Wetland restoration in an agricultural basin. Linking hydrologic response, optimal location, ecosystem services and stakeholder opinions.

A case-study for the Le Sueur River Basin, USA

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I. Acknowledgements

This master thesis and research has been performed as part of my master Sustainable Development at the University of Utrecht. Part of this research has been conducted in the USA as a collaborative effort between University of Utrecht and Johns Hopkins University in Baltimore. I would like to express my gratitude towards several persons that have critically evaluated my work and showed continuous enthusiasm throughout the project. First and foremost, I would like to express my gratitude to my supervisors at JHU and UU; Peter Wilcock, Se Jong Cho and Aat Barendregt. Second, I would like to thank Chuck Regan from MPCA who has provided guidance on HSPF which has been invaluable for completion of this thesis. Lastly, I would like to express my gratitude to the larger research community working on similar issues in the Greater Blue Earth River Basin that together have provided valuable reflection, discussion and input for my work.

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III. List of Abbreviations

(US)EPA –Environmental Protection Agency

AGWETP – Active Groundwater Evapotranspiration Potential

AGWRC – Active Groundwater Recession Constant

BASETP – Potential Evapotranspiration from Baseflow

CEPSC – Interception storage capacity

CFS – Cubic Feet per Second

CISSR -Collaborative Implementation Strategy for Sediment Reduction

CTI – Compound Topographic Index or Topographic Wetness Index

DU – Ducks Unlimited

ET - Evapotranspiration

EV- Evaporation

EVT – The combined effect of EV + ET

FDC – Flow Duration Curves

FWMC – Flow weighted mean concentration

GBERB – Greater Blue Earth River Basin

HBN- HSPF Binary Output

HSPF – Hydrologic Simulation Program FORTRAN

INFILT – Index to Infiltration capacity

INTFW – Interflow Inflow

IPCC – Intergovernmental Panel on Climate Change

IRC – Interflow Recession Constant

IVOL – Inflow to the stream

KVARY – parameter for nonlinearity of AGWRC

LSRB – Le Sueur River Basin

LZETP – Lower Zone Evapotranspiration Potential

LZS- Lower Zone Storage

LZSN – Lower Zone Storage Nominal

MDA – Minnesota Department of Agriculture

MN - Minnesota

MPCA – Minnesota Pollution Control Agency

MRB – Minnesota River Basin

NSUR – Roughness of overland flow plane

TMDL – Total Maximum Daily Load

SRES – Special Report on Emissions Scenarios

TSS – Total Suspended Solids

UCI – User Control Input

USDA – United States Department of Agriculture

USGS – United States Geological Survey

UZS- Upper Zone Storage

UZSN – Upper Zone Storage Nominal

1. Introduction

Climate change will affect precipitation patterns on a global and regional scale. In northern latitudes climate change is very likely to lead to an increase in precipitation and to more extreme precipitation events in general (IPCC, 2007). For instance, from three SRES scenario runs a precipitation event in North America with a return period of 20 years is predicted to have a return period of 8-15 years between 2080-2100, clearly showing the effect climate change is likely to have on extreme rainfall events (Field et al. ,2012).

Changes in the precipitation pattern affect stream flow in rivers. Previous research has correlated higher flows to observed increases in precipitation in the Midwest of the USA (Nangia et al., 2010; Johnson et al., 2009; Novotny & Stefan, 2007). However, other studies have argued that the increase in stream flow is mainly driven by land-use changes. Cropping patterns in the USA and especially in the Midwestern Corn Belt underwent major changes in the last half of the 20th century (USDA, 2011). In the wetland-rich landscapes of the upper Mississippi basin, 20th century crop conversions have led to an intensification of artificial drainage which is now a critical component of modern day agriculture (Schottler et al., 2013). However, the role of tile drainage in altering the hydrology of a large basin is still poorly understood (Blann et al., 2009). Previous research has found that at the plot scale, tile drainage tends to increase rapid drainage during storm events through surface inlets and macropores in the unsaturated soil zone (Magdalene, 2004; Chapman et al., 2005). Schottler et al. (2013) constructed a water budget for 21 Minnesota watersheds and concluded that watersheds with large land-use changes had increases in seasonal and annual water yields of >50% since 1940. On average, changes in precipitation and crop evapotranspiration explained less than one-half of the increase, with the remaining highly correlated with artificial drainage and loss of depressional areas (Schottler et al., 2013). A set of studies performed in Iowa found that storm flows, annual baseflows, and minimum flows have commonly increased over the second half of the 20th century, and the increase is more than can be accounted for by climatic changes. The hydrologic changes in Iowa have been partly attributed to artificial drainage but also to the incision and widening of streams (Schilling & Libra, 2003; Zhang & Schilling, 2006; Schilling & Helmers, 2008).

Although the main source of stream flow alterations is still highly debated consequences can already be observed. Schottler et al. (2013) conclude that increases in stream flow lead to more erosive rivers and a recent study by Belmont et al. (2011) in the Minnesota River Basin (MRB) concludes that the dominant source of sediment has shifted from agricultural soil erosion to accelerated erosion of nearby river sources. The erosion of nearby river sources is driven by increased river discharge (Lenhart et al., 2011; Donner et al., 2004; Raymond et al., 2008). Rivers located in the Corn Belt region with intensively row-cropped agriculture are often impaired by excessive sediment loads leading to a degradation of their habitat and recreational value, increasing the cost of water treatment and negatively impacting downstream surface waters (Payne, 1994; Thoma et al., 2005; Engstrom et al., 2009; US EPA, 2013). High erosion rates in the upper Mississippi basin eventually contribute to the creation of a hypoxic zone in the Gulf of Mexico (Johnson et al., 2009). For the above presented reasons sediment load and consequently turbidity is a leading cause of impairment in U.S. rivers and streams (US EPA, 2011; Palmer et al., 2000; Johnson et al., 2009).

Management of excessive sediment loads and watershed hydrology in many agricultural basins in the Upper Midwest is a necessity. Given the link between increased stream flow patterns and increased erosion rates in many agricultural landscapes in the Upper Midwest efforts to mitigate excessive sediment loads and turbidity must include strategies to manage watershed hydrology and reverse conditions contributing to higher flows. Restoration of degraded upland storage capacities of watersheds by 're-creating' or restoring spatially distributed networks of storage systems has been proposed as a promising strategy for runoff management, and for supporting downstream runoff control structures (Babbar-Sebens et al., 2013). Lemke and Richmond

(2009) suggest that re-naturalization of the hydrologic cycle using ecological solutions can solve both water quantity and water quality problems. Wetland ecosystems are considered as such a potential ecological solution for increasing the capacity of watersheds to store runoff water upstream. Especially in tile-drained landscapes, wetlands constructed to intercept tiles can serve as storage basins for agricultural runoff, leading to both reduction in peak stream flow and diminished transport of agricultural nutrients (Babbar-Sebens et al., 2013). A critical remark is provided by a research on coastal wetlands, Sun et al. (2002) challenge the general conception that wetlands always store water because one must consider the antecedent soil moisture when evaluating wetland hydrologic functioning. They conclude that wetland storage capacity is finite and when it is exceeded wetlands could behave similarly to uplands in terms of response to rainfall events. The above discussion clearly shows that it is important to determine the effect wetland restoration will have on stream flow, and especially on peak stream flow.
Besides the net effect on stream flow other research has focused on the optimal location of wetlands within the landscape. Loucks (1989) argued that wetland restoration far upstream is preferable over downstream wetland restoration. Anderson and Kean (1994) suggest in a study for the Red River Basin in Minnesota that wetlands are most effective when implemented in areas contributing latest to the stream flow, which often coincide with the furthest away areas. For a watershed in Indiana a recent study shows that fewer wetland sites and smaller areas are needed when spatial optimization of wetland area, location and drainage areas are part of the consideration (Babbar-Sebens et al. 2013). The rationale behind optimal wetland location in the latest contributing areas is that increasing storing capacity in these areas will delay or even diminish the peak stream flow contributions. Precipitation that falls directly in early contributing areas should be routed to the outlet area at a faster pace. Wetland restoration in these areas could delay the timing of the peak stream flow event making it intersect with the peak in stream flow from the later contributing areas. This in effect could lead to an increase in the magnitude of peak stream flow events. At the other hand, one could argue that for peak stream flow events most water is generated by the fast responsive areas and these areas constitute a far larger portion of the peak stream flow event. A modeling study performed by Ogawa and Male (1986) concluded that downstream wetland restoration is the most beneficial for flood control. Differences between locations are not only related to distance to a point in the watershed. Jones and Winterstein (2000) compare wetland restoration at different watersheds in the Red River Basin and conclude that differences in stream flow response can be observed even in a relatively homogenous basin and attributed these differences to small changes in localized conditions, like land-use, soil characteristics and antecedent moisture conditions. This study will address the differences between wetland restoration at different locations to determine whether there is an optimal location for wetland restoration. It will study the optimal location based on both distance upstream and localized differences.
Currently the Minnesota Pollution Control Agency (MPCA) is considering a set of management options to reduce sediment load in the Greater Blue Earth River Basin (GBERB) in southern Minnesota. The GBERB is part of the MRB, which is the largest source of sediment to the upper Mississippi River (Engstrom et al., 2009) and the GBERB contributes a disproportional high portion of sediment to the MRB, of up to 50% (Wilcock et al. 2009). In this area increases in sediment have been linked to increases in stream flow and therefore the first step in managing sediment loads is in stream flow management. Wetlands are seen as one of the most promising solutions in this landscape given the historical abundance of wetlands in the area and because of the high percentage of excessively tile drained agricultural land. Since 2009 Minnesota has a state-wide wetland restoration strategy as a supplement to the 1997 Wetland Conservation Plan for Minnesota (Minnesota Board of Water and Soil Resources, 2009). Earlier on implementation of Federal and state legislation in the 1980s led to the establishment of the Conservation Reserve Program and Reinvest in Minnesota Program resulting in the restoration of many wetlands throughout Southwestern Minnesota (Galatowitsch & van der Valk, 1994). Several studies have been performed on the effect of management options on turbidity. A previous study for the entire MRB found that with conventional scenarios it is unlikely to reduce peak stream flow and consequentially sediment load to a satisfactory level (Tetratech, 2009). However, the

scenarios in this report did not include wetland restoration. In a recent report by Baskfield et al. (2013) looking at the effect of different management scenarios on Total Maximum Daily Load (TMDL) for the MRB water storage was mentioned as one of the effective practices. Although wetland restoration was not explicitly taken into account for this study increasing water storage capacity is a clear hydrological function of wetlands. To deal with the sediment issues in the GBERB a multidisciplinary research group has been formed that in collaboration with local stakeholders will determine the effect and extent of possible management options and the linkages between stream flow and erosion of near-channel sources. The combination of the stakeholders and the researchers is referred to as Collaborative Implementation Strategy for Sediment Reduction (CISSR).
This research is part of this larger effort and aims to contribute by assessing the effects of wetland restoration on stream flow and to determine whether an optimal location for wetland restoration within this basin can be determined. This would provide the hydrological background for evaluation of wetland restoration as a management option. However, solely looking at the hydrological effects of wetland restoration is too limited an approach and other considerations to evaluate wetland restoration have to be taken into account. Therefore, this research will discuss several other aspects related to wetland restoration that will likely influence the decision-making process and that allow for a broader comparison between wetland restoration and other management options. First, it will look at the costs of wetland restoration and factors influencing those costs. Second, it will look at the stakeholder support for wetland restoration and the requirements set by stakeholders on wetland restoration. Third, wetlands provide not only flood control but multiple ecosystem services and this study will highlight some of those ecosystem services and factors affecting high or low provision of this service. Moreover, possible conflicts and linkages will be identified between the requirements of local stakeholders and costs and ecosystem services related to wetland in order to assess whether wetland restoration has the potential to be a successful management strategy in an agricultural basin. In order to meet the aim of the study the following two research question are posed:

What is the effect of wetland restoration on stream flow and what is the optimal location for wetland restoration?

What are the requirements set by local stakeholders on wetland restoration and how does this affect cost of wetland restoration and provision of a set of ecosystem services?

A set of sub-question has to be answered to be able to answer the two main research questions:

- What is the effect of wetland restoration on stream flow in general and especially peak stream flow?
- What is the optimal location for wetland restoration within this basin?
- What are important requirements set by farmers on wetland restoration and how does this influence the form and cost of wetland restoration?
- What are the possible linkages and trade-offs between stakeholder demands and the provision of a set of ecosystem services provided by wetland restoration?

It is hypothesized that wetlands will have a significant effect on stream flow and peak stream flow reduction. For stream flow management the most efficient strategy is to implement wetlands in the furthest upstream areas. Wetland restoration is often in direct conflict with farming practices and will therefore be viewed critically by stakeholders. Requirements set for wetland restoration will increase costs and limit the provision of other ecosystem services through wetland restoration.

In order to answer the above posed research questions this study will present a case-study in the Le Sueur River Basin (LSRB), part of the GBERB. The LSRB is selected based on several characteristics; the basin produces the highest sediment yield of all tributaries to the MRB (Wilcock et al., 2009). Next to that, the basin is primarily agricultural and heavily tile-drained and before agricultural intensification the area had a high percentage of wetland areas. This results in a high potential for wetland restoration. Lastly, the area mainly consists of flat uplands whereas most sediment is generated in the downstream areas from near-channel sources driven by increased river discharge (Belmont et al., 2011). The LSRB therefore provides an ideal site for a case-study on wetland restoration in an agricultural and results from this study will not only provide guidance for management in this particular but can also be partly used throughout the entire Corn Belt region.

The first chapter will provide an overview of the watershed and discuss sediment sources, trends in land-use history and the history of the landscape. Then this study will continue with an overview of the methodology used for the analyses and present the main results. The results will be divided in two chapters; in the first chapter the hydrologic effects of wetland restoration will be presented and in the second chapter costs, stakeholder views and ecosystem services related to wetland restoration will be analyzed. The study will end with a discussion on the results and a short and final conclusion on the main findings.

2. Study Area and background

2.1 Study area

The LSRB is located in Southern Minnesota and covers an area of 2.880 km^2. The LSRB combined with the Watonwan River Basin and the Blue Earth River Basin are commonly referred to as the GBERB. The majority of the LSRB lies within the Western Corn Belt Plains ecoregion and a small portion in the North Central Hardwoods Forest ecoregion (Spindler et al., 2012). The 2.880 km^2 area is drained by the Le Sueur River and its two major tributaries the Cobb River and Maple River. The three branches come together within a span of 3 km~10 km upstream of the Le Sueur confluence with the Blue Earth River. The Blue Earth joins the Minnesota River at Mankato, MN, 5 km downstream from the junction with the Le Sueur River and finally drains into the Mississippi River and Lake Pepin. The LSRB has a total of 8 gages all located in the downstream western part of the watershed (figure 1).

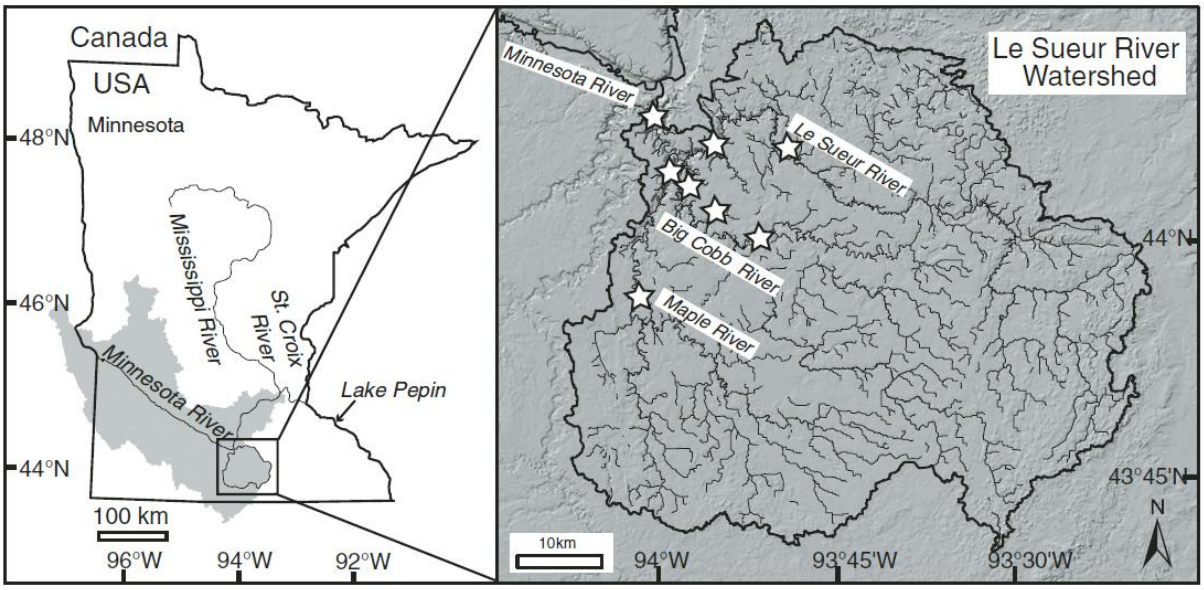


Figure 1: Map showing the location of the Le Sueur River Basin in south-central Minnesota, USA. The shaded area represents the drainage area of the Minnesota River Basin. On the right inset watershed map stars indicate the location of the 8 stream gages (Map from Gran et al., 2009).

The geological substrate is a 60m thick package of semi consolidated but soft fine-grained (67% silt and clay, 33 % sand, < 1% gravel and boulders) tills and glaciofluvial sands (Jennings, 2010). There are no major urban areas although the municipality of Mankato is expanding into the northern part of the watershed (Gran et al., 2009). The area is predominantly agricultural land, of which the vast majority is in soybean-corn row crops (>87%) (Musser et al., 2009). According to the 2001 land cover data-set land cover percentages in the area are: forest (1.5%), rangeland (3.8%), wetland (3.5%), cropland (82.7%), developed/urban (6.4%) and open water (2%) (Spindler et al., 2012). The basin is located in the Corn Belt region, a vast predominantly agricultural region in the Midwestern United States dominated by corn and soybean row-crops (figures 5 and 6).

Average daily stream discharge per year at the mouth of the LSRB between 1995-2009 ranges from 346-1.148 cubic feet per second (cfs) and the maximum daily discharge recorded in this time period is 12.800 cfs on April 6, 2001 (figure 2). Discharge is highest in the months April, May and June. Total average annual precipitation at the closest rain station between 1995-2009 ranges from 22.5-41.5 inches/day and the maximum precipitation event observed is 6.36 inches/day on September 9, 2004 (figure 3). Monthly average precipitation over the entire record is highest in June. Stream flow peaks in early spring (March-April) are mainly driven by snow melt runoff whereas later during the year the stream flow is governed by precipitation and evapo(transpi)ration ratios. For the remainder of this report evaporation will be referred to as EV, evapotranspiration will be referred to as ET and the combination of EV and ET as EVT.

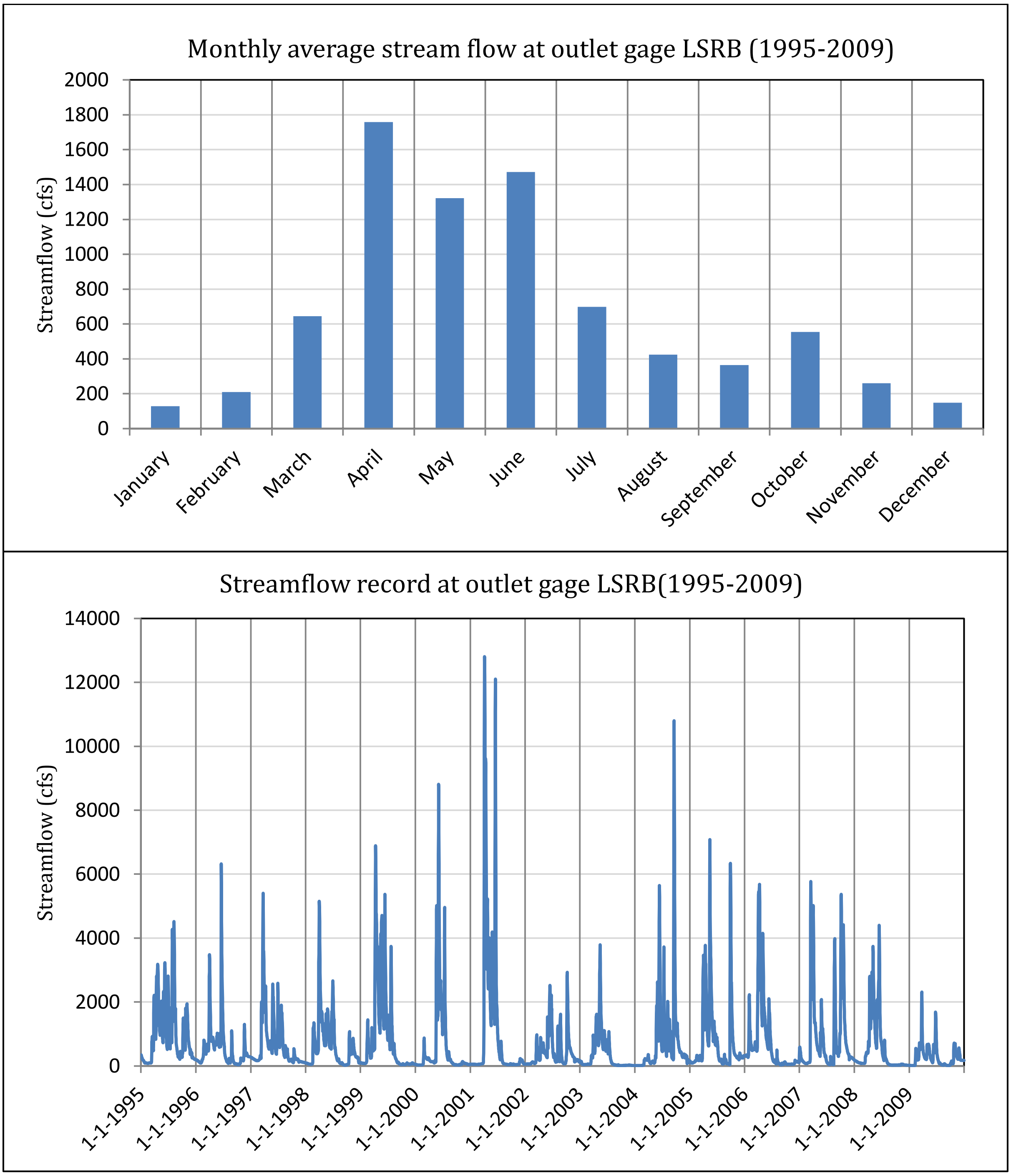


Figure 2: Observed monthly average stream flow and stream flow record for the outlet gage of the LSRB from 1995-2009.

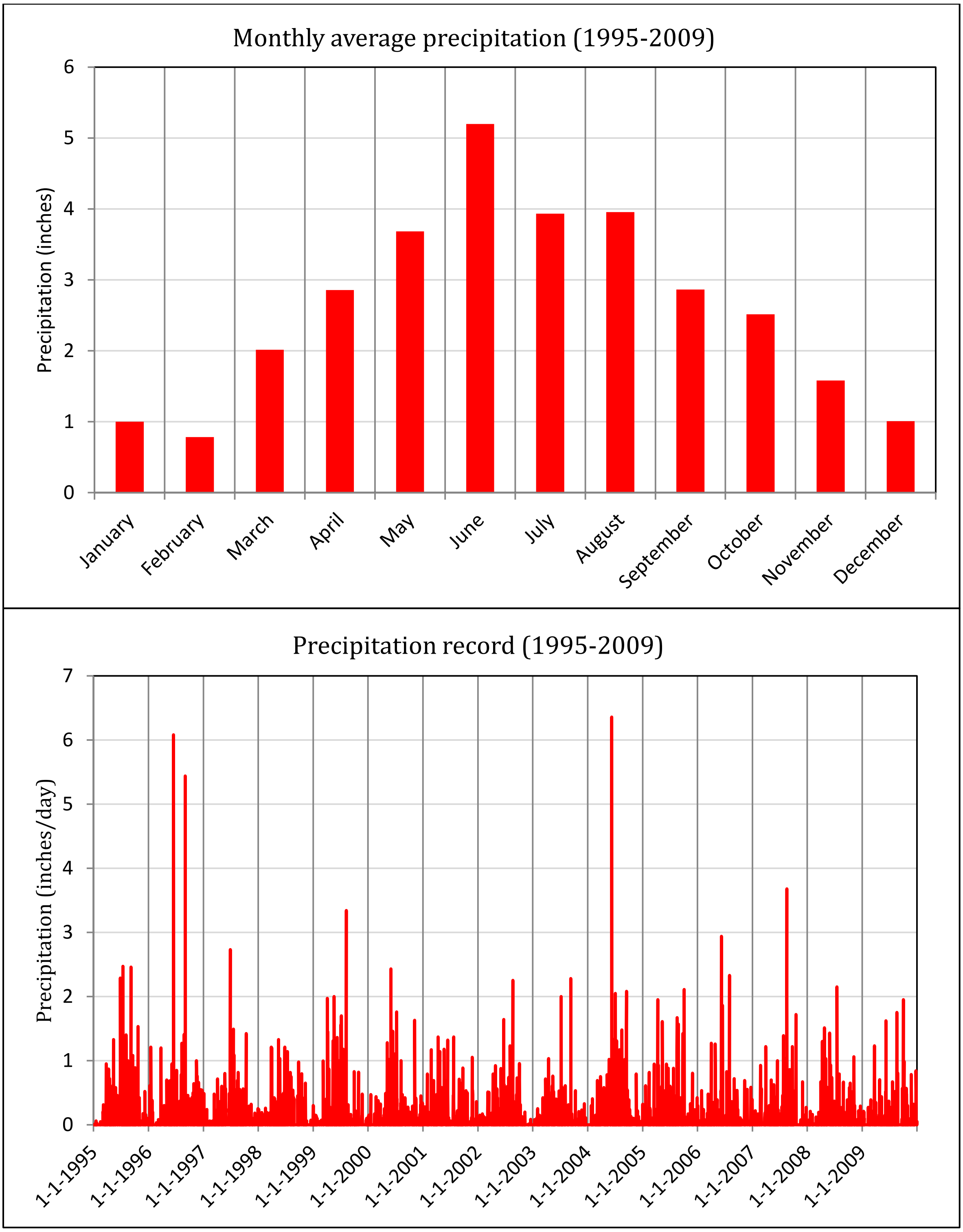


Figure 3: Observed monthly average precipitation and precipitation record for rain station at Mankato (MN), closest to outlet gage, from 1995-2009.

The LSRB produces the highest sediment yield (73.5 Mg/km^2) of all tributaries of the Minnesota River, accounting for 24-30% of the Minnesota River total suspended solids (TSS) load and making it a primary contributor to Lake Peppin (Wilcock et al., 2009; Gran et al., 2009). Given the fact that the LSRB only constitutes 7% of the area of the MRB this is a disproportionately high contribution. The target values set by the MPCA for this region are 58-66 mg L^{-1} (McCollor & Heiskary, 1993). From 2000-2006 concentrations in the LSRB were between 4-16 times as high as the target value (table 1). Engstrom et al. (2009) conclude from research on coring records of the past 500 years that sedimentation rates in Lake Pepin may have increased by as much as an order of magnitude over the past 500 years. The relatively high sediment yield of the MRB stems from a combination of quaternary landscape history and human land and water management. The landscape of the LSRB is due to this history naturally primed for rapid geomorphic change and large sediment supply (Belmont et al., 2011). The following paragraphs will discuss the quaternary landscape history, the relatively recent human alterations to land and water and the relative contribution of sources of sediment for the LSRB.

Table 1: TSS Loads and FWMC in the LSRB, 2000-2006. (Data from Gran et al., 2009)

	2000	2001	2002	2003	2004	2005	2006	Mean
TSS	5.8 x 10^5	4.2 x 10^5	1.1 x 10^5	8.6 x 10^5	4.1 x 10^5	2.7 x 10^5	1.5 x 10^5	2.9 x 10^5
FWMC*	918	355	318	245	475	356	270	420

**FWMC – Flow weighted mean concentration*

2.2 History of the landscape

The Minnesota River Basin is thickly mantled by glacial deposits remaining from large ice sheets that occupied the region as recently as 12.000 years ago (Wilcock et al., 2009). At the end of the last glacial period meltwater formed glacial Lake Agassiz which covered western Minnesota, eastern North Dakota, Manitoba, and western Ontario (Matsch, 1983). For a long time the only outlet was the glacial river Warren, a valley now occupied by the Minnesota River. The large volume of meltwater in combination with occasional extreme floods created a valley much larger than would be associated with a river the size of the Minnesota (Wilcock et al., 2009). River Warren incised creating a valley that was 45 m deep at the mouth and 70 m deep at Mankato (MN), where the rivers of the GBERB drain into the Minnesota River. As a consequence of this history, today the channel is incised 70 m in a valley up to 800 m wide at the mouth of the Le Sueur. High bluffs border many of the outer bends along the channel, and steep ravines snake into the uplands (Gran et al., 2009).

The initial river incision occurred approximately 11,500 years before present (Clayton and Moran, 1982; Matsch, 1983). Prior to the incision, tributaries to the ancestral Minnesota River were low-gradient streams of glacial meltwater origin. With the down cutting of River Warren, these streams were stranded above the master stream and began to cut deeper valleys (Figure 4). On a geologic time scale, this incision is just getting underway and is readily visible in air photos in portions of the basin along the Minnesota River (Wilcock et al., 2009). As the tributary rivers cut down, incision of their tributaries together with erosion of the valley sides increases the supply of sediment to the river. This sharp incision at the mouth of the LSRB is in sharp contrast with the rest of the area. By far the largest portion of the LSRB is dominated by low gradient flat-uplands, sometimes referred to as 'the grand surface'. In these areas sediment rates are mainly governed by agricultural practices. The knick-point is the location in the landscape where the rivers start incising. This knick-point is characterized by a sharp drop in base-level or a sharp increase in channel gradient. The knickpoint has migrated 40 km up the Le Sueur river network leading to a rapid vertical incision (Belmont, 2011; Gardner, 1983). This migrating reach is commonly referred to as the knick zone. The landscape history in the MRB and the LSRB has created two distinctive zones each with its own main sediment sources. These zone will be referred to as the uplands, the area upstream of the incision characterized by very low relief, and the incised or knick zone, the area at and downstream of the knickpoint with sharp relief, high bluffs and ravines moving into the uplands. The incised zone, although small in area, can supply large amounts of sediment from erodible glacial deposits in a setting of steep slopes and incising river channels (Wilcock et al., 2009). In the relatively very large area of the upland zone erosion rates are generally smaller but the rates of erosion have increased considerably since European settlement due to changing land practices (Mulla & Sekely, 2009). The changing land practices will be discussed in the next paragraph.

