

DEVELOPING NEW MODELING TECHNIQUES TO EVALUATE THE
ENVIRONMENTAL AND ECONOMIC IMPACTS OF INDIVIDUAL MANAGEMENT
PRACTICES AT THE FIELD AND WATERSHED SCALES

By

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ABSTRACT

DEVELOPING NEW MODELING TECHNIQUES TO EVALUATE THE ENVIRONMENTAL AND ECONOMIC IMPACTS OF INDIVIDUAL MANAGEMENT PRACTICES AT THE FIELD AND WATERSHED SCALES

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Excess sediment yield as nonpoint source pollution from agricultural lands is a major threat to water quality in lakes and streams in the United States. Watershed modeling can provide data about the source and amount of excess sediment yield. Identifying sediment yields at the field and watershed scales allows watershed planners to make better decisions concerning water quality. The specific objectives of this study were to: 1) compare three watershed-scale models (Soil and Water Assessment Tool (SWAT), Field SWAT, and High Impact Targeting (HIT)) against a calibrated field-scale model (RUSLE2) in estimating sediment yields from fields in the River Raisin Watershed; 2) evaluate the statistical significance among models; 3) assess the watershed models' capabilities in identifying the areas of concern at the field level; 4) evaluate the reliability of the watershed-scale models for field-scale analysis; 5) design and test multiple methods for quantifying the impacts of field-scale management changes the watershed outlet; 6) compare the true costs of BMPs and those from government programs. SWAT was the only model found to be not significantly different from the calibrated RUSLE2. All the models were incapable of identifying priorities areas similar to RUSLE2. SWAT provided the most estimates within the uncertainty bounds of RUSLE2 (51%). A hybrid RUSLE2-SEDMOM-SWAT model proved to be the best method to predict the effects of field-scale management decisions at the watershed outlet. The true costs of sediment reduction at the field and watershed scales were greater than government defined program costs in five out of six BMP categories.

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This thesis is dedicated to my family and friends, thank you.

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LIST OF ABBREVIATIONS

AIC	Akaike Information Criterion
AMA	Agricultural Management Assistance Program
AnnAGNPS	Annualized Agriculture Non-Point Source Model
ANOVA	analysis of variance
APEX	the Agricultural Policy Environmental Extender
ArcSWAT	The Soil and Water Assessment Tool for ArcGIS Software
AWEP	Agricultural Water Enhancement Program
BIC	Bayesian Information Criterion
BMP(s)	Best Management Practice(s)
BOD5	5-day biological oxygen demand
CBWI	Chesapeake Bay Watershed Initiative
CCPI	Cooperative Conservation Partnership Initiative
CDL	Crop Data Layer
CIG	Conservation Innovation Grants
CLU	Common Land use Unit
CPGL	Conservation of Private Grazing Land Program

CRP	Conservation Reserve Program
CSP	The Conservation Stewardship program
CWA	Clean Water Act
EPA	United States Environmental Protection Agency
EQIP	Environmental Quality Incentives Program
FRPP	Farm and Ranch Lands Protection Program
GIS	Geographic Information Systems
GRP	Grassland Reserve Program
ha	hectare
HIT	High Impact Targeting
HRU	Hydraulic Response Unit
HSPF	Hydrologic Simulation Program-FORTRAN
km	Kilometer
km ²	Square Kilometers
L-THIA	the Long-Term Hydrologic Impact Assessment model
LSD	Least Significant Difference
m	Meter

MI	Michigan
NASS	National Agricultural Statistics Service
NED	National Elevation Dataset
NED10	10m Resolution National Elevation Dataset
NHDPLUS	National Hydrology data set PLUS
NLCD	National Land Cover Database
NPS	Nonpoint Source
NRCS	Natural Resource Conservation Service
NSE	Nash-Sutcliffe efficiency
PBIAS	Percent Bias
PLOAD	GIS Pollutant Load Application
RSR	Root Mean Square Error to the Standard Deviation
RUSLE2	the Revised Universal Soil Loss Equation
SAS	Statistical Analysis Software
SCS	Soil Conservation Service
SEDMOD	Spatially Explicit Delivery Model
SDR	Sediment Delivery Ratio

SOM	Soil Organic Matter
SSURGO	The Soil Survey Geographic Database
STEPL	Spreadsheet Tool for Estimating Pollutant Load
STORET	Storage and Retrieval database
SWAT	Soil and Water Assessment Tool
t	Tons
TSS	Total Suspended Solids
USDA	the United States Department of Agriculture
USGS	United States Geological Survey
USLE	Universal Soil Loss Equation
WEPP	the Water Erosion Prediction Project
WRP	Wetlands Reserve Program
WSS	Web Soil Survey
WQT	Water Quality Trading
yr	Year

1. INTRODUCTION

1.1 Background

Non-point source (NPS) pollution from agricultural lands is a significant problem that threatens water quality throughout the United States (EPA, 2005). Excess sediment load is a major portion of this NPS pollution and directly links agricultural land use to water quality (Bossio et al., 2010). Publically sponsored programs exist that have a goal of reducing NPS pollution from agricultural lands (Shortle, 2012). These programs usually experience mixed success, and have difficulty meeting their goals of NPS pollution reduction, in part due to a lack of quality data about the sources and amounts of excess sediment loads (Thomas and Froemke, 2012). By collecting more data from a watershed, water quality programs can be improved and experience a higher level of effectiveness. Monitoring sediment loads in a watershed is not always feasible due to economic and time constraints. Watershed models can fill the gap in available data for decision makers without being a heavy economic or time burden.

Government sponsored water quality improvement projects often rely on watershed scale analysis. Watershed models such as the Soil and Water Assessment Tool (SWAT), the Water Erosion Prediction Project (WEPP), the Annualized Agricultural Non-Point Source (AnnAGNPS) model, the Long-Term Hydrologic Impact Assessment model (L-THIA), the PLOAD model, and the HSPF model have been extensively used to quantify NPS and the effects of BMP implementation (Shen et al., 2009; Parajuli et al., 2009; Im et al., 2009; Nejadhashemi et al., 2011; Giri et al., 2012) are often used for these purposes. However, the main concern with the above modeling approach is the scaling issue. Scaling data from watershed models to the

individual field level is a challenge, however most government programs require this type of data for targeting conservation practices. The scaling challenge arises because the majority of the watershed models outputs are generated at the subbasin level or smaller scales but lumped together based on the physiographical characteristics.

A series of watershed model interfaces were developed in the past few years claiming to provide field-scale information from watershed models, such as High Impact Targeting-HIT (O'Neil, 2010; Bartholic, 2009) and Field SWAT (Pai et al., 2011). The results of these new downscaling techniques have not been tested against calibrated, detailed field-scale models such as the Revised Universal Soil Loss Equation, Version 2 (RUSLE2). The first of the main objectives of this study was to compare sediment yield estimations from Field SWAT, HIT, and a detailed SWAT model that was delineated according to the field boundary map in the River Raisin Watershed in Michigan.

The modeling exercises mentioned above provide important information for making informed watershed management decisions. However, execution of a large-scale BMP implementation plan is infeasible due to a lack of enforcement of NPS regulations and loosely defined contract requirements in conservation programs. BMPs are most often implemented on individual fields, and due to the voluntary nature of most government programs, installation of many BMPs covering a significant portion of a watershed is unlikely. Understanding the true cost and effectiveness of individual BMPs both at the field and watershed scales is important to guide informed, realistic decision making for conservation programs such as the BMP Auction (Smith et al., 2009).

Field-scale models are available for the evaluation of BMP effectiveness, such as the Revised Universal Soil Loss Equation 2 (RUSLE2) and Agricultural Policy Environmental

Extender (APEX). These models are useful for field scale analysis but lack the ability to quantify the effects field scale management changes have at the watershed outlet. Results obtained from watershed scale models such as SWAT can be unreliable at the field scale due to the limitations of land use, topography, and soil input data resolutions for field-scale study (Daggupati et al., 2011). There is need for a modeling framework that is capable of assessing the impact of field-scale management strategies have at the watershed scale.

1.2 Objectives

The main objective of this research is to evaluate the effectiveness of BMPs at the field and watershed scales. The Raisin River watershed was chosen as the study area, which located in southeast Michigan roughly 100 km south west of Detroit, MI. First, the three watershed scale models capable of providing field scale sediment yield estimations were tested against a calibrated, detailed field scale model, RUSLE2. The information gained from this exercise was used to advise the design of 4 methods aimed at quantifying field scale management changes at the watershed outlet. Four techniques were proposed and tested with the goal of providing realistic sediment yield savings at the watershed outlet based on field scale management changes. Watershed scale sediment reduction loads from 80 field scale BMP scenarios defined by actual producers were evaluated at the field scale and at the watershed outlet in the Raisin River watershed. The applicability, advantages, and disadvantages of these approaches are discussed in this study. Finally, an economic analysis was performed to compare producer requested prices versus the prices defined by the USDA's Environmental Quality Incentives Program (EQIP) for BMP implementation. The specific objectives of this research were to:

- compare three watershed-scale models (SWAT, Field SWAT, and HIT) against calibrated field-scale model (RUSLE2) for estimating sediment yield from 41 randomly selected farms within the River Raisin watershed
- evaluate the statistical significance among models
- assess the watershed models' capabilities in identifying the areas of concern at the field level
- evaluate the reliability of the watershed-scale models to predict sediment erosion values within the uncertainty of a calibrated field-scale model
- design and evaluate multiple methods to evaluate the effects of individual BMPs at the watershed outlet
- perform an economic analysis comparing the true cost of sediment reduction defined by producers and the EQIP program at the field and watershed-scales

2. LITERATURE REVIEW

2.1 The Importance of Agriculture

Agriculture is becoming increasingly important as world population swells. The United States is the 3rd largest country in the world, totaling 9,826,675 km², (CIA, 2012). Agriculture land takes up about 40% of this total area (Encyclopedia of the Nations, 2012). The United States is the second largest agricultural producer, and the largest agricultural exporter in the world (Alston et al., 2010). Agriculture is an important part of the United States economy, growing from a \$17 billion industry in the late 1920's, to \$98 billion early this century (Alston et al., 2010). There are many benefits due to the high level of agricultural activity in this country including food security. Food security is relatively high in the United States compared to some other countries, with 85.3% of households being food-secure; a main reason for this level of security is a successful agricultural sector (Nord, 2009). Expanding the amount of available cropland and increasing crop yields will be necessary to keep food available and affordable to the growing population (Tweeten, 2008; Von Braun, 2008).

2.2 Environmental Problems Due to Agricultural Practices

Substances used as inputs in agricultural activities can be harmful to the environment. Pesticides caused an estimated \$12 billion in environmental and social damages in recent years (Pimentel, 2009). Agricultural activities can be damaging to drinking water supplies, even in industrialized countries (Olmstead, 2010). Agricultural systems have the potential to severely damage freshwater ecosystems (Moss, 2008). In comparison to land in its natural state,

agricultural land significantly increases water pollution by allowing, on average, a far greater amount of sediment and nutrients to enter surface waters (Moss, 2008).

External costs, or externalities, are involuntary consequences from agricultural practices for which there is no formal market trading; these consequences are put through the process of valuation, assigning monetary values to events and actions that exist outside the marketplace (Tegtmeier and Duffy, 2004). Agricultural activities usually involve high external costs (Porter et al., 2009). Examples of agricultural externalities include decreased stream health due to excess nutrient runoff, decreased biodiversity due to mono-cropping, decreased water quality from pesticide drift, decreased soil fertility from poorly planned crop rotations, and increased sediment runoff due to tillage practices, among many others. External costs due to agricultural practices have been estimated to be between \$5.7 and \$16.9 billion annually in the United States (Tegtmeier and Duffy, 2004).

2.3 Water Pollution

In lakes and streams, impaired water quality is due mostly to agricultural nonpoint source (NPS) pollution (EPA, 2005). NPS pollution is defined as: "...pollution that reaches receiving waters through diffuse and complex pathways..." (Shortle, 2012) Agricultural NPS pollution also contributes significantly to impaired ground water (EPA, 2005). Pollutants from agricultural practices include excess sediment, nitrogen, phosphorus, and pesticide chemicals (EPA, 2005). Environmental concerns from these pollutants include eutrophication, fish kills, habitat destruction, and ground water contamination (EPA, 2005). Downstream pollution specifically from sediment and nutrient runoff is a significant problem (Vitousek et al., 2009). Marine ecosystem impacts from these pollutant sources include eutrophication in fresh water and

hypoxic zones marine systems (Gordon et al., 2010). Direct and indirect impacts from agricultural water pollution cause dangerous changes in natural ecosystems that have negative social and economic effects (Gordon et al., 2010). Instances of impaired drinking water, reduction or destruction of natural fisheries, wetland and coastal ecosystem loss, decline in waterfront property values, and loss of revenue from water related recreation can all be traced back to agricultural water pollution (Gordon et al., 2010). There are three main categories for agricultural NPS pollution: excess sediment, nutrient, and pesticide runoff (Dowd et al., 2008).

2.3.1 Sediment Pollution

Though agricultural NPS water pollution comes in many forms, sediment run off stands out as an important link between land use and water quality (Bossio et al., 2010). As a general rule, soil erosion increases with the application of agricultural practices (Bossio et al., 2010). Agricultural practices put excess stress on the land in comparison to native grass stands and during a rain event sediment can reach a stream by rain splash or rill erosion, and can be accelerated by the presence of drain field tile, or ditches (Deasy et al., 2009).

An important component of agricultural land is soil organic matter (SOM), which is likely to be degraded due to mismanagement or lost in a rainfall event do to runoff (Bossio et al., 2010). As SOM decreases in agricultural soil, runoff begins to increase at an accelerated rate due to increased soil compaction (Bossio et al., 2010). Thus, a decreasing amount of SOM concentration in soil increases the rate of removal (Bossio et al., 2010).

Excess sediment in streams and lakes can be deposited a great distance from the source, causing far reaching problems including increased turbidity, and disturbing aquatic ecosystems (Deasy et al., 2009). In addition, sediment runoff harbors a host of associated pollutants that bind

to particles such as metals, pesticide and herbicide chemicals, and particulate nutrients including nitrogen and phosphorus (Deasy et al., 2009). These sediment bonded contaminants reduce benthic macro invertebrate species diversity, lowering food availability for fish and other organisms, thus causing disruption in aquatic the ecosystem environment (De Lange et al., 2004). Maintenance cost of dams, reservoirs, and lakes may increase due to excess sediment runoff by causing shorted lifespan of components or more frequent dredging (Deasy et al., 2009). Increased turbidity can also contribute to a loss of revenue from businesses associated with recreation centered on a water body (Gordon et al., 2010). Consequences like these can be quantified by externalities. The external costs due to excess sedimentation have been estimated at 13.4 billion per year in the United States (Tegtmeier and Duffy, 2004). Examples of external costs from sediment pollution that have not yet been mentioned include damages to recreational activities, fisheries, and wildlife preservation areas among many others.

In the Great Lakes Region, increased levels of sediment loads into lakes and streams poses a major threat to water quality (Steinman et al., 2009). The great lakes legacy act allocated \$270 million dollars towards efforts in reduction contaminated sediment erosion in the great lakes region (EPA, 2012a). According the EPA (2012), 14 of the 31 key sediment erosion problem areas around the great lakes basin are located in Michigan. The United States Army Corps of Engineers spends about \$40 million in the removal of 3 million m³ of sediment from navigable waterways in the Great Lakes Basin (Miller, 2012).

Excess sediment runoff is dangerous and has potentially far reaching environmental, social, and economic consequences. Since the main source of sediment runoff is associated with agricultural land use, management practices on these lands make a large impact on total sediment pollution in a watershed (Tegtmeier and Duffy, 2004).

2.3.2 Nutrient Pollution

The cycle of nutrients in a terrestrial system is greatly disturbed by human activities (Howarth et al., 2002). Nutrient pollution, namely nitrogen (N) and phosphorus (P) from agricultural sources into lakes and streams reduces environmental quality and the well being of humans (Vitousek et al., 2009).

The most common source of excess nutrients from agricultural lands is fertilizer in synthetic, organic, or manure form (Vitousek et al., 2009). There has been a significant global increase in (N) and (P) ground application of fertilizers in the recent past (Vitousek et al., 2009). Over fertilization contributes to a host of environmental problems downstream of the source (Vitousek et al., 2009). Significant environmental costs including rising levels of nitrous oxide, a greenhouse gas, poor water quality, and photochemical smog are all consequences of excessive nutrient balances on agricultural land (Vitousek et al., 2009).

Increases in N and P have lead to 60% of coastal rivers showing some level of water quality impairment, making it one of the largest pollution problems in the united states (Howarth et al., 2002). In the northeastern United States, nitrogen in streams has increased by a factor of eight in the 40 years prior to 2002 (Howarth et al., 2002). Field scale nutrient losses contribute significantly to overall fertilizer pollution through leaching (Howarth et al., 2002).

2.3.3 Pesticide Pollution

Agricultural pesticides are dangerous to fish and aquatic invertebrates environmental even in low concentrations, (Schulz, 2004; Shortle, 2012). Over sixty studies reporting pesticide presence in surface water have been published over the last 20 years (Schulz, 2004). These studies combine to form a general theme and conclusion, stated by Darbowski and Schulz (2003)

as: “nonpoint-source pesticide pollution from agricultural areas is widely regarded as the one of the greatest threats to contamination of natural surface waters...” Pesticides, especially insecticides cause problems in aquatic populations including fish, macro invertebrates, amphibians and birds by affecting reproductive cycles, growth and development (Schulz, 2004).

There are three main application techniques used to apply agricultural pesticides: soil incorporation, spraying, and fumigation (Gregoire et al., 2009). Once applied, pesticides enter waterways from agricultural lands by runoff or spray drift (Dabrowski and Schulz, 2003). Of these two, runoff is a greater threat to environmental health, due to the far reaching consequences pesticides can have in solution in moving bodies of water compared to atomization in the air (Dabrowski and Schulz, 2003). Runoff from fields can result in 1% to 10% of the total amount of pesticide application, and as much as 60% can drift away during spraying (Dabrowski and Schulz, 2003). Once these chemicals are mobilized in stream networks, their removal is difficult (Gregoire et al., 2009).

2.4 Methods to Reduce Pollution from Agricultural Practices

Sediment, nutrient and pesticide NPS pollutants from agricultural lands pose a threat to human and aquatic environmental health; addressing this issue is important to ensure continuing protection of water resources. Common logic points out are two basic ways to achieve unimpaired water resources: removing pollutants after they have entered waterways, or, to control pollutants to prevent them from entering waterways in the first place. Prevention is a much more attractive solution to the problem than remedial efforts (Kay et al., 2009). Although remediation techniques are necessary in waterways that are already polluted, the approach for a

permanent solution to agricultural NPS pollution in the future is to mitigate this pollution at the source.

There are various techniques available that are employed to reduce NPS pollution from agricultural lands (Kay et al., 2009). These practices can be applied at three different levels: reduction of pollutant input into the system, reducing the transportation ability of applied pollutants, and by collecting and degrading mobilized pollutants before they leave the source site (Kay et al., 2009).

2.4.1 Best Management Practices

Best Management Practices (BMPs) are actions aimed at reducing agricultural NPS pollution and improving water quality; they can be structural or non-structural practice (Kaplowitz and Lupi, 2009). A non-structural practice is one that does not require construction, but modifies actions taken on agricultural land, such as reducing the number of tillage passes in a field (Kaplowitz and Lupi, 2009). Structural BMP's require construction and more permanent land use change, such as a filter strip or an artificial wetland to capture runoff (Kaplowitz and Lupi, 2009). Recently, some of the most common BMPs specifically mentioned in the literature are: native grass, terraces, filter strips, grassed waterways, crop rotation modifications, conservation tillage, no till, nutrient management plans, cover crops, detention ponds, riparian buffer, stream bank stabilization, grade stabilizations, constructed wetlands, and contour farming. (Duncan and Bradshaw, 2007; Kaini et al., 2012; Maringanti, et al., 2009; Hsieh et al., 2010; Maringanti et al., 2011; Makarewicz, 2009; Woznicki and Nejadhashemi, 2009; Chen et al., 2010; Baumgart-Getz et al., 2012; Diebel et al., 2008; Georgas et al., 2009; Herendeen and Glazier, 2009; Maxted et al., 2009; Short, 2011; Sharpley et al., 2009; Tuppad et al., 2010;

NRCS, 2012a). The following practices have the specific goal of reducing NPS sediment pollution: terraces, native grass, filter strips, grassed waterways, riparian buffers, no till, conservation till, contour farming, stream bank stabilization, and grade stabilizations.

2.4.2 Common Sediment BMP Descriptions Specific to Michigan

The Natural Resource Conservation Service in Michigan has a technical guide describing the purpose and specifications of many BMPs (NRCS, 2012a; NRCS, 2012b). Definitions of these common sediment focused BMPs in addition to a few others not mentioned above are presented in Table 2-1. The main source of these definitions is the NRCS Technical Guide.

Table 2-1: Common BMPs Applied in Michigan as Describe in the NRCS Technical Guide

BMP	Definition	NRCS CODE	Purpose	Condition of Application	Specifications
Terraces	Terraces are repeating, horizontal, earthen ridges in a hillside across the slope. There are three types of terraces, broadbase, steep backslope, and impound. A broadbase terrace is built on gentle slope. Steep backslope terraces have a sodden backslope; these are the most common on extreme slopes. Terraces with underground outlets are given the term "impound". (NRCS, 2010f) (Nejadheshemi)	CODE 600	* Reduce erosion of soil *conserve moisture by retaining runoff (NRCS, 2010f)	*Excess soil erosion due to extreme slope length *Excess runoff causes problems *Water conservation is important *Reasonable farming can continue after construction *Reasonable outlet is available (NRCS, 2010f)	*Multiple terraces are parallel *Capacity to control a 10 year, 24 hour storm event *Capacity to continue effective operation through 10 years of sediment accumulation
Filter Strip	Herbaceous vegetation in a strip or area between agriculturally managed land and sensitive areas that removes contaminates in overland flow (NRCS, 2010b)	CODE 393	*Reduce sediment, suspended solids, and dissolved contaminants in runoff *Reduce contaminates associated with irrigation (NRCS, 2010b)	*Sensitive areas that need protection from sediment and other runoff associated pollutants	*10 year life span *minimum flow length equal to 20 feet *Located immediately down slope from source of contaminants

Table 2-1 (cont'd)

					*Slope above filter strip is no more than 1% *Vegetation must tolerate partial burial from sediment and herbicides used in nearby agricultural practices (NRCS, 2010b)
Irrigation Water Management	Determining and controlling frequency and volume of irrigation water in a planned manner (NRCS, 2010h)	CODE 449	*Manage soil moisture *Optimize use of available water *Decrease NPS pollution into surface and groundwater(NRCS, 2010h)	*Applicable to all irrigated land	*Applied in accordance with federal, state, and local laws *Water cannot be applied in excess of need *An “Irrigation Water Management Plan” must be developed to assist in consistent decision making

Table 2-1 (cont'd)

Grassed Waterways	A channel that is established with vegetation and graded so as to transport water at an appropriate velocity to be non-erosive (NRCS, 2010i)	CODE 412	*Transfer runoff away from water concentrations while avoiding flooding or erosion *Combat gully erosion *Improve and/or protect water quality (NRCS, 2010i)	*Areas of water concentrations where added vegetation and designed grading could reduce erosion from runoff (NRCS, 2010i)	*Minimum capacity to effectively transport runoff and associated sediment from 0 a 10 year, 24 hour storm event *Width less than 100 feet *Side slopes at a value unobtrusive to current agricultural practices *Stable outlet with appropriate capacity (NRCS, 2010i)
Riparian Buffers	An area dominated by shrubs, trees, or a combination of both located directly adjacent to a sensitive body of water (NRCS, 2010j)	CODE 391	*maintain or lower water temperatures by creating shade *Reduce sediment and other contaminants associated with excess runoff *Reduce pesticide drift into surface waters *Increase carbon storage	*Areas adjacent to intermittent or permanent streams, lakes ponds and wetlands *Not applied to stabilize shorelines or stream banks	*Must have sufficient width, length, height, and plant density to achieve goals *Vegetation must be dominated by

Table 2-1 (cont'd)

			in biomass and soils (NRCS, 2010j)		planted trees and shrubs that are able regenerate in the given soil, weather conditions and geographical area *Overland flow through the buffer is kept to sheet flow *Native and non invasive vegetation only (local cultivars acceptable)
No Till	Limiting disturbing soil operations to only those that are absolutely necessary and management the amount and distribution of plant residue (NRCS, 2010k)	CODE 329	*Reduce sheet and rill erosion of sediment and other runoff associated pollutants *Reduce wind erosion of soils *Improve SOM content in soils *Reduce particulate emissions from soil *Increase plant available moisture *Provide food and habitat	*All agricultural land where crops are planted	*No residue is burned *Uniform distribution of residues *includes all operations between and including harvesting the previous crop and harvesting the current

Table 2-1 (cont'd)

			for various wildlife		crop in the Soil Tillage Intensity Rating (STIR) value *STIR not greater than 30 *Residue amount needed are determined with current approved water erosion prediction technology (NRCS, 2010k)
Stream Bank Stabilization (Streambank Protection)	Stabilizing treatments used to protect streams or constructed channels (NRCS, 2010e)	CODE 580	*Prevent damage to facilities, land uses, or significant loss of land near streams and constructed channels *Maintain flow capacities of streams and constructed channels *Reduce downstream effects of sediment erosion *Enhance wildlife habitat near stream or channel banks (NRCS, 2010e)	*Natural streambanks or constructed channel banks that are susceptible to erosion (NRCS, 2010e)	*All treatments must be in agreement with local, state and federal laws *Avoid detrimental effects to endangered or threatened species and their habitat *Proposed treatments

Table 2-1 (cont'd)

					interface effectively with existing structure
Windbreak	Shelters made of one or more linear rows of trees or shrubs for the purpose of reducing wind erosion of soil (NRCS, 2010d)	CODE 380	*Reduce soil erosion due to wind *Protect plants *Improve air quality	*Where linear stands of trees or shrubs are desirable *Where aesthetics hold significant value *Wind protection is needed	*20 year function period *Species must be adapted to the region *Orientation as close to perpendicular to average wind direction as possible
Sediment Basin	A basin with an outlet formed by and excavation or embankment to collect excess sediment runoff (NRCS, 2010c)	CODE 350	*Capture and store excess sediment runoff for a sufficient period of time to allow settling (NRCS, 2010c)	*When treatments that target the source of sediment runoff are deemed inappropriate *Where basin failure will not result in extensive damage to structures or loss of life (NRCS, 2010c)	*Hazard class of constructed basin is low *Total dam height is less than 11 meters *Minimum sediment storage of 60 m³ per ha of treated area - Minimum detention storage of 250 m³ per ha of drainage area

Table 2-1 (cont'd)

					(NRCS, 2010c)
Conservation Crop Rotation	Crops grown in a planned sequence on the same field aimed at improving environmental quality by reducing sediment erosion and other pollution (NRCS, 2010b)	CODE 328	*reducing sheet and rill erosion *reducing irrigation-induced erosion *reducing wind erosion of soil *maintaining or improving soil organic matter content *managing nutrient balances *managing plant pests including weeds *insects and disease *providing food for domestic livestock *providing food or habitat for wildlife *Reducing the use of pesticides (NRCS, 2010b)	*Anywhere there are agricultural lands on which crops are grown *However, pasture and grazing lands, or any land on which crops are grown occasionally are inappropriate for this BMP (NRCS, 2010b)	* Specific crops and practices associated with these rotations must agree with those outlined by approved university publications * Crops chosen in the rotation must provide ample biomass when needed to effectively reduce soil erosion to within thresholds defined by current approved erosion prediction

Table 2-1 (cont'd)

					technology * If selected crops fall short of biomass requirements a conservation crop rotation should include cover crops (NRCS, 2010b)
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2.5 Existing BMP (conservation practice) Public Programs

BMP programs as they are known today are considered to begin with the Clean Water Act-CWA (Shortle et al., 2012). Under the act, point source (PS) pollution was defined as a federal responsibility and NPS pollution decisions were placed under state control (Shortle et al., 2012). These policies are described as a complicated system of varying initiatives (Shortle et al., 2012). Because of this, NPS management programs vary from state to state. Although each state may have differing policies, most use voluntary programs to address NPS pollution control. (Shortle et al., 2012; NRCS, 2012a)

2.5.1 Main USDA BMP Programs

Currently, the main United States Department of Agriculture (USDA) nationwide conservation programs that fund and promote BMPs are the Conservation Reserve Program (CRP), The Environmental Quality Incentive Program, (EQIP), and The Conservation Stewardship program (CSP) (NRCS, 2012b). These programs are run by the Natural Resource Conservation Service (NRCS, 2012b) a subset of the USDA (NRCS, 2012b). The following is a summary of each NRCS program currently in place that is directly related to water quality improvement through the use of BMPs.

