

Methods for Modelling Wetlands in Hydrologic Models

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Wetlands are abundant natural systems that serve as important ecosystems, mechanisms for nutrient filtering and storage, and providers of flood mitigation services. Wetlands strongly influence the hydrologic response and water balance on a landscape. The practice of water resources management often relies on numerical computer models that represent hydrologic features within a watershed like wetlands, lakes, and rivers, to accurately simulate the movement of water. However, representation of wetlands in hydrologic models is challenged by their small-scale nature, numerous classification schemes that are not readily associated with a water balance conceptual model, and sometimes complex hydrology. A shortcoming of existing wetland modelling studies includes the lack of multiple wetland types being represented, often due to the complexity that accompanies wetland classification schemes.

In this study, we address three research objectives: 1) to inventory existing wetland modelling methods and develop a catalogue of conceptual-numerical wetland modelling methods in hydrology based on wetland classifications and numerical water balance equations, 2) to implement conceptual-numerical wetland modelling methods in a regional hydrologic model case study and evaluate model performance to determine the impact of wetlands on simulation results, and 3) to examine how available wetland mapping products can inform wetland modelling.

A hydrologic model of the Nipissing watershed in Ontario was built using the Raven Hydrologic Framework and calibrated in a multi-objective calibration to both high and low flow objective functions in three modelling scenarios. The first modelling scenario (Scenario 1) contained no wetland representation; the second modelling scenario (Scenario 2) contained explicit wetland representation of one wetland conceptual-numerical model type; and the third modelling scenario contained explicit wetland representation of three wetland conceptual-numerical model types based on connectivity of wetlands to modelled streams and lakes.

Calibration results indicated good model performance for all model scenarios, as an adequate performance threshold of 0.50 for the Kling Gupta Efficiency (KGE) and log transformed Nash Sutcliffe Efficiency (logNSE) was achieved for both performance metrics. In calibration, Scenario 2 most often outperformed Scenario 1 (no wetland scenario) at individual calibration gauges and

Scenario 3 (most complex wetland scenario) due to pareto solution uncertainty and site-specific properties.

Validation results indicated that Scenario 3 most often outperformed the other two scenarios across multiple performance metrics at individual flow gauges and handled low flows especially well when analyzing low flow performance metrics and hydrographs. This is attributed to Scenario 3 storing the most water in wetland depressions out of all modelling scenarios from abstraction, lateral diversion of water accounting for wetland contributing areas, and groundwater process parameters set up for each simulated wetland type. Percent bias median and spread across all flow gauges significantly decreased by 15% from Scenario 2 to Scenario 3, highlighting the importance of low flow accuracy to hydrologic model performance.

Flow duration curves and hydrographs plotted by flow gauge demonstrated that site-specific properties of the entire study area and individual gauge drainage areas can impact simulation results. There was no relationship found between gauge drainage area, wetland coverage percent by area, and model performance at individual gauges in this study.

Four wetland mapping datasets in Ontario were compared to select a wetland data input to the Nipissing model. By comparing each wetland dataset, a formalized checklist is provided for modellers to use as a reference when making similar comparisons between their own wetland mapping products. It is recommended that wetland mapping product comparisons for project suitability be performed by first comparing wetland coverage between datasets using the wetland polygon coverage by area, then comparing spatial variability between datasets by inspecting areas of overlap and non-overlap, and finally comparing data attributes, particularly wetland classifications and any discrepancies between dataset attributes.

While the results of this study demonstrate the importance of low flow accuracy to model performance through the representation of wetlands, improvements could be made to aid future studies. It is recommended that future studies select a watershed with high quality flow and meteorological data, basins with varying wetland coverage, and little to no water regulation influence (e.g., hydroelectric dams). It is also recommended that the wetland conceptual-numerical models presented in this thesis be further tested on watersheds of different sizes, different combinations of wetland types, and varying degrees of complexity.

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Dedication

I dedicate this thesis to my loving family who have supported me endlessly throughout my life, raised me into who I am today, and believed in my ambitions. Mom & Dad, Damian, Grandma Linda, Col, Grandma & Grandpa Liko - thank you, and I love you all more than words can express.

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Chapter 1

Introduction

1.1 What is a Wetland?

Ponds, bogs, swamps, marshes, mires, and sloughs; these are some terms commonly used to refer to wetland systems. Wetlands can often be seen by roadsides, within communities, in parks, by streams, and many other areas in nature. The reader can likely picture their local pond with lily pads and ducks. These wetlands, both large and small, play an important role for ecosystems and water management on the landscape.

Canada contains a mosaic of wetlands with different types, sizes, and hydrologic function. The definition of a wetland varies between sources. The Canadian Wetland Classification System defines wetlands as land either permanently or temporarily saturated by water, which sustains poorly drained soils, water-loving vegetation, and aquatic life adapted to wet environments (The National Wetlands Working Group, 1997). Similar definitions exist elsewhere in the literature (Glooschenko et al., 1993; MacKenzie & Moran, 2004; Environment and Climate Change Canada, 2016). However, definitions of individual wetland types or even whether a water body should be considered a wetland can vary with location-specific terminology. This variability in definitions can make characterizing specific wetlands nuanced. The fundamental importance of wetlands, however, remains the same.

1.2 The Importance of Wetlands

Wetlands provide multiple ecological and hydrological benefits to a landscape. Perhaps the most obvious benefit is providing habitats and ecosystems for many species of birds, fish, mammals, amphibians, and vegetation. Wetlands are also widely researched for their water treatment benefits to downstream water courses and waterbodies. The long retention times of wetlands due to stagnant and slow-moving water allows nutrients like phosphorus and nitrogen to settle, reducing downstream concentrations (Marton et al., 2015; Basu et al., 2022).

Wetlands are also hydrologically significant to the landscape. Wetland depressions can act as storage compartments for water flowing from rivers or lakes, overland flow, or groundwater. These compartments can store excess water from the landscape and slow its movement. Thus, wetlands have been studied as providing flood mitigation benefits, as they reduce peak flows at the watershed outlet

and increase the time of concentration during storm events (Golden et al., 2014; Ameli & Creed, 2019; Goyette et al., 2023).

Often, the specific functions of wetlands within a study area are unknown and unmeasured. The knowledge of wetland function can be useful for decision makers when thinking about wetland management and conservation. As a demonstration, consider a simplified example: suppose a land developer is looking to build a subdivision in a small catchment. To build the subdivision, they are looking to remove several natural wetlands. While reviewing their construction application, the local environmental authority flags the wetlands as sensitive ecosystems and has evidence that the wetlands attenuate peak flows during precipitation events. As a result, the subdivision gets rejected in support of wetland conservation efforts. In this simplified example, having knowledge of local wetland systems can influence their conservation. At a larger scale, water resources management is influenced by knowledge of the watershed. A cascading effect is observed, as water resources management is influenced by the watershed, and the watershed is influenced by its components (e.g., wetlands).

To better understand complex systems, like watersheds, models are often used. Models are tools developed to be simplified representations of real systems. In the water resources field, watershed models are called hydrologic models. Hydrologic models are useful for representing water balances in a watershed and making predictions about the quantity of water and the timing of its appearance. Ultimately, hydrologic models are expected to predict the watershed response to answer water resources management questions, such as: How much water will move past this river segment in response to a given storm? In what ways will the watershed respond to urbanization where natural forest becomes cropland or impervious urban areas? In what ways do wetlands contribute to the hydrology of a study area?

The latter question was posed as the inspiration for this thesis. Many previous studies have also posed this question, which are reviewed in later chapters (Evenson et al., 2015; Fossey et al., 2015; Blanchette et al., 2019). To further explore this question, this study uses the representation of wetlands in hydrologic models to provide insight into wetland contributions to the watershed. A discussion and investigation of wetland hydrology and wetland modelling methods is presented in this thesis guided by research goals outlined in the following Section.

1.3 Thesis Objectives

The objectives of this thesis are to:

1. Inventory existing wetland modelling methods and develop a catalogue of conceptual-numerical wetland modelling methods in hydrology.
2. Implement conceptual-numerical wetland modelling methods in a hydrologic model and evaluate model performance.
3. Examine how available wetland mapping products can inform wetland modelling.

1.4 Thesis Organization

This thesis is organized in the following manner:

Chapter 1 provides motivation for this thesis work and outlines the thesis objectives.

Chapter 2 provides background information on wetland types, classification schemes, hydrologic modelling, and the application of wetlands in hydrologic modelling based on past studies.

Chapter 3 contains an inventory of existing wetland modelling methods based on a thorough literature review.

Chapter 4 discusses the study area and methods for the hydrologic modelling performed in this thesis.

Chapter 5 presents calibration and validation results of the hydrologic modelling performed in this thesis.

Chapter 6 provides a discussion of calibration and validation results, along with the assessment of wetland mapping product usage in hydrologic modelling.

Chapter 7 concludes this thesis and provides recommendations for future studies.

Chapter 2

Background

2.1 Types of Wetlands

This section introduces the most common ways to classify wetlands using wetland ecological type and hydrogeomorphic forms. These classification schemes are encountered in literature, wetland mapping datasets, and even study site descriptions, thus being important for modellers to understand for their own study areas. These classification schemes are later used in Chapter 3 to influence the compilation of an improved classification scheme of better utility to hydrologic modellers.

2.1.1 Ecological Wetland Types

Wetlands are most often classified in terms of ecological features, such as soil and vegetation type. Just as vegetation types are biologically diverse, soil types are also diverse in that their physical and chemical properties vary with their ecosystems. The degree of organic matter determines, in part, the dominant vegetation in the wetland. The degree of organic matter is determined by soil categories including either mineral or organic soils. Mineral soil wetlands are located in areas where little to no organic matter (peat) is accumulated, often because of frequent water level fluctuations and external influences on water quality such as sedimentation, vegetation, upstream flow, or dams. Organic wetlands, also called peatlands, include wetlands with a peat accumulation of more than 40 cm, on top of which organic soils develop (The National Wetlands Working Group, 1997). Wetlands can be further classified into wetland class, form, and type, which can be dependent on climate, hydrology, chemistry, geomorphology, and biology (The National Wetlands Working Group, 1997). The broadest category used to describe wetland systems is by wetland class, and includes bogs, fens, marshes, swamps, and shallow/open water wetlands. These five wetland classes are summarized with respect to soil type categories in Figure 2.1.

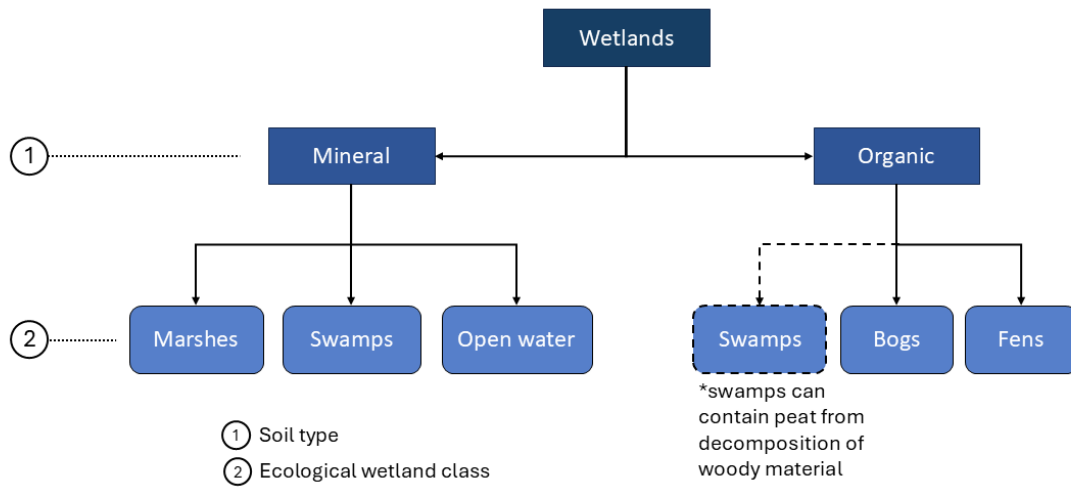


Figure 2.1. Wetland classes categorized by soil type. Based on classifications defined in Ducks Unlimited Canada, 2025; The National Wetlands Working Group, 1997

Each ecological wetland class is defined using vegetation and hydrological characteristics described in the following subsections.

2.1.1.1 Bogs

Bogs are a type of organic wetland, or peatland, characterized by a surface that is either raised (like a dome) or level with the surrounding terrain and contains anywhere from 40 cm to 5 m of peat (The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004; Metro Vancouver, 2007; Ducks Unlimited Canada, 2025). The accumulation of peat above the ground surface results in bogs only receiving incoming water via precipitation (this is termed ombrogenous) with the water table being raised above the ground surface as well. Accumulated water is acidic and very low in dissolved minerals due to being fed solely through precipitation. Thus, the vegetation growing in bogs is well adapted to low nutrient and acidic environments. Bogs are generally dominated by Sphagnum moss and lichens but can be either treed or treeless and can include species like black spruce and bog cranberry (The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004; Metro Vancouver, 2007; Price et al., 2023; Ducks Unlimited Canada, 2025). Peat accumulation can be divided into two distinct layers: the acrotelm and catotelm. The acrotelm is the upper layer consisting

of highly porous and poorly decomposed organic material. It can be seen as the “living layer”, where vegetation grows and ultimately contributes to the peat layer as it decomposes (Ingram, 1978; The National Wetlands Working Group, 1997; Metro Vancouver, 2007; Baird et al., 2008; Morris, 2010). Water moves freely through the acrotelm and may flow towards the outside edges of the bog to contribute to surface flow. The catotelm is the lower soil layer consisting of dense peat with very low permeability and well-decomposed organic materials. This lower layer is saturated and anaerobic, allowing only very slow water movement (Ingram, 1978; The National Wetlands Working Group, 1997; Metro Vancouver, 2007; Baird et al., 2008; Morris, 2010). The acrotelm and catotelm layers are sustained by the production of new organics at a faster rate than the decomposition of existing organic materials, resulting in peat layers that accumulate over time (Bragg, 2002). Water flows through the acrotelm of the bog along its sloped sides and exits the bog in transition zones called the lagg. The lagg can also be thought of as a discharge zone, acting as a buffer between nutrient-poor water and fen-like areas with increasing nutrient levels (Bragg, 2002; Metro Vancouver, 2007; Langlois et al., 2015).

2.1.1.2 Fens

Fens are a type of peatland containing at least 40 cm of organic material accumulation and can be located in topographic low points, thus receiving water from uplands, often from bogs (MacKenzie & Moran, 2004; Sampath et al., 2015; Ducks Unlimited Canada, 2025). Fens receive water through precipitation, groundwater, overland flow, and open stream channels, leading to a fluctuating water table (The National Wetlands Working Group, 1997; Sampath et al., 2015). Due to fens receiving water from groundwater and overland flow in addition to precipitation, the water is more nutrient rich than in bogs and can support vegetation with higher nutrient requirements (The National Wetlands Working Group, 1997; Bragg, 2002; Ducks Unlimited Canada, 2025). Fens often contain mosses like Sphagnum moss and brown mosses in addition to black spruce and dwarf birch. Shrubs tend to occur at drier locations, and sedges (grass-like vegetation) at wetter locations (The National Wetlands Working Group, 1997; Ducks Unlimited Canada, 2025).

2.1.1.3 Swamps

Swamps are forested or wooded wetlands with at least 30% woody vegetation cover and are underlain by mineral or peat soils (The National Wetlands Working Group, 1997; Price et al., 2023; Ducks Unlimited Canada, 2025). Swamps can be categorized as both a mineral soil wetland and a peatland

since well-decomposed wood-rich organics can accumulate on top of mineral soils in raised mounds and dips called hummocks (The National Wetlands Working Group, 1997; McKillop et al., 1999; MacKenzie & Moran, 2004; Price et al., 2023; Ducks Unlimited Canada, 2025). Water levels can fluctuate seasonally as net water storage is dependent on groundwater exchange, overland flow, evapotranspiration, and precipitation. Swamps may be seasonally flooded depending on snowmelt or responses to high precipitation events leading to increased streamflow (Price et al., 2023). The water table in swamps is at or below the surrounding ground surface, with treed swamps tending to be drier due to canopy interception and transition into upland forest, while shrub swamps tend to be wetter and can transition into marshes or other wetland classes (The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004). The seasonality in water table level can also be attributed to many of these wetland systems occurring as headwater swamps, where groundwater discharge dominates hydrologic processes (McKillop et al., 1999; Waddington et al., 2015; McCarter et al., 2024). Swamps can also be located in coastal regions, acting as a buffer zone between open water, marshes, and upland forest (Hoeppner & Rose, 2011).

2.1.1.4 Marshes

Marshes are intermittently flooded wetlands with water levels that can fluctuate daily, seasonally, or annually, with fluctuating water tables. Temporal variation in marsh water levels can be caused by fluvial or tidal flooding, overland flow, evapotranspiration, movement of water through the soil, and groundwater exchange (Harvey et al., 1987; The National Wetlands Working Group, 1997; Montalto et al., 2002; MacKenzie & Moran, 2004; Temmerman et al., 2005; Ducks Unlimited Canada, 2025). Marshes are mineral wetlands dominated by emergent aquatic vegetation like graminoids (bulrushes, grasses) or floating vegetation (lily pads, duckweed), which contribute very little organic material to the wetland bottom (The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004; Ducks Unlimited Canada, 2025). Many marshes are located adjacent to rivers, lakes, oceans, or floodplains and are dominated by groundwater discharge and intermittent flooding (Harvey et al., 1987; The National Wetlands Working Group, 1997; Montalto et al., 2002; Temmerman et al., 2005). Thus, in this research, the wetland class “marsh” includes both salt marshes and fresh water marshes.

2.1.1.5 Shallow/Open Water Wetlands

Shallow or open water wetlands are permanently flooded mineral wetlands containing standing water less than 2 m deep (The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004;

Ducks Unlimited Canada, 2025). These types of wetlands, often labelled as ponds, polders, or potholes, are dominated by floating-leaved aquatic vegetation and submerged aquatic vegetation, with less than 10% of the total plant species being emergent vegetation (The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004; Ducks Unlimited Canada, 2025). Open water wetlands can be thought of as slow-flowing, transitional zones part of larger permanent water bodies (e.g., lakes), and other wetland classes while maintaining a depth too deep for most emergent vegetation to grow (The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004; Bian et al., 2021). Hydrologic processes are often controlled by groundwater and flow to/from other water bodies, with the addition of evaporation which increases as open water surface area increases (The National Wetlands Working Group, 1997; Geurink et al., 2007).

2.1.2 Hydrogeomorphic Wetland Classifications

Wetlands contain distinct hydrologic processes and pathways to transport water depending on their position and connection to other waterways in the landscape. These distinctions are referred to as hydrogeomorphic classifications and allow wetlands to be classified by their driving mechanisms for water movement. Hydrogeomorphic classifications include riverine, lacustrine, palustrine, estuarine, and isolated wetlands (Cowardin et al., 1979; Whiteley & Irwin, 1986; Brinson, 1993; The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004).

Riverine or fluvial wetlands are those that are directly adjacent to and influenced by rivers and their flowing water (Whiteley & Irwin, 1986; MacKenzie & Moran, 2004). These types of wetlands can act as a floodplain during periods of streambank overflow, and thus their water levels can also fluctuate with changes in streamflow and groundwater (Whiteley & Irwin, 1986). Pathways for water movement include precipitation, evapotranspiration, groundwater exchange, interflow, and overland flow. The term “riparian” is often used to refer to riverine wetlands and can also include lacustrine wetland systems (The National Wetlands Working Group, 1997).

Lacustrine wetlands are directly adjacent to and influenced by lakes and their hydrological processes, including wave action (Cowardin et al., 1979; Whiteley & Irwin, 1986; MacKenzie & Moran, 2004; Albert et al., 2005). As such, wetland water levels are influenced by (or identical to) adjacent lake levels and can also be subject to tidal effects. Pathways for water movement include precipitation, evapotranspiration, groundwater exchange, interflow, and overland flow.

Palustrine wetlands are sites upland of lakes or rivers that convey water through groundwater or overland flow (Whiteley & Irwin, 1986; The National Wetlands Working Group, 1997; MacKenzie & Moran, 2004). Often, palustrine wetlands are associated with small wetland areas such as Prairie potholes or ponds (MacKenzie & Moran, 2004). Pathways for water movement can include precipitation, evapotranspiration, groundwater exchange, interflow, and overland flow, but not always. Ombrogenous bogs are an example of a palustrine wetland that only receive water through precipitation and lose water through evapotranspiration and the lateral movement of water towards the edges of the bog.

Estuarine wetlands are located at the confluence of riverine and marine environments (Cowardin et al., 1979; Brinson, 1993; MacKenzie & Moran, 2004). These wetlands are directly affected by tides and can include saltwater wetlands. Pathways for water movement include precipitation, evapotranspiration, groundwater exchange, interflow, and overland flow.

Isolated wetlands are sites with no overland flow outlet into the watershed (Whiteley & Irwin, 1986). These types of wetlands are completely isolated from the rest of the watershed and do not contribute to streamflow. Some types of bogs are an example of an isolated wetland. The main water movement pathways within isolated wetlands include precipitation, evapotranspiration, and overland flow into the wetland. Some isolated wetlands may be groundwater-fed while not contributing any water to the rest of the landscape.

Some distinction also needs to be made between wetlands and other water bodies such as ponds and lakes. As described previously, the presence of hydrophytic vegetation and hydric soils supported by frequent inundation is used to distinguish wetlands from other water bodies. Water bodies that do not adhere to these criteria are defined as non-wetland. There is a gray area, however, as wetland systems may be physically part of non-wetland systems. Larger water bodies, such as ponds and lakes, may contain wetland-like areas where hydrophytic vegetation is found which then transitions to non-wetland area as water depth increases and vegetation is not able to grow (The National Wetlands Working Group, 1997). Thus, most lakes and ponds are not considered wetlands but shallower areas along the edge of these water bodies may be classified as wetland area.

Constructed wetlands and storm water management (SWM) ponds, most often located in urban areas, can fit into wetland categories based on their vegetation and soil types. Hydrophytic vegetation can be planted in constructed wetlands and SWM ponds to mimic natural wetland ecosystems. Additional

hydrophytic vegetation also begins to grow as time passes and natural transport vectors, such as animals and wind, spread seeds and soil from other wetland sites (The National Wetlands Working Group, 1997). Constructed wetlands and SWM ponds, or parts of them that contain hydrophytic vegetation and hydric soils, can be categorized based on hydrogeomorphic and/or ecological wetland classifications.

2.2 Hydrologic Modelling

A model is a simplified representation of a real-life system. Models are used to make predictions about the represented system, specifically in cases where there is a lack of knowledge (no data) or when predicting into the future (Beven, 2012). No model is ever perfect, as the tools we use are not able to capture the complexity of the real world. However, the well-known quote by the statistician George Box, “all models are wrong, but some are useful”, recognizes that despite their imperfections, models can still have practical value and be used to inform decision making.

Hydrologic models are simplified representations of a watershed used to predict quantities of water storage and fluxes at points of interest. Hydrologic models make predictions using water balance equations comprised of mathematical expressions for water cycle processes (such as precipitation, snowmelt, evapotranspiration, runoff, etc.). The complexity of hydrologic models varies between types and the intended purpose of the model. Models can vary by both process complexity and spatial complexity (Clark et al., 2017). Process complexity refers to the extent to which models explicitly represent processes, and spatial complexity refers to the extent to which models explicitly represents landscape details and the movement of water across the model (Clark et al., 2017).