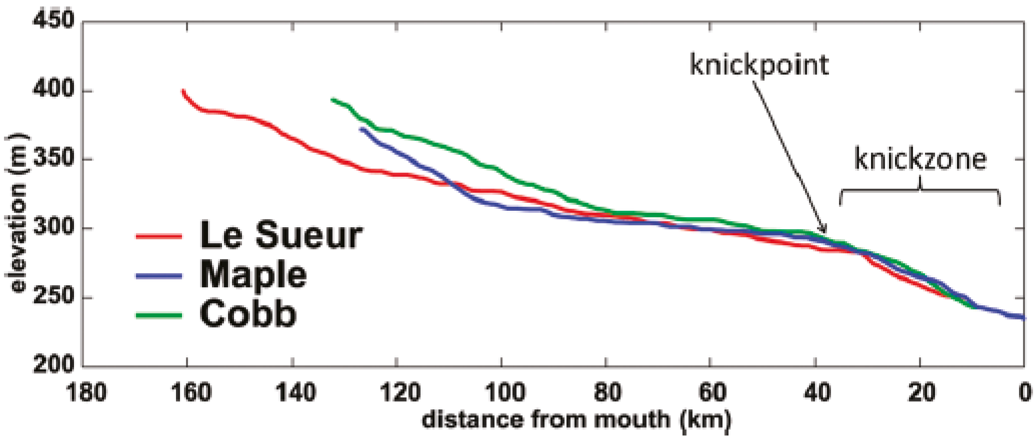


Figure 4: Profile of the Le Sueur River and its two main tributaries showing the drop in base-level at roughly 40km from the mouth (Belmont et al., 2011)

2.3 History of the land-use

Before human influence the dominant land cover in the area was prairie plus wet prairie and some hardwood forests limited to river corridors and the northeastern part of the LSRB (Gran et al., 2009). The two main changes introduced in the landscape after Euro-American settlement starting from the mid-1800s are the conversion of prairie fields to agricultural land and the introduction of artificial tile drainage. This has notable impacts on the hydrology. Wetlands and storages in the landscape were drained and previously unconnected bodies of water in the area were connected to the river network via tile drains and ditches. The conversion from wetlands to agricultural fields also impacted ET rates. The second period of major change in land is the conversion of small grains and forage crops to soybeans-corn annual row cropping in the latter half of the 20th century (figure 5). This conversion went hand-in-hand with an intensification of the tile drain and ditch network, again impacting ET rates in the LSRB. Eventually, this conversion in Minnesota and other provinces in Midwest USA led to the creation of the region now commonly referred to as the Corn Belt stretching a vast area of land in the Midwest. The land-use in this vast amount of land is predominantly agricultural and the landscape is dominated by soybeans and corn (figure 6). These two major periods of land-use change have resulted in one of the most productive agricultural areas worldwide but together also resulted in the almost complete disappearance of wetlands due to land conversion and artificial tile drainage.

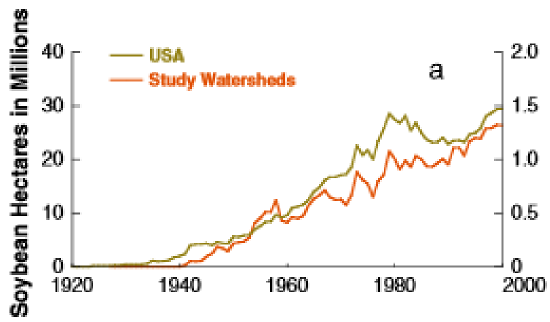


Figure 5: Increase in soybean hectares in the USA and in 21 Minnesota study watersheds from 1920-2000 (from Schottler et al. 2013).

The exact extent of wetland conversion and artificial drainage density are difficult to determine because of a lack of adequate data-sources. In the Elm and Center Creek watershed of the MRB wetlands covered more than half of the area prior to the conversion from wetland to agriculture (Jones & Winterstein, 2000). In his dissertation Burns (1954) concluded from a soil survey map of 1906 that Blue Earth County consisted of 58% wet prairie and 42% good drained land. Already some of the lands attributed to good drained lands were tile drained during that time indicating that the poorly drained part of the landscape before human involvement is likely to have even been larger. The fact that currently only ~4% of the area in the LSRB is wetland clearly underlines the impact humans have had on the landscape in only 150 years. Although public drainage enterprises tend to be recorded, but only recently, the largest amount of land is drained by private parties or still have tile drains dating back before records were set up making these datasets at the best incomplete. Burns (1954) found that in a 1920 census report for Blue Earth County 70.000 acres needed draining for reclamation and 100.000 acres were drained. In the 1930 census report 75.000 acres of the drained land were now fit for crop production hinting at the extent of artificial tile drainage. Moreover, the crop conversion to soybeans and corn is often accompanied by an even further intensification of artificial tile drainage (Sugg, 2007; Schilling & Helmers, 2008; Blann et al., 2009). After 1985 provisions of the Food Security Act were enacted discouraging drainage and conversion of wetlands to tillable agricultural lands (Mitsch & Gosselink, 1993; pp. 545-549). Although it is hard to estimate the pre-settlement wetland areas and the extent of artificial tile drainage the changes in the past 150 years regarding wetland disappearance, tile drain introduction and crop conversion are likely to have significantly changed the hydrology of the LSRB, and many other agricultural basins. This is underlined by a recent research by Wang and Hejazi (2011) estimated that in the Midwestern USA human activities contributed more to increased flow than climate. The increases in flow correlated well with the percentage of cropland in an area.

2.4 Sediment sources

The four main sediment sources in the GBERB are upland field erosion, bluff erosion, valley erosion and stream bank erosion (Hansen et al., 2010), of which bluff erosion contributes the highest portion. Upland field erosion is sediment from agricultural fields in the uplands and comes from a combination of erosion from direct precipitation and overland flow, erosion from concentrated flow in rills or gullies, and wind-blown erosion (Gran et al. 2011). Upland field erosion contributes 15-40% of the sediment in the entire MRB (Schottler et al., 2010). Ravines form the connection between the uplands and the lower incised river. Many ravines in the LSRB have threshold side slopes, meaning that incision or widening of the channel at the bottom of the ravine will lead to oversteepening of hill slopes and additional erosion as side slopes adjust. Often the timing of storm events plays an important role in ravine erosion because of potential previously stored sediment and erosion control by seasonal vegetation. Ravine erosion and sediment transport is tightly coupled to hydrology in the ravines leading to the concern that additional flow from subsurface drainage outlets that flow directly into ravines could increase the rate of erosion and sediment transport within these features. Ravines have an average contribution to the sediment budget of 8-9% (Gran et al., 2011). Bluffs and stream banks are often hard to separate out, and literature on this issue is often unclear, but most commonly bluffs are stream banks more than 3m in height. The three primary erosion processes are similar for stream banks and bluffs namely sapping, undercutting and freeze-thaw. Undercutting is the most important of these three processes. Erosion of banks and bluffs is highly variable over time, tending to be non-linear and concentrated around high stream flow events (Knighton, 1998). Bluffs contribute as much as 50-60% to the total sediment budget (Gran et al., 2011). Bluffs are mainly concentrated in the lower incised part of the watershed and a total of 480 bluffs exist in the entire LSRB (Day et al., 2013). The exact percentage of sediment load in the Minnesota River contributed by stream banks cannot be computed from the results of previous studies (Hansen et al., 2010).

Bluff erosion, valley erosion and stream bank erosion are nearby river sources of sediment underlining the importance of stream flow characteristics. Erosion of these sources is often governed by high stream flow conditions and erosion is expected to increase if stream flow of the rivers in the GBERB increases. A study by Belmont et al. (2011) in the MRB concludes that the dominant source of sediment has shifted from agricultural soil erosion to accelerated erosion of nearby river sources. The erosion of nearby river sources is driven by increased river discharge (Lenhart et al., 2011; Donner et al., 2004; Raymond et al., 2008). In a study by Cho et al. (2013) a topo-filter is used to determine the areas in the landscape that contribute most of the sediment. For different parameter values the areas that contribute 90% of the sediment 90% of the times were identified. These areas were dominantly located in the incised part of the watershed closest to the stream outlet. The contribution from the upland zone was far more limited.

All of the above shows that the LSRB has disproportionate high erosion rates and recent research has concluded that near channel sources are the main contributors. Erosion of near-channel sources is driven by increases in stream flow attributed to a combination of climate change and land-use change altering EVT and the hydrology. For management of sediment in this basin stream flow management is a prerequisite. Wetland restoration is seen as a promising management strategy for stream flow management. The following chapter will describe the strategy followed and later on the outcomes of this study will be presented to determine the effect of wetland restoration, wetland location and other factors influencing the opportunities for wetland restoration.

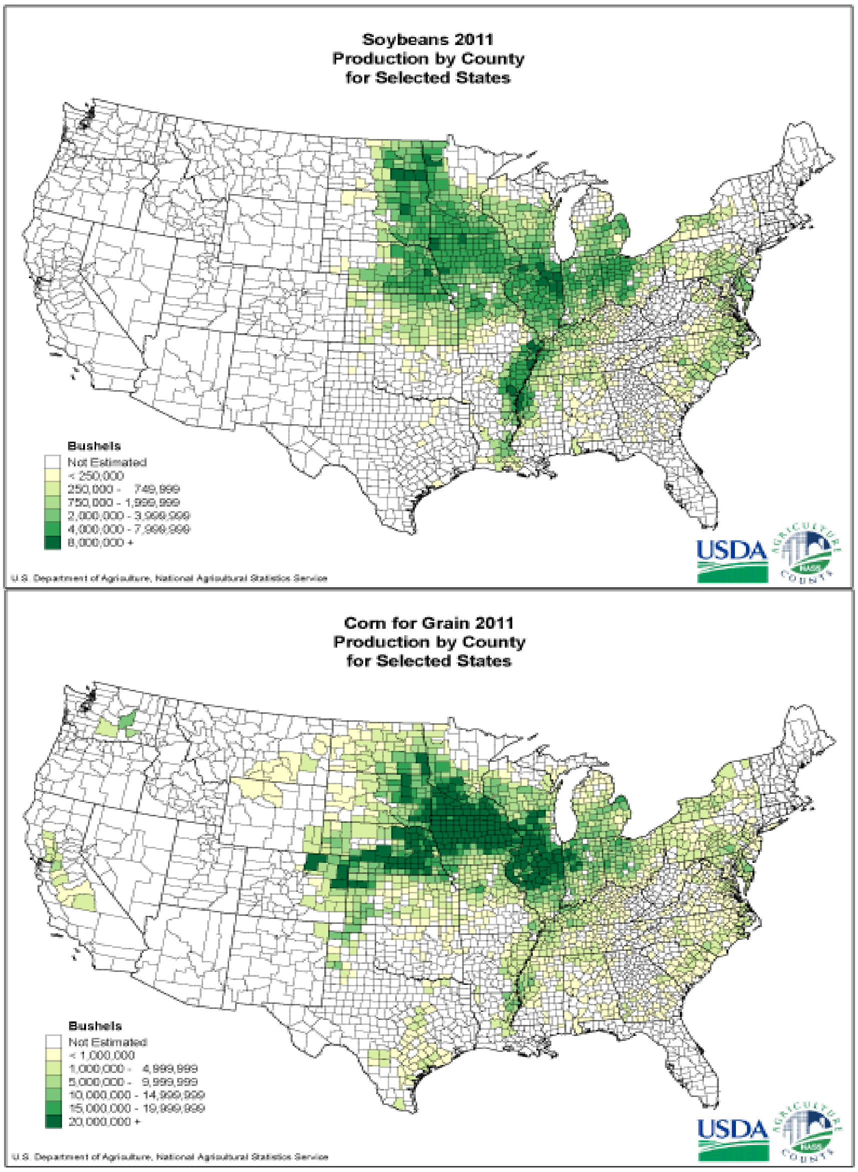


Figure 6: Production of soybeans and grains by county in the United States for the year 2011 (USDA & NASS, 2011)

3. Methodology

The largest part of this research has focused on hydrological simulation of wetland restoration. For this, the Hydrological Simulation Program FORTRAN (HSPF) was used to assess the effects of wetland restoration. Recently, a HSPF-model has been calibrated for the LSRB by RESPEC, commissioned by MPCA. MPCA has been generous in providing the calibrated model for this research. HSPF has been used for several studies on both water quantity and quality issues in Minnesota (e.g. Jones & Winterstein, 2000), the USA (e.g., Johnson et al., 2003; Saleh & Du, 2004, Miglaccio & Srivastava, 2007) and throughout the world, e.g., Canada (Al-Abed & Whiteley, 2002), China (Chen et al., 2004), Ireland (Nasr et al., 2007), South-Korea (Chung & Lee, 2009) and Turkey (Albek et al., 2004). Although HSPF has been used in several studies several caveats were discovered while using the model. This sometimes related to the program structure of HSPF and sometimes to the set-up of the calibrated model by RESPEC. This methodology section will present the strategy for the final runs. However, several analyses have preceded these runs and have provided insights for structuring the final runs. The reader will come across several references to Appendix A. In this appendix the previous runs and insights will be discussed in more detail for the interested reader. Given extent of this analysis and to improve the readability of this thesis these separate analysis will not be discussed in detail in this section.

3.1 HSPF

HSPF is a continuous, deterministic lumped-parameter simulation model and an extensive description of the model has been provided by Bicknell et al. (2001). HSPF is based on the original Stanford Watershed model IV from Crawford and Linsley (1966) for which the most important relations are schematically depicted in figure 8, and has been supported by the USEPA and the United States Geological Survey (USGS). Nowadays, HSPF is a combination of the Agricultural Runoff Management model (Donigian & Davis, 1978), Nonpoint-Source Runoff Model (Donigian & Crawford, 1976), and Hydrologic Simulation Program (Hydrocomp, 1977; Donigian & Huber, 1991; Donigian et al., 1995). It can also be used as a distributed parameter model as it reproduces spatial variability by dividing the basin in hydrological homogenous land segments and simulates runoff for each segment independently. In each land segment the primary reach is identified and for the LSRB this results in a total of 94 reaches (figure 7). HSPF distinguishes between three types of modeling segments: pervious land segment (PERLND), impervious land segment (IMPLND), for areas with insufficient infiltration, and stream/ reach/reservoir (RCHRES) to simulate the processes in a single reach of an open channel or a completely mixed lake. Flow routing in channels is computed by the continuity equation and storage routing or kinematic wave. The channel is represented by a user-defined fixed relationship between depth, surface area and volume and is eventually represented by an F-TABLE for each individual reach.

The model requires several inputs. Meteorological inputs are generally supplied in the form of a Watershed Data Management (WDM) file and can often be retrieved through BASINS, a GIS program, and processed by WDMUtil, both software packages that are freely obtainable from the EPA website. Data on land-use and all parameter values are provided through a User-Control-Input file (UCI). Wetlands are characterized as a specific land-use type in the UCI-file and per land segment the amount of acreages of wetlands is defined and linked to a unique set of parameters. For the LSRB the UCI-file distinguishes 9 different land-use categories namely; urban, forest, cropland – conventional till, cropland – conservational till, pastures, grassland, feedlots, ravines, bluffs and wetlands. Conventional tillage on cropland is a farming practice where all crop residue is removed from the field before new crops are planted. In contrast, conservation tillage is any method of soil cultivation that leaves the previous year's crop residue (such as corn stalks or wheat stubble) on fields before and after planting the next crop, to reduce soil erosion and runoff. Conservation tillage is promoted by the Minnesota Department of Agriculture (MDA, 2013). Most parameters are input on a fixed basis but HSPF provides the opportunity for monthly variation on a subset of parameters.

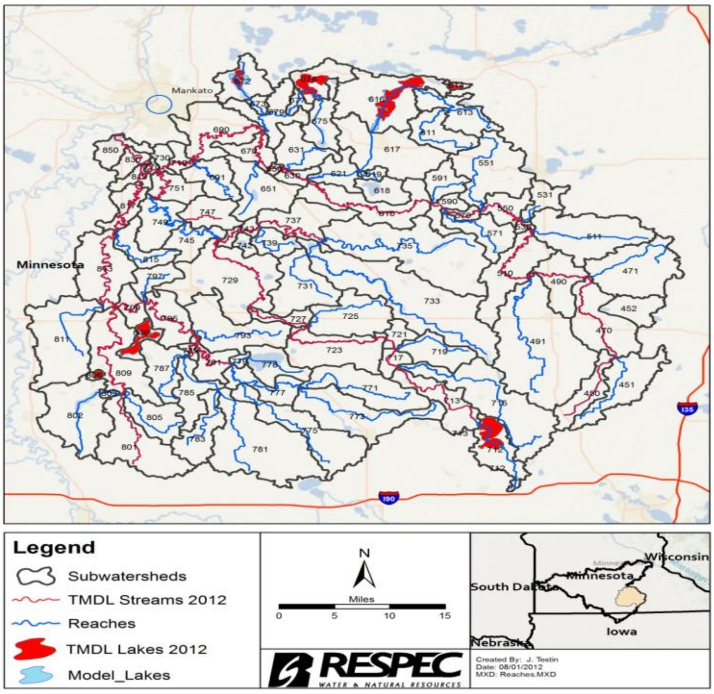


Figure 7: Identification of primary reaches and land segments in the HSPF-model for the LSRB developed by RESPEC. In red are the streams or lakes that are impaired based on a Total Maximum Daily Load (TMDL) study in 2012 (Regan(a), 2013, personal communication).

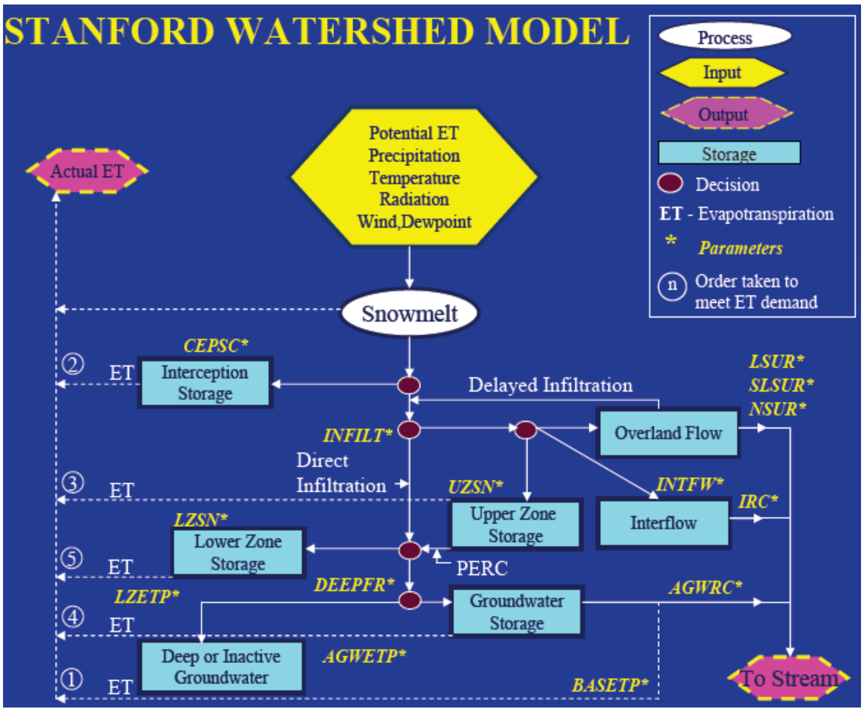


Figure 8: Schematic depiction of the most important relations in the Stanford Watershed Model used as a basis for development of HSPF. In yellow letters are the parameters used by the model to simulate the most important processes. For a description of the parameters please see appendix, table A.1.

3.2 HSPF and the LSRB

The original calibrated UCI-file for the LSRB was provided by the MPCA and adjustments are made with the free software EditPad Lite. The original model simulated both water quality and quantity processes and the model was edited to only simulate water quantity processes. This greatly reduced simulation time and only simulated processes relevant for the scope of this study. HSPF is run using a daily time-step and output is generated using HSPF Binary Output files (HBN). The calibrated model for the LSRB covers the period from 1995-2009. However, the model needs one year of start-up time in order to calibrate all values and therefore in all analysis the year 1995 is dropped. Output is generated at all 8 gage locations in the LSRB. These gages can be divided in several categories. First of all, there is one outlet gage at the mouth of the Le Sueur River. This gage will be referred to as the Red Jacket gage or outlet gage. Second, there are three knick-zone gages located on the Le Sueur River, Maple River and Cobb River. These will be referred to as the lower gages. Third, there are three gages located just before the knick-point capturing all flow from the upland zone. These gages will be referred to as the upper or upland gages. Recently, these gages have been removed by the MPCA. The Maple River and Le Sueur River upper gage roughly capture all flow from the upland zone. However, the gage on the Cobb River is located on the Little Cobb before the junction with the Big Cobb River. Thus, for the Little Cobb River the upper gage does not capture the flow from the entire upland zone. Therefore, analysis on the Cobb River will be performed on the lower gage and the upper gage will mostly be dropped from the results. The location of the upland gages just before the transition from upland to knick-zone provides the opportunity to measure the effect of wetland restoration for the entire upland zone. Since the majority of bluffs are located in the knick-zone it is of interest to lower the water levels flowing into this area. The location of the 3 upland gages provides us with the opportunity to analyze the inflow to the knick-zone and will therefore primarily be used in the analysis. The eight and last gage is located at Beauford Ditch. This gage is uniquely located since it only captures stream flow from a single headwater stream. This sub-watershed is located in the upper zone and is an ideal test side for wetland restoration in the upland zone. The initial analyses presented in appendix A.2 are performed solely on this gage.

This research is concerned with the effect of wetland restoration on stream flow in an agricultural basin. The focus on wetland restoration means that this research will not take treatment wetlands into consideration. Furthermore, wetlands can be restored on land and in-stream. Given the difficulties involved with in-stream wetland restoration in HSPF this research solely focuses on wetland restoration on land. Previous research has also solely focused on wetland restoration on land (Jones & Winterstein, 200; Babbar-Sebens et al., 2013).Wetland restoration in HSPF can be simulated by adjusting the wetland acreages per area connected to a reach. Each of these areas is linked to a distinctive parameter set. However, there are two important issues with the set-up of wetlands in the current calibrated model for the LSRB. There are two approaches to simulate wetlands in HSPF (NRC, 2010; Kong, 2009; Bicknell et al., 2001). The first approach is to have distinctive parameter sets linked to different land-uses. The second approach involves a wetland module, introduction of new parameters and a different set-up of the UCI-file. The wetland module tracks dynamic variation in groundwater level; models interaction between groundwater storage, soil storages, and infiltration/runoff processes; accommodates ponded conditions on the land surface; allows evaporation from ponded surface storage and surface runoff; and allows additional options for surface runoff when it is not gravity driven (Bicknell et al., 2001). In the current set-up of the LSRB calibrated model the first approach is used. Given the time available and extent of this research it was not reasonable to adjust the model structure for wetland restoration. However, for future usage of this model it is strongly recommended to look at these possible adjustments. Nonetheless, previous research on wetland restoration has made use of the first approach before (Jones & Winterstein, 2000) and, although the latter approach might be more sophisticated, the first approach should be able to capture wetland functioning. The second issue is that in the traditional set-up of HSPF water is routed from each land-use category directly to the reach. This means that wetlands cannot intersect surface runoff or tile drains from other land-use categories in the sub-watershed before it flows into the reach. Thus, wetlands only treat precipitation. There are possibilities to connect different land-uses in HSPF to route water from croplands to wetlands. Currently, this is not the set-up of the LSRB calibrated model. However, again given the time available in this study it was not deemed reasonable to adjust the UCI-file for this. Even more so, this study focuses on general extent of wetland restoration and lumped responses of the stream flow. Adjusting flow routing to allow croplands to drain into wetlands is only reasonable for studies focusing on individual wetland sites, which is not the case in this study. Otherwise for each individual wetland site the routing has to be adjusted which would lead to over 10.000 adjustments for the entire LSRB. The implication of the set-up of the current UCI-file is that the effect of wetland restoration is likely to be underestimated since wetlands are only capable of treating direct precipitation falling on their land surface and the current model does not capture the effect of water storage by wetlands from runoff of cropland. However, given the fact that this study aims to compare wetland sites at different locations possible differences should already be observable for wetlands only treating precipitation.

3.3 Wetland scenarios

Based on a sensitivity analysis and a comparison of the parameters assigned to croplands and wetlands (Appendix, paragraph A.1 and A.2) it was concluded that the initial wetland parameterization was largely similar to the cropland parameterization and resulted in almost no differences in stream flow following wetland restoration. Currently hardly any wetlands exist in the LSRB and therefore small calibration errors in the wetland parameterization are unlikely to influence the stream flow output. Therefore, model calibration has mainly focused on correct parameterization of croplands. For wetland parameterization the current parameter set was compared to other research on wetland restoration in the area (Jones & Winterstein, 2000) and to a wetland parameter set of a previous version of the LSRB model. The parameters assigned to wetlands were remarkably similar between J&W and Butcher models but were clearly different from the current calibrated LSRB model. A simplified version of the HSPF model was built in Excel and the parameter sets of the current model and of Butcher are analyzed and compared (Appendix, paragraph A.3). From this analysis it was concluded that the Butcher parameter set was better suited to simulate wetland restoration and the UCI-file was adjusted accordingly. Currently, the actual calibrated model for the LSRB has been updated by RESPEC with the new wetland parameter set from Butcher.

The next step to set-up the simulation runs is to determine the extent of possible wetland restoration per sub-watershed. Ducks Unlimited (DU), an organization focused on wetland restoration and conservation throughout the US, build a dataset to calculate potential wetland area for each individual sub-watershed in several counties in the MRB, including the LSRB (Ducks unlimited, 2013). Their methods involve manually outlining depressional wetland sites using stereoscopic photo interpretation on high-altitude color infrared aerial photographs. A previous study by Ozesmi and Bauer (2002) showed that wetlands have spectral signatures from which extents can be calculated using satellite imagery. A downside of this approach is the possible overlap with upland areas. To overcome this, aerial photographs can be used to verify the results. The DU dataset has been verified at multiple locations using aerial images. Another prerequisite for wetland restoration is the presence of hydric soils (Minnesota Department of Agriculture, 2012; Lenhart et al., 2008; Lenhart et al., 2010). For the DU database it has been verified whether the potential sites indentified corresponded to sites with hydric soils. If not, the sites were dropped from the potential wetland restoration list. Potential sites for wetland identification can also be based on topographic features. Babbar-Sebens et al. (2013) use a combination of soil data (SSURGO dataset) and the topographic wetness index (CTI) to identify potential wetland sites. However, the advantage of the DU method for this study compared to methods based on topographic features is twofold; first, this method is explicitly focused on identifying sites for wetland restoration since only areas where previous wetlands existed are identified as candidate areas and second, this method can identify both wetland areas with concentrated flow as well as areas were precipitation regenerates the wetland. A methodology based on topography can only identify potential areas based on concentrated flow patterns and therefore this research prefers the approach used to construct the DU database.

The DU database identifies individual wetland sites per sub-watershed. In HSPF, however, it is not possible to make a distinction between locations of wetland sites within a sub-watershed. Therefore, all individual wetland sites per area were summed to create a total potential wetland area in a sub-watershed. This research is not interested in determining the optimal individual location of wetlands since a lot of different considerations have to be taken into account besides hydrologic effectiveness. For actual wetland restoration on the spot hydrological functioning will likely be overruled by criteria concerning productivity of the land, willingness of stakeholders to participate and other considerations that influence individual wetland site restoration. This study merely focuses on the effect of wetland restoration and differences between wetland restoration at different locations. For the aim of this research it is therefore not an issue that individual wetland location cannot be simulated.

Selection of areas where wetland restoration will be simulated is based on a set of criteria. Bluffs are mainly located in the incised zone of the LSRB and erosion is therefore most prone in this region. Management focused on reducing flows in the LSRB should therefore concentrate on reducing stream flow entering the incised zone. This study therefore exclusively focuses on wetland restoration in areas upstream of the incised zone. The upland gages on the Maple River and Le Sueur River are located at the start of the incised zone and therefore all areas downstream of these gages are dropped from the analysis. For the Cobb River only areas upstream of the junction between the Little and Big Cobb are included. Next to this areas that are treated as a lake in HSPF or areas upstream of a lake are dropped from the analysis. In-stream lakes have a delaying effect on stream flow and in a sense function like wetland areas. Therefore, wetland restoration in these specific areas is not likely to result in an additional delay in stream flow. Lastly an analysis is performed calculating the total cropland area, both conventional and conservational cropland, per sub-watershed compared to the total area in order to ascertain that all areas incorporated are predominantly agricultural. No areas were dropped based on this analysis since all areas had at least 50% cropland.

After selection of the areas for wetland restoration distance to the outlet gage per individual outlet point of a reach is calculated. Stream length defined in the UCI-file is used for this calculation. Areas for all three rivers are ranked from furthest to closest to the Red Jacket gage. The final selection of areas and ranking is presented in table 2. Loucks (1989) argued that wetland restoration far upstream is most beneficial for stream flow management. Anderson and Kean (1994) hypothesized that wetland restoration in late contributing areas has the largest effect on stream flow. Although both hypotheses are not entirely similar the distinction between late and early contributing areas resulted in early contributing areas being downstream and late contributing areas being upstream. The critical assumption in this research is that by comparing areas further and closer upstream it is possible to simultaneously test both hypotheses. Areas closer to the gage are then typified as early contributing areas. Given the homogeneity in land-use, slope and soil type in the upland zone of the LSRB this is a valid assumption.

Wetland restoration is implemented by adjusting the amount of acreages attributed per sub-watershed to a land-use category. It is assumed that wetland restoration goes primarily at the cost of conventional cropland. Thus, for every acre of restored wetland one acre of cropland disappears. If the amount of wetland acreages restored exceeded the amount of conventional cropland the additional acreages were subtracted from conservational cropland. Wetlands per sub-watershed are always restored to the full potential. All adjustments were manually implemented and for easier implementation the amount of acreages are rounded towards the nearest hundred. To compare wetland location at different locations a distinction is made between far and close upstream wetland restoration. Far upstream wetland restoration starts with wetland restoration at the furthest away sub-watershed, and from there on moving downstream. This resembles wetland restoration in upstream and late contributing areas. Close upstream wetland restoration start at the sub-watershed closest to the gage and moves further upstream resembling wetland restoration downstream and in early contributing areas. A check was performed indicating that the rounding towards the nearest hundred did neither effect the average potential wetland restoration for the individual rivers nor the average potential wetland restoration far and close upstream per river and thus did not create a bias in the analysis.

The Le Sueur River has the largest potential for wetland restoration. In this sub-basin a maximum of 15.000 acreages of wetlands could be restored within two non-overlapping zones. However, this means that for the Cobb River and the Maple River parts of the far and close upstream zone overlap for the largest wetland restoration scenario. Next to this, there are two others issues concerning these sub-basins. For the Cobb River the location of the upland gages prohibits the opportunity to solely measure the inflow into the knick-zone. In the Maple River several tributaries make the distinction between far and close upstream zones difficult and relatively few acres for potential wetland restoration are available. This results in large overlaps for the 15.000 acreages of wetland restoration and makes distinction of a far and close upstream scenario intricate. Because of the above reasons the upper Le Sueur gage will be leading for evaluation of the wetland scenarios.

Besides a wetland simulation with 15.000 acres of wetland restored two additional runs were performed for 10.000 and 5.000 acres of wetland restoration. These runs are performed to make a comparison between location of wetland restoration in the Maple and Cobb River. Also smaller amounts of wetland restoration result in the far and close upstream zone being more clearly distinguishable. Furthermore, one would expect to see at least similar and possibly stronger diverging stream flow responses after wetland restoration in these zones. It therefore provides an additional check on the findings for the 15.000 acres wetland restoration scenario in the Le Sueur River.

