

**Assessing Local and Regional Wind-Driven Sediment
Transport Across the Canadian Prairie Pothole Region
Using a Mass-Conservation Modelling Framework**

By

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Abstract

Wind erosion is one of the erosive processes contributing to land degradation, declines in agricultural productivity, and air quality deterioration across the Canadian Prairies. Recent field-based studies provide limited evidence to support substantial sediment deposition or extensive transport distances within this region.

This study assessed wind erosion by quantifying deposited sediment and examining its spatial distribution from local (<1 km) to regional (hundreds of kilometers) scales across the Prairie Pothole Region (PPR). A Mass-Conservation Wind Erosion Model (MCWEM) grounded in fundamental physical principles was developed by integrating outputs from the Single-event Wind Erosion Evaluation Program with spatially explicit soil, land-use, land-cover, and wind data to track sediment production, redistribution, and loss. Using the MCWEM, a Worst-Case Complex Scenario (MCWEM-WCS) was implemented to evaluate sediment redistribution under sustained high-erosion conditions across three spatial scales: field (0.6 km), landscape (3 km), and regional (1,100 km).

Results indicated that the total quantity of wind-eroded sediment ultimately deposited in the final sink, Lake Winnipeg, was low across all spatial scales. Sediment transport was predominantly local, with most eroded sediment depositing near source areas rather than contributing to regional-scale transfer. Fine particles (PM_{10}) exhibited the greatest transport distances (up to 3,000 m), whereas coarse fractions (> 0.1 mm) were transported only a few meters and remained largely confined to source areas. The MCWEM-WCS predictions were evaluated against field-based observations and measurements from recent studies conducted within the Prairie Pothole Region and were found to be broadly consistent with those findings.

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This thesis represents far more than a research project; it reflects a long journey of learning, persistence, uncertainty, and growth. I would like to express my sincere gratitude to those who supported me academically, professionally, and personally along this path.

I am sincerely and warmly grateful to my advisor, Dr. David Lobb, for his steady guidance, patience, and commitment throughout this research. Even after I moved away from Winnipeg, distance never became a barrier. I will always remember our meetings with him holding a pen and paper, drawing quick and very unartistic figures, yet those simple sketches often made the most difficult concepts finally make sense.

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At the time of writing this thesis, my homeland, Iran, is going through difficult and uncertain times. I hold the sincere hope that when I look back on these pages in the future, it will be in a time of peace, freedom, and well-being for its people.

Table of Contents

Abstract.....	ii
Acknowledgments	iii
Table of Contents	v
List of Tables	vii
List of Figures.....	viii
Copyright Notice	xi
CHAPTER 1: Introduction.....	1
1.1. Overview of Larger Research Project	1
1.2. Scope of the Study.....	2
1.3. Research Questions and Hypothesis	4
CHAPTER 2: Literature Review	6
2.1. Background on Wind Erosion	6
2.1.1. Mechanisms of Wind Erosion	6
2.1.2. Factors Influencing Wind Erosion.....	8
2.2. Impacts of Wind Erosion in Agricultural Landscapes	11
2.2.1. Wind Erosion in the Canadian Prairie	12
2.2.2. Quantification of Wind Erosion in the Canadian Prairies	13
2.3. Overview of Wind Erosion Models	15
2.4. Problem Statement and Significance of the Study	18
2.5. Aims and Research Objectives.....	20
CHAPTER 3: Materials and Methods.....	21
3.1. Study Area.....	21
3.2. Analysis of Wind, Soil and Land Data.....	22
3.2.1. Wind.....	23
3.2.2. Soil	28
3.2.3. Land.....	30
3.3. Mass-Conservation Wind Erosion Model (MCWEM)	31
3.3.1. Model Principles.....	31
3.3.2. Model Structure	32
3.3.3. Model Parameters	37

3.4. Model Scenarios	39
3.4.1. Structure of Scenarios.....	39
3.4.2. Classification of Scenarios	42
3.4.3. Worst-Case Complex Scenario Model (MCWEM-WCS)	43
3.5. Fractional Deposition at Terminal Sink	46
3.6. Performance Assessment of MCWEM-WCS	46
CHAPTER 4: Results and Discussion.....	53
4.1. Model Results.....	53
4.2. Discussion	62
4.2.1. Addressing the Research Questions	62
4.2.2. Comparison of Model Predictions with Observed Field Data.....	66
4.2.3. Implications of the Study.....	81
CHAPTER 5: Conclusion	83
CHAPTER 6: Research Synergy and Contribution	85
6.1. How This Study Supports the Goals of the Research Project	85
6.2. Recommendations for Future Research	86
REFERENCES.....	91
APPENDICES	101
SUPPLEMENTARY MATERIALS	106

List of Tables

Table 2.1. Size classes of primary soil particles in Canadian Soil Information System (AAFC, 2013) and the primary transport modes defined in the SWEEP manual (USDA-ARS, 2007)	7
Table 3.1. Field unit types present in each MCWEM-WCS pathway, simulated in SWEEP to estimate soil loss from each field unit within the pathway. Codes represent combinations of land use/cover (first part: C = Cropland, G = Grassland/Forest/Shrubland, B = Bare Soil, W = Wetland/Water, U = Road/Building) and soil texture (second part: Cl = Clay, HCl=Rich clay, L = Loam, S = Sand, Si=Silt). Each type may occur multiple times within a pathway.	45
Table 4.1. Comparison of surface ^{137}Cs inventories between windward and leeward corners in Field 1 and Field 2; $\alpha = 0.10$. Reproduced from Baldwin (2015).....	70
Table 3.A. Particle-size classes, dominant transport modes, governing equations, and maximum transport distances applied in the WCS model.	101
Table 4.A. SWEEP-simulated soil loss by transport mode for each field unit type.....	104
Table 4.B. Model results for the three MCWEM-WCS pathways, summarizing total net erosion-deposition along each pathway and sediment delivery and deposition within the final sink (Lake Winnipeg).	105
Table S1. Fifteen-minute meteorological observations at the Minnedosa weather station on 31 May 2022. Source: Manitoba Agriculture Weather Program (2024).....	106
Table S2. Summary of SWEEP input parameters. Parameters under the Field and Weather categories were fixed across all simulations, while inputs under the Biomass, Soil Layers, and Soil Surface categories were varied among WCS scenarios.....	108

List of Figures

Figure 1.1. Dust storm in an agricultural landscape of the Prairie Pothole Region. Source: The Encyclopedia of Saskatchewan.....	3
Figure 1.2. Wind-blown soil deposited on snow (“snirt”) in a roadside ditch. Source: Field Crop News (2023).....	3
Figure 3.1. The Prairie Pothole Region (PPR). The study area spanning the Canadian portions of Alberta, Saskatchewan, and Manitoba. Source: Esri; TomTom; Garmin (2025).	22
Figure 3.2. Annual distribution of hourly wind speeds across the PPR (1953-2023). Data source: ECCC (2023).....	24
Figure 3.3. Probability distribution of hourly wind speeds across the PPR (1953-2023). Data Source: ECCC (2023).....	25
Figure 3.4. Hourly wind speed distribution across the PPR during May and June (1953-2023). Data Source: ECCC (2023).	26
Figure 3.5. Probability distribution of 15-minute wind speeds in Manitoba during the May-June period (2008-2024). Data Source: MAWP (2024).....	27
Figure 3.6. Wind direction frequency across the PPR based on hourly data from 1953 to 2023: all seasons (upper panel) and May-June only (lower panel). Data source: ECCC (2023).	28
Figure 3.7. Soil texture composition in the Canadian PPR. Data source: SLC (2013) and SLC-SWAT Canada Soils Dataset (Cordeiro et al., 2018).....	29
Figure 3.8. Mass-balance transfer function for estimating net soil mass change between 1-m boundary width of sequential field units under uniform conditions, with mode-specific transport limits and distance-decay weighting.	35
Figure 3.9. Structure of the Mass-Conservation Wind Erosion Model (MCWEM), including pathway discretization, source-data inputs, SWEEP-based erosion estimation, and the mass-balance redistribution component.	37
Figure 3.10. Layout of parallel transects in Fields 1 and 2. Reproduced from Baldwin (2015).	47
Figure 3.11. Flowchart of sediment transfers among fields, roadside ditches, and streams, showing erosion, transport, and deposition driven by aeolian, fluvial, and mass-movement processes. Reproduced from Brooks (2022).	50

Figure 4.1. Model result of unit-level soil mass balance across the three WCS pathways: (a) regional (0-1,100 km), (b) landscape (0-3 km), and (c) field scale (0-600 m). Positive values indicate net erosion, while negative values indicate net deposition.	55
Figure 4.2. Model result of soil mass balance for the WCS regional, landscape, and field pathways. Positive values indicate net erosion, while negative values indicate net deposition.....	56
Figure 4.3. Model result of unit-level soil mass balance within the first 600 m of Lake Winnipeg, including the riparian area, for the WCS landscape- and field-scale pathways.	56
Figure 4.4. Model result of remaining mass in transport along the three WCS pathways: (a) regional-scale, (b) landscape-scale, and field-scale.	58
Figure 4.5. Model result of remaining mass in transport at the end of the WCS landscape- and field-scale pathways, extending 3,000 m into Lake Winnipeg.	59
Figure 4.6. Model result of deposition by particle-size fractions (sand, silt, clay, fine clay) at the terminal sink (Lake Winnipeg) along the WCS landscape-scale pathway: (a) first 30 m, including the 0-5 m riparian area and shoreline; (b) lake interior from 30 to 3,000 m (note different y-axis scale).	61
Figure 4.7. Comparison of ^{137}Cs inventories between cultivated in-field areas and regional reference sites for Field 1 and Field 2. No statistically significant differences were found, supporting the model's assumption of minimal net sediment export by wind erosion. Adapted from Baldwin (2015).	68
Figure 4.8. Comparison of ^{137}Cs inventories between in-field corner areas and their adjacent grassed corner areas in Field 1 and Field 2. P-values indicate no statistically significant difference. Adapted from Baldwin (2015).	69
Figure 4.9. Surface ^{137}Cs activity and elevation profiles from cultivated in-field and undisturbed reference transects in Field 1 and Field 2. Reproduced from Baldwin (2015).	71
Figure 4.10. Annualized sediment transfers to roadside ditches in fine-textured catchments of sub-watersheds 05OF024 and 05OG008, showing that wind-driven features (snirt and blow dirt) contribute a small fraction compared to other sediment sources. Blow dirt values were adjusted for a 30-year return interval. Adapted from Brooks (2022). ...	73
Figure 4.11. Annual wind-driven (snirt + blow-dirt) sediment transfers by soil texture in sub-watershed 05OF024 Adapted from Brooks (2022). Delivery remains low across textures, indicating proximal trapping within roadside ditches.	75

Figure 4.12. Mean ^{137}Cs inventory along the toposequence for wetlands surrounded by croplands and native prairie. Adapted from Zarrinabadi (2023). 77

Figure 4.13. Sediment budgets based on ^{137}Cs transect data for the studied wetland catchments in Alberta and Manitoba. Gross erosion is the total mass exported per unit cultivated area; net erosion equals gross erosion minus gross deposition within cultivated fields. Values represent cumulative redistribution by wind, water, and tillage processes. Adapted from Zarrinabadi (2023). 78

Figure 4.14. Mean soil loss or deposition ($\text{kg m}^{-2} \text{yr}^{-1}$) along the toposequence for wetland catchments in Manitoba and Alberta. (a) uses ^{137}Cs reference inventories from individual local stable sites; (b) uses a catchment-level ^{137}Cs reference inventory. Adapted from Zarrinabadi (2023). 79

Figure 3.A. Generalized mass-balance transfer function illustrating soil redistribution along a hypothetical 6 km pathway under uniform conditions. The upwind section (3,000 m) acts as a source zone (red), while the downwind section (3,000 m) functions as a sink where sediment progressively deposits (green). Arrows indicate the direction and relative magnitude of soil fluxes. The function maintains mass conservation by ensuring total erosion equals total deposition. 102

Figure 3.B. Mass-balance response to extending the source length to 6000 m under fixed transport limits. Beyond 3,000 m within the source, the mass balance $M(x)$ remains zero: soil loss is fully offset by sediment arriving from upwind, indicating a dynamic equilibrium. Consequently, lengthening the source beyond this distance does not change the overall mass balance; it only extends a downwind no-exchange segment. Deposition in the sink still declines with distance and stops before the end of the sink zone. 102

Figure 3.C. Mass-balance response under heterogeneous surface conditions (mosaic land use and soil texture). Along a 6-km pathway, spatial heterogeneity produces discontinuous transport: loss and deposition occur in alternating pulses, with several localized sink segments appearing inside the land source. Compared with uniform-condition runs, heterogeneity shortens effective transport distances and increases near-source deposition. 103

Figure 3.D. Redistribution of particle-size fractions under uniform vs. heterogeneous surfaces. Heterogeneity increases near-source deposition and shortens effective transport distances, though the overall pattern remains in sink land. 104

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CHAPTER 1: Introduction

1.1. Overview of Larger Research Project

This study is part of a broader national research initiative titled “Wetlands as Nature-Based Climate-Change Solutions: Quantifying Carbon-Capture Potential While Building a Stronger Green Economy.” Funded through Environment and Climate Change Canada's Climate Action and Awareness Fund, this interdisciplinary project brings together academic institutions, government agencies, and conservation organizations to address key uncertainties surrounding the role of freshwater wetlands in mitigating climate change. It aims to provide the scientific foundation and policy tools necessary to include agricultural wetlands in Canada’s greenhouse gas (GHG) accounting and carbon offset frameworks.

The project was conceived in response to Canada's commitment under the Paris Agreement to reduce GHG emissions by 30% below 2005 levels by 2030 and to reach net-zero emissions by 2050. Although Canada’s peatlands have been widely studied in this context, freshwater mineral wetlands, particularly those embedded in agricultural landscapes, have received less attention despite their considerable potential for organic carbon (OC) sequestration. Historically, these wetlands have been subject to extensive drainage for agriculture, yet emerging research suggests that protecting and restoring them could significantly enhance carbon capture and reduce national emissions.

To address this knowledge gap, the project seeks to develop authoritative estimates of wetland OC accumulation, GHG fluxes, and carbon transport, while also evaluating the multiple ecosystem

services these wetlands provide. It adopts a multi-scale approach encompassing wetland, watershed, and national levels, with monitoring activities conducted at over 100 sites across Canada. The research also integrates remote sensing, hydrological modeling, and ecosystem service valuation to quantify the contributions of wetlands as nature-based climate solutions (NbS). Ultimately, this project aimed to support the design of carbon-smart agricultural landscapes, inform wetland conservation and restoration efforts, and enhance Canada's capacity to meet its climate targets.

1.2. Scope of the Study

This study focused on a specific component of the broader NbS research initiative by assessing the potential impact of wind erosion on carbon storage in wetlands across the Prairie Pothole Region (PPR) and in Lake Winnipeg. While the larger project examines multiple processes influencing carbon fluxes in freshwater wetlands, this study specifically evaluated wind erosion as a mechanism for redistributing organic carbon associated with wind-transported sediment from cultivated uplands to downstream aquatic environments.

Although other erosion processes such as water and tillage erosions have been more extensively studied in the context of wetland sedimentation, wind erosion can also transfer carbon-rich soil particles. These particles may be deposited locally within a few meters of their source or transported over longer distances ranging from hundreds of meters to several kilometers (Bagnold, 1941; Chepil et al., 1963; Li et al., 2015; Singh, 2023), contributing to sediment accumulation and the formation of carbon-rich depositional zones within aquatic environments (Figures 1.1-1.2). Despite this potential, the role of wind erosion in shaping wetland carbon dynamics remains largely unquantified, particularly in large-scale agroecosystems such as the Canadian Prairies.



Figure 1.1. Dust storm in an agricultural landscape of the Prairie Pothole Region. Source: The Encyclopedia of Saskatchewan.



Figure 1.2. Wind-blown soil deposited on snow (“snirt”) in a roadside ditch. Source: Field Crop News (2023).

This research sought to address this gap by modeling wind-driven sediment transport across the Canadian PPR, with Lake Winnipeg and its riparian areas treated as the terminal depositional area for sediment accumulation. Using a mass-conservation framework, this study simulated transport processes to evaluate the magnitude and spatial redistribution of wind-eroded sediment. Particular

attention was given to depositional areas such as field edges, riparian margins and wetland interiors, where sediment-bound carbon is likely to accumulate. Key variations in soil texture, land use and land cover, and field length were incorporated to assess their influence on wind transport capacity and redistribution patterns. Simulations were conducted under worst-case conditions by assigning these parameters their maximum influence, thereby capturing the upper bound of plausible wind-driven sediment and carbon redistribution. By doing so, the research provides insight into how wind erosion contributes to wetland carbon stocks. This insight is critical for improving carbon accounting methods in agricultural landscapes and complements the broader project's efforts to quantify greenhouse gas fluxes and carbon sequestration potential across diverse wetland types.

1.3. Research Questions and Hypothesis

This study aimed to evaluate the role of wind erosion in transporting sediment and associated organic carbon across agricultural landscapes in the PPR, with particular attention to delivery to prairie wetlands and Lake Winnipeg. Wind erosion has the potential to detach and mobilize carbon-rich soil particles from cultivated uplands, especially under dry surface conditions and during high-wind events common to the Canadian Prairies. These eroded particles may be deposited in lowland landscape elements, including riparian zones, shallow wetland margins, central wetland areas, and nearshore areas of Lake Winnipeg, where they may accumulate and potentially influence carbon storage within these aquatic systems. The PPR landscape is characterized by extensive agricultural land use and the high density of wetlands. Understanding the spatial extent and intensity of wind-eroded sediment movement is essential for accurately estimating carbon accumulation in wetlands and for improving landscape-scale carbon accounting in agroecosystems.

To explore these dynamics, the study was guided by the following research questions:

1. To what extent does wind erosion contribute to sediment transport at local and regional spatial scales across the PPR?
2. How much wind-eroded sediment is transported and ultimately deposited within prairie wetlands and Lake Winnipeg?

Based on these questions, the landscape characteristics of the region, and the widespread assumption that wind erosion plays a key role in sediment redistribution across prairie landscapes, the study evaluated the following hypotheses:

1. Wind erosion is a significant driver of sediment transport at both local and regional scales.
2. A substantial proportion of wind-transported sediment is deposited within central areas of wetlands and Lake Winnipeg.

By addressing these questions through a mass-conservation modeling framework, the study provides new insights into the spatial patterns of wind-driven sediment redistribution and its implications for wetland carbon sequestration.

CHAPTER 2: Literature Review

2.1. Background on Wind Erosion

2.1.1. Mechanisms of Wind Erosion

Wind erosion, as a dynamic physical process, is the interaction of the wind with the soil surface including three phases of detachment, transportation, and eventual deposition of soil particles. This detachment occurs as the wind exerts drag and lift forces, overcoming the gravitational and cohesive forces that hold particles to the soil surface (Toy et al., 2002; Zobeck & Pelt, 2015; Singh, 2023). The initiation of soil particle movement requires a critical wind velocity, known as the threshold wind velocity, which varies with soil properties. Individual soil aggregates become aerodynamically unstable, entrained into the wind stream, and subsequently transported downwind after the threshold velocity required to initiate soil movement is exceeded. The detachment of these airborne particles depends on the wind velocity, as well as the geometric characteristics and density of the soil particles (Fryrear, 1990; Lal, 1998; Singh, 2023).

Soil particles with diameters less than 2 mm are subject to three principal modes of transport during wind erosion. The dislodged particles or aggregates ranging from 0.84 to 2 mm (USDA-ARS, 2007), are too large to become airborne. Consequently, they move across the soil surface through rolling or sliding motions. This mechanism, known as surface creep, constitutes approximately 5-25% of the total soil mass transported. Individual particles or aggregates, measuring 0.1 to 0.84 mm (USDA-ARS, 2007), move by jumping and bouncing over the ground surface, rarely exceeding heights of a few meters. These particles can be lifted off the soil surface by the wind's

force or be expelled as other bouncing particles dislodge them upon impact. Saltation process can carry approximately 50% -75% of total transported soil mass. If the aggregates are very small, less than 0.1 mm (USDA-ARS, 2007), they may be transported over considerable distances in suspension mode. Suspended particles mainly come from larger aggregates being abraded by saltating sediment or striking the soil surface, similar to a sandblasting process. A portion of suspension particles with a diameter of 0.01 mm (10 μm) or less, commonly referred to as PM_{10} , can remain airborne, travel over distances of several kilometers, and be inhaled into the human respiratory tract. (Chepil et al., 1963 ; Li et al., 2015 ; Singh, 2023). Table 2.1 presents the size classes of primary soil particles and their corresponding dimensions, as defined by the Canadian Soil Information System (AAFC, 2013), along with the dominant transport modes as specified in the USDA-ARS Single-Event Wind Erosion Evaluation Program (SWEEP) manual (2007).

Table 2.1. Size classes of primary soil particles in Canadian Soil Information System (AAFC, 2013) and the primary transport modes defined in the SWEEP manual (USDA-ARS, 2007)

Size Range (mm)	Name of Separate (CanSIS)	Dominant Mode of Wind Transport
≥ 2	gravel	no movement
2.0-1.0	very coarse sand	creep
1.0-0.84	coarse sand	creep
0.84-0.50	coarse sand	saltation
0.50-0.25	medium sand	saltation
0.25-0.1	fine sand	saltation
0.1-0.05	very fine sand	suspension
0.05-0.01	silt	suspension
0.01-0.002	silt	suspension PM_{10}
0.002-0.0002	clay	suspension PM_{10}
≤ 0.0002	fine clay	suspension PM_{10}

2.1.2. Factors Influencing Wind Erosion

Wind erosion is widely understood to result from the combined influence of both natural and anthropogenic factors (Shao, 2008), which can be categorized into four main groups: climate and weather conditions, soil properties, land surface characteristics, and land use practices. Wind erosion and dust emissions are closely linked to climatic drivers, especially near-surface wind regimes (Tegen et al., 2004; Shao, 2001). Precipitation, temperature, humidity, wind velocity, and wind direction are the parameters associated with weather and climate conditions that influence wind erosion. Seasonal variations significantly influence the structural condition and wind erodibility of soils, largely due to biological activity and alternating cycles of wetting, drying, freezing, and thawing. These fluctuations cause measurable changes in soil stability and susceptibility to erosion throughout the year (Chepil, 1958). Singh (2023) noted that areas with low precipitation and high temperatures are more susceptible to wind erosion due to the high susceptibility of dry soil particles to wind blowing.

The erosive potential of wind is largely determined by wind speed and the moisture content of the soil (Chepil et al., 1962) and is typically represented through an erosivity index. Soil moisture levels are controlled by the balance between potential evaporation and precipitation over defined timeframes. This influence of moisture can be integrated into wind erosion models by either elevating the threshold wind velocity required for sediment detachment or by applying a correction factor that reduces the estimated erosivity (Bagnold, 1971).

Soil erodibility refers to the inherent vulnerability of soil particles to detachment and subsequent transport by erosive forces, including those induced by wind (Woodburn & Kozachyn, 1956) and it is predominantly governed by the intrinsic physicochemical properties of the soil and can be estimated through empirical relationships (Zhang et al., 2004). Organic content, surface roughness,

soil structure, and texture are critical parameters for quantifying soil erodibility, which in turn influences the magnitude of wind erosion (Tejada & Gonzalez, 2006). Soil erodibility varies substantially across different soil types due to its strong dependence on the physicochemical characteristics of the surface layer (Hevia et al., 2007). The susceptibility of soils to crust formation and wind erosion is largely determined by physical attributes such as aggregate stability, bulk density, and infiltration capacity (Duiker et al., 2001).

Surface-layer characteristics, including organic matter, calcium carbonate content, and particle size distribution, also play a critical role in enhancing or reducing soil resistance to wind erosion processes (Tejada & Gonzalez, 2006). Organic matter enhances aggregate stability and cohesion, thereby limiting the availability of loose, erodible particles; conversely, its depletion weakens soil structure and increases erodibility (Tejada & González, 2006; Wu et al., 2024). Calcium carbonate enhances soil resistance to wind erosion by cementing particles, stabilizing aggregates, and promoting crust formation (Dagliya, 2022). Particle size distribution governs the detachment and transport potential of soil: finer particles are more easily lifted and carried by wind, while coarser particles contribute to surface armoring. A well-balanced distribution of particle sizes enhances aggregation and reduces the availability of easily entrained sediment. As a result, fine- and medium-textured soils, such as silt loam and clay loam, tend to exhibit greater resistance to wind erosion than coarse-textured soils like sandy loam, owing to their higher cohesion and structural stability (Li et al., 2010; Chepil, 1951a; Singh, 2023).

The topography and landform as well as vegetation coverage are land surface characteristics which also considerably influence wind erosion. Landscapes with gentle slopes are generally more susceptible to wind erosion than undulating landscapes due to their lower resistance to wind. Numerous studies have demonstrated an inverse relationship between vegetation cover and soil

wind erosion rates (Zhang et al., 2009). While above-ground plant biomass mitigates wind energy and thereby reduces erosive forces, root systems enhance soil cohesion and structural stability, minimizing surface disturbance (Li et al., 2002; Hoffmann et al., 2008). Studies, including field and wind tunnel experiments, have investigated the impact of vegetation cover on wind-induced erosion. In general, 30% ground cover is required to prevent wind erosion (Singh, 2023; Yan et al., 2011). Breshears et al. (2008) examined sediment transport rates across various land covers such as forests and shrublands. They concluded that sparse shrublands have a more pronounced effect on wind profiles by directing airflow and enhancing turbulence. Forests, dense shrublands, and grasslands provide more uniform protection to the soil surface and typically generate a wind profile characterized by "skimming flow" (Pelletier, 2006).

The magnitude and spatial variability of wind erosion are also governed by land-use practices (Ginoux et al., 2012; Tegen et al., 2004). The interplay between land use, meteorological factors, and aeolian processes is often complex, particularly in regions highly susceptible to wind erosion and subject to frequent land-use transitions. Alterations in climate and land use can significantly influence wind erosion dynamics by increasing landscape susceptibility to disturbance, thereby heightening vulnerability to extreme weather events or periods of climatic stress (Boardman & Poesen, 2007). Human activities can modify surface characteristics, thereby influencing erosion dynamics. Various land management practices, such as terracing, conservation tillage, and slope abandonment, can either mitigate or exacerbate wind erosion by altering soil physicochemical properties, surface topography, and vegetation cover (Belnap et al., 2007). Although, the basic mechanism of wind erosion remains consistent across all types of land, the factors influencing sediment transport rates, such as soil characteristics, surface roughness, and management practices, can vary among these environments (Webb & Strong, 2011). The presence of non-erodible

materials, such as coarse fragments larger than 2mm, large stable aggregates, and vegetation, varies across different land uses. These materials refer to surface elements that resist wind movement and remain stable under wind forces. They act as physical barriers, shielding the soil surface from the direct impact of wind flow and trapping particles moving by creep and saltation. The effectiveness of this protection depends on the quantity of non-erodible material present in different land uses. Covering the soil with 20% surface cover of non-erodible material (e.g., crop residue, clods, or stones) resulted in a 57% reduction in soil loss, while covering it with 50% non-erodible material reduced soil loss by 95% (Fryrear, 1985).