Two commonly used model types that vary in process complexity include empirically and physically based models. Empirical models contain equations that are based on experimentally derived relationships (Kampf & Burges, 2007). Empirically based models are also referred to as “black box” models for the way they rely on statistics and data relationships rather than physical processes. Conversely, physically based models contain equations that are based on physics and governing rules of conservation (Kampf & Burges, 2007). Common model types that vary in spatial complexity include lumped and distributed models. Lumped models are one-dimensional in that they spatially aggregate processes over the entire catchment area to produce a flow estimate at the outlet, whereas distributed model processes and parameters are two-dimensional and outputs are a function of space and time (Singh & Woolhiser, 2002; Kampf & Burges, 2007). Fully distributed models require all

aspects to be distributed across the watershed (Singh & Woolhiser, 2002), thereby increasing model complexity. Yet, limitations of fully distributed models also exist and include increased data requirements, processing power, parameterization, and run times (Singh & Woolhiser, 2002; Clark et al., 2017). Many hydrologic models, therefore, are semi-distributed rather than fully distributed.

2.3 Wetlands in Hydrologic Modelling

Watersheds are composed of many different landcover types. Each landcover type influences how water moves and interacts with the landscape. For instance, precipitation is intercepted by canopy cover in forested areas of a watershed, meanwhile precipitation falls directly onto lakes and is routed to the lake outlet. Not only do different landcover types influence the physical pathways for water to move but also dictate the time of concentration and quantity of water reaching the watershed outlet at any given time. Thus, to accurately represent a watershed and create a useful model, it is important for hydrologic models to capture and represent the different landcover types across the watershed. In addition to common landcover types like forests and lakes, watersheds can be greatly influenced by the presence of wetlands. Wetlands have been a heavily studied topic in recent years in both field and modelling studies for their flood mitigation and water regulation services (Gleason et al., 2007; Golden et al., 2014; Ameli & Creed, 2019; Balliston & Price, 2022; Goyette et al., 2023; Wu et al., 2023). The hydrologic importance of wetlands has been explored in studies evaluating the benefits of wetland restoration versus removal on the landscape, especially in light of climate change increasing the frequency and intensity of storm events. Goyette et al. (2023) built and deployed a semi-distributed hydrologic model to assess the effects of wetland restoration scenarios under future climate scenarios. The study found that increasing the wetland coverage via restoration scenarios was able to mitigate and compensate for high and low flows induced by climate change (Goyette et al., 2023). Ameli and Creed (2019) deployed a coupled subsurface-surface fully distributed and physically based hydrologic model developed in Ameli and Craig (2014) to test riparian and non-riparian wetland loss scenarios in an Alberta watershed (Ameli & Craig, 2014). It was demonstrated that an increase in wetland loss resulted in an increase in stream peakflow and a decrease in baseflow discharged to the main stream network (Ameli & Creed, 2019). In another modelling study, this time assessing the impact of wetlands on drought in two separate watersheds, Wu et al. (2023) used a physically based, semi distributed hydrologic model to demonstrate that modelled wetlands had an overall mitigation effect on drought, reducing their duration and severity (Wu et al., 2023). It is

recognized that models are not a perfect representation of reality, however they can provide insight on the variability of the water balance with and without wetlands present. It is clear that the presence of wetlands can alter the hydrological regime of a landscape and ultimately influence the quantity and speed of water moving through the watershed.

Despite the existence of many wetland studies, an understanding of appropriate methods for physically representing wetlands in hydrologic models remains underdeveloped. The complexity of wetlands in terms of their classifications, often small surface areas compared to other water bodies like lakes, and abundance across the landscape have made accurately representing wetlands in hydrologic models a challenging task. For this reason, modelling studies simulate wetlands using a variety of different representations of wetland hydrologic processes. Wetland modelling methods are fundamentally based on water balance equations, which quantify the net water storage in individual wetlands or systems of wetlands using an assumed set of inflow and outflow processes. Five main categories of models are used to represent wetlands in hydrologic models. These categories include standalone wetland models, spatially implicit methods, spatially explicit methods, equivalent lumped wetland methods, and groundwater-based methods.

The first category of wetland representation method, the standalone wetland model, is used to simulate individual wetlands and are not connected to a greater watershed model. Examples of standalone wetland models are provided by both Hopkinson and Day (1980) and McKillop et al. (1999) where they used finite difference methods to solve coupled surface and subsurface differential equations to model forested wetlands (Hopkinson & Day, 1980; McKillop et al., 1999). In these models, the focus was to simulate hydrologic processes in a specific wetland of interest, which are represented as a depression bucket where water is allowed to fill to capacity and overflow. All wetlands in the model are assumed to function the same and are dictated by the same water balance equations whereas in real systems, wetland processes and area of impact change dynamically in space and time. However, standalone wetland models are still able to simulate hydrologic processes of specific wetland types in simple models and provide a base for more complex modelling methods.

In spatially implicit methods, wetlands are treated probabilistically in a lumped manner by characterizing things like the storage deficit or wetland contributing area distributions. These methods are spatially implicit as they do not explicitly track the water in each wetland nor are wetland locations represented. Spatially implicit methods are mainly applied to depressions in the Prairie

Pothole region of North America. Following the introduction of the Probability Distribution Model based RunOff generation (PDMROF) algorithm (Mekonnen et al., 2014), Ahmed et al. (2020) introduced the HYdrological Model for Prairie Region (HYPR) algorithm that represents wetland frequency and depth across a watershed using a Pareto distribution and PDMROF transformation parameters (Ahmed et al., 2020). Similarly, other wetland frequency distribution-based studies that aim to improve Prairie Pothole modelling by representing wetland fill and spill processes also exist (K. Shook et al., 2021; Ahmed et al., 2023; Taheri et al., 2023). Frequency distribution-based modelling of wetlands has been successfully applied to some Prairie basins but are not appropriate for watersheds with many different types of wetlands due to their inability to represent different wetland processes and dynamics.

Spatially explicit methods for modelling wetlands have been explored in recent studies and found to provide insight into wetland hydrology and its role in a watershed. Spatially explicit methods map physical wetland locations and track water movement in each wetland. Some models have attempted to simulate wetlands using storage volume and interception area relationships using remotely sensed data products. These models often treat wetland systems as a single lumped storage compartment and lack representation of dynamic wetland processes, such as fill and spill behaviour (Gleason et al., 2007). Gleason et al. (2007) built a hydrologic model using a remotely sensed wetland dataset in the Prairies to explicitly map wetland locations, including wetland catchment area to improve wetland water storage estimates. Spatially explicit wetland mapping has also been applied to models that perform wetland water balances within discretized units, such as grids or hydrologic response units (HRUs) (Golden et al., 2014). HRUs are areas defined by a unique combination of land use type, vegetation cover, and terrain type, all underlain by vertical soil profiles (Craig et al., 2020; Craig & Raven Development Team, 2024). Vertical water and energy balances are applied to each HRU within a finite number of hydrologic storage compartments (e.g., soil, canopy) (Beven, 2012). The advantage of explicitly mapping wetland location in hydrologic models is that each wetland unit can be classified and parameterized, and its connectivity to the rest of the surface water network can be represented directly.

Another application requiring the use of spatially explicit wetland models is the representation of wetland connectivity in the watershed. The concept of wetland connectivity, closely related to wetland hydrogeomorphic classifications, has been studied in hydrologic modelling applications (Golden et al., 2014; Evenson et al., 2015; Lane et al., 2018; Ameli & Creed, 2019). Recent literature

has tried to distinguish between the hydrologic importance of geographically isolated wetlands (GIWs, also referred to as non-floodplain wetlands (NFWs) or non-riparian wetlands) and riparian wetlands, also referred to as floodplain wetlands. Lane et al. (2018) describes floodplain wetlands as those that are hydrologically connected to the stream network and influenced by river processes, while describing non-floodplain wetlands as exhibiting primarily unidirectional, lateral flow from the wetland to the stream network through overland or groundwater flow (Lane et al., 2018). Through spatially explicit modelling of riparian and non-riparian wetlands, Ameli and Creed (2019) demonstrated that wetland attenuation of peak flows varied with proximity of wetlands to the stream networks. Additional studies, such as in Evenson et al. (2015), used the Soil and Water Assessment Tool (SWAT) model to represent GIWs with spatially explicit HRUs and demonstrate that explicitly modelling wetlands by connection to the watershed produces substantial effects on the basin. The complexity of spatially explicit modelling of wetlands can vary depending on the study purpose and available resources. In Golden et al. (2014), a model comparison study revealed a continuum of model types that can be used to explicitly represent wetlands, specifically GIWs. Coupled surface-subsurface models were recommended for the most complicated but accurate modelling of GIWs, whereas individual surface water or groundwater models may provide adequate representation of GIW hydrology in sites dominated by either surface or groundwater (Golden et al., 2014). It follows that the more complex the model, the more data and processing resources are required. Spatially explicit modelling of wetlands is more resource intensive than using a spatially implicit or standalone wetland modelling approach, as many HRUs or wetland data require processing, storage, and longer model run times.

Equivalent lumped wetland modelling methods are those that are spatially explicit but either lump all wetlands together or wetlands with similarities in location or type as one modelled unit. In one example study, Pomeroy et al (2010) and (2014) modelled depression-rich landscapes by lumping spatially explicit wetlands together by subbasin so that all wetland storage is aggregated by subbasin. While an advantage to using equivalent lumped wetland modelling is the simplification in wetland resolution from a computational perspective, this method lacks the accurate representation of wetland geometry, surface outlets, and connections (Evenson et al., 2018).

Groundwater-based models, such as those mentioned in Golden et al. (2014), are an additional method for wetland modelling. These include both subsurface and coupled surface-subsurface models. The commonly used groundwater model MODFLOW has been used in studies such as in

Wilsnack et al. (2001) and Bradley (1996) to simulate wetland storage in groundwater-dominated areas (Bradley, 1996; Wilsnack et al., 2001) and in bog modelling studies (Baird et al., 2012; Bleuten et al., 2020). Additional studies have used 3D groundwater-surface water interaction models, such as HydroGeoSphere, to represent both surface and subsurface processes (Frei et al., 2010; Ameli & Creed, 2019). The disadvantage of using groundwater-based models to simulate wetlands is the increased level of complexity and computational resources required to construct and execute the model (Golden et al., 2014).

The existing wetland modelling approaches have built a foundation for the representation of wetlands in hydrologic models. However, a commonality between studies is the lack of unification among wetland classifications. The terms and definitions for wetland types are often described using ecological characteristics, making it difficult for modellers to interpret the hydrology of their own wetland sites. This leaves a research opportunity to further explore and demonstrate how ecological and hydrogeomorphic wetland classifications can be clarified to inform the best choice of modelling approach. The work presented in this thesis addresses this research gap in part by synthesizing an inventory of wetland modelling approaches and providing a wetland classification scheme that allows easier integration of wetland types into hydrologic models based on ecological and hydrogeomorphic definitions.

Chapter 3

Inventory of Wetland Conceptual-Numerical Models

Following the establishment of wetland classifications and review of existing wetland modelling motivations and approaches, this thesis aims to provide an inventory of wetland modelling methods based on wetland types and relate their hydrologic characteristics to numerical models. These wetland modelling methods will herein be referred to as wetland conceptual-numerical models for the way they combine wetland classifications and hydrologic process equations. This chapter will introduce a combined ecological and hydrogeomorphic wetland classification scheme that simplifies wetlands into conceptual-numerical model categories based on a literature review of existing wetland modelling studies. Wetlands are typically defined in terms of ecological classifications, which are often not directly useful to modellers. The categorization of wetlands by conceptual-numerical model is intended to provide practical value to modellers representing wetlands in their study areas. The grouping of wetlands by similar hydrologic behaviour is more easily applied to models in terms of water balance than ecological characteristics. The literature highlighted in this Chapter provides modellers with wetland modelling applications and conceptual-numerical model category examples. The inventory focuses solely on water quantity modelling and does not address the additional complications of simulating water quality in wetland systems. In addition, the conceptual-numerical model categories mainly focus on compartment/standalone wetland models, spatially explicit, and equivalent lumped models. Some discussion on groundwater-surface water continuum models and spatially implicit probabilistic wetland models is provided, however is not the main focus of the conceptual-numerical model categories.

3.1 Wetland Conceptual-Numerical Models

As most wetland classification schemes are ecological in nature rather than hydrological, the identification of these ecological wetland types may not be directly useful for modellers looking to integrate wetland sites into their models. To establish a more unified classification scheme for wetland implementation in hydrologic models, ecological and hydrogeomorphic classifications defined in Section 2.1 are grouped into conceptual-numerical model categories based on similarities in hydrologic behaviour. Hydrogeomorphic wetland classifications (riverine, lacustrine, palustrine, estuarine, and isolated) are the broadest wetland category for the purposes of hydrologic modelling as

these types describe hydrologic connectivity of wetlands to other water features in the surface water network. Because they are based upon connectivity, hydrogeomorphic classifications are also the easiest to link to differences in water balance characteristics. Here, hydrogeomorphic classifications are linked to ecological classifications. Ecological wetland classifications, more commonly found in geospatial data sets, are typically of a finer granularity than the 5 hydrogeomorphic classifications. This is because they use ecological indicators to discriminate between wetlands which, in terms of hydrologic function, are not distinct. Figure 3.1 summarizes the hydrogeomorphic wetland types in relation to ecological wetland types.

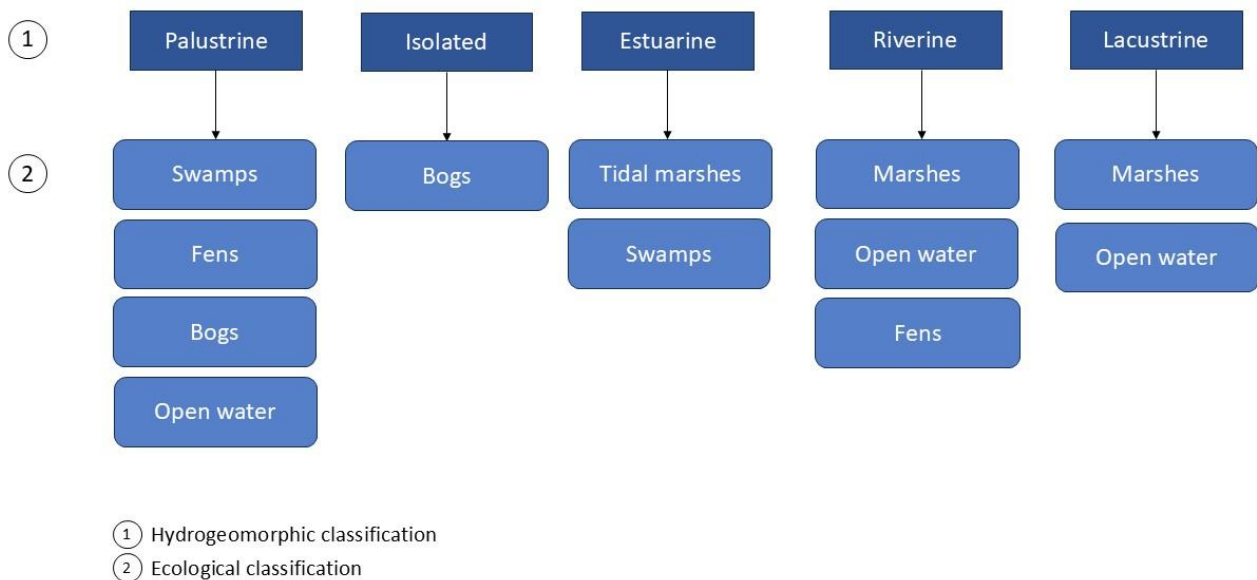


Figure 3.1. Hydrogeomorphic wetland classifications linked to ecological wetland classifications based on hydrological processes in common.

It is recognized that the merging of ecological wetland types (e.g., riverine marshes and riverine open water wetlands) into categories may lose some unique characteristics of individual wetland types. There is also the possibility of not every wetland site behaving as illustrated in Figure 3.1. However, rather than acting as a rigid framework, Figure 3.1 aims to provide a generalized starting point for wetland classification to be used in hydrologic modelling. Further modification to wetland characteristics is encouraged on a site-by-site basis.

A literature review was performed to inventory the various forms of wetland representations found in hydrologic modelling studies and subsequently categorize into six distinct conceptual-numerical model categories: riparian wetlands, lake fringe wetlands, geographically isolated wetlands, separated fill-and-spill wetlands, ombrogenous wetlands, and seasonally flooded wetlands. Amendments to the conceptual-numerical models may be used for forested wetlands or permafrost-underlain wetlands. Both forested and permafrost-underlain types are not conceptual-numerical model categories themselves, as amendments can be made to any conceptual-numerical category to support forested or non-forested and permafrost-underlain or non-permafrost-underlain features. The conceptual-numerical model categories were identified during the literature search process based on the unique wetland model types encountered in modelling studies and were assigned names representative of their hydrologic behaviour.

The conceptual-numerical wetland model categories were then related to the hydrogeomorphic and ecological classification schemes based on similarities in hydrologic behaviour. Between all conceptual-numerical model categories, the inventory is intended to include examples of the previously discussed wetland modelling methods (standalone wetland, implicit methods, explicit methods, equivalent lumped, and groundwater-based). Some conceptual-numerical model categories contain blurry boundaries between other ecological, hydrogeomorphic, and other conceptual-numerical classifications due to the variability of site-specific conditions. However, other than special case wetlands (with behaviour divergent from known behaviour of similar wetland types), the wetland conceptual-numerical model categories were related to ecological and hydrogeomorphic classifications by comparing hydrologic characteristics described by literature. Table 3.1 provides a thorough inventory of wetland modelling studies found in literature used to inform the wetland conceptual-numerical modelling methods, along with the logic behind each wetland classification. A total of 53 studies were included in this inventory. A summarization and discussion of each wetland conceptual-numerical model category follow

Table 3.1 Inventory of Wetland Modelling Studies in Literature

Literature Reference	Hydrogeomorphic Type	Ecological Type	Wetland Modelling Method	Wetland Conceptual-Numerical Model	Rationale/Notes
(Ahmed et al., 2020, 2023)	Palustrine	Open water	Spatially implicit (HYPR, HDS)	Separated fill-and-spill	Prairie Potholes, no indication of ecological type. Probabilistic method for representing wetlands.
(Alizad et al., 2016)	Estuarine	Marsh	Standalone wetland (Hydro-MEM)	Seasonally flooded	Coastal salt marsh in Florida. Hydraulic water level model only modelling wetland area.
(Al-Khudhairi et al., 1999; Thompson, 2004)	Riverine/Estuarine	Marsh	Spatially explicit (MIKE SHE)	Riparian/Seasonally flooded	Kent Elmley Marshes, drained inland marsh beside coastal inlet. Wetland land cover mapped.
(Ameli & Creed, 2019)	Palustrine/Riverine	Multiple	Groundwater-based (HydroGeoSphere)	GIW/Riparian	Focus of study is riparian vs. non-riparian wetlands. No wetland ecology indicated. Assumed to be multiple types. 3D groundwater-surface water interaction model.
(Baird et al., 2008, 2012; Morris, 2010)	Palustrine	Bog	Groundwater-based/Standalone wetland (Digibog)	Ombrogenous	Domed bog is the focus of the study. Digibog model was built for modelling individual domed bogs and has built in MODFLOW components.
(Blanchette et al., 2019)	Palustrine/Riverine	Multiple	Spatially explicit (HYDROTEL)	GIW/Riparian	Study distinguishes between geographically isolated and riparian wetlands. No wetland ecology indicated. Assumed to be multiple types. Wetland land cover mapped.

(Bleuten et al., 2020)	Palustrine	Bog	Groundwater-based (MODFLOW)	Ombrogenous	Raised bog complex in Siberia. Groundwater-focused modelling and analysis.
(Bradley, 1996)	Riverine	Marsh ¹	Groundwater-based (MODFLOW)	Riparian	Modelled wetland is described as a floodplain wetland with marshland vegetation. Groundwater-focused modelling and analysis.
(Chescheir et al., 2008)	Palustrine	Swamp	Standalone wetland (DRAINMOD)	GIW (unconfined)	Study notes palustrine, forested wetland in site description. Only the forested wetland is modelled.
(Evenson et al., 2015)	Palustrine	Swamp	Spatially explicit (SWAT)	GIW (unconfined)	Focus of study is geographically isolated wetlands. Study describes wetlands as hardwood forested wetlands and depressional swamps. Includes spatial mapping of wetlands and wetland HRU discretization.
(Evenson et al., 2018)	Palustrine	Open water	Spatially explicit (SWAT)	Separated fill-and-spill (confined)	Focus of study is fill-and-spill geographically isolated wetlands (Prairie Potholes). No indication of wetland ecology. Includes spatial mapping of wetlands and wetland HRU discretization.
(Fossey et al., 2015)	Palustrine/Riparian ²	Swamp/Fen/Bog	Spatially explicit (HYDROTEL)	GIW (unconfined)/Riparian	Study distinguishes between geographically isolated and riparian wetlands. Ecological types specified. Uses Hydrologic Equivalent Wetland (HEW) concept.

(Frei et al., 2010)	Riverine	Fen	Groundwater-based (HydroGeoSphere)	Riparian	Focus of study is riparian fen in German catchment. Uses surface-subsurface integrated model.
(Geurink et al., 2007)	Riverine/Palustrine	Multiple	Groundwater-based (MODFLOW/IHM)	Riparian/GIW	Study distinguishes wetland connection to streams and lakes. Mentions forested, open water, and marsh wetlands. Uses surface-subsurface integrated model.
(Giraud et al., 1997)	Riverine/Estuarine	Marsh	Standalone wetland/ Groundwater-based (SIDRA)	Riparian/Seasonally flooded	Coastal marsh on the Atlantic coast of France. Agricultural channel routing through the wetland.
(Gleason et al., 2007)	Palustrine	Open water	Spatially explicit	Separated fill-and-spill	Prairie Potholes with no ecological types indicated. Wetlands mapped with National Wetlands Inventory dataset.
(Golden et al., 2014)	Palustrine	Multiple ³	Multiple	GIW (confined and unconfined)	Focus of the study is geographically isolated wetlands. Review paper with various studies and model types.
(Grapes et al., 2006)	Palustrine/Riverine	Swamp/Marsh/ Fen	Groundwater-based (MODFLOW)	GIW/Riparian	Study distinguishes wetland connection to streams and lakes. Wetland ecology mentioned. Groundwater focused modelling and analysis.
(Harvey et al., 1987)	Estuarine	Marsh	Standalone wetland (FORTRAN)	Seasonally flooded	Chesapeake Bay coastal marsh. Modelling of marsh system alone.

