THE FUNCTIONAL ASSESSMENT OF WETLANDS

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ABSTRACT

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Wetlands play a diverse and important role in the ecosystem. They provide numerous environmental, economic, cultural, recreational, aesthetic and ecological benefits to society. Meanwhile, wetlands are lost at an alarming rate due to human actions such as deforestation, expansion of agricultural land, pollution, and climate change. Quantifying wetland functionality is the first step to protect these valuable and biologically diverse ecosystems. However, current functional assessment techniques only provide a general overview of wetland functions in large and diverse watersheds. In addition, due to the qualitative nature of these techniques, they cannot be used to develop future management and restoration plans, which require solid understanding of hydrological and water quality characteristics of the wetlands. The goal of this research is to address some of these limitations by examining the impacts of wetland size, depth, and placement on flow and sediment transport at subbasin and watershed scales. The Soil and Water Assessment Tool (SWAT), a physically-based watershed model, was used along with the System for Urban Stormwater Treatment and Analysis Integration (SUSTAIN) to examine flow and sediment transport in two watersheds in Michigan, the Shiawassee Watershed (southeastern Michigan) and the River Raisin Watershed (southeastern Michigan and northeastern Ohio). Both watersheds were selected because they have experienced significant conversion of land from wetlands to agriculture since European settlement. Wetland area was found to be more influential in controlling streamflow rate than wetland depth. Meanwhile, wetland implementation has limited impacts of daily peak flow rates and frequency of peak flow events at the watershed outlet. In general, rate of streamflow reduction is higher than sediment reduction at the subbasin level but more comparable at the watershed level. These results reveal the importance of wetland size, depth and placement as part of restoration efforts. This study introduces an alternative approach to the functional assessment of wetlands that is more accurate and quantitative.

This doctoral thesis is dedicated to my beloved family. To my parents for all their hard work, sacrifices, support, advices, and for always giving me unconditional love. “Los amo y estoy muy orgulloso de ustedes”

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KEY TO ABBREVIATIONS

ARS – Agricultural Research Service

BASINS – Better Assessment Science Integrating point and Nonpoint Sources

BMP – Best Management Practice

Canmx – Maximum Canopy Storage (mm)

CMSR – Completely Mixed Stirred Reactor

CH_N1 – Manning's "n" value for the tributary channel

Ch_N2 – Manning's "n" value for the main channel

CH_K2 – Channel effective hydraulic conductivity (mm/hr)

CN2 – Initial SCS runoff curve number for moisture condition II

CO2 – Carbon Dioxide (ppm)

CWA – Clean Water Act

DEM – Digital Elevation Model

EFTMIX – Tillage mixing efficiency

Epco – Plant uptake compensation factor

Esco – Soil evaporation compensation factor

ET – evapotranspiration (mm)

GIS – Geographic Information Systems

Gw_Delay – Groundwater delay time (days)

HRU – Hydrologic Response Unit

HSPF – Hydrologic Simulation Program-FORTRAN

HUC – Hydrologic Unit Code

Kg – Kilogram

Km2 – Square Kilometer

MUSLE – Modified Universal Soil Loss Equation

NCDC – National Climatic Data Center

NLCD – National Land Cover Database

NHD – National Hydrography Data

NPS – Non-Point Source

NRCS – Natural Resources Conservation Service

NSE – Nash-Sutcliffe Efficiency

PLOAD – GIS Pollutant Load

PRISM – Parameter-elevation Regressions on Independent Slopes Model

R2 – Coefficient of determination

Rchrg_Dp – Deep aquifer percolation fraction

RCM – Regional Climate Model

RMSE – Root Mean Square Error

RUSLE – Revised Universal Soil Loss Equation

SCS – Soil Conservation Service

SDSM – Statistical Downscaling Model

Sol_Alb – moist soil albedo

SOL_AWC – available water capacity of the soil layer (mm H2O/mm soil)

Sol_K – saturated hydraulic conductivity (mm/hr)

Spcon – Linear re-entrainment parameter for channel sediment routing

Spexp – Exponential re-entrainment parameter for channel sediment routing

SRES – Special Report on Emissions Scenarios

SSURGO – Soil Survey Geographic

STATSGO – State Soil Geographic Database

STORET - STOrage and RETrieval

SUSTAIN – System for Urban Stormwater Treatment and Analysis INtegration

SURLAG – Surface runoff lag coefficient

SWAT – Soil and Water Assessment Tool

TCLW – Tuttle Creek Lake Watershed

Timp – Snow pack temperature lag factor

TKN – Total Kjeldahl Nitrogen

TN – Total nitrogen

TP – Total phosphorus

TSS – Total Suspended Solids

USDA – United States Department of Agriculture

USEPA – United States Environmental Protection Agency

USGS – United States Geological Survey

USLE – Universal Soil Loss Equation

USLE_C – Universal Soil Loss Equation cover factor

USLE_P – Universal Soil Loss Equation support practice factor

WetSpa – Water and Energy Transfer between Soil, Plants and Atmosphere

WSS – Web Soil Survey

1 INTRODUCTION

Wetlands provide important hydrologic, geochemical, and biological functions in a watershed (De Laney, 1995; Hart, 1995; NRC, 1995). In addition, wetland systems directly support millions of people throughout the world by providing fertile soils for agriculture production, food, shelter for wildlife, trees for timber and fuel, recreation areas, and many other benefits for humans.

Meanwhile, wetlands have been lost at an alarming rate. The impacts of lost wetlands can affect health and productivity of water bodies such as streams, lakes and rivers downstream (Meyer et al., 2003). Destruction and degradation of wetlands can also reduce groundwater levels. For example, it has been estimated that groundwater resources would decline by 45% if 80% of Florida's cypress swamps were drained to accommodate other land uses (Ewel, 1990).

In order to protect valuable wetland ecosystems, first we need to better understand their functions and roles, especially within large and diverse areas. However, this is a challenging task due to the complexity of wetlands' pollutant removal and transport processes at the watershed scale. Meanwhile, current functional wetland assessment techniques only provide rough estimations, which in most cases are site specific and qualitative.

The overall goal of this project is to quantify some water quality and quantity benefits of wetland implementation scenarios in large and diverse watersheds. The specific objectives of this project are as follows:

1) Evaluate the impacts of wetland depth, area, and wetland placement in the watershed on streamflow and peak flow reduction at the watershed scale.
2) Determine the role of wetland placement in watershed sediment dynamics by considering the distance to the outlet and stream order concept

3) Identify the most appropriate sites for wetland implementation by considering the environmental and economic aspects of restoration scenarios

2 LITERATURE REVIEW

2.1 WETLANDS OVERVIEW

Wetlands are diverse environments which are defined by several factors; these include the soil type, hydrology, topography, climate and vegetation. Natural and human landscape disturbance activities could potentially negatively affect the above mentioned factors. Abiotic and biotic characteristics of a wetland are controlled by the hydrology of that wetland (National Research Council, 1995). Wetlands have a scientific and legal definition. A scientific definition was provided by Cowardin et al., 1979 who defined wetlands as “lands where saturation with water is the dominant factor determining the nature of soil development and the types of plant and animal communities living in the soil and on its surface.” A legal definition of wetlands under the Clean Water Act is: "those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas" (U.S. EPA, 1995). Wetlands can be situated in saline, brackish and freshwater environments along coastlines, within estuaries, rivers and lakes, on slopes where ground water breaks out as a spring or seeps, in abandoned ditches or stream channels, as well as other locations.

2.2 TYPE OF WETLANDS

We have several major types of wetlands in the United States due to the unique and diverse climate, land features, vegetation and land management activities. A very important characteristic of wetland types is the hydrologic conditions of wetlands. This factor will determine the wetland's geomorphology, habitat quality, water quality, and biodiversity according to: Mitsch and Gosselink, 1993; Kadlec and Knight, 1996.The EPA described the most common types of wetlands in the United States and organized them into four general categories: marshes, swamps, bogs and fens (U.S. EPA, 2001).

The following is a description of each of these types (U.S EPA, 2001; Mitsch and Gosselink, 2000; Mitsch and Gosselink, 1993):

1. **Marshes** are periodically saturated, flooded, or ponded with water and characterized by herbaceous (non-woody) vegetation adapted to wet soil conditions. Marshes are further characterized as tidal marshes and non-tidal marshes. Tidal (coastal) marshes occur along coastlines and are influenced by tides and often by freshwater from runoff, rivers, or ground water. Non-tidal (inland) marshes are dominated by herbaceous plants and frequently occur in poorly drained depressions, floodplains, and shallow water areas along the edges of lakes and rivers. Major regions of the United States that support inland marshes include the Great Lakes coastal marshes, the prairie pothole region, and the Florida Everglades.

    i. Freshwater marshes are characterized by periodic or permanent shallow water, little or no peat deposition, and mineral soils.

ii. Wet meadows commonly occur in poorly drained areas such as shallow lake basins, low-lying depressions, and the land between shallow marshes and upland areas.

iii. Wet prairies are similar to wet meadows but remain saturated longer.

iv. Prairie potholes develop when snowmelt and rain fill the pockmarks left on the landscape by glaciers. They provide excellent habit and breading grounds for migratory birds.

v. Playas are small basins that collect rainfall and runoff from the surrounding land.

vi. Vernal pools have either bedrock or a hard clay layer in the soil that helps keep water in the pool.

2. **Swamps** are fed primarily by surface water inputs and are dominated by trees and shrubs. Swamps occur in either freshwater or saltwater floodplains. They are characterized by very wet soils during the growing season and standing water during certain times of the year. Swamps are classified as forested, shrub, or mangrove.

    i. Forested swamps are found in broad floodplains of the northeast, southeast, and south-central United States and receive floodwater from nearby rivers and streams.

   ii. Shrub swamps are similar to forested swamps except that shrubby species like buttonbush and swamp rose dominate.

   iii. Mangrove swamps are coastal wetlands characterized by salt-tolerant trees, shrubs, and other plants growing in brackish to saline tidal waters.

3. **Bogs** are freshwater wetlands characterized by spongy peat deposits, a growth of evergreen trees and shrubs, and a floor covered by a thick carpet of sphagnum moss.
4. **Fens** are ground water-fed peat forming wetlands covered by grasses, sedges, reeds, and wildflowers.

2.3 FUNCTIONS, SERVICES AND VALUES OF WETLANDS

Wetlands are valuable, limited, dynamic and unique ecological habitats in the world. Wetland functions are properties that a wetland naturally provides, services are properties that are valuable to humans, and values are attributes that humans assigned to wetland services. These ecological areas have multiple functions, services and values that are defined by the location, agricultural expansion, urban sprawl, environmental laws and regulations. Some of these factors (i.e. industrialization, agricultural conversion and timber harvest) have significantly contributed to the loss and degradation of wetland ecosystems (Mitch and Gosselink, 1993). It is important to notice that not all wetlands will perform all functions and services nor do they have the same values. Wetland properties are determined by several factors including: climate and ecological conditions, location and size of the wetland, type of wetland and water availability among others.

Unfortunately "while wetland functions are natural processes of wetlands that continue regardless of their perceived value to humans, the value people place on those functions in many cases is the primary factor determining whether a wetland remains intact or is converted for some other use" (National Audubon Society, 1993).

2.3.1 Wetland Functions

Wetland functions are defined by Novitzki et al., 1997 as a process or series of processes that take place within a wetland. Wetland functions are ecological processes which include but are not limited to surface water storage, subsurface water storage, ground water recharge, sediment and other particulate retention, shoreline stabilization, stream shading, and other environmental functions (Novitzki, 1979; Luecke, 1993; Tiner, 1998; Ramsar, 2004; Keddy, 2000; Mitsch and Gosselink, 2000; Hassan et al., 2005; Verhoeven and Setter, 2010). In addition, wetland functions include transformation of nutrients, growth of living matter, and diversity of wetland plant. All of these values are for the wetland itself, for surrounding ecosystems, and for people (USGS, 2004). In many locations, such as the United Kingdom, South Africa, Canada and the United States, wetlands are part of conservation efforts due to the ecological, environmental, nutritional and recreational values (Keddy, 2009).

2.3.2 Wetland Services

Wetland ecosystems provide a diversity of services for the well-being of humans, as well for wildlife species. Some of these services are: food, fiber, biomass production, wildlife habitat for terrestrial and avian species, retention, removal and transformation of nutrients, flood control, flood water storage and storm buffering, biodiversity, maintenance or improvement of water quality, carbon sequestration, reduction of soil erosion and sedimentation, discharge and recharge of ground water. Several studies have been conducted showing the capacity of wetlands to abate flooding, improve water quality, and support biodiversity (Neely and Baker, 1989; Crumpton et al., 1993; Richardson and Craft, 1993; Bedford, 1999; Keddy, 2000).

Zedler (2003) attributed the decline of three ecosystem services (flood abatement, water quality improvement, and biodiversity support) in the Upper Midwestern area to the drainage of approximately 60% of the regions historical wetlands for agriculture.

2.3.3 Wetland Values

The value of a wetland lies in the benefits that it provides to the environment or to people, something that is not easily measured Novitzki et al., 1997. Wetlands can have ecological, social, or economic values. These values include multiple categories such as population (animals harvested for pelts, waterfowl and other birds, fish and shellfish, timber and other vegetation harvest, and endangered species), ecosystem (flood mitigation, storm abatement, aquifer recharge, water quality improvement, aesthetics, and subsistence use) and the biosphere (nitrogen , sulfur, carbon and phosphorus cycles) as described by Mitsch and Gosselink, 2000. Many studies have discussed wetland values in an extensive matter (e.g. Wharton, 1970; Gosselink et al., 1974; Mitsch, 1977; Costanza et al., 1997; Costanza et al., 1989; Turner, 1991; Barbier, 1994; Gren et al., 1994; Gren, 1994; Bell, 1997; Mitsch and Gosselink, 2000). Values that are well recognized by humans are productivity of downstream fisheries, recreational observation and hunting of wildlife, reduced cost of water purification, production of valuable sources of food and fiber, reduced damage due to flooding, and erosion reduction which is directly linked to maintenance.

Barbier (1989; 1993; 1994) classified the total economic value for tropical wetlands and divided it between direct use value, indirect use value and existence value. Direct use values are forest resources, wildlife resources, fisheries, forage resources, agricultural resources and water supply. Indirect use values are nutrient retention, flood and flow control, storm protection, ground water recharge, external ecosystem support and micro-climatic stabilization. Existence values are biodiversity, culture and bequest values. This study shows how important wetland values are and how extensive these natural system characteristics are.

2.4 DESCRIPTION OF FUNCTIONS, SERVICES AND VALUES OF WETLANDS

2.4.1 Water Quality

Extensive research has shown that wetlands play an important role in improving water quality (Wolverton et al., 1983; Neely and Baker, 1989; Reed, 1993; Larson et al., 1989; Crumpton et al., 1993; Richardson and Craft, 1993; Bedford, 1999; Keddy, 2000). Wetland vegetation will substantially slow the flow of runoff water causing deposition of mineral and organic particles with adsorbed nutrients (Nitrogen and Phosphorus) (Carter, 1996). The ability of wetlands to improve water quality has been extensively studied and these systems have been shown to lower concentrations of water contaminants including Nitrogen, Phosphorus, suspended solids, biochemical oxygen demand, trace metals, trace organics, and pathogens (Mitsch, 1994; Hammer 1992, 1993).

Due to the effectiveness of wetlands on nutrients filtration and transformation, artificial wetlands have been constructed for water quality restoration (e.g., Hammer 1992, Mitsch et al., 2001). According to these studies, natural and constructed wetlands performing this function will help improve local water quality of streams and other watercourses through the reduction of nutrients and sediments loads.

Studies have shown that wetlands act as a natural filter that can improve water quality (Kadlec and Knight, 1996). Wetland capacity to improve water quality has been shown in several studies. The capacity of wetlands to reduce nutrients (N and P) allows the reduction of eutrophication (Mitsch et al., 2001) in addition to the capacity to store large amounts of sediments (Day et al., 2007).

Johnston et al. (1990) developed a method to evaluate the cumulative effect of wetlands on stream water quality and quantity. A stepwise multiple regression analysis was used to evaluate relationships between stream water quality variables and wetlands. The results showed that the proximity of wetlands in relationship with the sampling station was associated with a decrease of several parameters including: inorganic suspended sediments, Nitrates (NO_3-), flow-weighted ammonium (NH_4), flow-weighed total P (TP). This study clearly shows the efficiency of the wetlands' capacity to improve water quality. The method used describes the efficiency of wetlands removing suspended sediment solids, TP and Ammonia during high flow periods and the efficiency of wetlands at removing Nitrates during low flow periods.

2.4.2 Water Quantity

Water is a resource that every known form of life on earth depends on and it affects every aspect of our lives. While water is abundant in many regions, approximately one billion people around the world don't have clean drinking water, and 2.6 billion still lack basic sanitation. In the United States we can observe water as an abundant resource in places like Michigan, but at the same time we can observe water as a limited resource in places like Northern California.

Wetlands play an important role in the hydrologic cycle and in some regions a vital role on water quantity. Wetlands receive, store and release water in numerous ways (e.g. surface water, ground water, and plant intake). In some regions wetlands will maintain streamflow during dry periods providing a water source for wildlife. Wetlands possess multiple hydrological functions related to gross water balance, ground water recharge, base flow and low flow, flood response and river variability. All of these functions will vary depending on the type of wetland, location, vegetation, and size, in addition to other important wetland characteristics.

Wetlands have the capacity to catch, retain, filter and release runoff water generated from heavy rainfall or snowmelt events allowing an increase in ground water infiltration which will help to reduce river flow downstream and agricultural runoff (Luecke 1993; Comin et al., 1997; Keddy, 2000). Wetlands will provide short term and long term water storage functions and assist with the reduction of downstream flood peaks (Ramsar, 2004).

2.4.3 Nutrient Transformation

Nitrogen (N) and phosphorus (P) are the primary macro-nutrients that enrich streams, lakes and rivers. Phosphorus is the main nutrient controlling productivity and the primary cause of the excess algal biomass in surface waters (Correll, 1998). The directly available forms of N and P are mostly inorganic (NO_3^- and NH_4^+). Total N and total P include soluble fractions, particulate and dissolved organic fractions. Total N and total P concentrations are used to predict algal biomass in lakes and reservoirs. Nutrient concentrations can differ from stream to stream because of differences in land use, geology, streamflow, point sources and other factors in the drainage basin. Wetlands have been shown to be very efficient in removal of nutrients from agricultural runoff (Comin et al., 2001).

Current studies show that wetlands have the capacity to significantly reduce nutrients, sediments and other pollutant concentrations produced from agricultural runoff under different environmental conditions (Kadlec, 1993; Mander et al., 2000; Trepel and Palmeri, 2002; Jordan et al., 2003; Archeimer et al., 2004; Skagen et al., 2008). Kelly and Harwell (1985) nutrients supplied to wetland areas from discharge or runoff are present in the soluble and particulate forms (i.e. dissolved nitrogen (N) will be introduced as nitrate (NO_3^-), ammonium (NH_4^+) or soluble organic forms and dissolved phosphate (PO_4^{-3}) or soluble organic Phosphorus).

Wetland vegetation will remove pollutants by slowing runoff and through pollutant plant uptake. Jordan et al. (2003) studied nutrient and sediment reduction capacity of constructed wetlands and found a reduction in non-point source pollution (approximately 25% of Ammonium and 52% of Nitrate were significantly removed from the studied area). A study conducted in the Houghton Lake wetland system in Michigan, shows the capacity of this wetland system to remove up to 90% of the Phosphorus load (Kadlec, 1993). Several studies have found that forested wetlands near rivers and streams are important for nutrient retention and sedimentation during flood events (Whigham et al., 1988; Yarbro et al., 1984; Simpson et al., 1983; Peterjohn and Correll, 1982).

Studies show that a combination between buffer strips and wetland areas will effectively control the nutrient fluxes (Vought and Lacoursiere, 1998). In addition, wetlands can act as filters removing particulate material, as sinks accumulating nutrients and also as transformers converting nutrients to different forms (Richardson, 1989).

Healy and Cawley (2002) studied the reduction capacity of a constructed surface-flow wetland in nutrients P and N from a waste treatment system in Ireland. They found an average percentage reduction of approximately 51% for total N and 13% for total P. In addition to the reduction in nutrients, a reduction of approximately 87% of suspended sediments was observed along with a 49% reduction in biological oxygen demand (BOD). This study clearly shows the nutrient reduction capacity of wetlands.

Several recent studies show the restored wetland nutrient (N and P) removal capacity (Fleischer et al., 1994; Reinelt and Horner, 1995; Raisin et al., 1997, Hunt et al., 1999, Kovacic et al., 2000; Braskerud 2002; Jordan et al., 2003). During each of these studies a reduction in nutrients was found, but the observations are different due to environmental and time factors (i.e. temporal variability of water inflow, timing required to filter or transform nutrients, hydraulic loading rates and hydraulic efficiency).

Borin et al. (2001) demonstrated that wetlands reduced Nitrates (NO_3^-) by 95% and total dissolved solids by 30% in runoff from cropland. Borin and Tocchetto (2007) found a nitrogen removal efficiency of approximately 90% during a 5 year study conducted on water and nitrogen balance for a constructed surface flow wetland treating agricultural drainage waters.

Weller et al. (1996) studied the role of wetlands in reducing P loading in surface waters. This study showed the calculation of different variables that summarized a variety of characteristics of wetlands using a geographic information system (GIS) and regression analysis to measure Phosphorus loading. The wetland variables that were developed for this study were: quantity of wetland (area number and perimeter), wetland type, land use and the relationship between wetlands and streams. Significant results were found in this study suggesting that a hectare of riparian wetland may be many times more important in reducing Phosphorus than an agricultural hectare is in producing Phosphorus (Weller et al., 1996). This study shows the capacity of wetlands reducing P loading from surface waters which is beneficial to improve and protect surface water quality.

Recent researches demonstrate the capacity of constructed or restored wetlands to remove nutrients and sediments from non-point source pollution (i.e. Mitsch, 1994; Raisin and Mitchell, 1995; Whigham, 1995; Jordan et al., 1999). Restoration of wetlands in agricultural watersheds will provide wildlife habitat as well as improve water quality (Whigham, 1995). This study shows multiple environmental benefits of wetland restoration. Wetland restoration could have a domino effect depending on the location (i.e. water quality and wildlife could have an impact in agricultural production and/or recreation activities (hunting) which could have an economical effect). Wetland conservation and restoration could improve if all of these valuable environmental benefits are promoted, recognized and better understood by the general public.

Crumpton et al. (1993) demonstrated the capacity of wetlands to trap nutrients upstream and downstream. Upstream wetlands trap few nutrients, while down-stream wetlands can potentially remove up to 80% of inflowing Nitrates.

2.4.4 Flood Water Storage

Numerous studies show that wetlands can decrease flooding, improve water quality, and support biodiversity (Campbell and Johnson, 1975; Novitski, 1978; Thomas and Hanson, 1981; Neely and Baker, 1989; Larson et al., 1989; Crumpton et al., 1993; Demissie and Kahn, 1993; Richardson and Craft, 1993; Bedford, 1999; Keddy, 2000). Wetlands that are located close to rivers, streams or adjacent water bodies can slow down runoff coming from storm water or snow melt and to a certain degree contribute to the protection of populations in floodplains areas. Wetland vegetation (e.g. trees, shrubs, and other wetland plants) will slow down runoff water by slowing the flow which will allow ground water resources to recharge and sediments to be trapped (Holden et al., 2007). The capacity for wetlands to decrease flooding is limited by several factors such as: water level fluctuations, plant community and density, habitat elements, ground water hydrology among other physical factors.

Wetlands will provide short term and long term water storage functions and assist with the reduction of downstream flood peaks (Ramsar 2004; Mitsch, 1992; Potter, 1994; Hey et al., 2002). Wetlands will moderate or prevent floods along a watershed area according to their distribution and size. Ewel (1997) showed that maintaining integrity of wetlands by leaving vegetation, soils, and natural water regimes intact could potentially reduce the severity and duration of flooding along rivers. Investigations have shown that small wetlands could potentially reduce and delay flood peaks serving as storage areas, while larger wetlands could potentially reduce peak flow levels (Potter, 1994; Hey et al., 2002). Novitzki (1978) studied the relationship between wetlands and flooding at a watershed scale in Wisconsin and found that watershed areas with approximately 40 % coverage by lakes and wetlands had significantly reduced flood flows (80 % less flood flow than similar watersheds having no or few lakes and wetlands).

Wu and Johnson (2008) performed a hydrologic comparison between a forested and a wetland/lake dominated watershed in northern Michigan using SWAT as the watershed modeling tool. The results suggest an important storage function provided by wetlands and lakes, in which they increase the ability of a watershed to moderate extreme flows and gradually release water as baseflow.

Acreman and Holden (2013) described how wetlands affect floods from a hydrological perspective of two wetland types; upland rain-fed wetlands and floodplain wetlands. The study explained how multiple factors such as: landscape location and configuration, soil characteristics, topography, soil moisture status and management will the ability of these wetlands to provide flood reduction services.

2.4.5 Sediment Retention

Sediment particles are often vehicles for transporting pollutants such as nutrients (e.g. nitrogen and phosphorus), pesticides and heavy metals. The accumulation of sediments at the bottom of water bodies (e.g. reservoirs) could potentially have an impact on fish and aquatic life, and could also reduce storage capacity in water reservoirs (Soler-Lopez et al., 2001a; 2001b; 2001c).

One of the principal external dynamic agents of sedimentations is that water serves as a source of pollutant transport. The detachment of particles in the erosion process occurs through the kinetic energy of raindrop impact or by the forces generated by the flowing water (Vanoni, 1997). Sediments are detached particles carried by rain water into streams, lakes, rivers and bays. Sedimentation problems are observed in streams, lakes and other important water bodies used by humans and wildlife. Some of the problems associated with sediment transport and deposition are: movement of soil particles, loss of soil fertility, reduction of sun light penetration through the water column, reduction in the reservoirs water storage capacity and reduction of dissolved oxygen concentration. Sediments can also carry concentrations of pollutants that contaminate waterways, including nutrients such as phosphorus and nitrogen which promote eutrophication in surface waters.

The role wetlands play in trapping sediments and preventing them from reaching surface water bodies is important. When positioned in stream networks, wetlands also mitigate hydraulically driven variables including sediment, nutrients, temperature, and disturbance

(Richards et al., 1996). Wetlands play a unique and important role in improving environmental quality as discussed previously. However, according to Kuenzler, 1990 wetlands should only be used to remove sediments and other agricultural pollutants after agricultural best management practices have been implemented.

Many studies show the capacity of wetlands to trap sediments as a water quality benefit. Sediment input from agricultural fields has potential to completely fill wetlands and shorten their effective life-span. Thus, the value placed on wetlands to trap sediments is in conflict with maximizing the effective topographic life of wetlands (Gleason and Euliss, 1998).

2.4.6 Carbon Sequestration

Wetlands connected to rivers and slope locations are very productive. Their interaction with streams make them significant sources of dissolved and particulate organic Carbon for aquatic ecosystems and biogeochemical processes in downstream aquatic habitats (Sedell et al., 1989 and Vannote et al., 1980).

It has been found that forested wetlands can offer a number of options for reducing Green Houses Gases (GHGs), particularly carbon (C) emissions. They aid in the removal of CO_2 from the atmosphere into carbon pools (Cui et al., 2005). Several studies (Nieveen et al., 1998; Schreader et al., 1998; Waddington and Roulet, 2000; Aurela et al., 2001; Lafleur et al., 2001, 2003; Bubier et al., 2004) related to forested wetlands have been developed to observe the interactions between abiotic and biotic environmental factors and processes (e.g. methane production and transport from wetland to the atmosphere). In addition, in wetland water level is the major factor controlling carbon allocation, organic matter decomposition and C fluxes in wetland (Kettunen et al., 1999; Weltzin et al., 2000).

2.4.7 Wildlife

Many studies have been done showing the benefits provided to wildlife by wetlands (Shaw et al., 1956; Cowardin et al., 1979; Kantrud et al., 1989; Yerkes 2000). Wetlands are a very significant ecosystem for wildlife. They provide shelter, food, and fish habitat, in addition to other important factors (e.g. water and diversity). For example, wetlands and surrounding upland areas provide breeding ducks and other waterfowl with the diverse habitat they need for feeding, breeding, and nesting (Batt et al., 1989; Kantrud et al., 1989; Yerkes, 2000). These wetlands can also have a high level of endemism, extensive plant zonation, and high biodiversity. The high biological productivity of wetlands among other factors has produced a rich biota associated to these ecological sties (Gibbs, 1995).

Annual flooding in low-gradient rivers and their adjacent flood plain wetlands constitute a significant subsidy to physical habitat, vegetative communities, and populations of aquatic organisms (Benke et al., 2000). In addition, seasonal water exchanges between lakes and coastal wetlands and tidal fluxes between salt marshes, estuaries, and shallow marine areas create and maintain productive habitats for a variety of plants and animals (Stevens et al., 2006). Most freshwater and many marine aquatic organisms (birds, fish, insects) utilize wetland environments at some stage of their development (Mitsch and Gosselink, 1993).

2.5 WETLANDS AND AGRICULTURE

Wetland issues (e.g. conservation, regulation, policies, and degradation) have been an important part of agricultural and environmental policy debates at Federal, State and Local levels for more than 25 years, when the Food Security Act regulated/protected wetlands. This has not stopped the significant loss of these unique and valuable ecosystems in the United States (Dahl, 1990, 2007). Society has increased the value it places on the services provided by wetlands (e.g. water quality improvement, flood control, wildlife habitat, and recreation), however owners of wetlands often don not gain a profit directly from these services because the benefit is freely enjoyed by many (Heimlich et al., 1998). These factors and the fact that wetlands are fertile and productive soils for agricultural production trigger a land use change impact (wetland to agricultural land) in many locations of the United States (e.g. Midwest Region: Michigan, Wisconsin, Indiana).

Agriculture is the production of food and goods through farming. Land use changes are required in order to perform agricultural activities in some locations (e.g. tiling for drainage, creation of irrigation or drainage ditches), which has been one of the biggest causes of the degradation of wetlands in the USA (Mitsch and Gosselink, 1993). Due to the high fertility of wetland soils (e.g. rich content of organic matter) farmers converted over 28 million acres of wetlands into high-quality cropland in nine Midwestern States since settlement (Heimlich et al., 1998; Heimlich and Gadsby, 1994). Over 50 % of the area of depressional wetlands, riparian zones, lake littoral zones and floodplains has been lost, mainly due to land use change as a conversion to increase agricultural land use, in North America, Europe and Australia (Millennium Ecosystem Assessment, 2005).

This conversion has not stopped in today's society, where land use changes are dynamic and agriculture and urbanization seem to be increasing every day. Increasing population and its associated increase in demand for food and economic development will continue to create the pressure to convert wetlands for farm use over the next several decades (Wood and van Halsema, 2008). Maintaining and improving the quality of wetlands is a very important goal because wetlands provide multiple services to society, wildlife and the environment as it has been discussed in previous sections of this document.

2.5.1 Crop Production

Wetlands are one of the primary sources of crop production, fiber and proteins in some locations in the world (e.g. Africa and South Brazil). The value given to wetlands for agricultural activities could be observed in any location of the world at any given time. For example, in the United States during dry periods farmers can farm wetland areas within their farms without changing the hydrology. In other locations farming activities are major economic pursuits in and around wetland areas due to the high agricultural production value of these ecosystems. An example of this would be areas where crops such as rice, maize, and various vegetables and fruit are cultivated (Omari, 1993).

The use of chemicals and changes in crop production could potentially affect: soil erosion, sedimentation in streams and reservoirs, pollution of surface and ground water waters (NRC, 1989). Wetlands act as filters that can improve water quality (Kadlec and Knight, 1996) and store sediments (Day et al., 2007) and will potentially have a positive impact in crop production areas.

2.5.2 Livestock Grazing

Another agricultural activity that wetlands provide is livestock grazing areas. In locations where seasonal wetlands are part of the ecosystem livestock grazing can continue during dry seasons facilitated by the large source of biomass associated with these productive areas. These areas could be grazed directly or used for hay production to feed livestock.

2.6 WETLANDS AND LANDUSE CHANGE

Land use/cover changes can alter watershed properties, such as: water infiltration rate, water velocity, peak flow fluctuations, water storage capacity, and vegetation, as well other important hydrological factors. This will affect sediment loads which could potentially have an impact on downstream wetland ecosystems.

Chen et al. (2006) modeled the impacts of land use and land cover change on sediment loads in wetlands in the Pouyang Lake Basin in Asia. GIS and Remote Sensing technologies were used for the prediction of annual soil loss in the basin area in addition to a mathematical model to study the relationship between land use/cover changes and sediment loads. They found that the land use/cover change in the watershed reduced sediment sources and led to a reduction in the suspended sediment concentration loads entering the wetland. Significant land use alterations including the decrease of wetland by 43.55 percent for the study time period was observed. The authors identified population growth, economy development and urbanization as the social problems faced by the study area as well as the driving forces that led land use changes in the studied watershed. Wetlands are also very sensitive ecosystems that are subject to much stress from human activities (Bergh, 2001).

Previous studies related to wetland ecosystems have found a significant impact of the land use/cover change on hydrological and fluvial processes. However, there is limited information available regarding the effect of land use/cover change on water quality, especially sediment load (Chen et al., 2006). Wetland ecosystems are susceptible to external influences because of functional relations between hydrological characteristics in the ecosystem and the surrounding area (Barendregt et al., 1995; De Mars and Garritsen, 1997).

2.7 WETLANDS CONSERVATION PROGRAMS

The hydrologic, geochemical and biological functions of wetlands are very important in a watershed (De Laney, 1995; Hart, 1995; NRC, 1995). For that reason federal and state governments, in addition to local and regional groups created programs for wetland conservation. The United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS) has a conservation program to conserve, enhance and create wetlands in the United States (USDA-NRCS, 2007). The USDA Farm Service Agency also has a conservation program available to enhance wetlands in the United States.

Implementation of the USDA, CRP and WRP has resulted in the restoration of approximately 2,200,000 ha (5,436,200 acres) of wetland and grassland habitats in the Prairie Pothole Region (Gleason et al., 2008).

2.7.1 Wetland Restoration Program (WRP)

The Wetland Restoration Program (WRP) is a voluntary program which offers landowners the opportunity to protect, restore, and enhance wetlands on their property (USDA-NRCS, 2007). The WRP provides technical and financial assistance to enable qualified landowners to address wetland, wildlife habitat, soil, water, and related natural resource concerns on private lands. The WRP program focuses on: enrolling marginal lands that have a history of crop failures or low yields, restoring and protecting wetland values on degraded wetlands, maximizing wildlife benefits, achieving cost-effective restoration with a priority on benefits to migratory birds, protecting and improving water quality, and reducing the impact of flood events.

In addition to the WRP, the USDA restores, enhances, and protects wetlands through other conservation programs such as: the Small Watershed/Watershed Rehabilitation Program, the

Wildlife Habitat Incentives Program (WHIP), and the Emergency Watershed Program (EWP). All of these conservation programs are promoted every year nationwide as part of the USDA's conservation strategy.

2.7.2 Conservation Reserve Program (CRP)

The Conservation Reserve Program (CRP) is a voluntary program which offers landowners the opportunity to protect, restore, and enhance wetlands on their property (USDA-NRCS, 2007). Under this program land is taken out of production and long-term, resource conserving cover vegetation is established to control soil erosion, improve water and air quality, and enhance wildlife habitat. The CRP Wetland Restoration Initiative is designed to restore the functions and values of wetland ecosystems that have been devoted to agricultural use. These wetlands prevent degradation of the wetland area, increase sediment trapping efficiencies, improve water quality, prevent erosion and provide vital habitat for waterfowl and other wildlife.

2.8 WETLANDS CONSERVATION PRACTICES

Currently conservation practices are well promoted by government agencies and local conservation groups (e.g. USDA-NRCS, US Fish and Wildlife Service, EPA, and Conservation Districts) and accepted by landowners (e.g. dairy, fruits and cash crops farmers). Federal, state and local agencies with a conservation vision will promote conservation and best management practices within their target region. All of these conservation practices have something in common which is addressing resource concerns (e.g. water quality and quantity, air quality, soil erosion and plant deficiencies). These conservation efforts will assist landowners in improving air and water quality, controlling/reducing erosion and sedimentation, reducing nutrient and pesticide pollution; control/reduce impacts to stream ecology, morphology and habitat (USDA-

NRCS, 2006). Some of the conservation practices related to wetland ecosystems are discussed below.

2.8.1 Wetland Restoration

Wetland Restoration is defined as the rehabilitation of a degraded wetland or the re-establishment of a wetland so that soils, hydrology, vegetative community, and habitat are a close approximation of the original natural condition that existed prior to modification to a practical extent. The purpose of this practice is to restore wetland function, value, habitat, diversity, and capacity to a close approximation of the pre-disturbance condition by: restoring hydric soil, restoring hydrology and restoring native vegetation (USDA- NRCS, 2003).

2.8.2 Wetland Enhancement

Wetland Enhancement is defined as the rehabilitation or re-establishment of a degraded wetland, and/or the modification of an existing wetland, which augments specific site conditions for specific species or purposes, possibly at the expense of other functions and other species. The purpose of this practice is to provide specific wetland conditions to favor specific wetland functions and targeted species by: hydrologic enhancement, vegetative enhancement (including the removal of undesired species, and/or seeding or planting of desired species) (USDA-NRCS, 2003).

2.8.3 Wetland Creation

Wetland creation is defined as the creation of a wetland on a site that was historically non-wetland. The creation will provide wetland hydrology on a geomorphic setting that was not originally wetland. Wetland creations usually have the highest cost and management requirements. They are usually done for only one function such as providing wildlife habitat, educational opportunities, or improving the quality of water from nonpoint source runoff. The purpose of this practice is to create wetland functions and values (USDA-NRCS, 2003; USDA-NRCS, 2006).

2.8.4 Wetland Wildlife Habitat Management

Wetland wildlife habitat management refers to retaining, developing, or managing habitat for wetland wildlife. These practices evaluate several wildlife elements including: food (type and amount), cover (type, amount and quality), water (quality, quantity and accessibility) interspersion and distance to crops, grasses or legumes, shrubs and trees and open areas), and migration (routes, season of use and corridors). The main purpose of this practice is to maintain, develop, or improve habitat for waterfowl, shorebirds, fur-bearers, or other wetland associated fauna and flora (USDA-NRCS, 2003).

2.8.5 Wetland Construction

Constructed wetlands have been used for wastewater treatment for nearly 40 years and have become a widely accepted technology available to deal with both point and non-point sources of water pollution. They offer a land-intensive, low-energy, and low-operational-requirements alternative to conventional treatment systems, especially for small communities and remote locations (Vymazal et al., 1995).

2.9 WETLAND ASSESSMENTS

In today’s world, efficiency and cost are factors that are imperative to consider by private and government agencies, planners, regulators, and the general public prior to conducting field work in potential areas of concern, including wetlands. These factors have had a significant impact in the scientific community including an interest in the development of tools to facilitate environmental evaluations. Proper assessments of these potential areas aids in providing a clear decision planning process and ensures that the most valuable wetlands are preserved, restored or enhanced.

Several methods have been developed by the scientific community to better understand wetland functions (Cowardin et al., 1979; Brinson, 1993; Gilver and McInness, 1994; Hruby, 1999.; Adamus et al., 1987; Leibowitz et al., 1992; Kent, 2001; Amman and Stone, 1991; Hruby et al., 1995). These wetland assessment methods have been developed for multiple purposes (e.g. planning and development, conservation projects and restoration practices). The process of developing wetland assessment tools is not an easy task. However, these methods allow the scientific community to examine and protect sensitive wetland areas within a short period of time. These processes are very valuable for Federal and State Agencies that are currently working together to protect, restore, enhance or create wetlands as part of a national initiative (e.g. United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS), U.S. Army Corp, U.S. Fish and Wildlife Service, Michigan Department of Natural Resources and the Environment (MDNRE), among others).

Some examples of methods used to analyze wetlands are: the Cowardin method, the Washington State rating system, the Oregon method, the EPA synoptic method and the Evaluation of planned wetlands method (Cowardin et al., 1979; Brinson, 1993; Gilvear and McInnes 1994; Kent, 2001; Amman and Stone, 1991; Smith et al., 1995).

Hruby (1999) describes several of the above mentioned assessment methods that have been used to better understand wetland systems. These methods include characterization, classification/categorization, rating, assessment, and evaluation. Each data set obtained from these methods is analyzed with a “model” or “algorithms” which uses either logic or a mechanistic approach. Hruby also describes how the lack of model validation negatively affects the model data output quality. He recommends finding ways to validate models that are more time efficient and cost effective. He describes how multiple methods that generate a numeric assessment of performance or value of wetland functions depend on the mechanistic approach of constructing models. The author also describes how rapid assessment methods based on mechanistic models do not assess the rates or dynamics of ecological processes taking place in wetlands. These assessments provide a clear way of organizing our knowledge about wetland functions.

Other wetland assessment methods that are used by either private and/or government agencies, planners, regulators, or the general public include the Wetland Evaluation Technique (WET), the Environmental Monitoring Assessment Program (EMAP-Wetlands) and the Hydro-geomorphic approach (HGM). The WET method assigns values to specific functions of individual wetlands, was developed for the Federal Highway Administration, and has been used widely (e.g. highways, restore/create wetlands). The WET method is applicable to all types of wetlands in the contiguous United States (Adamus et al., 1991). EMAP-Wetlands, developed by the EPA, focuses on determining the ecological condition of a population of wetlands in a region. The HGM combines features from the aforementioned methods by measuring the functions of individual wetlands and also by comparing them to functions performed by other wetlands, this approach was developed by the U.S. Army Corps of Engineers for assessing wetland functions.

Wheeler and Shaw (2000) used an extensive amount of data from over 80 wetlands in Eastern England to develop a classification and assessment system called WETMECS. This system combines landscape situation (location and condition), water supply mechanism, hydro-topographical elements, acidity and fertility to classify the hydrological process on wetlands.

The Army Corp of Engineers has a program dedicated to ecosystem management and restoration research. Under this program, they have created multiple regional guidebooks for applying the HGM approach to assess functions in multiple regions of the US. The HGM approach is a method for developing functional indices and the protocols used to apply these indices to the assessment of wetland functions at a site-specific scale. This approach was initially designed by Brinson (1993) to be used in the context of the Clean Water Act, Section 404 Regulatory Program to analyze project alternatives, minimize impacts, assess unavoidable impacts, determine mitigation requirements, and monitor the success of compensatory mitigation. Smith et al. (1995) expanded the HGM concept including wetland functional assessments using Brinson's HGM classification system. Today the HGM has multiple potential uses including the design of wetland restoration projects and the management of wetlands. The hydro-geomorphic classification of wetlands is intended to lay a foundation for, and support ongoing efforts to develop methods for assessing the physical, chemical, and biological functions of wetlands. The strengths of this classification approach include its ability to clarify the relationship between hydrology and geomorphology and wetland function, and its open structure, which allows adaptation in various types of wetlands and geographic regions of the country.

In 1997 several federal agencies: the U.S. Army Corps of Engineers (USACE); U.S. Department of Transportation, Federal Highway Administration; USDA, NRCS; U.S. Department of Interior, U.S. Fish and Wildlife Service (USFWS), and the U.S. Environmental

Protection Agency (USEPA) agreed to use the HGM approach as a basis for wetland functional assessments. Their methodology was to use the HGM classification as a tool for measuring changes in the functions of wetland ecosystems due to impacts by proposed projects, restoration, creation, and/or enhancement (Brinson et al., 1997).

2.10 WETLAND COMPUTER MODELS

Hydrological computer models are a very powerful and commonly used tool for environmental studies. Hydrology-based computer models allow the scientific community to better understand complex environmental processes such as the hydrological cycle and its components. To represent these processes, modelers have adopted multiple approaches/systems such as empirical or theoretical models. The empirical models are functional relationships defined in terms of statistical data analysis of observed data, while theoretical models are classified as functional relationships defined from physical laws and relationships (Heatwole, 1998). Empirical models have the limitation of being site specific which makes the applicability limited or not applicable to other locations. On the other hand, theoretical models are adaptable to different locations if properly calibrated.

Several computer models have been developed to simulate water quality and quantity at a watershed scale (e.g. Soil and Water Assessment Tool (SWAT) by Arnold et al., 1998; Neitsch et al., 2002; Hydrological Simulation Program-Fortran (HSPF) by Donigian et al., 1995; Agricultural NonPoint Source Pollution Model (AGNPS) by Young et al., 1987,1989, 1994; Areal Nonpoint Source Watershed Environment Response Simulation (ANSWERS) by Beasley et al., 1980; ANSWERS-Continuous by Bouraoui and Dillaha, 1996; Bouraoui et al., 2002; Dynamic Watersheds Simulation Model (DWSM) by Borah et al., 2002 and the Precipitation-Runoff Modeling System (PRMS) by Leavesley et al., 1983; Leavesley and Stannard, 1995).

In addition, hydrologic computer models are commonly used to estimate the impact of Best Management Practices at a watershed scale. These models can potentially simulate the characteristics of Best Management Practices (BMPs) as close to its physical conditions and functional design (Renschler, 2007).

Borah and Bera (2003) reviewed some of the above mentioned models (AGNPS, AnnAGNPS, ANSWERS, ANSWERS-Continuous, DWSM, HSPF, KINEROS, PRMS, and SWAT) in their publication “Watershed-Scale Hydrologic and Non-point Source Pollution Models: Review of Mathematical Bases”. AnnAGNPS, HSPF and SWAT are described as long-term, continuous simulation models that contain all of the three major components (hydrology, sediment, and chemical), which are applicable to watershed-scale catchments (Arnold and Fohrer 2005; Neitsh et al., 2002; Neitsh et al., 2001; Arnold et al., 1998; Young et al., 1987, 1989; Donigian et al., 1995). AGNPS and DWSM are storm event simulation models that also contain all three major components.

SWAT is a river-basin or watershed-scale model for continuous simulations in predominantly agricultural watersheds, and HSPF is suitable for mixed agricultural and urban watershed conditions (Arnold et al., 1998; Neitsch et al., 2002). AGNPS is similar to SWAT and it is an event-based model simulating runoff, sediment, and transport of Nitrogen, Phosphorous, and chemical oxygen demand (COD) resulting from single rainfall events (Young et al., 1987, 1989). Conversely, we have the single-event models like DWSM, which is a potential model for agricultural and rural watersheds. It simulates distributed surface and subsurface storm water runoff, propagation of flood waves, upland soil and streambed erosion, sediment transport, and agrochemical transport in agricultural and rural watersheds during single rainfall events (Borah et al., 2002).

2.10.1 Computer Models Strengths and Deficiencies

The use of computer models in today's civilization is advanced, extensive and diverse. This makes finding more than one computer watershed model for addressing any practical problem much simpler. Computer models for watershed simulation allow researchers and scientists to evaluate and compare data outputs for validation using more than one available model, which provides a more broad approach. The variety in watershed computer models is one of the major strengths we have in today's technology.

The integration of soils, water, animal, plant, and air (SWAPA) data, ecosystem and ecology, environmental components, in addition to other factors with hydrology is an additional strength of watershed computer models. The wide range of computer models that are available and the applications of these models make them flexible and robust for the water quality study area.

Even though watershed computer models have become an important part of the scientific and research community there are many deficiencies that need to be addressed. Some of these deficiencies are the lack of user-friendliness, extensive data inputs, data management, the complexity of model calibration procedures and the integration of social, political, economic, and environmental systems within the models.

The inefficiency of computer models in the prediction of multiple water quality parameters has been a deficiency according to recent studies (Benaman and Shoemaker, 2004; Conan et al., 2003). Nasr et al. (2007) compared SWAT, HSPF and SHETRAN/GOPC for modeling phosphorus export from three catchments in Ireland. The three mathematical models: SWAT, HSPF and a European model known as "système hydrologique Européen TRANsport" (SHETRAN)/grid oriented Phosphorus component (GOPC) were used for this study. The results after model calibration of daily flows and total (TP) outputs were compared and assessed. It was found that the HSPF model was the best at simulating the mean daily discharge while SWAT gave the best calibration results for daily TP loads. The study showed that no single model is consistently better in estimating the annual TP export for all three catchments.

In addition, models have some downfalls with temporal resolution. For example, SWAT operates only at a daily time step compared to HSPF which can simulate at any time step from 1 min up to 1 day. This will also be linked to data availability and management because data is not always available at intervals equal to or less than the simulation time step.

All of these deficiencies require a solution to obtain better performance of the model. The ability of users to modify these models for specific climate and locations is essential during this process.

2.10.2 Factors to Consider in Wetland Modeling

Selecting a hydrological computer model that will fit a specific research area is a process that takes time and dedication. The ability to understand the hydrology of wetlands and their characteristics (e.g. wetland type, soils, plants) is very important and can assist during the model selection process.

According to the Hydrologic Engineering Center 1988, the factors that should be taken into consideration when modeling a wetland include: (1) the location of the wetland within the watershed, (2) hydrology, (3) water retention periods of the wetlands (entire year, or just during wet periods), (4) amount of vegetation in the wetland (this can affect evapotranspiration), (5) storage and infiltration capabilities of the wetland, and (6) spatial variation of the landscape. Other factors are also important to consider, such as wetland age.

Hydrologic models often have different modeling capabilities. In order to find the best fit for a hydrological research interest, one will need the model components. There are multiple components incorporated within each hydrologic model including the following parameters: precipitation, snow accumulation and snowmelt, evapo-transpiration, interception, infiltration, surface drainage and runoff, depression storage and routing, subsurface soil water flow and channel routing (U.S. Army Corps of Engineers, 1998). Computer models offer multiple options and capabilities for modelers to select the most appropriate for their specific needs and project goals. It is up to the modeler/scientist to determine which model meets their criteria. A limitation during the selection of a computer model could be data availability, quality of data and quantification of data in addition to other restrictions.

