009. Denitrification in Wood chip bioreactors at different water flows. *J. Environ. Qual.* 38:1664–1671.

Gumaa, Y. T., Haffar, I., and Al-Afifi, M. A. 1998. Financial appraisal of date-frond mat fence systems for wind erosion control and sand dune stabilization in the arid region of the UAE. *J Arid Environ* 39:549–557.

Gupta, H. V., S. Sorooshian, and P. O. Yapo. 1999. Status of automatic calibration for hydrologic models: Comparison with multilevel expert calibration. *J. Hydrologic Engr.*, ASCE, 4(2): 135–143.

IEPA. 2004. *Illinois 2004 Section 303(d) List*. Illinois Environmental Protection Agency, Springfield, IL.

IEPA. 2007. *Sangamon River/ Lake Decatur watershed final approved TMDL*. Illinois Environmental Protection Agency, Springfield, IL.

Jaynes, D. B. and T. M. Isenhart. 2014. Reconnecting tile drainage to riparian buffer hydrology for enhanced nitrate removal. *J. Environ. Quality*. 43:631–638. doi:10.2134/jeq2013.08.0331.

Jha, M., P. W. Gassman, S. Secchi, R. Gu, and J. Arnold. 2004. Effect of watershed subdivision on SWAT flow, sediment, and nutrient predictions. *J. Ameri. Water Resou. Assoc.* 40(3): 811–825.

Keefer, L. and E. Bauer. 2011. Upper Sangamon River Watershed Monitoring Data for the USEPA Targeted Watershed Study: 2005-2008.

Keefer, L., E. Bauer, and M. Markus. 2010. *Hydrologic and nutrient monitoring of the Lake Decatur watershed*. Illinois State Water Survey, Champaign, IL.

Klemperer, W. D. 1996. *Forest resource economics and finance*. New York: McGraw-Hill Inc.

Lemke, M. and D. A. Kovacic. 2008. *Ecological and economic benefits of conservation based BMP's in the Mackinaw River Watershed*. DNR Project # MRP002-04.

MCCC. 2014. Midwest Cover Crops Council website (Available at http://www.mccc.msu.edu/ accessed 07 January 2014).

McElroy, A. D., S. Y. Chiu, J. W. Nebgen, A. Aleti, and F. W. Bennett. 1976. *Loading functions for assessment of water pollution from nonpoint sources.* EPA document EPA 600/2-76-151. USEPA, Athens, GA.

Moriasi, D. N., J. G. Arnold, M. W. Van Liew, R. L. Binger, R. D. Harmel and T. L. Veith. 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of the ASABE*, 50(3): 885–900.

Muleta, M. K., J. W. Nicklow, and E. G. Bekele. 2007. Sensitivity of a distributed watershed simulation model to spatial scale. *J. of Hydrolo. Eng.*, ASCE, 12(2): 163–172.

Nash, J. E. and J. V. Sutcliffe. 1970. River flow forecasting through conceptual models: Part I – A discussion of principles. *J. Hydrology* 125: 277–291.

Neitsch, S. L., J. G. Arnold, J. R. Kiniry, and J. R. Williams. 2011. *Soil and water assessment tool: Theoretical documentation*. TR-406. Grassland, Soil and Water Research Laboratory, Agricultural Research Service, College Station, TX.

Nicklow, J. W., and L. M. Mays. 2000. Optimization of multiple reservoir networks for sedimentation control. *J. Hydraulic Engineering* 126(4): 232–242.

NRCS. 2013. Drainage Water Management Fact Sheet. (Available at http://www.nrcs.usda.gov/-wps/portal/nrcs/detailfull/national/technical/references/?&cid=nrcsdev11_000182 accessed 07 January 2014)

Reed, P. and B. Minsker. 2004. Striking the balance: Long-term groundwater monitoring design for conflicting objectives. *J. Water Resour. Plng. and Mgmt., ASCE*, 130(2): 140–149.

Robertson, W. D., G. I. Ford, and P. S. Lombardo. 2005. Wood-based filter for nitrate removal in septic systems. *Transactions of the ASAE* 48:121–128.

Schipper, L. A., W. D. Robertson, A. J. Gold, D. B. Jaynes, and S. C. Cameron. 2010. Denitrifying bioreactors:An approach for reducing nitrate loads to receiving waters. *Ecological Engineering* 36:1532–1543.

SCS. 1972. Section 4: Hydrology. In *National engineering handbook*. Soil Conservation Service, Washington, D.C.

Singh, A., B. S. Minsker, and D. E. Goldberg. 2004. Combining reliability and Pareto Optimality: An approach using stochastic multiobjective genetic algorithm. *Proceedings of the 2004 World Congress on Water and Env. Resour.,* ASCE, June 27-July 1, Salt Lake City, UT.

Skaggs, R. W., N. R. Fausey, and R. O. Evans. 2012. Drainage water management. *J. Soil and Water Cons.* 6:167A–172A.

Tiwari, S. 2008. AMGA: an archive-based micro genetic algorithm for multi-objective optimization. *Proceedings of the Genetic and Evolutionary Computation Conference*, July 12–16, Atlanta, GA.

Tiwari, S., G. Fadel, and K. Deb. 2011. AMGA2: Improving the performance of the archive-based micro-genetic algorithm for multi-objective optimization. *Engineering Optimization*, 43(4): 377–401.

USEPA. 2013. Nutrient Management (Available at http://www.epa.gov/oecaagct/-ag101/cropnutrientmgt.html, accessed 20 December 2013).

Verma, S. et al. 2010. Evaluation of conservation drainage systems in Illinois – bioreactors. *2010 ASABE Annual International Meeting*, 20-23 June, Pittsburgh, PA.

Williams, J. R. 1969. Flood routing with variable travel time or variable storage coefficients. *Transactions of the ASAE* 12(1): 100–103.

Williams, J. R. 1995. The EPIC Model. In *Computer models of watershed hydrology*. Water Resources Publications, Highlands Ranch, CO, p. 909-1000.

Williams, J.R. and R.W. Hann. 1978. *Optimal operation of large agricultural watersheds with water quality constraints*. Texas Water Resources Institute, Texas A&M Univ., Tech. Rept. No. 96.

Zitzler, E. and Thiele, L. 1999. Multiobjective evolutionary algorithms: A comparative case study and the strength pareto approach.IEEE *Transactions on evolutionary Computation*, 3(4): 257–271.

Appendix A. Suitable HRUs in Big Ditch and Big/Long Creek Watersheds for Implementation of Selected BMPs

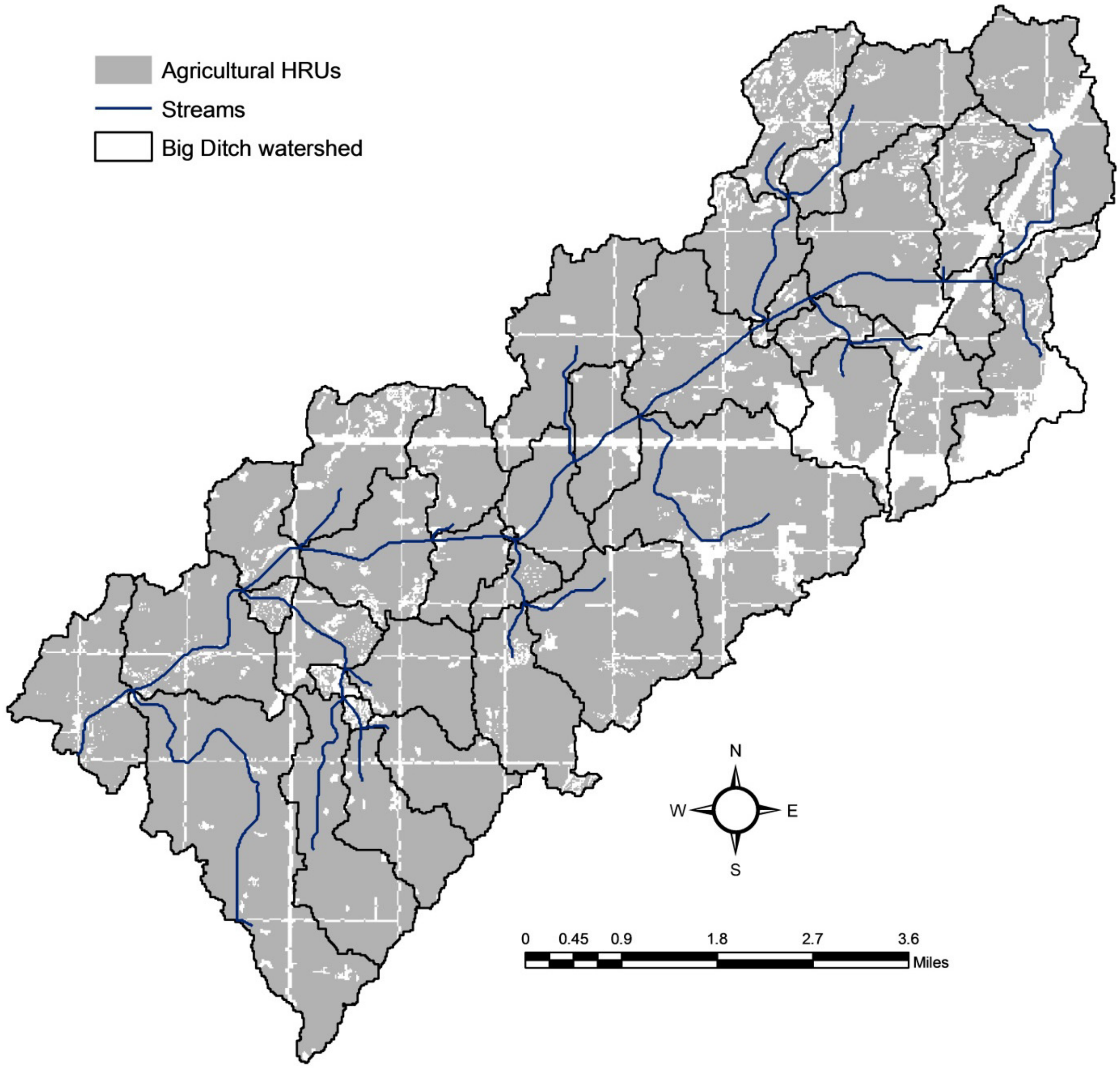


Figure A.1 Big Ditch HRUs suitable for nutrient management, cover crops, perennial crops, and filter strips

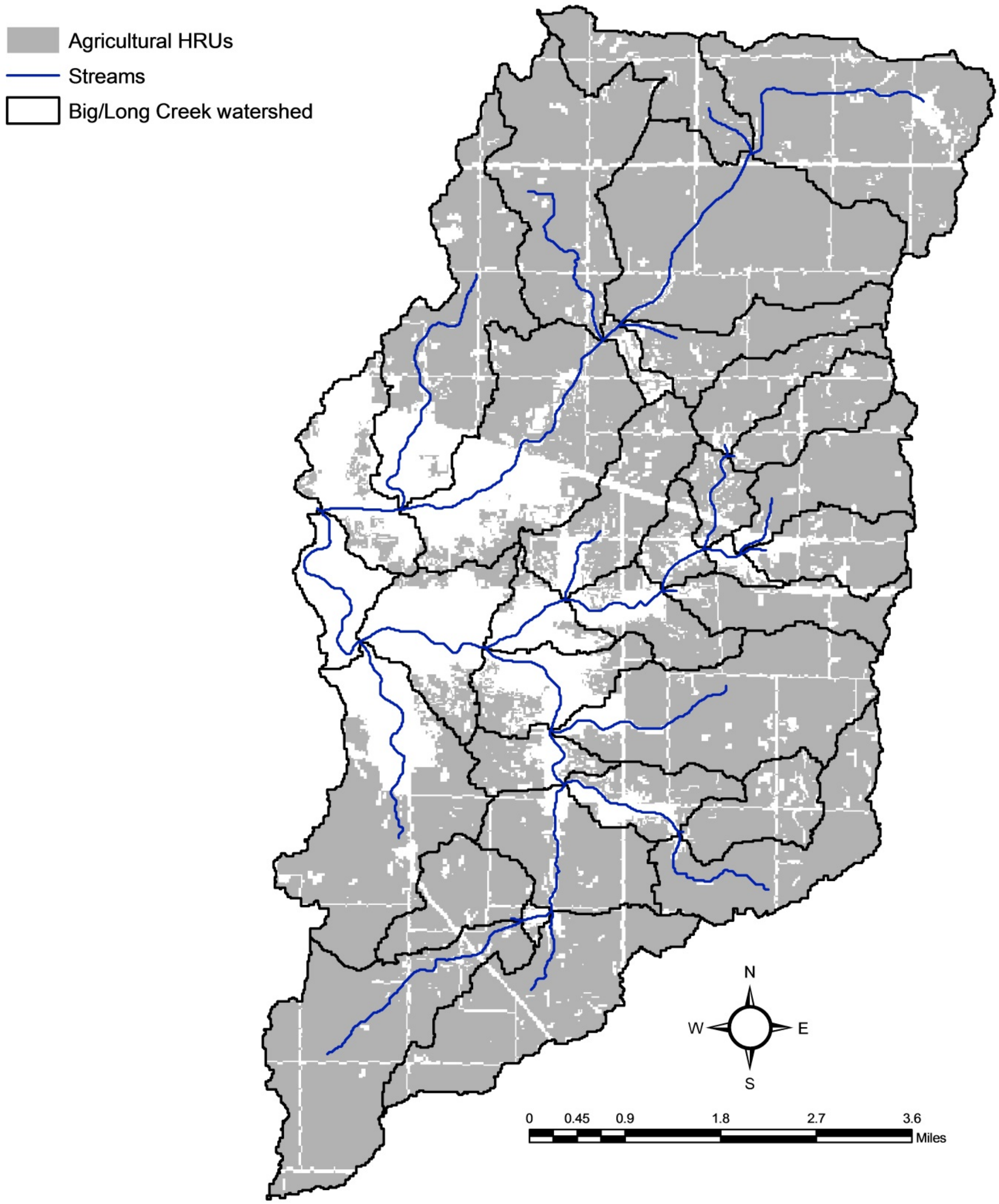


Figure A.2 Big/Long Creek HRUs suitable for nutrient management, cover crops, perennial crops, and filter strips