(Hopkinson & Day, 1980)	Palustrine	Swamp	Equivalent lumped (EPA SWMM RECEIVE)	GIW (confined)	Upland swamp in Louisiana. Wetland routing is lumped.
(Krasnostein & Oldham, 2004)	Palustrine	Open water	Standalone wetland (bucket model)	GIW (unconfined)	Australian depressional wetland. Bucket model mimics depressional wetland.
(Lapen et al., 2005)	Palustrine	Bog/Fen	Groundwater-based (FLOTRANS)	GIW (unconfined)	Focus of study is upland blanket peat complexes. 2D steady state finite element groundwater modelling.
(Lee et al., 2018)	Palustrine/Riverine	Multiple	Spatially explicit (SWAT)	GIW (unconfined)/ Riparian	Distinguishes between geographically isolated and riparian wetlands. No wetland ecology indicated, assumed to be multiple types. Includes spatial mapping of wetlands and wetland HRU discretization.
(Liu et al., 2008)	Palustrine/Riverine	Multiple ⁴	Spatially explicit (SWAT)	GIW (unconfined)/ Riparian	Distinguishes between geographically isolated and riparian wetlands. Forested wetlands mentioned, assumed to be multiple types. Includes spatial mapping of wetlands and wetland HRU discretization.
(Mackin et al., 2017)	Palustrine	Bog	Standalone wetland	Ombrogenous	Focus of study is raised bogs with modelling of peat accumulation.

(Mansell et al., 2000)	Palustrine	Swamp	Standalone wetland (VS2DT)	GIW (unconfined)	USGS-developed model of upland pine forest swamp.
(McKillop et al., 1999)	Palustrine	Swamp	Standalone wetland	GIW (unconfined)	Treed headwater swamp in Ontario watershed. Finite difference model with wetland routing only.
(Mekonnen et al., 2014)	Palustrine	Open water	Spatially implicit (PDMROF)	Separated fill-and-spill	Prairie Potholes with no indication of ecological type. Probabilistic method for representing wetlands.
(Mishra et al., 2010)	Lacustrine/Riverine	Marsh/Open water	Spatially explicit (VIC) ⁵	Lake fringe	Wetlands simulated along Great Lakes described as agricultural ponds. All ponds and wetlands mapped.
(Montalto et al., 2002, 2007)	Estuarine	Marsh	Standalone wetland/ Groundwater-based	Seasonally flooded	Tidal marshes modelled using groundwater equations.
(Mozafari et al., 2023)	Isolated/Palustrine/ Riverine	Bog/Fen/Swamp	Standalone wetlands/ Groundwater-based ⁶	Ombrogenous/GIW/ Separated fill and spill/ Riparian	Study compares peatland models.
(J. Pomeroy et al., 2010; J. W. Pomeroy et al., 2014)	Palustrine	Open water	Equivalent lumped (PHM)	Separated fill-and-spill	Prairie Potholes with no indication of ecological wetland type. Individual wetlands subset into lumped wetland and modelled using statistical connectivity relationships.

(Sampath et al., 2015)	Palustrine	Fen	Groundwater-based	Separated fill-and-spill	Geographically isolated fen with focus of the study on modelling groundwater sources to the fen.
(Shook & Pomeroy, 2011; Shook et al., 2013, 2021)	Palustrine	Open water	Spatially implicit, equivalent lumped (WDPM, PCM)	Separated fill-and-spill	Prairie Potholes with no indication of ecological type. Probabilistic method for representing wetlands.
(Taheri et al., 2023)	Palustrine	Bog/Fen/Open water	Spatially implicit (UWFS)	Separated fill-and-spill	Cascading water features in the prairies, Taiga plains, and Boreal regions. Probabilistic method for representing wetlands.
(Temmerman et al., 2005)	Estuarine	Marsh	Standalone wetland (Delft3D)	Seasonally flooded	Tidal marsh in the Netherlands with modelled water levels.
(Vining, 2002; Viger et al., 2010)	Palustrine	Swamp/Open water	Equivalent lumped (PRMS)	Separated fill-and-spill (unconfined)	Surface depression modelling of swamp and agricultural ponds. Wetlands are lumped by subbasin, with one wetland HRU per subbasin.
(Von Christierson et al., 2017)	Riverine	Marsh/Open water	Standalone wetland/spatially explicit (MIKE SHE)	Riparian	Study focuses on riparian floodplain wetlands.
(Wang et al., 2008)	Palustrine/Isolated	Bog/Fen	Spatially explicit (SWAT)	GIW	Study area contains channel fens, flat bogs, and peat plateaus. Includes spatial mapping of wetlands and wetland HRU discretization.

(Wen et al., 2013)	Riverine	Marsh	Spatially explicit (MIKE FLOOD/MIKE 11)	Riparian	Macquarie Marshes, Australia (floodplain wetland). Wetland storages included in model are based on spatial mapping of wetlands.
(Wilsnack et al., 2001)	Palustrine	Open water	Groundwater-based (MODFLOW)	GIW (unconfined)	Wetlands described to be hydrologically connected to aquifer with agricultural ponds influenced by regulated canals. Groundwater-focused modelling and analysis.
(Wu et al., 2023)	Palustrine/Riverine	Multiple	Spatially explicit (PHYSITEL/HYDR OTEL)	GIW (unconfined)/ Riparian	Distinguishes between geographically isolated and riparian wetlands. No wetland ecology indicated, assumed to be multiple types. Includes spatial mapping of wetlands and wetland HRU discretization.
(Zacharias et al., 2004)	Lacustrine	Fen ⁸	Spatially explicit (MIKE SHE)	Lake fringe	Calcareous fen located along southern edge of Trichonis Lake, Greece. Wetland storages included in model are based on spatial mapping of wetlands.
(Zelevnik & Globevnik, 2014)	Riverine	Marsh	Spatially explicit (MIKE FLOOD/MIKE 11)	Riparian	Tributaries flow through studied wetland. Studied wetland described as marshland. Distributed modelling of wetland areas based on spatial mapping.

¹ Bradley, 1996 refers to the wetland type as a bog due to the assigned name “Narborough Bog”, however describes marshland vegetation and notes the wetland is connected to the floodplain.

² Fossey et al., 2015 models both geographically isolated and riparian wetlands using HRUs and lumping by Hydrologic Equivalent Wetlands (HEWs) (Wang et al., 2008).

³ Golden et al., 2014 reviews a number of modelling methods for GIWs and provides example references.

⁴ Liu et al., 2008 models both geographically isolated and riparian wetlands by creating a SWAT routine for riparian wetlands and classifying wetlands based on connectivity to the modelled stream network. The study mentions all ecological wetland types present within the study area.

⁵ Mishra et al., 2010 treat all wetlands and lakes the same; as VIC water body grid cells. Each water body is spatially mapped within the study area.

⁶ Mozafari et al., 2023 provides a comparison of different peatland models and provides peatland model references for bogs, fens, and swamps, which are predominantly treated as standalone or groundwater models.

⁷ Von Christierson et al., 2017 built a MIKE SHE model for the river and floodplain domain, and a separate MIKE SHE model for just the riparian wetland extents.

⁸ Zacharias et al., 2004 models a catchment in Greece containing calcareous fen habitat (a rare type of wetland) along Trichonis Lake. This is a rare case of a fen habitat being a lake fringe wetland. Calcareous fens are typically located in areas with upwelling of groundwater (Minnesota Department of Natural Resources, n.d.).

The conceptual-numerical wetland modelling categories presented in Table 3.1 are further summarized in Figure 3.2 with respect to the ecological and hydrogeomorphic wetland classification schemes.

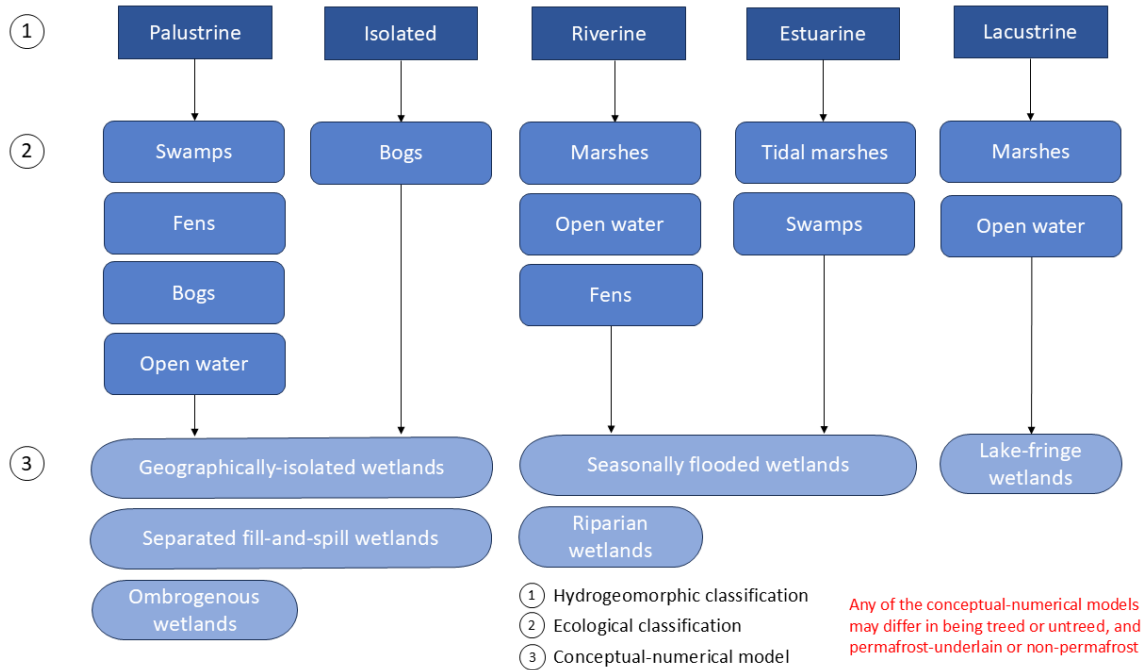


Figure 3.2. Hydrogeomorphic and ecological wetland classification summarized into wetland conceptual-numerical model categories based on literature review and example wetland hydrologic models. (1) Hydrogeomorphic classifications refer to wetland types based on their position and connection to other waterways in the landscape, (2) ecological classifications refer to wetland types based on their soil and vegetation properties, and (3) conceptual-numerical models refer to wetland modelling methods identified by the combination of (1) and (2) wetland classifications and hydrologic process equations used in literature.

The conceptual-numerical model categories shown in Figure 3.2 were created by identifying common hydrologic characteristics of modelled wetland types in the literature presented by Table 3.1. For example, many existing wetland modelling studies make distinctions between GIWs and riparian wetlands based on their hydrologic connection in the watershed (Golden et al., 2014; Evenson et al., 2015; Fossey et al., 2015; Lane & D’Amico, 2016). Another example is Prairie Pothole models that include cascading of water (fill-and-spill behaviour) to represent wetlands that are separated until overflow dynamics connect individual wetlands (Ahmed et al., 2020, 2023; Taheri et al., 2023). Areas

in which the categories in Figure 3.2 become fuzzy include the bubbles shown to span multiple ecological and hydrogeomorphic classifications. These conceptual-numerical model categories that span multiple classifications can be used to describe the hydrologic behaviour of different ecological and hydrogeomorphic wetland types. At a large watershed scale, variation of hydrologic behaviour within any conceptual-numerical category based on fuzzy ecological or hydrogeomorphic classification boundaries are expected to be minimal. Moderately to large sized watersheds will likely not produce noticeably different results if using one conceptual-numerical model category for two different ecological wetland types (e.g., marsh and fen both using a riparian wetland conceptual-numerical model). Additional care may need to be taken with the variation of hydrogeomorphic types within one conceptual-numerical model category. For instance, modelling palustrine and isolated bogs both using a geographically isolated wetland conceptual-numerical model may require the palustrine bog to have a groundwater connection and the isolated bog to have no groundwater connection. However, with regional (large scale) models and the use of wetland mapping products, it is often difficult to distinguish between ecological and hydrogeomorphic features let alone specific hydrologic differences between two of the same wetland type. Smaller sites or studies modelling a standalone wetland may experience a greater need for distinguishing between ecological and hydrogeomorphic classifications.

The following subsections describe each of the six wetland conceptual-numerical model categories and two additional configurations, along with examples of model implementation from literature. It should be noted that these wetland conceptual-numerical models are likely not a complete inventory of all possible wetland classifications. It is also recognized that there are many wetland systems that could be simulated by more than one conceptual-numerical model category. In these cases, any of the conceptual-numerical models could be modified as a combination of multiple types to suit the study area. Indeed, many of the literature examples also fall into multiple categories, and thus these conceptual-numerical model categories are intended to be a flexible abstraction that helps modellers to better understand wetland systems and provide an inventory of wetland modelling references for the reader to review. The literature reviewed in this thesis is a thorough inventory of available wetland modelling papers to the author's knowledge, although there may be additional studies that are also applicable.

3.1.1 Riparian Wetlands

Riparian wetlands are located on the boundary edge of rivers and streams and thus their water levels are dictated by water levels in the stream. For this reason, they can be seen as hydrologically connected to the stream network in hydrologic models and thus also classified as riverine wetlands (Hattermann et al., 2006). Ecological wetland types related to riverine and riparian wetlands include marshes, open water wetlands, and fens, all of which can transport water via stream channel flow as modelled in numerous studies (Bradley, 1996; Giraud et al., 1997; Al-Khudhairi et al., 1999; Quinton et al., 2003; Geurink et al., 2007; Frei et al., 2010; Wen et al., 2013; Zeleznik & Globevnik, 2014; Von Christierson et al., 2017). For hydrologic modelling purposes, riparian wetlands are considered a part of the floodplain and water levels are the same as the stream channel. An example diagram of a riparian wetland is shown in Figure 3.3, along with its associated hydrologic processes.

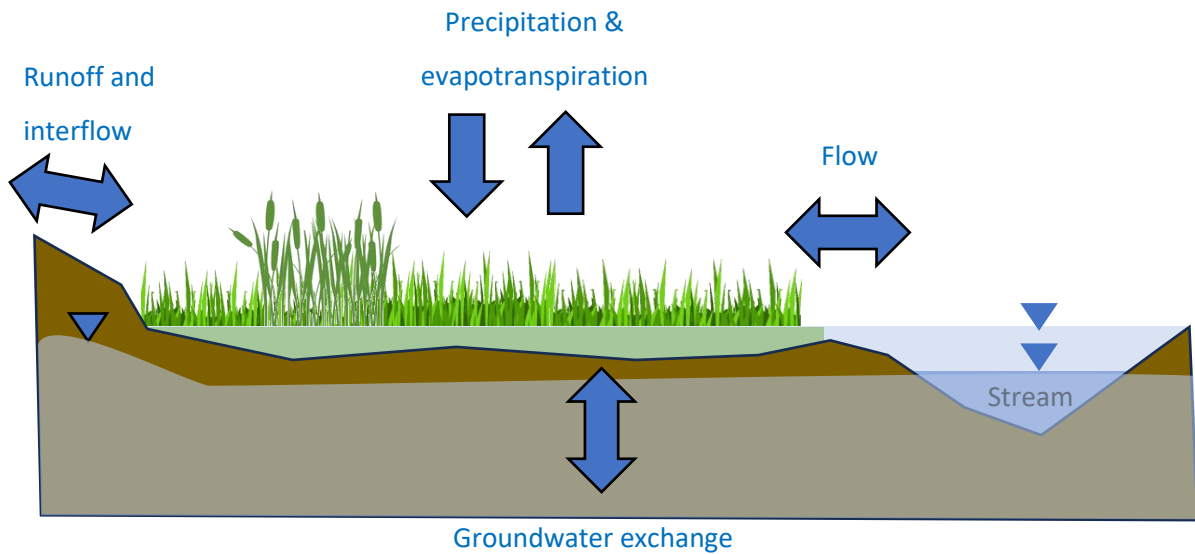


Figure 3.3. Conceptual-numerical model of a riparian wetland.

The water balance equation for riparian wetlands can thus be written as Equation 3.1:

$$\frac{dS}{dt} = P - ET \pm R \pm G \pm F \quad (3.1)$$

where dS/dt is the change in wetland storage [mm/d], P is the precipitation [mm/d], ET is evapotranspiration [mm/d], R is overland runoff and/or interflow [mm/d], G is groundwater recharge or discharge [mm/d], and F is the flow between the stream and the wetland [mm/d].

Due to the fluctuation of water levels supplied by the adjacent streams, the water level in riparian wetlands is also subject to change throughout the year. It is possible for the wetland to dry out during periods of low streamflow in the adjacent streams, and conversely flood during wet seasons (Giraud et al., 1997; Al-Khudhairi et al., 1999; Frei et al., 2010; Wen et al., 2013; Zeleznik & Globevnik, 2014). Water flowing through riparian wetlands is slower than water flowing through the stream due to increased roughness in vegetated areas. Thus, riparian wetlands can be represented in hydrologic models using increased Manning's roughness values and increased evapotranspiration within the riparian zone as performed in studies modelling riparian marshes (Wen et al., 2013) and riparian fens (Quinton et al., 2003; Frei et al., 2010). Frei et al. (2010) concludes that the non-linear routing of water between wetlands and channels can result from microtopographies created by soil and vegetation types, riparian groundwater interaction, and residence time of water travelling slower through riparian wetlands. Considering this, one strategy for parameterization of riparian wetlands is to classify riparian wetland areas as part of the stream network and adjust Manning's roughness and evapotranspiration rates separately from other land cover and wetland types (Frei et al., 2010).

Studies modelling riparian wetlands have also made use of coupled hydrologic-hydraulic and surface-subsurface models and routines such as MIKE FLOOD, MIKE SHE, MIKE 11, SIDRA-MAGE, and SWIM to capture both floodplain and groundwater dynamics (Giraud et al., 1997; Al-Khudhairi et al., 1999; Frei et al., 2010; Zeleznik & Globevnik, 2014; Von Christierson et al., 2017). For example, Von Christierson et al. (2017) used a coupled hydraulic and hydrologic modelling approach to simulate a riparian wetland in Denmark. MIKE SHE was used to represent two-dimensional diffusive overland flow, while MIKE 11 was used to represent floodplain flows, channel flows, and finite difference groundwater flows. A separate wetland model using MIKE SHE was then created to represent underlying peat material of the riparian wetland and dynamic boundary conditions dependent on the floodplain and stream channel model (Von Christierson et al., 2017). Al-Khudhairi et al. (1999) also made use of MIKE SHE and MIKE 11 to model a riparian marsh in the UK, arguing that semi and fully-distributed models like MIKE SHE, although containing parameters and complex calibration requirements, offer a more accurate representation of riparian wetland processes (specifically in the subsurface) compared to lumped modelling approaches or conceptual water

balance equations (Al-Khudhairy et al., 1999). Between these modelling studies, groundwater representation and wetland water level fluctuations based on river modelling is a common modelling approach used to capture the role of riparian wetlands acting as part of the floodplain.

3.1.2 Lake Fringe Wetlands

Lake fringe wetlands are located on the boundary edge of lakes and can be extensions of the lake body with characteristic wetland vegetation (hydrophytic vegetation). Water levels in lake fringe wetlands are equilibrated with lake levels and for modeling purposes, are considered the same as lake levels. An example diagram of a lake fringe wetland is shown in Figure 3.4, along with its associated hydrologic processes.

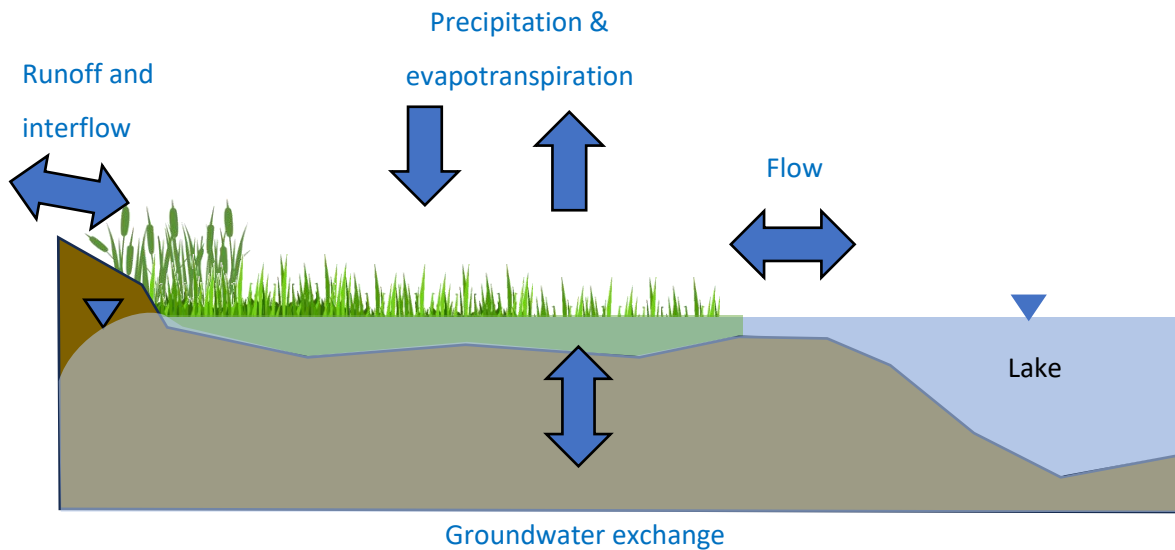


Figure 3.4. Conceptual-numerical model of a lake fringe wetland.

The water balance equation for lake fringe wetlands can thus be written as Equation 3.2:

$$\frac{dS}{dt} = P - ET \pm R \pm G \pm F \quad (3.2)$$

where dS/dt is the change in wetland storage [mm/d], P is the precipitation [mm/d], ET is evapotranspiration [mm/d], R is overland runoff and/or interflow [mm/d], G is groundwater recharge or discharge [mm/d], and F is the flow between the lake and the wetland [mm/d].

Few studies exist that explicitly model lacustrine wetlands, likely because in many cases they can be simulated as a vegetated extension of the lake. Albert et al. (2005) suggests that lacustrine wetlands existing in large lake environments, such as offshore from the Great Lakes, are subject to high energy wave action that destroys wetland vegetation and erodes soils (Albert et al., 2005). Since lake fringe wetlands are effectively extensions of the lake, and are subject to dynamic boundaries, treating them as such in hydrologic models means assuming they are part of the simulated lake. In other words, lake fringe wetlands are assumed to be contained within the modelled lake extents and receive no further treatment. A rare case is observed in Zacharias et al. (2004), where a calcareous fen exists along the lakeshore of Trichonis Lake in Greece. Calcareous fens are rare habitats typically located in areas with groundwater upwelling (Minnesota Department of Natural Resources, n.d.). Zacharias et al. (2004) explicitly includes the lakeside wetland area in their MIKE SHE model of the catchment, as the purpose of the study is to make comparisons between remotely sensed landcover data over time. In addition, the wetland areas within the catchment, despite being part of the lake system, are ecologically important habitats that were the focus of the study. The Zacharias et al. (2004) study is a unique example of a lacustrine wetland modelled explicitly because of the modelling purpose. Thus, for watershed models only interested in representing the wetland hydrology, lacustrine wetlands are assumed to be contained within the modelled lake extents and not treated explicitly.

3.1.3 Geographically Isolated Wetlands

Geographically isolated wetlands (GIWs) are palustrine wetlands that are not directly connected to the stream network via surface drainage but are presumed to have subsurface hydraulic connections and can receive overland flow and interflow from surrounding uplands or emit interflow downstream (Evenson et al., 2018; Golden et al., 2014). Ecological wetland classes within this category can include swamps, open water, fens, and some types of bogs, which can be headwater wetlands without any direct connection to the stream network. Two types of GIWs exist – unconfined and confined GIWs. Unconfined GIWs are underlain by permeable soil, allowing groundwater exchange to occur, while confined GIWs are underlain by impermeable soil (permafrost, clay/silt), isolating the wetland

depression from the local groundwater (Golden et al., 2014). Figure 3.5 provides an example diagram of an unconfined GIW.