Previous scenario reports on wetland restoration often restore wetlands up to a certain percentage in each individual sub-watershed. This might be sufficient if one only looks at stream flow response but would create a bias between analyzing the effects of wetland restoration between zones. In a relative sense the scenarios would be similar, f.i. both having 10% wetland restoration, but in the largest region more acreages of wetlands would be restored. As can be observed in figure 9 there seems to be a correlation between distance to the outlet and sub-watershed size. Although the correlation is not very strong the graph clearly shows that in both sub-basins after a certain distance basin area is always larger than 5.000 acreages. Results for the Cobb River are not presented because of the placement of the upper gage. A similar trend can be observed if all areas in the entire LSRB upstream of the upper gages are combined. Therefore, to create a bias in comparing different locations of wetland restoration this research uses fixed amounts of wetland restoration, similar to previous research by Jones and Winterstein (2000).

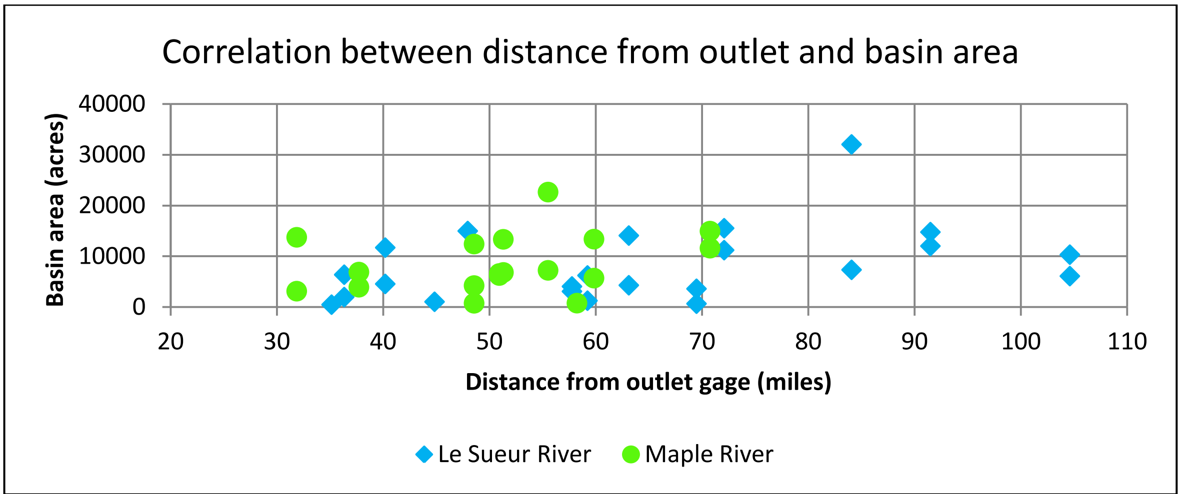
A total of 11 simulation runs are performed with a daily time-step (table 3). The first is the base scenario with current land-use pattern based on the 2006 census data. Then three scenarios for both the far upstream and close upstream wetland restoration exist. These include; large wetland restoration (15.000 acreages per sub-basin), medium wetland restoration (10.000 acreages per sub-basin) and small wetland restoration (5.000 acreages per sub-basin). HSPF calculates the effect per acre on runoff from a land segment. Thus, increasing the amount of acreages should result in a linear response. HSPF is therefore not suited to compare between scenarios with amount of acreages but nonetheless the outputs will be presented for all scenarios for comparison. Thus, this study will not make a comparison between amounts of acreages of wetland restored but will use the trends in all these analyses to highlight commonalities. In addition a sub-set of runs was performed for the small wetland restoration scenario. For the Le Sueur River 5.000 acreages were implemented in areas in-between far and close upstream. This run was performed to test if wetland restoration in an in-between zone would influence stream flow differently and relates to the concept of intermediate contributing areas defined by Anderson and Kean (1994). These simulations will be referred to as medium close and medium far upstream. The 11th scenario run is performed for the Little Cobb River. The areas upstream of the upper Cobb gage are divided in close and far upstream areas. The total potential wetland restoration area in the close upstream areas totals 2.700 acres. For the far upstream area 900 acres per reach were implemented to allow comparison. These simulation runs are explicitly performed to test the inflow into the incised zone at the upper Cobb gage.

Besides the simulation runs with the daily time-step additional runs with an hourly time-step are performed for the 15.000 far and close upstream wetland restoration scenarios. These runs are performed for the years 2002-2007. Running time increases exponentially if the model is run on an hourly time-step and therefore not all years are incorporated in this simulation. The years 2002-2007 are selected because this subset encompasses one of the wettest years, one of the driest and the largest precipitation event on the record.

Table 2: Wetland restoration table per reach based on database from DU (2013)

reach_id	main drainage basin	distance from outlet	Basin area	cropland	potential wetland sites	total area
		miles	*acres*	*%*	*#*	*acres*
713	Cobb River	75	7990,86	0,92	763	1540,23
715	Cobb River	75	20013,32	0,89	1613	4312,46
717	Cobb River	71	2242,84	0,91	267	158,34
719	Cobb River	71	6365,62	0,90	646	1382,16
721	Cobb River	69	2328,69	0,91	263	677,40
725	Cobb River	54	10663,17	0,91	773	2383,00
723	Cobb River	54	12030,66	0,80	915	1586,66
731	Cobb River	50	10433,42	0,84	973	2573,79
735	Cobb River	50	27090,14	0,87	2708	5203,03
733	Cobb River	50	30429,63	0,89	2702	6184,43
727	Cobb River	47	2563,33	0,82	263	364,65
739	Cobb River	39	3616,35	0,87	234	797,64
737	Cobb River	39	8826,63	0,84	947	1579,05
743	Cobb River	32	1921,49	0,70	119	285,24
729	Cobb River	32	20977,84	0,82	1743	3367,44
450	Le Sueur River	105	6084,5	0,91	553	1450,43
451	Le Sueur River	105	10309,33	0,88	1025	2180,67
470	Le Sueur River	92	12004,67	0,83	1244	1439,65
471	Le Sueur River	92	14750,35	0,87	1634	2828,46
490	Le Sueur River	84	7321,9	0,89	749	912,85
491	Le Sueur River	84	32023,06	0,91	2911	4934,07
510	Le Sueur River	72	11191,8	0,83	1015	2302,47
511	Le Sueur River	72	15511,36	0,77	1432	3238,53
530	Le Sueur River	69	651,4	0,78	51	109,66
531	Le Sueur River	69	3591,09	0,78	260	1030,57
550	Le Sueur River	63	4284,2	0,75	366	764,72
551	Le Sueur River	63	14069,36	0,83	1368	3378,24
570	Le Sueur River	59	1242,75	0,62	68	184,53
571	Le Sueur River	59	6207,92	0,92	646	1413,20
590	Le Sueur River	58	3068,6	0,91	336	611,80
591	Le Sueur River	58	4045,57	0,87	397	950,22
617	Le Sueur River	48	14965,17	0,82	979	2873,13
619	Le Sueur River	45	1025,91	0,91	74	171,82
621	Le Sueur River	40	4544,88	0,87	352	922,72
610	Le Sueur River	40	11692,85	0,80	1152	2010,09
630	Le Sueur River	36	1953,07	0,82	176	269,55
631	Le Sueur River	36	6359,17	0,88	456	1637,43
650	Le Sueur River	35	461,25	0,59	36	52,72

771	Maple River	71	11587,43	0,93	1110	1137,53
773	Maple River	71	14940,21	0,84	1100	1269,97
777	Maple River	60	5690,65	0,85	448	1234,90
775	Maple River	60	13367,71	0,93	1026	1343,64
779	Maple River	58	759,93	0,94	99	167,28
783	Maple River	56	7220,72	0,91	488	1691,79
781	Maple River	56	22636,01	0,91	1410	3302,81
805	Maple River	51	6808,84	0,88	753	966,33
801	Maple River	51	13328,12	0,91	1272	1737,15
785	Maple River	51	6203,04	0,94	470	1119,80
787	Maple River	51	6598,24	0,91	541	1039,98
789	Maple River	49	735,69	0,85	54	221,48
791	Maple River	49	4220,16	0,76	370	702,24
793	Maple River	49	12402,28	0,92	1192	2181,52
797	Maple River	38	3899,02	0,93	390	665,30
795	Maple River	38	6876,66	0,84	564	1121,44
799	Maple River	32	3117,09	0,78	240	411,19
809	Maple River	32	13728,21	0,84	1107	2401,53



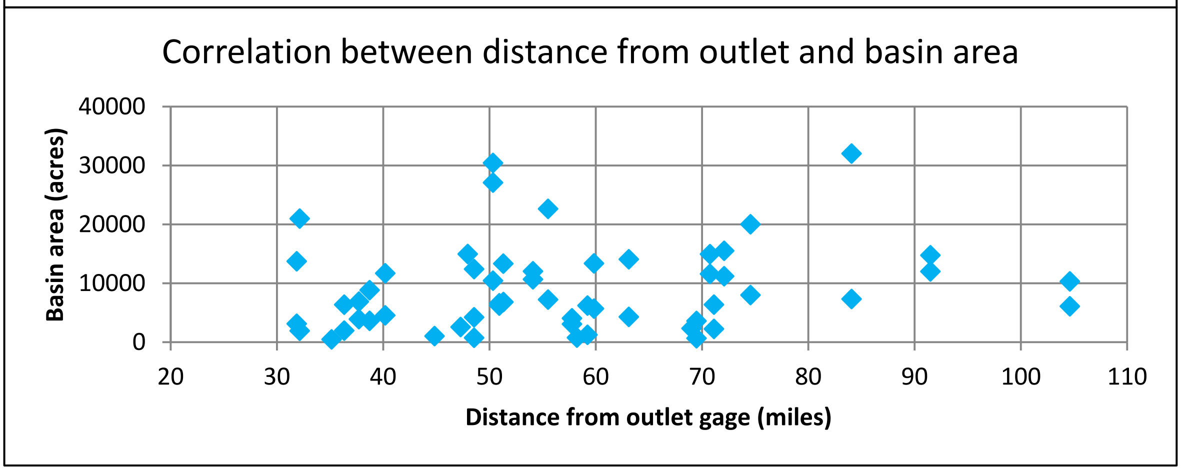


Figure 9: Relationship between distance from outlet and basin area for Maple River and Le Sueur River. Although the correlation between distance and basin area is not very strong both rivers clearly show that after certain distance basins are always larger than 5.000 acreages.

3.4 Stream flow response of wetland restoration

Stream flow output from HSPF is generated for all gages but most analyses will focus on the upper Le Sueur gage. Output will be produced for all months from 1995-2009. However, output will be analyzed for the months March-October from 1996-2009. Other months are not taken into considerations since during the winter most streams are frozen and gages are not in place. Since the HSPF model needs a year of start-up time results for 1995 are not taken into account. Stream flow output analyzed is in acre-feet per day, referred to as ROVOL in HSPF, since no aggregation issues occur for this type of output. The output is sometimes later on converted to cubic feet per second (cfs). According to a research by Carter et al. (1978) wetlands can reduce down gradient floods and peaks in two ways. The first effect is that wetlands reduce the volume of water released total annual runoff, because water evaporates from wetland surfaces. To test the 'evaporation effect' of wetlands on total runoff it is sufficient to look at stream flow output on monthly or yearly basis. The simulation runs on a daily time-step will be used for this part of the analysis generating average daily streamflow output. First, our analysis will focus on the effect of wetland restoration on peak stream flow without looking at differences between locations. The following output will be produced; 1) entire stream flow record, 2) Flow Duration Curves (FDC) for individual years, 3) tables depicting changes per month for each individual year, 4) peak stream flow graphs and 5) Flow exceedence curves. Yearly FDC, as suggested by Vogel and Fennessey (1994), will be used to analyze the effect of wetland restoration on both low and high flows. Aggregation of FDC over multiple years bears the risk of averaging out increases and decreases hiding differences between individual scenarios. Peak stream flow for the 50 days with the highest stream flow record under current land-use will be compared before and after wetland restoration to assess the effect of wetland restoration on peak stream flow. However, from a geomorphologic perspective 'channel forming flows' are of main interest. These flows are generally not the highest peak flows but can be better typified as common floods (Belmont, 2013). An analysis by Schottler et al. (2010) showed that both low and high flows have increased in the LSRB. But the analysis concluded that the highest peak floods have decreased whereas the common floods have increased. To study the common floods graphs will be generated showing the volume of discharge that exceeds a continuum of discharge volumes before and after wetland restoration. This gives a broader perspective on the stream flow response and is likely to be more relevant for erosion studies. Further research focused on the relationship between erosion and stream flow in the GBERB is likely to set a threshold on flood volume above which erosion of near-channel sources will consequentially lead to elevated levels of turbidity. These graphs can then easily be used to determine the effect of wetland restoration on the volume of discharge exceeding this threshold. All graphs and tables are generated for each individual scenario but not all results will be presented in this thesis. For determining the effect of wetland restoration on peak stream flow only results for the two 15.000 acres wetland restoration scenarios at the upper Le Sueur gage will be presented. It should be noted that analysis on the other rivers showed more or less similar trends. Nonetheless to assess whether all wetland restoration under all scenarios results in positive results at all gages all output will be evaluated based on the following criteria:

Eq. (1) $$Stream\ discharge^{N}_{(1996-2009)} > Stream\ discharge^{W}_{(1996-2009)}$$

Eq. (2) $$Stream\ discharge^{N}_{(y)} > Stream\ discharge^{W}_{(y)}$$

Eq. (3) $$Stream\ discharge^{N}_{m-j} > Stream\ discharge^{W}_{m-j}$$

In these equations N stands for current land use, W stand for land use after wetland implementation, (1996-2009) stands for entire study period, y stands for individual years and m-j stands for the months May and June. The first criterion states that for the entire study period stream flow after wetland implementation has to be lower than the stream flow under the base scenario. Previous analyses at Beauford Ditch demonstrated that wetland restoration leads to reductions in stream flow in most years, except for 2007 and 2009. These are the direst years on record with most precipitation falling at the end of the year instead of during the May-July period (results not presented here). The second criterion states that for each individual year, excluding the years 2007 and 2009, the stream discharge after wetland implementation has to be lower than under current land-use. Therefore, although likely, it is not necessarily the case that the first criterion is automatically met if the second criterion is passed since increases in 2007 and 2009 could possibly off-set decreases in the rest of the study period. The third criterion states that the sum of the May and June discharge has to be lower after wetland implementation. The third criterion is chosen since the highest turbidity values in the stream per year are often observed during the months May and June (Regan(a), personal communication). It is expected that either all or most of the wetland restoration scenarios pass these criteria. If not, the scenario is dropped from further analysis and it has to be concluded that wetland restoration will not always result in reduction of stream flow throughout the LSRB.

3.5 The effect of wetland location on stream flow

Besides looking at the overall effect of wetland restoration this study is also interested in comparing wetland restoration between locations. Differences in stream flow response based on wetland location can be based on localized conditions. This might be related to small differences in extent of wetland and cropland, differences in the radiation budget and precipitation resulting in differences in EVT and antecedent moisture conditions prior to a precipitation event. This has been previously identified by Jones and Winterstein (200) for the Red River Basin. Combined these differences might make one area more suited for wetland restoration compared to the other. These differences should be observable for both individual events but also on a more aggregated basis and can be attributed to the 'evaporative effect' identified by Carter (1978). Therefore, the same output generated for the analysis on the effect of wetland restoration can be used to compare differences in location of wetland restoration based on local conditions. Next to that, two metrics will be used to compare far and close upstream wetland restoration that might highlight possible trade-offs. These metrics will not only be compared for the Le Sueur River but for all three rivers in the basin for all scenarios. The first metric calculates the stream flow reduction per acre in the months May-June. This highlights the per acre efficiency of wetland restoration scenarios in the months with currently the most elevated turbidity levels. The higher the reduction per acre in these months per year the more suitable the location. In an initial analysis at Beauford Ditch wetland restoration is likely to result in decreases in stream flow in some months and increases in other months (results not presented here). The second metric therefore focuses on the increases and decreases per wetland scenario using the following equation:

Eq. (4) $$Ratio = \frac{\sum volume\ of\ increased\ discharge_d}{\sum volume\ of\ decreased\ discharge_d}$$

In this equation d stands for day. This equation relates the sum of the volume on days with increased stream flow over the entire study period to the sum of the volume of days with decreased discharge. The lower the value of the ratio the smaller the increases are compared to the decreases. The combination of the two metrics presented above and the analysis of the other output generated should highlight possible differences between far and close upstream wetland restoration based on localized conditions.

Possible differences between wetland restoration are not only related to localized differences between two areas but can also be related to the fact whether a wetland is restored far or close upstream (Klouck, 1989; Anderson & Kean, 1994). In essence, the distance to the gage determines the effectiveness of wetland restoration. However, these differences are not related to difference in the evaporative effect of wetlands. The amount of EVT over a fixed amount of time is not dependent on distance to the gage. EVT in areas further upstream might be larger in total since water is EVT over a longer time period compared to close upstream. This is true with and without wetland restoration. However, storage time in a wetland is not depending upon distance to the gage. Thus, wetlands that store water for an additional hour in both far and close upstream regions have similar EVT, if all other conditions are similar, and will thus not result in difference in stream flow response between these two locations.

However Carter et al. (1978) identified a second effect of wetland restoration, namely that wetlands can reduce peak stream flow by delaying the release of water to ditches and streams after snowmelt or rainfall (Carter et al., 1978). This 'time-delay effect' of wetland restoration can affect peak stream flow volume differently based on location. If water is later released in close upstream areas due to wetland restoration the peak in stream flow from close and far upstream areas might be combined resulting in possible higher peaks in stream flow or at least in less reduction of peak stream flow for close upstream areas compared to far upstream areas. For a comparison of differences in peak stream flow volume the hourly scenario runs will be used. For all 5 years the event generating the highest peak in stream flow will be selected. For these events the hydrographs for current land-use and far and close wetland restoration will be compared.

Table 3: Different scenario runs for wetland restoration. Note that the current land-use simulation used for comparison and the runs with an hourly time-step are not included in this table.

	Number of acres restored	Location upstream	River	Comparison at gage
1	15.000	Far	Le Sueur River	Upper Le Sueur gage
2	15.000	Close	Le Sueur River	Upper Le Sueur gage
3	10.000	Far	Le Sueur, Cobb and Maple River	Upper Le Sueur Gage Upper Maple Gage Lower Cobb Gage
4	10.000	Close	Le Sueur, Cobb and Maple River	Upper Le Sueur Gage Upper Maple Gage Lower Cobb Gage
5	5.000	Far	Le Sueur, Cobb and Maple River	Upper Le Sueur Gage Upper Maple Gage Lower Cobb Gage
6	5.000	Close	Le Sueur, Cobb and Maple River	Upper Le Sueur Gage Upper Maple Gage Lower Cobb Gage
7	5.000	Medium Far	Le Sueur River	Upper Le Sueur Gage
8	5.000	Medium Close	Le Sueur River	Upper Le Sueur Gage
9	2.700	Far	Little Cobb River	Upper Cobb gage
10	2.700	Far	Little Cobb River	Upper Cobb gage

3.6 Effects of wetland restoration at the plot scale

The above analyses can already answer the research question on how wetland restoration effects stream flow. However, HSPF output also provides the opportunity to analyze responses to wetland restoration at a sub-watershed scale. At this smaller scale changes in EVT, contribution of different outflow compartments, volume in different storage compartments and total outflow from the land segment will be compared before and after wetland restoration. This will provide a more detailed overview of the drivers of changes at the sub-watershed after wetland restoration. Comparison will be presented for the sub-watershed with reach id 491, a tributary of the Le Sueur River. This reach is selected because it is a headwater stream meaning that the inflow to the stream equals the outflow of the land segment. Moreover, this sub-watershed has the lowest percentage of wetlands of all headwater streams on the Le Sueur River. Therefore, the base scenario is almost similar to a scenario with only cropland. In this basin the full extent of potential wetlands will be restored, equaling 4.900 acres. The analysis will only include wetland and conventional and conservational cropland and will for simplicity not incorporate other land-use categories. The cropland and wetland combined cover over 95% of the basin area and therefore other land-uses will likely have a diminishable effect.

3.7 A broader framework for wetland restoration

As already stated in the introduction of the methodology, the largest part of this research is concerned with hydrological modeling. A smaller part of the research is focused on other aspects of wetland restoration in the LSRB. At a stakeholder meeting for the GBERB agreement was reached on the fact that different criteria should be incorporated to asses a single management option or to compare a set of management options for management of stream flow and turbidity. This research will provide a short overview of different characteristics of wetland restoration in the LSRB affecting the possibility for successful implementation of this management option. For this the outcomes of a small literature analysis, the input during an stakeholder meeting and a study on costs and stakeholder views regarding different management options for the GBERB by Heitkamp (unpublished data) will be combined to provide an overview of costs, stakeholder views and ecosystem services. Comments from stakeholders will be treated on an anonymous basis and will be referred to as CSSIR, 2013. If of importance the profession of the particular stakeholder will sometimes be mentioned. In this part of the research first costs, stakeholder views and ecosystem services will be discussed in a general manner and then specific aspects of wetland restoration in an agricultural basin will be highlighted.

Next to that this research will look at possible linkages and trade-offs between the interest of stakeholders, the costs and the provision of ecosystem services. In HSPF it is not possible to make a distinction between small and large wetlands. However, previous research has shown that optimally located small wetlands can outperform larger wetlands and are likely to be cheaper to implement (Babbar-Sebens, 2013). Given the importance and abundance of agriculture in the LSRB management options implemented should be 'farm-friendly' solutions (CSSIR, 2013). Therefore, this research will identify the attitude of farmers towards wetland restoration and possible requirements set on wetland restoration. An example of this is the question whether farmers prefer smaller of larger wetlands and why. These requirements are likely to affect costs of wetland restoration and the provision of ecosystem services. Optimizing for one ecosystem service often goes at the cost of others (Mitsch & Gosselink, 2000) and costs and stakeholder views might be in direct conflict with ecosystem services. However, it should also be possible to identify possible linkages for optimization of ecosystem services given budget considerations and stakeholder requirements. This research will identify the linkages and trade-offs between these different interests.

Previous research has identified a total of 23 ecosystem services that can be ordered into three functional groups (De Groot et al., 2002; De Groot et al., 2010). This research will not discuss impacts of wetland restoration on all 23 ecosystem services but will look at biodiversity conservation, nutrient recycling, recreation and carbon sequestration or climate regulation. This selection has been based on available research, the relevance of these ecosystem services in agricultural basins and the fact that the four services encompass all functional groups. A lot of the research on ecosystem services has been performed in agricultural landscapes. This is especially true for biodiversity conservation since this ecosystem service has been traditionally viewed as being in direct conflict with agricultural production (Macfadyen et al., 2012; Phelan et al., 2011). This has led to the recent and extensive debate on 'land-sparing' and 'land-sharing' in agricultural landscapes. This research takes a rather neutral standpoint in this debate and starts off from the opinion that wetland restoration is neither a typical land-sparing or land-sharing management option Although certain less productive areas in the landscape will be reserved for wetland restoration (typical for land-sparing) the functioning of the wetland and the ecosystem services provided cannot be decoupled from the surrounding land (land-sharing). For example, wetland restoration will influence agricultural practices in the proximity of the wetland but agricultural practices, and especially the use of fertilizers, will likely also influence the ecosystem services provided by the wetland.

4. Wetland restoration – hydrologic effects

4.1 Stream flow response to wetland restoration

All scenarios pass the criteria of equations 1-3. Thus, wetland restoration always leads to a reduction in stream flow for each individual year. Even more so, wetland restoration always leads to a reduction in stream flow in the months with the highest turbidity levels in each individual year (May-June). This section will first discuss the general response of stream flow to wetland restoration. All figures and tables discussed in this section will have two scenarios of wetland restoration. However, a distinction will only be made later on in the analysis and in this section the scenarios will be treated together and referred to as simply wetland restoration. Figure 12 depicts the stream flow record for the entire study period before and after wetland restoration. Several observations can be made from these graphs. To start with, stream flow before and after wetland restoration is not that different. For this figure, differences are most easily observed in the higher peaks of the stream flow. Throughout the year most peaks in the stream flow are reduced by wetland restoration, but some peaks are more reduced than others. In the last months of the year some smaller events are however increased after wetland restoration.

Analyzing the FDC highlights similar trends (figure 10). The largest 25% of the flows in 2009 are reduced by wetland restoration. For the rest of the stream flow record it is either difficult to separate the individual FDCs or wetland restoration occasionally results in increased stream flow. The results presented here are for 2009, the driest year on the record. Differences in FDC in other years were often smaller. Note that on the y-axis stream flow is not in cfs but in acre-feet per day maximizing the differences between the scenarios. Thus, similar to the observation made on the stream flow record, differences between yearly stream flow records before and after wetland restoration are small. It should be noted that increases in stream flow and decreases in stream flow can be balanced out in a FDC.

Interestingly, the flow volume exceeding a certain discharge clearly highlights the effects of wetland restoration. Over the entire study period both for the lower and higher flows depicted in these graphs wetland restoration results in a smaller flow volume exceeding a defined discharge level. This is even more pronounced for the highest discharges (figure 11). This result indicates that independent of the discharge level assigned to common floods or channel forming flows wetland restoration will result in a lowering of the volume exceeding this threshold. Consequently, this will likely result in a reduction of erosion from near-channel sources and lower turbidity levels.

Table 4 shows the sum of the changes in cfs per month between the two wetland restoration scenarios and the current land-use for the entire study period. Green indicates that stream flow is reduced after wetland implementation whereas red indicates stream flow has increased after wetland restoration. The bottom row is the sum of the stream flow change for each individual year whereas the outer right column depicts the average stream flow change per month over the entire study period. For all years wetland restoration results in a decrease in stream flow, even for the driest years on the record (2007 and 2009). Moreover, wetland restoration in most months results in a decrease in stream flow. Stream flow is on average most reduced during the months May-June, currently the months with the highest turbidity levels. Increases during the end of the year might be related to the fact that wetland storage capacity is low because earlier on in the year wetlands have stored most of the precipitation and because evaporation from cropland increases at the end of the year sometimes even exceeding the evaporation from wetlands. This table introduces an interesting trade-off related to wetland restoration, namely that decreases in stream flow at the beginning of the year are likely to result in increases in later periods of the year. However, decreases are always much larger compared to the increases in stream flow and reduction in stream flow are highest during the months with the highest turbidity levels. Even more so, given the fact that stream flow levels at the end of summer and the beginning of autumn are relatively low increases in stream flow in these months could be actually evaluated as being beneficial (C Regan, 2013(c), personal communication).

Lastly the changes in peak stream flow volume after wetland restoration for the 50 highest stream flow events are depicted (figure 13). For 49 of the 50 days wetland restoration results in either a decrease or a similar peak stream flow volume. Only for the 11th highest stream flow event wetland restoration result in an increase in stream flow. . This event is on the 17th of June, 2004 following the largest precipitation event on record and with several days of high stream flow prior to this event. A reduction in peak stream flow is achieved on all days preceding the 17th of June and the increase in stream flow observed is small. If instead of looking at individual days multiple days with peak stream flow would be treated as one event this would result in a decrease in the peak stream flow volume from this event. Moreover, given the fact that wetland restoration is likely to delay the outflow to the stream it should not be surprising that high stream flow volumes days after the largest stream flow event on record would be increased by wetland restoration. Water that was not released on previous days is released on this day through higher base flow levels and combines with the stream flow generated by precipitation during this day. Moreover, it is possible that the wetlands are (fully) saturated after the large precipitation event and are not capable of storing more water. Thus the combination of an increased base flow and no additional storage capacity can perfectly explain the observed increase in peak stream flow on this particular day. Given the fact that this particular basin does not have any flooding problems the small increase in peak stream flow volume is unlikely to result in large damages downstream. Thus, wetland restoration will in general reduce peak stream flow.

From the above analysis it can be concluded that wetland restoration results in a decrease in annual stream flow. Although differences in FDC are small other analyses clearly highlight differences in stream flow volume after wetland restoration. Moreover, wetland restoration results in a reduction of peak stream flow volume in almost all cases and reduces the flow volume exceeding both higher and lower discharges over the entire study period. This is likely to result in a decrease in erosion of near-channel sources. Per month the reductions are highest during May-June, currently the months with the highest turbidity levels. However, reduction in the early months of the year comes at the cost of increases in stream flow during the months August-October. But increases are always far smaller compared to decreases and increases during the later months of the year is not necessarily a problem in the LSRB. Although not presented here it has to be noted that the observations made on the Le Sueur River also hold for the other rivers in the LSRB.

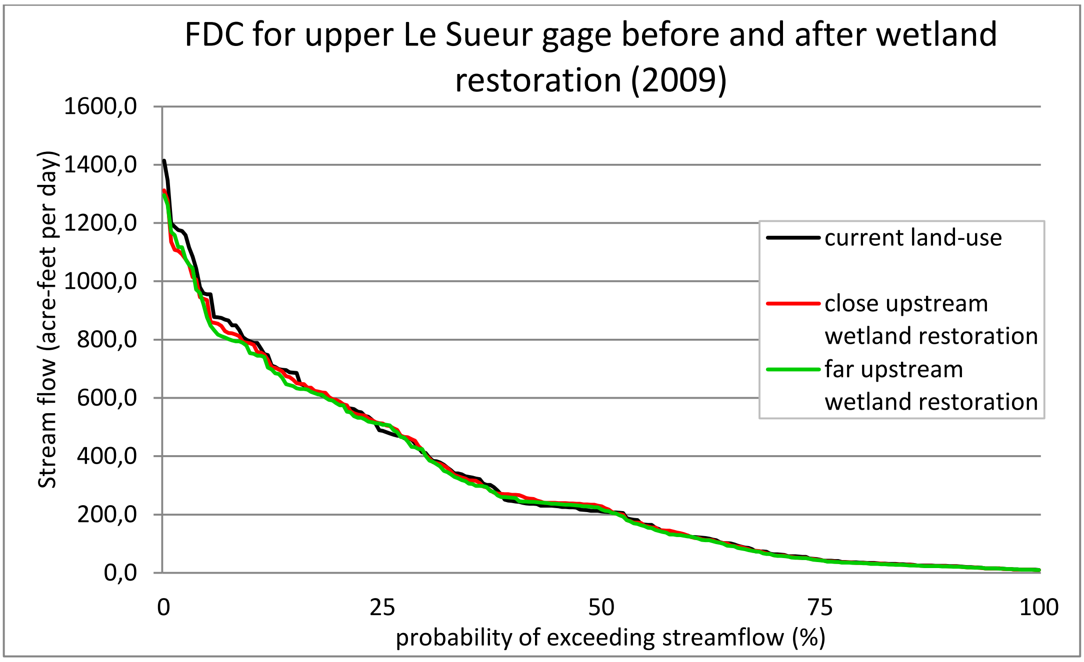


Figure 10: FDC for the upper Le Sueur gage before and after wetland restoration in 2009.