2.2. Impacts of Wind Erosion in Agricultural Landscapes

Wind erosion is a natural hazard that contributes to land degradation and desertification, affecting approximately 28% of the world's land surface. The sustained loss of soil resulting from wind erosion is expected to become a critical environmental concern (Buschiazzo & Zobeck, 2008) particularly in regions characterized by arid and semi-arid climates where precipitation is inadequate and vegetation cover is sparse (Singh, 2023). Wind erosion is commonly viewed as a process that removes soil particles, but it also involves soil-particle sorting, long-distance nutrient transport, and the fertilization of aquatic and terrestrial ecosystems far from the source. These factors contribute to soil heterogeneity and have local to global impacts on soil properties and the carbon cycle. In agricultural landscapes, this erosion diminishes crop productivity by reducing soil nutrient levels and water retention capacity, often necessitating increased fertilizer application (Oldeman, 1991; Visser & Sterk, 2007). Crop damage occurs when seedlings are exposed to sand blasting or abrasion, and when they are buried under sand during storm events. The movement of particles during wind erosion can cause damage to plant tissue, making them more susceptible to sunburn and pathogenic infection. Young plants that are buried under sand during storms suffer

from subsequent problems, including the physical stress of sand accumulation, reduced sunlight penetration, and increased soil temperatures during daylight hours (Michels et al., 1995; Bennell et al., 2007). Beyond reducing soil fertility and agricultural productivity, wind erosion also contributes to atmospheric pollution, as fine soil particles detached from the surface are entrained and transported through the air (Sharratt et al., 2015). Airborne dust contains aerosols that can affect the atmospheric radiation balance and global climate. In severe cases, it can also pose a threat to human health, causing irritation of the eyes and skin and increasing the risk of various diseases and cancers (Shao, 2008).

2.2.1. Wind Erosion in the Canadian Prairie

Wind erosion is an environmental problem in many regions of Canada; however, it is considered to be a major soil issue in the Canadian Prairie Region, which extends from central Alberta to southwestern Manitoba (Rennie, 1985; Lobb et al., 2017). The Canadian Prairies and the larger Northern Great Plains experience wind erosion and blowing dust, with land use and land cover changes resulting from human activities acting as important contributing factors. Studies have been conducted on the role of human activities on blowing dust generation. Neff et al. (2008) reported a significant 500% increase in sediment deposition in the collected core of Colorado Lake corresponding to settlement activities in the late 1800s. Wheaton (1992) states that wind erosion is more prevalent in the south-central region of the Prairie provinces, while it is relatively low in the northern agricultural lands. This is due to the spatial pattern of dust storms. In the more northern areas, where wind speeds are lower, temperatures are lower, and precipitation is higher, the risk of erosion is comparatively lower. Additionally, the reduced utilization of summer fallow and increased amount of residue in the northern regions contribute to a lower risk of erosion (Huffman et al., 2000).

Furthermore, climate change plays a key role in the susceptibility of soil to erosion in the Prairies. Recent weather records indicate a trend towards drier conditions globally, which contributes to a decline in vegetation cover and an increase in severe erosion events across the Prairies (Leys et al., 2018; World Meteorological Organisation, 2019). The combination of sparse ground cover, arid climates, and gusty spring winds may increase the frequency of dust storms in the PPR (Field et al., 2011). Huffman et al. (2000) modeled wind erosion across the Prairies and reported that, over the ten-year period from 1981 to 1991, the overall risk of wind erosion declined by approximately 7%, which is contrary to the expected trend. According to the authors, about one-third of this reduction can be attributed to changes in cropping systems, particularly a reduction in summer fallow, while the remaining decrease is a result of the wider implementation of conservation tillage practices (Huffman, 2000).

2.2.2. Quantification of Wind Erosion in the Canadian Prairies

Chepil and Milne (1941) were likely among the earliest researchers to measure wind-induced sediment directly within Canadian agricultural fields. They employed a slotted catcher to quantify soil movement occurring within 5 cm above the surface at different distances from the field's windward edge, though limited information was recorded about wind or soil conditions. Between 1938 and 1944, Chepil extended this research across several major soil types using Bagnold-style sand catchers (Chepil, 1946), aiming to relate surface saltation and creep flux to fetch distance (Zobeck et al., 2003). Studies to quantify wind-eroded sediment intensified in the 1990s. To support the transition from the 1960's Wind Erosion Equation, to process-based models such as the Wind Erosion Prediction System (WEPS), wind erosion monitoring sites were established in the Canadian prairies (Hagen, 1991). In 1990 and 1993, Larney et al. (1998) conducted a wind

erosion study in two experimental field sites near Lethbridge, Alberta, to quantify wind erosion and its effects on soil nutrients. Using circular erodible plots surrounded by protected stubble and equipped with Big Spring Number Eight (BSNE) windblown sediment samplers (Fryrear, 1986), they monitored 29 natural erosion events. Windblown sediments collected at 25 cm height were compared with surface soils, revealing nutrient enrichment in the transported material. The study linked this redistribution to reduced wheat yields on eroded areas and increased fertility in depositional zones, highlighting the agricultural implications of wind erosion on the Canadian prairies.

At broader scales, Agriculture and Agri-Food Canada developed the Soil Erosion Risk Indicator (SoilERI), within which the wind component (WindERI) estimates wind-erosion risk using climate, soil texture, topography, and farming-practice inputs summarized at the Soil Landscapes of Canada (SLC) polygon scale. Although the WindERI time series is back-cast to 1981, AAFC first reported the wind-erosion risk indicator nationally in 2000, following its development under the Agri-Environmental Indicator program initiated in 1993. Reported in five risk classes and compiled from 1981 to 2016, WindERI results indicate substantial improvement across the Prairies, most cropland shifting into the very-low wind-risk class by 2016, changes attributed by AAFC to conservation tillage, increased residue retention, and the decline of summer fallow (Agriculture and Agri-Food Canada, 2024).

Baldwin (2015) conducted a field-based assessment of wind erosion in two cultivated potato fields located in southern Alberta using a radionuclide tracer method (^{137}Cs) and SWEEP model simulations. The study showed that wind erosion led to spatial redistribution of soil mostly within fields, with erosion and deposition patterns varying across the landscape. Brooks (2022) conducted a sediment budget estimate in two agricultural sub-watersheds in southern Manitoba to quantify

sediment redistribution across multiple landforms influenced by wind, water, and tillage processes. The results indicated that wind erosion contributed a relatively small portion of the total sediment budget, with much of the eroded sediment deposited in nearby landscape features such as roadside ditches. A recent study by Ehsan Zarrinabadi (2023) provided a detailed assessment of wind erosion using field measurements, including wind-eroded sediment samplers and ^{137}Cs tracer methods, to quantify soil loss and deposition rates within agricultural lands and wetland landscapes in the PPR. The study found that soil loss and sedimentation were very small in agricultural lands, and the ^{137}Cs results indicated minimal enrichment in central wetland areas, with deposition concentrated in lower slopes and outer riparian zones.

2.3. Overview of Wind Erosion Models

Wind-induced erosion can be quantified through field measurement, laboratory experiments, and modeling (Heuvelink et al., 2006). Soil erosion models mathematically illustrate erosion mechanisms, including detachment, transport, and deposition across the landscape (Nearing et al., 2017). These models consider the influence of different factors simultaneously to provide detailed information on particle movement at specific temporal and spatial scales (Bhuyan et al., 2002).

The goal of wind erosion modeling is to estimate soil erosion rates at various scales, from catchment to regional and national levels, and to assess the on-site and off-site effects of wind erosion. Wind erosion models assist land managers in monitoring and predicting the impact of contributing factors on wind erosion, as well as in implementing conservation policies (Bhuyan et al., 2002; Blanco-Canqui et al., 2008).

The suitability of a wind erosion model depends on the application's objectives. Different models have varying approaches to representing erosion processes, complexity, scale of assessment, input data requirements, and outputs (Merritt et al., 2003). Furthermore, there are other factors that can

affect the selection of a model for a particular purpose including data requirements, model accuracy and validity, model components, and required hardware. Each erosion model comes with its own set of constraints, leading to a certain level of uncertainty regarding the accuracy of predicted outcomes. Wind erosion modelling research in North America commenced in the 1930s with the introduction of the Wind Erosion Equation (WEQ) model (Woodruff et al., 1965). Despite the challenges faced by early wind erosion models, including high data requirements, unrealistic assumptions, and limited validation across diverse geographical regions, advancements have been made to refine these approaches. These enhancements seek to provide more reliable estimates of wind-induced soil erosion and offer practical recommendations for erosion control measures (Jarrah et al., 2020).

Chepil's (1946) initial research on wind mechanics and dynamics forms the basis of many of today's wind erosion models. In 1959, Chepil proposed an equation that expands the theoretical framework of wind erosion models and emphasizes key factors contributing to the erosion process, as follow:

$$E = IRKFBWD$$

where E = soil loss due to wind erosion; I = soil cloddiness factor; R = vegetative material factor; K = ridge roughness equivalent factor; F = soil abrasability factor; B = wind barrier factor; W = field width factor; and D = wind direction factor.

In general, models can be classified into three main groups based on the physical processes it simulates, algorithms employed to describe these processes and the dependencies of the model on the data as follow:

- Empirical or statistical/metric
- Conceptual

- Physical (Merritt et al., 2003)

WEQ, WEPS and SWEEP are the most applied wind erosion models (Borrelli et al., 2021). WEQ is the first empirical wind erosion model developed from wind tunnel and field measurements for assessing annual soil loss (Woodruff & Siddoway, 1965). It considers several factors, including soil erodibility, ridge roughness factor, climate factor, unprotected field length, vegetation cover index, and the interaction between these factors, to predict average wind erosion in mass per unit area per year (Jarrah et al., 2020).

Although the WEQ provides valuable estimations of wind erosion, it is constrained by several limitations. For instance, this model only considers average wind and precipitation conditions, thus failing to account for variations from this mean. Furthermore, it lacks the capability to predict soil erosion rates for individual storm events or on a daily temporal scale. Moreover, its performance is not optimal in regions characterized by extreme rainfall patterns or fields of exceptionally large or narrow dimensions. Consequently, continuous revisions and refinements have been undertaken, leading to the development of the Revised Wind Erosion Equation (RWEQ). RWEQ predicts soil loss based on wind capacity to transport soil, regardless of soil type. For a given period in a single event, it estimates the horizontal mass transported by wind within the first 2 meters (Fryrear et al., 1998; Jarrah et al., 2020).

The Wind Erosion Prediction System (WEPS) model was developed in the 1980's. It was designed to address the limitations of the WEQ and RWEQ models (Hagen, 1991; Tatarko et al., 2016). The model was formulated through field and laboratory experiments with a process-based approach, incorporating correlations between surface conditions and erosion. It's capable of continuous simulation at a daily time-step, replicating overall wind erosion events, encompassing saltation, suspension, and surface creep processes, and estimating air pollution through dust emissions,

specifically particles smaller than 10 μm in diameter (PM_{10}). This model aids in developing soil conservation strategies to manage wind erosion in agricultural landscapes (Singh, 2023).

SWEEP is a standalone, process-based version of the WEPS erosion model. Although it operates independently of WEPS, it uses the same computational framework and is designed to predict soil loss during a single wind event (within 24 hours) based on user-defined surface and wind conditions (Tatarko et al., 2016). Model inputs are grouped into five main categories (Table S2): field geometry (field dimensions, orientation, and barriers), vegetative cover (residue, biomass height, and leaf and stem area index), soil characteristics (particle and aggregate size distribution, rock content, and aggregate stability), soil surface conditions (crust and loose material cover, surface roughness, and soil moisture), and wind parameters (speed and direction), with average wind speed intervals ranging from 5 to 60 minutes (USDA-ARS, 2007).

Despite the widespread use of these models, a significant limitation arises when attempting to apply them at larger scales, such as the regional level. These models have largely been developed for field-scale applications; therefore, their extrapolation from plot-level assessments to regional-scale analyses presents considerable practical challenges. These challenges include the requirement for detailed, spatially explicit input data, which are often costly and time-consuming to obtain, as well as the difficulty of calibrating and validating model performance across large and heterogeneous geographic areas (Zarrinabadi, 2023).

2.4. Problem Statement and Significance of the Study

Across the PPR, wind erosion has long been identified as an important process influencing soil loss, land degradation, and nutrient redistribution. Nevertheless, despite extensive research efforts and the development of multiple wind erosion models, significant knowledge gaps persist. There is limited field-based evidence on the magnitude of wind-eroded sediment accumulating in

depressional areas such as prairie wetlands. Recent field studies in the PPR, including Baldwin (2015), Brooks (2022), and Zarrinabadi (2023), report little or no evidence of substantial wind-eroded sediment accumulation within depressions such as roadside ditches and wetland areas. These observations contrast with model-based predictions, including regional wind erosion risk predicted by WindERI. This discrepancy between modelled predictions and observed data raises important questions regarding the actual impact of wind-driven sediment redistribution in the region.

In addition, most existing studies have been conducted at the plot or field scale, or under controlled conditions such as wind tunnels, providing limited insight into how wind erosion operates across larger and more complex landscapes. Consequently, uncertainty remains regarding the extent to which sediment can be transported over longer distances, whether wind erosion contributes meaningfully to sediment accumulation at broader regional scales, and whether sediment deposited in depressional areas is derived from regional-scale transport or primarily from nearby, localized sources. This distinction is important because it helps determine whether sediment accumulation in these areas is governed by long-distance transport processes or by local redistribution within the surrounding landscape.

Resolving these gaps is critical for understanding the role of wind erosion in shaping wetland sediment budgets and its implications for carbon sequestration. An evidence-based understanding of wind erosion, particularly one that considers multiple spatial scales and is supported by field observations, is needed to improve the management of prairie landscapes and the protection of vulnerable wetland areas.

2.5. Aims and Research Objectives

To address the existing knowledge gaps discussed in the previous section, this study aimed to improve our understanding of wind-driven sediment transport and its implications for sediment deposition and carbon input in prairie wetlands. In particular, it focused on evaluating the magnitude and spatial extent of wind erosion, and the potential contribution of sediment sources at different scales, from local to regional, to aquatic system infilling across the PPR.

The primary objectives of this study were to:

- Estimate the magnitude of wind-driven sediment transport at local and regional scales across the Canadian Prairie Pothole Region using a transfer-function based model.
- Analyze the spatial distribution of wind-driven sediment deposition within aquatic depressional features (i.e., Lake Winnipeg).

To achieve these aims, the study applied a mass-conservation modeling framework under worst-case scenarios, integrating key erosion-related variables including soil texture, land use and land cover, and field length. Simulations were conducted across three spatial scales: field scale (600 m), landscape scale (3 km), and regional scale (1,100 km), to assess scale-dependent differences in wind-induced sediment transport. The model outputs were compared with field observations and measurements reported in recent wind erosion studies to evaluate model validity and refine assumptions about sediment-transport magnitude and long-distance sediment delivery.

By addressing these objectives, this research sought to clarify the magnitude of wind-driven sediment deposition and to determine whether sediment deposited within Lake Winnipeg originates from regional-scale wind transport or primarily from nearby sources. It also aimed to contribute new insights into the role of wind erosion in sediment and carbon dynamics within prairie agricultural landscapes.

CHAPTER 3: Materials and Methods

3.1. Study Area

The Prairie Pothole Region (Figure 3.1) spans approximately 770,000 square kilometers across five U.S. states (Montana, North Dakota, South Dakota, Minnesota, Iowa) and three Canadian Prairie provinces: Alberta, Saskatchewan, and Manitoba. It is named after the vast number of mineral wetlands known as "prairie potholes". These shallow depressions are the result of the melting of underground ice masses that were deposited as glaciers receded at the end of the last ice age (Doherty et al., 2018). The vast extent of the PPR and its high density of wetlands make it one of the most productive, diverse, and distinctive wetland-grassland ecosystems in the world (Baldassarre & Bolden, 2006; Doherty et al., 2018). Elevations vary from 200 meters above sea level in the eastern region to 1200 meters in the western part of Canadian PPR (ESTR Secretariat, 2014). The PPR has semi-arid and highly dynamic climate (Zhang et al., 2021; Renton et al., 2015). The region features a pronounced west-east precipitation gradient, increasing from ~300 mm yr⁻¹ in the west to ~900 mm yr⁻¹ in the east, alongside a north-south temperature gradient with colder conditions in the north and warmer conditions in the south. In the western Canadian Prairies, the aridity implied by lower annual precipitation is partly tempered by cooler temperatures and a shorter growing season (Millett et al., 2009).

The Canadian Prairies have experienced vast modifications for agricultural purpose since European settlement in the late 19th century (Floate & Shorthouse, 2010). Soil erosion by tillage is

the main cause of erosion in agricultural landscapes across the PPR and as a nonselective process moves a large quantity of soil within cultivated landscapes (Lobb, 2011).



Figure 3.1. The Prairie Pothole Region (PPR), the study area spanning the Canadian portions of Alberta, Saskatchewan, and Manitoba. Source: Esri; TomTom; Garmin (2025).

3.2. Analysis of Wind, Soil and Land Data

Wind, soil and land data are central to understanding the spatial and temporal variability of wind erosion across the PPR. These three factors interact to influence both the susceptibility of the landscape to erosion and the magnitude of sediment transport. Wind data provide insight into the energy available to initiate erosion, while soil characteristics, with focus on texture, govern the ease with which particles can be detached and entrained. Land use and land cover conditions further shape the extent to which soil surfaces are exposed or protected during wind erosion events. This section analyzes each of these variables using long-term meteorological records, national soil

datasets, and land use/cover inventories to inform the wind erosion assessment and support model parameterization.

3.2.1. Wind

The historical hourly wind speed and direction data used in this study were obtained from 97 Environment and Climate Change Canada (ECCC) weather stations distributed across the PPR. These stations provide broad spatial coverage and long-term records that capture variability in the region's wind regimes. The analysis examined the distribution and probability of hourly wind speed and direction over a 70-year period, with particular attention given to the months of May and June. These months fall within the late spring period, when agricultural landscapes are generally more susceptible to wind erosion due to greater bare soil exposure, dry surface conditions, and limited vegetation cover. Together, these factors increase the potential for soil detachment and transport (Chepil, 1958; Fryrear, 1985). As shown in Figure 3.2, the annual distribution of hourly wind speeds in the PPR has remained relatively stable over the 70-year period, with median values typically ranging between 5 and 7 m s⁻¹ and consistent interquartile ranges across decades. This indicates that the central tendency of wind conditions has not changed significantly over time. However, the frequency and magnitude of extreme wind events, represented by outliers exceeding 20 m s⁻¹ (72 km hr⁻¹), appear more frequent and intense after 1995, with several years showing values above 30 m s⁻¹ (108 km hr⁻¹). While this may suggest an increase in wind speed extremes, it is also possible that the rise in recorded values reflects improvements in instrumentation and data resolution, rather than an actual increase in wind intensity.

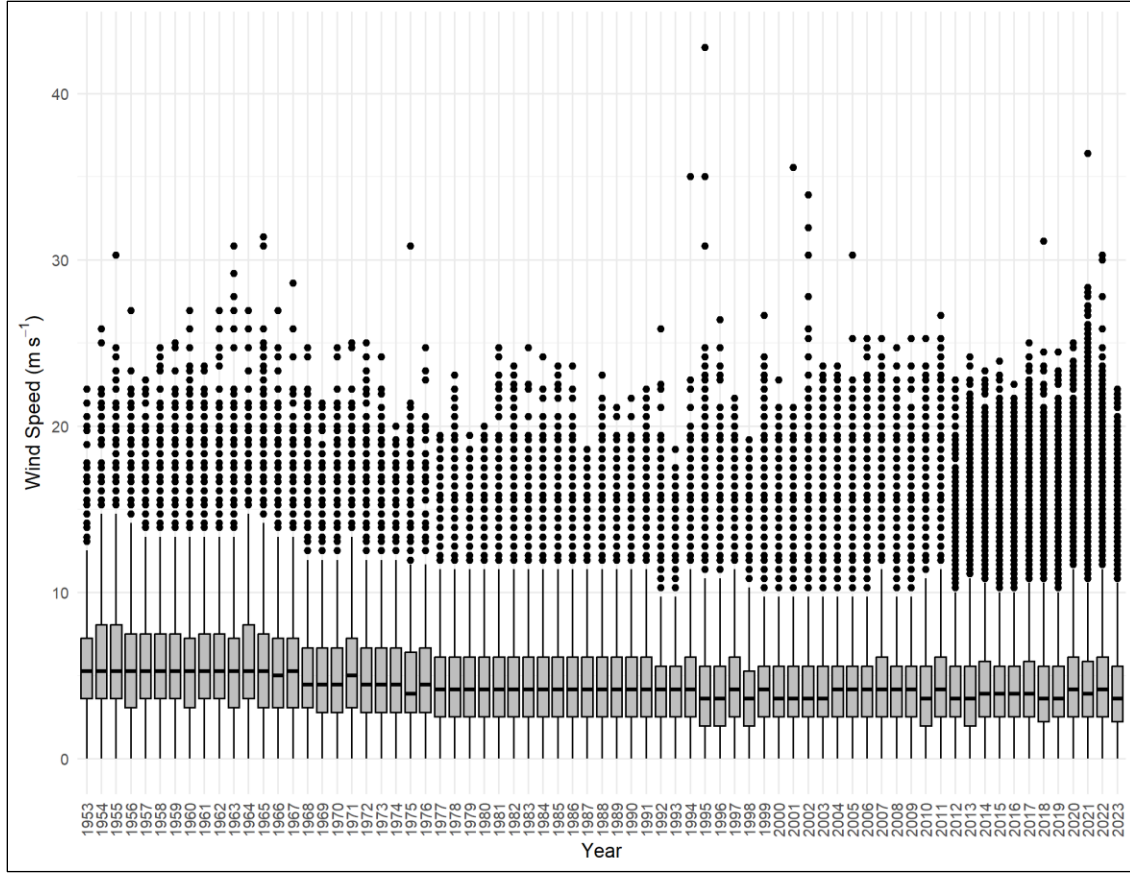


Figure 3.2. Annual distribution of hourly wind speeds across the PPR (1953-2023). Data source: ECCC (2023)

Complementing this, Figure 3.3 displays the probability distribution of hourly wind speeds, showing that the vast majority of observations fall below the critical erosive threshold of 11m s^{-1} (40 km hr^{-1}). This threshold was identified through SWEEP simulations across all major soil textures in the PPR. Nevertheless, approximately 3% of all recorded wind speeds exceed this critical limit, suggesting that while erosive winds are relatively rare in statistical terms, they occur regularly enough to pose erosion risk across the region.

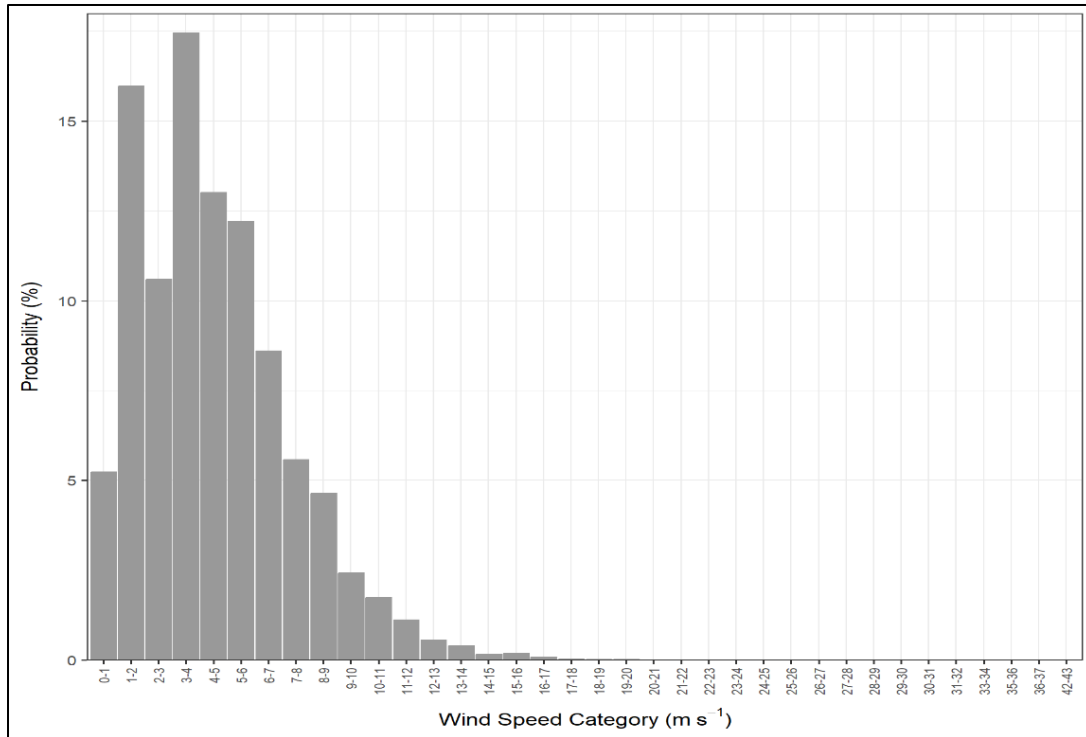


Figure 3.3. Probability distribution of hourly wind speeds across the PPR (1953-2023). Data Source: ECCC (2023).

During the months of May and June, periods often associated with high wind erosion risk, the distribution of hourly wind speeds in the region is skewed toward lower velocity ranges (Figure 3.4). Most wind speeds fall within the 2-6 m s⁻¹ categories, with a noticeable concentration in the 4-6 m s⁻¹ range for both months. Wind speeds between 6 and 10 m s⁻¹ occur less frequently but still represent a substantial portion of the seasonal wind climate. The distribution suggests that most winds are moderate and non-erosive while wind speeds exceeding 11 m s⁻¹ erosivity threshold account for approximately 5% of total observations in each month.

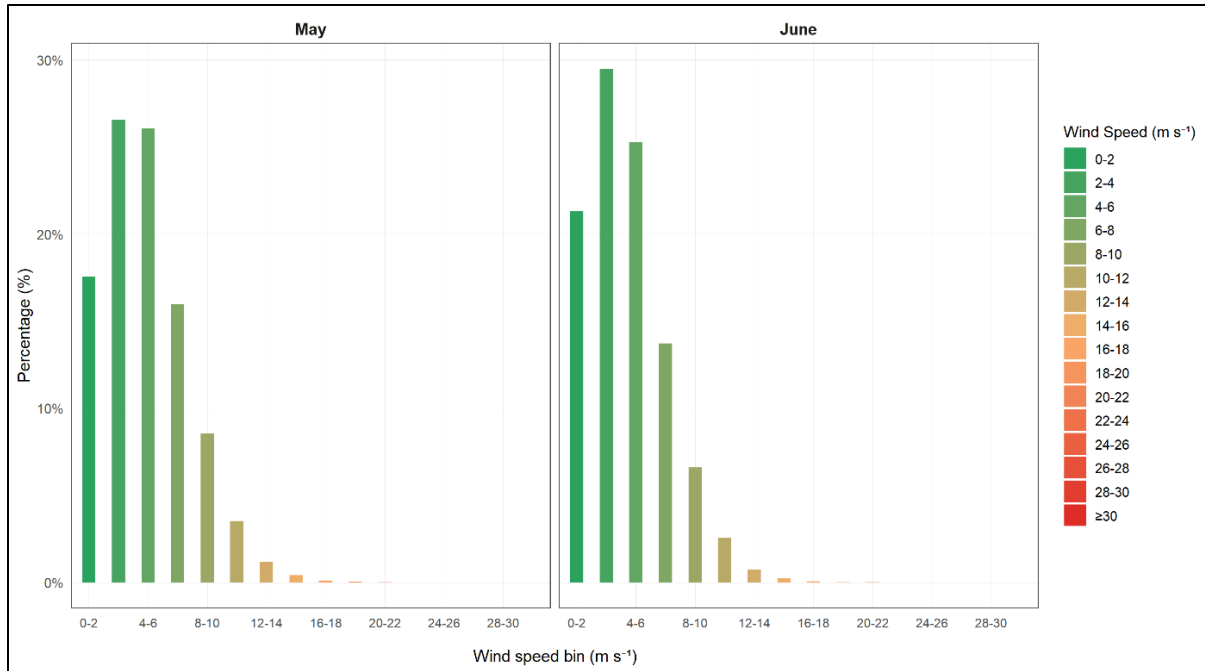


Figure 3.4. Hourly wind speed distribution across the PPR during May and June (1953-2023). Data Source: ECCC (2023).