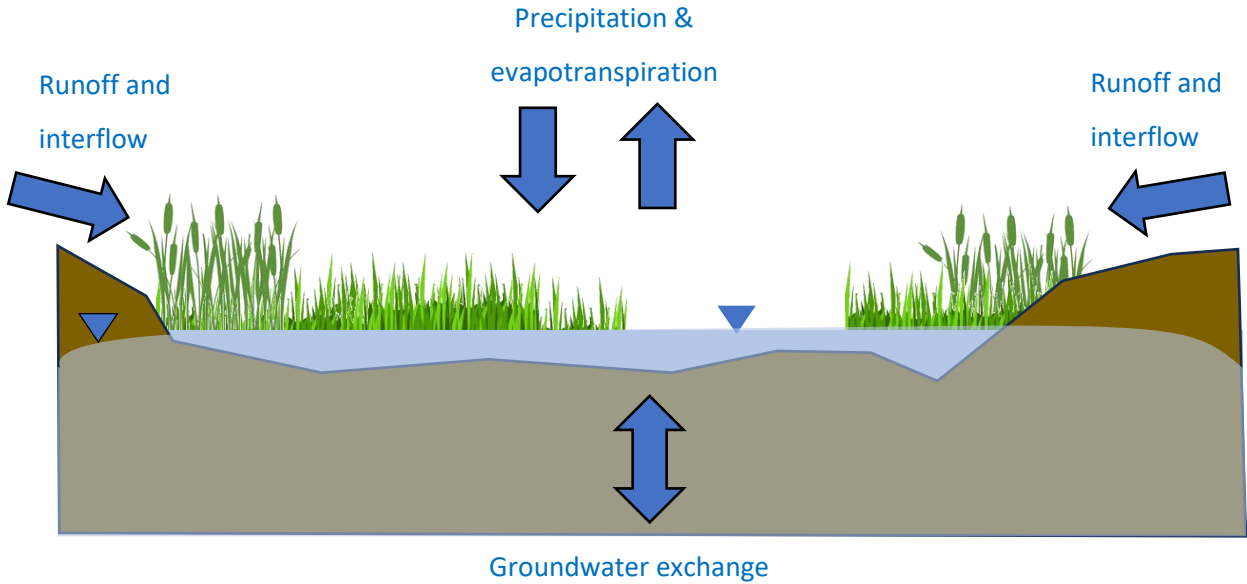


Figure 3.5. Conceptual-numerical model of an unconfined geographically isolated wetland

The water balance in unconfined GIWs is dominated by evapotranspiration losses, precipitation and overland runoff and/or interflow, but water can also be gained or lost through groundwater exchange. Similar water balance conceptual diagrams are described in literature (McKillop et al., 1999; Krasnostein & Oldham, 2004; Golden et al., 2014; Lane et al., 2018). From Figure 3.5, the water balance equation can be written as Equation 3.3:

$$\frac{dS}{dt} = P - ET + R \pm G \quad (3.3)$$

where dS/dt is the change in wetland storage [mm/d], P is the precipitation [mm/d], R is the overland runoff and/or interflow [mm/d], ET is evapotranspiration [mm/d], and G is groundwater recharge or discharge [mm/d].

An example diagram of a confined GIW, with an impermeable layer beneath the wetland, is shown in Figure 3.6.

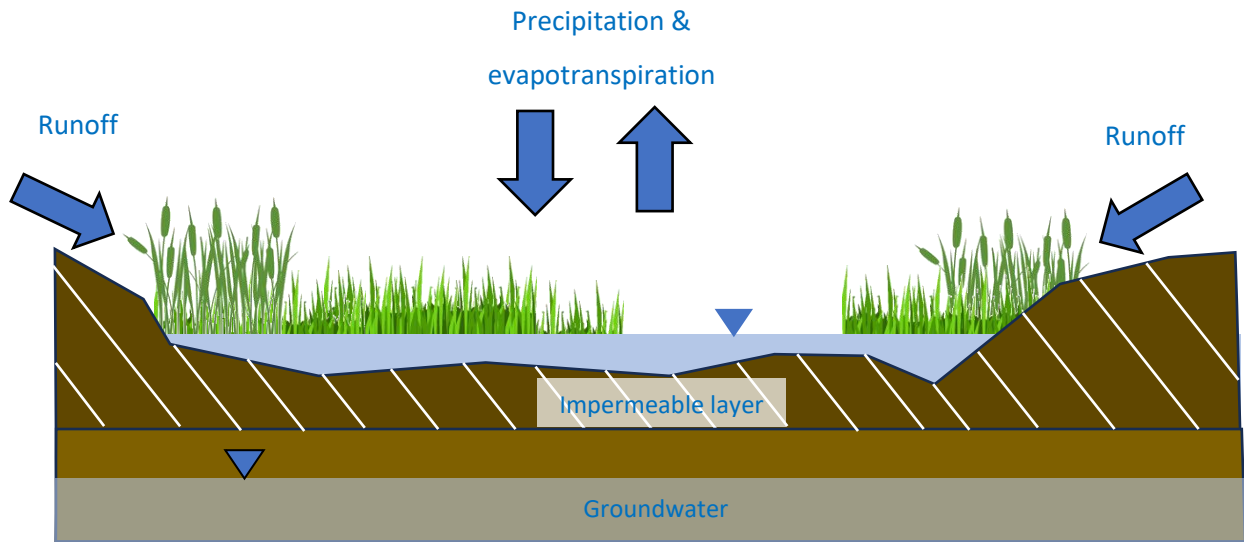


Figure 3.6 Conceptual-numerical model of a confined geographically isolated wetland

The water balance in confined GIWs is dominated by evapotranspiration losses, precipitation and overland inflows, but has limited to no groundwater exchange. From Figure 3.6, the water balance equation can be written as Equation 3.4:

$$\frac{dS}{dt} = P - ET + R \quad (3.4)$$

where dS/dt is the change in wetland storage [mm/d], P is the precipitation [mm/d], R is the overland runoff [mm/d], and ET is evapotranspiration [mm/d].

For implementation in hydrologic models, geographically isolated wetlands can be identified by their lack of intersection with any modelled streams or lakes. These areas of non-connectivity to the stream network or lakes can then be assumed to be geographically isolated and can be parameterized as such. Golden et al., 2014 recommends modelling GIWs using coupled surface-subsurface flow models due to groundwater-dominated hydrology in unconfined GIWs. In one example study, Sampath et al. (2015) applied groundwater modelling to a watershed containing predominately groundwater-fed geographically-isolated fen complexes (Sampath et al., 2015). In the Sampath et al. (2015) study, groundwater modelling was used to improve the understanding of groundwater interactions between the fen complex and surrounding aquifers. The study also included GIWs that were known to be

predominately groundwater-fed (Sampath et al., 2015). However, depending on the spatial scale and purpose of the hydrologic model (e.g., surface water focused case study versus groundwater focused case study), the use of both or either surface water/groundwater models may be used to represent GIWs (Golden et al., 2014). Parameterization of GIWs is dependent on characterizing dominant flow pathways, physical properties of the GIWs (volume, retention times, soil types), and GIW location mapping (Golden et al., 2014). Models such as the one presented in Evenson et al. (2015), use the SWAT model to explicitly represent Prairie GIWs with HRUs (Evenson et al., 2015). As noted by Evenson et al. (2015), spatially explicit modelling of GIWs requires the characterization of local contributing areas of each simulated wetland. Due to many GIWs being primarily groundwater-fed rather than surface-fed like other wetland conceptual-numerical model categories, it is important that the upgradient drainage area of the GIWs be represented. HRU boundaries on their own only spatially represent the wetland surface and are unable to capture additional contributing area fed by groundwater and topography. Thus, the local drainage areas of GIWs should be delineated with topography and used to represent upland surface and/or subsurface contributions (Evenson et al., 2015).

Standalone wetland modelling studies of upland headwater swamps also exist that represent GIW depression stores as unconfined GIWs connected to both surface and groundwater (Waddington et al., 1993; McKillop et al., 1999; McCarter et al., 2024). In one instance, McKillop et al. (1999) model an unconfined GIW using a series of wetland units (cells) connected to channel routing reaches that route water towards the wetland outlet. Both surface and subsurface flows are represented in water balance calculations.

Drawing on these studies, we learn that modelling GIWs as standalone wetlands requires the use of groundwater or coupled models. Conversely, for large scale watersheds or those including many wetlands, simulated GIWs are most often represented using spatially explicit methods.

3.1.4 Separated Fill-and-Spill Wetlands

Separated fill-and-spill wetlands are palustrine wetlands that have no directly adjacent connection to surface water (streams or lakes) during dry periods of the year, but are characterized by a tight groundwater connection and fill-and-spill behaviour (Spence & Woo, 2003) into adjacent wetlands when overtopping occurs. Separated fill-and-spill wetlands typically act as GIWs but have the capacity to become connected complexes when storage is high.

Separated fill-and-spill wetlands can include Prairie Potholes which exhibit fill-and-spill mechanics between wetlands (K. R. Shook & Pomeroy, 2011; K. Shook et al., 2013; Ahmed et al., 2020; K. Shook et al., 2021; Ahmed et al., 2023). Some fens, swamps, bogs, or open water wetlands may be characterized as separated fill-and-spill wetlands, as they can be tightly knit by groundwater connection and overtopping into adjacent wetlands. Northern wetland complexes, often dominated by fens and bogs, are also examples of fill-and-spill wetlands connected through overtopping and become strongly influenced by freezing and melting of soil water (Balliston & Price, 2022). An example diagram of a separated fill-and-spill wetland along with its hydrologic processes is shown in Figure 3.7.

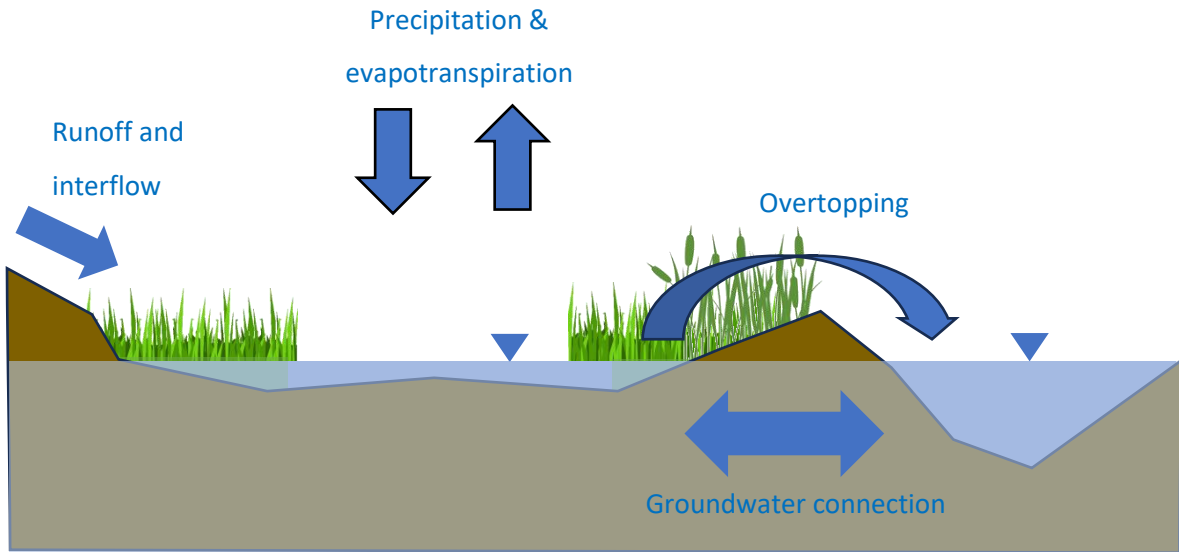


Figure 3.7. Conceptual-numerical model of a separated fill-and-spill wetland.

From Figure 3.7, the water balance equation can be written as Equation 3.5:

$$\frac{dS}{dt} = P + R - ET \pm G \pm O \quad (3.5)$$

where dS/dt is the change in wetland storage [mm/d], P is the precipitation [mm/d], R is the overland runoff and/or interflow [mm/d], ET is evapotranspiration [mm/d], G is the groundwater recharge or discharge [mm/d], and O is fill and spill overtopping of the wetland [mm/d].

Separated fill-and-spill wetlands are most often represented as wetlands complexes rather than individual wetlands due to their interannual connectivity. The water balance in separated fill-and-spill wetlands can be heavily influenced by the time of year. For example, some northern separated fill-and-spill wetland complexes are governed by disconnection and drying periods, with disconnection being more frequent during periods of freezing in the winter and drying periods being more frequent during the summer with higher evapotranspiration rates (Balliston & Price, 2022). Considering the complexity of interannual connectivity, the process of overtopping referred to in Figure 3.7 is rather challenging to quantify, especially in flat terrains with hundreds or thousands of separated wetlands. Overtopping of a wetland into another wetland is described as fill-and-spill behaviour and was first introduced by Spence & Woo (2003) to describe the behaviour of northern wetlands located in the Canadian Shield. Fill-and-spill has since also been used to describe similar behaviour observed in fen and Prairie Pothole complexes. The fill-and-spill dynamics within wetland complexes, where many wetlands fill and overtop into downstream wetlands, is called cascading. Figure 3.8 illustrates the fill-and-spill process between wetlands and the cascading effect of a wetland complex.

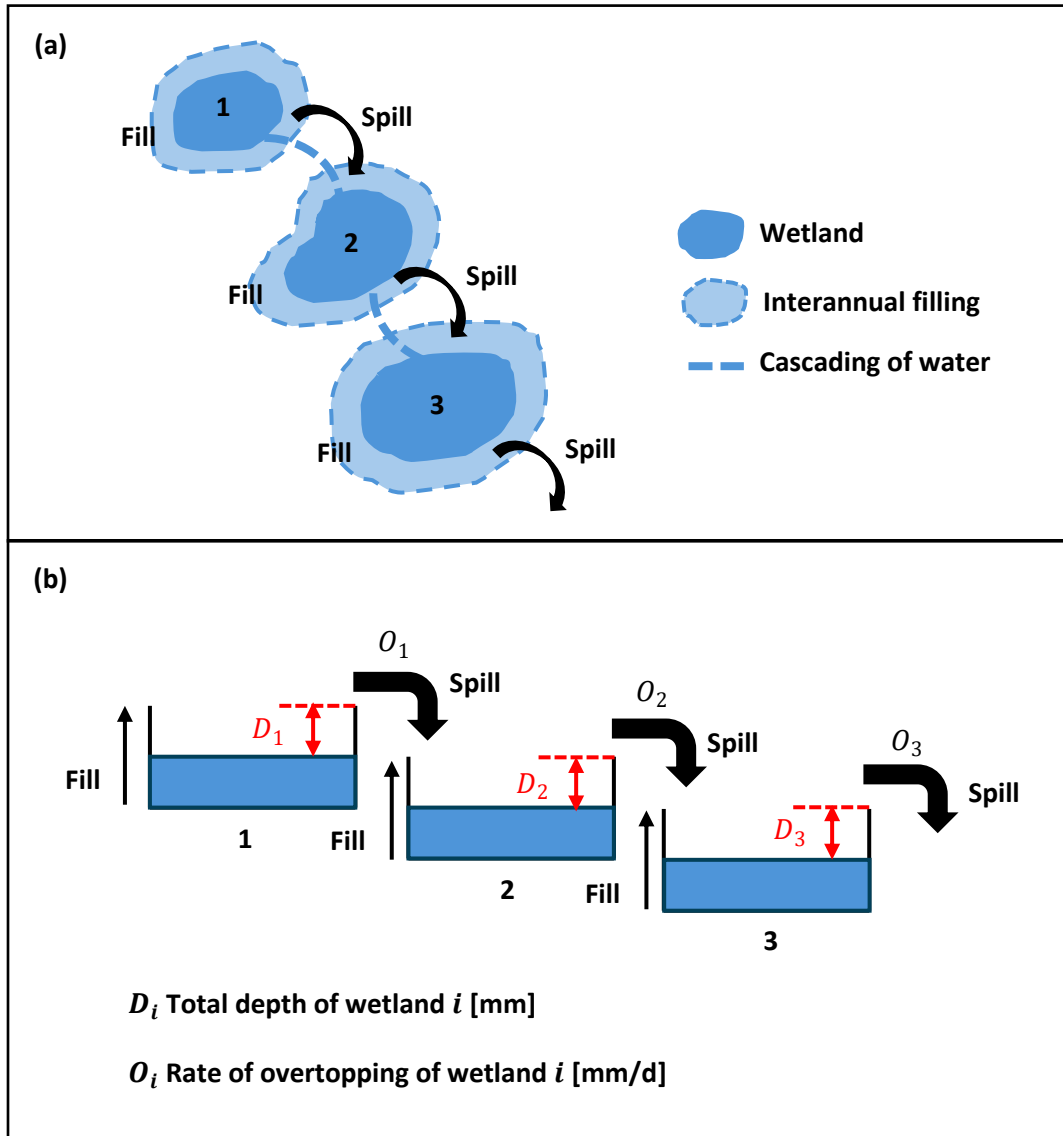


Figure 3.8 (a) Plan view of a wetland complex with interannual connectivity and cascading processes, and (b) conceptual diagram of wetland complex fill-and-spill dynamics (both figures adapted from Taheri et al. (2023))

Due to separated fill-and-spill wetland complexes containing hundreds or thousands of individual wetlands that are all interannually connected, they are typically modelled using spatially implicit methods. These algorithms characterize the wetland storage distribution as a probability distribution and solve for the net runoff and storage from the wetlands. These include the HYPR algorithm which

incorporates the Probability Distribution Model-based RunOff generation (PDMROF) method (Mekonnen et al., 2014; Ahmed et al., 2020). However, it has been argued that modelling Prairie Pothole depressions using probability distributions is not sufficient for representing individual depressions and their hydrologic processes (K. Shook et al., 2021). Fully distributed hydraulic models such as the Wetland DEM Ponding Model and Pothole Cascade Model (K. R. Shook & Pomeroy, 2011; K. Shook et al., 2013) have been used to simulate wetland-dominated Prairie landscapes and represent water fill and spill mechanics of thousands of individual wetlands, although their long computation time and simplification of hydrologic processes by lumping wetland storages together are well noted (K. Shook et al., 2021; Taheri et al., 2023). Another difficulty lies in determining the explicit connectivity of wetlands as separated fill-and-spill wetland connectivity changes interannually. Additional Prairie modelling algorithms such as the Upscaled Wetlands Fill-and-Spill (UWFS) method (Taheri et al., 2023) and Hysteretic Depressional Storage (HDS) algorithm (Ahmed et al., 2023) have aimed to better capture the cascading fill-and-spill effects of separated fill-and-spill wetland complexes.

As separated fill-and-spill wetlands behave similarly to GIWs for most of the year (when not in a spilling stage), separated fill-and-spill wetland complexes may be explicitly identified similar to GIWs by their non-connection to modelled streams and lakes. Individual wetland areas can then be parameterized using depression storage routines that are allowed to overtop or exhibit fill-and-spill behaviour for watersheds where the connection between wetlands is known or has been studied. For large scales or where many (hundreds or thousands) of separated fill-and-spill wetlands exist (e.g., the Prairies), spatially implicit modelling methods are used to deal with the difficulty of determining explicit connectivity of many wetlands over large areas.

3.1.5 Ombrogenous Wetlands

Ombrogenous wetlands are palustrine wetlands that only receive water through precipitation due to their peat soil forming a dome above the surrounding landscape. Ombrogenous wetlands lose water through evapotranspiration and lateral flow through the soil towards the edges of the wetland. The only ecological wetland class in this category is a domed bog. As discussed in Chapter 2, ombrogenous wetlands have two characteristic soil layers in which the acrotelm has a high hydraulic conductivity (K) and the catotelm has a low hydraulic conductivity (Ingram, 1978). The catotelm is

saturated and raised above the surrounding ground surface. An example diagram of an ombrogenous wetland is shown in Figure 3.9.

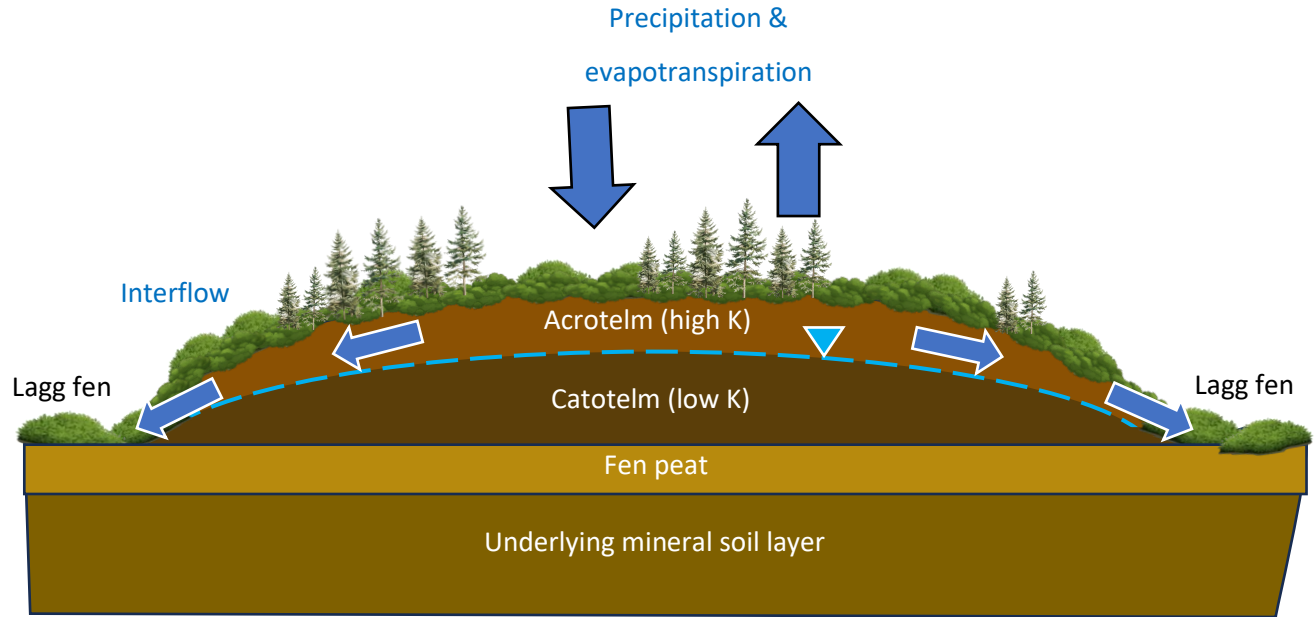


Figure 3.9. Conceptual-numerical model of an ombrogenous wetland.

From Figure 3.9, the water balance equation can be written as Equation 3.6:

$$\frac{dS}{dt} = P - ET - I \quad (3.6)$$

where dS/dt is the change in wetland storage [mm/d], P is the precipitation [mm/d], ET is evapotranspiration [mm/d], and I is interflow [mm/d].