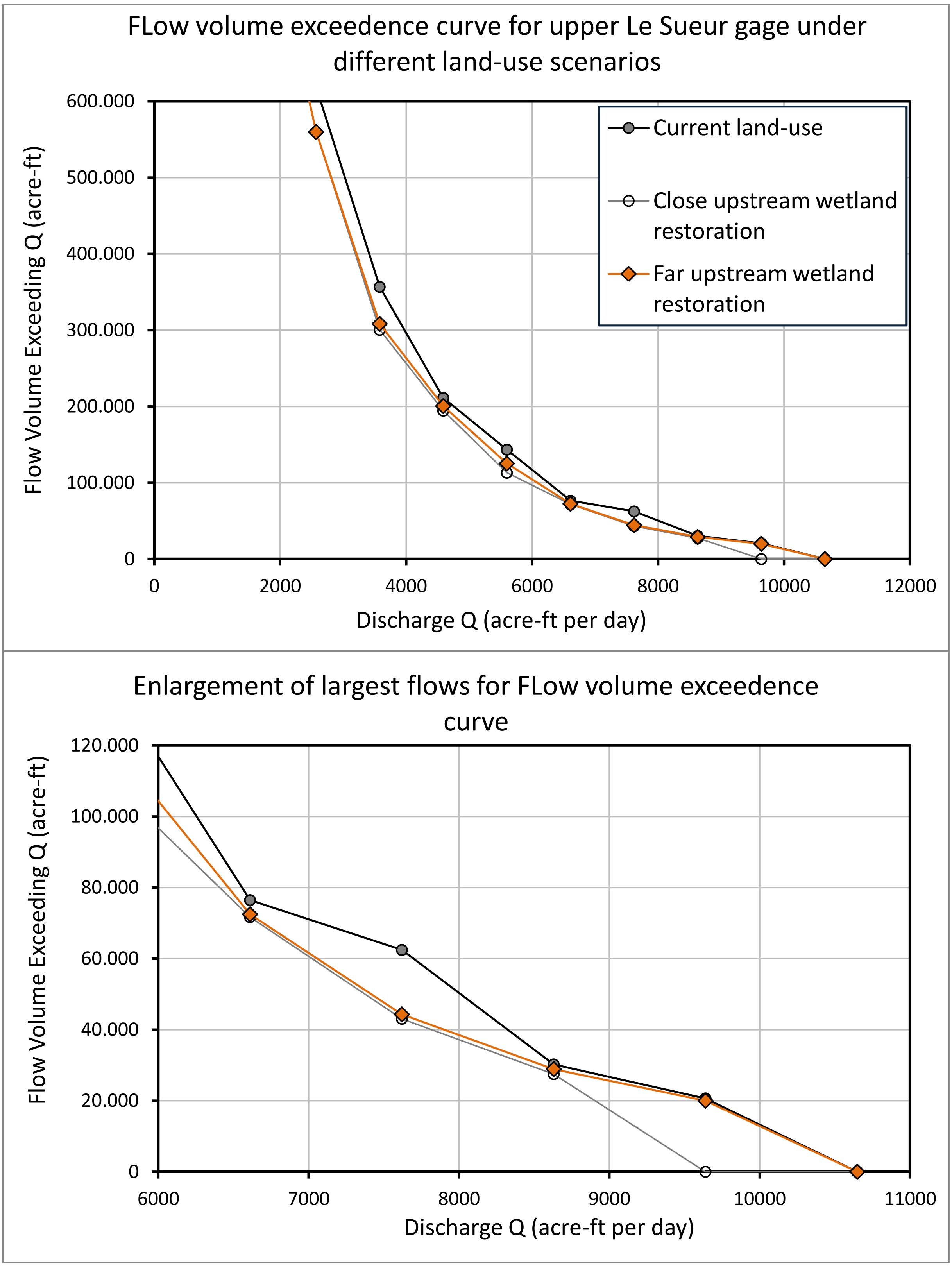


Figure 11: Flow volume exceedence curve for the entire study period at the upper Le Sueur gage before and after wetland restoration. The bottom graph only shows the results for the higher discharges

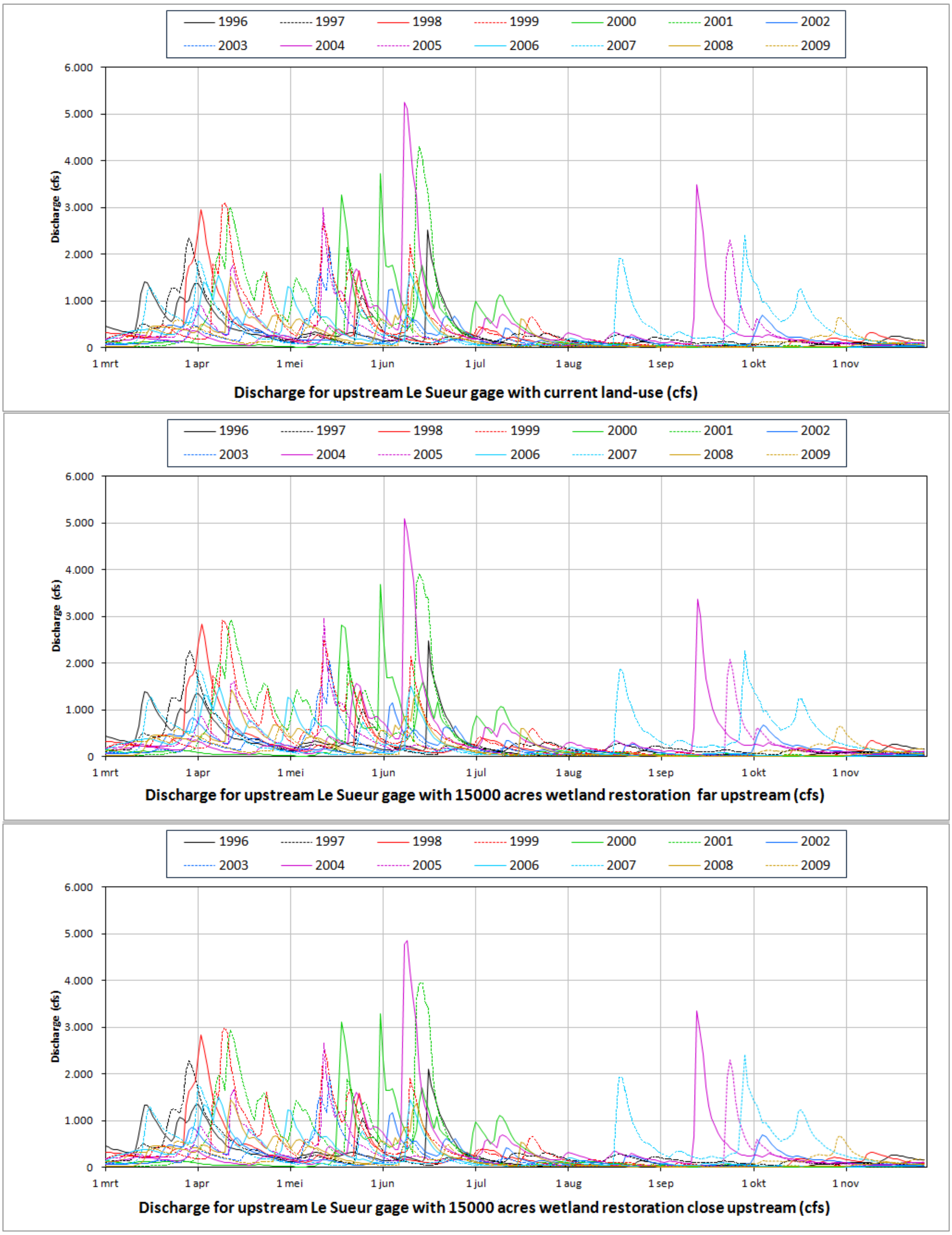


Figure 12: Yearly stream record at upper Le Sueur gage for current land-use and 15.000 acres of wetland restoration far and close upstream from 1996-2009

Table 4: Changes in total stream flow per month after wetland restoration. Green values indicate decrease in stream flow after wetland restoration whereas a red value indicates increases in stream flow after wetland restoration.

Difference per month between current land-use and 15.000 acres wetland restoration far upstream (cfs)

upper Le Sueur	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	Average monthly change
March	-634,57	-433,30	-220,47	-38,03	-7,76	112,37	-116,94	-77,75	196,81	-218,84	-268,49	-219,89	-7,48	109,70	-130,33
April	-151,30	-98,54	-644,68	-1584,87	-70,72	-574,38	-387,81	-209,55	-227,44	-864,61	-853,18	-192,99	-509,99	-246,58	-472,62
May	-202,17	-605,20	-1248,46	-1052,53	-1199,63	-950,21	-166,45	-1091,80	-1032,69	-524,46	-329,18	-514,40	-313,87	-374,74	-686,13
June	-496,66	-244,46	-453,85	-393,47	-1041,87	-1893,08	-949,98	-530,02	-1533,76	-217,54	-628,76	-299,92	-1080,88	-833,71	-757,00
July	-144,66	-185,58	-363,24	-471,74	-650,09	-85,38	-483,93	-262,11	-66,71	-166,45	-79,95	-121,90	-187,70	-128,85	-242,73
August	24,73	114,91	-42,81	91,12	15,88	-24,37	338,95	-38,86	104,17	-28,44	17,65	-154,43	-34,50	3,84	27,70
September	9,91	190,67	-13,89	-24,02	-15,85	13,32	32,11	-8,79	-360,03	-877,32	29,69	168,99	-7,47	-5,00	-61,98
October	-1,51	-29,19	53,38	-20,33	-13,04	3,78	85,56	-5,22	125,41	149,29	-3,59	-409,90	-9,64	310,63	16,83
total change per year	**-1596,23**	**-1290,70**	**-2934,03**	**-3493,87**	**-2983,07**	**-3397,96**	**-1648,49**	**-2224,12**	**-2794,24**	**-2748,36**	**-2115,82**	**-1744,44**	**-2151,53**	**-1164,72**	

Difference per month between current land-use and 15.000 acres wetland restoration close upstream (cfs)

upper Le Sueur	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	Average monthly change
March	-553,65	-404,41	-339,39	-165,02	46,72	22,58	-36,79	-37,00	155,62	-113,03	-219,26	-138,16	10,45	-3,32	-126,76
April	-98,01	-147,57	-738,33	-1113,85	-78,06	-546,47	-181,01	-317,52	-178,36	-719,27	-605,94	-496,19	-518,31	-137,24	-419,72
May	-191,89	-415,58	-323,37	-743,17	-831,15	-889,00	-200,01	-1044,80	-690,75	-882,99	-545,54	-203,37	-251,98	-403,60	-544,08
June	-1089,83	-473,68	-322,18	-806,07	-1799,40	-1311,11	-858,24	-429,08	-1715,09	-167,37	-1047,04	-451,74	-968,30	-373,06	-843,73
July	-120,97	-222,56	-320,69	-254,77	-112,07	-120,12	-433,58	-154,86	-201,24	-257,68	-123,70	-131,83	-425,50	-71,64	-210,80
August	10,82	146,71	-58,11	62,30	-27,76	-5,79	25,89	-37,59	163,45	-47,13	23,26	156,49	-47,04	-6,12	25,67
September	69,40	115,68	-17,97	19,15	-20,33	-3,56	89,21	-8,93	-14,89	-3,19	104,23	362,91	-18,18	1,75	48,23
October	-9,35	-39,33	89,68	-17,12	-6,24	17,97	183,71	-5,85	299,51	-20,63	112,69	-136,99	-8,19	344,12	57,43
total change per year	**-1983,47**	**-1440,73**	**-2030,36**	**-3018,55**	**-2828,29**	**-2835,50**	**-1410,82**	**-2035,63**	**-2181,76**	**-2211,29**	**-2301,30**	**-1038,88**	**-2227,06**	**-649,10**	

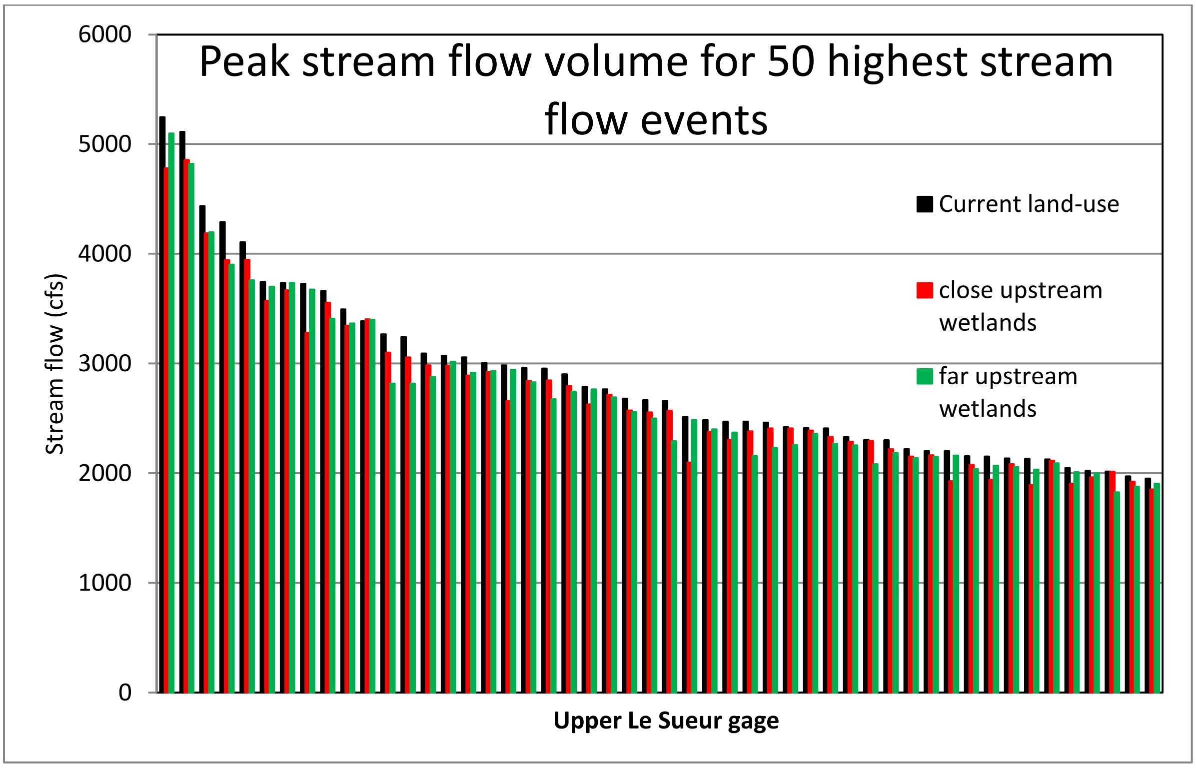


Figure 13: Peak stream flow volume for 50 largest events before and after wetland restoration at the upper Le Sueur gage.

4.2 Differences in stream flow response between wetland restoration location

So far this research has discussed stream flow changes of wetland restoration without discussing differences between the locations of wetlands. This section will discuss differences in location of wetland restoration based on localized differences and distance to the gage. The first part of the analyses focuses on localized differences between locations of wetland restoration and looks at the differences in monthly and annual stream flow response between far and close upstream wetland restoration. First the differences between far and close upstream wetland restoration for the analyses presented in paragraph 4.1 for only the Le Sueur River will be discussed and then for all rivers a comparison will be made between far and upstream wetland restoration. Often differences between current land-use and wetland restoration are larger than differences between the locations of wetland restoration. First, looking at the FDC for 2009 no clear differences are observed between far and close upstream wetland restoration (figure 10). The flow volume exceedence curve also highlights the fact that both scenarios have rather similar responses (figure 11). However, here close upstream wetland restoration always results in a lower stream volume exceeding a certain discharge. This becomes even clearer for the highest flows where only close upstream wetland restoration reduces the largest stream flow events observed. Thus, although differences are small it seems that close wetland restoration results in larger total volume reductions exceeding a certain threshold discharge volume and will thus result in larger reductions in erosion of near-channel sources.

From the stream flow record it can be observed that the highest peaks in stream flow are often more reduced by close upstream wetland restoration but at the other hand it seems that far upstream wetland restoration results in lower stream flow events at the beginning and end of the year (figure 12). This trend is far more easily observed in table 4. Both far and close upstream simulations have reductions in annual stream flow for all years and results in most cases in reductions in stream flow per month. Increases are also observed in both scenarios but do not necessarily take place at similar times. The scenario leading to the largest reduction differs per year. Average monthly reductions in stream flow highlight an interesting trade-off between close and far wetland restoration. In the month June close upstream wetland restoration results in the highest reduction in stream flow. But during the other months with reductions in stream flow in both scenarios far upstream wetland restoration results in larger reductions in stream flow. This is most pronounced in the month May. Later on during the year both scenarios result in increases in the months August and October, but the increases observed in October are smaller under far upstream wetland restoration. The opposite is true for the month of August but differences over the entire study period are more or less diminishable although differences per year can be larger. In September close upstream wetland restoration results in an on average increase in stream flow whereas far upstream wetland restoration results in a decrease in stream flow. In general, far upstream wetland restoration outperforms for most months the closer upstream wetland restoration either resulting in more decreases or in smaller increases. However, in the month of June close upstream wetland restoration results in a larger decrease in stream flow. Based on yearly stream flow reductions the scenario leading to the largest reduction differs per year.

Figure 13 compares the 50 highest peak stream flow events in the basin. The results presented in this graph again show that the scenario resulting in the largest reductions depends on the day studied. Often differences between the two wetland scenarios are small. From the above analyses it is difficult to tell whether close or far upstream wetland restoration results in the most beneficial effects and mainly highlights that there are likely to be trade-offs between the two scenarios based on reductions per year, per month and per peak stream flow event. Moreover, differences observed between the two scenarios are relatively small.

Two other comparisons are made between close and far upstream wetland restoration. These comparisons are made for all daily scenarios at all three rivers. However, results for the medium close and medium far scenario runs are not presented since these runs always resulted in the worst performance and it can therefore be concluded that wetland restoration is always more beneficial either far or close upstream in the LSRB compared to wetland restoration in-between these zones. The first comparison calculates the reduction per acre in the months May-June (table 5) and the second calculates the ratio of increases in stream flow over decreases in stream flow (table 6). Again both far and close upstream wetland restoration for all three rivers do not show large differences. However, for the Le Sueur River far upstream wetland restoration results in the largest reductions per acre in the months May-June and also results in relatively smaller increases compared to decreases. Interestingly, for the Cobb and Maple River the results are less unequivocal. This might be related to the fact that the Cobb River and the Maple River have several tributaries close to the upper gage making a distinction into two zones slightly more difficult but it could also be related to localized conditions on the Cobb and Maple River. For the Maple River close upstream wetland restoration results in the largest reductions per acre in the months May-June. However, this comes at a cost of relatively larger increases in stream flow. But it has to be noted that both far and close upstream wetland restoration result in diminishable low ratio values and therefore one could argue that for the Maple River close upstream wetland restoration has most beneficial effects. For the Cobb River far upstream wetland restoration results in the largest reductions per acre in the months May-June at the lower gage. However, the opposite is true if wetlands are only restored upstream of the upper gage and the stream flow response is measured at the upper gage. Close upstream wetland restoration at all gages for all scenarios on the Cobb River does result in lower ratio values and differences are larger compared to the other rivers. Based on these results it is impossible to conclude whether far or close upstream wetland restoration is most beneficial on the Cobb River although wetland restoration on either of the two regions might result in diverging stream flow responses. These comparisons show that far or close upstream wetland restoration as different effects depending on the river studied. More importantly these results show that the differences between wetland restoration in these zones are very small. This is in line with the previous observations.

However, table 4 highlighted that differences per month or year can differ between the two scenarios and therefore an additional analysis is performed for the Le Sueur River. The reduction per acre in the months May-June is calculated per year and the ratio of increases over decreases is calculated for each month over the entire study period (table 7 & 8). The differences in reduction per acre in the month May-June are larger per year between the two scenarios. However, these results also highlight previous observations that the best performing scenarios changes per year. It is interesting to note that the dominant scenario does not change between large, medium and small wetland restoration except for the years 2002 and 2003. Although differences are small, in 2002 the small wetland restoration scenario differs from the others in most dominant scenario whereas in 2003 the large wetland restoration scenario differs from the other two. This might be related to the fact that these two years are the wettest on record. Thus far upstream wetland restoration performs best based on reduction per acre in the months May-June over the entire study period but per year the best performing scenario changes.

Table 8 depicts the ratio of increases over decreases averaged per month for the entire study period. For this calculation the dominant scenario changes per month. The months important observation is that for both far and close upstream wetland restoration increases in stream flow during the months April till July are diminishable and both scenarios will result in decreases in stream flow on almost all individual days during these months of the year. This table also highlights some interesting trade-offs between the two scenarios. In March close upstream wetland restoration is preferable over far upstream wetland restoration and results in decreases in stream flow for both scenarios. Later on during the year far upstream wetland restoration clearly outperforms wetland restoration located close upstream. Often resulting in smaller increases during these months but sometimes even resulting in decreases instead of increases in stream flow. This is especially true for the month September when far upstream wetland restoration has a ratio value below 1 and close upstream wetland restoration always has a ratio value bigger than 1. Also differences in this month between the ratio values are large and especially the increases compared to the decreases for the close upstream wetland restoration scenario are far larger than observed in the other periods. In the month of March stream flow is mainly governed by snow melt and close upstream wetland restoration then results in the largest reductions in stream flow for the Le Sueur River. Later on during the year stream flow is governed by EVT, and also somewhat by precipitation. During these months far upstream wetland restoration results in the lowest ratios. This is especially true for the month September. During this month the potential evaporation from croplands in the LSRB calibrated model suddenly increases a lot and evaporation from wetlands is only slightly smaller in this month.

Table 5: Reduction per acre in the months May and June over the entire study period for different scenarios of far and close upstream wetland restoration at different locations. * For this scenario only 2.700 acres of wetlands are restored

		Le Sueur	Maple	Cobb	
Reduction per acre (May-June)		*Upper gage*	*Upper gage*	*Upper gage*	*Lower gage*
Large wetland restoration	*Far*	-0,10	x	x	x
	Close	-0,09	x	x	x
Medium wetland restoration	*Far*	-0,10	-0,10	x	-0,10
	Close	-0,10	-0,11	x	-0,09
Small wetland restoration	*Far*	-0,10	-0,10	-0,08*	-0,10
	Close	-0,09	-0,11	-0,10*	-0,10

Table 6: Ratio of volume of days with increased discharge over volume of days with decreased discharge over the entire study period for different scenarios of far and close upstream wetland restoration at different locations. * For this scenario only 2.700 acres of wetland are restored.

		Le Sueur	Maple	Cobb	
Ratio increase/decrease		*Upper gage*	*Upper gage*	*Upper gage*	*Lower gage*
Large wetland restoration	*Far*	0,07	x	x	x
	Close	0,09	x	x	x
Medium wetland restoration	*Far*	0,06	0,01	x	0,15
	Close	0,07	0,01	x	0,11
Small wetland restoration	*Far*	0,06	0,02	0,13*	0,15
	Close	0,08	0,01	0,10*	0,10

In conclusion, differences between wetland restoration in far and close upstream based on localized conditions are very small. Trade-offs exist for largest reduction over the entire study period, largest reductions or increases per month, largest reduction per year and effect on peak stream flow. Based on the two comparisons calculated for the Le Sueur River far upstream wetland restoration is slightly more favorable over close upstream wetland restoration. However, these results do not hold for either individual years or months depending on the comparison studied. Also, for the Maple and the Cobb River differences between the two calculated comparisons are small resulting in no preferred area for the Cobb River and close upstream wetland restoration being slightly better performing on the Maple River. Based on localized conditions there is no basis to state that wetland restoration is more suited in either close or far upstream areas. Thus, based on localized conditions the entire area upstream of the upper gages is suitable for wetland restoration. This creates a larger potential for wetland restoration in the LSRB.

Table 7: Reduction per acre in May-June per year at the upper Le Sueur gage.

		Upper Le Sueur Gage													
Reduction per acre (May- June)		*1996*	*1997*	*1998*	*1999*	*2000*	*2001*	*2002*	*2003*	*2004*	*2005*	*2006*	*2007*	*2008*	*2009*
Large wetland restoration	*Far*	-0,05	-0,06	-0,11	-0,10	-0,15	-0,19	-0,07	-0,11	-0,17	-0,05	-0,06	-0,05	-0,09	-0,08
	Close	-0,09	-0,06	-0,04	-0,10	-0,18	-0,15	-0,07	-0,10	-0,16	-0,07	-0,11	-0,04	-0,08	-0,05
Medium wetland restoration	*Far*	-0,05	-0,06	-0,11	-0,10	-0,15	-0,19	-0,08	-0,11	-0,19	-0,05	-0,07	-0,05	-0,09	-0,08
	Close	-0,09	-0,06	-0,03	-0,12	-0,17	-0,16	-0,06	-0,12	-0,17	-0,08	-0,10	-0,03	-0,09	-0,06
Small wetland restoration	*Far*	-0,05	-0,06	-0,13	-0,10	-0,15	-0,21	-0,06	-0,11	-0,18	-0,05	-0,05	-0,05	-0,10	-0,09
	Close	-0,09	-0,06	-0,03	-0,12	-0,16	-0,15	-0,06	-0,12	-0,16	-0,07	-0,10	-0,03	-0,09	-0,06

Table 8: Ratio increase over decrease at the upper Le Sueur gage per month of the entire study period.

		Upper Le Sueur Gage							
Ratio increase/decrease		*March*	*April*	*May*	*June*	*July*	*August*	*September*	*October*
Large wetland restoration	*Far*	0,19	0,00	0,00	0,00	0,00	2,20	0,34	1,48
	Close	0,12	0,00	0,00	0,00	0,00	2,57	8,76	4,30
Medium wetland restoration	*Far*	0,19	0,00	0,00	0,00	0,00	1,86	0,38	1,19
	Close	0,05	0,00	0,00	0,00	0,00	2,29	6,49	2,92
Small wetland restoration	*Far*	0,20	0,00	0,00	0,00	0,00	0,87	0,21	1,41
	Close	0,06	0,00	0,00	0,00	0,01	1,93	7,81	3,39

However, differences in location of wetland restoration are not only dependent upon localized conditions but also on distance to the gage. As hypothesized by Loucks (1989) and Anderson and Kean (1994) far upstream wetland restoration would be more beneficial because of the 'time-delay effect' of wetland restoration. Hydrographs, on an hourly time-step, are depicted for events with the largest peak stream flow in the years 2003-2007 (figure 14). 4 out of the 5 graphs show that close upstream wetland restoration results in a larger reduction in peak stream flow volume. Only for the October 2007 event far upstream wetland restoration results in a larger reduction in peak stream flow. By far the largest portion of the peak stream flow events happen during the months May and June and only a few events occur at the end of the year. Therefore, it is likely that the 2007 event represents a rather unique case and that for most of the peak stream flow events wetland restoration close upstream results in larger reductions. Even so, the differences between the 2007 event and the other events are not due to localized differences and not due to how far upstream the wetland is restored. Moreover, both far and close upstream wetland restoration do not result in a delay of the peak stream flow. However, for all events the recession curve for both wetland restoration scenarios and the current land-use scenario converge over time. Thus water that is stored during peak stream flow is released at a later time as base flow. Water is stored for a long enough period that increases in base flow do not result in larger peaks in the stream flow but only in converging recession curves. This is highlighted by the 2004 event that had several smaller rainfall events after the initial largest rainfall event on June 9th. Because of the initial rainfall event the wetland are almost fully saturated. The combination of increases in base flow and the low storage capacity of the wetland result in larger peak stream flow during a short period of time for both wetland restoration scenarios. The fact that only multiple large rainfall events in rapid succession result in an increase in the stream flow due to the delay-time effect of wetland restoration makes it clear that this is for the LSRB a rather unique situation. The fact that close wetland restoration is most effective in reducing peak stream flow is likely due to the fact that most water during these events is generated by these areas. The further upstream later contributing areas hardly contribute a significant portion to peaks in the stream flow and wetland restoration in this area thus hardly influences peak stream flow. Only occasionally, most water is generated by the far upstream late contributing areas, as is the case in the October 2007 event.

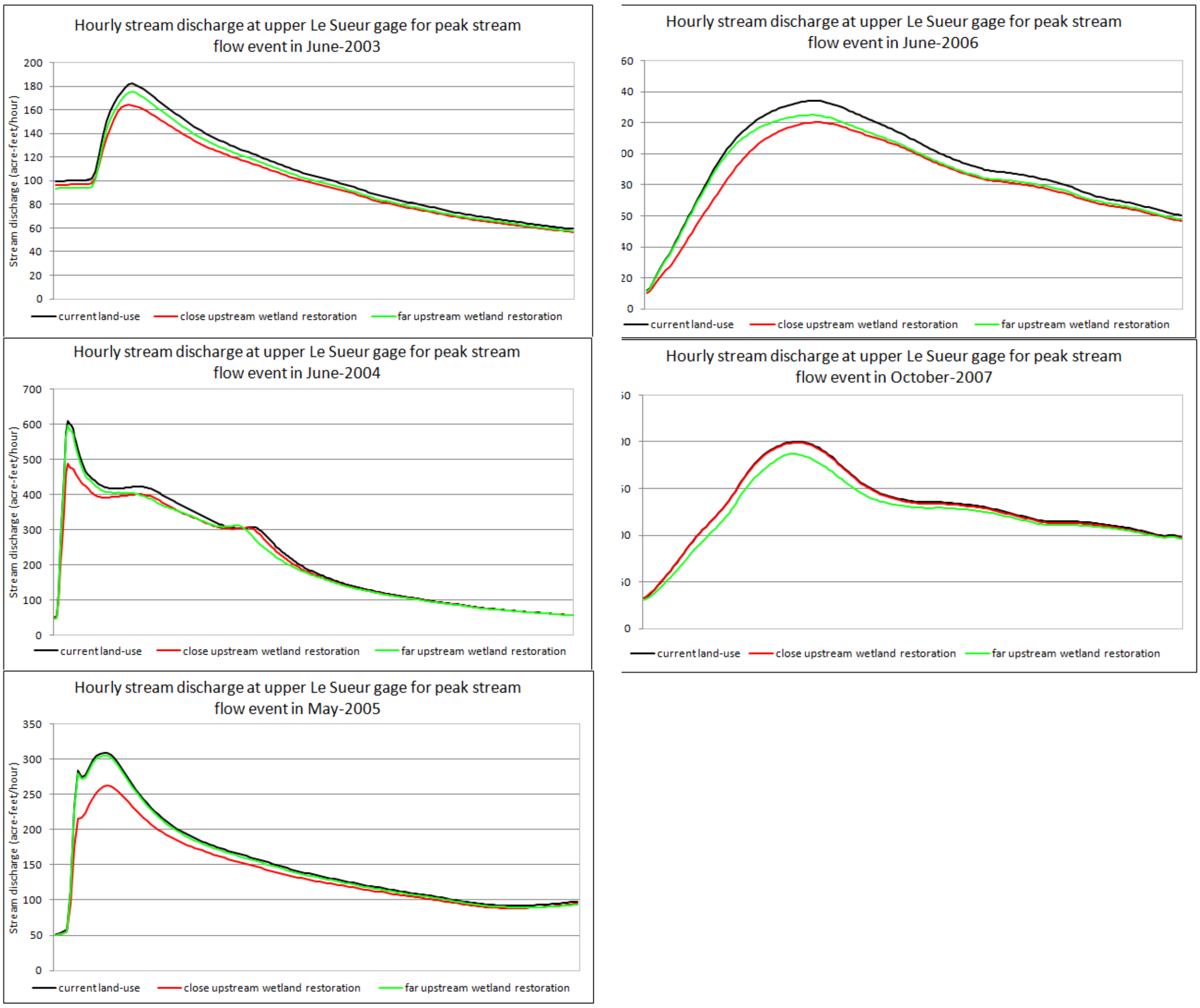


Figure 14: hydrographs with an hourly time-step for the largest peak stream flow event in 2003, 2004 and 2005 before and after far and close upstream wetland restoration.

Thus, based on these results the hypothesis that wetland restoration in far upstream late contributing areas has a more beneficial effect on peak stream flow has to be rejected. Even more so, the result suggest that close upstream wetland restoration will reduce peak stream flow more, in line with previous findings from Ogawa and Male (1986).

