A Characterization of Soil Erosion in Cultivated Watersheds in Manitoba's Red River Valley using Sediment Budgeting, and its Implications for Managing Soil Erosion's Impacts

by

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Abstract

Soil erosion accelerated by agriculture reduces agricultural productivity and compromises the function of drainage infrastructure and downstream water quality. In Manitoba, the relationship between soil erosion and water quality is of particular concern, following measurable declines in Lake Winnipeg's water quality since the 1990s. Understanding the state of soil erosion, transportation, and deposition in the Red River Valley is of interest, due to the extensive cultivation of the river's watershed and its contribution to total flow (and by extension, sediment flux) into the lake.

Sediment budgets were drafted for two sub-watersheds of the well-studied Boyne-Morris and La Salle River watersheds, located in the Red River Valley, and coded 05OF024 and 05OG008 by the Water Survey of Canada (WSC), respectively. To characterize soil erosion, the sediment budgets used published National Agri-Environmental Health Analysis and Reporting Program (NAHARP) soil erosion risk estimates calculated with the SoilERI model. Sediment transportation was quantified using flow and total suspended solids (TSS) measurements made by the WSC and other organizations. Deposition within the sub-watersheds was quantified through measurements made in road-side ditches (a common sediment sink in the Red River watershed) and inferred through imbalances in the sediment budgets.

Rates of soil erosion, deposition within, and transportation out of the 05OF024 sub-watershed were an order of magnitude greater than in the 05OG008 sub-watershed due to differences in basin scale, but relative differences in their rates in each basin were the same. Rates of erosion were 1 order of magnitude greater than rates of deposition in road-side ditches and 3 orders of magnitude greater than transportation past the sub-watershed outlets. Differences between rates of soil erosion and deposition in road-side ditches were noted and attributed to unmeasured deposition in

cultivated fields. Rates of deposition in such settings were of the same order of magnitude but less than rates of erosion. Both sediment budgets quantified rates of road-side ditch dredging, which were 1 order of magnitude greater than rates of deposition in road-side ditches and directed soil back into cultivated fields. Rates of water, wind, and tillage erosion characterized by NAHARP erosion risk estimates were not mirrored by related, measured rates of deposition in roadside ditches in either watershed.

The stark differences between rates of soil erosion and rates of sediment transportation past the outlets of both sub-watersheds suggested the downstream impacts of eroded soil on water quality may be minimal at coarse temporal scales in watersheds of similar or greater size in the Red River Valley. Greater degrees of sediment delivery to road-side ditches suggested that sediment may have more meaningful impacts on the function of drainage infrastructure, especially at the same temporal scales in smaller watersheds in the region. Differences in estimated rates of water, wind, and tillage erosion and related rates of deposition in road-side ditches suggest the SoilERI model may not adequately characterize rates of soil erosion. This does not invalidate the SoilERI model, but highlights its limitations which should be considered when it is used to estimate rates of soil erosion in such settings.

Acknowledgements

A thesis is an artifact, representing a snapshot of one's continuing education. Its my aim that the acknowledgements that follow not only thank those that made writing this thesis possible, but provide context for the environment it was written in. A description of a study is not enough to base a reflection of on one's time as a Master's student in. I hope by outlining this context and structuring my acknowledgements as I have, this deficiency can be amended. My acknowledgements can be divided according to those that address: people that have shaped how I think, people whose work supported the completion of this thesis, and groups that funded the work this thesis is based in.

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AAFC: Agriculture and Agri-Food Canada

AEI: agri-environmental indicator

CMP: component table

DSWMA: Deerwood Soil and Water Management Association

LAT: landscape area table

LDT: landform definition table

LET: landform extent table

LST landform segmentation table

MLI: Manitoba Land Initiative

N: Nitrogen

NAHARP: National Agri-Environmental Health Analysis and Reporting Program

NSDB: National Soil Database

P: Phosphorus

PAT: polygon attribute table

RM: rural municipality

SDR: sediment delivery ratio

SLC: Soil Landscapes of Canada

SLT: soil layer table

SNT: soil name table

STCW: South Tobacco Creek Watershed

TSS: total suspended solids

WSC: Water Survey of Canada

1. Soil Erosion, its Impacts, and its Characterization

1.1. An Introduction to Soil Erosion and its Impacts

Erosion is a catch-all term describing the natural leveling of landforms in a landscape. It encapsulates detachment of mechanically weathered sediment from erodible sources by erosive processes, loss of sediment from erodible sources through entrainment and downslope transportation of previously detached sediment, and, finally, deposition of the same sediment in sediment sinks. Rates of erosion have the potential to be accelerated by human activities, such as agriculture. Erosion accelerated by agriculture primarily affects soil sediment sources, and is referred to as soil erosion (Weil and Brady, 2016).

Soil erosion is commonly sub-divided into water, wind, and tillage erosion; each comprised of processes of detachment and entrainment (loss), transportation, and deposition unique unto them. Water erosion is initiated by detachment, entrainment, and transportation of soil by rain-splash and subsequent diffusive runoff between rills in the soil surface. These first stages of water erosion are collectively referred to as inter-rill erosion. As runoff increases and concentrates, so to do rates of erosion, with channelized flow detaching and entraining additional soil from the soil surface as the transportation capacity of the flow increases. Such increases occur within rills and are referred to as rill erosion. As rills deepen and coalesce, gullies that cannot be infilled by typical tillage practices form. Such deepening is referred to as gully erosion (Weil and Brady, 2016). The processes that compromise wind erosion, as they are commonly described, are less dependant on topography. Wind erosion is initiated by gusts of sufficient strength to detach mechanically weathered soil aggregates from the soil surface. Entrained, sand-sized soil aggregates saltate across the soil surface as they are transported, detaching additional soil aggregates with every impact. Sufficiently light aggregates are entrained by each impact of the saltating sediment load, while heavier soil aggregates creep forward along the soil surface. These processes account for the bulk of sediment transported by wind, with suspension of clay- and silt-sized sediment rarely accounting for more that 15% of the total sediment load (Weil and Brady, 2016). The processes that comprise tillage erosion also lack the spatial specificity of water erosion. Loss, transportation, and deposition of soil by tillage is dictated by the time that the soil profile is in contact with the implement passing through it and is described in terms of soil movement parallel and perpendicular to the direction of the implement's movement. Such movement is formally described as forward and lateral tillage translocation, respectively (Li *et al*., 2007; Lobb and Kachanoski, 1999). Though often discussed independently, there is a degree of interaction between water, wind, and tillage erosion, with the ability of tillage to exacerbate water and wind erosion well documented (Weil and Brady, 2016).

Compared to the erosion of other sediment sources (e.g., streambanks), soil erosion is particularly worrisome. This is due to the agri-environmental services soils provide. The agri-environmental impacts of soil erosion can be divided into two categories: on- and off-site. On-site, soil erosion leads to the loss of fine-grained sediment and nutrients from soil sediment sources. The impacts of these losses are cascading, reducing soil's water retention, cation exchange capacity, and biologic activity, in turn reducing crop yields (Weil and Brady, 2016). In extreme cases, such yield reductions can lead to farm abandonment. Lesser yield reductions related to soil erosion have been slowed though time by advances in agronomy, but have not been offset, resulting in overall losses in crop value (Krauss and Allmaras, 1982). Off-site, eroded soil can bury otherwise productive soils and backfill flow-control infrastructure when deposited downslope, leading to further yield reductions and increased risk of flooding, respectively (Weil and Brady, 2016). Finer soil aggregates and the nutrients sorbed to them can be transported further off-site, increasing total suspended solid (TSS), phosphorus (P), and nitrogen (N) loads in water-bodies before they deposit. Such changes in water quality commonly result in increased algae bloom size and frequency, followed by reduced dissolved oxygen concentrations upon the bloom's death. This process can lead to fish kills and impacts the viability of commercial fisheries (Boyd, 2020). The potential of soil erosion to increase TSS and P loads is particularly worrisome, as such conditions favor proliferation algae blooms dominated by cyanobacteria (Scheffer *et al*., 1997). Several genera of cyanobacteria produce microcystin-class toxins, which are linked to liver and colorectal cancers in humans when ingested (Ueno *et al*., 1996; Zhou *et al*., 2002) and have been shown to bioaccumulate in commercially fished fish species (Wilson *et al*., 2008).

In Canada, there is a long-standing appreciation of the impacts of soil erosion. This has motivated its management through a scientific characterization of its state.

1.2. The Impacts of Soil Erosion in Canada

There is a strong historic appreciation of the on-site impacts of soil erosion in the Canadian prairie provinces of Alberta, Saskatchewan, Manitoba. There, in the arid region spanning the prairie provinces named the Pallister Triangle, scientific investigation of the state of soil erosion, its on-site impacts, and mitigation began as early as the 1920s. These studies followed a period of severe wind erosion in Alberta from 1917 to 1919, inclusive, that led to high rates of farm abandonment (Gray, 1967). Over the next decade, research conducted at experimental farms in Lethbridge, Alberta and Swift Current, Saskatchewan demonstrated the contribution of summer-fallowing to soil erodibility, especially when paired with intensive tillage sequences, thought to be needed for moisture conservation in otherwise arid landscapes (Gray, 1967). The then contemporary practices of strip cropping and shelterbelt planting to manage soil erosion were questioned and, in their place, residue management to conserve soil moisture and increase soil roughness combined with tillage sequences that emphasized the importance of timing and implement design were recommended (Gray, 1967). The adoption of these beneficial management practices was at first slow and sporadic, but increased following another period of severe wind erosion and farm abandonment throughout the Pallister Triangle in the early 1930s (Gray, 1967). Broader adoption of these practices led to an overall decrease in the frequency and severity of notable wind erosion events (Lobb *et al*., 2016), but not before a loss of crop value estimated to be between 20 billion and 30 billion dollars prior to 1971 (Lobb *et al.*, 2017). Though rates of soil erosion in Canada are lower in recent decades, the cumulative effects of past soil erosion and today's lower rates of soil erosion continue to hamper agricultural productivity, with an additional 40 billion to 60 billion dollars in crop value lost between 1971 and 2011 (Lobb *et al*., 2017).

In Southern Manitoba the on-site impacts of soil erosion are still appreciated but of less concern today than soil erosion's potential off-site impacts. These concerns have been fueled by visually conspicuous increases in algae bloom size and frequency in the south basin of Lake Winnipeg since the mid-1990s. Such observations are supported by increasing measurements of phytoplankton biomass and chlorophyll concentrations between 1999 and 2007, inclusive, (Page, 2011b) and similar changes in estimates of phytoplankton biomass since the 1950s, inferred from phytoplankton fossil assemblages (Kling, 1998). These changes have tracked with measured increases in N and P concentrations between 1999 and 2007 (Page, 2011a) and estimated increases in N and P concentrations inferred from the same fossil assemblages (Kling, 1998). The impacts of these changes to date have been limited, with commercial fishermen noting reduced catches due to algae clogging fishing nets and beach fouling when algal mats blow ashore (Watson *et al.*, 2011). The response of commercial fisheries to these changes have been complex but do not yet indicate their immediate collapse. Between 1999 and 2007, TSS measurements rarely rose more than 25 mg L^{-1} above their baseline; a federal guideline for assessing the impact of TSS loads on aquatic life (McCullough and Lévesque, 2011). TSS measurements above this guideline correlated with disturbance of the lake's shallow bed by wave action and increased sediment transfers from the Red River when flooded (McCullough and Lévesque, 2011). These conditions have contributed to increasing cyanobacteria populations. Such changes in phytoplanktonic communities have not changed microcystin-class toxin concentrations dramatically, which are low but do occasionally rise to levels that pose a hazard to human health (Kotak *et al*., 2011). Though the outcomes of these changes in the water quality, namely eutrophication, do not yet match those of other polluted lakes, such as Lake Erie (Lévesque and Wassenarr, 2011), changes in the abundance and composition of phytoplanktonic communities in response to contaminant loading have been regarded as early indicators of future declines in lake health. As such, Lake Winnipeg's declining health has been described as an emerging issue. The contribution of soil erosion to this issue is not yet fully understood but its study has begun in the Red River Valley (Koiter *et al.*, 2013). Such attempts to characterize soil erosion in the Red River Valley follow an understanding of the substantive contribution of Red River to the sediment load of Lake Winnipeg's south basin and widespread cultivation of the landscapes within the river's watershed. Further characterization of the state of soil erosion in the Red River Valley is required.

1.3. Characterizing Soil Erosion

Assessment of the on- and off-site impacts of soil erosion and planning their management follows an understanding of soil erosion's state. Quantifying rates of soil loss and deposition can be difficult, due to their spatially dispersed nature. Therefore, transportation, which is spatially constrainable by flow concentration, is often measured and treated as representative of the whole of the processes which comprise erosion in the basin defined by transportation's point of measurement. Rates of deposition and soil loss in the basin are successively larger than rates of transportation, a relationship that scales with basin size (Walling, 1994). In small basins, such as those defined by experimental plots, these differences are small enough that rates of transportation are considered adequately descriptive of the state of soil erosion in the plot. In larger basins, such as a river's watershed, the disconnect between soil loss and deposition within the watershed and transportation past the watershed's outlet are large enough that means of estimating or directly quantifying rates of soil loss and deposition may be required. Typically, characterizations of erosion at these scales, plot- and watershed-, are used to explain the on- and off-site impacts of soil erosion, respectively. The methods that can be used to characterize soil erosion at these spatial-scales typically draw from long-term data sets to account for temporal variability. They are described as follows.

1.3.1. Plot-Scale Methods

Though they do not characterize the full gamut of erosive processes related to water, wind, and tillage erosion, empirical soil erosion models are valuable tools for researchers interesting in the on-site impacts of soil erosion and its management (Nearing *et al.*, 1994). Typically, these models are based in plot-scale measurements of soil erodibility, the erosivity of a process, and factors that affect erosivity and erodibility and are extrapolated to 2D, cross-sectional hillslopes representative of erodible landforms. The strength of such models is two-fold. First, the experimental basis of empirical soil erosion models allows them to illustrate the roles of the factors governing erodibility and erosivity, without making potentially unfounded assumptions regarding their mechanistic relationships with erodibility, erosivity, and each other (Weil and Brady, 2016). Second, by separating the erodibility of soil and the erosivity of a process from the factors that affect them, the specific effects of such factors can be investigated with a greater degree of confidence (Weil and Brady, 2016). These characteristics are of particular value to researchers interested in soil erosion, allowing them to not only characterize soil erosion's state, but to propose strategies for its management rooted in the factors governing erodibility and erosivity. For these reasons, the USLE, WEQ, and TillEM models – characterizing water, wind, and tillage erosion, respectively – have found wide-spread use.

The USLE model characterizes water erosion as the product of soil's erodibility and the measured erosivity of rainfall and runoff. This empirical estimate of water erosion is then adjusted by largely independent factors characterizing hillslope length, hillslope gradient, (vegetative) cover management practices, and supporting (agronomic) practices (Wischmeier and Smith, 1965). Specifically, it characterizes average annual rates of rill and inter-rill erosion at the order of a 22-year average rainfall cycle (Wischmeier and Smith, 1965) using time-averaged inputs. Adjustments have been made to USLE since its inception that allow it to accept time-integrated inputs, improving the precision of outputs (USDA, 2013). This improvement of the basic USLE model has been named RUSLE2. Gullying is not characterized by USLE-based models (USDA, 2013), instead being characterized on a case-by-case basis (Reid and Dunne, 1996).

Much like USLE model, the WEQ begins with a baseline, empirical estimate that is then adjusted using relevant factors. In the case of WEQ, the baseline estimate is of soil erodibility is subsequently adjusted in a stepwise manner using factors that characterize (in order of application) soil ridge roughness, climate, field length along the direction of the prevailing wind, and equivalent vegetative cover (Woodruff and Siddoway, 1965). Unlike USLE factors, those of WEQ are interdependent, making commentary on their effects on calculated rates of wind erosion less certain (Woodruff and Siddoway, 1965). Rates of wind erosion calculated with WEQ are average annual rates of suspension, saltation, and creep; though such averages are based in a smaller data set than USLE (99 dust storms between 1954 and 1956, inclusive, near Garden City, Kansas as described in 1963 by Chepil and Woodruff) and are based in time-averaged inputs.

The TillEM model, too, begins with a baseline, empirical estimate that is subsequently adjusted. However, the manner in which these adjustments are made differs from USLE and WEQ, as does the scope of the estimate provided by TillEM. TillEM describes the forward movement of soil by tillage with a multiple linear equation (Li *et al*., 2007; Lobb and Kachanoski, 1999). The model's coefficients represent a baseline rate of soil erodibility on level ground, additional erodibility related to hillslope gradient, and additional erodibility related to hillslope curvature (Li *et al*., 2007; Lobb and Kachanoski, 1999). Slope gradient and curvature act as independent variables (Li *et al*., 2007; Lobb and Kachanoski, 1999). This approach to modelling lumps together factors governing the erosivity of tillage, including implement design, implement operation, implement-tractor match, and farmer behaviour; acknowledging the difficulty in their disentanglement (Lobb and Kachanoski, 1999). Unlike ULSE and WEQ, which summarize the states of water and wind erosion over multiple years, TillEM calculates the mass movement of soil parallel to the direction of one tillage pass (Li *et al*., 2007; Lobb and Kachanoski, 1999). This temporally discrete mass movement has been named forward tillage translocation. If the net forward translocation is calculated for several tillage passes over time, the result is a precise estimate of the rate of tillage erosion. In this manner, rates of soil erosion characterized using TillEM are similar RUSLE2's outputs, as far as precision is considered. Tillage translocation perpendicular to the direction of tillage (lateral tillage translocation) is not characterized by TillEM, but has been estimated to be approximately 50% of forward tillage translocation (Lobb and Kachanoski, 1999).

Typically, these soil erosion models are applied to surveyed landforms in fields exhibiting problematic rates of soil erosion. In Canada, these models have also been used to provide insight into the state of soil erosion in landforms in landscapes across the nation. In 2003, Agriculture and Agri-Food Canada (AAFC) established the National Agri-Environmental Health Analysis and Reporting Program (NAHARP) to assess the many environmental impacts of agriculture in Canada (Clearwater *et al.*, 2016). The deliverable of the NAHARP was a set of Agri-Environmental Indicators (AEIs), representative of individual environmental impacts of agriculture that are of interest to policy-makers. The soil erosion AEI was calculated with modified RUSLE2, WEQ, and TillEM models, renamed WatERI, WindERI, and TillERI, respectively (Lobb *et al*., 2016). These models, collectively referred to as SoilERI, were applied to landforms described by the Soil Landscapes of Canada (SLC) geodatabase. The SLC geodatabase describes landscapes as spatially defined, ecologically uniform entities composed of any number of representative landforms. These representative landforms are encoded to the geodatabase as 2-dimensional hillslope cross-sections that are in turn composed of upper-, middle-, lower-, and depositional-hillslope segments. This data is a 1:1,000,000-scale recompilation of existing, detailed, 1:20,000- to 1:63,360-scale, soil survey data (AAFC, 2021). Additional, time continuous data used by SoilERI are input as 5-year averages, synchronous with the Canadian Census of Agriculture. Rates of soil erosion were calculated for each hillslope segment in each landform in the geodatabase, treating the rate of soil erosion as a steady-state mass balance function. This function was defined as the difference between rates of soil deposition and loss in the hillslope segment, the later estimated using SoilERI. Deposition was inferred from the rate of soil erosion immediately uphill of the hillslope segment of interest, multiplied by a sediment delivery ratio (SDR); a concept described in Section 1.3.2. in greater detail. The highest estimated rate of erosion within each landscape was reported by NAHARP, as such rates have greater bearing on relevant resource management policy (Lobb *et al*., 2016).

Though the SoilERI model provides valuable insight into the plot-scale state of soil erosion in landforms across Canada's landscapes, it does not adequately characterize soil erosion at broader scales. This is due to the previously mentioned divergence between rates of soil loss and deposition within a basin and transportation past the basin's outlet as basin size increases. Methods developed to characterize soil erosion at the watershed-scale take into account this divergence while aiming to relate the watershed-scale state of soil erosion to its plot-scale state.

1.3.2. Watershed-Scale Methods

Methods used to describe soil erosion at the watershed-scale broaden its characterization to include processes not addressed by plot-scale soil erosion models. This can include gullying and lateral tillage translocation that may affect the erodible landforms these models are applied to, or processes that only affect features outside of such landforms, like flow control infrastructure. Watershed-scale characterization of soil erosion is achieved through quantification of transportation past the watershed outlet paired with estimation of soil loss and deposition within the basin's boundaries, either through inference or measurement. These methods aim to link soil erosion at the watershed-scale back to soil erosion at the plot-scale to better serve erosion management planning. As such, studies that characterize soil erosion at the watershed-scale often (and somewhat ambiguously) refer to estimates of soil loss from soil sediment sources as estimates of soil erosion. Soil erosion at the watershed-scale can be characterized in three ways, through calculation of sediment yields, sediment delivery ratios (SDRs), and sediment budgets.

Sediment yield is the quotient of the suspended sediment discharged through a watershed's outlet divided by basin area, and is measured in t ha^{-1} yr^{-1} (USDA, 2008b). It is a summary of sediment loss and deposition within a watershed's boundaries; averaging rates of erosion across the basin's area (Walling, 1994). This makes sediment yield a low-resolution summary of sedimentary activity within a watershed, and is useful for drawing comparisons between basins (Walling and Webb, 1987). It is simple to calculate and often convenient, with the necessary data collected by many nations' hydrometric monitoring programs (Walling, 1994). In Canada, such data are collected and managed by the Water Survey of Canada (WSC, 2021), Agriculture and Agri-Food Canada (AAFC, 2017), and other organizations; though spatial- and temporal-coverage of such data varies. Despite its ease of calculation and convenience, sediment yields are often too coarse to serve as the sole basis for assessing the impacts of and managing soil erosion in a watershed. This is due to their broadly aggregative nature. Such a characterization of sedimentary activity dilutes the apparent severity of erosion within a basin and, in turn, its on-site impacts. Though deposition is alluded to by the tendency of sediment yields to scale inversely with basin size, this characteristic is not adequate to quantify rates of deposition within the basin or its impacts (Walling, 1994). Therefore, when richer data are available or can be collected, it is recommended that sediment yields are complemented by sediment delivery ratios (SDRs).

The SDR of a watershed is a unitless quotient, calculated by dividing the basin's sediment yield by the aggregate rate of erosion within its boundaries (USDA, 2008b). As its name suggests, it describes the efficiency which eroded sediment is delivered from sediment sources within the basin to the basin outlet, which aggregate rates of deposition can be inferred from. In effect, the SDR constrains rates of erosion – otherwise averaged across the watershed by calculation of the sediment yield – to erodible landforms in the basin, in turn increasing the resolution of sediment yields. Aggregated rates of erosion can be derived following one of two methods, depending data availability. In basins where gross rates of erosion have not been established, erosive processes that are active in the basin are first identified, then measured if they are measurable (e.g., channel erosion and gullying of non-soil sediment sources) or modelled if they are not (e.g., rill and inter-rill of soil sediment source), following the recommendations of the USDA (2008a). Models that can be used include ULSE, RUSLE, RUSLE2, and – in Canada – WatERI. In regions where aggregated rates of erosion are well understood, models relating SDR to watershed characteristics can be applied. Such models typically treat watershed area or relief and channel network length as predictor variables (USDA, 2008b), and are based in previously noted relationship between sediment yield and basin size and the understood influence topography and channel density have on sediment delivery (Roehl, 1962). Though SDRs improve characterization of sedimentary processes within a watershed, their relationship with sediment yield limits their scope; being best used to draw comparisons between watersheds. This is due to their aggregative treatment of erosion and deposition. Resource managers will often ask pointed questions, such as "What sediment sources are most impacted by soil erosion?", "Which erosive processes are most responsible for soil erosion?", "Where is eroded soil deposited?", or "Have past erosion management practices reduced the impacts of soil erosion?". Such questions require process-specific characterization of sedimentary processes at the watershed-scale that can account for discontinuities in their rates over time (Walling, 1994). Sediment budgets can help answer such questions.

A sediment budget is a balanced account of sediment production (i.e., loss) in a basin, sediment deposition in a basin, and sediment delivery past a basin's outlet, parsed by sediment sources, sinks, and the specific processes that affect them (Reid and Dunne, 1996). Because sediment budgets are not a spatially averaged summary of erosion, rates characterized by a sediment budget are expressed in t yr^{-1}. Due to their granular nature, a variety of methods can be employed to quantify processes characterized by a sediment budget. These methods are better detailed by literature reviews specific to sediment budgeting, such as Rapid Evaluation of Sediment Budgets, by Reid and Dunne (1996), but a few common methods can be summarized as follows. Rates of erosion can be quantified through measurement or modelling, as when calculating SDRs. Erosive processes other than those related to water can be included in sediment budgets, provided their lower rates of delivery to the watershed outlet are accounted for (Reid and Dunne, 1996). Such processes may include wind and tillage erosion, modellable with the WEQ and TillEM models or, in Canada, the WindERI and TillERI models, respectively. Rates of deposition within the basin can be quantified through direct measurement or inference. Direct quantification of rates of deposition is possible through measurement of aggradation of natural depressions or purpose-built flow control structures, provided the trapping efficiency of such structures is known. Rates of deposition are commonly inferred from the residuals between rates of measured processes in otherwise unbalanced budgets. A sediment budget can be temporally differentiated, provided the data it is based in is sufficiently long term (e.g., Trimble, 1983). However, there is still validity in sediment budgets rooted in shorter-term data. Such sediment budgets tend to be less certain than those based in longer-term observations, but such uncertainty is often less than that of sediment yields and delivery ratios calculated from the same data (Reid and Dunne, 1996). None the less, the uncertainty does limit the scope of inferences that can be drawn from a sediment budget. In particular, exact rates presented by a sediment budget are less meaningful than their orders of magnitude. This is not problematic, as many erosion management decisions are based in the relative severity of erosion and its impacts (Reid and Dunne, 1996). Though sediment yields are a more precise measure of sedimentary activity in a watershed than sediment yields and delivery ratios, they by no means supplant these methods. By comparison, sediment budgets are time consuming and resource intensive exercises, often baring their broad implementation. As such, when large-scale characterization of sedimentary activity is desired sediment budgets are most appropriate for identifying key erosion management issues. The findings of such studies can be extrapolated to similar basins in the region (Reid and Dunne, 1996), which sediment yields and delivery ratios can be used to support. This is illustrative of the need for a wide range of studies when investigating erosion at such scales, including supporting means of characterizing erosion (e.g., quantifying of ^{137}Cs distribution and sediment fingerprinting), investigation of soil erosion's impacts, and evaluation of relevant erosion management practices.

The study that follows attempts to combine these plot- and watershed-scale methods, along with past, relevant studies, in a manner that speaks to the overall sedimentary characteristics of cultivated watersheds in Manitoba's Red River Valley, with a particular focus on sediment budgeting.

2. Characterizing Soil Erosion in Cultivated Watersheds in Manitoba's Red River Valley using Sediment Budgeting

2.1. Introduction

It has been estimated that soil erosion has led to a cumulative loss of between 40 billion and 60 billion dollars-worth of potential agricultural productivity in Canada since the 1970s (Lobb *et al.*, 2017). This estimate considered only the direct, on-site impact of soil loss on crop yields. Costs not included in this estimate were those associated with the increased fertilizer and pesticide requirements where soil is lost, nor did it consider the economic cost of sediment transported and deposited off-site. The later cost can be attributed to burial of productive soils downslope, channel, and reservoir in-filling, increased turbidity, and transportation of sorbed contaminants and can be equal or greater than soil erosion’s on-site costs (Weil and Brady, 2016). This reflects the need to understand the gamut of sedimentary processes affecting soil in agricultural landscapes: loss, deposition, and transportation.

In respect to the on-site state of soil erosion, Agriculture and Agri-Food Canada (AAFC) has reported on the trend in its risk between 1981 and 2011 though the National Agri-Environmental Health Analysis and Reporting Program’s (NAHARP) Agri-Environmental Indicators series. The soil erosion agri-environmental indicator (AEI) is based in the SoilERI model, which estimates soil erosion risk in cultivated fields by taking into account erosive processes related to water, wind, and tillage. Over the years, SoilERI has demonstrated substantial decreases in soil erosion risk across Canada, especially in the prairie provinces of Manitoba and Saskatchewan. This change has been attributed to widespread adoption of conservation tillage practices, fewer fields left in summerfallow, and increased planting of high-residue crops (Lobb *et al.*, 2016). In Southern Manitoba specifically, NAHARP soil erosion risk estimates have been generally supported by measurement and modelling of water (Wright, 1994) and tillage erosion (Li, 2006); however, research regarding wind erosion has been lacking. Though the model has been validated (Li and Lobb, 2009; Baldwin and Lobb, 2012; Ali *et al.*, 2013; Huang *et al.*, 2017), there are factors ignored by SoilERI that are suspected to influence soil erosion risk. These include landform complexity, anthropogenic landforms including fence lines, tree lines, ditches, and roads, erosion management practices including grassed waterways, strip cropping, terracing, contour cultivation, winter cover cropping, and shelterbelt planting, and the interaction of erosive processes (Lobb *et al.*, 2016). In light of these limitations, soil erosion risk estimates made using SoilERI have been described as qualitative and best used to guide government policy (Lobb *et al.*, 2016). However, the scope of SoilERI is limited to a generalization of cultivated fields at the landscape-scale, defined by the Soil Landscapes of Canada (SLC) geodatabase. As such, policy decisions guided by it may not adequately address the watershed-scale downstream impacts of sediment previously mentioned. This has motivated broader, landscape-scale research in Manitoba which considers the sedimentary processes of transportation and deposition.

Regarding off-site transportation and deposition, research has been conducted in Southern Manitoba to better understand the processes in Canada's cultivated prairie landscapes. In 2013, a study by Koiter *et al*. sought to establish the contribution of sediment sources (topsoil, stream banks, and bedrock) to suspended sediment loads in streams using a technique known as sediment fingerprinting. A unique characteristic of the study was its use of sediment fingerprinting in successively smaller basins, nested within each other in the study area. It was found that topsoil sources (i.e., cultivated fields and riparian areas) made more substantial contributions to suspended sediment loads in smaller, headwater basins. The study identified this as an example of sediment cascade connectivity, a concept formalized by Fryirs (2013) that builds upon the sediment delivery ratio (SDR) described by Walling (1994). Sediment cascade connectivity describes sediment delivery from source to sink as a series of discrete steps, with temporary deposition in intermediate sediment sinks before final delivery. Connectivity was judged to be greater between topsoil sources and streams in small basins than in large ones, with fewer opportunities for intermediate deposition. Interestingly, the study also identified a slight increase in connectivity near the outlet of the study area. It was speculated this change was due to increased connectivity related to the local ditch network; simultaneously acting as a conduit for fine-grained sediment while trapping coarser soil aggregates. Identifying and understanding the role of these sediment sinks, their role in sediment transfer, and relating them to the established knowledge of soil erosion in cultivated Canadian prairie landscapes can provide a richer understanding the sedimentary processes affecting soil. In turn, erosion control policy decisions that better reflect the wide range of impacts related to soil erosion can be made.