2.11 COMPUTER MODELS FOR WETLANDS AND WATER QUALITY MODELING

Wetland models have been developed to assist the scientific community in understanding processes that occur in multiple wetland types (e.g. costal, swamps, peatland and ponds). Some of these studies were conducted for hydrology processes (Hammer and Kadlec, 1986; Walton et al., 1996), while others studied nutrients (e.g. N and P) (Widener, 1995; 1994; Dorge et al., 1994). Even when studies have been conducted for these types of ecosystems, modeling wetland processes is relatively new as compared to other ecosystems (Mitsch et al., 1988).

There are three approaches used to model wetland hydrology: single event models, stochastic models, and comprehensive water budgets (Koob et al., 1999). These models are classified into different categories: one-dimensional, two-dimensional or three-dimensional.

These hydrologic computer models have been used in the development and implementation of total maximum daily load (TMDL) standards and guidelines that are required by the Clean Water Act (CWA), simulation of floods, and modeling the hydrodynamics of wetlands for flood detention, water quality, water quantity and climate change as well as others (Neitsch et al., 2002, Borah et al., 2002; Arnold and Fohrer, 2005; Hattermann et al., 2008;Watson et al., 2008; Liu et al., 2007; Gassman et al., 2005; Conan et al., 2003).

Although some studies have attempted to describe wetland hydrology (Konyha et al., 1995; Reinelt and Horner, 1995; Hawk et al., 1999; Arnold et al., 2001; Zhang and Mitsch, 2005), there is a limitation in the availability of computer models capable of describing wetland water flows (e.g. urban) (Drexier et al., 1999; Raisin et al., 1999). This limitation is a critical factor because most current wetland modeling applications are derived from traditional pond design engineering (Konyha et al., 1995), which is denoted to be used for wetland modeling. There are serious limitations to this approach when modeling wetland water fluctuations, which are characteristically more subtle than water movement captured by pond models (Obropta, 1998). One example of this particular case is the SWAT model which has a pond simulation capability and lake/wetland algorithms integrated for wetland simulations, but not a specific application for these types of ecosystems (Wu, 2007). The modification of a hydrological computer model's applications related to wetlands or similar areas is an alternative to improve hydrological simulation process obtain better results.

Some of these computer models (e.g. SWAT, HSPF) have incorporated wetland modules as part of their applications (Wang et al., 2010; Hattermann et al., 2008; Watson et al 2008; Liu et al., 2007; Gassman et al., 2005, Conan et al., 2003). For example, the Topographic-based Nitrogen Transfer (TNT) and transformation model has been used to assess the effect of riparian wetlands, (Beaujouan et al., 2001). Having these computer models available to assist those in the scientific community who are interested in wetlands is a great approach for protection, conservation and restoration efforts. Unfortunately, representation of wetland processes in these computer models is not sufficient and requires more consideration.

Okruszko et al. (2006) used the GIS platform for combining the results of hydrological models and assessing the ecological consequences of a restoration project on wetlands in Poland. The author identified three hydrological models: linear optimization model for control of hydraulic structures, 1-D hydrodynamic model of the surface waters for flood simulation, and a regional groundwater model (SIMGRO) for predicting the changes in groundwater level. GIS was used as a data management tool for all of the hydrological models. Their goal was to simulate the impact of different restoration measures in the area on hydrological regimes of rivers and riparian wetlands.

The authors found an affirmative relationship between land use changes and potential restoration projects, which could be used as a tool to demonstrate the benefits of ecological restoration to farmers.

The following section will describe general modeling approaches and techniques, in addition to presenting the detailed modeling processes for wetlands used by researchers.

A variety of computer models used to model wetlands or wetlands properties will be discussed. The following wetland model descriptions are included to present an overview of approaches taken to model multiple processes related to wetlands.

2.11.1 System for Urban Stormwater Treatment and Analysis INtegration

The System for Urban Stormwater Treatment and Analysis INtegration (SUSTAIN) model is a decision support system developed by the United States Environmental Protection Agency (USEPA) to evaluate alternatives for water quality management. The SUSTAIN model incorporates the best available research that could be practically applied to decision making processes, including the tested algorithms from other watershed scale hydrologic and water quality simulation computer models such as the Storm Water Management Model (SWMM) and the Hydrologic Simulation Program – FORTRAN (HSPF) (EPA, 2009). The SUSTAIN model simulates the ability of individual or a combination of BMP's in reducing stream peak flow, nutrients and sediment concentrations taking in consideration watershed characteristics (e.g. soil type, precipitation, temperature, and landuse). SUSTAIN includes algorithms for simulating urban hydrology and pollutant loading. The incorporation of these methods provide effectiveness and balance between computational complexity and practical problem solving (USEPA, 2009, 2010). SUSTAIN also incorporates an advanced approach that allows for cost effectiveness evaluation of both individual and multiple nested watersheds to address the needs of both local- and regional-scale applications (USEPA, 2009, 2010).

2.11.2 Soil and Water Assessment Tool

The Soil and Water Assessment Tool (SWAT) model (Arnold and Fohrer, 2005; Neitsh et al., 2001, 2002; Arnold et al., 1998) is a river basin, or watershed, scale model developed for the USDA Agricultural Research Service (ARS). SWAT could be used as part of the USEPA's Better Assessment Science Integrating point and Nonpoint Sources (BASINS) (Lahlou et al., 1998) for TMDL analyses (Di Luzio et al., 2002; Mausbach and Dedrick, 2004). BASINS is a watershed and water quality−based assessment system that integrates geographical information system (GIS), national watershed data, and environmental assessment and modeling tools like SWAT into one package (U.S. EPA, 2001). SWAT has proven to be very efficient in predicting impacts of management practices on water, sediment, and agricultural chemical yields in large, un-gauged watersheds (Gassman et al., 2007).This model was developed to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large, complex watersheds with varying soils, land use, and management conditions over long periods of time. The model is physically based and computationally efficient; it uses readily available inputs and allows users to study long-term impacts.

The model features include watershed hydrology, sediment and water quality, pesticide fate and transport simulation, channel erosion simulation, and rural and agricultural management practices (e.g. agricultural land planting, tillage, irrigation, fertilization, among others). SWAT subdivides a watershed into a number of sub-basins. Portions of a sub-basin that possess unique land use/management/soil attributes are grouped together and defined as one hydrologic response unit (HRU; Neitsch et al., 2002). Each sub-basin is simulated as a homogenous area in terms of climatic conditions, and each HRU is assumed to be spatially uniform in terms of soils, land use, and topography.

The model possesses multiple strengths such as: great documentation, physically based, GIS interface (BASINS), high-quality land management modules and databases and is suitable for studying watersheds ranging in scale from small to very large. As any other modeling application the SWAT model has limitations such as it could not be used for simulating sub-daily events (e.g. single storm events), it is useful only for simulating conservative metal species from the point source input, it cannot specify actual areas to apply fertilizers, and it may require extensive data input and management. Some of the data inputs are: land uses, soils, topography (Digital Elevation Models (DEM)), sub-watersheds, point sources of pollutant discharge, climate and weather data, crop data, and long term water quality and flow data. SWAT requires a significant amount of data and empirical parameters for development and calibration (Benaman et al., 2001).

SWAT has been modified to improve the simulation of specific processes at a watershed scale for different conditions. Gassman et al., 2005 discussed several examples of these modifications (e.g. Extended SWAT (ESWAT)), which features enhanced in-stream kinetics and other modifications (van Griensven and Bauwens, 2001; 2005), the Soil and Water Integrated Model (SWIM), which is partially based on SWAT (Krysanova et al., 1998, 2005) and SWATMOD, a version of SWAT that has been linked to MODFLOW to simulate detailed surface/groundwater interaction (Sophocleus et al., 2000). All of these are examples of how this powerful watershed modeling tool has been advanced to resolve more complex environmental evaluations.

Borah and Bera (2002) reviewed several fully developed hydrologic and non-point source pollution models (e.g. SWAT, HSPF, AGNPS, AnnAGNPS, ANSWERS, ANSWERS-Continuous, PRMS, KINEROS, DWSM, and CASC2D among others) and found SWAT to be one of the most capable models for long-term continuous simulations in predominantly agricultural watersheds. The model was found suitable for predicting yearly flow volumes, sediments and nutrient loads. Borah and Bera found that the model daily predictions were generally poor. In addition, it was found that SWAT should be combined with other single event models (e.g. DWSM, ANSWERS, KINEROS) for adequately simulating the extreme single storm events.

2.11.3 SWAT Applications on Wetlands

SWAT applications have been used to study wetland hydrology, functions, and relationships with water quality and quantity. However, appropriately represent wetlands in models is challenging, and few SWAT applications reported in the literature have considered wetlands (Wang et al., 2008). For that reason the incorporation of SWAT into wetland studies is an important issue to be addressed and more research involving the simulation of wetlands ecosystems should be conducted.

Wang et al. (2010) simulated the effects of wetland conservation and restoration on water quality and quantity at a watershed scale in Minnesota. Wang used SWAT in addition to the hydrologic equivalent wetland (HEW) developed by Wang et al., 2008. This combination of modeling tools can be used to consistently predict effects of wetland conservation/restoration scenarios and to logically prioritize restoration efforts (Yang et al., 2008). SWAT treats wetlands as water bodies located within sub-basins (Arnold et al., 2001; Neitsch et al., 2002a) and allows one wetland at a time per subbasin modeled. The objective of this study was to use the HEW concept in SWAT to assess effects of wetland restoration within the Broughton's Creek watershed in addition to wetland conservation within the upper portion of the Otter Tail River watershed. It was found that the HEW concept allows non-linear functional relations between watershed processes and wetland characteristics (e.g. morphology) to be accurately represented in the models. It was found that a reduction (loss) of approximately 10 to 20 percent of the wetlands in the study area would considerably increase the peak discharge and loadings of sediment, total Phosphorus, and total Nitrogen. The author compared wetland conservation versus wetland restoration and described that wetland conservation deserved a higher priority compared to wetland restoration.

Hattermann et al. (2008) compared two approaches that allow integration of important wetland processes using the Soil Water Infiltration and Movement (SWIM) model. The SWIM model allows addition of water to the system as precipitation and removal by runoff, drainage, evaporation from the soil surface and transpiration by vegetation. These approaches evaluate water and nutrients fluxes at different levels of complexity allowing modeling results to be improved in terms of seasonal river discharge and nutrient loads in catchments with wetlands. In addition, these approaches evaluated by Hattermann et al., 2008 are compatible to other models and can be used for the integration of wetland processes for regional applications.

Wu and Johnson (2008) performed a hydrologic comparison between a forested and a wetland/lake dominated watershed in northern Michigan using SWAT as the watershed modeling tool. The specific objectives of this study were to calibrate SWAT to simulate streamflow and compare the effects of wetland and lake abundance on the magnitude and timing of streamflow from two watersheds (East branch and Middle branch of the Ontonagon River basin). SWAT treats wetlands as water bodies located within sub-basins (Arnold et al., 2001; Neitsch et al., 2002a). The study shows that the watershed containing greater wetland and lake areas had lower spring peaks and higher sustained flows during summer and fall.

Conan et al. (2003) found that SWAT adequately simulated land use change from wetlands to dry land in Spain (Upper Guadiana river basin) and evaluated the impact of ground water withdrawals in the studied basin. The model showed misrepresentation of data for certain conditions, which could be related to the lack of sufficient data (e.g. rainfall data). Another limitation was that the model was unable to represent all of the discharge details impacted by land use changes.

Arnold et al. (2001) studied a hydrologic model for design and constructed wetlands near Dallas, Texas (Trinity River). For this study the SWAT model was used to assess flow following a heavy precipitation event and base flow in regards to a wetland ecosystem. The model was run for 14 years and was compared to observed data from the nearby watershed. The model results indicate that the wetland should be at or above 85 percent capacity over 60 percent of the time, showing no signs of dryness during the 14 years of simulation.

Wang et al. (2008) used hydrologic-equivalent wetland (HEW) concepts within the SWAT model to estimate streamflow in a watershed with numerous wetlands in Minnesota. One of the objectives of this study was to demonstrate how to incorporate wetlands into a SWAT model using the HEW concept. The HEWs were defined in terms of six calibrated parameters: the fraction of the sub-basin area that drains into wetlands (WET_FR), the volume of water stored in the wetlands when filled to their normal water level (WET_NVOL), the volume of water stored in the wetlands when filled to their maximum water level (WET_MXVOL), the longest tributary channel length in the sub-basin (CH_L1), Manning's n value for the tributary channels (CH_N1), and Manning's n value for the main channel (CH_N2). Statistical methods (the Nash-Sutcliffe coefficient (E j 2), the coefficient of determination (R 2), and the performance virtue (PVk)) were used to evaluate the model's performance. The results indicated that the HEW concept is superior in incorporating wetlands into SWAT for the study area. Overall, the SWAT model with the HEW assumption had an acceptable or satisfactory performance in simulating the streamflows at daily, monthly, seasonal, and annual time steps.

Yang et al. (2008) completed a research for Ducks Unlimited in Canada related to water quantity and quality benefits from wetland conservation and restoration. The SWAT-based modeling system was applied to examine the effects of wetland conservation and restoration in the Broughton's Creek watershed. Multiple scenarios were used for a better simulation; all scenarios included wetland areas ranging from 2,379 ha in the year of 2005 to 2,998 ha in the year 1968. The results show that the peak discharge at the watershed outlet was predicted to be reduced by 1.6% to 23.4%, and the sediment loading was reduced by up to 16.9%. This study estimated reductions of TP and TN at the watershed outlet as a result of wetland restoration.

These estimated reductions each are equivalent to 2.4% to 23.4% of the existing TP or TN export out of the study watershed.

2.11.4 Hydrologic Simulation Program-Fortran

The Hydrological Simulation Program – Fortran (HSPF) (Donigian et al., 1995) is a U.S. EPA program for simulation of watershed hydrology and water quality for both conventional and toxic organic pollutants. HSPF is a comprehensive, continuous, lumped parameter, watershed−scale model that simulates the movement of water, sediment, pesticides, and nutrients on pervious and impervious surfaces, in soil profiles, and within streams and well−mixed impoundments. HSPF allows the user to simulate selected water quality constituents by specifying their sources, sinks, chemical properties, and transport behavior. (Bicknell et al., 2000).

HSPF has been incorporated as a non-point source model (NPSM) into the U. S. EPA’s BASINS. BASINS is use to analyze and develop TMDL standards and guidelines nationwide (U.S. EPA 2009, Whittemore and Beebe, 2000). HSPF has become a useful tool for water resource planners, because it is more comprehensive than other modeling systems available which allows more effective planning.

The model features include: time-series-oriented model for easier data management, the unified structure makes it simple to operate, easy to modify and extend, and extensive research and uses of the model are available to use as guidelines.

2.11.5 HSPF Applications on Wetlands

HSPF is one of the two most widely used water quality models for simulation of hydrology and water quality nationwide (Bicknell et al., 1997). The model has been modified to take into account multiple physical and agricultural databases, which makes it more accessible and robust.

Schwar et al. (1998) studied the restoration of rivers and wetlands using hydrologic design. HSPF was used as the continuous hydrologic modeling tool. This computer model offers the potential to design and evaluate restoration projects based upon the goals of the project (e.g. magnitude, duration and seasonality of streamflows or water levels). The application of computer models for restoration goals is very valuable providing data during the decision making process.

Davis (2001) studied the integration of two hydrologic models: HSPF and MODFLOW in Florida. The results shows the advantages of the integrated hydrologic model, and how the results of the model can be used to help predict future impacts of groundwater production on wetland conditions in those areas where wetlands interact with the shallow groundwater environment.

2.11.6 Storm Water Management Model

Obropta et al. (2008) simulated urban wetland hydrology for the restoration of a forested riparian wetland ecosystem. The U. S. EPA Storm Water Management Model (SWMM) was used to characterize water movement through multiple sub-basins in the Teaneck Creek. The main objectives were to develop a conceptual restoration plan for the site and predict surface water movement through the re-established wetlands using multiple data inputs (e.g. surface water flow, ground water flow, vegetation and soils data). The authors developed a methodology for analyzing water budget in urban wetlands. The results showed that the SWMM model can be used to analyze water budgets for individual wetland basins. In addition, the model can be used to analyze each wetland basin, separately or in combination. This will allow the evaluation of the effects of restoration projects (e.g. installing water-control structures). The model was effective at analyzing nutrient loadings in wetland areas.

Tsihrintzis et al. (1998) simulated wetlands hydrodynamics for flood detention in South Florida. For this study the SWMM-EXTRAN link-node model (Roesner et al., 1989) was used. The objectives of this study were: to test the applicability of the SWMM model on wetland hydrodynamics and to apply the calibrated model, using it with synthetic storms in wetland design for flood control. Similarities were found between predicted flows and water surface elevations when compared to observed values. The results showed that the model could potentially be used as a design tool to optimally size hydraulic structures connecting wetland areas.

2.11.7 WETLANDS-2D

WETLANDS-2D (flow and transport in variably saturated porous media) is a mathematical model for one or two dimensional water flow and solute movement in variably-saturated multi-layered porous media featuring optional surface water bodies (ponds) and multiple root zones. The model supports axially symmetric 2D systems as an approximation to three-dimensional systems. The model is a modification of the Variably Saturated 2-D Flow and Transport Model (VS2DT). The VS2DT is a two-dimensional finite difference simulator for cross-sectional or cylindrical variably saturated flow in porous media. The model allows consideration of non-linear storage, conductance, sink terms and boundary conditions. Processes included are infiltration, evaporation and plant root uptake.

2.11.8 Wetland – DNDC

Wetland-DNDC is a computer simulation model of water, Carbon and Nitrogen biogeochemistry in forested wetland ecosystems. The model can be utilized for estimating forest production, ecosystem C dynamics and emissions of trace gases including methane (CH_4), nitrous oxide (N_2O), nitric oxide (NO), Nitrogen gas (N_2), and ammonia (NH_3).

Wetland-DNDC was constructed by integrating hydrological and forest biogeochemical processes at site and watershed scales. Wetland-DNDC has been calibrated and validated at site scale against numerous field data sets measured in forest ecosystems in North America and Europe (Stange et al., 2000; Butterbach-Bahl et al., 2001; Zhang et al., 2002). In Wetland-DNDC, forest growth is simulated by tracking photosynthesis, respiration, C allocation, N uptake and water demand at a daily time step.

Cui et al., 2005 linked two models together (MIKE SHE to Wetland-DNDC) for carbon budgeting and anaerobic biogeochemistry simulation in forested wetlands. The wetland-DNDC model was modified by utilizing parameters for management measures, refining anaerobic biogeochemical processes, and linked to the hydrological model – MIKE SHE. Simulation results from this model show that water table changes had a notable effect on Green House Gases' (GHGs) fluxes. It was found that anaerobic conditions in forested wetland soils reduce organic matter decomposition and stimulate CH_4 production. Results show that average long-term carbon storage in ecosystem pools increased with increasing rotation length: Soil carbon showed only minor, long-term responses to harvesting events. It was shown that linking these two models would allow the assessment of GHGs mitigation strategies, carbon budgeting and forest management. The basic functions adopted by Wetland-DNDC for simulating forest growth and soil biogeochemistry processes have been well validated in previous publications (Li et al., 1992; Aber et al., 1996; Li et al., 2000).

2.11.9 Hydrologic Engineering Center – Hydrologic Modeling System

Hydrologic Engineering Center-Hydrologic Modeling System (HEC-HMS) was developed by the U.S. Army Corps of Engineers (U. S. Army Corps of Engineers 1998). HEC-HMS essentially replaces HEC-1; it provides numerous options for simulating precipitation runoff processes. This new program offers the ability to perform continuous hydrograph simulations over long periods of time. It accomplishes this through the use of a "single-reservoir soil-moisture representation". It also computes spatially distributed run-off values using a 'grid cell' depiction of the watershed. HMS can simulate the rainfall-runoff at any point within a watershed if the physical characteristics of the watershed are utilized. It is a tool for watershed management in that an HMS model can be developed to account for the human impact and to determine the effect on the magnitude, quantity, and timing of runoff at points of interest. Results from an HMS model can be used by a number of other programs to determine impact in areas such as water quality and flood damage.

2.11.10Precipitation-Runoff Modeling System

PRMS (Precipitation-Runoff Modeling System) is a distributed watershed model that simulates precipitation and snowmelt driven movement of water through the basin via overland flow, interflow, and baseflow. The model was developed by the United States Geological Survey (Leavsley and Stannard, 1995). Watershed response can be simulated at a daily time step or more frequently over the course of a storm. Kinematic routing of the unidirectional flow and the transport of sediments through a receiving network of well-mixed channel reaches can be simulated when the model is in "storm mode". Simulation of the energy balance in the snowpack and the water balance is based on many theoretically and empirically developed relations. The resulting model is comprehensive and flexible, but also very complex and requires a large number of parameters. The model contains procedures for parameter optimization and sensitivity analyses. Some of the model inputs include: daily precipitation, maximum and minimum air temperature and solar radiation data. This model takes into account snowmelt and uses air temperature and solar radiation data to compute this process as well as the processes of evaporation, transpiration and sublimation.

2.11.11Environmental Fluid Dynamics Code

The EFDC (Environmental Fluid Dynamics Code) is a 3-dimensional surface water model for hydrodynamic and water quality simulations in rivers, lakes, reservoirs, wetland systems, estuaries, and the coastal ocean. The basic physical process simulation capabilities of EFDC are similar to the Blumberg-Mellor Model (ECOM3D) and CH3D-WES. Notable extensions included in EFDC include representation of hydraulic structures for controlled flow systems, vegetation resistance for wetlands, and high frequency surface wave radiation stresses in near shore zones. The model solves the hydrostatic, turbulent-averaged equations on an orthogonal curvilinear horizontal grid and a sigma-stretched vertical grid. EFDC transports salinity, heat, cohesive or non-cohesive sediments (only one sediment class at a time), and toxic contaminants that can be described by equilibrium partitioning between the aqueous and solid phases.

2.11.12TABS-2

TABS-2 Horizontal two-dimensional model (Thomas and McAnally, 1990) is a generalized numerical modeling system for open-channel flows, sedimentation, and constituent transport. This model is a two-dimensional, depth averaged, finite element hydrodynamic numerical model developed by the Waterways Experiment Station of U.S. Army Corps of Engineers. It computes water surface elevation and horizontal velocity components for subcritical, free-surface flow in two-dimensional flow fields. This model has been applied to a variety of waterways, including rivers, estuaries, bays, and marshes (wetlands).

Lee and Shih (2004) studied the impacts of vegetation changes (mangrove removal) on the hydraulic and sediment transport characteristics in a wetland in the Guandu Natural Reserve, Taiwan. A horizontal two-dimensional model, TABS-2, was applied as part of this study to simulate the hydraulic and sediment transport characteristics of this wetland. The authors found that the optimal removal ration was between 10% and 20% according to the variations of the hydraulic and sediment transport simulation from TABS-2.

NHCP, CODE 600, 2010). Sediment reduction in runoff is achieved through decreasing length of the hill slope to decrease peak runoff rate, increased settling of sediments in surface runoff, and interception and retention of water (Arabi et al., 2007; Tuppad et al., 2010).

2.12 WETLANDS AND GEOGRAPHICAL INFORMATION SYSTEMS (GIS)

Many studies related to wetlands and the integration of GIS have been conducted as part of efforts to better understand these ecosystems (Lyon and Adkins, 1995; Lyon and McCarthy, 1995; Ji and Mitchell 1995; Ramsey and Jensen, 1995; Williams and Lyon 1995; Cedfeldt et al., 2000; Tiner, 2003; Van Lonkhuyzen et al., 2004, White and Fennessy, 2005; Gibson, 2006; Torbick et al., 2008).

Lyon and Adkins (1995) developed a wetland prediction model for the St. Clair River in Michigan. The model used GIS and RS technologies to identify wetland locations and then estimate landuse changes for a period of time (1974 to 1985). The prediction model was based on a linear equation which combines the data layers and a constant error term to produce a predicted wetland type. The author proved that GIS/RS technologies could be integrated into wetland research.

Cedfeldt et al. (2000) developed an automated wetland assessment methodology using GIS technologies to identify functionally significant wetlands in Vermont. Cedfeldt used wetlands functions to designate a wetland as functionally important (flood flow alteration, surface water quality improvement and wildlife habitat). This model showed the ability to identify potential restoration or mitigation sites.

Gibson (2006) developed a GIS model for potential riparian wetland restoration sites in Ohio. The model used secondary GIS data to identify and prioritize potential restoration sites with a restoration category value. The classification for restoration categories showed 68.1% of the studied sites being classified correctly. According to the author, this model is a potential planning tool that should be employed in the future in restoration projects.

Torbick et al. (2008) studied the application and assessment of a GIS model for jurisdictional wetlands identification in Northwestern Ohio. During this study it proved the effectiveness of the GIS tool's application to wetlands (accuracy values ranged from 55% to 84%). A wetland classification system was developed to identify the dwindling wetland land cover types which provided accurate and detailed maps for the county.

2.13 WETLANDS AND REMOTE SENSING APPLICATIONS

Remote sensing (RS) technologies in combination with GIS applications allow the scientific community to better understand environmental processes (e.g. hydrology and climate change). The utilization of remote sensing technologies for multiple purposes including wetland inventory and identification has proven to be a useful and commonly used application (i.e. Lunetta and Balogh, 1999, Townsend and Walsh, 2001).

Remote sensing technologies have been used for years to assess multiple environmental conditions in the landscape (e.g. sedimentation, loss of vegetation index, and land use change). An example of RS applications could be to correlated precipitation or runoff data with aerial photography, commonly U.S. Department of Agriculture (USDA) crop history slides. Using this tool, it is possible to determinate the number of times that wet signatures (e.g. standing water, soil saturation, and stressed crops) are visible at a specific site on a series of aerial photographs taken over a number of years. Using aerial photography and observed data allows RS classifications to have accuracy levels of (69%-88%). A study conducted by Townsend and Walsh (2001) showed an accuracy of approximately 90% when classifying thematic mapper (TM) data for forested wetlands using a series of three season imagery.

3 INTRODUCTION TO METHODOLOGY AND RESULTS

This dissertation is in the form of two research papers that have been submitted to scientific journals. The first paper is entitled “Modeling the hydrologic significant of wetland restoration scenarios”, and the second paper is entitled “Assessing the significance of wetland restoration scenarios on sediment mitigation plan”. Both papers are related to the development of large-scale wetland restoration plans. All of the wetlands simulated in these studies are based on restoration of natural wetland systems (hydrological restoration).

The first paper utilized a physically-based model, the Soil and Water Assessment Tool (SWAT), to quantify the impacts of wetland restoration scenarios in the Shiawassee River watershed located in Michigan. Initially, field surveys were performed in multiple locations in the watershed to parameterize the wetland model. After model calibration and validation, the restoration scenarios included 1424 model runs for all possible combinations of wetland depths (15, 30, 61, and 91 cm), areas (50, 100, 250, 500 ha), and placements (89 subbasins) were evaluated. Each wetland area and depth combination was implemented one-at-a-time for each subbasin. Wetland volume in SWAT was calculated on a daily time-step using the water balance concept over 19-year-period. The reduction in streamflow and peak flow at the watershed outlet was calculated. A statistical analysis was performed to identify significant differences regarding the impact of wetland area and depth on streamflow reduction at the watershed outlet. Finally, high impact areas for wetland restoration were identified for two separate goals: (1) flow reduction at the watershed outlet and (2) flow reduction at a specific location within the watershed.

The goal of the second paper was to examine the sediment reduction benefit of wetland restoration scenarios at the subbasin and watershed levels in the River Raisin watershed of

Michigan. To accomplish this goal, two models were utilized. The System for Urban Stormwater Treatment and Analysis Integration (SUSTAIN) was used to evaluate the local scale benefits of wetland installation in reducing sediment loadings, while SWAT was used to estimate sediment loads to and from wetlands and at the watershed outlet. Multiple wetland restoration scenarios (4480 simulations) were developed to assess the importance of wetland areas (0.4, 0.81, 2 and 4 ha) on sediment reduction. Statistical analyses were performed to explore the effect of stream order, distance from the outlet, and wetland surface area on mean annual sediment reduction. Finally, the analytic hierarchy process (AHP) was used to incorporate environmental and economic aspects of wetland restoration scenarios into the selection of the most cost effective wetland surface area for implementation in each subbasin.

4 MODELING THE HYDROLOGICAL SIGNIFICANCE OF WETLAND RESTORATION SCENARIOS

4.1 ABSTRACT

The Soil and Water Assessment Tool (SWAT) was used to study the impact of wetland restoration on streamflow in the Shiawassee watershed of Michigan. Wetland restoration scenarios were developed based on combinations of wetland area (50, 100, 250, and 500 ha) and wetland depth (15, 30, 61, and 91 cm). Increasing wetland area, rather than depth, had a greater impact on annual streamflow. Meanwhile, wetland depth had a limited impact on streamflow. Wetland implementation resulted in negligible reductions in daily peak flow rates and frequency at the watershed outlet. In developing high priority areas for wetland restoration, similar locations were identified for reduction of subbasin and watershed outlet streamflow. However, the best combinations of area/depth were quite different depending on the goal of the restoration plan.

4.2 INTRODUCTION

Wetlands play a diverse, unique, and important role in the health and conservation of vital ecosystems. Wetland systems directly support millions of people throughout the world by providing such benefits as fertile soils for agricultural production (food and fiber), wildlife habitat, clean water, trees for timber and fuel, and recreation areas. In addition, wetlands provide important hydrologic, geochemical, and biological functions in a watershed (De Laney, 1995; Hart, 1995; NRC, 1995). However, wetlands are an extremely vulnerable environmental system and have significantly vanished in the past century (Nejadhashemi et al., 2012). According to Dahl (2000), approximately 2,606 km^2 of wetlands were lost in the United States between 1986 and 1997 with an estimated loss distribution of: urban development (30%), agriculture (26%), silviculture (23%) and rural development (21%). Furthermore, in recent years, wetlands in the U.S. are disappearing at a rapid rate of 243 km^2 per year (Dahl, 1990; Dahl 2000). Some examples of the possible major causes of wetland losses and degradation in the United States are: artificial drainage, deposition of fill material, diking and damming, conversion to crop production, construction, induced erosion, changing nutrient levels, increases in urbanization, and natural causes such as erosion, droughts, hurricanes and climate change (Carter, 1961; Leopold, 1968; U.S. EPA, 1993, Wray et al., 1995; Burkett and Kusler, 2000; U.S. EPA, 2009).

To protect wetlands, various regulations have been developed. One example is the Clean Water Act (CWA), administered by the United States Environmental Protection Agency (EPA). The CWA Section 404 established a program to regulate the discharge of dredged and fill material into waters of the United States, including wetlands (Copeland, 2006). In addition, many efforts have been developed to conserve, preserve, and restore wetlands. These efforts include the development and use of tools to identify wetland restoration and conservation areas, demonstrate wetland services, and perform wetlands classifications. Although some studies have attempted to describe wetland functions using watershed models (Konyha et al., 1995; Reinelt and Horner, 1995; Hawk et al., 1999; Arnold et al., 2001; Kirk et al., 2004; Zhang and Mitsch, 2005, Liu et., al 2008, Wang et al., 2008, Melles et al., 2010, Yang et al., 2010), there are limitations primarily in over-simplification of wetland processes and understanding flood mitigation benefits based on wetland placement in a watershed (Drexier et al., 1999; Raisin et al., 1999, Wang et al., 2008, Yang et al., 2008). Efforts to simulate wetlands at the watershed scale are discussed below.

Conan et al. (2003) found that the Soil and Water Assessment Tool (SWAT) adequately simulated land use change from wetlands to dry land in Spain (Upper Guadiana river basin). The model represented the impact of groundwater withdrawals throughout the basin and showed misrepresentation of certain conditions that could be related to lack of sufficient data (e.g. rainfall data). Wu and Johnson (2008) performed a hydrologic comparison between a forested and a wetland/lake dominated watershed in northern Michigan using SWAT. The specific objective was to compare the effects of wetland and lake abundance on the magnitude and timing of streamflow from two watersheds (east and middle branches of the Ontonagon River basin). The study showed that the watershed containing greater wetland and lake areas had lower spring peaks and higher sustained flows during summer and fall. Wang et al. (2010) simulated the effects of wetland conservation and restoration on water quality and quantity for a 4506 km^2 watershed in Minnesota. In this study, the concept of hydrologic equivalent wetlands (HEWs) was utilized. A HEW was defined in terms of six calibrated parameters: fraction of the subbasin area that drains into wetlands, volume of water stored in the wetlands when filled to their normal water level, volume of water stored in the wetlands when filled to their maximum water level, longest tributary channel length in the subbasin, Manning's n value for the tributary channels, and Manning's n value for the main channel (Wang et al., 2008). This study showed that the HEW concept allows non-linear functional relations between watershed processes and wetland characteristics (e.g. morphology). A reduction of approximately 10 to 20 percent of the wetlands in the study area resulted in a considerable increase in peak discharge and loadings of sediment, total phosphorus, and total nitrogen. They concluded that wetland conservation is a higher priority than wetland restoration (Wang et al., 2010). Yang et al. (2008) studied water quantity and quality benefits from wetland conservation and restoration scenarios using SWAT in the Broughton's Creek watershed (251 km^2). Multiple wetland restoration scenarios were examined, including: 0%, 10%, 25%, 50%, 75%, 90% and 100%. The optimal scenario determined for peak flow reduction in this study was 90% restoration. However, when compared with cost effectiveness, scenarios ranging from 50% to 80% were the most cost effective in terms of the benefit to the wetland acreage ratios. Hattermann et al. (2008) compared two approaches that allow integration of important wetland processes using the Soil and Water Integrated Model (SWIM). They compared a simple supply/demand approach versus an advanced hydrotopes approach and concluded that using the advanced approach significantly improved seasonal river discharge and nutrient load predictions in catchments with wetlands.

Placement of a wetland for streamflow reduction is an important consideration in the planning process. Understanding the relationship between stream order and wetland area and depth allow for targeting stretches of river in a watershed in which restoration will be most beneficial when project goals involve streamflow reduction. As described above, a number of studies have explored watershed-scale wetland modeling. However, none of these studies systematically examined the impact of wetland area, depth, and placement on streamflow and peak flow reduction in a watershed. This study is also unique in terms of the number of scenarios and the length of study performed to assess the hydrological function of wetlands. The Shiawassee watershed was selected for planning of wetland conservation activities because historically, the majority of the watershed was covered by wetlands (57%). However, vast land use change has reduced the wetland area to 11% of the watershed. Therefore, this watershed was considered to be a good candidate for development of wetland conservation and restoration strategies. The hypothesis is that by introducing wetlands onto the landscape, we can significantly reduce peak flow rate, which ultimately decreases environmental and economic losses due to flooding.

We utilize SWAT to evaluate the impacts of wetland depth (15, 30, 61, and 91 cm from normal water surface level to wetland bottom, the average depth of standing water), wetland area (50, 100, 250, 500 ha), and wetland placement in the watershed on streamflow and peak flow reduction at the watershed scale. The findings of this study will provide scientific understanding of wetland functions in controlling and altering the hydrologic cycle of a watershed.

4.3 MATERIALS AND METHODS

4.3.1 Study Area

The Shiawassee watershed (hydrologic unit code 04080203) is located southwest of Saginaw Bay in the central portion of Michigan's Lower Peninsula and is part of the Saginaw watershed **(**Figure 4-1). It drains approximately 3,000 square kilometers through the Shiawassee River to the Saginaw River, which ultimately drains to the Saginaw Bay of Lake Huron.

The land use in the Shiawassee watershed during pre-settlement was composed by approximately 57% woody wetlands and approximately 38% of deciduous/mixed forest (Apfelbaum et al., 2007). Currently land use in the watershed is 57% agricultural (primarily corn, soybean, wheat, and pasture), 14% deciduous/mixed forest, 11% woody wetlands, 7% grassland, and 5% urban **(**Figure 4-2 and Figure 4-3). The main significant land use change in the watershed was the conversion from marshes, forested bog wetlands and mixed/deciduous forests into agricultural land by logging, filling and draining (tiling) wetland areas.

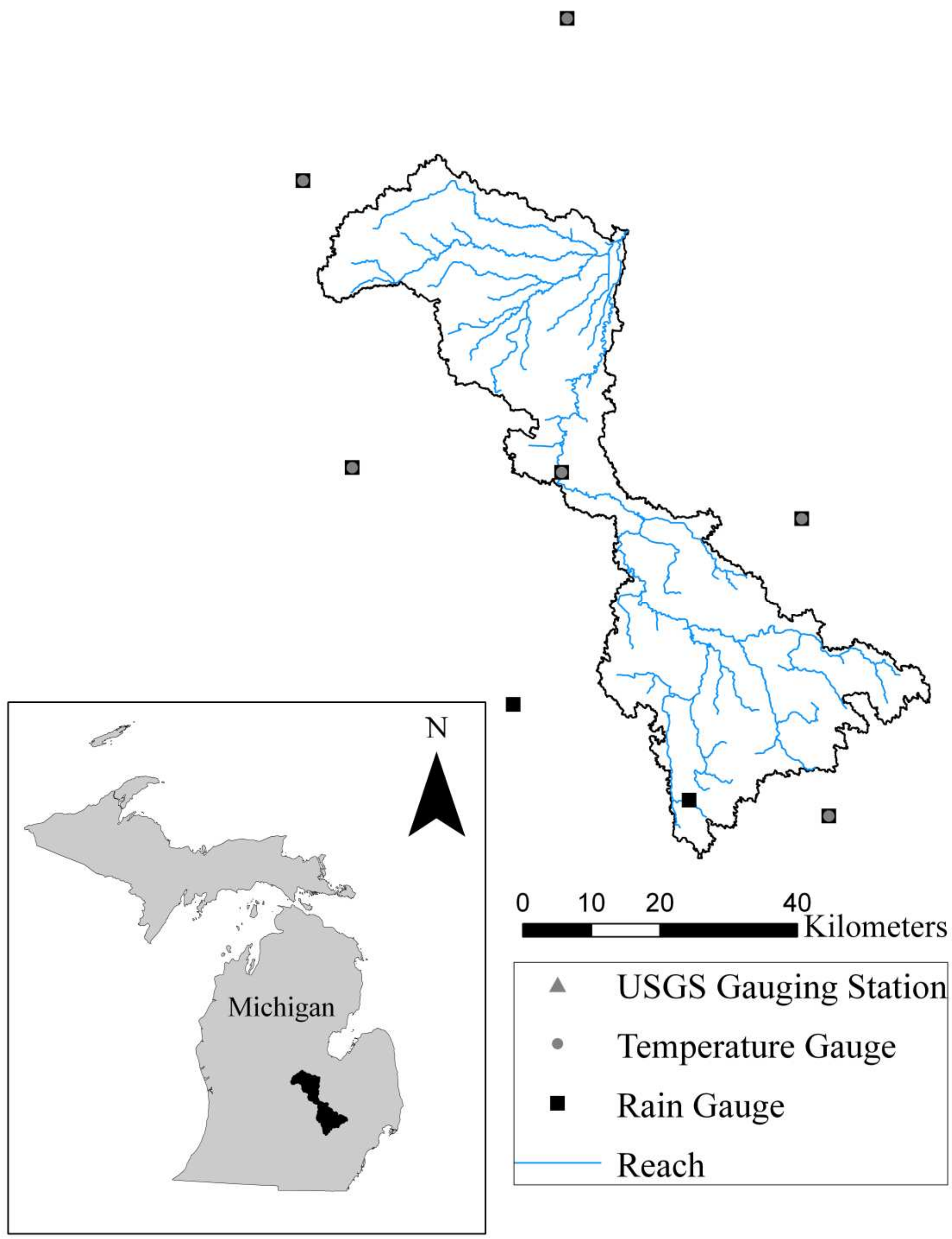

Figure 4-1. Study Area.

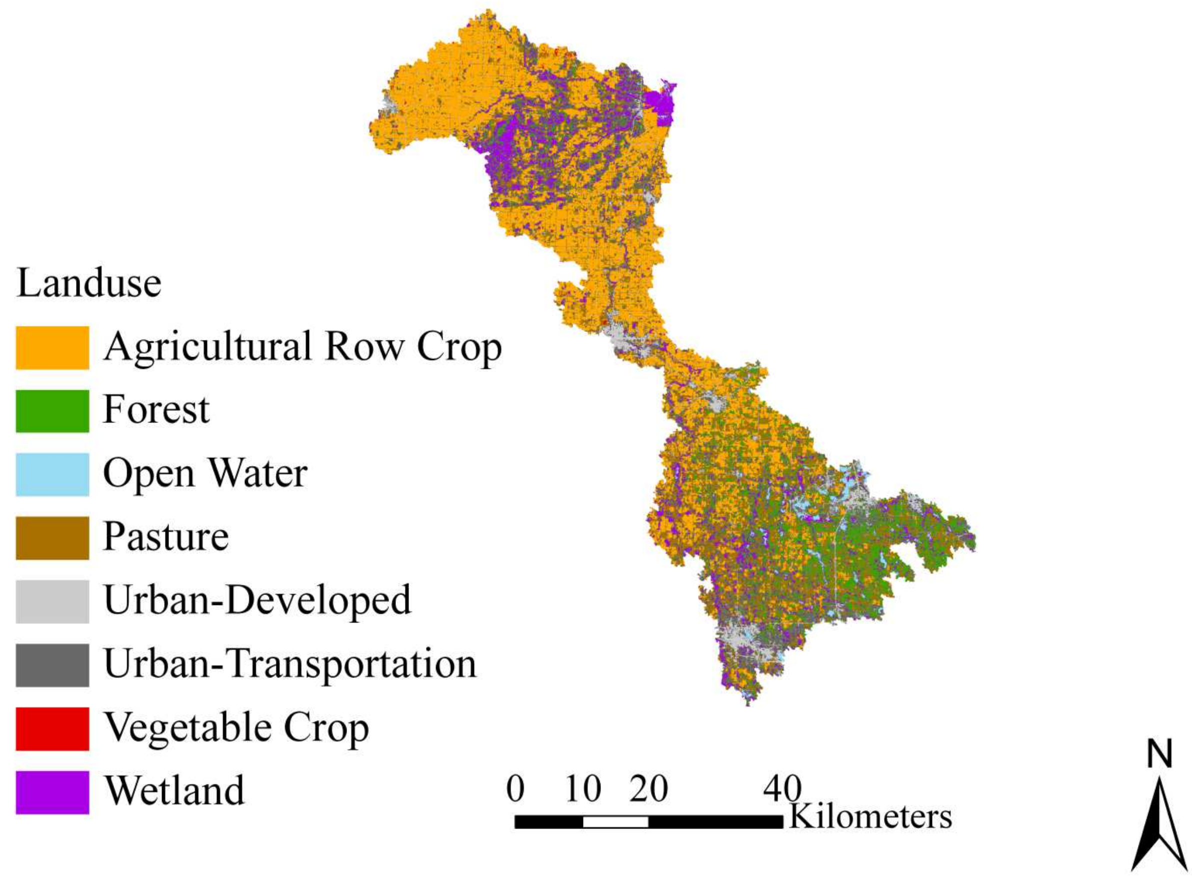

Figure 4-2. Shiawassee watershed land use

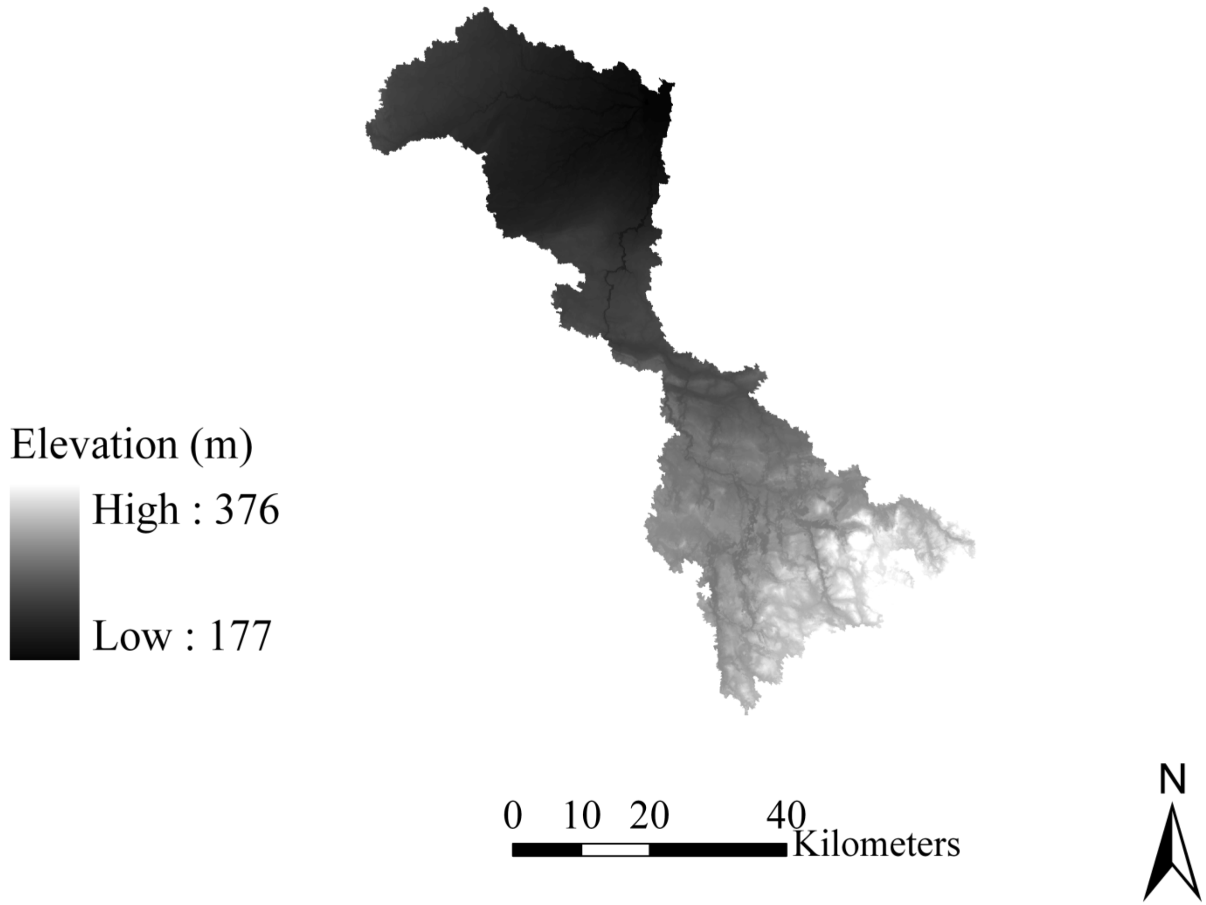

Figure 4-3. Shiawassee watershed topography.

4.3.2 SWAT Model Description

The SWAT model is a watershed scale model developed by the US Department of Agriculture (USDA)-Agricultural Research Service (Arnold et al., 1998). In this study ArcSWAT2009.93.7a was used. SWAT has proven to be a robust model capable of predicting impacts of land use change and management practices on water, sediment, and agricultural chemical yields in large un-gauged watersheds over long periods of time (Gassman et al., 2007). The model features include watershed hydrology, sediment and water quality modeling, pesticide fate and transport simulation, channel erosion simulation, and rural and agricultural management practices (e.g. agricultural land planting, tillage, irrigation, fertilization, among other). SWAT subdivides a watershed into a number of subbasins based on topography and river network. Portions of a subbasin that possess unique land use/slope/soil attributes are grouped together and defined as one hydrologic response unit (HRU) (Neitsch et al., 2005). In this study, the Shiawassee watershed was delineated into 110 subbasins in SWAT. The average subbasin area is approximately 2,000 ha, while maximum and minimum subbasin areas are 6,800 ha and 75 ha, respectively.

4.3.3 SWAT Model Wetland Processes

SWAT has multiple wetland algorithms that simulate water quality and quantity within a watershed. Below is a description provided by Arnold et al. (2001) and Neitsch et al. (2005) regarding representation of wetland processes in SWAT.

SWAT models four types of water bodies including reservoirs, ponds, wetlands, and depressions. Reservoirs are placed on the main channel, although the model does not recognize the difference between naturally occurring and man-made impoundments. Ponds and wetlands are modeled similarly, but are different in outflow calculation. Ponds model outflow as a function of flood season and soil water content, while in wetlands outflow occurs when water volume exceeds normal storage capacity. In contrast to reservoirs, ponds and wetlands are located off the main channel, receiving flow from a portion of the subbasin in which it is located. Finally, depressions are simulated at the HRU level, and runoff generated in these areas does not contribute to the main channel. Water flowing into wetlands must originate from the subbasin in which the water body is located. SWAT wetland simulations are based on one wetland per subbasin, which will have a predefined catchment area to capture streamflow discharge within the subbasin (Neitsch et al., 2005). SWAT employs the water balance for wetlands, presented in Equation 1.

$$V = V_{stored} + V_{flowin} - V_{flowout} + V_{pcp} - V_{evap} - V_{seep} \qquad (1)$$

where, V is the volume of water in the impoundment at the end of each day (m^3), V_{stored} is the volume of water stored in the wetland at the beginning of each day (m^3), V_{flowin} is the volume of water entering the wetland during each day (m^3), $V_{flowout}$ is the volume of water flowing out of the wetland during each day (m^3), V_{pcp} is the volume of precipitation falling on the wetland area during each day (m^3), V_{evap} is the volume of water removed from the wetland by evaporation during each day (m^3), and V_{seep} is the volume of water lost from the wetland by seepage (m^3).