2.5.1.1 Environmental Quality Incentives Program (EQIP)

The EQIP program is designed to assist agricultural producers by helping to fund the implementation and maintenance of BMPs for up to 10 years per contract (NRCS, 2012b). In providing assistance through the EQIP program, the NRCS plans to help producers meet federal, state and local regulations (NRCS, 2012b). EQIP attempts to work with producers to identify the most suitable BMPs to use, and helps producers with a conservation plan in agreement with

locally adapted NRCS technical standards (NRCS, 2012b). This is the largest of the programs, with an annual budget of about \$1.3 billion, the majority of which was spent on projects with water quality improvement goals (Shortle et al., 2012). Over \$487 million worth of requests were sent in by farmers applying to this program in 2008 (Shortle et al., 2012).

2.5.1.2 Conservation Reserve Program (CRP)

Although not specifically mentioned in the 2008 NRCS Farm Bill, the CRP program is the largest program based on budget (NRCS, 2012b). CRP promotes a specific BMP, native grass, by paying producers to leave qualified land fallow, thus reducing NPS agricultural pollutants (NRCS, 2012b). The goal is to influence farmers to convert erosion and pollution sensitive crop lands to more secure native grass stands (NRCS, 2012b). These native grass stands come in the form of filter strips, riparian buffers, or entire fallow fields, among others, and participating producers receive an annual payment for participation (NRCS, 2012b).

2.5.1.3 Conservation Stewardship Program (CSP)

The conservation stewardship program is a program promoting BMPs through payments made to producers who are making operation-level modifications towards environmental improvement (NRCS, 2012b). No size limit is imposed on operations, so a great number of producers in the United States are eligible to apply for a contract (NRCS, 2008). Participants are paid based on performance: an annual payment for adopting BMPs, and a special payment for implementing an environmentally friendly crop rotation (NRCS, 2008).

2.5.2 Other USDA Programs

In addition to EQIP, CRP, and CSP, the NRCS runs other voluntary programs with water quality improvement goals (NRCS, 2009). There are a total of 14 such programs included in the 2008 NRCS Farm Bill (NRCS, 2009).

2.5.2.1 Agricultural Management Assistance Program (AMA)

The AMA program funds producer's voluntary efforts to improve water quality and erosion control on their lands (DOA, 2008a). Specific actions suggested under this program include structural BMPs, tree planting, soil erosion control, pest management , or replacement of traditional farming with organic methods (DOA, 2008a). Total budgeting for this program includes \$15 million be distributed yearly from 2008 through 2012 (DOA, 2008a). AMA is not nationwide, it is only "...available in 16 states where participation in the Federal Crop Insurance Program is historically low: Connecticut, Delaware, Hawaii, Maine, Maryland, Massachusetts, Nevada, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Utah Vermont, West Virginia, and Wyoming (DOA, 2008a).

2.5.2.2 Chesapeake Bay Watershed Initiative (CBWI)

The Chesapeake Bay Watershed Initiative Designed to assist producers in BMP implementation on agricultural lands within the Chesapeake Bay watershed. The initiative aims to control erosion and nutrient levels in ground and surface water (DOA, 2008b). Actions aimed at prevention of further water quality degradation, and remedial efforts to restore original environmental health can be funded under the CBWI (DOA, 2008b). Funding priority within the CBWI is given to the Susquehanna, Shenandoah, Potomac, and Patuxent River basins, which

outlet into the Chesapeake Bay in Maryland (DOA, 2008b). If selected, participants agree to “improve water quality and quantity, and restore, enhance, and preserve soil, air, and related resources” (DOA, 2008b).

2.5.2.3 Cooperative Conservation Partnership Initiative (CCPI)

The CCPI “provides targeted assistance to producers for enhancing conservation outcomes on agricultural and nonindustrial private forest land” (DOA, 2008c). Addressing conservation priorities, encouraging regulatory compliance and cooperation of producers, and promotion and demonstration of novel, innovative BMPs comprise the listed goals of the CCPI. State Conservationists are responsible for how 90% of CCPI funds are spent (DOA, 2008c). All CBWI funds come from allocating 6 percent of each of the total funds from the EQIP, CSP, and WHIP programs (DOA, 2008c). Local, State, and Tribal governments, producer cooperatives, universities, and non government organizations are all eligible to apply to the CCPI for funding of project (DOA, 2008c).

2.5.2.4 Conservation of Private Grazing Land Program (CPGL)

Addressing natural resource concerns and “enhancing the economic and social stability” on grazing and pasturelands are the major goals of the CPGL (DOA, 2008h). A program specifically for grazing land was deemed necessary by the NRCS because much of the agricultural land in the United States is grazing land and many “ecological and economic benefits” can be traced back to conservation efforts on these lands (DOA, 2008h). Opportunities for producers listed under this program include funding for improving the aesthetic level of grazing land, implementing and maintaining BMPs, improving both water quality and water

quantity, and wildlife habitat (DOA, 2008h). Also included are funding opportunities for creating recreational options and increasing diversification (DOA, 2008h).

2.5.2.5 Agricultural Water Enhancement Program (AWEP)

Financial and technical assistance for ranchers and farmers under the AWEP program is directed at conserving ground and surface water on agricultural lands (DOA, 2008f). AWEP funds projects that fit into the following categories: conservation plans that include modeling and/or “condition assessment,” projects that reduce the amount of water necessary for agricultural practices, projects that focus on water quality remediation, irrigation efficiency improvement, “activities designed to mitigate the effects of drought,” and any other project deemed to add to water conservation efforts (DOA, 2008f). Producers who are eligible for EQIP can apply to the AWEP (DOA, 2008f). Decisions on AWEP fund dispersal are partial to applications that include a large area, “allow for monitoring or evaluation,” include irrigation management changes, or help one or more producers comply with a regulation (DOA, 2008f).

2.5.2.6 Conservation Innovation Grants (CIG)

CIG are focused on supporting innovative BMPs that work to protect the environment and natural resources (DOA, 2008g). This is a competitive grant program where non-Federal and Tribal governments as well as non-government organizations can compete for award money (DOA, 2008g). Projects that receive CIG funding are often focused on large areas, such as the watershed or multi-state scale (DOA, 2008g). Every year, specific concerns are stated as eligible grant awards, sub competitions are also allowed under CIG (DOA, 2008g). CIG can cover up to 50% of project proposed (DOA, 2008f). Disadvantaged producers, such as Tribes or community organizations can receive preference in CIG decisions (DOA, 2008g).

2.5.2.7 Farm and Ranch Lands Protection Program (FRPP)

The FRPP program purchases conservation easements for entities with land enrolled in other conservation programs (DOA, 2008i). This action attempts to keep agricultural land that is deemed conservatively operated from changing land use (DOA, 2008i). These agreements may last up to 5 years for producers that have a “proven record of acquiring and monitoring conservation easements” (DOA, 2008i). More than 216,000 ha were enrolled in this program in 2008, including over 400 contracts and spanning 49 states (DOA, 2008i).

2.5.2.8 Grassland Reserve Program (GRP)

Stated under the goals for GRP are restoration, protection, and enhancement of grassland, with a focus on working grazing operations (DOA, 2008d). The GRP is designed to enhance “...plant and animal biodiversity...” and to protect “...grassland[s] and land[s] containing shrubs and forbs under threat of conversion...” (DOA, 2008d). From 2003 through 2007, the GRP was involved in 250 easements including 47,000 ha located in 38 states (DOA, 2008d). There is a \$50,000 limitation on annual payments through this program given out in up to 50% cost share payments (DOA, 2008d). Priority is given to land with expiring CRP contracts; GRP contracts can be 10, 15, 20 years, or the maximum length of time allowed by the state in which the contract is signed (DOA, 2008d).

2.5.2.9 Wetlands Reserve Program (WRP)

Aimed at private landowners, the WRP “...provides technical and financial assistance to private landowners and Tribes to restore, protect, and enhance wetlands” (DOA, 2008e). In an attempt to meet these goals, the WRP specifically funds BMPs that reduce sediment and

chemical runoff, help to increase ground water levels and decrease flooding, and "...provide opportunities for educational, scientific, and limited recreational activities (DOA, 2008e). Three options are available for landowners to join the program, they are: "Permanent Easement," "30 year Easement," or a "Recreational Cost Share Agreement" (DOA, 2008e). The USDA pays the entire easement in 30 year and permanent easements, and up to 75% of restoration costs for the Recreational Cost Share Agreement (DOA, 2008e).

2.6 Status of BMP's in the United States and Michigan

Although many different BMPs are known to be effective, and multiple public programs funding and promoting them exist, "it has been well established that agricultural NPS policies are not having the desired outcomes" (Shortle et al., 2012). Often, these policies and programs fall short of their goals (Gilinsky et al., 2009). Further criticisms deem these programs inefficient (Office of the Press Secretary, 2009; Shortle et al., 2012). The major public programs aimed at improving water quality EQIP, CRP, and CSP are falling under increasing scrutiny for inefficiency and ineffectiveness (Shortle et al., 2012; NRCS EQUIP; Gilinsky et al., 2009) .

Inability of government officials to monitor producer's progress and establish accountability in the program caused almost 50% of EQIP funds to be spent on unfulfilled contracts (USDA, 2009). Land retirement, though in theory is a sound solution to pollution reduction, may be an ineffective tool in real agricultural systems. The low efficiency of these USDA and NRCS programs is due partly to the fact that while technical support for BMP projects has increased 60% from 1985 to 2006, expenditures have increased roughly 500% (USDA, 2009). Further criticism states all three of these programs have ineffective targeting strategies or none at all, thus missing a vital component to efficient spending towards water

recourse protection (Shortle et al., 2012). The USDA has also been accused of poor allocation of funds to the most important problems water quality problems in the nation (Shortle et al., 2012).

THE USDA and NRCS BMP programs have shown some results by reduced NPS pollution in multiple systems throughout America including the Chesapeake Bay and the Great Lakes (NRCS, 2011a; NRCS, 2011b; NRCS, 2010k; Shortle et al., 2012). This has, however, cost billions of dollars and clear instances of waste can be pointed out (USDA., 2006; Shortle et al., 2012).

2.7 Attitude of Producers

All USDA, NRCS programs promoting and funding BMPs discussed thus far have been voluntary. Therefore, it is important to review the attitude of producers to BMPs. There is a sizable body of literature dedicated to describing factors that lead to producers adopting BMPs, and nationwide, specific conclusions can be difficult to draw from the mass of data (Reimer et al., 2012)

However difficult, understanding producer's attitudes towards BMP implementation is very important to future water quality and natural resource protection, as billions of dollars are spent on BMP related voluntary program funding (Baumgart-Getz et al., 2012; Greiner et al., 2009). Baumgart-Getz et al. (2012) found the most important factors that influence producer's adoption of BMPs are access to quality of information, financial capacity, and a sound connections with watershed and community groups, by surveying 46 BMP studies that spanned 26 years from 1982 to 2007 (Baumgart-Getz et al., 2012). Prokopy et al. (2008) reviewed 55 studies covering over 25 years and reported that education level, capital income, farm size,

access to information, positive environmental attitudes, environmental awareness, and utilization of social networks all show positive correlations to BMP adoption (Prokopy et al., 2008).

Producers who consider themselves strongly economically motivated are less likely to adopt BMPs, while those who listed themselves as predominantly lifestyle and conservation driven are more likely (Greiner et al., 2009). Because each geographical area has a unique set of social attitudes and influences, "...a sound understanding of farmer's motivations and risk attitudes is required—in a regional, industrial, and environmental context..."(Greiner et al., 2009).

Smith et al. report that the main reasons producers do not adopt BMPs are too much government control, few options for producers, heavy paperwork load, low payments, and complicated program information, insufficient information available about programs, and high penalties for failing to meet program requirements. This study is unique compared to the two previous studies mentioned, because rather than profiling producers and finding descriptive factors to correlate with BMP adoption, it lists specific reasons producers do not adopt BMPs from a survey (Prokopy et al., 2008; Greiner et al., 2009). Since "...farmer adoption rates can be improved by focusing on the generally consistent determinates of agricultural BMP adoption," a BMP programs should take into account the regional social climate in the area (Prokopy et al., 2008). Programs should also address the main reasons BMPs are not adopted, e.g., economics (Greiner et al., 2009). Most of the current NRCS programs do not specifically address these important points in an effective way.

2.8 Incorporation of Best Science and Understanding for BMP Implementation

Most BMP programs include a statement requiring prediction technology, and as stated above, the social component of understanding producers attitudes is paramount to program success (NRCS, 2010a; Greiner et al., 2009). Therefore, the most effective BMP programs should combine the best scientific methods of prediction available with an integral understanding of the producer's main concerns about BMPs. A few recently implemented programs have attempted to use this understanding to increase the efficiency of water quality programs (EPA, 2003).

2.8.1 Water Quality Trading

According to the United States Environmental Protection Agency (EPA), roughly 40% of rivers and 50% of lakes have insufficient water quality to support their designated uses, in spite of multiple government sponsored programs with water quality improvement goals (EPA, 2012b; NRCS, 2010a). The EPA further states that "...solutions to...complex water quality problems requires innovative approaches that are aligned with core water programs" (EPA, 2003).

In an effort to improve the effectiveness of water quality programs in this way, the EPA established Water Quality Trading (WQT), a market-based approach (EPA, 2003). WQT is similar to a carbon trading system, where facilities whose pollution reduction costs are higher are allowed to purchase equivalent pollution reductions from another facility or source at a lower cost (EPA, 2012c). The basis of WQT is "... the fact that sources in a watershed can face very different costs to control the same pollutant" (EPA, 2012c). WQT is designed to provide assistance and guidance to states, Tribes, and interstate agencies for developing their own water quality trading programs (EPA, 2003). According to the EPA market-based approaches are more

flexible and are more likely to achieve water quality improvement goals than traditional regulations (EPA, 2003).

As of 2005, there were 30 trading programs sponsored by WQT in various states, and currently there are forty eight (Peterson et al., 2005; EPA, 2012d). WQT trading programs exist in 26 states, seven of which have a statewide trading framework in place (EPA, 2012d).

Pollutants are not necessarily being reduced simply because a WQT program is in place (Peterson et al., 2005). Although programs exist, trading may not actually be taking place (Peterson et al., 2005). In fact as of September 2011, thirteen of the programs had not led to any trading (EPA, 2012d). Many of the other programs have few trades, in fact, Peterson et al. state that "...the most commonly noted feature of existing [WQT] programs is low trading volume" (Peterson et al., 2005). Low trading volumes are due primarily to high trading ratios, and limited information between trading agreements (Peterson et al., 2005). High trading ratios simply mean that each calculated unit of pollution in the trade system must be offset by a similar reduction (Peterson et al., 2005). Though this system helps to ensure loading levels within the WQT program bounds, economists liken it to transaction costs, and in turn with low trading volumes (Peterson et al., 2005). Information between parties is limited in the trade agreements, as potential traders often do not know all the trade prices available within the WQT framework, and so are less likely to make a trade (Peterson et al., 2005). Although WQT addresses some of the problems with current BMP programs, it fails to gain enough participation to be effective and efficient.

2.8.2 BMP Auction

The BMP auction was created in an effort to address low levels of participation in voluntary water quality programs (Smith et al., 2009). The BMP auction is built to address producers' main concerns, listed by Smith et al. (2009), in order to increase participation. A BMP auction is a program in which producers submit bids by defining their own prices for management and implementation of BMPs. These bids are ranked by effectiveness and funded accordingly (Smith et al., 2009). A BMP that is cost effective is one that shows a high environmental benefit per dollar spent (such as highest sediment runoff reduction per dollar) (Smith et al., 2009). The intent of the program was to ensure the most efficient bids will receive funding (Smith et al., 2009).

In addition to assessing efficiency, the BMP auction allows the bidders to request their own prices for implementing BMPs (Smith et al., 2009). This creates a competitive environment, which will drive BMP prices down, allowing the government agency applying the program to decrease water pollution using less funding (Smith et al., 2009). By given price naming to the producers, the government control over the programs seems less intrusive, thus addressing the producer concern of invasive government management of programs and low payouts. Although producers can name their own price, a maximum is usually set; this is set to a reasonable amount for both producer and agency (Smith et al., 2009). Using the BMP auction, agency managers can create competition for funding, and pay for the most effective BMPs (Smith et al., 2009).

The BMP auction relies heavily on stakeholder support (Smith et al., 2009). In fact Smith et al. suggest using local personnel with rapport among producers to market a proposed auction

(Smith et al., 2009). This group of people is called the “stakeholder leadership group” (Smith et al., 2009). By using this leadership group, producers will be more likely to join the auction and bid for BMP funding (Smith et al., 2009).

A specific list of potential BMPs is used to simplify the auction, and a universal, yet simple signup sheet is created for producers to send in bids, thus reducing paperwork to a minimum (Smith et al., 2009). Producers are encouraged to submit multiple bids, but not exceed a set maximum payment per farm (Smith et al., 2009). This streamlined approach addresses the producers’ concerns of complicated programs and too much paperwork.

In recent literature, the auction format for water quality programs has received praise (Thurston et al., 2008; Rolfe and Windle, 2011; Smith et al., 2009). A BMP auction has received funding and has been implemented successfully in the Pomona Lake Watershed in Kansas (Smith et al., 2009). The BMP auction is a novel approach that addresses producers’ main concerns about joining voluntary BMP programs and allows funding agencies to make effective decisions on how funds should be allocated to reduce pollution most efficiently.

2.9 Available Modeling Tools

The BMP auction requires ranking bids comparatively by modeling their outcomes in some way (Smith et al., 2009). Almost every NRCS program dedicated to improving water quality states the need for predictive technologies in funding decisions (NRCS, 2012a). Therefore, in order for the BMP auction to be as successful as possible, the best modeling tools and strategies should be used to rank the bids. The following are descriptions and explanations of advantages and disadvantages of the most common modeling tools used in water quality prediction applications.

2.9.1 Spreadsheet Tool for Estimating Pollutant Load (STEPL)

STEPL is a watershed scale spreadsheet tool for estimating pollutant loads at stream outlets (EPA, 2012e). Simple algorithms are used to estimate loads for sediment and nutrients (EPA, 2012e). The output of the model is a spreadsheet quantifying surface runoff, nutrient loads, 5-day biological oxygen demand (BOD5), and sediment delivery (EPA, 2012e). Also included are estimated reductions in these loads based on the implementation of BMPs in the watershed (EPA, 2012e). An online data access system exists for users to define the input data (Tetra Tech, 2012).

2.9.1.1 STEPL Model Components

STEPL requires user input for land use, animals present in the watershed, precipitation and irrigation, soil and USLE parameters, and presence of septic systems or point discharges (Tetra Tech, 2005). Model components include runoff, groundwater, sheet and rill erosion, gully and stream bank erosion, and pollutant transport (Tetra Tech, 2005). Annual sediment load estimation is calculated with the universal soil loss equation and sediment delivery ratio (Nejadhashemi and Mankin, 2007; Nejadhashemi et al., 2011). Pollutant concentrations in runoff volume are used to calculate annual nutrient loading; these factors are influenced by management practices and land (Nejadhashemi and Mankin, 2007; Nejadhashemi et al., 2011). Load and sediment reductions are calculated with predefined BMP efficiencies (Nejadhashemi and Mankin, 2007; Nejadhashemi et al., 2011).

2.9.1.2 STEPL: Published Work and Applications

There are not many articles published that employ STEPL. It was designed for the EPA to make management decisions on a watershed scale (EPA, 2012e). Since the model only calculates these loads on an annual basis is usually used for estimations over a long period of time (Nejadhashemi and Mankin, 2007). STEPL is appropriate for assessing current situations in management practice strategies in preliminary planning stages. (Nejadhashemi and Mankin, 2007)

2.9.2 GIS Pollutant Load Application (PLOAD)

CH2M HILL designed the PLOAD model for calculating NPS pollutant loads in watersheds on an annual and seasonal basis (EPA, 2001). Any best management practices applied within the watershed the watershed boundary and land-use including event mean concentrations or export coefficients for each land-use are required to run the model (POLAD1). (Nejadhashemi and Mankin, 2007; Nejadhashemi et al., 2011) This model is appropriate to use when there is a significant amount of uncertainty in the effectiveness of watershed management practices or the reliability of total maximum daily load threshold (EPA, 2001; Nejadhashemi and Mankin, 2007). Some modifications to this model include tools that show nonpoint source pollution as tables and maps and can compare different model runs (Nejadhashemi and Mankin, 2007).

2.9.2.1 PLOAD Model Components

The EPA Simple Method and the curve number method are used to calculate mean event concentrations and runoff (EPA, 2001). Event mean concentrations are calculated for each land use are calculated and used as inputs in the EPA Simple Methods to calculate pollutant

loads (EPA, 2001). Other inputs included in the EPA Simple method are ratio of runoff producing storms and a land use specified runoff coefficient (EPA, 2001).

2.9.2.2 PLOAD: Published Work and Applications

PLOAD is not very prevalent in published studies, though some water quality and NPS pollution studies exist that discuss the PLOAD model (Nejadhashemi and Mankin, 2007; Nejadhashemiet al., , 2011) (Bosch et al., 2004; Endreny and Wood, 2003; Renaud et al., 2006) Although PLOAD is relatively easy to use, it is not recommended for extensive modeling (Nejadhashemiet al., , 2011).

2.9.3 Long Term Hydrologic Impact Analysis (L-THIA)

L-THIA was designed by the College of Engineering at Purdue University to compare the impacts of differing land-use scenarios (Bhaduri et al., 2000). The model uses daily runoff from climate records the current number soil data and the event mean concentrations and land-use to calculate average annual runoff for each land-use and management combination. L-THIA is a model dedicated to long-term analysis reporting yearly averages of these long-term periods. The model is focused on comparing the impacts of different land-use scenarios (Bhaduri et al., 2000). L-THIA outputs runoff and nonpoint source pollution in graphs and tables (Bhaduri et. al., 2001).

2.9.3.1 L-THIA Model Components

The structure for L-THIA is a lumped parameter model (Bhaduri et al., 2000). Annual runoff is calculated with the curve number method, but does not consider snow melt in the

calculations (Bhaduri et al., 2000). Daily runoff from climate records, soil type, and the event mean concentrations, and land-use are used to calculate average annual runoff for each land-use and management combination (Bhaduri et al., 2000).

2.9.3.2 L-THIA: Published Work and Applications

A number of studies report using L-THIA for NPS pollution prediction in various watersheds (Bhaduri et al., 2000; Tangi et al., 2005; Choi et al., 2005; Muthukrishnan, 2002; Yang et al., 2006; Lu et al., 2009). L-THIA is easy to use and requires simple, available inputs; however since it employs the current number method it is not as effective an area where snow and permanently frozen soils significantly impact the watershed (Bhaduri et. al., 2000). Also, it is not effective in areas with highly variable moisture conditions (Bhaduri et. al., 2000).

2.9.4 Hydrologic Simulation Program-FORTRAN (HSPF)

The EPA designed the HPSF model for predicting watershed effects to land use change, as well as the effects of NPS and point source pollution (Bhaduri et al., 2000). In-stream processes are combined with overland flow to estimate loads. The HSPF model uses hydrologic response units to model overland flow (Bhaduri et. al., 2000). A hydrologic response unit (HRU) is a unit of the watershed with uniform characteristics in each descriptive category: land use type, soil type and slope (USGS, 2010).

2.9.4.1 HSPF Model Components

For each HRU, surface flow and interflow and ground flow are calculated based on infiltration and meteorological estimations (Johnson et. al., 2003; EPA, 2000). These results are averages and represent each hydrological response unit's response to existing conditions. This

model allows the processes within the watershed that are most important to be exemplified by the user. Actual physical data from watersheds concerning surface and ground water flow can be hard or impossible to measure. Therefore, for this model, like many other large-scale watershed hydrological models, the parameter values are often found through calibration. Calibration involves adjusting the necessary parameters in order to make the predicted values satisfactorily similar to the measured values (Johnson et. al., 2003).

2.9.4.2 HSPF: Published Work and Applications

This model has been widely used in documented in literature (USGS, 2010; Donigian, 2002; Chen et al., 1998; Laroche et al., 1996; Johnson et al., 2003; Van Liew et al., 2003; Albek et al., 2004; Ackerman et al., 2005). The EPA has used this model for total maximum daily load (TMDL) analysis through the watershed data management software BASINS. Watersheds of small and large scale have been modeled successfully with HPSF. It is appropriate for predicting flow and sediment for yearly and monthly intervals, however severe weather significantly reduces the model's effectiveness (Borah et. al., 2004). Point and nonpoint sources can be calculated, however as the time interval is reduced the model becomes less and less accurate. (Borah et. al., 2004). HPSF can be helpful to study land-use change and pollution management scenarios (Borah et. al., 2004).

2.9.5 Annualized Agriculture Non-Point Source Model (AnnAGNPS)

AnnAGNPS estimates NPS pollution through continuous simulation daily time step calculations (Bosch et al., 2001). This model is based on the idea of flow between discrete cells; these cells effectively form a raster set dividing the entire watershed with uniform square areas (Polyakov et. al., 2007). By dividing the watershed into discrete cells of similar land use, soil

type, and slope, surface runoff, sediment runoff, nutrients and pesticides can be calculated at multiple places throughout the watershed (Bosch et al., 2001). AnnAGNPS models conservation practices, including some BMPs, and their effects on pollution in the watershed (Bosch et al., 2001). The AnnAGNPS model has many input parameter categories, 34 in all (Polyakov et al., 2007). Model outputs can be obtained from any cell at a user-specified time (Bosch et al., 2001). The model also includes an input editor to easily manipulate parameters after model set up, and can be integrated with GIS (Polyakov et al., 2007).

2.9.5.1 AnnAGNPS Model Components

AnnAGNPS includes components for hydrology, sediment nutrient and pesticide transport, irrigation, precipitation, and snowmelt (Bosch et al., 2001). Chemical oxygen demand is used as an indicator of the degree of pollution (Polyakov et al., 2007). The curve number method is used to calculate surface runoff, and Manning's equation for channel runoff (Bosch et al., 2001). The weather data used in the model can be either observed or simulated (Bosch et al., 2001). Darcy's equation is used for lateral subsurface flow, and Hooghoudt's equation for flow through tile drains (Bosch et al., 2001). AnnAGPS incorporates the Revised Universal Soil Loss (RUSLE) equation to estimate sediment loads (Das et al., 2007). The transport of sediment through channels is modeled with the revised Einstein equation (Bosch et al., 2001).