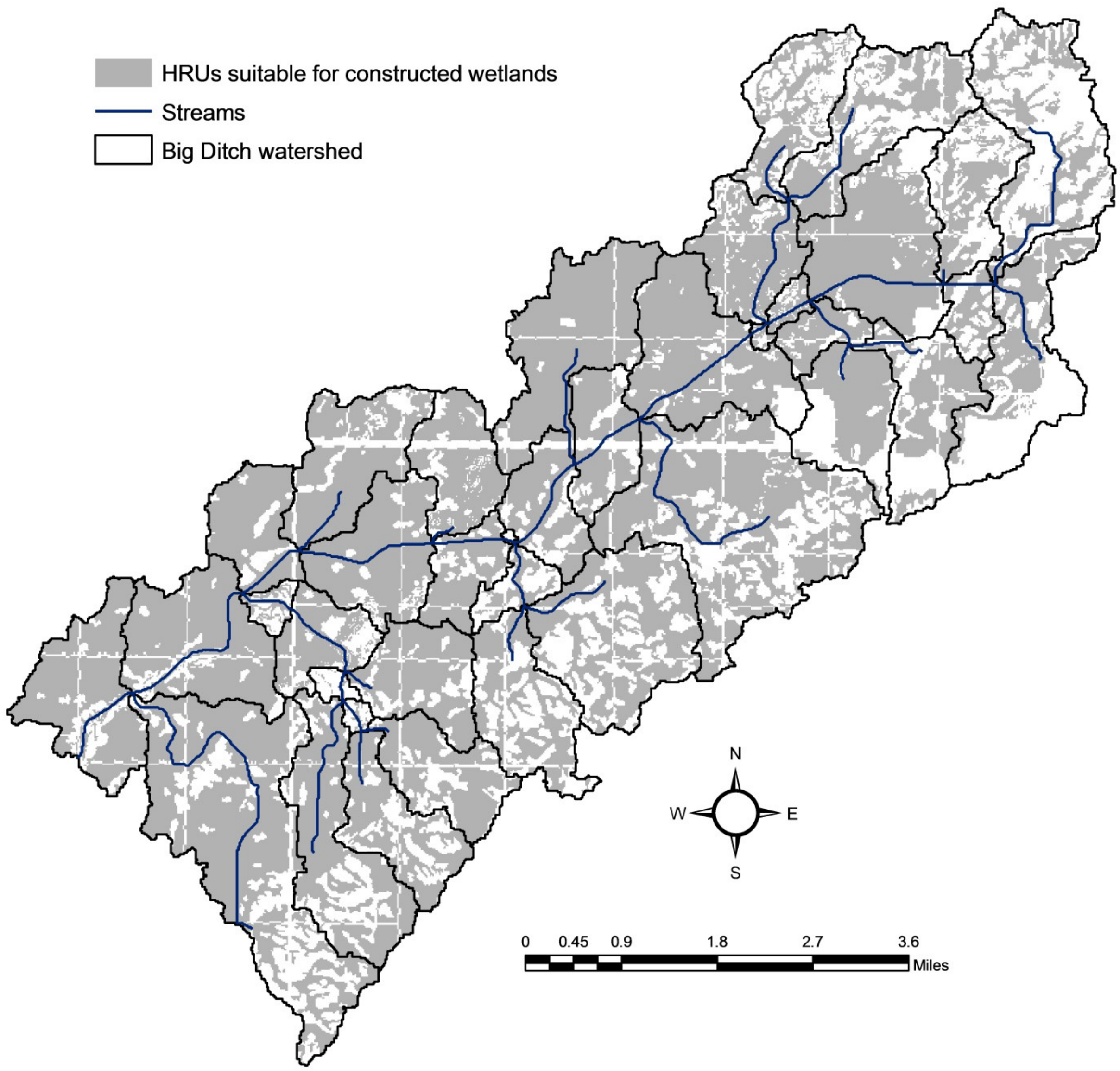


Figure A.3 Big Ditch HRUs suitable for constructed wetlands

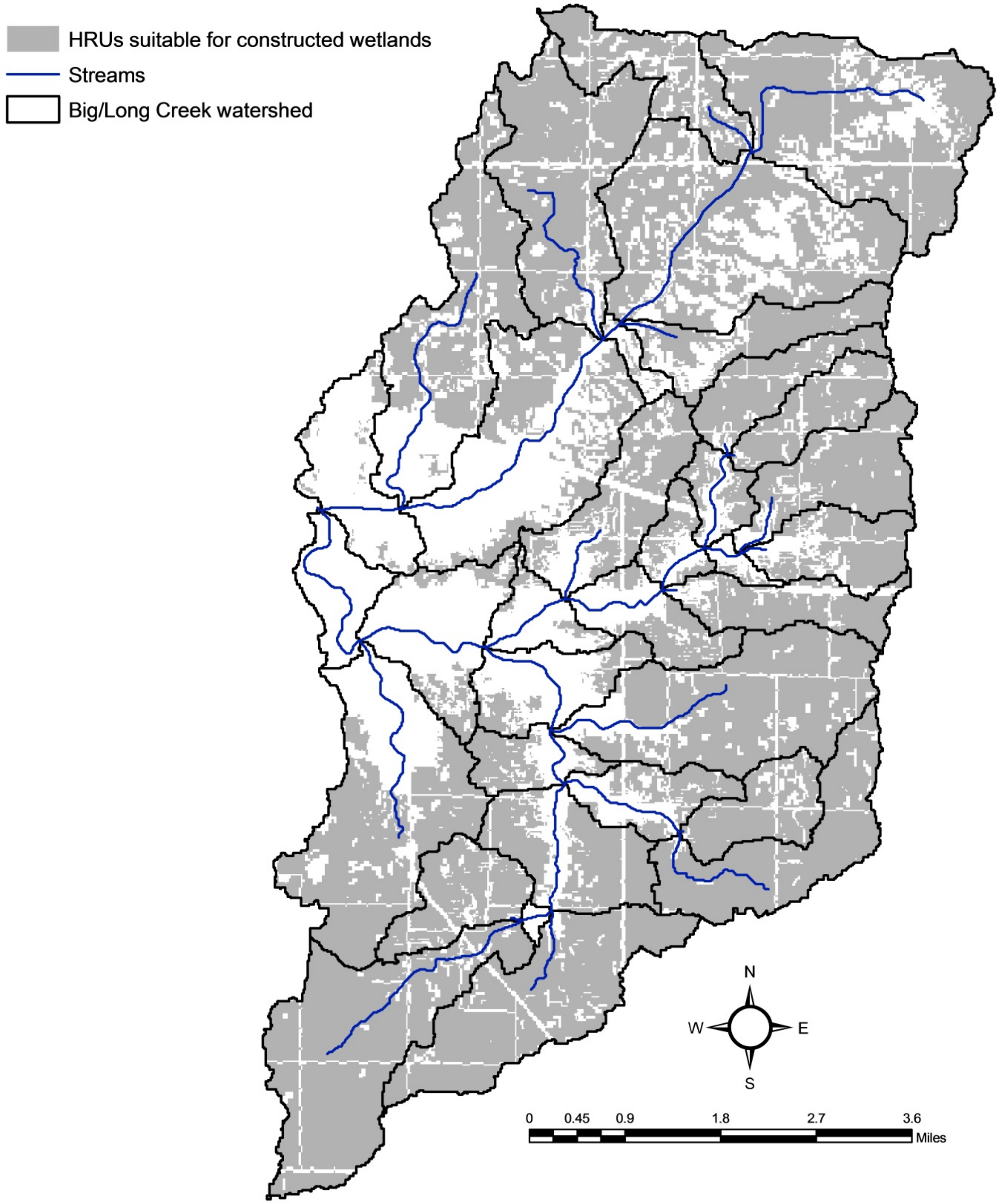


Figure A.4 Big/Long Creek HRUs suitable for constructed wetlands

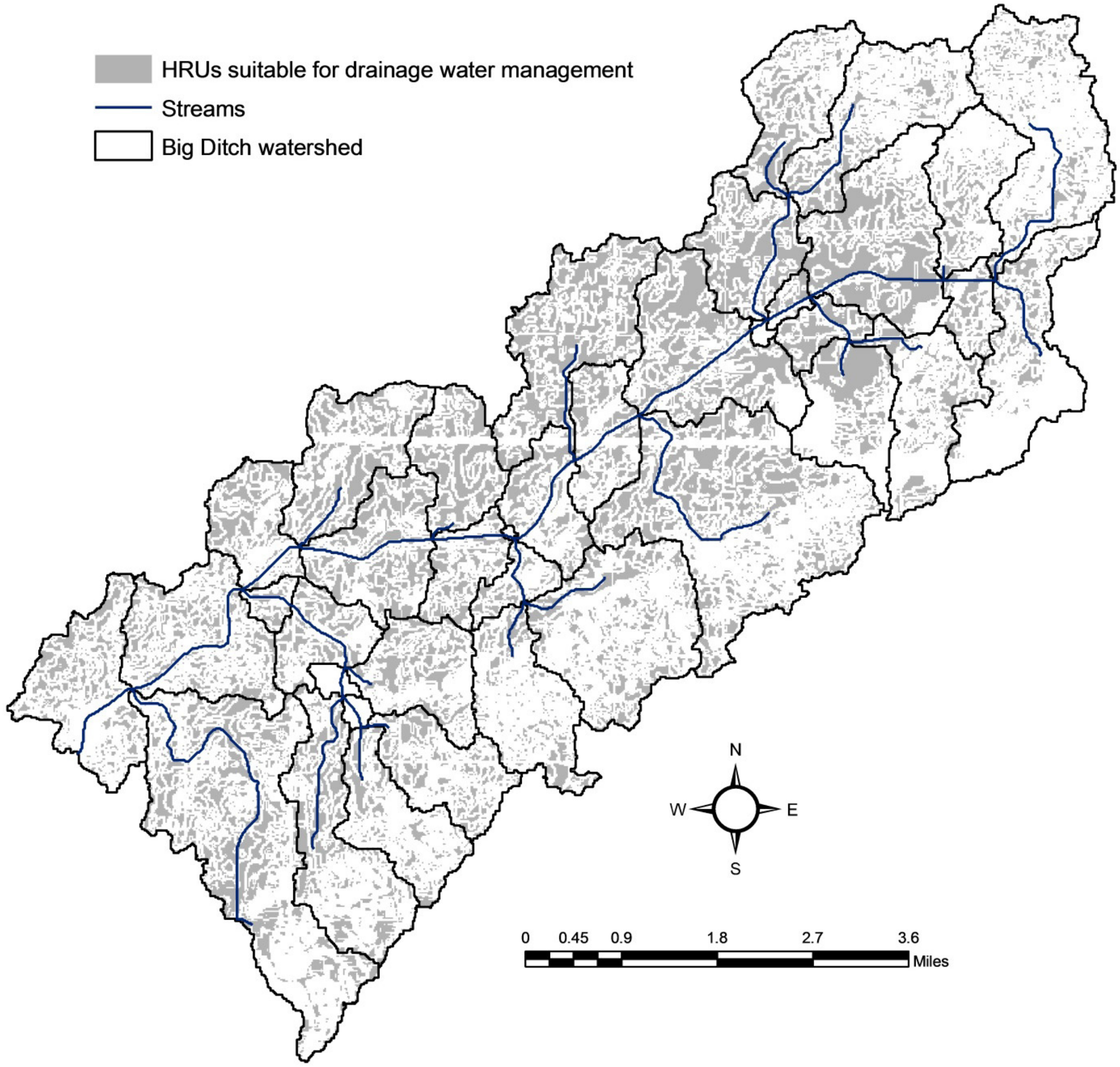


Figure A.5. Big Ditch HRUs suitable for drainage water management

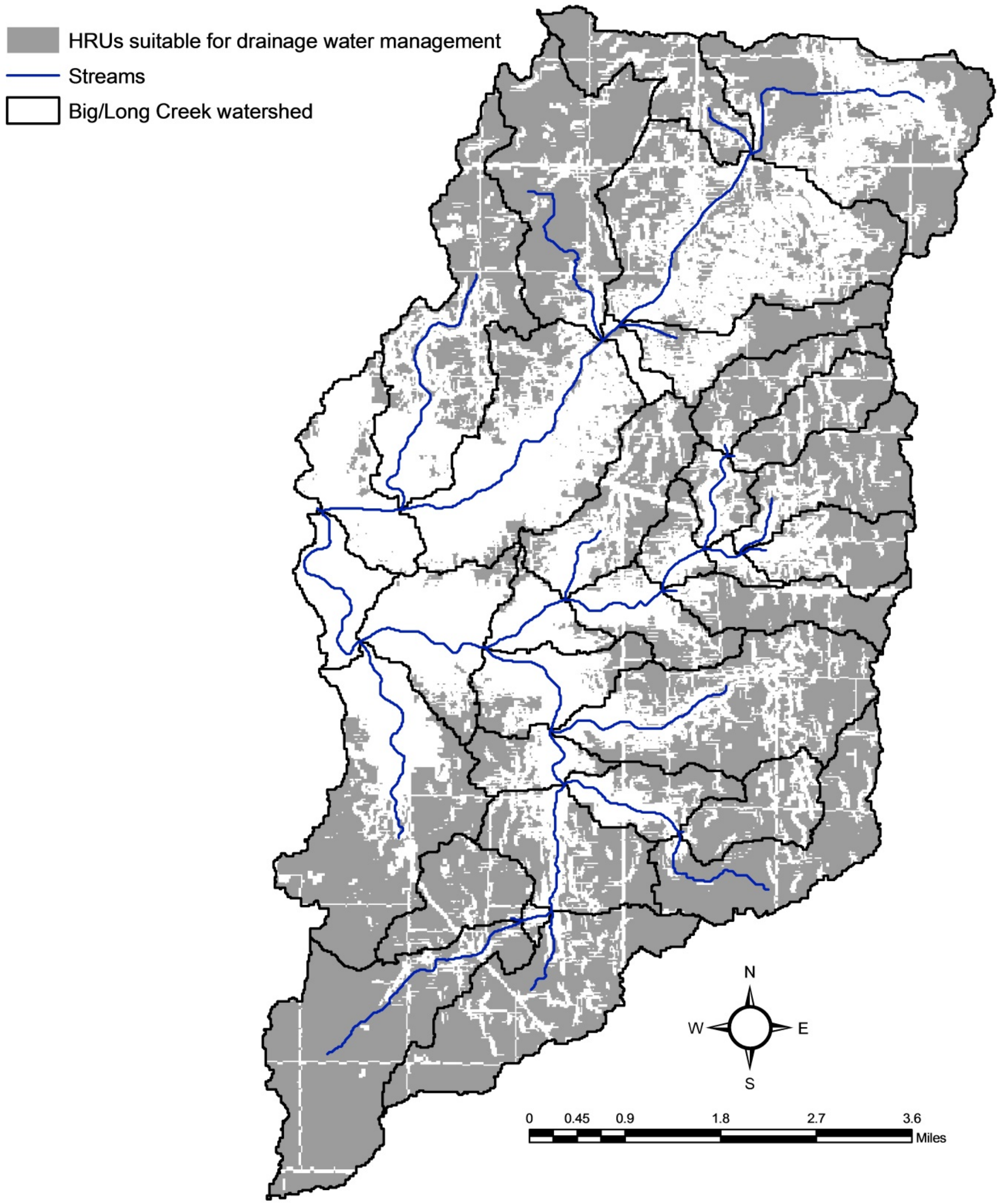


Figure A.6 Big/Long Creek HRUs suitable for drainage water management

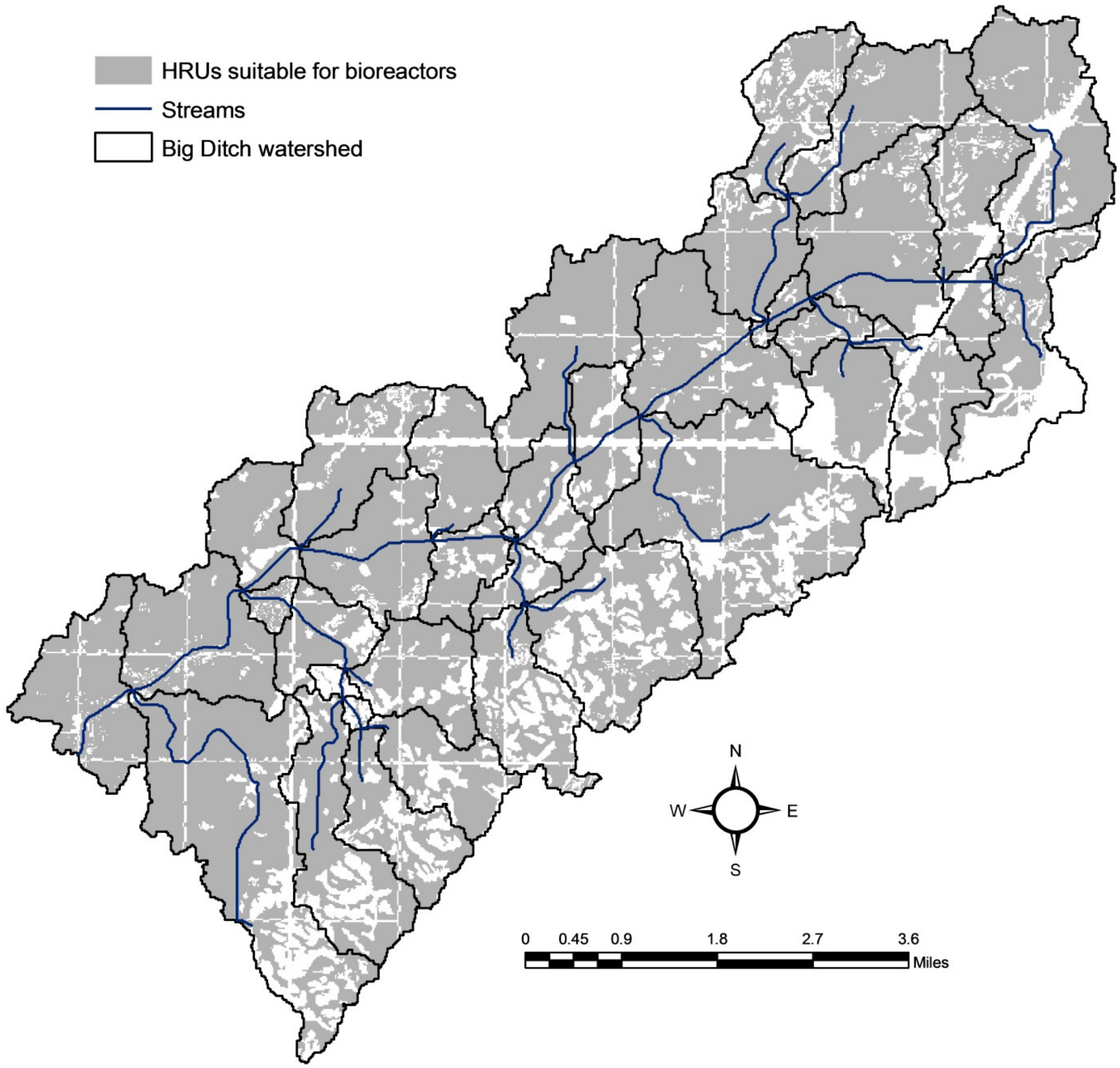


Figure A.7 Big Ditch HRUs suitable for bioreactors

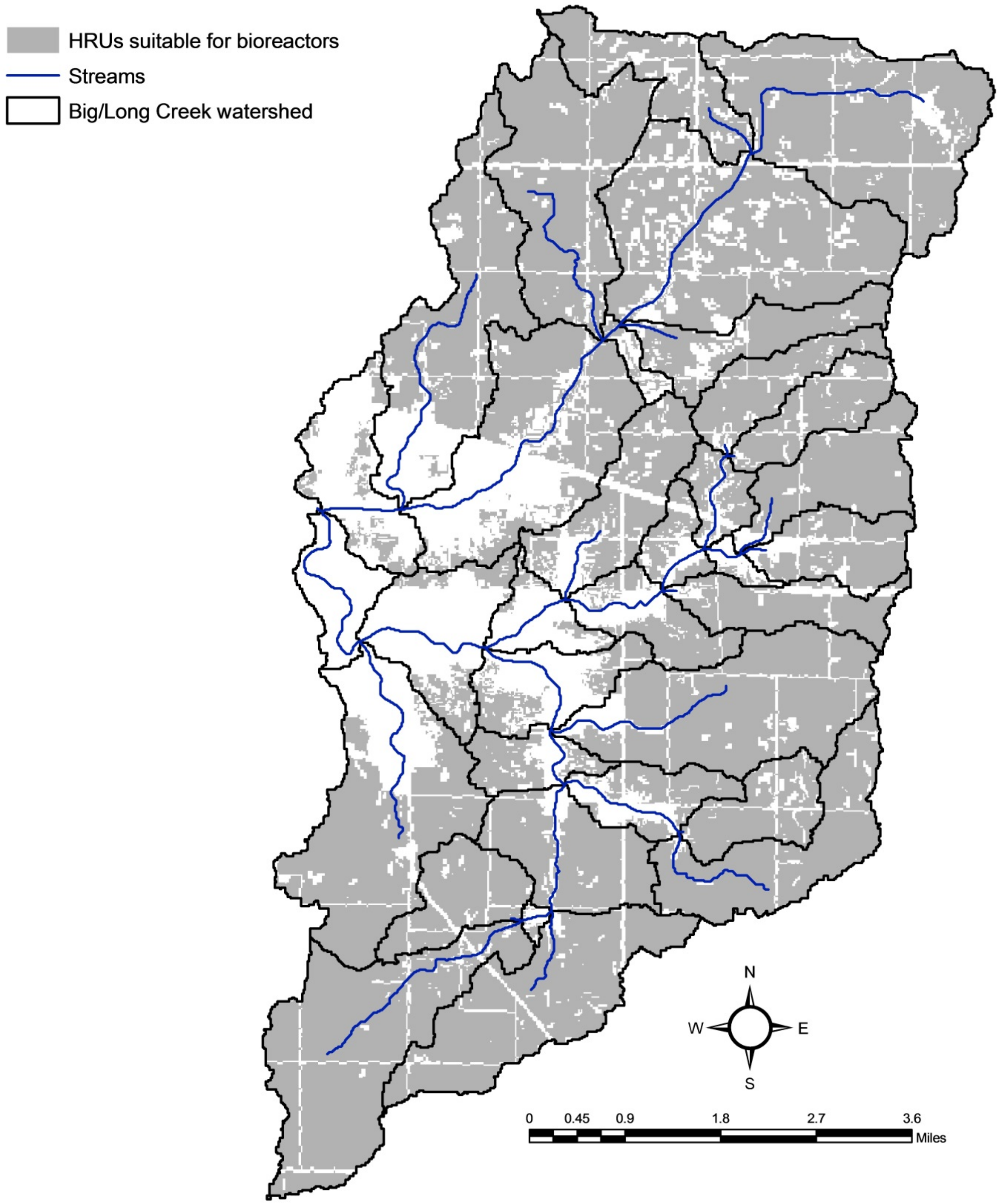


Figure A.8 Big/Long Creek HRUs suitable for bioreactors