In this study, the Boyne-Morris and La Salle River watersheds in Southern Manitoba are investigated. Both watersheds contain the three Prairie Ecozones found in Manitoba which, when combined with their predominantly agricultural economies, improves the extensibility of soil erosion research conducted within their boundaries to other watersheds in the region. Their representativeness of agricultural catchments in the Canadian Prairies in part motivated their selection by Wright (1994), Li (2006), and Koiter *et al.* (2013) in their respective studies, previously mentioned. Sediment budgeting can be used to relate these studies to each other, illustrating nature of soil erosion in a holistic manner. In addition, sediment budgeting can strengthen validation of past studies and provide further insight into sedimentary processes affecting soil that have not been investigated to date.

Sediment budgeting describes erosion in a basin as a balanced account of sediment production (loss) in the basin, sediment deposition in the basin, and sediment delivery (transportation) past the basin outlet. First proposed by Dietrich and Dunne (1978) and further developed by others (Swanson *et al.*, 1982; Phillips, 1986), sediment budgeting requires identification of relevant sedimentary processes and their controls from which rates of sediment production, deposition, and delivery can be calculated. Though sediment budgets portraying the full gamut of sedimentary processes in a watershed can be of academic interest, simplified budgets including only a few processes can be drafted to address specific land management issues (Reid and Dunne, 1996). Provided the sedimentary processes considered by simplified budgets adequately relate to the management issues of interest, management decisions related to their results can be applied to similar catchments (Reid and Dunne, 1996). In the Boyne-Morris and La Salle River watersheds, where understanding downstream transfer of sediment to inform agri-environmental policy is of interest, the objectives of the study were:

1. to identify the processes producing sediment in sources, depositing it in sinks, and delivering it to the watershed outlet, and
2. to calculate the rates of sediment production in sources, deposition in sinks, and delivery to watershed outlets, based on an understanding of the identified sedimentary processes and their controls.

Erodible landforms in cultivated fields were assumed to be the sediment source responsible for sediment production of interest in the watersheds, with rates of sediment production inferred from published NAHARP erosion risk estimates. Relevant sediment deposition was suspected to be active in two landforms: sediment sinks within cultivated fields (such as in-field surface drains and ditches) and in road-side ditches along field edges. Assumption of the importance of the later landform reflects Koiter *et al*.'s (2013) speculation that ditches can influence sediment cascade connectivity, previous studies of sediment accumulation in road-side ditches (Pease *et al.*, 2003; Lecce *et al.*, 2006), the proximity of road-side ditches to eroding fields, and the sheer extent of Southern Manitoban road-side ditch network. Lastly, sediment delivery was inferred through the suspended sediment load, transported in streams. To address infrequent, high-magnitude sedimentary events, this study considered data spanning from 1981 through to 2018, though not all sedimentary processes are documented over the entirety of this period of time.

A noteworthy deficiency of the sediment budgeting technique is the degree of uncertainty associated with a given budget's components. None the less, sediment budgets provide valuable insights in the roles of sediment sources, sinks, and transportation pathways in a watershed. This does, however, mean that the exact rates calculated when drafting a sediment budget are less noteworthy than their orders of magnitude. This is not a problem when judged against the basis of land management decisions, which typically consider the effect of sedimentary processes relative to each other (Reid and Dunne, 1996), making the sediment budget a valuable among many for understanding the sedimentary character of a catchment. This assertion extends to sediment budgeting as is applied to the Boyne-Morris and La Salle River watersheds in this study.

2.2 Study Area Description

The Boyne-Morris and La Salle River watersheds, the foci of this study, are located adjacent to one another in Southern Manitoba. The watersheds span from the Pembina Mountain region of the Manitoba Escarpment, across the Red River Valley, and approach the western outskirts of the City of Winnipeg; covering 407,113 ha and 240,769 ha, respectively. With respective effective drainage areas of 364,437 ha and 221, 562 ha, the watersheds both drain eastward – the Boyne-Morris River watershed through the Shannon Creek, Little Morris River, and Morris River systems and the La Salle River watershed through the La Salle River – into the north-flowing Red River (Fig. 2.1). The landscape is predominantly of glacial origin and related to the retreat of the Laurentide Ice Sheet. Atop the Manitoba Escarpment landforms are hummocky, undulating, and inclined – as is typical of ice stagnation topography – and overlain by medium-textured Black Chernozems. Eastward – at the base of the escarpment – ridged, undulating, and level landforms cover ancient beach ridges and bird's foot deltas from which coarse-textured Dark Grey Chernozems, Black Chernozems, and Dark Grey Luvisols are derived. Further east, in the Red River Valley, undulating and level landforms sit atop glacio-lacustrine deposits and are characterized by fine-textured Black Chernozems and Humic Vertisols (Fig. 2.2).

The Boyne-Morris and La Salle River watersheds contain the three prairie ecoregions found in Manitoba: Aspen Parkland, Lake Manitoba Plain, and Manitoba uplands. These ecoregions are characterized by native vegetation that includes tall-grass prairie and meadow grass communities, aspen groves, and forests containing white elm, basswood, cottonwood, Manitoba maple, and green ash (AAFC, 2004a; AAFC, 2004b). Climate can be described as sub-humid with an average of 545.0 mm of precipitation annually: 445.3 and 100.3 mm falling as rain and snow, respectively (Fig. 2.3). Temperatures can range from less than -30°C in the winter to greater than 30°C in the summer, with a mean annual temperature of 3.5°C. Winds typically blow from the NW year-round, with average wind speeds of 14.7 km h^{-1} gusting up to 93 km h^{-1} as illustrated by Figure 2.4 (ECCC, 2022). Agriculture is widespread in the Boyne-Morris and La Salle River watersheds, with 1164 and 524 farms presently covering 313,274 and 191,094 ha of their respective total areas (Fig. 2.5), a figure that is consistent with past census data (Fig. 2.6). Cereals and oilseeds are presently the predominant crops planted (133,940 and 79,189 ha and 143,703 and 96,748 ha, respectively) with lesser amounts of forages (1485 and 1825 ha), pulses (9247 and 2924 ha), and potatoes (3748 and 84 ha) cultivated (AAFC, 2016e). This contrasts with historical, cereal-centric production (Fig. 2.7).

Though intensified by agriculture, the contemporary combined risks of wind, water, and tillage erosion within the Boyne-Morris and La Salle River watersheds are typically very low (< 6 t ha^{-1} yr^{-1}), as estimated by the SoilERI model (Fig. 2.8). These modelled estimates portray wind as the primary driver of soil erosion in the region (AAFC, 2016d). This reflects historic concerns regarding soil erosion and efforts made to reduce it, with the use of windbreaks and cover crops reported by 560 and 215 and 176 and 61 farms in the Boyne-Morris and La Salle River watersheds, respectively (AAFC, 2016e). Tillage erosion has been estimated to be of less concern, with 171,410 and 125,049 ha of the present-day Boyne-Morris and La Salle River watersheds conventionally tilled, respectively (AAFC, 2016b; AAFC, 2016e). Conservation tillage practices are slightly less common (100,744 and 48,332 ha) and no-till practices are uncommon (29,996 and 12,153) (AAFC, 2016e). In both watersheds, conventional tillage is typically defined by one primary tillage operation, one to two secondary tillage operations, one to two harrowing operations, and seeding and has been described as having more in common with a conservation tillage sequence than the formal definition of a conventional tillage sequence (Li *et al.*, 2007). Such sequences have the capacity to translocate soil in the till layer 0.12-0.34 m forward along and 0.06-0.17 m lateral to the direction of tillage over level land (Lobb, 2011). Only 672 and 482 ha of the watersheds are left in fallow each year, though weed control in these areas is predominantly mechanical (461 and 86 ha) (AAFC, 2016e). Past adoption of specific tillage practices is summarized in Figure 2.9. Water erosion has the lowest quantifiable risk and is generally unmanaged (AAFC, 2016c). Water management strategies instead focus on enhancing agricultural capability through promotion of run-off, reducing ponding in otherwise poorly drained fields. This is achieved with extensive ditch networks.

The drainage infrastructure of the Boyne-Morris and La Salle River watersheds consists of a very large, unquantified surface drain network in fields, 550 and 343 km of larger in-field ditches, 7627 and 4469 km of road-side ditches following 3598 and 2174 km of unpaved roads, and 1128 and 1439 km of provincial drains; all draining into the watersheds' 453 and 113 km natural stream networks (Fig. 2.10). Routine maintenance of the networks is required to ensure their continued function, often taking the form of dredging to remove both vegetation and deposited sediment. In respect to road-side ditches, such operations are handled by rural municipalities (RMs) with a turnover rate of approximately 20 years for an RM's cumulative road-side ditch system (Justin Shaw, pers. comm., RM of MacDonald Public Works). Once dredged out, ditch sediment is piled along the edges of adjacent fields and the spoil bulldozed to return it to its presumed source. The ditch networks of the Boyne-Morris and La Salle River watersheds convey water seasonally; reflecting the flashy flow regimes of the watersheds' natural streams. Average daily discharges at the mouths of the Shannon Creek, Little Morris River, Morris River, and La Salle River systems are 1.4, 3.7, 5.6, and 4.5 $m^3 s^{-1}$, respectively; measured between April and November when their channels are thawed. Discharges typically peak in April and are driven by the freshet, with average annual peak discharges of 41.4, 61.3, 60.0, and 54.5 $m^3 s^{-1}$ respective to the four rivers (WSC, 2021).

Although records of stream discharge in the Boyne-Morris and La Salle River watersheds are abundant and long-term, fewer measurements of total suspended solids (TSS) have been made. This is problematic from a budgeting standpoint as such data are required to determine sediment discharges from the watersheds. Therefore, budgets for smaller sub-watersheds within the Boyne-

Morris and La Salle River watersheds with adequate data were drafted. The two sub-watersheds selected were defined by Water Survey of Canada (WSC) hydrometric monitoring stations 05OF024 and 05OG008, respectively, and will be referred to by their station codes forward. Both sub-watersheds exhibit drainage patterns, roadway patterns, and general geologic, pedologic, ecologic, climatic, and agricultural characteristics similar to those of their respective larger watersheds, supporting their treatment as representative. This is illustrated by the inset sub-watershed boundaries in Figures 2.2, 2.5, 2.8, and 2.10.

2.3. Materials and Methods

Drafting a sediment budget requires identification of relevant sedimentary processes and understanding their controlling factors to calculate their respective rates of sediment loss or accumulation. These requirements can be difficult to satisfy, due to the spatial- and temporal-variability of the sedimentary processes included in a sediment budget and the interaction of these processes at the watershed-scale. This can impact the applicability of any number of measurement techniques that can otherwise be easily applied to sedimentary processes when investigated independently in other geomorphological studies. As such, when planning a sediment budget, it is less useful to emulate the methods of past sediment budgeting studies than it is to follow the broader guidelines suggested by literature reviews. Such literature reviews not only summarize a number of measurement techniques that may be considered when drafting a sediment budget, but contextualize the techniques against the characteristics of watersheds they've been applied to and the problems they've been used to address. In addition, literature reviews of this nature may present project planning strategies and workflows that simultaneously ease drafting of a sediment budget and enhance the researcher's understanding of the sedimentary system being described. When drafting the sediment budgets for the 05OF024 and 05OG008 sub-watersheds, one such review was frequently consulted: Rapid Evaluation of Sediment Budgets by Reid and Dunne (1996). Section 2.3.1. summarizes how the budgets were organized following their example. Section 2.3.1. also provides an overview of how the sediment budget was calculated while subsequent sub-sections detail the specific methods used to quantify rates of sediment discharge or accumulation associated with sedimentary processes active in fields (Section 2.3.1.1.), road-side ditches (Section 2.3.1.2.), and streams (Section 2.3.1.3.). These settings generally relate to sediment production in the sub-watersheds, deposition within the sub-watersheds, and sediment delivery past the sub-watershed outlets. A description of how the project's sediment budgets were validated follows in Section 2.3.2. All analyses were performed using R 4.1.2 (R Core Team, 2021), and the tidyverse 1.3.1 (Wickham *et* al., 2019), units 0.7-2 (Pebesma, *et al.*, 2016), sf 1.0-4 (Pebesma, 2018), and raster 3.5-2 (Hijmans, 2015) packages unless otherwise noted.

2.3.1. Sediment Budget Organization and Calculation

The first step in drafting sediment budgets for the 05OF024 and 05OG008 sub-watersheds was to plot flowcharts; illustrating relevant processes of loss, transportation, and deposition, as well as their interactions (Fig. 2.11). From these diagrams primary and secondary data collection was planned and methods were devised.

Secondary data was collected and, where gaps existed, primary data collection was planned for. Publicly available secondary data that was used to construct the sediment budgets included Manitoba Land Initiative (MLI) shapefiles describing land use, roadways, watershed boundaries, and hydrographic features (MLI, 2000; MLI, 2004; MLI, 2005; MLI, 2013); the National Soil Database (NSDB) SLC geodatabase describing soil and landform characteristics (AAFC, 1996; AAFC, 2010; NRC, 2017); NAHARP soil erosion risk estimates calculated at the scale of the SLC geodatabase (AAFC, 2016b; AAFC, 2016c; AAFC, 2016d); WSC hydrometric data (WSC, 2021);

and AAFC TSS data (AAFC, 2017). Unpublished secondary data was also consulted and included TSS measurements made by the Deerwood Soil and Water Management Association (DSWMA) and records and anecdotal evidence of ditch and road maintenance from the RM of MacDonald Public Works. These data were sufficient to determine rates of sediment discharge from erodible landforms in cultivated fields (loss) and past sub-watershed outlets suspended in streams (transportation). However, these secondary data sources were insufficient to solely quantify the sedimentary processes active road-side ditches. Therefore, a survey of road-side ditches was conducted between 2015 and 2018, inclusive, and focussed on collecting primary data related to the sedimentary processes active in them (detailed in Section 2.3.1.2.). The specific sedimentary processes these primary and secondary data apply to are summarized in Table 2.1. With data in hand, methods to calculate rates of sediment loss or accumulation could be devised.

Because a sediment budget is a balanced account of sediment production in a basin, deposition in a basin, and delivery (transportation) to the basin's outlet, a simple sediment budget characterizing a single sediment transfer can be quantified as:

$$\varphi = I - \Delta S \qquad \text{(Eqn. 2.1)}$$

where,

φ is the rate sediment is delivered to the basin outlet in t yr^{-1},

I is the rate sediment is produced by sediment sources in the basin in t yr^{-1}, and

ΔS is the changing rate sediment is deposited in the basin in t yr^{-1},

as described by Roberts and Church (1986). When land management questions are very focussed (e.g., Derose *et al.*, 1998), this basic relationship adequately characterizes the sediment budget of the basin of interest. However, this cannot be said of all sediment budgeting exercises. By their very nature, some land management questions require sediment budgets that simultaneously consider multiple sediment transfers, with each transfer characterized by Equation 2.1. These transfers can be related, with the rate of sediment delivery (φ) of one transfer acting as the rate of sediment production in the next (I). This was true of the 05OF024 and 05OG008 sub-watersheds' sediment budgets which, as illustrated by Figure 2.11, accounted for sediment transfers from field to road-side ditch, from road-side ditch to field, from road-side ditch to stream, and from field to stream. In these sub-watersheds, it was expected that the rates of sediment production and deposition through any of these transfer paths would be the sum of a number of sedimentary processes. By time-averaging these rates, ΔS can be simplified to S and the rates of sediment production and deposition can be quantified as:

$$I = \sum_{a}^{n} \bar{Q}_a$$

or (Eqn. 2.2)

$$S = \sum_{a}^{n} \bar{Q}_a$$

where,

$\bar{Q}_a$ is the average annual rate of discharge or accumulation associated with a sedimentary process, a, in t yr^{-1} and

n is the number of relevant sedimentary processes.

By these steps alone the sediment budgets for the 05OF024 and 050G008 sub-watersheds can be said to account for the sedimentary processes active within their borders and any interactions that may exist between them. However, the sediment budgets do not take into consideration the spatial-variability of the activity of these processes. To do so, one must consider the spatial distribution of factors that control erosion in the sub-watershed, deposition in the sub-watersheds, and transportation past the sub-watershed outlets. For any given sedimentary process, doing so portrays the basin in question as a collection of catchments, each defined by contrasting expression of these factors. The average annual rate of sediment discharge or accumulation associated with a sedimentary process can then be described as:

$$\bar{Q}_a = \sum_b^o \bar{Q}_{ab} \qquad \text{(Eqn. 2.3)}$$

where,

$\bar{Q}_{ab}$ is the average annual discharge of an input or output process, a, active in a catchment, b, in t yr^{-1} and

o is the total number of catchments.

This means of accounting for spatial-variation has been advocated for (Reid and Dunne, 1996) and was applied by dividing the 05OF024 and 05OG008 sub-watersheds into catchments characterized by coarse-, medium-, and fine-textured soils; following and understanding of the relationship between sediment grain-size and erosion. Catchments were based in soil texture data derived from the SLC geodatabase's Polygon Attribute (PAT), Component (CMP), Soil Name (SNT), and Soil Layer (SLT) tables (Fig. 2.12). These catchments applied to sediment transfers from field to road-side ditch and from road-side ditch to field. Rates of sediment transfer from road-side ditches to streams and from fields to streams were calculated at the sub-watershed scale, reflecting the stronger control watershed boundaries have over such transfers as well as the previously mentioned scarcity of TSS data.

To account for temporal-variability of the activity of sedimentary processes, rates of sediment discharge or accumulation over several years were considered when possible and averaged, as has been alluded to in Equations 2.2 and 2.3. It should be stated that because the sediment discharge or accumulation associated with any sedimentary process is a rate, its average should be treated as a harmonic mean (Steel *et al.*, 1997), which can be expressed as:

$$\bar{Q}_{ab} = \frac{p}{\sum_c^p \frac{1}{Q_{abc}}} \qquad \text{(Eqn. 2.4)}$$

where,

Q_{abc} is the discharge of an input or output process, a, active in a catchment, b, in the year, c, in t yr^{-1}, and

p is the total number years.

How the rates (Q_{abc}) were calculated varies from process to process, but specific methodologies can be generally grouped and detailed according to the portion of the landscape they are active in, fields, road-side ditches, or streams, as is illustrated by Figure 2.11.

2.3.1.1. Sedimentary Processes in Fields

Rates of sediment discharge from erodible landforms in cultivated fields were calculated from sediment yields and the proportion of a landscape's area such landforms were known to occupy. These values were derived from NAHARP water, wind, and tillage erosion risk estimates and the spatial framework these risk estimates were calculated in, the SLC geodatabase, respectively. The SLC geodatabase describes landscapes as spatially defined, ecologically uniform entities, composed of any number of representative landforms. These representative landforms are presented as 2-dimensional hillslope cross-sections, which are typically composed of upper-, middle, lower-, and depositional-hillslope segments. NAHARP erosion risk estimates are calculated for each hillslope segment in a landscape, but for each erosive process only the greatest rate of erosion in the landscape is reported. These rates are annual averages, representative of the

5-year period between census years from 1981 to 2011. Understanding this, and because sediment budget catchments were defined following the SLC geodatabase's description of soils, sediment discharge from erodible landforms in a catchment can be calculated as follows:

$$Q_{abc} = \sum_{e}^{r} Y_{abcd} * A_{bd} * P_{bde} \qquad \text{(Eqn. 2.5)}$$

where,

Y_{abcd} is the soil erosion risk of sedimentary process a in landscape d in catchment b and year c in t ha^{-1} yr^{-1},

A_{bd} is the terrestrial area of landscape d in catchment b in ha,

P_{bde} is the proportion of landscape d in catchment b occupied by hillslope segment e, and

r is the number of hillslope segments in catchment b.

A number of adjustments to the SLC geodatabase must first be made before this equation can be applied and even when made, the equation can only be applied selectively to the geodatabase. This reflects the scope of the SLC geodatabase and how NAHARP erosion risk estimates are reported, respectively.

Because SLC landscapes are partitioned on the basis of ecological characteristics, their spatial extents do not conform to watershed boundaries. As such, the spatial extents of SLC landscapes were first adjusted to sub-watershed boundaries. To do so, the SLC V3R2 PAT and Landscape Area (LAT) tables were joined and intersected with SLC V2R1 HYDRO and CanVec Political Boundary vectors to spatially define hydrographic and political features that are otherwise numerically, but not spatially, defined within the unmodified SLC geodatabase. The resulting vector was then clipped to the boundaries of the 05OF024 and 05OG008 sub-watersheds. The terrestrial, fresh-water, and oceanic areas of the vector were recalculated, simultaneously updating the PAT and LAT tables as they applied to the sediment budgeting exercise. From the updated LAT table, values of A_{bd} could be derived. Further adjustments to the SLC geodatabase were then made to account for how NAHARP erosion risk estimates were reported.

Recalling that NAHARP erosion risk estimates are only representative of the hillslope segment in a landscape undergoing the greatest rate of erosion, there is a degree of uncertainty when determining what SLC hillslope a reported rate of erosion applies to. Generally, the upper-, middle-, and lower-hillslope segments all experience some degree of erosion, with the middle-hillslope segment most affected by water erosion and the upper-hillslope segment most affected by wind and tillage erosion. As such, Y_{abcd} values derived from NAHARP erosion risk estimates were only paired to P_{bde} values related to cultivated upper- and middle-hillslope segments, depending on erosive process. By doing so, the calculated rates of sediment discharge from cultivated hillslopes can generally be treated as conservative. This conservativeness is reduced by the decision to apply NAHARP water erosion risk estimates to every middle-hillslope segment and NAHARP wind and tillage erosion risk estimates to every upper-hillslope segment in a landscape. This was deemed acceptable, as many landscapes in the 05OF024 and 05OG008 sub-watersheds are characterized by a single landform (Tab. 2.2).

Table 2.2 also illustrates that some landscapes are characterized by only one landform. This is problematic when such landforms are level with little or no slope. As defined by the SLC V3R2 geodatabase's Landform Definition (LDT) table, such landforms lack upper-hillslope segments. Despite this, wind and tillage erosion risk rates greater than 0 t ha^{-1} yr^{-1} are reported for these landscapes. A 2008 NAHARP document discussed plans to calculate erosion risk rates using updated landform definitions for such landforms (Li, 2008). As such, it was assumed NAHARP erosion risk rates were calculated using these definitions. The LDT was updated to conform to these definitions and the changes were propagated through to the Landform Segmentation (LST) table drawing from data encoded in the Landform Extent (LET) table. From the resulting updated LST, values of P_{bde} could be derived. It should be stated that calculation of P_{bde} assumes no changes in land use between 1981 and 2011. This assumption is acceptable when weighed against the Canadian Census of Agriculture's characterization of land use change in the Boyne-Morris and La Salle River watersheds, which is minimal (Fig. 2.6).

The rate of deposition in cultivated fields was calculated using Equation 2.1, with the previously calculated rates of sediment production used to characterize I and the rates of deposition and transportation calculated following the methods to be outlined in sections 2.3.1.2. and 2.3.1.3. summed to give φ.

2.3.1.2. Sedimentary Processes in and near Road-Side Ditches

Rates of sediment transfers into and out of road-side ditches were calculated as follows:

$$Q_{abc} = l_{ab} * p_{abc} * m_{abc} \qquad \text{(Eqn. 2.6)}$$

where,

l_{ab} is the length of the road-side ditch network adjacent to an erodible sediment source relating to transfer process *a* in catchment *b* in km,

p_{abc} is the proportionality between the length of the road-side ditch network adjacent to a sediment source eroded by transfer processes *a* in catchment *b* and l_{ab}, over year *c*, and

m_{abc} is the mass of sediment transferred from an eroding sediment source into or out of road-side ditches by transfer process a in catchment b over year c in t km^{-1} yr^{-1}.

Though primary data collection was required to quantify sediment transfers into and out of road-side ditches (Q_{abc}), determination of such rates was not possible without consideration of data from secondary sources. The extent of the road-side ditch network potentially impacted by such transfers (l_{ab}) could only be calculated from secondary data sources while a combination of primary and secondary data was used to determine eroding proportion of the road-side ditch network (p_{abc}) and the mass of sediment transferred into and out of road-side ditches (m_{abc}).

To determine the length of the road-side ditch network associated with each sediment transfer (l_{ab}), the potential erodible sediment sources related to each transfer were first considered. Sediment transfers from road to ditch or that involve reworking of ditch sediment have the potential to impact the entire ditch network. Therefore, the raw length of ditches characterized by MLI Designated Drain Index shapefiles sufficiently quantify l_{ab} for such transfers. Transfers from field to ditch, however, only have the potential to operate in ditches adjacent to cultivated fields. To determine l_{ab} relating to these transfers, MLI Land Use rasters were consulted. When MLI Designated Drain Index shapefiles are plotted over MLI Land Use rasters, many ditches map over transportation infrastructure cells. Therefore, a means to interpolate the contents of surrounding cells onto transportation infrastructure cells was required. To achieve this, all transportation infrastructure cells were reassigned to null. The resulting gaps in the raster were then reassigned to valid land use values based on their surroundings through nearest neighbour interpolation, targeting only null values and drawing from an 8-cell neighbourhood using Grass GIS' r.fill.stats module (Ducke, 2021). The modified raster was then vectorized, filtered on the basis of agricultural land use, and used to clip ditches characterized by MLI Designated Drain Index shapefiles accordingly. The lengths of ditches clipped as such could then be recalculated.

To determine the proportion of l_{ab} that, in a given year, was subject to an erosive transfer process (p_{abc}) a large-scale survey of evidence of erosive activity in and near road-side ditches was conducted. The evidence considered by this survey was the presence or absence of morphologically distinct depositional features that could be readily linked to the sediment transfers proposed to be active in and near road-side ditches, per unit length of ditch. Seven such features were identified (Fig. 2.13). The presence or absence of sedimentary features was generally counted at a 1.609 or 0.805 km resolution, reflecting the operative scale of erosive processes that transfer sediment from road to ditch, from ditch to road, or that rework ditch sediment; or that transfer sediment from field to ditch, respectively. At the large-scale the presence of ditches affected by these features ranged from ubiquitous to scarce, necessitating unique surveying methods for each. A detailed summary of these methods can be found in Table 2.3.

Simultaneous with the survey to determine p_{abc}, measurements were made between 2015 and 2018, inclusive, to determine the annual rate of sediment transfers into or out of road-side ditches where sediment transfers were observed (m_{abc}). The survey assumed:

$$m_{abc} = \bar{m}_f * \bar{n}_f \qquad \text{(Eqn. 2.7)}$$

where,

$\bar{m}_f$ is the arithmetic mean or median mass of a sedimentary feature in t and

$\bar{n}_f$ is the arithmetic mean or median number of sedimentary features per unit length of ditch the feature was observed in.

For each documented sedimentary feature, m_f was determined through measurements of soil bulk density (BD_f) and feature morphometrics while n_f was determined through feature counts. Because the 7 observed types of sedimentary features were morphologically distinct, each required unique means of measuring m_f and n_f. Similar considerations were required to calculate m_f from bulk density and feature metrics and, in turn, sediment transfer rates from $\bar{m}_f$ and $\bar{n}_f$. These particularities are detailed in Table 2.4. Qualitative and quantitative tests of normality (histograms and Q-Q plots and Shapiro-Wilks tests at a confidence interval of 95%, respectively) were performed to assure an appropriate measure of central tendency was selected to represent m_f and n_f. If normality was not present, outliers were searched for, with outliers treated as 1.5 times the inter-quartile range relevant to first and third quartiles, respectively. If outliers were detected metadata was consulted to determine if they should be removed and, if removed, normality was re-evaluated. As an interesting aside, because sediment transfer from field to ditch was attributable to several sedimentary features specifically related to water, wind, and tillage erosion, measurements of sediment transferred from field to ditch could be compared to parsed rates of sediment production in fields. This appears to be novel among the few papers investigating sediment transfers into road-side ditch networks (Pease *et al.*, 2003; Lecce *et al.*, 2006).

Due to time constraints and the number of sediment transfers considered, a long-term survey documenting the sediment transfers active in and near road-side ditches, while desirable, was not possible. If a particular sediment transfer was observed throughout the survey period (but not necessarily measured), its annual occurrence was assumed to be routine. However, its important to remember that a routine transfer may be in part characterized by sedimentary features less routinely observed. This necessitates that the infrequency of such sedimentary features be considered and their associated transfer rates aggregated with the transfer rates of routinely observed features for a given sedimentary transfer process. In the case of this study, this applied to blow dirt, which was assumed to be representative of years characterized by high rates of wind erosion. It was assumed such years only occurred once over the 30-year period considered by NAHARP soil erosion risk estimates. Other, routine sediment transfers characterized by routinely observed sedimentary features were measured over several years but, because rates of sediment transfer associated with such processes were generally assumed to be steady, the measurements were treated as temporally equivalent and analyzed together to improve "annual" sample size.

2.3.1.3. Sedimentary Processes in Streams

Annual suspended sediment discharges through streams were calculated from daily stream discharges and event-based TSS concentration measurements. Daily stream discharge data was retrieved from the WSC, and event-based TSS concentration data was retrieved from the DSWMA and AAFC for the 05OF024 and 05OG008 sub-watersheds, respectively. Total suspended solid concentration sampling was conducted from 2014 to 2016 in the 05OF024 sub-watershed and from 2010 to 2015 in the 05OG008 sub-watershed, inclusive. These samples were collected at a higher-intensity basis (multiple samples per week) during high-flow events and scaled back to sub-monthly frequency under low-flow conditions. Measurements and samples were collected at each sub-watersheds' respective WSC station with the exception of TSS concentration measurements in the 05OF024 sub-watershed, which were collected at the DSWMA site RSE-1. This site is 200 m downstream of the WSC station and was judged to be representative of sediment flux through that portion of the La Salle River (*Pers. Comm.*, Les McEwan, DSWMA).

The mass of sediment transported as bedload was not measured. This limitation of the available data is likely a minor issue in the overall characterization of sediment discharge at the sub-watershed outlets. Previous study of spring sediment loads in the Red River and a sub-set of its tributaries near Fargo, North Dakota found only 0.1 to 4.9‰ of total sediment carried under high-flow conditions to be attributable to bedload transportation (Blanchard *et al.*, 2011). This observation tracks with the assertion by Walling (1994) that bedload transportation is often trivial when characterizing soil erosion at the watershed-scale.

Annual suspended sediment discharges were calculated by summing daily suspended sediment discharges; determined by multiplying daily stream discharge and TSS concentration measurements as described by Porterfield (1972). Because only event-based TSS concentration data were available, it was necessary to estimate unmeasured daily values. Meals *et al.* (2013) have advocated two basic estimation methods that have been implemented by others (Blanchard *et al.*, 2011): linear regression and interpolation. Both were explored to determine which was most appropriate.