2.9.5.2 AnnAGPS: Published Work and Applications

AnnAGPS is widely used and documented to estimate NPS pollution loads in watersheds (Bosch et al., 2001; Das et al., 2007; Zema et al., 2012; Pease et al., 2010; Polyakov et al., 2007; Borah, 2011; Parajuli et al., 2009; Kliment et al., 2008). These studies focus on the evaluation of conservation practices, land use change and BMPs. Different watersheds can be compared in

therefore problem areas can be identified. The effect of management operations can then be assessed using the model as well. This is easily done by adjusting the input parameters to be consistent with the proposed management practices (Polyakov et. al. 2007).

2.9.6 Soil and Water Assessment Tool (SWAT)

The Soil and Water Assessment Tool (SWAT) is a semi-distributed, continuous simulation model with a daily time step (Arnold et al. 1998). Many management and land use situations can be model with SWAT in gauged and ungauged watersheds (Karamouz et al. 2010). The SWAT model can simulate sediment, nutrient, and pesticide movement in multiple forms through all bodies of water in a watershed (Arnold et al. 1998). Point and non-point sources of many types of pollutants can be evaluated using the SWAT Model (Arnold et al. 1998). The SWAT model is a widely used and accepted tool for whole watershed scale modeling applications for tracking sediment, nutrient, and other pollutant transport (Arabi et al. 2006).

2.9.6.1 SWAT Model Components

Included in the SWAT model are components for hydrology, soil and vegetative processes (Kemanian et al., 2011). The model splits whole watersheds into sub watersheds and further yet into HRUs (Kemanian et al., 2011). Water flow and pollutants are routed to watershed outlets through streams (Kemanian et al., 2011). Inputs include data (measured or simulated) for temperature and precipitation, soil and land use maps, slope classification, and management practices (Nejadhashemi and Mankin, 2011). Surface runoff is calculated using the modified SCS curve number method, and the Modified Universal Soil Loss Equation (MUSLE) estimates sediment yields for each HRU (Saleh and Du, 2004; Neitsch et al., 2005). Flow is modeled with Mannings equation (Neitsch et al., 2005).

2.9.6.2 SWAT: Published Work and Applications

The SWAT model is used often for variety of watershed modeling applications (Kemanian et al., 2011; Saleh and Du, 2004; Neitsch et al., 2005; Gassman et al., 2010; Yanga et al., 2009; Vieth et al., 2010) (Parajuli et al., 2009; Bosch et al., 2010; Luo et al., 2011; Boscha et al., 2011; Schuola et al., 2008; Cibin et al., 2010; Zhang et al., 2011; Gassman et al., 2007). By 2007, 250 peer reviewed publications used or discussed the SWAT model (Gassman et al., 2007). The number of published works that employ this model have grown significantly since. Although SWAT has some problems with accurate digital elevation model (DEM) subbasin delineation and does not identify braided streams, it is a very useful and effective watershed modeling tool (Luo et al., 2011; Gassman et al., 2007).

2.9.7 Revised Soil Loss Equation 2 (RUSLE2)

RUSLE2 is a process and empirically based model that predicts rill and inter-rill erosion caused by runoff and rainfall (Muthukrishnan, 2002). It is used to predict annual average rates of erosion per unit area over long periods of time (NRCS, 2003b). RUSLE2 is intended to inform conservation planners and estimate sediment deliveries on a field scale, It is not a whole watershed scale model (NRCS, 2003b).

2.9.7.1 RUSLE2 Model Components

RUSLE2 requires database inputs for soil type, climate, and crop management zones, and inputs of slope length, percent slope, and management operations per field (NRCS, 2003b). RUSLE2 is based on the Universal Soil Loss Equation (USLE), which includes aggregate factor values for rainfall erodibility, soil erodibility, slope length and percent, and the impacts of

cropping management systems (RSULE4). RUSLE2 is designed to be used with available databases constructed for the USLE equation (Yoder et al., 2003).

2.9.7.2 RUSLE2: Published Work and Applications

Some publications employ RUSLE2 for sediment load calculations (Foster et al., 2001; Pal and Al-Tabbaa, 2009). It is mainly used by the NRCS to catalogue soil erosion and evaluate and predict results from conservation efforts (NRCS, 2003b). RUSLE2 is effective for one-field-at-a-time conservation evaluation and planning, where fields have homogenous management practices, soil types, and slope properties. Since the output, average annual soil loss per acre, is normalized over area, a large area such as a watershed cannot be modeled with RUSLE2. The reason for this is as area increases, there are more and more heterogeneous inputs, and the prediction becomes impossible.

2.9.8 High Impact Targeting (HIT)

HIT is a combination of two models, the RUSLE model and the Spatially Explicit Delivery Model (SEDMOD) (IWR, 2010). RUSLE estimations are used for soil load estimation; these estimations are then used as inputs to the SEDMOD model in order to simulate stream routing (IWR, 2010). The model outputs field scale maps highlighting problem areas and sediment load estimations at the watershed scale (IWR, 2010).

2.9.8.1 HIT Model Components

SEDMOD is cell based, and uses flow path slope gradient, flow path slope shape, flow path hydraulic roughness, stream proximity, soil texture, and overland flow to model stream processes (Ouyang et al., 2005). The SEDMOD outputs are delivery ratio estimates for a

spatially explicit areas in a watershed given clay content, elevation, and land use inputs (Ouyang et al., 2005). Sediment delivery ratio is calculated using the drainage area of the specified field and extra adjustment made based on nearby conditions (Ouyang et al., 2005). The RUSLE model (very similar to RUSLE2) calculates an annual sediment erosion value normalized over the area defining the input boundaries (O’Neil, 2011). The targeting maps are found by combining these results (O’Neil, 2011).

2.9.8.2 HIT: Published Work and Applications

This hybrid RUSLE/SEDMOD model approach was first presented in 2005, though it was not yet deemed “high impact targeting,” and the data frame work was not yet built (Ouyang et al., 2005). However, no peer reviewed studies were found that use the HIT model for sediment erosion and delivery studies since its introduction. Although SEDMOD calculates delivery ratios for sediment in a watershed, it does not fully account for in-stream processes that vary throughout the year.

2.9.9 Agriculture Policy/Environmental eXtender (APEX)

The purpose of the APEX model is to evaluate the effects of management practices on agricultural land (Williams and Izaurralde, 2005). The model runs on a daily time step, and can run simulations for very long periods of time (Williams and Izaurralde, 2005). APEX can simulate a number of management practices including irrigation, drainage, furrow diking, buffer strips, terraces, waterways, fertilization, manure management, lagoons, reservoirs, crops rotations, pesticide applications, grazing, and tillage (Williams and Izaurralde, 2005). APEX was designed to model on a farm scale, which can be sub-divided into fields (Williams and Izaurralde, 2005).

2.9.9.1 APEX Model Components

APEX includes components for weather, hydrology, soil erosion, manure erosion, pesticide fate, soil temperature, crop growth, tillage, plant environmental control, and economics (Williams et al., 2008). Surface runoff is predicted with the SCS curve number, and the Green and Ampt Method is used for infiltration (Williams et al., 2008). The TR-55 Method estimates peak runoff and both vertical and horizontal flow patterns are considered in subsurface flow (Williams et al., 2008). Multiple options are available to the user for evapotranspiration calculations: the Hargreaves and Samani, Penman, Priestley-Taylor, Penman-Monteith, and Baier-Robertson methods (Williams et al., 2008). Soil erosion is estimated with seven different methods, including modifications of the RUSLE, USLE, and Modified Universal Soil Loss Equation (MSULE) (Williams et al., 2008).

2.9.9.2 APEX: Published Work and Applications

Many studies have used the APEX model to assess management practices, and estimate NPS pollution loads on the farm and small watershed scale (Santhi et al., 2008; Gassman et al., 2002; Gassman et al., 2004; Mudgal et al., 2008; Tuppad et al., 2009; Seleh et al., 2003; Wang et al., 2008). APEX is a sophisticated tool with many user options (Williams and Izaurrealde, 2005; Williams et al., 2008; Santhi et al., 2008).

2.10 Model Comparison Summary

The following table compares the previously described models over six categories: model scale, time step, major inputs, major outputs, BMP simulation capability, and whether or not the

model has significant support for its stated application in literature. These categories help compare the models' operation, BMP evaluation potential, and effectiveness.

Table 2-2. Comparison of Currently Used Watershed and Field Scale Models

MODEL	Scale	Time Step	Major Inputs	Major Outputs	BMP Simulation Capability	Relative Prevalence in Literature
STEPL	Watershed	Single Event	*Landuse *Herd Presence *USLE Parameters *Point Source Presence	Annual sediment load estimation	Some BMPs	Low
PLOAD	Watershed	Annual	*Management practices applied in watershed *Soil Data	Annual Nutrient and sediment loads	Management BMPs	Very Low
L-THIA	Watershed	Daily	*Daily weather *Soil Type *land use data	Annual pollutant loads	Management BMPs	High
HSPF	Watershed	Single Event	*Weather *Soil *Elevation *Land use	Single Event Pollutant loads	Management BMPs	High
AnnAGNPS	Watershed	Daily	*Land Use *Soil type *Elevation	Sediment, nutrient and pesticide loads	Some BMPs	High

Table 2-2 (cont'd)

SWAT	Watershed	Daily	*Daily Weather *Land use *Elevation *Soil Type *Management Practices	Sediment, nutrient, pesticide loads on an annual, monthly, or daily basis	Many BMPs	High
RUSLE2	Field	Annual	*Weather *Soil type *Management practices	Average Annual soil loss per unit area	Comprehensive list of BMPs	Medium
HIT	Watershed	Annual	*Weather *Soil Type *Management Practices *Clay percentage	Average annual sediment delivered to stream	Some BMPs	None
APEX	Farm (small Watershed)	Daily	*Daily Weather *Soil type *Management practices *Elevation *Land use	Annual, monthly, or daily pollutant loads in stream or from subbasin	Many BMPs	High

3. INTRODUCTION TO METHODOLOGY AND RESULTS

The content of this thesis is in the form of two research papers that were submitted to scientific journals. The first paper is titled “Evaluating the Impact of Field-scale Management Strategies at the Watershed Outlet” and had a goal of quantifying the reliability of some widely used environmental models to make field-scale sediment runoff estimations. The Raisin River watershed was chosen as the study area, which is located in southeast Michigan roughly 100 km southwest of Detroit, MI. Three watershed scale models capable of providing field-scale sediment yield estimations (SWAT, Field_SWAT, and HIT) were tested against RUSLE2, a calibrated, detailed field scale model. Forty-one fields with site-specific management practice information were randomly selected to test the reliability of the three watershed models in quantifying sediment yield for every field. Information gained from this exercise was used in the next study to advise the design of the best method aimed at quantifying the environmental and economic impacts of field-scale management changes at the watershed outlet.

The second paper is entitled “Evaluating the Impact of Field-scale Management Strategies at the Watershed Outlet.” In this study, four methods were proposed and tested with the goal of providing realistic sediment yield savings estimations at the watershed outlet based on field-scale management operations. Watershed-scale sediment reduction loads from 80 field-scale BMP scenarios were evaluated at both field and watershed outlets in the River Raisin watershed. The applicability, advantages, and disadvantages of these approaches are discussed in this study. Finally, an economic analysis was performed to compare producer requested prices versus the prices defined by the USDA’s Environmental Quality Incentives Program (EQIP) for BMP implementation.

4. EVALUATING THE CAPABILITIES OF WATERSHED-SCALE MODELS IN ESTIMATING SEDIMENT YIELD AT FIELD-SCALE

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4.1 ABSTRACT

Many watershed model interfaces have been developed in recent years for predicting field-scale sediment loads. They share the goal of providing data for decisions aimed at improving watershed health and the effectiveness of water quality conservation efforts. The objectives of this study were to: 1) compare three watershed-scale models (Soil and Water Assessment Tool (SWAT), Field_SWAT, and the High Impact Targeting (HIT) model) against calibrated field-scale model (RUSLE2) in estimating sediment yield from 41 randomly selected agricultural fields within the River Raisin watershed; 2) evaluate the statistical significance among models; 3) assess the watershed models' capabilities in identifying areas of concern at the field level; 4) evaluate the reliability of the watershed-scale models for field-scale analysis. The SWAT model produced the most similar estimates to RUSLE2 by providing the closest median and the lowest absolute error in sediment yield predictions, while the HIT model estimates were the worst. Concerning statistically significant differences between models, SWAT was the only model found to be not significantly different from the calibrated RUSLE2 at $\alpha = 0.05$. Meanwhile, all models were incapable of identifying priorities areas similar to the RUSLE2 model. Overall, SWAT provided the most correct estimates (51%) within the uncertainty bounds of RUSLE2 and is the most reliable among the studied models, while HIT is the least reliable.

The results of this study suggest caution should be exercised when using watershed-scale models for field level decision-making, while field specific data is of paramount importance.

4.2 INTRODUCTION

Agricultural non-point source (NPS) pollution significantly threatens the water quality of lakes and streams in the United States (EPA 2005). Excess sediment loads are an important link between agricultural land use and water quality (Bossio et al., 2010). Therefore, in the United States and worldwide, several publicly sponsored programs have been established with the goal of reducing NPS from agricultural lands (Shortle, 2012). These programs have had mixed success, and have met obstacles in effectively reducing NPS primarily due to a lack of data concerning the sources and amounts of excess pollution loads (Thomas and Froemke, 2012). By collecting and synthesizing watershed data, water quality improvement projects can be made more effective, as more information allows for better decision-making. Meanwhile, conducting monitoring to quantifying pollution sources throughout a watershed is usually infeasible due to economic and time constraints. Under these conditions, watershed models can be employed as an alternative method capable of providing data on NPS pollution quantities and sources (Daggupati et al., 2011).

Many government sponsored water quality improvement projects rely on watershed scale analysis. For this reason, watershed models such as the Soil and Water Assessment Tool (SWAT), the water Erosion Prediction Project (WEPP), the Annualized Agricultural Non-Point Source (AnnAGNPS) model, the Long-Term Hydrologic Impact Assessment model (L-THIA), the PLOAD model, and the Hydrological Simulation Program - FORTRAN (HSPF) model have been used extensively to quantify NPS and the effects of BMP implementation on water quality

(Shen et al., 2009; Parajuli et al., 2009; Im et al., 2009; Nejadhashemi et al., 2011; Giri et al., 2012). However, there are scaling concerns with these modeling approaches. Downscaling the watershed-scale results to individual fields, which are the target of conservation practices, is difficult because the majority of watershed model outputs are generated at the subbasin level or smaller scales but lumped together based on physiographic characteristics. For example, in the SWAT model, areas with homogeneous land use, soil type, and slope are lumped into hydrologic response units (HRUs), which can vary in size from less than one hectare to hundreds of hectares. At the same time, these lumped units do not follow actual field boundaries and therefore, it is difficult and likely inaccurate to provide field specific recommendations regarding conservation practice implementation strategies. In addition, the most common agricultural practice information is incorporated at subbasin or HRU levels in watershed models, which is not site or field specific.

In the past few years, a series of watershed model interfaces were developed claiming to provide field-scale information from watershed models, such as the High Impact Targeting (HIT) model (O'Neil, 2010; Bartholic, 2009) and Field_SWAT (Pai et al., 2011). However, the results of these new downscaling techniques have not been tested against calibrated field-scale models such as the Revised Universal Soil Loss Equation, Version 2 (RUSLE2), which is the goal of this study. The primary objective of this study is to compare sediment yield estimations from Field_SWAT, HIT, and a detailed SWAT model that was delineated according to agricultural field boundary maps in the River Raisin Watershed in Michigan. The specific objectives of this study are to: 1) compare watershed-scale modeling approaches against a calibrated field-scale model (RUSLE2) in estimating sediment yield from 41 agricultural fields randomly selected within the River Raisin Watershed; 2) evaluate the statistical differences among modeling

estimations; 3) assess the models' capabilities in identifying areas of concern; and 4) evaluate the reliability of the watershed models in estimating field-scale sediment loads.

4.3 METHODOLOGY

4.3.1 Study Area

This study was performed using data from the River Raisin Watershed, located in southeast Michigan and northern Ohio (Figure 1). The main reach in this watershed is the River Raisin, which flows into Lake Erie near Monroe, Michigan. The watershed is comprised of 268,000 ha of land located in Hillsdale, Jackson, Lenawee, Monroe, Washtenaw, and Fulton counties. Most of the land in the River Raisin watershed is used for agriculture (60%), while the remaining land is categorized as forest (13%), urban (12%), wetlands (7%), and range grass and shrubs (1%), according to the National Land Cover Database (NLCD, 2009). Corn, soybeans, wheat, and pasture are the most common agricultural uses, with corn and soybeans produced on over half of the agricultural area. The watershed elevation ranges from 121 m to 391 m above sea level, with an average elevation of 300 m.

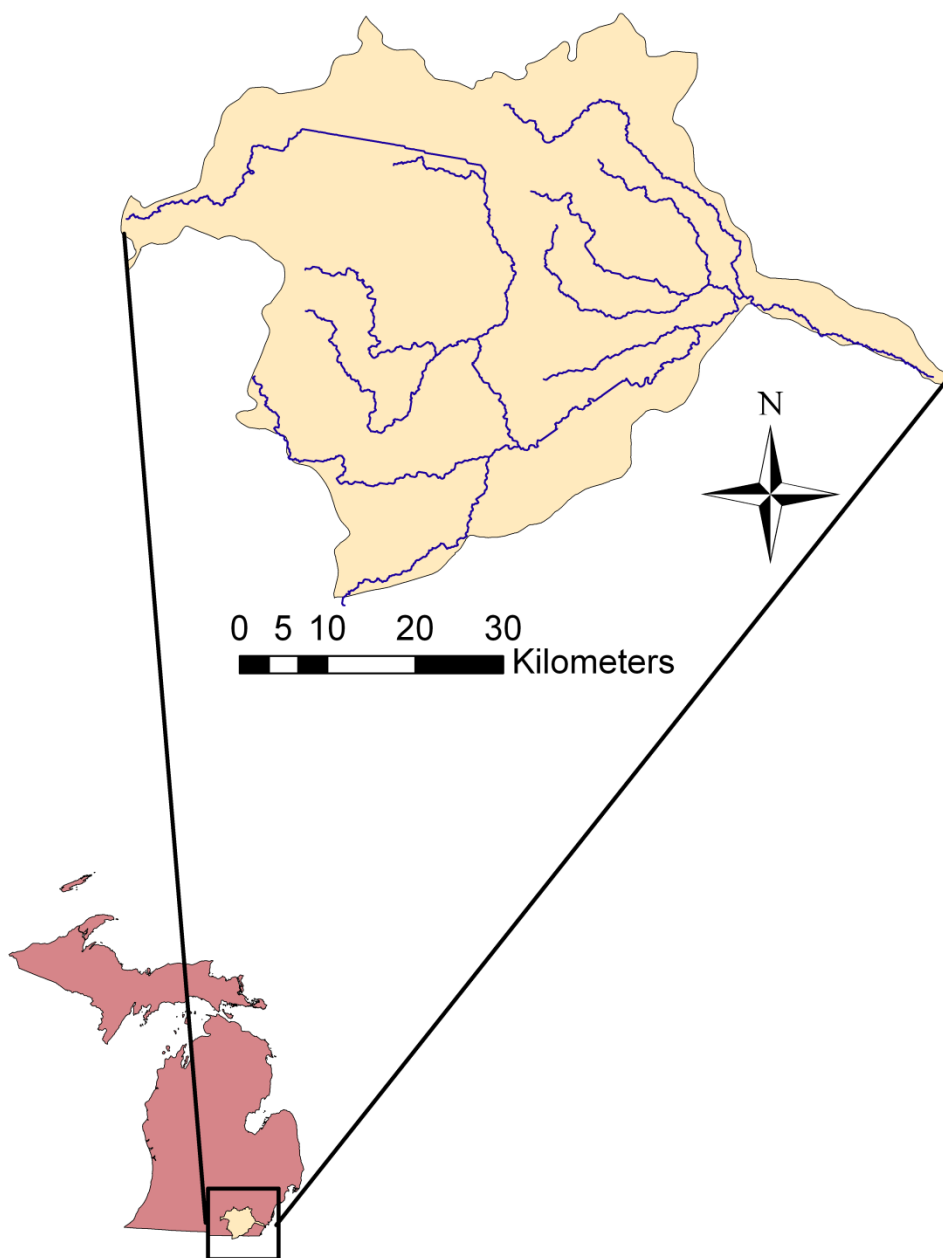


Figure 4-1: Saginaw River watershed. For Interpretation of the references to color in this and all other figures, the reader is referred to the electronic version of this thesis.

4.3.2 Data Inventory

Various datasets were collected, including streamflow, stream sediment concentration, physiographical characteristics of the watershed, climatological data, and both specific and common agricultural operations and rotations.

Two sets of data (streamflow and sediment concentration) were collected for watershed model calibration. The mean daily streamflow data was collected from USGS station 04176500 near Monroe, Michigan, for the time period of January 1, 1983 through December 31, 2011. In addition, total suspended solids (TSS) measurements were obtained from the US Environmental Protection Agency (USEPA) Storage and Retrieval (STORET) database. The STORET database contains a wide range of water quality monitoring data from many sampling locations across the United States. There were 129 measurements available at sampling location 580046 located near the River Raisin watershed outlet. Daily TSS data were collected from July 6, 1998 to October 3, 2005. The data were not continuous, averaging 1.5 observations per month.

Physiographic characteristics that were used include elevation, land use, agricultural field boundaries, and soil datasets. Elevation data was obtained from the US Geological Survey (USGS) National Elevation Dataset (NED). The USGS provides a 30 m resolution elevation map that is available for the continental United States. For land use, the Cropland Data Layer (CDL 2007) is available in a 56 m resolution raster format. Aerial maps and Common Land Unit (CLU) boundaries created by the NRCS were used to catalogue agricultural fields. Soil data was obtained from the from the Natural Resources Conservation Service (NRCS) National Geospatial Management Center in the form of the Soil Survey Geographic Database (SSURGO). The soil

map is available in raster format at 1:24,000 scale and consists of 433 unique soils in the study area.

Daily precipitation and maximum and minimum temperatures were collected from the National Climatic Data Center (NCDC) Cooperative Data Network (COOP). Data from two stations (209218 and 200032) located within the boundaries of the watershed were used. Climatological data was available for these two stations from January 1, 1970 through December 31, 2011.

Field specific data was collected from 41 fields in the River Raisin watershed. Crop rotations and management operations were obtained for each field through a one-by one producer survey with the help of local NRCS staff. In addition, common land use practices were obtained from US Department of Agriculture (USDA) 2007 Censuses of Agriculture (USDA, 2007).

4.3.3 Model Descriptions

Four models (RUSLE2, SWAT, Field_SWAT, and HIT) were used to calculate sediment yield for 41 fields randomly selected within the River Raisin Watershed. These four models are described in the following sections.

4.3.3.1 RUSLE2

The USDA Agricultural Research Service (USDA-ARS) developed the RUSLE2 model in 2003 for the NRCS to assist with field-scale sediment erosion estimation. RUSLE2 was designed to guide conservation planning by estimating rill and interrill erosion caused by rainfall and runoff. The model is based on the Revised Universal Soil Loss Equation (RUSLE), which

estimates average annual sediment yield per unit area based on soil properties, land use, and daily precipitation and temperature data.

$$A = R \cdot K \cdot L \cdot S \cdot P \quad (4-1)$$

The average annual soil loss (A) per unit area is calculated using the following parameters: climate erodibility (R) that is based on rainfall and temperature, soil erodibility (K) measured under standard conditions, the slope length (L), the slope steepness (S), the land cover management factor (C), and the support practice factor (P) (Foster, 2003). User inputs include weather data, soil database, location database, and management operations. RUSLE2 includes many agricultural practices for creating management operations. Validation of the RUSLE2 model was performed on simulated and natural plots using 2,000 plot-years and 10,000 plot-years of data, respectively.

4.3.3.2 SWAT

The SWAT model is a spatially distributed, physically based watershed scale model developed by the USDA-ARS (Arnold et al., 1998; Gassman et al., 2007). The modeling procedure begins with watershed delineation, which consists dividing the watershed into subbasins based on stream network and topography, and further into HRUs based on homogeneous land use, soil type, and slope. SWAT outputs are calculated at the HRU level, aggregated to the subbasin level, and routed through stream network to the watershed outlet. The SWAT model is capable of simulating a broad list of components including hydrology, land management, weather, plant growth, chemical transport, and nutrient transport (Gassman et al., 2007).

Detachment, transport, and degradation/deposition are the three stages of sediment erosion modeled in SWAT. The MUSLE equation (4-2) is used to estimate the detachment of sediment particles from the land during a rainfall event.

$$sed = 11.8 \cdot (Q_{surf} \cdot q_{peak} \cdot area_{hru})^{0.56} \cdot K_{usle} \cdot C_{usle} \cdot P_{usle} \cdot LS_{usle} \cdot CFRG \quad (4-2)$$

The variable *sed* is the daily sediment yield (metric tons), Q_{surf} represents the surface runoff volume in (mm/ha), q_{peak} is peak runoff rate (m³/s), $area_{hru}$ is the area of HRU (ha), the USLE soil erodibility factor, K_{usle} , assumed as 0.013 m² hr/(m³-metric ton cm), the remaining variables (unitless) C_{usle} , P_{usle} , LS_{usle} , $CFRG$, represent the USLE land cover factor, the USLE support practice factor, the USLE topography factor, and the coarse soil fragment factor, respectively (Neitsch et al., 2005).

The transport stage, or the amount of detached sediment that enters the reach, is modeled using equation 4-3.

$$sed = (sed' + sed_{stor,i-1}) \cdot \left(1 - \exp \left[\frac{-surlag}{t_{conc}} \right] \right) \quad (4-3)$$

The variable *sed* is equal to the sediment discharged into the main channel on a given day (metric tons), sed' represents sediment yield from the HRU on a given day (metric tons), the mass of sediment stored from the preceding day is represented by $sed_{stor,i-1}$ (metric tons), the surface runoff lag coefficient is *surlag*, and t_{conc} is the time of concentration for a subbasin (hr) (Neitsch et al., 2005).

This final stage of sediment transport in a watershed is dynamic, as sediment may be deposited or move about in a stream network depending on environmental factors. In order to model this phenomenon, SWAT employs a maximum sediment concentration value, which acts as a threshold. This value determines whether deposition or degradation will occur as a function of the peak runoff rate (Neitsch et al., 2005).

4.3.3.3 Field_SWAT

Field_SWAT is a graphical user interface tool for mapping SWAT HRU outputs to field boundaries. The MATLAB program environment was used to develop the interface but is not required for operation of Field_SWAT. Field_SWAT aggregates water runoff and sediment yield based on HRUs from an existing SWAT project. The main purpose of Field_SWAT is to aid users in visualization of SWAT outputs at the field-scale and is listed as a potential tool for field-scale targeting of conservation programs (Pai et al., 2011). The field level HRU aggregation output is described by equations 4-4 and 4-5 below.

$$\sum_{i=1}^m x_i dt = v \quad (4-4)$$

In the above equation, x_i represents a value in a vector $X(t) = (x_1, x_2, \dots, x_i)$ of a particular SWAT response variable for a subbasin (the value of water or sediment runoff from HRU i in the subbasin at time t). The variable v represents the daily, monthly or annual SWAT output from a subbasin. The number of HRU is the subbasin of interest, represented by m . Therefore, equation 4 computes the summation of all HRU outputs at time step t in the subbasin.

In order to capture the model output for individual fields for the above subbasin, equation 4-5 is introduced:

$$\int_{j=1}^n y_j dt = w \quad (4-5)$$

In the above equation, y_j represents the j th value of a vector $V(t) = (y_1, y_2, \dots, y_j)$ containing instantaneous values of the SWAT model outputs for a subbasin. The number of fields (from the field boundary layer created by the user) within the subbasin is n , and w is the daily, monthly, or annual SWAT output from a subbasin. Equations 4 and 5 comprise the mapping algorithm in Field_SWAT, and are subject to the constraint that the value of v be equal to the value of w .