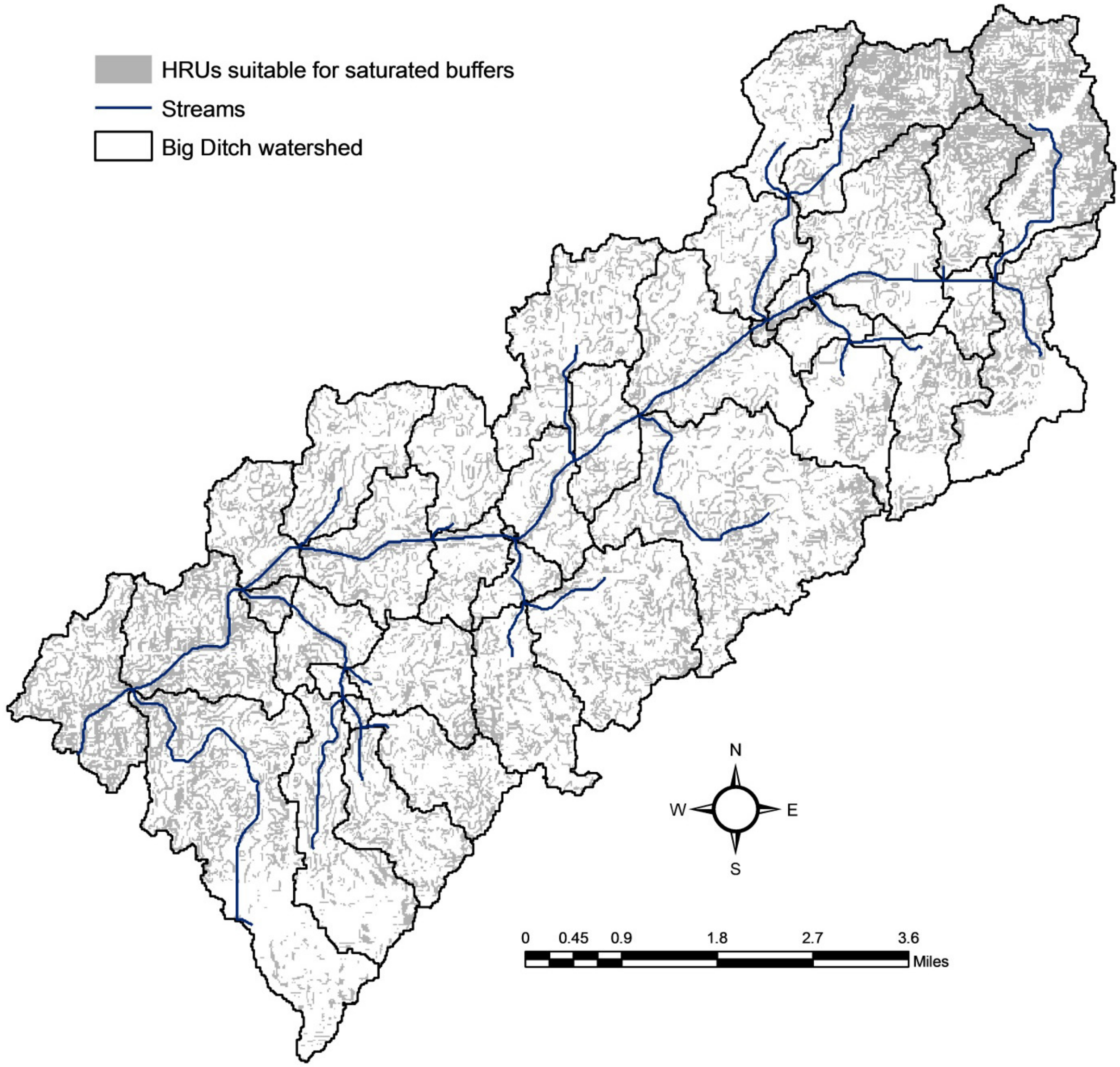


Figure A.9 Big Ditch HRUs suitable for saturated buffers

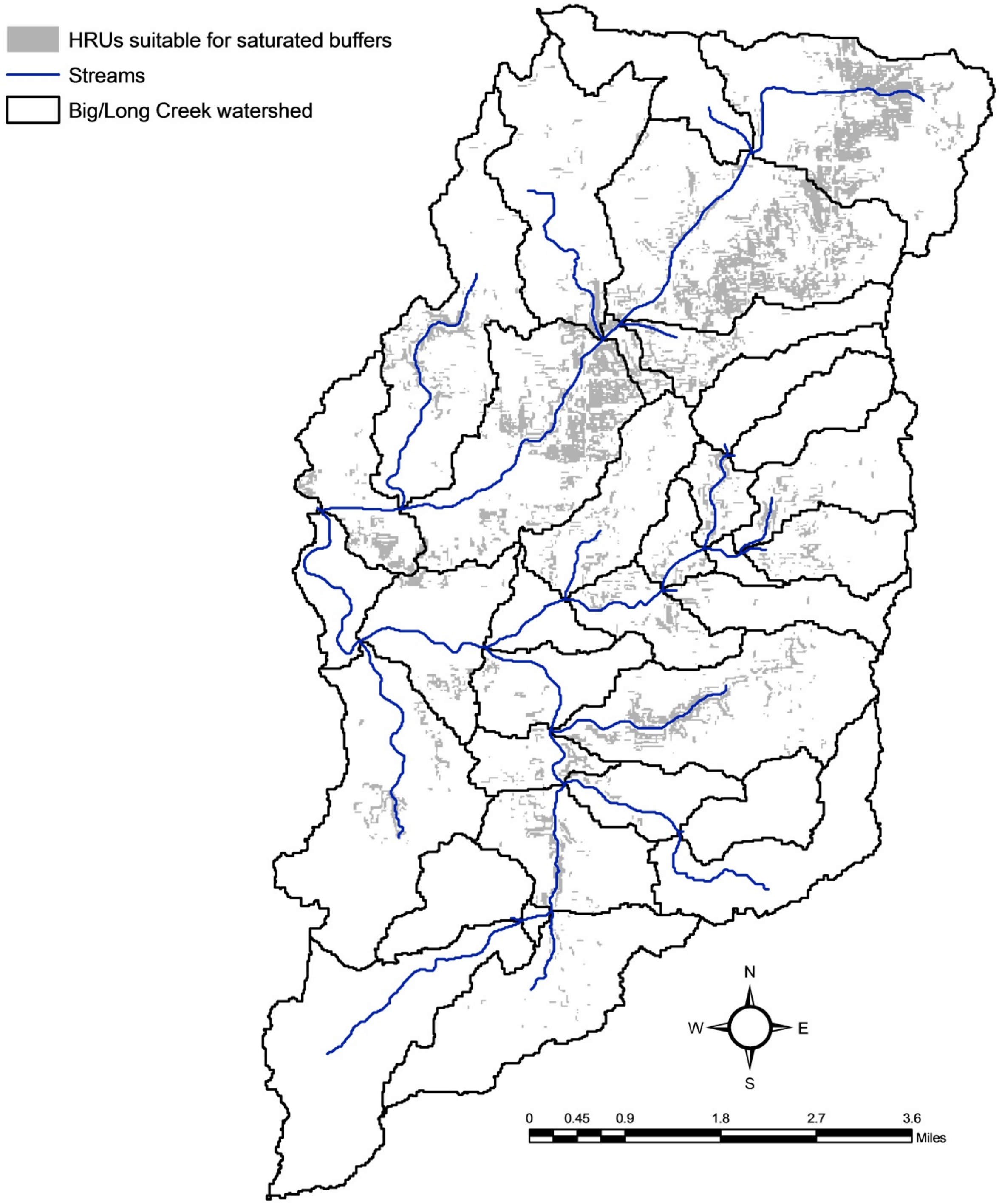


Figure A.10 Big/Long Creek HRUs suitable for saturated buffers

Appendix B. Optimal Placements of BMPs for Cost-Effective Reduction of NPS Pollutants in Big Ditch Watershed

Table B.1. Pollutant reduction for optimal placement of Cereal Rye in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 4.5	47.3	5.834	16.7	0.050	8.4	0.025	9.1	509,788
area 2 = 15.1		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 27.7		17.30		2,000		4,078		101