Wetland surface area is used to calculate the amount of precipitation falling on the wetland as well as the amount of evaporation and seepage. SWAT updates surface area on a daily basis using Equation 2:

$$SA = \beta_{sa} \cdot V^{\exp(sa)} \tag{1}$$

where, SA is the wetland surface area (ha), β_{sa} is a coefficient, *V* is the volume of water in the wetland (m^3), and *exp(sa)* is an exponent. The coefficient, β_{sa}, and the exponent *exp(sa)* are calculated by solving Equation 3 and Equation 4 using surface area and volume known values.

$$\exp(sa) = \frac{\log_{10}(SA_{mx}) - \log_{10}(SA_{nor})}{\log_{10}(V_{mx}) - \log_{10}(V_{nor})} \tag{3}$$

$$\beta_{sa} = \left(\frac{SA_{mx}}{V_{mx}} \right)^{\exp(sa)} \tag{4}$$

where, SA_{mx} is the surface area of the wetland when filled to the maximum water level (ha), SA_{nor} is the surface area of the wetland when filled to the normal water level (ha), V_{mx} is the volume of water held in the wetland when filled to the maximum water level (m^3), and V_{nor} is the volume of water held in the wetland when filled to the normal water level (m^3).

The volume of precipitation falling on the wetland during a given day is calculated in SWAT using Equation 5:

$$V_{pcp} = 10 \cdot R_{day} \cdot SA \tag{5}$$

where, V_{pcp} is the volume of water added to the wetland by precipitation during the day(m^3), R_{day} is the amount of precipitation falling on a given day (mm), and *SA* is the surface area of the wetland (ha).

The volume of water entering the wetland on a given day is calculated using Equation 6:

$$V_{flowin} = fr_{imp} \cdot 10 \cdot \left(Q_{surf} + Q_{gw} + Q_{lat} \right) \cdot \left(Area - SA \right) \quad (6)$$

where fr_{imp} is the fraction of the subbasin area draining into the wetland. In this study, the total wetland area within each subbasin was calculated using the CDL (2009) and subtracted from the total subbasin area. Then, it was assumed that half of the remaining area drained into the wetland, similar to the technique used by Wang et al. (2010) for SWAT model calibration. Q_{surf} is the surface runoff from the subbas in on a given day (mm), Q_{gw} is the groundwater flow generated in a subbasin on a given day (mm), and *Area* is the subbasin area (ha). The volume of water entering the wetland is subtracted from the surface runoff, lateral flow and groundwater loadings to the main channel.

The volume of water lost to evaporation on a given day is calculated using Equation 7:

$$V_{evap} = 10 \cdot \eta \cdot E_0 \cdot SA \quad (7)$$

where, η is an evaporation coefficient (0.6) and E_o is the potential evapotranspiration for a given day (mm).

The volume of water lost by seepage through the bottom of the wetland on a given day is calculated as shown in Equation 8:

$$V_{seep} = 240 \cdot K_{sat} \cdot SA \tag{8}$$

where, K_{sat} is the effective saturated hydraulic conductivity of the wetland bottom (mm).

Wetlands in SWAT will release water whenever water volume exceeds the normal storage volume, V_{nor}. Wetlands have the ability to retain surface floodwaters, releasing the excess water slowly to downstream areas (Jiang et al., 2007). Wetland soil provides a considerable amount of floodwater mitigation, holding 3 to 9 times the weight of the soil per unit volume (Jiang et al. 2007).

Equations 9, 10, and 11 demonstrate this behavior:

$$V_{flowout} = 0 \qquad if\ V < V_{nor} \tag{9}$$

$$V_{flowout} = \frac{V - V_{nor}}{10} \qquad if \qquad V_{nor} \leq V \leq V_{mx} \tag{10}$$

$$V_{flowout} = V - V_{mx} \qquad if \qquad V > V_{mx} \tag{11}$$

4.3.4 SWAT Model Setup

The SWAT model data input for this project included landuse, topography, soils, wetland field data, and climate. The 2009 Cropland Data Layer (CDL, 56 m resolution) was acquired from the USDA - National Agriculture Statistics Service. Elevation and soil datasets were obtained from the National Elevation Dataset (NED, 30 m resolution) and the USDA State Soil Geographic dataset (STATSGO), respectively. The STATSGO dataset includes soil physical and chemical properties that are commonly used for hydrologic modeling and (STATSGO2, 2006). Information obtained from this dataset also can help in wetland classification and placement within the watershed. Daily climatic data (precipitation and temperature) was obtained from the National Climatic Data Center (NCDC) for eight precipitation and six temperature stations from 1988 to 2009. A model SWAT data frame for wetland modeling processes is shown on Figure 8-1.

4.3.5 Wetlands Digital Data Assessment

Wetland data was obtained from an extended data analysis of the National Hydrography Database (NHD), Cropland Data Layer (2009 CDL, 56 meters resolution), and the National Wetland Inventory (NWI). The NHD is a digital vector dataset used by geographic information systems (GIS) to map and analyze surface-water systems. The NWI is a United States Fish and Wildlife Service program that provides an inventory of wetlands primary for scientific purposes. The combination of these three data sources allowed for development of more accurate wetland locations and descriptions.

4.3.6 Wetland Field Data Assessment

Multiple wetland locations were assessed and survey data was incorporated into the model's wetland parameters for a more accurate representation of wetland conditions in the study area. Fifteen percent of the watershed area (28 sampling locations) was surveyed to obtain wetland average area, depth and approximate maximum water storage volume. The locations of the surveyed locations are presented in Figure 8-27. Standard survey techniques were used to obtain wetland profiles within the landscape (Figure 8-28, Figure 8-29, and Figure 8-30). During field data collection, the following wetlands were identified and evaluated: farm wetland, forested wetland, and restored wetland. Multiple parameters were considered while completing the field data assessment, including wetland hydric vegetation, hydric soils and hydrology.

For the surveyed wetlands, the average depth was measured. This information was used with the NWI surface area data to calculate wetland volume. From this, a relationship between wetland volume and wetland area was developed (Figure 4-4) to calculate the volume and depth of the remaining wetlands that were not surveyed. This approach was initially described and employed by Yang et al., (2008).

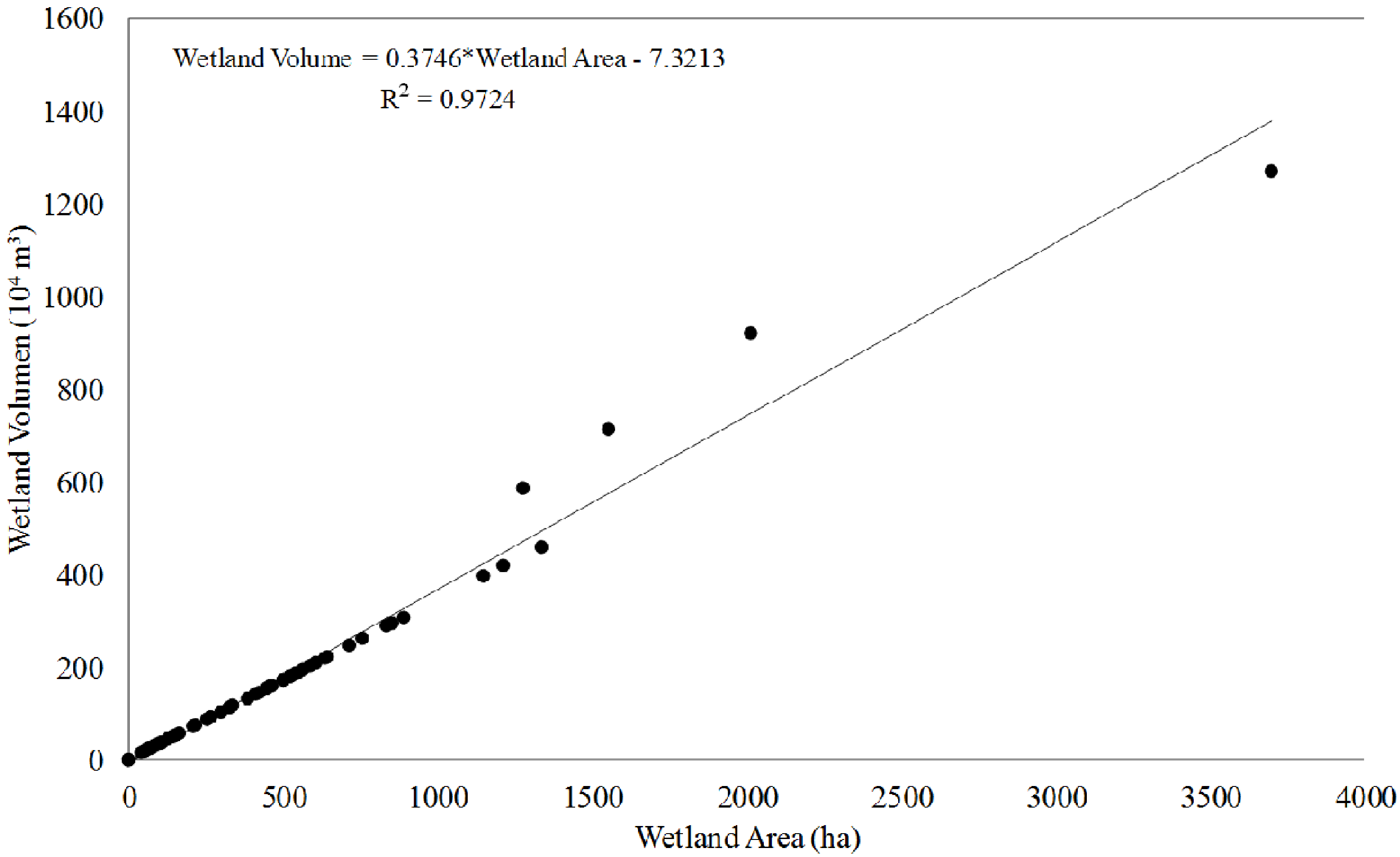

Figure 4-4. Relationship between wetland storage areas and volumes in the Shiawassee watershed.

4.3.7 SWAT Calibration and Validation

Model calibration and validation provides credibility to model simulated outputs and is a fundamental component of hydrological modeling. Calibration is important in hydrological studies to reduce model simulation uncertainty (Engel et al., 2007). During the calibration process, sensitive parameters that could potentially have an effect on wetland hydrologic processes (e.g. infiltration, evaporation and routing) were manipulated within acceptable ranges to improve model predictions (Table 4-1). Manual calibration on a daily basis was performed for streamflow followed by multiple statistical analyses for the validation period. This allowed both a qualitative and quantitative verification of the SWAT model's accuracy on flow predictions. Model outputs were compared to observed streamflow data.

Three statistical indices were used to evaluate model performance: Nash–Sutcliffe coefficient of efficiency (NSE), root mean square error (RMSE)-observations standard deviation ratio (RSR), and percent bias (PBIAS). NSE ranges from $-\infty$ to 1 (where 1 is optimal) and is a normalized statistic that determines the relative magnitude of the residual variance compared to the measured data variance. NSE is also used to quantitatively describe the accuracy of model outputs, and is considered satisfactory at values at greater than 0.5 on a monthly basis (Moriasi et al., 2007). RSR standardizes RMSE using the observations standard deviation, and ranges from zero to large positive values; with zero being a perfect model simulation (indicating zero residual variation). Monthly RSR values are considered satisfactory at less than or equal to 0.7 (Moriasi et al., 2007). PBIAS measures the tendency of simulated data to over- or under-predict the observed data. The optimal PBIAS is 0, with low positive/negative values indicating accurate model simulation, and values less than ±25% being acceptable (Moriasi et al., 2007). Ideal model calibration should consist of three to five years of data that includes average, wet, and dry years to emulate significant hydrological events that will trigger model constituent processes during calibration (Moriasi et al., 2007). The SWAT model streamflow calibration was performed on a daily basis from 2002-2005 and validated from 2006-2009 at USGS gauging stations 04144500 (Shiawassee River near Owosso, MI (drainage area of 1246 km^2) and 04145000 Shiawassee River near Fergus, MI (drainage area of 1440 km^2).

Table 4-1. Model Calibration parameters

USGS Station	Parameter	Description	Method	Calibrated Value
04144500	ALPHA_BF	Baseflow alpha factor	Replace	0.85
	CH_K(2)	Effective hydraulic conductivity in tributary channel alluvium	Replace	150
	CH_N(2)	Manning's n value for the main channel	Replace	0.02
	ESCO	Soil evaporation compensation factor	Replace	0.7
	GW_DELAY	Ground water delay time	Replace	12
	RCHRG_DP	Deep aquifer percolation fraction	Replace	0
	SOL_AWC(1)	Available water capacity of the first soil layer	Multiply	0.95
	SURLAG	Surface runoff lag time	Replace	1
	WET_K	Hydraulic conductivity through bottom of wetland	Replace	0.01
04145000	ALPHA_BF	Baseflow alpha factor	Replace	0.8
	CH_K(2)	Effective hydraulic conductivity in tributary channel alluvium	Replace	50

Table 4-1 (cont'd)

CN2	SCS curve number for soil moisture condition II	Multiply	0.9
ESCO	Soil evaporation compensation factor	Replace	0.7
GW_DELAY	Ground water delay time	Replace	5
RCHRG_DP	Deep aquifer percolation fraction	Replace	0
SURLAG	Surface runoff lag time	Replace	1
WET_K	Hydraulic conductivity through bottom of wetland	Replace	0.01

4.3.8 SWAT Wetland Restoration Scenarios for the Shiawassee Watershed

The capacity for wetlands to decrease flooding is determined by several factors such as water level fluctuations, plant community and density, habitat elements, and ground water hydrology, among other physical factors such as area and depth (Carter and Novitzki, 1988; Weller, 1994). Multiple scenarios were developed to assess these physical factors during wetland restoration processes. These scenarios were formulated after analyzing wetland physical factors utilizing GIS datasets in addition to collected field data. This allowed for evaluating the effect of wetland restoration area and depth. Overall, the impacts of four wetland depths and four wetland areas on streamflow and peak flow reduction were studied.

The evaluation of wetland area as part of restoration scenarios is an important factor to consider during the planning process to account for potential environmental and ecological benefits, in addition to financial reasons. Wetland restoration scenarios were represented in SWAT by two parameters: surface area of wetlands at normal water level (WET_NSA) and volume of water stored in wetlands when filled to normal water level (WET_NVOL). Restoration scenarios include sixteen combinations of four areas and four depths. Restoration area scenarios were 50, 100, 250, and 500 ha, while depth scenarios were 15 cm (6 in), 30 cm (12 in), 61 cm (24 in), and 91 cm (36 in). The simulated depths were selected to reflect field assessments. However, wetland depth is not a SWAT parameter, so it was adjusted using the relationship between area (WET_NSA) and volume (WET_NVOL) (Figure 4-4).

Each wetland area and depth combination was implemented one-at-a-time for each subbasin in SWAT and the reduction in streamflow and peak flow at the watershed outlet was determined. In the case that a wetland already existed in a subbasin, the restoration scenario area and volume were added to the existing wetland area and volume. However, when subbasin area was less than 500 ha, no wetland restoration scenarios were implemented. With four wetland areas, four wetland depths, and 89 subbasins with area greater than 500 ha in the Shiawassee watershed, this resulted in 1424 model simulations. Each SWAT model simulation was run from 1990 to 2009, with a two year warm-up period (1988-1989).

A statistical analysis was performed using the SAS 9.2 statistical package (refer to Figure 8-2 for the utilized statistical data frame). Wetland area and depth were compared using Fisher's Least Significant Difference (LSD) test (α=0.05), which allowed for examination of statistically significant differences regarding the impact of wetland area and depth on streamflow reduction at the watershed outlet.

4.3.8.1 Impact of Wetland Placement by Stream Order on Streamflow

Stream orders within the watershed were classified using the ArcGIS Spatial Analyst stream order classification tool. The stream order classification is a numerical denomination that classifies stream segments of a drainage network. This classification used topographical map features to classify each stream order. The stream order distribution for the Shiawassee watershed is: 55% of the streams are order one, 20% of streams are order two, 10% of streams are order three, and 15% of streams are order four. Stream order was used as a reference to evaluate the impact of wetland location on streamflow reductions. Stream order was considered for this study because it provides a visual idea of the area and strength of streams in the watershed (headwater versus main channel).

4.3.8.2 Impact of Wetland Placement on Peak Flow

Streamflow data at the watershed outlet was analyzed to observe the impact of wetland area and depth on peak flow events during the modeling period. Daily peak flow events were identified as a day where streamflow is greater than the previous and following days. Average annual maximum peak flow was defined as the 20-year average of the maximum peak flow event occurring in each year of the study period.

4.4 RESULTS AND DISCUSSION

4.4.1 Model Calibration and Validation

Statistical indices for the SWAT model calibration (2002-2005) and validation (2006-2009) are presented in Table 4-2. According to the model evaluation criteria described by Moriasi et al. (2007) was satisfactory in simulating streamflow. Time-series plots of observed versus simulated streamflow are presented for USGS station 04144500 and 04145000 in Figure 4-5 and Figure 4-6**,** respectively.

Table 4-2. Model Performance summary

	USGS Station 04144500		USGS Station 04145000	
Statistical Measures	Calibration	Validation	Calibration	Validation
NSE	0.69	0.67	0.71	0.58
PBIAS	7.18%	-8.40%	17.77%	0.69%
RSR	0.56	0.57	0.54	0.65

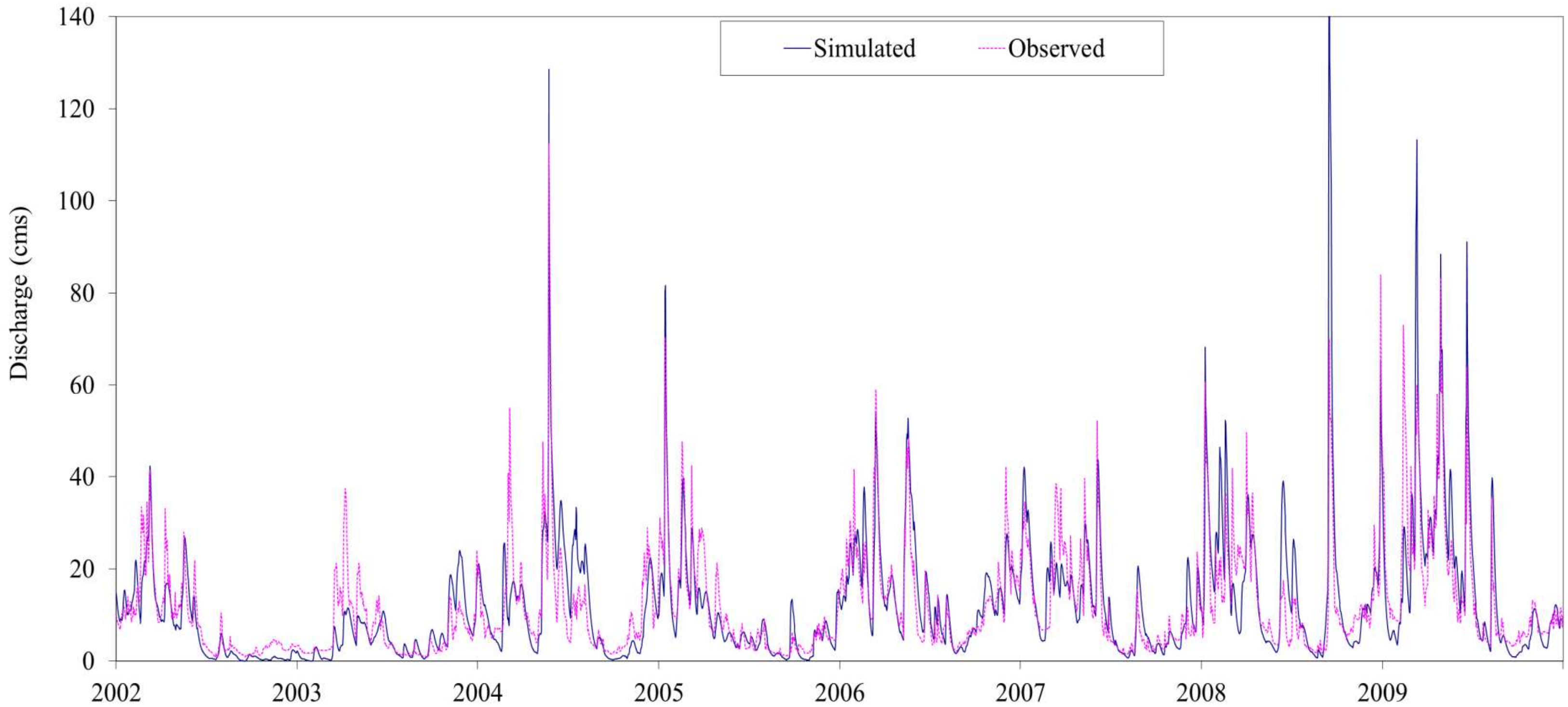

Figure 4-5. Observed versus simulated daily discharge at USGS station 04144500

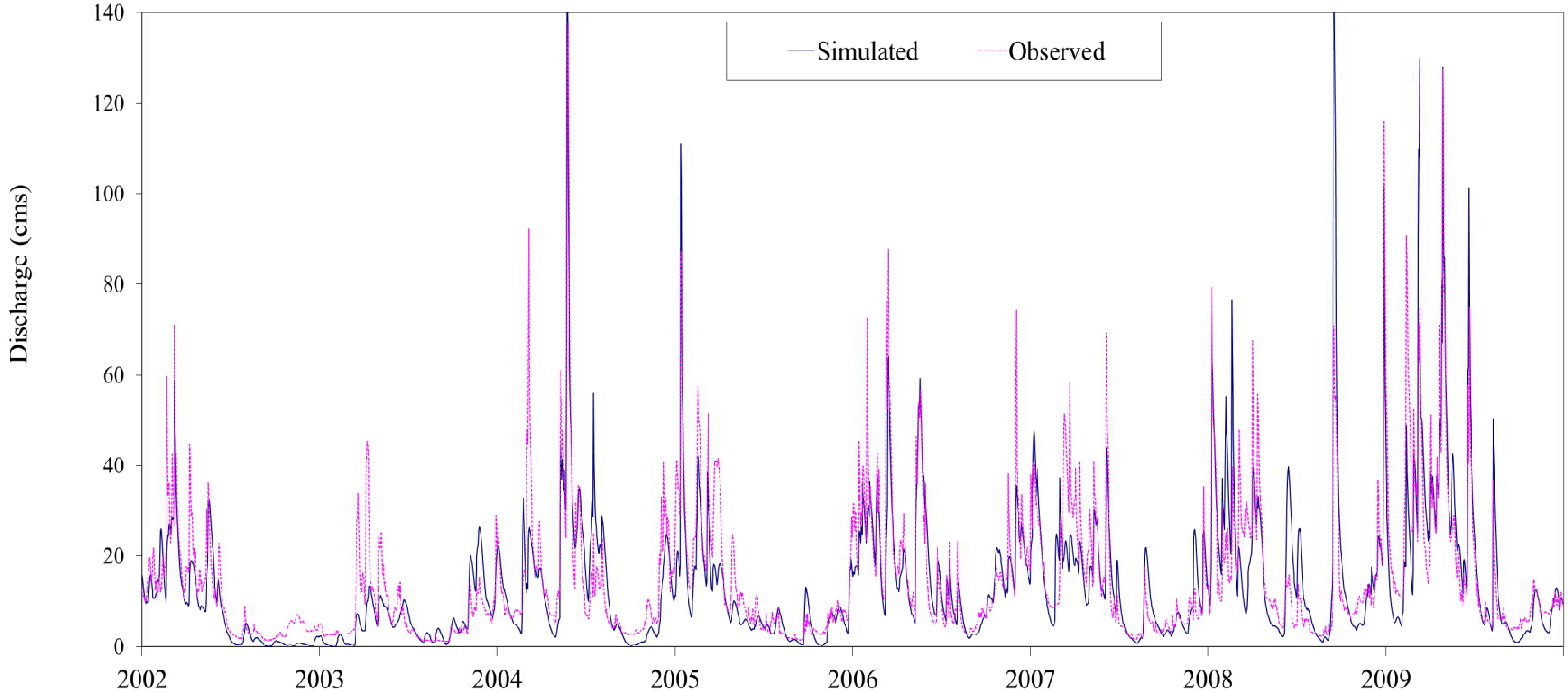

Figure 4-6. Observed versus simulated daily discharge at USGS station 04145000

4.4.2 *Wetland Restoration Scenarios*

4.4.2.1 Impact of Wetland Area on Streamflow

Streamflow changes from the base scenario at the outlet of the Shiawassee watershed were estimated based on different wetland areas (50 ha, 100 ha, 250 ha and 500 ha) with constant depth. For example, a comparison of streamflow change based on wetland area with a constant depth of 91 cm is presented in Figure 4-7 **(**Figure 8-3**,** Figure 8-4**,** and Figure 8-5 in the appendix**)**. Overall, average annual streamflow changes ranged from 0 to 0.35 percent. As wetland area in a subbasin increased, the streamflow change at the watershed outlet increased. For instance, subbasins in the 50 ha scenario had negligible impact on streamflow, while 500 ha wetland implementation resulted in the greatest changes in streamflow for all depths, generally in the form of streamflow reductions. In addition, as wetland area increases the number of subbasins contributing to streamflow reduction at the watershed outlet increases. These results are similar to those found by Yang et al. (2008), which demonstrated that larger flow reductions require more wetlands to be restored.

Comparing average daily streamflow across the 20-year study period resulted in negligible changes at the watershed outlet. However, the results of the Fisher's LSD indicate statistically significant, but not necessarily relevant (from a stakeholder standpoint), differences between the wetland area scenarios (Table 4-3).

Table 4-3. Statistical comparison of streamflow means for wetland area

Wetland Area (ha)	Streamflow at watershed outlet (m^3/s)
50	20.320 a
100	20.318 b
250	20.309 c
500	20.306 d

*Streamflow means followed by the same letters are not significantly different (α=0.05).

Streamflow changes from the base scenario at individual subbasin outlets on which wetlands were implemented were estimated based on different wetland areas (50 ha, 100 ha, 250 ha and 500 ha) with constant depth. With a constant wetland depth, area is varied to evaluate its impact on streamflow (e.g. Figure 4-8 represents 91 cm depth across all wetland area scenarios; additional areas shown in Figure 8-6**,** Figure 8-7 and Figure 8-8 in the appendix**)**. Streamflow changes at the subbasin level range from 0 to 25 percent depending on the area of wetland restoration. Based on visual observation, regardless of depth, the 250 ha and 500 ha wetland area restoration scenarios result in a majority subbasins with greater than 5% streamflow change at the subbasin outlet. Finally, restoring 50 ha of wetland has negligible impact on subbasin level streamflow.

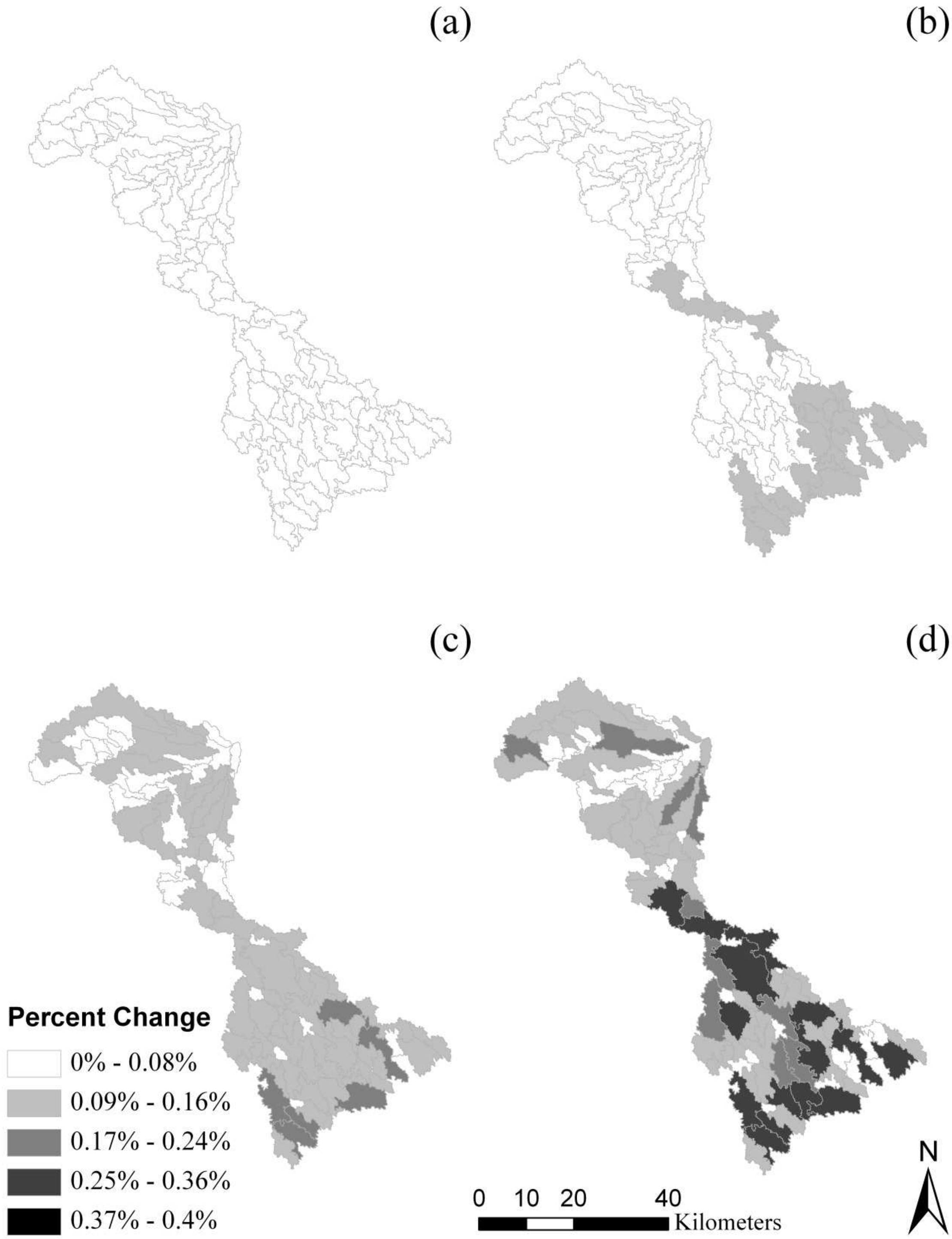

Figure 4-7. Percent flow change at the watershed outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 91 cm

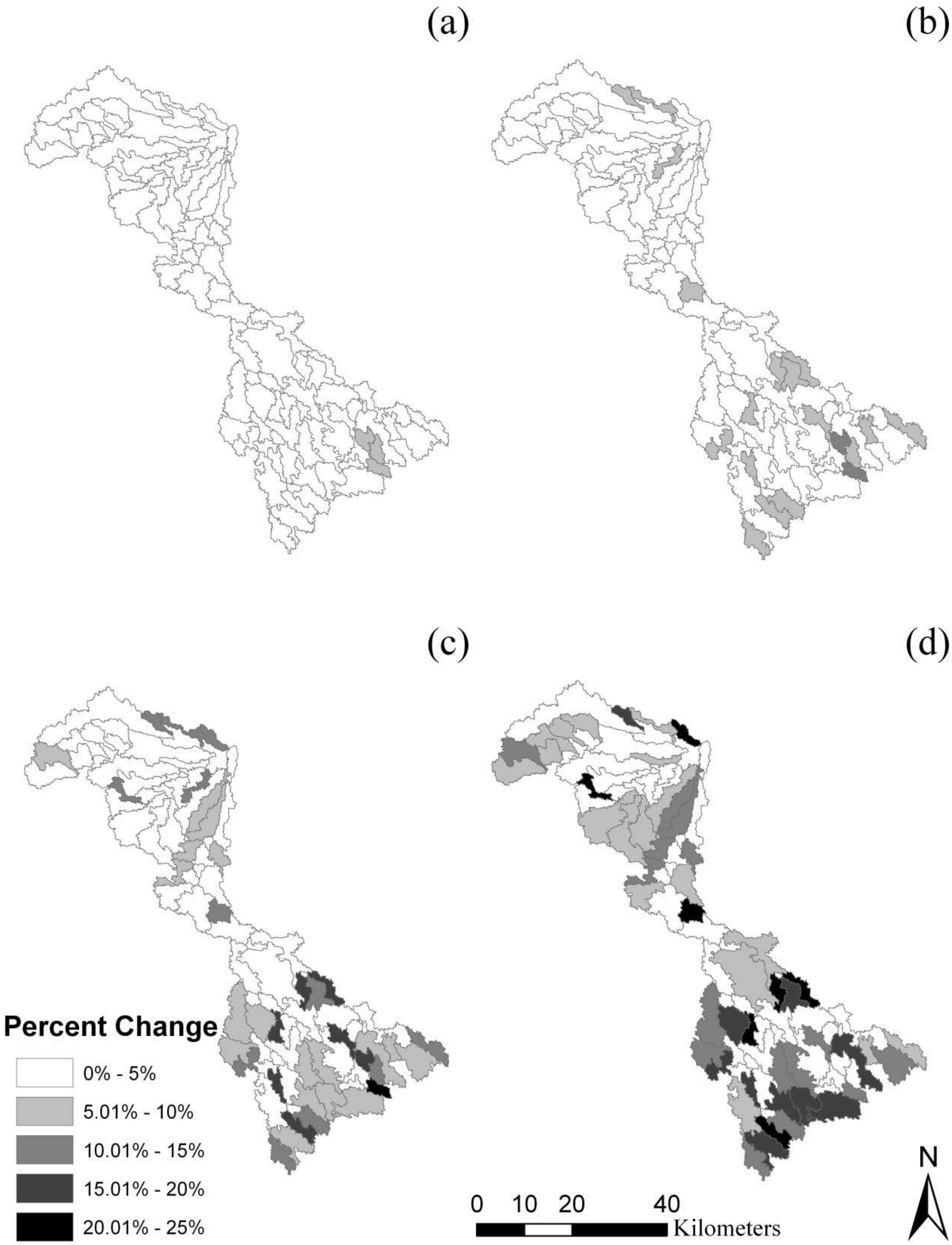

Figure 4-8. Percent flow change at the subbasin outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 91 cm

4.4.2.2 Impact of Wetland Depth on Streamflow

The impact of wetland depth in each subbasin on streamflow change at the watershed outlet was evaluated using a constant wetland restoration area for each wetland depth. For example, all depths at a constant area of 500 ha are presented in Figure 4-9 **(**Figure 8-9**,** Figure 8-10**,** and Figure 8-11 in the appendix**)**. Overall, streamflow at the watershed outlet is minimally impacted among all depths when area is constant. Wetland depth scenarios showed streamflow reduction for wetland areas greater than 100 ha, while no impacts on streamflow reduction were observed in any of the 50 ha scenarios when examining the depth effect. Under the 500 ha wetland restoration scenario, most subbasins influence streamflow at the watershed outlet regardless of depth. Therefore, in developing wetland restoration scenarios, wetland area is more important than depth in affecting streamflow at the watershed outlet.

The statistical analysis comparing treatment means (wetland depth) across all studied wetland areas using Fisher's LSD (α=0.05) is presented in Table 4-4. All wetland depths were found to be statistically similar. This supports the conclusion that selection of wetland depth does not have as much impact on streamflow at the watershed outlet as selecting a suitable wetland area.

At the subbasin level, the trend among wetland depths in streamflow change was similar to the outlet, although the percent changes were greater. For example, as demonstrated in Figure 4-10 **(**Figure 8-12**,** Figure 8-13**,** and Figure 8-14in the appendix), all depths have a similar impact on subbasin level streamflow. This was the case for depth across all wetland areas.

Table 4-4. Statistical comparison of streamflow means for wetland depth

Wetland Depth (cm)	Streamflow at watershed outlet (m^3/s)
15	20.314 a
30	20.314 a
61	20.313 a
91	20.313 a

*Streamflow means followed by the same letters are not significantly different (α=0.05).

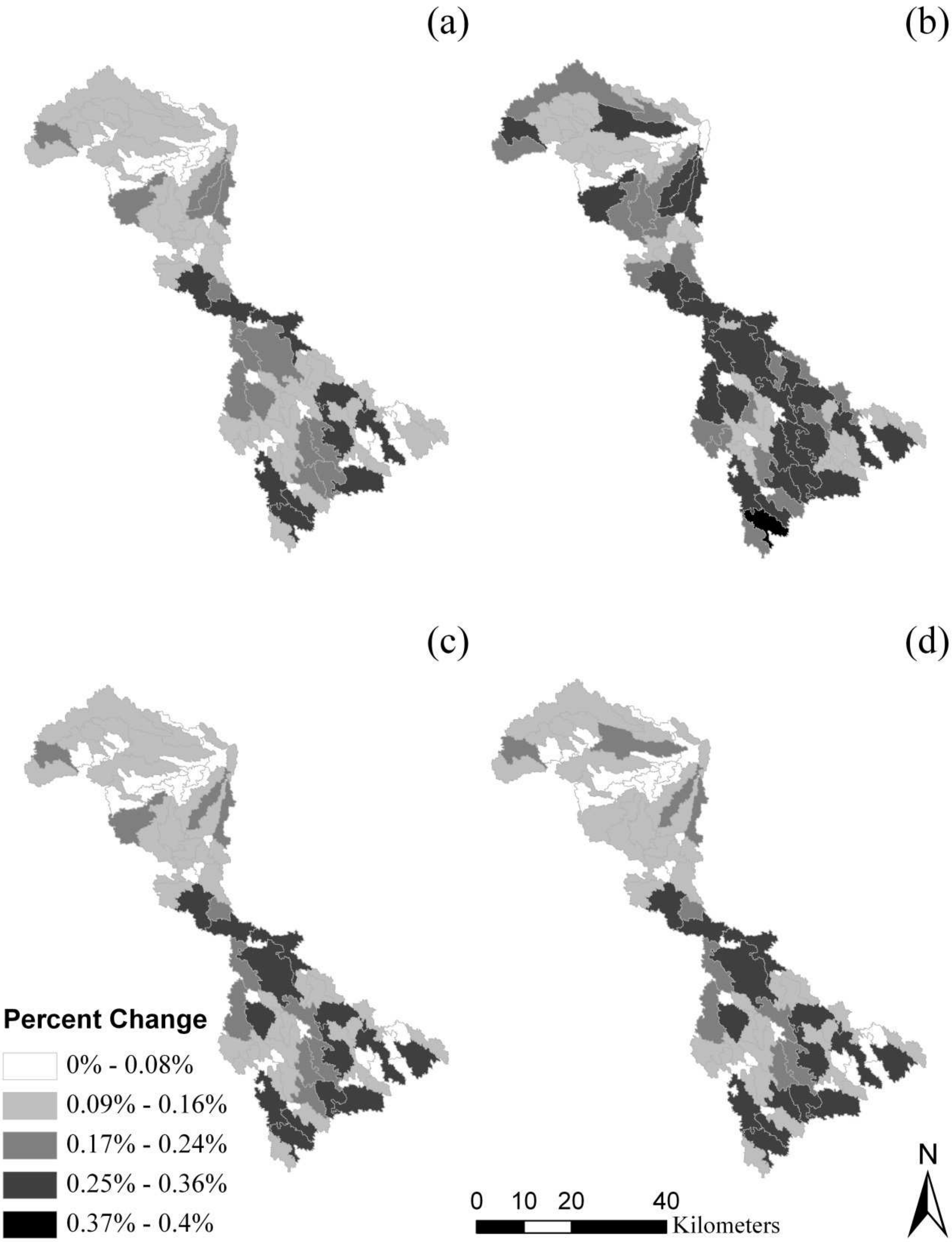

Figure 4-9. Percent flow change at the watershed outlet compared to base scenario – 500 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

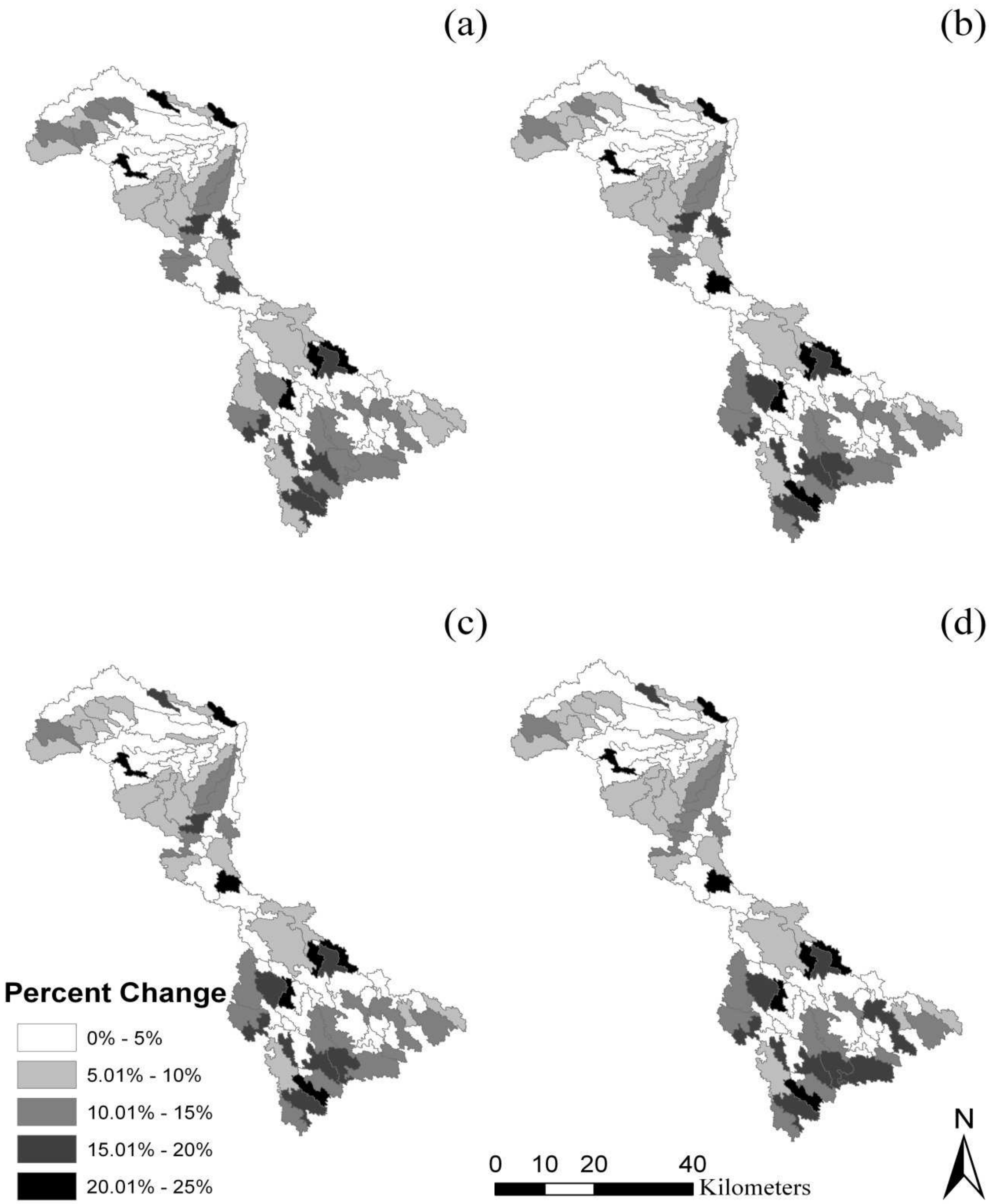

Figure 4-10. Percent flow change at the subbasin outlet compared to base scenario – 500 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

4.4.2.3 Impact of Wetland Placement by Stream Order on Streamflow

The wetland performance in percent streamflow change at the watershed outlet was classified based on the stream order in which the wetland restoration scenario was implemented. For example, Figure 4-11presents percent flow change at the watershed outlet with a constant depth of 91 cm (Figure 8-15**,** Figure 8-16, and Figure 8-17 in the appendix**)**, while Figure 4-12 presents percent flow change at the watershed outlet with a constant area of 500 ha (Figure 8-18, Figure 8-19, and Figure 8-20 in the appendix**)**. Average streamflow changes were insensitive to stream order at restoration areas up to 250 ha, while at 500 ha wetland placement on stream order one and four has a greater impact on streamflow (Figure 4-11). Under constant area a similar trend was observed in which wetland placement in stream orders one and four resulted in greater streamflow changes at all depths (Figure 4-12). Therefore, it can be concluded that wetland restoration on first and fourth order streams may result in higher streamflow reduction when wetland size is greater than 250 ha.

In terms of the impact of wetland restoration scenarios on streamflow change at the subbasin level, wetlands placed on subbasins with stream order one are more effective than other orders. As stream order increases moving from one to four, effectiveness of wetland restoration scenario decreases among all depths and areas as shown in Figure 4-13 and Figure 4-14 **(**Figure 8-21 through Figure 8-26 in the appendix). This can be explained by assuming that the flow rate of first order streams are generally less than those of higher stream order. Therefore, the same wetland size is more effective in reducing flow in subbasins with first order streams than those with orders two, three, or four. This indicates that specific subbasin location in which the wetland was implemented plays an important role in local streamflow reduction.