To address the effectiveness of linear regression, an attempt was made to draft sediment rating curves regressing known, $\log_{10}$-transformed TSS concentrations onto corresponding $\log_{10}$-transformed daily stream discharge measurements. To validate the assumptions required for linear regression a pair of Shapiro-Wilk tests were performed for each $\log_{10}$-transformed data set to ensure normality, followed by additional visual inspection of the data plotted as histograms and Q-Q plots. Once transformed data were shown to be sufficiently normal at a significance level of 95%, outliers were searched for and, if absent, the relationship between daily stream discharge and measured TSS concentrations were quantified by calculating their Pearson correlation coefficients. This was done to verify that stream discharge was potentially predictive of TSS concentration. Only $\log_{10}$-transformed data from the 05OF024 sub-watershed was sufficiently normal and free of outliers to permit regression of stream discharge onto TSS and analysis of correlation between the same variables. The later characteristic of the data was found to be moderate ($r = 0.39$ and $p =$

5.5×10^{-4}). Regression was performed to explore this relationship further (Fig. 2.14). A coefficient of determination (R^2) was calculated for the resulting sediment rating curve to evaluate model strength. The calculated R^2 value was low, suggesting daily discharge is a poor predictor of TSS. Therefore, the interpolation was used to estimate TSS concentration. Following this method, known TSS concentrations were arranged in a time-series, and linear interpolation was used to estimate values between them. It is likely that poor performance of modeled estimates can be linked to hysteretic effects (Tab. 2.5) and abnormally high TSS concentrations during the freshet (Fig. 2.15). These effects were expected to be greater in the smaller, unmodelled 05OG008 sub-watershed. Understanding that there are hysteretic effects between related measured stream discharges and TSS concentrations, it is possible that linear interpolation may miss peak TSS concentrations and, in turn, lead to underestimation of daily TSS values. Such underestimation is likely minimal, due to autocorrelation between sequential measurement of TSS concentrations.

2.3.2. Sediment Budget Validation

Even when carefully constructed, sediment budgets possess a high degree of uncertainty. Though it is not unheard of to perform uncertainty analysis in conjunction with other types of budgeting exercises (LaBaugh and Winter, 1984) and has been advocated for when sediment budgeting (Minella *et al*., 2014), such supplemental analyses are only possible when studies are based in extremely mature monitoring programs. In the case of this study, insufficient data was available for a detailed uncertainty analysis. However, the sediment budgets for the 05OF024 and 05OG008 sub-watersheds can still be validated. This was achieved through quantitative and qualitative trend analysis.

To verify that comparisons between the sediment transfers considered by the 05OF024 and 05OG008 sediment budgets were valid, trends in the processes related to these transfers were searched for. If such trends (or a lack thereof) were similar, it could be said that there was a degree of certainty in the sediment budgets, even if the degree of certainty could not be quantified. How trends were assessed depended on the temporal breadth and resolution of the data used to quantify the sedimentary processes considered by the sediment budgets.

Trends in processes quantified using broad, high-resolution data sets could be identified statistically. This applied to sediment transfers from field to stream and from ditch to stream inferred through measurement of suspended sediment loads in streams, with one important proviso. Because suspended sediment loads were calculated from interpolated daily TSS concentrations, daily suspended sediment discharge could not be used to directly identify trends. Instead, knowing that daily suspended sediment discharge was the product of daily stream discharge measurements and interpolated daily TSS values, daily stream discharge measurements were analyzed and considered alongside the strength of the previously discussed relationships between daily stream discharge and event-based TSS measurements. Daily stream discharge values between 1981 (the first census year of NAHARP soil erosion risk estimates) and 2018 (the final year of the road-side ditch survey) were analyzed for the 05OG008 sub-watershed while the range of dates analyzed for the 05OF024 sub-watershed were truncated from 2000 to 2018 due to data availability. Prior to statistical analysis stream discharges were plotted to search for gaps in the data or evidence of seasonal cycles, both which have bearing on the statistical methods to be selected. Both were found. In the 05OF024 sub-watershed, conspicuous daily gaps were noted from 2000 to 2009 and in the 05OG008 sub-watershed, from 2002 to 2009 with further annual gaps in 1989 and from 1997 to 2001 (Figs. 2.16 and 2.17). Discharges are seasonal, being highest in April with smaller discharges following in the summer and fall. Following American Environmental Protection Agency guidelines (Meals *et al.*, 2011), a seasonal Mann-Kendall trend test is at first glance best suited to these data sets. However, further examination of plotted stream discharge data reveals a longer dry-wet-dry cycle over the monitored years. This cycle is most apparent as a wet period between 2004 and 2009 in the 05OG008 sub-watershed and mirrored in the 05OF024 sub-watershed in 2005. As such, to better address the goal of trend analysis within the scope of this study and to avoid complications associated with disentangling trends of different temporal scales, a Mann-Kendall trend test was performed using the Kendall 2.2 package in R (McLeod, 2011). To prepare discharge data for analysis daily values were re-aggregated into annual medians, simultaneously smoothing seasonal stream discharge cycles and reducing autocorrelation. Results of Mann-Kendall trend tests were evaluated at a 95% confidence interval.

When the rates of processes were derived from long-term data which lacked sufficient temporal resolution to permit statistical trend analysis, trends were identified visually. This approach was applied to sediment transfers in fields via water, wind, and tillage erosion. To identify such trends, maps illustrating the difference in erosion risk between 1981 and 2011 were plotted, similar to those presented by Lobb *et al.* (2016) when discussing changes in soil erosion risk over the same period across Canada. In the case of this study, mapping differences in erosion risk (i.e., yields) is acceptable because it has been assumed the extent of erodible landforms in cultivated fields is invariant with time. If this assumption was not made, it would be more appropriate to consideration changes in discharges. Furthermore, by considering differences in soil erosion risk, differences can be discussed in terms of established figures describing acceptable rates of change in soil loss (Lobb *et al.*, 2016), landscape erodibility, and sedimentary process erosivity.

Trend analysis was not attempted when the data characterizing a sedimentary process lacked breadth (sedimentary processes active in or near road-side ditches) or was unmeasured, instead being inferred through the differences between measured sedimentary processes (depositional processes in fields). The former scenario follows the assumption that such processes were steady, as previously detailed, while the latter acknowledges that trends in processes inferred through differences will track with trends in the measured rates they are derived from, making any discussion of such trends in the process meaningless.

2.4. Results

2.4.1. Sediment Transfers in Fields

Average annual sediment transfers from eroding hillslopes in cultivated fields between 1981 and 2011 in the 05OF024 and 05OG008 sub-watersheds are presented in Table 2.6. Maps illustrating the differences in soil erosion risk reveal little difference in rates of sediment transfers over this period, at the resolution used by Lobb *et al.* (2016). Across the sub-watersheds, rates of water erosion have remained constant (Fig. 2.18). Similarly, rates of wind and tillage erosion have remained constant in all but a small number of landscapes, which have increased and decreased, respectively (Figs. 2.19 and 2.20). Increases in wind erosion are limited to landscapes in the lower reaches of the 05OF024 and 05OG008 sub-watershed, while decreases in tillage erosion are limited to head-water landscapes of the 05OF024 sub-watershed. In the 05OF024 sub-watershed these differences have been 1.03 t ha^{-1} yr^{-1} for wind erosion and range from -3.63 to -1.07 t ha^{-1} yr^{-1} for tillage erosion, both between 1981 and 2011. In the 05OG008 sub-watershed, the difference in wind erosion over the same period has been 1.03 t ha^{-1} yr^{-1}. Accounting for landscape area, this amounts to differences in sediment discharged from erodible hillslopes by wind erosion of 148 and 956 t yr^{-1} in the 05OF024 and 05OG008 sub-watersheds, respectively, and differences in discharges related to tillage erosion ranging from -6445 to -1 t yr^{-1} in the 05OF024 sub-watershed. While inconsequential within the scope of this study, brief discussion of the driving forces behind these trends (or lack thereof) can enhance discussion of sediment transfers in fields and their relationship with their corresponding sediment budgets.

In both the 05OF024 and 05OG008 sub-watersheds, the majority of landforms have little to low slope gradient. The influence of slope, as characterized by its respective factor in USLE-based water erosion models, is described as a quadratic function. As such, slope has a greater impact on soil erosion (the product of erosivity and erodibility) than slope length, (vegetative) cover-management practices, and supporting (agronomic) practices. Therefore, in any changes relating to WatERI's slope length, cover-management practices, and supporting practices factors between 1981 and 2011 are likely to have been muted by the sub-watersheds' naturally low relief. Differences in rates of wind and tillage erosion can also be related to their respective model factors and their relationships with soil erodibility and process erosivity. In both sub-watersheds, the landscapes where wind erosion risk has increased are characterized by level landforms and clayey soils. These characteristics, as encoded in the SLC geodatabase, are used to calculate WindERI factors representing soil erodibility, soil ridge roughness, and field length along the prevailing wind direction. These factors do not change with time, and are similar in nearby landscapes where wind erosion risk has not changed. Therefore, it is more likely that changes in wind erosion risk are related to WindERI's time-continuous factors. Such changes are unlikely to be related to WindERI's climate factor, with wind speeds and directions remaining constant in the La Salle River watershed (Fig. 2.4) and average annual precipitation in the 2011 census year being similar to (but more variable than) the 1981 census year (Fig. 2.3). Changes in wind erosion risk may be more related to changes in WindERI's equivalent vegetative cover factor. This is evidenced by changes in cropping patterns between 1981 and 2011, in particular increased planting of oilseeds since the mid-2000s (Fig. 2.7). Such changes have the potential to increase soil erodibility, while

WindERI's other factors remain comparatively constant. This scenario, where increased wind erosion risk is driven by increasing landscape erodibility is contrasted by changes in tillage erosion risk in the 05OF024 sub-watershed. There, the risk of tillage erosion has decreased in landscapes characterized by hummocky landforms, which in turn are characterized by slopes with high gradients and degrees of curvature compared to other landforms in the region. These changes in tillage erosion risk may be attributable to increased adoption of less erosive tillage sequences characterized by conservation and no-till tillage operations (Fig. 2.9), the impacts of which would be most pronounced in the sub-watershed's highly erodible head-water landscapes.

2.4.2. Sediment Transfers in and near Road-Side Ditches

Parameters used to calculate average annual sediment transfers into, out of, and within road-side ditches are summarized in Tables 2.7 and 2.8, and the resulting average annual rates are presented in Table 2.9. Because these transfer rates are based in measurements made between 2014 and 2018, and thus lack temporal breadth, no attempt was made to qualify or quantify trends in these rates. However, the nature of the data collected to quantify these rates merits discussion, as it may have bearing on how these rates can be compared to sediment transfer rates in fields and through streams.

The number of samples used to calculate $\overline{m}_f$ varies greatly between documented sedimentary features (Appendix B). Typically, small, easily sampled sedimentary features were characterized by a greater number of samples than larger features or features that were rarely observed. This was not deemed to be problematic, as calculated values of $\overline{m}_f$ span four orders of magnitude, which any impacts of small sample sizes are unlikely to overshadow. Values of $\overline{n}_f$ show little cause for concern in regards to sample size and value of l_{ab} were derived from well-vetted geospatial datasets. The cause for greatest concern lies with the data used to derive some values of p_{abc}. Data used to characterize values of p_{abc} for dredged spoil piles, sediment fans, and blow dirt in particular were based in records kept by the RM of MacDonald Public Works. These data are of high temporal and spatial quality, detailing ditch maintenance and blow dirt clearing from 2000 to 2017 and account for all road-side ditches in the RM. These data are representative of the sedimentary processes related to these features in the eastern-most reaches of the Boyne-Morris and La Salle River watersheds, where soils are predominantly fine-textured. Similar data could not be found for areas characterized by medium- and coarse-textured soils, necessitating values of p_{abc} for fine-textured soils being extrapolated to these regions. This may be problematic, as sub-watersheds were divided into catchments on the basis of soil texture following the assumption soil acted as a broad control over rates of sediment transfers within fields and in and near ditches. As such, it is more than likely that true values for p_{abc} for coarse- and medium-texture catchments differ from those which were extrapolated from fine-texture catchments. This does not entirely discount transfer rates calculated for these processes, as both sub-watersheds are predominantly underlain by fine-textured soils.

2.4.3. Sediment Transfers in Streams

Average annual sediment transfers through the outlets of the 05OF024 and 05OG008 sub-watersheds summarized in Table 2.10. Mann-Kendall trend testing does not suggest a significant trend in stream discharge in the 05OG008 sub-watershed between 1981 and 2018 (τ = -0.11 and p = 0.25) nor in the 05OF024 sub-watershed between 2000 and 2018 (τ = 0.0064 and p = 0.92). The relationships between $\log_{10}$-transformed stream discharge and $\log_{10}$-tranformed TSS measurements were only modellable for the 05OF024 sub-watershed, and the predictive power of that model was found to be poor (Fig. 2.14). This seems to suggest that while it can be said that stream discharge has not changed, it cannot be said confidently that sediment transfer through streams has behaved in a similar manner. However, it is important to recall it was previously proposed the poor performance of the 05OF024 sub-watershed's sediment rating curve (and for that matter, failure to draft a sediment rating curve for the 05OG008 sub-watershed) was due to high TSS measurements through the freshet and hysteretic effects between model variables. These discharge characteristics are seasonal, while trend testing focussed on aggregated year-to-year patterns. It is then within the realm of possibility that, if longer records TSS measurements existed, related model performance would both improve and better speak to year-to-year sediment discharge patterns. However, this point may be moot, as longer records of TSS measurements would permit direct Seasonal Mann-Kendall trend testing of suspended sediment discharges. Knowing this, it was assumed (albeit, conveniently) that no trends in sediment discharge out of the 05OF024 and 05OG008 sub-watersheds existed between 1981 and 2018, following the absence of significant trends in stream discharge.

This stability is interesting when considered alongside more temporally-dynamic watershed and sub-watershed characteristics. Watershed-wide changes in precipitation, cropping patterns, and tillage systems between 1981 and 2011 do not seem to have meaningful impacts on sediment transfers through streams; nor do the lesser (through previously judged to be inconsequential) sub-watershed-wide differences in sediment transfers from erodible landforms in cultivated fields over the same period. This contrast is meaningful, as it suggests a buffer between sediment discharges in fields and sediment discharges through streams and has implications for how the following sediment budgets are interpreted.

2.4.4. Sediment Budgets

The previously presented annual average annual sediment transfers were tabulated in terms of sediment inputs (I), outputs (φ), and storage (S) that relate to them (Tab. 2.11). These tabulated values were then used to draft Sankey diagrams illustrating the sediment budgets for the 05OF024 and 05OG008 sub-watersheds (Figs. 2.21 and 2.22, respectively). Sankey diagrams were drafted in LaTeX using code written by Gaborit (2013). On their own, any one component of the sediment budgets reveals little about the nature of sedimentary processes in the sub-watersheds. It is through comparison of these components, made possible by the sediment budgeting technique, that a fuller understanding of the nature of sedimentary processes in the sub-watersheds can be gained.

2.4.4.1. Sediment Delivery Patterns

Patterns in sediment delivery were found to be similar in the 05OF024 and 05OG008 sub-watersheds, be it from field to stream, road-side ditch, or in-field depression. Sediment inputs from erodible landforms in cultivated fields were 3 orders of magnitude larger than sediment outputs through streams. This finding was expected, given the size and generally low relief of the sub-watersheds, and its anticipation motivated this study's investigation of intermediate sediment storage in the sub-watersheds. Sediment storage in road-side ditches was found to be 1 order of magnitude smaller than sediment inputs from erodible landforms in cultivated fields. Much of this stored sediment was related to these inputs, with sediment storage in road-side ditches related to inputs from adjacent gravel roads and reworking of ditch sediment being 1 and 2 orders of magnitude smaller than sediment storage in road-side ditches related to inputs from erodible landforms in cultivated fields, respectively. Storage in depressions in cultivated fields (calculated as the difference between inputs from erodible landforms in fields and the sum of storage in ditches related to inputs in fields and sediment outputs through streams) was found to be smaller than inputs from erodible landforms in fields but of the same order of magnitude. This presents a pattern of decreasing sediment delivery with increasing distance between sediment inputs and outputs. Such a finding is, again, expected following the understanding that SDRs are inversely proportional to basin area (Walling, 1994). By following this expected pattern, the sediment budgets' SDRs contribute to the apparent validity of the sediment budgets, however such an interpretation should not be considered without caution. First, it can be argued that by determining sediment storage in cultivated fields through subtraction, the magnitude of this budgetary component is led to a value that is naturally between that of sediment inputs from erodible landforms in fields and outputs downstream. While this is undeniable, by selecting a conservative estimate of sediment production in fields, the likelihood of sediment storage in cultivated fields being substantively larger than its true value is reduced. Second, this assessment of the sediment delivery assumes sediment deposited in road-side ditches by water, wind, and tillage is directly related to wind, water, and tillage erosion of erodible hillslopes in cultivated fields characterized by the SLC geodatabase. This assumption denies the possibility that other landforms in fields (e.g., surface drains and field edges) may be eroded by water, wind, or tillage. Though the first of these two precautions is difficult to address without measuring sediment deposition in cultivated fields, the second can be addressed within the scope of this sediment budgeting study. This can be achieved by first considering the relationships between the sedimentary processes producing sediment in cultivated fields.

2.4.4.2. Sediment Production Patterns

Sediment produced by erodible hillslopes in cultivated fields can be better understood by parsing soil erosion into its components driven by water, wind, and tillage; as they were calculated. In the 05OF024 and 05OG008 sub-watersheds, the proportionality between these components differs, though all are of the same order of magnitude. In 05OF024 sub-watershed rates of soil erosion decrease from wind erosion to tillage erosion to water erosion, while in the 05OG008 sub-watershed wind erosion still dominates, but water erosion is greater than tillage erosion. This difference can be attributed to differences in the proportionality of landforms in the watersheds and the proportionality between the upper- and middle-slope segments in said landforms. The landscapes of the 05OG008 sub-watershed are predominantly characterized by level landforms with little to no slope (Tab. 2.2), of which upper- and middle-slope segments comprise 10% and 35% of the landform's total slope length. Though there are a number of similar landscapes in the 05OF024 sub-watershed, other landscapes are primarily characterized by hummocky, gently sloping landforms, characterized in turn by upper- and middle-slope segments that each account for 32% of the landform's total slope length. Recalling that rates of sediment discharge from eroding hillslopes in cultivated fields were calculated assuming wind and tillage erosion only impacted upper-slope segments and water erosion only impacted middle-slope segments, these differences in landscape characteristics would explain differences in the proportionality between sediment produced by water, wind, and tillage erosion in the two watersheds. However, the proportionality between these values should be considered against published limitations of the models they are derived from. It has been stated that there is likely a degree of overestimation of water and wind erosion by SoilERI due to the model ignoring erosion control practices including grassed waterways, strip cropping, terracing, contour cultivation, shelterbelts, and winter cover crops (Lobb *et al.*, 2016). Shelterbelts are commonly adopted in both the Boyne-Morris and La Salle River watersheds and winter cover crops are planted, though to a lesser degree. Consequently, wind erosion rates may be overestimated. It has also been stated that SLC geodatabase landform definitions may contribute to additional overestimation of water erosion and underestimation of tillage erosion of erodible hillslopes in cultivated fields (Lobb *et al.*, 2016). Though the impact of SoilERI ignoring certain erosion practices cannot by commented on, due to this study's methods, the impact of SoilERI's reliance on SLC landform definitions can be. To do so, the previously presented rates of sediment production can be compared with rates of sediment storage in ditches in a process-wise manner. If the patterns are similar, the assumption that deposition is related to erosion of erodible hillslopes in cultivated fields defined by the SLC geodatabase stands. If not, it is possible that sediment deposition in road-side ditches has a stronger relationship with water, wind, and tillage erosion of other erodible landforms in cultivated fields not defined by the SLC geodatabase. This, ultimately affects how sediment delivery illustrated by the budgets is discussed within the scope of land-management.

2.4.4.3. Sediment Storage Patterns

As when interpreting patterns in sediment production, the patterns in sediment storage in road-side ditches are most easily understood when rates of sediment deposition are separated into components related to erosive processes, namely: water, wind, and tillage erosion in the adjacent cultivated field, erosion of adjacent road surfaces, and bioturbation within road-side ditches by burrowing mammals. The proportionality between these modes of deposition is the same in both the 05OF024 and 05OG008 sub-watersheds. The bulk of deposition in road-side ditches can be related to tillage erosion, followed in descending order by deposition related to wind and water erosion. The rate of sediment deposited by tillage erosion was approximately 4-times greater than that deposited by wind erosion which, in turn, was approximately 20-times greater than the rate which sediment was deposited by water erosion. As was previously stated, road surface erosion and bioturbation made additional small, but not trivial, contributions to sediment storage in road-side ditches. The proportionality between rates of deposition related to water, wind, and tillage erosion do not agree with the previously presented rates of erosion impacting erodible hillslopes in cultivated fields. This suggests that sediment deposited in road-side ditches may not be related to sediment produced in the erodible landforms characterised by the SLC geodatabase. This raises the question of the degree of erosion risk in these uncharacterized landforms in cultivated fields. It also raises questions about sediment delivery from landforms in cultivated fields in general to road-side ditches and streams. In both cases, these questions are based in empirical measurements of sediment deposition in road-side ditches, which links to and lends credibility to all prior interpretation of the presented sediment budgets. However, before the implications of these findings can be discussed within the context of land-management one last feature of the 05OF024 and 05OG008 sub-watershed sediment budgets requires summary: the inclusion of measured ditch maintenance activities (discussed below).

2.4.4.4. Other Patterns

A unique characteristic of the 05OF024 and 05OG008 sub-watershed sediment budgets is the inclusion of ditch dredging by RMs to maintain connectivity between fields and streams. In both sub-watersheds, this process input an order of magnitude more sediment into cultivated fields than was output from erodible landforms in the same fields by wind, water, and tillage erosion. Though the exact values ascribed to ditch dredging should be treated with caution, due to limitations related to their calculation described in Section 2.4.2., the magnitude of the process raises further questions about sediment delivery in 05OF024 and 05OG008 sub-watersheds.

2.5. Discussion

Potential sources of sediment budget uncertainty were identified when calculating components of the 05OF024 and 05OG008 sub-watershed sediment budgets. These sources can be categorized as uncertainty in values of Q_{abc} and uncertainty in values of $\bar{Q}_{ab}$, and are generally summarized in Sections 2.3.1.1.-2.3.1.3. and 2.3.2.-2.4.3., respectively. Such uncertainty was deemed acceptable, having little impact on sediment budget interpretation. However, further uncertainty was identified when balancing the sediment budgets and related to imbalances between water, wind, and tillage erosion in cultivated fields and related deposition in road-side ditches (Sections 2.4.4.2. and 2.4.4.3). It was hypothesised in Section 2.4.4.3. that such an imbalance was due to erosion of landforms in cultivated fields not characterized by the SLC geodatabase and processes that may or may not be characterized by NAHARP soil erosion risk estimates. Though the sediment budgets presented by this study are of sufficient breadth to discuss the state, impacts, and management of soil erosion in the 05OF024 and 05OG008 sub-watersheds, such a discussion should not take place without first discussing this imbalance, as its understanding bears on the state of soil erosion on which the assessment of impacts and mitigation practices follow. Discussion of the state of soil erosion, its impact, and its management can be further enhanced by considering past, relevant studies.

Landforms and erosive processes relating to water and tillage erosion not characterized by NAHARP soil erosion risk estimates can be identified by considering the processes that are (or are not) characterized by the SoilERI model, its relevant inputs, and their relationship with related measurements of soil erosion and deposition. This approach cannot be applied to uncertainty in wind erosion estimates, due to its risk being better predicted by cropping patterns not used to draft the sediment budgets presented by this study. As such, potential inaccuracy in the wind erosion component of SoilERI (WindERI) will not be discussed further, as it is better addressed by other types of studies. The water erosion component of SoilERI (WatERI) characterizes rill and inter-rill erosion, while the tillage erosion component (TillERI) of the same model characterizes forward tillage translocation. Both WatERI and TillERI predict high rates of soil erosion in highly erodible landscapes characterized by steeply sloping, convex landforms. Surface drains in cultivated fields exhibit these landform characteristics perpendicular to their direction of flow, suggesting they may be subject to rill erosion, inter-rill erosion, and forward tillage translocation. Such slopes insufficiently concentrate flow to permit gully erosion and lateral tillage translocation is likely to occur but reduced by opposing tillage passes in a tillage operation. The severity of water and tillage erosion affecting these landforms is not known. The proportion of landscapes occupied by surface drains in the Boyne-Morris and La Salle River watersheds is not precisely known, but a 2013 study by MacLoed suggested a typical Dominion Land Survey section in the region is drained by approximately 20 cumulative kilometers of surface drains, which the Government of Manitoba suggests should not be graded more the 10% along their shoulder slopes (MAFRD, 2013). Field edges may also contribute to water and tillage soil erosion in cultivated fields not characterized by NAHARP erosion risk estimates. Though field edges do not differ substantively from surrounding landforms characterized by the SLC geodatabase in terms of erodibility, the manner in which fields edges erode do. As surface drains approach field edges, flow accumulation can lead to localized gully erosion unaccounted for by WatERI. Similarly, over multiple tillage passes lateral tillage translocation not modelled by TillERI can drive soil to the field edge's boundary with the road-side ditch network. Gullying is both measurable and modellable, but is typically done on a study-by-study basis. Generalized modelling of lateral tillage translocation has been explored, with Li *et al*. (2009) linking lateral tillage translocation along field edges, modelled with the DirTillEM model, to measured "bowling-out" of cultivated fields in the Red River Valley in south-eastern Manitoba. The "rims" around such fields were predicted to be 0.2 m tall and 5.0-10.0 m wide after 20 years of continuous tillage, compared with measured heights and widths of 0.15 and 15.0 m derived from LIDAR data, respectively. These results were based in the assumption that the rate of lateral tillage translocation is approximately one half of the rate of forward tillage translocation (Lobb and Kachanoski, 1999). Future characterization of surface drain and field edge landforms as well as evaluation of the rates which they erode when subject to erosive processes driven by water and tillage may be pertinent. However, within the scope of this study their identification is sufficient to discuss the state soil erosion, its impacts, and its management in the 05OF024 and 05OG008 sub-watersheds.

Past studies investigating soil erosion in the Red River watershed have been motivated by the potential of eroded soil to impact the water quality of Lake Winnipeg. Of particular interest has been the relationship between eroded soil and the increased frequency and extent of cyanobacterial blooms in the lake's south basin since the late 1990s. Low SDRs measured at the outlets of the 05OF024 and 05OG008 sub-watersheds over the period represented by this study's sediment budgets suggest downstream sediment transfers from the sub-watersheds, and, by extension, their impact on Lake Winnipeg's water quality, are minimal. This assessment is strengthened by a previous sediment fingerprinting study in the 05OF017 sub-watershed; nested within the 050F024 sub-watershed (Koiter *et al.,* 2013). In Koiter *et al.*'s 2013 study, it was demonstrated that the contribution of soil to suspended sediment loads in streams decreased with increasing basin size. In small, headwater basins upstream of WSC station 05OF017, soil contributed to 64-85% of suspended sediment load. This contribution steadily decreased with increasing basin size, with soil only contributing to 13-33% of suspended sediment load ~3 km upstream of the watershed's outlet. At the outlet, soil contributed to 31-44% of total suspended sediment load. These observations were interpreted as a general decrease in soil delivery to the basin outlet, proportional to sediment delivery from other sediment sources, as the length of channels feeding the outlet increased with basin size. The reversal at the watershed outlet was assumed to be due to a greater number of road-side ditches linking fields and streams, reducing the distance travelled by sediment between source and outlet. These were cited by the researchers as examples of decreasing and increasing sediment cascade connectivity, a concept described by Fryirs (2013). A re-examination of the study by Barthod *et al*. (2015) using a different set of tracers suggested the contribution of sediment sources to total suspended sediment load measured near the watershed outlet was closer to 3%. If one considers the sediment yields of the 05OF017 and 05OF024 sub-watersheds (which relate to sediment delivery and are 0.08 t ha^{-1} yr^{-1} and 0.008 t ha^{-1} yr^{-1}, respectively) it becomes apparent that on average the potential impact of eroded soil, and not just sediment, on water quality downstream of WSC station 05OF024 may be minimal. The sediment yield of the 05OG008 sub-watershed (0.004 t ha^{-1} yr^{-1}), suggests this assessment holds true for both basins. This presents a disconnect between the public's perception of the severity of soil erosion at the watershed-scale (based in unquantified observations of sediment transfers without comparison) and its state quantified by this study.

An interesting side effect of the relationship between basin size and SDR is that as SDR decreases, so too does the impact of changes in soil erosion on downstream water quality. This was previously alluded to in Section 2.4.3., where it was noted that suspended sediment discharges through the outlets of the 05OF024 and 05OG008 sub-watersheds remained constant, despite changes in rates of soil erosion and its drivers. This presents a critical question that must be asked when planning management of downstream impacts of soil erosion: how large is the basin upstream of where the impact is assessed? Basins as large as the 05OF024 and 05OG008 sub-watersheds may naturally mitigate the impacts of soil erosion downstream of their outlets through scale alone, whereas smaller, headwater basins may require management strategies that directly reduce their naturally higher SDRs. In basins that are smaller still (such as those defining sediment delivery from cultivated fields to road-side ditches or even within cultivated fields themselves), the most pertinent means of managing the downstream impacts of soil erosion may be directly managing soil erosion in cultivated fields. From this, it can be concluded that while soil erosion's impacts on water quality downstream of WSC stations 05OF024 and 05OG008 may typically be minimal, soil erosion may have meaningful impacts downstream of smaller basin outlets within the sub-watersheds. What these impacts may be are alluded to by stream discharge management decisions that have been implemented in the 05OF017 sub-watershed.