Interflow through the sloped sides of the acrotelm leads to the lagg, which are buffer zones between the bog and surrounding land usually in the form of a fen (Bragg, 2002; Langlois et al., 2015). Water within the catotelm moves very slowly due to poorly decomposed material (Ingram, 1978), thus in hydrologic modelling it can be assumed that there is minimal connection to groundwater below. As a result of the domed water table, ombrogenous bogs tend to drain water and contribute baseflow to runoff hydrographs rather than attenuate large volumes of water (Bragg, 2002).

In standalone wetland models of ombrogenous bogs, water movement has been modelled using equations that describe the flow accumulation in the soil (Mackin et al., 2017) and horizontal

groundwater flow using Dupuit-Forchheimer (D-F) approximations (Freeze & Cherry, 1979; Ingram, 1987). Spatially distributed models, such as the hydrological submodel Digibog (Morris, 2010; Baird et al., 2012) have also been used to simulate water movement within the acrotelm using MODFLOW routines and D-F groundwater approximations. Groundwater modelling has also been performed in similar studies using MODFLOW, SWAT, and other coupled surface-subsurface models to model groundwater flow through domed bogs (Bleuten et al., 2020; Mozafari et al., 2023), although due to the presence of the impermeable catotelm layer, groundwater moves through the shallow subsurface rather than deep layers. The importance of hydraulic conductivity to water movement through domed bogs has also been highlighted in modelling and field studies, as the hydraulic conductivity can be transient and is highly influential on flow pathways (Lapen et al., 2005; Baird et al., 2008; Bleuten et al., 2020).

3.1.6 Seasonally Flooded Wetlands

Seasonally flooded wetlands are estuarine or riverine wetlands that are described by changing flow directions due to their variability in water levels with tides, wave action, or seasonal water level fluctuations (Temmerman et al., 2005). For this reason, ecological wetland classifications in this category include marshes, swamps, open water wetlands, and occasionally fens which can be seasonally dependent on streamflow. Tidal marshes and coastal swamps are included as estuarine wetlands which are seasonally or diurnally flooded by tides. An example diagram of a seasonally flooded wetland is shown in Figure 3.10.

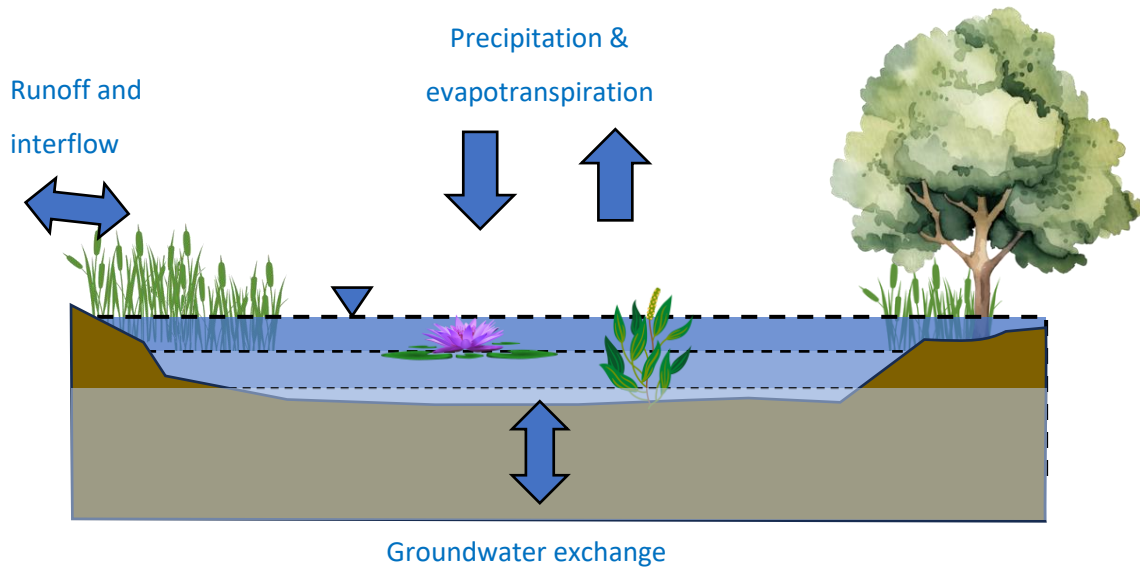


Figure 3.10. Conceptual-numerical model of a seasonally flooded wetland.

From Figure 3.10, the water balance equation can be written as Equation 3.7:

$$\frac{dS}{dt} = P - ET \pm R \pm G \quad (3.7)$$

where dS/dt is the change in wetland storage [mm/d], P is the precipitation [mm/d], ET is evapotranspiration [mm/d], R is the runoff and/or interflow [mm/d] and G is the groundwater recharge or discharge [mm/d].

Depending on the season or time of day, the wetland may be completely saturated or dry. Fluctuating water levels are mostly dependent on groundwater connection to adjacent water bodies and overland flow. Between different types of models used to simulate seasonally-flooded wetlands such as tidal marshes and swamps, shallow groundwater flow using Boussinesq's equation and Dupuit-Forchheimer approximations are used to simulate fluctuating water levels (Harvey et al., 1987; Giraud et al., 1997; Al-Khudhairi et al., 1999; Montalto et al., 2002; Thompson, 2004; Montalto et al., 2007). Seasonally flooded wetlands are typically modelled using hydraulic models capable of predicting water levels. Coupled hydrologic/hydraulic models such as MIKE SHE, MIKE 11, and Hydro Marsh Equilibrium Model (Hydro-MEM) have been used in studies that deploy spatially-explicit and three-dimensional groundwater and lateral soil water movement calculations that predict the seasonal and diurnal flooding of each modelled grid cell (Al-Khudhairi et al., 1999; Thompson, 2004; Alizad et al., 2016). Modelling of coastal marsh and swamp seasonal flooding has also been

performed using coupled vegetation models such as the swamp individual-based tree model (swamp IBM) and Delft3D model to account for plant growth and three-dimensional flow pathways created by plant structures and vegetation coverage (Temmerman et al., 2005; Hoeppepner & Rose, 2011).

3.1.7 Forested Wetlands

Forested wetlands are a special category of conceptual-numerical model that is an amendment to the previously mentioned categories, particularly GIWs. Most often, forested wetlands are palustrine, estuarine, or isolated wetlands that are characterized by a greater percentage of forest cover than open water. Swamps are characterized by more than 30% woody vegetation cover, meanwhile other wetlands like fens and bogs can include treed types. Flooding can be seasonal in coastal swamps, and in temperate regions many treed wetlands are headwater regions dominated by groundwater discharge (McKillop et al., 1999). An example diagram of a forested wetland as an amendment to a GIW is shown in Figure 3.11, along with its hydrologic processes.

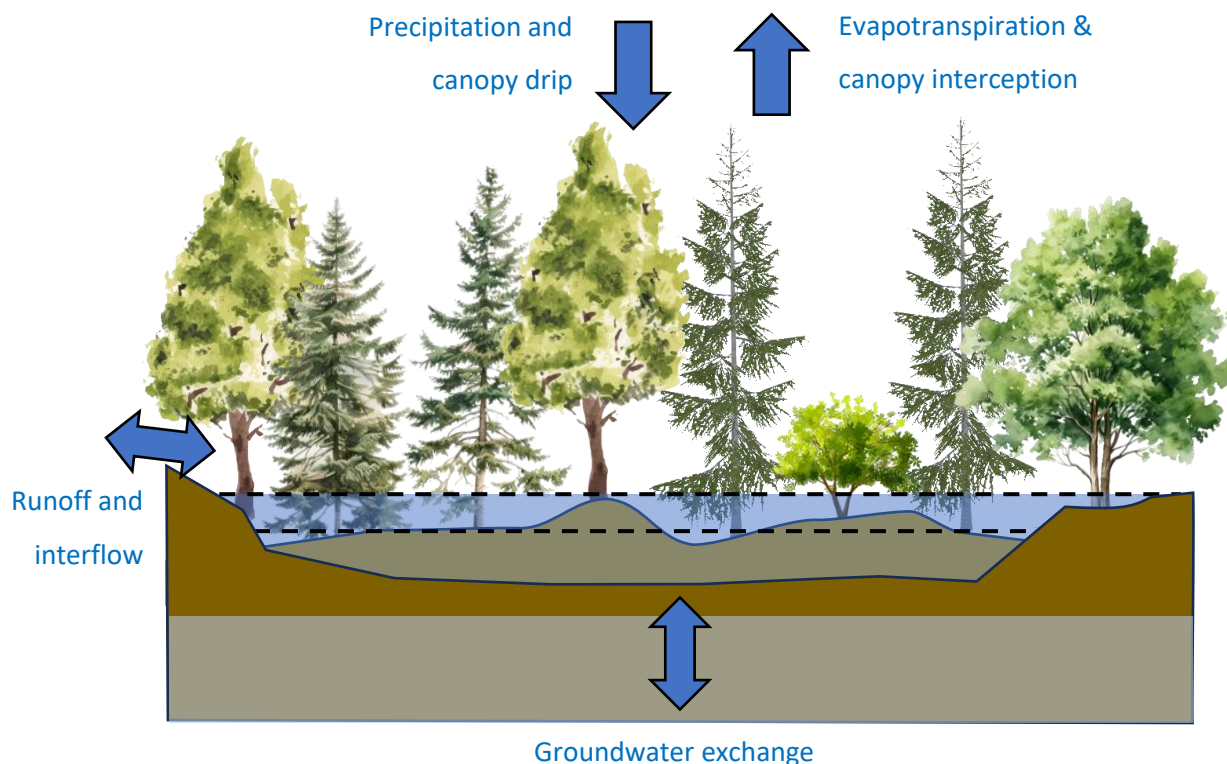


Figure 3.11. Diagram of a forested wetland example added to an unconfined GIW.

A water balance equation is not defined for forested wetlands as they can be augmented to different types of conceptual-numerical models. For instance, Figure 3.11 shows a forested unconfined GIW, however confined GIWs (with little or no groundwater interaction) or any of the other conceptual-numerical models can also be forested.

The key changes to wetland conceptual-numerical models when added with the forested wetland category include consideration for canopy interception, evapotranspiration, canopy drip, and shading effects. Forested wetlands, due to tree and shrub cover, experience a greater amount of canopy interception than unforested wetlands, and thus experience a larger magnitude of evapotranspiration rather than open water evaporation compared to unvegetated wetlands. Studies such as in Zacharias et al. (2004) found that by modelling a catchment with vegetated and forested fen complexes, an increased evapotranspiration rate was observed compared to other landcover types (Zacharias et al., 2004). Chescheir et al. (2008) also found that the majority of water loss (86%) occurred as evapotranspiration in a DRAINMOD model of a forested wetland in North Carolina, USA, highlighting the importance of vegetation abundance in forested wetlands (Chescheir et al., 2008). Forested wetlands, such as headwater swamps, have been modelled in studies using finite difference models that solve groundwater differential equations (Hopkinson & Day, 1980; McKillop et al., 1999). Hopkinson & Day (1980) made use of the EPA Storm Water Management Model (SWMM) RECEIVE finite difference model to represent a coastal Louisiana swamp with two-dimensional flow equations. Contrary to other forested wetland models, the model built using EPA SWMM is not a hydrologic model, but rather a hydraulic model, and did not consider evapotranspiration or canopy interception influenced by vegetation cover. A similar hydrologic finite difference model was created in McKillop et al. (1999) which involved coupling surface and subsurface flow equations to model a headwater swamp in Southern Ontario. Evapotranspiration, canopy interception, and canopy drip processes were included in model calculations, as the importance of canopy and vegetation cover in forested wetlands was recognized. The hummocky terrain within swamps was also identified as an important flow pathway, and as a result, the model contained a two-layer soil profile with the first layer representing hummocks and hollows for allowing surface water channel routing and the second layer representing saturated organic material used for routing interflow (McKillop et al., 1999). Additional forested wetland models have integrated evapotranspiration routines and plant species parameterization, such as leaf area index (LAI) to further differentiate between forested and vegetated wetlands (Mansell et al., 2000).

3.1.8 Permafrost-Underlain Wetlands

Permafrost-underlain wetlands are another special amendment to the previously mentioned wetland conceptual-numerical model categories that are most often associated with bogs and fens but can occasionally be found in other ecological wetland classes (The National Wetlands Working Group, 1997). Permafrost-underlain wetlands are found where the soil is frozen for at least 2 years underneath ponded water and acts as a boundary condition for water storage (Harris et al., 1988). The deeper soil remains frozen all year round, with an active layer of frozen soil on top that freezes in the fall/winter and thaws in spring/summer (Harris et al., 1988; Quinton et al., 2009). For this reason, water storage in permafrost-underlain wetlands is driven by freeze-thaw processes in the active layer, (soil layer that freezes and thaws annually) (Devoie et al., 2019). Deeper liquid water storage is expected during the summer when the active layer thaws, and less liquid water storage is expected in the winter when the active layer freezes. Fill and spill mechanics between permafrost-underlain wetlands exists and is controlled by permafrost berms, which can act similarly to shallow bedrock and can lead to cascading of water between wetlands (Quinton et al., 2009; Connon et al., 2015; Mack et al., 2023). Connectivity between depressions is dependent on surface water connection or active layer interflow (Quinton et al., 2009; Devoie et al., 2019). An example diagram of a permafrost-underlain wetland is shown in Figure 3.12 along with its hydrologic processes.

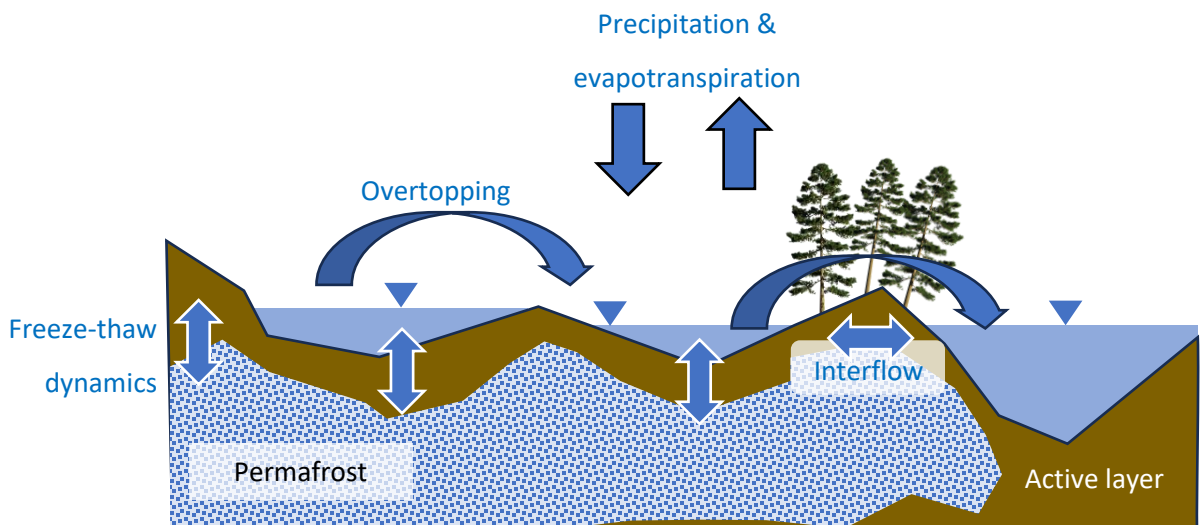


Figure 3.12. Diagram of an example permafrost-underlain wetland addendum to previously mentioned conceptual-numerical model categories.

Permafrost-underlain wetlands contain many complex and varying hydrologic processes. As such, this type of wetland does not have a unique water balance equation and can be simulated as part of different conceptual-numerical models. For example, permafrost-underlain wetlands may be represented as separated fill-and-spill wetlands to capture overtopping dynamics that occur above the permafrost layer.

Permafrost-underlain wetland modelling studies typically simulate freeze-thaw dynamics in the active layer, as well as permafrost thaw over time (Quinton et al., 2009; Devoie et al., 2019; Devoie & Craig, 2020). In large scale watersheds, permafrost-underlain wetlands can be broadly represented in hydrologic models by parameterizing their connectivity to the modelled stream network.

Parameterization at both large and smaller scales can include a multi-layer soil profile with the active layer experiencing freeze-thaw dynamics and the deeper permafrost acting effectively as a bedrock layer to constrain flows. Additional layers, such as the perennially-thawed soil layers called taliks, may also be represented (Harris et al., 1988; Devoie et al., 2019).

3.2 Limitations to be Addressed in this Thesis

The first objective of this thesis, to inventory existing wetland modelling methods and develop a catalogue of conceptual-numerical wetland model categories, was accomplished in this Chapter.

While the presented inventory is intended to make the integration of wetlands into watershed models more intuitive for modellers based on hydrologic behaviour rather than ecology alone, the organization approach taken to define the conceptual-numerical model categories still contains shortcomings. The organization approach assumes a linear relationship between the hydrogeomorphic classifications, ecological classifications, and conceptual-numerical model categories. This assumption can lead to some similarities between wetland categories being unrepresented or missed, due to wetland site-specific properties and variability of wetland properties (even within the same conceptual-numerical model category). The overlapping of conceptual-numerical model bubbles with multiple ecological and hydrogeomorphic classifications presented in Figure 3.2 aims to address some of the uncertainty related to the variability of wetland site properties. Still, the presented organization approach does not consider site-specific details and thus poses limitations for its use. For this reason, it is recommended that the organization approach to define wetland conceptual-numerical models be used as a starting reference for practical examples of wetland water balances and wetland hydrology.

Modellers can then build on their wetland representations by choosing the most appropriate conceptual-numerical model(s) based on their available site-specific information.

The existing modelling methods in the inventoried literature also pose gaps and limitations which are further addressed by the rest of this thesis. One limitation of the existing wetland modelling literature is their constrained modelling scopes. The existing wetland modelling literature either focuses on modelling one type of wetland per study, or a binary classification system (riparian vs. isolated) with missing consideration for treating multiple types of wetlands across multiple conceptual-numerical modelling categories. This provides a research opportunity for testing the representation of multiple wetland conceptual-numerical model categories within a single study site. Additionally, the existing studies are relatively small in scale and many lack the use of spatially explicit wetland-specific datasets for wetland representation. In recent years, the emergence of large-scale wetland datasets has presented opportunities for explicitly mapping wetlands in watershed models, however, their value in wetland modelling is poorly understood and not well researched.

To help fill the research gaps identified in existing modelling studies, a regional hydrologic model, or semi-distributed, large-scale watershed of a wetland-abundant region was built and used to test the representation of multiple types of wetland conceptual-numerical models. The second objective of this thesis, to implement conceptual-numerical wetland modelling methods in a hydrologic model, is addressed by the explicit representation of multiple wetland types over different modelling scenarios. The third objective of this thesis, to examine how available wetland mapping products can inform wetland modelling, is addressed by using a wetland mapping dataset to explicitly represent wetlands in a large-scale watershed and comparing similar wetland mapping products. The model setup, structure, and methods used to build the model are discussed in Chapter 4.

Chapter 4

Methods

4.1 The Nipissing Watershed

This section describes the physical characteristics of the Nipissing watershed as a background for the model created in subsequent sections.

4.1.1 Site Description

The Nipissing watershed is located in north-central Ontario with Sudbury to the west and North Bay to the east. The watershed spans 19,273 km² and is dominated by forest cover, lakes, and wetlands. The watershed lies within the Canadian Shield, with much of the basin being underlain by shallow bedrock (Ontario Geological Survey, 2000).

The headwaters start in the northwestern and southeastern tips of the watershed and drain through Lake Nipissing where water then flows southwest into Georgian Bay via the French River delta. The largest water body in the watershed is Lake Nipissing, with a surface area of around 878 km². Other notably sized water bodies in the watershed include Wanapitei Lake and Lake Temagami. Figure 4.1 shows the Nipissing watershed extent, lakes, and location relative to the Great Lakes in Ontario.

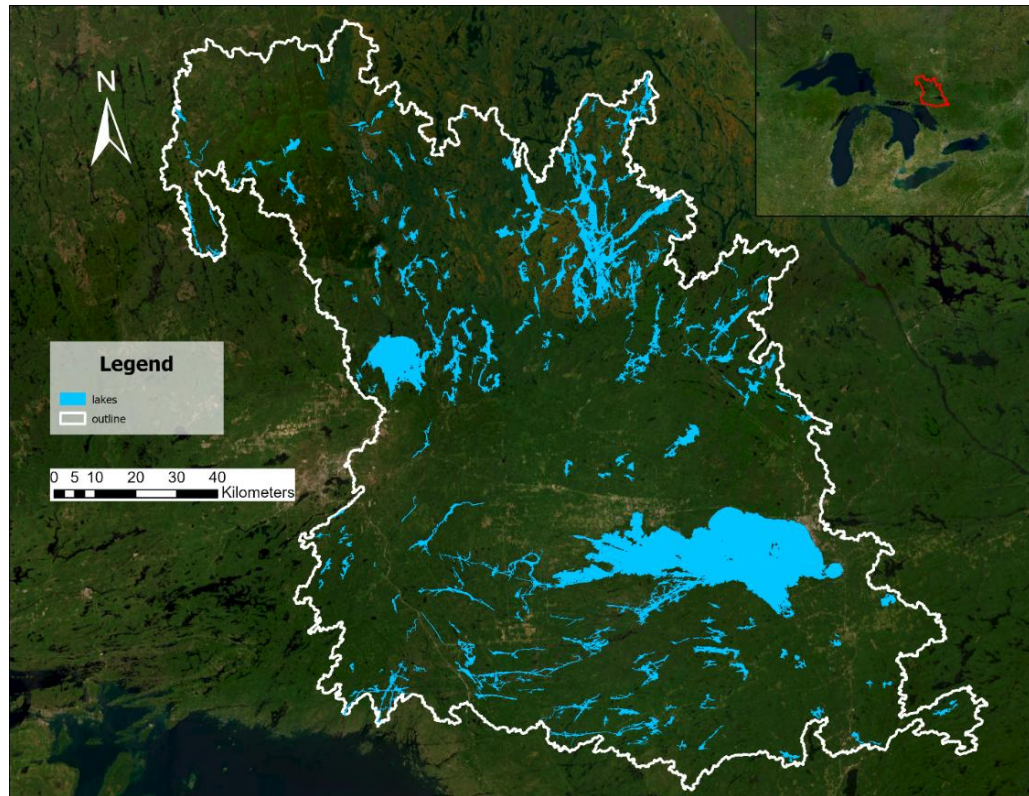


Figure 4.1. The Nipissing watershed in Ontario

The Ontario Ministry of Natural Resources and Forestry (MNRF) has divided the Province into Ecozones and Ecoregions based upon the climate, bedrock formations, and vegetation. By these classifications, the Nipissing watershed lies within the Ontario Shield Ecozone and is characterized by a cold, moist climate with mixed forest, deciduous forest, coniferous forest, and sparse forest as the dominant vegetation types. Annual precipitation within the Georgian Bay Ecoregion, which includes the Nipissing basin, ranges from around 770 mm to 1,100 mm (Crins et al., 2009).

Topography in the watershed is relatively flat, with 500 m of elevation relief between the highest and lowest points. The highest points in the watershed include the northwestern tip and southeastern tip, then decreasing in elevation towards Lake Nipissing and to the southwestern outlet into Georgian Bay. Many areas within the Nipissing watershed, as part of the Canadian Shield and Ontario Shield Ecozone, contain exposed bedrock outcrops (Crins et al., 2009).