To complement the long-term hourly dataset, 15-minute interval wind data from the Manitoba Agriculture Weather Program were analyzed for May and June over the period 2008-2024, from weather stations located between 47.5°-54° N latitude and 95°-115° W longitude. This higher-resolution dataset captures short-duration wind gusts that are often missed in hourly records but can still contribute to significant soil transport. These brief but intense events, though infrequent, play a key role in total wind-driven sediment loss. This analysis was conducted to evaluate whether higher temporal resolution data capture a greater frequency of threshold-exceeding wind events and to assess the robustness of conclusions derived from the hourly dataset. As shown in Figure 3.5, wind speeds most frequently occur in the 1-6 m s⁻¹ range, while erosive winds exceeding 11 m s⁻¹ are rare, comprising less than 2% of all observations. This proportion is lower than that observed in the Prairie-wide hourly dataset; however, the two datasets are not directly comparable due to differences in spatial coverage and temporal extent. In particular, the 15-minute dataset is

limited to stations in Manitoba and covers a shorter time period, which may exclude some high-wind years. Nevertheless, the results confirmed that threshold-exceeding wind events remained relatively infrequent even at higher temporal resolution, while also highlighting the importance of regional variability in wind regimes across the PPR.

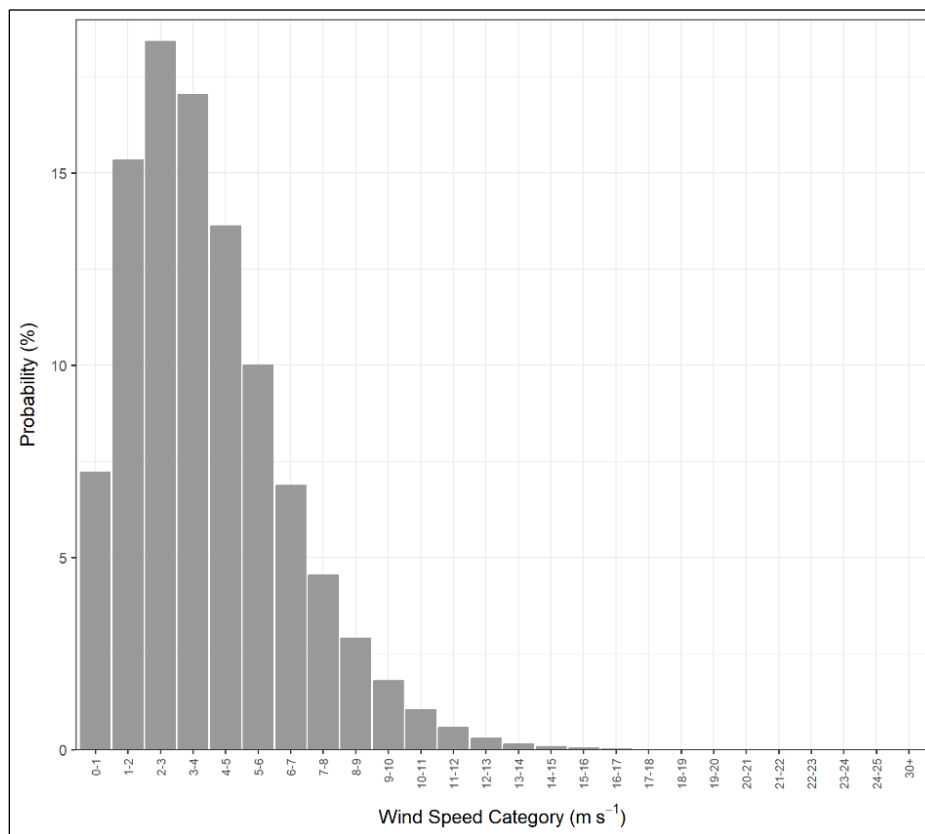


Figure 3.5. Probability distribution of 15-minute wind speeds in Manitoba during the May-June period (2008-2024). Data Source: MAWP (2024)

Analysis of hourly wind direction revealed seasonal variability in directional dispersion. Across the full year, winds predominantly originate from the west and northwest, reflecting a consistent directional regime. During May and June, when surface conditions favour wind erosion, the directional distribution becomes more evenly distributed across sectors. However, westerly and northwesterly winds remain among the most frequent directions. Wind direction frequency is

illustrated in Figure 3.6, which compares all-season and May-June based on historical hourly wind records from 1953 to 2023.

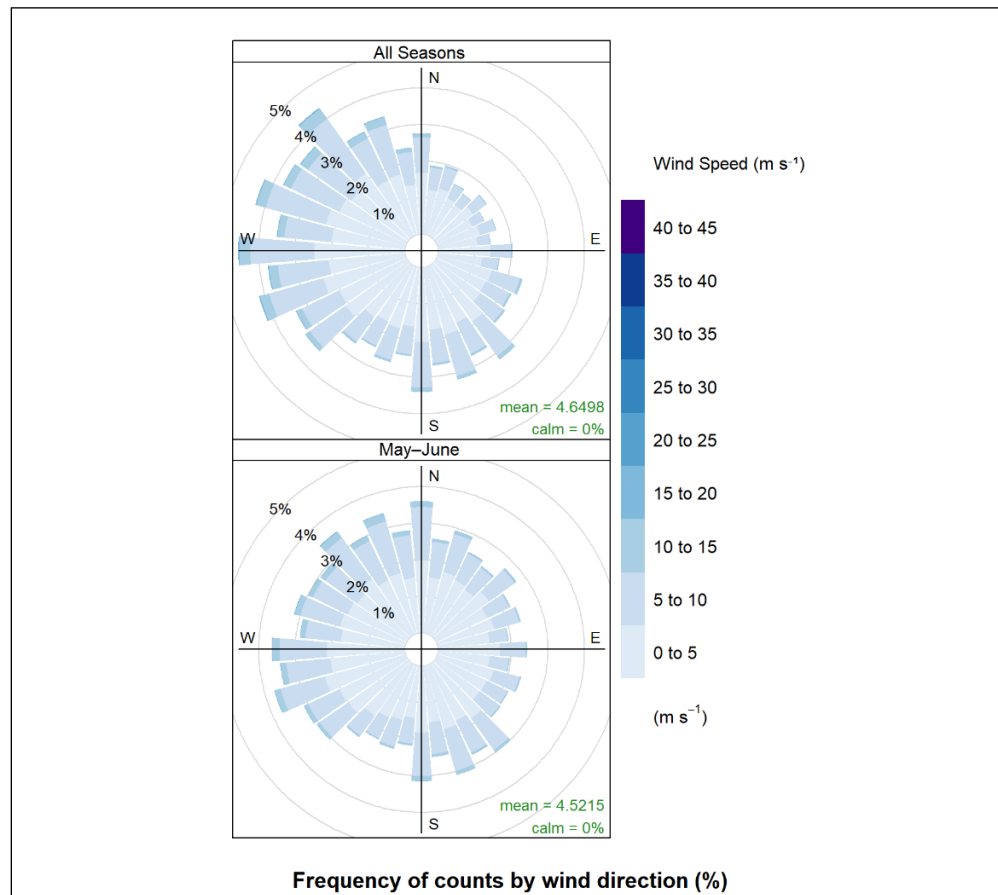


Figure 3.6. Wind direction frequency across the PPR based on hourly data from 1953 to 2023: all seasons (upper panel) and May-June only (lower panel). Data source: ECCC (2023).

3.2.2. Soil

The agricultural soils of the PPR are shaped by a complex interplay of climate, vegetation, glacial history, and topography (Anderson & Cerkowniak, 2010). Soil formation in the region began following the retreat of the glaciers, with southern Saskatchewan becoming ice-free approximately 17500 years ago (Christiansen, 1979). The region displays a clear zonal pattern, with Brown Chernozems prevalent in the dry Mixed Grassland zone, transitioning to Dark Brown and Black Chernozems in the more humid Moist Mixed Grassland and Aspen Parkland, and eventually to

Gray Luvisols in northerly forested lands of the Boreal. These soils have primarily formed on glacial till and glaciolacustrine deposits left behind by the Laurentide and Cordilleran glaciers, with parent materials ranging from carbonate-rich limestone-derived tills in Manitoba to more acidic, shale-influenced materials in Alberta (Moss, 1944; Rutherford et al., 1978). Grassland vegetation increases the accumulation of organic matter within the mineral topsoil (A horizon), particularly in Chernozemic soils, whereas forest vegetation more commonly contributes to the development of distinct organic surface layers (O horizons) above the mineral soil. Additional variability arises from local topography, climatic moisture gradients, and groundwater movement across the region (Anderson & Cerkowniak, 2010). In general, the prairie soils are characterized by high natural fertility and high moisture-holding capacity, factors that contribute to the region's agricultural productivity (Shorthouse, 2010). The distribution of soil texture classes in the Canadian PPR is presented in Figure 3.7, based on the Soil Landscapes of Canada (SLC) data from Agriculture and Agri-Food Canada (2013) and the SLC-SWAT Canada Soils Dataset (Cordeiro et al., 2018).

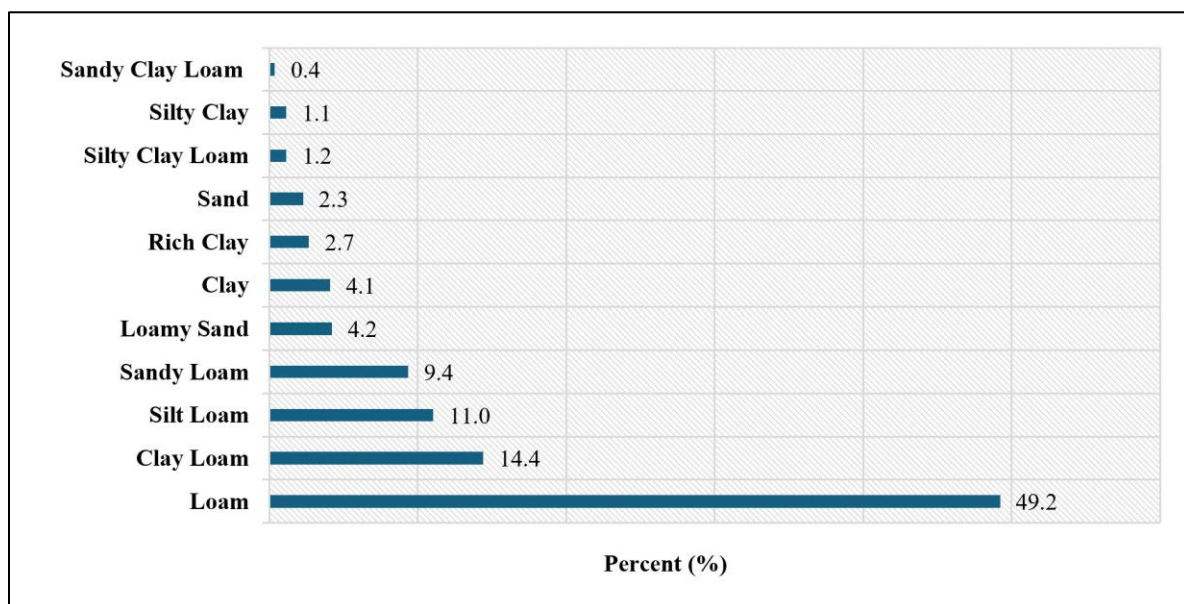


Figure 3.7. Soil texture composition in the Canadian PPR. Data source: SLC (2013) and SLC-SWAT Canada Soils Dataset (Cordeiro et al., 2018)

Soil moisture, a key factor influencing wind erosion, exhibits seasonal patterns across the PPR. Moisture levels are generally higher in early spring and late fall, while significantly lower values are observed during mid to late summer (Reuter & Bell, 2001; Khakural et al., 1998). In addition, the presence of wetlands with varying degrees of hydrological permanence can strongly affect the moisture dynamics of adjacent upland soils, further contributing to spatial variability in surface conditions (Van der Kamp et al., 2003; Rosenberry & Winter, 1997).

3.2.3. Land

The PPR is defined as the Prairies Ecozone in the Canadian Ecostratification system (Ecological Stratification Working Group, 1996; ESTR Secretariat, 2014) and comprises seven distinct ecoregions. In southern Alberta and Saskatchewan, these include the Mixed Grassland, Cypress Upland, Moist Mixed Grassland, and Fescue Grassland. North of the Moist Mixed Grassland lies the Aspen Parkland, which forms a broad arc extending from southwestern Alberta toward the Edmonton region and continuing southeast into Manitoba. In southern Manitoba, the ecozone encompasses the Lake Manitoba Plain and the Boreal Transition ecoregion (Shorthouse, 2010). Since European settlement, the Prairies Ecozone has undergone extensive land transformation, becoming one of the world's largest agricultural regions. This expansion has been accompanied by widespread wetland drainage, resulting in a significant reduction in wetland abundance compared to pre-settlement conditions, although millions of wetlands still remain across the landscape (Dahl, 1990; Shorthouse, 2010; Doherty et al., 2016). According to the 2023 Annual Crop Inventory, cropland dominates the Canadian PPR, covering approximately 56% of the landscape. This is followed by grassland (16%), pasture and forage lands (8%), forest (7%), and shrubland (3%). Other land covers include waterbodies (3%), wetlands (4%), urban or developed areas (2%), and exposed or barren lands (1%). In the context of this study, the term “bare soil” is

adopted in place of barren and exposed lands. Among primary subclasses, spring wheat and canola are the most widely cultivated crops, accounting for 24% and 17% of the PPR, respectively. A detailed description of land cover categories is provided in Section 3.4.1.

3.3. Mass-Conservation Wind Erosion Model (MCWEM)

3.3.1. Model Principles

The Mass-Conservation Wind Erosion Model (MCWEM) developed in this study is grounded in two fundamental physical principles: mass conservation and continuity. These principles form the basis for describing sediment transport processes in dynamic landscapes. The mass conservation principle, originally formulated in physics and chemistry, states that in a closed system, mass can neither be created nor destroyed. Therefore, the total sediment mass within the system is conserved over time. This principle ensures that any sediment detached, transported, or deposited is accounted for in both space and time, with no loss or gain left unaccounted for in the modeling framework (Paola & Voller, 2005; Olmsted & Williams, 1997; Pedrizzetti, 2022; Julien, 2010).

The continuity equation is a core principle that forms the basis of numerous concepts in physics, engineering, and related disciplines. It has wide-ranging and significant applications, serving as a crucial tool for analyzing the movement and behavior of dynamic systems (Singh, 2023). In the context of aeolian sediment transport, this implies that the rate at which sediment enters and exits a given control volume must be reflected in the change in mass within that volume over time. Mathematically, this is expressed as a balance between sediment influx, outflux, and accumulation within a given spatial unit (Pedlosky, 2003; Childs, 2011; Julien, 2010; Shao, 2008). Wind-induced sediment transport occurs from a source to a sink (i.e., depositional area) through processes of detachment and movement across spatial and temporal scales. Each zone within the

landscape includes morphological systems, referring to distinct landforms, and cascading systems, which describe the energy and material flows through those zones (Chorley, 1971; Peters et al., 2008).

The MCWEM applies these principles to simulate wind erosion events, capturing the potential for sediment detachment, transport, and deposition across spatially heterogeneous landscapes. By integrating these foundational laws, the model ensures physical realism in estimating sediment transport and redistribution.

3.3.2. Model Structure

The MCWEM was designed to simulate wind-driven sediment transport across heterogeneous landscapes under erosion conditions. The model structure was rooted in a mass balance framework governed by the continuity equation, ensuring that sediment mass was conserved from detachment to deposition along one-dimensional, spatially explicit pathways.

The model included three core components (Figure 3.9):

1. Pathways and field unit characterization

Linear west-to-east pathways, aligned with the prevailing wind direction, were defined to capture spatial variability in key land characteristics, including land use, land cover, and soil texture, at a pixel resolution of $30\text{ m} \times 30\text{ m}$ for model simulations. Pathways were divided into a sequence of adjacent field units. Each field unit was defined by its land use, land cover, and soil texture, derived from geospatial datasets such as the Annual Crop Inventory (2023), Soil Landscapes of Canada (2013), and the Detailed Soil Survey (2013). These units were spatially arranged along the prevailing wind direction and form the basis for tracking sediment movement. Although each unit

was simulated independently, they were linked through directional sediment flux, governed by transport-mode thresholds and the mass-balance function.

2. Estimation of soil loss with SWEEP

For each type of field unit (e.g., cropland with sandy loam soil), SWEEP model was applied to simulate wind erosion under specified surface and meteorological conditions. SWEEP outputs erosion as grams per meter of boundary width (g m^{-1}), partitioned into distinct transport modes: creep/saltation, suspension, and PM_{10} .

In this study, a boundary width of one meter was assumed, allowing SWEEP outputs in g m^{-1} to be treated directly as soil mass in grams (g). This convention simplified the modeling framework by expressing results as soil mass per 1 m slice of the downwind edge, which made them easier to interpret and present as soil mass (g) versus pathway distance (m). Because all pathways shared a constant 30-m width, this normalization emphasized differences driven by pathway length rather than cross-sectional width. Where necessary, results were scaled to the full 30-m cross-section by multiplying by 30 (Table 4.B). These unit-specific erosion outputs were then incorporated into the mass-balance function, where they were used to calculate net sediment mass and redistribution along the pathways.

3. Mass-balance function and spatial budgeting

A mass-balance transfer function was developed to provide a framework for estimating how eroded sediment is redistributed between sequential field units along each pathway. For each unit, the function calculates net mass balance as the sum of soil loss and deposition received from upwind units, partitioned into creep/saltation, suspension, and PM_{10} modes. Each mode was assigned a maximum transport distance based on particle size, with a distance-decay weighting applied so

that the transferred proportion decreases with increasing distance from the source. This produces alternating zones (lands) of loss and deposition while enforcing mass conservation (total erosion equals total deposition). In its generalized form (Figure 3.7), the function assumes uniform conditions in which all units share the same surface characteristics. Further illustration of this function's behaviour is provided in Figures 3.A- 3.C.

In reality, however, the PPR is characterized by a heterogeneous mosaic of land uses, soil textures, and surface conditions. In this context, the field units along each pathway differ in their erodibility and transport potential. Accordingly, the ML (mass loss) and MD (mass deposition) terms vary from unit to unit, and the transfer function is applied in a more complex form that accounts for spatial variability in surface properties and sediment transport dynamics. To implement the model under heterogeneous conditions, the function was constructed in Excel, where SWEEP-derived soil loss values, expressed as grams per 1-m boundary width, were assigned to each unit, and mass balances were calculated iteratively along the pathway. This provided a practical and transparent means of representing unit-level variability while remaining consistent with the theoretical mass-balance framework.

Mass-balance function

$$M(x) = \begin{cases} -(\text{ML}_{\text{CS}} + \text{ML}_{\text{S}} + \text{ML}_{\text{P}}), & x = 0, \\ -\left(1 - \frac{n_1}{N_{\text{S}}}\right)\text{ML}_{\text{S}} - \left(1 - \frac{n_1}{N_{\text{P}}}\right)\text{ML}_{\text{P}}, & \Delta x \leq x < L_{\text{S}}, \\ -\left(1 - \frac{n_1}{N_{\text{P}}}\right)\text{ML}_{\text{P}}, & L_{\text{S}} \leq x < L_{\text{P}}, \\ 0, & x = L_{\text{P}}, \\ \text{MD}_{\text{CS}} + \text{MD}_{\text{S}} + \text{MD}_{\text{P}}, & x = L_{\text{P}} + \Delta x, \\ \left(1 - \frac{n_2}{N_{\text{S}}}\right)\text{MD}_{\text{S}} + \left(1 - \frac{n_2}{N_{\text{P}}}\right)\text{MD}_{\text{P}}, & L_{\text{P}} + \Delta x < x \leq L_{\text{P}} + L_{\text{S}}, \\ \left(1 - \frac{n_2}{N_{\text{P}}}\right)\text{MD}_{\text{P}}, & L_{\text{P}} + L_{\text{S}} < x \leq 2L_{\text{P}}, \\ 0, & x > 2L_{\text{P}}. \end{cases} \quad (1)$$

- $M(x)$ – Net soil mass (g) in the field unit at distance x from the source.
- Δx – Length of each field unit (m).
- x – Distance from the source (m).
- L_i – Assumed maximum transport length (m) for mode i ($i \in \{\text{CS}, \text{S}, \text{P}\}$). Here L_{S} and L_{P} denote the suspension and PM_{10} limits, respectively.
- ML_i – Total mass of loss (g) by mode i , obtained from SWEEP for the source unit.
- MD_i – Total mass of deposition (g) received by a unit for mode i after redistribution.
- CS – Creep/saltation transport mode.
- S – Suspension transport mode.
- P – PM_{10} mode.
- $n_1 = \lfloor x/\Delta x \rfloor$ – Number of field units from the start of the loss zone.
- $n_2 = \lfloor (x - L_{\text{P}})/\Delta x \rfloor$ – Number of field units measured from the start of the deposition zone.
- $N_{\text{S}} = \lceil L_{\text{S}}/\Delta x \rceil$, $N_{\text{P}} = \lceil L_{\text{P}}/\Delta x \rceil$ – Total units spanning the suspension and PM_{10} transport lengths.

(1) This function assumes uniform conditions, where all field units share similar surface properties. In the heterogeneous mosaic-like patterns of the landscape of the PPR, the values of ML and MD vary between field units due to differences in land use, cover, soil texture and other surface characteristics.

Figure 3.8. Mass-balance transfer function for estimating net soil mass change between 1-m boundary width of sequential field units under uniform conditions, with mode-specific transport limits and distance-decay weighting.

In this study, each pathway was defined with a fixed width of 30 m and discretized into sequential field units, creating pathways of varying length aligned with the prevailing wind direction. For sediment redistribution, maximum transport distances for each mode of transport (creep, saltation, and suspension) were derived from established relationships between particle size and transport distance under a wind speed of 30 m s^{-1} . This wind speed represents a high-intensity wind erosion condition, substantially exceeding the typical erosive threshold of 11 m s^{-1} , and was selected to estimate the upper limits of transport distance, consistent with the scenario assumptions described later in the study. These estimates were calculated using governing equations from Stokes' law, kinematic transport theory, the Owen and Bagnold models. Detailed particle-size classes, governing equations, and primary references used to determine these thresholds are provided in Table 3.A. Based on these assumptions, coarse particles, including sand (2-0.1 mm) transported by creep and saltation, were assigned a maximum transport distance of 30 m; particles transported primarily in suspension, including very fine sand and silt (0.1-0.01 mm), were assigned a 600 m limit; and PM_{10} particles, including fine silt and clay ($<0.01 \text{ mm}$), were assigned a maximum transport distance of 3,000 m. These distances represent conservative transport-distance assumptions within defined pathways and are consistent with the objective of this study to quantify sediment redistribution across local, landscape, and regional scales rather than global atmospheric dust transport. Although extremely fine particles may travel substantially longer distances under favourable atmospheric conditions, their relative mass contribution within the mass-balance framework applied in this study is extremely small and was therefore considered negligible.

Using SWEEP outputs in combination with the mass-balance function, the model simulated sediment movement and net mass balance along each pathway. This generated a spatially explicit

profile of erosion, deposition, and net soil change by transport mode, including the amount ultimately delivered to the terminal land sink, Lake Winnipeg, at the downwind end of the pathway.

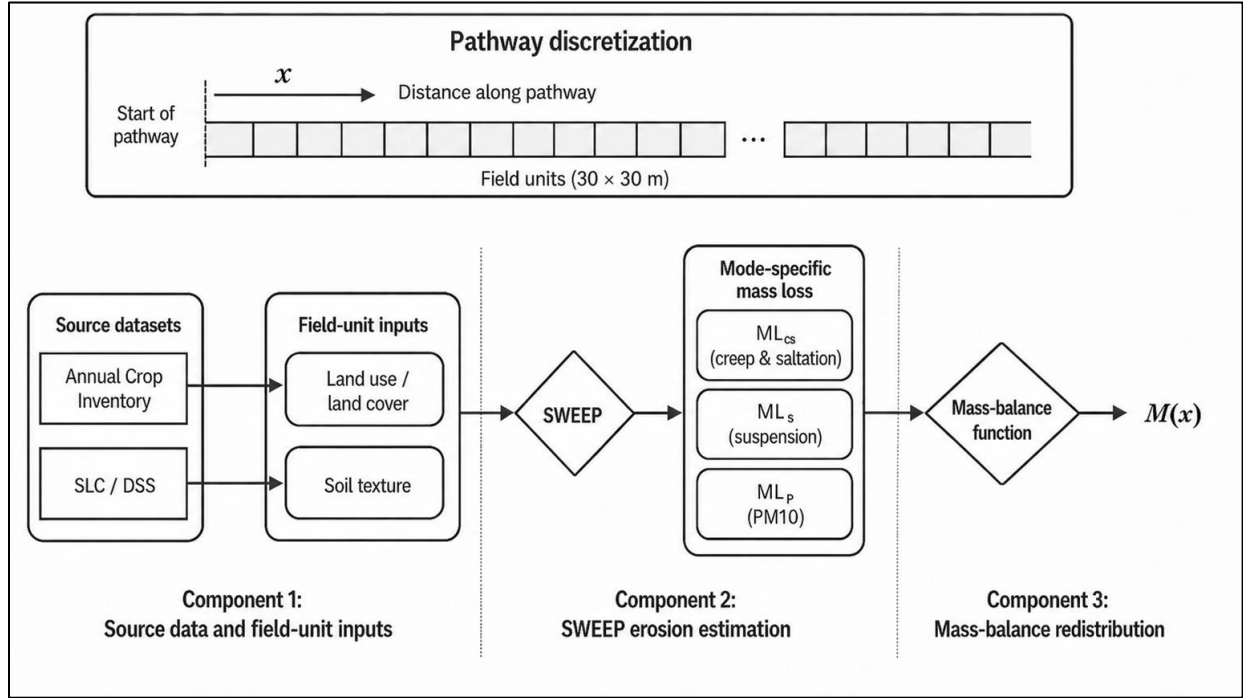


Figure 3.9. Structure of the Mass-Conservation Wind Erosion Model (MCWEM), including pathway discretization, source-data inputs, SWEEP-based erosion estimation, and the mass-balance redistribution component.