4.3.3.4 The High Impact Targeting (HIT) Model

The High Impact Targeting (HIT) model was designed by the NRCS, the Michigan Department of Agriculture (MDA), the Huron Conservation District, and the Michigan State University Institute of Water Research to identify highly erosive areas in a watershed (Koches, 2010). It is an online tool that uses geographic information systems (GIS) to display high impact area maps. The purpose of the HIT model is to give conservation districts, watershed groups, or other watershed stakeholders the necessary information to make decisions and create plans to reduce sediment erosion within a watershed (Koches, 2010).

The HIT model inputs include land use, soil clay content, and elevation, soil erodibility, rainfall, and support practice factors to calculate the mass of soil erosion (VanderMolen, 2010). The main outputs of the model are field-scale maps identifying high risk areas for sediment erosion and loading, and total erosion and sediment estimations at a watershed scale. (vanderMolen, 2010).

The HIT model is a combination of the Revised USLE model and the Spatially Explicit Delivery Model (SEDMOD), and estimates annual erosion and sediment loads entering streams (Koches, 2010). The SEDMOD model is used to estimate the percentage of eroded soils that enter the reach system in a given area, and the Revised USLE model is used to estimate the amount of soil erosion (Vandermolen, 2010).

4.3.4 Model Calibration/Validation

In this study, the calibration procedure involved calibrating two SWAT projects. In the first SWAT project, predefined subbasin and stream network maps were used, while in the second SWAT project (Field_SWAT), the subbasin and river network maps were delineated by the SWAT automatic watershed delineation tool. Both the SWAT and the Field_SWAT models were calibrated according to guidelines described by Moriasi et al. (2007). Based on the average monthly values, for flow and sediment the Nash-Sutcliffe coefficient of efficiency (NSE) should be greater than or equal to 0.5, while the ratio of the root mean square error to the standard deviation of measured data (RSR) should be less than or equal to 0.70. In addition, the percent bias (PBIAS) should be within $\pm 25\%$ for flow, and within $\pm 50\%$ for sediment. The calibration period for flow and sediment for both SWAT models was January 1, 1998 through December 31, 2001, and the validation period was January 1, 2002 through December 31, 2005. Additional calibration is unnecessary for the RUSLE2 model because it has been calibrated using over 10,000 plot-years of data (NRCS, 2003a), while the HIT model cannot be calibrated because model results are preprocessed and incorporated in the model at 10 m resolution.

4.3.5 Field-Scale Sediment Yield Estimations

Among the studied models, RUSLE2 is the only model that was calibrated and validated at the field scale; therefore, the results obtained from the three watershed models will be tested against the RUSLE2 model in order to quantify their effectiveness at predicting sediment yields at the field-scale. According to NRCS guidelines, at least 15 years of climate data and preferably 20 to 30 years should be used to estimate average annual sediment yield at the field-scale (NRCS, 2003b). In addition, this data should be obtained as close as possible to the period of study. Therefore, the most recent climatological data was obtained from two weather stations within the region of study from 1983 to 2011. Among the studied models, only the HIT model does not allow the user to adjust the climate database. However, because the Revised USLE model is an integrated component of the HIT model, it is fair to assume that 30-years of daily climatological data were used. Below, the procedure to obtain long-term average sediment yields from 41 fields within the River Raisin watershed is described for each model.

4.3.5.1 RUSLE2

The most detailed, field-scale data was used in RUSLE2 model in order to create a point of comparison with other models. All slope length, slope percentage, and soil types were identified within the boundary of each field. As described earlier, crop rotations and management operations were also obtained for all 41 fields through the one-by-one producer survey. The results of one field survey are presented in Table 4-1.

In the next step, the RUSLE2 model was run for all combinations of slope, slope length, and soil type within the field of interest. In order to accelerate the process, a spreadsheet toolbox

was designed using Microsoft Excel to assist with RUSLE2 estimations. RUSLE2 outputs were averaged for each field and the results were presented in tons/ha/yr.

Table 4-1. Comparison of SWAT and RULSE2 Management Operations for field number 1

Date	RUSLE2 Operation	RUSLE2 Crop	SWAT Operation	SWAT Crop	Year
21-Apr	Fertilizer application anhydrous knife 1	Corn, grain	Fertilizer application	CORN	1
5-May	Planter, double disk opener w/fluted coulter,		Plant/begin growing season		
5-May	Sprayer, post emergence		Pesticide application		
1-Nov	Harvest, killing crop 50pct standing stubble		Harvest and kill operation		2
13-May	Fertilizer application surface broadcast		Fertilizer application		
13-May	Disk, tandem heavy primary op.		Tillage operation		
13-May	Cultivator, field 6-12 in sweeps	Soybean	Tillage operation	SOYB	
14-May	Planter, double disk opener w/fluted coulter		Plant/begin growing season		
7-Jun	Sprayer, post emergence		Pesticide application		
1-Oct	Harvest, killing crop 20pct standing stubble	Corn, grain	Harvest and kill operation	CORN	3
21-Apr	Fertilizer application shank low disturbance,		Fertilizer application		
5-May	Planter, double disk opener w/fluted coulter		Plant/begin growing season		
5-May	Sprayer, post emergence		Pesticide application		4
1-Nov	Harvest, killing crop 50pct standing stubble		Harvest and kill operation		
13-May	Fertilizer application surface broadcast		Fertilizer application		
13-May	Disk, tandem heavy primary op.	Soybean	Tillage operation	SOYB	
13-May	Cultivator, field 6-12 in sweeps		Tillage operation		
14-May	Planter, double disk opener w/fluted coulter		Plant/begin growing season		
7-Jun	Sprayer, post emergence		Pesticide application		
1-Oct	Harvest, killing crop 30pct standing stubble		Harvest and kill operation		

4.3.5.2 SWAT Model

In general, SWAT is incapable of estimating sediment yield for individual fields within the watershed; the results of several fields (HRUs) are aggregated at the subbasin scale. At the same time, the automatic delineation tool in SWAT defines subbasins according to a topography map (digital elevation model) that is not detailed enough to capture field boundaries. To solve this problem and obtain SWAT model results for the individual fields, the automatic delineation tool in SWAT was used to create the most detailed subbasin map possible by selecting the minimum number of cells required for stream definition. In the next step, the field soil map was superimposed on the subbasin map to create a new subbasin map that includes the fields of interest. The total number of subbasins created at this stage was 163. In the next step, the predefined stream network was created by using the NHDPLUS map and the 10 m resolution digital elevation model. The process of field-scale delineation for the SWAT model is unique, as no similar work was found in the literature. The main challenge in this process is the need for varying degrees of detailed delineation, which is time consuming.

In the next step, unique management operations and crop rotations corresponding to RUSLE2 were created in SWAT for all fields of interest. An example of a management operation/rotation is provided in Table 1. This is unique, as it is a common modeling exercise to apply non-unique management operations for each crop to all subbasins in a SWAT project. Applying field-scale data to the SWAT project ensured close similarity to the calibrated RUSLE2 model, although not all management operations created in SWAT were identical to the management operations created with RUSLE2, due to SWAT model limitations. After the model set up, calibration, and validation, the SWAT model was run for the 30-year study period.

Finally, the subbasin/field level outputs were averaged to estimate long-term sediment yields per area for the fields of interest.

4.3.5.3 Field_SWAT

As described earlier, Field_SWAT uses a field boundary map to aggregate the sediment yield estimation obtained at HRU level. Therefore, Field_SWAT requires a calibrated SWAT model, and a shapefile containing all field boundaries with a unique record corresponding to each field. Like the previous SWAT modeling exercise, the automatic delineation tool in SWAT was used to create the most detailed subbasin map by choosing the minimum number of cells required for stream definition. In the next step, SWAT generates a stream network map. The difference between Field_SWAT and SWAT in this case was that only the most common management practices and rotation operations can be introduced for each crop in each subbasin because the introduction of field specific management operations is not feasible. After calibration and validation, the SWAT model was run for the 30-year period. Long-term average sediment yields were obtained for all fields of interest through the Field_SWAT user interface.

4.3.5.4 The High Impact Targeting (HIT)

Within the HIT model no data inventory or model run is required because all modeling was preprocessed. The only required procedure is to download the raster-based sediment yield map from the HIT model website (<http://35.9.116.206/hit2/home.htm>). In the next step, the predefined field map is superimposed on the sediment yield raster map and the average sediment yields are calculated for each field using the Zonal Statistics command in a GIS platform such as ArcGIS.

4.3.6 Data Analysis

Four techniques were used to evaluate SWAT, Field_SWAT and HIT models' effectiveness in estimating sediment yield and the applicability of these models in identifying areas of concern for excessive sediment generation at the field-scale. First, the range, median, and absolute error values for each model were compared to RUSLE2. Second, Fisher's least significant difference (LSD) test was performed to compare sediment yield estimation among the models. Third, the models' capabilities in identifying areas of concern were assessed. Finally, the reliability of the watershed models in estimating sediment yield was evaluated within the bounds of uncertainty of the RUSLE2 model results.

4.3.6.1 Evaluating Field-Scale Sediment Yield

A box and whisker plot was created based on the long-term average sediment yield to compare the overall range and median values for all four studied models (Figure 2). This graph was used for simple visual comparison. However, in order to make more informed decisions, the absolute error in estimating sediment yield was also calculated for all fields. Both procedures were performed using the R software version 2.15.1.

Statistical analysis was performed using the SAS 9.2 statistical package. The experiment was a completely randomized design with one factor (model), with four treatments: SWAT, Field_SWAT, HIT, and RUSLE2. Each model treatment was used to determine soil erosion on 41 fields within the watershed. The data were analyzed using one-way ANOVA. First, the statistical assumptions of normality of residuals and homogeneity of residual variances were assessed. Normality of the residuals was assessed using the Shapiro-Wilk test for normality and normal probability plots. Residuals were found to deviate from normality at $\alpha=0.05$. Therefore,

the response variable (sediment yield) was transformed using both log and square root transformations, where the log transformed was selected for use in the final model. Homogeneity of residual variances was assessed using Levene's test, which determined that the variances were significantly different between treatments at $\alpha = 0.05$. Therefore, the analysis was completed with heterogeneous variances using the GROUP option of the REPEATED statement in the mixed procedure of SAS 9.2. The Akaike Information Criterion (AIC) and Bayesian Information Criterion (BIC) values were found to be lower with heterogeneous variances than the analysis with homogeneous variance, thus the analysis considering unequal variances was used to draw conclusions regarding differences in the models. After the final statistical model was selected (log transformed, heterogeneous variances), Fisher's LSD was used to determine statistically significant differences in sediment yield calculations between models. More detailed information on the statistical analysis method can be found in the appendix.

4.3.6.2 Comparing Identification of Areas of Concern

Areas of concern are defined as areas that relative pollution yields generation per unit area are significantly higher than the surrounding areas (Giri et al., 2012). These areas are usually divided to three priority levels: high, medium, and low. In this study, the Jenks natural breaks method of classification was used on the results of all four models in order to classify the 41 randomly selected fields into priority levels. The Jenks natural breaks method ensures that the data from any two priority levels are significantly different (Jenks, 1967).

4.3.6.3 Uncertainty Analysis

Each of the three watershed models (Field_SWAT, SWAT, and HIT) were evaluated against RUSLE2 estimates with uncertainty bounds. The calibrated RUSLE2 model has

uncertainty bounds dependent on the value of the model output prediction. For predictions between 9.0 and 67.3 tons/ha/yr, RUSLE2 has the lowest uncertainty, ± 25 percent. For prediction values between 2.2 and 8.9 tons/ha/yr, or between 67.4 and 112.1 tons per ha per year, there is an uncertainty level of ± 50 percent; RUSLE2 predictions have the highest uncertainty level ($> \pm 100$) when they are less than 2.2 tons/ha/yr. Finally, for model predictions greater than 112.1 tons/ha/yr, the uncertainty bound is less than ± 50 (NRCSb, 2003). These uncertainty bounds were applied to the RUSLE2 data set, and upper and lower uncertainty bounds were calculated, creating a confidence region around the RUSLE2 predictions that the watershed models were evaluated against. The entire data set (data for every field from all four models) was organized in increasing order by RUSLE2 prediction value, and graphed (Figure 4).

In addition to the uncertainty bounds, the performance of the three models was evaluated using P-factor and R-factor analysis. The P-factor is defined as the percentage of estimated values that fall within the uncertainty bounds. The R-factor is defined as the average width of the uncertainty bounds divided by the standard deviation of the corresponding measured variable. In an ideal case for a set of model predictions, the P-factor would be equal to 1, and the R-factor equal to 0, meaning that all estimations from the model fall within a narrow uncertainty bracket. The P-factors and R-factors were calculated for each of the three watershed models with respect to RUSLE2 uncertainty bounds.

4.4 RESULTS AND DISCUSSION

4.4.1 Model Calibration/Validation

As described earlier, the calibration procedure involved calibrating two SWAT projects. The calibration and validation was satisfactory for both models according to criteria described by

Moriasi et al. (2007). The statistical results for the overall calibration and validation period are displayed in Table 4-2.

Table 4-2. Combined calibration and validation for SWAT and Field_SWAT

Models	Streamflow			Sediment		
	NSE	RSR	PBIAS	NSE	RSR	PBIAS
SWAT	0.54	0.67	2.68%	0.53	0.69	34.9%
Field_SWAT	0.50	0.70	5.02%	0.52	0.69	-1.94%

4.4.2 Evaluating Field-Scale Sediment Yield

The long-term average sediment yields obtained from the 41 field-scale analysis are presented in Figure 2. The box and whisker plot provides the range and median values for each model. This allows comparison between three watershed-scale modeling approaches that claim to provide field-scale results versus the calibrated RULSE2 model.

Among the studied models, SWAT produced the greatest maximum value (60 t/ha/yr) and largest range (59.7 t/ha/yr). Field_SWAT had the second highest range and maximum values (20.65 and 20.66 t/ha/yr, respectively). The difference in range between SWAT and Field_SWAT is that in Field_SWAT, the most common management practices and rotation operations were introduced in each subbasin. However, in the SWAT model the use of the detailed predefined subbasin map allowed the introduction of field specific management operations. Therefore, it is expected that the results of the SWAT model are closer to the RUSLE2 model. Comparing the median sediment yield values of 41 fields confirmed the above hypothesis, the median SWAT value of 2.32 t/ha/yr is the closest to the median RUSLE2 model of 3.36 t/ha/yr and the Field_SWAT is the second best with a median value of 1.36 t/ha/yr.

Meanwhile, the very large estimation of sediment yield in the SWAT model (60 t/ha/yr) resulted in distortion of the mean value (6.15 t/ha/yr), which is almost twice as large as the RUSLE2 model average (3.84 t/ha/yr).

The HIT model produced surprisingly low estimates compared to the other models, with a range almost 16 times lower than the RUSLE2 estimation. The RUSLE2 maximum value of 13 t/ha/yr was 12.18 t/ha/yr greater than the maximum HIT prediction of 0.82 t/ha/yr. Meanwhile, these maximums were obtained from two different fields between models, which creates more doubt about the reliability of the HIT model. This discrepancy is likely due to input data. RUSLE2 requires very detailed and site specific information while the HIT model only requires county-level agricultural census data for crop operations/rotations.

Regarding the absolute error between the three watershed models and RUSLE2, SWAT had the lowest median value of 1.28 t/ha/yr and the highest maximum value of 53.5 t/ha/yr. The second best model is Field_SWAT with a median error of 2.63 t/ha/yr while for the HIT model this value is 3.00 t/h/yr. However, both of these models have lower maximum error (19.21 and 12.54 t/ha/yr, respectively) than SWAT. On average, SWAT and Field_SWAT over-predicted sediment yield by 154% and 173%, while the HIT model under predicting by 2084%. Among all studied models the worst estimation for a single field was produced by the HIT model, which under-predicted the sediment yield by 75 times. The worst-case scenarios for SWAT and Field_SWAT were 9 and 14 times higher than the RUSLE2 perditions, respectively.

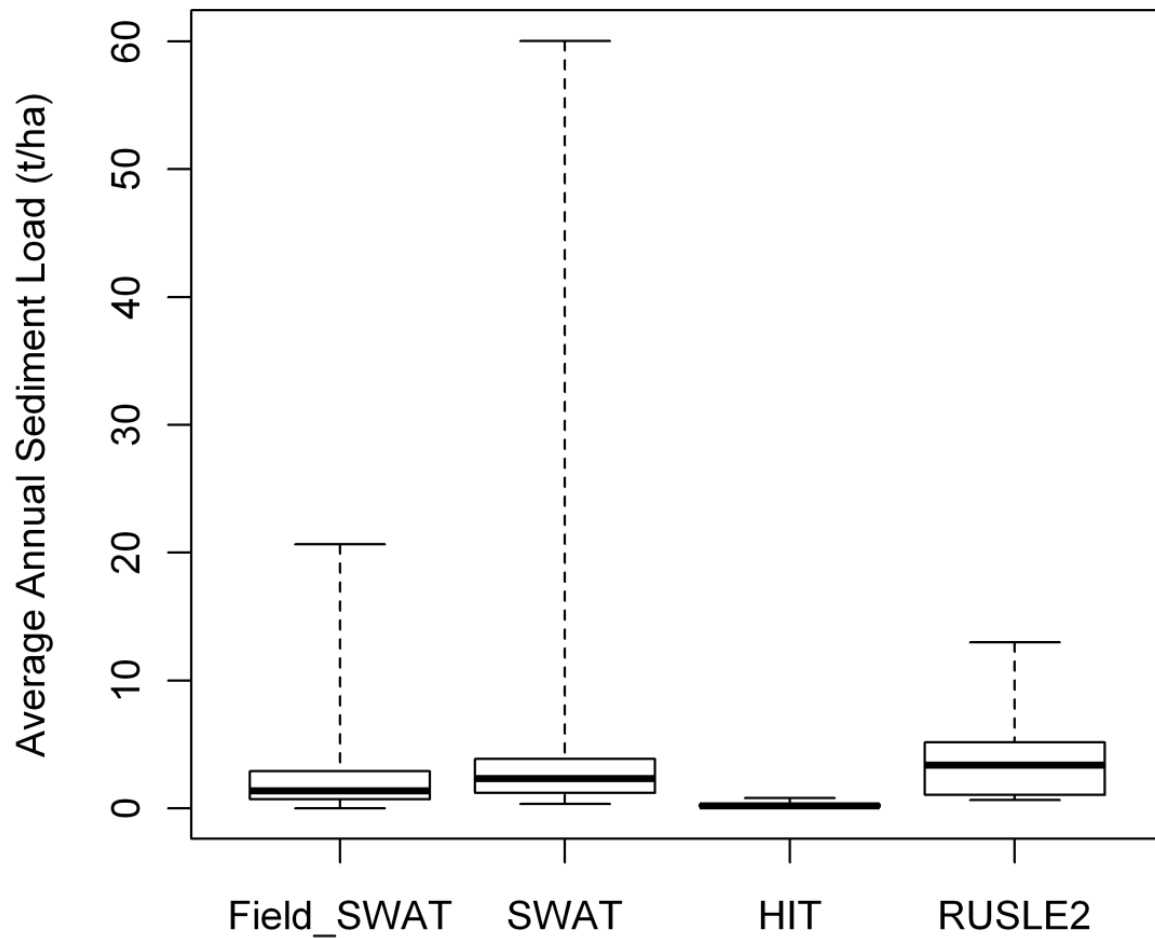


Figure 4-2. Comparison of four modeling strategies for estimating sediment runoff from 41 randomly selected fields in the River Raisin Watershed. In each box plot, the bold line is drawn at the median of the sample set, the top and bottom box ends are drawn at the 75 and 25 percentile of the sample set, respectively. RUSLE2 estimations were considered benchmark values.

In the Type 3 test of fixed effects, the models were found to be highly statistically significant ($P < 0.0001$) at $\alpha = 0.05$. Therefore, Fischer's Least Significant Difference (LSD) test was performed for all pairwise comparisons of the treatments (Table 4-3). All comparisons were found to be significantly different at $\alpha = 0.05$ except for the SWAT versus RUSLE2 comparison. More detailed information on the results from the statistical analysis can be found in the appendix.

Table 4-3. P-values for differences of least squares means between models

Model	Field_SWAT	HIT	RUSLE2	SWAT
Field_SWAT		<0.0001	0.0118	0.0344
HIT	<0.0001		<0.0001	<0.0001
RUSLE2	0.0118	<0.0001		0.9938
SWAT	0.0344	<0.0001	0.9938	

4.4.3 Comparing Identification of Areas of Concern

Identifying areas of concern is an important step toward developing an implementation plan to control and mitigate non-point source pollution, because pollution yields from different fields within a watershed are disproportionate. Meanwhile, measuring pollution yields from all fields through monitoring is expensive and impractical. Therefore, watershed managers rely on watershed models to guide them in the decision making procedure. In this section of study, our goal was to examine the reliability of three watershed models in identifying areas of concern, which can be later used for targeting best management practices. Based on the sediment yield results obtained from 41 randomly selected fields and the Jenks natural breaks method, the areas

of concern were divided into high, medium, and low priority. However, different ranges were identified for the three propriety areas because different models provided different results that varied in range (Figure 4-3).

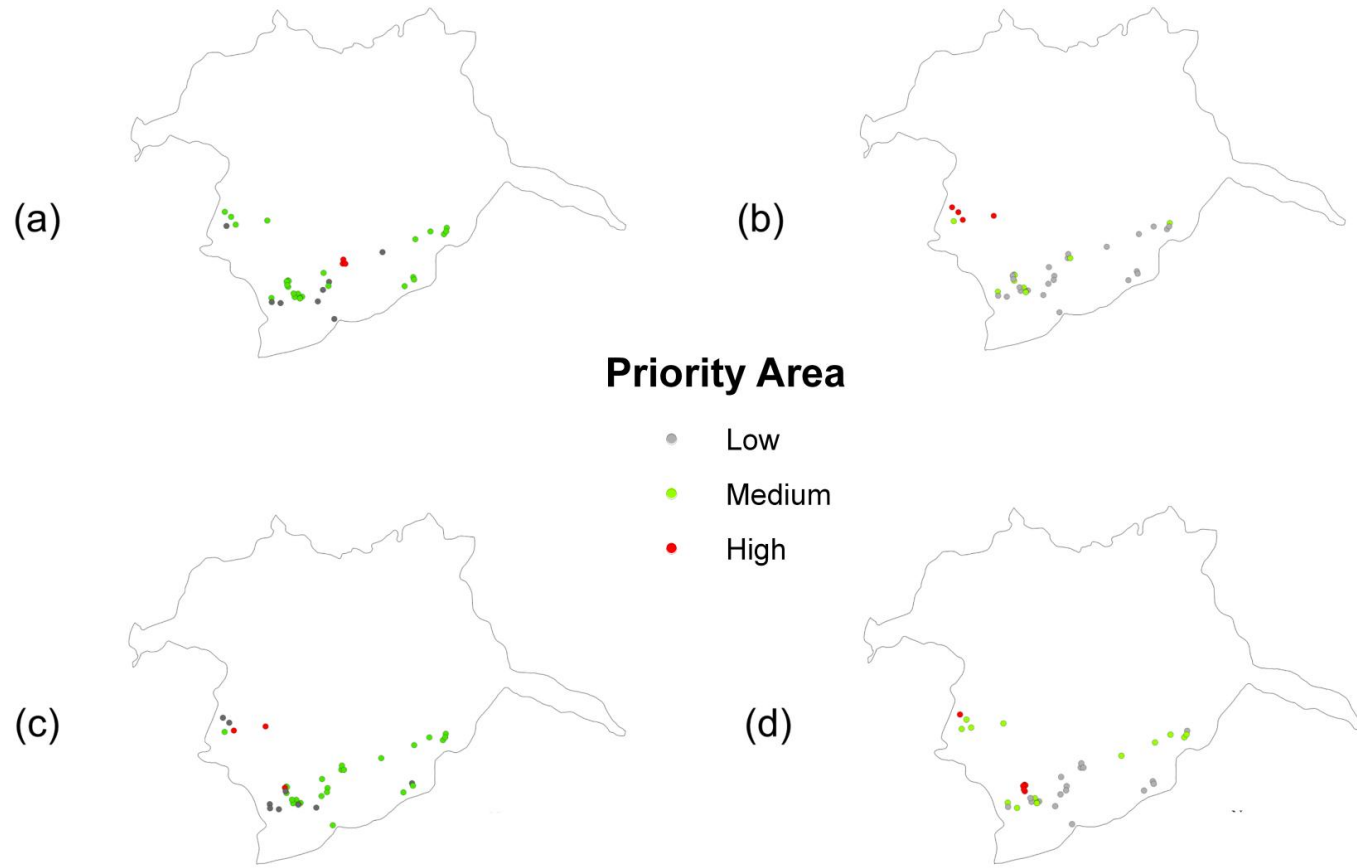


Figure 4-3. Spatial distribution of fields identified as “high,” “medium,” and “low” priority within the 41 fields randomly selected for analysis in the River Raisin watershed for (a) Field_SWAT, (b) SWAT, (c) HIT, and (d) RUSLE2

The fields within each priority area were compared to those of RUSLE2 in order to assess the effectiveness of each model at correctly identifying areas of concern (Table 4-4). The first value in each cell is the number of fields that the given model categorized the same as RUSLE2. For example, in the first row and column, the “0” indicates that Field_SWAT did not have any high priority fields in common with RUSLE2. The “0%” in the same cell is calculated by dividing the number of fields with identical priority assignments to RUSLE2 (zero in this case) in the given category (for this example, high) to the total number of fields that labeled as a “high” priority from the RUSLE2 model. Overall, RUSLE2 identified seven fields as high priority, 17 as medium priority, and 17 as low priority. However, none of the models were similar to the RUSLE2 priority classifications. The SWAT model was able to assign one high priority field in common with RUSLE2. In addition, the SWAT model has the most similar medium and low priority predictions with RUSLE2. Meanwhile, the low priority areas had the best estimates comparing to RUSLE2 for all watershed models. The HIT model had the worst performance of any model, failing to identify any correct priorities for the high or medium categories. Overall, the results show that all three of the watershed models are ineffective at field-scale identification of areas of concern.

Table 4-4. Total number of the fields and the percentage of correctly identified fields within each category compared to RUSLE2.

Priority Level	Field_SWAT	SWAT	HIT
High	0 (0%)	1 (14%)	0 (0%)
Medium	1 (6%)	4 (24%)	0 (0%)
Low	9 (53%)	15 (88%)	7 (41%)

4.4.4 Uncertainty Analysis

Since the results of the calibrated RUSLE2 model were not exact, uncertainty bounds can provide a better understanding of the true value of sediment yield for each field. With this information, a more realistic comparison can be made regarding the reliability of the Field_SWAT, SWAT, and HIT models. This information is presented in Figure 4-4.