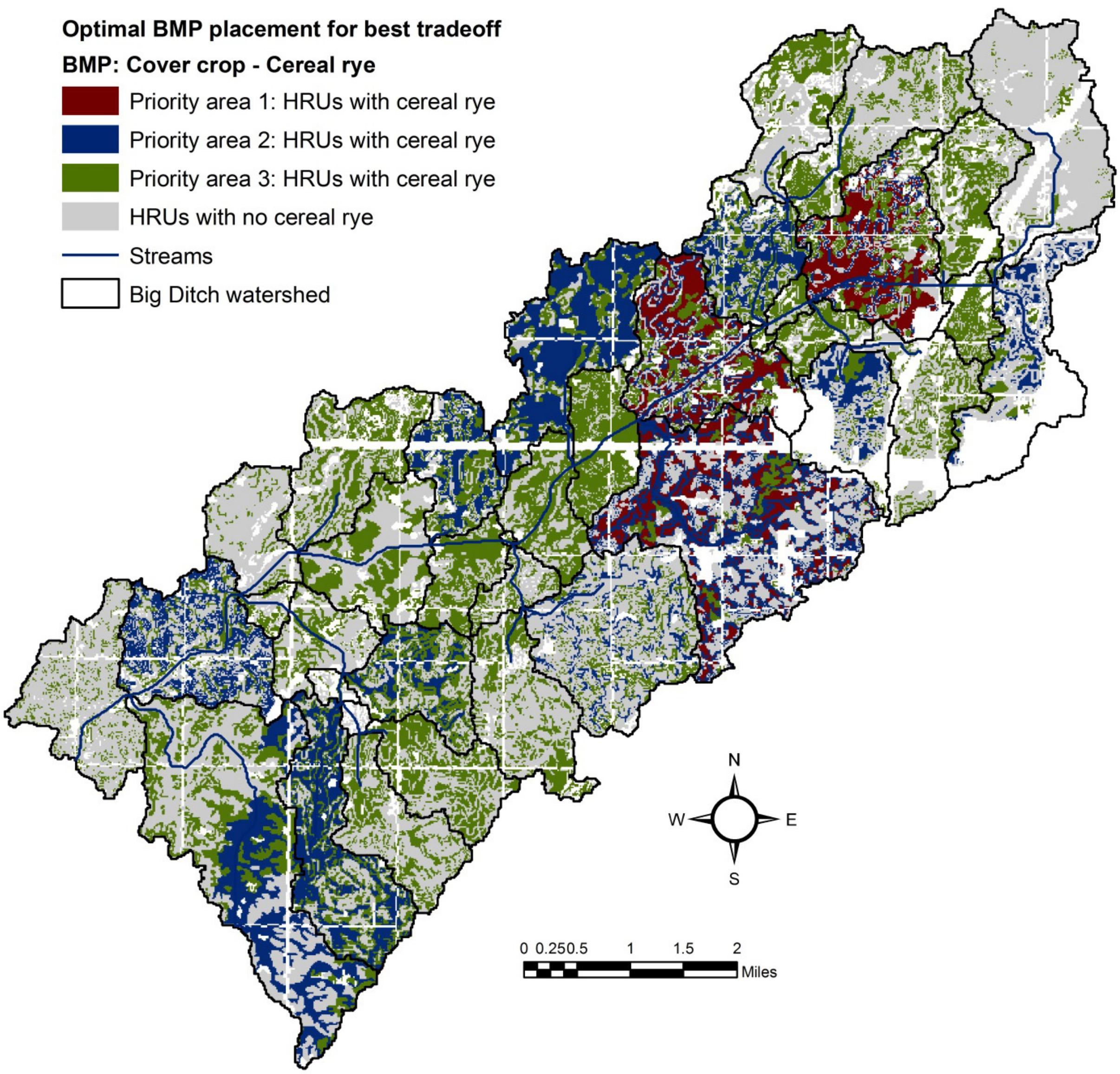


Figure B.1 Optimal placement of cereal rye in Big Ditch Watershed

Table B.2 Pollutant reduction for optimal placement of annual ryegrass in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 4.5	43.3	4.666	13.3	0.047	7.8	0.023	8.4	423,172
area 2 = 14.7		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 25.5		19.59		1,956		3,982		91

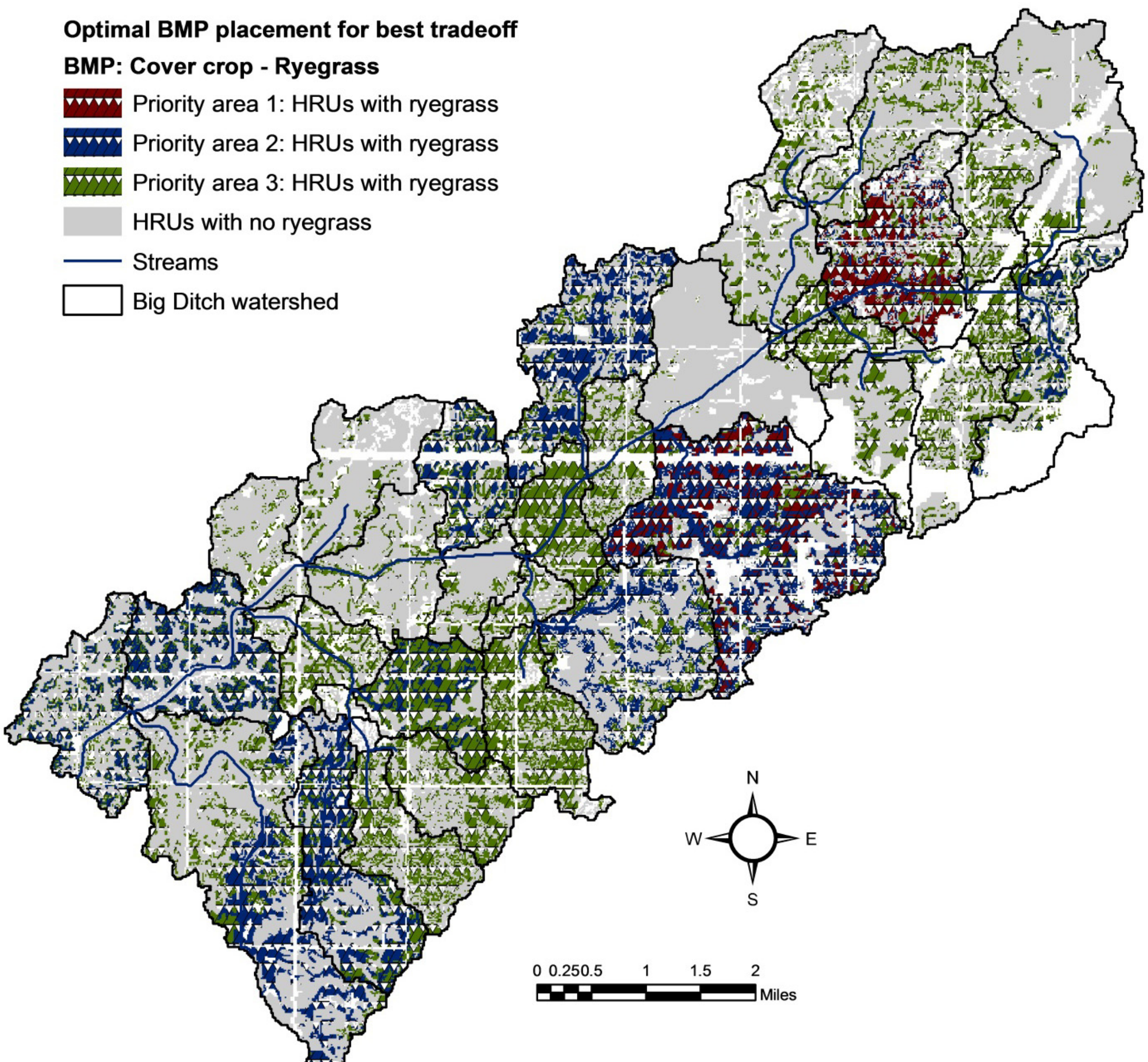


Figure B.2 Optimal placement of annual ryegrass in Big Ditch Watershed

Table B.3 Pollutant reduction for optimal placement of crimson clover in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 4.5	44.6	3.707	10.6	0.052	8.7	0.023	8.4	684,736
area 2 = 15.6		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 24.5		38.74		2,739		6,252		144

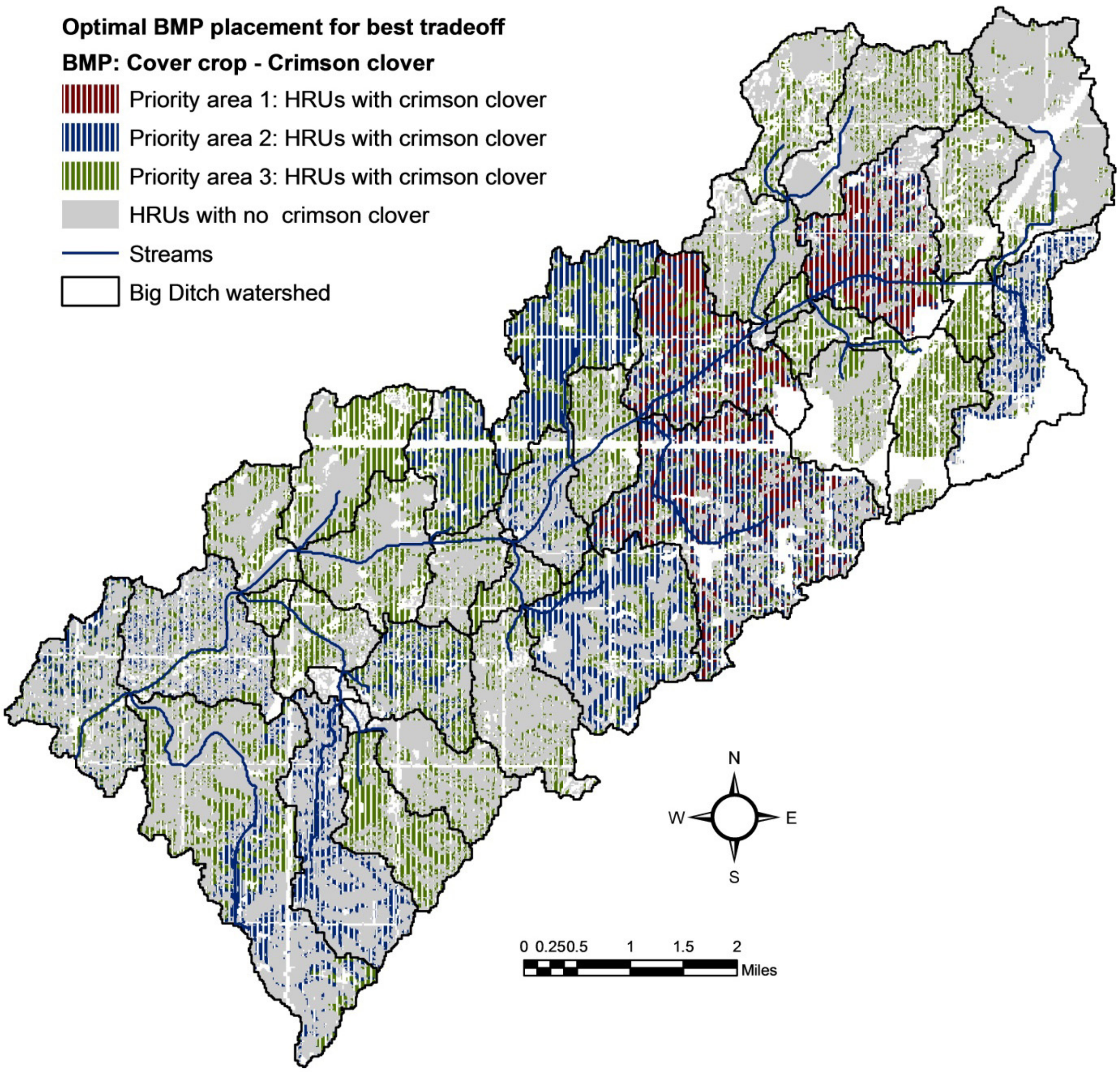


Figure B.3 Optimal placement of crimson clover in Big Ditch Watershed

Table B.4 Pollutant reduction for optimal placement of alfalfa in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 4.5	45.4	16.349	46.8	0.124	20.6	0.042	15.4	4,176,638
area 2 = 15.6		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 25.5		52.61		6,943		20,430		860

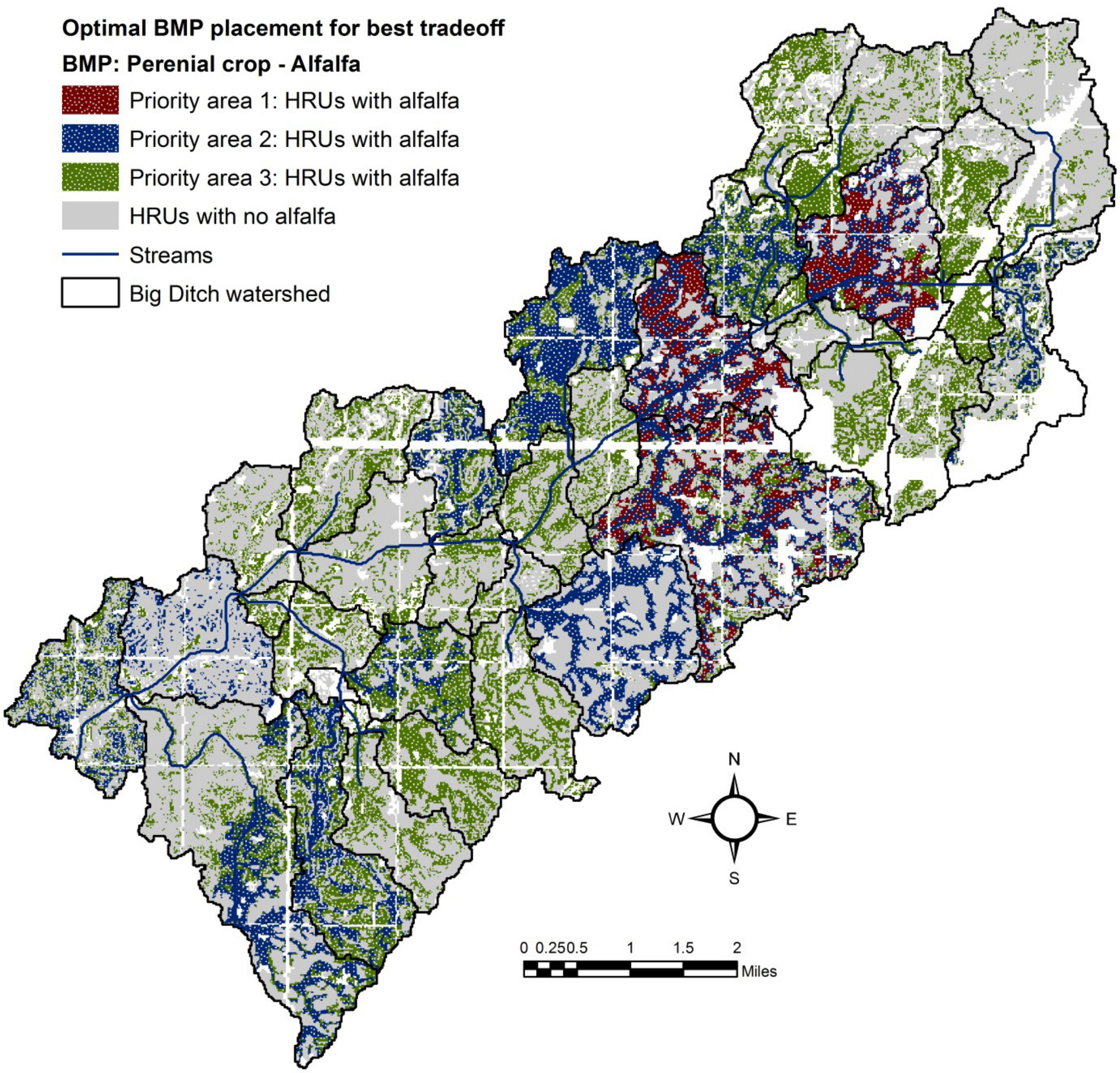


Figure B.4 Optimal placement of alfalfa in Big Ditch Watershed

Table B.5 Pollutant reduction for optimal placement of constructed wetlands in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 2.2	18.6	5.987	17.1	0.043	7.2	0.014	5.0	109,426
area 2 = 7.5		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 8.9		9.18		1,279		3,997		55