3.3.3. Model Parameters

The parameters used in the MCWEM were classified as either constant or variable, reflecting their role and flexibility within the simulations. This distinction is important because constant parameters establish the baseline physical environment and modeling constraints, whereas variable parameters represent the heterogeneity of conditions along each pathway.

Constant parameters remained fixed across all simulations and defined the fundamental conditions of the modeled environment. These included the spatial resolution of the field units, set at 30 m × 30 m to match the resolution of the Annual Crop Inventory (2023) and soil datasets (i.e., SLC and DSS). Wind conditions were fixed, assuming a constant velocity and direction over a single wind

event during a 24-hour period, consistent with the SWEEP model configuration. This duration was considered sufficient for wind to traverse the entire pathway. Wind flow was assumed to occur along the west-east axis of hypothetical pathways to reflect predominant erosive wind patterns observed in May and June. Terrain characteristics were simplified to a uniform and gentle slope across the entire pathway, thereby minimizing elevation-induced wind variability and isolating the effects of surface properties on sediment transport. Soil moisture was similarly held constant at a dry condition to maximize erodibility potential.

Variable parameters, in contrast, differed among field units and across simulations to reflect the inherent heterogeneity of the PPR landscape. Land use/cover was a primary variable, derived from the Annual Crop Inventory (2023) and grouped into five main categories: bare soil; cropland; grassland, forest and shrubland; roads and buildings; and wetlands and water bodies. Each category was associated with different management practices, vegetation cover, surface roughness, and biomass, all of which influence erodibility and wind transport potential. Soil texture was another critical variable, obtained from the SLC and DSS databases. Textures such as loam, clay loam, sand, and loamy sand differ in particle size distribution and cohesion, which directly affect threshold friction velocities and transport-mode partitioning (Shao, 2008). Pathway length, indicated by the total number of sequential field units in a simulation, also varied and determined the simulation scale. Taken together, these constant and variable parameters allowed the MCWEM to capture how combinations of land use, soil texture, and spatial scale influence wind-driven sediment transport, while realistically representing the heterogeneous mosaic patterns characteristic of the PPR landscape.

3.4. Model Scenarios

3.4.1. Structure of Scenarios

The design of model scenarios in this study was guided by the need to capture the variability of environmental conditions across the PPR while allowing a systematic evaluation of how specific variable parameters influence wind-eroded sediment transport. To achieve this, the scenarios were structured around three primary groups of variable parameters: (1) land use/cover class, (2) soil texture, and (3) simulation scale. Each group represents a key factor affecting wind erosion processes and sediment redistribution patterns, and their combinations define the full range of possible simulation conditions.

Land use/cover class strongly influences the susceptibility of soils to wind erosion by modifying aerodynamic and near-surface wind flow, surface roughness, vegetative protection, and soil stability. Five primary land use/cover categories were considered based on the Agriculture and Agri-food Canada (AAFC) classification:

1. Croplands (C): refers to land actively managed to produce agricultural crops, including both annual and perennial species. This group encompasses undifferentiated agricultural land, cereals, oilseeds, pulses, vegetables, fruits, and other specialty crops. Vegetative cover is seasonal and varies according to crop type and phenological stage and cropland is often associated with reduced vegetative cover during certain periods of the year, making it highly vulnerable to wind erosion, particularly after tillage or during post-harvest periods.
2. Grassland, forest, and shrublands (G): refers to areas dominated by perennial vegetation that generally provides substantial and persistent surface protection against wind erosion.

Grasslands consist primarily of native grasses and herbaceous vegetation, while shrublands contain low-stature woody vegetation and may include regenerating forests or shrub-dominated wetlands. Forested areas, classified as coniferous, broadleaf, or mixedwood, form dense canopies that intercept wind, enhance surface roughness, and stabilize soils. Collectively, these covers act primarily as erosion barriers and, in many cases, as sediment sinks.

3. Bare soil (B): refers to non-vegetated surfaces that are highly susceptible to wind erosion. In the AAFC classification, this corresponds to exposed land/barren areas and includes glacier ice, rock, unconsolidated sediments, burned areas, rubble, mines, and other naturally occurring non-vegetated surfaces. It excludes fallow agricultural land, which is classified under cropland. These surfaces offer minimal resistance to wind shear, allowing direct entrainment of particles and rapid transport under erosive wind conditions.
4. Wetlands and water bodies (W): refers to both aquatic environments and saturated terrestrial ecosystems that act as terminal sediment sinks. Wetlands, such as marshes, bogs, fens, swamps, and peatlands, are characterized by high water tables and vegetation adapted to saturated conditions. Water bodies include lakes, rivers, reservoirs, and streams. These environments are inherently non-erodible by wind and play a critical role in trapping and storing wind-transported sediment.
5. Road/Building (U): refers to urban and developed lands are characterized by impervious or compacted surfaces, including buildings, roads, railways, industrial sites, and associated vegetation. This group also includes greenhouse facilities. Such surfaces are generally non-erodible; however, their structural features can alter local wind fields, create turbulence, and influence the spatial patterns of sediment deposition in adjacent erodible areas.

The choice of land use/cover type for each field unit directly affects its erosion potential and its role as a source, transport zone, or sink within the modeled domain.

Soil texture influences wind erosion primarily by controlling particle detachment thresholds and surface cohesion, thereby regulating the wind energy required to initiate erosion and the availability of particles for subsequent transport. Its interaction with land use strongly influences the erosion potential of a field unit; for instance, a bare sandy loam field is far more erodible than a clay loam grassland under identical wind conditions. Each soil texture is characterized by distinct parameters in the SWEEP configuration, including soil layer thickness, bulk density, aggregate density, and particle-size fractions. These properties collectively determine how soil responds to erosive forces and ultimately shape the magnitude and distribution of sediment transport.

Simulation scale is determined by the number of field units (n) into which the pathway is divided.

Three scales were considered:

- Small scale (field level): Represented by a low number of field units, typically corresponding to a catchment or single agricultural field.
- Medium scale (landscape level): Consists of a larger number of units covering a mosaic of land uses and soil textures, typical of a local watershed or a section of land.
- Large scale (regional level): Encompasses extensive landscapes, potentially spanning hundreds of kilometers and incorporating major physiographic and land cover diversity.

The number of field units not only defines scale of the simulation but also determines the total number of possible scenarios. As the number of field units increases, the number of potential land use/cover–soil texture combinations increase exponentially.

The total number of possible scenarios depends on three main factors: (1) the number of field units (n), (2) the possible land use/cover and soil texture combinations assigned to each field unit, and (3) the spatial arrangement of these units along the simulation pathway. The interaction of these factors can generate an extremely large number of scenarios. For instance, even with only five land use categories and three soil texture categories, a 10-unit pathway could produce approximately 2 million unique compositions if spatial arrangement is ignored, but 15^{10} distinct pathways when spatial arrangement is considered. This highlights the necessity of defining representative scenario types rather than attempting to model every scenario.

3.4.2. Classification of Scenarios

To provide a structured approach to scenario analysis, the model scenarios were categorized into two broad classes: basic scenarios and complex scenarios. Basic scenarios were defined by the use of a single variable from each parameter group. In these scenarios, all field units were assigned the same land use/cover class and soil texture for a given simulation scale. This results in a uniform spatial arrangement, with no variation across the domain. Such scenarios are valuable for isolating the effects of individual parameters, enabling the quantification of the influence of a specific land use-soil texture combination without interference from other variables. They also provided a baseline or benchmark against which the results of more complex, heterogeneous configurations would be compared.

Complex scenarios represented heterogeneous landscapes in which multiple land use/cover classes and soil textures were distributed across the simulation domain. Individual field units would differ in one or both attributes, producing a spatial mosaic. This approach more accurately reflected the

inherent variability of the PPR landscape. Complex scenarios enabled the investigation of the role of spatial configuration in wind erosion transport.

3.4.3. Worst-Case Complex Scenario Model (MCWEM-WCS)

To capture the spatial heterogeneity of the PPR, complex scenarios were selected to represent hypothetical pathways extending from west to east across the region. These scenarios incorporated all primary land use and land cover groups identified in this study, combined with dominant and erosion-susceptible soil texture classes present across the PPR at different scales. To do this, the pattern of land use/cover, soil texture and spatial arrangement of units were derived from high-resolution aerial imagery from Google Maps, spatial analysis of the Annual Crop Inventory (2023), and soil data obtained from the SLC (2013) and the DSS (2013). These complex scenarios provided a realistic representation of the PPR's heterogeneous mosaic, where cropland, natural vegetation, wetlands, and developed areas occur in varying proportions and spatial arrangements. To assess scale-dependent responses, three representative simulation scales were implemented:

- Field scale (0.6 km): Represents localized field conditions along a short pathway terminating at Lake Winnipeg, capturing fine-scale variability and short-distance sediment transport.
- Landscape scale (3 km): Represents landscape conditions along a longer pathway terminating at Lake Winnipeg, incorporating greater heterogeneity and more extensive sediment transport processes.
- Regional scale (1,100 km): Represents regional conditions across the PPR, extending from west to east and terminating at Lake Winnipeg.

These complex scenarios were evaluated under extreme, or worst-case, conditions to assess the model's predictive capacity under the most severe environmental circumstances plausible in the

PPR. Testing scenarios under maximum wind erosion susceptibility allowed the simulations to capture the upper limit of potential sediment detachment, transport, and deposition. This approach provided a test of the study hypotheses. If the model predicted minimal sedimentation under these extreme conditions, then more moderate and realistic configurations would be expected to produce even lower sedimentation rates, supporting rejection of the hypotheses. Conversely, substantial sedimentation under worst-case conditions would warrant further sensitivity analyses under less severe settings.

To run the selected complex scenarios under worst-case conditions, all parameters were configured to maximize erosion potential. To provide a realistic wind time series input, wind data for the Mass-Conservation Wind Erosion Model–Worst Case Complex Scenarios (MCWEM-WCS) were obtained from the Manitoba Agriculture Weather Program (2024), which maintains a network of over 100 automated weather stations across the province and provides high-resolution meteorological data at 15-minute intervals. Data from May and June between 2008 and 2024 were examined to capture the most extreme wind events. All days within this period were screened to identify the date with the highest recorded frequency of erosive winds while preserving temporal variability in the input dataset. The selected record was from 31 May 2022 at the Minnedosa weather station (Climate ID: 5011760; 50° 16' N, 99° 50' W; elevation: 521 m), which recorded erosive wind conditions in 58 of the 96 intervals that day (see Table S1), with a predominant westerly wind direction (270°) and an average daily temperature of 12 °C. The simulated landform was represented as flat terrain, with an average slope of 0.4°, to eliminate topographic sheltering effects. Land use included all five study categories arranged in a heterogeneous but realistic mosaic. Soil texture assignments combined dominant classes with highly erodible types, identified

from provincial DSS datasets. Table 3.1 shows the unique field unit types present in each WCS pathway.

Table 3.1. Field unit types present in each MCWEM-WCS pathway, simulated in SWEEP to estimate soil loss from each field unit within the pathway. Codes represent combinations of land use/cover (first part: C = Cropland, G = Grassland/Forest/Shrubland, B = Bare soil, W = Wetland/Water, U = Road/Building) and soil texture (second part: Cl = Clay, HCl=Rich clay, L = Loam, S = Sand, Si=Silt). Each type may occur multiple times within a pathway.

WCS Pathway	Field Unit Types (Land use/cover – Soil Texture)	Total Number of Field Units
Field Scale (0.6 km)	C-L, U-L, G-L, G-S, W, B-LS	20
Landscape Scale (3 km)	C-Cl, C-L, C-S, G-L, U-Cl, C-CIL, C-LS, G-Cl, G-S, W, U-L, B-LS	100
Regional Scale (1,100 km)	C-Cl, C-CIL, C-HCl, C-L, C-LS, B-S, C-S, C-SCIL, C-SiL, C-SiCl, G-L, G-Cl, G-CIL, G-S, W, U-Cl, U-L, B-LS	~36,665

During SWEEP runs, all field units classified as grassland, regardless of soil texture, were configured with a minimum of 30% surface residue cover, consistent with typical conditions in perennial grasslands of the PPR. Under these configurations, predicted soil loss was very low, with total erosion rates approaching zero across all tested soil textures. This finding is consistent with established wind erosion research, which indicates that maintaining vegetative or residue cover substantially increases aerodynamic roughness, dissipates wind energy, and shields the soil surface from particle detachment (Skidmore & Siddoway, 1978; Fryrear et al., 2000). Given the consistently minimal erosion potential, grassland field units were assigned a soil loss value of zero in the model. This simplification reduced computational complexity while maintaining model accuracy.

3.5. Fractional Deposition at Terminal Sink

At the terminus of each pathway, Lake Winnipeg was considered the final land sink for wind-driven sediment. To assess how the incoming sediment load was redistributed by particle-size fraction within this terminal sink, the exponential decay function of Summation-Curve framework developed by Lobb et al. (1995, 2001) was applied. The function (Lobb et al., 1995; Lobb & Kachanoski, 1999) is given by:

$$k_f(x) = \exp(-x/\lambda_f)$$

where x is the downwind distance, f is the particle-size fraction, and λ_f is the characteristic transport length for that fraction. This function was used to estimate the fraction of mass redistributed at each distance. The value of k_f was multiplied by the total mass of each fraction to determine the amount of sediment present at distance x . This produced a spatially explicit profile of deposition and sediment delivery to Lake Winnipeg for each particle-size fraction (see Figure 3.D for redistribution under uniform and heterogeneous surface conditions).

3.6. Performance Assessment of MCWEM-WCS

MCWEM-WCS performance was assessed by comparing scenario outputs (model predictions) with observed field data from recent studies. These studies were selected because they provide quantitative field-based evidence and are closely aligned with the study context, particularly wind-induced sediment redistribution in agricultural landscapes of the PPR. Their datasets include ^{137}Cs radionuclide inventories, sediment flux measurements, and sediment budget analyses, which were used to evaluate erosion magnitudes, redistribution processes, and depositional patterns. Results from studies by Baldwin (2015), Brooks (2022), and Zarrinabadi (2023) provide an empirical basis

for evaluating the model's assumptions and its predicted temporal and spatial trends under worst-case scenarios. The outcomes of each of these studies were compared in detail with the model results in Chapter 4.

Baldwin (2015) conducted a field-based investigation of wind erosion on highly sensitive agricultural soils in southern Alberta, between Lethbridge and Medicine Hat. Motivated by the apparent disconnect between national wind erosion risk models (WindERI) and limited visual evidence of erosion, the study aimed to provide more accurate erosion estimates using ^{137}Cs tracing. The research focused on two cultivated fields, referred to as Field 1 and Field 2; both were subject to intensive tillage in a potato-crop rotation known for low post-harvest residue cover, conditions that heighten vulnerability to wind erosion. The experimental design involved a systematic grid of parallel transects aligned with prevailing southwest winds to capture spatial variability in ^{137}Cs inventories. Sampling targeted both cultivated in-field areas and grassed field corners, with additional transects established in two undisturbed pasture reference sites to obtain a regional reference value. Each sampling location included two distinct soil samples: a surface sample (0-5 cm) and a sectioned core sample collected in 15 cm increments to a depth of 45 cm in fields and 60 cm in reference areas (Figure 3.10).

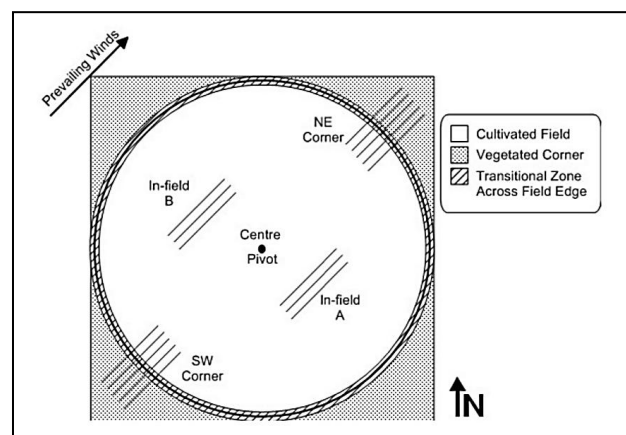


Figure 3.10. Layout of parallel transects in Fields 1 and 2. Reproduced from Baldwin (2015).

Based on the study's methodological design, several expectations were established regarding the spatial distribution of ^{137}Cs under conditions dominated by wind erosion. It was hypothesized that upper slope positions, more exposed to wind and lacking vegetative cover, would exhibit depletion of ^{137}Cs due to preferential removal of fine, radionuclide-associated soil particles. In contrast, lower and depressional slope positions were expected to accumulate eroded soil, resulting in higher ^{137}Cs concentrations. Similarly, cultivated field areas were anticipated to show reduced inventories compared to undisturbed reference sites that have remained vegetated and untilled since the 1960s. Additional spatial expectations were linked to field edges and prevailing wind directions: the vegetated corners of each field, which serve as roughness elements and deposition zones, were expected to retain higher ^{137}Cs levels than the adjacent cultivated corners. Moreover, due to dominant southwest winds in the region, the northeast (leeward) corners were anticipated to accumulate more ^{137}Cs than the southwest (windward) corners. These expectations guided the interpretation of spatial ^{137}Cs patterns across various slope positions and land cover types, allowing the study to test whether observed distributions aligned with those typically associated with wind-driven sediment transport.

By applying ^{137}Cs radionuclide tracing, Baldwin (2015) quantified net soil loss across cultivated fields, generating an integrative measure of erosion that aligns closely with the objectives of mass-conservation modeling. Through systematic sampling across slope positions in both cultivated fields and adjacent non-cultivated margins, Baldwin (2015) captured the spatial variability necessary to identify patterns of sediment redistribution.

Brooks (2022) conducted a comprehensive watershed-scale assessment of soil erosion processes in the agricultural landscapes of Manitoba's Red River Valley. The study aimed to characterize the magnitude and spatial extent of erosion associated with tillage, water, and wind forces by

constructing process-separated sediment budgets for two representative prairie sub-watersheds: 05OF024 (Boyne–Morris River) and 05OG008 (La Salle River). These watersheds were chosen to reflect key differences in land management, drainage infrastructure, and erosion susceptibility, and together represent typical conditions in intensively cultivated regions of the Canadian prairies.

To achieve this, Brooks employed a sediment budget framework that disaggregated erosion into its primary drivers (tillage, water, and wind) and quantified three major components: gross sediment production, deposition in roadside ditches, and net sediment delivery to stream outlets. The structure of the sediment budget framework is depicted in Figure 3.11, which summarizes the sediment sources, transfer mechanisms, and sinks across cultivated fields, roadside ditches, and stream networks. This methodology integrated three core data sources: field-based surveys of ditch-deposited sediment, streamflow and total suspended solids (TSS) data from Water Survey of Canada stations, and erosion-risk indices derived from the NAHARP SoilERI model. This combination enabled spatial attribution of sediment sources and estimation of erosion intensity across the landscape.

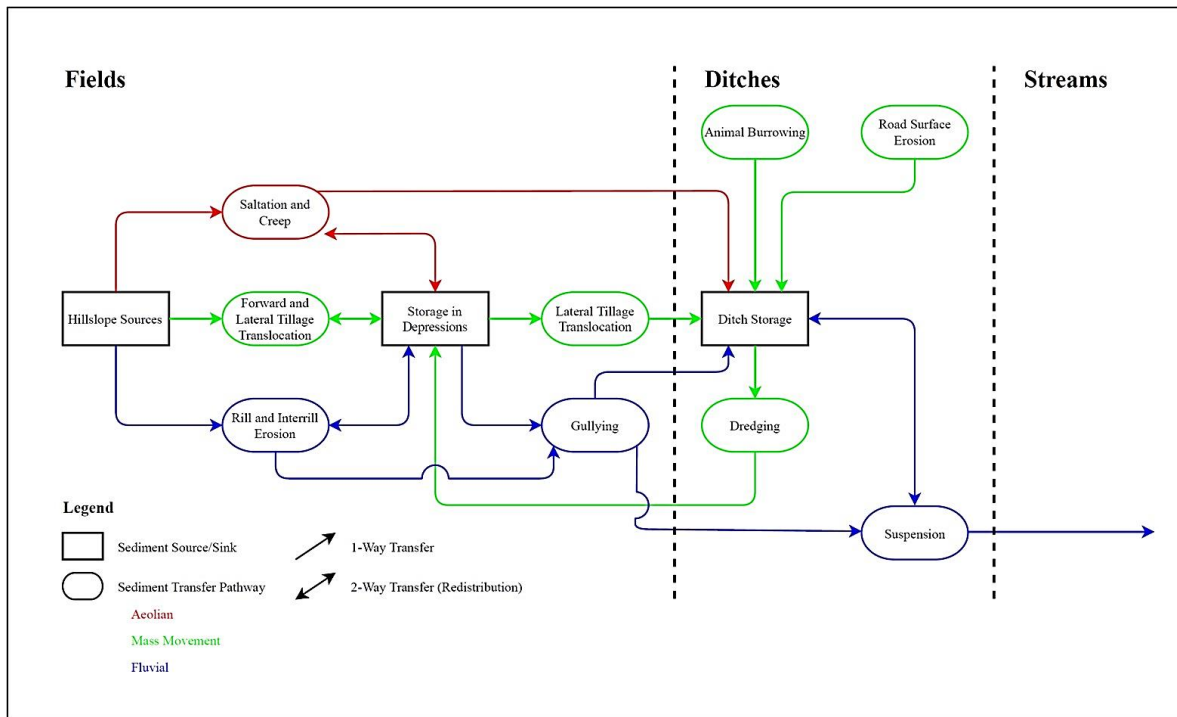


Figure 3.11. Flowchart of sediment transfers among fields, roadside ditches, and streams, showing erosion, transport, and deposition driven by aeolian, fluvial, and mass-movement processes. Reproduced from Brooks (2022).

Wind erosion was specifically assessed through detailed field surveys of roadside ditches to identify and quantify depositional features. Brooks focused on two wind-driven sediment deposits: snirt (a mixture of snow and soil), which was assumed to occur in all roadside ditches, and blow-dirt, which was recorded and mapped based on municipal maintenance records following major dust storm events. For each depositional feature, Brooks measured its length, width, and height, collected soil cores to determine bulk density, and calculated feature volume and mass using geometry-based equations. Mean feature mass was then combined with the observed frequency of that feature per kilometer of ditch to estimate sediment transfer rates and the average annual mass of wind-driven deposition to ditch sinks. By explicitly quantifying the contribution of wind to localized redistribution, Brooks' work provides a valuable empirical benchmark for understanding wind-driven sediment dynamics at the watershed scale. The findings offer an important point of comparison for evaluating the MCWEM-WCS.

Zarrinabadi (2023) conducted a comprehensive assessment of soil erosion across the Canadian PPR, focusing on sediment redistribution from wind, water, and tillage erosion in agricultural landscapes. The research combined multiple methodological approaches to capture erosion dynamics at different spatial and temporal scales across Prairie landscape. Although Zarrinabadi's study addressed multiple erosion processes and employed a range of methodologies, the comparison was primarily focused on the field-derived ^{137}Cs measurements to evaluate the performance of the MCWEM-WCS. The ^{137}Cs tracer method provides a cumulative record of soil redistribution and thus offers a robust empirical benchmark for assessing modeled predictions of erosion and deposition under heterogeneous Prairie conditions.

The ^{137}Cs measurements were conducted within wetland landscapes in the Broughton's Creek and Bigstone Creek watersheds in southwestern Manitoba and central Alberta, respectively. In this region, transects were established to extend from cultivated uplands downslope toward riparian corridors and wetland basins. This sampling strategy was designed to capture the full sequence of erosion and deposition across the landscape. Soil cores were collected during the 2018-2019 field seasons. Cultivated fields were sampled to a depth of 45 cm, while undisturbed reference sites, such as permanent grasslands or wetlands, were sampled to 60 cm. The reference sites established the baseline ^{137}Cs inventory, reflecting the relatively uniform fallout from 1950s -1960s atmospheric nuclear testing. Because ^{137}Cs binds strongly to fine soil particles and is immobile once deposited, deviations from these baseline inventories provide a direct measure of cumulative soil redistribution over the past five to six decades. Cores were sectioned, dried, homogenized, and analyzed by gamma spectrometry to quantify ^{137}Cs activity. Inventories were compared with reference levels to determine net soil loss or gain at each sampling position. Sites with depleted inventories indicated long-term erosion, reflecting soil removal from upland fields. Conversely,

enriched inventories indicated depositional accumulation, typically observed in downslope depressional areas, riparian zones, or wetlands where eroded sediment was trapped. For the purposes of this study, Zarrinabadi's field measurements were used to test the assumptions of the MCWEM-WCS, particularly regarding transport distances, particle-size partitioning, and the delivery of sediment to terminal sinks.

CHAPTER 4: Results and Discussion

4.1. Model Results

As noted in Section 3.3.2, SWEEP reports event-based erosion as mass per meter of the downwind boundary. Because pathway width was fixed at 30 m in all simulations, a 1 m boundary width was adopted to standardize results across scales and enable direct comparison among pathways; accordingly, all quantities were reported as soil mass in grams (g) unless noted otherwise. Full-width totals for each 30×30 m field unit were reported in Table 4.A, and pathway-level totals were reported in Table 4.B.

Model simulations produced mass-balance values for each spatial unit along the three WCS pathways representing the regional, landscape, and field scales (Figures 4.1-4.2). Mass balance is defined as the net soil balance for each field unit, calculated as the mass leaving the unit minus the mass entering the unit. Positive values represent soil loss, while negative values represent deposition. At the regional-scale pathway (0-1,100 km), soil mass balance fluctuated around zero with frequent alternations between loss and deposition. Most units remained within $\pm 1,000$ g per 1-m boundary width, while some showed larger deviations, reaching 3,930 g of soil loss and 4,456 g of deposition. At the landscape-scale pathway (0-3 km), mass-balance values were also centered near zero, with fewer extreme values than the regional-scale pathway. Most values remained within ± 500 g per 1-m boundary width, while the largest deviations were represented by a pair of adjacent units, reaching 3,444 g of soil loss and 2,551 g of deposition. At the field-scale pathway (0-600 m), values were tightly clustered around zero, with fewer outliers than the regional- and landscape-

scale pathways. Maximum values reached 1,664 g of soil loss and 1,211 g of deposition. The corresponding distribution in Figure 4.2 shows narrow interquartile ranges across all three scales, with the main differences occurring in the magnitude and frequency of outliers. In general, the model results showed alternating patterns of gain and loss across all three pathways, with unit-level variation decreasing from the regional scale to the field scale.

Figure 4.3 shows the modeled soil mass balance at the end of the landscape- and field-scale pathways, extending up to 600 m into Lake Winnipeg. Because the regional- and landscape-scale pathways produced identical outputs, only the landscape-scale pathway was reported for comparison with the field-scale pathway. At the landscape-scale pathway, the model simulates a net soil mass balance of approximately 1,221 g of deposition within the first 30 m of Lake Winnipeg including the riparian area. Thereafter, values decrease gradually, reaching approximately zero by 600 m. At the field-scale pathway, results are comparable, with approximately 1,211 g of deposition in the first 30 m followed by a progressive decline to zero g by 600 m. The results indicated that the three pathways produced nearly identical magnitudes, with deposition concentrated within the first 30 m and steadily declining toward zero as the pathways extended into the central part of the lake.