The Nipissing watershed is dominated by Precambrian and Paleozoic bedrock with glacial deposits of sand and clay (Ontario Geological Survey, 2000; Kelly-Hooper, 2001; Crins et al., 2009; Government

of Canada, 2013). Much of the basin not underlain with bedrock contains Ortho Humo-Ferric Podzol (O.HFP) (Crins et al., 2009). Podzolic soils are an order of soil type classified using a combination of morphological and chemical criteria of the soil layers. Podzolic soils are known to occur in parent materials with low pH, coarse to medium textured soils usually under forest or vegetation cover in cool to cold climates with moderate humidity (Government of Canada, 1998). The O.HFP type of podzolic soil found in the Nipissing basin is characterized by acidic soils underneath shrub and forest vegetation (Government of Canada, 1998). Podzolic order horizons are categorized by layer subgroups with varying soil type. O.HFP type Podzolic soils are characterized as containing topsoil layers with sandy loam (Government of Canada, 1998; Canadian Society of Soil Science, 2020), and subsoil layers are characterized by material that is a silty loam coarser than clay (Government of Canada, 1998; Canadian Society of Soil Science, 2020).

Soil properties were estimated for these sandy loam and silty loam soil layer types and used to establish soil parameter ranges in the Nipissing model. Using the USDA soil texture triangle and taking the approximate centroid of sandy and silty loam areas, percentages of sand, silt, and clay for each soil layer were estimated. Figure 4.2 shows the USDA soil texture triangle with approximate centroids used for estimating soil composition.

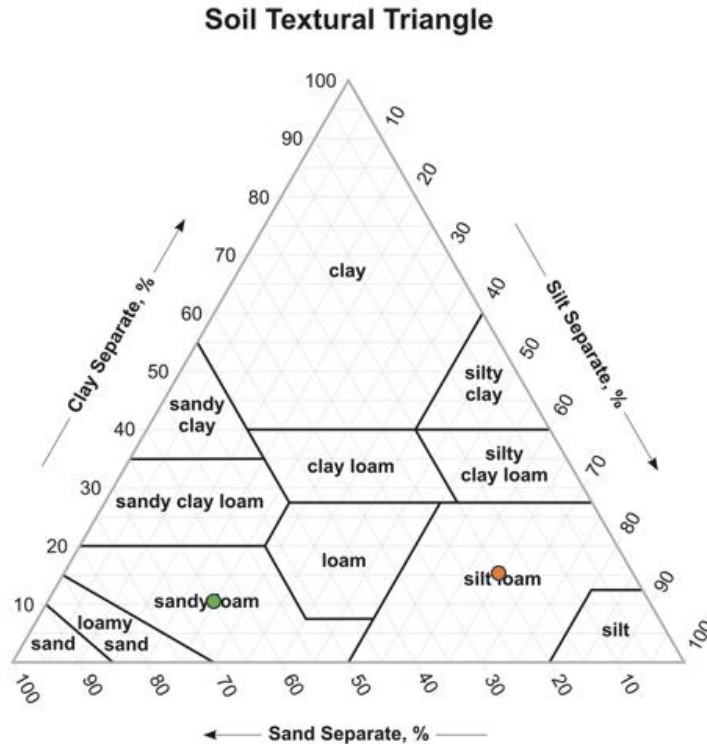


Figure 4.2. USDA soil texture triangle (Groenendyk et al., 2015) with sandy loam and silty loam approximate centroids for estimating Ortho Humo-Ferric Podzol soil composition

Sandy loam was estimated to have 25% silt, 65% sand, and 10% clay, while silty loam has 65% silt, 20% sand, and 15% clay. The parent material underneath horizon A and B soils is predominantly bedrock in the Nipissing watershed (Ontario Geological Survey, 2000; Crins et al., 2009), thus near-impermeable material was assumed for this deep layer. Using soil texture class properties published by Saxton & Rawls (Saxton et al., 1986; Saxton & Rawls, 2006), clay material was estimated to be composed of 25% sand, 50% clay, and 25% silt.

Field capacity and wilting point were also estimated for each soil layer. Using Table 3 of Saxton & Rawls (Saxton & Rawls, 2006):

- Topsoil layer: sandy loam with 65% sand and 10% clay has a field capacity of 0.18 and wilting point of 0.08.
- Middle soil layer: silty loam with 20% sand and 15% clay has a field capacity of 0.31 and wilting point of 0.11.

- Deep layer: clay with 25% sand and 50% clay has a field capacity of 0.42 and wilting point of 0.30.

Soil composition, field capacity, and wilting point estimates were used to establish soil property parameter ranges in the Nipissing model.

The Nipissing watershed contains several hydroelectric dams and flood controls. As such, the hydrologic regime of Lake Nipissing and the French River delta is highly regulated. Dams upstream of Lake Nipissing include West Nipissing Power Dam, Crystal Falls Generating Station, Canadore College Dam, and Nipissing Control Dam. Water flowing downstream of Lake Nipissing reaches the French River delta, which passes through dams including Big Chaudière Dam (North and South), Little Chaudière Dam, and Portage Dam. Figure 4.3 displays a map of the Nipissing watershed with points indicating dam locations.



Figure 4.3. Nipissing watershed dam locations denoted by squares with numbers. This figure is taken from the Sturgeon River/Lake Nipissing/French River Watershed Management Plan, Figure 1.1 (Acres International Ltd. & Ontario Ministry of Natural Resources and Forestry, 1992)

As a result of regulation, accurate flow simulations in Lake Nipissing and the French River delta are difficult to produce without representing hydroelectric reservoir outflows. The presence of regulation therefore determines what data can be used for calibration and validation of the model. Section 4.2.3

further describes flow and level gauge locations selected to benchmark model performance in the Nipissing watershed.

As is characteristic of parts of northern Ontario, the Nipissing watershed contains many wetlands. Using the Ontario GeoHub Wetlands dataset (Ontario Ministry of Natural Resources and Forestry, 2010a) to estimate wetland area coverage, approximately 2,171 km² of the watershed is covered in wetlands, or 11.3% of the total watershed area. It should be noted that this percentage may be larger or smaller, as the Ontario wetland polygon layer may overestimate wetland coverage in some areas while disregarding wetlands below a certain spatial scale.

4.2 Model Development

Using the Raven Hydrological Modelling Framework (Craig et al., 2020), a hydrologic model of the Nipissing watershed was built to assess the impacts of wetland representation methods on model performance. The Raven Hydrological Modelling Framework (Raven) is a flexible modelling toolbox containing multiple algorithms for the creation of lumped or semi-distributed hydrologic models (Craig et al., 2020). This section describes the input data, model set-up, and calibration of the Nipissing watershed Raven model to test three wetland model structure scenarios: a base scenario with no wetland representation, Scenario 2 with one wetland type, and Scenario 3 with three wetland types. These model structure scenarios are described in detail in this section after outlining data requirements and HRU discretization.

Model input data, including watershed delineation, meteorological, terrain, flow and water level, geology, land cover, and wetland data are described in the following subsections.

4.2.1 Watershed Delineation

Watershed delineation of the Nipissing watershed was performed using the Ontario Lake and River Routing Product Version 2.0 (OLRRPv2) BasinMaker Colab Notebook (Han et al., 2023; Shen et al., 2024), extracted for the Nipissing watershed and including a filter that removes lakes less than 1 km² in area and subbasins with a drainage area less than 50 km². These lake and subbasin filters are standard filters applied to other Ontario region models being developed the University of Waterloo Hydrology Research Group.

BasinMaker is a Python-based GIS tool used for watershed delineation and simplification, particularly for watersheds with complex lakes (Han et al., 2023). BasinMaker was also used to delineate OLRRPv2 regions, which covers drainage regions across Ontario according to the Ontario Integrated Hydrology (OIH) Quaternary Ontario Watershed Boundaries (Ontario Ministry of Natural Resources and Forestry, 2021). Stream networks are delineated using an enhanced flow direction raster and provincial DEM from the OIH dataset (Ontario Ministry of Natural Resources and Forestry, 2019). Lake polygons are obtained from the HydroLakes database (Messenger et al., 2016). The OLRRPv2 product delineated for the Nipissing watershed containing 545 subbasins, a simplified stream network, and lake polygons is shown in Figure 4.4.

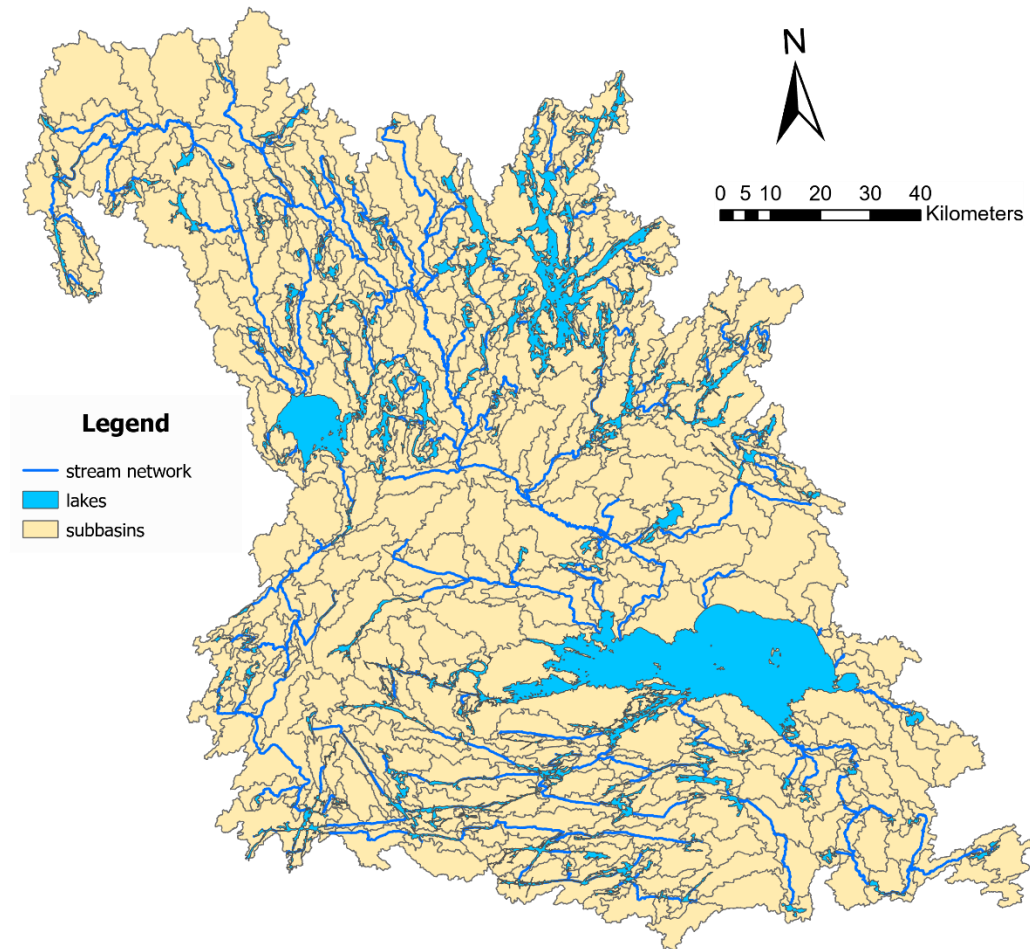


Figure 4.4. Nipissing watershed delineated using OLRRPv2 (Han et al., 2023; Shen et al., 2024) including subbasins, stream network, connected lakes, and non-connected lakes

The watershed subbasins, stream network, and lakes shown in Figure 4.4 were used to define the network and subbasin geometry within the numerical model.

The Provincial Digital Elevation Model (PDEM) provided by the Ontario MNRF (Ontario Ministry of Natural Resources and Forestry, 2019) was used to retrieve mean elevation, slope, and aspect values of HRUs discussed in Section 4.2.4.

4.2.2 Meteorological Data

Meteorological gauge data is very sparse throughout and around the Nipissing watershed. An evaluation of data availability and temporal coverage of all Meteorological Service of Canada (MSC) gauge stations within 75 km of Lake Nipissing was performed, identifying climate stations with sparse and inconsistent data records as shown in Figure 4.5. This analysis demonstrated the inability to use meteorological observations in this study, despite the use of observed climate data being preferred over gridded climate data products.

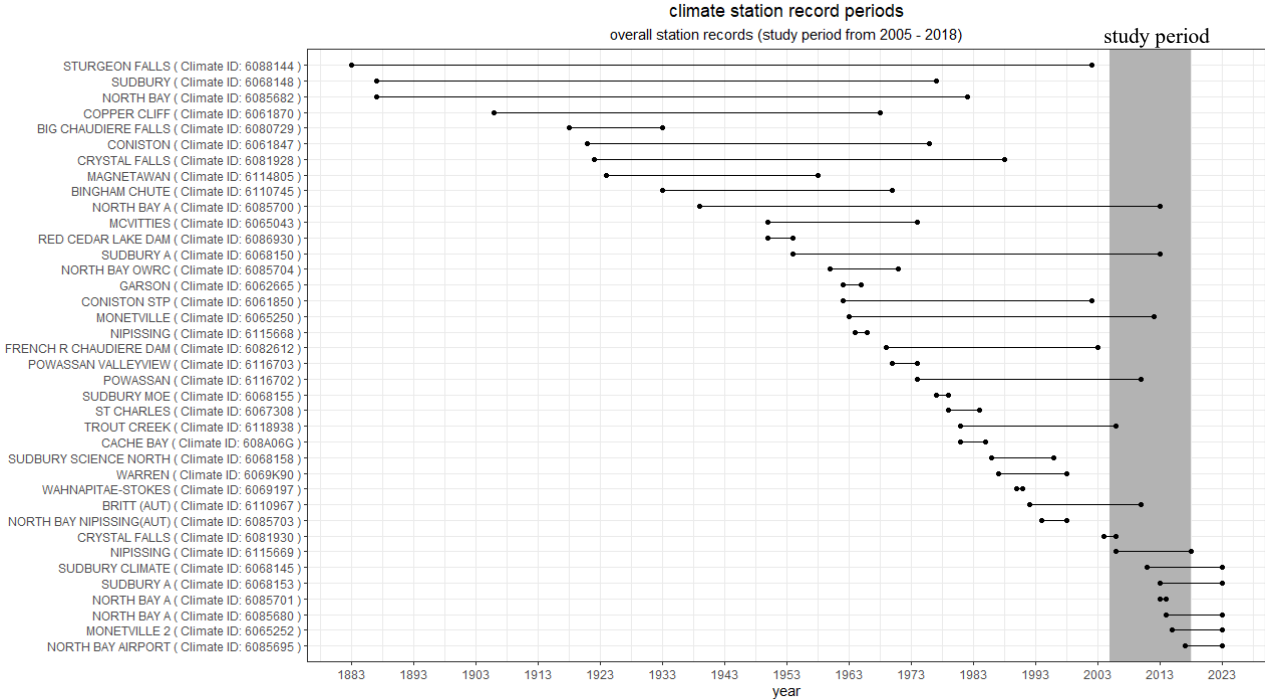


Figure 4.5. Climate gauge record plot summarizing length of data records for Environment and Climate Change Canada climate stations within 75 km of the centroid of Lake Nipissing. Complete data records for the study period of interest (2005 to 2018) are very sparse. Many of the gauges shown here also contain incomplete data records even during the reported start and end periods.

Due to lack of complete and proximate observation data with sufficient temporal coverage, gridded climate data was used instead. Gridded precipitation and temperature data was extracted from the Regional Deterministic Reanalysis System v2.1 (RDRS) data product (Gasset et al., 2021) downloaded from the Canadian Surface Prediction Archive (CaSPAr) (Mai et al., 2020). RDRS v2.1 was assessed as part of the Great Lakes Runoff Intercomparison Project (GRIP-GL), yielding between 1,000 to 1,050 mm of mean annual precipitation at calibration gauges within the Nipissing watershed and to the west in Sudbury (Mai et al., 2022). These mean annual precipitation values are consistent with the roughly 770 mm to 1,100 mm of annual precipitation estimated for the Georgian Bay Ecozone (Crins et al., 2009) and are thus appropriate for use in the Nipissing model.

A comparison between another popular gridded product, Daymet v4 (Thornton et al., 2022a, 2022b), and RDRS v2.1, was also performed. The findings of this comparison were that RDRS v2.1 more accurately predicted precipitation and mean temperature observations at Sudbury Airport gauge 6068145 (Climate ID) than Daymet v4. The full comparison of RDRS and Daymet products for the Nipissing study area is provided in Appendix A.

As a result of the RDRS versus Daymet comparison, spot checking of RDRS results in GRIP-GL, and standardized use of RDRS data in many Environment and Climate Change Canada (ECCC) and OLRRP projects, RDRS v2.1 precipitation and temperature data were used exclusively in the modelling experiment. RDRS data was processed for the Raven model using the GitHub repository RDRS processing toolbox created by Rezgar Arabzadeh from the University of Waterloo Hydrology Research Group (Arabzadeh, 2025).

4.2.3 Flow and Water Level Data

Six Water Survey of Canada (WSC) flow and two WSC level gauges were used for calibration and validation of the model. Gauge locations are shown in Figure 4.6.

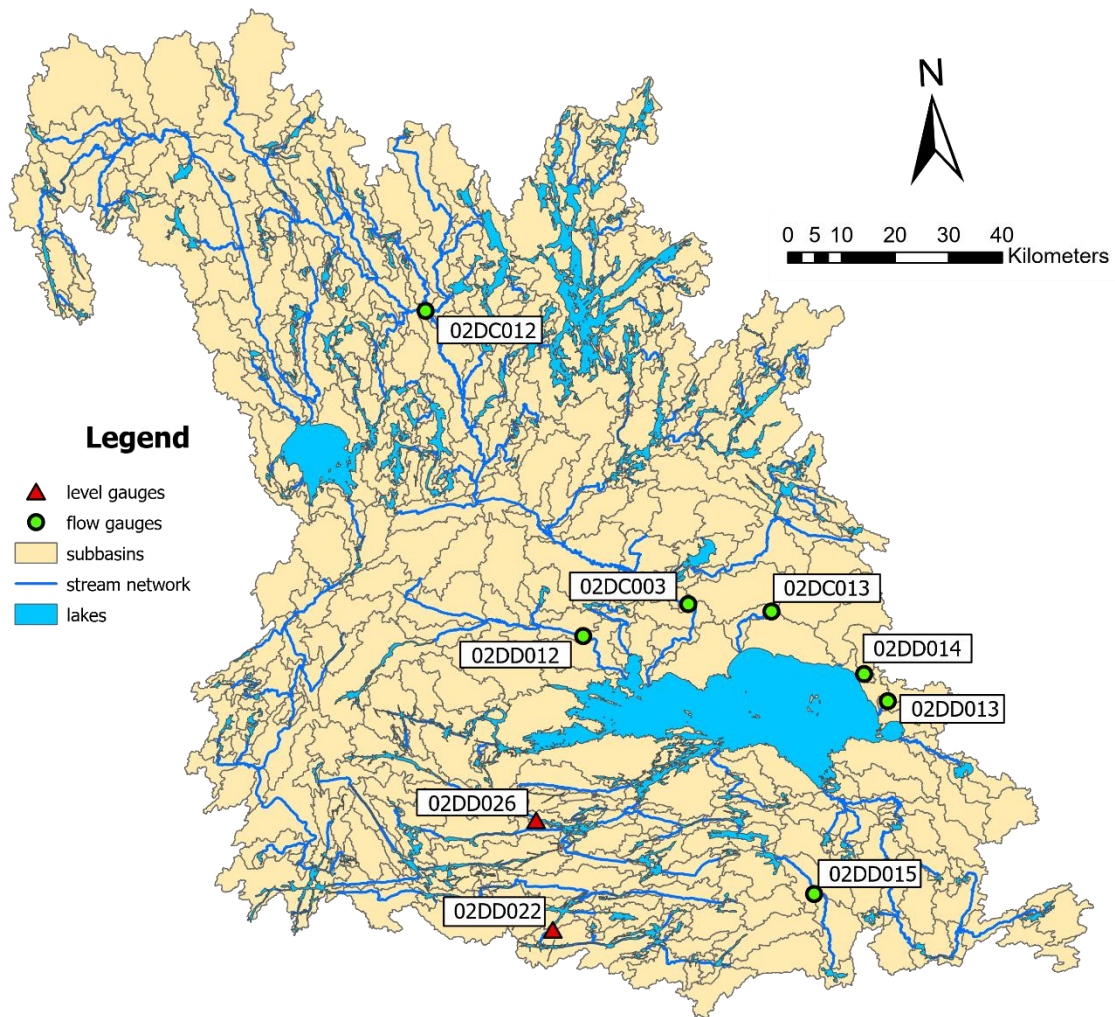


Figure 4.6. WSC Streamflow and water level gauge locations obtained for Nipissing Raven model evaluation

Flow gauge 02DC003 is located at the Crystal Falls generating station operated by Ontario Power Generation (OPG) and was not used for calibration or validation but included in the model for data assimilation purposes. Crystal Falls dam outflow data was obtained from OPG and used to replace simulated flows with observed flows leaving gauge 02DC003. This data assimilation process creates a merger of model results and observational data below Crystal Falls generating station to account for regulation impacts downstream of the dam. However, the impact of regulation on downstream gauges is not eliminated, as additional regulation downstream may be present.

As a result of regulation downstream of Lake Nipissing, flow and level gauges 02DD026 and 02DD022 were not used for calibration, but instead used for validation. WSC gauges used in calibration and validation are further discussed in Section 4.3.

4.2.4 Development of Hydrologic Response Units (HRUs)

HRUs are areas defined by a unique combination of land use type, vegetation cover, and terrain type, all underlain by vertical soil profiles (Craig et al., 2020; Craig & Raven Development Team, 2024). Vertical water and energy balances are applied to each HRU within a finite number of hydrologic storage compartments (e.g., soil and canopy) (Beven, 2012). Once all water balance calculations are applied to an HRU, the resulting runoff is shed and routed between subbasins. In Raven, HRUs are user-specified, meaning that the modeller can decide the degree of HRU complexity most appropriate for their model. For instance, HRUs can be as simple as only distinguishing between forest and water land cover types or be more complex by including unique combinations of land cover, vegetation types, elevation, soil type, or any other geospatial criterion.

The Nipissing model made use of land cover, geology, and wetland datasets for HRU discretization. For the initial Nipissing model, no attempt was made to discriminate between different land covers or soil types; the only distinction was made between land, lakes and other water bodies for land cover type and areas of bedrock versus non-bedrock for geology type, yielding a maximum of six HRUs per subbasin. Water HRUs were defined as water bodies other than modelled lakes that were identified by satellite land cover data. Simplified HRUs were used as a base representation of the watershed land cover, as most of the landcover in the basin is either forested or water. Urban areas, such as North Bay and parts of Sudbury, cover a relatively small area and likely have negligible impact on the overall watershed hydrologic response. This initial HRU discretization represents a base model scenario without wetland presence.

Wetland data was then introduced into the HRU discretization in two additional modelling scenarios. The addition of wetland HRUs after evaluating a base scenario allowed the impact of adding wetland land cover on model performance to be evaluated.