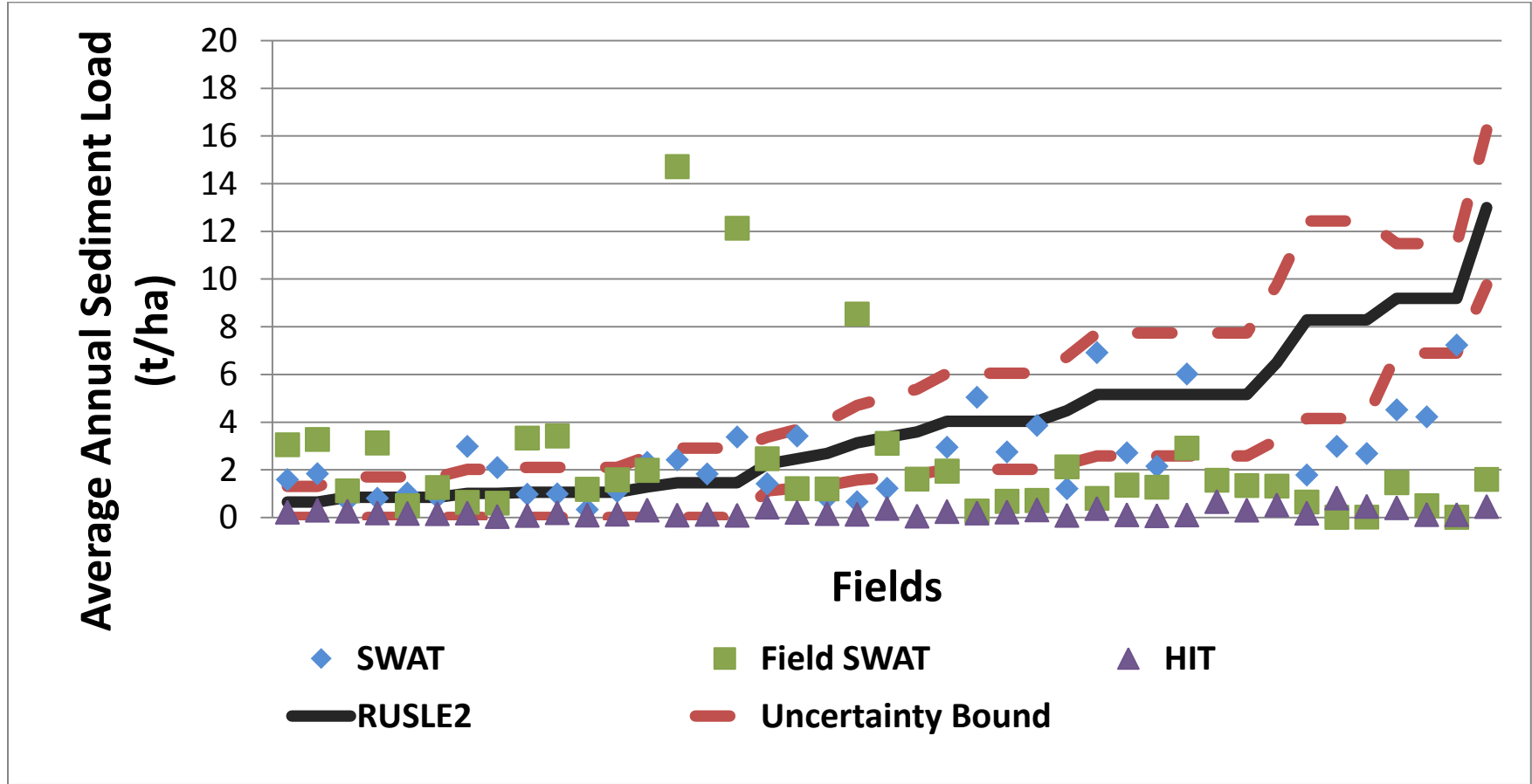


Figure 4-4. Estimations from all four models organized by increasing RUSLE2 sediment runoff predictions. Uncertainty bounds are provided for the RUSLE2 estimations (three outliers from the SWAT model not shown)

Overall, the SWAT model performed consistently across all levels of sediment yield; however, four of the estimations were significantly above the uncertainty bound and could be classified as outliers. The HIT model results are only acceptable within the uncertainty bounds for low predictions. Meanwhile, Field_SWAT performed better in the low and mid-range sediment yield estimations. In addition, because the data is organized by increasing RUSLE2 estimations, models that show a similar trend will likely make better estimates of priority areas than those that do not. The only model to demonstrate a prominent, increasing trend was SWAT. None of the models was able to provide a sufficient estimation of sediment yield on more erosive fields. Therefore, even though it may be acceptable to use watershed models in estimating sediment yields from low and medium priority areas, for the highly erosive areas, the results of more detailed field-scale model such as RUSLE2 is recommended.

In addition to the uncertainty bounds, the performance of the three models was evaluated using the P-factor and R-factor (Table 4-5).

Table 4-5. P-factors and R-factors for Field_SWAT, SWAT and the HIT Model

Uncertainty Factor	Field_SWAT	SWAT	HIT
P-factor	0.27	0.51	0.39
R-factor	0.91	0.31	22.06

According to Abbaspour (2009) P-factors close to 1 are desired and R-factors less than 1 are acceptable. None of the models performed exceptionally well, however SWAT and Field_SWAT had satisfactory R-values while the HIT model had an R-factor significantly greater than 1. The greatest P-value came from the SWAT and HIT models. However, Field_SWAT had fewer predictions within the uncertainty bounds than the two other models and therefore the value of

the P-factor was the worst. Overall, the SWAT model performed the best among the studied models by having the closest P-factor to 1 and a satisfactory R-factor less than 1. In general, the uncertainty analysis shows that SWAT predictions are most similar to those of RUSLE2, with Field_SWAT performing worse than SWAT but slightly better than the HIT model.

4.5 CONCLUSION

In recent years, various watershed models and interfaces have been developed with the goal of providing results that can be used for decision making at the small-scale and for individual farms. SWAT, Field_SWAT, and HIT are examples of these models. However, the results of these models have not been compared against detailed and calibrated field-scale models such as RUSLE2. Therefore, evaluating these models against reliable field-scale data is an important step towards using these models responsibly. The specific objectives of this project are to: 1) compare three watershed-scale models (SWAT, Field_SWAT, and HIT) against a calibrated field-scale model (RUSLE2) in estimating sediment yield from 41 randomly selected agricultural fields within the River Raisin watershed; 2) evaluate statistically significant differences among the models; 3) assess the watershed models' capabilities in identifying areas of concern at the field level; 4) evaluate the reliability of the watershed-scale models for field-scale studies.

The SWAT model produced the most similar estimates in comparison to RUSLE2 by providing the closest median and the lowest absolute error in sediment yield predictions. The worst predictions were associated with the HIT model that overall under-predicts sediment yield by about 1600%.

Concerning statistically significant differences between models, the results confirmed the superiority of SWAT as the only watershed model found to be statistically similar to RUSLE2 at $\alpha = 0.05$, with a p value equal to 0.9938.

The watershed models were evaluated regarding their capabilities in identifying the areas of concern at the field level. Results showed that almost all of the watershed models are incapable of identifying low, medium, and high priorities similar to RUSLE2 model classification. However, the SWAT model performed slightly better than the two other watershed models, while HIT performed the worst by completely missing all fields that were categorized as high and medium priority areas by the RUSLE2 model.

Finally, the reliability of the watershed-scale models for the field-scale study was evaluated using uncertainty bounds generated for the RUSLE2 model. SWAT provided the most correct estimates within the uncertainty bounds, at 51%, while Field_SWAT provided 27%, and HIT 39%. For highly erosive fields, all models performed poorly. The HIT model performed better in minimally erosive fields and SWAT performed reasonably well for low and medium erosive areas. The SWAT model predictions within RUSLE2 uncertainty bounds were somewhat consistent for the entire range of sediment yield estimations, with a few outliers. The HIT model was only able to estimate low sediment yields within the RUSLE2 uncertainty bounds, and Field_SWAT provided correct estimations within the bounds for mainly the low and mid-range sediment yield. Uncertainty analysis using the P and R-factors suggests that SWAT model is the most reliable among the studied models, while HIT is the least reliable.

Although SWAT is a watershed scale model, it provided estimations similar to RUSLE2 when delineated at the field-scale, likely due to incorporation of detailed management operations

similar to those used in RUSLE2. For Field_SWAT, field specific inputs are difficult/impossible to incorporate due to the large size of subbasins in comparison to average field size. Even though the HIT model generated the results at the highest resolutions among the studied models (10 m), usage of county level agricultural management and rotations information for each crop likely resulted in poor model performance. These results suggest that field specific data is of paramount importance when making field-scale sediment yield estimations. Therefore, caution should be exercised when using the watershed models for field level decision-making. This study demonstrated that none of the studied watershed models provided satisfactory field-scale targeting results, while the overall performance of the SWAT model is not significantly different from the RUSLE2 model in estimating the field-scale sediment yield. Meanwhile, SWAT generated several outliers that are significantly larger than the RUSLE2 model results. Therefore, it is recommended that field-scale models such as RUSLE2 should be used to evaluate the impact of management operations at the small scale. Future work should involve the integration of field-scale agricultural operations at the watershed level, which provide valuable information concerning the true costs of best management practice implementation scenarios.

5. EVALUATING THE IMPACT OF FIELD-SCALE MANAGEMENT STRATEGIES AT THE WATERSHED OUTLET

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5.1 ABSTRACT

Nonpoint source pollution from agricultural lands is a significant contributor of sediment pollution in lakes and streams in the United States. Therefore, quantifying the impact of individual field management strategies at watershed-scale provides valuable information to watershed managers and conservation agencies. In this study, four methods employing some of the most cited models in field and watershed analysis were compared in order to find a practical yet accurate method. The models used in this study include field-scale model (the Revised Universal Soil Loss Equation 2 -RUSLE2), the Spatially Explicit Delivery Model (SEDMOD), and a watershed-scale model (Soil and Water Assessment Tool - SWAT). These models were used to develop four modeling strategies (methods) as follows: Method 1: predefined field-scale subbasin and reach layers were used in the SWAT model; Method 2: subbasin-scale sediment delivery ratio was employed; Method 3: results obtained from the field-scale RUSLE2 model were incorporated as point source inputs to the SWAT watershed model, and; Method 4: a hybrid solution combining analysis from the RUSLE2, SEDMOD, and SWAT models. The analysis was performed on the River Raisin watershed in southeast Michigan. Method 4 was selected as the most accurate among the studied methods since it accounts for three stages of sediment transport from a field to the watershed outlet. In addition, the effectiveness of six types of best management practices (BMPs) both in terms of the amount of water quality improvement and associated cost was assessed. Economic analysis was performed using Method 4, and producer

requested prices for BMPs were compared with prices defined by the Environmental Quality Incentives Program (EQIP). The results show that on a per unit area basis, producers requested higher prices than EQIP in four out of six BMP categories. Meanwhile, the true cost of sediment reduction at the field and watershed scales were greater than EQIP in five out of six BMP categories according to producer requested prices.

5.2 INTRODUCTION

Non-point source (NPS) pollution from agricultural lands poses a significant threat to water quality in the United States. Runoff from agricultural lands is the main cause of water quality problems in rivers and lakes; a major component of this pollution is excess sediment runoff driven by rainfall events (EPA, 2005). Many publicly sponsored programs are aimed at reducing sediment runoff in an effort to protect and preserve water resources (Shortle, 2012). However, efforts to reduce water pollution have been mainly aimed at point sources, while NPS pollution remains predominantly uncontrolled (Thomas and Froemke, 2012). The lack of success in NPS pollution control is due largely to the difficulty of identifying specific problem areas that are significant sources of pollution (White et al., 2009). Lack of regulation and enforcement also cause NPS pollution to remain uncontrolled (EPA, 2005).

Monitoring projects aimed at quantifying water quality usually involve high implementation and operational costs and require long periods of time and a great amount of data to form conclusions. To address these difficulties, models can be employed to gain valuable knowledge faster than monitoring and at lower costs. Watershed models provide a way to quantify NPS, identify critical source areas of pollution, and compare management strategies

(Daggupati et al., 2011). Therefore, these models are useful and often necessary tools in the planning and evaluation stages of water quality improvement projects.

Several studies have been completed regarding the application of watershed models to quantifying NPS. For example, Shen et al. (2009) evaluated the performances of Water Erosion Prediction Project (WEPP) and the Soil and Water Assessment Tool (SWAT) for soil erosion prediction in the Zhangjiachong watershed. The WEPP model provided slightly better predictions than the SWAT model; however, both produced satisfactory results. Parajuli et al. (2009) used the Annualized AGricultural Non-Point Source (AnnAGNPS) model and the SWAT model to predict sediment yields (among other outputs) in the Cheney Lake watershed located in Kansas. SWAT performed better than AnnAGNPS for sediment yield prediction over the 45-month evaluation period. Im et al. (2007) compared predictions of sediment yield from the HSPF and SWAT models in the Polecat Creek watershed in Virginia. Both HSPF and SWAT produced satisfactory results, and HSPF performed slightly better for time steps greater than a month. However, all of the above models were found effective in NPS quantification. In addition, watershed models are widely used to identify critical source areas. For example, recently, Nejadhashemi et al. (2011) compared the applicability of Spreadsheet Tool for Estimating Pollutant Load (STEPL), the Long-Term Hydrologic Impact Assessment model (L-THIA), the PLOAD model, and the SWAT model to identify critical source areas. They concluded that SWAT was the only model capable of identifying critical source areas, and results from the other models were not satisfactory. In addition, Giri et al. (2012) performed a comprehensive study to compare different targeting techniques (based on various factors such as pollutant concentration, load, and yield) to identify the critical source areas using the SWAT model. They concluded that concentration based targeting is the most effective in reducing nutrients, while load based

targeting techniques are more effective in reducing sediment at the watershed outlet. Finally, watershed-scale impact assessment of best management practice (BMPs) implementation scenarios have been extensively studied (Gitau et al., 2008; Ullrich and Volk, 2009; Lee et al., 2010; Tuppad et al., 2010; Gassman et al., 2010; Betrie et al., 2011; Giri et al., 2012), demonstrating that a watershed-scale model is a powerful tool for use in management plan development.

The modeling exercises mentioned above are important and necessary to make informed watershed management decisions; however, execution of a large-scale BMP implementation plan is infeasible due to lack of rigorously enforced NPS regulations. In reality, BMPs are implemented on individual fields, and due to the voluntary nature of most BMP installation programs, many BMPs covering a significant portion of a watershed is unlikely. Under this condition, understanding the true cost and effectiveness of individual BMPs both at the field and watershed-scales is important to guide informed decision making for conservation programs such as the BMP Auction (Smith et al., 2009).

Many field-scale models are available for evaluation of BMP effectiveness, such as the Revised Universal Soil Loss Equation 2 (RUSLE2) and Agricultural Policy Environmental Extender (APEX). Although very useful for field-scale analysis, watershed-scale impacts cannot be quantified. Also, field-scale results obtained from watershed scale models such as SWAT may be unreliable due to the limitations of land use, topography, and soil input data resolutions for field-scale study (Daggupati et al., 2011). Therefore, there is need for an integrated modeling framework capable of assessing the impact of field-scale management strategies at the watershed scale, which is the main objective of this study. The study area is the River Raisin watershed, located in southeast Michigan. Four techniques were proposed and tested to evaluate watershed-

scale sediment reduction loads from 80 field-scale BMP scenarios. The methods tested were using: (1) predefined field-scale subbasin and reach layers in the SWAT model; (2) subbasin-scale sediment delivery ratio; (3) results obtained from the field-scale RUSLE2 model as point source inputs to the SWAT watershed model; (4) a hybrid solution combining analysis from the RUSLE2, the Spatially Explicit Delivery Model (SEDMOD), and SWAT models. The applicability, advantages, and disadvantages of these approaches are discussed in this study. Finally, an economic analysis was performed to compare producer requested prices versus the prices defined by the USDA's Environmental Quality Incentives Program (EQIP) for BMP implementation.

5.3 MATERIALS AND METHODS

5.3.1 Study Area

The River Raisin watershed (Hydrologic Unit Code 04100002) is located approximately 100 km south west of Detroit, Michigan. Almost the entire watershed is within Michigan boundaries, excluding a small portion in Ohio. The watershed is located partially in five counties: Hillsdale, Jackson, Lenawee, Monroe, Washtenaw, and Fulton, with most of the area in Lenawee County. The River Raisin flows east into Lake Erie near Monroe, Michigan. Sixty-six percent of the total 268,100 ha watershed area is used for crops and pastureland, according to the Nation Land Cover Database (NLCD, 2009). The remaining land cover is 13% forest, 12% urban, 7% wetlands, 1% range grass and brush, and 1% water. Major agricultural land use includes corn, soybeans, wheat, and pastureland. Mean elevation is 300 m above sea level with a maximum elevation of 391 m, and a minimum of 12 m, according to the United States Geological Survey.

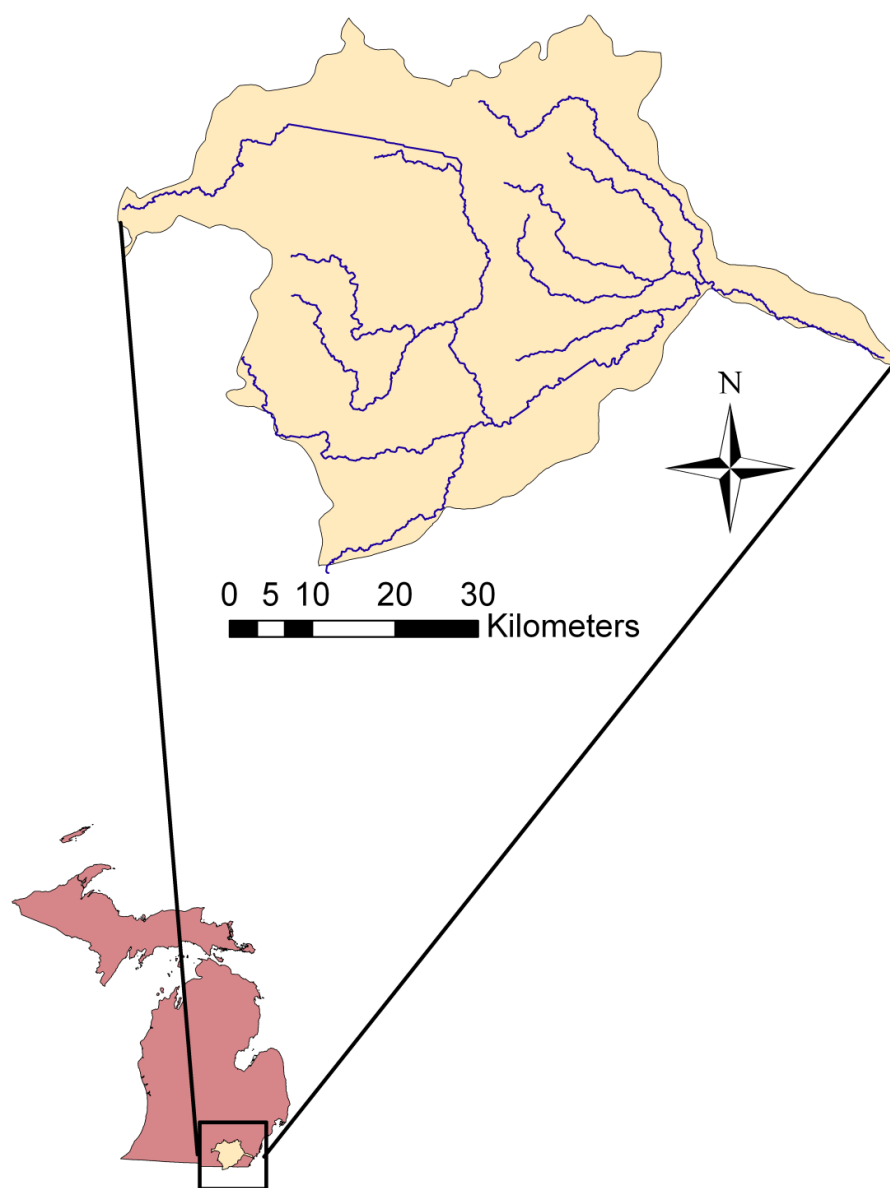


Figure 5-1. Study Area - River Raisin Watershed

5.3.2 Data Inventory

In this study, a wide range of data was required for the modeling practices. The following is a summary of all data collected.

Mean daily streamflow data was available from January, 1990 through December, 2009 from USGS station number 04176500 located on the River Raisin near Monroe, Michigan. A total of 7,305 records were collected. Ninety-nine percent of the data were accepted for publication, and 1% were labeled as preliminary. The United States Environmental Protection Agency (EPA) Storage and Retrieval (STORET) database contained 129 total suspended solid (TSS) measurements spanning irregularly from July 6, 1998 to October 3, 2005. The station number 580046 was located near Monroe, Michigan at latitude 42.07° , longitude -84.13° . For the months that had observations, there were on average 1.5 observations per month with a maximum of 5 and a minimum of 1. Meanwhile, weather data consisting of daily maximum and minimum temperatures and total daily precipitation were obtained from the National Climatic Data Center from two weather stations. Stations 209218 and 200032 are located within the watershed boundary. Data from station 209218 are available from January, 1990 through April, 2008, and data from station 200032 are available from January, 1990 through December, 2009.

Elevation data used in this study was obtained from the USGS National Elevation Dataset (NED) website. The USGS has digital elevation maps available for the entire continental United States at a resolution of 1 arc-second (30 m) and 1/3 arc-second (10 m). The 10 m resolution elevation map (NED10) was used in this study. Land cover data was obtained from the USDA National Agricultural Statistics Service (NASS) Crop Data Layer (CDL), 2007 edition. The land cover raster had a resolution of 56 m.

All fields boundaries were catalogued by common land use unit (CLU) created by the Natural Resources Conservation Service (NRCS) and aerial maps. Two sets of soil maps were used in this study. For the field-scale modeling, field coordinates were used to locate the field boundaries on the NRCS Web Soil Survey (WSS) online tool. WSS provides detailed descriptions of all soil map units within the area of interest. However, for the watershed-scale study, soil data in raster format were obtained from The Soil Survey Geographic Database (SSURGO). The SSURGO database provides high-resolution soil data maps. The soil raster map for the Raisin Watershed listed 433 unique soil types.

5.3.3 Models Used

Three types of models were used in this study: a field-scale model (RUSLE2), an overland sediment delivery model (SEDMOD), and a watershed-scale model (SWAT). Brief descriptions of these models are provided below.

5.3.3.1 RUSLE2

RUSLE2 is a sediment erosion estimation tool developed by the USDA Agricultural Research Service (ARS) for the NRCS in 2003. The main purpose of RUSLE2 is to guide in conservation planning, and to estimate rill and inter-rill erosion by rainfall and runoff using the Revised Universal Soil Loss Equation (RUSLE). The RUSLE (Equation 1) estimates average annual sediment yield per unit area based on disaggregated daily precipitation and temperature values.

$$A = R \cdot K \cdot L \cdot S \cdot P \quad (5-1)$$

where A is the average annual soil loss from rill and inter-rill erosion caused by rainfall, and overland flow, measured in U.S. tons per acre per year, R is the rainfall-runoff erosivity factor, K is the soil erodibility measured under a standard condition, L is the slope length, S is the slope steepness, C is the cover management factor, and P is the support practice factor (Foster et al., 2003). The S and L factors are defined directly by the user and RUSLE2 uses algorithms to calculate all of the remaining parameters based on the inputs. Inputs to RUSLE2 include user-built management operations from built-in events, weather data, soil database, and county location database. RUSLE2 has been validated using 10,000 plot-years of data from natural runoff sites, and 2,000 plot-years from simulated plots (Foster et al., 2003).

5.3.3.2 SEDMOD

The Spatially Explicit Delivery Model (SEDMOD) sediment delivery framework was first introduced by Fraser et al. (1999) to estimate sediment deposition from surface runoff before reaching a stream. Sediment delivery ratio (SDR) is estimated using six parameters: flow path slope gradient, flow path slope shape, flow path hydraulic roughness, stream proximity, soil texture and overland flow (Fraser, 1999). These parameters are provided by input raster files including those for elevation, soil texture, soil transmissivity, stream network, roughness, and soil loss (Kandel, 2010). SEDMOD uses a raster file for each of the six inputs parameters to create intermediate grids including streamline, flow direction, flow accumulation, gradient, path shape, profile curvature, moisture, cell length, path slope and proximity estimations. The SDR is then calculated using a linear weighting model, and a composite raster is created. Input parameters are used to calculate a delivery potential using Equation 2 (Kandel, 2010).

$$DP = SG_r SG_w + SS_r SS_w + SR_r SR_w + SP_r SP_w + ST_r ST_w + OF_r OF_w \quad (5-2)$$

where DP is delivery potential, SG is flow path slope gradient, SS is flow path slope shape, SR is flow path surface roughness, SP is stream proximity, ST is soil texture, OF is the overland flow index. Subscript r represents the rating of each parameter (ranging from 0-100) and subscript w is the relative weight of each parameter.

An empirical equation for delivery ratio-watershed area used by (Fraser, 1999) assumes that an “average” plot would have a delivery ratio of 100%. The average plot has an area of 0.00049 km^2 similar to the plot size used for the USLE study. Therefore, an intercept parameter used in Equation 3, (C) can be calculated (Kandel, 2010).

$$DRa = C \cdot A^{-1/8} \quad (5-3)$$

where DRa is the delivery ratio-watershed area (%) and A is the watershed area measured in km^2 . Finally, the spatially distributed delivery ratio, which is equal to SDR, is calculated using Equation 4 (Kandel, 2010).

$$SDR = DP + [DRa - \mu(DP)] \quad (5-4)$$

where SDR is the delivery ratio, and $\mu(DP)$ is the mean composite layer of DP .

5.3.3.3 SWAT

SWAT was developed by the USDA-ARS and is a physically based, spatially distributed watershed model (Arnold et al., 1998; Gassman et al., 2007). The SWAT model divides the watershed into subbasins, and further into HRUs, or hydrologic response units. These HRUs are

the basic land area unit that SWAT uses to calculate model outputs. The HRUs are represented as land areas containing homogeneous land cover, soil type, and slope. Model components include hydrology, land management strategy, weather, plant growth, chemical transport, nutrient transport (Gassman et al., 2007). The SCS curve number method is used to calculate runoff in SWAT, and sediment erosion is calculated using the modified universal soil loss equation (MUSLE) for each HRU. Because this study focuses on sediment, an in-depth explanation of sediment runoff and transport estimations is discussed.

Sediment runoff is calculated as an average annual erosion as a function of runoff, peak runoff rate, HRU area, soil characteristics, land cover, and topography (Neitsch et al., 2005). There are three main stages of sediment erosion: detachment, transport, and degradation and deposition. Detachment of sediment particles from land is calculated using the MUSLE Equation (5-5).

$$sed = 11.8 \cdot (Q_{surf} \cdot q_{peak} \cdot area_{hru})^{0.56} \cdot K_{usle} \cdot C_{usle} \cdot P_{usle} \cdot LS_{usle} \cdot CFRG \quad (5-5)$$

In the above equation *sed* is daily sediment yield in metric tons, Q_{surf} is the volume of surface runoff measured in mm per ha, q_{peak} is the peak runoff rate measured in cubic m per s, $area_{hru}$ is the area of the HRU measured in ha, K_{usle} is the USLE soil erodibility factor with units of 0.013 metric tons square meters hours per cubic meter metric ton centimeters, C_{usle} is the USLE land cover factor, P_{usle} is the USLE support practice factor, LS_{usle} is the USLE topography factor, and *CFRG* is the course soil fragment factor (Neitsch et al., 2005).

After the detachment stage of erosion, the amount of sediment transported and released into the main channel is calculated using Equation 6 (Neitsch et al., 2005).

$$sed = (sed' + sed_{stor,i-1}) \cdot \left(1 - \exp \left[\frac{-surlag}{t_{conc}} \right] \right) \quad (5-6)$$

where sed is the sediment discharged into the main channel on a given day measured in metric tons, sed' is the amount of sediment load from the HRU on given day measured in metric tons, $sed_{stor,i-1}$ is the mass of stored sediment from the preceding day measured in metric tons, $surlag$ is the surface runoff lag coefficient, which is a user defined, and t_{conc} is the concentration time measured in hours (Neitsch et al., 2005). By varying the $surlag$ coefficient, watersheds with high and low amounts of storage can be modeled. Lateral and base flow can also contribute to sediment in the main channel.

The third stage of erosion is degradation and deposition. Deposition occurs when sediment leaves the stream flow and settles on the streambed, due to high concentration. Degradation occurs when sediment concentration is low, allowing sediment on the streambed to become suspended and travel with stream flow. The threshold value for these processes, maximum sediment concentration, is a function of the peak runoff rate of the stream (Neitsch et al., 2005).