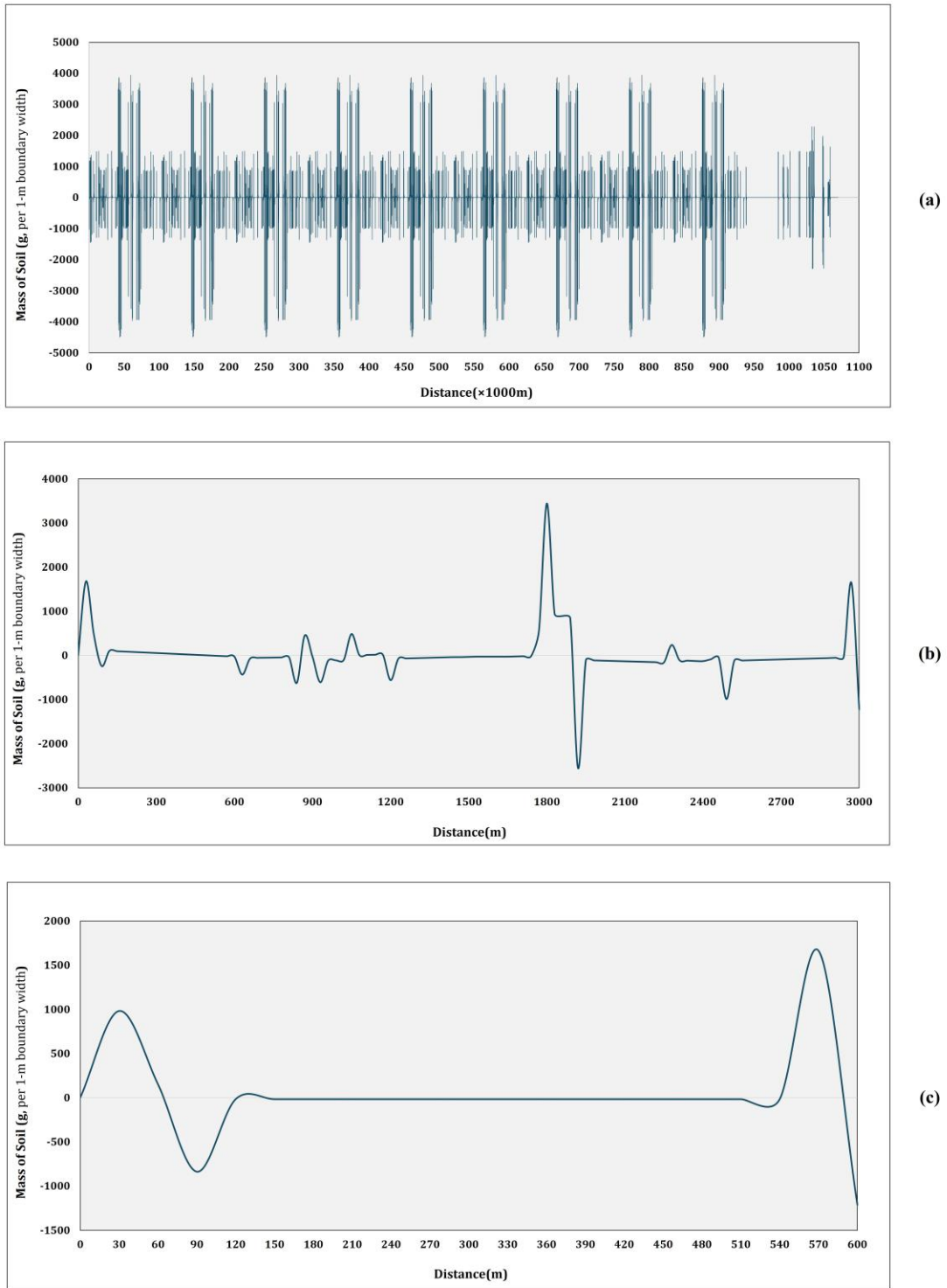


Figure 4.1. Model result of unit-level soil mass balance across the three WCS pathways: (a) regional (0-1,100 km), (b) landscape (0-3 km), and (c) field scale (0-600 m). Positive values indicate net erosion, while negative values indicate net deposition.

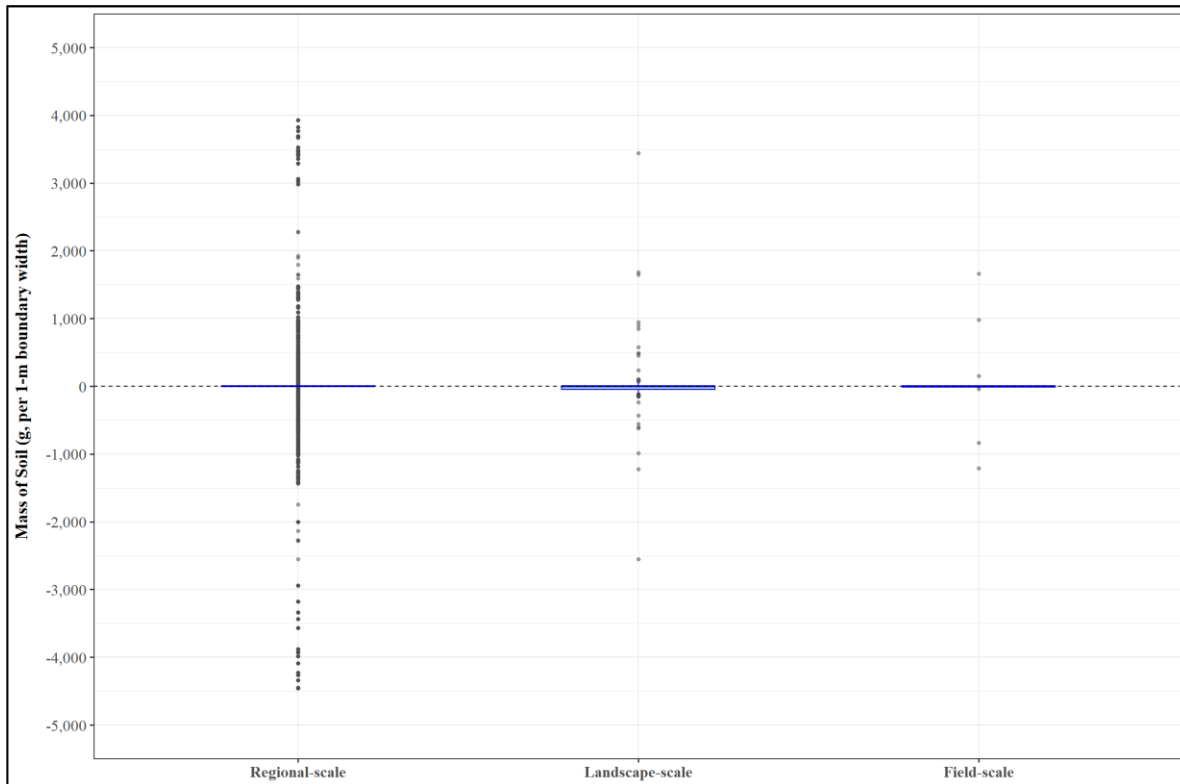


Figure 4.2. Model result of soil mass balance for the WCS regional, landscape, and field pathways. Positive values indicate net erosion, while negative values indicate net deposition.

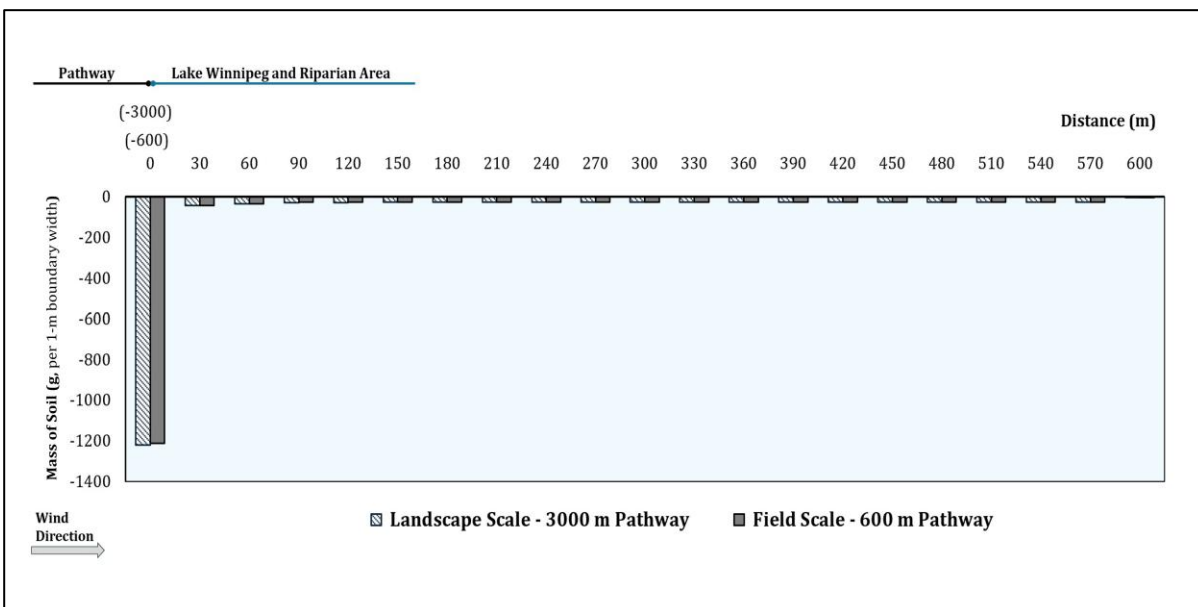


Figure 4.3. Model result of unit-level soil mass balance within the first 600 m of Lake Winnipeg, including the riparian area, for the WCS landscape- and field-scale pathways.

Figure 4.4 shows how the remaining mass in transport (sediment carried downwind) changes along the three WCS pathways. The curve rises where net erosion adds sediment to the transport load (erosion > deposition) and declines where deposition removes sediment from transport (deposition > erosion). The highest peak along each curve represents the maximum in-transit load for that pathway before deposition reduces the transported mass downwind. At the regional scale (Figure 4.4a), remaining mass in transport exhibits sharp pulses along the 1,100 km pathway, indicating intermittent additions from localized source units followed by rapid drawdown as sediment is redeposited downwind. The sequence of peaks shows episodic loading rather than sustained accumulation toward the lake boundary. At the landscape scale (Figure 4.4b), remaining mass in transport increases early where successive erodible units contribute sediment, then generally declines toward the lake boundary as entrainment weakens and deposition becomes dominant, particularly where the pathway intersects less erodible surfaces (e.g., grassland). A pronounced mid-pathway peak reflects a concentrated source segment, primarily composed of cropland units, where sediment input increases the transport load before it is reduced by downwind deposition. At the field scale (Figure 4.4b), remaining mass in transport stays near zero across much of the pathway and increases mainly within short segments near the beginning and near the downwind end, indicating that most detachment is confined to a limited portion of the field-scale pathway.

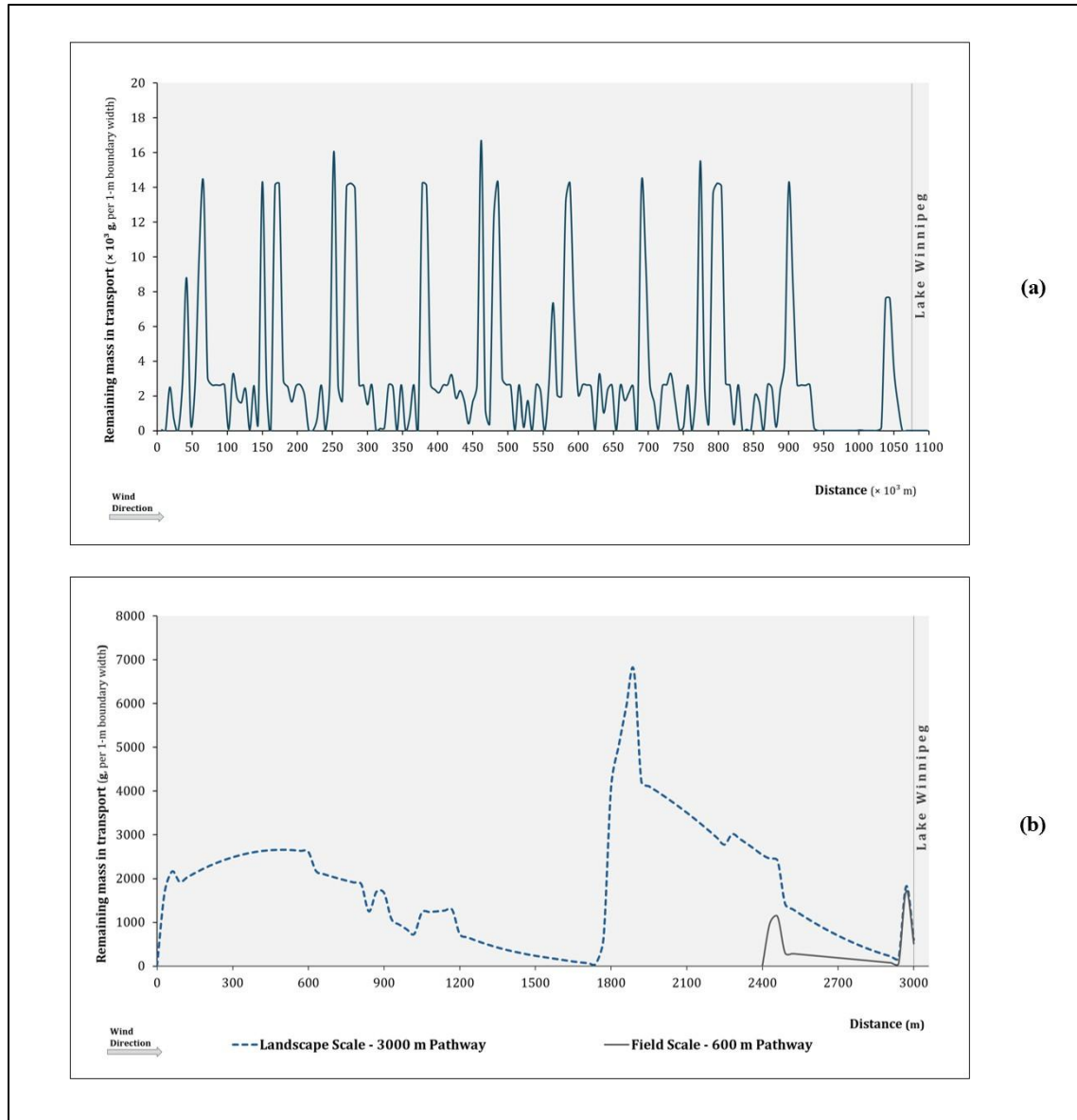


Figure 4.4. Model result of remaining mass in transport along the three WCS pathways: (a) regional-scale, (b) landscape-scale, and field-scale.

The model simulated sediment deposition at the end of the regional-, landscape-, and field-scale pathways, where the system terminates in the final sink area, Lake Winnipeg (Figure 4.5). The regional-scale pathway produced results identical to the landscape-scale pathway and, therefore, was not shown separately. Figure 4.5 shows that at the end of the landscape-scale pathway, at the onset of the final sink, the cumulative net mass reaches 1,827 g. Deposition begins immediately

within Lake Winnipeg, with the majority of the deposition occurring within the first 600 m of lakeward distance. Beyond 600 m, the mass declines gradually and approaches zero, indicating negligible delivery to more central lake areas.

Of this total, 1,170 g was delivered by creep and saltation processes, consisting primarily of very coarse sand (2-1 mm), coarse sand (1-0.5 mm), medium sand (0.5-0.25 mm), and fine sand (0.25-0.10 mm). A further 523 g was transported as suspended sediment, represented by very fine sand (0.10-0.05 mm) and silt (0.05-0.002 mm). Only 135 g consisted of particles smaller than 0.01 mm (PM₁₀), including clay (≤ 0.002 mm) and fine clay (≤ 0.0002 mm), which remained suspended in the air and exhibited the greatest potential for long-distance transport.

At the end of the field-scale pathway, delivered mass is similar, and deposition is likewise concentrated within the first 600 m before declining toward zero across the lakeward distance. At this scale, total deposition was slightly lower, at 1,724 g, with 1,170 g transported by creep and saltation, 540 g by suspension, and 14 g accounted for by PM₁₀.

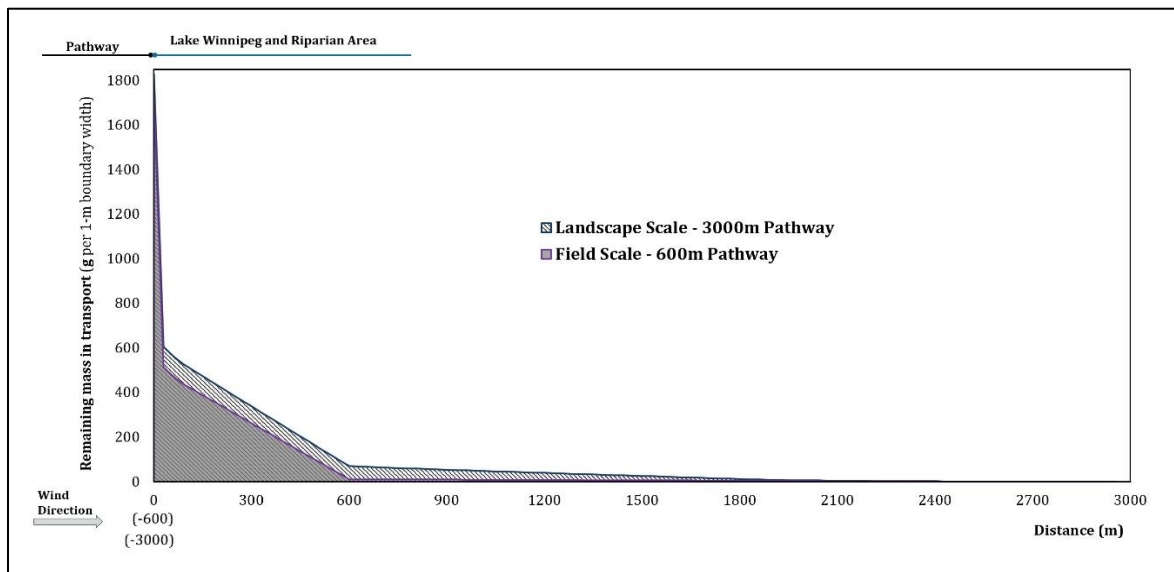


Figure 4.5. Model result of remaining mass in transport at the end of the WCS landscape- and field-scale pathways, extending 3,000 m into Lake Winnipeg.

The model output for particle-size redistribution of sediment within the first 3,000 m lakeward of Lake Winnipeg, is shown in Figure 4.6. Only the landscape-scale pathway is displayed, as the regional- and field-scale pathways yielded approximately the same deposition profile. Figure 4.6a includes the 5 m riparian zone and the immediate nearshore area of the lake; while Figure 4.6b covers the 30-3,000 m lakeward distance, representing deposition farther into the lake (note the different y-axis scale).

In Figure 4.6a, deposited mass declines steeply over the first tens of meters, with the greatest accumulation concentrated in the 0-5 m riparian zone. Sand (2-0.05 mm) shows the highest magnitude, starting near 750 g and diminishing rapidly to near zero by 30 m. Silt (0.05-0.002 mm) is the second most abundant fraction, an order of magnitude lower than sand but still visible across the 30 m span. Clay (0.002-0.0002 mm) and fine clay (≤ 0.0002 mm) remain negligible, at less than 2 g, throughout this nearshore distance.

In Figure 4.6b, the lakeward zone beyond 30 m exhibits low but detectable deposition, which decreases with distance from shore. Sand (2-0.05 mm) is about 1.83 g at the beginning but declines rapidly, reaching zero by 60 m. Silt (0.05-0.002 mm) is initially the most prominent fraction, at 27.8 g at 30 m, but drops sharply within the first 100 m and fully deposits by 300 m. Clay (0.002-0.0002 mm) begins at about 1.1 g at 30 m and declines gradually to 0.01 g. Fine clay (≤ 0.0002 mm) decreases more slowly, from 0.10 g at 30 m to 0.03 g at 3,000 m and persists throughout the full lakeward distance.

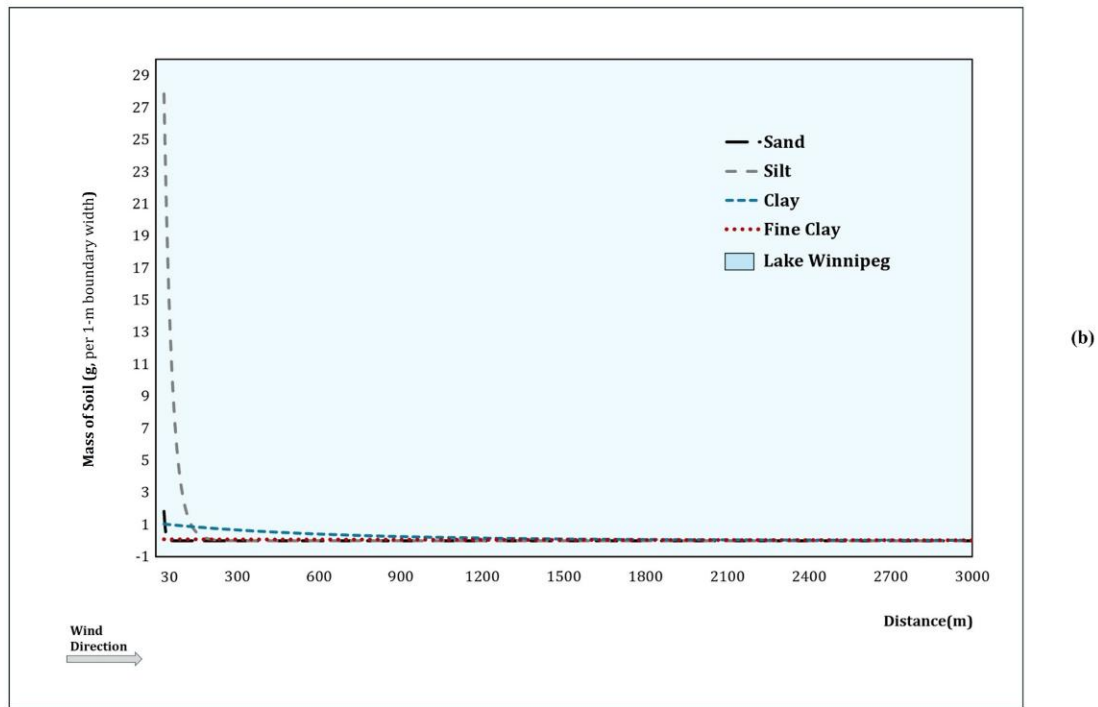
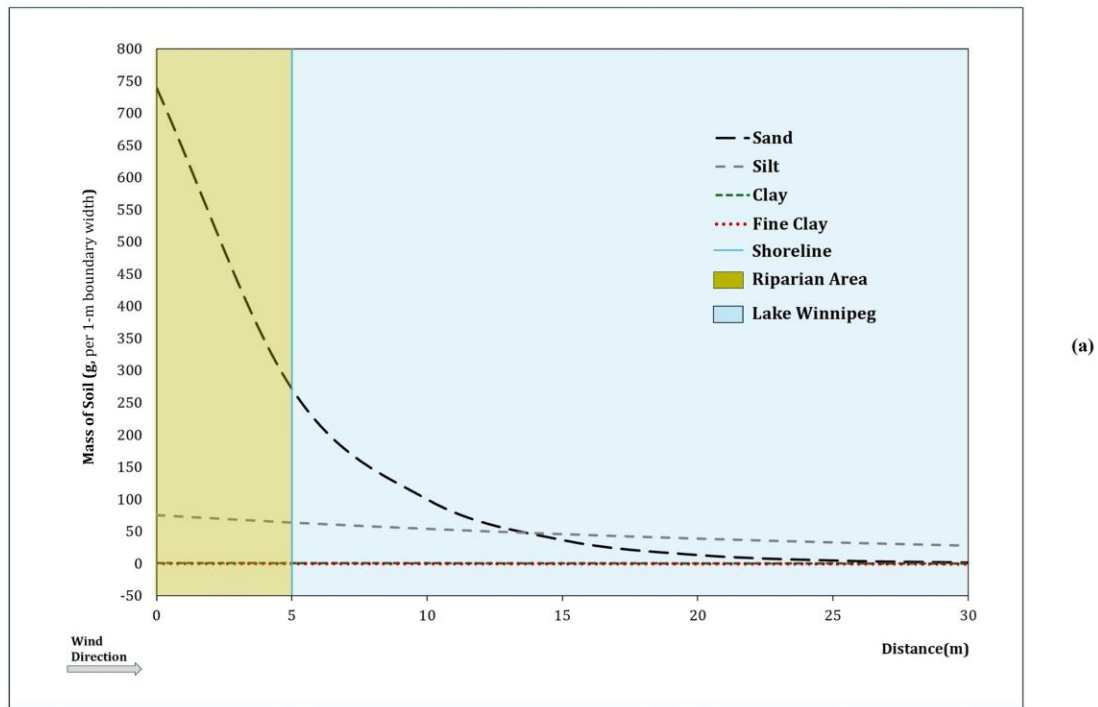


Figure 4.6. Model result of deposition by particle-size fractions (sand, silt, clay, fine clay) at the terminal sink (Lake Winnipeg) along the WCS landscape-scale pathway: (a) first 30 m, including the 0-5 m riparian area and shoreline; (b) lake interior from 30 to 3,000 m (note different y-axis scale).

4.2. Discussion

4.2.1. Addressing the Research Questions

In addressing the research questions outlined in Chapter 1, the results first illustrated the magnitude and spatial extent of wind-driven sediment transport across scales (RQ1), as shown in the unit-level balances and remaining mass in transport presented in Figures 4.1, 4.2, and 4.4. Patterns of deposition and particle-size redistribution within the final sink presented in Figures 4.3, 4.5, and 4.6 further clarified the magnitude and spatial pattern of deposition, namely the locations and amounts of sediment ultimately deposited within wetlands and Lake Winnipeg (RQ2).

Model results of unit-level soil mass balance along the three WCS pathways (Figures 4.1a-c and 4.2) were centred near zero at all scales, with variance that depended on scale. The frequent switches between net loss and net deposition arose from unit-to-unit variation in surface properties (i.e., soil texture, land cover and land use) and the mode-specific transport limits. Along the regional pathway, the series exhibited frequent fluctuations around zero, with most units lying within $\pm 1,000$ g. At the landscape scale, variability changed more gradually with distance, and the magnitude of deviations was smaller. Most values fell within ± 500 g, and across the 3,000 m pathway, many short-range exchanges offset one another, making extreme values less frequent and more localized. At the field scale, variability was smallest and was concentrated near the pathway ends. As pathway length decreased, the transect encountered a narrower range of surface heterogeneity, and short-range exchanges between adjacent units increasingly offset one another. Accordingly, the dispersion of unit-level mass-balance values contracted from the regional to the landscape and then to the field scale, and the mass-balance pattern shifted from high-frequency, unit-scale oscillations to low-amplitude runs centred near zero.

At the end of the landscape- and field-scale pathways, the mass balance (net deposition) exhibited a similar trend. Deposition was concentrated within the first 30 m of Lake Winnipeg, after which values declined gradually and approached zero by approximately 600 m (Figure 4.3). This pattern indicates that the initial 30 m, which includes the 5 m riparian area, functions as the primary depositional zone.