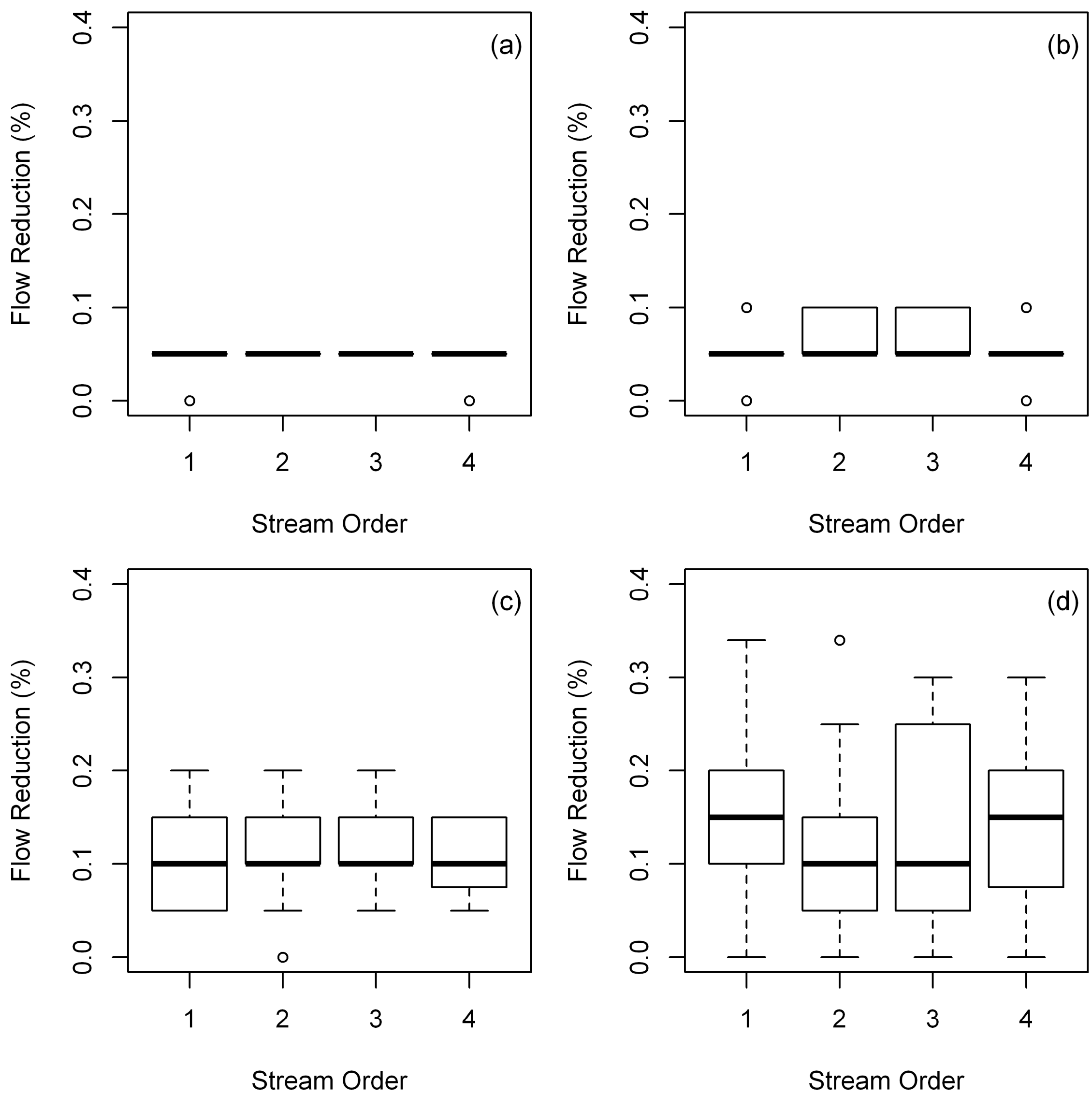

Figure 4-11. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 91 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

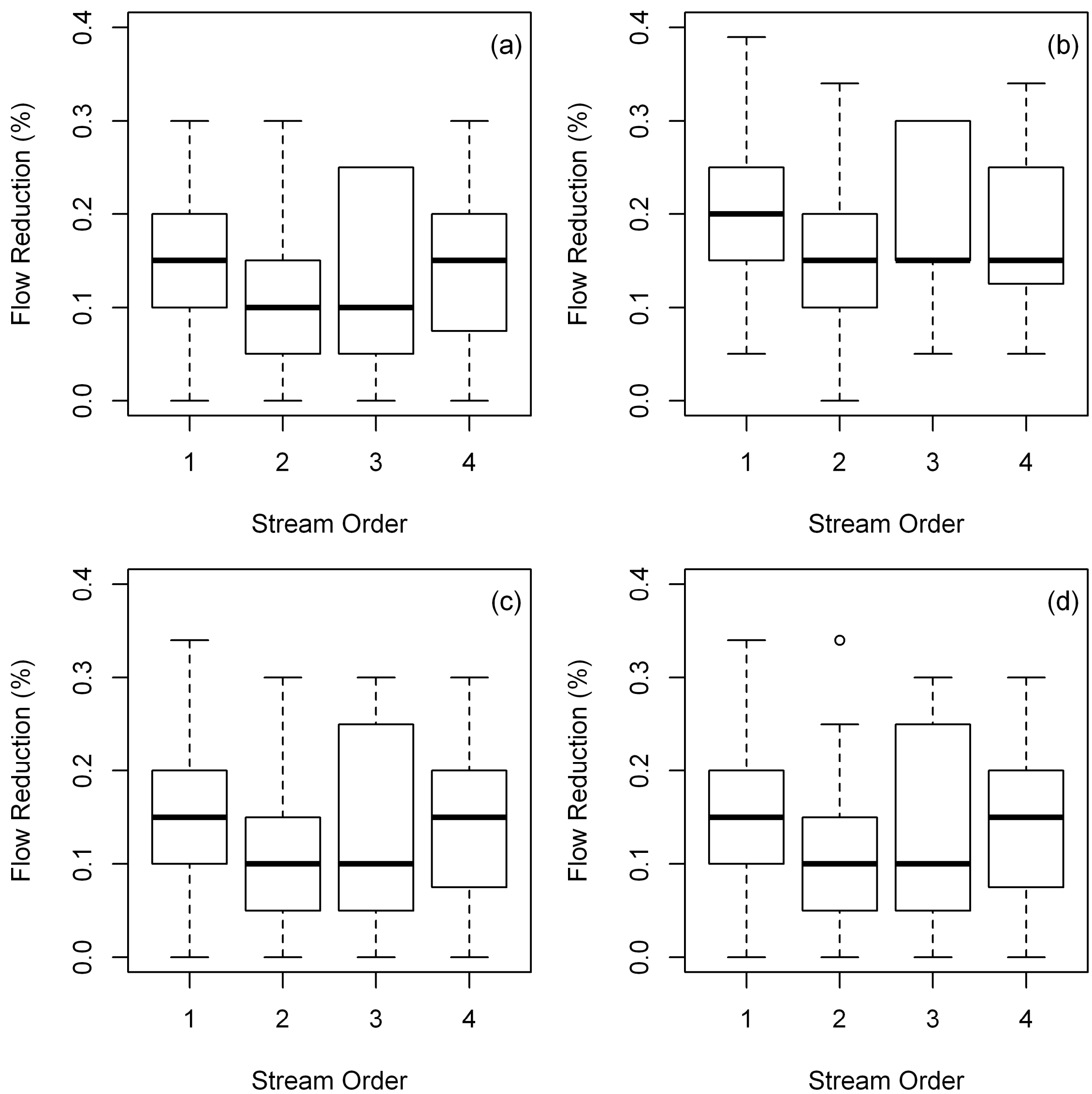

Figure 4-12. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 500 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

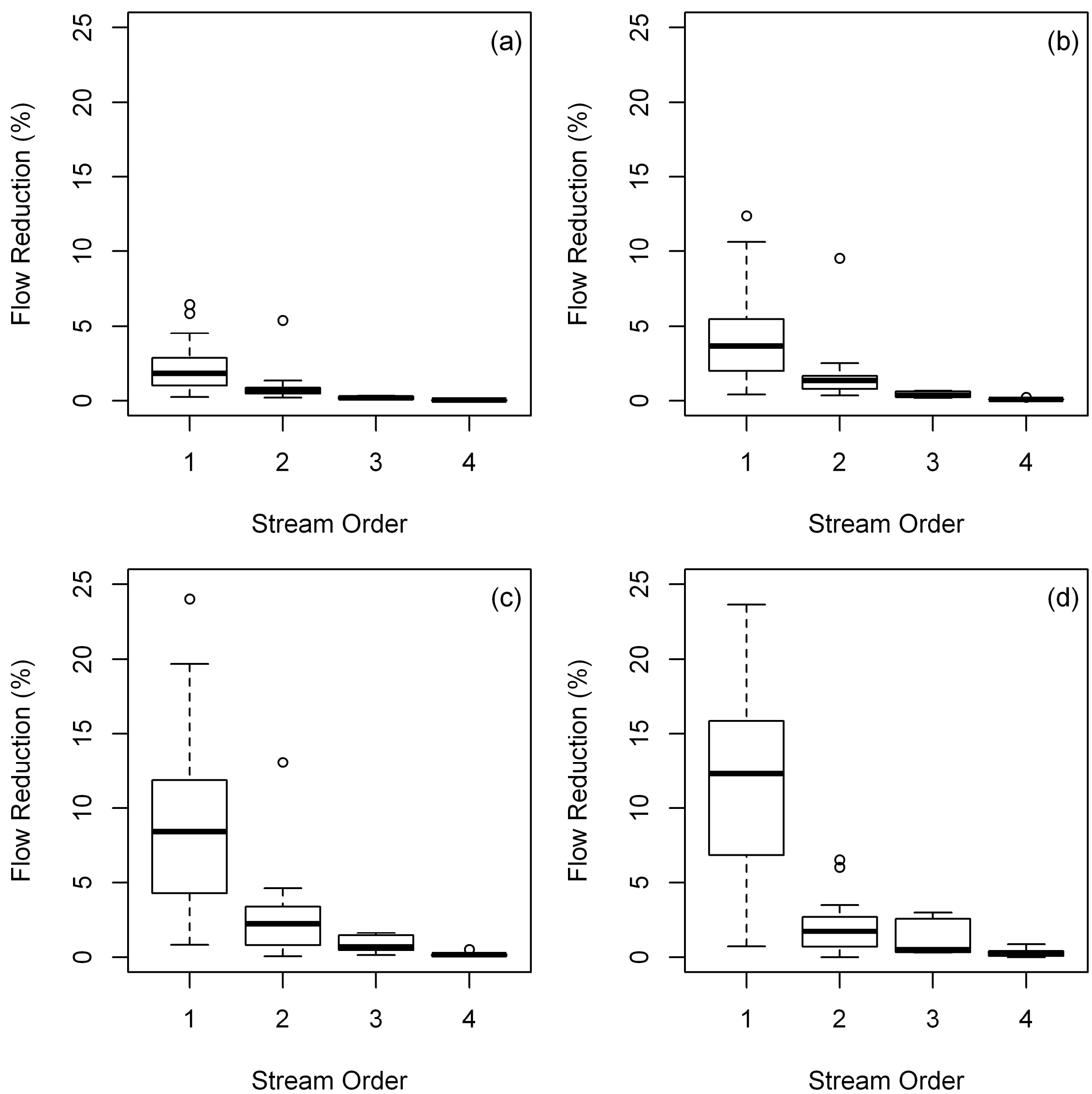

Figure 4-13. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 91 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

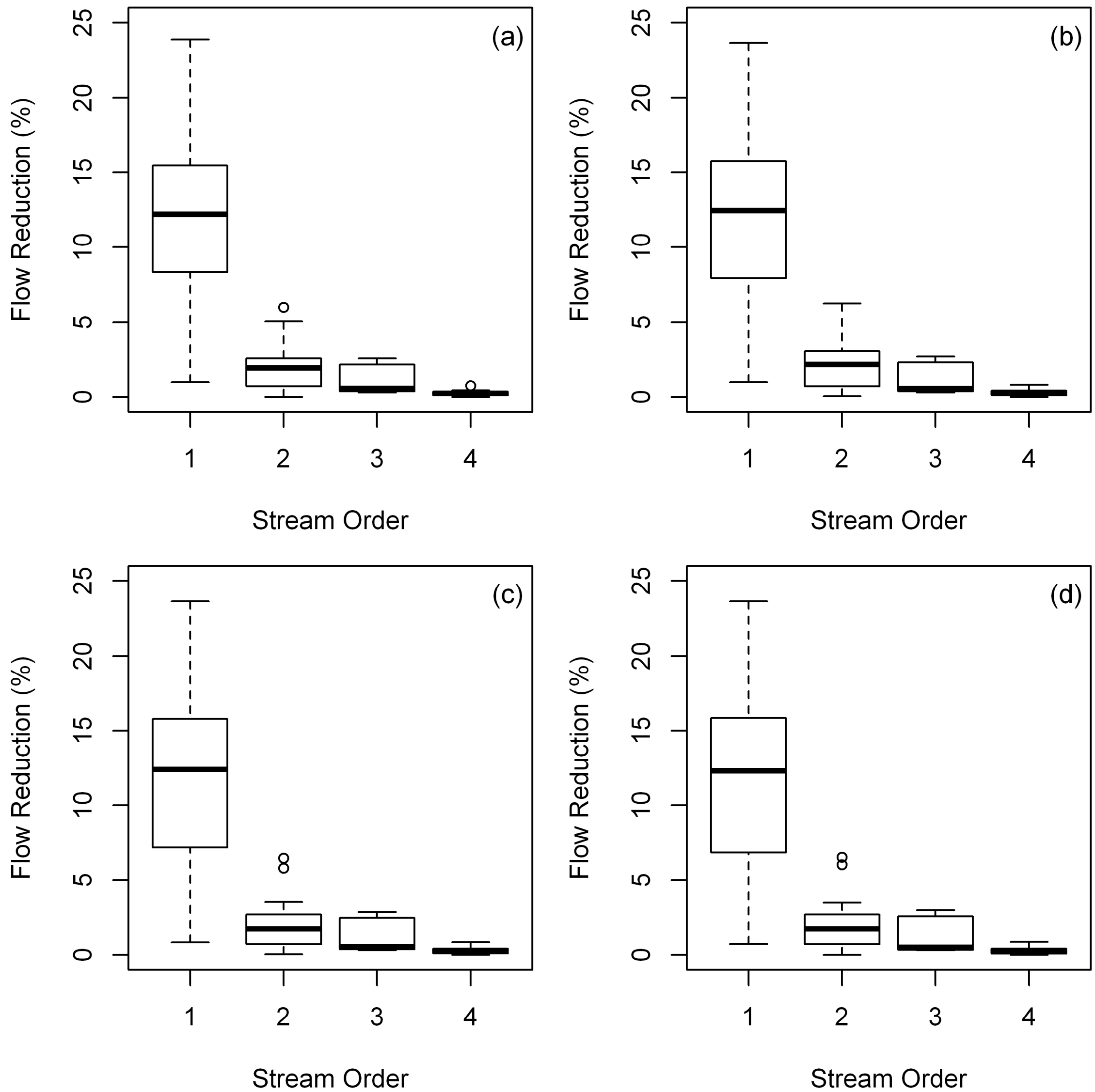

Figure 4-14. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 500 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

4.4.2.4 Peak Flow Reduction

Peak flow is defined peak as the maximum water level reached during a runoff event. This is identified as a day in which the streamflow rate is greater than both the preceding and following days. Two criteria for examining peak flow were used (daily and average annual maximum peak flow). Peak flow reduction was compared between the case in which no wetlands are restored (base scenario) and the scenarios containing wetlands (all combinations of area/depth one-at-a-time in each subbasin). The base scenario produced 553 daily peak flow events with a 20-year average peak flow rate of 40 m^3s^{-1} at the watershed outlet. Negligible changes in peak flow rate were observed, while the number of peak flow events was reduced by one to eight events, depending on the wetland restoration scenario and location. There was a greater reduction in peak flow events with increasing wetland depth and area.

The base scenario average annual maximum peak flow was 157 m^3s^{-1}. Overall, implementation of wetland scenarios resulted in reductions of 0.2 m^3s^{-1} to 1.9 m^3s^{-1}, depending on the combination of depth, area, and location (Table 4-5). Similar to the previous observation concerning wetland area and depth, increasing area results in a greater decrease in annual maximum peak flow. However, increasing depth does exhibit decreases in annual maximum peak flow, although they are not as large as increasing area.

Table 4-5. Wetland area/depth combinations for reduction of average annual maximum peak flow (m^3/s) events

Area/depth	15 cm	30 cm	61 cm	91 cm
50 ha	0.21	0.25	0.32	0.43
100 ha	0.34	0.41	0.55	0.67
250 ha	0.87	0.85	1.12	1.33
500 ha	1.17	1.35	1.71	1.85

4.4.3 High Priority Areas for Wetland Restoration

High priority areas for wetland restoration were developed with two implementation goals: flow reduction at the watershed outlet or flow reduction at a specific location within the watershed. These areas were identified by selecting the combination of wetland area and depth for each subbasin that resulted in the greatest streamflow reduction at the watershed outlet (Figure 4-15**a**) and individual subbasin level (Figure 4-15b). This ultimately allows for strategic identification of locations for wetland restoration and the specific area and depth of wetland that should be implemented. High, medium, and low impact areas for wetland restoration were identified using the Jenks natural breaks classification method, which arranges values into classes based on natural groupings inherent in the data, while maximizing differences between classes (Jenks, 1967). In this specific watershed, high priority locations for wetland restoration were similar when targeting streamflow reduction at the watershed outlet and individual subbasin outlets. However, this may not be the case when applied to different geographic regions.

The number of subbasins selected for each wetland area/depth combination resulting in the greatest streamflow reduction is presented for the watershed outlet (Table 6) and individual subbasin outlets (Table 7). Wetlands of 500 ha were selected most often for both the watershed outlet and subbasin outlets (45 and 55 subbasins, respectively). Meanwhile, the depths selected most frequently were 15 cm for the watershed outlet and 91 cm for subbasin outlets. The most frequently selected combination was a 500 ha wetland of 15 cm depth for flow reduction at the watershed outlet, while for subbasin outlets 500 ha with 15 cm or 91 cm were selected most often. Despite the identification of similar high priority locations for the watershed and subbasin outlets, the depth and area of wetland implementation can be different to achieve maximum flow reduction. Finally, there is not a definitive optimal wetland area and depth combination, but it is apparent that wetlands of larger area and smaller depth were selected most frequently as being effective in streamflow reduction.

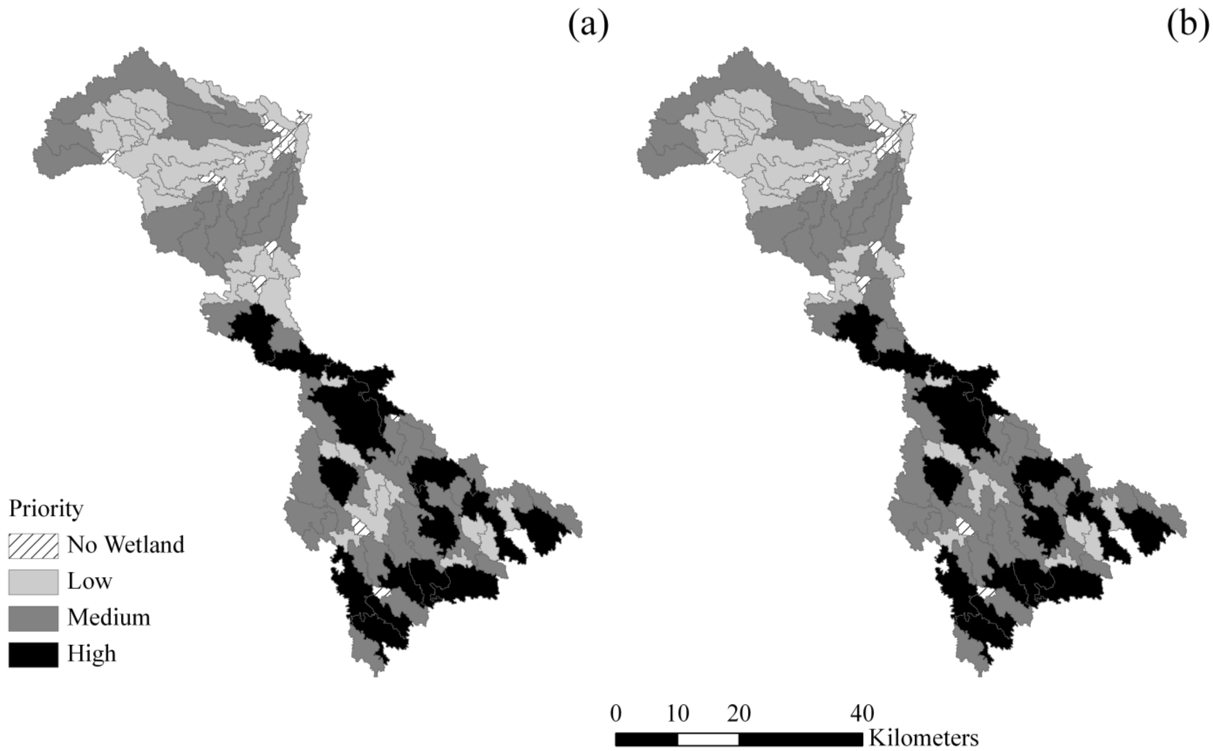

Figure 4-15. High, medium, and low impact areas for wetland restoration scenarios

Table 4-6. Wetland area/depth combinations selected as optimal for streamflow reduction at the watershed outlet

Area/depth	15 cm	30 cm	61 cm	91 cm	Total
50 ha	14	0	0	0	14
100 ha	0	0	2	0	2
250 ha	17	2	7	2	28
500 ha	33	3	5	4	45
Total	64	5	14	6	89

Table 4-7. Wetland area/depth combinations selected as optimal for streamflow reduction at the individual subbasin outlet

Area/depth	15 cm	30 cm	61 cm	91 cm	Total
50 ha	2	0	1	0	3
100 ha	1	0	1	6	8
250 ha	3	1	4	15	23
500 ha	26	3	3	23	55
Total	32	4	9	44	89

4.5 Conclusion

This study examined the impact of wetland area, depth, and placement on streamflow and peak flow reduction at the watershed scale. Using the SWAT model, the impacts of various wetland depths, areas, and location placements were examined. The specific objectives of this study were to evaluate the impacts of wetland areas (50, 100, 250, and 500 ha), depths (15, 30, 61, and 91 cm), and (3) wetland placement on streamflow characteristics at the watershed and individual subbasin outlets.

Wetland area was determined to be an important factor in reducing streamflow at the watershed and subbasin level. Greater wetland areas (250 and 500 ha) resulted in greater streamflow reductions, likely because of increased storage volume, while the smallest area (50 ha) had negligible impacts on streamflow reduction. In addition, there were statistically significant differences between all wetland area scenarios at the watershed outlet. However, wetland depth was found to have minimal impact on reducing streamflow. In the statistical analysis, it was found that all depths resulted in no significant difference in streamflow at the watershed outlet.

Wetland placement based on stream order demonstrated that average streamflow changes at the watershed outlet were insensitive for wetland areas less than 500 ha. However, at 500 ha wetland restoration scenarios on first and fourth order streams had greater streamflow reductions. At the subbasin level, wetland implementation on first order streams was most effective for all depths and areas. Therefore, wetland restoration in headwater streams and locations near the watershed outlet were determined to be most beneficial for streamflow reduction. However, due to the comparatively large variation in subbasin level streamflow reduction on first order streams, caution should be exercised when implementing wetlands in these areas.

Average daily peak flows and events at the watershed outlet were negligibly impacted by wetland implementation. However, average annual maximum peak flow was slightly reduced due to wetland restoration. In addition, both increasing area and depth were somewhat effective in reducing average annual maximum peak flow magnitude.

Finally, high priority areas for wetland restoration were determined based selection of the smallest area and depth that resulted in the greatest streamflow reduction at the watershed and individual subbasin outlets. Despite the similarity between watershed and subbasin level high priority maps, the area and depth combinations to achieve to maximum streamflow reductions were often different. This distinction is important for the decision making process, as wetland implementation size will change based on project goals and resources, although location may not.

The findings of this study will provide an understanding of wetland functions in controlling streamflow. In addition, it explores the manner in which optimal wetland area and depth can be selected for maximum benefit at the watershed and subbasin scale. Although optimal wetland area and depth were identified, their impacts on streamflow and peak reduction at the watershed outlet are generally minimal. However, these results may vary regionally based on watershed physiographical characteristics. Meanwhile, streamflow reductions are just one reason for wetland restoration; water quality improvements are an even more widespread motivation. Therefore, future studies should incorporate water quality parameters to compare restoration scenarios in achieving watershed-scale management goals.

5 ASSESSING THE SIGNIFICANCE OF WETLAND RESTORATION SCENARIOS ON A SEDIMENT MITIGATION PLAN

5.1 ABSTRACT

Wetlands have many environmental, social, and economic values. However, due to accelerated land use change and lack of understanding of the functions of wetland ecosystems, they have deteriorated, if not lost in many areas worldwide. Meanwhile, current functional wetland assessment techniques only provide rough estimation, where in most cases are site specific and qualitative. The overall goal of this project is to examine the sediment reduction benefit of wetland implementation scenarios both at subbasin and watershed scales. Two set of models were used to accomplish this goal. First, a watershed model – the Soil and Water Assessment Tool (SWAT), was employed to estimate sediment load at the subbasin scale. However, due to limitations of wetland functions of SWAT model, a second model – the System for Urban Stormwater Treatment and Analysis Integration (SUSTAIN) was used. The sediment load generated for each subbasin was incorporated in the SUSTAIN model. This allows for evaluating sediment reduction capability of wetlands at subbasin level. Next, a portion of sediment not treated by a wetland was fed back to the SWAT model and routed to the watershed outlet. The impacts of four different wetland surface areas (0.40, 0.81, 2, and 4 ha) on sediment load mitigation were examined one-at-the-time for all subbasins within the River Raisin watershed located in southeastern Michigan and northeastern Ohio. Comparison of the sediment reductions due to different wetland restoration scenarios reveals the importance of wetland placement in a watershed. In general, rate of streamflow reduction resulting from wetland implementation is higher than sediment reduction at the subbasin level but more comparable at the watershed level. In addition, clusters of wetlands installed at the distance of 150 - 200 stream

km from the outlet outperformed other clustered wetlands at closer and farther distances. Wetlands associated with 1st order streams performed better at the subbasin level, while wetlands located at 4th order streams performed better at the watershed level. Considering environmental and economic issues of wetland restoration scenarios revealed that the 0.4 ha wetland was the most suitable for subbasin and watershed levels outcomes due to the significantly lower cost of 0.4 ha wetland installation and maintenance than other scenarios.

5.2 INTRODUCTION

Wetlands perform essential hydrological, geochemical, and biological functions at the watershed level (De Laney, 1995; Hart, 1995). They have the capacity to significantly reduce nutrients, sediments, and other pollutant concentrations produced from runoff under different environmental conditions at the watershed scale (Kadlec, 1993; Mander et al., 2000; Trepel and Palmeri, 2002; Jordan et al., 2003; Arheimer et al., 2004; Skagen et al., 2008, Wang et al., 2010, Fan et al. 2012). Processes that contribute to pollutant removal in wetlands are chemical, physical, and biological in nature (Kadlec et al., 2008). Chemical processes include precipitation of phosphorus by iron, aluminum or calcium, and precipitation of heavy metals (Nilsson et al., 2011). Additionally, wetlands facilitate chemical transformation of nitrogen, which leads to the release of nitrogen to the atmosphere (Vymazal, 2007). Physically, wetland vegetation substantially slows runoff, leading to deposition of mineral and organic particles and adsorbed contaminants (Carter, 1996). Wetland microbial activity is a biological process that results in decomposition of organic matter removal of nitrogen through microbial transformation (nitrification-denitrification) (Brix, 1993). Plant uptake of organic chemicals into plant tissue is another biological process attributed to wetlands' ability to treat pollutants (Ryan et al., 1988). Furthermore, wetlands provides many ecosystem services, especially related to water quantity

(Luecke 1993; Comin et al., 1997; Keddy, 2000, Ramsar, 2004). Wetlands are effective in catching, retaining, and filtering runoff water generated from heavy rainfall or snowmelt events and promoting groundwater infiltration, which helps reduce river peak flow (Luecke 1993; Comin et al., 1997; Keddy, 2000; Ramsar, 2004).

Wetland restoration and construction technologies for the treatment of pollutants is an emerging field (EPA, 2000; Schröder et al., 2007). A robust understanding of pollutant removal processes and wetland environmental characteristics is needed for conservation, restoration, planning, and design purposes. For this reason wetland water quality improvement capabilities have been studied for different types of wetlands in specific settings, as described above. However, the challenge to optimize ideal restoration conditions on a larger scale persists due to the complexity of wetlands and pollution transport processes at the watershed scale.

Researchers use numerous models to simulate pollutant transport at catchment and watershed scales. The Soil and Water Assessment Tool (SWAT) is one of the most widely used model in watershed and river basin simulation (Gassman, 2014). Arnold et al. (2001) used SWAT to simulate the water budget in a constructed wetland in Texas, where the model was modified to include the interaction between ponded water in the wetland and the soil profile and shallow aquifer. Wang et al. (2008) developed the hydrologic equivalent wetland (HEW) method to represent wetlands in SWAT model and applied the method to successfully simulate streamflow in a watershed located in Minnesota. Liu et al. (2008) developed a SWAT extension to simulate flow and sediment in a riparian wetland, but did not validate the model due to the limitation of observed data. Wu and Johnston (2008) compared SWAT performance between forested and a wetland/lake dominated watershed in Michigan, and reported satisfactory model calibration but discrepancies in summer streamflow prediction. Wang et al. (2010) applied the

HEW method to estimate streamflow, sediment, total nitrogen (TN), and total phosphorus (TP) loads under wetland restoration and conservation scenarios in Manitoba, Canada. However, the authors only calibrated the model for streamflow. Therefore, evaluation of wetland performance for sediment, TN, and TP is highly uncertain. Feng et al. (2013) incorporated a wetland module into SWAT to simulate wetland hydrology in northeast China, where the method performed well in reconstructing wetland hydrological processes. Martinez-Martinez et al. (2014) used SWAT to simulate streamflow rates and peaks under wetland restoration scenarios and reported that average streamflow fluctuation at the watershed outlet is more sensitive to wetland area than depth. Numerous other researchers have used different modeling approaches to incorporate wetlands in their simulations such as the Hydrological Simulation Program-Fortran (HSPF) (Schwar et al., 1998; Zhang et al., 2009), MIKE-SHE (Thompson et al., 2004; Zacharias et al., 2005; Dai et al., 2010), DRAINMOD (Caldwell et al., 2007; Skaggs et al., 1995; Jia and Luo, 2009), and SWMM (Obropta et al., 2008; Tsihrintzis et al., 1998; Koo et al., 2013). Overall, most studies have only considered wetland hydrology in watershed scale modeling and have either ignored or not calibrated the model for sediment and nutrients to simulate the impact of wetland restoration scenarios on pollutant treatment, as such a task is still a challenge to scientists (Wang et al., 2008).

Among physically-based watershed/water quality models, SWAT is a comprehensive model that combines spatial and temporal analysis, is open source, and has strong model support, making it one of the most widely used water quality models in watershed and river basin modeling (Gassman, 2014; Srinivasan et al., 1998). However, a major drawback of using SWAT for watershed scale wetland modeling is that SWAT assumes a completely mixed wetland system in pollutant routing. In addition, SWAT ignores nutrient transformation in simulating nutrient removal in wetlands, ponds, and reservoirs and considers settling as the sole method of nutrient removal (Neitsch et al., 2011). In order to solve this problem, we proposed to couple SWAT with a second model capable of addressing these issues. The System for Urban Stormwater Treatment and Analysis INtegration (SUSTAIN) model (USEPA, 2009) allow users to simulate wetlands as either a plug flow reactor or a continuously stirred tank reactor (CSTR) in series with a user-defined number of CSTRs. In addition, SUSTAIN models pollutant removal by either first-order order decay or a modified kinetic model (K-C*) (Shoemaker et al., 2009). This research addresses the challenges of using a hybrid of two water quality models to examine the sediment reduction benefits of wetland implementation scenarios at subbasin and watershed scales. The specific objectives of this project are to: 1) assess the impacts of wetland restorations scenarios on flow and sediment, 2) determine the role of wetland placement in watershed sediment dynamics by considering the distance to the outlet and stream order concept, and 3) evaluate the environmental and economic aspects of wetland restoration scenarios at the subbasin and watershed scale.

5.3 MATERIALS AND METHODS

5.3.1 Study Area

The River Raisin watershed (Hydrologic Unit Code 04100002) is located primarily in southeastern Michigan, with a small portion located in northern Ohio (Figure 5-1**).** The River Raisin watershed drains approximately 2681 km^2 into Lake Erie. The watershed is predominantly agricultural, covering approximately 66% of the total watershed area (CDL, 2012). The primary crops grown in the watershed are corn, soybeans, and wheat. The remaining land cover is 13% forest, 12% urban, 7% wetlands, 1% range grasses, and 1% water (CDL, 2012).

The River Raisin watershed is characterized by hilly to moderately rolling topography in the western and northwestern regions and by relatively flat terrain in the southeast. Soils are characterized as having slopes of 0-5 percent. (Knutilla and Allen, 1975). Sandy loams, loams, and clay loam soils with moderate to high infiltration rates dominate the upstream northwestern portion of the River Raisin watershed. The streams in this portion of the watershed have more stable flows and consistent groundwater recharge. Meanwhile, the southeastern portion of the watershed is dominated by primarily clays, clay loams, and silty clays with low to very low permeability and slow infiltration rates (Dodge, 1998).

The River Raisin watershed was selected due to its significant variation in soil types, land use patterns, topography and geology. Historically, this watershed was a swamp (wetland) with flat topography and muck and clay soils (Dodge, 1998). Comparison of the current land use map (NLCD, 2001) with prehistoric land use (MNFI, 2014) reveals a loss of 59% of the woody wetlands and 91% of emergent herbaceous wetlands since the mid-1800s. Due to land use changes (tiling and drainage) the watershed is now highly agricultural.

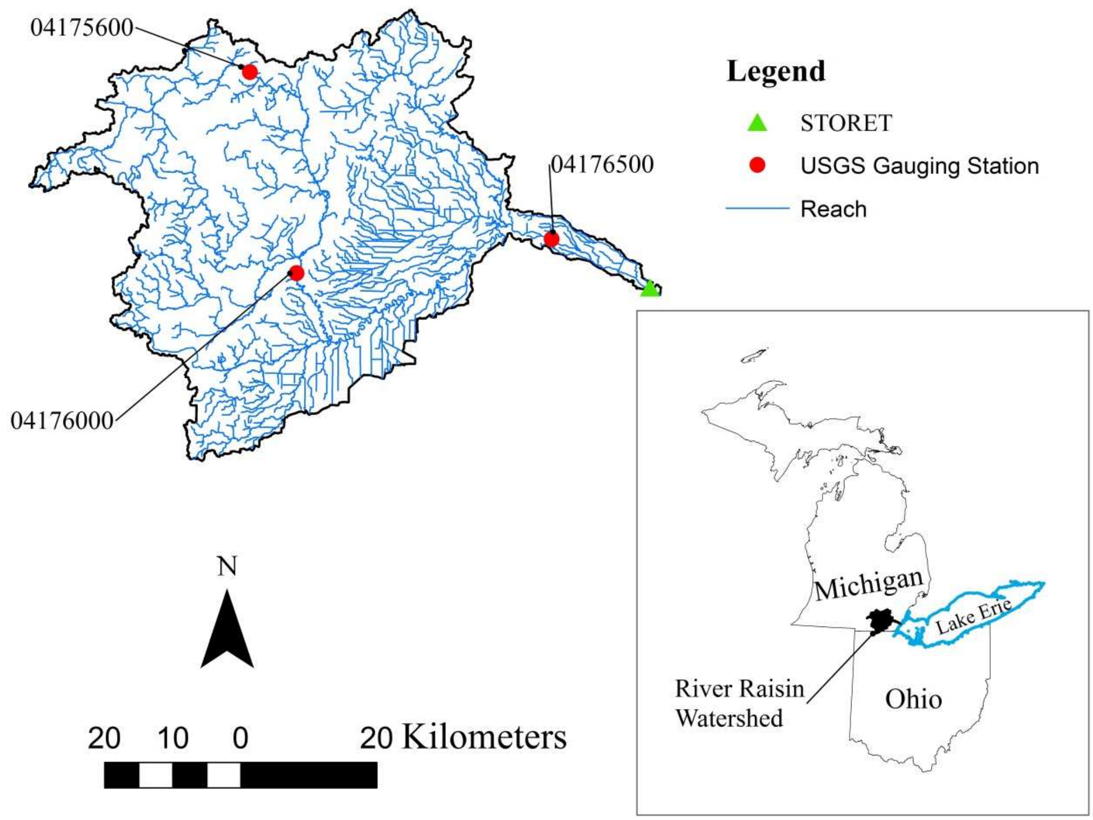

Figure 5-1. Study area and location of the monitoring stations

5.3.2 Models

Wetlands are complex, diverse, and dynamic ecosystems, and watershed-scale wetland assessment is a challenge currently faced by watershed managers and conservationists. Due to limitations of watershed models in quantifying the benefits of wetland restoration strategies, SUSTAIN was used to evaluate the local scale benefit of wetland installation in reducing runoff and sediment loadings, while SWAT was used to estimate sediment loads and runoff to and from wetlands all the way to the watershed outlet. Linking SUSTAIN and SWAT to model water quantity and quality is a unique approach to evaluate wetland behavior at multiple spatial scales.

5.3.2.1 SUSTAIN Model

The SUSTAIN model (version 1.2) is a decision support system developed by the Tetra Tech for the United States Environmental Protection Agency (USEPA) to evaluate water quality management alternatives (USEPA, 2009). SUSTAIN simulates the ability of individual or a combination of best management practices to reduce peak streamflows and sediment and nutrient concentrations. Data required by SUSTAIN includes: climate, soils, landuse, topography, delineated subbasins, stream network, stream geometry, streamflow, water quality, and best management practices.

SUSTAIN provides users with multiple BMP process simulation methods. Flow routing processes can be simulated either by a stage-outflow storage routing using weir and/or orifice equations or by kinematic routing through solving the Coupled Continuity equation and Manning's equation (Linsley et al., 1992). Infiltration processes can be simulated by either the Holtan-Lopez equation or the Green-Ampt method (Maidment, 1993; Huber and Dickison, 1988). Evapotranspiration (ET) processes can either be simulated using a constant ET rate or through calculating potential ET and actual ET (Bicknell et al., 1997; Maidment, 1993). Pollutant routing processes are simulated using completely mixed, continuous stirred tank reactors in series (CSTRs) (Wong et al., 2002), or plug flow reactors (Persson et al., 1999). The first-order decay, Kadlec and Knight's (K'-C*) method (Kadlec and Knight, 1996), or sediment settling under quiescent and turbulent conditions (James et al., 2002) can be used to simulate pollutant removal.

Wetland simulation processes were performed according to the SUSTAIN user's manual recommendations (Alvi et al. 2009). Wetlands were simulated as wet ponds and flow and sediment output from SWAT were used as wetland inputs. Wetlands in SUSTAIN are defined by size specification, pollutant removal method, and removal capacity. In this study all wetland alternatives were simulated with a relatively shallow depth (0.3048 m); and therefore, it is reasonable to assume vegetation is present. The Holtan infiltration method accounts for vegetation effects and therefore was selected for infiltration simulation. The *K*-*C** model, proposed by Kadled and Knight (1996), is a widely used pollutant reduction model in wetlands (Park and Roesner, 2009) and was used to estimate sediment removal. In this model, the total suspended solids removal rate (*K*) in wetlands was considered as 5,000 m/yr and the background concentration value (*C**) was adjusted to 6 mg/L (eWater, 2012). Sediment routing was modeled using 1.5 CSTR in series as recommended by the SUSTAIN model manual (USEPA, 2009). Depth of the wetland was defined by the height of the weir. As suggested by USDA-NRCS (2009), length-to-width ratio of the wetlands was maintained at 2:1. Percentage of sand, silt and clay in the inflow were obtained from the Soil Survey Geographic (SSURGO) dataset for the upper layer soils (USDA-NRCS, 2013b). Additional parameters and their sources are listed in Table 9-1. A sensitivity analysis of key SUSTAIN parameters was performed and the results are presented in Table 9-2.

5.3.2.2 SWAT Model

The SWAT model was developed by the US Department of Agriculture (USDA)-Agricultural Research Service (Arnold et al., 1998) and is able to simulate watershed hydrology, sediment and water quality, pesticide fate and transport, channel erosion, and agricultural management practices. SWAT delineates a watershed into subbasins based on a digital elevation model and river network. Following delineation, the subbasins are further divided into hydrologic response units (HRUs). An HRU is an area of homogeneous land use, soil type, slope, and management practices. The SWAT model has multiple algorithms that simulate water quality and quantity within the watershed, as described in Arnold et al. (2012). Land-based calculations in SWAT (hydrology and pollution generation) are completed at the HRU level and aggregated to the subbasin level. Pollutants are then transported to the nearest reach and routed through the river network.

In this study SWAT 2012 (v. 591) was used. The SWAT model data inputs include: precipitation and temperature data obtained from the National Climate Data Center (NCDC), soils (SSURGO), landuse (Cropland Data Layer (CDL), 30 m resolution), DEM (US Geological Survey (USGS) National Elevation Dataset (NED), 30 m resolution), watershed and subbasins, stream networks from the Michigan Institute of Fisheries Research based on the USGS National Hydrography Dataset), streamflow measurements (US Geological Survey (USGS) gauging stations), and water quality grab samples (Environmental Protection Agency STOrage and RETrieval (STORET) data warehouse).

The River Raisin SWAT model was calibrated for streamflow at three locations (USGS stations 04175600, 04176000, and 04176500) and sediment concentration at the STORET site 580046 (figure 1). During the calibration process model parameters were adjusted to improve model predictions of observed streamflow and sediment concentration data (Table 5-8 and Table 5-9). The SWAT model streamflow calibration was performed on a daily basis from 1996-2000 and validated from 2001-2005. The sediment calibration and validation was performed on grab sample data from 2000-2002 and 2003-2005, respectively. This period was selected based on limited availability of sediment data.

Three goodness-of-fit criteria were used to evaluate SWAT model performance: Nash-Sutcliffe coefficient of efficiency (NSE), root mean square error-observations standard deviation ratio (RSR), and percent bias (PBIAS). NSE ranges from - ∞ to 1 (where 1 is optimal) and is a normalized statistic that determines the relative magnitude of the residual variance compared to the measured data variance (Moriasi et al., 2007). NSE is considered satisfactory when greater than 0.50 for a monthly time step. RSR standardizes RMSE using the observations standard deviation, and ranges from zero to large positive values, with zero representing perfect model simulation (indicating zero residual variation). RSR values are considered satisfactory at less than 0.70 for a monthly time step (Moriasi et al., 2007). PBIAS measures the tendency of simulated data to over- or under-predict compared to observed data. The optimal PBIAS is 0, with low values indicating accurate model simulation. Values between ±25% are satisfactory for streamflow and between ±55% for sediment at a monthly time step (Moriasi et al., 2007).

5.3.3 SUSTAIN/SWAT Hyrbid Model Approach

This study integrates SWAT and SUSTAIN to evaluate the impacts of wetland restoration on sediment loads at subbasin and watershed scales**.** Figure 5-2 **shows** the SWAT and SUSTAIN model integration schema. First, the calibrated/validated SWAT model was used to estimate surface runoff and sediment yield for all 1,235 subbasins in the River Raisin watershed. Surface runoff and sediment yield from the SWAT model were input into the SUSTAIN model wetlands.

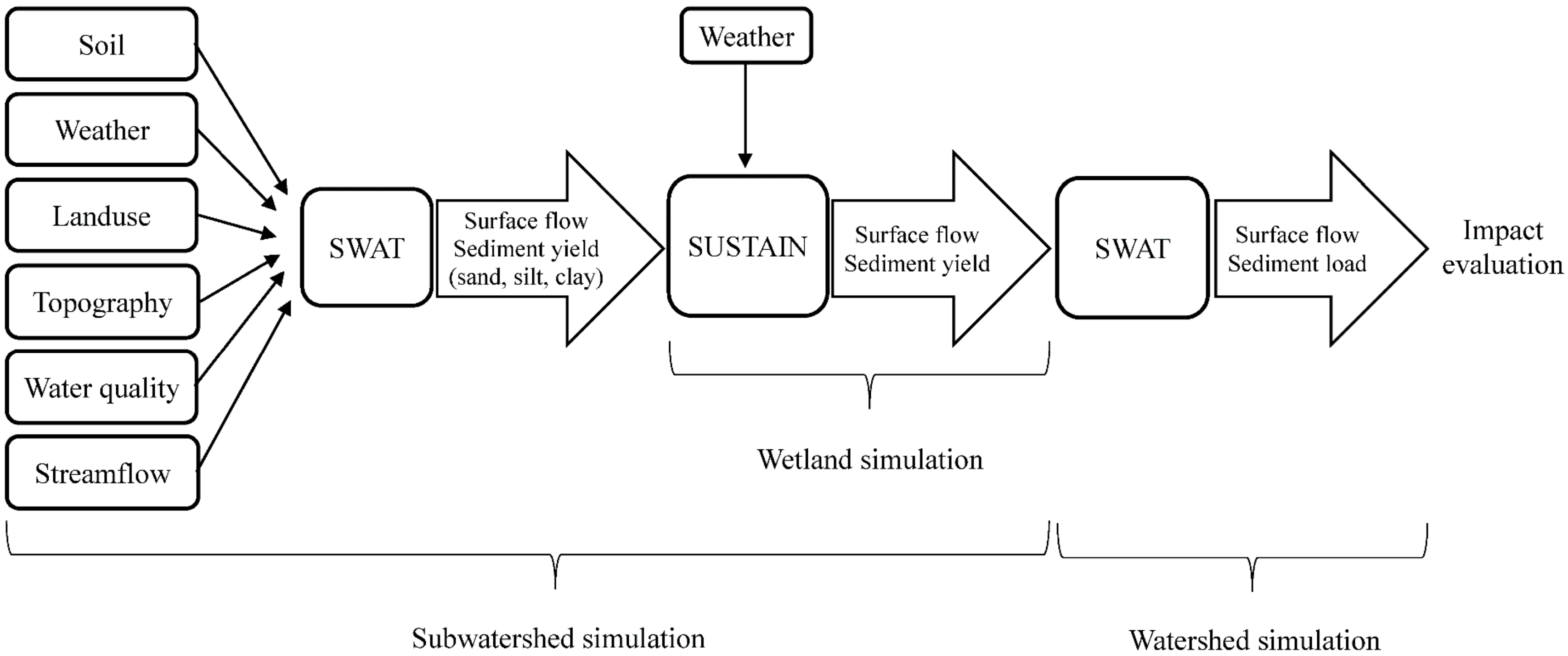

Figure 5-2. SWAT – SUSTAIN models integration schema

Sediment removal and surface runoff attenuation were simulated using SUSTAIN. Finally, the SUSTAIN outputs (surface runoff and sediment load) were input back into the calibrated SWAT model as a point source for each subbasin and routed to the watershed outlet. However, it is challenging to seamlessly integrate the results of these two models. As SWAT runs on daily time step and SUSTAIN requires hourly input data, SWAT outputs for each subbasin were converted to average hourly data prior to use in SUSTAIN. Finally, after completing the simulation, SUSTAIN outputs were aggregated to daily loads and used as point source inputs in SWAT. This process was repeated one-at-a time for all wetland sizes for each subbasin. The SWAT and SUSTAIN models were run for 10 years from 1996 to 2005.

5.3.4 Wetland Restoration Scenarios

Wetland surface area, location, and depth are critical components of wetland restoration planning to abate sediment loads at the watershed scale. Multiple wetland restoration scenarios were developed to assess importance of wetland physical factors under restoration conditions. Restoration scenarios were formulated by using a combination of wetland areas and placement within the River Raisin watershed. This approach was based on 0.4, 0.81, 2 and 4 ha (1, 2, 5 and 10 acres) of wetland restoration with an average wetland depth of 30.48 cm (12 in). Wetland surface area selection was based on wetland restoration and management literature resources obtained from government entities and guidance from the USDA-NRCS National Engineering Handbook (USDA-NRCS, 2003, USDA-NRCS, 2009). Different wetland restoration depths were not explored based on findings from Martinez- Martinez, et al. (2014), where wetland depths were found to be less important than surface area with respect to surface runoff reduction at the subbasin and watershed scales. Each wetland surface area was implemented one at a time for each subbasin in SUSTAIN and a sediment reduction capacity was generated per individual scenario. Wetlands were not implemented in subbasins with a total area less than 4 ha (the maximum wetland restoration size determined for the study), which resulted in 4480 simulations. SUSTAIN simulation of the wetlands were performed at an hourly time step and were based on the assumption that the restored wetlands were placed at the outlet of each subbasin.

5.3.5 Environmental and Economic Aspects of Wetland Implementation

Environmental and economic factors were considered during the selection of the most effective wetland surface area for each subbasin based on sediment reduction at the watershed outlet using analytical hierarchy process (AHP). AHP is an optimization algorithm that can solve complex decision-making problems (Linkov and Steevens, 2013) by using evaluation criteria based on expert knowledge and pairwise comparison (Young et al., 2009). The AHP algorithm was developed by Saaty (1980) and has been widely used in diverse fields such storm water BMP selection (Young et al., 2011), environmental vulnerability assessment of river basins (Chang and Chao, 2011), land suitability assessment for irrigation (Chen et al., 2010), integrated watershed management (Biswas et al., 2012), landfill site selection (Sener et al., 2010), and solvent selection in pharmaceutical processes (Perez-Vega et., 2011). However, to the best of our knowledge, using AHP for wetland placement in large and complex watersheds such as the River Raisin Watershed has not been explored.

5.3.6 Environmental Aspects of Wetland Implementation

Four different wetland surface areas (0.40, 0.81, 2 and 4 ha) were implemented in each subbasin one at a time, and the resulting sediment load was estimated at the watershed outlet. The sediment reduction at the watershed outlet following wetland implementation in each subbasin was calculated by subtracting the sediment load after implementation from the base sediment load where wetlands were not implemented in the watershed.

5.3.7 Economic Aspects of Wetland Implementation

The implementation costs of the four wetland scenarios were obtained from the United States Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS) Field Office Technical Guide (USDA-NRCS, 2013a). The USDA-NRCS implementation cost includes, but is not limited to, site preparation, excavation, backfilling, grading and finishing, vegetation planting, trees and shrubs establishment, and operations and maintenance (Biebighauser, 2007; Biebighauser, 2011; USDA-NRCS, 2013). The cost and needs of each implement component is site specific and the costs can be lower or higher than listed. The SUSTAIN model includes a wetland restoration cost database; however, it was developed using national sources and is not site specific (Alvi et al. 2009). In addition, SUSTAIN does not account for costs associated with wetland installation and maintenance. Therefore, economic analysis was performed independently (Table 9-6, Table 9-7, Table 9-8, and Table 9-9). The tables include three different cost estimates (low, average, and high) based on the 2013 statewide survey. The cost differences are due to site locations, existing site conditions, labor cost, and other factors. The average wetland implementation costs over a 10-year lifespan of wetland operation for 0.4, 0.81, 2 and 4 ha with a depth of 30.48 cm are $1,490, $2,874, $6,947, and $13,656, respectively.

5.3.8 Wetland Selection Considering Environmental and Economic Factors

AHP was used to integrate two different factors (environmental benefits and economic cost) with different units in the selection of most suitable wetlands for sediment reduction at the watershed outlet. The AHP is a structured technique consisting of four steps to convert the complex decision making processes into an algorithm: 1) constructing a pairwise matrix, 2) computing the priority vector, 3) calculating the consistency ratio, and 4) ranking the alternatives (Saaty, 1980; Young et al., 2009; Giri and Nejadhashemi, 2014). In order to better describe the AHP process, subbasin 2 (Figure 5-3) was selected as an example.

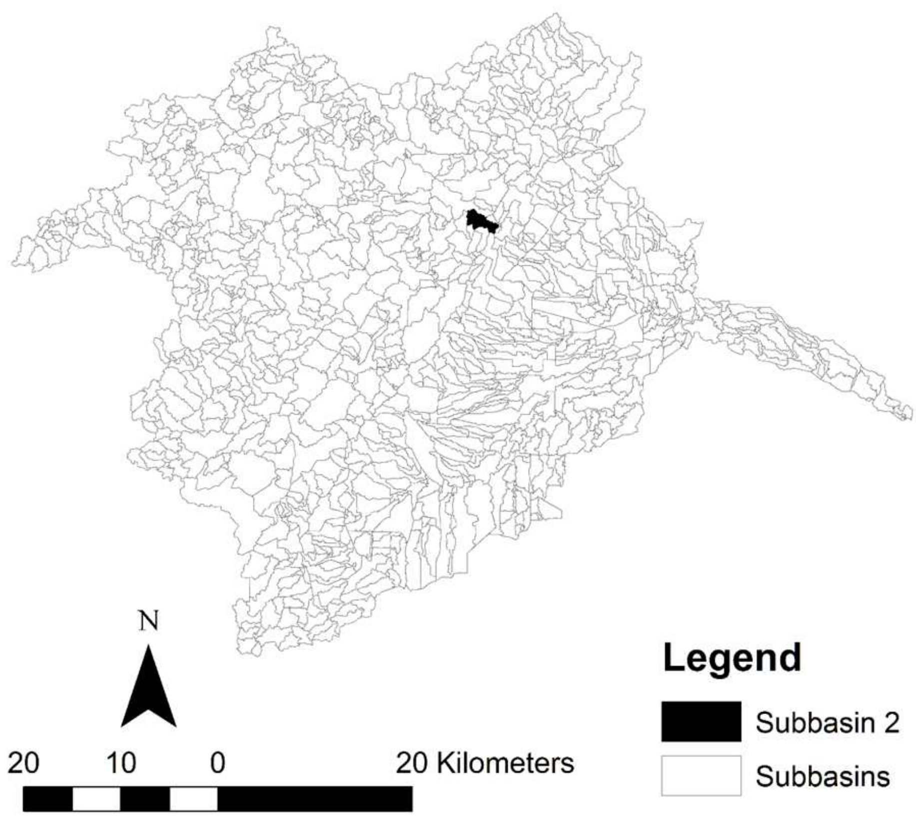

Figure 5-3. River Raisin watershed subbasin 2.