The following subsections describe the geology, land cover, and wetland datasets used for HRU discretization along with the discretization process.

4.2.4.1 Geology and Soil Data

The Quaternary Geology, Seamless Coverage of the Province of Ontario dataset (Ontario Geological Survey, 2000) was clipped to the Nipissing watershed extents and used to inform HRU delineation.

The “geology_II” layer within the dataset contains the type and age of geologic materials across Ontario. Due to the majority of the Nipissing being underlain by bedrock (Crins et al., 2009; Ontario Geological Survey, 2000), the geology layer was simplified into two categories: bedrock (Paleozoic and Precambrian periods) and non-bedrock (all other historical periods). Figure 4.7 shows the geology layer clipped and simplified for the Nipissing watershed model.

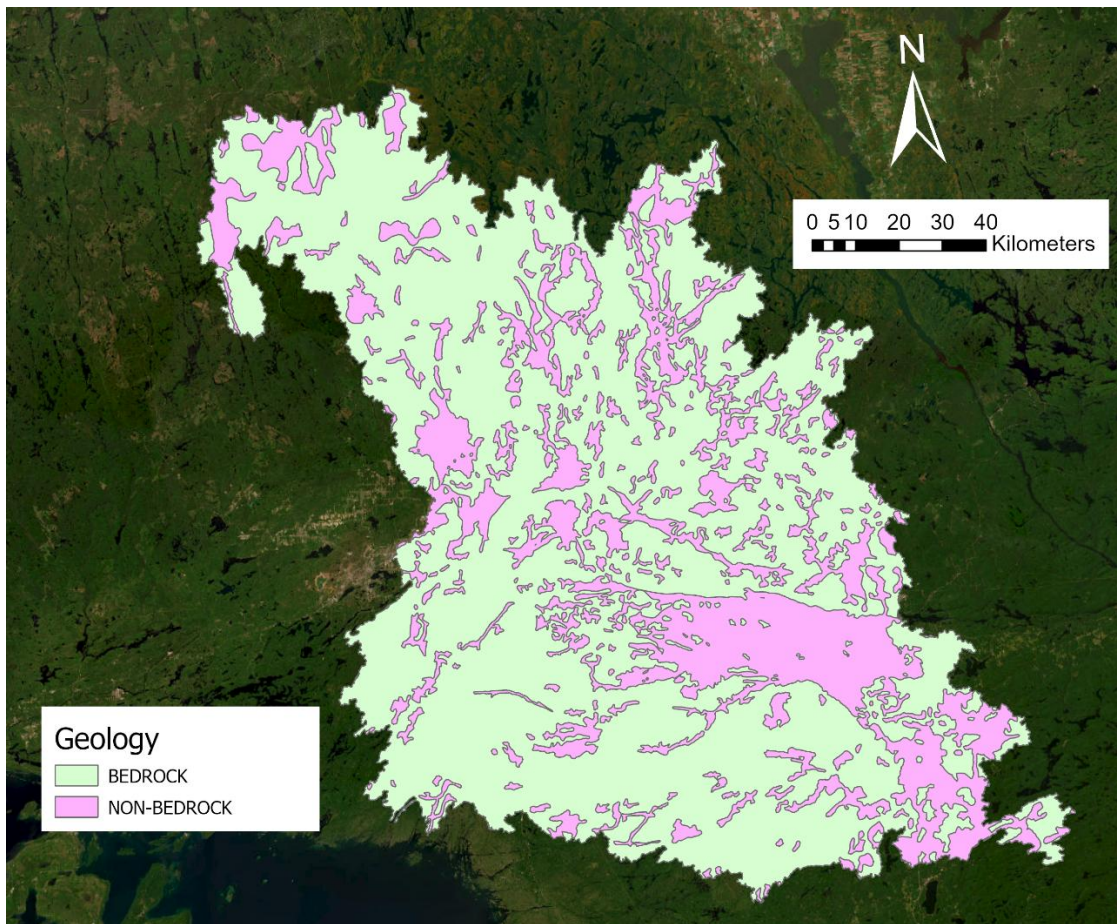


Figure 4.7. Quaternary Geology, Seamless Coverage of the Province of Ontario dataset (Ontario Geological Survey, 2000) clipped to the Nipissing watershed and simplified into bedrock and non-bedrock geologic material types

It is important to note that bedrock exists below the ground surface across the watershed. Areas mapped as “bedrock” in the model represent locations where the bedrock is close to the ground surface compared to areas of “non-bedrock” where there is a much larger soil column above the bedrock.

4.2.4.2 Land Cover Data

The Ontario Land Cover Compilation (OLCC) v.2.0 (Ontario Ministry of Natural Resources and Forestry, 2016) land cover raster was obtained for land use representation. The dataset is a combination of the Far North Land Cover v1.4, Southern Ontario Land Resource Information System (SOLRIS) v1.2 (Ontario Ministry of Natural Resources and Forestry, 2000a), and the Provincial Land Cover 2000 Edition (Ontario Ministry of Natural Resources and Forestry, 2000b). Each of these datasets cover different extents of the province. The Nipissing watershed is fully covered by the Provincial Land Cover 2000 dataset (Ontario Ministry of Natural Resources and Forestry, 2000b), as shown in Figure 4.8.

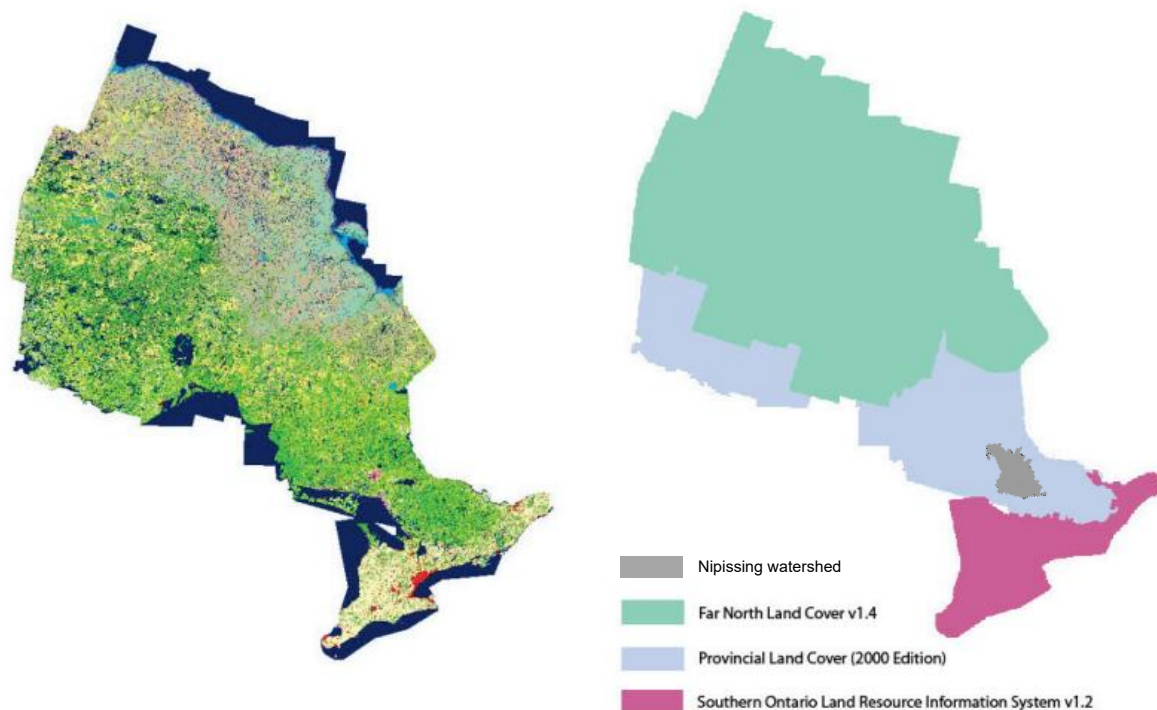


Figure 4.8 (a) Ontario Land Cover Compilation v2 for Ontario and (b) Source Land Cover Datasets. Figure adapted from the OLCC v2 Data Specifications Document (Ontario Ministry of Natural Resources and Forestry, 2014)

The OLCC v2 raster layer was clipped to the Nipissing watershed extents, converted to a shapefile, and used for HRU discretization.

4.2.4.3 Wetland Polygon Data

Wetland polygon data was used in the development of the Nipissing model to define the extents of wetland HRUs in the model. A moderate level of uncertainty exists among wetland datasets, as not all wetlands are mapped (especially at very small scales). Additionally, wetland boundaries are often dynamic and subject to change over time. There also exists the possibility for overestimating wetland boundaries. To investigate the uncertainty related to available wetland mapping products, four datasets were compared. These datasets include the Ontario GeoHub Wetlands dataset (MNRF dataset) (Ontario Ministry of Natural Resources and Forestry, 2010a), Ontario Provincial Land Cover

Dataset (Ontario Ministry of Natural Resources and Forestry, 2000b), Forest Resource Inventory (FRI) dataset Version 2 (Ontario Ministry of Natural Resources and Forestry, 2020), and the National Terrestrial Ecosystem Monitoring System for Canada (NTEMS) (Hermosilla et al., 2016; Wulder et al., 2018, 2025). Upon comparing each wetland mapping product using GIS software, it was found that there were significant discrepancies between datasets, particularly in the overlapping of wetland polygons. A spatial comparison of the MNRF dataset, Ontario Provincial Landcover dataset, FRI dataset, and NTEMS dataset was performed and is included in Appendix B. Further discussion and recommendations for the use of wetland mapping products is provided in Chapter 6. Through the comparison of various wetland datasets, it was found that the MNRF dataset is the most complete and accurate dataset offered by the government of Ontario, and to maintain consistency with the use of other Provincial datasets, the MNRF wetlands dataset was used in the Nipissing model.

The MNRF wetland dataset contains wetland polygons across Ontario, as well as attribute information for each polygon. One useful shapefile attribute includes wetland ecological type (bog, fen, marsh, swamp, open water, or unknown, consistent with the ecological types discussed in Section 2.1.1). The wetland polygon attributes also identify wetlands evaluated with the Ontario Wetland Evaluation System (OWES) and those that are designated as Provincially Significant Wetlands (PSW) (Ontario Ministry of Natural Resources and Forestry, 2010a). A variety of sources inform the Ontario GeoHub wetlands dataset, including contractor surveys, the Wetland Interim (OBM/NTS), Forest Resource Inventory (FRI), Southern Ontario Land Resource Inventory System (SOLRIS), and MNRF district data (Ontario Ministry of Natural Resources and Forestry, 2024).

The wetland polygons dataset, which is accessed as a polygon feature dataset, contains wetland shapefiles explicitly mapping wetland locations across Ontario. The wetland shapefile obtained from the dataset was clipped to the Nipissing watershed using ArcGIS Pro 3.0.3 (ArcGIS). A total of 48,399 wetland polygons were retained from the dataset for use in the Nipissing model, with a total area of approximately 2,171 km².

4.2.4.4 Classification of Wetland Polygons

The MNRF wetland polygon data was used to build wetland model structure scenarios by including these wetlands as spatially explicit wetland HRUs. To fulfill the second objective of this thesis, wetland HRUs in the Nipissing model were distinguished according to the conceptual-numerical models discussed in Chapter 3. To investigate the impact of explicitly modelled wetlands versus no

wetlands in Raven, the wetland data was sorted into three conceptual-numerical model categories based on hydrologic connectivity in the watershed. Ecological classifications were included in the MNRF wetland dataset, however, were not used in this model to classify wetlands because of the introduction of additional uncertainty related to the dataset’s ability to accurately identify ecological wetland types. As the main hydrologic features of the Nipissing model are lakes and streams, it is useful to distinguish wetlands by their connectivity to the river and lake network in GIS operations:

- Riparian wetlands - Wetlands that are physically connected to the modelled stream network. Referred to as “connected wetlands” in GIS operations and model files for brevity.
- Lake fringe wetlands - Wetlands that are on the fringe of a modelled lake or physically connected to a modelled lake. Referred to as “lake-adjacent wetlands” in GIS operations and model files for brevity.
- Geographically isolated wetlands – Wetlands that are not physically connected to the modelled stream network or lakes. All GIWs in the Nipissing model are assumed to be unconfined. Referred to as “isolated wetlands” in GIS operations and model files for brevity.

Wetland polygons within the clipped wetland shapefile were classified according to the above definitions by physically overlapping the wetland polygons with the modelled lake polygons and river network using ArcGIS. This process allows individual wetland polygons to be tagged with their corresponding classification. Figure 4.9 illustrates each wetland classification type defined by overlapping the wetland polygon layer with modelled lakes and river segments.

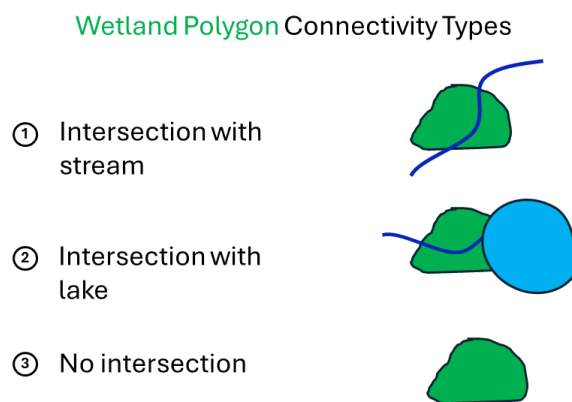


Figure 4.9 Wetland polygon classification based on hydrologic connection in the modelled watershed

By distinguishing between wetland connectivity type in Raven, the manipulation of Raven parameters to represent the different hydrologic behaviour of wetland conceptual-numerical models is possible. In this study, wetlands were explicitly represented in Raven by defining new HRUs using wetland location and connectivity to the modelled network.

Part of the HRU building process involved pre-processing of the wetland polygon data to classify wetlands based on their hydrologic connectivity in the watershed, as defined in Figure 4.9. The following workflow was developed using the R programming language (R Core Team, 2023) to classify wetland polygons based on their connectivity type:

1. River, lake, wetland, and watershed polygons are read into the script using the “sf: Simple Features for R” package (Pebesma, 2016). The script reads in the watershed outline, stream network, and connected/non-connected lakes from the OLRRP v2.0 watershed files described in Section 4.2.1. The MNRF wetland polygons, clipped to the Nipissing watershed, are also read into the script environment. Wetland connectivity type columns are added to the wetland polygons shapefile. All shapefile data is transformed to the coordinate reference system (CRS) EPSG:3161 for MNR Lambert for consistency.
2. Wetland polygon geometry is then intersected with the stream network geometry using the *st_intersects* function (Pebesma, 2016), creating a logical matrix where each row represents one wetland polygon and each column represents one stream segment geometry. The *st_intersects* function returns a value of 1 in a given cell if the geometry of the wetland polygon in that row and the stream segment in that column intersect and returns a 0 if they do not intersect. For each wetland polygon row, the sum of all columns is calculated and added to the wetland data frame as a new column labelled “connect_stream”. Any value within the *connect_stream* column that is greater than or equal to 1 indicates that the wetland polygon is connected to the stream network.
3. Wetland polygon geometry is then intersected with the lake polygon geometry, creating another logical matrix where the same workings as the stream network intersection matrix apply. A new column in the wetland data frame called “connect_lakes” is created. Any value within the *connect_lakes* column that is greater than or equal to one indicates that the wetland polygon intersects with a lake polygon. A third column in the wetland data frame, “isolated”, is created to represent wetlands that are connected to neither a stream nor lake. The columns *connect_stream* and *connect_lakes* are repopulated with a conditional statement that assigns a

- value of 1 if the value for any wetland was greater than or equal to 1, and a value of 0 otherwise. The *isolated* column is then populated with a value of 1 if both the *connect_stream* and *connect_lakes* column contain a value of 0 and keep the value as 0 otherwise.
4. The MNRF wetland shapefile, clipped to the Nipissing region, with three additional columns indicating connectivity of each wetland polygon is then saved. A value of 1 in the *connect_stream* column indicates that the wetland polygon is connected to the river network, a value of 1 in the *connect_lakes* column indicates that the wetland polygon is connected to a lake, and a value of 1 in the *isolated* column indicates that the wetland polygon is geographically isolated from the modelled stream network and lakes. It should be noted that some wetland polygons may be connected to both a stream and lake. In this case, the wetland is still classified as lake adjacent. It should also be noted that any given wetland polygon can receive a different classification depending on the representation of lakes and rivers in the model.

A separate R script was created to prepare wetland shapefiles for integration into the land cover shapefile. To ensure compatibility of the final HRU shapefile with BasinMaker colab scripts later used to produce Raven input files, the attribute column “Landuse_ID” must be included in HRU shapefiles to distinguish between land cover types using unique integer values. A *Landuse_ID* column was appended to the classified wetlands shapefile.

Prior to HRU delineation, one final step was required to process the land cover and wetland shapefiles. The classified wetlands shapefile needed to be stitched into the land cover shapefile to define wetland locations. All previous land cover wetland classifications within the OLCC v2 dataset are reclassified as either forest, lake, or water during the HRU building process and only the newly classified wetland polygons from the MNRF wetland dataset are used to identify wetland land cover.

ArcGIS was used to erase areas in the land cover shapefile that overlap with wetland polygons using the Erase tool (similar to using the wetland geometries as cookie cutters). The wetland polygons were then merged into the land cover shapefile using the Merge tool. The resulting land cover shapefile contained wetland polygons as new land use types based on their hydrologic connectivity in the watershed. An illustration of the erase and merge process is shown in Figure 4.10.

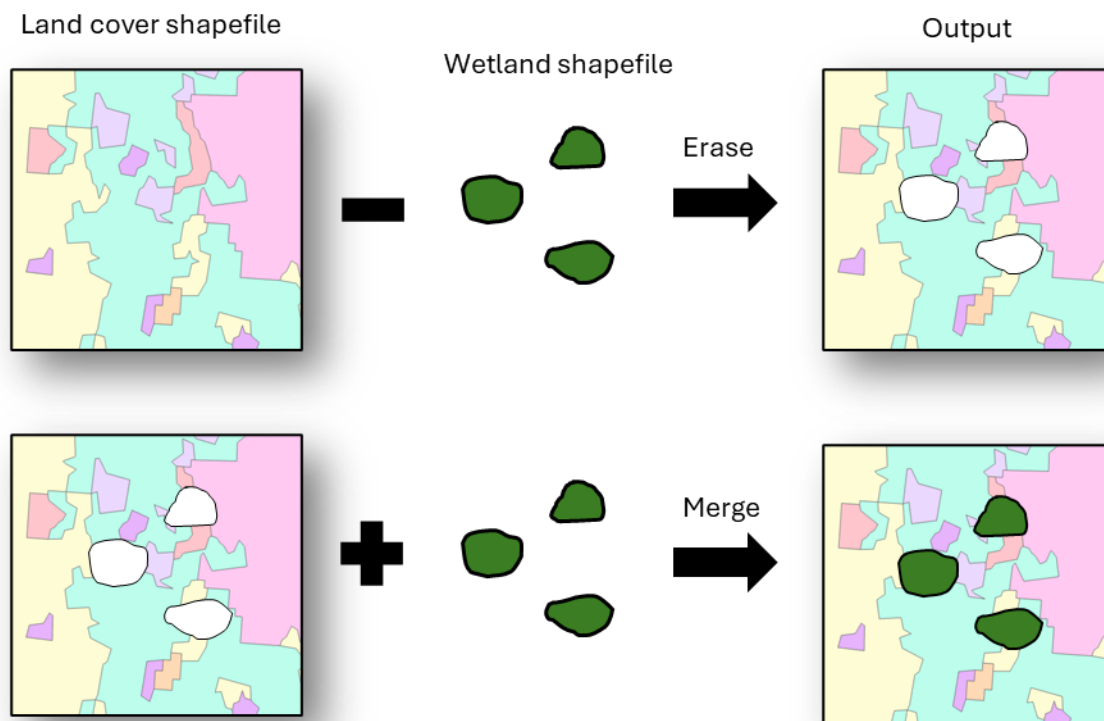


Figure 4.10 ArcGIS Erase and Merge process used to integrate classified wetland polygons into land cover shapefile before HRU delineation

This same process of using Erase and Merge tools were also used to add OLRRP v2 lakes into the land cover shapefile, overriding the previous land cover such that lake extents from BasinMaker are respected in the final model. OLRRP v2 lakes were added separately into the land cover dataset due to their built-in attributes that would then be used in Raven to identify these water bodies as reservoirs.

The attribute table of the new land cover shapefile including classified wetlands then required manipulation. Each land cover type (labelled “gridcode” in the land cover shapefile attribute table) was assigned a *Landuse_ID* value. The OLCC v2 land cover types were redefined to only contain lakes, forest, water, and the merged wetland classification types. This merging of all other land cover types is justified based on the small (<5% coverage in the watershed) impact from other land cover types such as urban or agricultural. The final land cover shapefile was then saved for HRU

delineation. Table 4.1 summarizes the land cover classes assigned to the newly classified land cover shapefile.

Table 4.1. Integration of MNRF wetland polygons classified by hydrologic connectivity and reclassification of Ontario Land Cover Compilation v2 land cover types for HRU discretization

OLCC v2 Land Cover Code	Reclassified <i>Landuse_ID</i>	Reclassified Land Cover Class Name	Description
-	-1	LAKE	Lake areas covered by OLRRP v2 connected and non-connected lake polygons.
1	1	WATER_SIMPLE	Water that is included in OLCC v2 but is not covered by OLRRP v2 lake polygons.
2 to 28	2	FOREST	Areas of various land cover types simplified into forest coverage for the purposes of the Nipissing model.
-	29	WETLAND_CON	Wetland area classified as riparian wetlands connected to a modelled stream network (connected wetlands).
-	30	WETLAND_ISO	Wetland area classified as geographically isolated and not connected to modelled lakes or streams (isolated wetlands).
-	31	WETLAND_LAKE	Wetland area classified as lake fringe wetlands connected to a modelled lake (lake connected wetlands)

4.2.4.5 HRU Discretization

HRUs were discretized using ArcGIS by overlapping geometry from the reclassified land cover shapefile described in Section 4.2.4.4, geology shapefile, and subbasin delineations. Elevations, slope, and aspect for each HRU centroid were extracted from the DEM described in Section 4.2.1.

The land cover and geology shapefiles were dissolved based on their respective attributes (*Landuse_ID* for the land cover shapefile and geology type for the geology shapefile) in order to prepare for layer intersections. Then, land cover, subbasin, and geology shapefiles were intersected using the Pairwise Intersect tool.

A separate R script was used to identify sliver HRUs (those with $\leq 3\%$ of the subbasin area) and merge the slivers with the largest HRU in that subbasin. HRU land cover types *LAKE*, *WETLAND_CON*, *WETLAND_ISO*, and *WETLAND_LAKE* were not simplified due to either their reservoir properties or many wetlands being lost in the simplification process. Simplified HRUs were then dissolved again in ArcGIS by *Landuse_ID*, subbasin, geology type, and HydroLake ID to merge similar types together and reduce the HRU count.

HRU attributes were processed for Raven input files using the BasinMaker attributes python script written by Ming Han (Han et al., 2023), using the required “landuse_info.csv” file defining *Landuse_ID* codes. These same categories were also defined in the required “veg_info.csv” file for vegetation type and “soil_info.csv” for geology types. For the purposes of testing model structure with wetland hydrologic connection only, HRU vegetation types were kept the same as landcover types. A total of 2,943 HRUs were generated. The Raven RVH file was created using the BasinMaker Version 2.0 Google Colab Notebook (Han et al., 2023; Shen et al., 2024).