Since 1979, 26 small dams have been constructed in the 05OF017 sub-watershed. Though specifically designed to reduce peak stream discharges to prevent damage to downstream infrastructure, they have also stored eroded sediment. Measured at two large multi-purpose dams in the sub-watershed between 1999 and 2007, inclusive, such storage amounted to 68-69% yr^{-1} and 65-83% yr^{-1} less suspended sediment delivered downstream of the dams after the freshet and rainstorms, respectively (Tiessen *et al.*, 2011). In Koiter *et al.*'s 2013 sediment fingerprinting study, it was noted that road-side ditches were periodically dredged to remove deposited soil and vegetation to promote rapid drainage and increase ditch capacity, in turn enhancing agricultural productivity of otherwise poorly drained land. These practices suggest that in the smaller basins of the 05OF017 sub-watershed, the impact of soil erosion on the continued function of flow-control infrastructure (i.e., loss of capacity through deposition in ditches and reservoirs) is of greater concern than its impact on water quality; impacts that may extend to other small basins in the 05OF024 and 05OG008 sub-watersheds. In the case of deposition in reservoirs, this impact of soil erosion may already be addressed by the capacity of the 24 smaller dry flood-control and back-flood dams constructed upstream of the studied multipurpose dams to reduce SDRs to the multipurpose dams' reservoirs. Addressing ditch sedimentation is more difficult. Ditch dredging to promote rapid drainage and increase ditch capacity was identified in the 05OF024 and 05OG008 sub-watersheds and quantified in their respective sediment budgets. Koiter *et al.* (2013) noted the ability of ditch dredging to affect SDRs to road-side ditches. Though the 05OF024 and 05OG008 sub-watershed sediment budgets demonstrate ditch dredging does reduce the SDR between cultivated fields and road-side ditches, its recommendation as a management practice to reduce sediment delivery cannot be made for two reasons. First, removal of ditch vegetation by dredging increases the erodibility of the sediment that remains in the ditch, increasing downstream SDRs until vegetation can re-establish. This may negatively impact downstream flow-control infrastructure, such as culverts or the previously mentioned small dams. Second, as quantified by this study, dredging by excavators far exceeded rates of sediment deposition in road-side ditches. This may be problematic, as excessive dredging may cut into lower, nutrient poor soil horizons. Once bulldozed into adjacent fields, spoil may bury or dilute more productive soils. As such, the impact of sediment delivery to road-side ditches from cultivated fields may be better addressed through direct management of soil erosion in cultivated fields.

Past management of water and tillage erosion in cultivated fields has focussed on reducing soil erosion to a tolerable rate. These efforts have included many practices summarized by the Canadian Census of Agriculture, including: grassed waterways, strip cropping, terracing, contour cultivation, and adopting of conservation or no-till tillage systems. The efficacy of these practices has been judged against a tolerable rate of soil erosion balanced with liberal rate of A horizon formation: 5 t ha^{-1} yr^{-1} in medium- to moderately coarse-textured, cultivated soils. This rate of A horizon formation is lower in fine-textured, cultivated soils (Hall *et al.*, 1979). Such guidelines far exceed a more meaningful rate of pedogenesis of 1 t ha^{-1} yr^{-1}, based in the development of favorable rooting zone from unconsolidated parent material (McCormack *et al.*, 1979). It stands to reason that while management practices implemented to date have slowed rates of water and tillage erosion, they have not effectively mitigated its cumulative in-field impacts. Topsoil replacement may be a management practice capable of addressing this problem.

Topsoil replacement is any management practice that relocates deposited topsoil within a landscape upslope to its source; simultaneously reducing downslope SDRs and restoring soil fertility in eroded landforms. At small scales, this is already practiced by landowners in the Boyne-Morris and La Salle River watersheds. There, it is common practice to routinely dredge surface drains in cultivated fields to maintain desirable peak discharges, with dredged soil deposited upslope of the surface drain's flowpath. This is comparable with road-side ditch dredging observed by Koiter *et al.,* (2013), but is typically performed using rotary ditchers or lighter equipment capable of precise, shallow dredging. Li *et al.* (2009) suggested tillage patterns that laterally translocate soil towards the center of fields from field edges may reduce "bowling-out" of fields, which in turn may reduce soil deposition in road-side ditches and rates of road-side ditch dredging. Such applications of topsoil replacement require accurate estimation of rates of erosion impacting surface drains and field edges to determine appropriate degrees of re-grading and, in the case of restoration of field edges, requires mowing and baling to manage ditch vegetation. More concretely, topsoil replacement has been demonstrably effective for addressing erosion of larger landforms, such as those classified by the SLC geodatabase. In a 2005-2011 study by Schneider *et al.* (2021) in Minnesota, wheel-tractor scrapers were used to translocate soil deposited in depositional-slope segments to mechanically weathered upper-slope segments. Landforms studied were undulating with gentle slopes and subject to tillage sequences predominantly characterized by mouldboard plowing with secondary tillage operations over the past century. Summed rates of water and tillage erosion and deposition over this period were estimated to be 130 t ha^{-1} yr^{-1} and >100 t ha^{-1} yr^{-1}, respectively. Fifteen to twenty centimeters of soil was scraped from the 46 cm of eroeded soil deposited over the original A horizon in depositional-slope segments and translocated upslope. Substantive crop biomass and yields increases in upper-slope segments were observed while small decreases were noted in depositional-slope segments and attributed to soil removal and plot design. Improvements in crop biomass were noted when experiments by the same researchers were conducted at a less eroded site in South Dakota. If applied to less rugged landscapes in Red River Valley, the depth and frequency of scraping would likely be less, due to lower known rates of soil erosion risk. Actual rates of soil erosion from field to field in a landscape may differ from the estimated rates of soil erosion risk within that landscape. As such, the exact depth and frequency of scraping operations when needed cannot be determined in a prescriptive manner.

Even when the state of soil erosion is understood, its impacts are assessed, and management options have been identified, the implementation of a watershed-wide erosion management plan can be difficult. Management of one impact of soil erosion may exacerbate another, requiring decisions to be rooted in a sound understanding of the sedimentary processes that impact soil in the watershed of interest. Sediment budgeting is one of many research methods that can be applied alongside soil erosion modelling (Li *et al*., 2009), fingerprinting studies (Koiter *et al.*, 2013), environmental impact assessment (Tiessen *et al*., 2011), and development of management techniques (Schneider *et al.*, 2021) to guide such decisions. The economic cost to employ management decisions to address one impact of soil erosion may need to be balanced with the costs of addressing or neglecting others. In the studied watersheds, these may include the costs of constructing and maintaining small dams, keeping RM ditch dredging efforts under budget, purchasing equipment to reduce or revert rates of soil erosion, or reduced crop yields. While the presented sediment budgets for the 05OF024 and 05OG008 sub-watershed cannot be said to give absolute answers to the questions that may be asked when planning to manage soil erosion in the Boyne-Morris and La Salle River watersheds, they can provide important context for the information at hand and required to guide such efforts.

2.6. Conclusions

The objectives of this study were satisfied through the application of the sediment budgeting technique in the 05OF024 and 05OG008 sub-watersheds of the Boyne-Morris and La Salle River watersheds in Manitoba's Red River Valley. Despite uncertainties inherent to this method, a number of meaningful conclusions were drawn from the sub-watershed sediment budgets when interpreted alongside the conclusions of past, relevant studies in the Red River Valley.

Differences in the relationships between modeled rates of water, wind, and tillage erosion in cultivated fields and measured rates of water, wind, and tillage deposition in road-side ditches suggested the SoilERI model used did not adequately characterized soil erosion risk. From the drafted sediment budgets, there was sufficient evidence to suggest this may be attributed to water and tillage erosion of landforms in cultivated fields not characterized by the SLC geodatabase, namely surface drains and field edges. Discrepancies related to wind erosion in cultivated fields could not be commented on from the data used to draft the sediment budgets. This finding mirrors published concerns regarding possible inaccuracies in NAHARP soil erosion risk estimates (Lobb *et al.*, 2016), but points to specific landform-related causes. This finding affects how the impacts of soil erosion are assessed and managed in landscapes characterized by the SLC geodatabase, especially landscapes that have had their agricultural capability enhanced through flow control infrastructure. The risk of erosion facing such uncharacterized landforms could not be assessed.

Very low delivery of eroded sediment to the outlets of the 05OF024 and 05OG008 sub-watersheds was illustrated by the sub-watersheds' sediment budgets. This suggests the contribution of eroded soil to declining water quality downstream of the sub-watersheds' outlets is often minimal. This assessment is strengthened by past assessment of the proportional contribution soil sediment sources to suspended sediment loads in streams, which were found to be low (Koiter *et al*., 2013). This assessment likely applies to other watersheds of similar or larger size in the Red River Valley, especially those characterized by landforms described by the SLC geodatabase as level with little to no slope. This is due to the inherently low SDRs of such watersheds. It was speculated that sediment transported downstream of erodible hillslopes in cultivated fields was more likely have negative impacts in smaller headwater basins of the 05OF024 and 05OG008 sub-watersheds, due to generally accepted inverse relationship between SDR and basin size. Observed of rates of sediment delivery to road-side ditches were higher than to sub-watershed outlets, supporting this. Backfilling of flow control infrastructure by eroded soil in small headwater basins was suggested to be of concern in the 05OF024 and 05OG008 sub-watersheds, supported by demonstrated sediment retention behind local small dams (Tiessen *et al*., 2011) and observed road-side ditch and in-field surface drain dredging. It was suggested that management practices that reduce SDRs have the most potential to mitigate such impacts. Present ditch dredging practices were judged to be an ineffective means of reducing SDRs to flow control infrastructure, with the potential to increase SDRs over short periods of time. Additionally, because dredged sediment is returned to adjacent fields, excessive dredging carries the risk of burying or diluting productive soil in field edges. Instead, it was suggested SDRs to flow control infrastructure may be more effectively reduced through topsoil restoration in cultivated fields. Such practices can be applied to hillslopes characterized by the SLC geodatabase, surface drains, and field edges and have the added benefit of directly addressing the impacts of soil erosion at the point of sediment production, namely reduced crop yields. It was stressed such management practices demand support by other management practices, specifically those which reduce rates of soil erosion in cultivated fields and mowing to remove vegetation in road-side ditches.

3. Implications for Managing the Impacts of Soil Erosion in Manitoba's Red River Valley

In Chapter 2, the 05OF024 and 05OG008 sub-watershed sediment budgets highlighted the limitations of National Agri-Environmental Health Analysis and Reporting Program (NAHARP) erosion risk estimates and our understanding of sediment delivery. Management practices that have the potential to address the on- and off-site impacts of soil erosion in the sub-watersheds (and by extension, in the respective Boyne-Morris and La Salle River watersheds they reside in) were discussed. Without a fuller understanding of the nature of sedimentary processes in the studied basins, the efficacy of such practices is uncertain and they cannot be prescribed with confidence. As such, recommendations for future research stemming from the study detailed in the previous chapter should focus on improving our understanding of soil erosion in Manitoba's Red River Valley. Such recommendations can be broadly categorized as those that focus on improving our understanding of soil erosion at the plot- and watershed-scales. These categories reflect the identified limitations of NAHARP erosion risk estimates and our understanding of sediment delivery.

The 05OF024 and 05OG008 sub-watershed sediment budgets point to two ways of refining NAHARP erosion risk estimates. First, additions can be made to the inputs used to calculate NAHARP erosion risk estimates, namely the Soil Landscapes of Canada (SLC) geodatabase. These additions are directly pointed to by the findings of the sediment budgeting study. Second, changes to the models that handle these inputs – SoilERI and its component WatERI, WindERI, and TillERI models – can be made. These changes are inferred from the findings of the preceding study and the relevant body of literature, though are recommended with a lesser degree of certainty.

Regarding additions to inputs used to calculate NAHARP erosion risk estimates, the SLC geodatabase may benefit from the inclusion of surface drain and field-edge landforms. Such landforms should be defined in the geodatabase's landform definition (LDT), landform extent (LET), and landform segmentation (LST) tables, with appropriate linkages made to other tables in the geodatabase. These landforms should be added to the geodatabase as unique entities, not through modification of existing landforms, as such changes harm the integrity data encoded to the SLC geodatabase and are difficult to trace and speak to the validity of. Rates of erosion can still be calculated using the SoilERI model. Lateral tillage erosion can be calculated and presented separately from forward tillage erosion, characterized by the TillERI model. Past studies have treated the rate of lateral tillage translocation of primary tillage operations as half that of forward tillage translocation (Li *et al*., 2009). Following these suggestions, the greatest rate of erosion for each erosive process active in the landscapes of the SLC geodatabase can then be reported, following the example of past National Agri-Environmental Health Assessment and Reporting Program (NAHARP) reports. However, reporting all calculated rates of erosion may be advantageous, as the resulting data product can better illustrate the relationship between the SoilERI model and the SLC geodatabase. Regardless of presentation, changes to the SLC geodatabase of this nature have the capacity to refine the outputs of all three of SoilERI's component models within the context of erosion risk estimation. Such changes are likely to have the greatest affect on the outputs of WatERI and TillERI, due to their sensitivity to changes in landform morphometrics.

There may be room for refinement of the models used to calculate NAHARP soil erosion risk estimates, though such recommendations are largely inferential. Further refinement of WindERI, if needed, may require attention to inputs not characterized by the SLC geodatabase or adjustment of model factors that are not calculated with it. However, such a recommendation is made in the absence of evidence of a strong landform-related control over wind erosion made by the study in the previous chapter, and not an understanding of the other factors that govern the outputs of the WindERI model. In respect to WatERI and TillERI, an emerging issue relating to them is the suspected interaction between water and tillage erosion (e.g., Zheng *et al*., 2021). Direct incorporation of the findings of such studies into WatERI and TillERI is not recommended, especially when the models are used to calculate NAHARP soil erosion risk estimates. This is due to the structures of WatERI and TillERI, which describe water and tillage erosion as a product and a linear equation of independent factors that govern water and tillage erosion, respectively. If these models are adjusted to account for interactions between water and tillage erosion, the independence of their respective factors is reduced, in turn diminishing the value of the models as tools to guide soil erosion management policy. Instead, it may be more pertinent that the findings of such studies are used to quantify uncertainty in present water and tillage erosion risk estimates. This may speak to a more general need for quantification of uncertainty in NAHARP soil erosion risk estimates.

Insight into the nature of sediment delivery in the Red River Valley cannot be gained from modelling alone. Additional sediment budgeting studies – complemented by sediment fingerprinting studies, environmental and economic impact assessments, and evaluations of management practices – can answer questions relating to sediment delivery raised by the 05OF024 and 05OG008 sub-watershed sediment budgets. The findings of such studies can then be extended to other basins in the Red River Valley through calculation of and comparison between the sediment yields and delivery ratios of the studied basins and other basins of interest; limited by the availability of total suspended solid (TSS) data.

Sediment budgets should be drafted for basins smaller and larger than the 05OF024 and 05OG008 sub-watersheds, recalling the inverse relationship between sediment yield and basin size. For smaller basins, sediment budgets should be designed to address questions related to sediment delivery to drainage infrastructure such as surface drains, ditches, culverts, and small dams. Particular attention should be paid to transfers from soil sediment sources, following the findings of the preceding study of the Boyne-Morris and La Salle River watersheds. Sediment budgets for larger basins should continue to focus on the issue of how sediment is delivered to Lake Winnipeg. Such budgets should focus on the role of sediment sources with high degrees of connectivity with natural stream networks, including channel beds, channel banks, and urban areas. It may be worth conduction such studies at finer temporal resolutions. This may increase the likelihood of budgets characterizing infrequent, high-magnitude sediment fluxes, which have a bearing on management decisions. Regardless of basin size or temporal resolution, multiple means of drafting sediment budgets may be worth investigating. For instance, the IROWC-P downstream phosphorus (P) contamination risk model uses a number of sediment delivery ratios (SDRs) to determine particulate P delivery. These SDRs – based in the RUSLE2 water erosion model and best reckoning (van Bochove, 2010) – can be used in place of measured sediment transfers in existing sediment budgets that include soil sediment sources. The resulting revised sediment budget, when compared with the existing sediment budget based in measured sediment transfers, can then be used to validate IROWC-P SDRs, refining the model.

To extend the findings of sediment budgeting studies, sediment yields and SDRs for other basins require calculation. Calculation of sediment yields is limited to basins with sufficient records of TSS measurements. Sediment delivery ratios can be calculated from models that treat basin size or stream network length and basin relief as predictor variables, or calculated directly from modelled estimates of soil erosion based in SoilERI. The use of the latter method underscores the importance of a sound understanding of the plot-scale state of soil erosion and the uncertainty that surrounds the models used to characterize it. With this information, existing sediment budgets can be extended to basins exhibiting similar patterns in aggregate sediment delivery, surficial geology, pedology, and ecologic, climatic, and agricultural characteristics.

These recommendations underscore the importance of budgeting studies. While they cannot independently address all issues pertaining to contaminant flux through a basin, the holistic manner they portray contaminant production, deposition, and transportation provides valuable insight into contaminant delivery to the point in the landscape that its impact is assessed.

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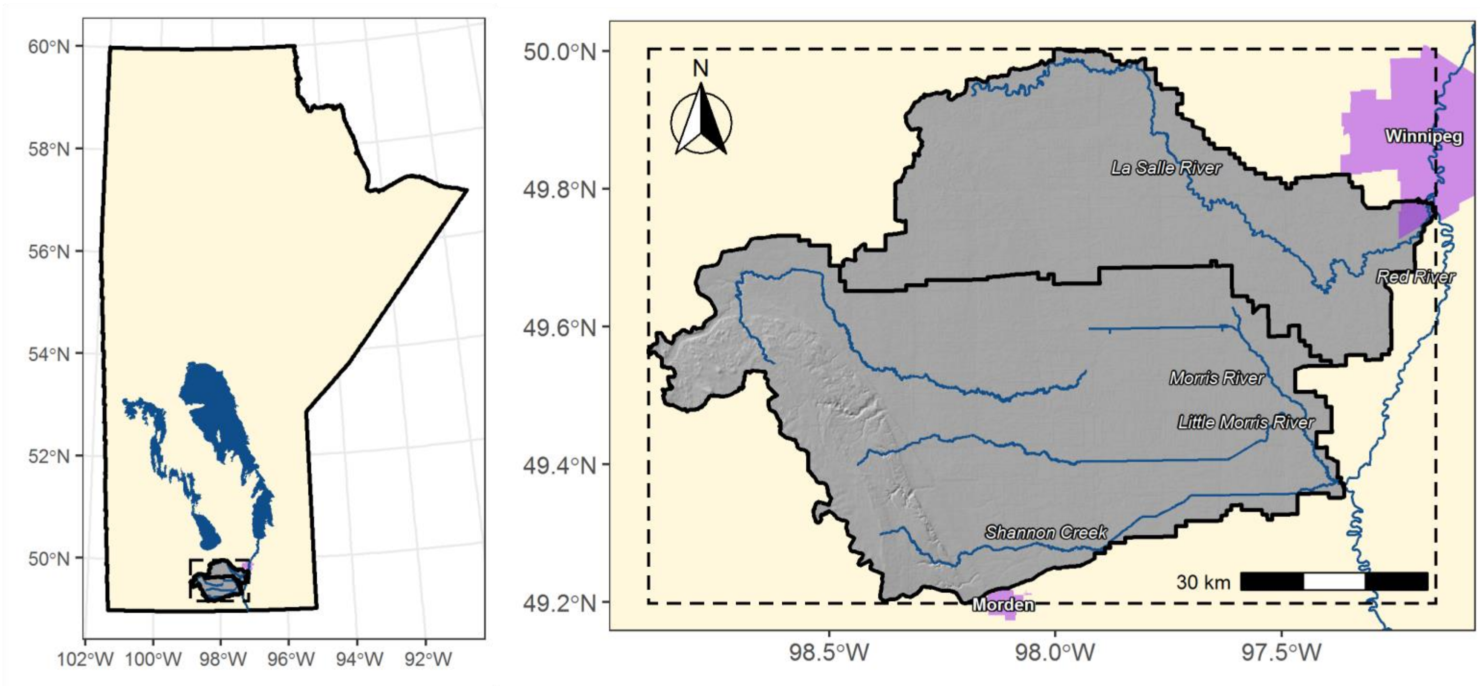

Figure 2.1. Location of the Boyne-Morris and La Salle River watersheds in Manitoba and their respective major tributaries (MLI, 2014).

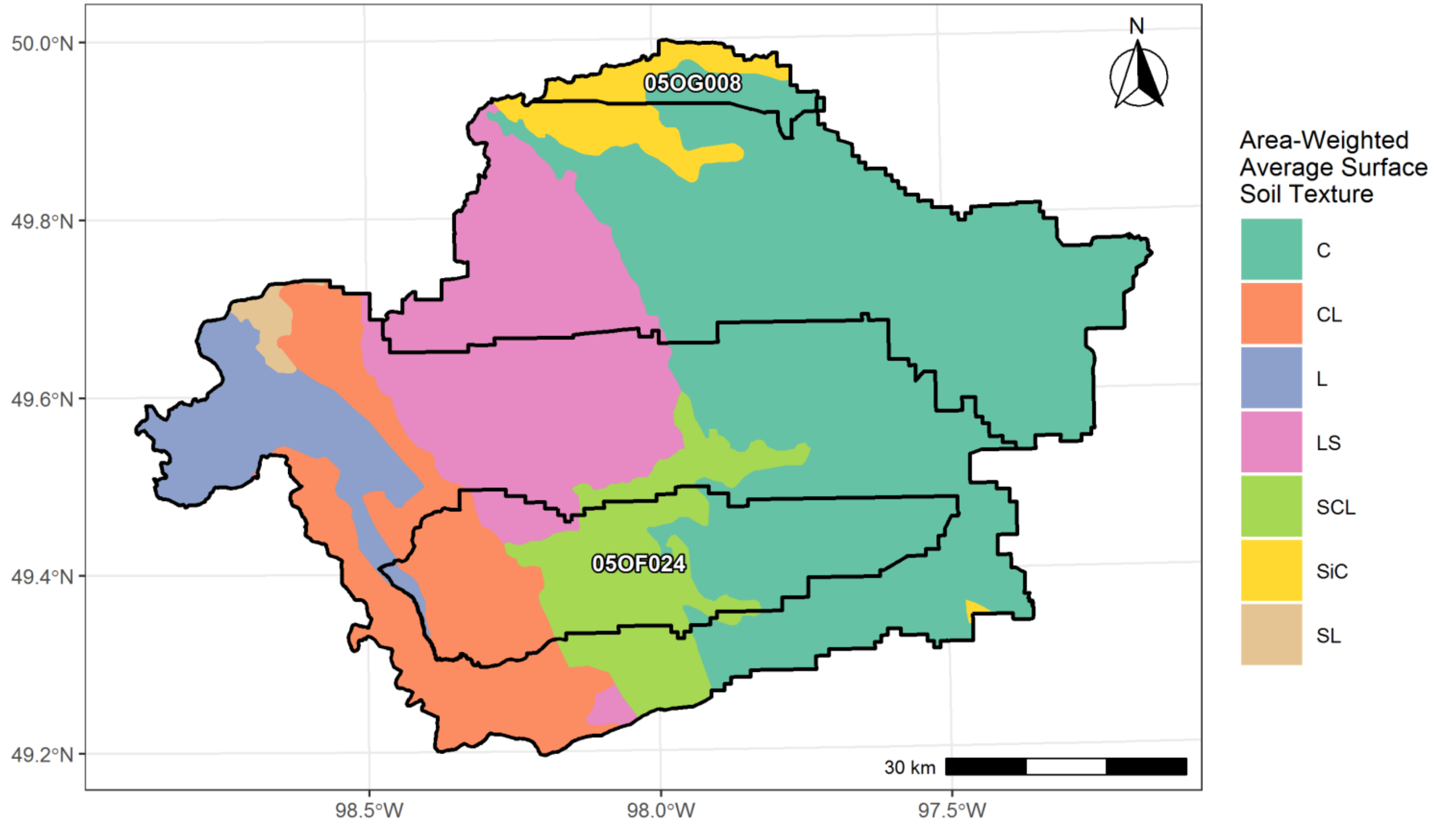

Figure 2.2. Area-weighted average surface soil textures and terrain in the Boyne-Morris and La Salle River watersheds and their respective 05OF024 and 05OG008 sub-watersheds (AAFC, 2010; MLI, 2005).

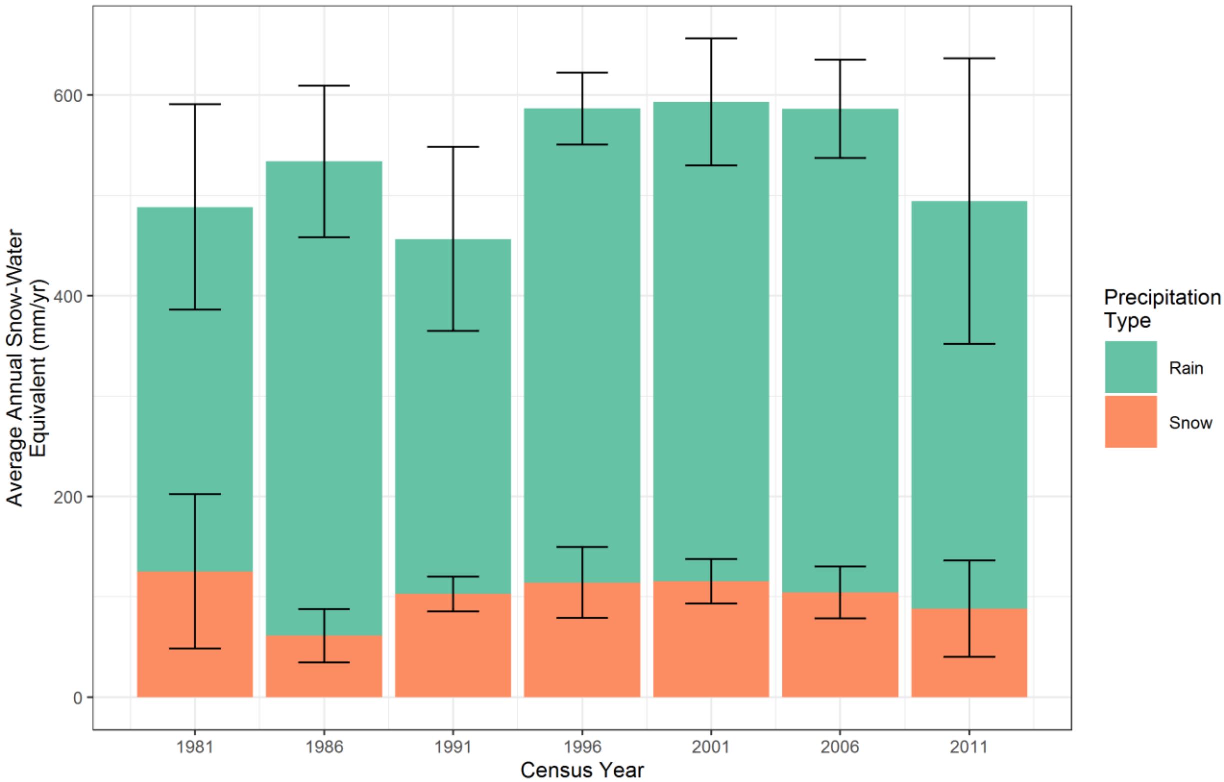

Figure 2.3. Annual average total precipitation in the combined Boyne-Morris and La Salle River watersheds, measured at ECCC station number 3582 near Carman, MB (ECCC, 2022). Annual average totals are calculated per census year. Error bars equal 1 standard deviation about the mean.

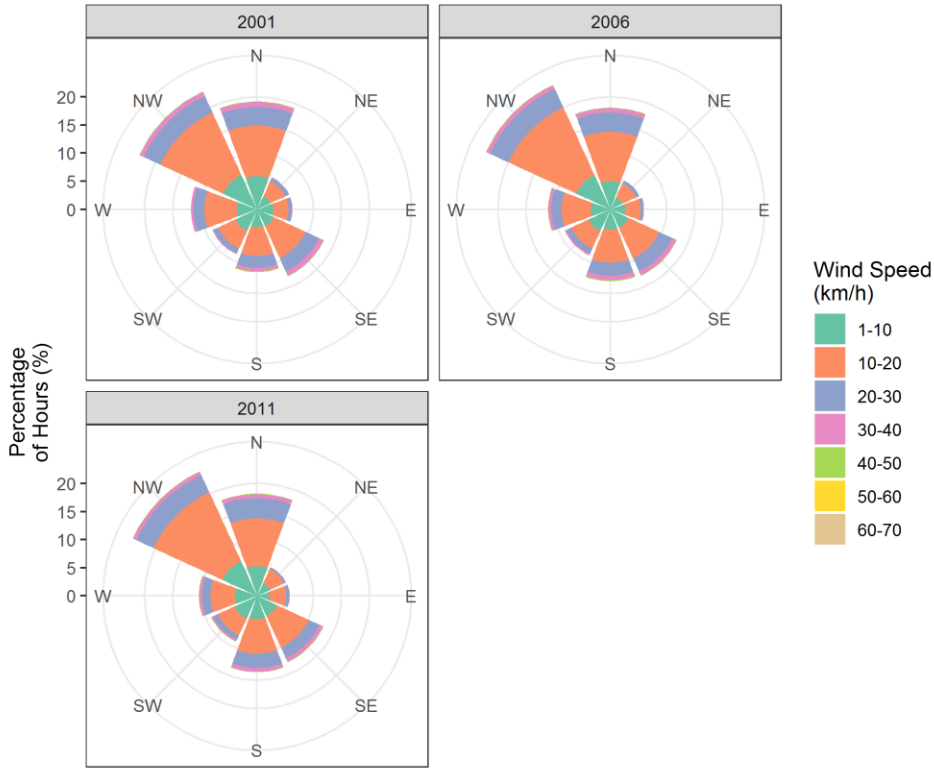

Figure 2.4. Distribution of wind speeds and directions per census year in the combined Boyne-Morris and La Salle River watersheds, measured at ECCC station number 26857, near Carman, MB (ECCC, 2022). Records for the 2001, 2006, and 2011 census years are 3.5, 5.1, and 10.7% incomplete or windless, respectively.