Step 1) Constructing the pairwise comparison matrix: A pairwise comparison matrix was constructed for each subbasin based on environmental benefits of sediment reduction at the watershed outlet by four wetland surface areas. Sediment reduction was calculated by subtracting sediment loads at the outlet before and after wetland implementation. The process was repeated for all subbasins. In the pairwise matrix all rows and all columns were compared. This allows that relative importance of each alternative to be compared to other alternatives on a scale of 1/9 to nine. In this matrix, one means that two alternatives are equally important. Nine indicates that one alternative is absolutely more important than the other, while the reverse is true for 1/9. The pairwise comparison matrix for subbasin 2 is presented in Table 5-1.

Table 5-1. Pairwise comparison matrix developed for Subbasin 2 based on sediment reduction at the watershed outlet.

Wetland Size (ha)	0.4	0.81	2	4
0.40	1.0000	0.9735	0.9168	0.8475
0.81	1.0272	1.0000	0.9417	0.8705
2.00	1.0907	1.0619	1.0000	0.9243
4.00	1.1800	1.1488	1.0819	1.0000

Step 2) Computation of priority vector: Before calculating the priority vector, the principal eigenvector is constructed. To construct the principal eigenvector, first each column within the pairwise comparison matrix developed in the step 1 was summed. Then each element in the matrix is divided by the sum of the value for the corresponding column by dividing each column in the pairwise comparison matrix by sum of its values. Therefore, the total sum of each

column should be one (Table 5-2). The average of each row/column in the matrix was calculated, giving the priority vectors (Table 5-2). The priority vectors were calculated as 0.2327, 0.2390, 0.2538, and 0.2746 for 0.4 ha, 0.81 ha, 2 ha and 4 ha wetlands, respectively. The remaining priority vectors were calculated using the AHP extension developed for ArcGIS by Marinoni (2009).

Table 5-2. Priority vector calculation based on sediment reduction at watershed outlet for the Subbasin 2.

Wetland Size (ha)	0.4	0.81	2	4	Row Average (Priority Vector)
0.40	0.2327	0.2327	0.2327	0.2327	0.2327
0.81	0.2390	0.2390	0.2390	0.2390	0.2390
2.00	0.2538	0.2538	0.2538	0.2538	0.2538
4.00	0.2746	0.2746	0.2746	0.2746	0.2746
Total	1.000	1.000	1.000	1.000	1.000

Step 3) Computation of consistency ratio: The consistency ratio is a measure of the consistency of importance between different elements of a priority matrix. This step verifies whether the priory vectors obtained in the previous step are acceptable. If the consistency ratio is less than 0.1, then the pairwise comparison matrix developed in step 1 is consistent and the priority vectors are acceptable (Saaty, 1980). If it is greater than 0.1, the pairwise comparison matrix should be adjusted and checked for consistency and logical relationships. To calculate the consistency ratio, the original pairwise matrix constructed in step 1 was multiplied with the priority vector, resulting in a new matrix (Matrix A, shown below). This new matrix was divided by the priority vectors to create Matrix B, shown below.

$$\underbrace{\begin{pmatrix} 1.0000 & 0.9735 & 0.9168 & 0.8475 \\ 1.0272 & 1.0000 & 0.9417 & 0.8705 \\ 1.0907 & 1.0619 & 1.0000 & 0.9243 \\ 1.1800 & 1.1488 & 1.0819 & 1.0000 \end{pmatrix}}_{\text{Pairwise Comparison Matrix}} \times \underbrace{\begin{pmatrix} 0.2327 \\ 0.2390 \\ 0.2538 \\ 0.2746 \end{pmatrix}}_{\text{Priority Vector}} = \underbrace{\begin{pmatrix} 0.9307 \\ 0.9560 \\ 1.0151 \\ 1.0982 \end{pmatrix}}_{\text{Matrix A}}$$

$$\underbrace{\begin{pmatrix} 0.9307 \\ 0.9560 \\ 1.0151 \\ 1.0982 \end{pmatrix}}_{\text{Matrix A}} \div \underbrace{\begin{pmatrix} 0.2327 \\ 0.2390 \\ 0.2538 \\ 0.2746 \end{pmatrix}}_{\text{Priority Matrix}} = \underbrace{\begin{pmatrix} 4.0000 \\ 4.0000 \\ 4.0000 \\ 4.0000 \end{pmatrix}}_{\text{Matrix B}}$$

The maximum eigenvalue (λ_{max}) was calculated by averaging all elements in Matrix B. Then, the consistency index (*CI*) was calculated based on equation (1) (Saaty, 1980):

$$CI = \frac{(\lambda_{max} - n)}{(n-1)} \tag{1}$$

Where *n* is to the total number of alternative solutions that are examined (number of rows/columns in the pairwise comparison matrix). In this study, there are four alternatives for wetland surface area. After obtaining the *CI* value, which in this case is zero, the consistency ratio (*CR*) can be calculated using equation 2 (Saaty, 1980):

$$CR = \frac{CI}{\text{Random index}} \tag{2}$$

The random index value for four wetland alternatives was 0.9 from the Saaty scale (Forman, 1990). Therefore, the consistency ratio for subbasin 2 was zero, which indicates that priority vector for subbasin 2 is acceptable.

This procedure was repeated to develop and test the priority vectors for each subbasin based on the average implementation costs for different surface areas of wetlands Table 5-3 to Table 5-6). The priority vector for the wetland implementation costs were calculated as 0.5428, 0.2815, 0.1164, and 0.0592 for wetland surface areas of 0.4, 0.81, 2 and 4 ha, respectively. In addition, the consistency ratio for economic criteria was also zero, which indicates that priority vector for subbasin 2 is acceptable.

Table 5-3. Daily streamflow calibration/validation results

Statistical Measures	**USGS Station 4175600**		**USGS Station 4176000**		**USGS Station 4176500**	
	Calibration	Validation	Calibration	Validation	Calibration	Validation
NSE	0.48	0.47	0.61	0.47	0.72	0.57
PBIAS	23.07%	8.63%	16.35%	13.35%	8.50%	0.13%
RSR	0.72	0.73	0.62	0.73	0.53	0.66

Table 5-4. Daily sediment calibration/validation results

Statistical Measures	**USGS Station 4175600**	
	Calibration	Validation
NSE	0.65	0.46
PBIAS	17.49%	-2.57%
RSR	0.59	0.74

Table 5-5. Statistical comparison of mean flow and sediment reduction for different wetland surface areas at watershed and subbasin level

Wetland area (ha)	Mean reduction at watershed level		Mean reduction at subbasin level	
	Streamflow* (cms)	Sediment* (tons)	Streamflow* (cms)	Sediment* (tons)

0.4	0.010^{a}	17.1^{a}	21.5^{a}	118.9^{a}
0.81	0.010^{ab}	17.4^{a}	40.4^{b}	131.2^{a}
2	0.010^{ab}	18.3^{ab}	90.2^{c}	139.9^{a}
4	0.011^{b}	19.6^{b}	159.2^{d}	141.9^{a}

[*]Streamflow and sediment means followed by the same letters are not significantly different ($\alpha = 0.05$).

Table 5-6. Distance from outlet and sediment reduction

Distance from outlet (km)	Number of subbasins	Average subbasin area (ha)	Average annual sediment yield (ton)	Average slope (%)	Average sediment reduction Watershed level (tons)				Average sediment reduction Subbasin level (tons)			
					0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)	0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)
0-50	159	197.3^{a}	55.6^{a}	1.0^{a}	13.3^{a}	13.6^{a}	14.5^{a}	15.9^{a}	52.0^{a}	54.5^{a}	55.3^{a}	55.4^{a}
50-100	320	237.8^{ac}	166.1^{b}	1.6^{b}	14.9^{ab}	15.2^{ab}	16.2^{a}	17.8^{a}	143.3^{b}	157.0^{b}	164.9^{b}	166.1^{b}
100-150	229	273.9^{c}	204.1^{b}	1.9^{b}	16.2^{ab}	16.5^{ab}	17.5^{a}	19.0^{a}	165.4^{b}	184.3^{b}	199.0^{b}	203.3^{b}
150-200	278	274.6^{c}	176.8^{b}	3.9^{c}	21.9^{c}	22.3^{c}	23.1^{b}	24.3^{b}	143.1^{b}	159.7^{b}	173.0^{b}	176.2^{b}
200-250	134	214.3^{a}	10.7^{a}	5.0^{d}	18.5^{bc}	18.7^{bc}	19.1^{ab}	19.6^{a}	10.3^{a}	10.5^{a}	10.6^{a}	10.6^{a}

[*]Values followed by the same letters are not significantly different ($\alpha = 0.05$).

Step 4) Ranking of alternatives: The priority vectors for different criteria were entered into a decision matrix (Table 5-7), in which the importance weights of the criteria are determined.

Table 5-7. Decision matrix of wetland size alternatives for all criteria developed for watershed analysis (Subbasin 2).

Wetland size (ha)	Sediment reduction	Implementation Cost
0.4	0.2327	0.5428
0.81	0.2390	0.2815
2	0.2538	0.1164
4	0.2746	0.0592

For this example, we assumed a weight of 0.9 for environmental factor (sediment reduction at the outlet) and a weight of 0.1 for economic factor (wetland implementation cost). The final rank of the wetlands were calculated by multiplying the decision matrix by the weights of importance of the criteria.

$$\begin{pmatrix} 0.2327 & 0.5428 \\ 0.2390 & 0.2815 \\ 0.2538 & 0.1164 \\ 0.2746 & 0.0592 \end{pmatrix} \times \begin{pmatrix} 0.9 \\ 0.1 \end{pmatrix} = \begin{pmatrix} 0.2637 \\ 0.2432 \\ 0.2400 \\ 0.2530 \end{pmatrix}$$

From the above result, it can be concluded that 0.4 ha wetland is the most desirable (because it has the largest weight) followed by 4 ha, 0.81 ha, and 2 ha as the final weights are 0.2637, 0.2530, 0.2432, and 0.2400, respectively.

Five scenarios, or combinations, of different environmental and economic weighting factors were developed. The five different scenarios were environmental/economic (0.1/0.9), environmental/economic (0.25/0.75), environmental/economic (0.5/0.5), environmental/economic scenario (0.75/0.25), and environmental/economic (0.9/0.1). For example, the environmental/economic scenario (0.1/0.9) indicates that the environmental factor is given less preference (0.1), while the economic factor is given more importance (0.9). This indicates that when selecting a wetland surface area for sediment reduction the cost of implementation is more important than environmental benefit in the context of watershed scale restoration strategy. The AHP algorithm was developed for all scenarios and the most desirable wetland were selected.

5.3.9 Statistical Analysis

Annual sediment load generated under different wetland restoration scenarios were compared to the base annual sediment load (before wetland implementation) to calculate sediment load reduction. The sediment load reductions obtained by four wetland restoration scenarios were then averaged over the simulation period to obtain the mean annual sediment reduction at both subbasin and watershed level. Statistical analyses were performed to explore the effect of stream order, distance from the outlet, and wetland surface area on mean annual sediment reduction. Mean input sediment loads across different stream orders were also compared. In order to compare the mean differences between sediment loads, the Fisher's Least Significant Difference test was used to perform a multiple pairwise comparison at 5% significance level (α=0.05) using MATLAB 7.12 (R2011a).

5.4 RESULTS AND DISCUSSION

5.4.1 Model Calibration

The results for SWAT streamflow and sediment concentration calibration are presented in Table 5-8 and Table 5-9, respectively. The calibration and validation was evaluated based on the criteria established by Moriasi et al. (2007) and a visual comparison (Figure 5-4**,**Figure 5-5**,** Figure 5-6**,** and Figure 5-7).

Table 5-8. Daily streamflow calibration/validation results

Statistical Measures	**USGS Station 4175600**		**USGS Station 4176000**		**USGS Station 4176500**	
	Calibration	Validation	Calibration	Validation	Calibration	Validation
NSE	0.48	0.47	0.61	0.47	0.72	0.57
PBIAS	23.07%	8.63%	16.35%	13.35%	8.50%	0.13%
RSR	0.72	0.73	0.62	0.73	0.53	0.66

Table 5-9. Daily sediment calibration/validation results

Statistical Measures	**USGS Station 4175600**	
	Calibration	Validation
NSE	0.65	0.46
PBIAS	17.49%	-2.57%
RSR	0.59	0.74

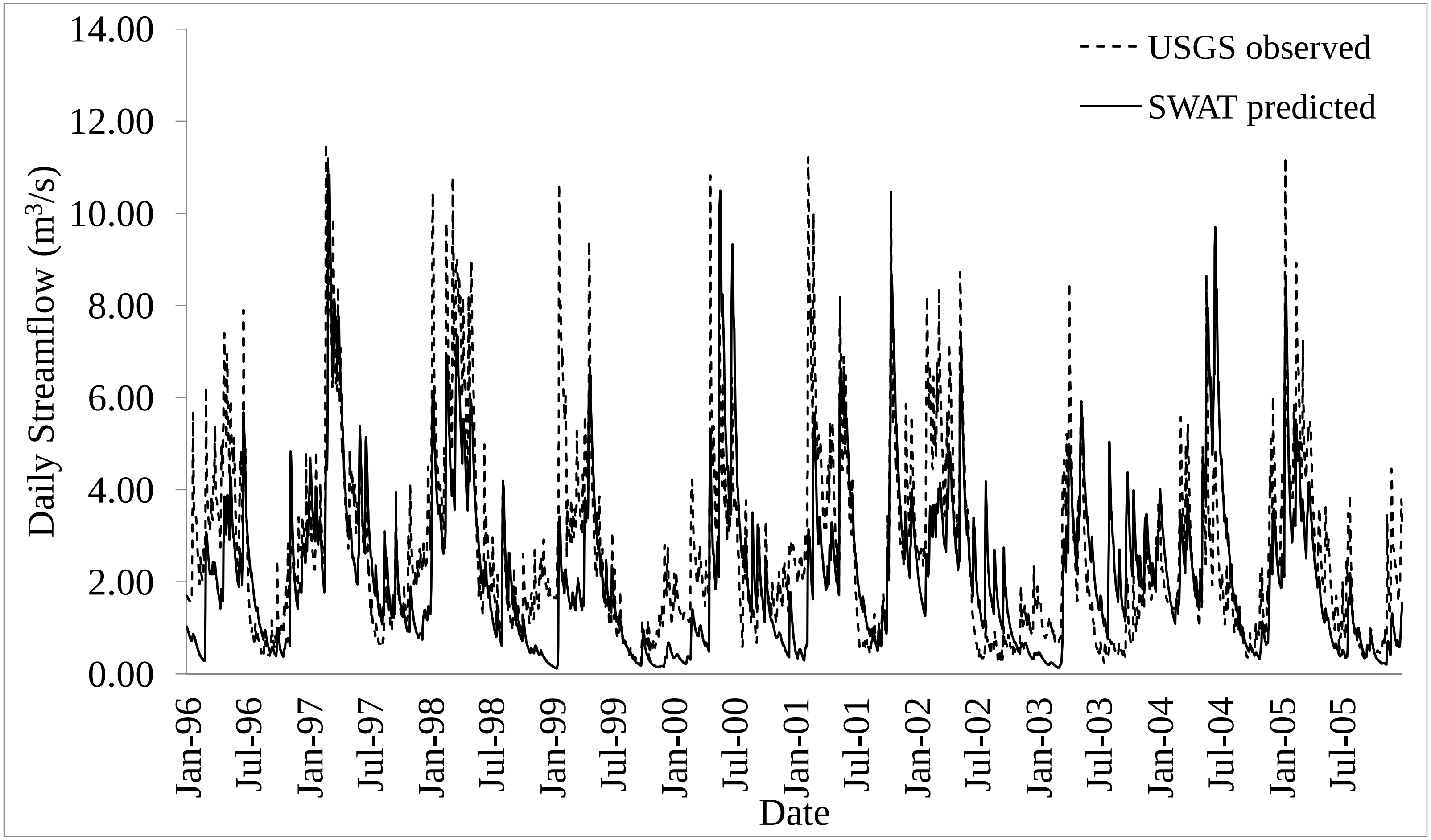

Figure 5-4. Streamflow calibration versus validation for the USGS 04175600 gauging station.

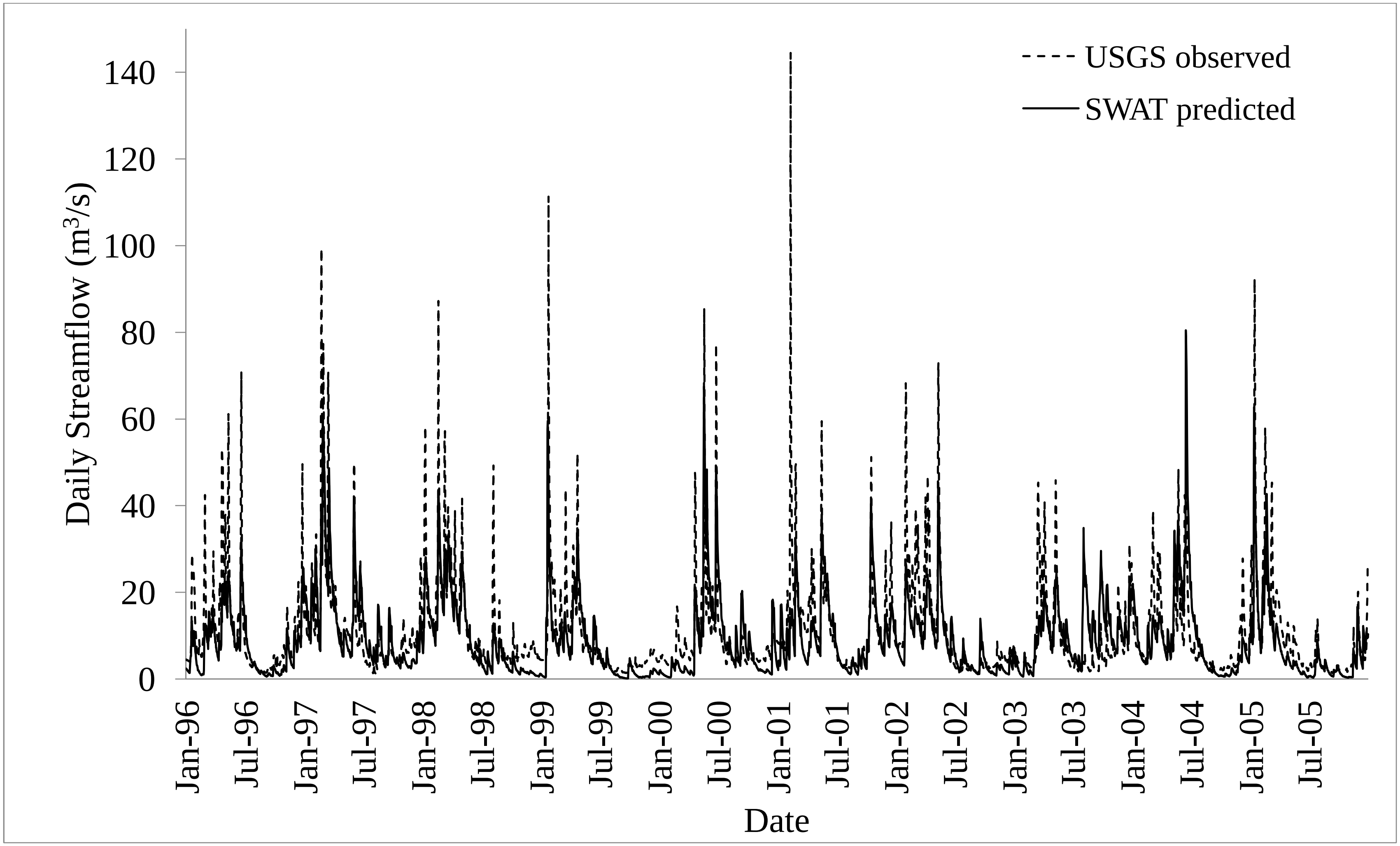

Figure 5-5. Streamflow calibration versus validation for the USGS 04176000 gauging station.

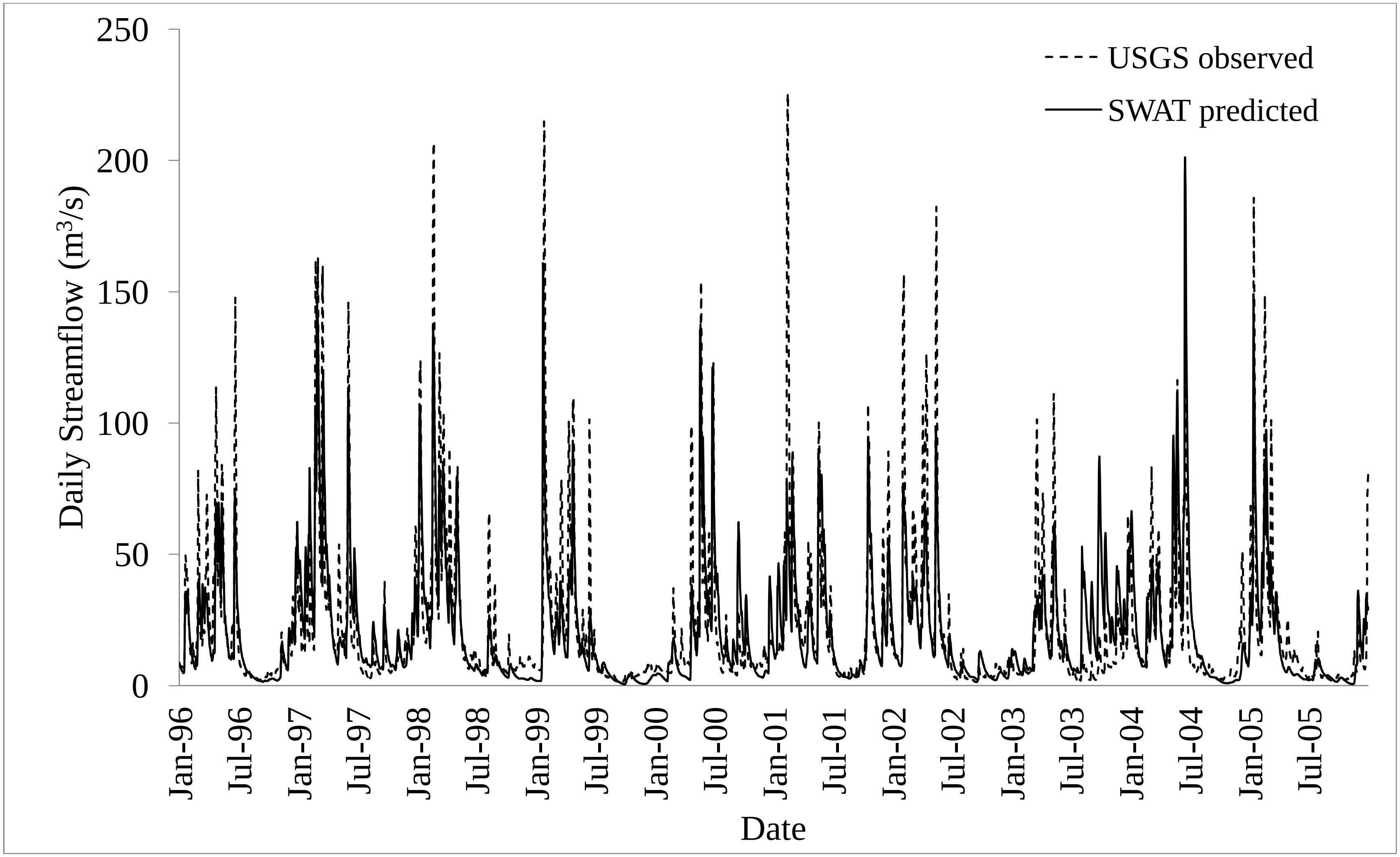

Figure 5-6. Streamflow calibration versus validation for the USGS 04176500 gauging station.

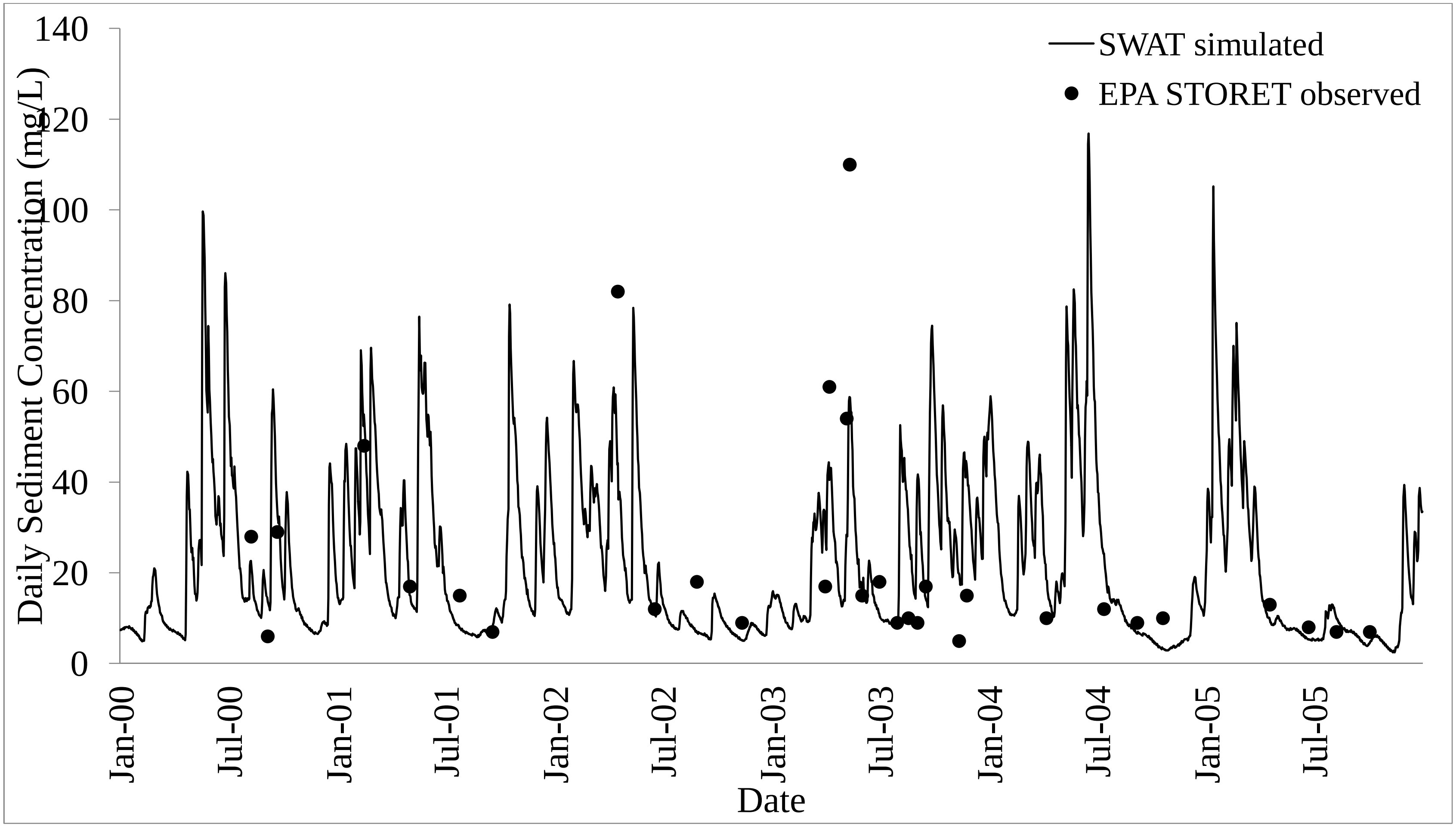

Figure 5-7. Sediment Concentration calibration versus validation for the STORET station 580046.

5.4.2 Wetland Impacts on Streamflow

Overland and streamflows are the main drivers of sediment transport in a watershed. Therefore, we first discuss the overall impacts of wetland restorations scenarios on streamflow at both watershed and subbasin levels.

In general, runoff water is captured and retained in a wetland for a short or long time period of time based on the wetland capacity. This will ultimately reduce downstream flood peaks and flow magnitude. However, based on the results of this study wetland installation resulted in minimal streamflow reductions at the watershed level. Figure 5-8 shows the percent average annual streamflow reduction at the watershed outlet was below 1.2% under all wetland restoration scenarios. Minimal flow reduction might be due to the small surface area of the wetlands (0.4 to 4 ha) when compared to the average subbasin area of 233 ha. These results are consistent with the findings by Martinez-Martinez et al. (2014), where wetlands with an area less than 50 ha did not significantly reduced streamflow at the outlet of a watershed of similar surface area. In addition, no visual difference was observed between wetland restoration scenarios although it was obvious that wetlands located in the headwaters of the watershed (first order streams) perform better than ones in the mid-reaches of the watershed (Figure 5-8), which is in line with other wetland studies (Mitsch, 1993; Martinez-Martinez et al, 2014).

Meanwhile, the impacts of wetland restoration scenarios on runoff at the subbasin level were significantly different from the watershed level. Streamflow was substantially reduced at the subbasin level but the range was very large, varying from 0% to 100% at different locations Figure 5-9). There was a substantial increase in streamflow reduction at the subbasin level with increasing wetland surface area. The average annual streamflow reductions for the four modeled wetland restoration scenarios (0.4, 0.81, 2 and 4 hectares) at the subbasin level were: 21.45 cms, 40.42 cms, 90.22 cms and 159.17 cms, respectively. Only 2% of the 0.4 ha wetlands reduced greater than 50% streamflow, which increased to 33% when the wetland surface area increased to 4 ha. Likewise, none of the 0.4 ha wetlands reduced more than 90% of the streamflow, but 12% of the 4 ha wetlands accomplished this.

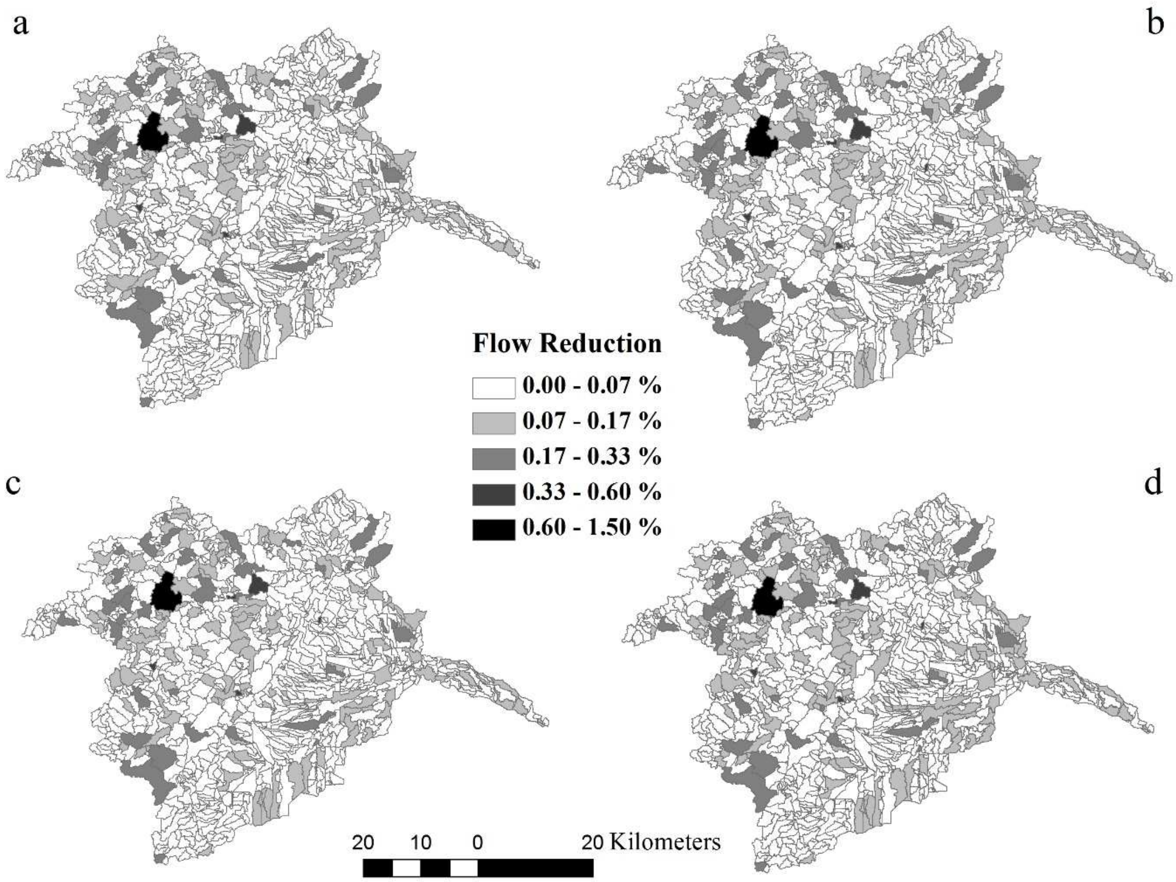

Figure 5-8. Percentage flow reduction at the watershed outlet. (a) 0.40 ha wetland scenario, (b) 0.81 ha wetland scenario, (c) 2 ha wetland scenario, and (d) 4 ha wetland scenario.

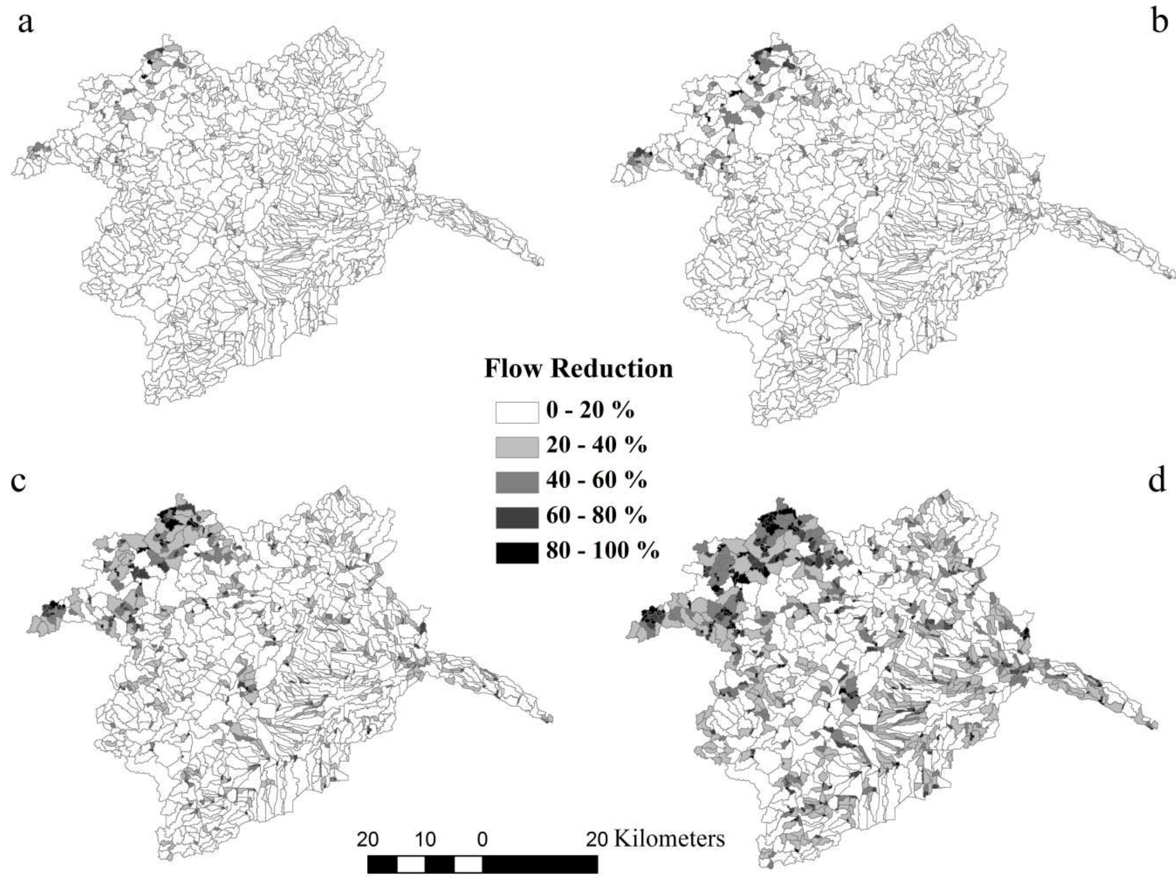

Figure 5-9. Percentage flow reduction at the watershed subbasin level. (a) 0.40 ha wetland scenario, (b) 0.81 ha wetland scenario, (c) 2 ha wetland scenario, and (d) 4 ha wetland scenario.

5.4.3 Wetland Impacts on Sediment Load

The SWAT/SUSTAIN hybrid model approach captured sediment reduction at the outlet and subbasin levels. In general, wetland restoration scenarios were determined to be more effective in reducing sediment at the subbasin level than at the watershed outlet. Watershed level sediment reduction ranged from 0 to 189 tons/year for all wetland surface areas during the simulation period. However at the subbasin level, sediment reduction ranged from 0 to 4015 tons/year. Figure 5-10 shows the impact of wetland restoration on sediment reduction at the watershed level averaged over the simulation period while Figure 5-11 shows the sediment reduction for the same period at the subbasin level. At the watershed level no visual difference was observed among wetland scenarios (Figure 5-10). In addition, the subbasins with highest impact on sediment reduction at the watershed outlet (Figure 5-11) did not align with the subbasins that resulted in the most flow reduction (Figure 5-9) at watershed level (Figure 5-9). Similar results were observed for the flow and sediment reductions at the subbasin level (Figure 5-9 and Figure 5-11, respectively). For example, installation of a 4 ha wetland resulted in significant sediment reduction (>80%) for most subbasins. However, the highest flow reduction (>80%) was only observed in less than 20% of the watersheds. Significant sediment reduction can be explained by relatively small subbasin size to wetland area. Therefore, it can be concluded that the streamflow reduction was not highly correlated with sediment reduction at the subbasin level (R = - 0.25 to - 0.33) or the watershed level (R = 0.30) for all wetland restoration scenarios.

At the subbasin level, streamflow reduction increased at a faster rate than the sediment reduction with the increase in wetland area. However, at the watershed level, streamflow and sediment reduction increased at a comparable rate with the increase in wetland area. At subbasin level, a 100% increase in wetland area (0.81 ha scenario) from 0.4 ha resulted in a 70% increase in streamflow reduction and a 10% increase in sediment reduction, whereas a 900% increase in wetland area (4 ha scenario) resulted in 357% increase in streamflow reduction and 19% increase in sediment reduction. At the watershed level, a 100% increase in wetland area from 0.4 ha resulted in a 2% increase in both streamflow and sediment reduction, while a 900% increase in wetland area resulted in a 17% increase in streamflow reduction and 15% increase in sediment reduction.

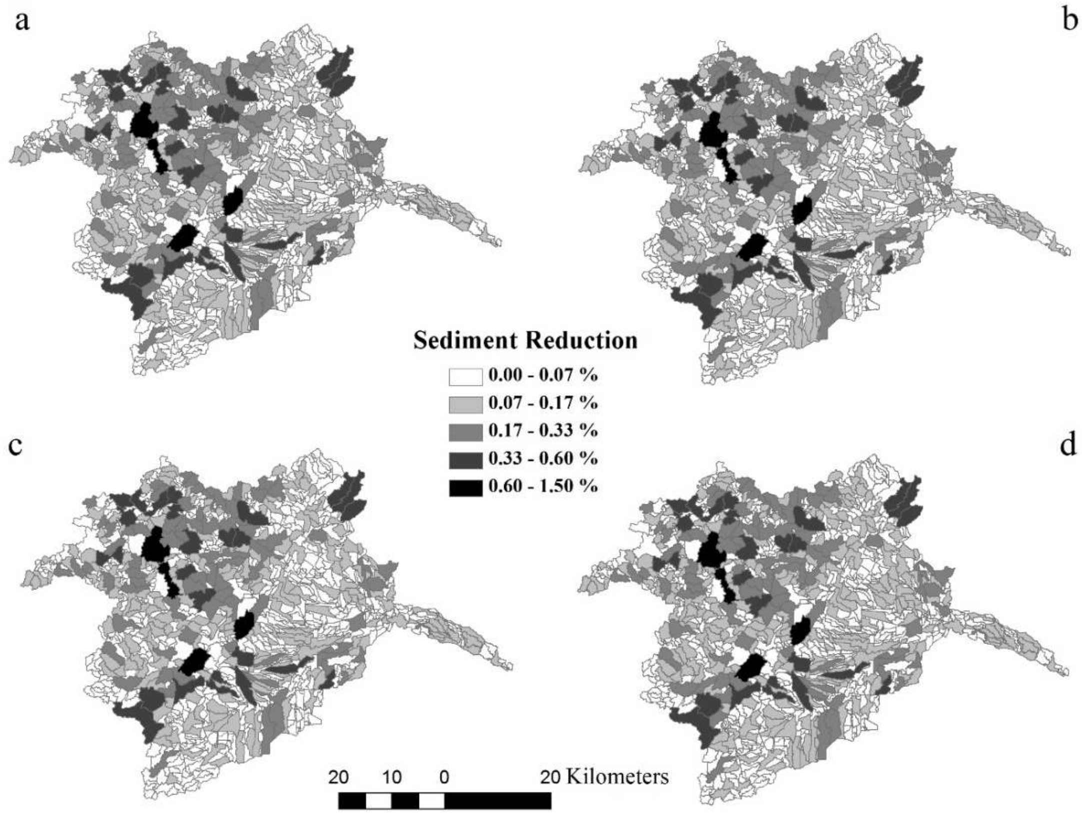

Figure 5-10. Sediment reduction at the watershed outlet. (a) 0.40 ha wetland scenario, (b) 0.81 ha wetland scenario, (c) 2 ha wetland scenario, and (d) 4 ha wetland scenario.

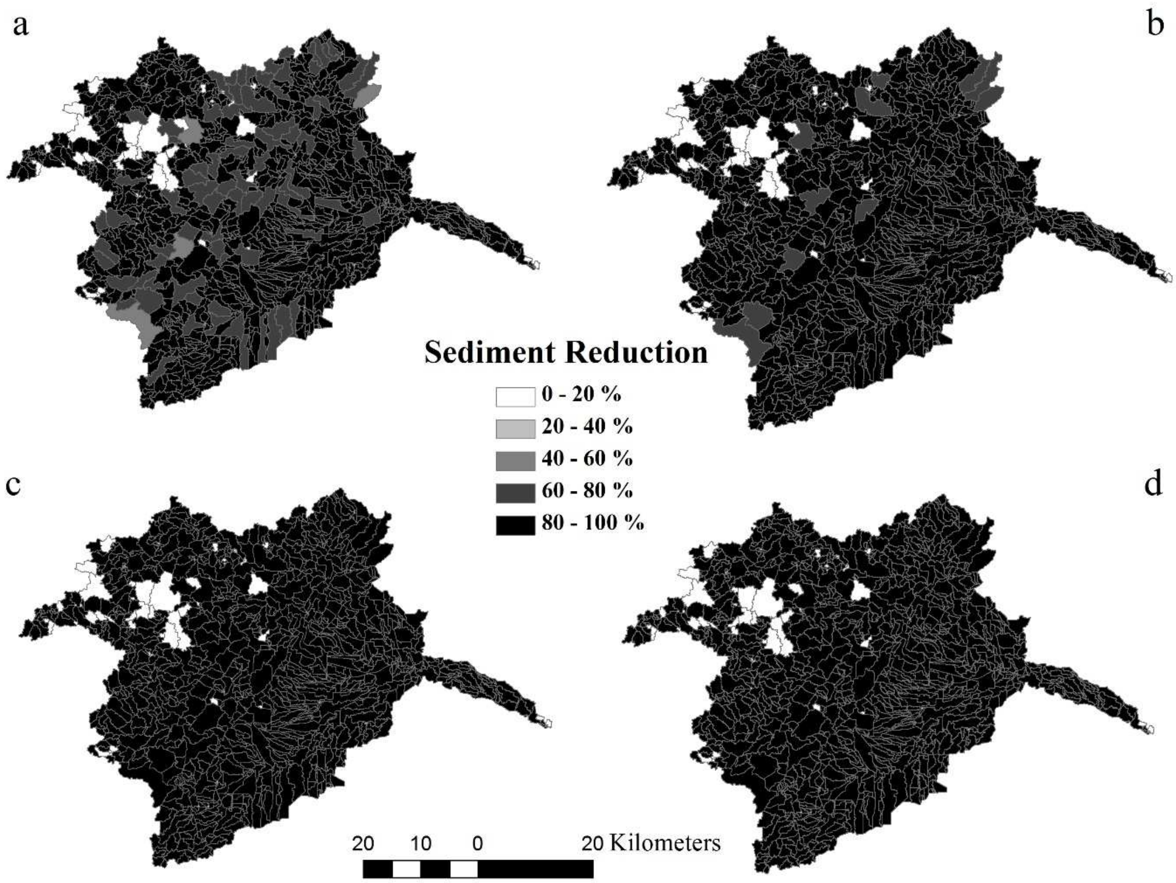

Figure 5-11. Sediment reduction at the subbasin level. (a) 0.40 ha wetland scenario, (b) 0.81 ha wetland scenario, (c) 2 ha wetland scenario, and (d) 4 ha wetland scenario.

Table 5-10 compares the average annual streamflow and sediment reductions based on different wetland surface areas at the watershed and subbasin levels. At the watershed level, only the mean annual sediment reduction provided by 4 ha wetland was significantly higher than the sediment reduction by 0.4 ha and 0.81 ha wetlands. As sediment transport is inextricably tied to streamflow, significant differences in sediment reduction at the watershed level might be due to the larger flow attenuation caused by the 4 ha wetland. As shown in Table 5-10, both flow and sediment were significantly reduced between 0.4 ha and 4 ha wetlands at the watershed level, confirming the above hypothesis. However mean sediment reductions at the subbasin level were not significantly different despite that flow rates were significantly different for each wetland scenario.

Table 5-10. Statistical comparison of mean flow and sediment reduction for different wetland sizes at watershed and subbasin level

Wetland area (ha)	Mean reduction at watershed level		Mean reduction at subbasin level	
	Streamflow* (cms)	Sediment* (tons)	Streamflow* (cms)	Sediment* (tons)
0.4	0.010[a]	17.1[a]	21.5[a]	118.9[a]
0.81	0.010[ab]	17.4[a]	40.4[b]	131.2[a]
2	0.010[ab]	18.3[ab]	90.2[c]	139.9[a]
4	0.011[b]	19.6[b]	159.2[d]	141.9[a]

*Streamflow and sediment means followed by the same letters are not significantly different (α = 0.05).

In the next two sections we explore whether characteristics of a wetland implementation site can be used to better explain sediment dynamics in a watershed. The two characteristics under considerations are the river distance of the wetland implementation site to the watershed outlet and the order of the stream in which the wetland was installed.

5.4.4 Distance From Watershed Outlet and Sediment Reduction

Figure 5-12 (a) shows the relationship between sediment reduction and the distance from the implementation site to the watershed outlet for the 0.4 ha wetland scenario. No recognizable trend was observed. Similarly, no significant trend was observed for the remaining wetland restoration scenarios.

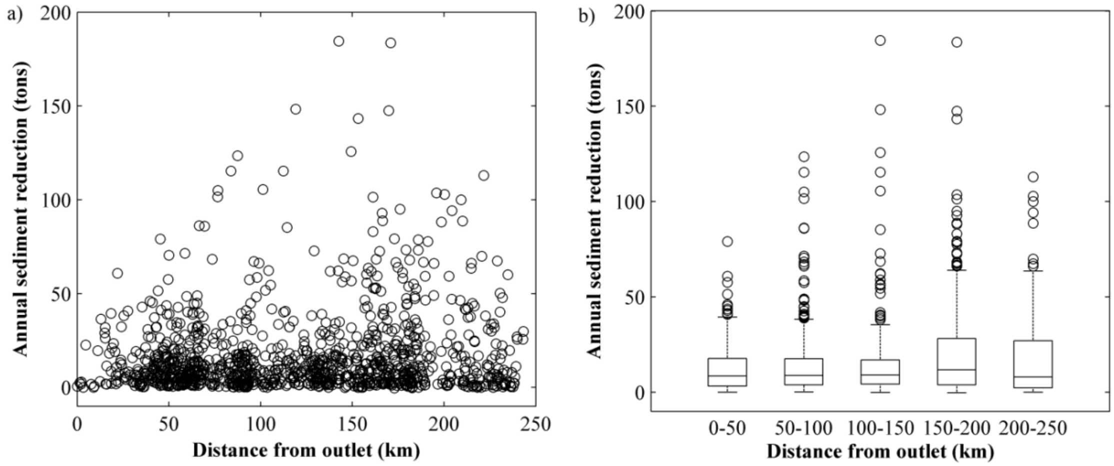

Figure 5-12. Relationship between distance of the wetland implementation site and the watershed outlet. (a) individual wetlands and (b) clustered wetlands for 0.4 ha wetlands.

Further, the subbasins were grouped into five categories (distance from outlet) as shown in Figure 5-12 (b) and pairwise comparison between the categories was performed on the difference in sediment reduction at the subbasin and watershed levels. Table 5-11 compares sediment reduction with average subbasin area, annual sediment yield, and slope. Average annual sediment yield was the greatest for subbasins located between 100-150 km from the watershed outlet and was significantly different from the subbasins between 0-50 or 200-250 km from the outlet. However, the 100-150 km distance was on par with the subbasins at distances of 50-100 or 150-200 km from the outlet. Average annual sediment reduction for different wetland surface areas at the subbasin level showed similar trend to the total sediment yield. Unlike the sediment reduction at subbasin level, the wetlands at the subbasins located at a distance of 150-200 km from the outlet showed the highest sediment reduction at the watershed outlet in all wetland restoration scenarios. In addition, the outlet sediment reductions were significantly higher than all shorter distances (0-50 km, 50-100 km, 100-150 km). Further exploration of the subbasins with 150-200 km distance revealed that most subbasins belonged to stream order 1 (153 of 178), dominant soil type C (151 of 178), and forested land cover (103 of 178). Also, mean slope of the subbasins associated with greatest sediment reduction at the watershed level (150-200 km from the watershed outlet) had significantly greater slope than the subbasins that were situated 0-50, 50-100 and 100-150 km but significantly less than the subbasins situated at 200-250 km from the watershed outlet.