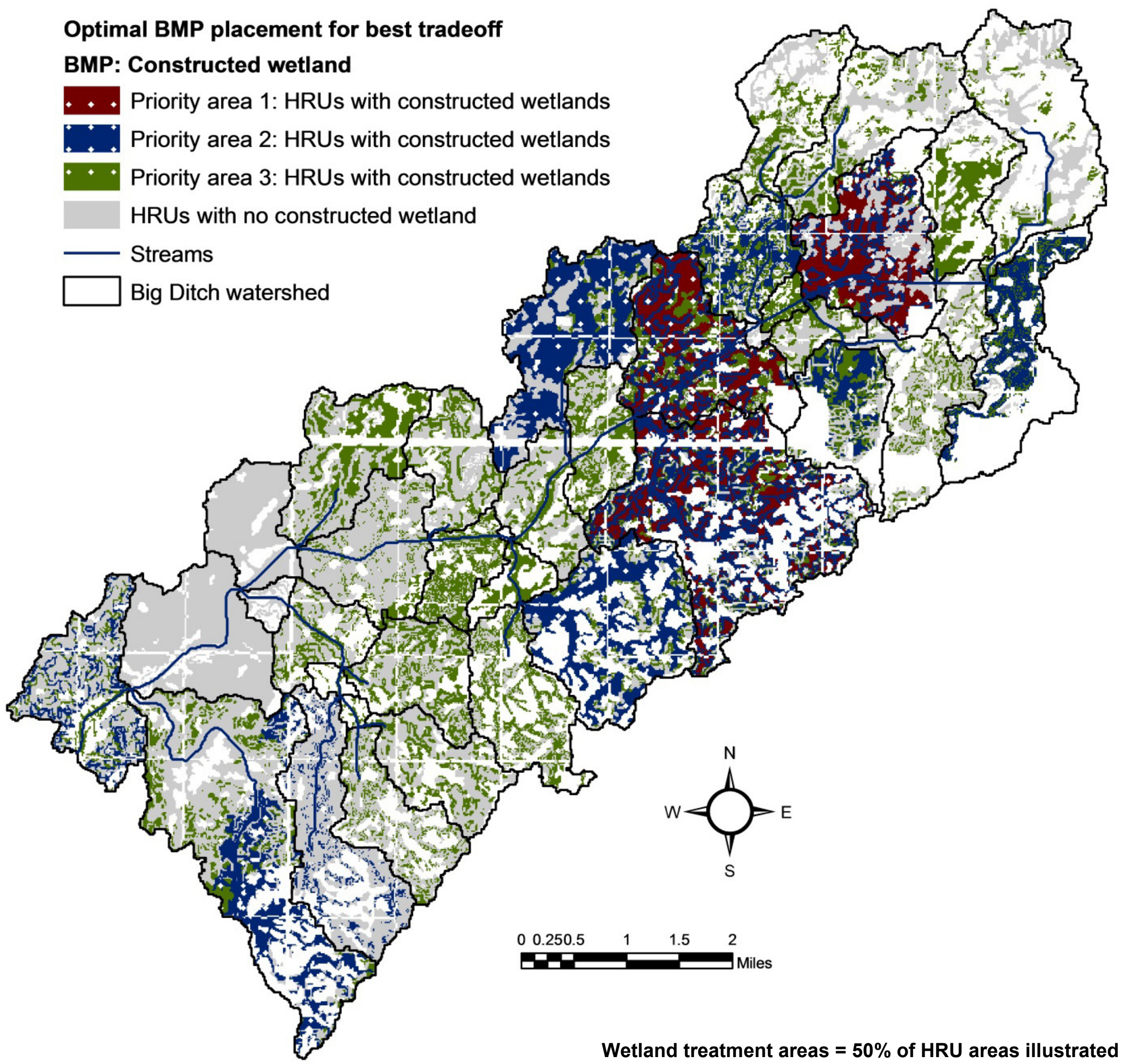


Figure B.5 Optimal placement of constructed wetlands in Big Ditch Watershed

Table B.6 Pollutant reduction for optimal placement of DWM in Big Ditch Watershed

<table>
<tr><td colspan="2">BMP treament area
[% of watershed area]</td><td colspan="6">Pollutant load Reduction</td><td rowspan="2">Equal annual
cost (EAC)</td></tr>
<tr><td colspan="2"></td><td colspan="2">Nitrate-N</td><td colspan="2">Total phosphorus</td><td colspan="2">Sediment</td></tr>
<tr><td>Priority</td><td>Total</td><td>[kg/ha/yr]</td><td>[%]</td><td>[kg/ha/yr]</td><td>[%]</td><td>[t/ha/yr]</td><td>[%]</td><td>[$/yr]</td></tr>
<tr><td>area 1 = 2.9</td><td rowspan="3">7.8</td><td>3.216</td><td>9.2</td><td>n/a</td><td>n/a</td><td>n/a</td><td>n/a</td><td>25,629</td></tr>
<tr><td>area 2 = 1.9</td><td colspan="2">EAC [$/kg/ha]</td><td colspan="2">EAC [$/kg/ha]</td><td colspan="2">EAC [$/t/ha]</td><td>$/ha/yr</td></tr>
<tr><td>area 3 = 3.0</td><td colspan="2">9.62</td><td>n/a</td><td>n/a</td><td>n/a</td><td>n/a</td><td>329</td></tr>
</table>

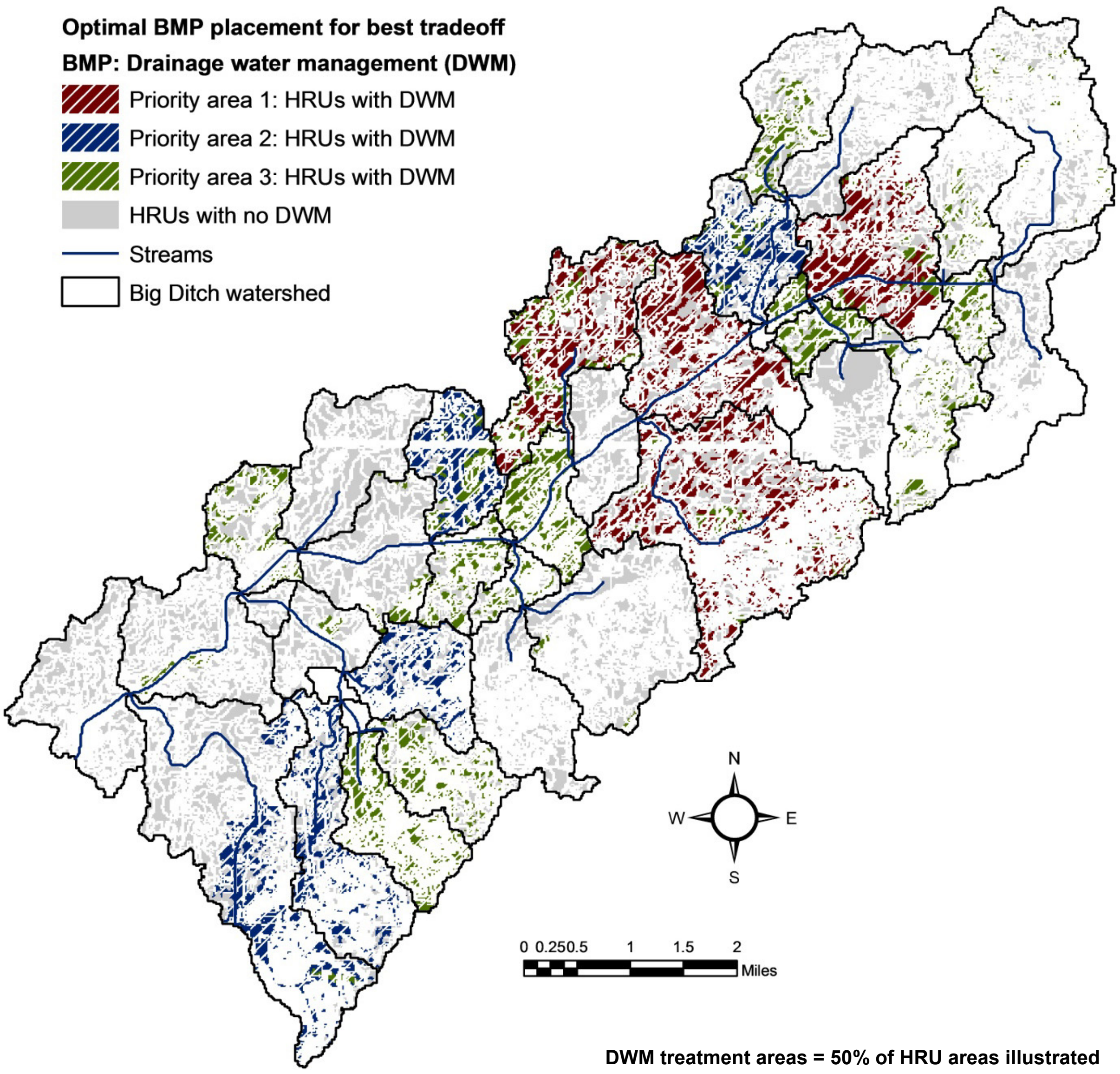


Figure B.6 Optimal placement of drainage water management in Big Ditch Watershed

Table B.7 Pollutant reduction for optimal placement of bioreactors in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 5.8	39.8	14.623	41.8	n/a	n/a	n/a	n/a	108,195
area 2 = 12.2		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 21.8		1.74		n/a	n/a	n/a	n/a	25

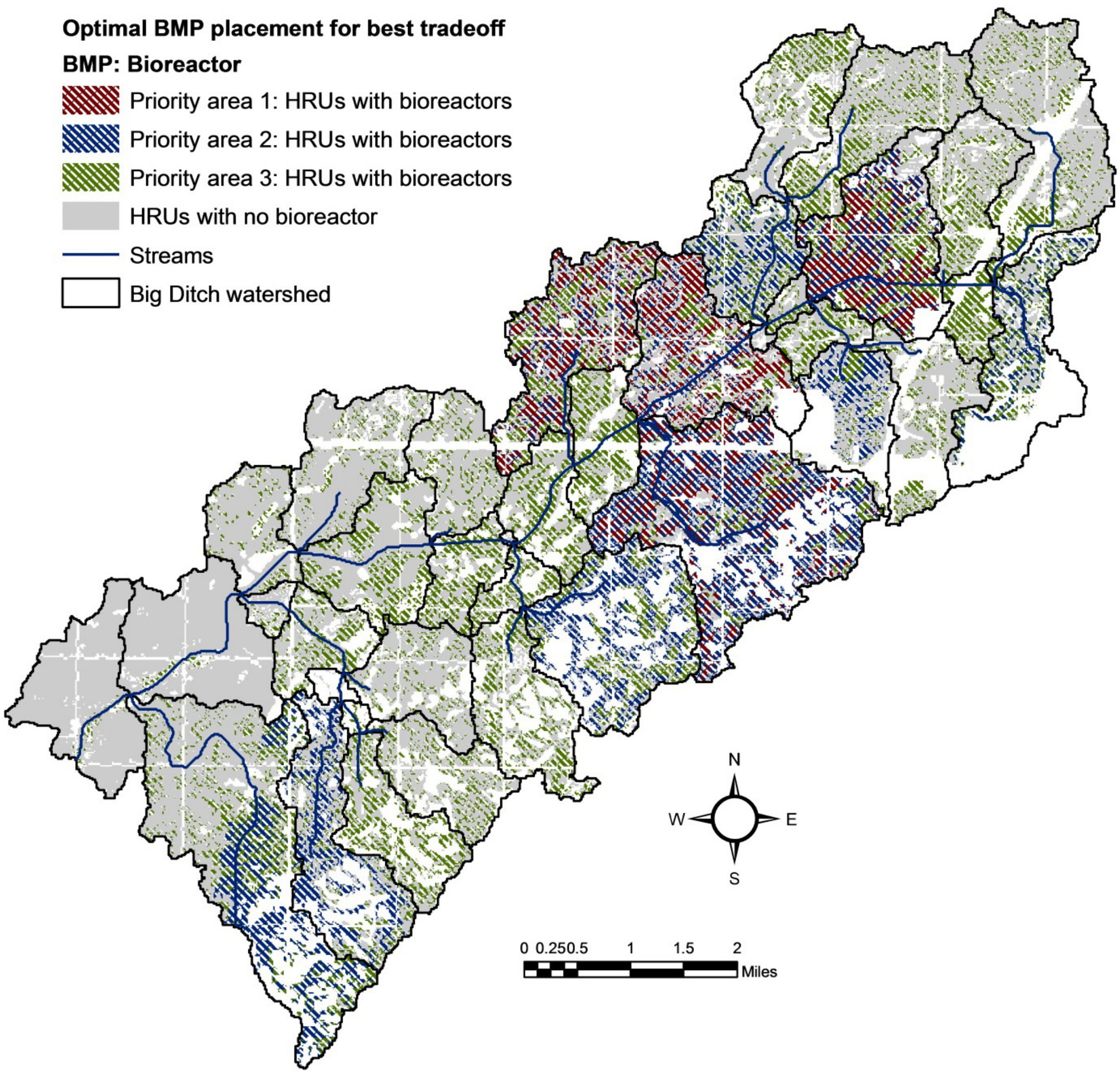


Figure B.7 Optimal placement of bioreactors in Big Ditch Watershed

Table B.8 Pollutant reduction for optimal placement of saturated buffers in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 0.6	5.9	2.095	6.0	n/a	n/a	n/a	n/a	17,175
area 2 = 2.6		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 2.7		13.00		n/a	n/a	n/a	n/a	27