The combination of the difference in stream flow response due to location of wetlands based on localized differences and distance to the gage results in the following conclusion. Localized differences between wetland restoration are small and provide no real basis for selecting an optimal location. Differences in wetland restoration from the distance to the gage suggest that for most peak stream flow events wetland restoration closer to the gage will result in larger reductions in peak stream flow. The hypothesis by Loucks (1989) that the optimal location for wetland restoration is far upstream and the hypothesis by Anderson and Kean (1994) that the optimal location for wetland restoration is in late contributing areas both have to be rejected. Sun et al. (2002) concluded that saturated wetlands respond similarly to a precipitation event as uplands. However, the fact that wetland restoration resulted in decreases in the peak stream flow indicate that the wetlands were not fully saturated even during the most extreme events per year. This strongly suggests that these observations and conclusions do not only hold for these particular events but also for smaller peak stream flow events in the LSRB.

4.3 Effects of wetland restoration at the plot scale

This part of the analyses will focus on the effect of wetland restoration at a single sub-watershed. First, this study will analyze the difference in storage. Note that for this analysis also the storage in the month November is depicted given an insight in the recession outflow from the different compartments. The spread in total storage per acre is larger for wetland compared to conventional cropland resulting in both larger and smaller storages (figure 15). This includes both short time and long time storage compartments. Figures 16 and 17 show the differences between short term and long term storage compartments for surface storage and interflow storage (short term storage compartments) and upper zone, lower zone and active groundwater storage (long term storage compartments). Wetland restoration clearly reduces the amount of water in short term storage and increases the amount of water in long term storage. Especially storage in the upper zone and active groundwater storage is increased by wetlands whereas interflow storage is most reduced.

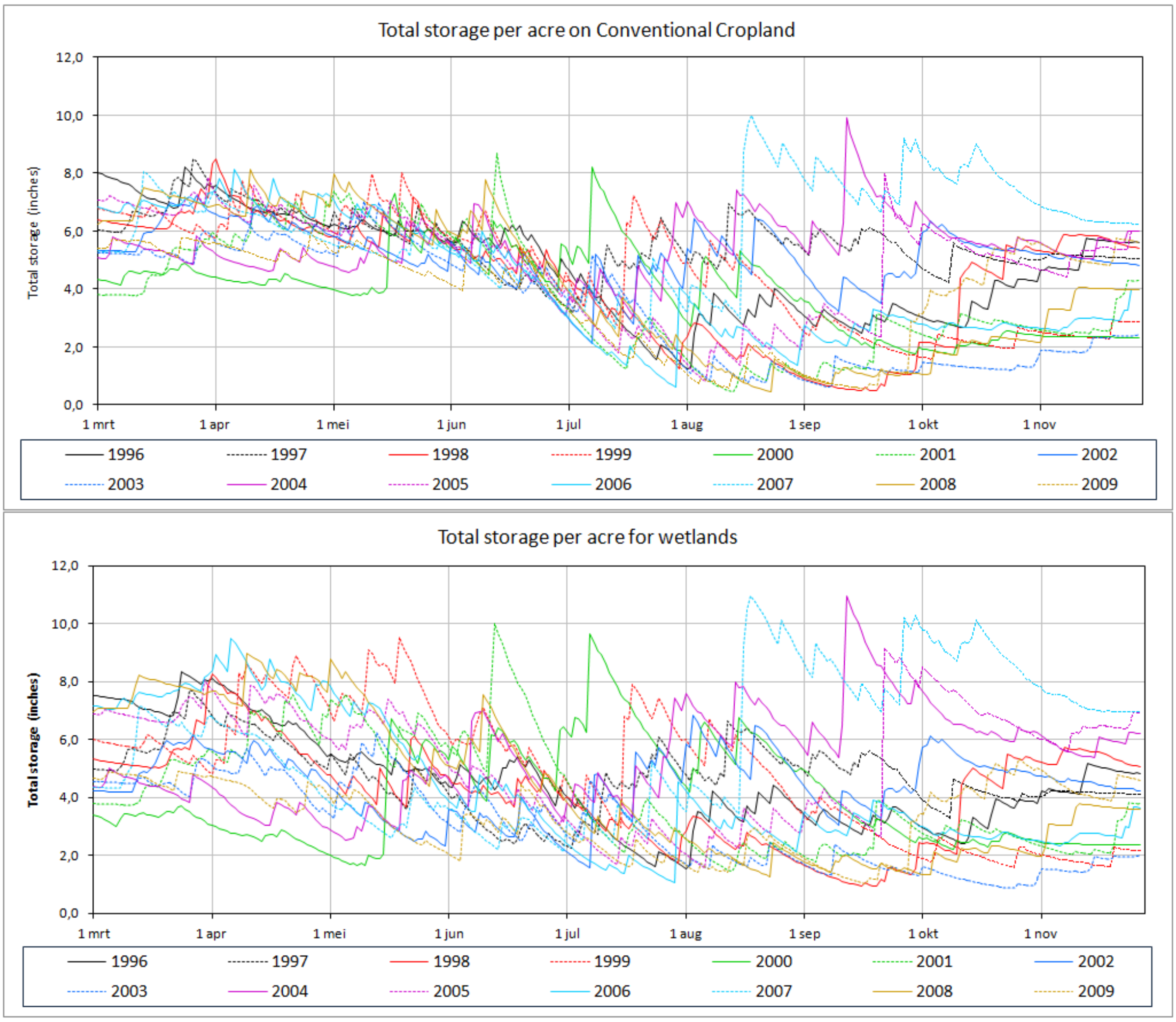


Figure 15: total storage on an acre of wetland or conventional cropland for one sub-watershed in the LSRB. Note that the legend presented at the bottom of these graphs is similar to the legend for all graphs concerning differences in storage.

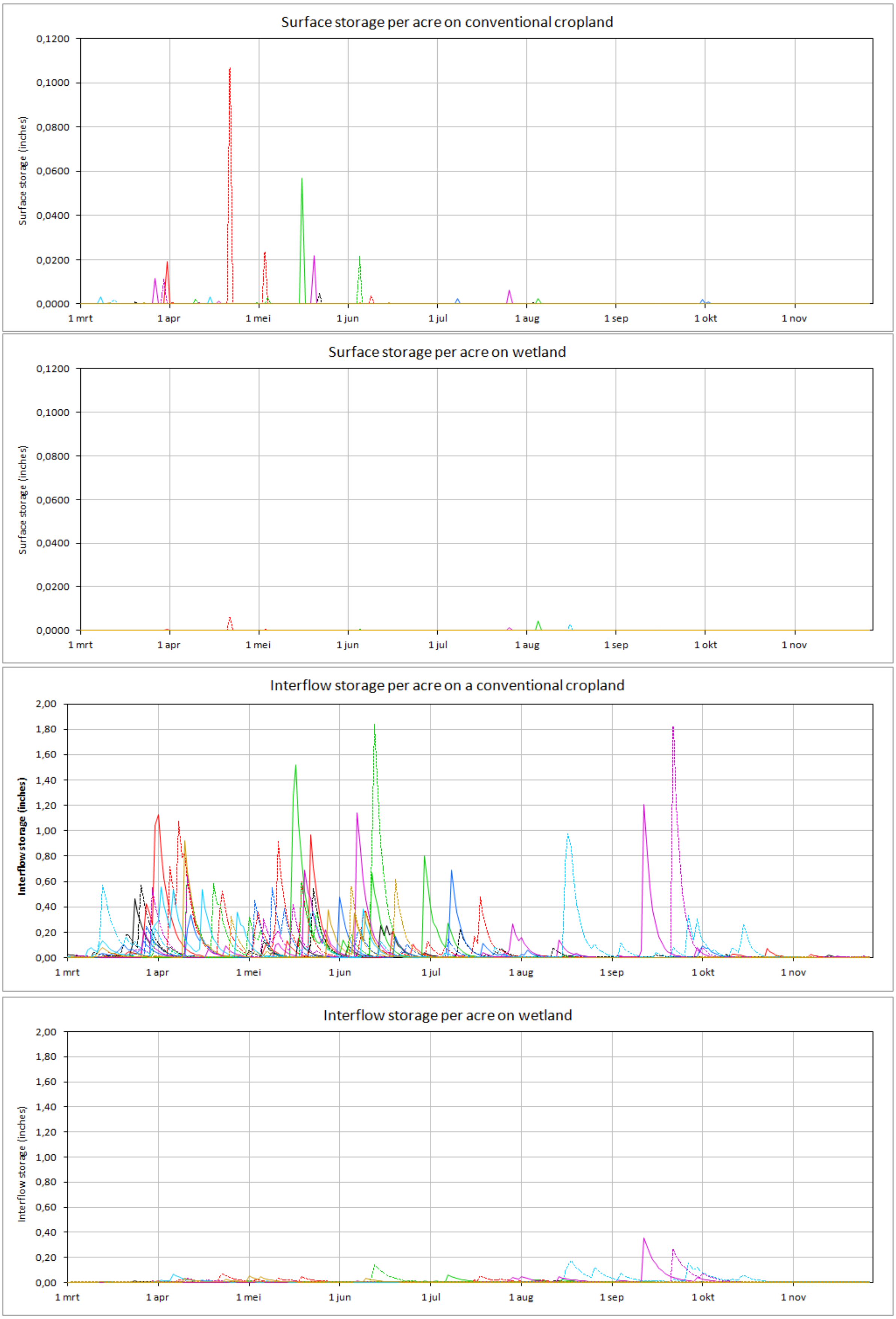


Figure 16: Short term storage per acre on conventional cropland and wetland. Different colors represent individual years. Given the fact that the legend is similar for all five figures it is only depicted in figure 14.

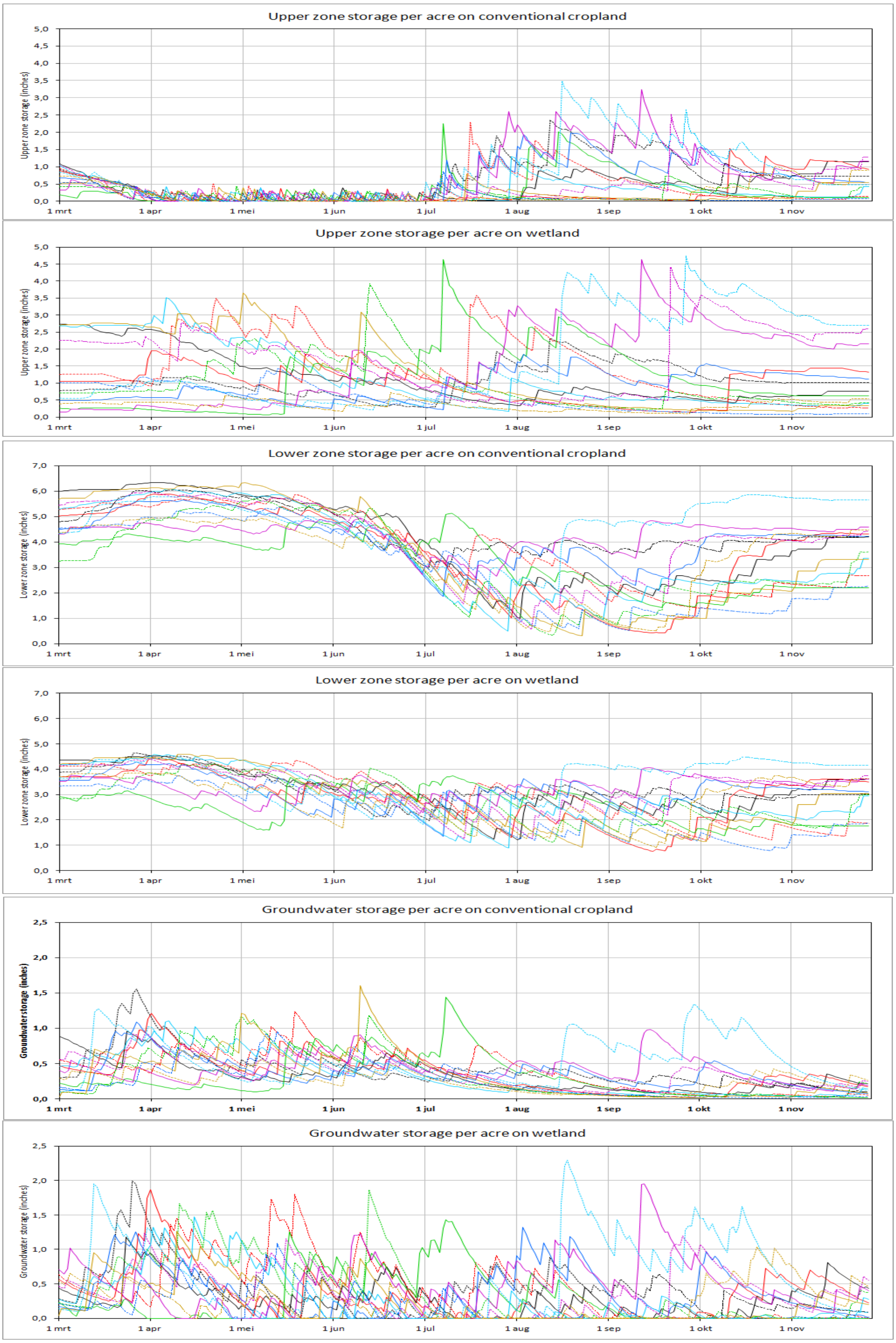


Figure 17: Long term storage per acre on conventional cropland and wetland. Different colors represent individual years. Given the fact that the legend is similar for all five figures it is only depicted in figure 14.

Total outflow to the stream from a wetland is clearly lower than total outflow from conventional cropland (figure 18). This is true for all years and for all months. And even for the largest outflow events on a cropland an acre of wetland would almost result in no outflow. All the water not flowing to the stream is for a smaller part kept in storage and for the largest part exits the system through EVT. Also a slightly larger amount of water leaves the system into inactive groundwater. Figure 19 depicts the surface, interflow and groundwater outflow. Surface and interflow outflow combined can be seen as direct runoff. From the upper zone and the lower zone no outflow to the stream occurs. Water only exits these compartments through EV, ET or percolation towards the groundwater. Direct runoff is clearly reduced by wetland restoration but outflow from the groundwater is increased. The increases in outflow from groundwater are often off-set by the larger decreases in direct runoff. However, on days where groundwater flow is the only input to the stream wetland restoration likely results in increases in stream flow.

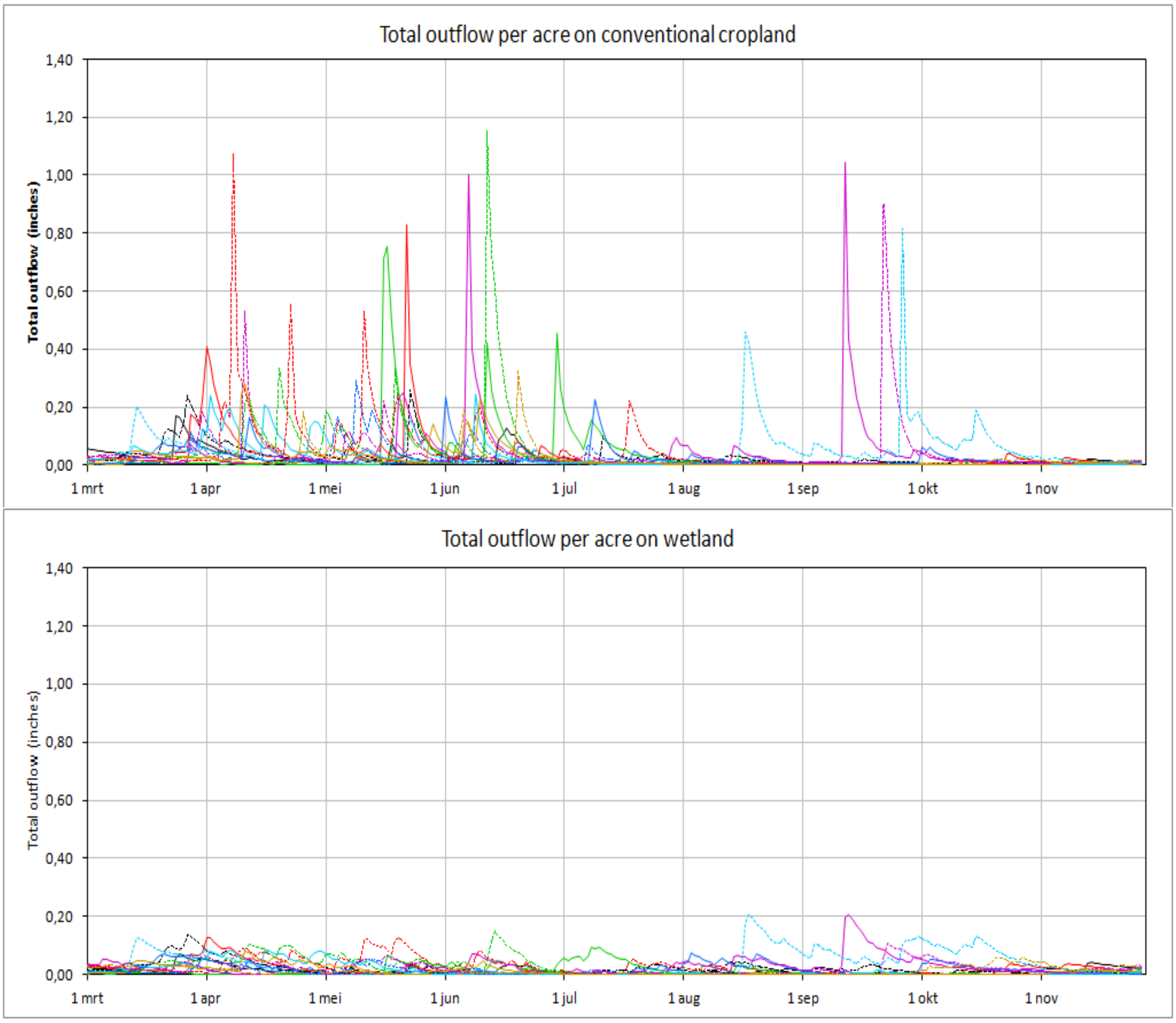


Figure 18: Total outflow per acre on conventional cropland and wetland. Different colors represent individual years. Given the fact that the legend is similar for all five figures it is only depicted in figure 14.

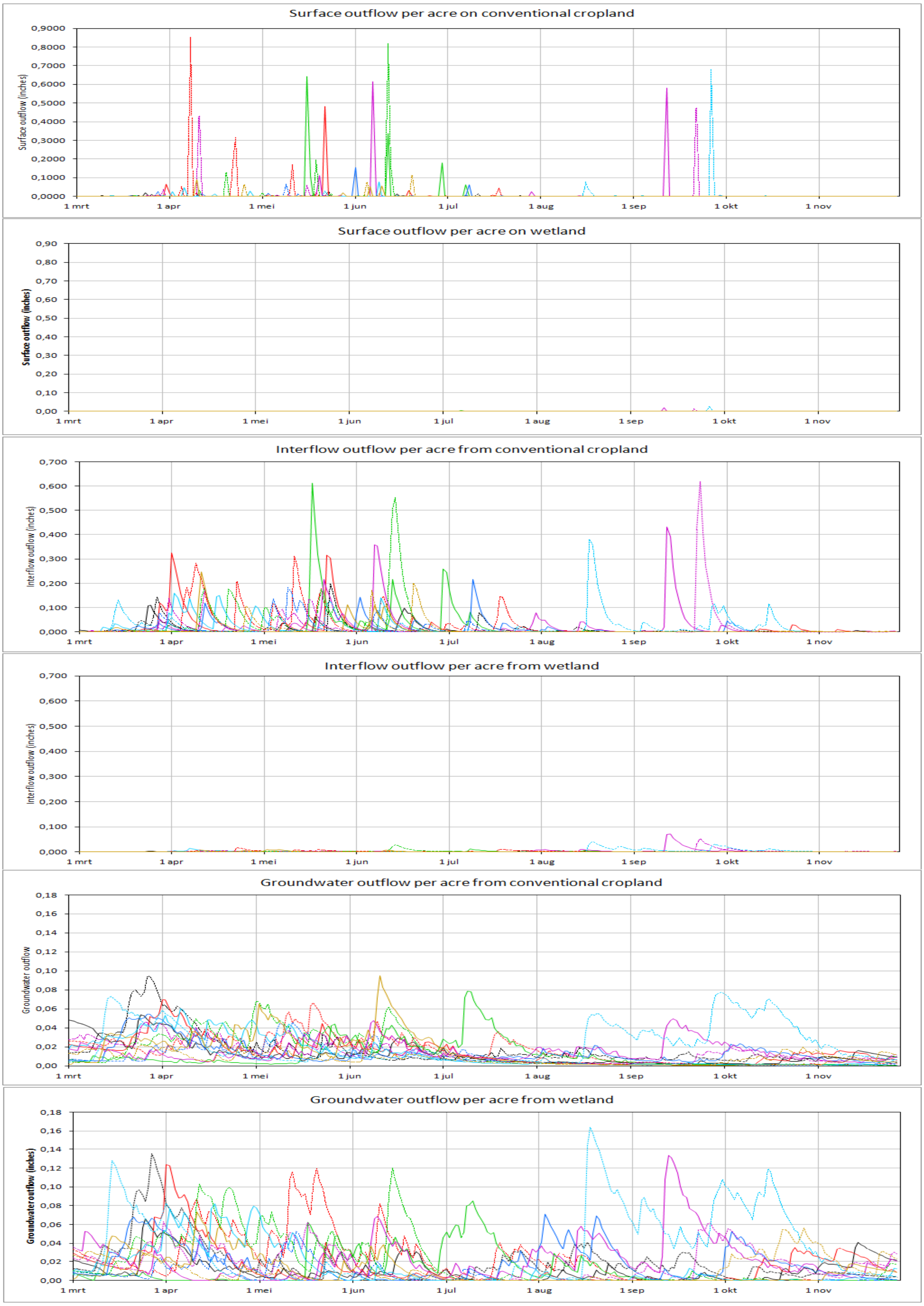
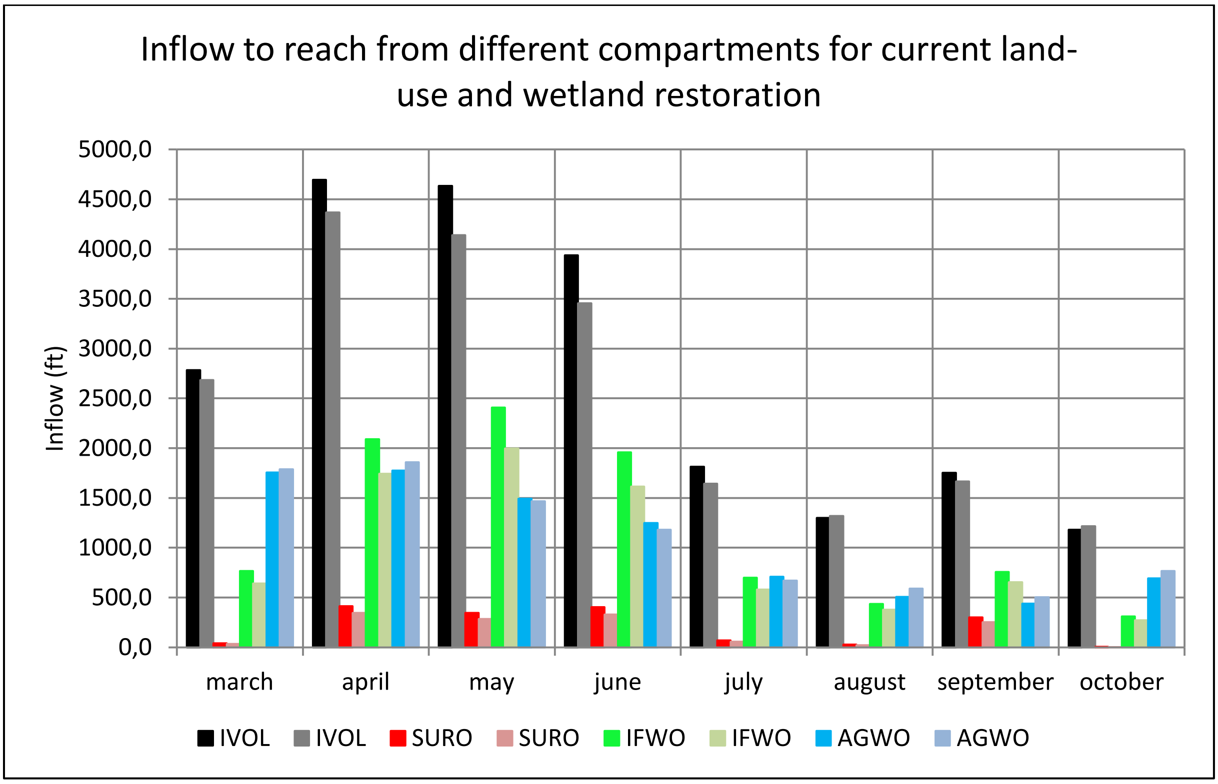


Figure 19: Outflow per acre on conventional cropland and wetland. Different colors represent individual years. Given the fact that the legend is similar for all five figures it is only depicted in figure 14.



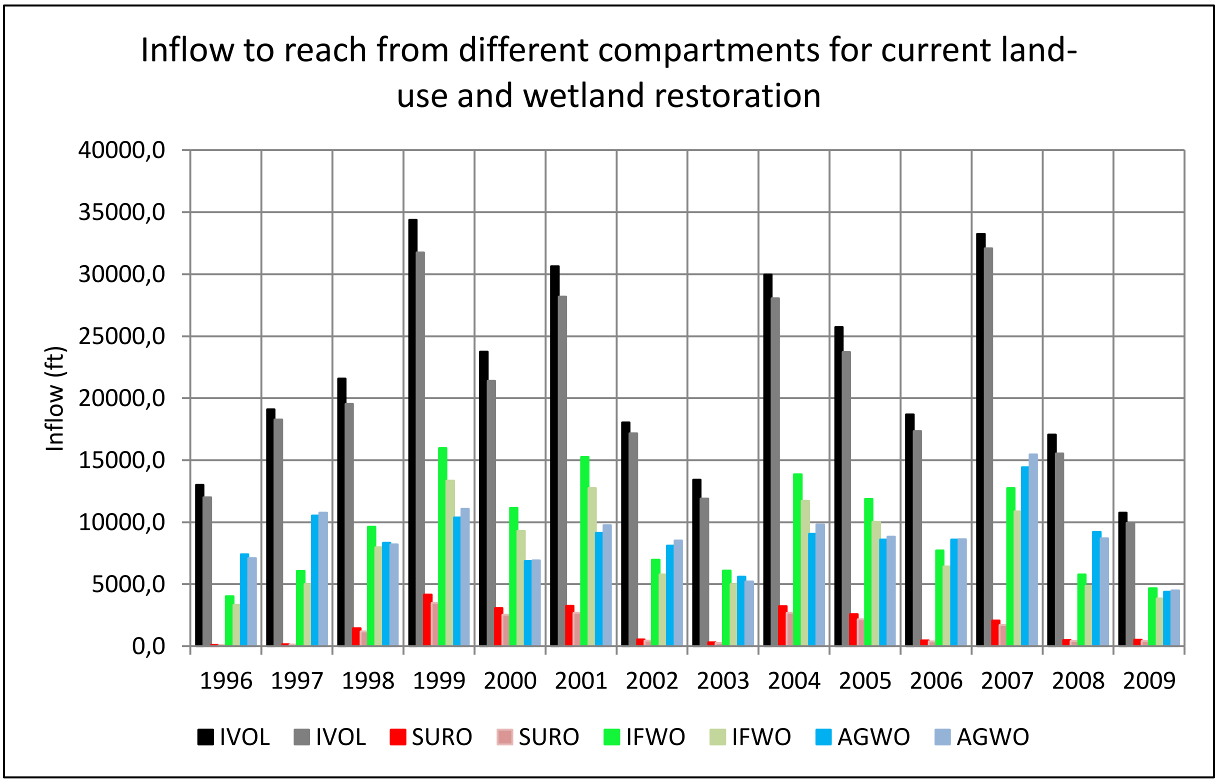


Figure 20: Inflow to the stream (IVOL) before and after wetland restoration. The dark colors represent the situation with current land-use and the lighter colors represent the situation after wetland restoration. For this analysis only the land-use categories conservational cropland, conventional cropland and wetland are taken into account. Combined these categories cover more than 95% of the total basin area.

This observation becomes even clearer if one looks at the total inflow to the stream (IVOL) before and after wetland restoration. The IVOL is the sum of the outflow from wetland and conventional and conservational cropland since the area chosen for study is a headwater stream and no water is contributed from upstream reaches. This analysis highlights some interesting features. In most years groundwater flow is increased after wetland restoration but this increase is offset by the decreases in surface runoff and more importantly interflow. Per year wetland restoration results in less water delivered to the stream. In most months and especially from April till June wetland restoration reduces inflow to the stream. In August and October wetland restoration results in small increases in inflow to the stream because the increases in groundwater outflow are larger compared to the decreases in direct runoff. This analysis shows that the increases and decreases in stream flow observed at the upper Le Sueur gage are driven by decreases in interflow and increases in groundwater flow. Although wetland restoration also reduces surface runoff this hardly impacts the overall stream flow since surface runoff constitutes only a small portion of the total outflow. Differences between the inflow to the stream and the different constituents is relatively small since still the largest portion of the sub-watershed studied is cropland, even after 5.900 acres of wetland restoration.

For an individual year (1996) and for each month within this year the relative contributions from the different constituents before and after wetland restoration are depicted (figure 21). These results highlight that except for the month of June the inflow to the stream is always dominated by groundwater. Since wetland restoration mainly results in a reduction of interflow this explains the fact that largest reductions at the gage are observed in the months May and June. This analysis shows even more clearly that without wetland restoration surface runoff only is a very small component of the total runoff. Furthermore, this analysis shows that the contribution from active groundwater is highest in September in both situations and wetland restoration does not increase the relative contribution of groundwater. This also explains why at least for far wetland restoration no increases in stream flow are observed for some years in the month of September.

The results presented so far show that wetland restoration results in a decrease in inflow to the stream. Thus water is either stored until after the study period or it exists the system through inflow to the inactive groundwater and EVT. Total actual EVT from all different compartments is higher after wetland restoration than under current land-use conditions (figure 22). For example, the decrease in inflow to the stream in the year 1999 can for 97% be explained by the increase in EVT after wetland restoration. The remaining likely being water exiting the system by inflow to inactive groundwater driven by an increase in infiltration.

Thus, wetland restoration at the sub-watershed scale results in more water being stored in the long-term storage compartments compared to the short term storage. This leads to an almost complete disappearance of direct runoff and an increase in groundwater outflow. Inflow to the stream is reduced after wetland restoration and this decrease can for the largest part be explained by an increase in total actual EVT. Per year wetland restoration always results in a decrease in inflow to the stream largely because of reductions in interflow. However, wetland restoration also results in increases in active groundwater outflow leading to increases in the inflow to the stream in some months. The above observations are nicely translated to actual stream flow changes at the gage. The observed reductions in stream flow in most months of the year are driven by decreases in interflow outflow to the stream whereas the increases in stream flow in the later part of the year are driven by increases in active groundwater outflow. The contribution of surface runoff compared to the other constituents is negligible both before and after wetland restoration.