Table 5-11. Distance from outlet and sediment reduction

Distance from outlet (km)	Number of subbasins	Average subbasin area (ha)	Average annual sediment yield (ton)	Average slope (%)	Average sediment reduction Watershed level (tons)				Average sediment reduction Subbasin level (tons)			
					0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)	0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)
0-50	159	197.3^{a}	55.6^{a}	1.0^{a}	13.3^{a}	13.6^{a}	14.5^{a}	15.9^{a}	52.0^{a}	54.5^{a}	55.3^{a}	55.4^{a}
50-100	320	237.8^{ac}	166.1^{b}	1.6^{b}	14.9^{ab}	15.2^{ab}	16.2^{a}	17.8^{a}	143.3^{b}	157.0^{b}	164.9^{b}	166.1^{b}
100-150	229	273.9^{c}	204.1^{b}	1.9^{b}	16.2^{ab}	16.5^{ab}	17.5^{a}	19.0^{a}	165.4^{b}	184.3^{b}	199.0^{b}	203.3^{b}
150-200	278	274.6^{c}	176.8^{b}	3.9^{c}	21.9^{c}	22.3^{c}	23.1^{b}	24.3^{b}	143.1^{b}	159.7^{b}	173.0^{b}	176.2^{b}
200-250	134	214.3^{a}	10.7^{a}	5.0^{d}	18.5^{bc}	18.7^{bc}	19.1^{ab}	19.6^{a}	10.3^{a}	10.5^{a}	10.6^{a}	10.6^{a}

[*] Values followed by the same letters are not significantly different ($\alpha = 0.05$).

5.4.5 *Stream Order and Sediment Reduction*

Two-way analysis of variance (ANOVA) showed that the interaction of stream order and wetland surface area was not significant ($p=1$) in sediment reduction at the subbasin and watershed levels. Hence, the interaction term was dropped and pairwise comparison was performed using Fisher's LSD separately for stream order and wetland surface area.

At the watershed level, wetlands at 4th order streams provided the highest annual average sediment reduction and the wetlands associated with 2nd order streams provided the lowest sediment reduction for all four wetland restoration scenarios. As shown in table 8, only sediment reduction by wetlands associated with 4th order streams was significantly greater than the sediment reduction by wetlands associated with 2nd order streams. Other than this case, stream order did not play a significant role in sediment reduction. Table 5-12 also showed that in general, wetland surface area does not play a major role in sediment reduction for a specific stream order except stream orders 1 and 5, in which 4 ha wetlands reduce significantly more sediment than the 0.4 ha wetland at the watershed level.

At the subbasin level, wetlands associated with 1st order streams produced the highest mean sediment reduction in all wetland restoration scenarios. Nonetheless, there was no significant difference in annual sediment reduction between the wetlands located in stream orders except for the wetlands at 4th and 1st order streams. Wetlands located at the subbasins associated with 1st order streams reduced significantly more sediment than wetlands located on 4th order streams, which is completely opposite to wetland performance on these streams at the watershed level. As shown in Table 5-12, mean sediment yield in subbasins associated with 1st order streams was the highest and was significantly higher than sediment yield in the subbasins associated with 4th order streams. As shown in Table 5-12, wetland surface area did not significantly impact sediment reduction at subbasin level in all of the wetland restoration scenarios at all stream orders.

Table 5-12. Statistical comparison of the sediment reduction provided by different wetland surface areas and stream orders at watershed level

Stream order	No. of subbasins	Average annual sediment load yield (tons)	Average annual sediment reduction (tons) for different stream order based on wetland sizes (vertical comparison)				Average annual sediment reduction (tons) for different wetland sizes based on stream orders (horizontal comparison)			
			0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)	0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)
1	571	162.7[a]	18.3[a]	18.6[a]	19.5[a]	20.9[a]	18.3[a]	18.6[ab]	19.5[ab]	20.9[b]
2	273	130.1[ab]	13.4[b]	13.8[b]	14.6[b]	15.8[b]	13.4[a]	13.8[a]	14.6[a]	15.8[a]
3	128	136.9[ab]	15.7[ab]	16.0[ab]	16.8[ab]	18.2[ab]	15.7[a]	16.0[a]	16.8[a]	18.2[a]
4	104	80.3[b]	21.1[a]	21.4[a]	22.3[a]	23.6[a]	21.1[a]	21.4[a]	22.3[a]	23.6[a]
5	31	135.8[ab]	19.1[ab]	19.5[ab]	20.5[ab]	22.1[ab]	19.1[a]	19.5[a]	20.5[ab]	22.1[b]
6	13	59.1[ab]	20.0[ab]	20.4[ab]	21.0[ab]	22.1[ab]	20.0[a]	20.4[a]	21.0[a]	22.1[a]

*Mean values followed by the same letters are not significantly different (α = 0.05).

5.4.6 Landuse and Sediment Reduction

Landuse type significantly impacted sediment generation per unit area, total sediment yield, and ultimately sediment reduction at subbasin and watershed levels as shown in Table 5-13. The subbasins with dominant agricultural landuse (> 50% of the subbasin area) produced significantly higher sediment per unit area and subbasin (sediment yield) compared to the other landuses. Urban landuse produced higher average sediment per unit area and total sediment compared to forest and water, but the three landuses were statistically similar for sediment generated per unit area and total sediment yield.

The wetlands implemented in agricultural subbasins also were more effective and reduced significantly higher sediment at the subbasin level than the wetlands in subbasins with forest, urban or water. Wetlands within agricultural subbasins reduced significantly higher sediment at the subbasin level, but no significant difference was observed at the watershed level between agricultural and forested subbasins. Wetlands implemented in urban and water dominated subbasins were not significantly different in sediment reduction at the subbasin and watershed level. The results show that wetlands are more effective in sediment reduction when implemented in subbasins with higher sediment yield, which, in this study were those dominated by agriculture. As demonstrated in Table 5-14, watershed level sediment reduction within forested subbasins is more than 20 times that of the subbasin level. The sediment produced in all the subbasins were routed to the wetlands as point sources, and SWAT model may not have adequately handled the point source input within forest dominated subbasins. Hence, caution should be exercised in interpreting the sediment reduction results from the wetlands within forested subbasins.

Table 5-13. Landuse type and sediment reduction

Landuse	Sediment generation per unit area (ton/ha)	Average sediment yield per landuse type and subbasin* (tons)	Annual average sediment reduction at watershed level (tons)				Annual average sediment reduction at subbasin level (tons)			
			0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)	0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)
Agriculture	0.72[a]	187.9[a]	18.3[a]	18.7[a]	19.6[a]	21.1[a]	157.0[a]	173.3[a]	184.7[a]	187.3[a]
Forest	0.01[b]	0.8[b]	15.9[a]	16.2[a]	16.7[a]	17.5[a]	0.7[b]	0.8[b]	0.8[b]	0.8[b]
Urban	0.09[b]	13.8[b]	8.2[b]	8.6[b]	9.5[b]	11.1[b]	11.3[b]	12.4[b]	13.4[b]	13.7[b]
Water	0.00[b]	0.0[b]	0.3[b]	0.3[b]	0.3[b]	0.3[b]	0.0[b]	0.0[b]	0.0[b]	0.0[b]

* Each subbasin contains only one landuse type

5.4.7 Selection of Most Suitable Wetland Considering Environmental and Economic Factors

As described earlier, five combinations of environmental and economic scenarios were studied using AHP to evaluate the importance of considering these factors when developing watershed-wide restoration scenarios. Figure 5-13 presents the number and surface area of wetland selected as most effective (primary y-axis) and cost of sediment reduction per ton (secondary y-axis) based on different environmental/economic scenarios.

Overall, the 0.4 ha wetland was selected as most suitable for almost all subbasins for three environmental/economic scenarios (0.1/0.9, 0.25/0.75, and 0.5/0.5). This is due the much lower cost associated with 0.4 ha wetland installation ($1490) versus $13,656 for the 4 ha wetland. Meanwhile, the environmental benefits associated with installation of larger wetland surface area is minimal (less than 2% between 0.4 ha and 4 ha wetlands). The breakeven point was found to be the scenario that gave equal importance to environment as well as to economy. The breakeven point is the location where the response variable changes sharply with respect to the predictor. This point provides the maximum sediment reduction with an associated minimum cost. In the environment/economic scenario (0.5/0.5), after 0.4 ha wetland, the 4 ha wetland was selected as the most effective in a small number of subbasins due to slightly higher sediment reduction efficiency compared to other wetland alternatives (0.8 and 2 ha). From environmental/economic scenario (0.5/0.5) up to environmental/economic scenario (0.9/0.1), the number of subbasins in which 4 ha was selected as optimal increased. In the 0.75/0.25 environmental/economic scenario the 0.4 ha wetland was most favorable for a majority of the subbasins while 4 ha wetland was the second most favorable and the two other wetland surface areas (0.8 and 2 ha) were rarely selected. For environmental/economic scenario (0.9/0.1), the 0.4 ha wetland was widely replaced by the 4 ha wetland as most suitable because the environmental factor was given much more importance than the economic factor. The 0.8 ha and 2 ha wetlands were selected as most effective in a limited number of subbasins only for environmental/economic scenarios 0.75/0.25 and 0.9/0.1. Overall, it can be concluded that smaller wetland surface area should be selected reduce sediment at the watershed outlet at lowest cost. However, if cost is not a limiting factor in a wetland restoration strategy, larger wetlands can be implemented.

The above analysis was repeated at the subbasin level (Figure 5-14). Contrary to the watershed scale analysis, the 0.4 ha wetland was selected as the most suitable option in all scenarios. And in four scenarios the 0.4 ha wetland was the only recommended wetland. The breakeven point was found to be the scenario that gave more weight to the environmental component than the economic component (0.75/0.25 environmental /economic scenario). For 0.9/0.1 environmental/economic scenario, the 0.4 ha wetland was the most suitable wetland, while 0.8 ha, 2 ha, and 4 ha wetlands were selected as most effective in a limited number of subbasins.

The spatial distribution of wetland placement based on different environmental /economic benefits (watershed versus subbasin levels) are presented in Figure 5-15 and Figure 5-16, respectively. As shown in these figures, the differences between 0.1/0.9 and 0.25/0.75, and 0.5/0.5 environmental/economic benefits at the watershed and subbasin levels were minimal and the 0.4 ha wetland was the preferred choice. However, the differences were magnified for 0.75/0.25 and 0.9/0.1 environmental/economic scenarios. Especially, for the 0.9/0.1 environmental/economic scenario, larger wetlands were selected throughout the watershed in the watershed scale analysis (Figure 5-15), while selection of larger wetland areas was limited to the middle part of the watershed for the subbasin level analysis. This shows the importance of level of analysis (watershed versus subbasin) for the appropriate placement of wetlands in a watershed.

These results provide a solution to policymakers based on the importance of environmental and economic factors. This procedure can be applied in selecting different wetland surface areas and using different weighting factors depending on the region or goals of a specific wetland restoration project.

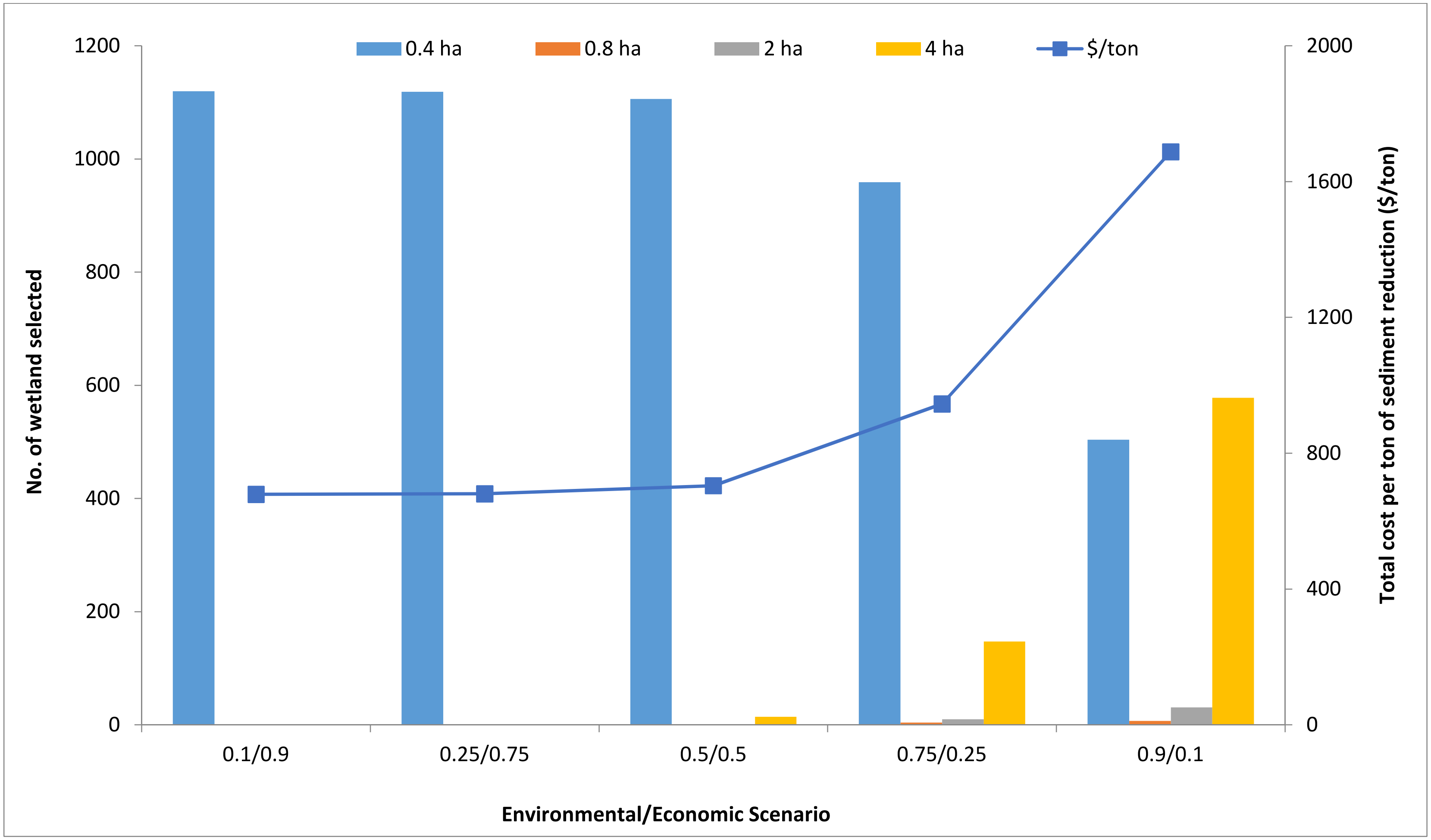

Figure 5-13. Most suitable wetland selected based on different environmental / economic scenarios (watershed scale).

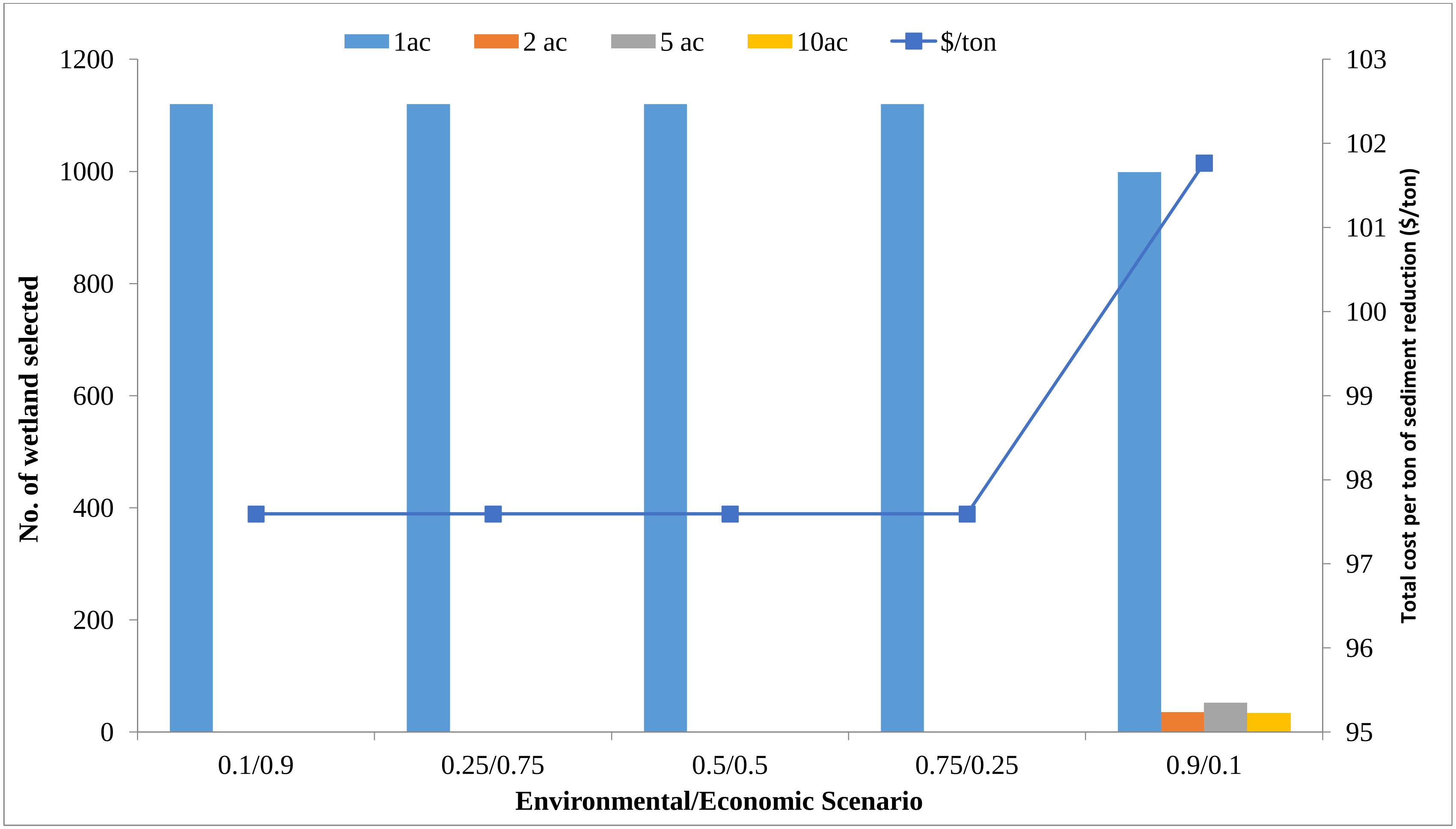

Figure 5-14. Most suitable wetland selected based on different environmental/economic scenarios (subbasin level).

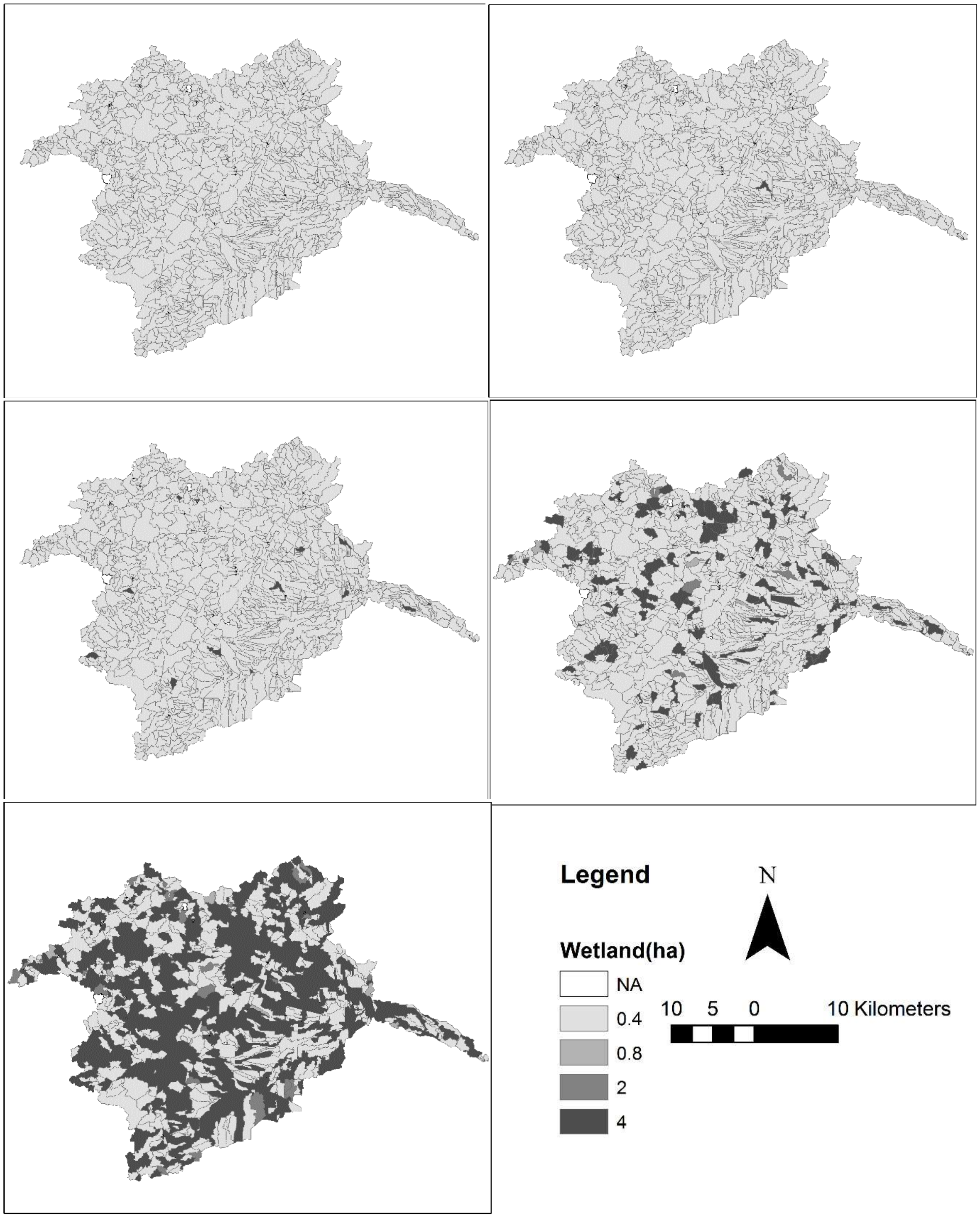

Figure 5-15 Most suitable wetland placement considering watershed level environmental/economic benefits a) 0.1/0.9, b) 0.25/0.75, c) 0.5/0.5, d) 0.75/0.25, and e) 0.9/0.1.

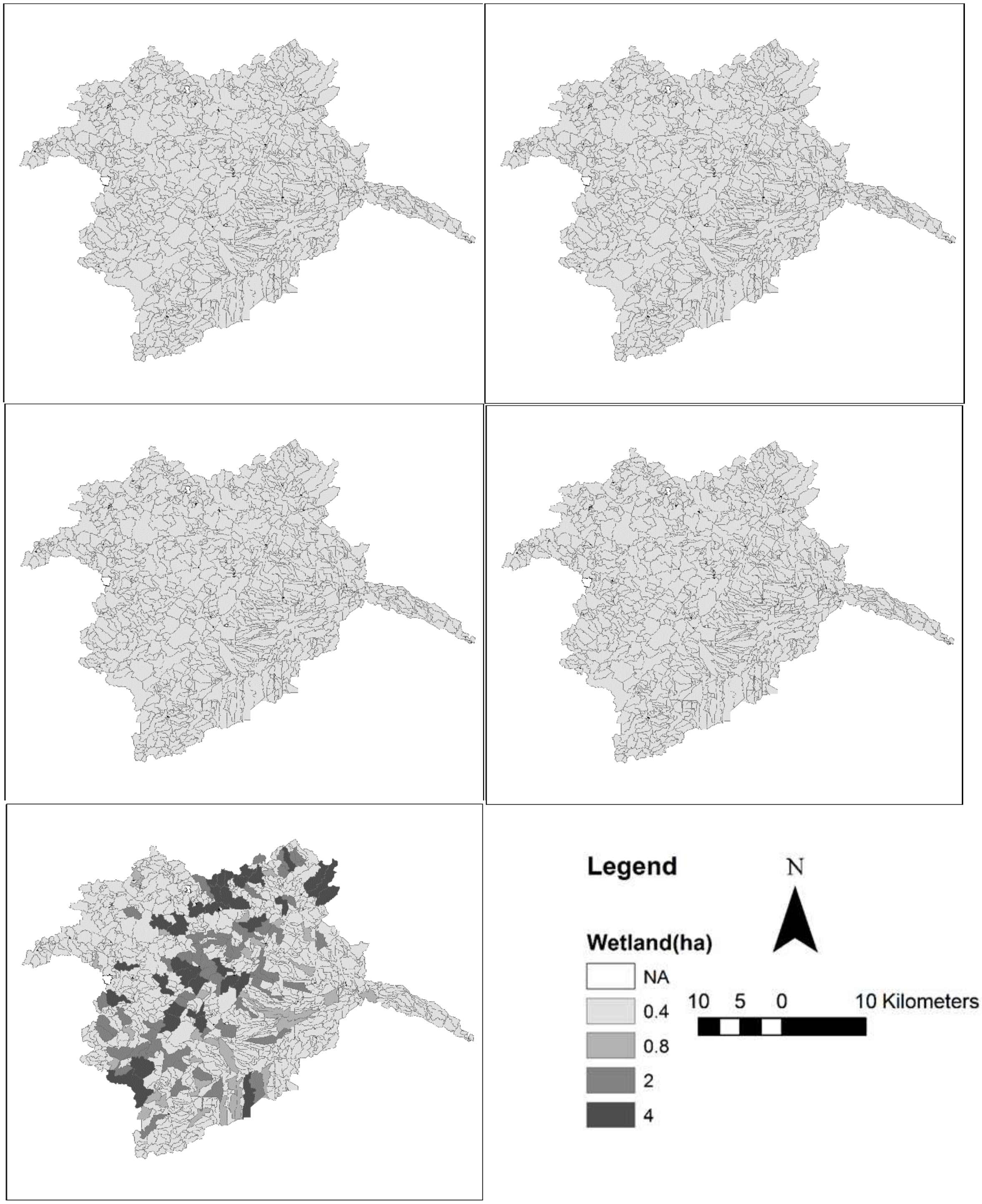

Figure 5-16. Most suitable wetland placement considering subbasin level environmental/economic benefits a) 0.1/0.9, b) 0.25/0.75, c) 0.5/0.5, d) 0.75/0.25, and e) 0.9/0.1.

5.5 CONCLUSION

Due to limitations of current modeling practices, functional assessment of wetland is widely used, which in most cases is site specific and qualitative. The goal of this study is overcome these disadvantages by coupling watershed (SWAT) and site scale (SUSTAIN) models for evaluating the sediment reduction benefits of wetland restoration scenarios. The SWAT model simulates land surface and river routing processes while SUSTAIN handles the implementation of restoration wetlands in subbasins. Finally, the analytic hierarchy process was used to incorporate environmental and economic considerations when selecting wetland restoration scenarios.

The results of this study showed that the impacts of wetland restorations scenarios on flow and sediment reduction at the watershed outlet was minimal (<1.5%), which can be attributed to small surface area of wetland restoration scenarios (0.4 to 4 ha) compared to the average subbasin size in the watershed. In addition, streamflow and sediment reduction rates were consistent with increases in wetland area. Conversely, wetland installation resulted in significant flow and sediment reductions at the subbasin level up to 100%.

In the next step the wetland restoration sites were examined to better explain sediment dynamics in the watershed. Two specific site characters (order of the stream and distance to the watershed outlet) were used for this analysis. The subbasins were grouped into five clusters based on their distance to the outlet and six classes based on the stream order number for the implementation site. Average annual sediment yield generated at subbasins located at 100-150 km from the watershed outlet were the highest. For the watershed level analysis, wetlands located at 150-200 km from the outlet performed the best while wetlands located at 100-150 km from the outlet performed the best at subbasin level. Wetlands associated with 1st order streams performed better at subbasin level. Conversely, wetlands located in subbasins with the 4th order streams resulted in the highest sediment reduction at the outlet. Overall, larger wetlands performed better in sediment removal at both watershed and subbasin levels, though the reductions were not statistically different.

Finally, both economic and environmental impacts of wetlands restoration scenarios were considered for final placement of wetlands in the study area. Overall, smaller wetlands should be selected to control the sediment reduction at the watershed outlet because of their low cost but acceptable performance. However, if cost is not a limiting factor in a wetland restoration strategy, larger wetlands can be implemented. Contrary to the watershed scale analysis, the 0.4 ha wetland was selected as the most suitable option in all scenarios for the subbasin level sediment reduction goal.

This study introduced an alternative approach to functional assessment of wetlands that is more accurate and quantitative. The approach can be used and adapted by local, state, and national level agencies for wetland restoration initiatives to identify priority areas for the lowest cost and the greatest environmental impact resulting from sediment control at subbasin and watershed scales.

6 OVERALL CONCLUSION

The model integration exercise in this study demonstrated the impacts of wetland restoration scenarios at the watershed scale. In addition, it quantified the benefits of implementation based on location, size, and, depth of wetlands into a more understandable concept compared to functional assessments. This will facilitate conversation among watershed stakeholders that may include farmers, conservation groups, and federal and local agencies. This study also assists watershed managers in making informed decisions and allocating conservation dollars based on environmental and economic benefits.

The following conclusions are based on the results of the first study, "Modeling the hydrologic significance of wetland restoration scenarios":

- Wetland size was determined to be an important factor in reducing streamflow at the watershed and subbasin level. Larger wetland area resulted in more streamflow reduction due to increased storage volume and evaporation. While differences in streamflow reduction among wetland sizes were statistically significant, they may not be relevant to implementation plans. Under these scenarios, a higher level of statistical significance should be considered.

- Long-term daily average streamflow reductions were insensitive across stream orders for wetlands with areas less than 500 ha. However, wetland restoration of 500 ha on first and third order streams were most effective in streamflow reduction. Wetland implementation on first order streams was most effective at the subbasin level regardless of depth and area, but reduction was minimal at the watershed outlet.

- While the high impact maps for the watershed (smallest area and depth combination resulting in greatest streamflow reduction) were generally similar, optimal area and depth combinations for streamflow reduction were often different. This is useful in the decision making process, where targeted location may not change but implementation size will change based on project goals and resources.

The following conclusions are based on the results of the second study, “Assessing the significance of wetland restoration scenarios on sediment mitigation plan”:

- Overall, larger wetlands removed more sediment at the watershed and subbasin levels. However, the impacts of wetland restoration scenarios on sediment reduction at the watershed outlet was minimal (<1.5%) compared to the subbasin level (up to 100%).

- The subbasin clustering based on the distance from the watershed outlet showed that the optimum wetland restorations sites (clusters) are be different based on the goal of the study (sediment reduction at the subbasin versus outlet)

- Wetland implementation in sites associated with 1st order streams performed better at the subbasin level. Conversely, wetlands located in subbasins with 4th order streams reduced the most sediment at the watershed outlet.

- After considering environmental benefits and economic costs of wetland restoration scenarios, the smallest wetland size was selected for both subbasin and watershed level analysis. The smallest wetland size was the best in terms of cost of sediment reduction in the study area.

7 RECOMMENDATIONS FOR FUTURE RESEARCH

This study examined the impact of wetland area, depth, and placement on streamflow, peak flow reduction, and sediment reduction at subbasin and watershed scales. However, there are significant gaps in key areas of knowledge concerning wetland ecosystems functions, services and values. Based on the results of this study areas of further research include the following:

- Many modeling tools exist to evaluate environmental and hydrologic benefits of wetlands, but due to their limitations (e.g. input intensive, evaluate a single or few water quality/quantity parameters, operate only at field scales, etc.) they cannot be used for development of watershed scale restoration plans. Integrating existing models or developing new models is necessary to account for the wide range of wetland functions that include additional water quality and quantity benefits.

- Evaluate the impacts of restoration scenarios beyond water quality and quantity to include other wetland functions such as wildlife habitat conservation, groundwater recharge, greenhouse gas sequestration, etc.

- Perform additional studies in unique physiographic regions to better understand wetlands role in achieving watershed-scale management goals in different environmental settings.

- Study the social challenges and benefits of wetland restoration in addition to the economic and environmental aspects of sustainable watershed management.

APPENDICES

APPENDIX A

ADDITIONAL MATERIAL TO SECTION 4 TITLED: "MODELING THE HYDROLOGICAL SIGNIFICANCE OF WETLAND RESTORATION SCENARIOS"

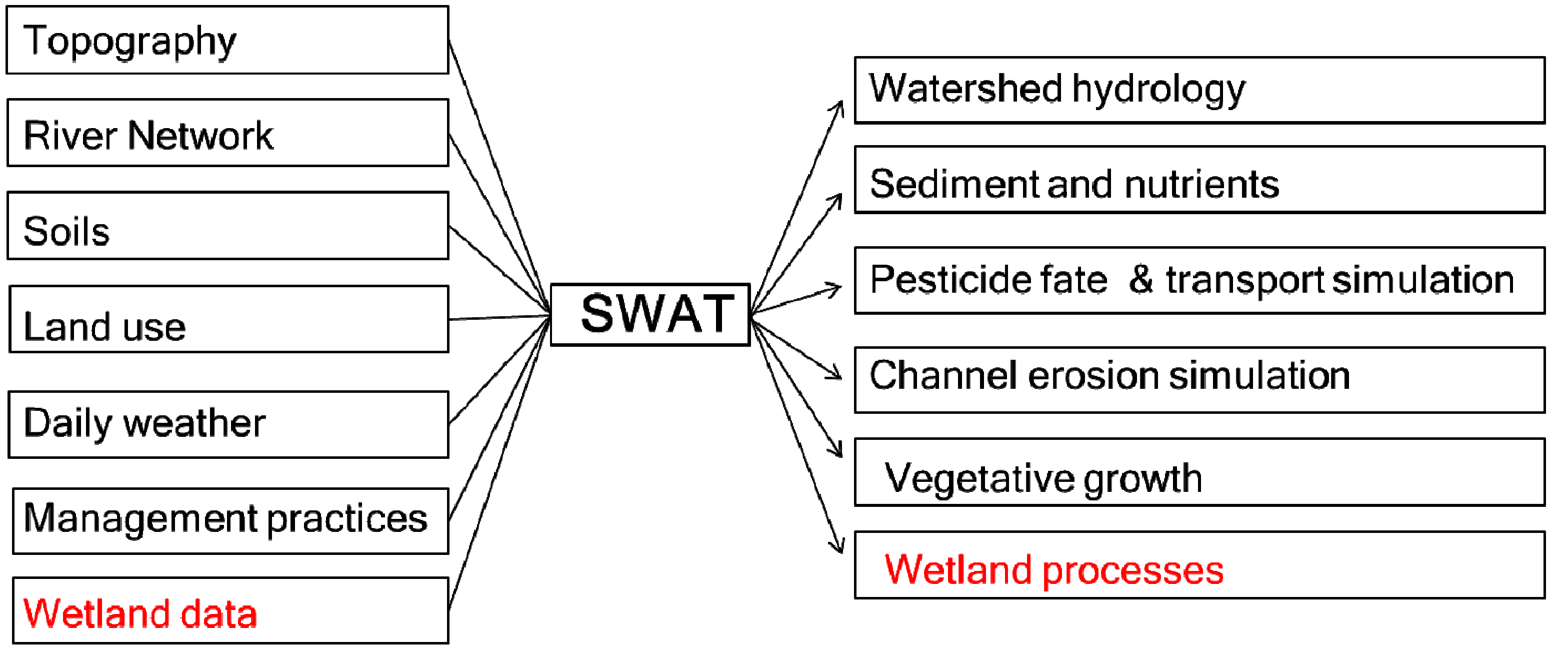

Figure 8-1. SWAT data frame for wetland modeling processes.

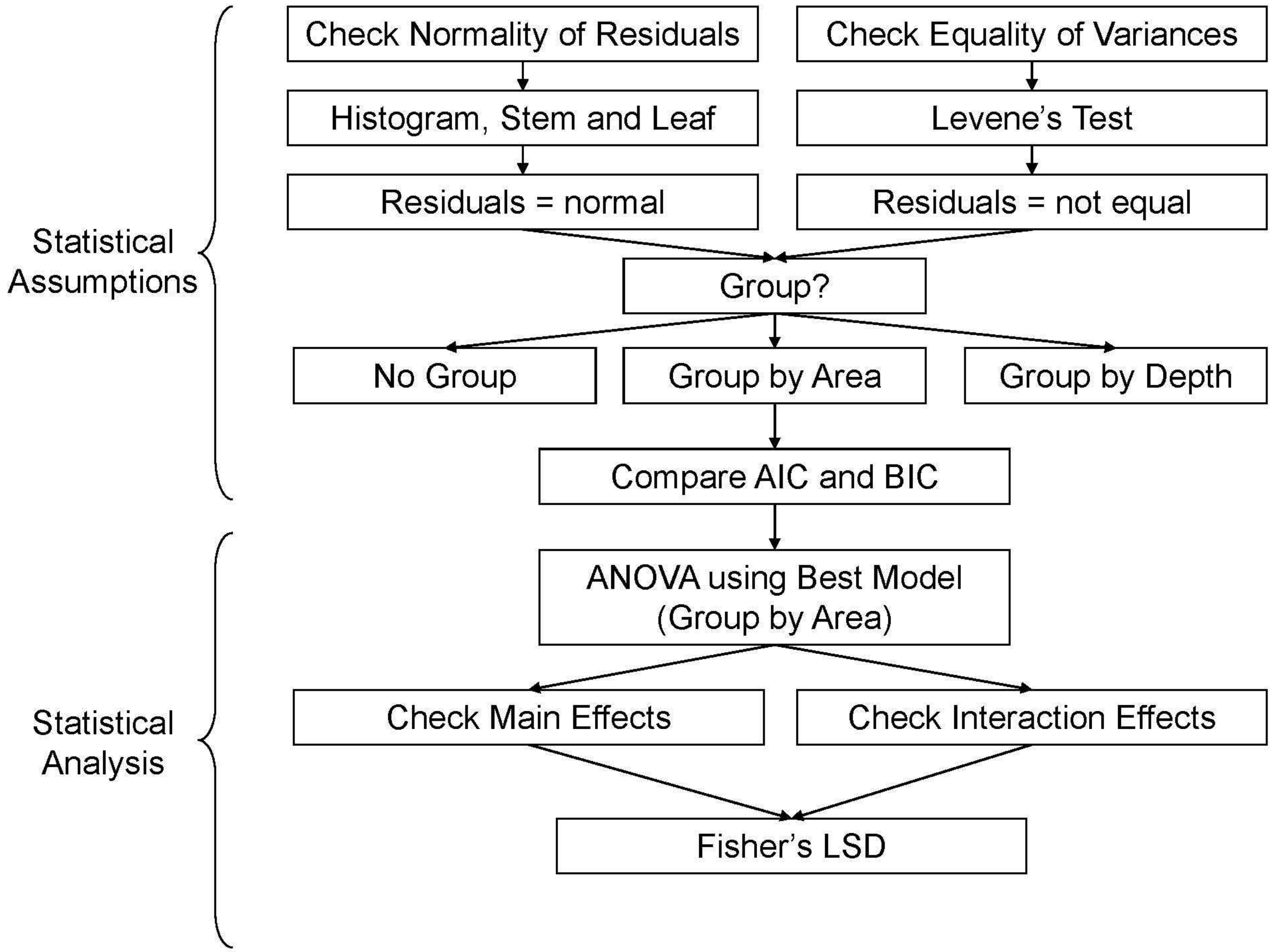

Figure 8-2. SWAT statistical data frame.

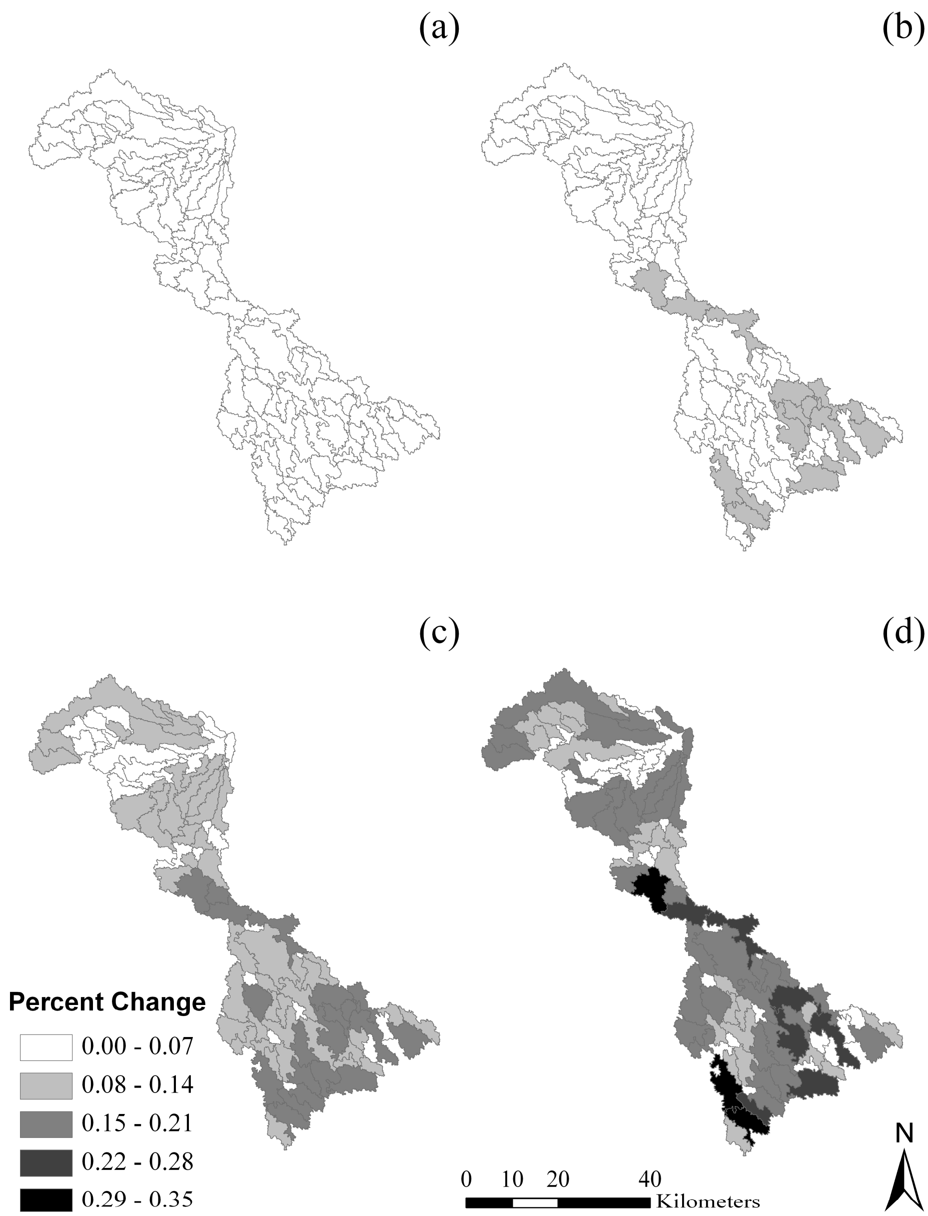

Figure 8-3.Percent flow change at the watershed outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 15 cm

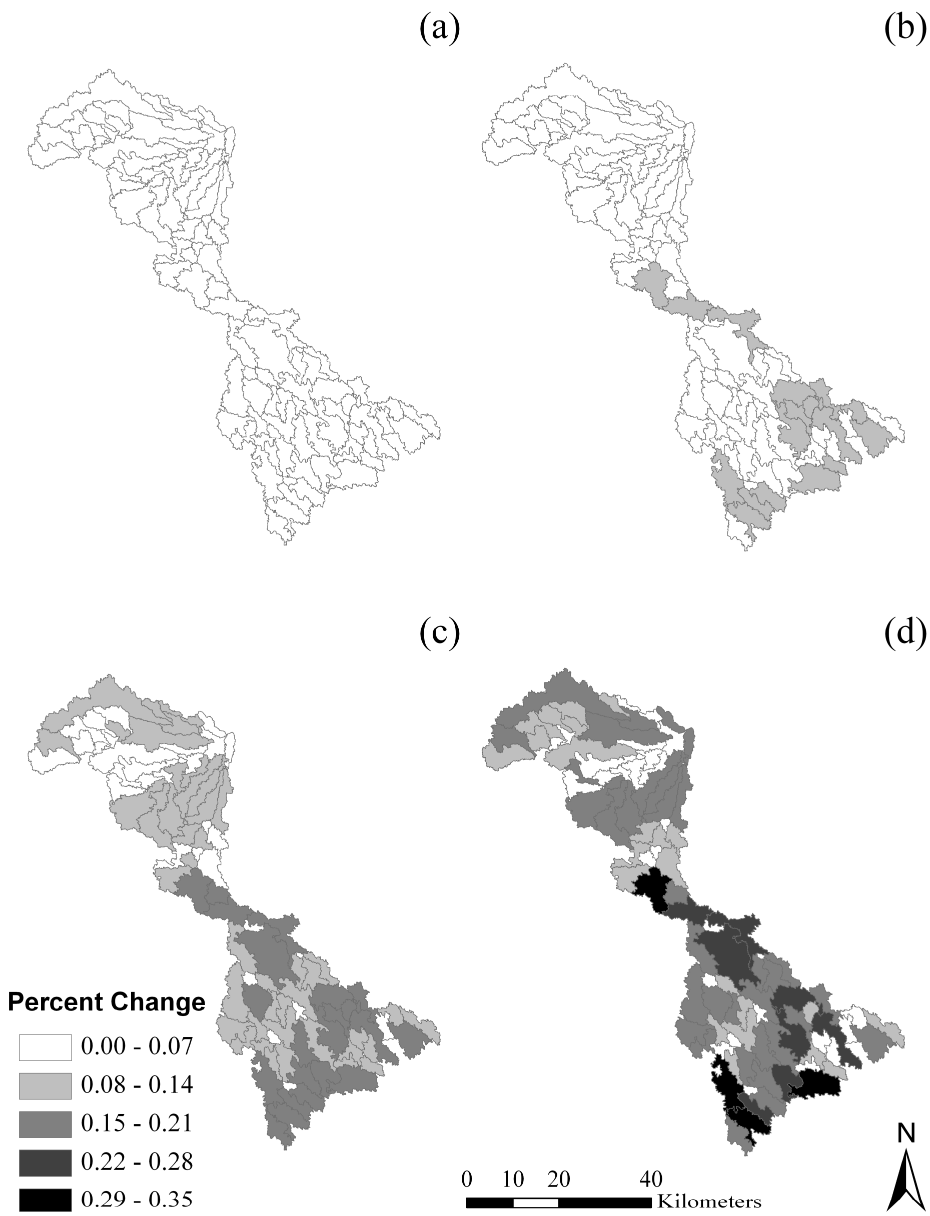

Figure 8-4.Percent flow change at the watershed outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 30 cm

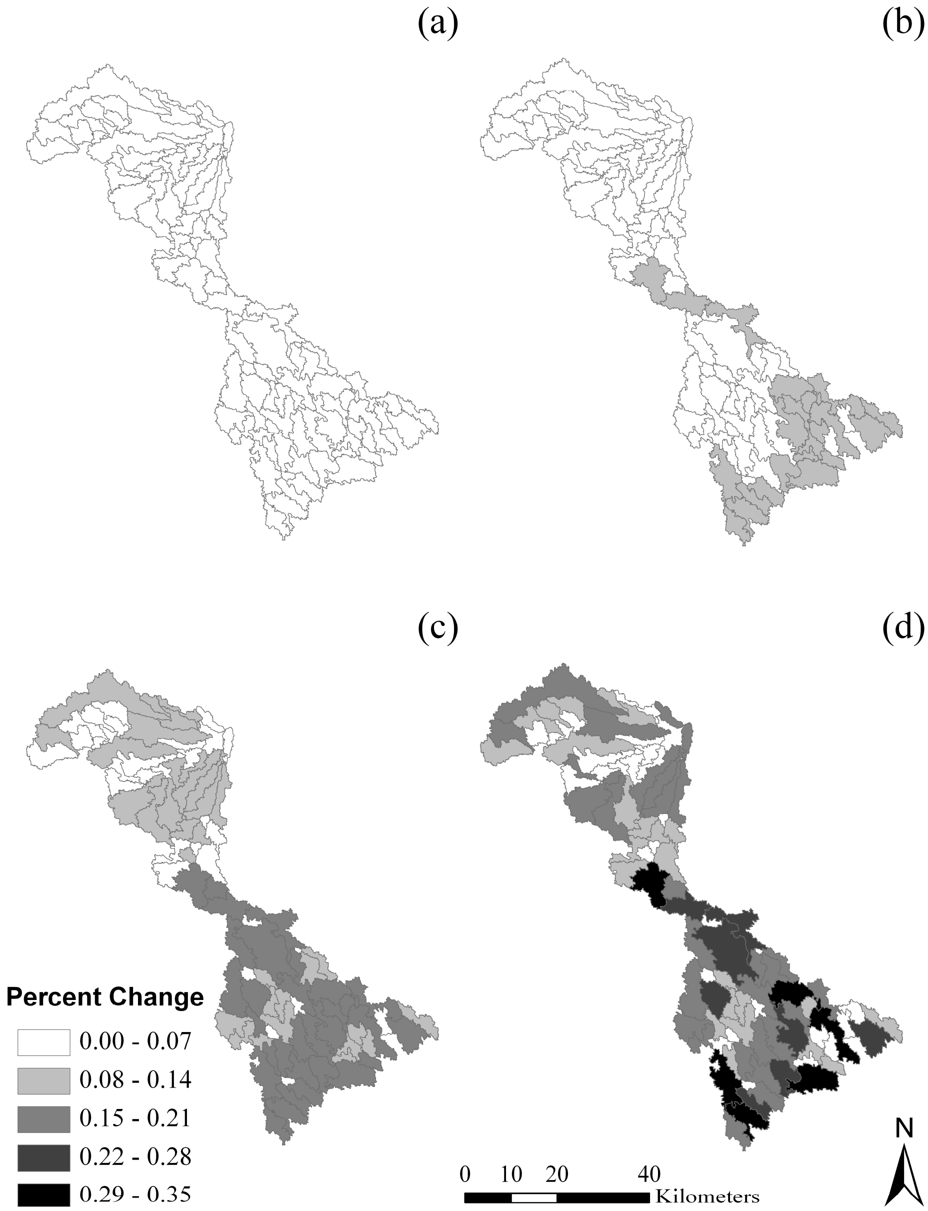

Figure 8-5.Percent flow change at the watershed outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 61 cm