5.4 SWAT Model Calibration

Among the models described above, only SWAT model can be calibrated due to lack of observed values at the edges of the fields and streams. Calibrating the SWAT model consists of

iterative model runs and parameter adjustments until model predictions are deemed satisfactory compared with observed data. Calibrations for flow and sediment were performed using manual calibration. The calibration/validation period was approximately eight years spanning from January 1, 1998 through September 30, 2005. Calibration was performed on a daily basis from January 1, 1999 to December 31, 2001 and the validation was performed from January 1, 2002 to September 30, 2005. These dates correspond to the limiting data source, in this case, the daily TSS values. Three statistical parameters were used to compare daily-observed data against daily model simulations: Nash-Sutcliffe efficiency (NSE), percent bias (PBIAS), and the ratio of the root mean square error to the standard deviation of measured data (RSR). A model predicting two or more process can be considered accurate if the $NSE > 0.5$, $PBIAS < \pm 25\%$, and $RSR \leq 0.70$ for streamflow, and the $NSE > 0.5$, $PBIAS < \pm 55\%$, and the $RSR \leq 70$ for sediment (Moriassi et al., 2007).

The NSE compares the magnitude of residual variance with data variance (Moriassi et al., 2007). NSE was computed using Equation 7:

$$NSE = 1 - \left[\frac{\sum_{i=1}^n (Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^n (Y_i^{obs} - Y^{mean})^2} \right] \quad (5-7)$$

The variable Y_i^{obs} represents the i th value from the observed data set and Y_i^{sim} represents the i th value from the simulated data set. The numerator in the expression is the sum of squares of the residuals. The denominator is the sum of squares of the observed values subtracted from the mean of all observed values. This portion of the expression consists of Y_i^{obs} ,

defined above, and Y^{mean} , the mean of the observed values. The NSE ranges from negative infinity to one, an optimal value of one.

PBIAS quantifies the tendency of simulated values to be greater or less than observed values (Moriassi et al., 2007). Zero is the optimal value of PBIAS, indicating simulated values are equally distributed above and below the observed values. Equation 8 is used to calculate PBIAS. In this equation the numerator in the expression sums the residuals, each multiplied by 100; the denominator sums the observed values.

$$PBIAS = \left[\frac{\sum_{i=1}^n (Y_i^{obs} - Y_i^{sim}) \times 100}{\sum_{i=1}^n Y_i^{obs}} \right] \quad (5-8)$$

The RSME-observations standard deviation ratio (RSR) parameter measures the degree of collinearity between the observed and simulated data sets (Moriassi et al., 2007). RSR ranges from 0 to 1. Equation 9 was used to calculate RSR.

$$RSR = \frac{\left[\sqrt{\sum_{i=1}^n (Y_i^{obs} - Y_i^{sim})^2} \right]}{\left[\sqrt{\sum_{i=1}^n (Y_i^{obs} - Y^{mean})^2} \right]} \quad (5-9)$$

RSR combines error index analysis with a scaling factor (Moriassi et al., 2007). A value of 0 for RSR indicates zero RMSE, or a perfect simulation. Therefore, RSR close to zero are satisfactory (Moriassi, et al., 2007).

5.5 Evaluating the Impact of Field-Scale Management Strategies at the Watershed Outlet

As described earlier, four techniques were proposed to evaluate the impacts of field-scale management strategies at the watershed outlet for a 10-year period: January 1, 2000 to December 31, 2009. The methods tested were using: (1) predefined field-scale subbasin and reach layers in SWAT model; (2) subbasin-scale sediment delivery ratio; (3) results obtained from the field-scale RUSLE2 model as point source inputs to the SWAT watershed model; (4) a hybrid solution combining analysis from the RUSLE2, SEDMOD, and SWAT models.

5.5.1 Predefined Field-Scale Subbasin and Reach Layers in SWAT Model (Method 1)

The goal of this section was to create a stand-alone SWAT model that could effectively quantify sediment loads at the field-scale and then track sediment through in-stream processes to the watershed outlet. The field-scale SWAT project required custom watershed delineation in order to ensure each field had quantifiable outputs and could be uniquely edited to guarantee correct field-scale data, since it is inadvisable to rely on the accuracy of NLCD cover data at such a small scale (Daggupati et al., 2011). For this reason, the subbasin map was made to include all the field boundaries. The first step in the process was to load the elevation map (NED) into the SWAT delineation framework and perform automatic delineation using minimum area for flow accumulation, thus rendering the most detailed subbasin map and reach network that the automatic delineation module could create. Then, the common land use unit (CLU) field boundary shapefile was used to draw the individual field in the subbasin map (figure 2). The newly created subbasins included all field boundaries, so that each field from the field map had a corresponding subbasin in the subbasin map. In the SWAT model, every unique subbasin/field must have a unique reach segment. Meanwhile, the NHDPlus stream network is

not detailed enough and does not pass through all individual fields (figure 3); therefore, for every extra subbasin created, a corresponding reach segment was added. The splitting of subbasins and adding of reach segments was done manually using the high-resolution elevation map (10 m resolution), NHDPlus stream network, and NHDPlus catchment map simultaneously as guides. The goal was to create a subbasin and stream network as close to reality as possible. After the shapefiles (subbasin and reach) were created, all records in the subbasin and reach files were edited manually to ArcSWAT predefined network guidelines, which include giving each subbasin and the reach a corresponding unique identification number. In addition, the reach flow direction had to be defined by listing the identification numbers of the reach or reaches each subbasin flowed into. The challenging nature of this task arises from the need for selective detail, which is unattainable using automatic delineation in ArcSWAT. The subbasin and reach network were designed to be very detailed where fields of interest existed and less detailed in all other locations. By delineating the watershed in this way, relatively small fields could still be included as unique subbasins and the calculation time would still be reasonable by keeping the total number of subbasins to a workable number (163 subbasins in this study). The remaining model set up included hydrologic response unit (HRU) definition. The HRU classification was set to 20% land cover, 10% soil cover, and 20% slope class by area. All agricultural land uses included in the field data (corn, soybeans, winter wheat, tomatoes, and cabbage) were exempted from the HRU classification. Finally, the model was calibrated and validated for the flow and sediment loads at the watershed outlet. The creation of the field-level subbasin in SWAT model is a unique aspect of this method, and to the best of our knowledge, has not been described in published work.

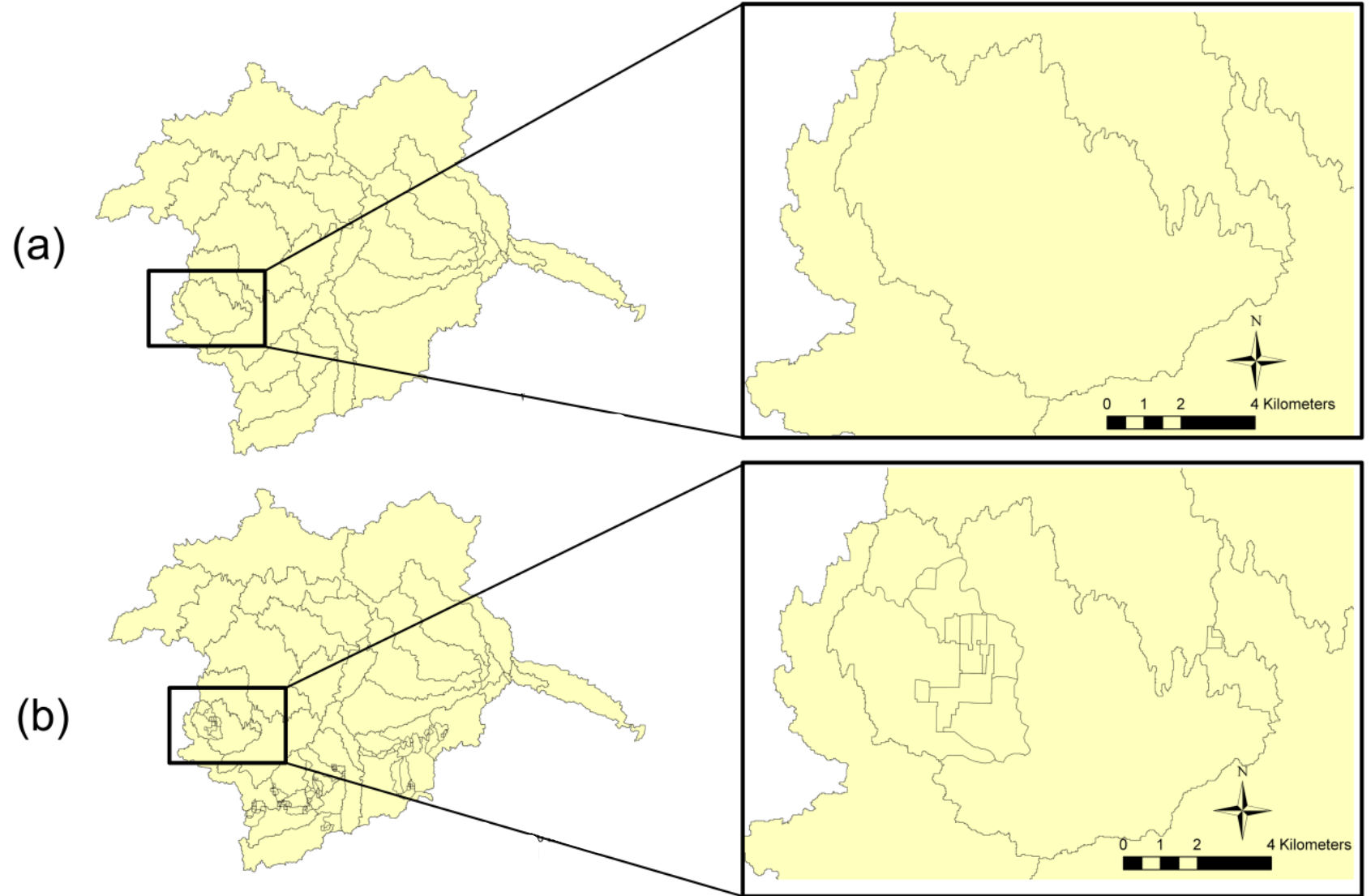


Figure 5-2: (a) Automatically delineated subbasin and (b) predefined subbasin map

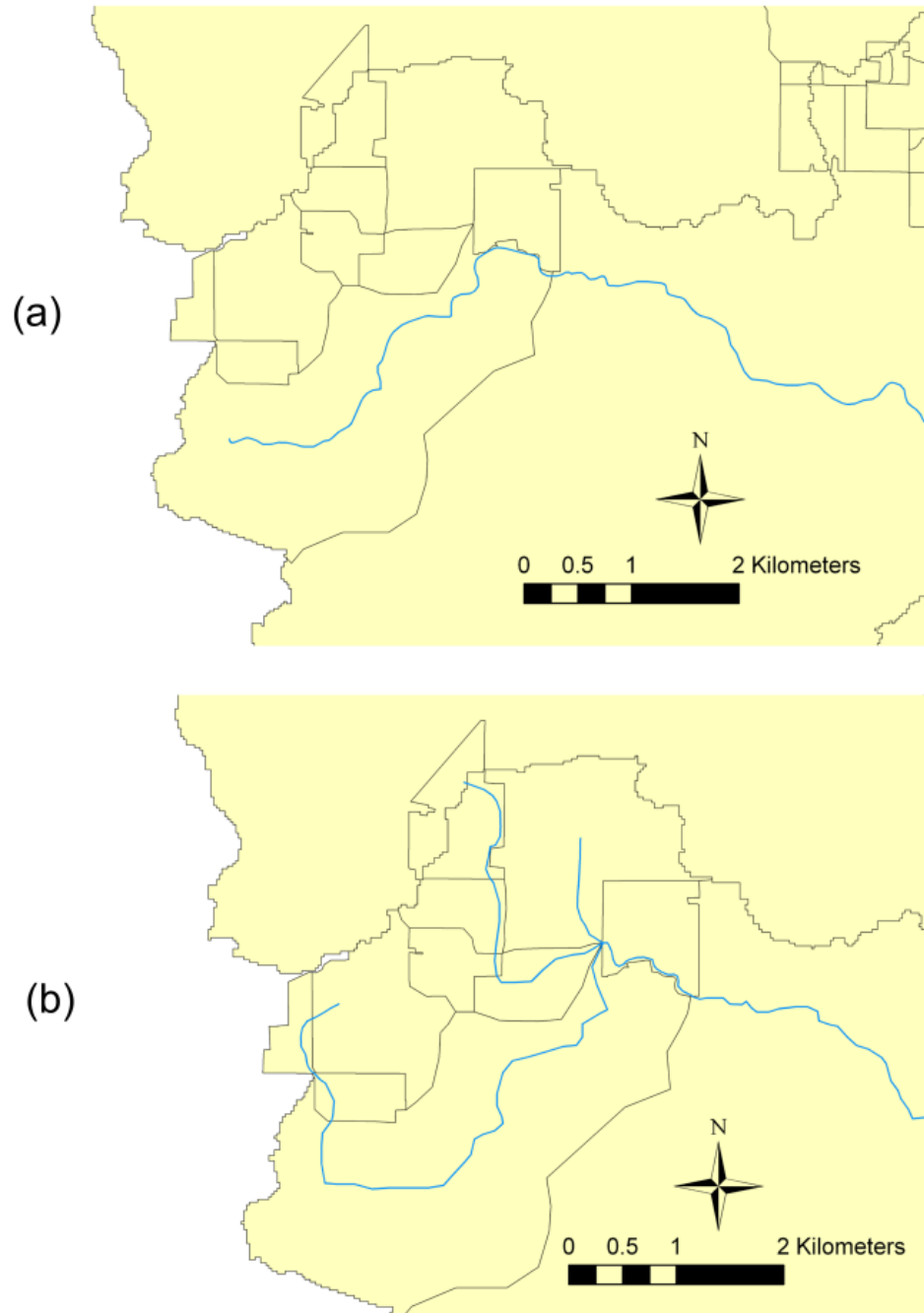


Figure 5-3: Comparing the NHDPlus stream network (a) versus the manual stream network delineation (b)

5.5.1.1 RUSLE2 Field-scale Sediment Loads

In order to provide consistency between the four methods comparison and estimate sediment load at higher accuracy level than SWAT model, RUSLE2 was used to calculate sediment loads from each field. Detailed information on crop rotations, management practices, and soil types are needed to attain satisfactory results. Producer-specific information can be difficult to acquire, but is necessary for best results. In this study, all management and crop rotation data were obtained in collaboration with the Lenawee County NRCS and Conservation District. Every field was assigned specific crop rotations and managements based on information provided by the farmers themselves. Although time consuming, this is a critical step in accurate quantification of sediment yields at the field-scale. For the RUSLE2 model, detailed soil descriptions were obtained from WSS. For a given field, between one to thirteen soil types were identified. The total sediment load for each field was estimated by summing sediment load from each soil type multiplied by the area coverage. For this reason, a spreadsheet tool was designed in Microsoft Excel to assist in total load calculations. The spreadsheet tool was designed around a matrix that included all possible RUSLE2 answers given the management operations and soil types. This matrix was created based on one soil type at a time, rather than one management at a time, which greatly reduced the amount of user time required for calculating sediment runoff values with RUSLE2. Then, the total sediment load for each field was calculated by summing the multiplications of sediment load (obtained from RUSLE2) by the number of hectares covered by the specific management and soil type combination occupied in the field. In this way, accurate total sediment load for each field could be calculated while reducing the amount of user effort and time spent in the RUSLE2 interface.

5.5.1.2 Replacement of SWAT Field-Scale Sediment Runoff Estimation with RUSLE2

In order to incorporate the sediment loads estimated from the RUSLE2 model into the calibrated SWAT model, the support practice factor was set to zero for each subbasin representing a field within the SWAT model to eliminate sediment erosion. In the next step, a point source input was added to each field equal to the amount of sediment runoff calculated by RUSLE2 one at the time. After running the current scenario, BMPs were applied by changing the point source value for each field from the current situation to the proposed BMP situation, also calculated with RUSLE2. After each model run (before or after BMP implementation), the sediment yield at the watershed outlet was obtained. Next, the difference between the average annual sediment loads from the current and BMP scenarios was calculated. This difference was defined as sediment savings at the watershed outlet due to the implementation of the BMP.

5.5.2 Subbasin-Scale Sediment Delivery Ratio (Method 2)

The second method was a combination of the RUSLE2 model and subbasin level SDR calculated using SWAT. The average SDR was calculated for each subbasin defined by the automatic delineation procedure. The method described by Woznicki and Nejadhashemi (2012) was employed in this study. They defined SDR for a subbasin as the ratio of long-term sediment yield at the watershed outlet to the total sediment load at the subbasin. This method involves forcing sediment load to zero for one subbasin at a time and comparing the total sediment yield at the watershed outlet to the amount of sediment at the watershed outlet with no subbasins sediment yield set to zero. Sediment load is forced to zero for a subbasin by changing the USLE support practice factor (USLE_P) to zero for every HRU in the subbasin of interest. The SDR was calculated using Equation 5-9.

$$SDR_i = \frac{Y_{O,P=1} - Y_{O,P<1}}{L_{i,P=1} - L_{i,P<1}} \quad (5-9)$$

The variable $Y_{O,P=1}$ represents the sediment yield at the watershed outlet under the base scenario, when all subbasins are contributing sediment runoff. The sediment yield at the watershed outlet when USLE_P is less than 1 is represented by $Y_{O,P<1}$. The sediment load from subbasin i contributed to reach i is $L_{i,P=1}$ and the sediment load contributed to reach i from subbasin i when USLE_P is less than 1 is $L_{i,P<1}$. This process was completed for all subbasins and SDR by subbasin map was created as shown in Figure 5-4.

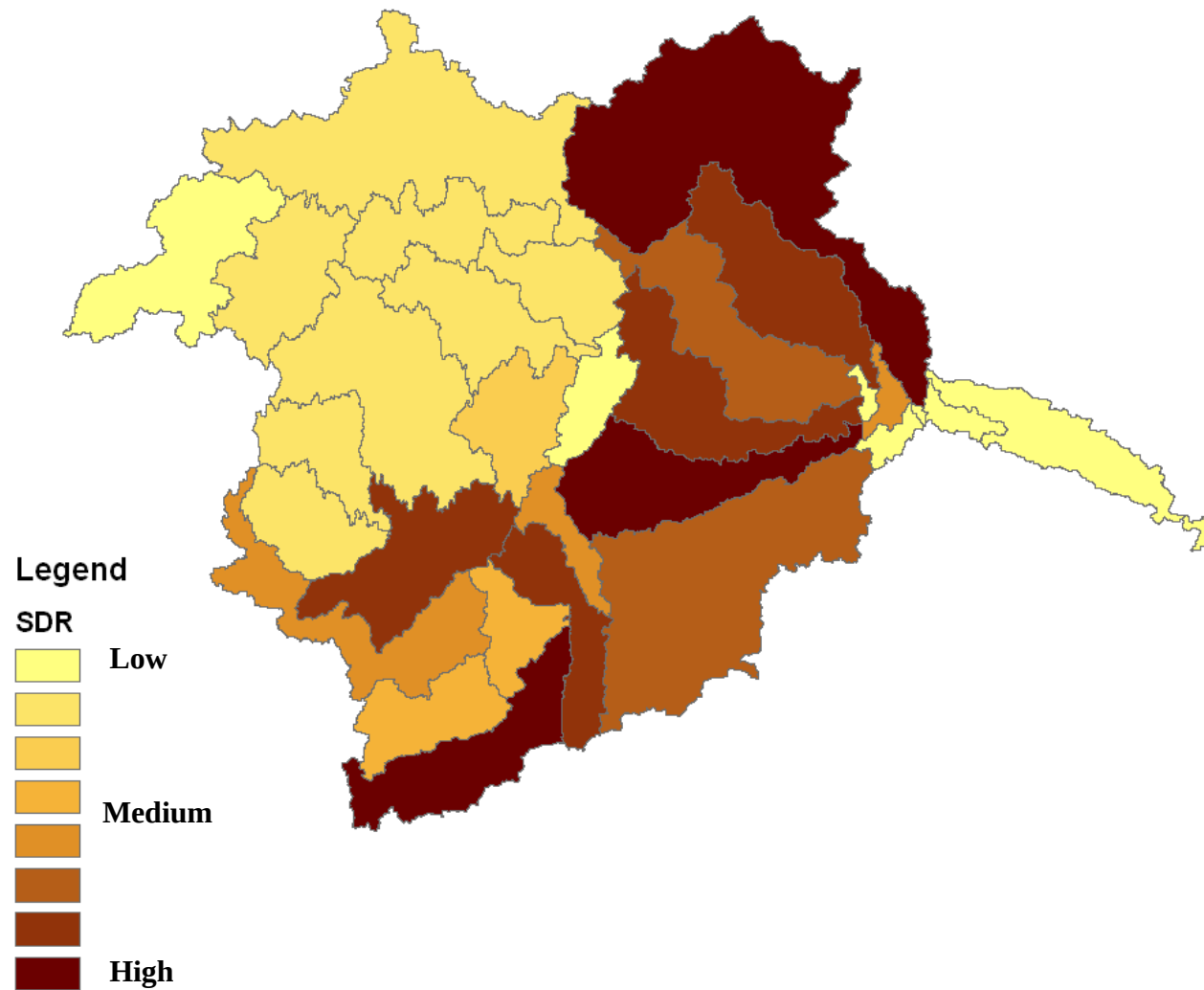


Figure 5-4. Subbasin-scale delivery ratio map

A SDR value was assigned to each field by layering the field map over the SRD map created in the previous step. The overlay command used in the ArcGIS software and fields contained within a certain subbasin were assigned that subbasin's SDR. This SDR was multiplied by the average annual sediment calculated for each BMP with the RUSLE2 model. Sediment output at the watershed outlet was obtained for the period of study (2000-2009) before and after BMP implementation on each field. The difference between these two values represents average annual sediment savings at the watershed outlet from each BMP.

5.5.3 SWAT without Field-scale Delineation (Method 3)

The third method employed a SWAT model for in-stream process calculation and RUSLE2 model to estimate sediment load at the edge of the field before and after the BMP implementation. Similar to Method 1, the process begins by performing automatic watershed delineation using minimum area for flow accumulation, thus rendering the most detailed subbasin map and reach network that the automatic delineation module can create. Already knowing the coordinate systems of all fields, the closest reach to each field was identified and point source discharge to the reach was added to the SWAT project. Therefore, this method is a simplified version of Method 1, in which the needs for manual extension of the NHDPlus reach to each field within the watershed is eliminated. The model was run for each field twice (before and after BMP implementation) and the 10 years (2000-2009) average sediment yield at the watershed outlet was calculated. The difference between the average annual sediment loads was defined as sediment savings at the watershed outlet.

5.5.4 Hybrid Solution Combining Analysis from the RUSLE2, SEDMOD, and SWAT (Method 4)

The fourth method is the most accurate among the studied methods and designed to account for three stages of sediment transport from a field to the watershed outlet: 1) estimate sediment runoff from the field using the RUSLE2 model, 2) account for deposition that occurs in overland flow between the field of interest and the closest NHDPlus reach using the SEDMOD model, and 3) calculate sediment transportation from stream entry point to watershed outlet using the SWAT model. This method is similar to Method 3, but employs SEDMOD to calculate the percent of sediment runoff from a field that actually enters a nearby stream, while Methods 2 and 3 assume all sediment leaving a field enters the stream network in the watershed. The field-scale estimate of sediment load was performed using the RUSLE2 model and spreadsheet tool similar to Methods 1 through 3. In the next step, the SEDMOD model was used to estimate SDR from each field to the closest stream. The SDR was calculated at 10 m resolution for the entire watershed (figure 5). In the next step, the common land use unit map containing the fields of interest was overlapped with the SEDMOD SDR map. Then, the average field level SDR was calculated for all fields within the watershed using the Zonal Statistic command in the ArcGIS software. The point sources were set in the SWAT model for the corresponding fields as a product of RUSLE2 model times the SDR from SEDMOD. A high-resolution delineation was needed for the point source placement. Since SWAT model only allows one point source per subbasin; therefore, an ideal SWAT model for this application would have one point source for every field. A Python program was developed that ran SWAT once for every unique point source before and after BMP implementation for each field. Once the runs were completed, the total sediment at the watershed outlet was compared

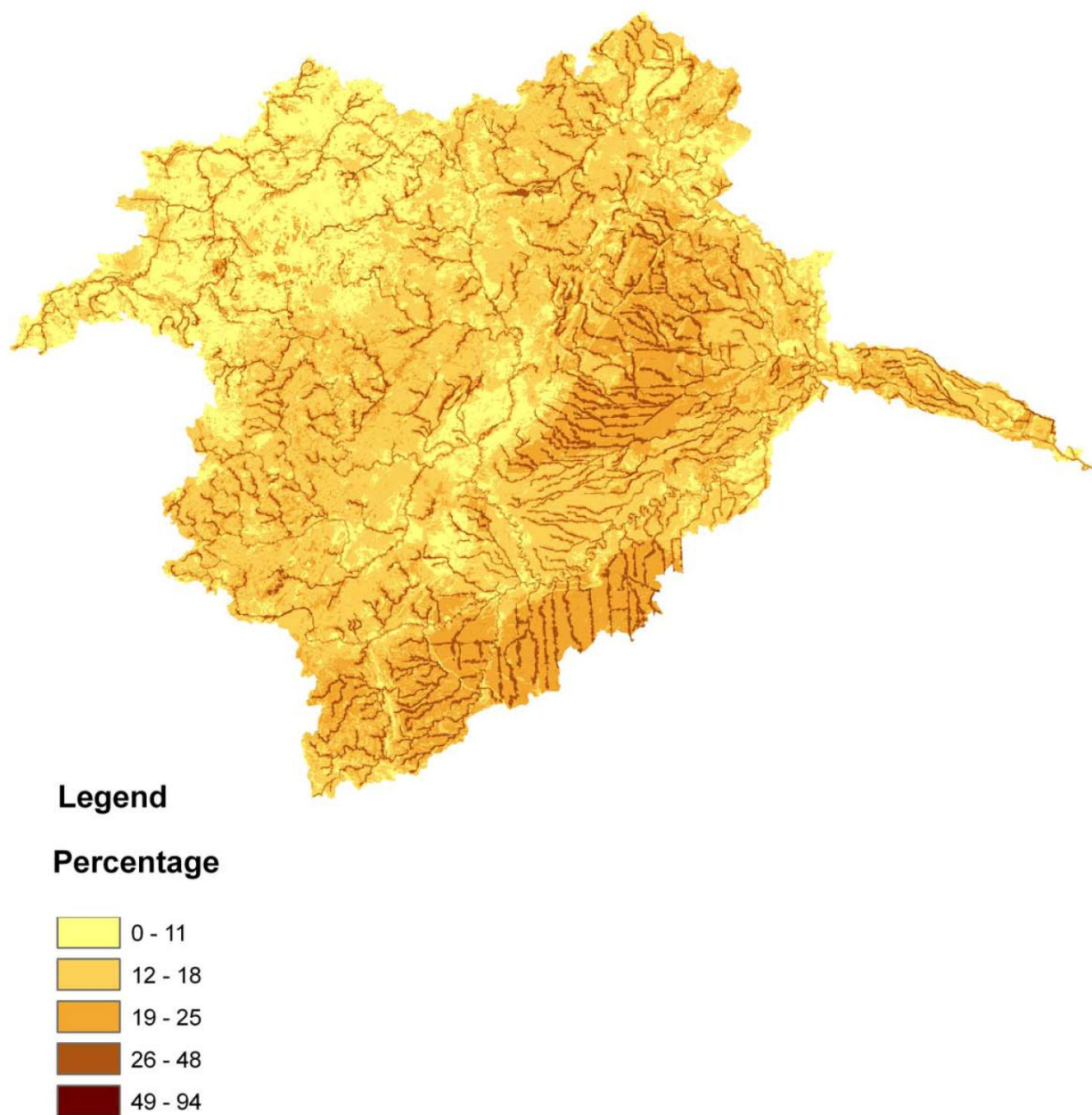


Figure 5-5. Grid-based delivery ratio map at 10 m resolution using the SEDMOD model

5.5.5 Best Management Practices (BMPs) Implementation Scenarios

The BMPs used in this study were proposed by the producers as part of the BMP Auction program (Smith et al., 2009). In the BMP Auction program, producers submit bids to a buyer, which would usually be a governmental agency involved in water resource management. The bids consist of a proposed BMP and requested price for installation. Producer defined costs create a competitive market and the costs are likely close to the true value of the BMP (Johansson, 2006). This is the reverse of the conventional process, in which the governmental agency defines the prices for a wide range of BMPs. Allowing producer to name their own prices gives watershed planners a unique opportunity to see a more realistic value of a BMP from the producers' standpoint. After performing water quality analysis for each field participated in the BMP Auction program, the bids are ranked based on dollar requested per ton of sediment reduction at the watershed outlet. In this study, the BMPs were proposed by the producers consist of a single or a group of BMPs. Six different BMP categories were included in this study: cover crop, cover crops and filter strips, filter strips, residue management, residue management and cover crops, and residue management and filter strips. All four methods were used to evaluate the sediment reduction at the watershed outlet for all 80 BMPs within the six BMP categories. This was performed to compare the inconsistency and variability in different BMP efficiency caused by using different simulation techniques (methods).