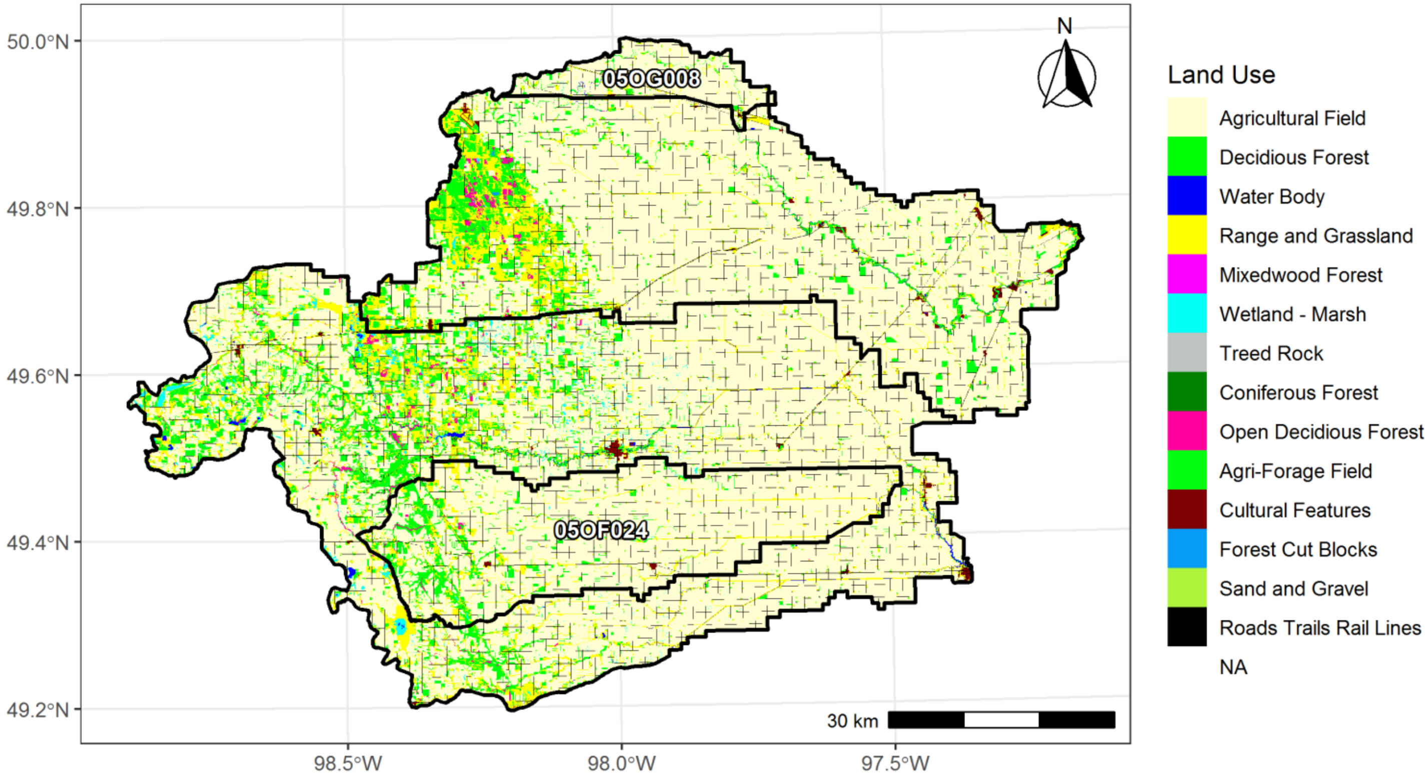

Figure 2.5. Land use patterns in the Boyne-Morris and La Salle River watersheds and their respective 05OF024 and 05OG008 sub-watersheds (MLI, 2005; MLI, 2013). Land classified as “Roads Trails Rail Lines” typically follows the 30 m-wide road allowances of the Dominion Land Survey grid.

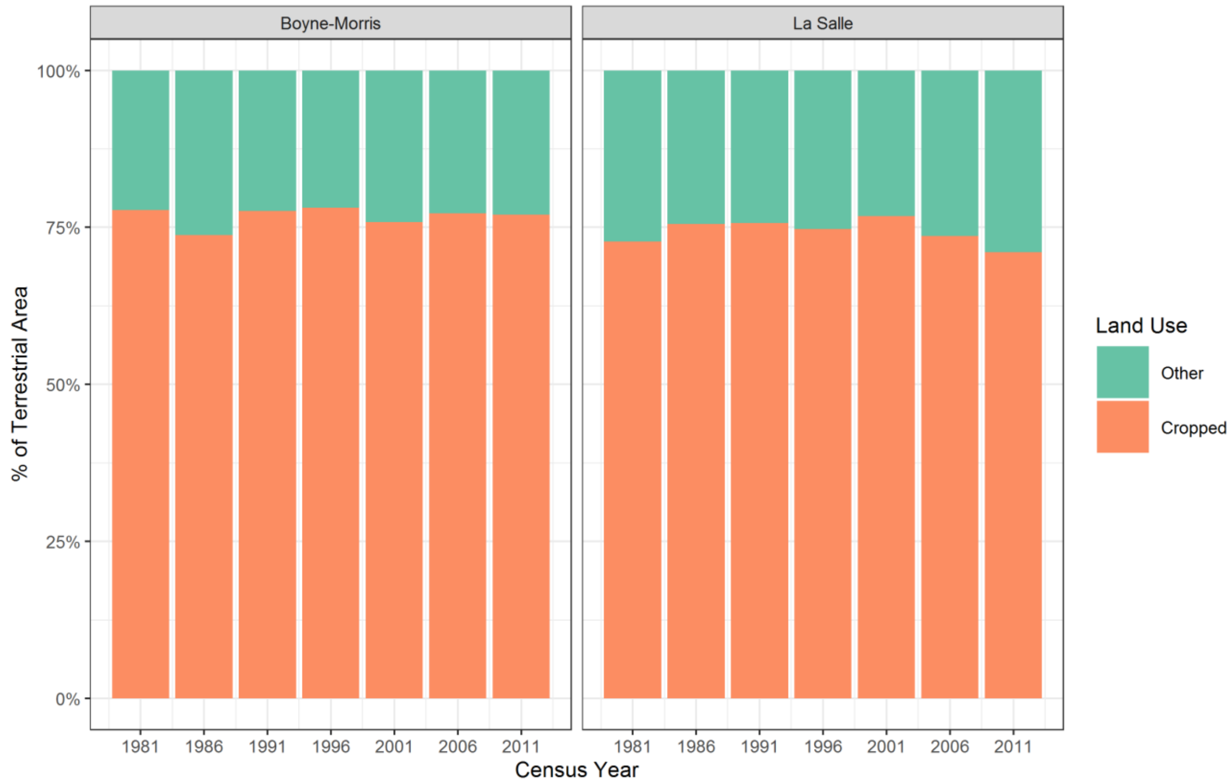

Figure 2.6. Past land use patterns in the Boyne-Morris and La Salle River watersheds as characterized by the Canadian Census of Agriculture (AAFC, 2016e).

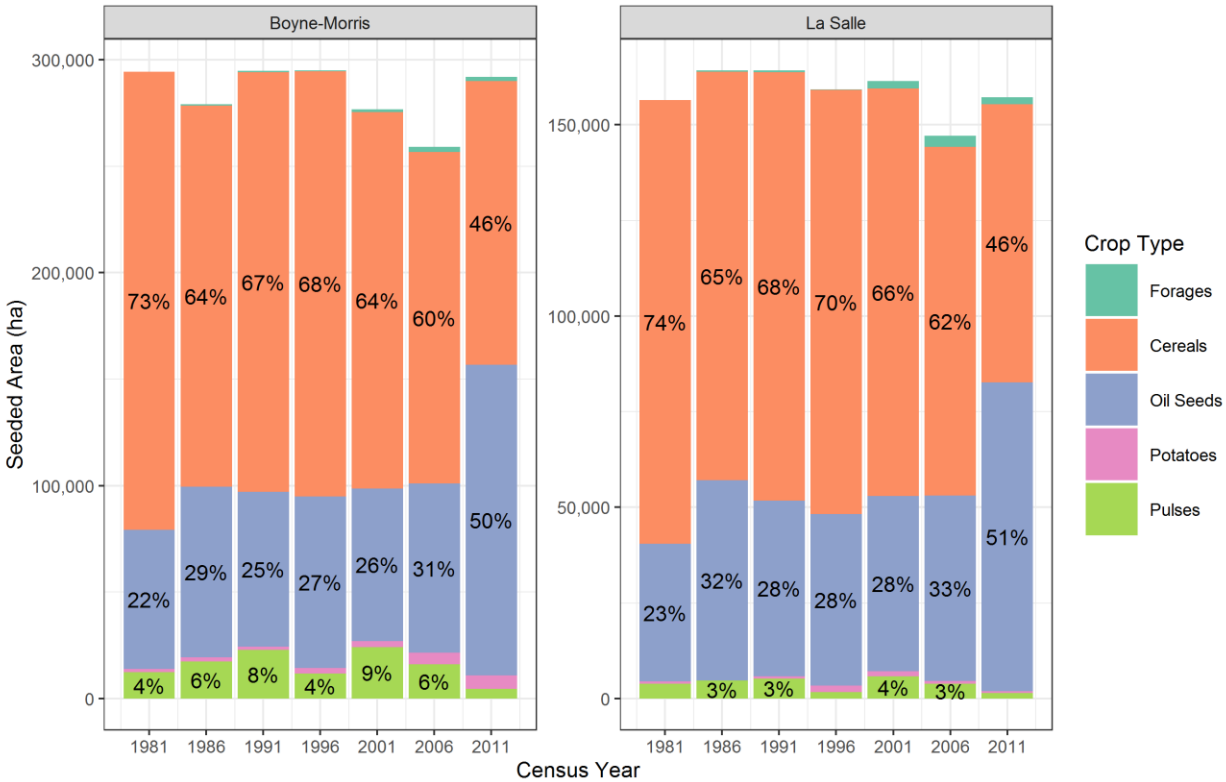

Figure 2.7. Past cropping patterns in the Boyne-Morris and La Salle River watersheds, as characterized by the Canadian Census of Agriculture (AAFC, 2016e). Crops accounting for less than 3% of seeded area are plotted with percentages omitted for clarity.

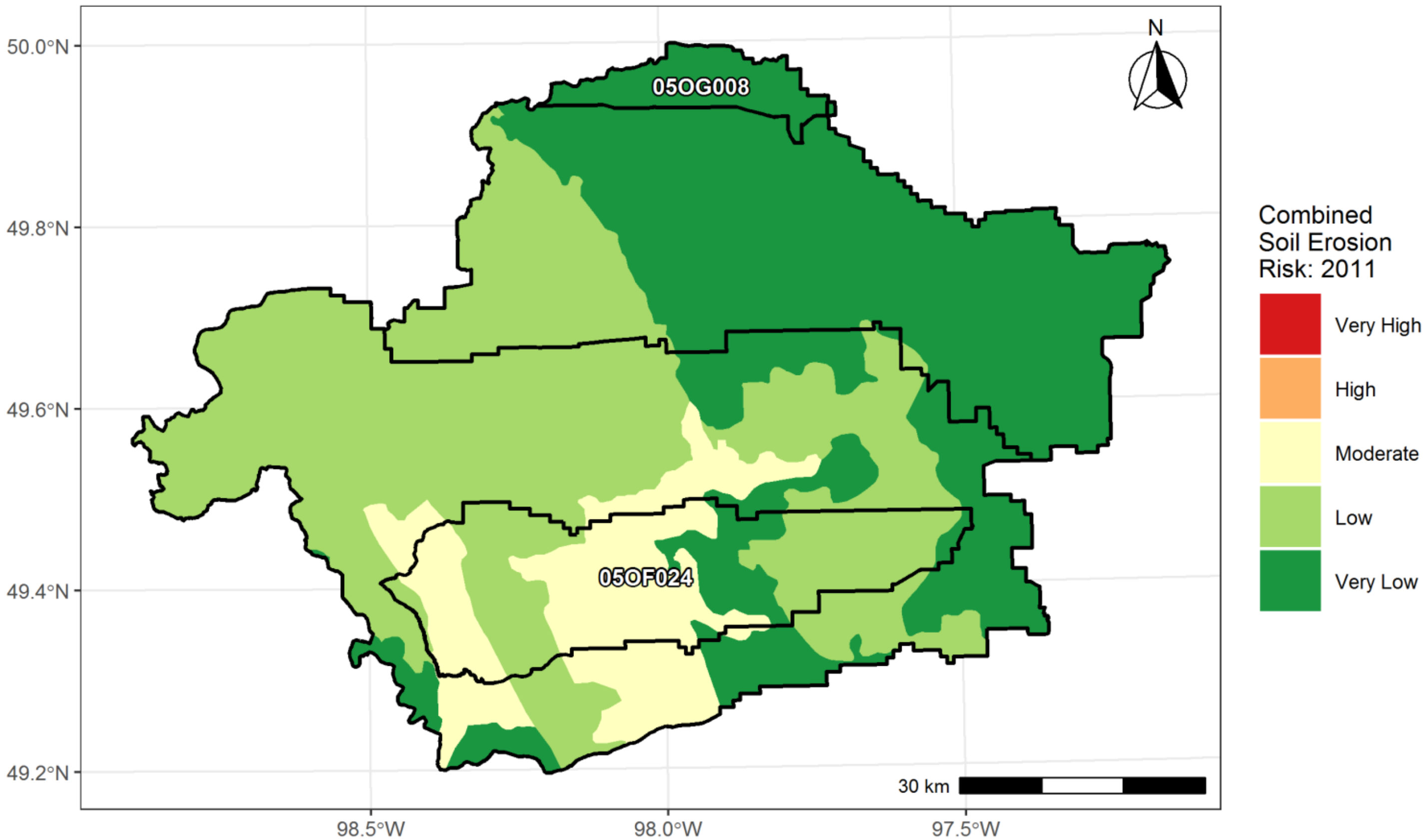

Figure 2.8. Recent combined soil erosion risk and terrain in the Boyne-Morris and La Salle River watersheds and their respective 05OF024 and 05OG008 sub-watersheds (AAFC, 2010; AAFC, 2016a; MLI, 2005).

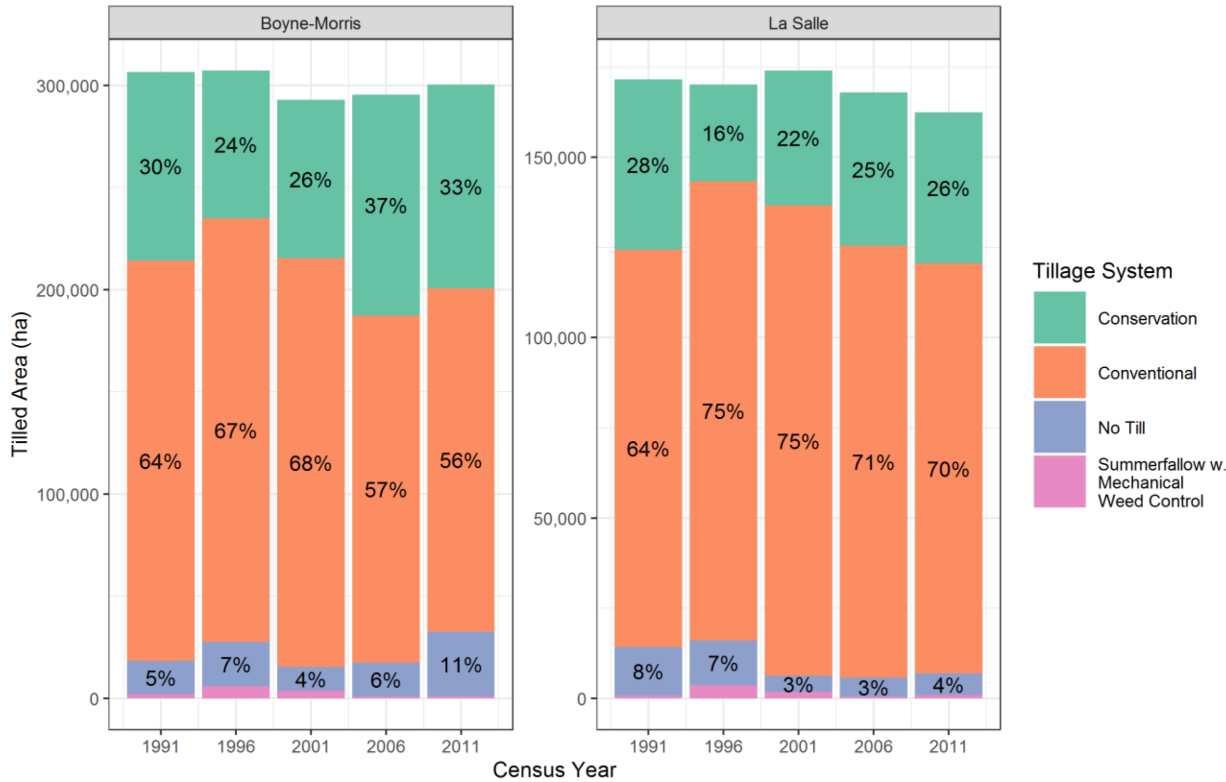

Figure 2.9. Past adoption of specific tillage practices in the Boyne-Morris and La Salle River watersheds, as characterized by the Canadian Census of Agriculture (AAFC, 2016e). Practices accounting for less than 3% of tilled area are plotted with percentages omitted for clarity.

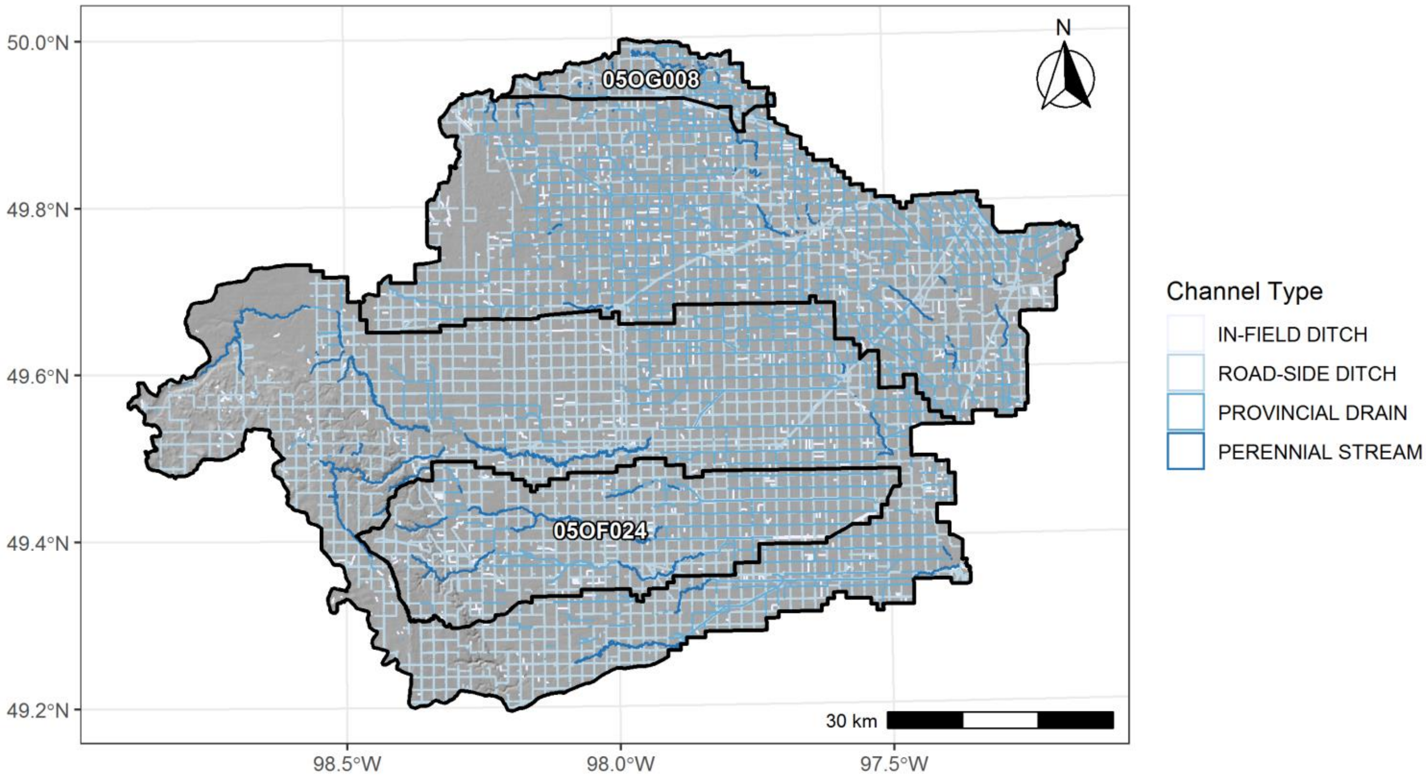

Figure 2.10. Drainage infrastructure patterns and terrain in the Boyne-Morris and La Salle River watersheds and their respective 05OF024 and 05OG008 sub-watersheds (MLI, 2004; MLI, 2005).

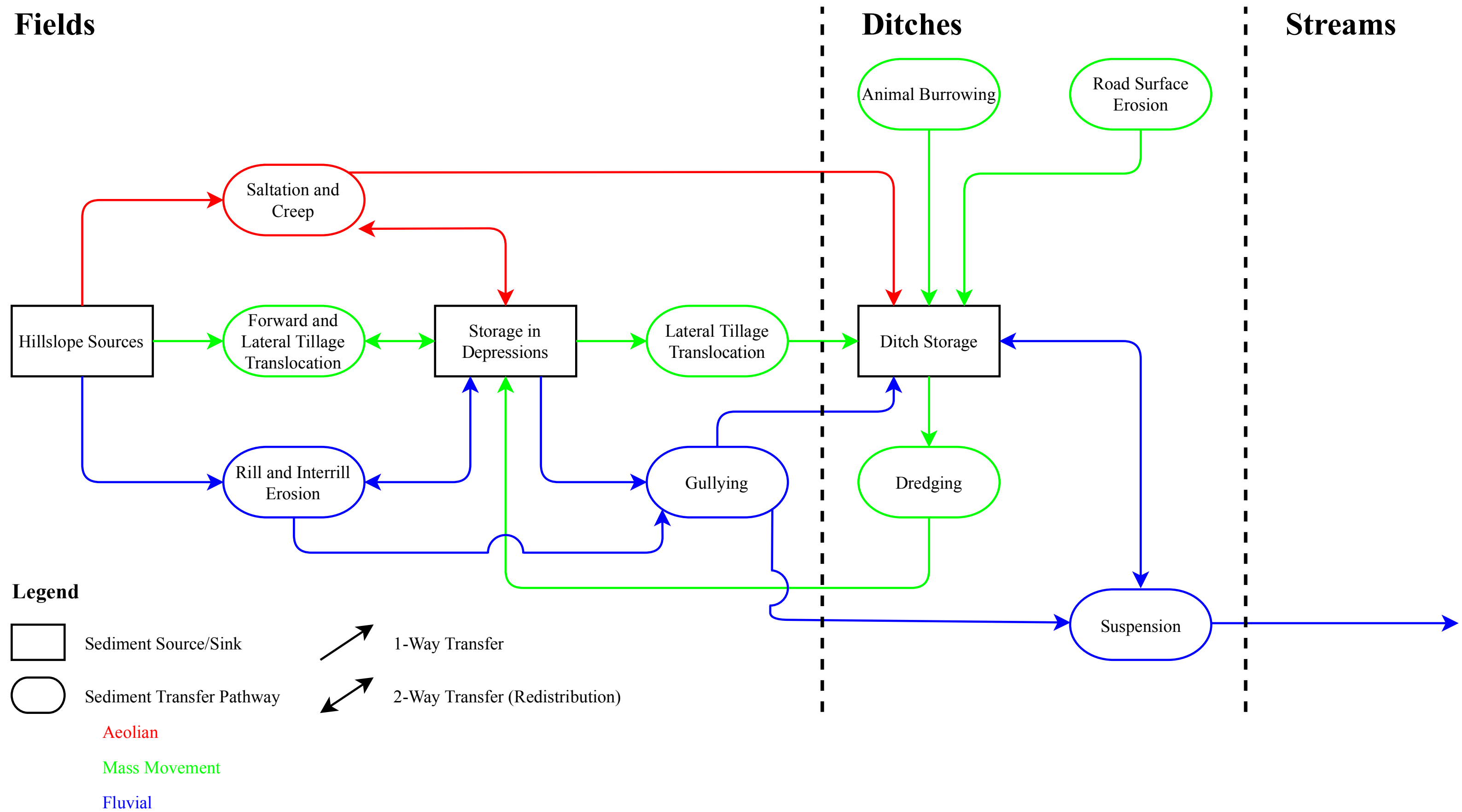

Figure 2.11. Flowchart illustrating sediment transfers between cultivated field, road-side ditch, and stream settings; as driven by the sedimentary processes of erosion (ellipses), deposition (rectangles), and transfers (arrows) in the 05OF024 and 05OG008 sub-watersheds of the Boyne-Morris and La Salle River watersheds.

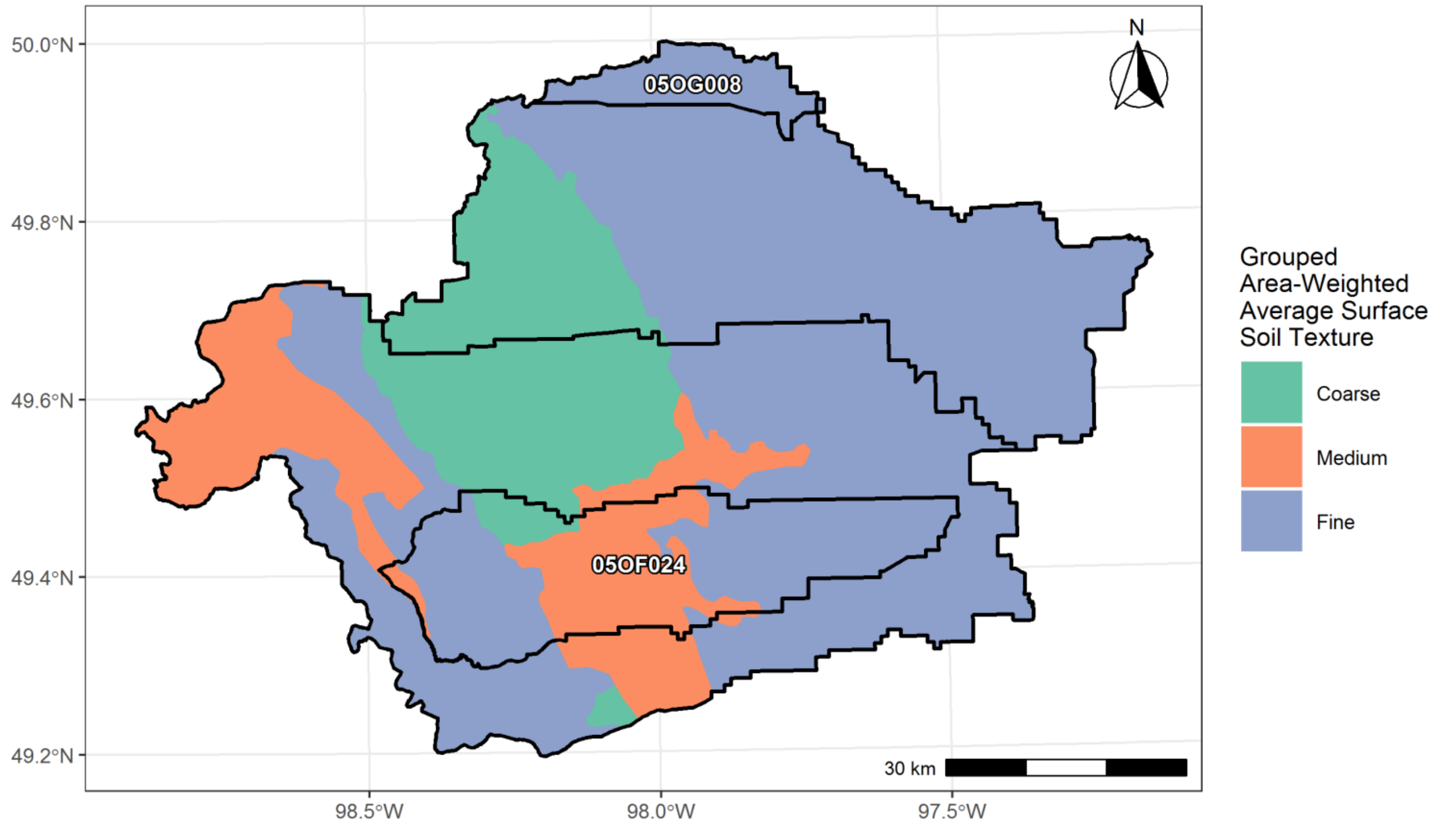

Figure 2.12. Area-weighted surface soil texture in the Boyne-Morris and La Salle River watersheds, grouped according to commonly held definitions of coarse-, medium-, and fine-textured soils (AAFC, 2010; MLI, 2005; SCWG, 1998). These generalized textual classes were used to further divide the Boyne-Morris and La Salle River watersheds' respective 05OF024 and 05OG008 sub-watersheds into uniformly erodible catchments for sediment budget calculation.

Figure 2.13. Sedimentary features related to sediment transfers in and near road-side ditches. Field to ditch transfers can be characterized by sediment fans (a); dirty snow, colloquially referred to as snirt (b), and blow dirt (c); and tillage throw (d), relating to water, wind, and tillage erosion in fields, respectively. Further sediment is transferred into ditches from roads in the form of gravel side-cast by traffic (e). Deposited sediment is reworked in the ditch by burrowing mammals (f), and ultimately dredged out by RMs (g) and transferred back into to fields.

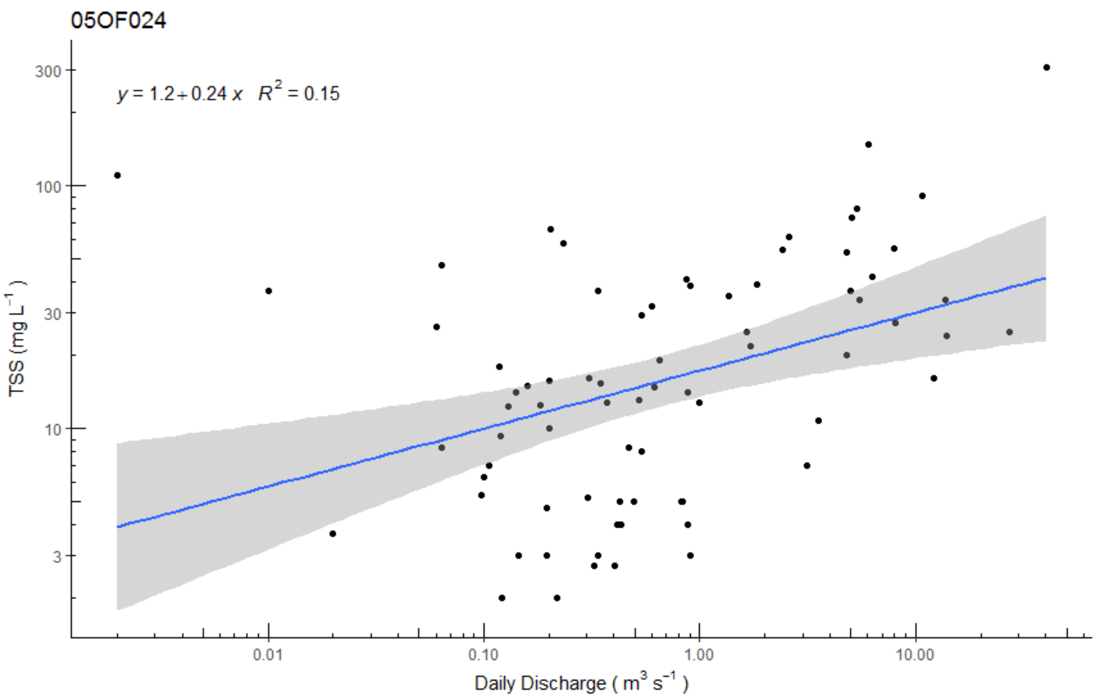

Figure 2.14. The $\log_{10}$-$\log_{10}$ linear relationship between stream discharge and TSS at the outlet of the 05OF024 sub-watershed. Standard error is represented by the shaded area. The low coefficient of determination suggest interpolation between known TSS concentrations may yield better estimates of daily TSS concentration.