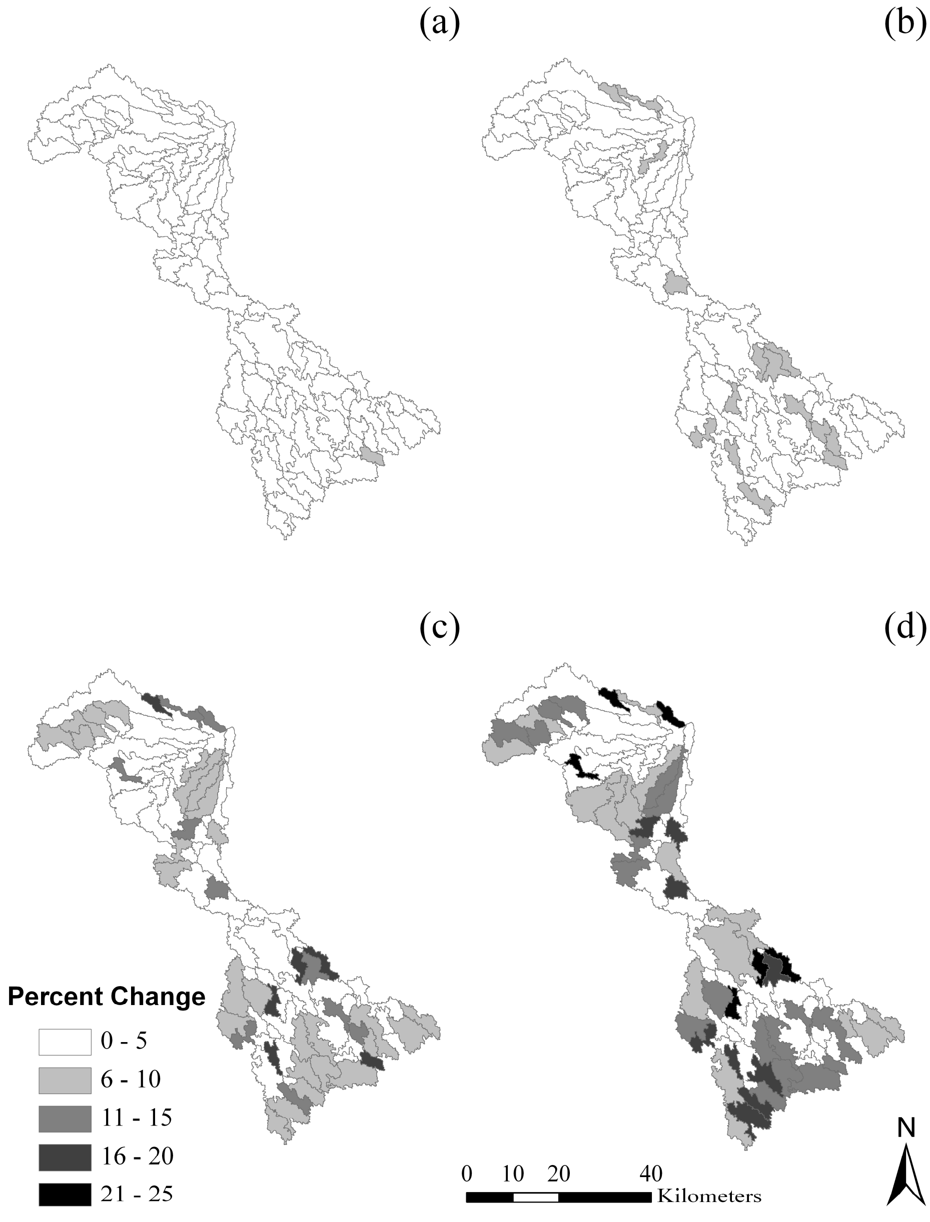

Figure 8-6. Percent flow change at the subbasin outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 15 cm

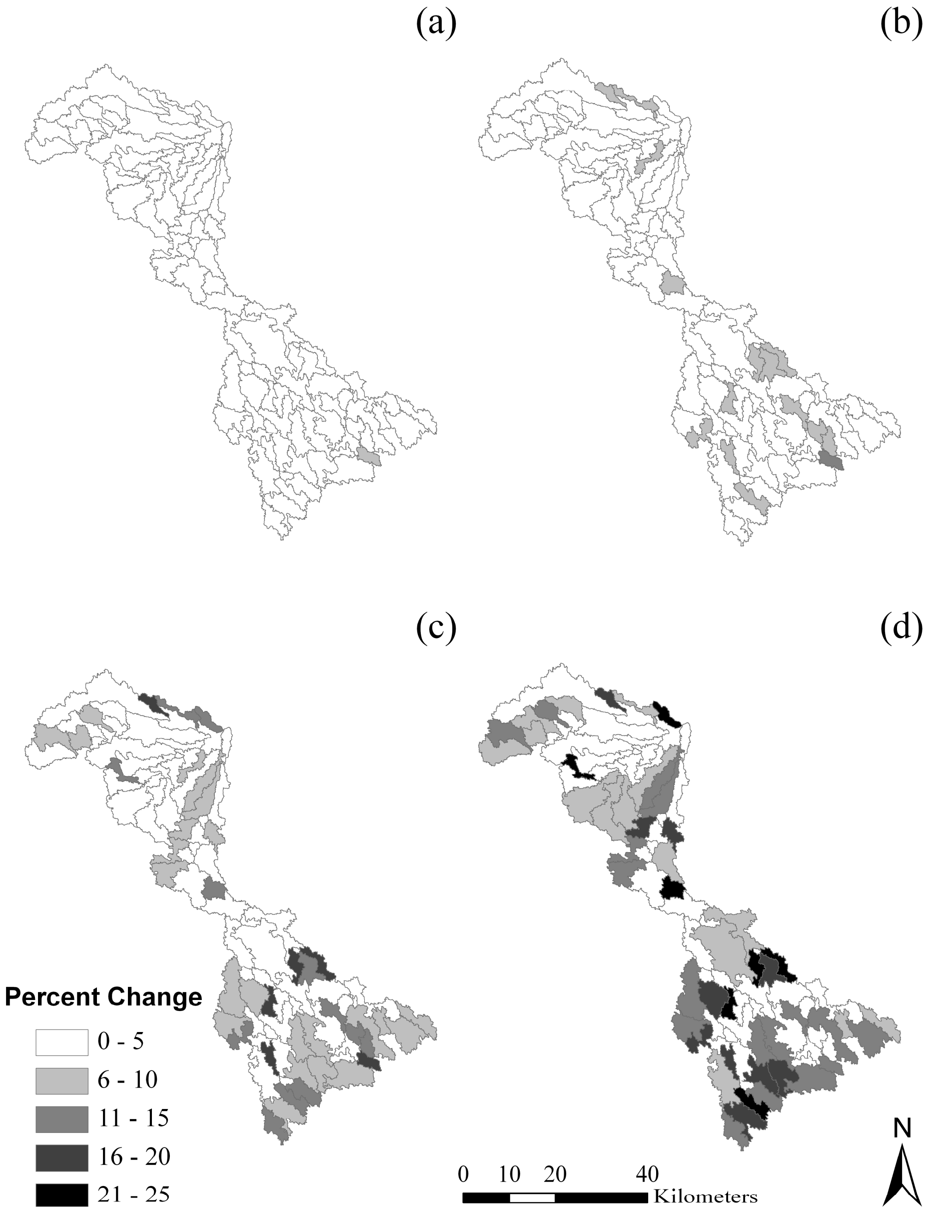

Figure 8-7. Percent flow change at the subbasin outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 30 cm

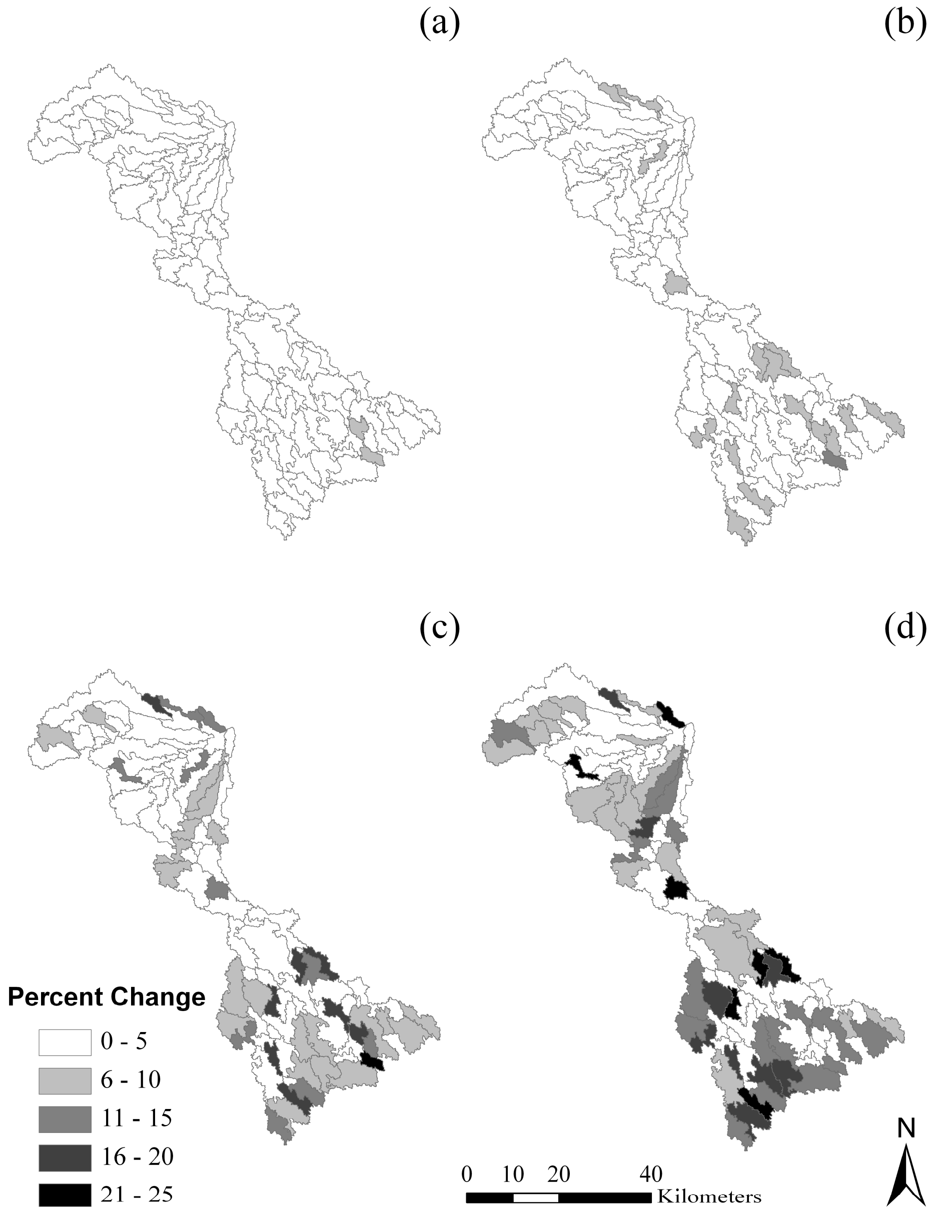

Figure 8-8. Percent flow change at the subbasin outlet compared to base scenario – wetlands areas: (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha at 61 cm

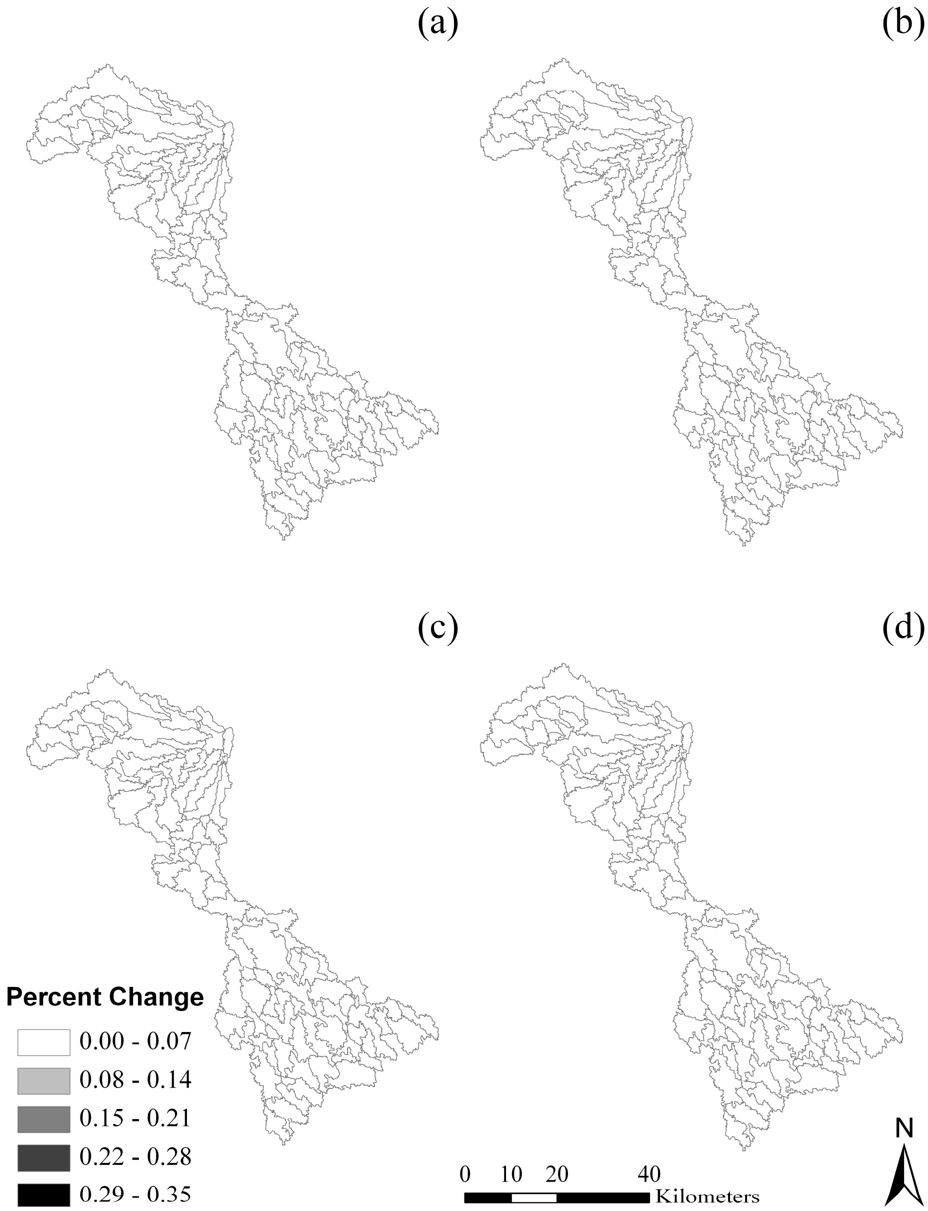

Figure 8-9. Percent flow change at the watershed outlet compared to base scenario – 50 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

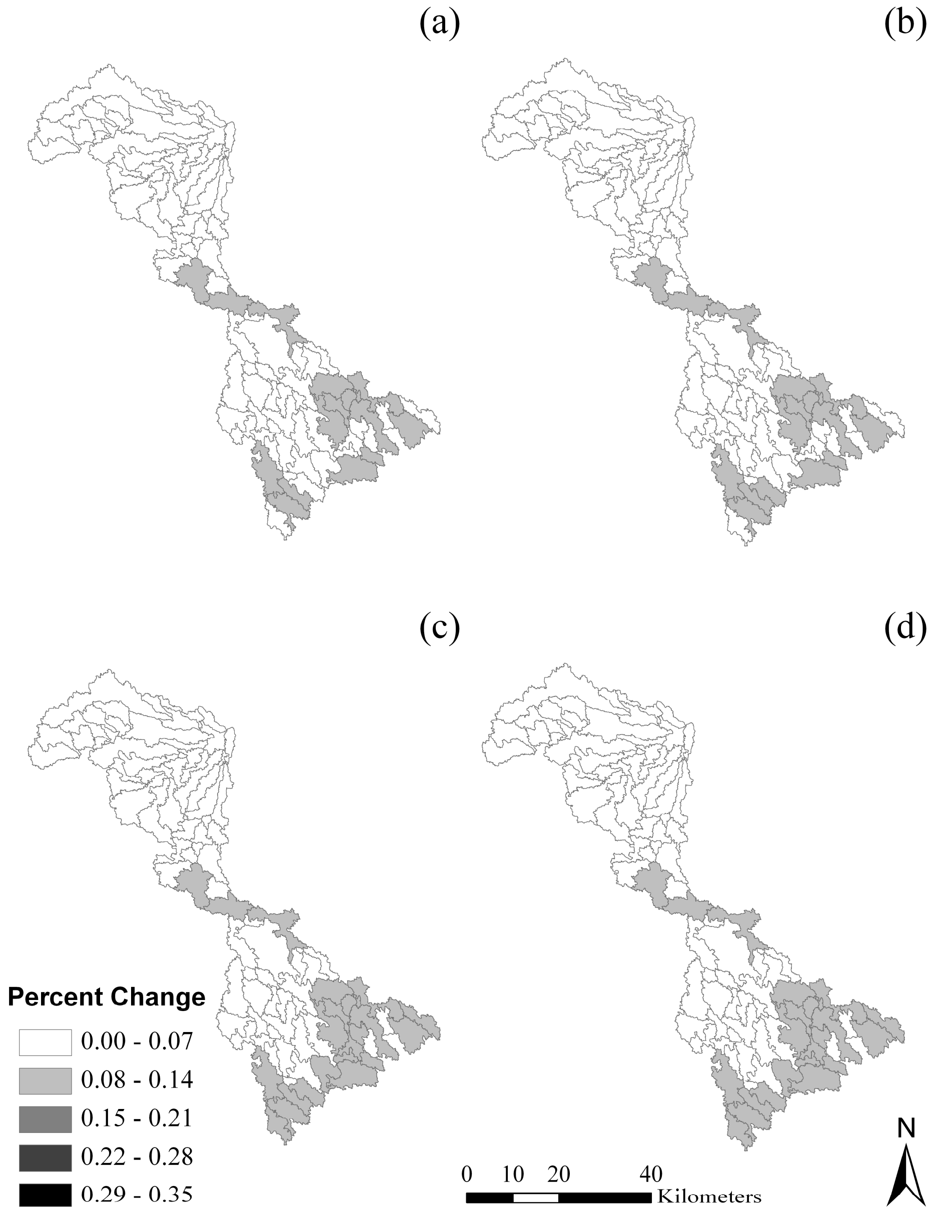

Figure 8-10.Percent flow change at the watershed outlet compared to base scenario – 100 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

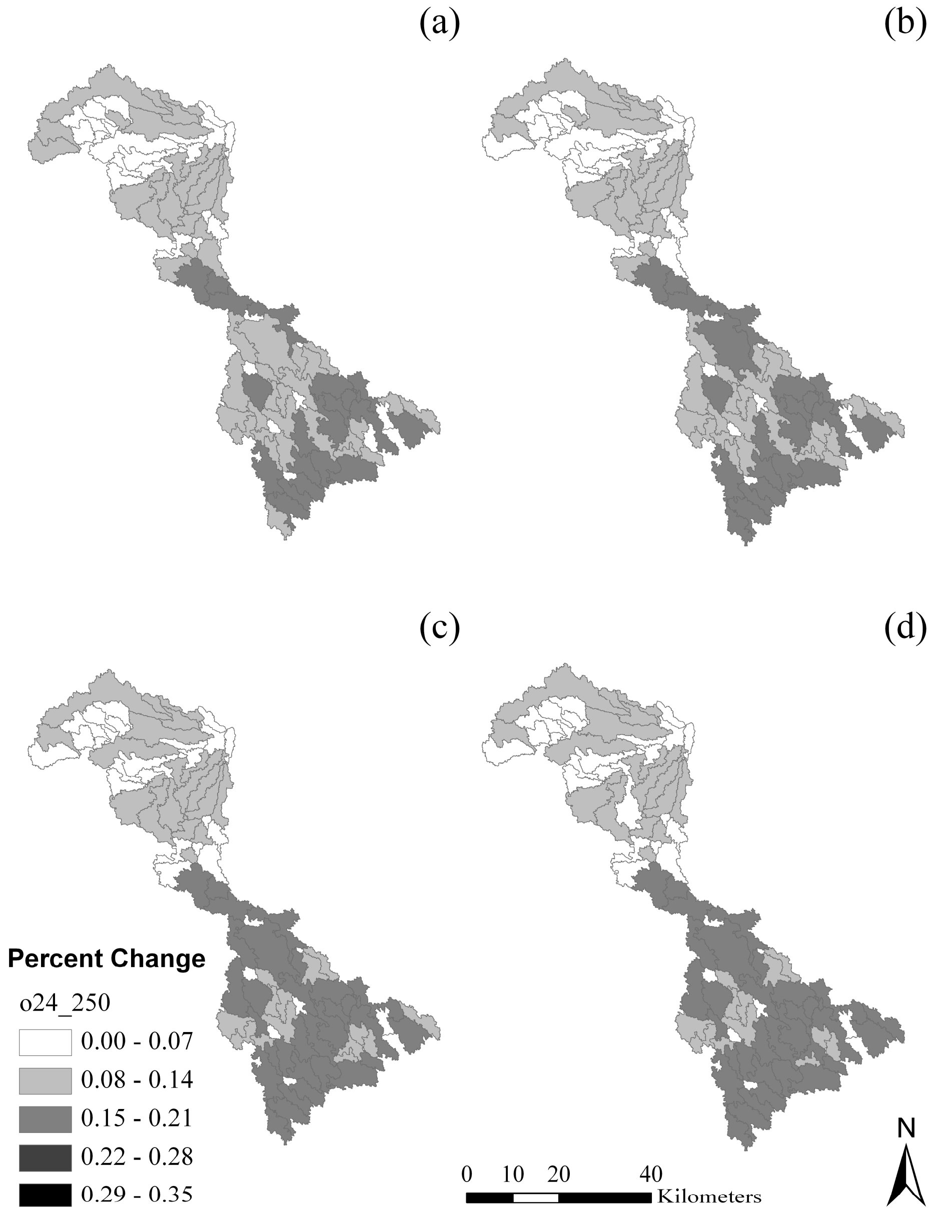

Figure 8-11. Percent flow change at the watershed outlet compared to base scenario – 250 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

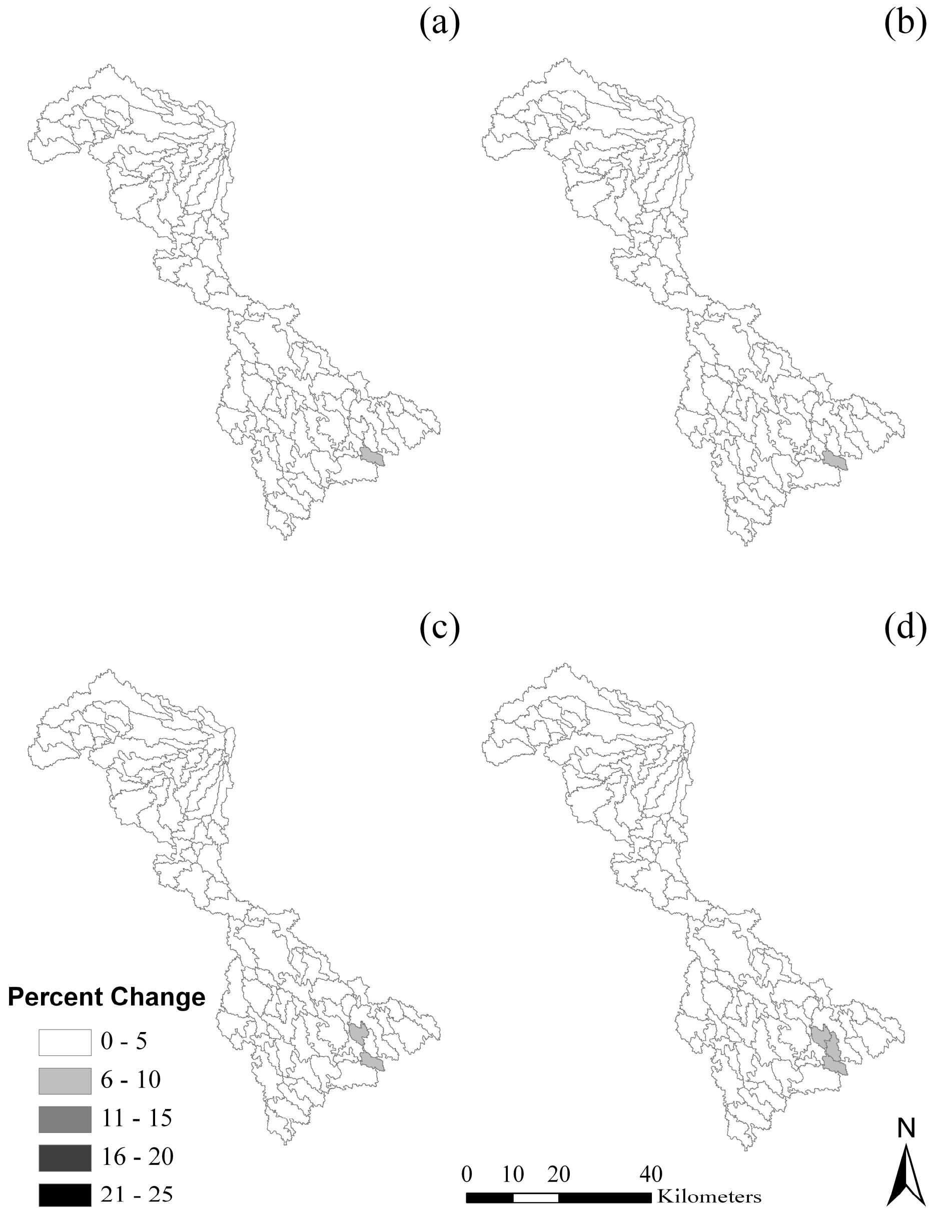

Figure 8-12. Percent flow change at the subbasin outlet compared to base scenario – 50 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

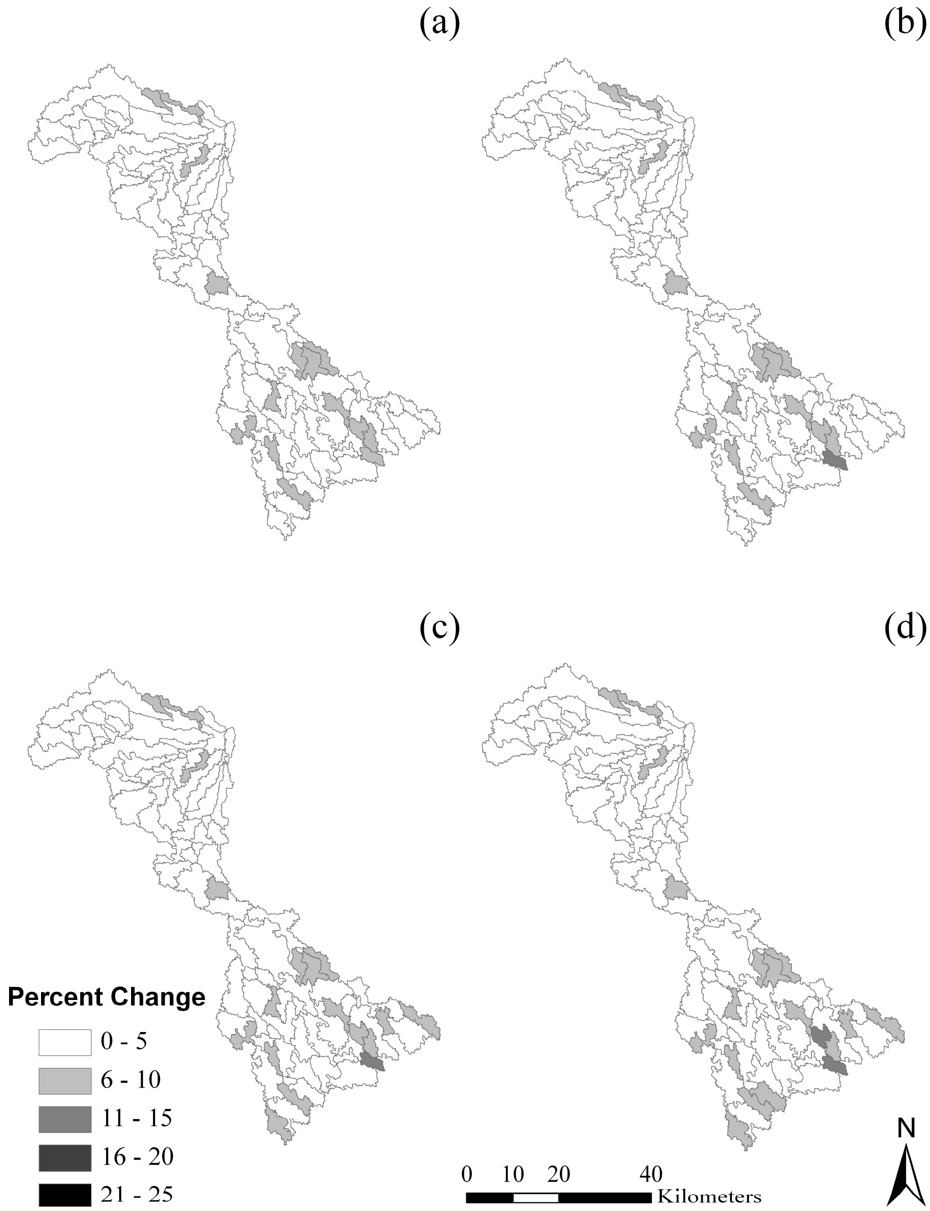

Figure 8-13. Percent flow change at the subbasin outlet compared to base scenario – 100 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

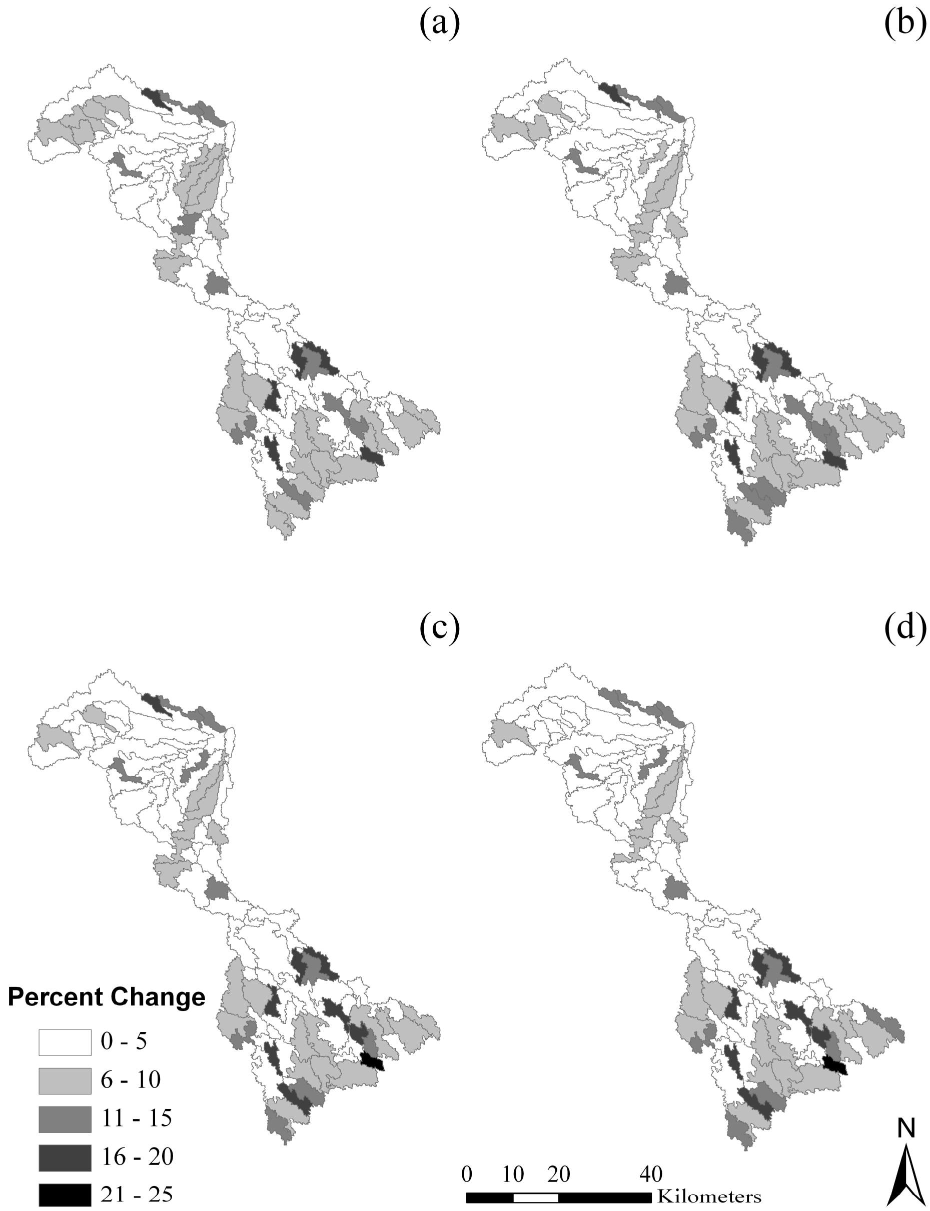

Figure 8-14.Percent flow change at the subbasin outlet compared to base scenario – 250 ha wetlands at depths of (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

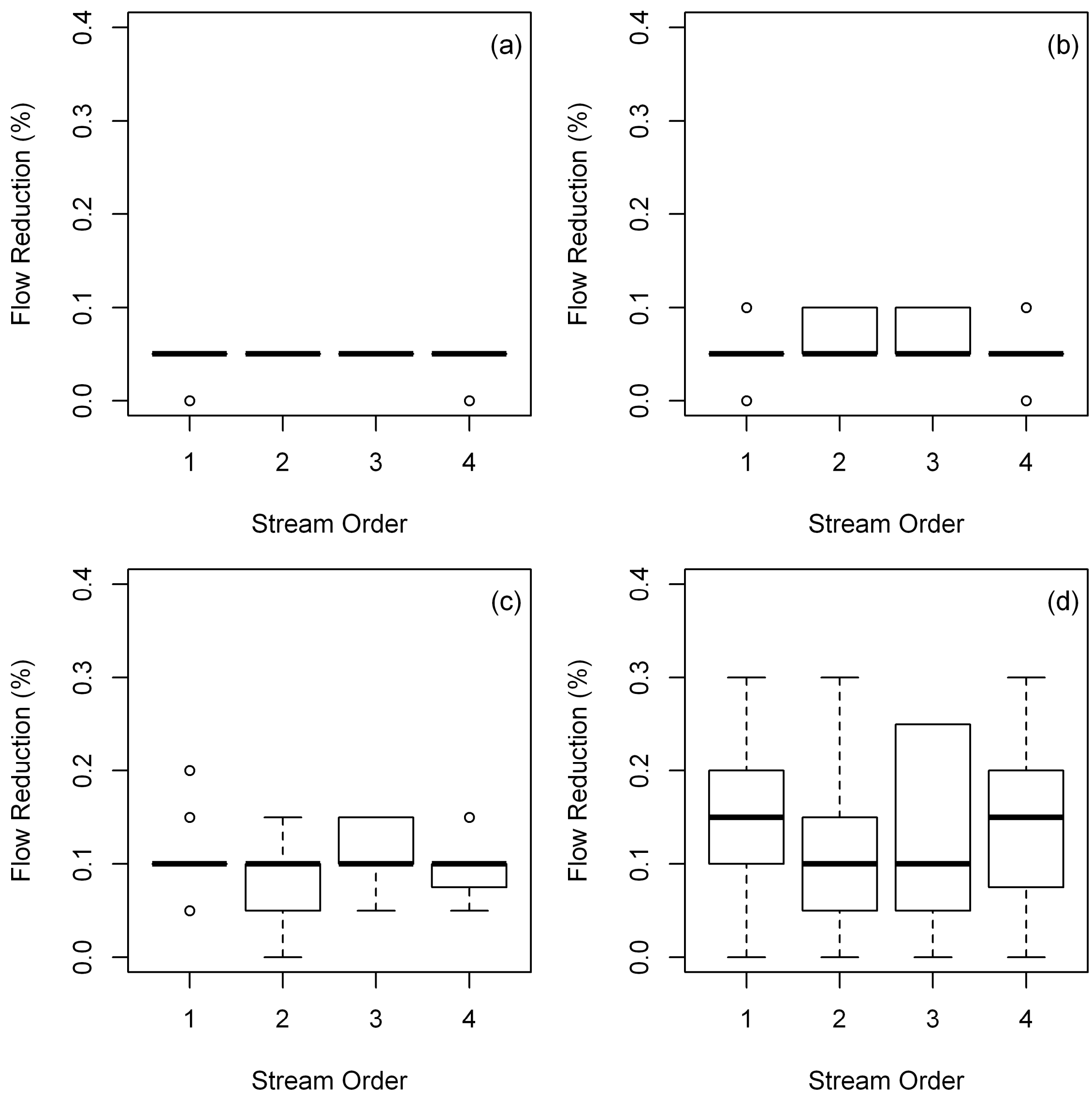

Figure 8-15. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 15 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

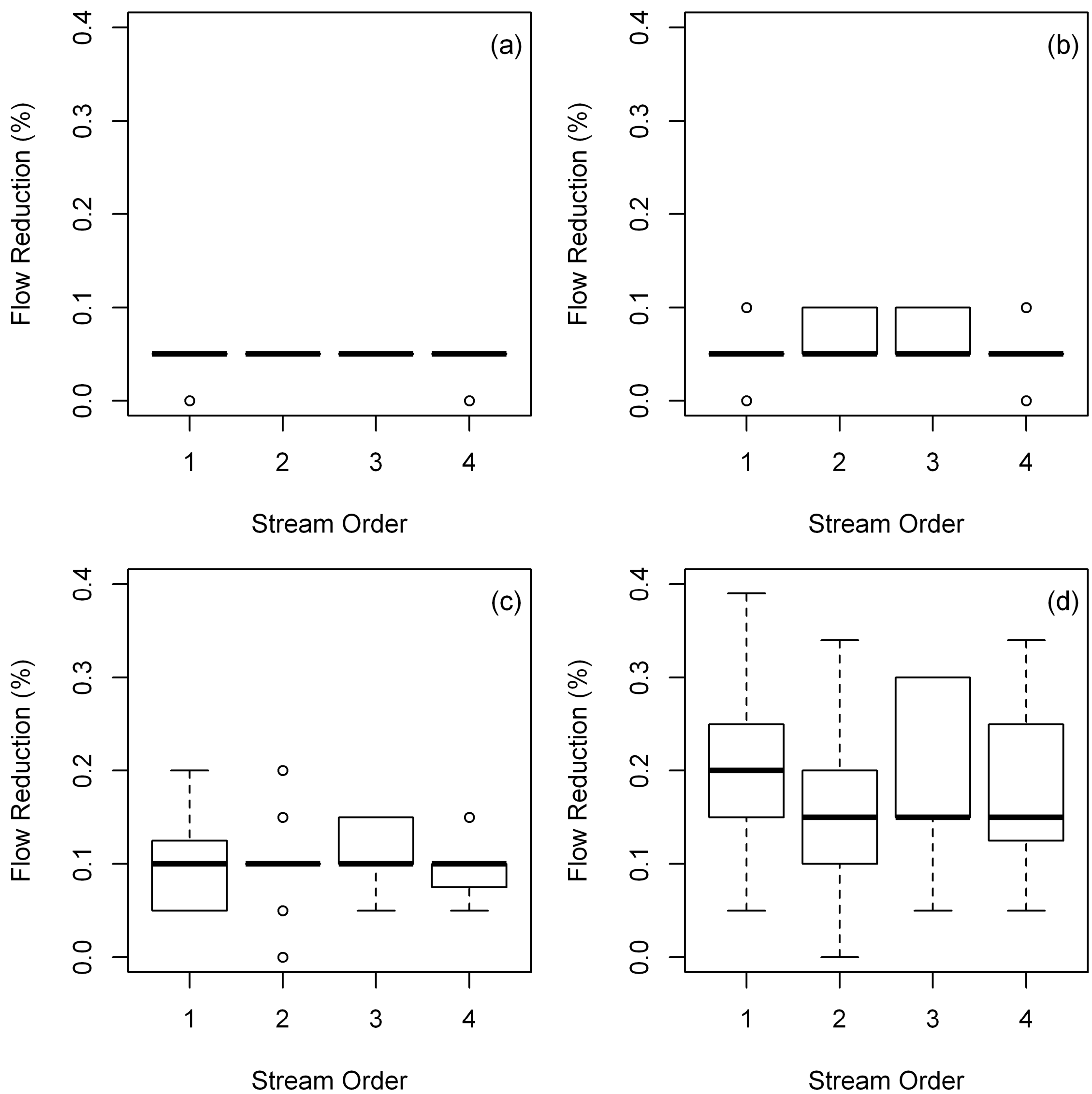

Figure 8-16. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 30 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

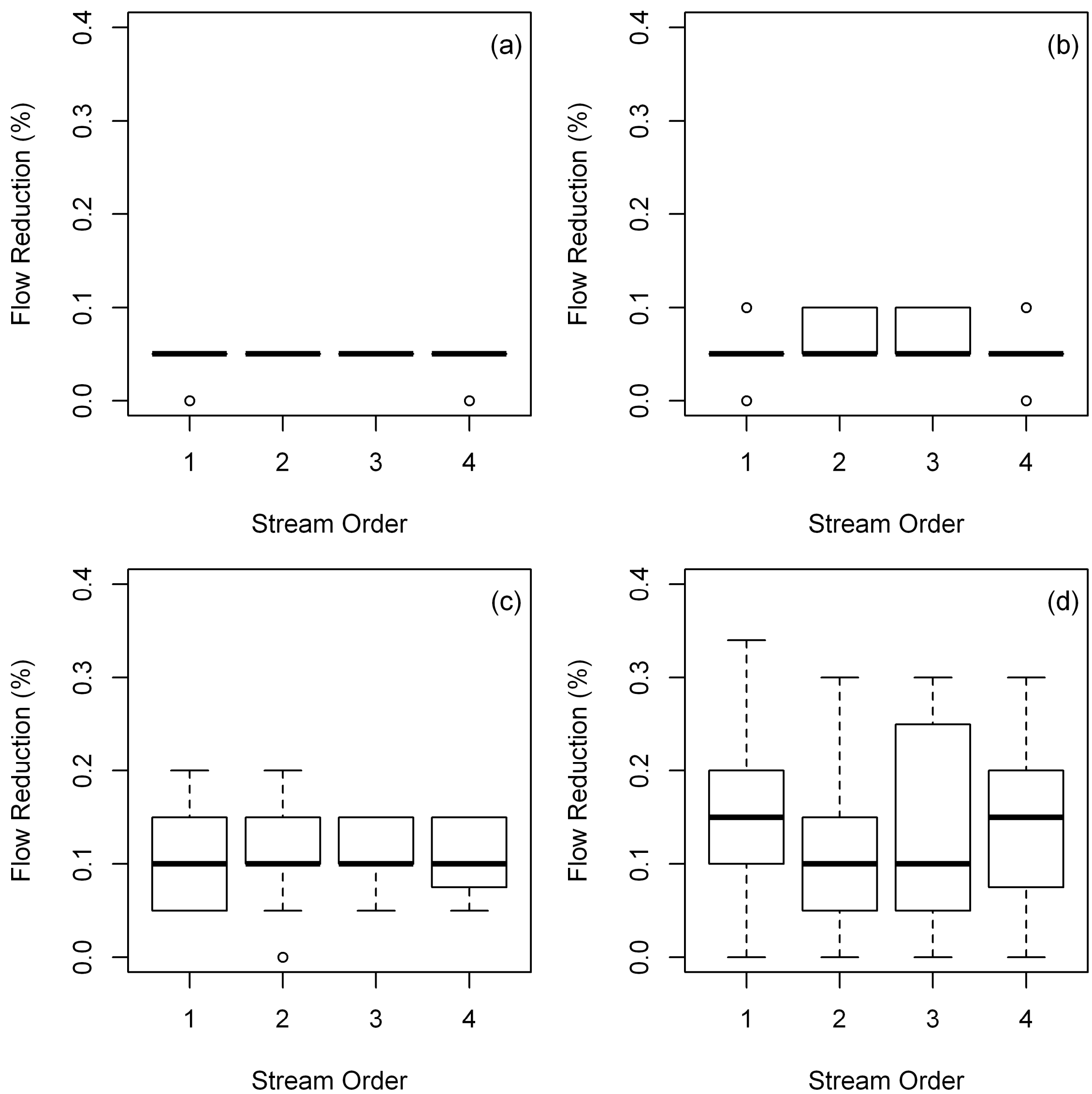

Figure 8-17. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 61 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

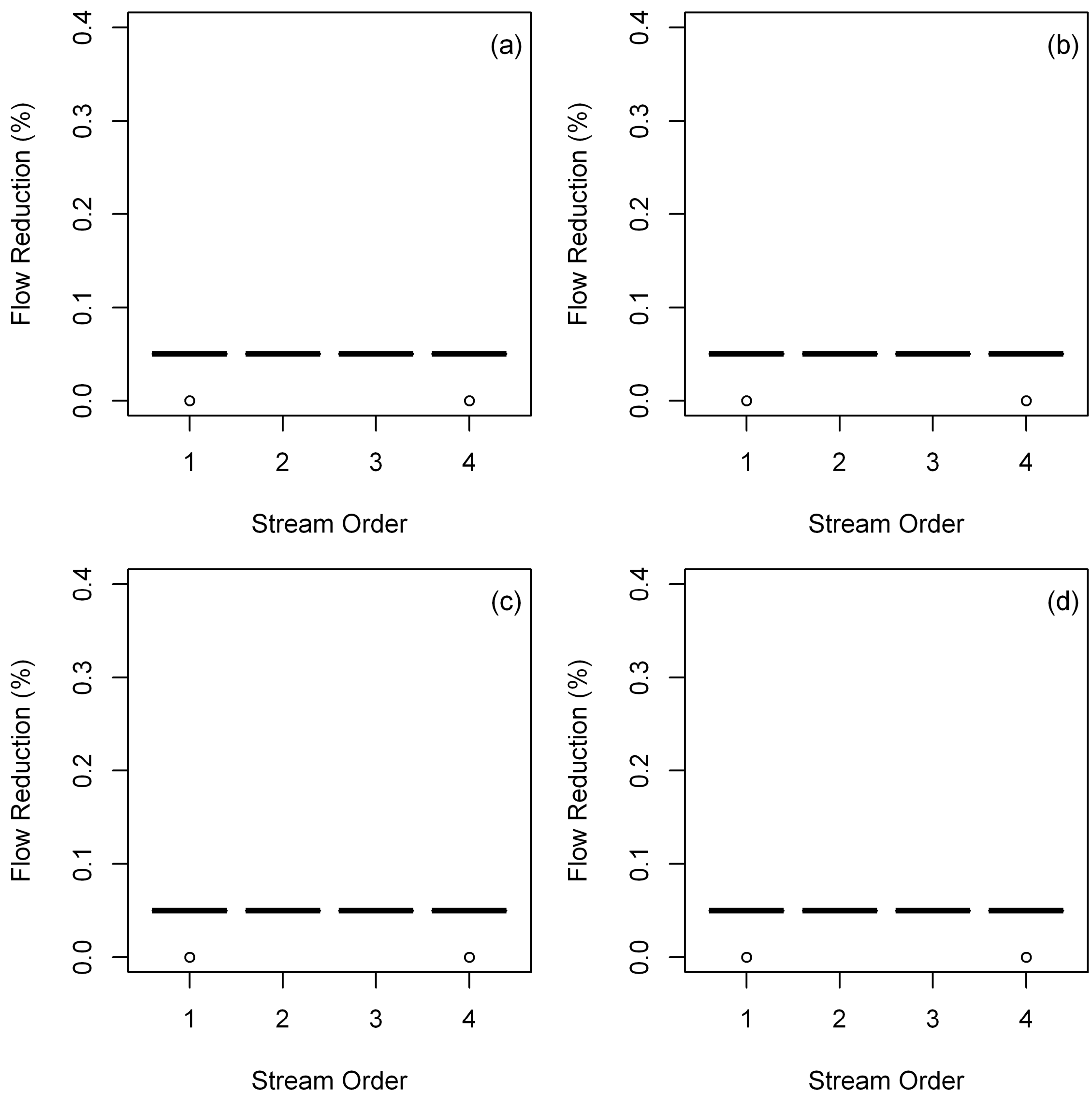

Figure 8-18. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 50 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