The remaining mass in transport (Figure 4.4) indicated that wind-driven sediment redistribution was spatially constrained, even under worst-case erosion scenarios. Across all scales, the transported load increased where erosion from erodible units exceeded deposition and declined where deposition removed sediment from transport downwind. Despite substantial differences in pathway length and the number of field units, all curves converged to similarly low end-point values at the sink boundary (final delivered mass: 1,827 g for the regional and landscape pathways and 1,724 g for the field pathway). When expressed as pathway-average equivalents by normalizing these delivered masses by the source pathway area, the resulting values were very small (0.00166 g m^{-2} for the regional pathway, 0.609 g m^{-2} for the landscape pathway, and 2.87 g m^{-2} for the field pathway). This indicates that most wind-eroded sediment was redeposited along the pathways, leaving only a small remaining load that ultimately reached Lake Winnipeg; therefore, long-distance delivery was minimal. These values help place wind-eroded sediment delivery in a broader sediment-budget context. The modeled areal deposition values were very small, indicating limited delivery of wind-eroded sediment to Lake Winnipeg even under worst-case erosion conditions. If the mean areal deposition predicted by the model were extrapolated across the south basin of Lake Winnipeg, the resulting approximate estimate of wind-derived sediment mass would still be low relative to sediment inputs from other processes, particularly sediment transported by rivers and streams.

The rise and subsequent decline in remaining mass in transport reflected the evolving balance between detachment and deposition. Differences in the position and magnitude of the maximum in-transit load resulted from pathway length, source availability, and the spatial arrangement of unit properties that control erodibility and deposition. Shorter pathways encountered source and depositional segments over shorter distances, whereas the regional pathway spanned a broader range of unit conditions and provided more opportunities for alternating erosion and deposition. This allowed intermittent load build-up over longer distances before the remaining mass declined through deposition.

The pattern displayed in Figure 4.5 further reinforced the spatial constraint of wind-eroded sediment transport and the highly localized nature of deposition at the terminal sink. Both the landscape- and field-scale pathways exhibited a steep decline in accumulated mass within the first 600 m of entry into Lake Winnipeg, after which the curves plateaued near zero. This consistent decline confirmed that most of the transported sediment was deposited within the near-shore zone, with only minimal mass reaching farther into the lake interior. In addition, the integrated totals indicated that cumulative deposition remained low and consistent across scales. Despite large differences in transect length, the totals varied only slightly. This indicated that sediment redistribution was primarily local, with sediment deposited in Lake Winnipeg originating almost entirely from units within approximately 3,000 m of the shoreline. Sediment eroded farther upwind was redeposited before reaching the lake and therefore did not contribute to terminal deposition.

As mentioned in Section 3.3.2, the maximum transport distances applied in this study were derived from particle-size-dependent transport relationships. Because each transport mode (creep/saltation, suspension, and PM_{10}) and particle-size group follows different governing equations, assigning mode-specific transport distances is consistent with established theoretical and empirical

descriptions of aeolian transport processes. In this framework, PM₁₀ particles, including fine silt, clay, and fine clay fractions smaller than 0.01 mm, were assigned a maximum transport distance of 3,000 m. While very fine particles may travel much farther through multi-event atmospheric transport, the objective of this study was to quantify sediment redistribution within local-, landscape-, and regional-scale pathways rather than global atmospheric dust circulation. Therefore, the maximum 3,000 m distance represents a conservative transport limit appropriate for both the spatial scale and the mass-balance framework applied in this study. To examine the basis of this assumption, the source pathway length was extended from 3,000 m to 6,000 m (Figure 3.B). Results indicated that increasing the upwind distance did not substantially change the cumulative mass delivered to the terminal sink, suggesting that most sediment contributing to terminal deposition originated within approximately 3,000 m of the shoreline. This indicates that the selected transport distances represent realistic contributing lengths controlling sediment delivery to the terminal sink, rather than arbitrary spatial constraints imposed on the model.

The concentration of deposition near the source was further reflected in the proportional breakdown of transport modes. At both regional and landscape scales, roughly two-thirds of the deposited mass (64%) was delivered by creep and saltation, about one-third (31%) by suspension, and only a minor fraction (7%) by PM₁₀. At the field scale, the pattern remained consistent, with 68% transported by creep and saltation, 31% by suspension, and only 1% as PM₁₀. The consistency in these proportions underscored the dominant role of creep and saltation processes, which consist primarily of sand particles ranging from very coarse (2-1 mm) to fine (0.25-0.10 mm), in controlling eroded sediment redistribution. The sand fraction rapidly exhausted its transport capacity and settled quickly, resulting in deposition close to the source and remaining largely unaffected by transect length.

By contrast, PM₁₀, including silt (0.05-0.002 mm), clay (0.002-0.0002 mm), and fine clay (≤ 0.0002 mm), traveled up to 3,000 m. Although PM₁₀ contributed little to the total mass, these fine particles extended the spatial reach of redistribution and played a key role in enabling long-distance transport during wind erosion events. This may be particularly important for contaminants such as nutrients and pesticides that can be concentrated in the colloidal (< 0.001 mm) fraction of mineral and organic particles (McDowell-Boyer et al., 1986).

Figure 4.6 clarified the particle-size redistribution and the role of the riparian zone in the final sink of landscape-scale pathway. In Figure 4.6a, deposition peaked sharply and was dominated by sand fractions (2-0.05 mm) transported by creep and saltation. These particles were efficiently trapped within the 5 m riparian zone and adjacent shoreline, while contributions from clay and fine clay remained negligible over this short distance. Beyond the shoreline, as shown in Figure 4.6b, only finer fractions persisted. Silt (0.05-0.002 mm) was initially the most prominent but declined rapidly, reaching zero by the first hundred meters. Clay (0.002-0.0002 mm) decreased more gradually, and fine clay (≤ 0.0002 mm) attenuated even more slowly; although both occurred in very small amounts, neither fraction reached zero across the first 3,000 m of the lake, indicating their capacity for long-distance transport.

4.2.2. Comparison of Model Predictions with Observed Field Data

To assess the performance of the MCWEM-WCS, its outputs were compared against field observations from the three studies introduced in Chapter 3: Baldwin (2015), Brooks (2022), and Zarrinabadi (2023). While differences in spatial scale, temporal resolution, and erosion processes may limit direct comparison, meaningful insights can still be gained by assessing whether the modeled magnitude of wind-induced sediment loss, spatial redistribution patterns, transport

distances, and depositional behavior are consistent with the general trends observed across landscapes in the PPR.

Baldwin (2015) conducted a ^{137}Cs -based assessment of wind erosion in two intensively cultivated potato fields in southern Alberta. The study aimed to evaluate actual erosion under Prairie conditions and compare measured redistribution patterns with predictions from two national models, WindERI and SWEEP. Although Baldwin incorporated simulation modeling, the ^{137}Cs field inventories provide the most direct empirical benchmark for evaluating the assumptions and outputs of the MCWEM-WCS.

The MCWEM-WCS predicts that wind-eroded sediment is primarily redeposited within short distances of its source, resulting in minimal net soil loss and very low delivery to terminal sinks. Baldwin's field measurements support this outcome. As illustrated in Figure 4.7, median ^{137}Cs inventories in cultivated fields were only marginally lower than those in nearby undisturbed reference sites: 1,036 vs. 1,073 Bq m^{-2} in Field 1 (3.4% reduction) and 940 vs. 1,073 Bq m^{-2} in Field 2 (12.4% reduction). These differences were statistically insignificant at $\alpha = 0.10$ ($P = 0.992$ and $P = 0.370$), respectively. This indicates that, despite the highly erosive conditions associated with potato production on these soils and limited residue cover, net wind-driven soil export from the fields was negligible.

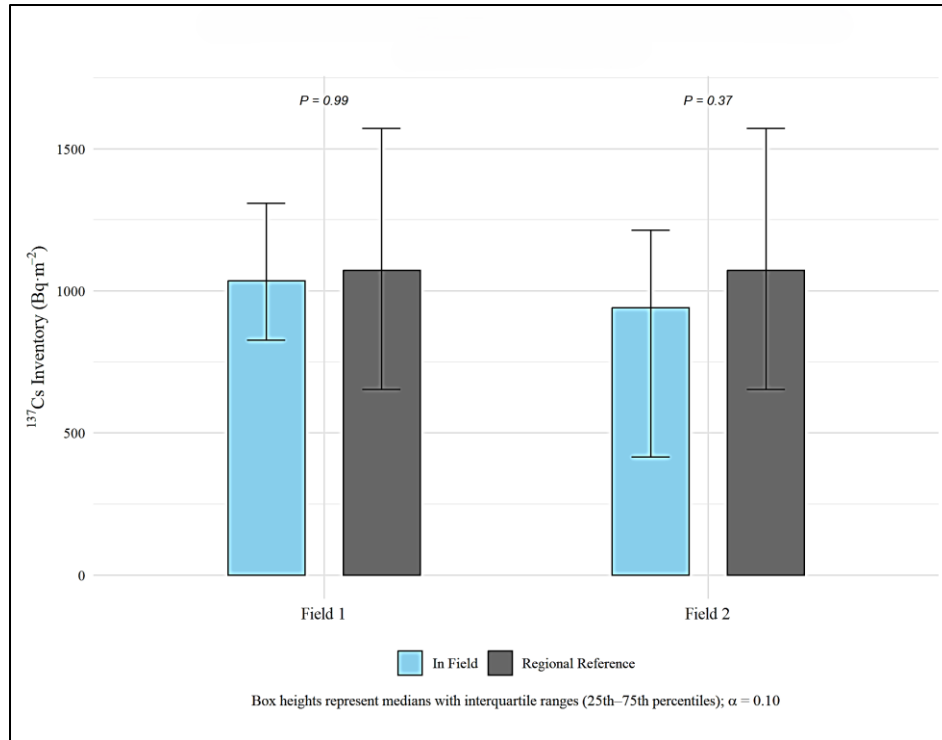


Figure 4.7. Comparison of ^{137}Cs inventories between cultivated in-field areas and regional reference sites for Field 1 and Field 2. No statistically significant differences were found, supporting the model's assumption of minimal net sediment export by wind erosion. Adapted from Baldwin (2015).

A comparison of ^{137}Cs inventories between in-field corner areas and their adjacent grassed corners, as shown in Figure 4.8, further reinforces the model's predictions. If wind erosion were a dominant erosive force, lower ^{137}Cs values would be expected for exposed in-field corners, with higher values accumulating in grassed margins due to their role as vegetative traps. However, across all four quadrant comparisons, no statistically significant differences were observed, with all P values > 0.1 . In two cases, Field 2 SW and NE, in-field corners exhibited higher median inventories than their adjacent grassed counterparts. This lack of accumulation in grassed margins suggests that wind erosion did not produce strong net delivery to field edges, but rather redistributed sediment within or near the source areas.

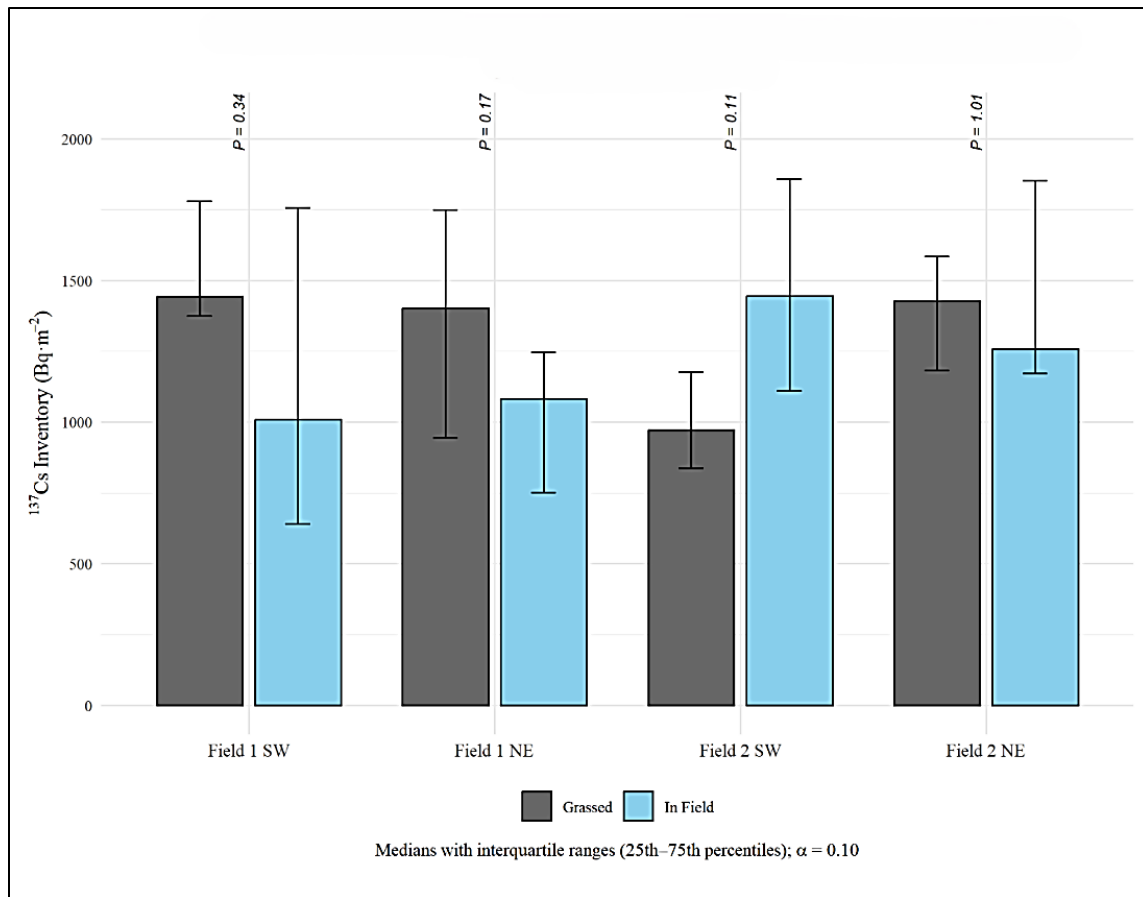


Figure 4.8. Comparison of ^{137}Cs inventories between in-field corner areas and their adjacent grassed corner areas in Field 1 and Field 2. P-values indicate no statistically significant difference. Adapted from Baldwin (2015).

Baldwin also investigated directional patterns in surface inventories under prevailing southwest winds, expecting northeast leeward corners to show preferential accumulation of fine, wind-eroded particles. However, as shown in Table 4.1, no statistically significant differences were found between windward and leeward corners in either field ($P = 0.476$ in Field 1; $P = 0.171$ in Field 2). In Field 1, the windward corner even had a higher median value than the leeward corner. These findings suggest that wind was not the primary driver of sediment redistribution and contributes minimally to sediment transfer across the Prairie landscape.

Table 4.1. Comparison of surface ^{137}Cs inventories between windward and leeward corners in Field 1 and Field 2; $\alpha = 0.10$. Reproduced from Baldwin (2015)

<i>Group</i>	<i>Position</i>	<i>Number of samples</i>	<i>Median (Bq m⁻²)</i>	<i>25% (Bq m⁻²)</i>	<i>75% (Bq m⁻²)</i>	<i>U</i>	<i>T</i>	<i>P Value</i>
Field 1	NE Grassed Corner	6	276	205	749	8	26	0.476
	SW Grassed Corner	4	425	363	510			
Field 2	NE Grassed Corner	6	229	214	241	5	29	0.171
	SW Grassed Corner	4	261	232	354			

As ^{137}Cs strongly adsorbs to fine colloidal particles, particularly clay and fine clay fractions (≤ 0.002 mm), its distribution in the upper soil layers can serve as an effective tracer for fine suspended sediment movement. In wind erosion processes, these fine particles are preferentially detached and transported, making surface ^{137}Cs concentrations a useful indicator of fine-particle redistribution when interpreted alongside soil properties and management history. Baldwin's (2015) results indicated that while total ^{137}Cs inventories did not differ significantly between cultivated and reference areas, surface activity levels were notably lower in the cultivated fields, with values of 233 and 187 Bq m⁻² in Fields 1 and 2, respectively compared with 772 Bq m⁻² in the reference areas ($P < 0.001$ in both cases; Figure 4.9).

Baldwin attributed this contrast primarily to tillage-induced vertical mixing in the ploughed fields rather than to surface-level removal by wind. In intensively managed lands, such as potato fields, practices including digging and hilling can substantially redistribute soil both vertically and laterally, thereby altering ^{137}Cs depth profiles. In both fields, surface ^{137}Cs concentrations in cultivated areas showed limited spatial variability and lower values, likely reflecting vertical redistribution by tillage. In contrast, reference transects displayed elevated and more variable surface ^{137}Cs concentrations, which may indicate the accumulation of fine, ^{137}Cs -enriched particles

in relatively undisturbed surface soils. These patterns are consistent with the model's prediction that longer-distance wind-driven redistribution is mainly associated with finer suspended particles, including PM₁₀, although Baldwin's results suggest that tillage effects were more important than wind-driven export in explaining the observed surface ¹³⁷Cs patterns.

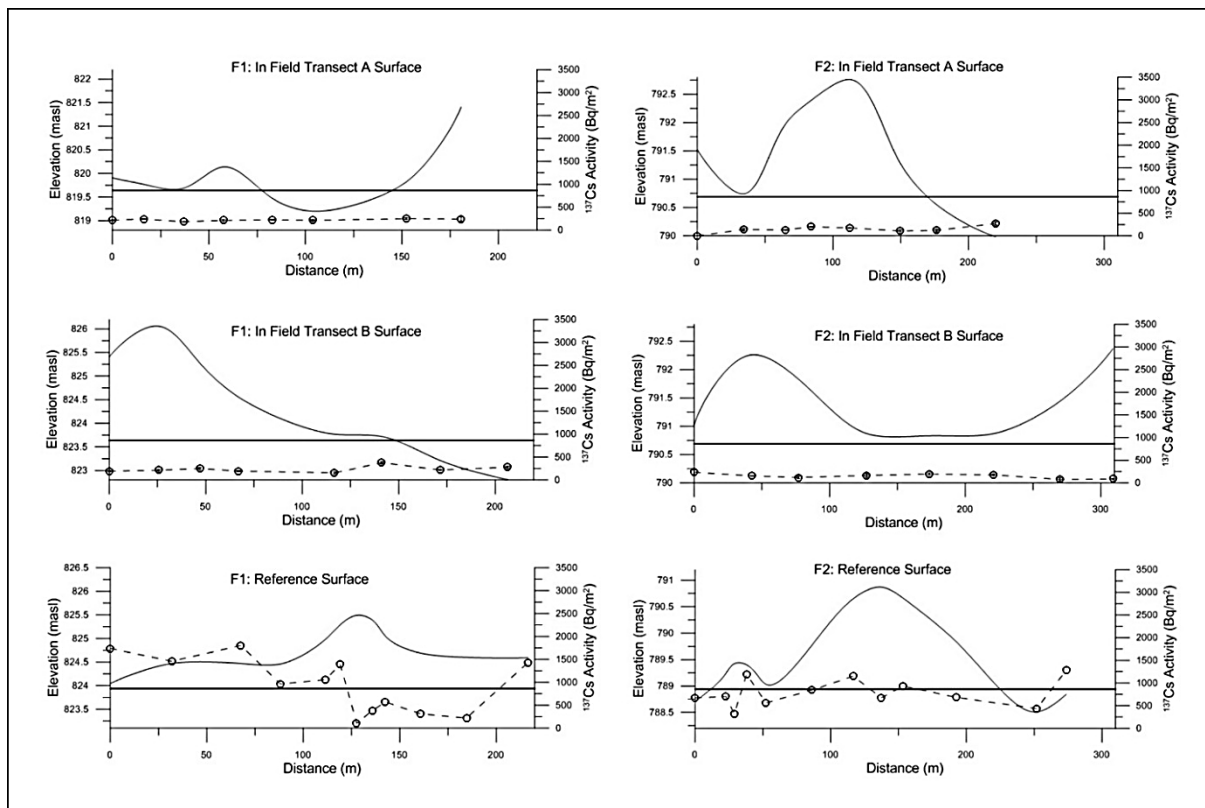


Figure 4.9. Surface ¹³⁷Cs activity and elevation profiles from cultivated in-field and undisturbed reference transects in Field 1 and Field 2. Reproduced from Baldwin (2015).

In addition, Baldwin's particle size analysis, performed by laser diffraction on surface soil collected from the in-field cultivated and grassed corner transects of Fields 1 and 2, showed a rapid decline in mean particle size from approximately 0.30 to 0.16 mm within the first 0-5 m of the grassed corner transects (e.g., F1-SW and F2-NE). This pattern indicates that the coarser fraction was deposited almost immediately, consistent with the modelled behaviour of creep and saltation.

Beyond 5 m, mean diameters continued to decline gradually, reflecting the progressive settling of finer fractions.

In summary, Baldwin's field study found limited inventory differences, no clear evidence of directional sediment trapping, surface activity patterns influenced primarily by tillage mixing, rapid coarse fraction deposition with progressive surface fining, and low-magnitude redistribution dominated by fine particle movement. Together, these findings are broadly consistent with the model results, which indicate localized sediment redistribution, rapid deposition of coarser particles, and limited long-distance delivery across Prairie landscapes.

Brooks (2022) developed field-based sediment budgets for sub-watersheds 05OF024 and 05OG008 in southern Manitoba, quantifying production, deposition, and delivery across cultivated fields, in-field drains, roadside ditches, and streams. Using morphometric measurements of depositional features attributed to wind, tillage, and water (e.g., sediment fans, tillage throw, dredged spoil piles, gravel sheets, bioturbated mounds), Brooks (2022) estimated average annual and total transferred masses of eroded sediment from fields into surrounding waterways. Annualized transfers indicated that most sediment deposited in roadside ditches of fine-textured catchments originated from non-aeolian processes. The features collectively accounted for approximately 181,470 Mg yr⁻¹ (1.9 Mg ha⁻¹ yr⁻¹) in 05OF024 and 71,284 Mg yr⁻¹ (3.8 Mg ha⁻¹ yr⁻¹) in 05OG008. By contrast, wind-derived contributions are small: snirt comprises 2,703 Mg yr⁻¹ (1.5%) and 1101 Mg yr⁻¹ (1.5%), while blow dirt contributes only 933 Mg yr⁻¹ (0.5%) and 380 Mg yr⁻¹ (0.5%) in 05OF024 and 05OG008, respectively (Figure 4.10). This distribution is consistent with the MCWEM-WCS's assumption that wind erosion contributes a relatively minor share to catchment-scale sediment budgets.

Although the MCWEM-WCS in this study focused on May-June conditions, while Brooks's dataset also included snirt accumulation from late winter/early spring, the small fractions represented by both blow dirt and snirt reinforce the model's prediction that wind-eroded sediment rarely makes a substantial contribution at the catchment scale. Brooks (2022) also observed that blow-dirt deposits remained within roadside ditches and adjacent lands, with no evidence of downstream delivery to aquatic systems, which aligns with the model's short-distance redistribution pattern and near-source deposition outcomes.

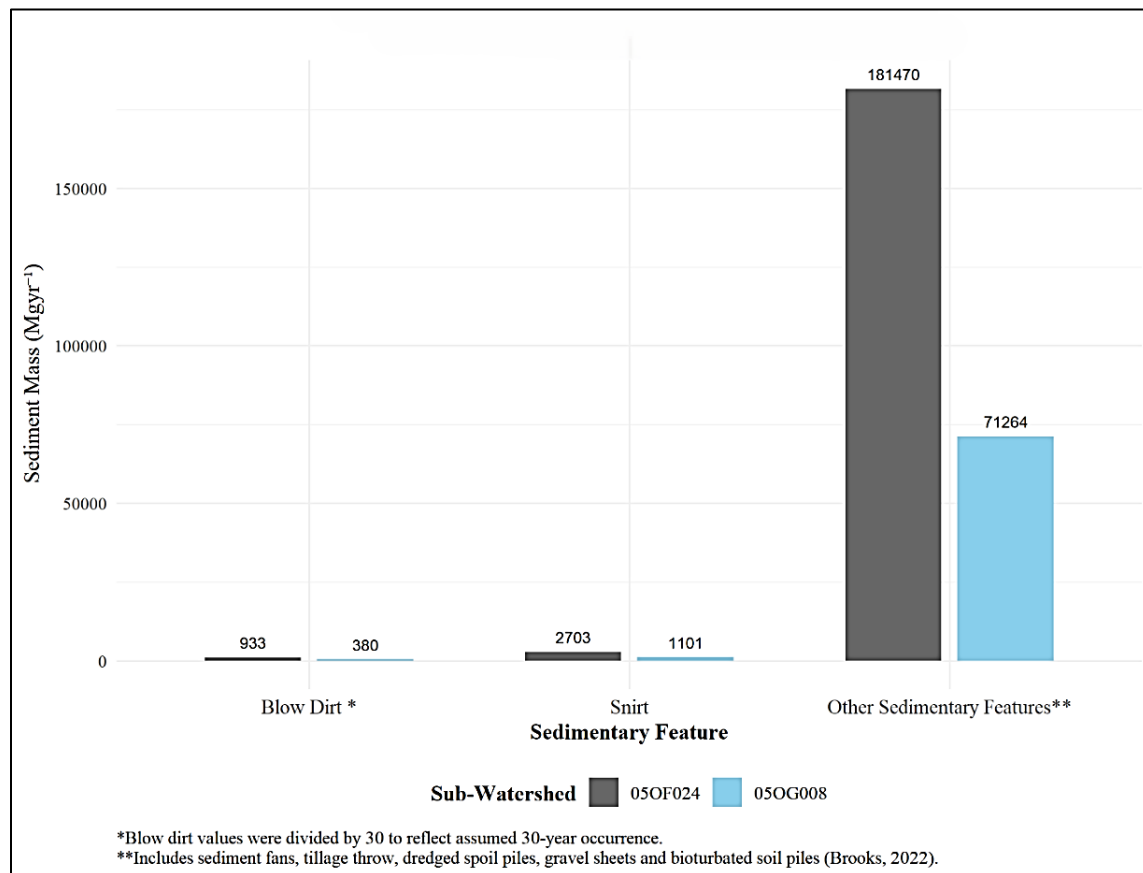


Figure 4.10. Annualized sediment transfers to roadside ditches in fine-textured catchments of sub-watersheds 05OF024 and 05OG008, showing that wind-driven features (snirt and blow dirt) contribute a small fraction compared to other sediment sources. Blow dirt values were adjusted for a 30-year return interval. Adapted from Brooks (2022).

To quantify sediment movement across the landscape, Brooks (2022) estimated sediment budgets for each sub-watershed using a mass-balance approach. Sediment transfers were grouped by

erosion process, including wind, water, and tillage, and classified by direction, including field to ditch, field to stream, and ditch to field. Within this structure, snirt and blow dirt were grouped as wind-driven features, with values reflecting sediment mass transferred from eroding fields toward drainage pathways and streams. The results showed that in both watersheds, sediment inputs from fields to ditches far exceeded outputs toward streams. In 05OF024, wind-driven sediment input from fields to ditches was approximately $69,910 \text{ Mg yr}^{-1}$, while only $6,975 \text{ Mg yr}^{-1}$, or 7%, was exported from ditches to the stream. In 05OG008, input from fields was $4,952 \text{ Mg yr}^{-1}$, with an output to the stream of $1,134 \text{ Mg yr}^{-1}$, or 16%. The significant decline in sediment mass from field edge to stream outlet further supports the model's prediction of short-distance redistribution and negligible terminal delivery.