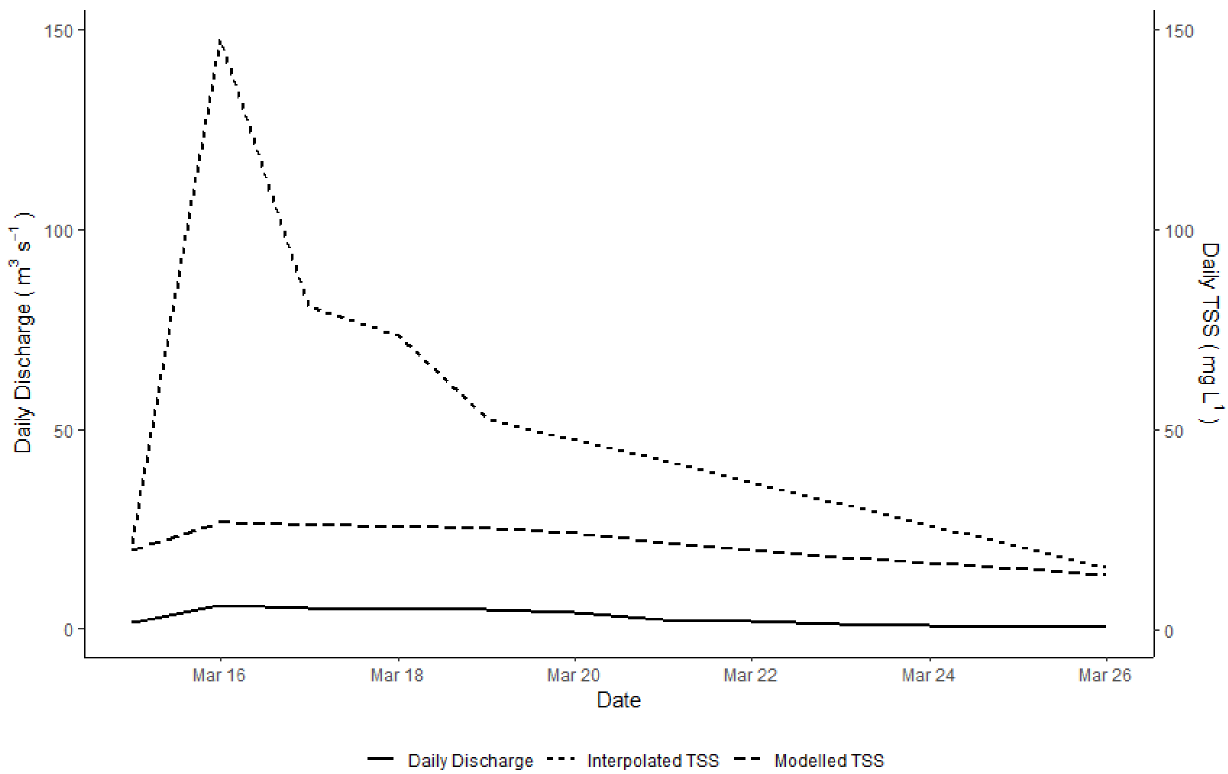

Figure 2.15. Comparison of daily TSS concentrations estimated with linear regression and interpolation. Plotted data was measured at WSC station 05OF024 early in the freshet of 2015. Underestimation of known TSS concentrations by the linear model is considerable, supporting the selection of interpolation as the preferred method of daily TSS concentration estimation.

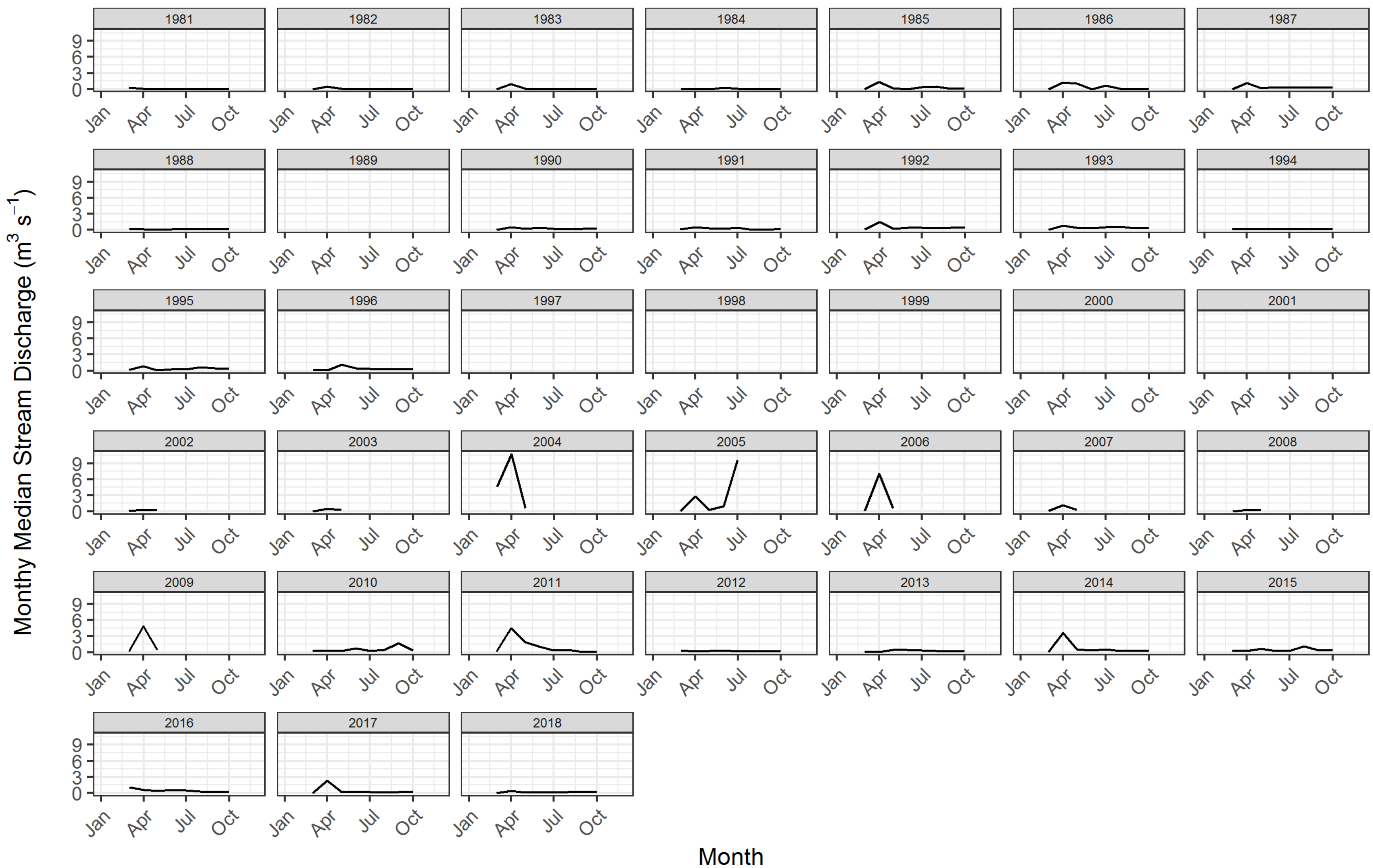

Figure 2.16. Historical monthly median stream discharges past WSC station 05OG008. Unusually high discharges characterize 2004 to 2009. Records from 2002 to 2009 are incomplete and records from 1989 and 1997 to 2001 are absent.

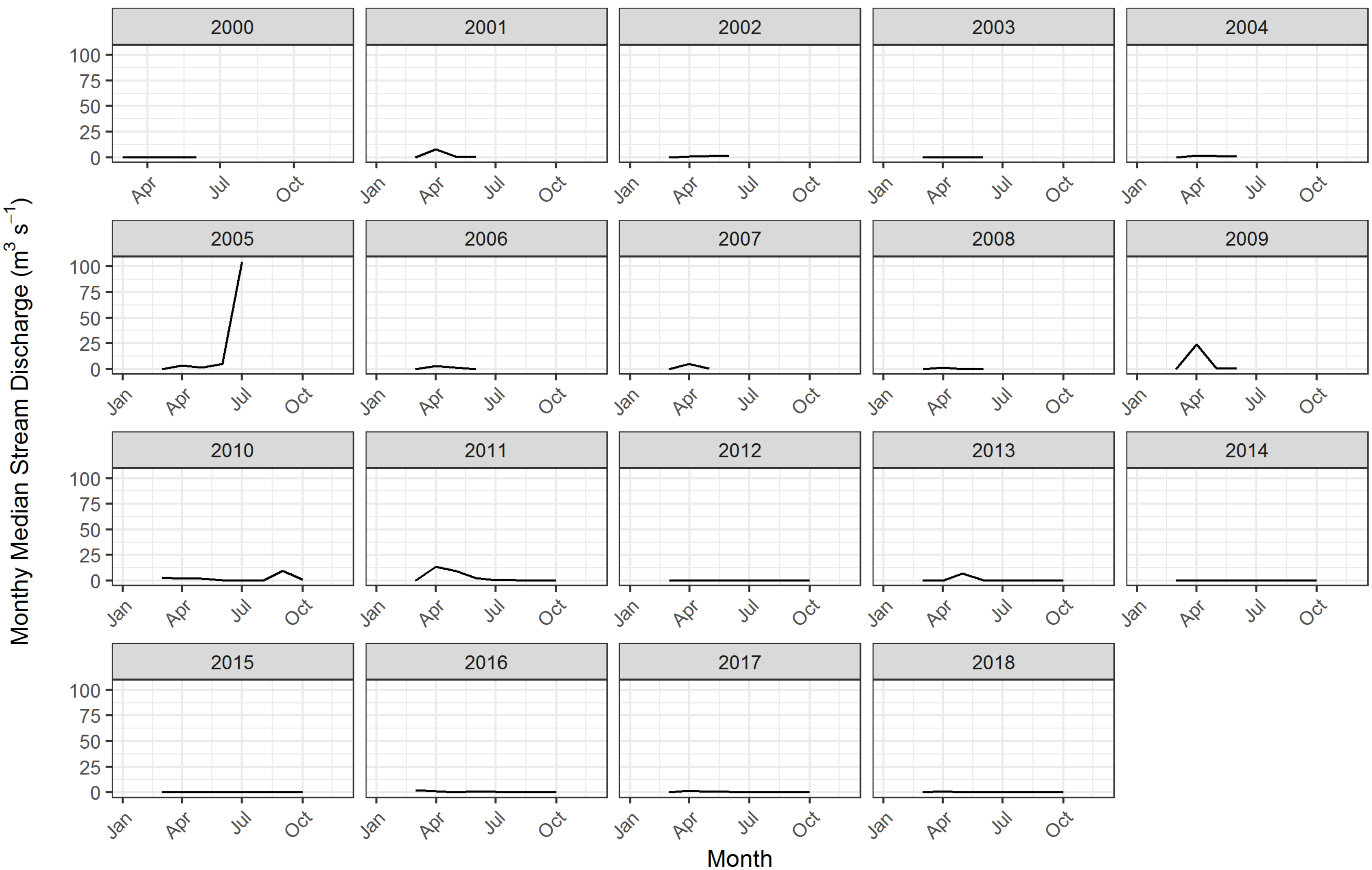

Figure 2.17. Historical monthly median stream discharges past WSC station 05OF024. Unusually high discharges characterize 2005. Records from 2000 to 2009 are incomplete.

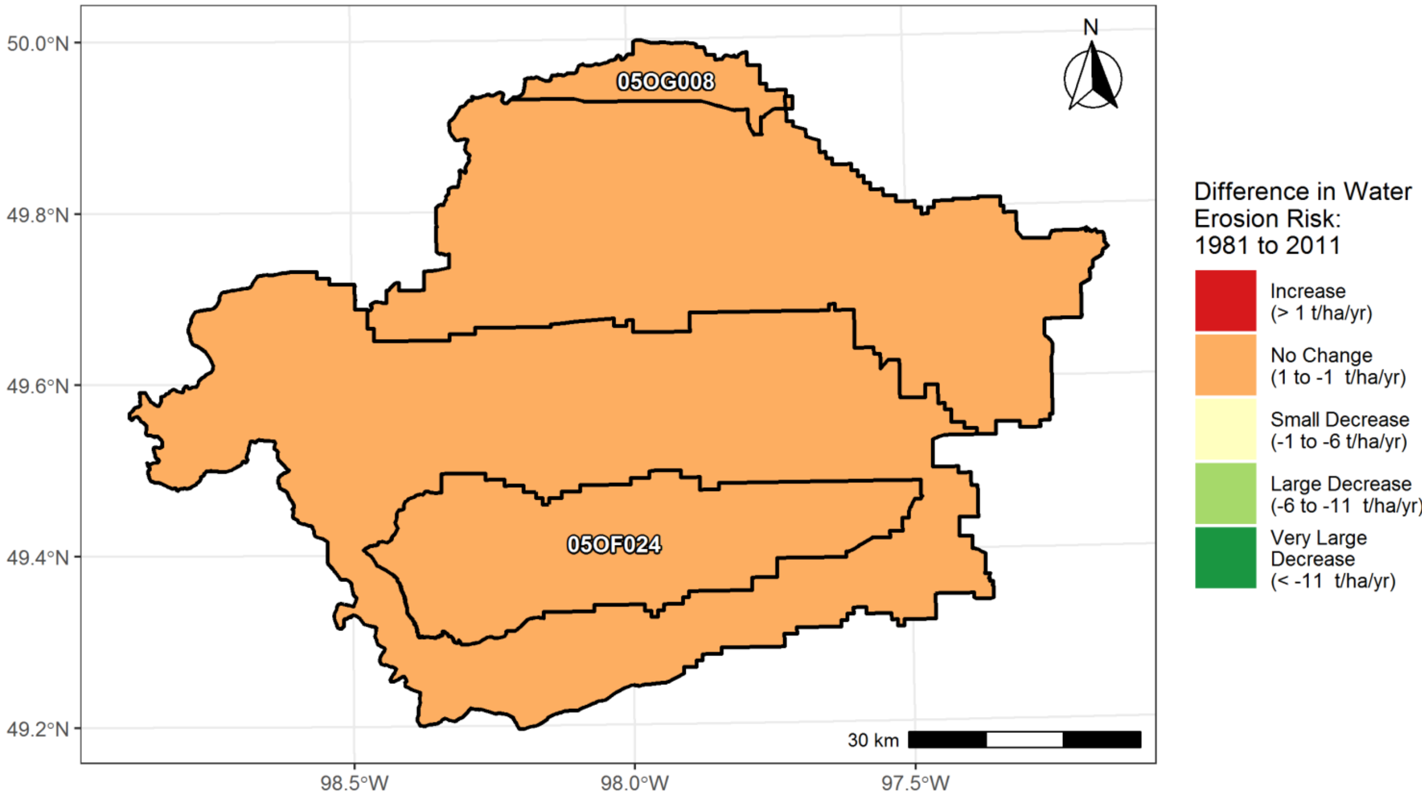

Figure 2.18. Differences in water erosion in the Boyne-Morris and La Salle River watersheds and their respective 05OF024 and 05OG008 sub-watersheds between 1981 and 2011.

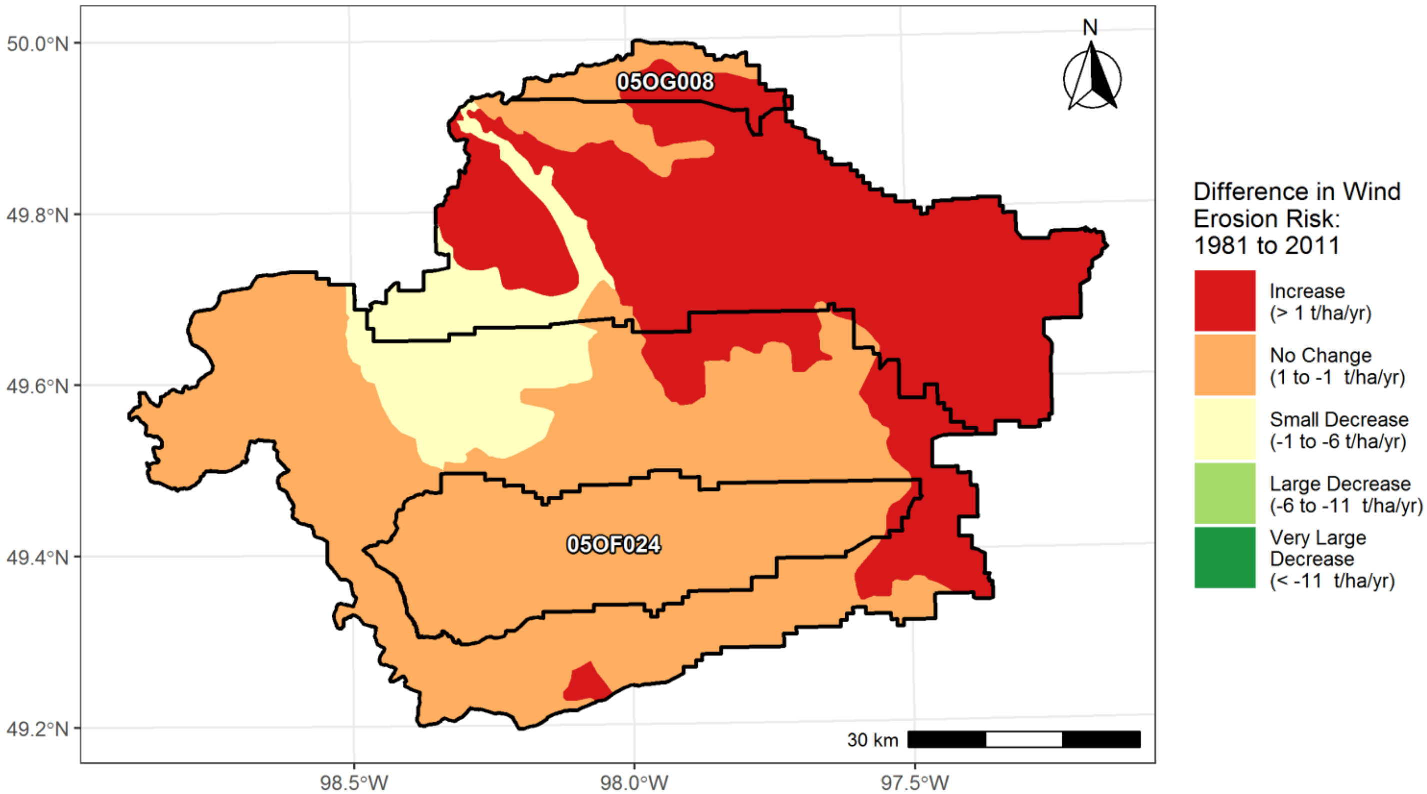

Figure 2.19. Differences in wind erosion in the Boyne-Morris and La Salle River watersheds and their respective 05OF024 and 05OG008 sub-watersheds between 1981 and 2011. A large proportion of sub-watershed 05OG008 is impacted by these changes. However, the changes themselves are small, amounting to an increase in water erosion slightly more than 1 t ha^{-1} yr^{-1} over the 30-year period characterized by NAHARP soil erosion risk estimates. These differences may be related to changes in cropping, increasing soil erodibility.

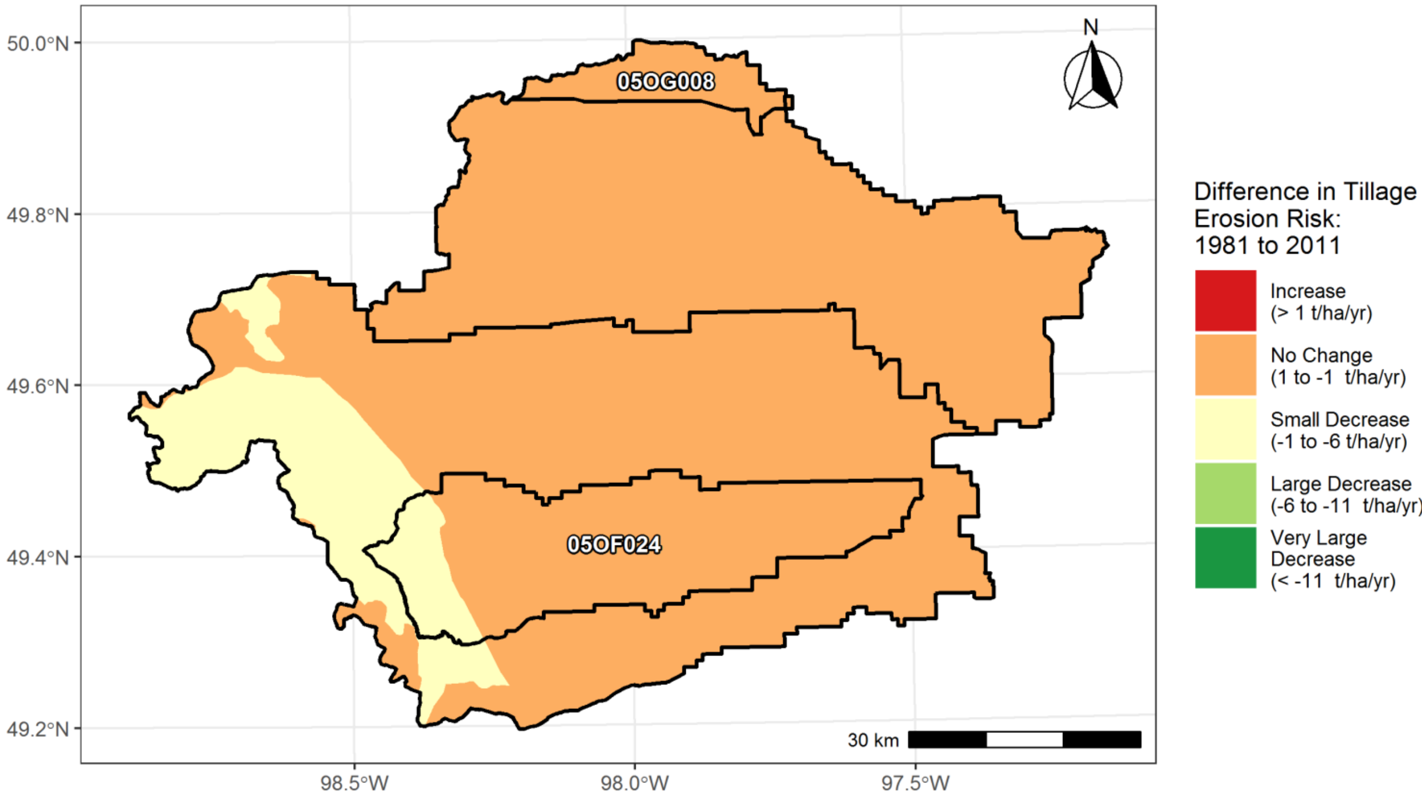

Figure 2.20. Differences in tillage erosion in the Boyne-Morris and La Salle River watersheds and their respective 05OF024 and 05OG008 sub-watersheds between 1981 and 2011. A small proportion of sub-watershed 05OF024 is impacted by these changes. The impact of differences in this portion of the sub-watershed are lessened by size of changes themselves, amounting to a decrease of slightly less than -1 t ha^{-1} yr^{-1} over the 30-year period characterized by NAHARP soil erosion risk estimates. These differences may be related to changes in tillage sequences, decreasing overall tillage erosivity.

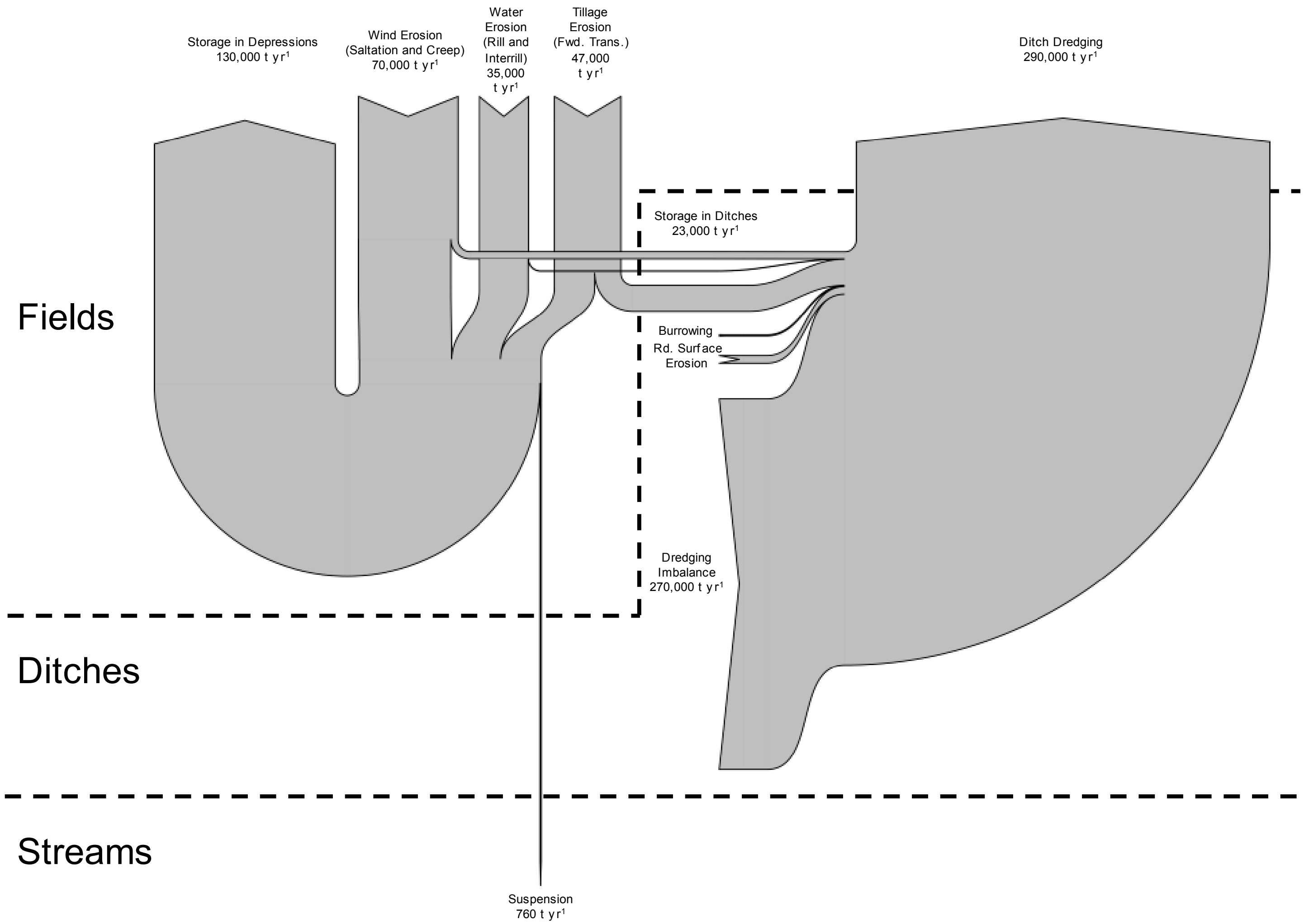

Figure 2.21. Annual average sediment budget of the 05OF024 sub-watershed between 1981 and 2011. The sediment yield of the 05OF024 sub-watershed was 0.008 t ha^{-1} yr^{-1} over the same period.

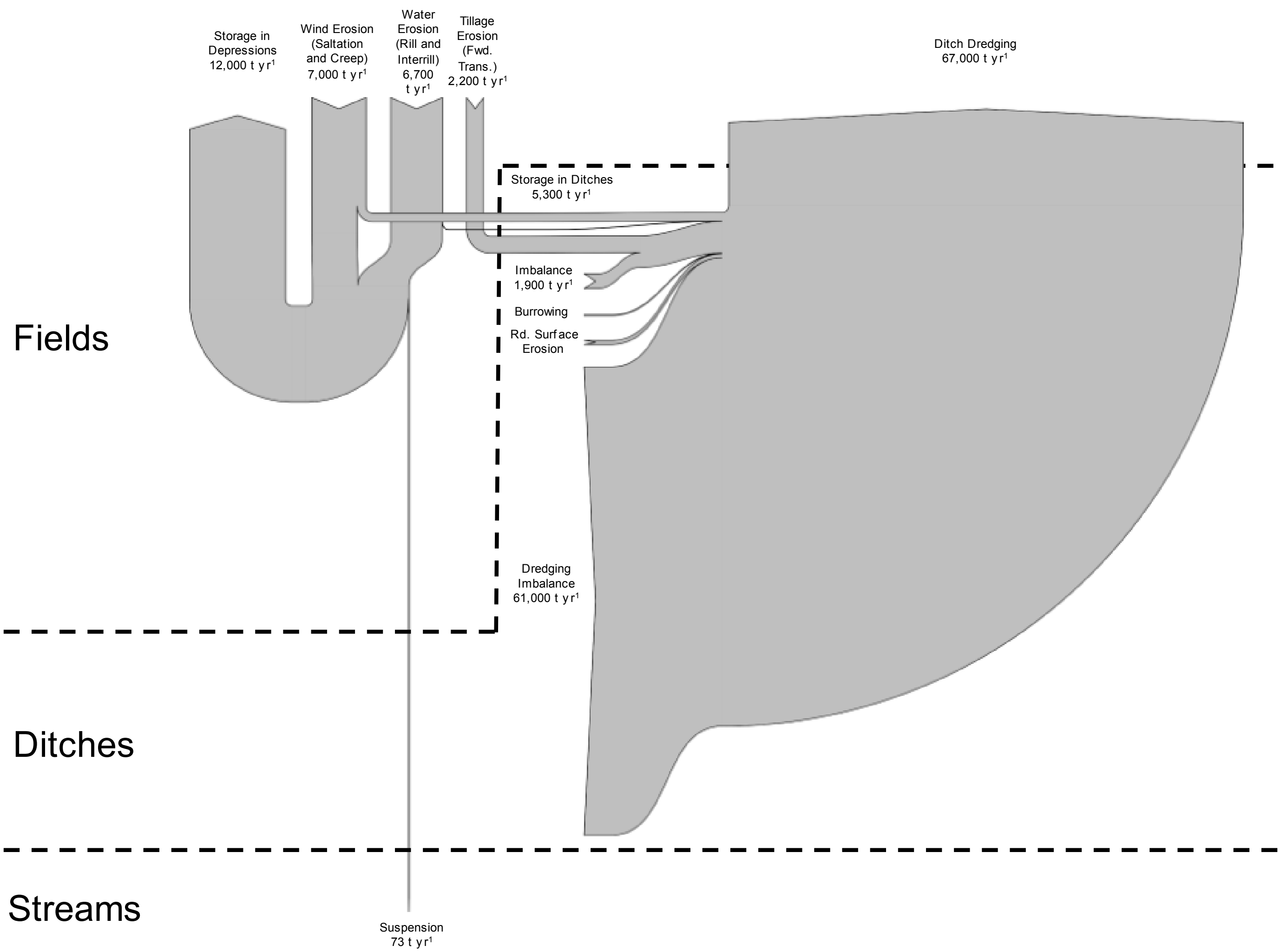

Figure 2.22. Annual average sediment budget of the 05OG008 sub-watershed between 1981 and 2011. The sediment yield of the 05OG008 sub-watershed was 0.004 t ha^{-1} yr^{-1} over the same period, similar the yield of the larger (though topographically similar) 05OF024 sub-watershed. This contrasts with the sediment yield of the similarly-size, but rugged, 05OF017 headwater sub-watershed nested in the 05OF024 sub-watershed (0.08 t ha^{-1} yr^{-1} between 1963 and 1977, inclusive).