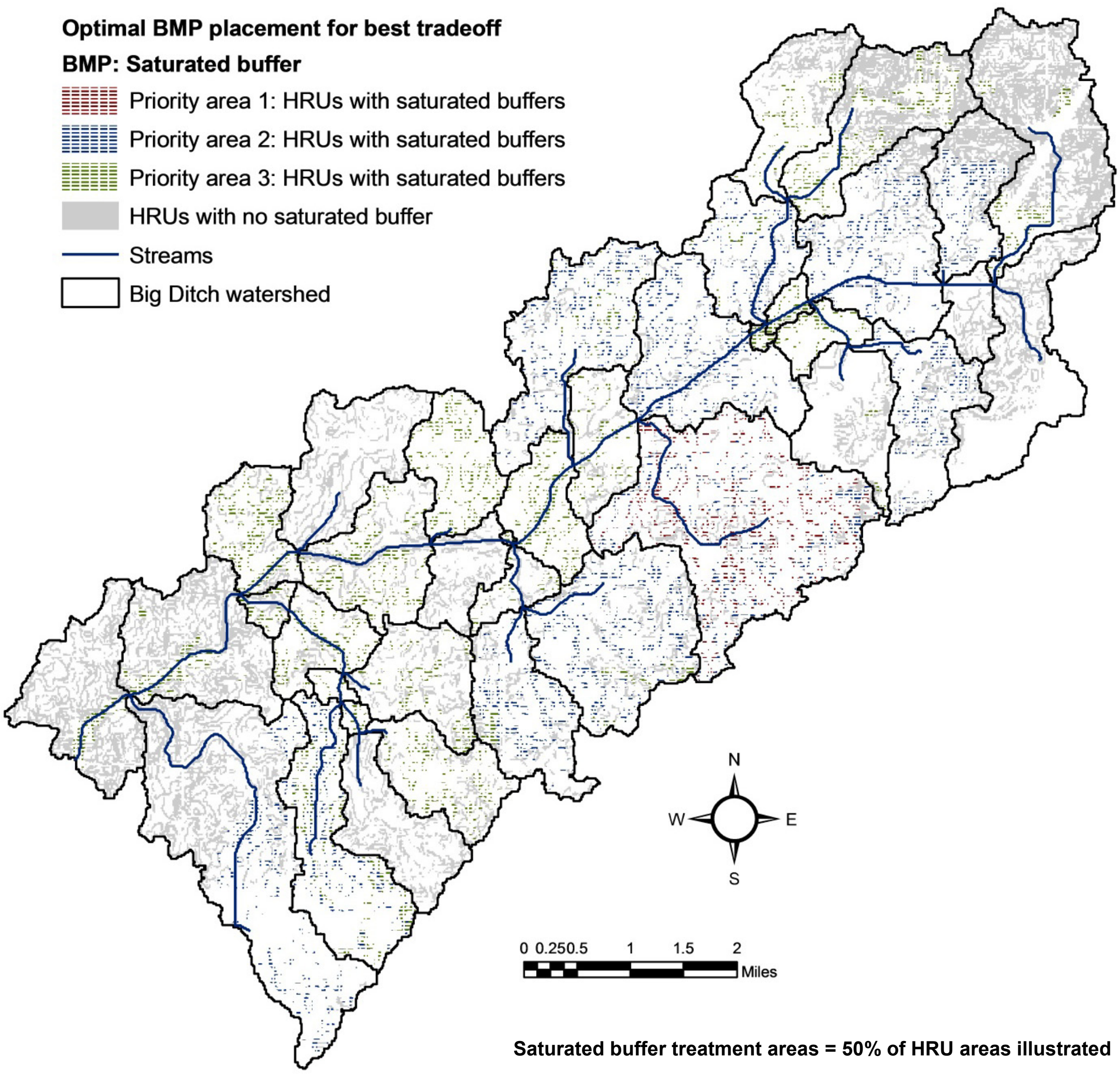


Figure B.8 Optimal placement of saturated buffers in Big Ditch Watershed

Table B.9 Pollutant reduction for optimal placement of filter strips in Big Ditch Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 0.8	42.0	3.460	9.9	0.155	25.8	0.059	21.5	25,622
area 2 = 18.9		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 22.3		1.65		36.88		97.29		6

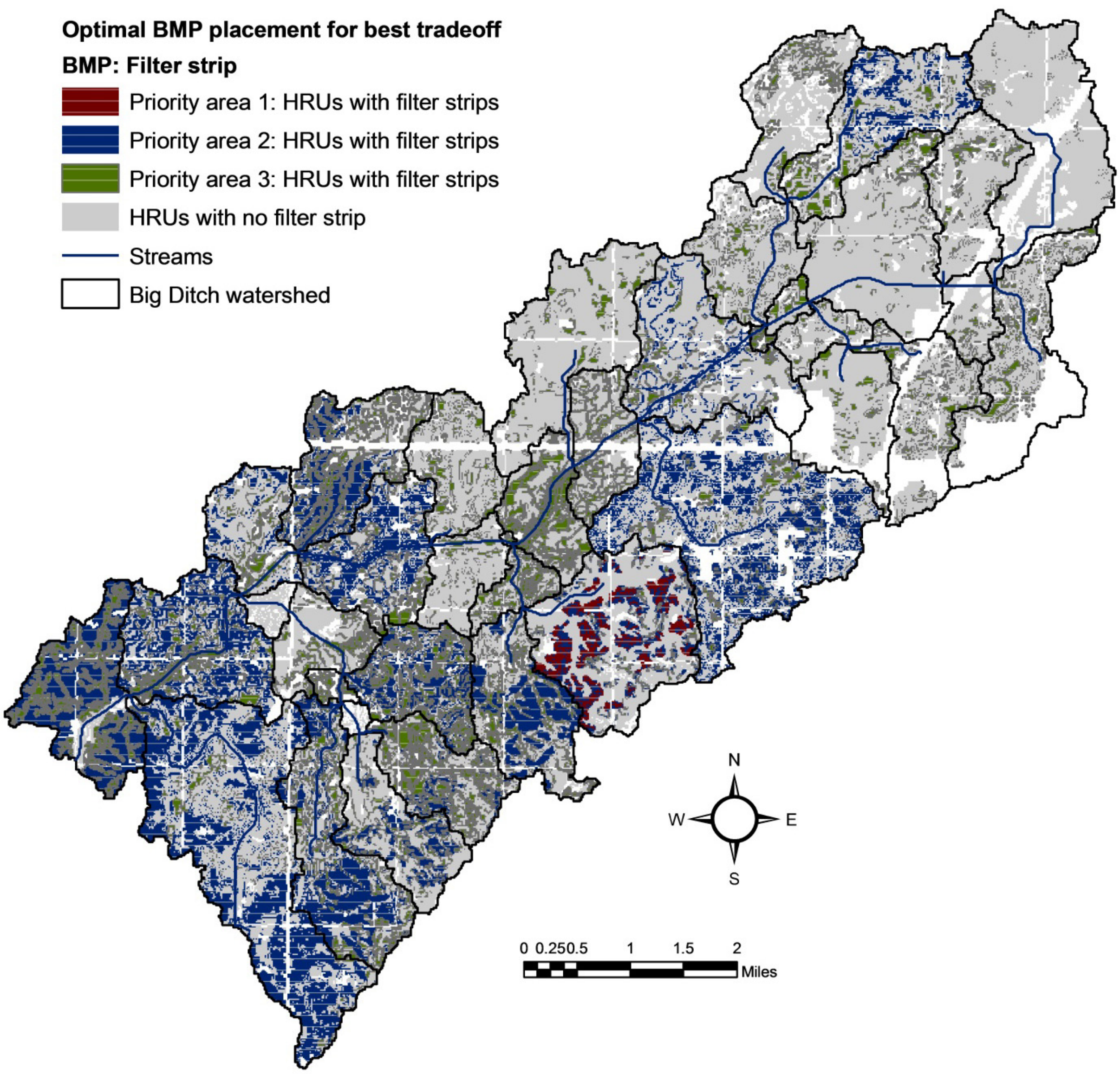


Figure B.9 Optimal placement of filter strips in Big Ditch Watershed

Appendix C. Optimal Placements of BMPs for Cost-Effective Reduction of NPS Pollutants in Big/Long Creek Watershed

Table C.1. Pollutant reduction for optimal placement of Cereal Rye in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 12.2	41.2	5.295	15.4	0.073	6.9	0.039	4.6	549,874
area 2 = 10.8		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 18.2		20.43		1,491		2,749		108

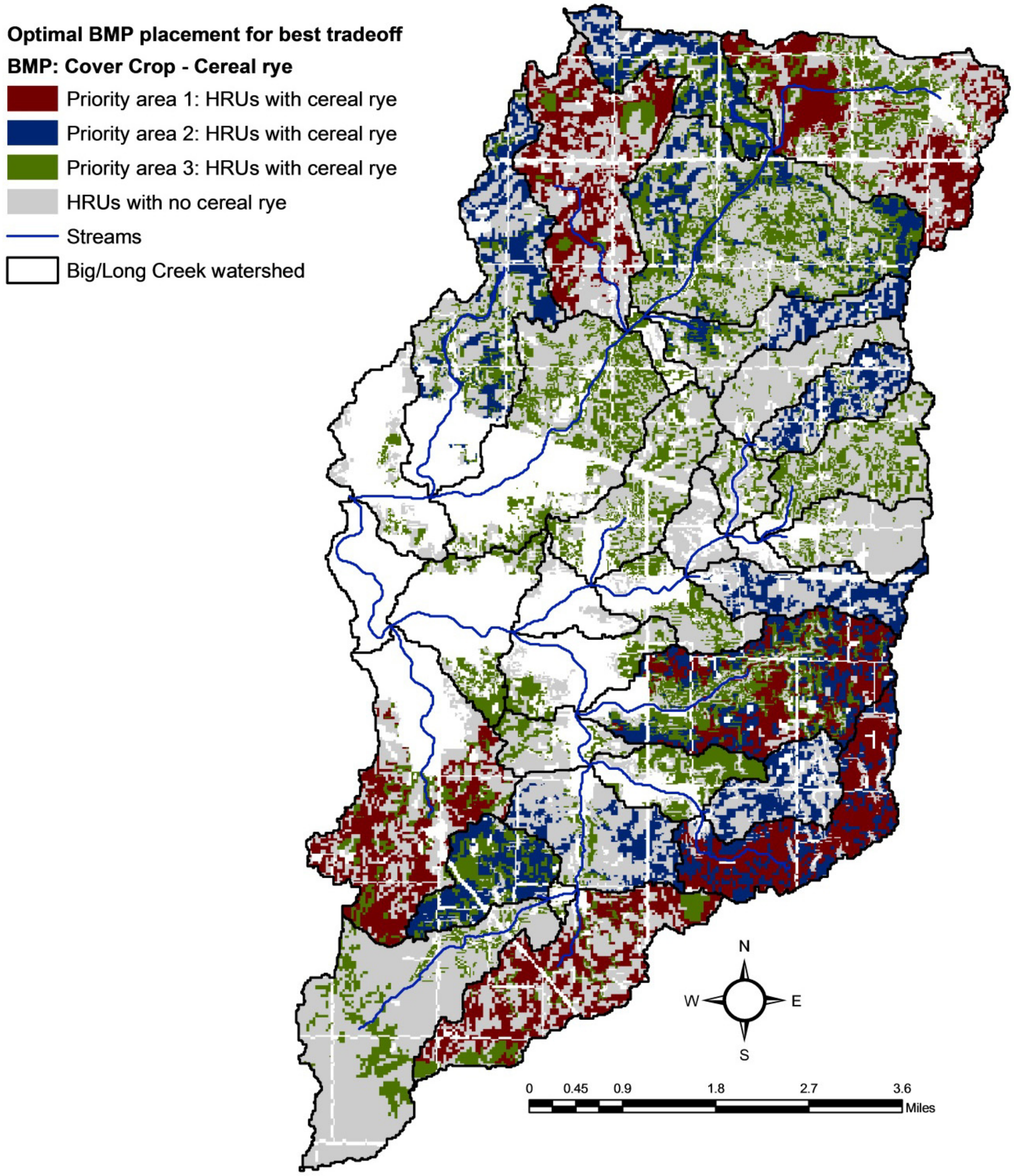


Figure C.1 Optimal placement of cereal rye in Big/Long Creek Watershed

Table C.2 Pollutant reduction for optimal placement of ryegrass in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 7.0	39.3	4.419	12.9	0.064	6.1	0.037	4.4	444,571
area 2 = 14.3		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 18.0		20.78		1,436		2,465		92

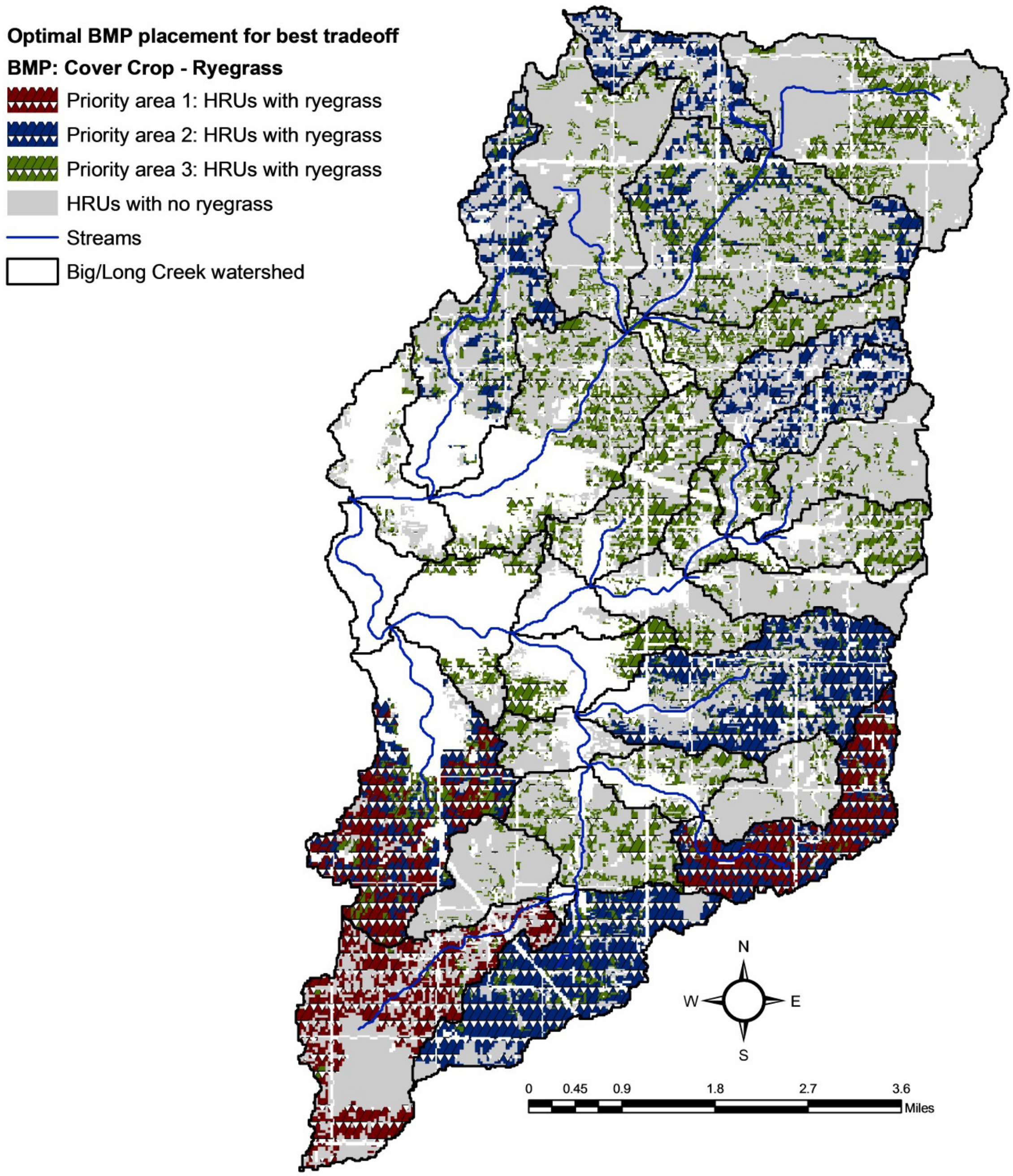


Figure C.2 Optimal placement of annual ryegrass in Big/Long Creek Watershed

Table C.3 Pollutant reduction for optimal placement of crimson clover in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 10.1	37.8	2.691	7.8	0.060	5.7	0.039	4.6	673,021
area 2 = 12.4		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 15.3		53.61		2,402		3,690		144