Figure 4.11 partitions the annual sediment budget of sub-watershed 05OF024 by soil texture and shows strong texture dependence in wind-driven transfers (snirt and blow-dirt). Field to ditch inputs increased as particle size decreased, $5,376 \text{ Mg yr}^{-1}$ (coarse), $30,706 \text{ Mg yr}^{-1}$ (medium), and $33,828 \text{ Mg yr}^{-1}$ (fine), whereas ditch to stream exports remained small across textures: 235 Mg yr^{-1} (4.4%), $1,932 \text{ Mg yr}^{-1}$ (6.3%), and $2,785 \text{ Mg yr}^{-1}$ (8.2%), respectively. These results indicate that most sediment was trapped locally rather than transported to streams, and that fine particles account for the highest proportion of sediment mobilized by wind. This pattern aligns with the MCWEM-WCS, which predicted low overall delivery to terminal sinks, rapid deposition of coarser fractions over short distances, and a proportionally greater role of fine particles in longer-distance wind transport.

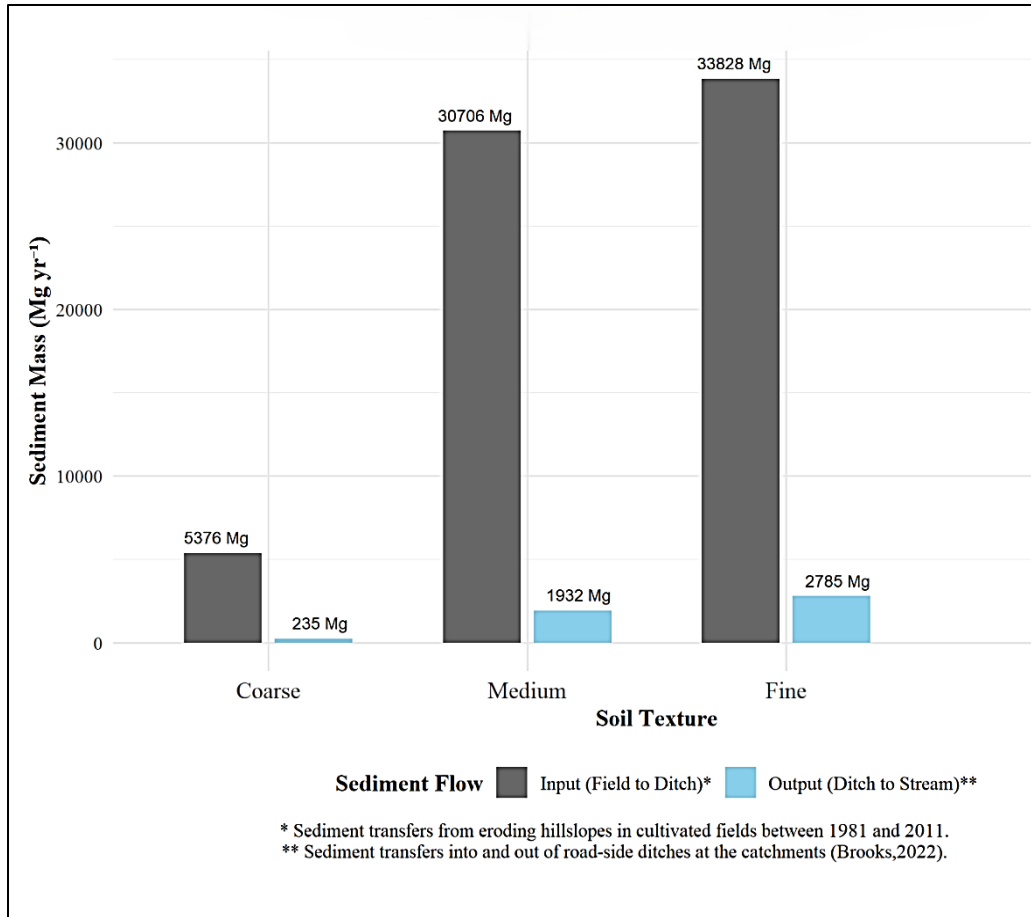


Figure 4.11. Annual wind-driven (snirt + blow-dirt) sediment transfers by soil texture in sub-watershed 05OF024. Adapted from Brooks (2022). Delivery remains low across textures, indicating proximal trapping within roadside ditches.

Field observations reported by Zarrinabadi (2023) align closely with the assumptions of the MCWEM-WCS, particularly regarding transport distances and depositional patterns. Zarrinabadi's surveys conducted during the vulnerable post-harvest spring period in roadside ditches and topographic depressions of the Red River Valley, focused on landscape features often assumed to trap wind-induced sediment, showed no measurable accumulation, even adjacent to soybean fields established on flat, sandy-loam soils with minimal residue cover. Inspections carried out after multiple erosive events and during early land preparation similarly revealed no deposition along ditches or vegetated margins. These observations are consistent with the model applied in this

study, which indicated that delivery to terminal sinks such as lakes or wetlands remained very low even under highly susceptible conditions.

Further insight into the spatial pattern of soil redistribution comes from Zarrinabadi's ^{137}Cs -based surveys across wetland catchments in cultivated and native landscapes of the PPR. Although ^{137}Cs inventories integrate the effects of wind, water, and tillage erosion, their spatial distribution provides valuable context for evaluating the MCWEM-WCS. As shown in Figure 4.12, in cultivated catchments, inventories showed clear downslope enrichment, with upper slope positions exhibiting depletion, indicating source areas, and lower slopes and riparian areas showing enrichment, indicating depositional areas. These trends closely match the depositional patterns predicted by the model at all scales. The absence of ^{137}Cs enrichment within the central parts of wetlands further supports the model's prediction of short-distance transport and very low delivery to terminal sinks. In contrast, native prairie catchments did not display the same downslope depositional signals; ^{137}Cs depletion occurred even at lower slope positions, suggesting minimal sediment redistribution under undisturbed vegetation. Zarrinabadi attributed this to the stabilizing influence of perennial cover, surface micro-relief, and site-specific sampling, which likely limited detachment and transport and produced relatively uniform or slightly depleted inventories.

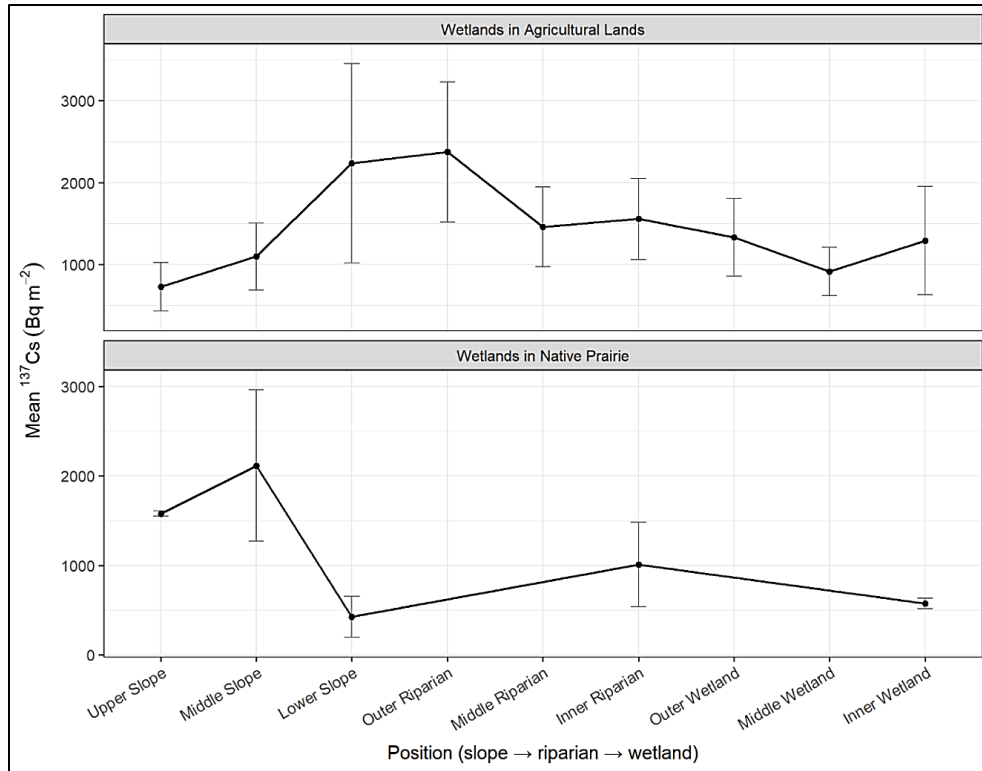


Figure 4.12. Mean ^{137}Cs inventory along the toposequence for wetlands surrounded by croplands and native prairie. Adapted from Zarrinabadi (2023)

Sediment budgets derived from the ^{137}Cs transect dataset provide additional support for the model's assumptions. Gross erosion, net soil loss, and sediment delivery were estimated across seven cultivated catchments in Manitoba and Alberta (Figure 4.13). Although these values reflect the combined influence of wind, water, and tillage, they remain a strong empirical benchmark for evaluating model predictions. Net erosion was consistently lower than gross erosion across all catchments, highlighting substantial intra-catchment redistribution. Mean net soil loss from cultivated fields was $0.6 \text{ kg m}^{-2} \text{ yr}^{-1}$ ($6 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) in Manitoba and $0.1 \text{ kg m}^{-2} \text{ yr}^{-1}$ ($1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) in Alberta, compared with gross erosion rates of 1.1 ($11 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) and $0.3 \text{ kg m}^{-2} \text{ yr}^{-1}$ ($3 \text{ Mg ha}^{-1} \text{ yr}^{-1}$), respectively. These differences yielded delivery ratios of 57% in Manitoba and 35% in Alberta, suggesting that much of the eroded sediment was redeposited within the catchments, particularly in adjacent riparian zones, rather than delivered to wetland interiors.

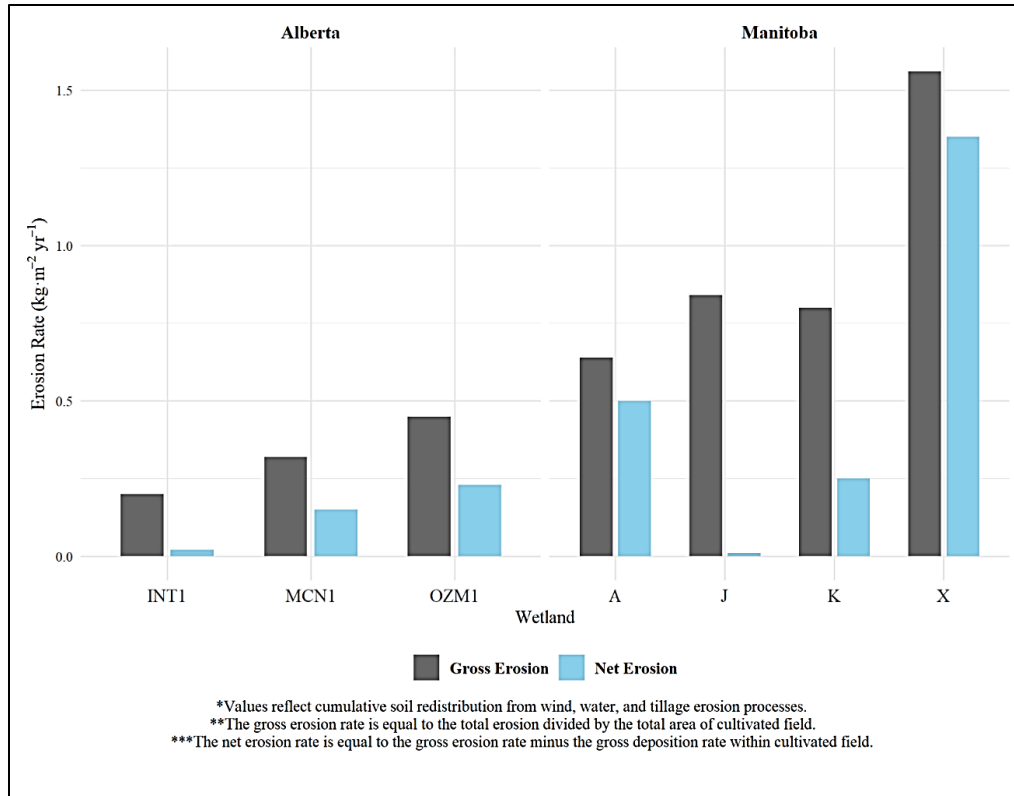


Figure 4.13. Sediment budgets based on ^{137}Cs transect data for the studied wetland catchments in Alberta and Manitoba. Gross erosion is the total mass exported per unit cultivated area; net erosion equals gross erosion minus gross deposition within cultivated fields. Values represent cumulative redistribution by wind, water, and tillage processes. Adapted from Zarrinabadi (2023).

Profiles of redistribution patterns, derived from point-scale soil loss and deposition rates, were constructed using (a) ^{137}Cs reference values estimated from individual local stable sites and (b) a catchment-level ^{137}Cs reference value (Figure 4.14). Both approaches revealed similar spatial trends: upper and middle slope positions consistently acted as erosion hotspots, while deposition was concentrated on lower slopes and in outer riparian areas. Inner riparian and wetland zones showed minimal gains or slight net losses. These findings align with the model's prediction that wind-driven sediment is largely confined within the field-to-riparian gradient and rarely transported further downslope.

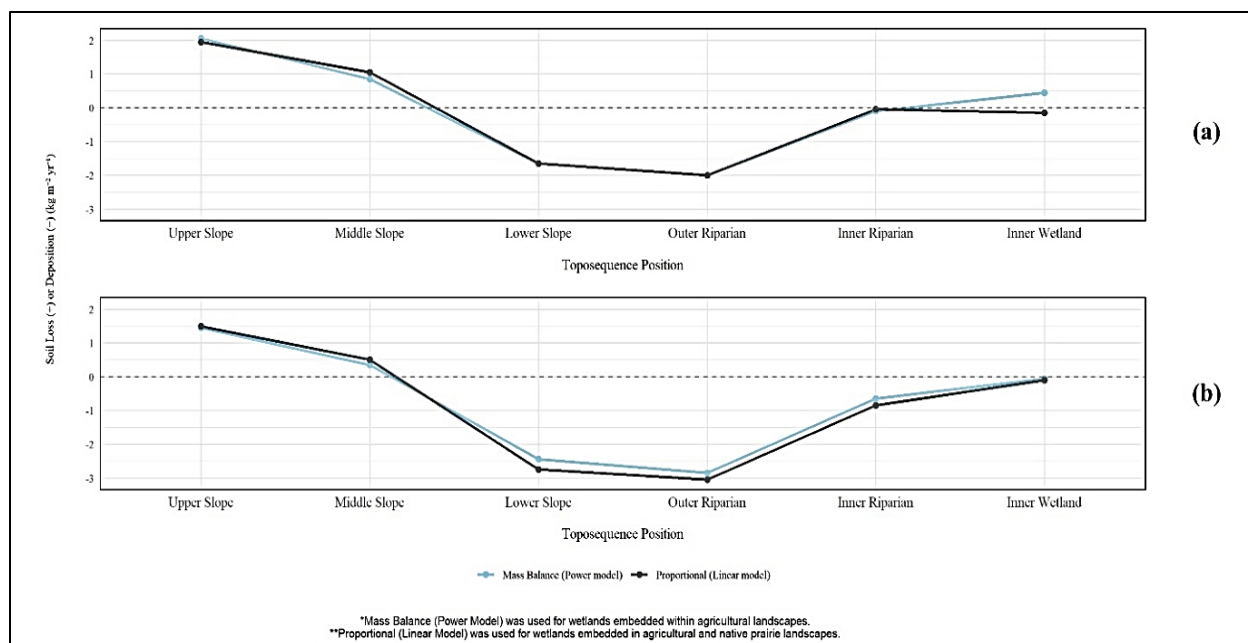


Figure 4.14. Mean soil loss or deposition ($\text{kg m}^{-2} \text{yr}^{-1}$) along the toposequence for wetland catchments in Manitoba and Alberta. (a) uses ^{137}Cs reference inventories from individual local stable sites; (b) uses a catchment-level ^{137}Cs reference inventory. Adapted from Zarrinabadi (2023).

Analysis of particle-size distributions across these seven wetland catchments provides additional context for evaluating the MCWEM-WCS. Results indicate that finer fractions, particularly silt and clay, are transported farther downslope than coarser particles such as sand, reaching central wetland areas during runoff and wind-erosion events. Furthermore, the particle-size composition of samples collected along toposequence transects demonstrated that silt was the dominant fraction within inner wetland positions. These findings are consistent with the MCWEM-WCS, which predicts rapid deposition of coarse particles near the source and extended redistribution of fine particles into central areas of wetland.

Zarrinabadi (2023) field surveys and ^{137}Cs -based analyses support that wind-driven sediment transport is highly localized, with most sediment redeposited near its source and only very low delivery to terminal sinks such as wetland interiors. Downslope ^{137}Cs enrichment in cultivated catchments corresponds with the model's prediction of maximum accumulation in riparian zones,

while the absence of enrichment in native prairie catchments reflects the stabilizing influence of undisturbed vegetation and surface micro-relief. Particle-size data further reinforce these outcomes by demonstrating rapid deposition of coarse fractions and extended redistribution of silt into wetland interiors. Sediment budget results also support the model by showing low net erosion rates, substantial local trapping, and minimal long-distance export.

Collectively, the findings of Baldwin (2015), Brooks (2022), and Zarrinabadi (2023) reinforce the model's predictions that wind erosion in prairie landscapes is largely a local process: coarse particles settle rapidly within a few meters of depositional areas (e.g., field edges and riparian margins), while finer fractions can travel farther but contribute only minimally to overall deposition. These lines of evidence support the conclusion that most sediment is retained near its source, with little delivered to terminal sinks.

For the model performance assessment, this study compared model outputs with three recent field-based studies (Baldwin, 2015; Brooks, 2022; Zarrinabadi, 2023) because of their direct relevance to wind-induced sediment redistribution in agricultural landscapes within the PPR. Although broader aeolian research may demonstrate different patterns under some environmental conditions, studies that explicitly quantify wind-driven sediment transport across regional scales and sediment delivery to wetland interiors in the PPR remain limited. This limits the extent to which the model outputs can be compared with field-based evidence across a broader range of conditions. Therefore, the results of this study should be interpreted within the context of the model framework, particularly the soil texture, land use, land cover, and transport-distance assumptions examined here. Further research, as discussed in Chapter 6, should incorporate a wider range of environmental conditions, variable model parameters, and field-based measurements to better investigate wind erosion processes and sediment delivery at local, landscape, and regional scales.

4.2.3. Implications of the Study

This study demonstrates that wind-driven sediment transport is constrained in both magnitude and distance. Even under worst-case conditions, most mobilized sediment is deposited close to its source. The first 5 m of a land sink acts as the primary depositional area, intercepting most of the transported sediment before it can travel farther. The cumulative mass-balance pattern, characterized by an initial rise to a peak followed by a downwind decline, highlights where detachment is greatest and where interception is most effective.

These outcomes suggest that much of the sediment accumulating in depositional areas is of local origin, challenging the hypothesis that wind erosion drives significant regional redistribution across the Prairie. Concerns that wetlands may be progressively filled with wind-eroded sediment are tempered by the finding that most sediment is trapped in riparian margins before reaching open water. Coarse and medium fractions settle rapidly near shore, while finer particles can travel farther but contribute little to the overall budget. The ecological risk of long-distance delivery therefore appears limited, although small inputs of fine particles may still be relevant in sensitive receiving environments, such as shallow wetlands, poorly drained depressions, or areas where fine particles carry nutrients or contaminants.

Because wind velocity was held constant in the simulations, differences in soil loss and gain during wind erosion events reflect variation in land cover, land use, and soil texture, factors that determine surface erodibility and deposition potential. This highlights the importance of surface condition in shaping erosion outcomes and points to two central strategies: reducing detachment where surfaces are most vulnerable and maintaining interception capacity where deposition naturally occurs. Croplands with little residue cover act as likely source areas, while water bodies, grasslands, forests, ditches, and riparian strips consistently function as depositional areas. Management actions

that maintain residue, limit soil disturbance, or increase surface roughness are therefore critical for wind erosion control. In addition, installing barriers to shorten the effective fetch length, such as shelterbelts, fences, vegetated strips, or grassed field margins, can further reduce the energy available for sediment detachment and transport, thereby suppressing cumulative erosion.

CHAPTER 5: Conclusion

This study evaluated wind's contribution to sediment transport at local and regional scales across the Canadian Prairie Pothole Region using a Mass-Conservation Wind Erosion Model that simulates sediment production, loss, transport, and deposition under worst-case erosion scenarios. By applying the model at field (0.6 km), landscape (3 km), and regional (1,100 km) scales, the study assessed the spatial extent of wind-driven redistribution and the magnitude of eroded sediment ultimately delivered to Lake Winnipeg.

Despite modelling conditions of maximum erosion susceptibility, the results show that net transported mass was low at all scales because most sediment was redeposited close to its source; consequently, delivery to terminal sinks was very low. Cumulative mass curves at each scale increased rapidly and then declined toward zero. Within the terminal sink, Lake Winnipeg, deposition peaked within the first few meters at the land–water interface, including the riparian margin and immediate shoreline, and declined sharply lakeward. Coarse particles, ranging from 2 to 0.1 mm and transported by creep and saltation, were largely deposited very close to their point of origin and did not reach central areas of water bodies acting as terminal sinks.

Fine particles ≤ 0.01 mm travelled the full 3,000 m across the final sink, but their contribution to cumulative deposition remained small, accounting for approximately 7% along the landscape-scale pathway and 1% along the field-scale pathway. Within the far-travelling fraction, particles in the PM_{2.5} range, approximately 0.0025 mm, persisted beyond 3,000 m; however, they accounted for only 4% and 0.5% of the total deposited mass along the landscape- and field-scale pathways,

respectively. These values do not support substantial regional redistribution across the PPR or major wind-eroded sediment delivery to water bodies, including prairie wetlands.

Recent field studies support these outcomes. Observations from Baldwin (2015), Brooks (2022), and Zarrinabadi (2023) consistently report strong near-source trapping, minimal net loss from agricultural fields, and limited sediment accumulation within sinks such as ditches, wetlands, and streams. In conclusion, wind-driven sediment redistribution in the Canadian Prairie Pothole Region is spatially limited and governed primarily by local-scale processes rather than regional-scale transport. Most eroded sediment is redeposited within tens to hundreds of meters of its origin, with very low delivery of fine particles to terminal sinks, even under extreme wind conditions.

CHAPTER 6: Research Synergy and Contribution

6.1. How This Study Supports the Goals of the Research Project

This study contributes to the Wetlands as Nature-based Solutions (NbS) research program, which aims to evaluate the role of wetlands as carbon and nutrient sinks in Canadian agricultural landscapes. The program supports national efforts to protect and enhance carbon sinks, including freshwater mineral wetlands in the Prairie region. Within this broader context, the study aligns with the goal of developing estimates of organic carbon accumulation and lateral carbon transport into wetlands.

Most previous research on wetland carbon inputs has focused on hydrological pathways, leaving wind-driven processes relatively unexamined. This study addresses that gap by developing the MCWEM, to simulate sediment production, transport, and deposition across three spatial scales under worst-case erosion scenarios. The model incorporates mode-specific and particle-size specific transport thresholds to estimate the flux of carbon-bearing sediment from upwind agricultural fields into potential depositional zones, including Prairie wetlands.

Results demonstrate that even under highly erosive conditions, most wind-eroded sediment is redeposited close to its source, with net delivery to wetland interiors remaining very low. Coarse and medium particles settle rapidly near field edges and riparian zones, while finer fractions (PM₁₀) travel farther but account for only a small proportion of total deposited mass.

These findings refine our understanding of lateral carbon inputs to wetlands. Although Prairie wetlands may serve as localized sinks for wind-driven sediment, most deposition occurs near the

land–water interface. Riparian margins and wetland edges function as primary interception zones, where sediment and associated nutrients are deposited and stabilized. These depositional zones often enhance organic matter accumulation and carbon storage through sediment burial and vegetation productivity under moist conditions (Bridgham et al., 2006; Tangen & Bansal, 2020). By quantifying the wind-driven pathway in sediment and carbon cycling, this study enhances the spatial resolution of carbon flux assessments across Prairie landscapes.

The model outputs can inform scenario-based analyses, wetland monitoring strategies, and modelling tools used in national GHG inventory frameworks and emerging carbon calculators. In this way, the MCWEM helps reduce uncertainty surrounding sediment-bound carbon inputs and contributes to more accurate ecosystem-service evaluations.

In summary, while wind erosion is not a dominant mechanism for carbon delivery to the central interior of wetlands, its quantifiable contribution, particularly at wetland margins and connected depositional zones, indicates that wind-driven sediment transport should be considered alongside other erosion pathways when evaluating wetland carbon budgets. This work advances the broader aims of the Wetlands NbS program by distinguishing among transport mechanisms, identifying carbon retention zones, and informing conservation and restoration efforts that support national climate objectives.

6.2. Recommendations for Future Research

This study presents a framework for estimating wind-driven sediment transport under worst-case conditions across Prairie landscapes. Wind erosion is governed by highly dynamic and interacting factors, including wind velocity and turbulence, surface roughness, soil moisture, and topographic variability, all of which fluctuate over time and space. Capturing this full complexity requires

extensive data and computational resources, which can obscure fundamental patterns and reduce model transparency. Therefore, the model was intentionally structured in a simplified form to facilitate sediment mass-balance tracking, ensure transparency in process representation, and enable consistent comparison across multiple spatial scales. Model assumptions were applied to isolate key processes, enhance interpretability, and establish a conservative foundation for sediment redistribution modeling. These assumptions, and the uncertainties they entail, highlight important areas for future research, model extension, and broader application:

1. Integrate variable wind regimes and seasonal variability

Future research should incorporate realistic wind behavior (e.g., turbulence and variable velocity) to simulate the cumulative impacts of erosive conditions over time and across different seasons.

2. Investigate sub-unit variability

Within-field heterogeneity, including variability in land use/cover mosaics, soil texture, and micro-topography, can influence wind erosion both directly, by affecting wind flow and sediment transport processes, and indirectly, by modifying key controlling factors such as surface condition and soil moisture. Incorporating finer-scale spatial variability in future work would allow a more realistic representation of wind erosion dynamics, particularly in transitional areas where mixed land cover and soil conditions influence sediment entrainment and deposition.

3. Assess the role of topography and landform

The influence of slope, landform configuration, and surface variability on wind energy and sediment redistribution requires further investigation. Topographic variation affects wind flow patterns and contributes to spatial differences in soil moisture, which in turn influence soil erodibility. Even slight elevation differences can alter moisture distribution and sediment trapping

potential, especially in transitional zones where changes in surface roughness and exposure modify local wind dynamics.

4. Simulate dynamic soil moisture and seasonal surface conditions

Future scenarios could test variable soil moisture regimes (dry, moist, frozen/thawed), allowing exploration of how seasonal conditions influence detachment thresholds and wind erosivity. Incorporating dynamic surface conditions such as residue cover, surface crusting, snow cover, surface drainage, and microtopography would improve representation of temporal variability in soil moisture and could be considered in future modelling.