Table 2.1. Summary of data used to draft sediment budgets for the 05OF024 and 05OG008 sub-watersheds, sorted by sedimentary setting and process

Sedimentary Setting	Sedimentary Process	Data Authorship	Data Description	Data Type	Data Sources
Fields	Water, Wind, and Tillage Erosion	Secondary	NAHARP Soil Erosion Risk Indicators	.SHP [a]	AAFC (2016b, 2016c, 2016d)
			NSDB SLC V3R2	.GDB [b]	AAFC (2010)
			NSDB SLC V2R1 HYDRO Overlay	.SHP	AAFC (1996)
			CanVec Political Boundaries	.SHP	NRC (2017)
			MLI Gross Watersheds	.SHP	MLI (2005)
	In-Field Deposition	-	-	-	-

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Table 2.1. (continued from previous page)

Ditches	Road Surface Erosion	Primary	Road-Side Ditch Survey	*Various*	-
		Secondary	MLI Municipal Roads	.SHP	MLI (2000)
			MLI Gross Watersheds	.SHP	MLI (2005)
	Road-Side Ditch Excavation	Primary	Road-Side Ditch Survey	*Various*	-
		Secondary	MLI Designated Drain Watercourses	.SHP	MLI (2004)
			RM of MacDonald Ditch Maintenance Records	*Various*	*NA*
			MLI Gross Watersheds	.SHP	MLI (2005)
	Animal Burrowing	Primary	Road-Side Ditch Survey	*Various*	-
		Secondary	MLI Designated Drain Watercourses	.SHP	MLI (2004)
			MLI Gross Watersheds	.SHP	MLI (2005)
	Ditch Deposition	Primary	Road-Side Ditch Survey	*Various*	-
		Secondary	MLI Designated Drain Watercourses	.SHP	MLI (2004)
			MLI Land Use	.TIF [c]	MLI (2013)
			RM of MacDonald Ditch Maintenance Records	*Various*	-
			MLI Gross Watersheds	.SHP	MLI (2005)

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Table 2.1. (continued from previous page)

Streams	Suspension	Secondary	MLI Designated Drain Watercourses	.SHP	MLI (2004)
			WSC Flow Measurements	.CSV	WSC (2021)
			AAFC Contaminant Measurements	.CSV	AAFC (2017)
			DSWMA Contaminant Measurements	.CSV	-

[a] .SHP: shapefile

[b] .GDB: geodatabase

[c] .TIF: tagged image file

[d] .CSV: comma-separated variable

Table 2.2. Proportionality between landforms in the landscapes characterized by the SLC geodatabase

Sub-Watershed	Landscape	Landform					Terrestrial Area (ha)
		HB [a]	IE [b]	UA [c]	LA [d]	RA [e]	
05OF024	766006	65	20	15	-	-	12135
	849006	-	-	-	100	-	21591
	849009	-	-	2	98	-	1399
	852001	-	-	-	100	-	12287
	852003	-	-	-	90	10	13219
	852004	-	-	-	100	-	5675
	852005	-	-	7	93	-	30410
	854001	70	-	30	-	-	1450
	854002	30	-	70	-	-	3
05OG008	849009	-	-	2	98	-	9054
	851003	-	-	30	70	-	9874

[a] HB: Hummocky, 4-9% gradient

[b] IE: Inclined and dissected, 31-60% gradient

[c] UA: Undulating, 0-3% gradient

[d] LA: Level, 0-3% gradient

[e] RA: Ridged, 0-3% gradient

Table 2.3. Description of data used to determine p_{abc}, sorted by sedimentary transfer direction, process, and associated sedimentary feature

Sedimentary Transfer Direction	Sedimentary Process	Sedimentary Feature	Data Description	Spatial Extent of Data	Temporal Extent of Data
Field to Ditch	Water Erosion	Sediment Fans	RM of MacDonald ditch maintenance records [a]	RM of MacDonald	2005 – 2016
Field to Ditch	Wind Erosion	Snirt	Assumed to affect all road-side ditches	-	-
Field to Ditch	Wind Erosion	Blow Dirt	RM of MacDonald ditch maintenance records after dust storms	RM of MacDonald	2018
Field to Ditch	Tillage Erosion	Tillage Throw	Survey along quarter-sections over 3, 8.045 km^2 (5 $mi.^2$) grids; each grid overlaying fine-, medium-, and coarse-textured soils	Boyne-Morris Watershed	2018
Ditch to Fields	Ditch Dredging	Dredged Spoil Piles	RM of MacDonald ditch maintenance records	RM of MacDonald	2005 – 2016
Road to Ditch	Road Surface Erosion	Gravel Sheets	Assumed to affect all road-side ditches	-	-
Reworking in Ditch	Animal Burrowing	Bioturbated Soil Piles	Survey along sections over numerous transects of varying length; each transect overlaying fine-, medium-, and coarse-textured soils	Boyne-Morris Watershed	2018

[a] Assumes sediment fans only form at the outlet of in-field surface drains intersecting recently dredged road-side ditches.

Table 2.4. Summary of the measurements and calculations made to determine m_{abc}, sorted by sedimentary feature

Sedimentary Feature	Feature Geometry	Survey Measurements			Survey and Lab Procedures	m_{abc} =	Spatial Extent Surveyed
		Volumetric	Mass	Abundance			
Sediment Fans	Discrete, up-ended triangular prism-shaped	l_f [a], w_f [b], h_f [c]	BD_f [d]	n_f [e]	1. l_f, w_f, and h_f measured 2. one 90.49 cm^3 soil core collected 3. n_f counted 4. soil air-dried 5. soil < 2 mm weighed to measure BD_f	$\overline{(\frac{1}{2} * l_f * w_f * h_f * BD_f)} * \overline{n_f}$	Boyne-Morris Watershed
Snirt	Continuous, sheet-shaped	-	m_f [f]	-	1. snirt collected along transect 2. snirt melted, decanted, and remaining soil air-dried 3. soil < 2 mm weighed to measure m_f	$\overline{m_f} * 1609$ [g]	Boyne-Morris Watershed
Blow Dirt	Continuous, sheet-shaped	h_f	BD_f	-	1. h_f measured at 1 m intervals along width of transect to measure s_f and b_f 2. one 90.49 cm^3 soil core collected 3. soil air-dried soil < 2 mm weighed to measure BD_f	$\overline{\left(\sum BD_f * \int (s_f^{h} * w_f + b_f^{i}) dw_f\right)} * 1609$	Boyne-Morris Watershed
Tillage Throw	Continuous, wedge-shaped	-	m_f	-	1. tillage throw collected along transect 2. soil air-dried soil < 2 mm weighed to measure m_f	$\overline{m_f} * 1609$	Boyne-Morris Watershed

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Table 2.4. (continued from previous page)

Dredged Spoil Piles	Continuous, triangular prism-shaped	w_f, h_f	BD_f	-	1. w_f and h_f measured 2. three 90.49 cm^3 soil cores collected 3. soil air-dried 4. soil < 2 mm weighed to measure BD_f	$\overline{(\frac{1}{2} * 1 * w_f * h_f * BD_f)} * 1609$	RM of MacDonald
Gravel Sheets	Continuous, sheet-shaped	-	m_f	-	1. soil > 2 mm from snirt samples weighed to measure m_f over 3-month period of snirt deposition	$\overline{(m_f * 4)} * 1609$	Boyne-Morris Watershed
Bioturbated Soil Piles	Discrete, cone-shaped	l_f, w_f, h_f	BD_f	n_f	1. l_f, w_f, and h_f measured 2. one 90.49 cm^3 soil core collected 3. n_f counted 4. soil air-dried 5. soil < 2 mm weighed to measure BD_f	$\overline{(\frac{1}{3} * \pi * h_f \frac{l_f}{2} * \frac{w_f}{2} * BD_f)} * \overline{n_f}$	Boyne-Morris Watershed

Notes. For all features, feature length and width were treated as the lengths parallel and perpendicular to the length of the road-side ditch, respectively. Feature height can be considered synonymous with depth where appropriate. The abundance of sediment fans per unit length of ditch was inferred from counts of in-field surface drain outlets, not of sediment fans themselves, reflecting the observed scarcity of sediment fans in the surveyed watersheds. Continuous features were measured along a 1 m long transect, extending width-wise away from the edge of the feature's sediment source to the feature's edge. Measurements of snirt and gravel sheets were made from 2015-2018.

[a] l_f = feature length

[b] w_f = feature width

[c] h_f = feature height

[d] BD_f = feature bulk density

[e] n_f = feature count per unit length of ditch

[f] m_f = feature mass

[g] meters in 1 mile

(continued on next page)

Table 2.4. (continued from previous page)

[h] s_f = feature slope

[i] b_f = feature y-intercept

Table 2.5. Hysteretic relationships between TSS and daily discharge at WSC stations 05OF024 and 05OG008

Watershed	Year	Flow Duration	Linear	Clockwise	Counter-Clockwise
05OG008	2010	Mar. 18 – Apr. 26			■
		Apr. 26 – Jul. 26			■
		Jul. 26 – Oct. 25	■		
	2011	Apr. 5 – Apr. 16	■		
		Apr. 26 – Jul. 17			■
	2012	May 22 – Jul. 30			■
	2013	Apr. 26 – May 13		■	
		May 28 – Jun. 18			■
		Jun. 18 – Jul. 22	■		
	2014	Apr. 20 – May 1			■
		May 1 – May 10			■
		Jun. 4 – Jun. 16			■
		Jun. 24 – Jul. 9	■		
		Jul. 9 – Jul. 28			■
	2015	Mar. 16 – Apr. 26		■	
		Apr. 30 – Jun. 10			■
		Jun. 23 – Aug. 9		■	
05OF024	2014	Apr. 10 – May 12		■	
		May 20 – Aug. 14			■
	2015	Mar. 15 – Apr. 15		■	
		Apr. 23 – Jun. 25	■		
		Jun. 29 – Jul. 30			■
	2016	Mar. 10 – Mar. 31			■
		Apr. 7 – May 24			■
		May 26 – Jun. 20		■	
		Jun. 27 – Jul. 11			■
		Jul. 11 – Oct. 11			■

Table 2.6. Average annual sediment transfers from erodible hillslopes in cultivated fields at the catchment- ($\bar{Q}_{ab}$) and sub-watershed-scales ($\bar{Q}_{a}$) from 1981-2011, inclusive

Sub-Watershed	Sedimentary Process	$\bar{Q}_{ab}$ (t yr^{-1})			$\bar{Q}_{a}$ (t yr^{-1})
		Coarse	Medium	Fine	
05OF024	Water Erosion	721	10,013	23,826	34,560
	Wind Erosion	5,376	30,706	33,828	69,910
	Tillage Erosion	484	7,179	39,183	46,846
05OG008	Water Erosion	-	-	6,682	6,682
	Wind Erosion	-	-	6,975	6,975
	Tillage Erosion	-	-	2,186	2,186

Note. Values of Y_{abcd}, A_{bd}, *and* P_{bde} *used to derive these averages can be found in Appendix A.*

Table 2.7. Masses of sediment transferred into and out of road-side ditches per kilometer of ditch in one year (m_{abc})

Sub-Watershed	Catchment	Sedimentary Feature	$\overline{m}_f$ (t)	$\overline{n}_f$ (count 1.609 km^{-1})	m_{abc} (t km^{-1})
05OF024	Coarse	Sediment Fans	0.510	5	1.6
		Snirt	0.003	1,609	3.0
		Blow Dirt	1.036	1,609	1,036.0
		Tillage Throw	0.016	1,609	16.0
		Dredged Spoil Piles	2.140	1,609	2,140.0
		Gravel Sheets	0.003	1,609	3.0
		Bioturbated Soil Piles	0.028	28	0.5
	Medium	Sediment Fans	0.510	5	1.6
		Snirt	0.003	1,609	3.0
		Blow Dirt	1.036	1,609	1,036.0
		Tillage Throw	0.016	1,609	16.0
		Dredged Spoil Piles	2.140	1,609	2,140.0
		Gravel Sheets	0.003	1,609	3.0
		Bioturbated Soil Piles	0.028	54	0.9
	Fine	Sediment Fans	0.510	5	1.6
		Snirt	0.003	1,609	3.0
		Blow Dirt	1.036	1,609	1,036.0
		Tillage Throw	0.016	1,609	16.0
		Dredged Spoil Piles	2.140	1,609	2,140.0
		Gravel Sheets	0.003	1,609	3.0
		Bioturbated Soil Piles	0.028	33	0.6
05OG008	Fine	Sediment Fans	0.510	5	1.6
		Snirt	0.003	1,609	3.0
		Blow Dirt	1.036	1,609	1,036.0
		Tillage Throw	0.016	1,609	16.0
		Dredged Spoil Piles	2.140	1,609	2,140.0
		Gravel Sheets	0.003	1,609	3.0
		Bioturbated Soil Piles	0.028	33	0.6

Note. Values used to derive $\overline{m}_f$ and $\overline{n}_f$ can be found in Appendix B.

Table 2.8. Masses of sediment transferred into and out of road-side ditches at the catchment-scale in one year (Q_{abc})

Sub-Watershed	Catchment	Sedimentary Feature	l_{ab} (km)	p_{abc} (%)	m_{abc} (t km^{-1})	Q_{abc} (t)
05OF024	Coarse	Sediment Fans	76	7	1.6	8.5
		Snirt	76	100	3.0	228.0
		Blow Dirt	76	3	1,036.0	2,362.1
		Tillage Throw	76	13	16.0	158.1
		Dredged Spoil Piles	99	7	2,140.0	14,830.2
		Gravel Sheets	86	100	3.0	258.0
		Bioturbated Soil Piles	99	100	0.5	49.5
	Medium	Sediment Fans	625	7	1.6	70.0
		Snirt	625	100	3.0	1,875.0
		Blow Dirt	625	3	1,036.0	19,425.0
		Tillage Throw	625	81	16.0	8,100.0
		Dredged Spoil Piles	720	7	2,140.0	107,856.0
		Gravel Sheets	602	100	3.0	1,806.0
		Bioturbated Soil Piles	720	78	0.9	505.4
	Fine	Sediment Fans	901	7	1.6	100.9
		Snirt	901	100	3.0	2,703.0
		Blow Dirt	901	3	1,036.0	28,003.1
		Tillage Throw	901	70	16.0	10,091.2
		Dredged Spoil Piles	1,120	7	2,140.0	167,776.0
		Gravel Sheets	1,100	100	3.0	3,300.0
		Bioturbated Soil Piles	1,120	30	0.6	201.6
05OG008	Fine	Sediment Fans	367	7	1.6	41.1
		Snirt	367	100	3.0	1,101.0
		Blow Dirt	367	3	1,036.0	11,406.4
		Tillage Throw	367	70	16.0	4,110.0
		Dredged Spoil Piles	444	7	2,140.0	66,511.2
		Gravel Sheets	174	100	3.0	522.0
		Bioturbated Soil Piles	444	30	0.6	79.9

Note. All transfer processes related to the observed sedimentary features are assumed to occur annually, with the exception of blow dirt, which was assumed to be deposited once every 30-years. Values used to derive p_{abc} can be found in Appendix B.

Table 2.9. Average annual sediment transfers into and out of road-side ditches at the catchment- ($\bar{Q}_{ab}$) and sub-watershed-scales ($\bar{Q}_{a}$)

Sub-Watershed	Sedimentary Process	Sedimentary Feature(s)	$\bar{Q}_{ab}$ (t yr^{-1})			$\bar{Q}_{a}$ (t yr^{-1})
			Coarse	Medium	Fine	
05OF024	Water Erosion	Sediment Fans	8.5	70.0	100.9	179.4
	Wind Erosion	Snirt and Blow Dirt	234.9	1,931.8	2,784.9	4,951.6
	Tillage Erosion	Tillage Throw	158.1	8,100.0	10,091.2	18,349.3
	Ditch Dredging	Dredged Spoil Piles	14,830.2	107,856.0	167,776.0	290,462.2
	Road Surface Erosion	Gravel Sheets	258.0	1,806.0	3,300.0	5,364.0
	Animal Burrowing	Bioturbated Soil Piles	49.5	505.4	201.6	756.5
05OG008	Water Erosion	Sediment Fans	-	-	41.1	41.1
	Wind Erosion	Snirt and Blow Dirt	-	-	1,134.4	1,134.4
	Tillage Erosion	Tillage Throw	-	-	4,110.0	4,110.0
	Ditch Dredging	Dredged Spoil Piles	-	-	66,511.2	66,511.2
	Road Surface Erosion	Gravel Sheets	-	-	522.0	522.0
	Animal Burrowing	Bioturbated Soil Piles	-	-	79.9	79.9

Note. Averages were calculated using values of Q_{abc} *aggregated by sedimentary transfer process, accounting for irregularity observed sedimentary features.*

Table 2.10. Annual (Q_{abc}) and average annual ($\bar{Q}_a$) sediment transfers through streams at the sub-watershed-scale

Sub-Watershed	Q_{abc} (t yr^{-1})								$\bar{Q}_a$ (t yr^{-1})
	2010	2011	2012	2013	2014	2015	2016	2017	
05OF024	-	-	-	-	461.4	389.1	2,116.0	14,565.5	757.8
05OG008	1,005.2	1,226.2	15.5	638.5	234.8	98.7	-	-	73.1

Table 2.11. Sediment budget transfers, presented in terms of inputs (*I*), storage (*S*), and outputs (φ)

Sub-Watershed	Sediment Transfer Direction	*I*		*S*		φ	
		Sedimentary Process/Feature	$\bar{Q}_a$ (t yr^{-1})	Sedimentary Process/Feature	$\bar{Q}_a$ (t yr^{-1})	Sedimentary Process/Feature	$\bar{Q}_a$ (t yr^{-1})
05OF024	Field to Stream	Water Erosion	34,560	Sediment Fans	179		
		Wind Erosion	69,910	Snirt and Blow Dirt	4,952		
		Tillage Erosion	46,846	Tillage Throw	18,349		
				In-Field Deposition	127,078		
						Suspended Sediment	758
	Field to Ditch	Water Erosion	34,560			Sediment Fans	179
		Wind Erosion	69,910			Snirt and Blow Dirt	4,952
		Tillage Erosion	46,846			Tillage Throw	18,349
				In-Field Deposition	127,078		
	Ditch to Field	Sediment Fans	179				
		Snirt and Blow Dirt	4,952				
		Tillage Throw	18,349				
		Gravel Sheets	5,364				
		Bioturbated Soil Piles	757				
				Dredging Imbalance	-265,813	Dredged Sediment Spoil Piles	290,462

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Table 2.11. (continued from previous page)

05OG008	Field to Stream	Water Erosion	6,682	Sediment Fans	41		
		Wind Erosion	6,975	Snirt and Blow Dirt	1,134		
		Tillage Erosion	2,186	Tillage Throw	4,110		
				Tillage Imbalance	-1,924		
				In-Field Deposition	12,408		
						Suspended Sediment	73
	Field to Ditch	Water Erosion	6,682			Sediment Fans	41
		Wind Erosion	6,975			Snirt and Blow Dirt	1,134
		Tillage Erosion	2,186	Tillage Imbalance	-1,924	Tillage Throw	4,110
				In-Field Deposition	12,408		
	Ditch to Field	Sediment Fans	41				
		Snirt and Blow Dirt	1,134				
		Tillage Throw	4,110				
		Gravel Sheets	522				
		Bioturbated Soil Piles	80				
				Dredging Imbalance	-60,624	Dredged Sediment Spoil Piles	66,511

Appendix A. Data used to Calculate Average Annual Sediment Transfers from Erodible Hillslopes in Cultivated Fields

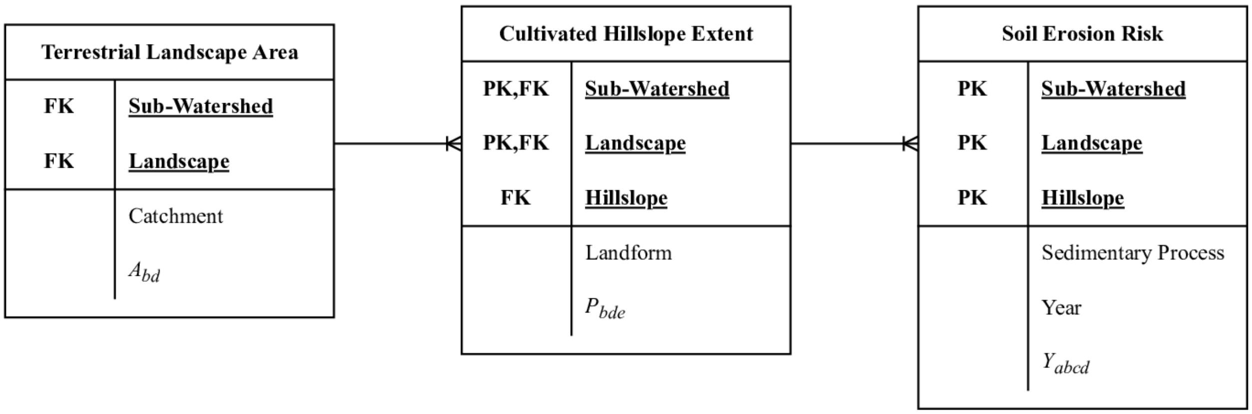

Figure A.1. Entity relationship diagram describing the linkages between the values of A_{bd}, P_{bde}, and Y_{abcd}, used to calculate Q_{abc} for erodible hillslopes in cultivated fields. Values of Q_{abc} were, in turn, used to calculate relevant values of $\bar{Q}_{ab}$ and $\bar{Q}_{a}$.

Table A.1. Tabulated values of A_{bd} used to calculate Q_{abc} for erodible hillslopes in cultivated fields

Sub-Watershed	Catchment	Landscape	A_{bd} (ha)
05OF024	Coarse	852004	5675
	Medium	852005	30410
		854001	1450
	Fine	766006	12135
		849006	21591
		849009	1399
		852001	12287
		852003	13219
		854002	3
05OG008	Fine	849009	9054
		851003	9874

Table A.2. Tabulated values of P_{bde} used to calculate Q_{abc} for erodible hillslopes in cultivated fields

Sub-Watershed	Landscape	Landform	Hillslope	P_{bde} (%)
05OF024	766006	[a]HB	[e]M	22.75
		HB	[f]U	19.50
		[b]UA	M	7.50
		UA	U	3.00
	849006	[c]LA	M	35.00
		LA	U	10.00
	849009	LA	M	34.30
		LA	U	9.80
		UA	M	1.00
		UA	U	0.40
	852001	LA	M	35.00
		LA	U	10.00
	852003	LA	M	31.50
		LA	U	9.00
		[d]RA	M	5.50
		RA	U	2.00
	852004	LA	M	22.75
		LA	M	12.25
		LA	U	6.50
		LA	U	3.50
	852005	LA	M	32.55
		LA	U	9.30
		UA	M	3.50
		UA	U	1.40
	854001	HB	M	24.50
		HB	U	21.00
		UA	M	15.00
		UA	U	6.00

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Table A.2. (continued from previous page)

05OF024	854002	HB	M	10.50
		HB	U	9.00
		UA	M	35.00
		UA	U	14.00
05OG008	849009	LA	M	34.30
		LA	U	9.80
		UA	M	1.00
		UA	U	0.40
	851003	LA	M	24.50
		LA	U	7.00
		UA	M	15.00
		UA	U	6.00

Note. Values of P_{bde} for level landforms with gradients between 0 and 3% follow updated SLC geodatabase LDT table definitions suggested by Li (2008) and used to calculated NAHARP water, wind, and tillage erosion risk indicators (AAFC, 2016b, AAFC, 2016c; AAFC, 2016d).

[a] HB: Hummocky, 4-9% gradient

[b] UA: Undulating, 0-3% gradient

[c] LA: Level, 0-3% gradient

[d] RA: Ridged, 0-3%

[e]M: Middle-slope

[f]U: Upper-slope

Table A.3. Tabulated values of Y_{abcd} used to calculate Q_{abc} for erodible hillslopes in cultivated fields

Sub-Watershed	Landscape	Hillslope	Sedimentary Process	Year	Y_{abcd} (t ha^{-1} yr^{-1})	Sub-Watershed	Landscape	Hillslope	Sedimentary Process	Year	Y_{abcd} (t ha^{-1} yr^{-1})
05OF024	766006	[a]M	Water Erosion	1981	3.57	05OF024	849006	M	Water Erosion	1981	0.62
				1986	3.48					1986	0.62
				1991	3.29					1991	0.59
				1996	3.41					1996	0.61
				2001	3.57					2001	0.63
				2006	3.47					2006	0.63
				2011	3.63					2011	0.66
		[b]U	Wind Erosion	1981	2.33			U	Wind Erosion	1981	4.35
				1986	2.13					1986	4.91
				1991	2.07					1991	3.98
				1996	2.12					1996	4.64
				2001	2.61					2001	4.51
				2006	2.57					2006	5.60
				2011	2.51					2011	5.27
			Tillage Erosion	1981	14.27				Tillage Erosion	1981	0.84
				1986	13.35					1986	0.78
				1991	12.92					1991	0.74
				1996	13.02					1996	0.77
				2001	12.91					2001	0.82
				2006	11.81					2006	0.78
				2011	11.91					2011	0.77

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Table A.3. (continued from previous page)

05OF024	849009	M	Water Erosion	1981	0.73	05OF024	852001	M	Water Erosion	1981	0.72
				1986	0.74					1986	0.72
				1991	0.69					1991	0.68
				1996	0.73					1996	0.72
				2001	0.74					2001	0.76
				2006	0.74					2006	0.74
				2011	0.78					2011	0.78
		U	Wind Erosion	1981	3.50			U	Wind Erosion	1981	3.59
				1986	3.93					1986	4.21
				1991	3.38					1991	3.36
				1996	3.97					1996	3.98
				2001	3.93					2001	4.22
				2006	4.83					2006	4.06
				2011	4.54					2011	4.31
			Tillage Erosion	1981	0.85				Tillage Erosion	1981	0.88
				1986	0.81					1986	0.82
				1991	0.74					1991	0.77
				1996	0.80					1996	0.82
				2001	0.82					2001	0.87
				2006	0.78					2006	0.82
				2011	0.79					2011	0.82

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Table A.3. (continued from previous page)

05OF024	852003	M	Water Erosion	1981	0.60	05OF024	852004	M	Water Erosion	1981	0.39
				1986	0.58					1986	0.36
				1991	0.56					1991	0.38
				1996	0.58					1996	0.36
				2001	0.58					2001	0.37
				2006	0.55					2006	0.35
				2011	0.58					2011	0.34
		U	Wind Erosion	1981	8.13			U	Wind Erosion	1981	10.19
				1986	8.27					1986	9.40
				1991	7.35					1991	8.39
				1996	7.51					1996	9.02
				2001	8.74					2001	9.46
				2006	7.77					2006	10.16
				2011	8.83					2011	9.69
			Tillage Erosion	1981	0.87				Tillage Erosion	1981	0.94
				1986	0.80					1986	0.84
				1991	0.80					1991	0.88
				1996	0.83					1996	0.86
				2001	0.81					2001	0.85
				2006	0.78					2006	0.84
				2011	0.76					2011	0.74

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Table A.3. (continued from previous page)

05OF024	852005	M	Water Erosion	1981	0.82	05OF024	854001	M	Water Erosion	1981	2.52
				1986	0.76					1986	2.36
				1991	0.75					1991	2.30
				1996	0.77					1996	2.47
				2001	0.81					2001	2.26
				2006	0.78					2006	2.21
				2011	0.84					2011	2.27
		U	Wind Erosion	1981	9.26			U	Wind Erosion	1981	1.56
				1986	9.79					1986	1.24
				1991	8.11					1991	1.19
				1996	8.83					1996	1.36
				2001	9.65					2001	1.21
				2006	9.22					2006	1.27
				2011	10.11					2011	1.15
			Tillage Erosion	1981	1.04				Tillage Erosion	1981	12.16
				1986	0.94					1986	11.32
				1991	0.91					1991	11.12
				1996	0.94					1996	11.28
				2001	0.94					2001	9.95
				2006	0.87					2006	9.34
				2011	0.93					2011	8.53

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Table A.3. (continued from previous page)

05OF024	854002	M	Water Erosion	1981	2.20	05OG008	849009	M	Water Erosion	1981	0.73
				1986	2.13					1986	0.74
				1991	2.02					1991	0.69
				1996	2.19					1996	0.73
				2001	2.05					2001	0.74
				2006	2.08					2006	0.74
				2011	2.18					2011	0.78
		U	Wind Erosion	1981	1.59			U	Wind Erosion	1981	3.50
				1986	1.26					1986	3.93
				1991	1.15					1991	3.38
				1996	1.32					1996	3.97
				2001	1.30					2001	3.93
				2006	1.52					2006	4.83
				2011	1.43					2011	4.54
			Tillage Erosion	1981	6.00				Tillage Erosion	1981	0.85
				1986	5.74					1986	0.81
				1991	5.51					1991	0.74
				1996	5.68					1996	0.80
				2001	5.03					2001	0.82
				2006	4.88					2006	0.78
				2011	4.93					2011	0.79

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Table A.3. (continued from previous page)

05OG008	851003	M	Water Erosion	1981	1.14
				1986	1.14
				1991	1.03
				1996	1.07
				2001	1.10
				2006	1.11
				2011	1.19
		U	Wind Erosion	1981	2.36
				1986	2.41
				1991	2.05
				1996	2.21
				2001	2.68
				2006	3.02
				2011	3.11
			Tillage Erosion	1981	1.21
				1986	1.16
				1991	1.07
				1996	1.13
				2001	1.16
				2006	1.09
				2011	1.10

[a]M: Middle-slope

[b]U: Upper-slope

Appendix B. Data used to Calculate Annual Sediment Transfers into and out of Road-Side Ditches

Table B.1. Sediment fan morphometrics measured following a rain storm on July. 29, 2015; used to calculate $\bar{m}_f$ for sediment fans

Section	Adjacent Road	l_f (m)	w_f (m)	h_f (m)	BD_f (t m^{-3})
21-7-1W	HWY 305	3.2	4.1	0.25	0.79
21-7-1W	HWY 305	3.6	2.7	0.20	0.80*
21-7-1W	HWY 305	10.1	2.4	0.14	0.80*
21-7-1W	HWY 305	3.8	1.8	0.20	0.79
21-7-1W	HWY 305	2.9	2.0	0.22	0.80*
21-7-1W	HWY 305	1.5	1.5	0.20	0.80
21-7-1W	HWY 305	5.5	2.4	0.21	0.80*
21-7-1W	HWY 305	3.3	2.2	0.12	0.80*
21-7-1W	HWY 305	4.0	3.3	0.08	0.80*
21-7-1W	HWY 305	3.5	2.2	0.15	0.80*
29-7-1W	HWY 305	3.0	2.4	0.06	0.80*
29-7-1W	HWY 305	3.0	2.3	0.09	0.79
29-7-1W	HWY 305	2.3	3.0	0.14	0.80*
29-7-1W	HWY 305	2.0	1.6	0.06	0.80*
29-7-1W	HWY 305	3.2	3.0	0.08	0.80*
29-7-1W	HWY 305	3.4	2.7	0.11	0.83
29-7-1W	HWY 305	1.3	2.0	0.06	0.77