5.5.6 Economic Analysis

Currently, the primary USDA nationwide conservation programs that fund and promote BMPs are the Conservation Reserve Program (CRP), the Environmental Quality Incentive Program, (EQIP), and the Conservation Stewardship program (CSP). These programs are

administrated by the NRCS a subset of the USDA (NRCS, 2012). However, in this study we are focusing on the EQIP program because it is the largest program, with an annual budget of about \$1.3 billion (Shortle et al., 2012). The EQIP program is designed to assist agricultural producers by helping to fund the implementation and maintenance of BMPs for up to 10 years per contract (NRCS, 2012).

In the River Raisin watershed, we identified a group of the producers who were willing to adopt the BMPs supported by EQIP program if they were given a chance to set their own price and select their own BMP through the BMP Auction program. EQIP prices are generally set based on dollars per unit area. This is an easy way to make payments to producers interested in BMPs implementation; however, it does not take into account actual sediment reduction cost at the field or the watershed outlet. In this analysis, prices defined by producers are compared to prices from the EQUP program for the six BMP categories.

In this economic analysis, no external costs were considered. The monetary values used represent the implementation and maintenance costs of the best management practices. In the case of producer requested prices, the monetary values were those asked for by the producers in the survey. In the case of the EQIP defined prices, the monetary values were payments defined under the EQIP program based on the type of best management practice. The reason externalities were not considered in this analysis was the goal of this economic exercise: to increase the cost effectiveness of BMP programs. When evaluating cost effectiveness, the monetary values of implementation and maintenance—the equivalent of what a government sponsored BMP program would pay to implement a practice—are the costs of interest. Introducing externalities would cause the results of the economic analysis to have prices that do not reflect actual money being paid to producers, and the producer and EQIP price definitions could not be directly compared.

5.6 RESULTS AND DISCUSSION

5.6.1 SWAT Model Calibration/Validation

The SWAT model was the only model used in this study that requires calibration. Since each method required a slightly different SWAT model setup, four separate calibration/validation procedures were performed. Table 1 summarizes the overall calibration of validation results for each of the four methods. Based on the criteria described in section 2.3.3.1, the model performance during calibration and validation periods are satisfactory.

Table 5-1. Overall SWAT model calibration and validation for all four Methods

Method	Streamflow			Sediment		
	NSE	RSR	PBIAS	NSE	RSR	PBIAS
1	0.58	0.64	9.25%	0.58	0.64	19.9%
2	0.69	0.56	3.25%	0.53	0.68	26.9%
3	0.55	0.67	5.05%	0.58	0.65	-7.28%
4	0.50	0.70	5.02%	0.52	0.69	-1.94%

5.6.2 Predefined Field-Scale Subbasin and Reach Layers in SWAT Model (Method 1)

The first method only produced a few results, all on fields near the watershed outlet. Among these fields, the farthest one located about 72 km upstream of the outlet. Implementation of BMPs on all other fields did not affect sediment load at the watershed outlet. This might be due to the fact that in SWAT model the small field sized subbasins did not produce enough surface runoff to carry sediment off-site. The point source loads (calculated independently from SWAT), resulted in extremely high sediment concentrations in the reach for each field. These concentration values were above the maximum sediment capacity for the flow rate, and thus, most of the sediment was settled within the subbasin reach and was not routed to the watershed

outlet. Thus, the limiting factor in this method is very low estimations of flow for very small subbasins. Meanwhile, for some fields (6 out of 80), a small sediment reduction at the watershed outlet was detected. A possible reason for these results is that the small amount of sediment that was leaving the subbasin was close enough to the outlet to not settle within the river network before reaching the watershed outlet.

The processes described above highlight the disadvantage of field-scale study in the SWAT model. This stems from the requirement of the SWAT model that all subbasins must contain a reach. Since field-sized subbasins were made, reaches that do not actually exist must be added to the river network, although they were created with scrutiny and followed likely patterns based on a high-resolution digital elevation model, the high-resolution NHDPlus stream network, and were combined with the subbasin drainage areas to make a robust SWAT project. In reality, the overland flow portion of the sediments' movement to the streams is much longer, and not all fields drain directly into streams. This method would perhaps be appropriate in a watershed with many open drainage systems adjacent to the fields. Although this network might be realistic, the same problem of insufficient flow for sediment transport would likely occur.

5.6.3 Subbasin-Scale Sediment Delivery Ratio (Method 2)

Method 2 was designed for ease of use and took a relatively shorter time to complete than the other methods. This low time requirement is because no field is directly modeled within the SWAT model. However, initial SWAT runs are still required to establish SDR values for each subbasin, which according to Woznicki et al. (2012) is equal to the number of subbasins in the SWAT project. Since in this study 80 BMPs scenarios were tested, at least 160 SWAT runs are required for Methods 1, 3, and 4 while just over 100 runs are required for Method 2 because

some fields are located within the same subbasin. In addition, the added time for defining, calculating, and applying point sources is avoided in Method 2. Overall, the second method was successful in quantifying sediment load at the watershed outlet for all fields. There are, however, limitations with this model. In Method 2, all fields within the same subbasin are all given the same delivery ratio, regardless of proximity to the stream, soil type, and a host of other factors that determine SDR. Therefore, this method ignores the net sediment deposition that occurs in overland flow process. On the other hand, 100% of sediment load at the edge of a field reaches the nearest stream, which is not a realistic representation.

5.6.4 SWAT without Field-scale Delineation (Method 3)

The third method built on the idea of Method 2, but rather than estimating SDR as a single value (long-term annual average of SDR per subbasin) this method included point sources, thus accounting for daily in-stream processes throughout the year. However, it was again assumed that 100% of the sediment leaving the fields entered the reach, therefore ignoring deposition during the overland processes. Similar to Method 2, automatic delineation was used in Method 3 and; therefore, in the case that multiple fields were contained within one subbasin, the fields were forced to share a point source. This method is also capable of capturing field-scale management practices at the watershed outlet.

5.6.5 Hybrid Solution Combining Analysis from the RUSLE2, SEDMOD, and SWAT (Method 4)

The final method accounts for three stages of sediment delivery from the field to the watershed outlet. Some of the most used and trusted models are employed for their respective applications, and the results are compiled. RUSLE2 effectively quantifies average field sediment

runoff, while SWAT tracks the sediment transport from its introduction into the stream to the watershed outlet. An estimation of the amount of sediment entering the nearest stream from the field is made using SEDMOD model. The SWAT model is used to quantify in-stream processes from the stream inlet to the watershed outlet. Overall, this hybrid solution had the most logically complete strategy to estimate the impacts of field-scale managements at the watershed outlet. The main difference in this method compared to the others is the inclusion of a sediment deposition after leaving the field and before reaching to the stream system affected by overland flow. The SEDMOD model is based on soil characteristics, elevation, and proximity to reach. Although this is not a comprehensive understanding of overland flow, it provides an approximation that is otherwise lacking in these procedures. Estimated percentage of sediment leaving the fields and entering the reach ranged from 13% to 29%, much lower than the previous assumption of 100%. Thus, the sediment point source values for Method 4 were much lower than the other methods.

5.6.6 Overall Method Comparison

In order to summarize our findings, a comparison of the advantages and disadvantages of each method described above is presented in table 5-2.

Table 5-2. Overview of the four methods

Method	Advantages	Disadvantages
1	*Field-scale subbasins allow for direct estimation of sediment load for each individual field	*Due to low runoff rate for small subbasins (fields) most sediment loads are deposited within the subbasins *Data preparation period is time-consuming *Unable to estimate the impact of field-scale management practices at the watershed outlet
2	*Simplest method tested *Fastest calculation time	* No field is directly modeled within the SWAT model *Assumes 100% of sediment leaving field enters the reach *SDR resolution is limited to the subbasin level *By using the long-term average SDR value, ignores the temporal SDR variation
3	*Uses the full potential of the SWAT model to estimate in-stream processes during sediment delivery *Requires fewer point sources than Method 2	*Ignores overland sediment deposition *Multiple fields can be represented by the same point source *Field location within subbasins is ignored
4	* Accounts for three stages of sediment delivery: field-scale, overland sediment deposition, and in-stream process *Considers field location	*Requires extra step for estimating overland sediment deposition using the SEDMOD model *SWAT snaps the close point sources and (correspondence to different fields) force them to share a similar point discharge to a reach

5.6.7 Overall Method Comparison in Evaluating BMP Implementation Scenarios

Each method was used to evaluate the sediment reduction at the watershed outlet for all 80 BMPs (six BMP categories). This was performed to compare the inconsistency and variability in BMPs efficiencies caused by using different simulation techniques (methods). As it was discussed earlier, the fourth method was considered as the most accurate among the studied methods and designed to account for three stages of sediment transport from a field to the watershed outlet. Therefore, the performances of different methods were compared to Method 4. The results are presented in Figure 6.

In the cover crop category, method 3 had the widest range of predicted sediment reductions at the watershed outlet; the minimum value was 1.74 tons and the maximum value was 64.3 tons. The median value for Method 3 was 17.7 tons. Method 2 predicted the next highest range (51 tons) and median (6.08 tons) for cover crop values. Therefore, comparing to Method 4 (median is 3.03 tons), Methods 2 and 3 overestimating the sediment reduction 101% and 484%, respectively. Method 1 predicted a minimum sediment reduction equal to 0 tons, a maximum of 0.86 tons, and a median of 0 tons. However, the range and median values for all other BMP categories are 0.

Method 2 had the highest range in cover crop and filter strip category and predicted a minimum of 0.536 tons, a maximum of 33.9 tons, and a median of 10.6 tons. Method 3 predicted the highest median value at 14.6 tons, with a range of 4.05 tons. Methods 2 and 3 over predicted sediment reduction at 303% and 455%, respectively, when compared to Method 4.

Similar trends were observed for filter strip, residue management, and residue management and filter strip categories. However, within the residue management and cover crop

category, smaller variation was observed between Method 2 through 4. The median values for Methods 2, 3 and 4 are 9.79, 8.80, and 4.21 tons, respectively. Meanwhile, the variations in BMPs' effectiveness are quite large and varies from 152.2 tons for Method 3 to 16.2 tons for Method 4.

Finally, in the residue management and filter strip category, the largest sediment reduction was observed for all methods except Method 1. In this BMP category, Method 3 had the highest median value at 38.7 tons, follow by 25.2 tons for Method 2 and 13.8 tons for Method 4. At the same time, the ranges for BMP efficiency are the smallest between all BMP categories.

Overall, evaluation of field-scale impact of management practices at the watershed outlet using Method 1 resulted in the lowest median values for all BMP categories and in many cases showed no impact at the watershed outlet. This highlights the disadvantage of field-scale study in the SWAT model that originates from a low estimation of flow for each field (subbasin) incapable of transporting the majority of eroded sediments to the reach and the watershed outlet. Meanwhile, Methods 2 and 3 consistently produced higher median sediment loads than method 4. This is likely due to ignoring the sediment delivery component from the field to the stream in these methods. Method 4 employs the SEDMOD model to account for this stage of sediment transport, and sediment loading values at the stream were reduced by over 70% in most cases. The lower median value for Method 4 is a trend across all BMP categories and proves that field overland flow and deposition between the field outlet and stream inlet is an important step to include in field-scale BMP evaluation.

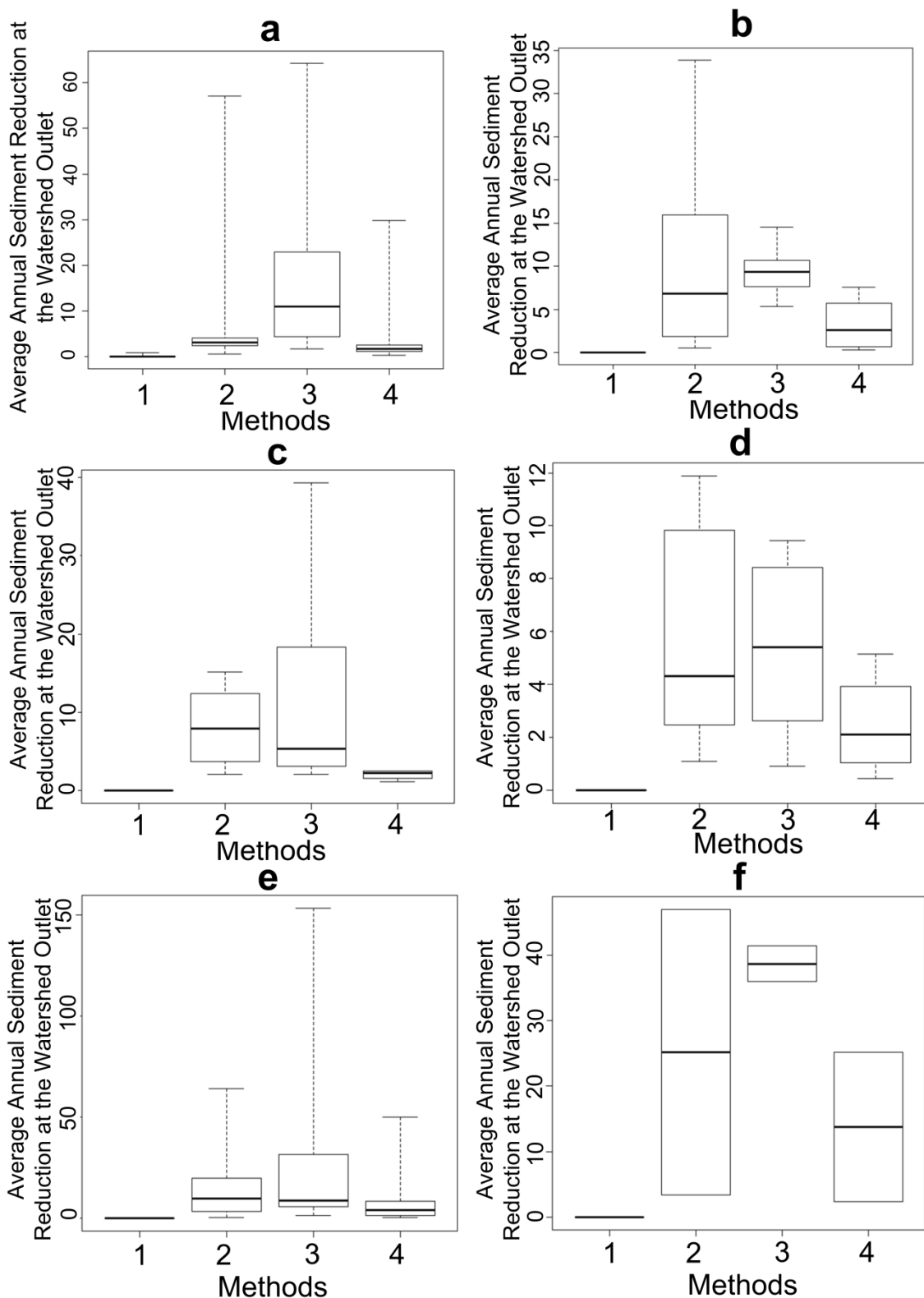


Figure 5-6. Comparison of four modeling strategies: a is cover crop, b is cover crop and filter strip, c is filter strip d is residue management, e is residue management and cover crop, and f is residue management and filter strip

5.6.8 Economic Analysis

The focus of an economic analysis of a watershed project is defined by the project goals. Three main goals addressed by this economic analysis were identified as: 1) increase producer participation, then develop an analysis focused on the price of BMP implementation per unit of application area to provide insight into how much producers believe certain BMPs to be worth; 2) sustain field productivity by keeping soil on site – a field-level analysis of BMP implementation will allow watershed stakeholders to find the BMPs most effective at erosion reduction; and 3) protection of lakes, downstream structures, and aquatic ecosystems – evaluating the price of BMPs on a per ton sediment reduced at the watershed outlet allows watershed planners to identify BMPs that will result in greatest water quality improvement.

5.6.8.1 Goal 1: Improve Producer Participation in Conservation Programs

The goal of improving producer participation is simple: increase interest and therefore the number and quality of agricultural practices aimed at improving water quality. The BMP Auction addresses the top reasons producers list that keep them out of government funded projects (Smith et al., 2009). Economically, this means allowing producers to define their own prices for BMP implementation. These prices are most often defined on a per unit area basis. In this study, producers defined their own prices, and, in the Table 3, they are compared to EQIP prices that would be paid for the same practices through the NRCS. The EQIP and producer prices are weighted based on area used for each BMP. For example, the filter strip EQIP price is \$1,032.90 per ha, but the cover crop and filter strip combination it is only \$61.57, which is the weighted average for the area used for cover crop and filter strip. Therefore, the \$/ha for cover crop and

filter strip is much less than just filter strip because the majority of the area is devoted to cover crops rather than filter strip, while the entire area for filter strip is under one BMP.

Table 5-3: Median average \$/ha for producer requested and EQIP defined prices

BMP Category	Median Producer Requested Price	Median EQIP price
Cover Crop	\$74.13	\$48.16
Cover Crop and Filter Strip	\$128.60	\$61.57
Filter Strip	\$97.41	\$1,032.90
Residue Management	\$24.71	\$22.61
Residue Management and Cover Crop	\$27.18	\$35.39
Residue Management and Filter Strip	\$45.20	\$36.76

In four out of six BMP categories, median producer requested prices were greater than that of EQIP. The residue management and cover crop combination and filter strip categories stand out as having lower prices than the EQIP. Meanwhile, EQIP is over-valuing most of the BMP categories compared to producers on per unit area basis. It is important to note that initially we were expecting that the producers' requested price for BMP implementation would higher than the EQIP program because they are currently participating in the EQIP program. However, the large discrepancy in price requested for the filter strip results from the fact that the filter strip program under EQIP is rigid (in terms of width, harvesting, and the length of the contract). However, the length of the contract under the BMP Auction can be one to several years. These differences suggest that producers may be willing to implement the filter strip under significantly lower cost if the regulations become less rigid.

5.6.8.2 Goal 2: Field Soil Conservation and Agricultural Sustainability

Evaluating prices based on sediment runoff reduction at the field-scale allows watershed planners to measure the effectiveness each BMP to save soil on site. This analysis is appropriate when project goals are related to soil conservation and agricultural sustainability. Both producers' requested and EQIP prices per unit sediment runoff reduction at the field-scale are presented in the Table 5-4. This comparison is important, because it illustrates the difference in true price of sediment reduction at the field-scale.

Table 5-4: Producer Requested and EQIP Median Prices per Ton Sediment Reduction at the Field Outlet

BMP Category	Median Producer Requested Price per Ton of Sediment Reduction (\$/ton)	Median EQIP Price per Ton of Sediment Reduction (\$/ton)
Cover Crop	\$73.59	\$47.22
Cover Crop and Filter Strip	\$199.29	\$98.10
Filter Strip	\$107.94	\$1,151.92
Residue Management	\$22.05	\$20.17
Residue Management and Cover Crop	\$23.10	\$18.90
Residue Management and Filter Strip	\$12.09	\$9.83

In five out of six BMP categories, the producer requested median cost per ton of sediment reduction was more than those defined by EQIP. The filter strip category was the exception, suggesting that, water quality can be significantly improved if the EQIP program becomes less rigid in its requirements. Meanwhile, the combined residue management and filter strip program is the most effective BMP category among the studied BMPs with the price of \$12.09 and \$9.83 for the producer requested and EQIP, respectively.

5.6.8.3 Goal 3: Watershed-Scale Sediment Reduction

In the previous sections of this paper, a comparison between four methods illustrated how different modeling techniques can have widely varying results, even when similar input data are used. For this reason, it is extremely important to select the simplest approach that not only accurate but also effectively address the goal of the study (Nejadhashemi et al., 2009). Ultimately, Method 4 was selected for economic analysis because the comparison and evaluation of results deemed this method as the most accurate that can answer the question in hand, which is the true cost of sediment reduction at the watershed outlet. In addition, Method 4 was the only method that directly accounted for all three stages of sediment transport and produced measurable results. Measuring the effectiveness of BMPs based on sediment reduction at the outlet gives watershed planners a way to measure BMP effectiveness based on overall watershed water quality improvement.

In figure 7, the “producer requested” prices are total prices submitted by the each producer divided by the calculated sediment reduction at the watershed outlet. The EQIP prices are the total cost for each BMP defined under EQIP guidelines divided by the reduction at the watershed outlet. Overall similar trends were observed at watershed-scale and field-scale. However, the box and whisker plots also allow examining the range in addition to the median values. Among the studied BMP categories, only filter strip has the higher range in EQIP program than the producer requested costs. Meanwhile, the median cost of sediment reduction at watershed outlet using filter strip is \$10,914 under the EQIP, significantly larger than producers requested price of \$811. However, the median price for the rest of the BMPs under the EQIP

program is varies from \$155 (residue management and filter strip) to \$1140 (cover crop and filter strip). These values are \$193 (residue management) to \$2041(cover crop and filter strip) per ton of sediment reduction under the BMP Auction program (producer requested price). This can provide valuable information to watershed manager and stakeholders to estimate the true cost of conservation programs.

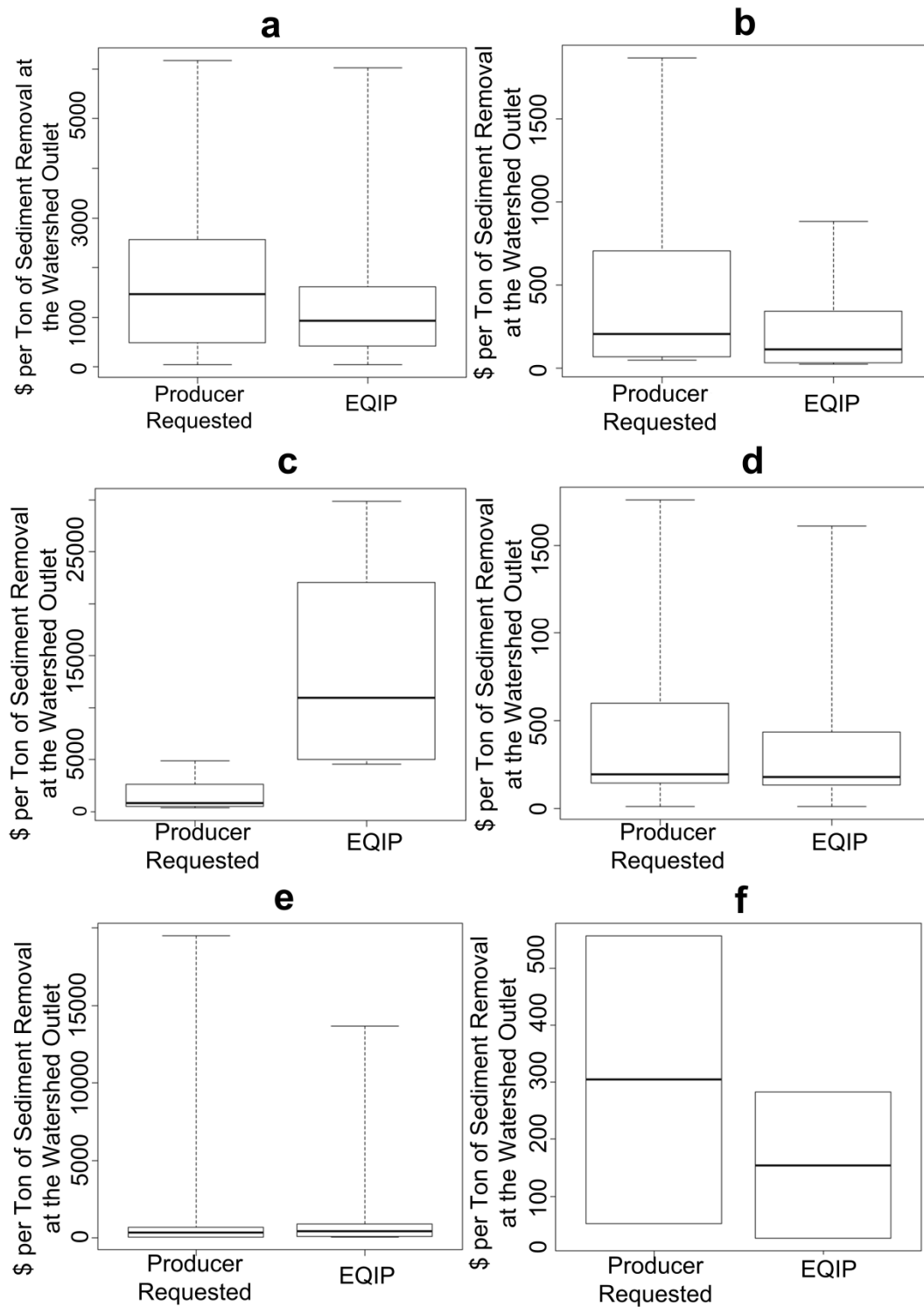


Figure 5-7. Comparison of dollar spent per ton sediment reduction at the watershed outlet under farmer defined and EQIP prices: a is cover crop, b is cover crop and filter strip, c is filter strip d is residue management, e is residue management and cover crop, and f is residue management and filter strip

5.7 CONCLUSION

In this study, four methods were compared in order to evaluate a simple but effective technique for quantifying the impact of field-scale management practices at the watershed outlet in the River Raisin watershed. The methods tested were using: (1) predefined field-scale subbasin and reach layers in SWAT model; (2) subbasin-scale SDR; (3) results obtained from the field-scale RUSLE2 model as point source inputs to the SWAT watershed model; (4) a hybrid solution combining analysis from the RUSLE2, the Spatially Explicit Delivery Model (SEDMOD), and SWAT models.

Method 1 proved to be an ineffective way to evaluate the field-scale BMP implementation strategy based on sediment reduction at the watershed outlet. The limiting factor in this method is very low estimations of flow for very small subbasins. Meanwhile, Methods 2 and 3 were able to quantify the field-scale management practices at the watershed outlet, but ignore the net sediment deposition occurs in overland flow process. Method 4 in the most accurate among the studied methods and designed to account for three stages of sediment transport from a field to the watershed outlet: 1) estimate sediment runoff from the field, 2) account for deposition that occurs in overland flow between the field of interest and the closet reach, and 3) calculate sediment transportation from stream entry point to watershed outlet. In this method, RUSLE2, SEDMOD, and SWAT hybrid model effectively quantified all the field-scale management actions at the watershed outlet. Overall, the results of this study showed that the SWAT model is not capable of capturing field-scale activities and delineation should be limited to reasonably sized subbasins suggested by automatic delineation or predefined NHD Plus catchments. In addition, the second stage of sediment transport, sediment deposition on

overland, should be included in future studies dealing with development of field-scale BMP implementation strategies.