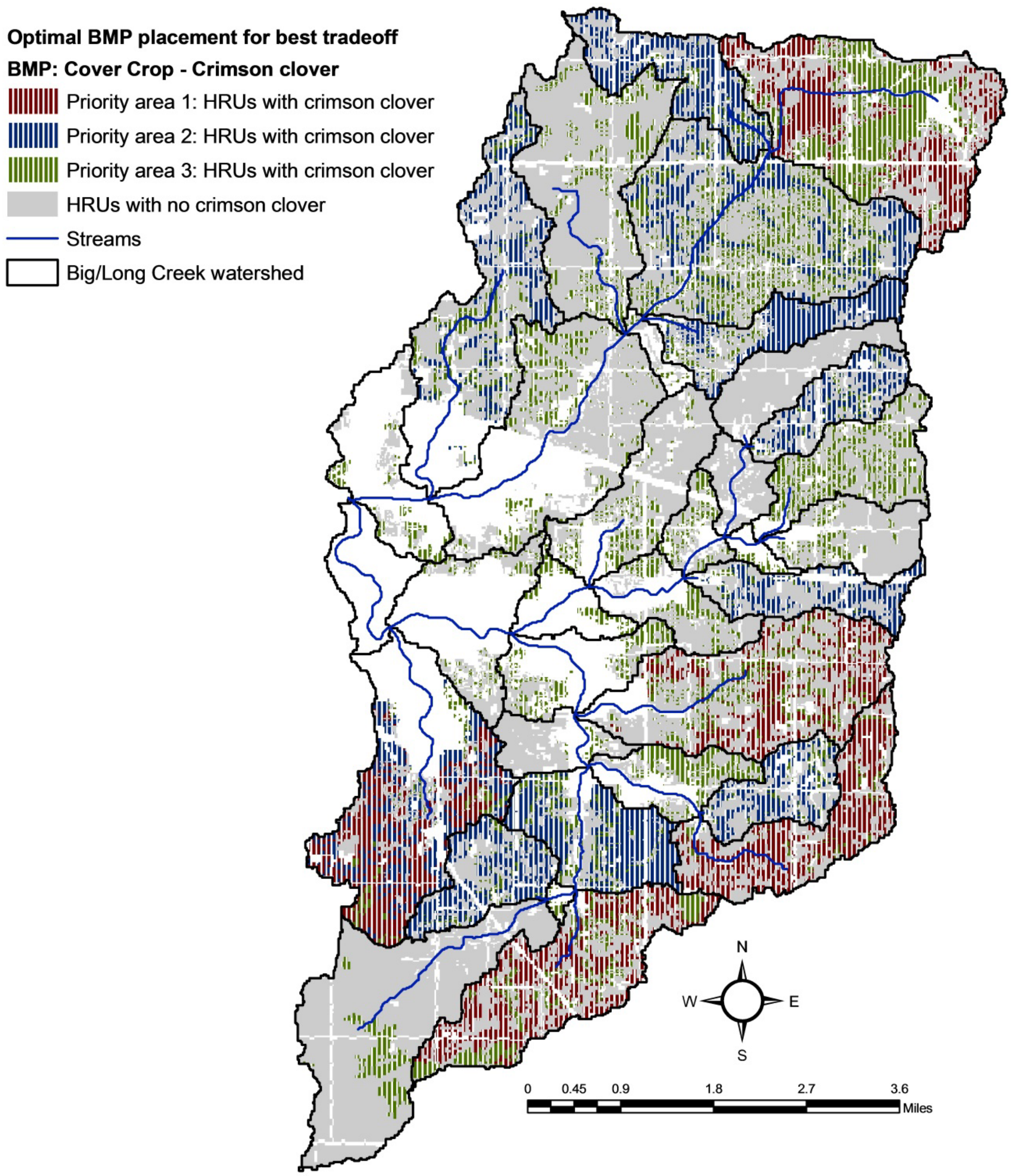


Figure C.3 Optimal placement of crimson clover in Big/Long Creek Watershed

Table C.4 Pollutant reduction for optimal placement of alfalfa in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 11.3	38.9	12.826	37.4	0.119	11.3	0.048	5.6	5,352,245
area 2 = 13.7		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 13.9		86.93		9,405		23,326		1115

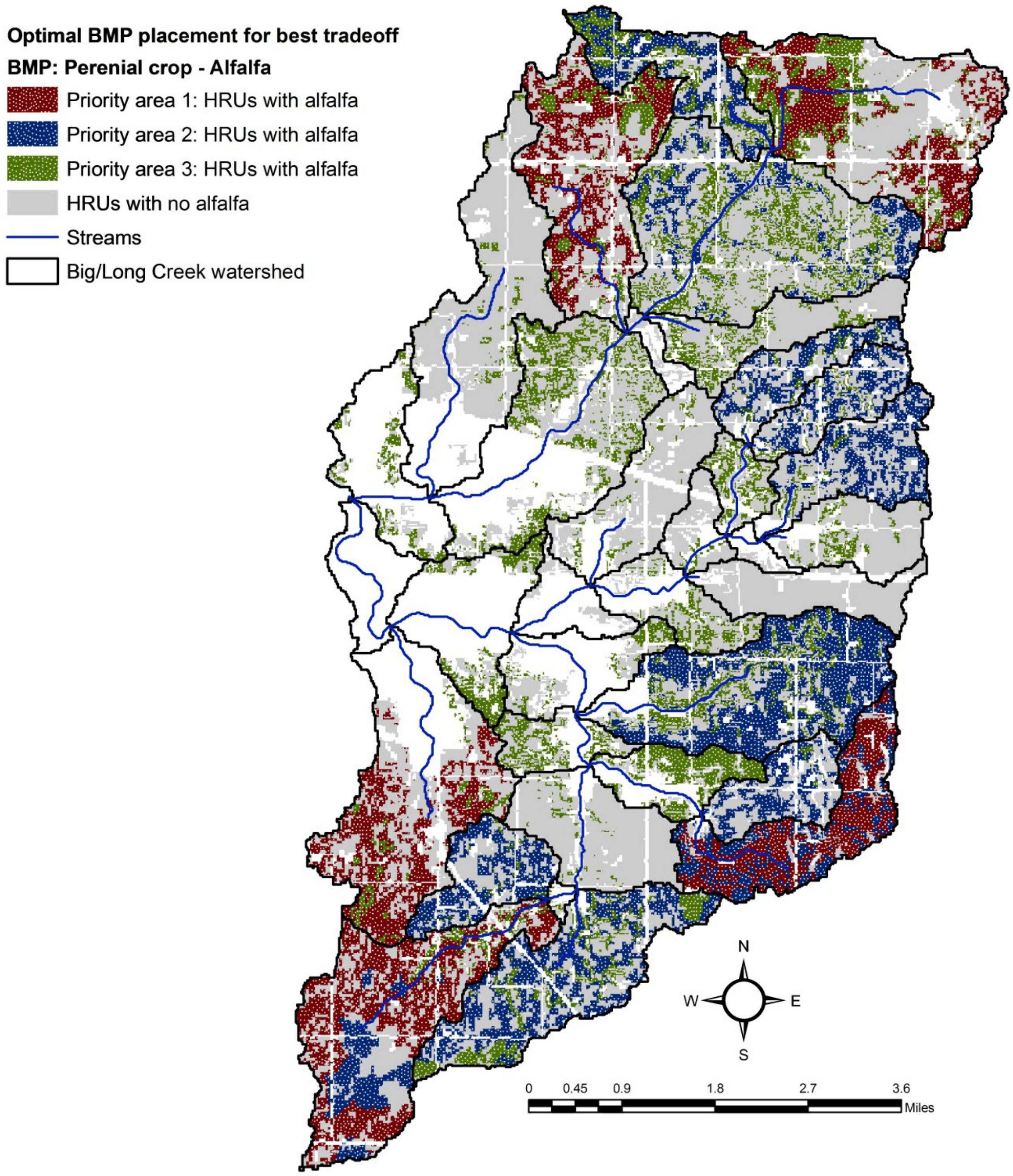


Figure C.4 Optimal placement of alfalfa in Big/Long Creek Watershed

Table C.5 Pollutant reduction for optimal placement of constructed wetlands in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 4.6	15.3	4.911	14.3	0.035	3.3	0.012	1.4	116,583
area 2 = 5.2		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 5.5		12.61		1,785		5,316		62

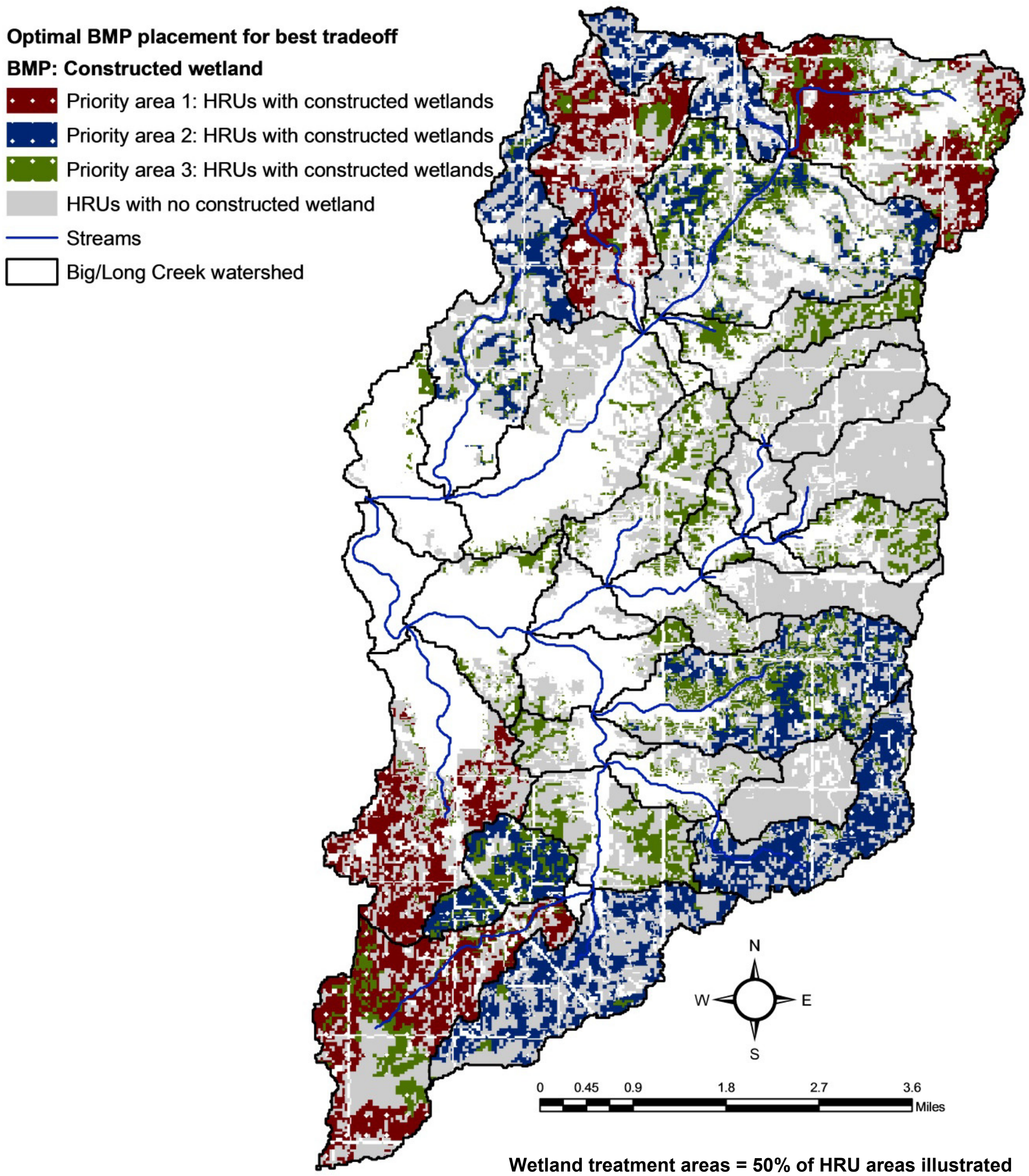


Figure C.5 Optimal placement of constructed wetlands in Big/Long Creek Watershed

Table C.6 Pollutant reduction for optimal placement of DWM in Big/Long Creek Watershed

BMP treament area		Pollutant load Reduction						Equal annual
[% of watershed area]		Nitrate-N		Total phosphorus		Sediment		cost (EAC)
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 5.6	13.8	5.579	16.3	n/a	n/a	n/a	n/a	52,977
area 2 = 4.7		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 3.5		5.57		n/a		n/a		31