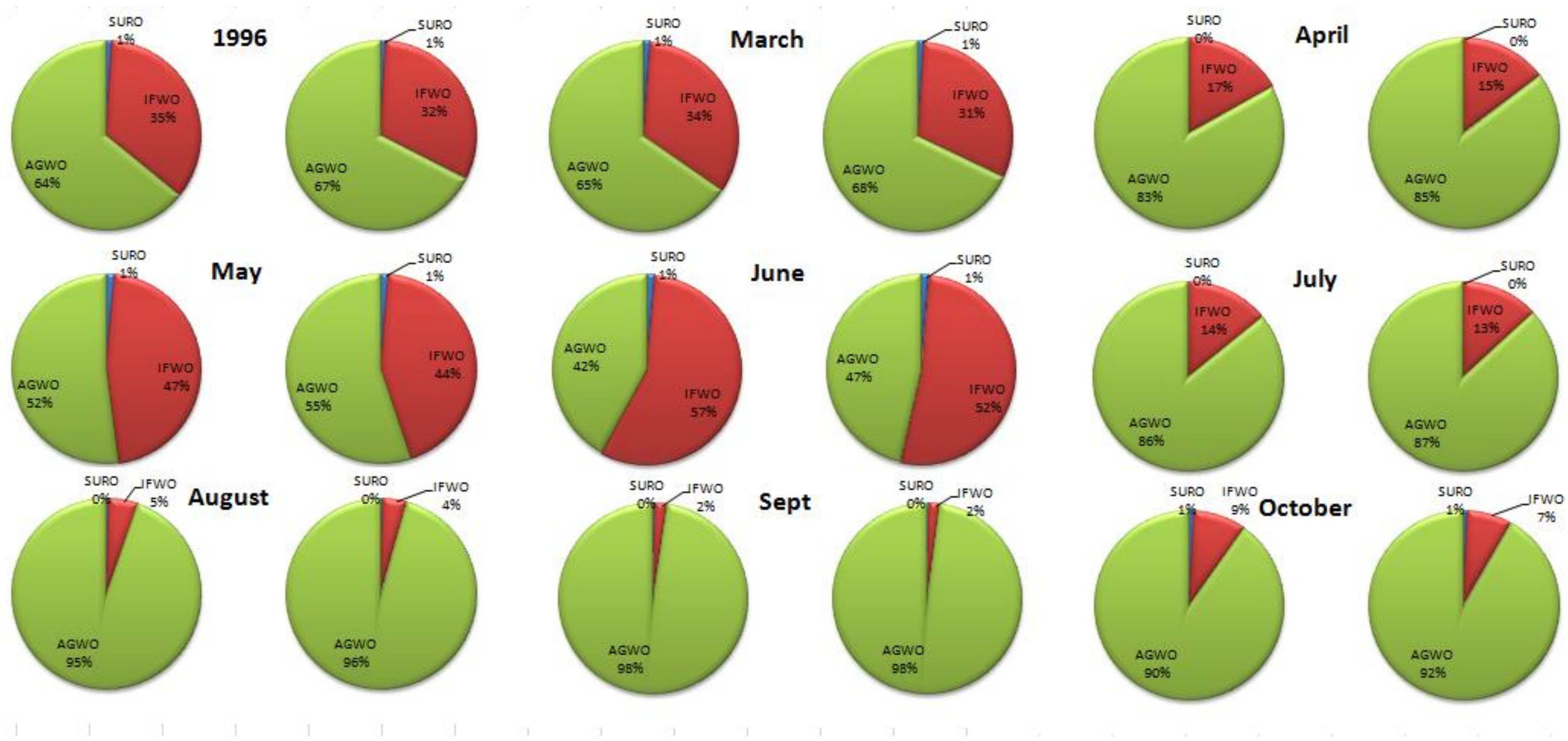


Figure 21: Relative contribution to inflow to the stream for different constituents for 1996 and for each month in 1996. On the right is the situation before wetland restoration and on the left is the situation after wetlands are restored.

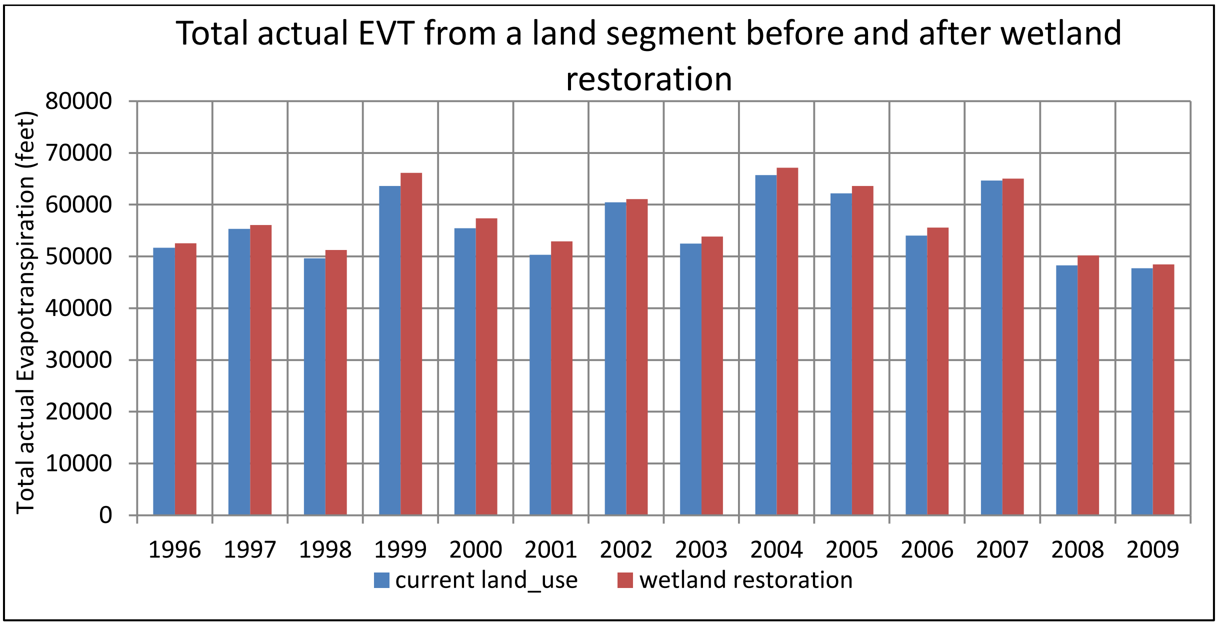


Figure 22: Total actual Evaporation and Evapotranspiration from a single sub-watershed before and after wetland restoration.

5. Wetland restoration – A broader framework

Currently a collaborative effort in the GBERB is under way to evaluate different management options for sediment reduction. A group of researchers from a set of U.S. universities performs research on several aspects of the sediment problem in the GBERB and a large part of the research is dependent upon inputs from stakeholders group consisting of farmers, industry representatives, researchers from nearby universities and lastly local and state government and agency officials. The goal of the research effort is to determine the possible extent of several management options and their effects on stream flow and/or sediment concentration. Although reduction of turbidity is the main focus in the GBERB the stakeholder group has also stated that other criteria could be important for evaluation and the decision-making process. The current set of management options under consideration includes, controlled drainage, alternative tile intakes, wetland restoration, treatment wetlands, two-stage ditch and culvert sizing, grade stabilization at side inlets, conservation tillage, filter strips, grassed waterways, terraces, water and sediments control basins (WASCOBs), streambank stabilization and bluff toe protection. This part of the research will focus first of all on the view of stakeholders regarding wetland restoration and the requirements set on wetland restoration by local stakeholders. Then an overview will be given of the costs of wetland restoration and other ecosystem services provided by wetlands. More importantly this part of the research will identify the consequences of the requirements set by local stakeholders on the costs of wetland restoration and the provision of a set of ecosystem services.

5.1 Stakeholder views and requirements for wetland restoration.

The LSRB and the GBERB are part of the larger Corn Belt region in the Midwest of the US and consist predominantly of agricultural land. Currently, the Corn Farmers Coalition is advertising the importance of corn farming for the US to policy makers in Washington, D.C. They state as part of their campaign that in 2012 corn farmers exported $10 billion dollar worth of corn making it one of the few products with a trade surplus, American corn farmers are the most productive in the world growing 20% more corn per acre than any other nation and 95% of the corn farms are family owned producing 90% of the U.S. corn (Corn Farmers Coalition, 2013). These numbers highlight the importance of the region for global food provision. Also it indicates that the vast majority of stakeholders involved in wetland restoration will be several generations of farmers. This last point is important since it indicates that farming is the way of life for the vast majority of the local land-owners and there is a critical attitude towards management options reducing the amount of arable land and a critical attitude towards agencies like MPCA in the LSRB (CSSIR, 2013). Although individuals might have differing preferences, the economic interests of the stakeholders are likely to be very homogenous. This means that essentially the stakeholders this research is dealing with are primarily farmers and the group does not consist of a mix of farmers and non-farmers as is often the case in environmental problems (see for an example Davenport et al., 2010). Kerry-Turner et al. (2000) identified nine groups of relevant users of wetlands and by far the vast majority of the stakeholders and land-owners involved in this basin can be typified as agricultural producers. This is not only true for the LSRB but for many basins in the Corn Belt region. This is of importance for two reasons; 1) value assessed to ecosystem services provided by a wetland is dependent upon the users' group and 2) land is privately owned in the basin thus land has either to be bought by the MPCA and other relevant agencies but more likely management options will have to be implemented in accordance with the land-owners. Previous stakeholder meetings have also stressed the importance of farm-friendly solutions in this area (CSSIR, 2013). Thus views of farmers towards wetland restoration and the possible requirements set by this group are therefore likely to be large influence on the decision-making process and on the question if and how wetlands will be restored.

Previous research on stakeholder perspectives regarding wetland restoration in the US has been mainly performed from the perspective of biodiversity conservation and less from the perspective of flood and erosion management. Support in general for wetlands restoration in Illinois, a state in which over 90% of original wetlands have been converted to agricultural or urbanized uses (Suloway & Hubbell, 1994), is uncertain. In a study of Illinois residents' attitudes toward open space, less than 43% of respondents rated wetlands as having high importance in the state (IDNR, 2003). In fact, wetlands were ranked 13th in importance out of the 16 types of open spaces listed, below parks, forests, playgrounds and sports fields. Interestingly, 90% of the respondents rated water quality very to extremely important, even outranking issues such as crime reduction and education. This high rate for water quality is in contrast to other studies on perceptions to wetland restoration that grant higher importance to wildlife habitat, possibilities for hunting and biodiversity conservation (Davenport et al., 2010; Milon & Scrogin, 2006; Lupi et al., 2002; Johnson & Pflugh, 2008; Kaplowitz & Kerr; 2003). Mixed findings have been reported on the perceived significance of wetland functions like flood control and water quality enhancement (Johnson and Pflugh, 2008). All above studies have been performed in the US but interestingly a literature overview of studies in the UK on wetland restoration stresses that citizens are less likely to select ecological criteria as of importance (Tunstall et al., 2000). Altogether, studies in the US have documented a high level of support for wetlands restoration in general but studies also suggest that many citizens may have limited knowledge about wetlands, especially their defining characteristics, and may even lack an awareness of wetlands existing in their own communities (Johnson & Pflugh, 2008).

The studies presented so far have focused on attitudes of the general public towards wetland restoration and less so on the attitudes of farmers. A research by Posthumus et al. (2008) focused particularly on farmers' attitudes towards wetland restoration in a basin where temporary storage of runoff water on farmland was found to have potential to mitigate flooding. They found that the participating stakeholders thought that this was beyond farmers' responsibility of good farming practice. Reducing runoff and soil erosion were considered least important by the interviewees. On the question whether farmers should take active part in flood risk management, 10 of the 25 farmers responded negatively. Fifteen farmers said they could contribute to flood risk management, but only if they received compensation to cover the costs as they considered this a service to society.

In the LSRB interviews with farmers on their perceptions towards several management options under consideration have been performed by Heitkamp (personal communication). Although the research has not been completed yet the results of the first interviews suggest that farmers prefer in-stream management of stream flow and erosion and are critical towards management on farm land. Regarding wetland restoration the results so far suggest that producers/farmers are leery of wetland restorations because of the large footprint and management issues involved. Wetland restoration goes beyond treating water quality and is truly looking to restore all aspects of wetlands. At the other hand farmers expressed that restored wetlands store more water on the landscape than treatment wetlands and have a comparable cost. Treatment wetlands were often considered to be preferred for farming practices since they require less land and are less regulated but they cost considerably more and are not always capable of processing enough water.

Besides input from interviews with farmers this research has also gathered input from a CSSIR stakeholder meeting in June, 2013 in Mankato, Minnesota. During the stakeholder meeting several interesting comments were made with regards to wetland restoration. A stakeholder challenged the impact of the storage capacity of wetlands on stream flow given the fact that wetlands would, according to the stakeholder, normally be fully saturated and evaporation is low compared to croplands during several periods of the year. It was also stated that wetlands of course trap sediment from overland flow but will also result in an increase in organic sediment production. These comments highlight the fact that there is not only a critical attitude towards wetland restoration but that also doubts exist on the actual effect of wetland restoration on stream flow reduction and erosion control.

More importantly there are several considerations to be made regarding wetland restoration and farming practices. First of all, wetlands will negatively impact crops in the proximity of the wetland because cropland areas close to a wetland will be wetter. Thus, wetlands not only take out productive land but also impact surrounding land still used for agricultural purposes. Second, pumping stations often pump several acres of land. Only restoring part of that pumped land into a wetland can result in considerable costs because possibly the entire pumping network has to be adjusted. Third, several small wetlands across an agricultural field are in conflict with farming practices since it makes plowing difficult or even impossible in certain areas. Therefore, if implemented, farmers would prefer one large wetland over a set of smaller wetlands. Fourth, wetland restoration in the proximity of ditches is preferred by stakeholders since this are already wet spots in the landscape and farmers are more willing to abandon these spots. Interestingly, another stakeholder performing research in the area stated that wetlands close to a water body (stream, lake or ditch) are preferred and result in more beneficial hydrologic effects. According to the stakeholders this should be an important criterion in ranking wetlands and might result in an ideal match between land that farmers are willing to give up and land that is best suited for wetland restoration.

Overall, given the high abundance of farmland in the GBERB management options should be farm-friendly. Farmers are critical towards wetland restoration because it often requires large parcels of land and goes beyond simple sediment and water management. The two most important requirements set by farmers on wetland restoration are that a few large sized wetlands are preferred over many small sized wetlands since the latter results in problems with plowing and areas closer to water bodies are preferred for wetland restoration both by farmers and from a hydrological perspective.

5.2 Costs of wetland restoration

For wetland restoration projects there is often a lack of accurate cost accounting resulting in considerable uncertainty in cost estimations of wetland restoration. This uncertainty often manifests itself in a significant difference between projected and actual costs of restoration projects leading to an equal set of projects resulting in cost overruns and projects being considerably under budget. The majority of cost overruns were related to construction rather than additional planning and on average, 95% of added cost went into construction (Wellmann, 1995).The uncertainty in cost estimates is highlighted by the large range in costs per acre for wetland restoration defined by previous research (table 9). It has to be noted that these costs ranges often only take into account construction and design of a wetland and do not always consider maintenance cost or land acquisition costs. However, what is and what is not taken into account is not explicitly stated for each individual source. Moreover, these costs cover all types of wetland implementation projects in all sorts of conditions in the US, unless specified explicitly.

Wetlands can be restored in different types of landscapes resulting in the high convergence in costs. For instance, the 1.500.000 cost per acre can be attributed to a project where wetlands were restored on a site where bed rock had to be lowered. King and Bohlen (1994) compared wetland restoration in different landscapes and show that agricultural conversion of wetlands is the cheapest with an average of a $1.000 per acre and a maximum of $20.800 per acre when both hydrology and plants had to be fully restored. It has to be noted that these cost estimates are in 1994 dollars and the costs have not been converted to Net Present Value (NPV). If for wetland sites only the hydrological storage function would be restored and no vegetation would be planted removal of tile drains is likely to be sufficient in the LSRB. King and Bohlen (1994) show that tile drain removal is cheapest of all methods to convert an agricultural land to a wetland and involves the lowest uncertainty in costs. Lastly, for agricultural conversion no preconstruction or post construction costs are involved and wetland restoration costs fully constituted of actual construction cost (King & Bohlen, 1994). Thus, wetland restoration on agricultural land is likely to be on the absolute lowest end of the cost ranges presented by previous research. This finding is supported by previous wetland restoration projects in the Red Devil Basin (North Dakota State Water Commission, 2003; Alice Lake Wetland Restoration project, 2003).

Table 9: Literature overview of costs of wetland restoration. In most cases land acquisition is not included.

Reference	$/acre	Comments
Argonne National Laboratory (unknown)	3.500 -80.000	
King & Bohlen (1994)	5 – 1.500.000 50 - >2.000	*For all types of wetland restoration* *For agricultural conversions*
Gutrich & Hitzhusen (2004)	14.000 - 58.000 11.000 - 71.000	*In Ohio* *In Colorado*
Minnesota board of Water and Soil Resources (1992)	95 – 3.000	*In Minnesota*
Shultz (2003)	1.500 2.750	*For previously drained wetlands* *Wetlands with outlet control devices*
Alice Lake Wetland Restoration Project (2003)	265	*North Dakota*
North Dakota State Water Commission (2003)	155	*For agricultural conversion in Red Devils Basin*
Guinon (1989)	1.626 – 240.000	*In 1995 US dollars*
MDA (2012)	10.000 (for total project)	*Minimum cost for wetland restoration project in Minnesota*

However, the costs for land acquisition in an agricultural basin have not been taken into account in the cost estimates presented so far. Land acquisition costs can be a considerable part of the total costs of wetland restoration, in one example being roughly 20% of total project costs (Shultz, 2003). Difficulties in land acquisition, because of conflict with a property owner over purchase or use of a particular property and adequate compensation, can result in time delays and consequently additional costs. This is often a large part of budget overruns in wetland restoration projects (Wellmann, 2003). If wetland restoration will be part of the management strategy the MPCA and possibly other organizations will buy up farmland to convert to wetland in the GBERB. This is likely to be farmland that is currently least productive. $5.000 per acre is seen as a reasonable lower estimate for the current price of land in Minnesota (CSSIR, 2013). However, as was pointed out by a local farmer the price paid for the land is likely not to be equal to the value of the least productive land but more likely to be equal to the costs involved to acquire and move to a different higher productive parcel of agricultural land (CSSIR, 2013). Thus estimates of the value of the least productive land or the average price of agricultural land might be too low to actually capture land acquisition costs. A previous restoration project in North-western Minnesota resulted in land owners selling their land for approximately 10% higher values than the appraised values (Shultz, 2003; HDR Engineering, 2003).

But it is unlikely that land will be purchased on a large scale given the fact that the MPCA is not a land acquiring agency. More likely the MPCA will strive for cooperation with land owners. From 1996-1999 cooperation between land owners and agencies was constructed through land rental in the Devils Lake Basin in North Dakota. However, renting values demanded by farmers and other land owners were far above market values and were 153 percent higher than average reported county values and 88 percent higher than the maximum reported county value (Shultz, 2003). Thus cooperation between land owners and the MPCA through rental of land bears the risk of unexpectedly large land rental rates demanded by farmers.

A different possibility for cooperation between land owners and agencies in land management could be achieved by incorporation of tax cuts for beneficial land management (CSSIR, 2013). This would also shift some of the burden of the cost from the MPCA towards farmers. Currently, farmers adjacent to a stream combined pay taxes for management of the stream. If wetland restoration would result in tax cuts for farmers adjacent to that stream this would create an incentive for land management by farmers and would possible result in shared costs of wetland restoration between all farmers benefiting from the tax cuts. However, possible issues that have to be overcome are the distribution of the costs imposed on the farmer owning the land where the wetland will be restored, the risk of free-riders and problems in regulations involving streams crossing state or county boundaries. This is especially true for the southern part of the Blue Earth River Basin which is partly located in Iowa. However, if these issues can be overcome tax cuts could be a promising strategy since it creates an incentive for voluntary land management by farmers and would partly compensate for the costs imposed on them.

Although not much research exist on the differences in cost between small and large wetland the large fixed costs associated with every wetland restoration results in economies of scale (King & Bohlen, 1994). For each 10% increase in wetland size a 3.5% decrease in costs per acre is achieved. However, this is especially true for non-agricultural conversion to wetland since fixed costs are highest for these categories (King & Bohlen, 1994). Thus the requirement for larger sized wetland to be restored could actually result in relatively cheaper wetland implementation due to economies of scale. Moreover, it might result in less conflict with adjustments made to pumping stations even reducing the costs further. The requirement for wetland restoration close to water bodies is not likely to influence cost.

Lastly, the above data and discussion can be used for a first estimate of wetland restoration cost. However, cost per acre is likely to be an unsuited criterion for comparison between different management options. Here it is suggested that comparison can best be based on cost for a typical size of a management option given a certain target reduction in stream flow and/or erosion of near-channel sources. Current ongoing research must then first determine the link between the individual management options and reduction in sediment delivery to the streams.

5.3 Ecosystem services

Wetlands are capable of performing many processes simultaneously and therefore they provide a suite of values to humans (Mitsch & Gosselink, 2000). The value of an ecosystem to humans is known as ecosystem services. The concept of ecosystem services, the goods and benefits provided by the environment, is becoming well established in research and is finding its way through policy to practice. Most research has focused on the positive impacts of wetland restoration but negative impacts of wetland restoration have received far less attention. Buckley and Crone (2008) acknowledged the possibility of negative off-site impacts associated with wetlands restoration to nearby landowners including increased abundance of pest species like mosquitoes and invasions of non-native weeds. In addition, Davenport et al. (2010) stated that the prospects of land use restrictions or hydrologic changes that may impact economic activities like forestry, agriculture or recreation based tourism may be deflating to local landowners and industries. To prevent disappointments after wetland restoration it is therefore important to inform stakeholders on both positive and negative impacts.

Although possible negative impacts of wetland restoration exist wetlands provide a host of other ecosystem services at multiple ecological scales (Mitsch & Gosselink 2000). These services are central to human health and societal well-being and encompass nutrient cycling, food and fiber, flood mitigation, water filtration, erosion control, aesthetics, outdoor recreation, (Millennium Ecosystem Assessment, 2005) and carbon storage (Euliss et al., 2006). The functions of wetlands are disproportionate to their area. Although wetlands cover <3% of the globe, they contribute up to 40% of global annual renewable ecosystem services (Zedler & Kercher, 2005). Of these, providing water of high quality ranks highest. Schuyt & Brander (75) suggested that the global annual value of wetlands is $70 billion, with an average annual value of $3000/ha/year and a median annual value of $150/ha/year. The 10 functions with the highest values (U.S. dollars per ha per year) include recreation ($492), flood control and storm buffering ($464), recreational fishing ($374), water filtering ($288), biodiversity support ($214), habitat nursery ($201), recreational hunting ($123), water supply ($45), materials ($45), and fuel wood ($14). Although above and other research has already proven the value of wetlands they are considered by many to be of little or no value, or even at times to be of negative value. This lack of awareness of the value of conserved wetlands and their subsequent low priority in the decision-making process has resulted in the destruction or substantial modification of wetlands, causing an unrecognized social cost (Kerry Turner et al., 2000). Wetland restoration could possible off-set this cost and in the US wetlands are currently the most regulated ecosystem and current policies state that no net loss of wetland may occur.

However optimizing for one ecosystem services often goes at the expense of another (Mitsch & Gosselink, 2000). But their might also be links between different ecosystem services given the requirements set by the local stakeholders. Although, this means that ecosystem services will not be optimized there might be changes for sub-optimization given the requirements set. Four ecosystem services are considered in this study, namely biodiversity conservation, nutrient and sediment cycling, recreation and climate regulation. An example of possible linkages is given by a study by Deckles et al. (2002) shows that for wetland restoration in the Prairie pothole region all criteria relating to water storage, nutrient removal and biodiversity and habitat are improved.

. Wetland restoration in the US is often implemented from an ecological perspective and has especially focused on habitat conservation for the water fowl (f.i. Ducks unlimited, 2013(b)). Boylan and Maclean (1997) have estimated that 46% of the US' endangered species are wetland-dependent suggesting that the continued loss or degradation of wetlands will likely have a negative impact on the nation's biodiversity but that wetland restoration will likely benefit endangered species. Thus, the above research suggests that wetland restoration might be important from a habitat conservation and biodiversity conservation perspective.

However there are also some critical remarks to be made. Wetlands support high productivity of plants but not always high plant diversity. Animal populations in wetlands are often more diverse (Zedler & Kercher, 2005). More importantly, wetland restoration is not likely to have similar effects on ecology as natural wetlands. From a biodiversity perspective, ongoing wetland protection policies may not be working because restored or created wetlands are often very different from natural wetlands (Whigham, 1999). Galatowitsch and van der Valk (1996) found that many species guilds were different between natural and restored wetlands and concluded that the seed bank is an important source of colonizers. Fewer species occurred in the seed bank of restored wetlands. A study in Wisconsin using a multi-variate approach found that revegetation occurred in the restored wetlands, but that the dominant genus in the restored wetlands differed from the dominant species in natural reference sites (Ashworth, 1997). A study in Iowa found that for all zones studied, except one, restored wetlands had fewer species than natural wetlands (Zedler, 1998). So although wetland restoration will have a positive effect on biodiversity and habitat conservation it should not be expected that restored wetlands behave similarly to natural wetland sites. However, the advantage in the LSRB is that wetlands will be restored in places where previously natural wetlands existed. Thus, likely having favorable conditions compared to sites were no wetland has been in place before.

The difference in species community and sometimes the failure of wetland restoration projects is attributed to the failure to recognize that wetlands are part of a larger landscape (Zedler, 1998; Zelder & Kercher, 2005). Especially, distance between wetlands is crucial for biodiversity (Semlitsch & Bodie, 1998). To establish connections between wetland sites and species populations existing small wetlands are of most importance for biodiversity of wetland flora and fauna. In the study by Semlitsch and Bodie wetlands smaller than 0.2 ha are considered to be small wetlands. In a study on wetland restoration in Minnesota size and spatial isolation of restored wetlands were found to be important predictors of species richness. The results of this study indicated that restored wetlands are a valuable habitat for at least a subset of the amphibian fauna of this region (Lehtinen & Galatowitsch, 2001). A study by Moler & Franz (1987) suggests that large wetlands may have a less diverse species population.

Thus, maximization of species richness through establishment of many small wetlands is in direct conflict with the interest of farmers that prefer larger wetland sizes that have a minimum impact on plowing practices. Thus, there seems to be a conflict between the requirements set by farmers on wetland restoration and biodiversity conservation. Possible there is also linkage between the requirement for wetland restoration close to water bodies and biodiversity conservation. These water bodies might currently be important corridors for animal species and wetland restoration close to these corridors might increase the area for wildlife habitat and strategically placed wetlands might reduce the distance between two currently unconnected or far away corridors. However, no research could be fined on this subject. More importantly, previous research has shown that is important to be clear on expectations of effects on biodiversity conservation from wetland conservation. Wetland restoration is likely to have differing effects on biodiversity conservation compared to natural wetlands.

Another important ecosystem service of wetland restoration is the beneficial effects wetlands have on water quality. This research so far has focused on water quantity management through wetland restoration from the perspective that lower flows will likely reduce erosion rates and turbidity in the stream from near-channel sources. However, 15-40% of the sediment is originated from the agricultural fields in the uplands. The biggest advantage of wetland restoration compared to other management options is that it influences sediment and nutrient delivery to the stream both directly and indirectly. It indirectly influences turbidity levels through stream flow and wetlands are also capable of storing sediment and nutrients in the wetland and filter the water directly impacting turbidity levels and water quality. In an agricultural landscape nutrient filtering, besides sediment reduction, is of much importance given the high use of artificial fertilizer and the consequently high concentration of nitrogen and phosphorus in runoff.

Wetlands are well known for their ability to remove sediments, nutrients, and other contaminants from water (Kadlec & Knight, 1996). This even has lead to the concept of 'nitrogen-farming' introduced by Hey et al. (2005). Nitrogen farming is the restoration of wetlands for the specific purpose of removing nitrates from agricultural and urban runoff. Nitrogen farming on a massive scale in the Mississippi River basin could be a successful management option to abate the hypoxic zone in the Gulf of Mexico. On the scale of individual sites, research suggests that even narrow bands of vegetation (as little as 4 m) immediately adjacent to streams can remove up to 85% to 90% of $NO_{x,}$ N, P, and sediments carried in runoff (Evans et al., 1996; Lenhart et al., 2010). Restoring wetlands to improve the quality of water that flows through a watershed requires a landscape approach, e.g., finding sites that can intercept a significant fraction of a watersheds' nutrient-rich runoff (Crumpton, 2001). Wetland protection policies may also be inadequate to preserve and restore ecological processes such as nutrient cycling because they mostly focus on individual wetlands and ignore the fact that wetlands are integral parts of landscape (Whigham, 1999). Examples of studies at the watershed level conclude that 1% to 5% of the total watershed would be needed to cleanse waters of the Des Plaines River in Illinois and up to 15% for the Great Lakes basin in Michigan, USA (Mitsch & Gosselink, 2000).

Although a landscape is required from the perspectives of water quantity, water quality management and biodiversity the resulting preferred locations for wetland restoration are likely to be highly similar for water quantity and quality management but could be highly divergent for biodiversity conservation. Thus, the fact that all three ecosystem services require a landscape approach this does not necessarily result in linkages between these ecosystem services. The requirements set for larger sized wetlands do neither result in direct conflict or linkages with sediment and nutrient filtering. This is highly dependent on the drainage area of the wetlands and the farming practices used on the adjacent farmlands. Lastly, from nutrient cycling wetland are preferably located close to the stream and this creates a linkage to the requirement set by local stakeholders that prefer wetland sites close to water bodies. Moreover, it also creates a linkage with the preferred locations from a water quantity management perspective.

Although wetland restoration is likely to increase water quality and to enhance wildlife habitat and species richness a trade-off exist between the two. It is often assumed that nutrient removal is highest where species richness is low; that is, wetlands cannot be both species rich and excel at nutrient removal because high nutrient loadings allow a few aggressive plants to displace many of the natives. The assumption of a trade-off is largely untested (Zedler & Kercher, 2005). For vascular plants diversity is especially high in wetlands that do not receive much surface water runoff (Bedford et al., 1999). Many species can coexist where nutrients are in short supply, total productivity is low, canopies are short, light penetrates through the canopy, and no species has a strong competitive advantage. Such wetlands are confined to landscape positions where the purest groundwater moves to the surface which is clearly not the case in the agricultural dominated GBERB. Thus the beneficial effects of nutrient and sediment removal by wetland restoration might negatively impact species diversity in the GBERB. However, Herr-Turoff and Zedler (2005) provide contrasting results showing that a single species dominated wetland did not remove more nitrogen compared to a wetland with a diverse prairie assemblage. Wetland restoration in an agricultural landscape that is efficiently located for nutrient removal thus might be in direct conflict with biodiversity conservation because less species are likely to be supported by wetlands.