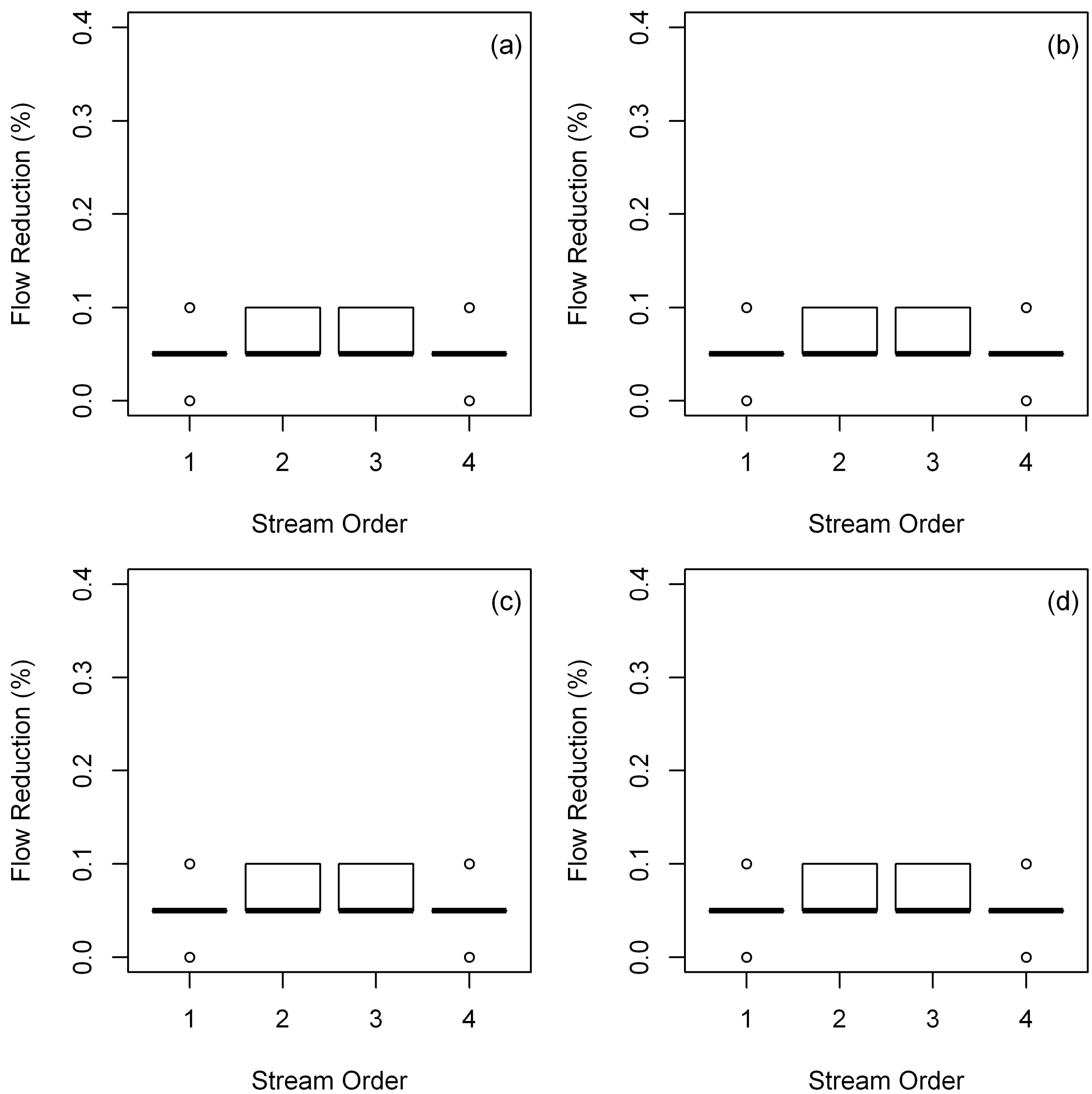

Figure 8-19. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 100 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

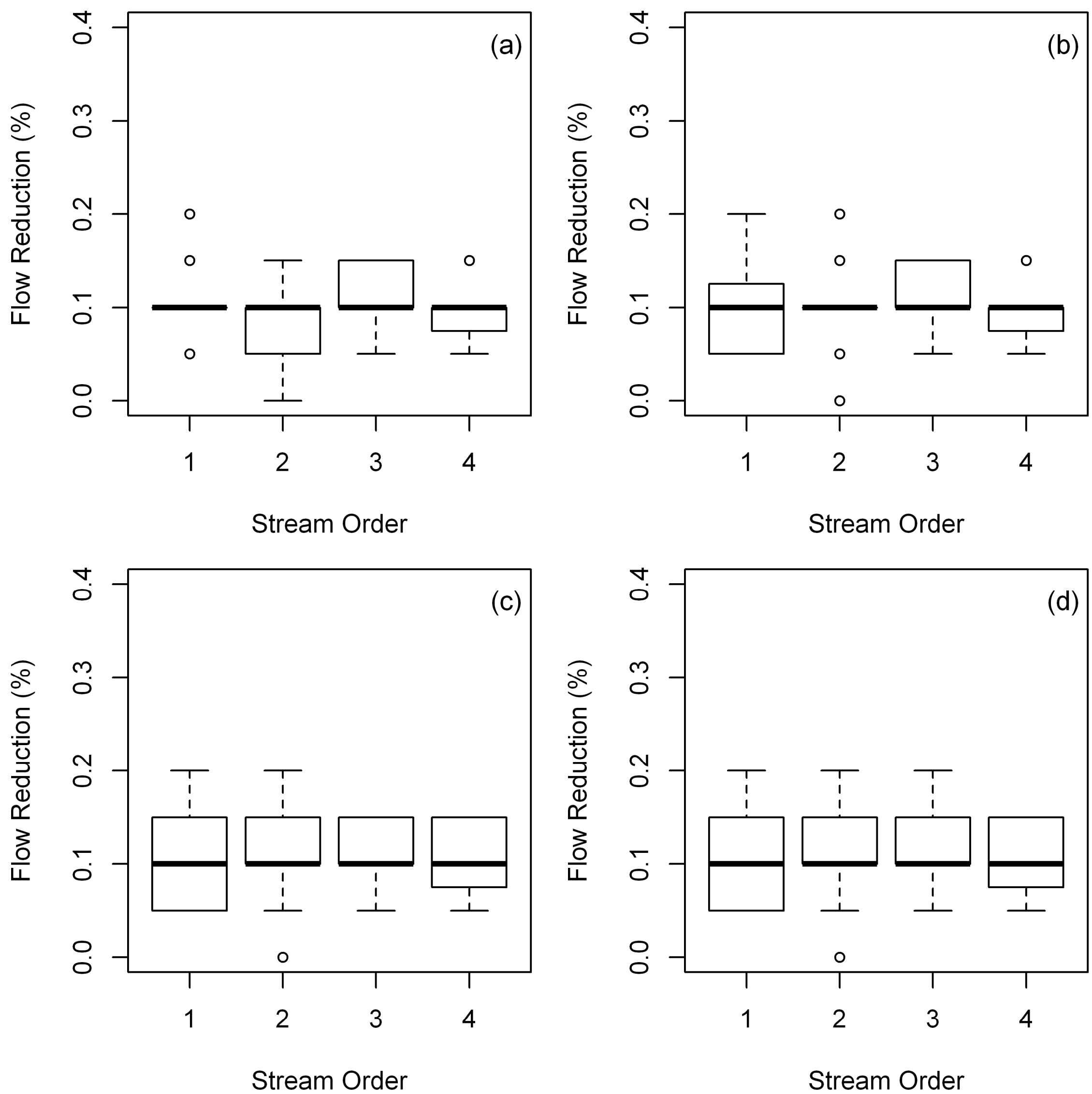

Figure 8-20. Percent streamflow change at the watershed outlet compared to base scenario by stream order for 250 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

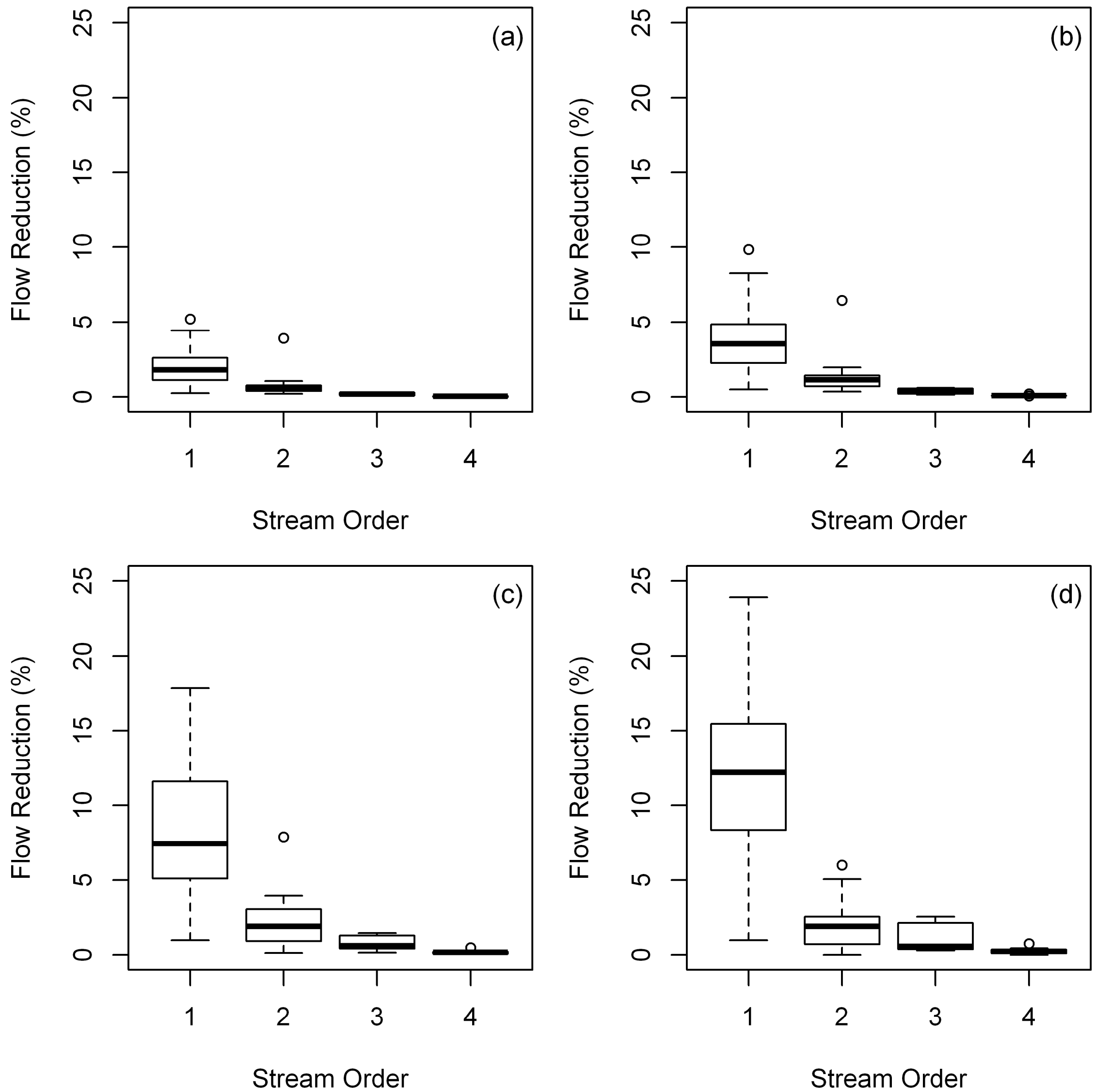

Figure 8-21. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 15 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

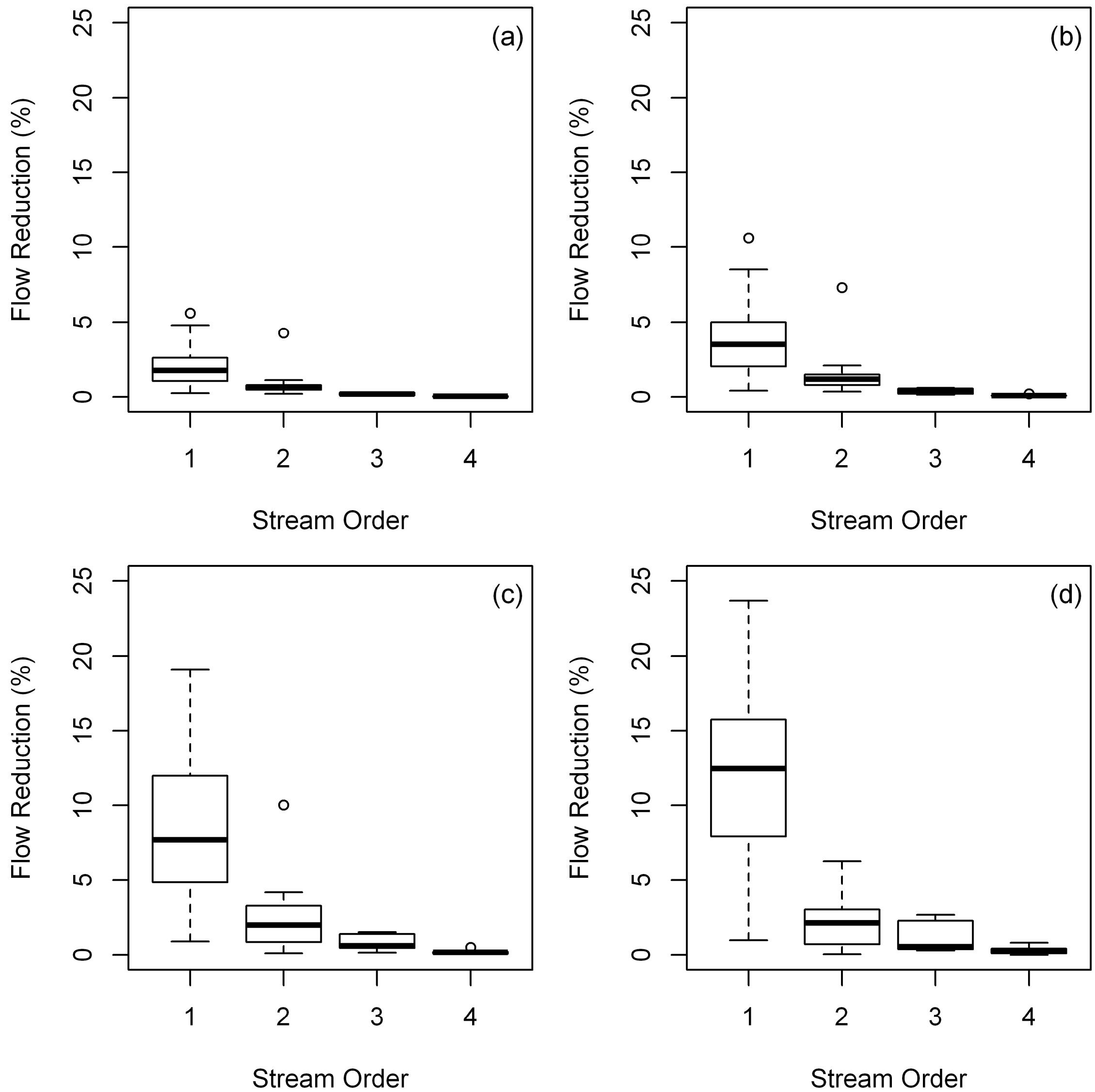

Figure 8-22. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 30 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

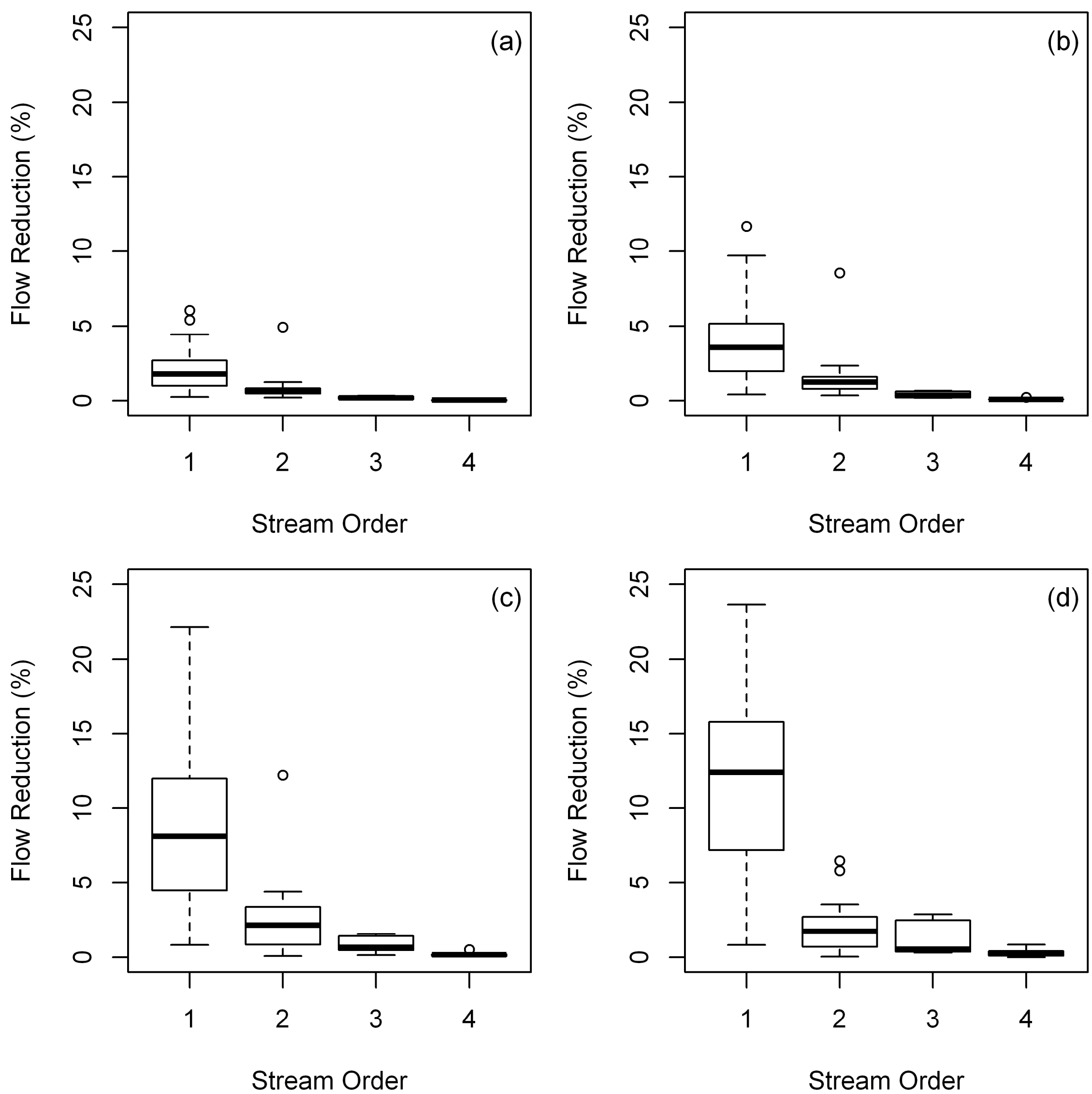

Figure 8-23. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 61 cm and (a) 50 ha, (b) 100 ha, (c) 250 ha, and (d) 500 ha

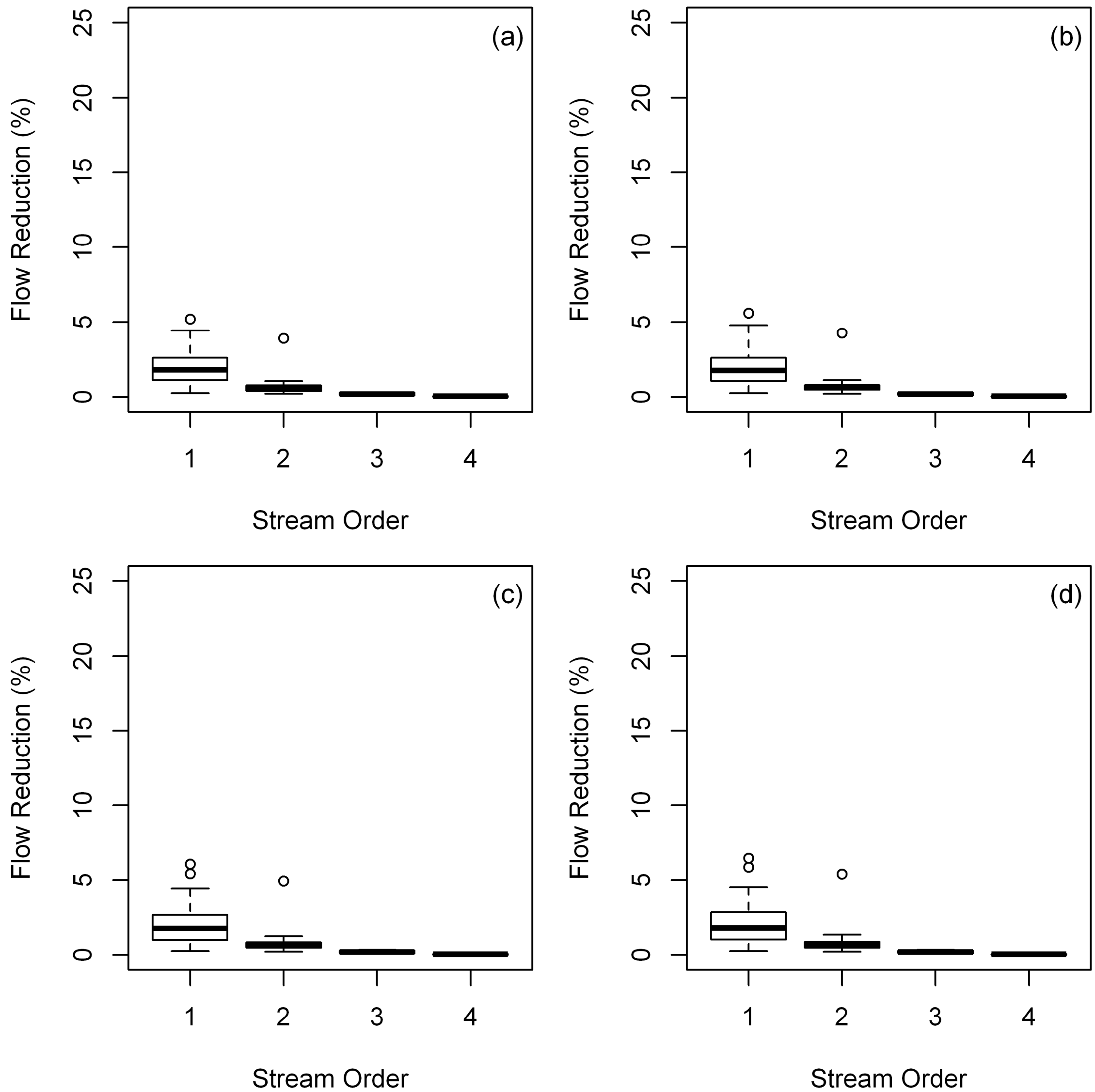

Figure 8-24. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 50 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

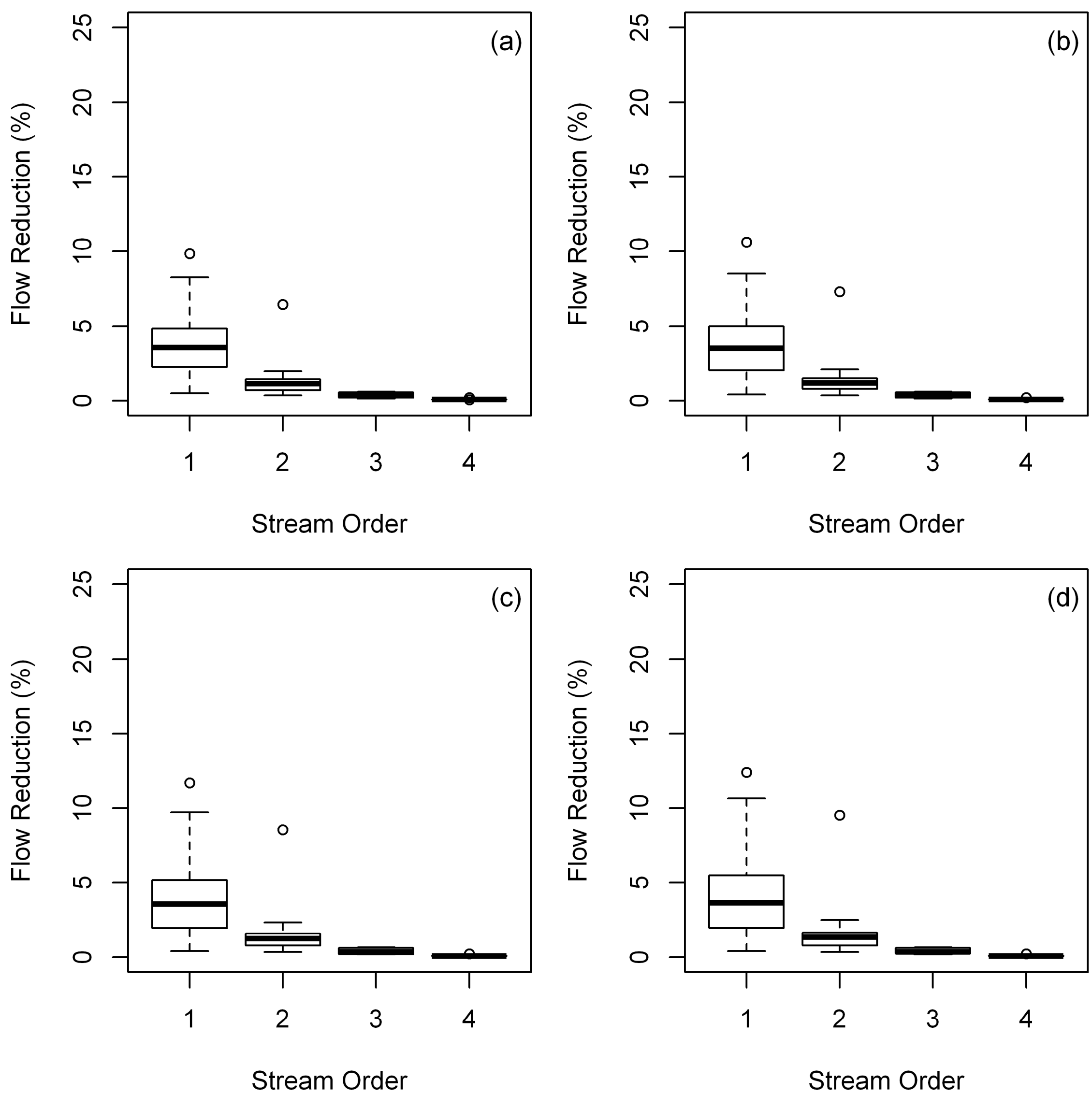

Figure 8-25. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 100 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

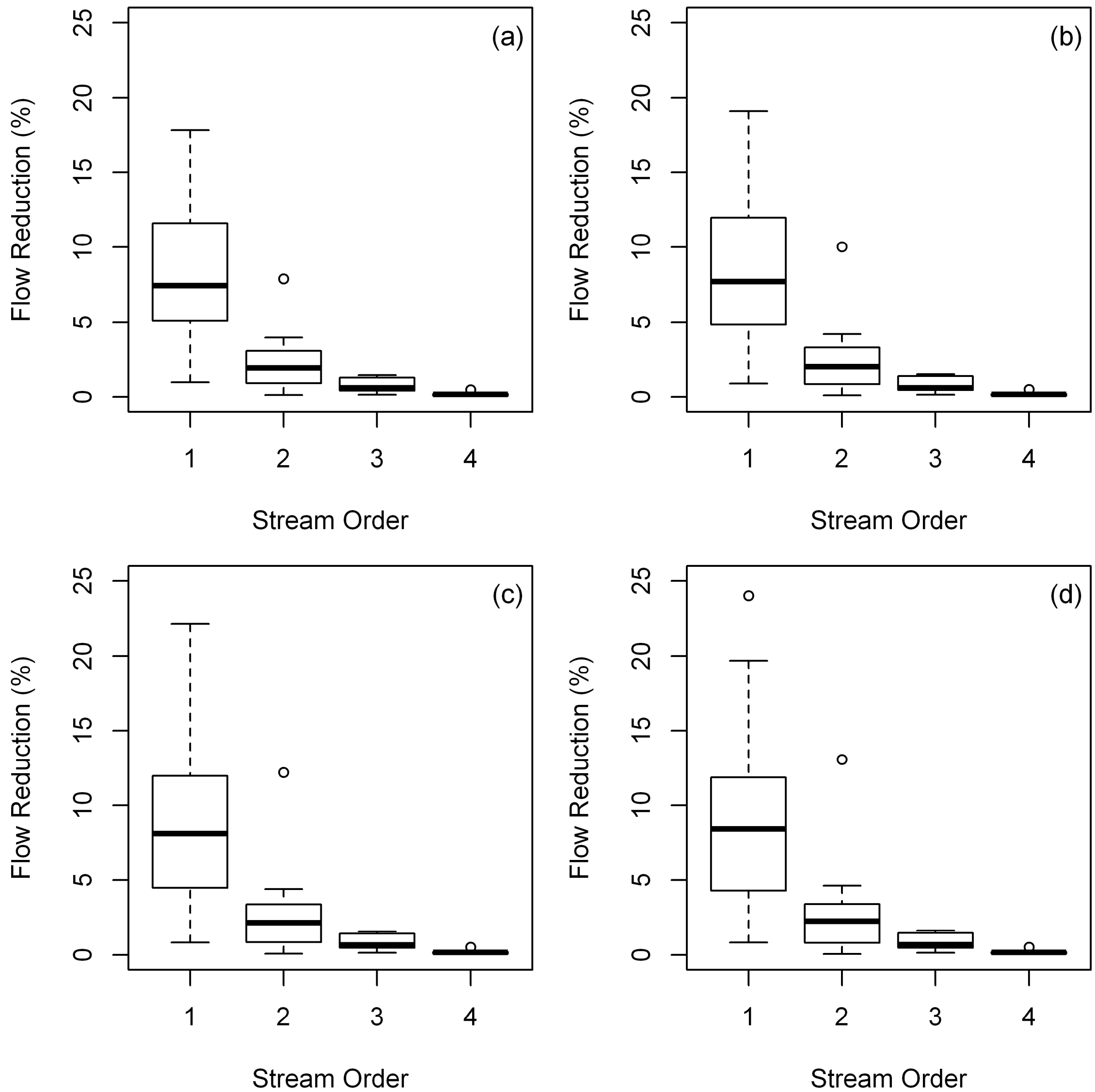

Figure 8-26. Percent streamflow change at the subbasin outlet compared to base scenario by stream order for 250 ha and (a) 15 cm, (b) 30 cm, (c) 61 cm, and (d) 91 cm

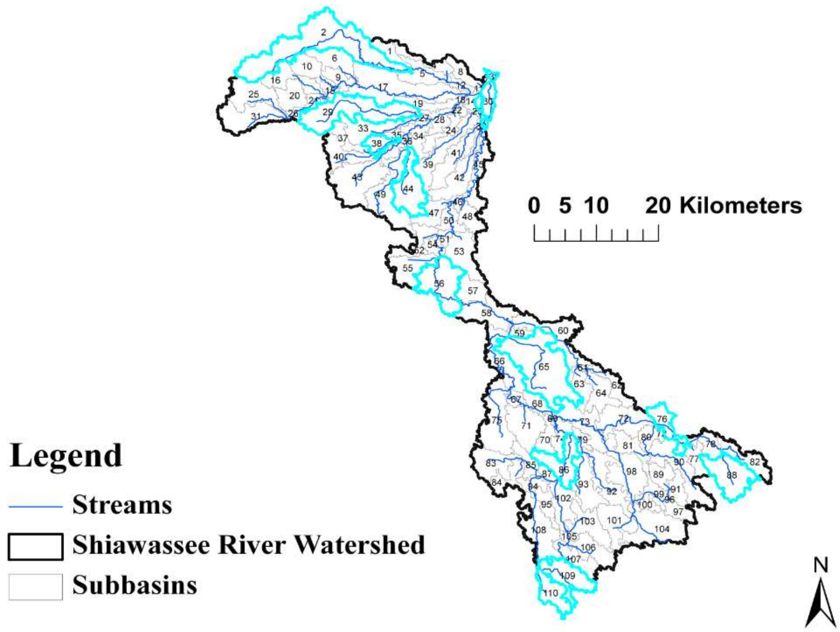

Figure 8-27. Sampled subbasins

Figure 8-28. Measuring wetland depth in the Shiawassee watershed

Figure 8-29. Collecting GPS points for surveyed areas in the Shiawassee watershed

Figure 8-30. Soil hydrology data collection for a farmed wetland

APPENDIX B

ADDITIONAL MATERIAL TO SECTION 5 TITLED: "Assessing the Significance of Wetland Restoration Scenarios on Sediment Mitigation Plan"

Table 9-1. SUSTAIN model parameters that were used in this study

Parameter	Value	Reference	Comments
Substrate properties			
Depth of soil	0.23 m		22.8 cm of media
Soil porosity	0.398	USEPA (2006)	Sandy clay loam soil
Soil field capacity	0.244	USEPA (2006)	Sandy clay loam soil
Soil wilting point	0.136	USEPA (2006)	Sandy clay loam soil
Initial surface water depth	0.09 m		Assumed
Initial moisture content	1		Saturated soil
Saturated soil infiltration	2.54 mm/hr	USEPA (2006)	Clay lining below the soil media
ET multiplier	1		Assumed
Infiltration parameters			
Vegetative parameter A	0.8	USEPA (2006)	Recommended value
Monthly growth index	0.1-1		Assumption based on vegetative growth in Michigan
Water quality parameters			
K	5000 m/year	eWater (2012)	Recommended value
C*	6 mg/l	eWater (2012)	Recommended value
Sediment			
Porosity	0.5	USEPA (2006)	Recommended value
Sand fraction	0.530	USDA-NRCS (2013b)	
Silt fraction	0.251	USDA NRCS (2013b)	
Clay fraction	0.219	USDA NRCS (2013b)	

Table 9-1 (cont'd)

Sand effective diameter	0.1 cm	USEPA (2006)	Average of typical value
Sand velocity	5.3 cm/sec	USEPA (2006)	Average of typical value
Sand density	2.45 g/cm^3	USEPA (2006)	Average of typical value
Sand coefficient	0.255	USEPA (2006)	Average of typical value
Sand exponent	2.5	USEPA (2006)	Average of typical value
Silt effective diameter	0.0034 cm	USEPA (2006)	Average of typical value
Silt velocity	0.013 cm/sec	USEPA (2006)	Average of typical value
Silt density	2.25 g/cm^3	USEPA (2006)	Average of typical value
Silt deposition stress	7.42 Pa	USEPA (2006)	Average of typical value
Silt scour stress	13.17 Pa	USEPA (2006)	Average of typical value
Silt erodibility	48.12 Pa/day	USEPA (2006)	Average of typical value
Clay effective diameter	0.002 cm	USEPA (2006)	Average of typical value
Clay velocity	0.0127 cm/sec	USEPA (2006)	Average of typical value
Clay density	2.25 g/cm^3	USEPA (2006)	Average of typical value
Clay deposition stress	7.42 Pa	USEPA (2006)	Average of typical value
Clay scour stress	13.17 Pa	USEPA (2006)	Average of typical value
Clay erodibility	48.12 Pa/day	USEPA (2006)	Average of typical value

Table 9-2. Sensitivity analysis of the SUSTAIN model parameters

Run	Parameter	Initial Value	New value	Conditions	Flow Change in outflow (%)	Sediments Change in outflow (%)
1	WIDTH	466.69	Not changed	Depends on planned wetland size	-	-
2	LENGTH	933.38	Not changed	Depends on planned wetland size	-	-
3	SAND_FRAC	0.53	0.22	Depends on Soil Type	-42.3 -0	-17.50 -0
4	SAND_FRAC	0.25	0.53	Depends on Soil Type	-42.3 -0	-17.50 -0
5	SAND_FRAC	0.22	0.25	Depends on Soil Type	-42.3 -0	-17.50 -0
6	SILT_FRAC	0.53	0.22	Depends on Soil Type	-42.3 -0	-17.50 -0
7	SILT_FRAC	0.25	0.53	Depends on Soil Type	-42.3 -0	-17.50 -0
8	SILT_FRAC	0.22	0.25	Depends on Soil Type	-42.3 -0	-17.50 -0
9	CLAY_FRAC	0.53	0.22	Depends on Soil Type	-42.3 -0	-17.50 -0
10	CLAY_FRAC	0.25	0.53	Depends on Soil Type	-42.3 -0	-17.50 -0
11	CLAY_FRAC	0.22	0.25	Depends on Soil Type	-42.3 -0	-17.50 -0
12	WEIRH	0.33	Not changed	Depends on planned wetland size	-	-
13	EXITYPE	1	Not changed	Best represent restored wetland	-	-
14	RELEASETYPE	3	Not changed	Best represent restored wetland	-	-
15	POROSITY	0.396	0	Depends on Soil Type	0	0
16	POROSITY	0.396	0.25		0	0
17	POROSITY	0.396	0.5		0	0
18	POROSITY	0.396	1		0	0
19	AVEG	0.8	0	Empirical value (Holtan infiltration Method)	-	
20	AVEG	1.8	0.5		0	0
21	AVEG	2.8	0.75		0	0
22	AVEG	3.8	1		0	0
23	FCAPACITY	0.244	0,0.5, 0.75, 1	Depends on Soil Type	0-0.5	0
24	FCAPACITY	1.244	0.5		0-0.5	0-.9
25	FCAPACITY	2.244	0.75		0-0.5	0
26	FCAPACITY	0.244	1	Depends on Soil Type	0-0.5	0
27	WPOINT	0.136	Not changed	Depends on Soil Type	-	-

Table 9-2 (cont'd)

28	INFILTM	2	0	Depends on Soil Type	0	0
29	INFILTM	2	1	Depends on Soil Type	0	0
30	INFILTM	2	2	Depends on Soil Type	0	0
31	POLROTM	3	1 >	Best represent restored wetland	0	0
32	POLREMM	1	0,1	Best represent restored wetland	0	0
33	SDEPTH	0.75	Not changed	Depends on Soil Type	-	-

Table 9-3. SUSTAIN and SWAT model parameters for models set up.

Parameter	Value	Source	Comments
DEM	NA	National Elevation Dataset (NED) at 30 meter resolution	Required for automatic delineation of drainage areas
Land Use data	NA	Cropland Data Layer (CDL) 56 meter resolution	Required for defining land use distribution
Streamflow data	NA	USGS	Required for calibration of internal modeling of runoff; recommended for system testing
Stream Network	NA	National Hydrography Dataset (NHD)	Required for automatic delineation of drainage areas and for placing on-stream management practices
Precipitation	NA	National Climatic Data Center (NCDC). NCDC Summary	Required for internal land simulation and for estimating storm sizes for the post-processor
Evapotranspiration	NA	National Climate Data Center (NCDC)	Calculated using daily temperature
Latitudes	43.361955	GIS	NA
Land simulation option	External	SWAT	NA
Pollutants	Sediment	NA	Sediment Flag: Sediment
Time series for land use	0 input	NA	Modeled as point source data
Sediment fraction:		Web soil survey	NA
Sand	0.530		
Silt	0.251		
Clay	0.219		
BMP template	**Wet pond**		
Infiltration method	Holtran	NA	To incorporate the effect of vegetation
Pollutant removal	K-C* method	NA	
Pollutant routing method	CSTR in series: 1.5 CSTR	SUSTAIN manual (pp. 3-58), Type B	Value in manual is 1.4, rounded to 1.5
Define BMP parameters			
Wetland area	1, 2, 5, 10 acres	Determinate based on watershed characteristics	
Aspect ratio	2:1	Crites, R. W., Middlebrooks, E. J., & Reed, S. C. (2010). *Natural wastewater treatment systems*. CRC Press.	0.25:1—4:1
		USEPA, 2000. Constructed wetlands treatment of municipal wastewaters.	>1:1

Table 9-3 (cont'd)

		EPA/625/R-99/010. Cincinnati, Ohio.	
Length	NA	Based on area and aspect ratio	NA
Width	NA 2087.1 ft	Based on area and aspect ratio	NA
Wetland depth	0.33 ft	Based on field data collection	Discussed on Martinez at al., 2013
Weir height	Depth of wetland	NA	NA
Weir width	Width of wetland	NA	NA
Substrate properties			
Depth of soil	0.75 ft	6-9 inch of media	
Soil porosity	0.398	Sandy clay loam soil (Table 3-8 Sustain manual)	Soil type found in the watershed
Soil field capacity	0.244	Sandy clay loam soil (Table 3-8 Sustain manual)	Soil type found in the watershed
Soil wilting point	0.136	Sandy clay loam soil (Table 3-8 Sustain manual)	Soil type found in the watershed
Initial surface water depth	0.3 ft	Assumed	NA
Initial moisture content	1	Saturated	NA
Saturated soil infiltration	0.01 in/hr	Clay lining (Table 3-8 Sustain manual)	Clay lining below the soil media
ET multiplier	1	NA	NA
Infiltration parameters			
Vegetative parameter A	0.8	Recommended in Sustain manual (pp 3-54)	NA
Monthly growth index	0.1-1	Assumed based on vegetative growth in Michigan	NA
Water quality parameters			NA
K	16400 ft/year	Sustain manual (Table 3-17, pp 3-59)	NA
C*	6 mg/l	Sustain manual (Table 3-17, pp 3-59)	NA
Sediment			
Bed width	Width of wetland	NA	NA
Bed depth	Depth of wetland	NA	NA
Porosity	0.5	HSPF manual recommended value	NA

Table 9-3 (cont'd)

Sand fraction	0.530	Web soil survey	NA
Silt fraction	0.251	Web soil survey	NA
Clay fraction	0.219	Web soil survey	NA
Sand effective diameter	0.05 in	HSPF manual recommended value	NA
Sand velocity	2.1 in/sec	HSPF manual recommended value	NA
Sand density	165.434 lb/ft^3	HSPF manual recommended value	NA
Sand coefficient	0.255	HSPF manual recommended value	NA
Sand exponent	2.5	HSPF manual recommended value	NA
Silt effective diameter	0.00135 in	HSPF manual recommended value	NA
Silt velocity	0.005 in/sec	HSPF manual recommended value	NA
Silt density	140.4629 lb/ft^3	HSPF manual recommended value	NA
Silt deposition stress	0.155 lb/ft^2	HSPF manual recommended value	NA
Silt scour stress	0.275 lb/ft^2	HSPF manual recommended value	NA
Silt erodibility	1.005 lb/ft^2	HSPF manual recommended value	NA
Clay effective diameter	0.0008 in	HSPF manual recommended value	NA
Clay velocity	0.005 in/sec	HSPF manual recommended value	NA
Clay density	140.4629 lb/ft^3	HSPF manual recommended value	NA
Clay deposition stress	0.155 lb/ft^2	HSPF manual recommended value	NA
Clay scour stress	0.275 lb/ft^2	HSPF manual recommended value	NA
Clay erodibility	1.005 lb/ft^2	HSPF manual recommended value	NA
			NA
Wet pond placement	On stream BMP	HSPF manual recommended procedure	NA
Watershed delineation	Import subbasin	HSPF manual recommended procedure	NA
Evaluation factor		HSPF manual recommended procedure	NA
Flow	Average annual flow volume ft^3/year	HSPF manual recommended procedure	NA
Flow	Peak discharge cfs	HSPF manual recommended procedure	NA
TSS	Average annual load (lb/year)	HSPF manual recommended procedure	NA
TSS	Average annual concentration (mg/l)	HSPF manual recommended procedure	NA

Table 9-3 (cont'd)

Point source	SWAT output data	Martinez-Martinez, 2013	NA
Create input file			
Simulation time period	01/01/1996-12/31/2005	Data availability	NA
Land time series time	60 min	NA	NA
BMP simulation time step	15 min	NA	NA
CRRAT	1.5	Default value	NA
Output time step	hourly	NA	NA

Table 9-4. SWAT model parameters that were adjusted during flow calibration procedure

USGS Station 04175600			USGS Station 04176500			USGS Station 04176000		
Parameter	**Value**	**Method**	**Parameter**	**Value**	**Method**	**Parameter**	**Value**	**Method**
CN2	0.75	Multiply	CN2	0.92	Multiply	CN2	0.88	Multiply
ESCO	0.85	Replace	ALPHA_BF	0.90	Replace	ALPHA_BF	0.99	Replace
ALPHA_BF	1.00	Replace	GW_DELAY	7.50	Replace	GW_DELAY	20.00	Replace
SOL_AWC	0.80	Multiply	ESCO	0.85	Replace	ESCO	0.90	Replace
OV_N	0.15	Add	CH_N2	0.01	Replace	CH_N2	0.25	Replace
GW_DELAY	35.00	Replace	CH_K2	40.00	Replace	OV_N	1.50	Multiply
CH_N2	0.10	Replace	OV_N	1.50	Multiply	RCHRG_DP	0.15	Replace
REVAPMN	5.00	Replace	GW_REVAP	0.08	Replace	GW_REVAP	0.05	Replace
EPCO	0.01	Replace	REVAPMN	0.01	Replace	Sol_AWC	0.75	Multiply
CH_K2	20.00	Replace	SOL_AWC	0.90	Multiply	SLSUBBSN	1.40	Multiply
SLOPE	1.30	Multiply	SLSUBBSN	1.25	Multiply	CH_K1	0.50	Replace
SURLAG	0.30	Replace	SLOPE	0.90	Multiply			
CH_N1	0.04	Replace						
CH_K1	0.01	Replace						

Table 9-5. SWAT model parameters that were adjusted during sediment calibration procedure for station STORET 580046

Parameter	Value	Method
CHCOV1	0.10	Replace
CHCOV2	0.65	Replace
USLE_P	0.82	Replace
ADJ_PKR	0.98	Replace
PRF	0.05	Replace
SPCON	0.01	Replace
SPEXP	1.47	Replace

Table 9-6. Cost summary for establishment and maintenance of a 0.4-hectare wetland over 10-year period (USDA-NRCS, 2013a)

Wetland Restoration Components	Unit	Required Amount	Cost Range (US $ in 2013)		
			Low	Average	High
Site Preparation	ha	0.40	175	181	250
Excavation	m^3	2.76	73	109	146
Backfilling/Grading and Finishing	m^3	2.76	49	73	97
Vegetation Planting	ha	0.40	190	383	575
Trees & Shrubs Site Preparation	ha	0.40	387	554	700
Operation and Maintenance	ha	0.40	190	190	190
		Total Cost	$1,064	$1,490	$1,958

Table 9-7. Cost summary for establishment and maintenance of a 0.81-hectare wetland over 10-year period (USDA-NRCS, 2013a)

Wetland Restoration Components	Unit	Required Amount	Cost Range (US $ in 2013)		
			Low	Average	High
Site Preparation	ha	0.81	350	362	500
Excavation	m^3	3.90	103	154	207
Backfilling/Grading and Finishing	m^3	3.90	69	103	138
Vegetation Planting	ha	0.81	380	766	1,150
Trees & Shrubs Site Preparation	ha	0.81	774	1,108	1,400
Operation and Maintenance	ha	0.81	380	380	380
		Total Cost	$2,056	$2,874	$3,774

Table 9-8. Cost summary for establishment and maintenance of a 2-hectare wetland over 10-year period (USDA-NRCS, 2013a)

Wetland Restoration Components	Unit	Required Amount	Cost Range (US $ in 2013)		
			Low	Average	High
Site Preparation	ha	2.00	875	905	1,250
Excavation	m^3	6.17	163	244	327
Backfilling/Grading and Finishing	m^3	6.17	109	163	218
Vegetation Planting	ha	2.00	950	1,915	2,875
Trees & Shrubs Site Preparation	ha	2.00	1,935	2,770	3,500
Operation and Maintenance	ha	2.00	950	950	950
		Total Cost	$4,982	$6,947	$ 9,120

Table 9-9. Cost summary for establishment and maintenance of a 4-hectare wetland over 10-year period (USDA-NRCS, 2013a)

Wetland Restoration Components	Unit	Required Amount	Cost Range (US $ in 2013)		
			Low	Average	High
Site Preparation	ha	4.00	1,750	1,810	2,500
Excavation	m^3	8.72	231	345	462
Backfilling/Grading and Finishing	m^3	8.72	154	231	308
Vegetation Planting	ha	4.00	1,900	3,830	5,750
Trees & Shrubs Site Preparation	ha	4.00	3,870	5,540	7,000
Operation and Maintenance	ha	4.00	1,900	1,900	1,900
		Total Cost	$9,805	$13,656	$17,920

Table 9-10. Statistical comparison of the sediment reduction provided by different wetland surface areas and stream orders at subbasin level

Stream order	No. of subbasins	Average annual sediment yield (tons)	Average annual sediment reduction (tons) for different stream order based on wetland sizes (vertical comparison)				Average annual sediment reduction (tons) for different wetland sizes based on stream orders (horizontal comparison)			
			0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)	0.4 (ha)	0.81 (ha)	2 (ha)	4 (ha)
1	571	162.7[a]	135.1[a]	149.7[a]	160.0[a]	162.1[a]	135.1[a]	149.7[a]	160.0[a]	162.1 [a]
2	273	130.1[ab]	108.3[ab]	119.4[ab]	127.8[ab]	129.8[ab]	108.3[a]	119.4[a]	127.8[a]	129.8[a]
3	128	136.9[ab]	113.7[ab]	125.1[ab]	134.1[ab]	137.0[ab]	113.7[a]	125.1[a]	134.1[a]	137.0[a]
4	104	80.3[b]	68.7[b]	76.0[b]	78.0[b]	80.4[b]	68.7[a]	76.0[a]	78.0[a]	80.4[a]
5	31	135.8[ab]	129.8[ab]	135.0[ab]	135.7[ab]	135.8[ab]	129.8[a]	135.0[a]	135.7[a]	135.8[a]
6	13	59.1[ab]	56.0[ab]	58.4[ab]	59.1[ab]	59.1[ab]	56.0[a]	58.4[a]	59.1[a]	59.1[a]

*Mean values followed by the same letters are not significantly different (α = 0.05).

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