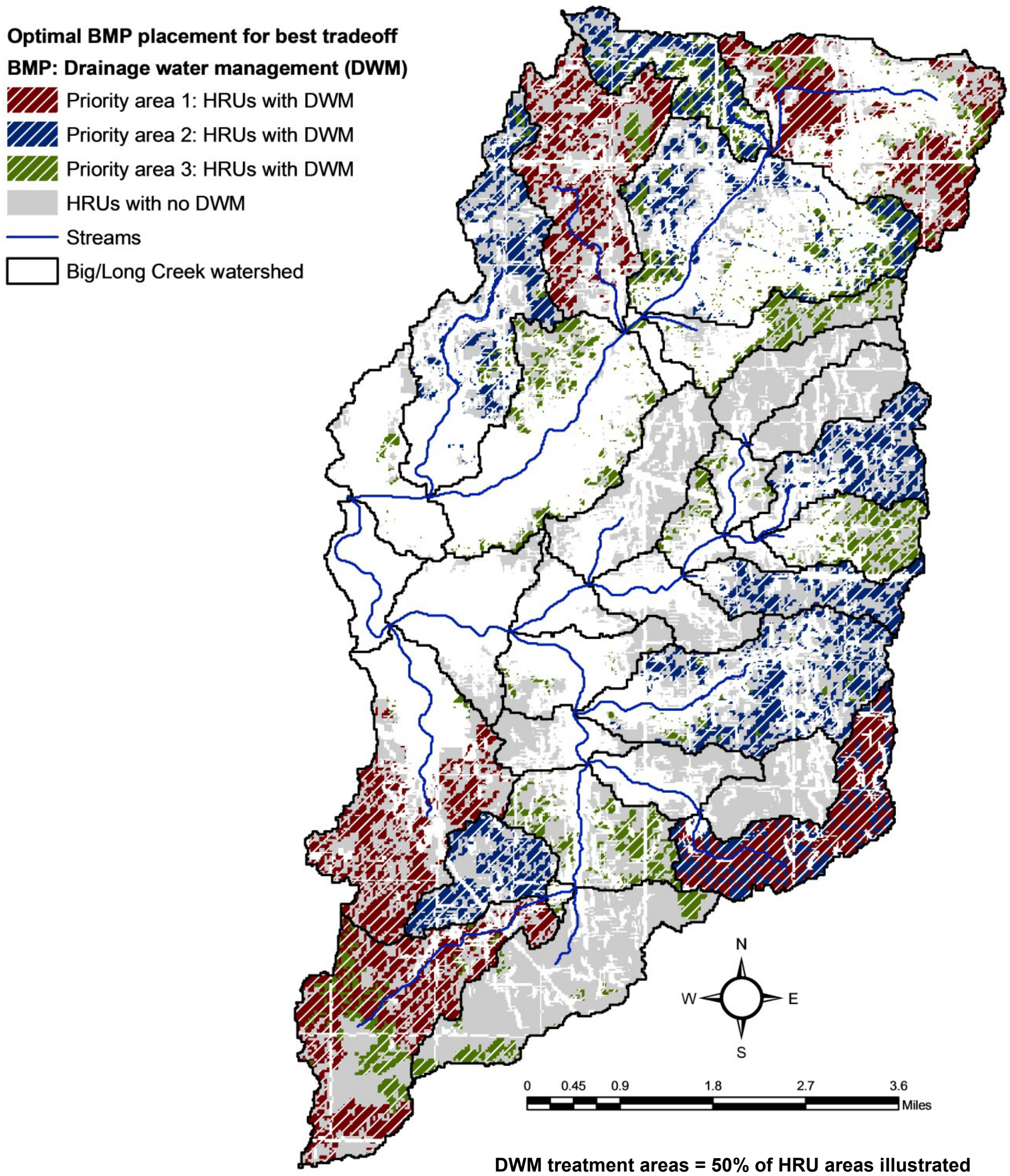


Figure C.6 Optimal placement of drainage water management in Big/Long Creek Watershed

Table C.7 Pollutant reduction for optimal placement of bioreactors in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 10.5	38.7	14.806	43.2	n/a	n/a	n/a	n/a	121,993
area 2 = 14.1		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 14.1		1.73		n/a	n/a	n/a	n/a	26

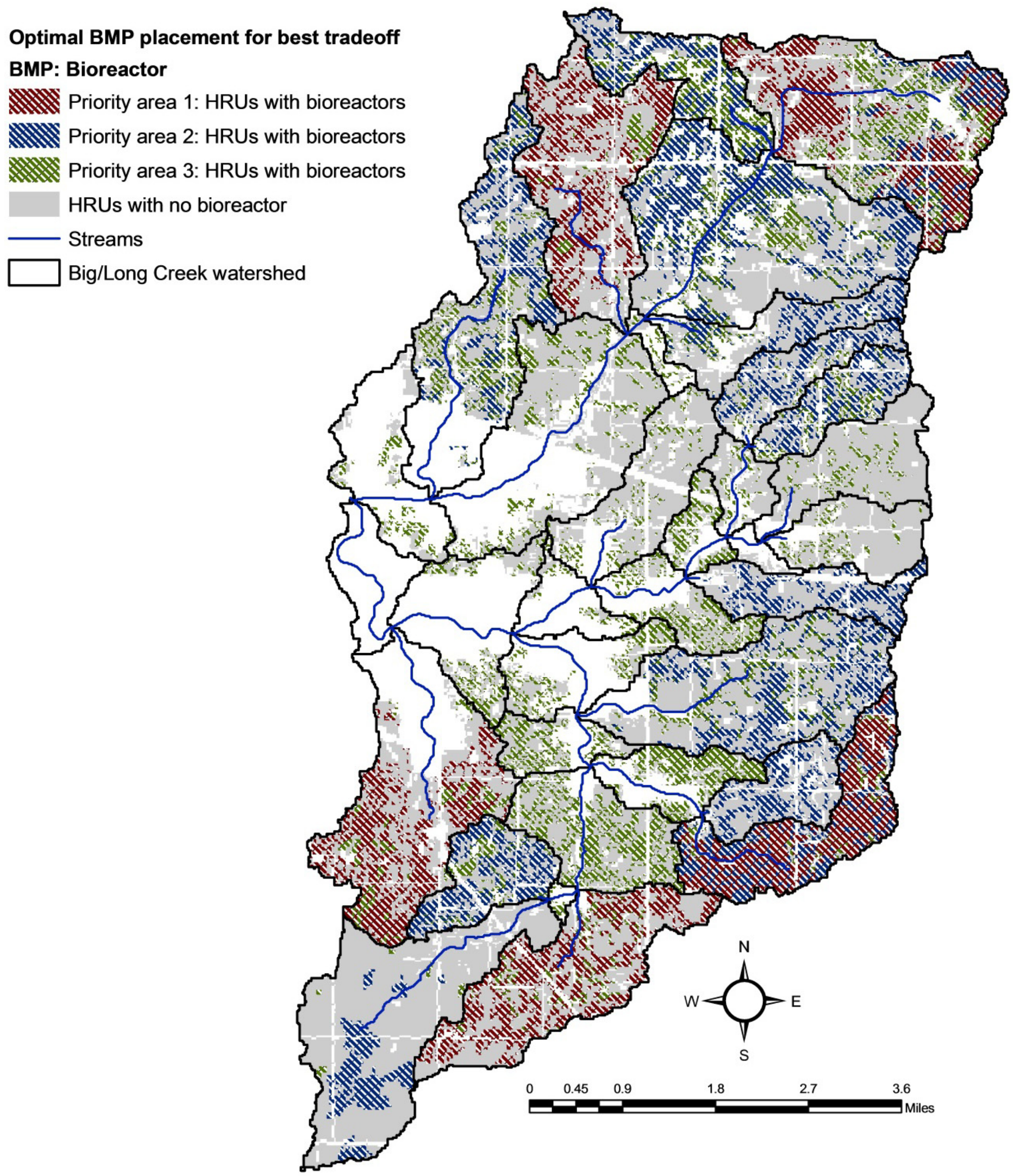


Figure C.7 Optimal placement of bioreactors in Big/Long Creek Watershed

Table C.8 Pollutant reduction for optimal placement of saturated buffers in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 0.8	2.1	0.718	2.1	n/a	n/a	n/a	n/a	7,017
area 2 = 0.4		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 0.9		38.09		n/a		n/a		27

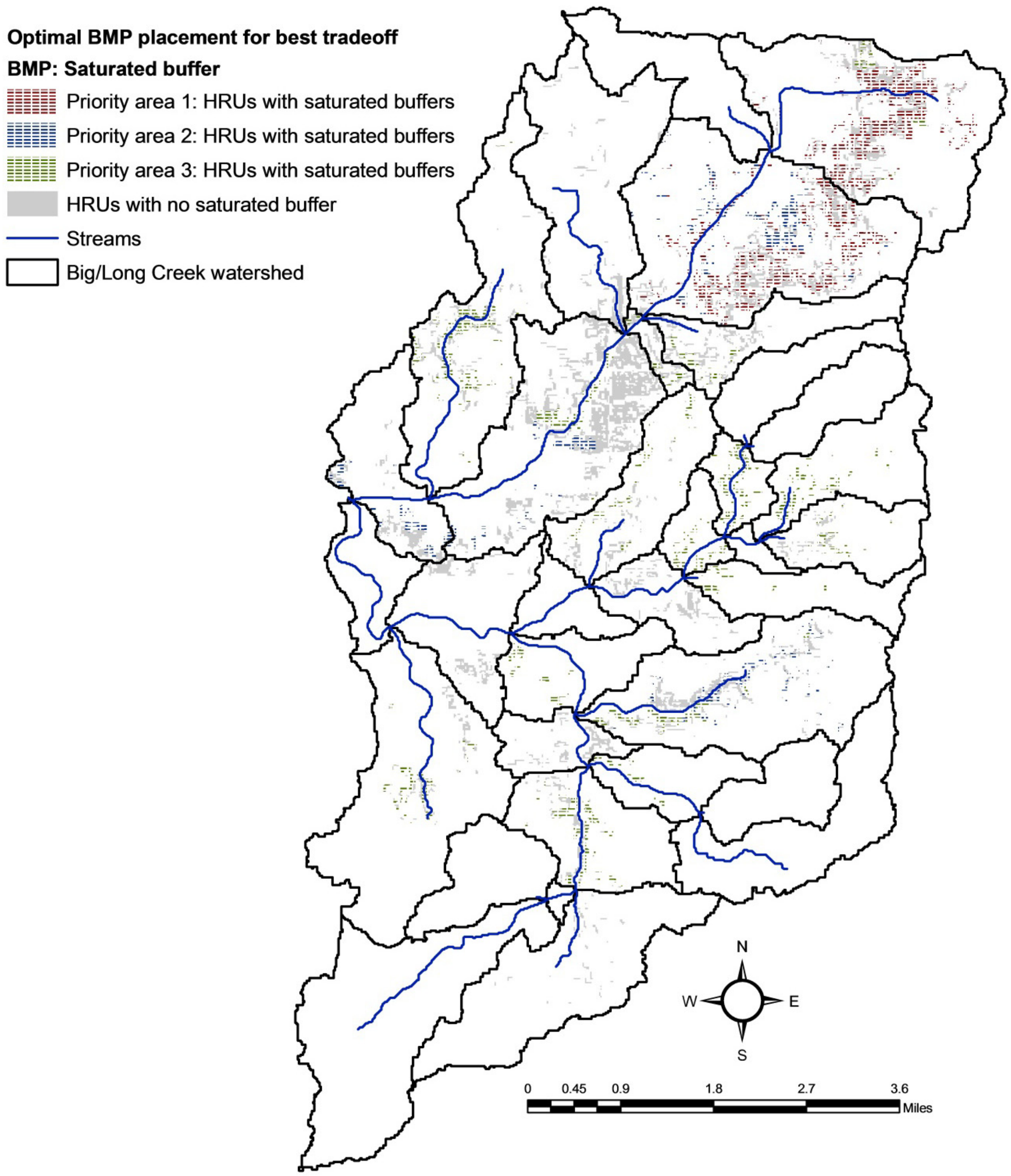


Figure C.8 Optimal placement of saturated buffers in Big/Long Creek Watershed

Table C.9 Pollutant reduction for optimal placement of filter strips in Big/Long Creek Watershed

BMP treament area [% of watershed area]		Pollutant load Reduction						Equal annual cost (EAC)
		Nitrate-N		Total phosphorus		Sediment		
Priority	Total	[kg/ha/yr]	[%]	[kg/ha/yr]	[%]	[t/ha/yr]	[%]	[$/yr]
area 1 = 7.9	40.4	2.204	6.4	0.138	13.2	0.070	8.2	33,868
area 2 = 11.9		EAC [$/kg/ha]		EAC [$/kg/ha]		EAC [$/t/ha]		$/ha/yr
area 3 = 20.6		3.09		49.3		97.0		7

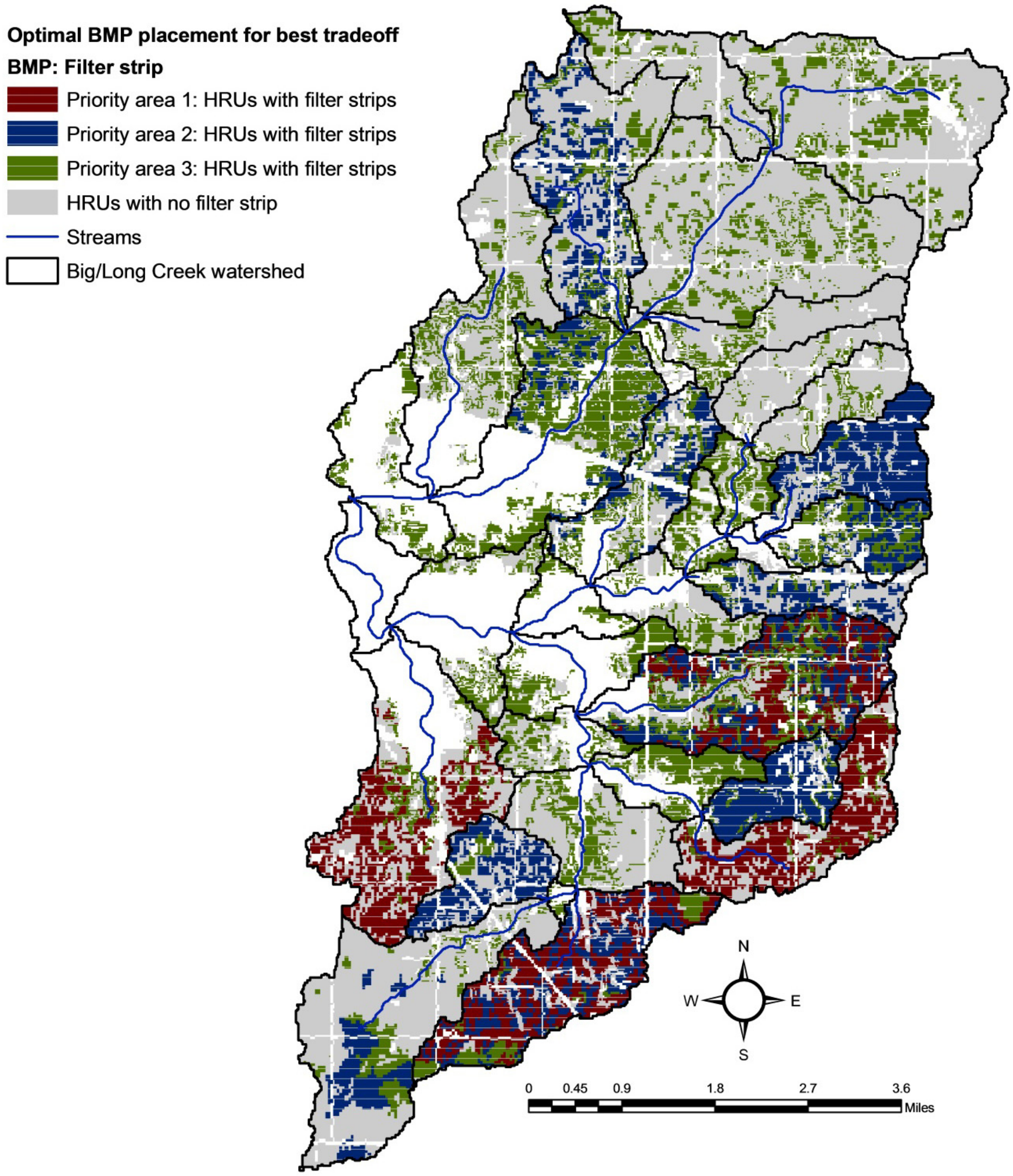


Figure C.9 Optimal placement of filter strips in Big/Long Creek Watershed