Another effect of wetland restoration might be the increase in recreation opportunities. Given the location of the GBERB and the proximity to the nearest urban center wetland restoration in these areas does not result in recreation opportunities for urban citizens. Also already a lot of streams and water bodies exist within the GBERB so wetland restoration only has a minimal effect on water related recreation. However, better water quality as a result of wetland restoration might provide better or new opportunities for recreation of local inhabitants. Possibly more interesting for the GBERB is the effect wetland restoration might have on hunting possibilities. A study on the Cache River Basin, Illinois, on stakeholder perspectives revealed that wetlands were of importance to them because of fishing and hunting opportunities (Davenport et al., 2010). A recent book on agricultural land-sharing and wetland conservation practices in the GBERB emphasizes several times the beneficial effects of wetland conservation on hunting possibilities and several farmers that voluntarily implemented wetlands on their lands mention hunting as an important reason for doing so (Shepard & Westmoreland, 2010). Increased possibilities for recreation of farmers might be an interesting way to spur voluntary interest in wetland restoration compared to several other management options considered in the GBERB. To maximize recreation opportunities through hunting wetland restoration should focus on larger sized wetlands providing a habitat for species or restore wetlands that link to wildlife corridors. This is likely to be in the proximity of the streams and links nicely to the fact that farmers are already more likely to accept wetland restoration if wetland are of larger size and in close proximity to water bodies. Thus there are linkages between the requirements set by farmers and recreation opportunities following from wetland restoration. There can also be a linkage identified between lower turbidity levels and improved opportunities for water recreation.

The last ecosystem service considered in this study is climate regulation due to uptake of green house gases. This is a relatively new and emerging body of research. However, this could be an ecosystem service of considerable interest from the view point of climate change but also for individual farmers, if they in the future would receive monetary subsidies for green house gas storage. Existing wetlands must be preserved to the greatest extent possible to prevent further releases of terrestrial C to the atmosphere, but it is less clear what role created and restored wetlands will play in climate regulation. The effect of wetland restoration on climate regulation is uncertain since wetlands are sources of CH_4 and C sequestration rates appear to vary across wetland types (Zedler & Kercher, 2005). In a study of Canadian peat lands, Roulet (2000) indicates that, once the "global warming potential" of CH_4 is factored in, many peat lands are neither sinks nor sources of greenhouse gases. CH_4 is a greenhouse gas 23 times stronger than CO_2 and can thus easily offset any CO_2 uptake by wetlands. Mitra et al. (2005) confirm the results of Roulet (2000) finding that although wetlands store vast quantities of C in vegetation and especially in their soils, they also contribute more than 10% of the annual global emissions of the greenhouse gas methane (CH4) and can even be a significant source of CO_2 under some conditions. Interactions involving the physical conditions in the soil, microbial processes, and vegetation characteristics seem to be largely responsible for determining whether wetlands will act as a net source or sink of green house gases (Smith et al., 2003). Thus, the actual effect of wetland restoration in the LSRB on climate regulation remains unclear.

Smith et al. (2003) conclude, based on a review of several experimental studies, that greenhouse gas exchange between soil and atmosphere increases exponentially with temperature but decreases with soil saturation. Currently temperatures in Minnesota are relatively low and soil saturation in wetlands is expected to be high. However, climate might change these current conditions. Johnson et al. (2005) assessed the effects of climate change in Minnesota on wetlands. Projections from their ensemble model suggest that Minnesota will experience a 3^{o}C rise in temperature statewide by 2069. They estimated that a 20% increase in precipitation is needed to compensate for a 3^{o}C rise in temperature to maintain water balance in wetlands in southern Minnesota, and thus in the GBERB. However, increases in moisture may be only one-third of what is needed to offset ET. Thus, climate change will lead to increases in temperature that will exponentially increase green house gas exchange from wetland areas. Moreover, increases in precipitation will not be enough to compensate for the additional EVT. As a consequence, soil saturation will decrease resulting in an additional increase in green house gas exchange from wetland areas. Thus, where under current climate conditions wetlands in the GBERB are likely to be net sinks of green house gas predicted climate change will reduce this sink or even turn wetlands into a net source of green house gases creating an additional positive feedback loop in the climate. This might an important consideration in the decision making process for wetland restoration.

This study remains unaware off previous research on the relationships between climate regulation, other ecosystem services, wetland size and proximity to the stream. Therefore, it is not possible to determine possible linkages and trade-offs of climate regulation following wetland restoration.

In conclusion, this research has identified the views of farmers regarding wetland restoration. In general, farmers are critical towards wetland restoration but are more likely to be open to wetland restoration if incentives are created through tax cuts for wetland restoration or through the provision of multiple ecosystem services. This research therefore identified linkages and trade-offs between wetland restoration. Essential to this are the requirements stated by local farmers that wetland restoration is preferred in the proximity of water bodies and wetland restoration should focus on restoring larger but fewer wetlands. Focusing on larger wetlands sites might result in cost savings. Furthermore, larger wetland sites are beneficial for recreation through hunting opportunities. Wetlands located closer to water bodies might be beneficial for water quality enhancement and might link to wildlife corridors enhancing the wildlife habitat and biodiversity conservation. Wetlands in general have a positive effect on water quantity management, water quality management, biodiversity conservation and recreation opportunities. Furthermore, wetland restoration might result in climate regulation under current conditions, but climate change might dampen or even reverse this effect in the future. For biodiversity conservation larger wetlands might be less beneficial compared to smaller wetlands and restored wetlands are not likely to support a similar species guild as natural wetlands. Moreover, wetlands in agricultural landscape are likely to receive large quantities of nutrients favoring only a few productive species and thus limiting the effect of wetland restoration on biodiversity conservation. These findings are likely to be transferrable to other agricultural basins in the US since farmers are, for plowing, likely to favor fewer larger wetland sites. In wetter, tile-drained, agricultural landscapes it is also expected that farmers prefer wetland restoration closer to water bodies.

6. Discussion

6.1 Surface runoff in HSPF

This research has shown that for the LSRB wetland restoration reduces annual stream flow, reduces stream flow in the months with the highest turbidity levels, reduces peak stream flow volume for individual days and for the entire event and reduces runoff from the land to the stream. Furthermore, this research has concluded that the hypothesis that wetland restoration in far upstream late contributing is most beneficial has to be rejected. To assess the effect of wetland restoration and location of wetland restoration a calibrated HSPF model was used. Here some discussion points will be brought up on the functioning of HSPF, both related to the model structure as well as to the decision made during calibration, and possible implications for the results of this research.

Currently, surface runoff from cropland is only a very minor part of contributions to the stream flow. Even during the largest storm events surface runoff is small compared to base flow and interflow. Although the watershed is extensively tile drained soils in the watershed still have low to very low infiltration. Thus, the absence of surface runoff is not in line with natural conditions. This is partly due to the structure of HSPF. HSPF is capable of both treating saturation excess, through storage ratios, and infiltration excess, through the infiltration parameter. During storm events in the GBERB surface runoff is still occurring due to infiltration excess (CSSIR, 2013). If precipitation falls on the land surface HSPF first stores water in the groundwater and lower zone. If moisture supply exceeds this storage capacity water goes into the upper zone. Excess water then first flows into the interflow. If, even then, not all water is assigned to storage compartment water finally goes to surface storage and runoff. Thus, only during the most extreme precipitation events will surface runoff occur.

Surface runoff could be increased by greatly reducing the infiltration value. However, in an extensively drained agricultural landscape, like the LSRB, tile drainage is an important part of the runoff from the land. In HSPF tile drainage is represented by interflow. However, reducing infiltration would reduce the amount of water transported by tile drains. This leads to a trade-off since low values of infiltration would result in more realistic surface runoff patterns during more storm events but this would result in lower interflow and thus less water transported by tile drains during days with lower precipitation. Higher values of interflow, as is currently the case, therefore seem reasonable since this would result in the best results on most days. However, both the set-up of HSPF and the relatively higher value of infiltration have an important implication. Namely, that croplands are capable of storing large amounts of water. This can be clearly observed in the graphs on lower zone storage capacity and in the appendix (appendix, paragraph A.3). Since wetlands most important function is to store water this limits the effect wetlands will have on stream flow since only so much more water can eventually be stored. In order to still have considerably more storage on wetland surfaces rather large increases in the infiltration parameter but especially in the parameter governing upper zone storage are necessary. The initial parameter set for wetlands in the LSRB was largely similar to croplands and thus did not result in reasonable effects on stream flow. This research has tested other parameter sets (appendix, paragraph A.2) and selected the parameter set suggested by Butcher. However, although this parameter might better represent wetland functioning than the initial parameter set it has to be stressed here that this parameter set is not calibrated for the LSRB and it is therefore impossible to tell whether it under- or overestimates wetland functioning. The Butcher parameter set results in the almost complete absence of direct runoff from wetland surfaces, both interflow and surface runoff. This is in contrast to a previous study by Sun et al. (2002) showing that saturated wetlands behave similarly as uplands. Wetlands in the LSRB, represented in HSPF, are never fully saturated prior to a storm event and are thus always capable of storing most to all of the water.

This might be related to the fact that wetlands in HSPF only treat precipitation and that the current set-up of the calibration model for the LSRB does not allow surface, interflow or groundwater outflow to be intercepted by wetlands. Thus the moisture supply to the wetland is likely to be far lower compared to the real world. If wetlands in HSPF were also capable of treating runoff they are likely to be more often fully saturated prior to a storm event and this might reduce the impact wetlands have on the stream flow and especially on peak stream flow. At the other hand, under unsaturated conditions wetlands would be capable of both treating precipitation and runoff from the land segment. This would result in a larger impact on stream flow and peak stream flow in particular from wetland restoration.

Since the extents of both misrepresentations of wetland functioning in HSPF are unknown it is impossible to assess whether the effect of wetland restoration on stream flow is over- or underestimated in this study. It is likely the case that the effect of wetland restoration on annual and monthly runoff is underestimated since wetlands would be capable of treating far larger quantities of water on average if runoff is also intercepted. But the effect on peak stream flow is likely overestimated since wetlands will have less storage capacity prior to a storm event resulting in more direct runoff from wetlands. In a research by Babbar-Sebens et al. (2013) it has been stated that wetland restoration is especially effective in tile-drained landscapes since wetlands constructed to intercept tiles can serve as storage basins for agricultural runoff, leading reductions in peak stream flow. However, wetlands intercepting tile drains are likely to treat large quantities of water and will thus be more saturated prior to storm events questioning the effect these wetlands will have on peak stream flow. Future research might test the statement by Babbar-Sebens et al. (2013) through field measurements, laboratory experiments and/or modeling efforts.

6.2 Representing tile drainage in HSPF

Regarding HSPF another point will be stressed. Tile drains in agricultural landscapes in HSPF are represented by interflow. Both interflow and tile drainage result in water flowing through the shallow sub-surface but processes are likely to be different. The interflow zone in HSPF has the opportunity to store water whereas tile drains have no storage capacity and transport water directly and quickly to the nearest water body. Minimizing storage and optimizing quick outflow as a consequence of tile drainage is governed by the Interflow recession constant (IRC). Values of this parameter can be varied between 0-1 and low values represent small storage and quick outflow whereas high values result in longer storage and slower outflow patterns. Thus, low values are best to represent tile drainage. In the current calibrated model monthly values for IRC vary between 0.65 and 0.8. For future models and for discussion on the LSRB calibrated model it would interesting to look into this since it likely results in unreasonable large storage in the interflow zone in tile drained landscapes.

Even more so, there is a conflict between representation of tile drainage and shallow subsurface flow through interflow. Representing one goes at the cost of representing the other. At the plot scale both interflow and flow through tile drains have a delay in response time compared to surface runoff. If water flows through the shallow subsurface as interflow there is friction between the flow and soil and possibilities for storage occur. However, once water enters a tile drain friction is reduced, no storage is possible and water is routed in the most direct way to the nearest water body. Thus, at the larger scale of the watershed or sub-watershed the response time of surface runoff and runoff through tile drains is likely similar. It might even be so that runoff through tile drains is faster since surface runoff, can be stored in depressions, follows topography instead of being routed directly to the stream and experiences friction from surrounding vegetation and the soil. Of course, the entire goal of tile drainage is to route water as quickly of the field as possible.

It is suggested here the process of flow through tile drains on a larger scale is similar to surface runoff and could be viewed as 'sub-surface surface runoff'. From a process mind of view it would then be better to represent tile drains not through interflow but as part of the surface flow. Thus instead of increasing the amount of water flowing through interflow in a agricultural tiled landscape it is suggested that the amount of water should instead be decreased for better process representation. Clearly, stating that tile drainage is located in the interflow zone and should thus be represented in this zone does not hold since the physical representation of other zones, upper and lower zone in HSPF, remains unclear.

More importantly, simulation of management options influences processes in HSPF and in the real world. For HSPF to be used under non-stationary conditions it is therefore important to not only be able to represent current stream flow, in essence be calibrated, but also to capture processes essential to the system. Management options will then influence these processes and a process based model provides better ground for linking management options to parameter changes. The above is hoping to start up a broader discussion not only on the question 'where' in the landscape water flows but also on 'how' water flows through the landscape because in order to use models under conditions of non-stationarity, like climate change, land-use change and management, process representations are most important. An influential paper by Milly et al. (2008) has stated that stationarity is dead and this implicates that the use and the relevance of models, like HSPF, will change.

The question remains whether enough alterations are possible within the current model structure and therefore a discussion between HSPF users and developers on these topics is necessary. Currently, no such platform exists. Even more worrisome in this regard is the fact that the most up-to-date version of the manual is already dated back to 2001 (Bicknell et al., 2001). It would be interesting to create an addition to this manual, often done through tecnotes, with a broader discussion on how to use HSPF for studies under non-stationary conditions and especially climate change. Currently, the available information provided on HSPF comes short of addressing these issues, but HSPF is already used for climate change impact studies.

6.3 Wetland location

Besides looking at the effect of wetland restoration this research has also focused on the effect of wetland location on stream flow. Loucks (1989) argued that a greater number of wetlands in the lower reaches are preferable over few larger wetlands downstream. In a slightly different setting Anderson and Kean (1994) argued that wetlands in late contributing areas are most efficient in stream flow reduction. However, the results from this study do not suggest that wetland restoration in areas further upstream that contribute later to stream flow are better locations for wetland restoration. This is in line with results from a modeling study by Ogawa and Male (1986) that suggested the opposite: the usefulness of wetlands in decreasing flooding increases with the distance the wetland is downstream. The results presented in this study suggest that this is at least the case for most peak stream flow events situated during the late spring and summer months. Thus, a previous simulation study in combination with this study have both rejected this hypothesis. This study is unaware of simulation studies proving the opposite. Furthermore, Jones and Winterstein (2000) mention in their research the possible difference between wetland restoration upstream and downstream related to distance to the gage. However, after careful study of their report it turns out that differences between basins are only assessed based on localized differences. This makes an explicit distinction between the evaporative and time delay effect of wetland restoration and thus between differences in location of wetland restoration attributed to localized differences and distance to the gage. It is strongly suggest that future research comparing location of wetland restoration explicitly mentions the effect tested to avoid indistinctness whether the differences between basins are based on distance to a certain point or on localized differences between the basins and thus on what is actually compared and tested.

6.4 Linking stakeholder perspectives, costs and ecosystem services

Besides looking at the hydrological effect of wetland restoration this research has focused on a broader framework for wetland restoration. Wetlands provide a wide set of ecosystem services. Mitsch and Gosselink (2000) have argued that optimizing for one ecosystem services often comes at a cost of other services. Consequentially, research has often focused on trade-offs between ecosystem services (f. i. between nutrient removal and biodiversity (Bedford et al., 1999)). From a management context, especially when local stakeholders are of essence in the land management, focusing on linkages between ecosystem services might be of more interest. This research has highlighted some of these possible linkages for the GBERB.

Identification of linkages is facilitated by a clear guiding criterion for wetland restoration based on the interest of local stakeholders and the main goal of wetland restoration. In many agricultural basins farm-friendly solutions might the most important guiding criteria. However, if these types of studies are implemented one must be wary for double counting of benefits which can be avoided by explicit allocation of benefits between functions (Kerry Turner et al., 2000) or by rank ordering (De Groot et al., 2002).

6.5 Generalization of results

Some of the results presented in this study are only applicable to the LSRB. This is especially true for the differences in stream flow related to localized conditions. However, although on the extreme end the LSRB is one of many agricultural basins suffering with erosion problems and elevated turbidity levels in the US. This is especially true for the Corn Belt region. Due to changing land management over the past decades and due to anticipated climate change the attribution of near-channel sources might already be large or gain importance in a multitude of basins. Therefore, parts of the results of this study are likely to be applicable to more agricultural basins in the US. This is true in general for the results on effect of wetland restoration, costs of wetland restoration, stakeholder views and requirements set by stakeholders, and the linkages identified between requirements set, costs of wetland restoration and the provision of ecosystem services. Moreover, the strategy and methodology developed and used in this study can be easily implemented in a wider range of basins. The finding that upstream wetland restoration does not result in more beneficial effects for stream flow management is supported by the only previous simulation studies on this subject (Ogawa & Male, 1986). Therefore, it is believed that the finding that distance to the gage is not a defining factor for selection of optimal wetland restoration location is likely to be the rule rather than the exception.

7. Conclusion

This report has been concerned with wetland restoration in an agricultural basin. More importantly, it has assessed the effect of wetland restoration on hydrology, the optimal location for wetland restoration, stakeholder views restoration plus requirements set by stakeholders regarding wetland restoration and the influence of these requirements on costs and linkages between ecosystem services. In order to do so this research has performed a case-study on the Le Sueur River Basin in Minnesota, USA. This basin has undergone major changes in the past century from a prairie area with a high abundance of wetlands to a typical corn belt region agricultural basin with intensively row-cropped agricultural of soybeans and corn, a high density of artificial drainage and an almost complete disappearance of wetland area. Combined this has resulted in one of the most erosive basin in the USA with very high turbidity levels being a primary contributor of sediment to the Minnesota River and further downstream the Mississippi River and the hypoxic zone in the Gulf of Mexico. These high erosion rates and turbidity levels are driven by erosion of near-channel sources, especially bluffs, following increases in stream flow due to climate change and land management. Wetland restoration is seen as a promising strategy in this and other agricultural basin to deal with these increases in stream flow and abate erosion. For the hydrologic component of this research the software HSPF was used.

This research has shown that wetland restoration results in decreases in annual stream flow and decreases in peak stream flow. Moreover, for most months of the year wetland restoration results in a decrease in stream flow but later during the year wetland restoration might result in increases in stream flow during lower flow periods. Reductions in stream flow following wetland restoration are largest in the months May-June that currently have the most elevated turbidity levels. Furthermore, wetland restoration results in a decrease in the flow volume exceeding the discharge for channel forming flows likely resulting in a reduction in erosion of near-channel sources. The decreases in stream flow are driven by decreases in water delivered to the stream flow. Wetland restoration results in more water being stored in long term storage and less water being stored in short term storage compartments compared to croplands. Consequently, wetland restoration reduces direct runoff, surface runoff and most importantly interflow, and slightly increases base flow. Over the year wetland restoration results in less water being routed from the land into the nearest stream and the decrease in water delivered to the stream can almost completely be explained by increases in evapo(transpi)ration. From these result it can indeed be concluded that wetland restoration is a promising strategy for reducing stream flow and erosion abatement.

Furthermore, this research has compared far and close upstream wetland restoration to determine the optimal location for wetland restoration. In order to do so this research made a distinction between differences in stream flow response between these two regions based on localized differences and based on distance to the gage. Differences in stream flow response between the two regions were very small and differed depending on the metric studied and the scale of aggregation. Based on the differences in stream flow response none of the two locations was more suited for wetland restoration. The finding that the entire upstream area in the upland zone is suitable for wetland restoration increases the potential area for wetland restoration greatly in the LSRB.

This research also tested the hypothesis that far upstream wetland restoration in late contributing areas has a more beneficial response on peak stream flow (Anderson and Kean, 1994; Loucks 1989). The differences in stream flow response are according to these hypotheses governed by distance to the gage. However, the results in this study show that close upstream wetland restoration in early contributing areas results in the largest reductions in peak stream flow. This is in line with the a previous simulation study on this topic (Ogawa & Male, 1986) and therefore the hypothesis that the optimal location for wetland restoration is in far upstream late contributing areas has to be rejected. Thus, for the Le Sueur River Basin the entire area upstream of the upper gages is suitable for wetland restoration and no optimal location can be defined based on the results of this research. However, if stream flow management in this basin would only be concerned with a sub-set of the stream flow response, f.i. only the highest peak stream flow events, the results of this study provide a basis for selection of the optimal location.

Wetland restoration in an agricultural basin involves cooperation with local farmers in order to come up with farm-friendly solutions. This is especially true in the Corn Belt region in the Midwestern US since this is one of the most productive agricultural areas in the world being essential for global food production. Therefore this research identified farmers opinions towards wetland restoration and possible requirements imposed. Farmers are critical towards wetland restoration since it has a large impact on the land. If wetlands are restored farmers prefer a few large wetlands over many smaller wetlands and prefer wetlands in closer proximity to the stream. This research has identified several linkages between the requirements set by farmers on wetland restoration and costs of wetland restoration and provision of ecosystem services. Example include, but do not cover all linkages, that larger wetlands are likely to result in cost savings due to economies of scale and wetland restoration closer to the stream is likely to result in more nutrient filtering of for instance nitrogen and phosphorus. Wetland restoration is preferable over other management because it influences turbidity directly through capturing of sediment and indirectly through reduction of stream flow. Focusing on linkages between ecosystem services provided by wetland restoration and requirements set by farmers offers the opportunity to identify additional benefits of wetland restoration and highlights more incentives for wetland restoration in an agricultural basin. More importantly, this research shows that requirements on wetland restoration from an agricultural perspective do not necessarily have to be in conflict with costs consideration and provision of other ecosystem services. And this research has highlighted that, given the requirements set by local stakeholder, several ecosystem services can simultaneously be optimized without necessarily resulting in large trade-offs.

However, a possible downside of wetland restoration is the possible effect on climate regulation. Wetlands regulate the climate due to carbon sequestration. However, based on local conditions sequestration of carbon can be offset by methane releases. In Minnesota wetlands are likely to be a net sink of green house gases under current climate conditions. However, anticipated climatic changes will lower the carbon uptake of wetlands reducing either the magnitude of the sink or reversing wetlands into a net source. This might create an additional positive feedback loop in the climate system.

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Appendix – Wetland functioning in HSPF

This appendix provides a background to the methodology used in the main report. It starts off with a small sensitivity analysis resulting in an assessment of the wetland parameter values and the governing functions in HSPF for a pervious land segment. Then a small HSPF model is build in Excel in order to test the effect of different parameters sets and changes. This together resulted in changes to the calibrated model and underpins part of the reasoning of the methodology used in this paper.

A.1 Sensitivity Analysis

This research started with a small sensitivity analysis of the HSPF LSRB model to understand the model structure, functioning and the effect of individual parameters. It would also provide a first insight in how wetland restoration effects stream flow. This section will first discuss the methodology of the sensitivity analysis. 12 parameters are selected for the sensitivity analysis (table A.1). The selection and range of the parameters is constituted from a combination of three sources: 1) a previous scenario report for the entire Minnesota River Basin using HSPF that provides information on parameter changes for different scenarios (Tetratech, 2009), 2) The HSPF manual version 12 and corresponding tecnotes that provide information on the physical meaning of the parameters and possible values from previous studies (Bicknell et al., 2001) and 3) expert advice from MPCA to relate the parameters to local conditions (Chuck Regan, personal communication(a)). It has to be noted that the sensitivity analysis does not adhere to the normal rigorous standards for a sensitivity analysis. However, since the interest was solely in getting a quick better understanding of HSPF the analysis performed is sufficient. To better understand the equations and the description below it is advisable to first look at table A.1 depicting the possible, likely and current calibrated values for the individual parameters. The following equation is used to generate two scenarios for each individual parameter:

Eq. (A.1)
$$NV_i = \left(\frac{MaxV_i - MinV_i}{4}\right) +/- CV_i$$

For the following criterion:

Eq. (A. 2)
$$MinPV_i \leq NV_i \leq MaxPV_i$$

In this formula subscript i denotes the parameter, NV is the new value used in the sensitivity analysis, MaxV and MinV are the maximum and minimum value of the likely range and CV is the calibrated value. Thus each new value is the current value +/- 25% of the range of the likely value of the parameter. This creates for each parameter a +25 and a -25 scenario and thus a total of 24 new parameter values and model runs. After the NV_i is calculated it is checked whether the criterion is met. MinPv and MaxPV represent the minimum and maximum possible values of each parameter. If the criterion is not met the value is set to the closest possible value of that parameter. The current maximum calibrated value for AGWETP is above the possible range of that parameter, as defined by the HSPF manual. For all land-uses AGWETP is equal to zero except for wetlands. It is decided to increase the value for AGWETP in wetlands and ignore the criterion for this analysis for this particular parameter. It remains unclear why this value is chosen outside the possible range but it does not result in model errors. Parameter values are adjusted for all land-uses in all 94 sub-watersheds. If parameter values are input on a monthly basis in HSPF only values that differ from zero are adjusted. The output of all 24 models was analyzed for the months March-October since during the winter period no stream flow is observed and gages are not in place.

Flow Duration Curves (FDC) are generated to assess the sensitivity of the model to parameter changes. This study analyzes stream flow changes based on yearly FDC, as suggested by Vogel and Fennessey (1994). Since this study is concerned with peak stream flow only the upper 20%

of the FDC will be used. Based on the daily stream flow output in cfs FDC are plotted for each individual year using Excel. Note that leap years do not require a different ranking formula since February is not part of the period under consideration. The area under the curve of the 20% highest flows is calculated. Sensitivity of peak stream flow to changes in parameter values is then calculated using the following formula:

Eq. (A. 3) $$Sensitivity_{i,j,g} = \frac{Area\ under\ FDC\ (top\ 20\%)\ for\ NV_{i,j,g}}{Area\ under\ FDC\ (top\ 20\%)\ for\ CV_{i,j,g}}$$

Where j represents the year, g represents the eight gage location and NV and CV respectively represent the model with the new value and the original calibrated model. The larger the deviation from 1 the more sensitive the model is to changes in a specific parameter.

Table A.1: Selected parameters for sensitivity analysis and the possible likely and current calibrated values for all land-uses.

parameter	explanation	possible range	likely range	calibrated value
AGWRC	Active Groundwater Recession Constant	0.85 - 0.99	0.92-0.99	0.925-0.95
AGWETP	Active Groundwater Evapotranspiration Potential	0 - 0.2	0 - 0.05	0 - 0.25
BASETP	Potential Evapotranspiration from Baseflow	0 - 0.2	0 - 0.05	0.005 - 0.01
CEPSC	Interception storage capacity	0.01 - 0.4	0.03 - 0.2	0.06
INFILT	Index to infiltration capacity	0.001 - 0.5	0.01 - 0.25	variable
INTFW	Interflow inflow	1 -10	1 - 3	3 - 4.5
IRC	Interflow recession	0.3 - 0.85	0.5 - 0.7	0.85
KVARY	nonlinearity of AGWRC	1 - 5	1- 3	1.5 - 3
LZETP	Lower zone Evapotranspiration potential	0.1 - 0.9	0.2 - 0.7	variable
LZSN	Lower zone nominal storage	2 - 15	3 - 8	3 - 5
NSUR	Roughness of overland flow plane	0.1 -1	0.15 - 0.35	variable
UZSN	Upper zone storage nominal	0.05 - 2	0.1 - 1	variable

A.2 Findings from the sensitivity analysis

19 of the 24 simulations resulted in sensitivity values different than 1. Table A.2 provides an overview of the changes for all parameters at all eight gage locations. If the changes in the ratio were less than 0.02 the value is replaced by a star-sign. Only the parameters of interception storage capacity (CEPSC) and roughness of overland flow plane (NSUR) were always equal to 1. Meaning that no response in stream flow is observed based on the changes opposed in the parameters in this analysis. For active groundwater evapotranspiration potential (AGWETP) the stream flow was insensitive when the value was increased but showed some minor (<0.02) changes when the value was decreased. This is rather surprising given the fact that the value is only adjusted for wetlands, which currently comprise a very small portion of the basin, and remains zero for all other land-uses. The parameters showing the largest responses are upper zone storage nominal (UZSN), lower zone storage nominal (LZSN) and lower zone evapo-transpiration potential (LZETP). Most parameters had a very small stream flow response often within the range of .98-1.02. For the parameters Index to infiltration capacity (INFILT) and Interflow recession (IRC) the direction of change matters. The KVARY parameter only resulted in significant changes at 3 of the 8 gages. All other parameters either had an effect or no effect at all gages except for the active groundwater recession constant (AGWRC) were changes are observed at all gages except for the Maple. The stream flow response did not differ between the different zones. This could be related to the large area contributing to the upland gages effectively dominating the effects in other zones. On average stream flow at the Maple River was slightly more sensitive compared to the other rivers.

Table A.2: Results of sensitivity analysis for 12 parameters at all eight gage locations. The - represent the model runs for which the NV<CV whereas the + represent the opposite case. The values presented are the ratios. * sign means that the outcome lies between 0.98 and 1.02 and is therefore not represented. The larger the deviation from 1 the more sensitive the stream flow responds to changes in the specific parameter.

	Upland gages								Knick-zone gages						outlet gauge	
	Maple River		*Cobb river*		*Le Sueur River*		*Beauford Ditch*		*Maple River*		*Cobb River*		*Le Sueur River*		*Le Sueur River*	
	-	+	-	+	-	+	-	+	-	+	-	+	-	+	-	+
AGWRC	*	.98	1.03	.94	1.02	.98	1.03	.95	*	.98	1.02	.95	1.02	.97	1.02	.97
AGWETP	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
BASETP	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
CEPSC	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
INFILT	1.12	*	1.12	*	1.15	*	1.11	*	1.12	*	1.11	*	1.15	*	1.13	*
INTFW	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
IRC	*	.98	*	.98	*	.98	*	.98	*	.98	*	.98	*	.98	*	.98
KVARY	*	*	.98	*	.98	*	*	*	*	*	.98	*	*	*	*	*
LZETP	1.15	.88	1.16	.87	1.14	.89	1.17	.87	1.15	.88	1.15	.88	1.14	.89	1.15	.88
LZSN	1.02	.95	1.03	.95	1.04	.96	1.02	.95	1.02	.94	1.04	.96	1.04	.96	1.03	.95
NSUR	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
UZSN	1.17	.87	1.18	.86	1.15	.87	1.19	.85	1.17	0.87	1.17	.87	1.14	.87	1.16	.87

The changes in stream flow per year at the outlet gage are depicted in figure A.1. More parameters increase peak stream flow than lower it and also the increases in stream flow are larger than the decreases. Graphs at other locations showed more or less similar general trends. This figure underlines the previous observation made that for most parameters the model is rather insensitive. For INFILT direction of change matters and the ratio responds far more strongly to a decrease in the parameter value. This can be explained by the functioning of INFILT. A decrease in INFILT leads to lower infiltration rate and an increase in surface runoff to the stream. This is a rather direct effect. For an increase in INFILT more water can potentially infiltrate. However, if lower zone storage is saturated an increase in INFILT will not lead to more infiltration, although it has more potential to infiltrate, and thus will not lead to a decrease in runoff. Therefore, an increase in infiltration has a more indirect effect on runoff possibly explaining the difference in the magnitude of the stream flow response. What can be uniquely observed from this graph are the large year to year differences in stream flow response. For decreases in peak stream flow the years with a larger reduction correspond to the wetter years in the dataset and the years with the smallest reductions correspond to the drier years in the dataset. It is not surprising that the stream flow responds less strongly to changing parameter values in drier years. For example, upper zone storage is likely to be less saturated during dry years and increasing the storage capacity thus likely results in smaller effects on stream flow.