*Average BD_f used in place of measured values

Table B.2. Snirt morphometrics; used to calculate $\overline{m}_f$ for snirt

Year	Section	Adjacent Road	m_f (t)
2015	29-9-4W	51N	0.003
	29-9-4W	51N	0.004
	29-9-4W	51N	0.003
2016	29-9-4W	51N	0.002
	29-9-4W	51N	0.002
	7-6-2W	HWY 336	0.001
	7-6-2W	HWY 336	0.002
2017	12-5-2W	26N	0.004
	29-9-4W	51N	0.001
	29-9-4W	51N	0.001
2018	24-6-4W	33N	0.007
	29-9-4W	51N	0.003

Table B.3. Blow dirt morphometrics measured following a dust storm on April 29, 2018; used to calculate $\overline{m}_f$ for blow dirt

Section	Adjacent Road	BD_f (t m^{-3})	s_f (m m^{-1})	w_f (m)	b_f (m)
24-7-5W	HWY 305	1.50	0.080	1.00	0.140
			-0.040	1.00	0.220
			-0.115	1.00	0.180
			0.010	1.00	0.065
			0.020	1.00	0.075
			0.015	1.00	0.095
			-0.040	1.00	0.110
			0.025	1.00	0.070
			0.020	1.00	0.095
			0.075	1.00	0.115
			-0.050	1.00	0.190
			-0.010	1.00	0.140
			-0.030	1.00	0.130
			-0.005	1.00	0.100
			-0.025	1.00	0.095
			-0.060	1.00	0.060
9-9-1E	50N	1.00	0.070	1.00	0.090
			0.040	1.00	0.160
			-0.060	1.00	0.120
			-0.120	0.50	0.060
7-6-3W	32N	1.01	0.020	1.00	0.160
			0.010	1.00	0.180
			-0.010	1.00	0.190
			0.015	1.00	0.180
			-0.065	1.00	0.195
			-0.025	1.00	0.130
28-8-1E	HWY 247	1.33	0.190	1.00	0.130
			-0.030	1.00	0.320
			0.050	1.00	0.290
			-0.200	1.00	0.340
			-0.080	1.00	0.140
			-0.060	1.00	0.060
12-6-3W	32N	1.24	0.030	1.00	0.100
			0.150	1.00	0.130
			0.230	1.00	0.280
			-0.310	1.00	0.510
			-0.020	1.00	0.200
			-0.060	1.00	0.180
			-0.030	1.00	0.120
			-0.020	1.00	0.090

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Table B.3. (continued from previous page)

22-6-4W	HWY 3	1.11	-0.010	1.00	0.110
			0.030	1.00	0.100
			-0.020	1.00	0.130
			0.030	1.00	0.110
			-0.040	1.00	0.140
8-9-3W	HWY 101	1.10	0.040	1.00	0.170
			-0.090	1.00	0.210
			-0.040	1.00	0.120
			-0.267	0.30	0.080
19-9-2E	HWY 2	1.04	-0.025	1.00	0.115
			0.090	1.00	0.090
			0.010	1.00	0.180
			-0.035	1.00	0.190
			-0.040	1.00	0.155
			-0.040	1.00	0.115
			-0.075	1.00	0.075

Table B.4. Tillage throw morphometrics measured in the autumn of 2018; used to calculate $\bar{m}_f$ for tillage throw

Quarter Section	Adjacent Road	m_f (t)
SW 5-6-2W	HWY 336	0.001
SW 3-6-2W	9W	0.020
NW 17-5-2W	HWY 336	0.004
NW 6-9-1E	49N	0.011
NW 3-9-1E	49N	0.003
SW 7-5-8W	HWY 244	0.053

Table B.5. Dredged spoil pile morphometrics measured in the autumn of 2018; used to calculate $\bar{m}_f$ for dredged spoil piles

Section	Adjacent Road	BD_f (t m^{-3})	w_f (m)	h_f (m)
33-9-1E	54N	1.08	2.8	1.0
			4.3	1.4
			5.2	1.6
			5.7	2.0
			5.0	1.6
4-10-1E	54N	0.95	3.8	0.1
			2.6	1.2
			3.5	1.3
			2.9	0.5
			2.5	0.9
18-8-2W	HWY 247	1.04	6.1	0.4
			4.8	0.6
			5.2	0.4
			6.2	0.6
			6.3	0.6
7-8-1W	HWY 332	1.05	2.0	0.6
			2.6	1.1
			3.4	1.1
			3.5	1.3
			2.6	1.0
19-7-1W	HWY 332	1.08	3.0	1.7
			3.1	1.2
			3.7	1.6
			3.5	1.4
			4.0	1.4

Table B.6. Gravel sheet morphometrics; used to calculate $\overline{m}_f$ for gravel sheets

Year	Section	Adjacent Road	m_f (t)
2015	29-9-4W	51N	0.000
	29-9-4W	51N	0.001
	29-9-4W	51N	0.000
2016	29-9-4W	51N	0.002
	29-9-4W	51N	0.002
	7-6-2W	HWY 336	0.001
	7-6-2W	HWY 336	0.001
2017	12-5-2W	26N	0.000
	29-9-4W	51N	0.001
	29-9-4W	51N	0.000
2018	24-6-4W	33N	0.000
	29-9-4W	51N	0.001

Table B.7. Bioturbated soil pile morphometrics measured in the autumn of 2018; used to calculate $\bar{m}_f$ for bioturbated soil piles

Section	Adjacent Road	BD_f (t m^{-3})	h_f (m)	l_f (m)	w_f (m)
29-7-4W	HWY 305	1.15	0.17	0.48	0.60
			0.13	0.50	0.70
			0.17	0.62	0.43
			0.19	0.77	0.83
			0.16	0.73	0.58
22-7-5W	HWY 305	1.14	0.21	0.71	0.88
			0.23	0.80	0.85
			0.21	0.56	0.57
			0.25	0.75	0.83
			0.18	0.57	0.48
30-7-5W	HWY 305	0.94	0.29	1.26	1.18
			0.24	0.89	0.76
			0.19	1.18	1.12
			0.19	0.62	0.77
			0.23	0.74	0.74
23-7-4W	HWY 305	0.77	0.20	0.75	1.10
		0.95	0.08	0.46	0.33
		0.74	0.14	0.54	0.63
		0.98	0.17	0.73	0.77
		0.83	0.13	0.40	0.72
		0.80	0.16	0.53	0.61
		0.36	0.07	0.63	0.50
		0.76	0.17	0.74	0.61
		0.65	0.10	0.40	0.59

Section	Adjacent Road	BD_f (t m^{-3})	h_f (m)	l_f (m)	w_f (m)
31-5-6W	HWY 338	0.67	0.13	0.87	0.70
			0.13	0.42	0.25
			0.16	0.65	0.54
			0.14	0.45	0.41
			0.11	0.55	0.30
18-6-6W	HWY 338	0.99	0.24	0.70	0.60
			0.20	0.84	0.72
			0.23	1.20	0.85
			0.19	0.65	0.50
			0.25	0.72	0.64
7-7-6W	HWY 338	1.31	0.21	0.67	0.75
			0.25	0.60	0.70
			0.18	0.67	0.67
			0.25	0.62	0.42
			0.20	0.77	1.30
19-7-6W	HWY 338	1.10	0.17	1.00	0.65
			0.20	0.95	0.84
			0.13	0.82	0.71
			0.22	1.00	0.70
			0.27	1.03	0.80

Table B.8. Surface drains per 1.609 km of road-side ditch; used to calculate $\bar{n}_f$ for sediment fans

Section	Adjacent Road	n_f	Section	Adjacent Road	n_f	Section	Adjacent Road	n_f	Section	Adjacent Road	n_f
36-9-2W	HWY 332	4	20-9-2E	51N	0	24-9-1W	HWY 334	5	22-8-2E	10E	7
17-7-1W	HWY 332	8	20-9-1E	2E	2	6-9-1E	HWY 334	6	23-8-2E	10E	4
6-7-1W	HWY 332	9	29-9-1E	HWY 334	7	31-8-1E	HWY 334	5	22-8-2E	9E	10
5-7-1W	HWY 332	10	33-9-1E	HWY 334	8	1-9-1W	HWY 334	4	21-8-2E	9E	5
7-7-1W	HWY 332	12	32-9-1E	HWY 334	5	36-8-1W	HWY 334	1	27-8-2E	9E	6
8-7-1W	HWY 332	7	24-9-1E	51N	3	25-8-1W	HWY 334	3	31-8-1E	1E	6
18-7-1W	HWY 332	15	24-9-1E	6E	4	30-8-1E	HWY 334	1	32-8-1E	1E	3
19-7-1W	HWY 332	8	24-9-1E	5E	1	19-8-1E	HWY 247	0	31-8-1E	48N	4
21-9-1E	2E	3	13-9-1E	51N	0	30-8-1E	HWY 247	0	6-9-1E	48N	4
7-9-1E	1E	5	19-9-2E	6E	9	31-8-2E	48N	5	1-9-1W	48N	7
17-9-1E	1E	6	23-9-1E	5E	2	6-9-2E	48N	4	36-8-1W	48N	3
5-7-2W	11W	11	19-9-2E	7E	7	32-8-2E	48N	9	35-8-1W	48N	7
8-9-1E	49N	2	20-9-2E	7E	4	33-8-2E	48N	9	2-9-1W	48N	1
10-9-1E	49N	5	35-9-2W	HWY 332	10	4-9-2E	48N	4	6-9-1E	49N	0
20-7-1W	HWY 332	5	35-9-1W	HWY 424	0	5-9-2E	48N	5	5-9-1E	49N	3
6-7-2W	11W	7	26-9-2W	HWY 332	5	19-8-3E	12E	8	4-9-1E	49N	2
9-9-1E	49N	3	25-9-2W	HWY 332	2	24-8-2E	12E	9	3-9-1E	49N	7
20-9-1E	1E	6	26-9-1W	HWY 424	2	25-8-2E	12E	3	5-9-1E	48N	7
19-9-1E	1E	5	27-9-1W	HWY 424	1	30-8-3E	12E	7	32-8-1E	48N	1
28-9-1E	HWY 334	8	7-9-1E	49N	2	26-8-2E	11E	8	8-7-2W	11W	1
18-9-1E	1E	7	12-9-1W	HWY 334	6	25-8-2E	11E	7	7-7-2W	11W	4
8-9-1E	1E	1	7-9-1E	HWY 334	4	24-8-2E	11E	4	17-7-2W	11W	8
20-9-2E	8E	4	18-9-1E	HWY 334	5	23-8-2E	11E	6	18-7-2W	11W	11
17-9-2E	51N	7	13-9-1W	HWY 334	6	27-8-2E	10E	8	20-7-2W	11W	7
34-9-1W	HWY 424	5	19-9-1E	HWY 334	5	26-8-2E	10E	6	19-7-2W	11W	6

Table B.9. Bioturbated soil piles per 1.609 km of road-side ditch measured in the autumn of 2018; used to calculate $\bar{n}_f$ for bioturbated soil piles

Catchment	Section	Adjacent Road	n_f	Catchment	Section	Adjacent Road	n_f
Coarse	21-7-6W	HWY 305	2	Fine	5-5-2W	HWY 336	0
	19-7-6W	HWY 338	11		8-5-2W	HWY 336	0
	18-7-6W	HWY 338	17		17-5-2W	HWY 336	0
	7-7-6W	HWY 338	109		20-5-2W	HWY 336	0
	5-7-6W	HWY 338	16		29-5-2W	HWY 336	21
	31-6-6W	HWY 338	15		32-5-2W	HWY 336	29
Medium	20-7-4W	HWY 305	76		5-6-2W	HWY 336	0
	30-7-4W	HWY 305	18		8-6-2W	HWY 336	0
	24-7-5W	HWY 305	50		17-6-2W	HWY 336	1
	23-7-5W	HWY 305	1		20-6-2W	HWY 336	0
	22-7-5W	HWY 305	56		19-7-1W	HWY 305	0
	21-7-5W	HWY 305	20		24-7-2W	HWY 305	0
	20-7-5W	HWY 305	138		23-7-2W	HWY 305	0
	30-7-5W	HWY 305	71		22-7-2W	HWY 305	0
	24-7-6W	HWY 305	3		21-7-2W	HWY 305	0
	23-7-6W	HWY 305	29		20-7-2W	HWY 305	0
	22-7-6W	HWY 305	9		19-7-2W	HWY 305	0
	29-5-6W	HWY 338	24		24-7-3W	HWY 305	0
	31-5-6W	HWY 338	158		23-7-3W	HWY 305	0
	6-6-6W	HWY 338	0		22-7-3W	HWY 305	0
	7-6-6W	HWY 338	0		21-7-3W	HWY 305	0
	18-6-6W	HWY 338	101		20-7-3W	HWY 305	0
	19-6-6W	HWY 338	0		19-7-3W	HWY 305	25
	30-6-6W	HWY 338	0		24-7-4W	HWY 305	7
Fine	20-5-6W	HWY 338	0		23-7-4W	HWY 305	134
	17-5-6W	HWY 338	1		22-7-4W	HWY 305	23
	8-5-6W	HWY 338	0		21-7-4W	HWY 305	59

Table B.10. Ditch dredging archived by the RM of MacDonald Public Works; used to calculate p_{abc} for sediment fans and dredged spoil piles

Years	Dredged Distance (km)
2005-2007	223.651
2008-2010	268.703
2011-2013	358.807
2014-2016	371.277

Table B.11. Blow dirt deposition archived by the RM of MacDonald Public Works following a dust storm on April 29, 2018; used to determine p_{abc} for blow dirt

Section	Adjacent Road	Infilled Distance (km)	Section	Adjacent Road	Infilled Distance (km)	Section	Adjacent Road	Infilled Distance (km)
35-9-2W	54N	1.609	29-7-1W	41N	0.201	10-7-1E	38N	1.006
35-9-2W	53N	1.609	28-7-1W	41N	0.805	5-7-1E	37N	0.402
27-9-2W	8W	0.805	27-7-1W	41N	0.402	23-9-2E	51N	0.547
27-9-2W	52N	0.547	22-7-1W	39N	0.805	11-9-2E	50N	0.805
11-9-2W	50N	0.805	14-7-1W	39N	0.805	10-9-2E	50N	0.805
34-8-2W	48N	0.805	13-7-1W	39N	0.547	6-9-2E	49N	0.402
27-8-2W	47N	0.805	1-7-1W	37N	1.609	4-9-2E	49N	0.402
11-8-2W	43N	0.805	13-9-1E	51N	0.402	31-8-2E	48N	0.402
8-7-2W	38N	1.609	15-9-1E	50N	0.805	34-8-2E	48N	0.805
8-7-2W	37N	0.805	8-9-1E	50N	0.805	35-8-2E	48N	0.805
2-7-2W	37N	0.402	11-9-1E	49N	1.207	25-8-2E	47N	1.207
32-9-1W	53N	0.402	4-9-1E	49N	0.547	16-8-2E	45N	0.603
34-9-1W	53N	1.609	6-9-1E	48N	0.402	9-8-2E	44N	0.805
20-9-1W	51N	0.805	31-8-1E	48N	0.805	10-8-2E	44N	1.006
14-9-1W	51N	0.805	35-8-1E	48N	0.402	11-8-2E	44N	0.201
14-9-1W	1W	1.609	36-8-1E	48N	0.805	12-8-2E	44N	0.805
35-8-1W	48N	0.805	5-8-1E	42N	0.805	3-8-2E	43N	0.402
25-8-1W	47N	0.805	31-7-1E	42N	0.805	2-8-2E	43N	0.805
26-8-1W	47N	0.805	32-7-1E	42N	0.805	1-8-2E	43N	0.805
27-8-1W	47N	0.805	28-7-1E	41N	0.805	33-8-2E	42N	0.805
27-8-1W	2W	0.805	25-7-1E	41N	0.805	30-8-3E	47N	1.207
30-8-1W	5W	1.609	17-7-1E	39N	0.805	19-8-3E	46N	0.805
7-8-1W	43N	0.805	15-7-1E	39N	0.805	18-8-3E	45N	0.402
2-8-1W	43N	0.805	7-7-1E	38N	1.609	6-8-3E	43N	0.402
32-7-1W	42N	1.207	9-7-1E	38N	0.805			

Table B.12. Tillage throw deposition surveyed between October 26 and 29, 2018; used to determine p_{abc} for tillage throw

Catchment	Quarter Section	Tilled	Catchment	Quarter Section	Tilled	Catchment	Quarter Section	Tilled	Catchment	Quarter Section	Tilled
Coarse	NW 8-5-8W	TRUE	Coarse	NE 4-5-8W	TRUE	Coarse	SW 34-4-8W	FALSE	Coarse	SE 26-4-8W	TRUE
	NE 8-5-8W	FALSE		SW 4-5-8W	TRUE		SE 34-4-8W	TRUE		NW 25-4-8W	TRUE
	SW 8-5-8W	TRUE		SE 4-5-8W	TRUE		NW 35-4-8W	TRUE		NE 25-4-8W	TRUE
	SE 8-5-8W	TRUE		NW 3-5-8W	TRUE		NE 35-4-8W	TRUE		SW 25-4-8W	TRUE
	NW 9-5-8W	TRUE		NE 3-5-8W	TRUE		SW 35-4-8W	TRUE		SE 25-4-8W	TRUE
	NE 9-5-8W	TRUE		SW 3-5-8W	FALSE		SE 35-4-8W	TRUE		NW 20-4-8W	TRUE
	SW 9-5-8W	TRUE		SE 3-5-8W	FALSE		NW 36-4-8W	TRUE		NE 20-4-8W	TRUE
	SE 9-5-8W	TRUE		NW 2-5-8W	TRUE		NE 36-4-8W	TRUE		SW 20-4-8W	TRUE
	NW 10-5-8W	TRUE		NE 2-5-8W	TRUE		SW 36-4-8W	TRUE		SE 20-4-8W	TRUE
	NE 10-5-8W	TRUE		SW 2-5-8W	FALSE		SE 36-4-8W	TRUE		NW 21-4-8W	FALSE
	SW 10-5-8W	FALSE		SE 2-5-8W	FALSE		NW 29-4-8W	FALSE		NE 21-4-8W	TRUE
	SE 10-5-8W	FALSE		NW 1-5-8W	FALSE		NE 29-4-8W	FALSE		SW 21-4-8W	TRUE
	NW 11-5-8W	TRUE		NE 1-5-8W	FALSE		SW 29-4-8W	TRUE		SE 21-4-8W	TRUE
	NE 11-5-8W	TRUE		SW 1-5-8W	FALSE		SE 29-4-8W	FALSE		NW 22-4-8W	TRUE
	SW 11-5-8W	TRUE		SE 1-5-8W	TRUE		NW 28-4-8W	FALSE		NE 22-4-8W	TRUE
	SE 11-5-8W	TRUE		NW 32-4-8W	TRUE		NE 28-4-8W	TRUE		SW 22-4-8W	TRUE
	NW 12-5-8W	FALSE		NE 32-4-8W	TRUE		SW 28-4-8W	FALSE		SE 22-4-8W	TRUE
	NE 12-5-8W	TRUE		SW 32-4-8W	TRUE		SE 28-4-8W	TRUE		NW 23-4-8W	TRUE
	SW 12-5-8W	TRUE		SE 32-4-8W	TRUE		NW 27-4-8W	TRUE		NE 23-4-8W	TRUE
	SE 12-5-8W	TRUE		NW 33-4-8W	TRUE		NE 27-4-8W	TRUE		SW 23-4-8W	TRUE
	NW 5-5-8W	TRUE		NE 33-4-8W	TRUE		SW 27-4-8W	TRUE		SE 23-4-8W	TRUE
	NE 5-5-8W	TRUE		SW 33-4-8W	TRUE		SE 27-4-8W	TRUE		NW 24-4-8W	TRUE
	SW 5-5-8W	FALSE		SE 33-4-8W	TRUE		NW 26-4-8W	TRUE		NE 24-4-8W	TRUE
	SE 5-5-8W	TRUE		NW 34-4-8W	TRUE		NE 26-4-8W	TRUE		SW 24-4-8W	TRUE
	NW 4-5-8W	TRUE		NE 34-4-8W	TRUE		SW 26-4-8W	TRUE		SE 24-4-8W	TRUE

(continued on next page)

Table B.12. (continued from previous page)

Medium	NW 17-8-5W	FALSE	Medium	NE 9-8-5W	FALSE	Medium	SW 3-8-5W	FALSE	Medium	SE 35-7-5W	FALSE
	NE 17-8-5W	FALSE		SW 9-8-5W	FALSE		SE 3-8-5W	TRUE		NW 36-7-5W	FALSE
	SW 17-8-5W	FALSE		SE 9-8-5W	FALSE		NW 2-8-5W	FALSE		NE 36-7-5W	FALSE
	SE 17-8-5W	NA		NW 10-8-5W	FALSE		NE 2-8-5W	FALSE		SW 36-7-5W	FALSE
	NW 16-8-5W	FALSE		NE 10-8-5W	FALSE		SW 2-8-5W	FALSE		SE 36-7-5W	TRUE
	NE 16-8-5W	FALSE		SW 10-8-5W	FALSE		SE 2-8-5W	FALSE		NW 29-7-5W	FALSE
	SW 16-8-5W	NA		SE 10-8-5W	TRUE		NW 1-8-5W	FALSE		NE 29-7-5W	FALSE
	SE 16-8-5W	FALSE		NW 11-8-5W	FALSE		NE 1-8-5W	FALSE		SW 29-7-5W	TRUE
	NW 15-8-5W	FALSE		NE 11-8-5W	FALSE		SW 1-8-5W	FALSE		SE 29-7-5W	FALSE
	NE 15-8-5W	FALSE		SW 11-8-5W	FALSE		SE 1-8-5W	FALSE		NW 28-7-5W	TRUE
	SW 15-8-5W	FALSE		SE 11-8-5W	FALSE		NW 32-7-5W	FALSE		NE 28-7-5W	FALSE
	SE 15-8-5W	FALSE		NW 12-8-5W	TRUE		NE 32-7-5W	FALSE		SW 28-7-5W	FALSE
	NW 14-8-5W	FALSE		NE 12-8-5W	TRUE		SW 32-7-5W	FALSE		SE 28-7-5W	FALSE
	NE 14-8-5W	FALSE		SW 12-8-5W	FALSE		SE 32-7-5W	FALSE		NW 27-7-5W	FALSE
	SW 14-8-5W	FALSE		SE 12-8-5W	FALSE		NW 33-7-5W	TRUE		NE 27-7-5W	FALSE
	SE 14-8-5W	FALSE		NW 5-8-5W	FALSE		NE 33-7-5W	FALSE		SW 27-7-5W	FALSE
	NW 13-8-5W	FALSE		NE 5-8-5W	FALSE		SW 33-7-5W	FALSE		SE 27-7-5W	FALSE
	NE 13-8-5W	FALSE		SW 5-8-5W	FALSE		SE 33-7-5W	FALSE		NW 26-7-5W	FALSE
	SW 13-8-5W	FALSE		SE 5-8-5W	FALSE		NW 34-7-5W	FALSE		NE 26-7-5W	NA
	SE 13-8-5W	FALSE		NW 4-8-5W	FALSE		NE 34-7-5W	TRUE		SW 26-7-5W	FALSE
	NW 8-8-5W	FALSE		NE 4-8-5W	FALSE		SW 34-7-5W	FALSE		SE 26-7-5W	NA
	NE 8-8-5W	NA		SW 4-8-5W	FALSE		SE 34-7-5W	FALSE		NW 25-7-5W	TRUE
	SW 8-8-5W	FALSE		SE 4-8-5W	FALSE		NW 35-7-5W	FALSE		NE 25-7-5W	TRUE
	SE 8-8-5W	FALSE		NW 3-8-5W	FALSE		NE 35-7-5W	FALSE		SW 25-7-5W	TRUE
	NW 9-8-5W	NA		NE 3-8-5W	FALSE		SW 35-7-5W	FALSE		SE 25-7-5W	TRUE

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Table B.12. (continued from previous page)

Fine	NW 20-6-2W	TRUE	Fine	NE 16-6-2W	TRUE	Fine	SW 10-6-2W	TRUE	Fine	SE 2-6-2W	FALSE
	NE 20-6-2W	TRUE		SW 16-6-2W	TRUE		SE 10-6-2W	TRUE		NW 1-6-2W	TRUE
	SW 20-6-2W	TRUE		SE 16-6-2W	FALSE		NW 11-6-2W	TRUE		NE 1-6-2W	TRUE
	SE 20-6-2W	TRUE		NW 15-6-2W	TRUE		NE 11-6-2W	TRUE		SW 1-6-2W	TRUE
	NW 21-6-2W	TRUE		NE 15-6-2W	TRUE		SW 11-6-2W	TRUE		SE 1-6-2W	TRUE
	NE 21-6-2W	TRUE		SW 15-6-2W	FALSE		SE 11-6-2W	TRUE		NW 32-5-2W	TRUE
	SW 21-6-2W	TRUE		SE 15-6-2W	FALSE		NW 12-6-2W	TRUE		NE 32-5-2W	TRUE
	SE 21-6-2W	TRUE		NW 14-6-2W	TRUE		NE 12-6-2W	TRUE		SW 32-5-2W	FALSE
	NW 22-6-2W	TRUE		NE 14-6-2W	FALSE		SW 12-6-2W	FALSE		SE 32-5-2W	FALSE
	NE 22-6-2W	TRUE		SW 14-6-2W	TRUE		SE 12-6-2W	TRUE		NW 33-5-2W	TRUE
	SW 22-6-2W	TRUE		SE 14-6-2W	FALSE		NW 5-6-2W	TRUE		NE 33-5-2W	FALSE
	SE 22-6-2W	TRUE		NW 13-6-2W	TRUE		NE 5-6-2W	TRUE		SW 33-5-2W	FALSE
	NW 23-6-2W	FALSE		NE 13-6-2W	TRUE		SW 5-6-2W	TRUE		SE 33-5-2W	TRUE
	NE 23-6-2W	FALSE		SW 13-6-2W	TRUE		SE 5-6-2W	TRUE		NW 34-5-2W	FALSE
	SW 23-6-2W	FALSE		SE 13-6-2W	TRUE		NW 4-6-2W	TRUE		NE 34-5-2W	FALSE
	SE 23-6-2W	TRUE		NW 8-6-2W	TRUE		NE 4-6-2W	TRUE		SW 34-5-2W	FALSE
	NW 24-6-2W	TRUE		NE 8-6-2W	FALSE		SW 4-6-2W	TRUE		SE 34-5-2W	TRUE
	NE 24-6-2W	TRUE		SW 8-6-2W	TRUE		SE 4-6-2W	TRUE		NW 35-5-2W	TRUE
	SW 24-6-2W	TRUE		SE 8-6-2W	FALSE		NW 3-6-2W	TRUE		NE 35-5-2W	TRUE
	SE 24-6-2W	TRUE		NW 9-6-2W	TRUE		NE 3-6-2W	TRUE		SW 35-5-2W	TRUE
	NW 17-6-2W	TRUE		NE 9-6-2W	FALSE		SW 3-6-2W	TRUE		SE 35-5-2W	TRUE
	NE 17-6-2W	FALSE		SW 9-6-2W	TRUE		SE 3-6-2W	TRUE		NW 36-5-2W	FALSE
	SW 17-6-2W	TRUE		SE 9-6-2W	FALSE		NW 2-6-2W	TRUE		NE 36-5-2W	FALSE
	SE 17-6-2W	TRUE		NW 10-6-2W	FALSE		NE 2-6-2W	FALSE		SW 36-5-2W	FALSE
	NW 16-6-2W	TRUE		NE 10-6-2W	FALSE		SW 2-6-2W	FALSE		SE 36-5-2W	FALSE

Note. NA values in the Tilled column are representative of quarter-sections that were uncultivated or had poor road access.

Table B.13. Animal burrowing surveyed in October 2018; used to determine p_{abc} for bioturbated soil piles

Catchment	Section	Adjacent Road	Burrowed	Catchment	Section	Adjacent Road	Burrowed
Coarse	21-7-6W	HWY 305	TRUE	Fine	5-5-2W	HWY 336	FALSE
	19-7-6W	HWY 338	TRUE		8-5-2W	HWY 336	FALSE
	18-7-6W	HWY 338	TRUE		17-5-2W	HWY 336	FALSE
	7-7-6W	HWY 338	TRUE		20-5-2W	HWY 336	FALSE
	5-7-6W	HWY 338	TRUE		29-5-2W	HWY 336	TRUE
	31-6-6W	HWY 338	TRUE		32-5-2W	HWY 336	TRUE
Medium	20-7-4W	HWY 305	TRUE		5-6-2W	HWY 336	FALSE
	30-7-4W	HWY 305	TRUE		8-6-2W	HWY 336	FALSE
	24-7-5W	HWY 305	TRUE		17-6-2W	HWY 336	TRUE
	23-7-5W	HWY 305	TRUE		20-6-2W	HWY 336	FALSE
	22-7-5W	HWY 305	TRUE		19-7-1W	HWY 305	FALSE
	21-7-5W	HWY 305	TRUE		24-7-2W	HWY 305	FALSE
	20-7-5W	HWY 305	TRUE		23-7-2W	HWY 305	FALSE
	30-7-5W	HWY 305	TRUE		22-7-2W	HWY 305	FALSE
	24-7-6W	HWY 305	TRUE		21-7-2W	HWY 305	FALSE
	23-7-6W	HWY 305	TRUE		20-7-2W	HWY 305	FALSE
	22-7-6W	HWY 305	TRUE		19-7-2W	HWY 305	FALSE
	29-5-6W	HWY 338	TRUE		24-7-3W	HWY 305	FALSE
	31-5-6W	HWY 338	TRUE		23-7-3W	HWY 305	FALSE
	6-6-6W	HWY 338	FALSE		22-7-3W	HWY 305	FALSE
	7-6-6W	HWY 338	FALSE		21-7-3W	HWY 305	FALSE
	18-6-6W	HWY 338	TRUE		20-7-3W	HWY 305	FALSE
	19-6-6W	HWY 338	FALSE		19-7-3W	HWY 305	TRUE
	30-6-6W	HWY 338	FALSE		24-7-4W	HWY 305	TRUE
Fine	20-5-6W	HWY 338	FALSE		23-7-4W	HWY 305	TRUE
	17-5-6W	HWY 338	TRUE		22-7-4W	HWY 305	TRUE
	8-5-6W	HWY 338	FALSE		21-7-4W	HWY 305	TRUE