The above-described methods were used to evaluate the impact of field-scale BMP implementation scenarios at the watershed outlet. Six different BMP categories were included in this study: cover crop, cover crops and filter strips, filter strips, residue management, residue management and cover crop, and residue management and filter strips. Overall, analysis using Method 1 provided the lowest median value for all BMP categories, and failed to measure any difference at for over 90% of the fields in this study. Using the SWAT model at the field-scale has the main disadvantage of the low flow estimations for small subbasins (fields), causing estimated sediment erosion to stay on-site, never reaching the stream network. Methods 2 and 3 produced higher median sediment loads than method 4. The lack of a sediment delivery component is the most likely reason for the higher estimates. Method 4 accounts for all stages of sediment transport to the watershed outlet. The SEDMOD model is used to estimate overland flow and deposition between the field outlet and stream inlet. This stage of sediment transport reduced loading values at the stream by over 70% in some cases. Method 4 predicted lower median values across all BMP categories, proving that overland flow and deposition between the field outlet and stream inlet is important to consider when evaluating the effects field-scale BMPs have at the watershed outlet.

An economic analysis using producer requested prices and EQIP program for BMP implementation has identified how producers value BMP on a per unit area basis compared to current water quality programs. The true costs of sediment reduction on both the field-scale and watershed scale from producer requested prices and current water quality improvement projects were quantified. On a per unit area basis, producers requested higher prices than EQIP in four

out of six BMP categories. Meanwhile, the true cost of sediment reduction at the field and watershed scales were greater than EQIP in five out of six BMP categories according to the producer requested prices. These results indicate that the true values of BMPs through the eyes of the producers are not being adequately addressed with the EQIP program. More producer input is needed when government agencies design water quality improvement programs. Or alternatively, true cost can be more directly addressed by implementing non-traditional programs such as the BMP Auction. In either case, it is clear that producer input and participation is paramount to the success of water quality programs.

Future work focusing on development of an all-in-one model capable of providing information at three stages of sediment transport from a field to the watershed outlet would give watershed planners an effective tool for decision making. In addition, it is important to perform similar economic analysis in different watersheds to better understand the true costs of conservation programs.

6. CONCLUSIONS

Recently, different methods for estimating field-scale sediment loads have been developed. These methods were designed to provide information to decision makers with the intention of improving water quality at the watershed level. SWAT, Field_SWAT, and the HIT model are a few of these methods. No information was provided in the literature about the reliability of the field-scale estimations from these models. Quantifying the effectiveness of these methods to produce accurate field-scale sediment yield estimations is an important step towards using the models responsibly. In this study, the reliability of each of the three models was tested against a detailed, calibrated field-scale model, RUSLE2, resulting in the following conclusions:

- the SWAT model produced the most accurate estimates in comparison to RUSLE2, by providing the closet median and the lowest absolute error in sediment yield predictions
- a statistical analysis found SWAT to be the only watershed that provided a data set insignificantly different from the calibrated RUSLE2 model, showing that SWAT performed best at field-scale analysis among the watershed-scale models
- all watershed-scale models were found to be incapable of identifying the three areas of priorities similar to RUSLE2 model
- the SWAT model provided the most correct estimates within the uncertainty bounds, at 51%, while Field_SWAT provided 27%, and HIT 39%, showing that none of the watershed models are satisfactory for field-scale analysis

Based on these conclusions, four methods were compared to evaluate techniques designed for quantifying the impact field-scale management changes have at the watershed outlet in the River Raisin watershed. Since none of the watershed-scale models alone were capable of field-

scale analysis, the field-scale component of each of the following methods was calculated using RUSLE2. The SWAT model was used for the in-stream dynamics portion of the analysis. The methods tested were: (Method 1) predefined field-scale subbasin and reach layers in SWAT model; (Method 2) subbasin-scale sediment delivery ratio; (Method 3) results obtained from the field-scale RUSLE2 model as point source inputs to the SWAT watershed model; (Method 4) a hybrid solution combining analysis from the RUSLE2, SEDMOD, and SWAT models. Each of these methods was used to evaluate impacts of BMPs at the watershed outlet. Six different BMP categories were included in this study: cover crop, cover crops and filter strips, filter strips, residue management, residue management and cover crop, and residue management and filter strips. The following conclusions can be drawn from the analysis:

- Method 4, the hybrid RUSLE2-SEDMOD-SWAT model was identified as the only reliable method of the four tested to estimate the effects of individual BMPs at the watershed outlet
- the true values of BMPs defined by producers are not being adequately addressed with the EQIP program; therefore, more producers inputs are needed when government agencies design water quality conservation programs

7. RECOMMENDATIONS FOR FUTURE RESEARCH

In this study, a detailed analysis of field-scale sediment yield estimations and their effects at the watershed outlet was performed using multiple models and methods. In addition, an economic analysis quantifying the true costs of BMPs both at field and watershed scale was performed. However, there is a clear need for research in order to integrate effective field-scale estimations and true costs of BMPs into watershed-scale analysis, decision making, and conservation planning. Suggestions for future research based on the findings of this study include:

- Investment in collecting and digitizing field-scale management operations, which found to be a key in reliable estimation of sediment load both at the field and watershed scales.
- The development of an all-in-one model capable of providing information at three stages of sediment transport from a field to the watershed outlet. This would be an effective tool for watershed planners. Depending of the goals of a water quality program, BMPs could be ranked and funded accordingly.
- Expanding the economic analysis outlined in this paper to different watersheds around the country. These actions would increase the understanding of the true costs of conservation programs in different localities. Studies such as this can help with the development of more effective conservation programs and shed light on the true costs of sediment reduction, building a knowledge base that can guide conservation planners in future water quality programs.

APPENDIX

APPENDIX

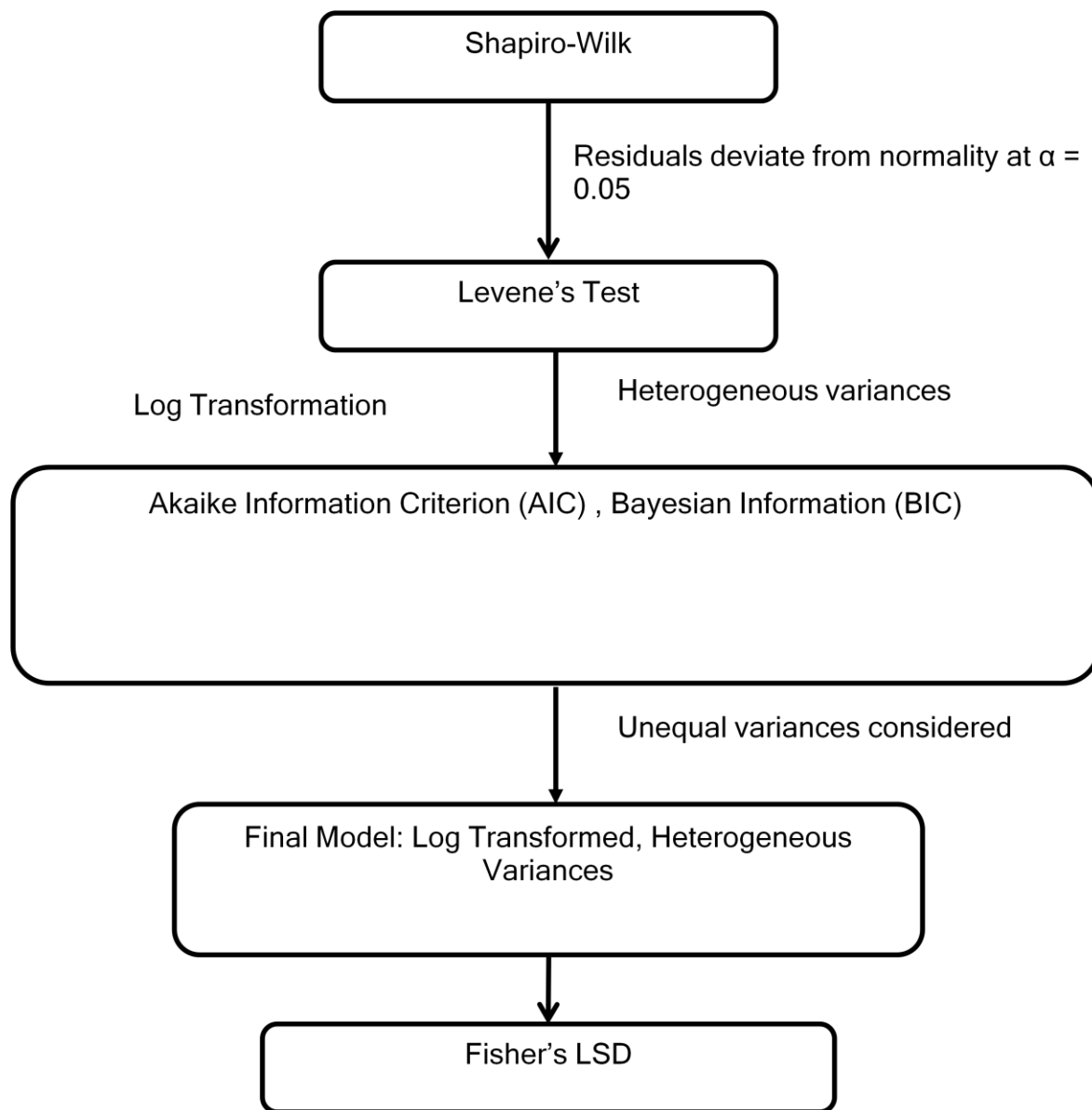


Figure A-1. Schematic of logical flow path for statistical analysis

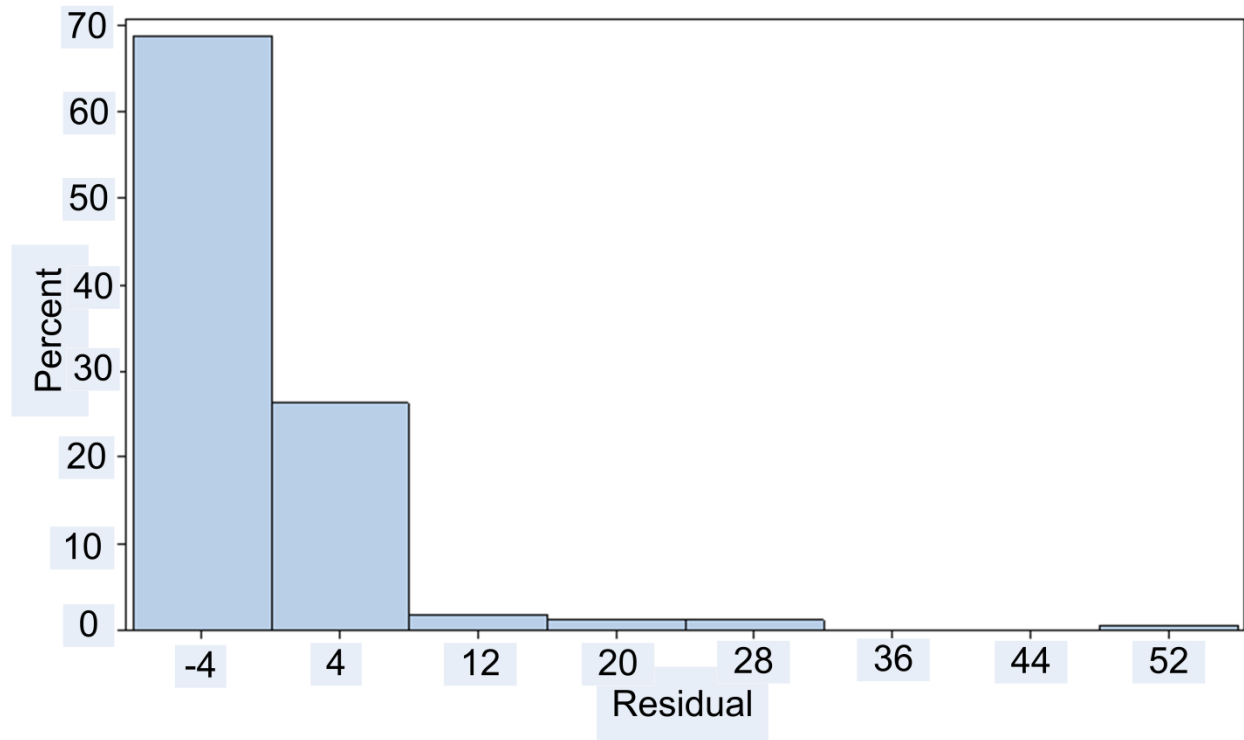


Figure A-2. Histogram of residuals (data not transformed)

Table A-1. Residuals (data not transformed), P-values

Test	--Statistic--		-----p Value-----	
Shapiro-Wilk	W	0.502371	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.322231	Pr > D	<0.0100

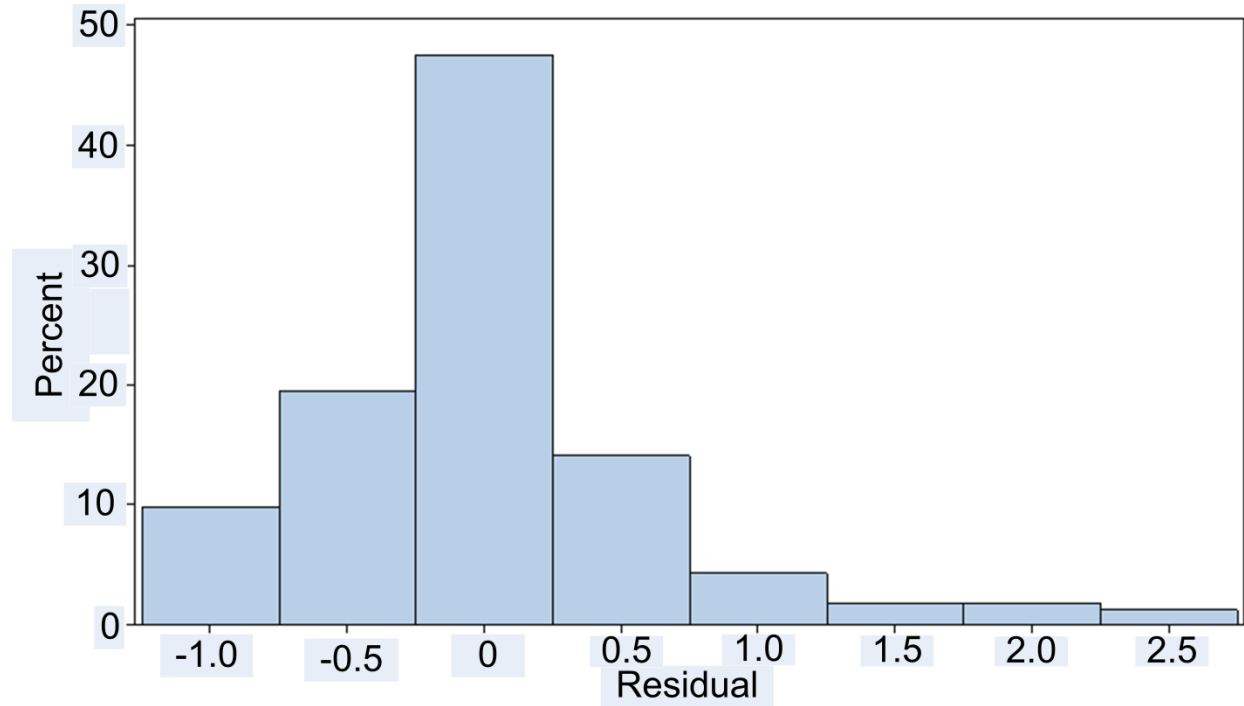


Figure A-3. Histogram of residuals (log transformed raw data)

Table A-2. Residuals (log transformed raw data), P-values

Test	--Statistic--		-----p Value-----	
Shapiro-Wilk	W	0.887311	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.125953	Pr > D	<0.0100

*1.4 check normality of residuals sqrt;

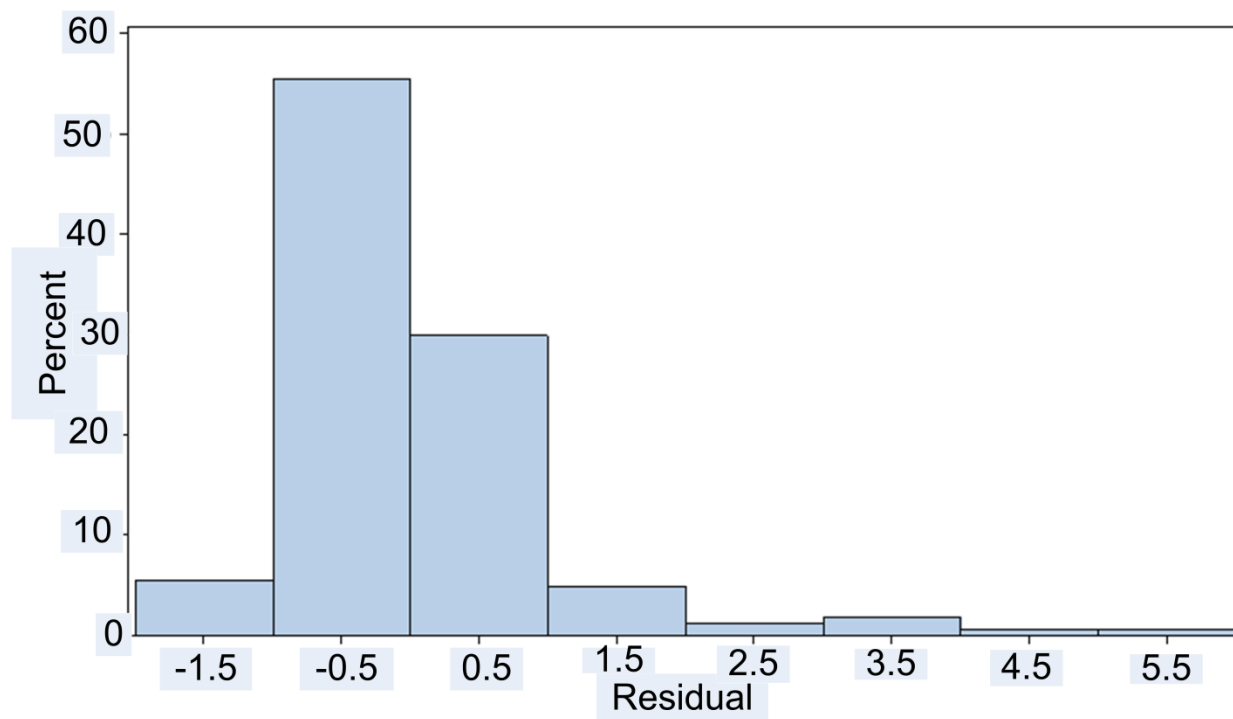


Figure A-4. Residuals (square root transformed raw data)

Table A-3. Residuals (square root transformed raw data), P-values

Test	--Statistic--		-----p Value-----	
Shapiro-Wilk	W	0.753258	Pr < W	<0.0001
Kolmogorov-Smirnov	D	0.198728	Pr > D	<0.0100

*Based on W-statistic, select log transform;

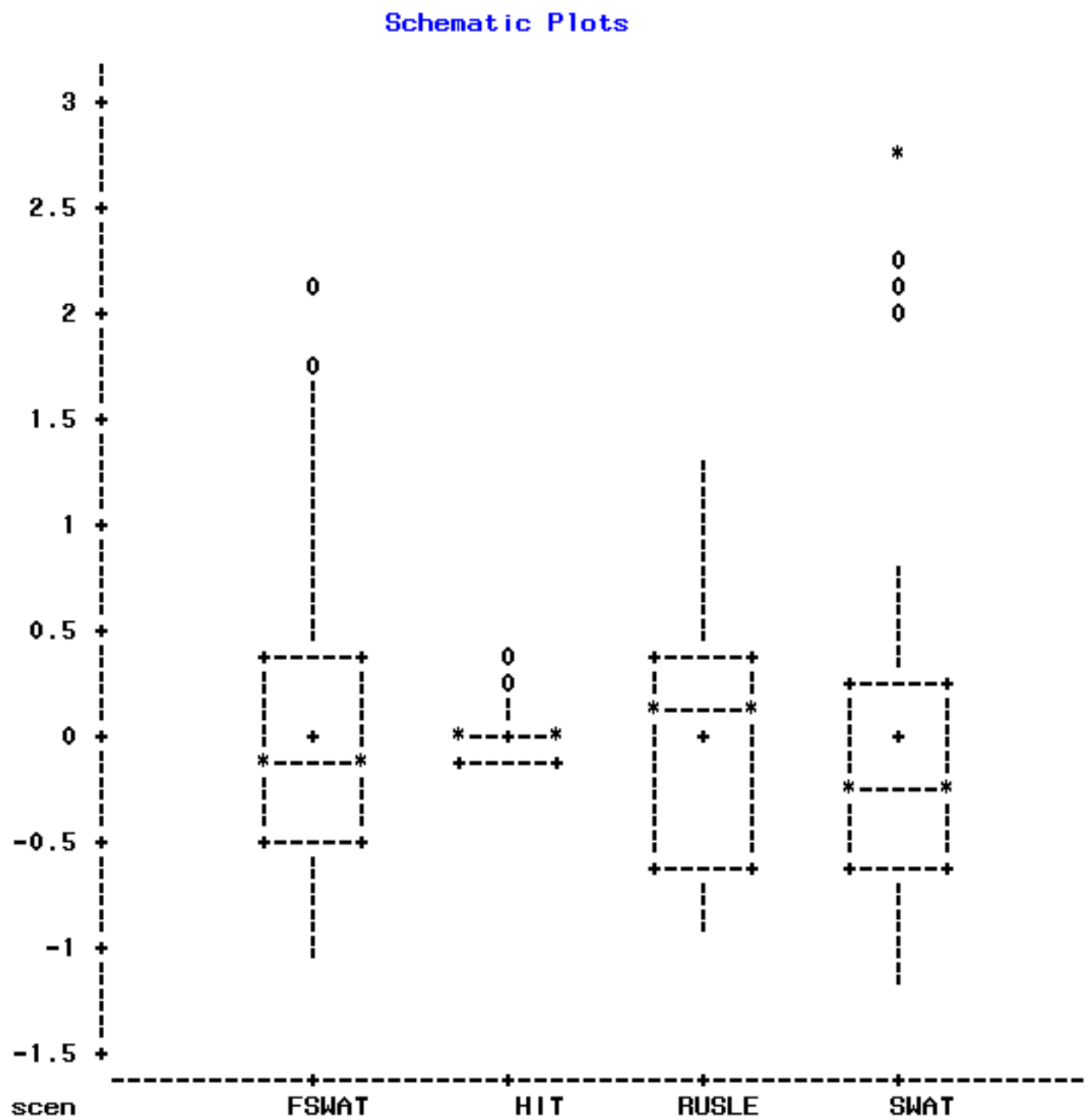


Figure A-5. Residual Variances for each model

Table A-4. Levene's test for equality of variances

Type 3 Tests of Fixed Effects				
Effect	Num		Den	
	DF	DF	F Value	Pr > F
scen	3	160	4.49	0.0047

Table A-5. Reject null hypothesis (alpha=0.05). The residuals are not homogeneous

Model	AIC	BIC
2.1	330.9	330.9
2.2	235.1	235.3

Table A-6. ANOVA Table

Type 3 Tests of Fixed Effects

Effect	Num		Den	
	DF	DF	F Value	Pr > F
scen	3	73	82.44	<.0001

Least Squares Means

Effect	scen	Standard		DF	t Value	Pr > t
		Estimate	Error			
scen	FSWAT	1.0097	0.1071	40	9.43	<.0001
scen	HIT	0.2101	0.01985	40	10.59	<.0001
scen	RUSLE	1.3849	0.09859	40	14.05	<.0001
scen	SWAT	1.3862	0.1382	40	10.03	<.0001

Differences of Least Squares Means

Effect	scen	_scen	Standard		DF	t Value	Pr > t
			Estimate	Error			
scen	FSWAT	HIT	0.7996	0.1089	42.7	7.34	<.0001
scen	FSWAT	RUSLE	-0.3752	0.1456	79.5	-2.58	0.0118
scen	FSWAT	SWAT	-0.3765	0.1748	75.3	-2.15	0.0344
scen	HIT	RUSLE	-1.1748	0.1006	43.2	-11.68	<.0001
scen	HIT	SWAT	-1.1761	0.1396	41.7	-8.43	<.0001
scen	RUSLE	SWAT	-0.00133	0.1697	72.3	-0.01	0.9938

Table A-7. Input data for statistical analysis

SWAT	1	2.424
SWAT	2	1.831
SWAT	3	3.381
SWAT	4	0.666
SWAT	6	6.919
SWAT	7	1.421
SWAT	8	0.815
SWAT	9	1.594
SWAT	10	1.841
	...	
HIT	1	0.101
HIT	2	0.134
HIT	3	0.096
HIT	4	0.137
HIT	6	0.379
HIT	7	0.437
HIT	8	0.166
HIT	9	0.226
HIT	10	0.309
	...	
RUSLE	1	1.457
RUSLE	2	1.457
RUSLE	3	1.457
RUSLE	4	3.138
RUSLE	6	5.156
RUSLE	7	2.242
RUSLE	8	2.690
RUSLE	9	0.650
RUSLE	10	0.650
	...	
FSWAT	1	14.717
FSWAT	2	20.668
FSWAT	3	12.129
FSWAT	4	8.535
FSWAT	6	0.799
FSWAT	7	2.466
FSWAT	8	1.201
FSWAT	9	3.050
FSWAT	10	3.275
	...	

List A-1. Code used to execute statistical analysis

```
data andy;
input scen$ sub sed;
cards;

;run;

proc print data=andy;
run;

*checking assumptions;
proc mixed data=andy;

class scen;
model sed=scen/ outp=mr;
proc print data=mr; run;

*1.1 checking normality of residuals;
proc univariate data=mr normal plot;
var resid; histogram resid;
run;

*1.2 data transform;
data andy2; set andy;
sedadd=sed+1;
sedlog = log(sedadd);
sedsqrt = sqrt(sed);
run;
proc print data=andy2;
run;

*1.3 check normality of residuals log;
proc mixed data=andy2;
class scen;
model sedlog=scen/ outp=mrlog;
run;

proc univariate data=mrlog normal plot;
var resid; histogram resid;
run;

*1.4 check normality of residuals sqrt;
proc mixed data=andy2;
class scen;
model sedsqrt=scen/ outp=mrsqrt;
run;

proc univariate data=mrsqrt normal plot;
var resid; histogram resid;
run;

*Based on W-statistic, select log transform;
```

List A-1 (cont'd)

```
*1.5 Checking equality of variances;  
proc sort data=mrlog; by scen; run;  
proc univariate data=mrlog normal plot;  
by scen;  
var resid;  
run;
```

```
*1.6 Levene's test;  
data mrlog; set mrlog;  
res2=resid*resid;  
absres=abs(resid);  
proc mixed data=mrlog;  
class scen;
```

```
model res2=scen;  
run;
```

```
*2.1 Model assuming equal variances;  
proc mixed data=andy2;  
class scen;  
model sedlog=scen;  
run;
```

```
*2.2 Model assuming unequal variances - this model is selected based on lower  
AIC and BIC;  
proc mixed data=andy2;  
class scen;  
model sedlog=scen/ddfm=satterthwaite;  
repeated /group=scen;  
run;
```

```
*3.1 Final model;  
proc mixed data=andy2;  
class scen;  
model sedlog=scen/ddfm=satterthwaite;  
repeated /group=scen;  
lsmeans scen/pdiff;  
run;
```

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