5. Redefine maximum transport distances using alternative approaches

In this study, a maximum transport distance of 3,000 m was assumed based on theoretical and empirical models describing the relationship between particle size and transport distance for different wind erosion transport modes. Sediment transported beyond this threshold would correspond to very fine fractions representing less than 1% of the total eroded soil mass; therefore, its contribution was considered negligible for the purposes of this study.

Under extreme atmospheric conditions, very fine dust particles (much smaller than 0.0002 mm), are observed to be transported at high altitudes over thousands of kilometers, enabling intercontinental transfer of dust from major desert regions such as the Sahara and Asian deserts across the Atlantic and Pacific Oceans (Kellogg & Griffin, 2006). While the assumptions applied in this study were based on transport mode behaviour and the mass-balance framework, future research could refine maximum transport distance assumptions using alternative approaches to better evaluate the potential range of sediment transport under varying environmental conditions.

6. Examine multi-directional sediment movement

The current west-east transect structure simplifies mass tracking but does not capture lateral or multi-directional transport. Future modeling could adopt different dimensional spatial grids to simulate crosswind movement, sheltering effects, and changes caused by direction shift.

In addition, this study focused on worst-case wind erosion scenarios to explore upper-bound transport and deposition dynamics. As discussed in Chapter 3, the scenarios were defined using three core variables: soil texture, land use/land cover, and spatial scale. Future work could broaden scenario design by incorporating additional drivers such as wind intensity classes, soil moisture conditions, freeze-thaw cycles, vegetation cover, the presence of conservation structures, and potential changes in wind regimes under climate change. This would allow more flexible simulations that better reflect seasonal patterns, management practices, and possible long-term environmental change.

Another valuable direction for future research is to estimate carbon and nutrient delivery to wetlands by linking modeled sediment fluxes with field-measured concentrations. In practice, the model would provide spatial and particle-size-specific sediment delivery (the carrier), while field sampling and laboratory analyses from independent programs would supply fraction-specific SOC, total N, and total P values (the content). Combining modeled fluxes with these measured coefficients would yield theoretical carbon/nutrient transfer estimates with uncertainty bounds. This integration would support stronger, policy-relevant estimates of wetlands' role as nature-based climate solutions in agricultural landscapes. In summary, future research can build on this model by incorporating environmental variability and dynamic inputs, extending transport representation (e.g., multi-dimensional grids, redefining maximum transport distance), and linking modeled sediment fluxes with field-measured carbon and nutrient concentrations. These advances will preserve the model's core framework while enabling more accurate estimates of sediment-

driven carbon and nutrient delivery, stronger guidance for site-specific management, and more robust assessments of wind erosion impacts across Prairie agroecosystems.

While this study isolated wind-driven sediment transport to better understand its individual contribution, wind erosion in agricultural landscapes does not occur in isolation. Interactions with other processes, particularly tillage and water erosion, can modify sediment redistribution patterns. For example, tillage can redistribute soil vertically and laterally, altering surface conditions and sediment availability for subsequent wind transport. Similarly, surface roughness and microtopography shaped by tillage and water processes can influence wind flow and particle entrainment. Although these interactions were not explicitly represented in the present modelling framework, they may affect the magnitude and spatial patterns of sediment redistribution observed in field conditions.

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APPENDICES

Table 3.A. Particle-size classes, dominant transport modes, governing equations, and maximum transport distances applied in the WCS model.

Size range (mm)	CanSIS class	Mode	Core equation / model	Primary sources	Maximum transport distance in the model (m)
2.0-0.5	very coarse & coarse sand	Creep	Owen shear-decay cutoff ¹	Owen (1964); Shao (2008)	5
0.5-0.1	medium & fine sand	Saltation	Bagnold–Owen hop ² kinematics	Bagnold (1941); Owen (1964); Shao (2008)	30
0.1-0.05	very fine sand	Suspension	Stokes settling + kinematic transport ³	Stokes (1851); Bagnold (1941); Shao (2008)	600
0.05-0.01	silt	Suspension	Stokes settling + kinematic transport	Stokes (1851); Shao (2008)	600
0.01-0.002	fine silt	Suspension PM ₁₀	Stokes + kinematic transport with single-gust time limit ⁴	Stokes (1851); Shao (2008)	3,000
0.002-0.0002	clay	Suspension PM ₁₀	Stokes + kinematic transport with single-gust time limit	Stokes (1851); Shao (2008)	3,000
≤ 0.0002	fine clay	Suspension PM ₁₀	Stokes + kinematic transport with single-gust time limit	Stokes (1851); Shao (2008)	3,000

1. Creep stops where wind shear drops below the threshold as flow enters roughness.

2. One short, ballistic jump of a particle in saltation.

3. Advection by the mean flow and gravitational settling.

4. Maximum time a particle remains airborne during a single gust event.

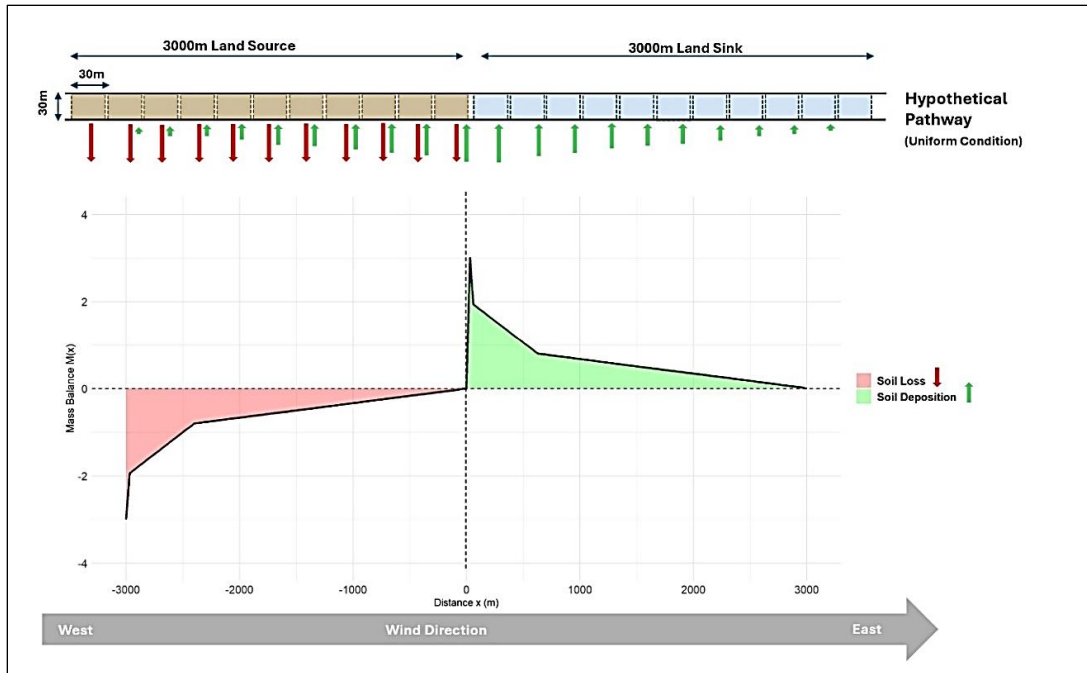


Figure 3.A. Generalized mass-balance transfer function illustrating soil redistribution along a hypothetical 6 km pathway under uniform conditions. The upwind section (3,000 m) acts as a source zone (red), while the downwind section (3,000 m) functions as a sink where sediment progressively deposits (green). Arrows indicate the direction and relative magnitude of soil fluxes. The function maintains mass conservation by ensuring total erosion equals total deposition.

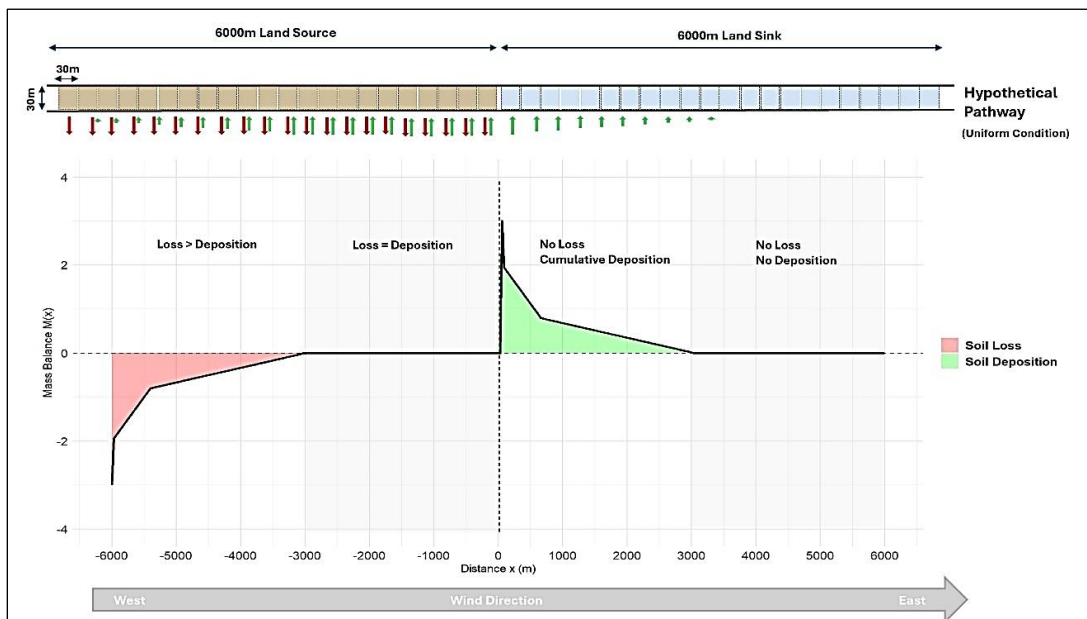


Figure 3.B. Mass-balance response to extending the source length to 6000 m under fixed transport limits. Beyond 3,000 m within the source, the mass balance $M(x)$ remains zero: soil loss is fully offset by sediment arriving from upwind, indicating a dynamic equilibrium. Consequently, lengthening the source beyond this distance does not

change the overall mass balance; it only extends a downwind no-exchange segment. Deposition in the sink still declines with distance and stops before the end of the sink zone.

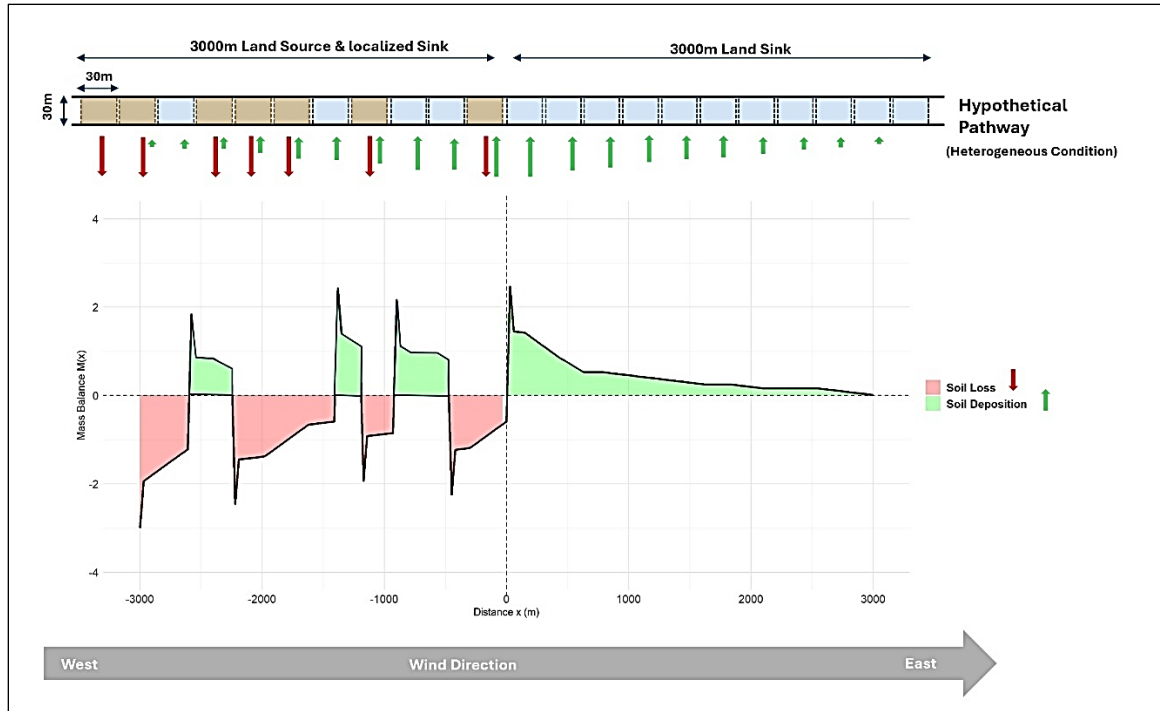


Figure 3.C. Mass-balance response under heterogeneous surface conditions (mosaic land use and soil texture). Along a 6-km pathway, spatial heterogeneity produces discontinuous transport: loss and deposition occur in alternating pulses, with several localized sink segments appearing inside the land source. Compared with uniform-condition runs, heterogeneity shortens effective transport distances and increases near-source deposition.

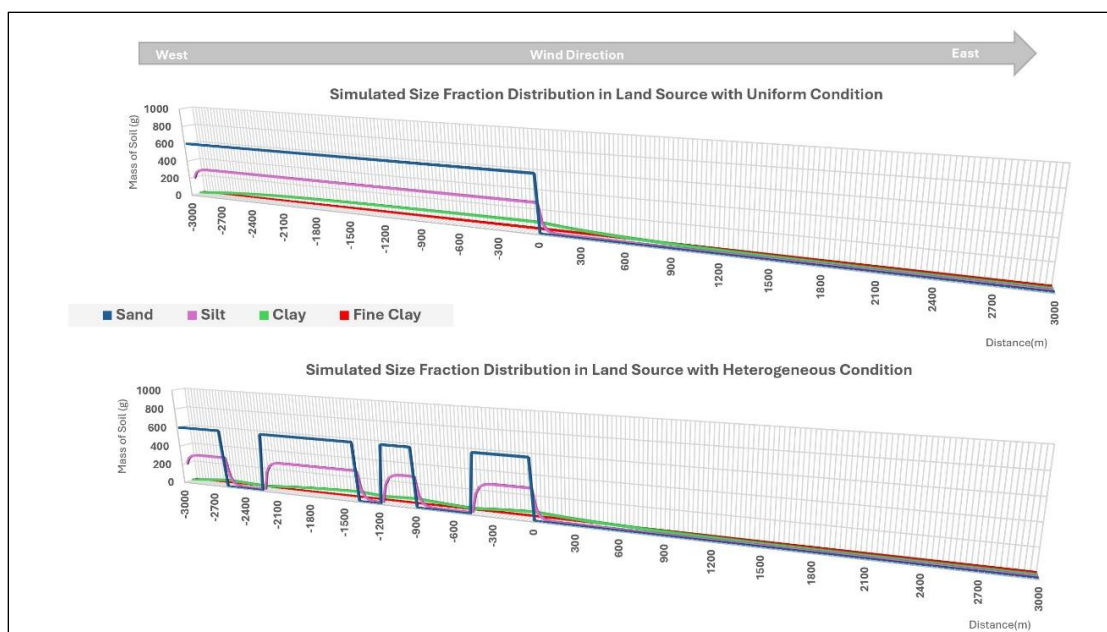


Figure 3.D. Redistribution of particle-size fractions under uniform vs. heterogeneous surfaces. Heterogeneity increases near-source deposition and shortens effective transport distances, though the overall pattern remains in sink land.

Table 4.A. SWEEP-simulated soil loss by transport mode for each field unit type.

Field Unit Type (see Table 3.2)	Creep & Saltation (g m ⁻²)	Suspension (g m ⁻²)	Suspension PM ₁₀ (g m ⁻²)	Mass of Loss leaving downwind boundary (g m ⁻¹)	Total Mass of Loss in Field Unit (g)
C-Cl	16.00	3.33	0.07	582.13	17,463.90
C-HCl	9.67	1.67	0.03	340.78	10,223.40
C-L	27.33	5.33	0.09	982.64	29,479.20
C-CIL	21.33	3.67	0.06	751.86	22,555.80
C-SiL	32.67	6.67	0.14	1,184.10	35,523.00
C-S	97.67	32.67	0.67	3,930.00	117,900.00
C-LS	41.33	7.67	0.13	1,473.78	44,213.40
C-SiCl	14.67	2.67	0.06	521.77	15,653.10
C-SCIL	58.67	17.00	0.33	2,280.00	68,400.00
B- LS	109.67	38.00	1.00	4,460.00	133,800.00
U- Cl	0.00	1.80	0.04	55.26	1,657.80
U-L	0.00	4.10	0.07	125.16	3,754.80
G	0.00	0.00	0.00	0.00	0.00
W	0.00	0.00	0.00	0.00	0.00

Table 4.B. Model results for the MCWEM-WCS pathways, summarizing total net erosion-deposition along each pathway and sediment delivery and deposition within the final sink (Lake Winnipeg).

WCS Pathways	Regional-scale	Landscape-scale	Field-scale
Pathway Dimensions			
Length (m)	1,100,000	3,000	600
Width (m)	30	30	30
Area (m ²)	33,000,000	90,000	18,000
Final Sink Dimensions			
Length (m)	3,000	3,000	3,000
Width (m)	30	30	30
Area (m ²)	90,000	90,000	90,000
Erosion across Modeled Pathway			
Delivered mass to final sink boundary (g m ⁻¹)	1,827	1,827	1,724
Total mass delivered to modeled sink (g)	54,810	54,810	51,720
Pathway-average equivalent soil loss (g m ⁻²)*	0.00166	0.609	2.870
Deposition within Modeled Final Sink **			
Mean areal deposition across sink (g m ⁻²)	0.609	0.609	0.575

Following SWEEP conventions, deposition delivered to the final sink is expressed as mass per meter of boundary width (g m⁻¹). Total sink mass equals (g m⁻¹) × 30 m width; mean areal deposition equals total mass ÷ Area of pathway.

*This value represents the uniform-source equivalent (total mass delivered ÷ source pathway area) and therefore assumes all units contribute equally to soil loss; it is used for normalization rather than to imply spatially uniform erosion.

**Final sink refers to the first 3,000 m lakeward of the Lake Winnipeg, including riparian margin.

SUPPLEMENTARY MATERIALS

Table S1. Fifteen-minute meteorological observations at the Minnedosa weather station on 31 May 2022. Source: Manitoba Agriculture Weather Program (2024)

TMSTAMP	Air_T	RH	AvgWS	AvgWD	MaxWS
0:15	9.97	93.6	19.1	341	25.62
0:30	9.82	93.1	19.31	339.3	25.21
0:45	9.76	93.6	17.84	340.8	23.21
1:00	9.79	92.1	17.98	343.4	22.76
1:15	9.89	90.3	18.2	343.1	23.95
1:30	9.77	89.9	18.3	344.3	24.46
1:45	9.79	87.9	19.22	342.7	24.7
2:00	9.61	88.4	19.2	337.6	24.5
2:15	9.45	87.6	19.02	338.3	23.77
2:30	9.05	87.7	19.75	335.6	25.62
2:45	8.74	88.1	20.41	335.1	28.17
3:00	8.51	87.5	19.57	333.6	25.66
3:15	8.26	87.8	19.29	333.7	25.24
3:30	8.07	86.7	20.33	332.9	25.24
3:45	7.87	86.4	20.06	332.8	25.38
4:00	7.71	86	20.14	332.7	25.26
4:15	7.54	86.5	19.38	332.6	26.56
4:30	7.2	88.5	18.4	332	23.38
4:45	6.96	89.7	17.15	332.3	23.17
5:00	6.78	89.5	17.8	331.4	23.34
5:15	6.77	86.2	20.9	332.2	29.2
5:30	6.55	85.3	21.07	330.1	26.62
5:45	6.28	86.5	20	331.8	24.28
6:00	6.25	86.1	20.14	329.1	25.07
6:15	6.21	86.6	19.08	327	24.17
6:30	6.04	85.6	20.44	325.5	26.24
6:45	5.82	85.3	21.66	325.3	27.81
7:00	5.62	86.7	19.66	323.2	23.97
7:15	5.53	87.5	19.43	319.7	25.87
7:30	5.52	88.5	19.61	317.4	24.15
7:45	5.7	88	20.51	319.4	27.73
8:00	5.76	88.7	20.02	319.3	27.13
8:15	5.92	87.7	22.37	317.7	28.97
8:30	6.18	87.4	21.26	322.7	25.95
8:45	6.41	85.2	22.82	325.9	29.03
9:00	6.64	83.6	21.53	327.4	27.69
9:15	6.91	82.1	21.01	329.2	26.5
9:30	7.05	81.7	20.49	327.5	26.99

TMSTAMP	Air_T	RH	AvgWS	AvgWD	MaxWS
9:45	7.23	80.5	20.03	325.5	26.87
10:00	7.4	81	18.23	320.4	24.48
10:15	7.47	80.6	19.08	323.5	23.46
10:30	7.66	79.43	19.06	321.8	24.75
10:45	7.87	78.67	19.34	320.8	24.46
11:00	8.02	78.7	19.72	316.6	25.26
11:15	8.19	78.39	19.07	317.5	24.17
11:30	8.19	77.17	19.76	313.6	26.71
11:45	8.2	76.85	18.99	315.1	24.03
12:00	8.1	76.13	19.63	314.6	25.42
12:15	8.42	75.11	19.84	314.5	25.73
12:30	8.79	73.45	19.39	316.6	24.93
12:45	8.88	71.07	20.85	311.3	26.52
13:00	9.2	70.38	18.08	313.8	24.44
13:15	9.8	70.12	17.4	310.3	24.32
13:30	10.19	68.4	17.1	312.1	21.44
13:45	10.42	66.87	16.81	309.5	21.81
14:00	10.88	67.88	15.18	311.8	19.31
14:15	11.14	66.36	15.41	309.9	20.42
14:30	11.6	64.22	14.93	308.4	18.62
14:45	11.91	62.59	14.85	301.7	20.34
15:00	12.05	60.78	14.25	311.3	22.01
15:15	12.43	59.57	13.31	307.3	17.6
15:30	12.94	56.97	13.36	305.9	18.4
15:45	13.15	56.06	14.67	296.7	19.13
16:00	12.94	55.93	14.75	305.1	17.93
16:15	12.88	55.34	13.89	297.3	17.93
16:30	13.2	53.66	14.01	294.5	19.44
16:45	13.16	50.95	14.9	301.7	18.97
17:00	13.43	49.73	13.69	303.6	19.74
17:15	13.55	45.98	14.46	303.3	19.44
17:30	13.78	42.8	12.62	301	17.19
17:45	14.06	41.74	12.65	295.2	15.86
18:00	14.02	40.32	11.89	285.2	15.72
18:15	14.16	38.71	12.77	289.3	17.29
18:30	14.15	38.66	11.41	296.6	14.8
18:45	14.14	37.39	13.05	306.8	16.86
19:00	13.99	37.73	12.04	298.6	14.72
19:15	14.05	37.75	11.56	300.3	15.95
19:30	13.89	37	10.99	303.9	14.76
19:45	13.71	35.22	11.91	301.8	15.31
20:00	13.45	36.02	11.49	306.7	15.03
20:15	13.19	37.43	10.66	312.2	13.48

TMSTAMP	Air_T	RH	AvgWS	AvgWD	MaxWS
20:30	12.93	38.5	10.46	307.9	13.43
20:45	12.74	37.95	10.57	304.7	13.88
21:00	12.35	39	8.88	302.3	13.21
21:15	11.98	39	9.08	306.9	12.31
21:30	11.31	42.02	6.63	311	10.47
21:45	10.53	45.51	4.91	304.6	6.06
22:00	10.11	48.08	4.42	294.7	5.55
22:15	10.03	48.62	3.72	284.1	5.21
22:30	10.27	46.56	3.63	277.3	4.67
22:45	10.35	45.03	3.06	260.8	4.23
23:00	10.08	46.4	3.38	255.7	4.67
23:15	9.98	46.89	3.44	253.5	4.74
23:30	9.91	47.74	3.16	253.4	4.59
23:45	9.5	49.99	3.31	259.5	4.82
0:00	9.19	52.04	3.41	260.8	5.72

TMSTAMP: time stamp of interval completion

Air_T: Avg of 5 sec temp vals prev 1 min (°C)

RH: Avg of 5 sec RH vals prec 1 min (%)

AvgWS: scalar mean wind speed prev 15 min (m s⁻¹)

AvgWD: derived vector wind dir prev 15 min- deg true (°)

Table S2. Summary of SWEEP input parameters. Parameters under the Field and Weather categories were fixed across all simulations, while inputs under the Biomass, Soil Layers, and Soil Surface categories were varied among WCS scenarios to represent differences in vegetation cover, land use and soil texture.

Category	Parameter	Unit	Description / Notes
Field	X-Length	m	Field length in wind direction
	Y-Length	m	Field width (cross-wind)
	Angle	deg	Field orientation relative to wind
	Barrier position (X ₁ , Y ₁ , X ₂ , Y ₂)	m	Start and end coordinates of barrier(s)
	Barrier height / width	m	Physical dimensions of barrier
	Barrier porosity	fraction	Fractional openness of barrier
Biomass	Residue average height	m	Mean height of standing residue
	Residue stem area index	m ² m ⁻²	Ratio of stem area to ground area
	Residue leaf area index	m ² m ⁻²	Ratio of leaf area to ground area
	Residue flat cover	m ² m ⁻²	Fractional cover of flat residue
	Growing crop average height	m	Mean height of live crop canopy
	Growing crop stem area index	m ² m ⁻²	Canopy density (stems)
	Growing crop leaf area index	m ² m ⁻²	Canopy density (leaves)
	Row spacing	m	Distance between crop rows
	Seed placement	N/A	Ridge / furrow option
Soil Layers	Layer thickness	mm	
	Sand, very fine sand, silt, clay fractions	Mg Mg ⁻¹	Particle-size distribution
	Rock volume fraction	m ³ m ⁻³	Coarse fragment content

Category	Parameter	Unit	Description / Notes
	Dry bulk density	Mg m ⁻³	
	Aggregate density	Mg m ⁻³	Mean particle density
	Aggregate stability	ln (J kg ⁻¹)	Resistance to breakdown
	GMD / GSD of aggregate sizes	mm	Aggregate size statistics
	Minimum / maximum aggregate size	mm	
	Soil wilting point water content	Mg Mg ⁻¹	
Soil Surface	Surface crust fraction	m ² m ⁻²	Fraction of crusted surface
	Surface crust thickness	mm	
	Loose material on crust	m ² m ⁻²	Fraction of exposed loose material
	Loose mass on crust	kg m ⁻²	
	Crust density	Mg m ⁻³	Density of surface crust
	Crust stability	ln (J kg ⁻¹)	Crust resistance to detachment
	Random roughness (Allmaras)	mm	Surface micro-relief
	Ridge height / spacing / width	mm	Oriented roughness geometry
	Ridge orientation	deg	Ridge direction relative to wind
	Dike spacing	mm	Cross-ridge spacing
	Snow depth	mm	
	Hourly surface water content	Mg Mg ⁻¹	
Weather	Air density	kg m ⁻³	
	Wind direction	deg	
	Anemometer height	m	
	Aerodynamic roughness (z ₀)	mm	Surface roughness at station
	z ₀ location flag	N/A	Station or Field option
	Number of intervals per day	N/A	Time resolution (e.g., 96 for 15-minutes)
	Wind table	m s ⁻¹	Wind speeds for each interval