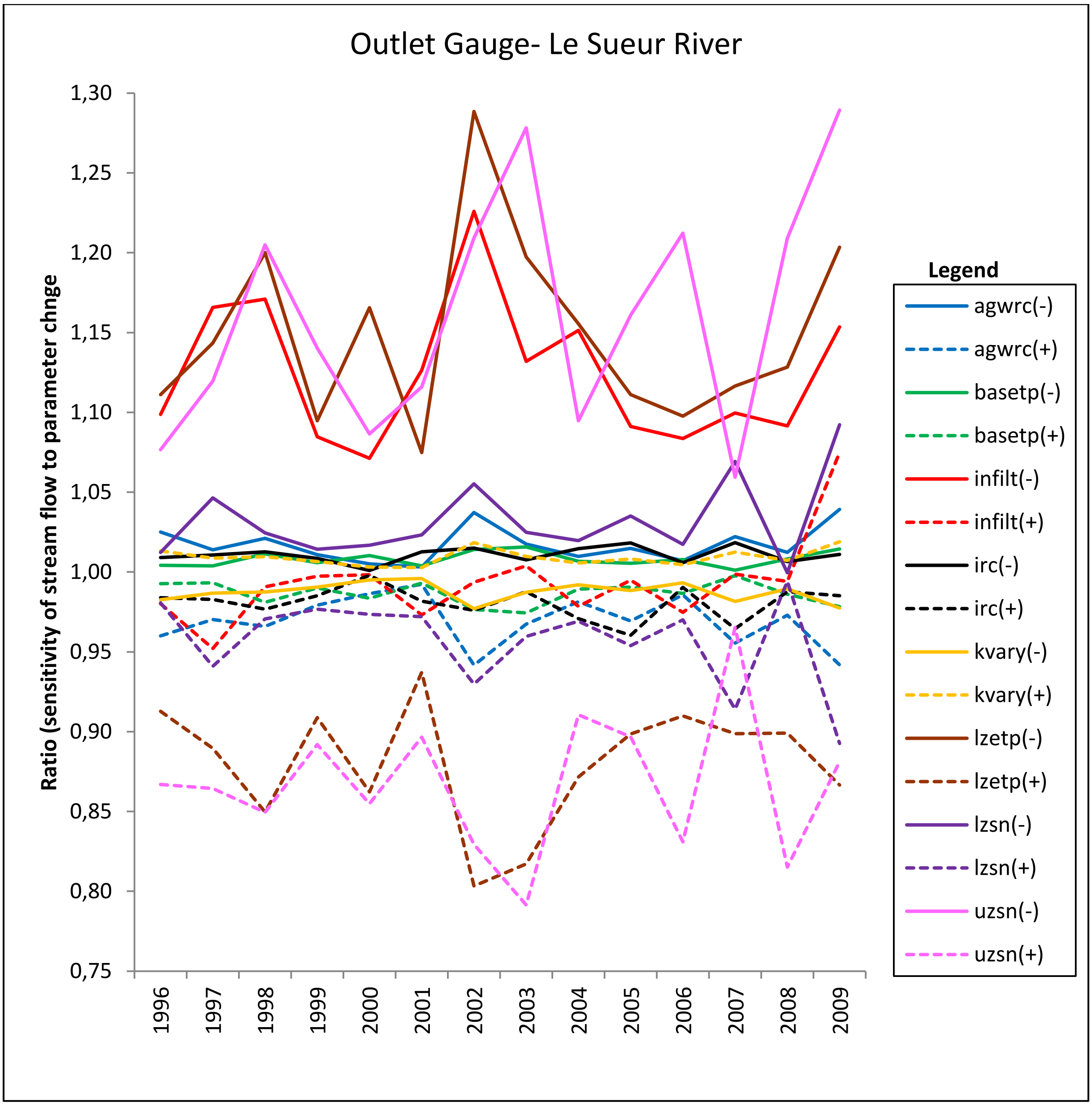

Figure A.1: Sensitivity of the highest stream flow events at the outlet gage to parameter changes for all years in the study period. The solid lines indicate decreases in parameter values and the dashed lines indicate increases in parameter values. Only 16 out of 24 parameter runs are depicted. Parameters that are not depicted had a value equal to 1, i.e. stream flow response being completely insensitive to parameter changes.

For wetland restoration it is important to note that parameters effecting storage capacity, LZSN and UZSN, are both parameters towards which stream flow is more sensitive. This indicates that wetlands will possibly have a significant effect on stream flow. Wetlands are furthermore likely to increase infiltration but, as can be concluded from this analysis, the model is relatively insensitive to an increase in infiltration. However, it might be that the increase in infiltration value is larger for wetlands than the current changes used in this analysis. Furthermore, a combination of an increase in INFILT and LZSN might result in higher infiltration rates but this analysis did not test the effect of multiple parameter adjustments for one simulation. It can be observed from figure A.1 that in 2009 an increase in infiltration led to an increase in peak stream flow. 2009 was by far the driest year in the dataset and therefore increased infiltration capacity likely resulted in relatively high base flow possibly increasing the peak stream flow during a storm event. Lastly it has to be noted that although sensitivity responses are in general small a minor change in the area under the top 20% highest flows can still result in large absolute stream flow reduction.

A.2 Preliminary wetland simulation

After the sensitivity analysis a couple of preliminary runs were performed for the entire basin and for Beauford Ditch separately. In these runs acreages of wetland were increasingly reduced resulting in a decrease in conventional and/or conservational cropland. Surprisingly, these results showed often increases in stream flow after wetland restoration. Although wetlands might occasionally increase stream flow due to an increase in base flow during a heavy precipitation event an on average increase in peak stream flow for a large portion of the events contradicts the general conception on wetland functioning. Currently, the acreages of wetland are very small in the LSRB. If wetland parameters would be 'poorly' calibrated the error related to this would be insignificantly small since wetlands only occupy a small percentage of the watersheds. Table A.3 provides an overview of some of the calibrated values for parameters related to wetlands and croplands in the current LSRB model. For comparison parameter values used in two other wetland simulation projects are also provided. Since some differences occur between model set-up, description of the land-uses and the individual basins also the difference between wetland and cropland parameters is provided. The first model is a preliminary version of the current HSPF model developed by J. Butcher, a scientist with great experience in modeling wetlands throughout his career (C. Regan, personal communication). Although this model has not been calibrated to stream flow records the values should indicate reasonable values for wetland parameterization. The second model used for comparison is from a research by Jones and Winterstein (2000) that looked at the effect of wetland restoration in the Lake Heron Basin in Minnesota. Since this study was focused on wetland restoration it would be expected that a lot of effort has gone into correct estimation of wetland parameter values. Furthermore, the Lake Heron Basin is located in the proximity of the LSRB and has roughly similar characteristics.

Table A.3: comparison between wetland and cropland parameter values for three wetland models

	Current LSRB model			**Butcher model**			**Jones & Winterstein**		
	cropland	wetland	Δ	Cropland	wetland	Δ	Cropland	wetland	Δ
LZETP	.15-.87	.2-.5	.05-.37	.15-.82	.2-.5	.05-.32	.2-.82	.2-.6	0-.22
INFILT	.065-.08	.095-.116	.03-.036	.024	.5	.476	.025-.06	.4	.44-.475
LZSN	3-4	3-4	0	5.4	3.7	1.7	4.2	3.0	1.2
UZSN	.05-1.5	.15	.1-1.35	.18-.65	2.5	1.85-2.32	.12-.14	2.5	2.36-2.38

Table A.3 shows that the Butcher and Jones & Winterstein model have similar parameter sets for wetlands and croplands, but that the current LSRB model has a very different set of parameters assigned to wetlands. A second observation is that the parameterization in the current LSRB model for wetlands and croplands is remarkably similar. This is not the case in the other two models. For LZETP values for both cropland and wetlands in all models are reasonably similar. However, for the other three parameters the differences between at the one hand the current HSPF model and at the other hand the other two models is huge. For INFILT cropland values are reasonable similar but the value for INFILT detached to wetlands is roughly four times as large in the latter two models. For both storage parameters the differences are also remarkably large between the current LSRB model and the two other models. For UZSN this even leads to the result that during most of the year wetlands have smaller upper zone storage capacity than cropland in the LSRB model. In the other two models capacity to store water in the upper zone is always larger for wetlands than for croplands. Less storage capacity in the upper zone could also explain the increases in stream flow observed in the preliminary analysis. From a comparison of parameter values between wetlands and croplands in the current HSPF model it must be concluded that wetlands hardly infiltrate (more) water, have only slightly more storage capacity in the lower zone and less storage capacity in the upper zone during several months of the year. In essence, the parameterization for wetlands in the current model does not represent the hydrologic functioning of wetlands. This observation is underlined by the discrepancy in the parameterization of wetlands in the current LSRB model compared to the two other models. Thus, the observation that stream flow increases after wetland parameterization is likely due to the current incorrect calibration for wetlands in the LSRB HSPF model.

From the manual of HSPF (Bicknell et al., 2001) and the corresponding equations it is difficult to clearly get a grasp on the meaning of parameter changes because most parameters have multiple effects and interact with one another. This makes an evaluation of the parameters tedious. A simplified excel model of the HSPF LSRB model for Beauford Ditch is built to get a better insight in the effects of parameter changes and to evaluate different sets of wetland parameters. The next sections will discuss the methodology behind the excel model. It has to be noted that an updated version of the LSRB was made available by RESPEC and MPCA after comparison of the wetland and cropland parameters. In this version parameters for wetland were set equal to the values suggested by Butcher. The HSPF model in Excel will be used to assess whether these changes result in more storage on wetland surfaces.

A.3 A simplified HSPF model in Excel

Figure A.2 provides a schematic overview of the different pathways precipitation on a pervious land surface can flow through to exit the system, according to the model structure of HSPF. Water can exit the system by flowing into the adjacent stream, EVT or infiltrate into deep inactive groundwater. Water that reaches the ground surface either directly infiltrates and goes to the lower zone or groundwater zones, or does not directly infiltrate. From the water that stays on the surface a part infiltrates into the upper zone, with no direct outflow to the stream, and the rest of the water is separated in interflow, shallow subsurface storage with outflow to the stream, and surface storage. Water enters the stream trough direct runoff via interflow outflow and surface outflow/runoff or through groundwater outflow. Water in the upper zone or lower zone cannot directly reach the stream. Water in the upper zone can leave this storage through EV or through percolation to lower zones. Water in the lower zone can only exit the system, and this zone, through ET. In figure 8 of the main report these fluxes and the parameters governing these fluxes are depicted as well.

From the HSPF manual the equations governing the fluxes into the different storage compartments were extracted and the formulas were written down in an Excel spreadsheet. Values for parameters and variables were looked up in the UCI-file. If values were location specific the values for Beauford Ditch were selected. Several assumptions had to be made in order to simulate the HSPF model and functioning of parameters. First, EVT from the storages is constant throughout the year and no EVT occurred from the groundwater outflow. Second, from the output generated by HSPF for the base model a low and large value for upper and lower zone storage was selected. These values were used as initial starting values for upper zone storage (UZS) and lower zone storage (LZS) and where used to simulate dry or wet antecedent moisture conditions prior to the 14 days period of simulation. The values are 0.2 and 1.5 for UZS and 0.5 and 4.5 for LZS respectively. Third, it is assumed that the ground is not frozen. Since we are only interest in stream flow in the months March till October this is a reasonable assumption.

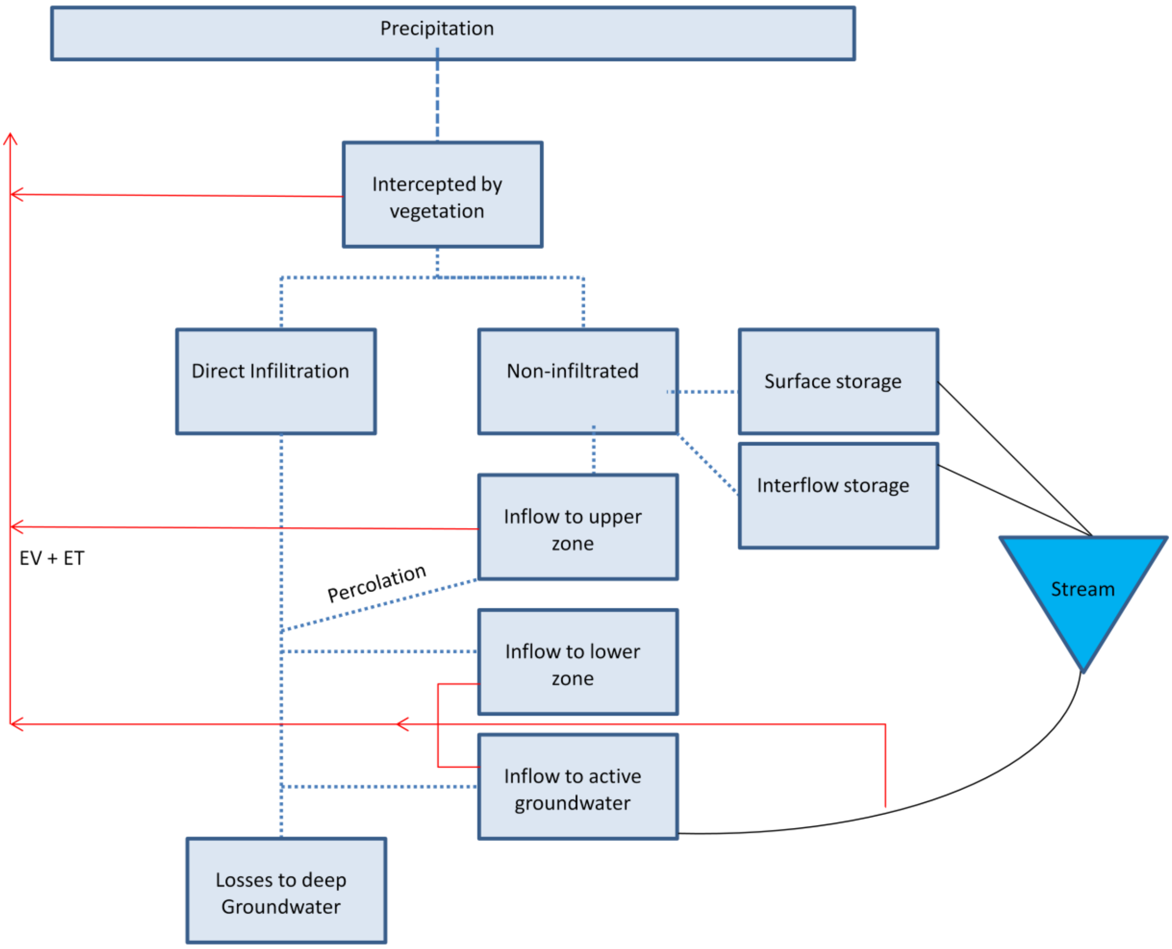

Figure A.2: Schematic overview of different pathways for precipitation to exit the system on a pervious land segment.

As a starting point parameter values were set to the current values in the calibrated model (Table A.3). Parameter values were then consistently increased until they were equal to the values suggested by Butcher. UZSN is a parameter that changes per month for wetlands. For most of the period between March and November the value is relatively low (0.05-0.3) but later on in the year the value increases (1.2-2.1). Since during the longest period of record the value is low the median value of 0.15 is chosen as a starting value. Output was generated for fourteen days. On the ninth day of these fourteen days a precipitation event is simulated of respectively 1, 2 or 4 inches. These values were chosen based on the distribution of the rainfall reaching the ground surface over the entire record of study. 1 inch of precipitation resembles an event that happens more than 10 times a year, 2 inches of precipitation represents an event happening only a few days (<5) per year and 4 inches of precipitation resemble an extreme storm event only happening once every couple of years. For comparison, the maximum moisture supply to the surface on a single day in the entire record is 5.44 inches. A following step was to take into account different antecedent wetness conditions. First, a distinction was made between dry and wet conditions at the start of the modeling period influencing the initial values of LZS and UZS. Then for the eight days preceding the precipitation event it could either be dry, zero precipitation on all eight preceding days, or wet, 0.8 inches of precipitation on all eight preceding days. This resulted in four combinations of antecedent conditions for all three rainfall events (dry-dry, dry-wet, wet-dry and wet-wet). For all runs four graphs were generated and compared depicting the inflow into the different storage compartments, the total volume of storage in the compartment, the total outflow from each storage and the cumulative outflow and moisture supply over the fourteen day record. These comparisons showed that the Butcher calibration for wetlands produced reasonable results leading to more water storage and less direct runoff.

A.4 A comparison between wetlands and croplands

Here a comparison will be presented for the differences in inflow, storage and outflow from the different compartments based on the Excel-HSPF model. The comparison will focus on at the one hand on a set of conventional cropland parameters and on the other hand on a set of parameters for wetlands based similar to the parameter set suggested by Butcher. The latter is also the parameter set assigned to wetlands in the updated LSRB HSPF model and should give insight in the possible effects of wetland restoration. Figure A.3 and A.4 show the results for the Excel-HSPF model for a four inch event under wet antecedent starting conditions and with rainfall of 0.8 inches on the preceding eight days. This is the highest simulated moisture supply with wet antecedent conditions. These figures clearly show diverging responses. First the upper zone inflow and storage is far greater for wetlands than for croplands. Second for croplands, on the day with the highest precipitation almost all moisture supply ends up as direct runoff. For a similar event on a wetland all water flows into the groundwater and upper zone. Since the lower zone is continuously full in both the cropland and the wetland no inflow is observed in this compartment. Consequently, on a cropland almost all moisture supply enters the stream mainly through interflow and especially surface runoff. For wetlands however only 4 out of 12 inches of total moisture supply goes to outflow and the rest is stored or leaves the system through EVT. Figures A.5 and A.6 show the outflow for the same 4 inch rainfall event on a cropland. However, the simulation now starts with dry prior antecedent moisture conditions. Figure A.5 and A.6 represent different simulations since the first has rainfall on the eight days preceding the larger rainfall event (dry-wet) whereas the latter only simulates rainfall on the ninth day (dry-dry). It is interesting to note that cropland can roughly store 4 out of 11 inches in the first case. In the second case almost no outflow occurs and the cropland is capable of storing all water. Both are surprising outcomes becomes it suggest that the current croplands can store most of the water from a extreme precipitation event only happening once every few years under dry antecedent moisture conditions. In the first wetland restoration can then only lead to slightly more storage and in the latter case wetland restoration is not likely to result in more storage. If croplands would actually be capable of storing such vast amounts of water during extreme precipitation events there would likely not be much turbidity issues in the GBERB. These results thus suggest that there might be issues with the current calibration for croplands as well likely resulting in an underestimation of the effect of wetland restoration on stream flow. However, given the fact that the cropland values have been calibrated it is not reasonable to also adjust those values. Moreover, it might be that the short simulation period and the assumptions on fixed EVT and the values for LZS and UZS have negatively influenced the outcome. Therefore, the model will be run with the current parameter set for croplands and with the wetland parameter set suggested by Butcher. First, several model runs have been performed at Beauford Ditch. These results will not be presented here since most of the outcomes and conclusions are in general similar to the results presented in paragraph 4.3 of the main report.

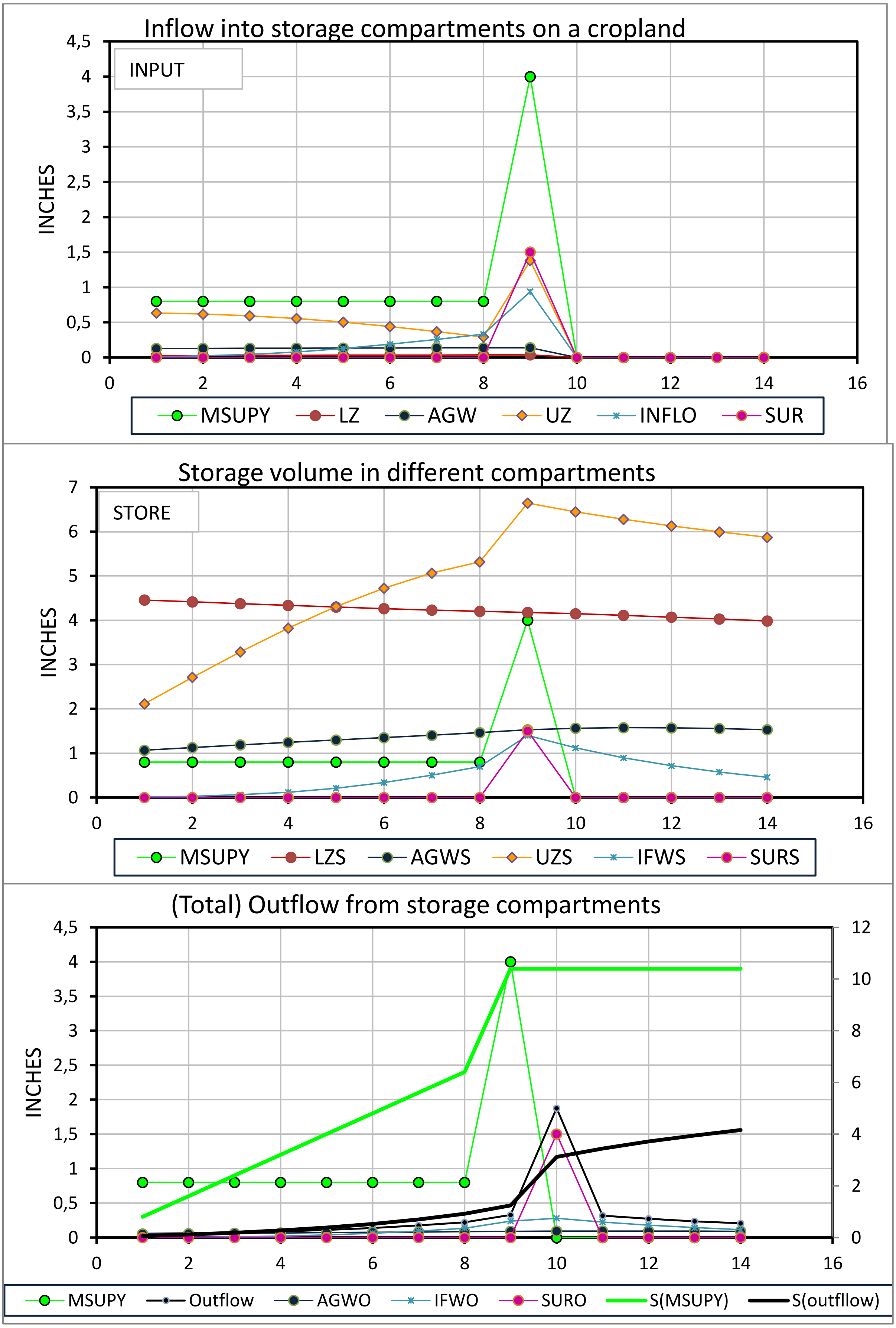

Figure A.3: Inflow, storage and outflow for different compartments in HSPF mimicked using a model built in Excel. The values are for wet antecedent conditions and a large rain storm for a wetland parameter set.

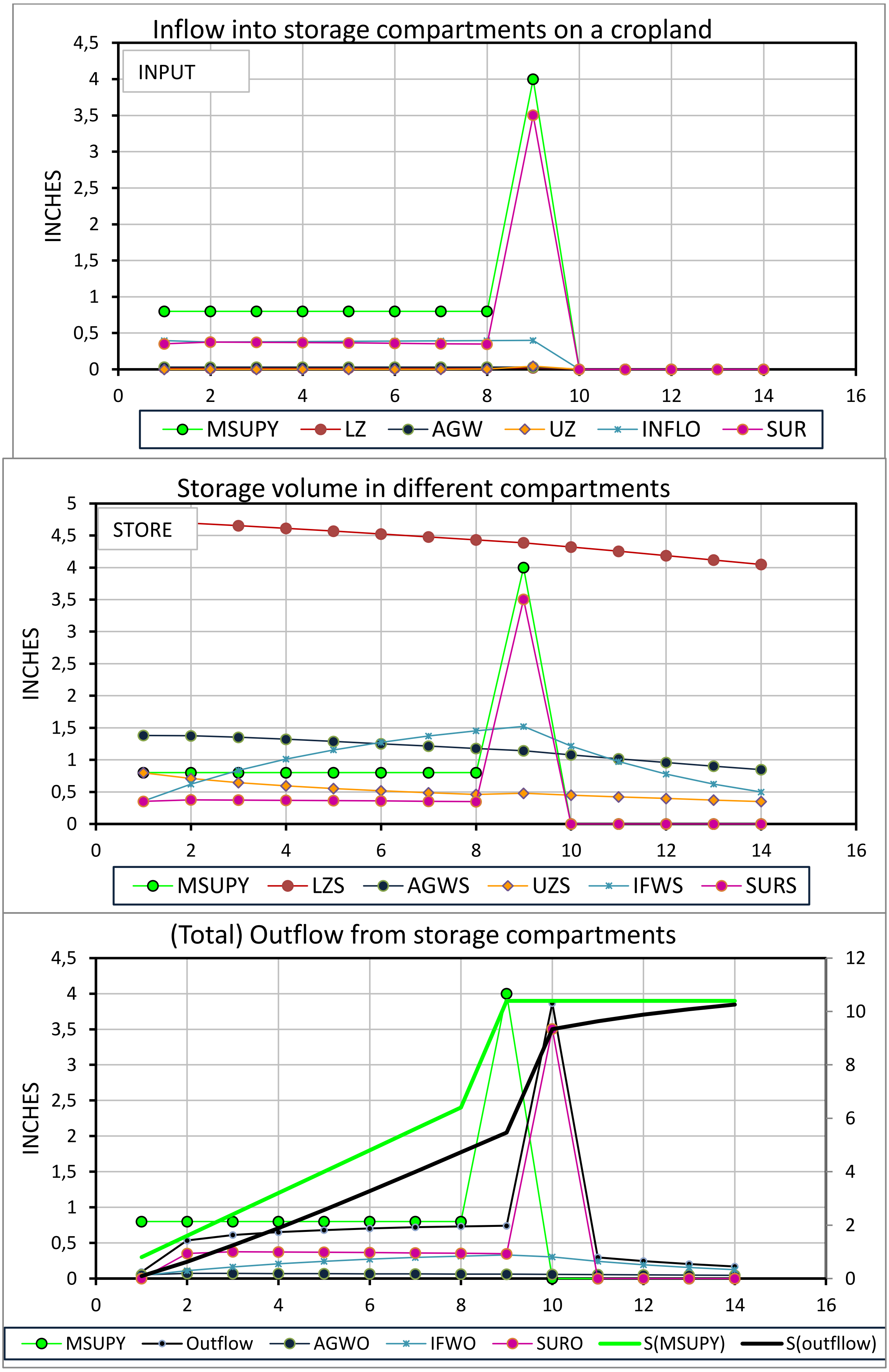

Figure A4: Inflow, storage and outflow for different compartments in HSPF mimicked using a model built in Excel. The values are for wet antecedent conditions and a large rain storm with a cropland parameter set.

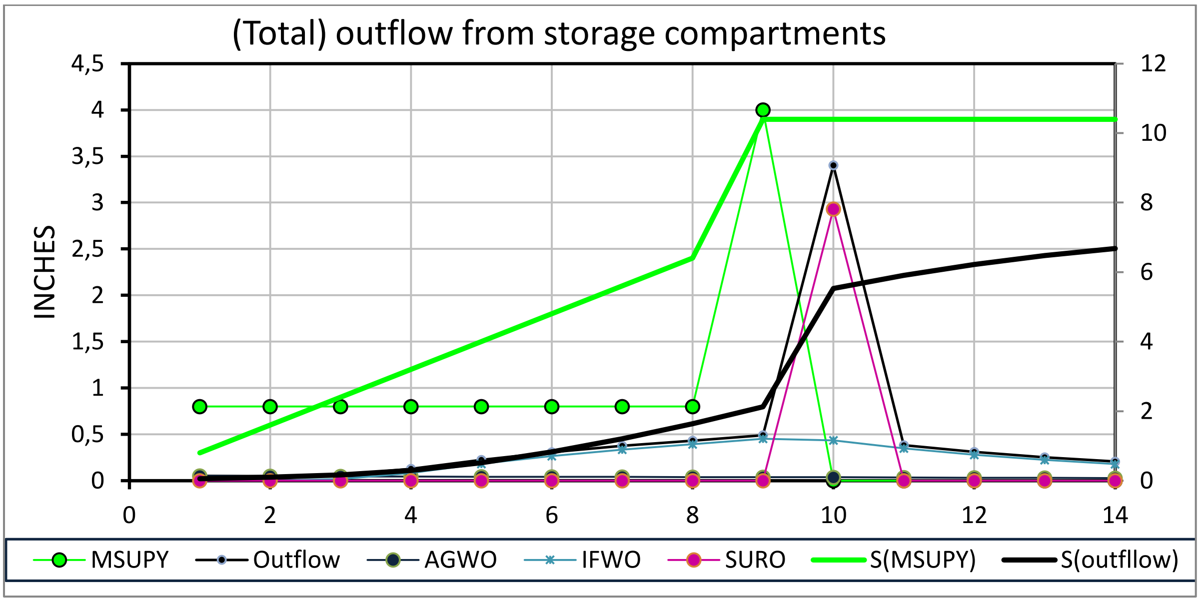

Figure A5: Total outflow from storage compartments from a cropland under dry starting antecedent conditions. Precipitation has fallen on the eight preceding days.

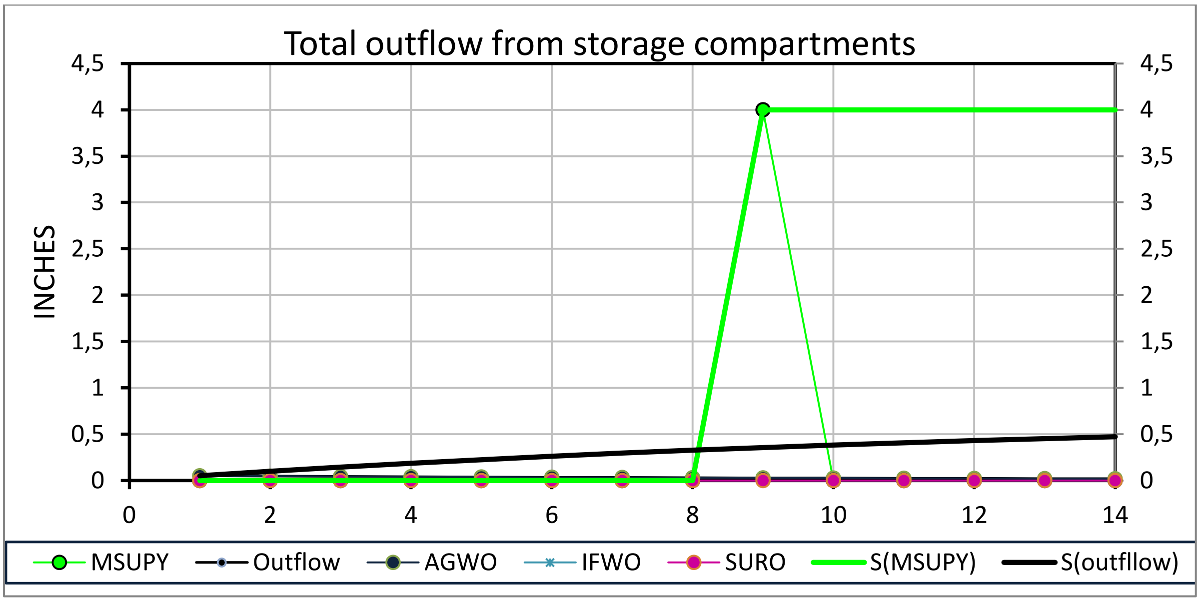

Figure A6: Total outflow from storage compartments from a cropland under dry starting antecedent conditions. No precipitation has fallen on the eight preceding days.