A final post-processing step was required to distinguish between lake fringe wetlands and lake adjacent wetlands. In some cases, wetland polygons that intersected with a lake polygon were also intersected by a subbasin boundary, thereby splitting the wetland between subbasins. This occurs due to a mismatch in model input sources, as the subbasin delineation and wetland mapping product are independent datasets. The part of the wetland polygon that lies within a lake subbasin was considered a lake fringe wetland and drains into the lake. The other part of the wetland polygon within the downstream subbasin is handled with two cases.

In Case A, the wetland polygon in the subbasin downstream of the lake polygon drains into a stream network with no additional lake polygon present within that subbasin. Case A wetland polygons were classified as a “lake adjacent wetland”, receiving similar treatment to a wetland only connected to the stream network. Lake adjacent wetlands were labelled “WETLAND_LAKE_ADJ” for land use and vegetation classes within the Raven RVH file. An example of a lake fringe and lake adjacent wetland is shown in Figure 4.11.

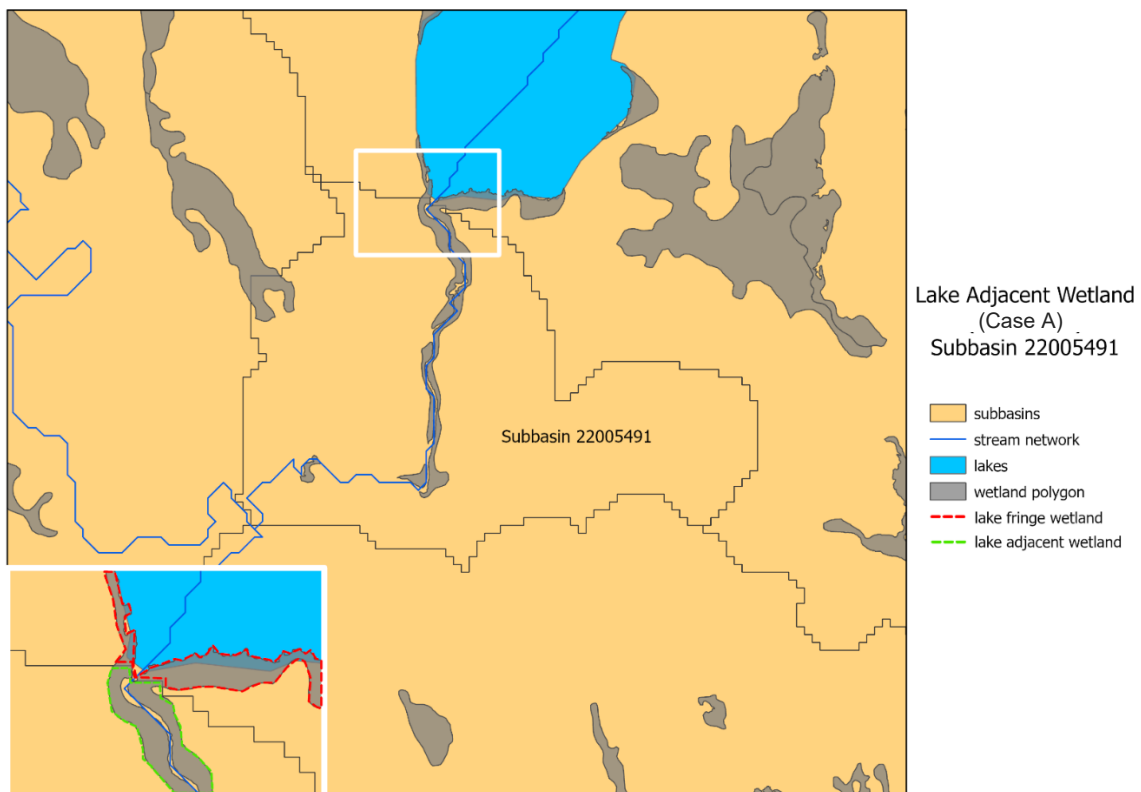


Figure 4.11 Lake adjacent wetland Case A, where the wetland polygon classified as *WETLAND_LAKE* becomes cut by a subbasin boundary and the subbasin does not contain a lake. *WETLAND_LAKE* then becomes reclassified as wetland *WETLAND_LAKE_ADJ* (lake adjacent wetland)

In Case B, the wetland polygon in the subbasin downstream of the lake polygon is assumed to be influenced by lake dynamics because there exists a lake polygon in that subbasin either connected to

the wetland polygon or directly downstream. It is recognized that the lake polygon downstream may not physically overlap with the wetland polygon, however, the processing becomes very complex when measuring the distance from the wetland polygon to the lake polygon and reclassifying based on proximity of the two. For simplicity, it is assumed that all Case B wetland polygons are influenced enough by lake processes to be classified as lake fringe wetlands and are labelled as *LAKE* for land use and vegetation classes within the Raven RVH file. Both Case A and Case B only apply to wetland modelling scenarios that include lake fringe wetlands.

4.2.5 Wetland Modelling Scenarios

All Raven input files were generated using Magpie v0.2 and OLRRP BasinMaker Colab v2 (Burdett & Craig, 2023; Han et al., 2023; Shen et al., 2024) and are provided in Appendix C. The Raven RVI and RVP files were initially created using the Canadian Shield template and default parameter values and then adapted to the Nipissing watershed using input from the Petawawa tutorial Raven model (Han, 2022), research on the Nipissing watershed, and manual calibration. With each additional wetland HRU type, land cover and soil parameters were added to the Raven RVP file.

The Nipissing model was set up using three different scenarios to test the impact of varying wetland model structures on simulation performance. Scenario 1 acts as a base model and assumes no representation of wetlands. Land use and vegetation types include *FOREST*, *LAKE*, and *WATER*, with all wetland HRUs from the discretization step being re-assigned to represent *FOREST* land use and vegetation. *BEDROCK* and *NON-BEDROCK* soil profiles were used. Figure 4.12 shows HRU landcover types delineated for Scenario 1, in addition to the hydrologic processes structure.

Scenario 2 introduces a single wetland type and wetland soil profile for all wetland polygons defined in Section 4.2.4.4. All wetlands were assumed to be stream-adjacent to test model performance after adding one type of wetland. Land use and vegetation types include *FOREST*, *LAKE*, *WATER*, and *WETLAND_CON*. Soil profiles include *BEDROCK*, *NON-BEDROCK*, and *WETLAND_CON*, using a common wetland soil profile for all wetland polygons. Figure 4.13 shows HRU landcover types delineated for Scenario 2, in addition to the hydrologic processes structure.

Scenario 3 introduces three wetland types and two wetland soil profiles. Using the wetland classifications defined in Section 4.2.4.4, wetlands connected to streams, isolated wetlands, and lake adjacent wetland types were included in HRU land use and vegetation classes. A *WETLAND_ADJ* soil profile was assigned to wetlands connected to stream and lake adjacent wetlands, and a

WETLAND_ISO soil profile was assigned to isolated wetlands. Land use and vegetation types in this scenario include *FOREST*, *LAKE*, *WATER*, *WETLAND_CON*, *WETLAND_ISO*, and *WETLAND_LAKE_ADJ*. As a large portion of isolated wetland inflows stems from local contributing areas based on topography, the contributing areas of each isolated wetland were delineated and used to estimate the percentage of the total watershed area contributing to isolated wetlands. Figure 4.14 shows HRU landcover types delineated for Scenario 3, in addition to the hydrologic processes structure. Thus, the complexity of wetland representation increases from Scenario 1 to Scenario 3. It is important to note that all wetlands in this study were conceptualized as groundwater recharge zones rather than groundwater discharge zones. Additionally, all wetlands are assumed to have some vegetation cover as canopy interception processes are included for all HRUs in the model.

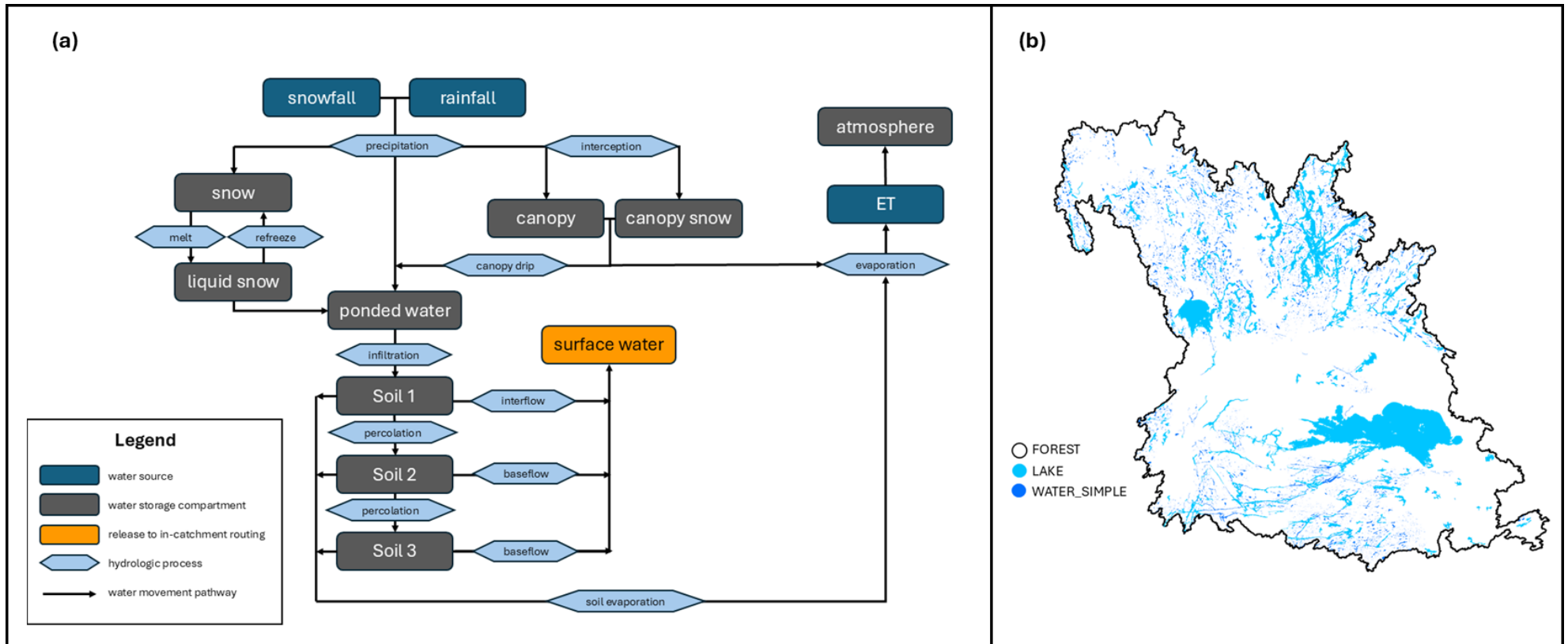


Figure 4.12 (a) Hydrologic processes flow chart for model Scenario 1, and (b) model Scenario 1 HRU land cover types

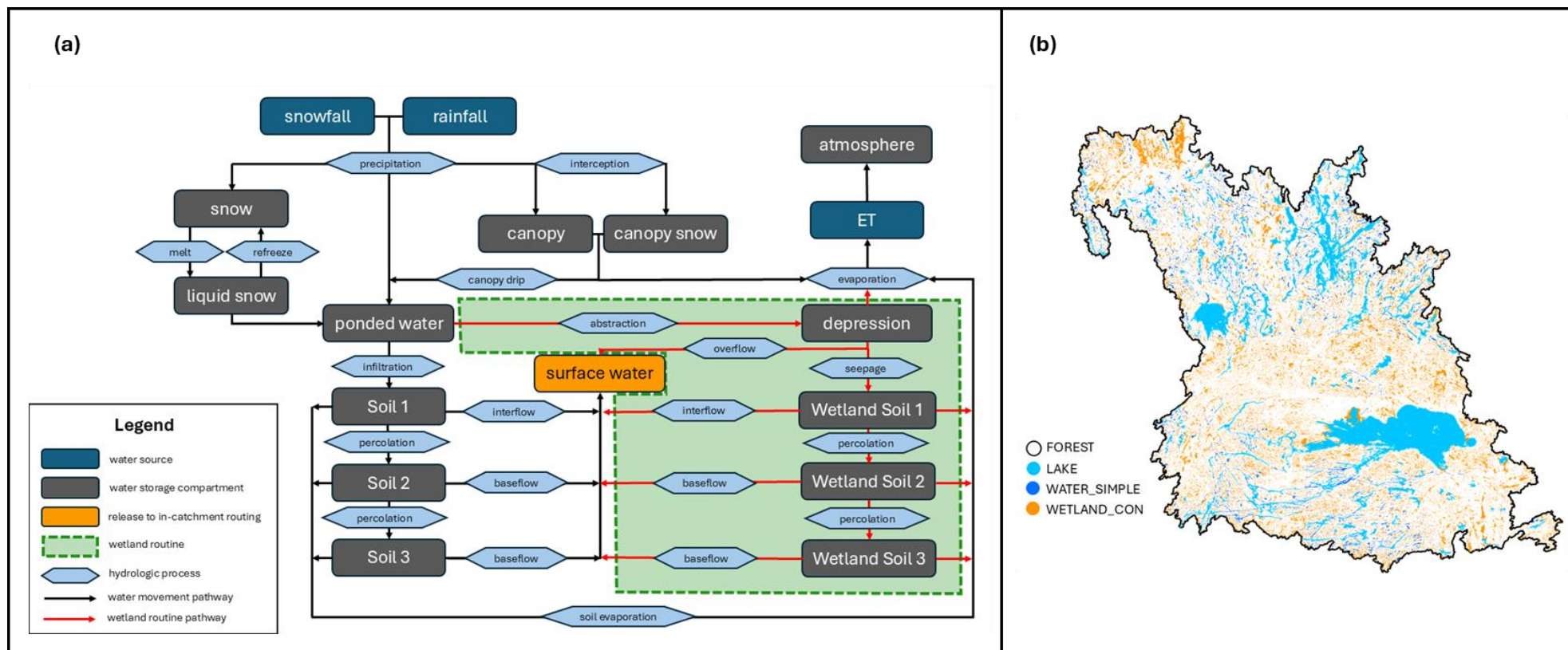


Figure 4.13 (a) Hydrologic processes flow chart for model Scenario 2, and (b) model Scenario 2 HRU land cover types

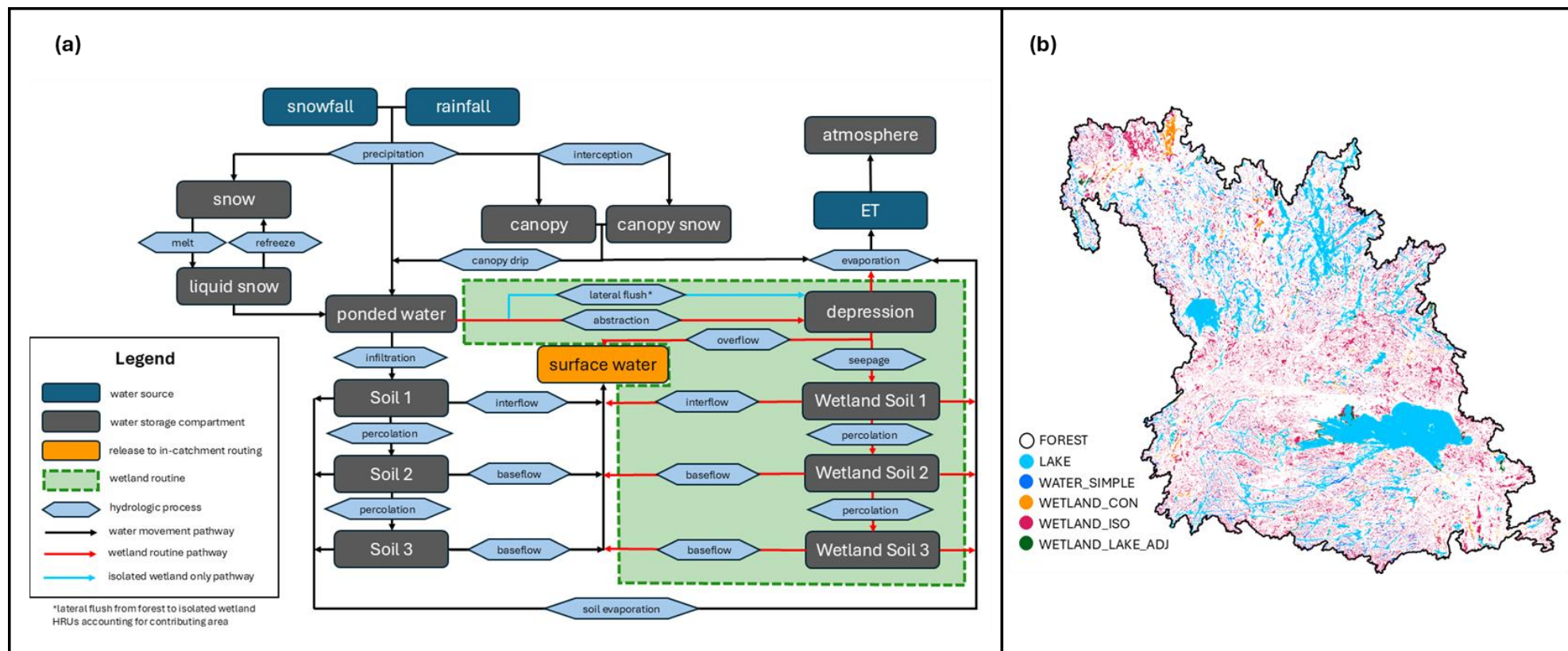


Figure 4.14 (a) Hydrologic processes flow chart for model Scenario 3, and (b) model Scenario 3 HRU land cover types

Although Scenario 3 contains two additional wetland HRU types compared to Scenario 2, the wetland routing processes shown in Figure 4.14 would be duplicated and applied to each wetland HRU type.

Isolated wetland contributing areas were delineated using the Ontario Hydro Network Watercourse flow accumulation raster (Ontario Ministry of Natural Resources and Forestry, 2010b) in ArcGIS. For each isolated wetland polygon centroid, the Watershed tool was used to determine the contributing area boundaries based on flow accumulation raster pixels draining to each isolated wetland centroid. The results indicated that 36% of the total watershed area contributes to isolated wetland polygons, compared to only 11% of the total watershed area being taken up by isolated wetland surface area included in the wetland polygon dataset. The importance of including isolated wetland contributing areas is emphasized here, as contributing areas are more than three times larger than the surface area of the isolated wetlands themselves. Thus, contributing areas of isolated wetlands were represented in Raven by including a lateral flush command which moves a percentage of water from *FOREST* HRUs to *WETLAND_ISO* HRUs. The percentage of water moved corresponds to the fraction of isolated wetland contributing area within the total area draining to flow gauges. Figure 4.15 displays the isolated wetland polygon centroids plotted with their calculated contributing areas.

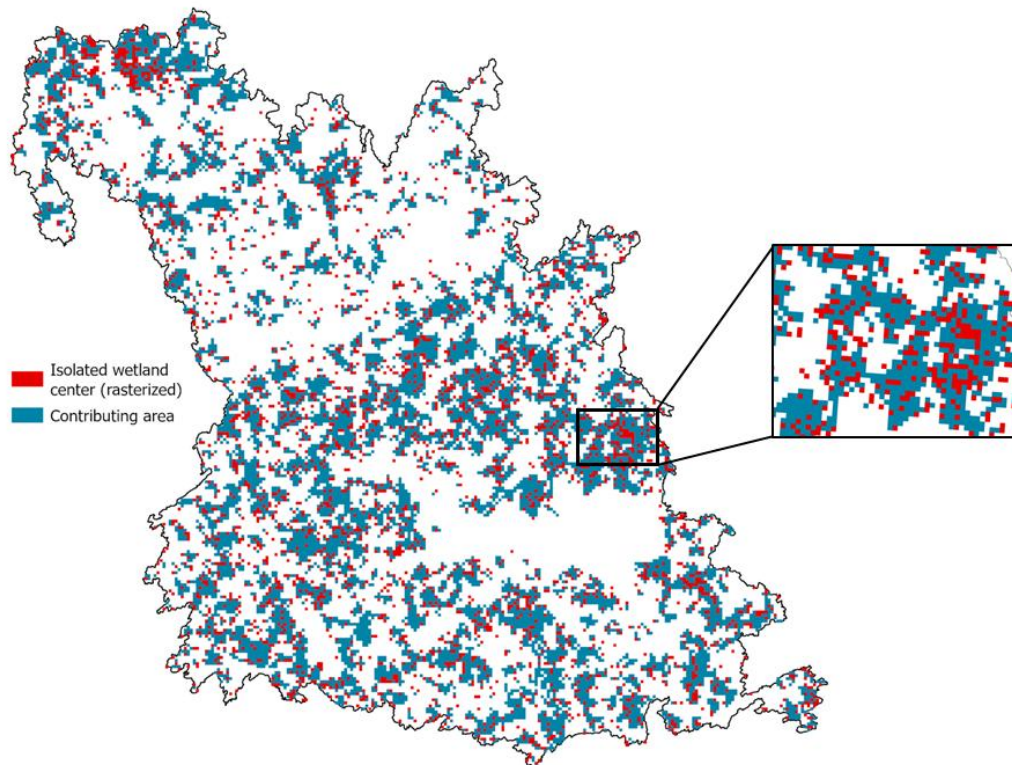


Figure 4.15 Isolated wetland polygon centroids plotted with their contributing areas

4.3 Model Calibration and Evaluation

Both manual parameter testing and sensitivity analysis were performed for each modelling scenario before calibration to determine sensitive parameters. An automatic multi-objective calibration was then performed for each modelling scenario, followed by validation using various performance metrics, level gauge performance, hydrographs, and flow duration curves.

4.3.1 Calibration

Calibration, simply put, is the process of tuning model parameters to minimize differences between simulated and observed data. During calibration, the model is trained to observation data using performance metrics that compare simulated to observed data.

Performance metrics are used to quantify the difference between the simulated model outputs and observation data and thereby evaluate model performance. However, there is no perfect performance metric, and each have their own advantages and disadvantages. It follows then, that the purpose of testing calibration experiments with various performance metrics is to diversify the model training by analyzing a variety of model performance indicators that make different comparisons between observed and simulated data. Depending on the calibration formulation, different aspects of the model (e.g., high flows, low flows, timing of peaks) will become the focus of its training and produce different simulation results.

Automatic calibration was performed using the Ostrich Optimization Software Toolkit (Ostrich) (Matott, 2017) with the goal of optimizing both high flows and low flows simultaneously. The purpose of optimizing both high and low flows was to train the model to simulate the full range of flow conditions, with expectations that wetland inclusion will influence both low flows and high flows.

The calibration experiment was performed using the Pareto Archived Dynamically Dimensioned Search (PADDs) multi-objective algorithm (Tolson & Shoemaker, 2006; Asadzadeh & Tolson, 2013). All scenarios were calibrated using three independent trials with random seeds in order to minimize uncertainty related to random selection of initial parameter values by the PADDs algorithm. A calibration budget of 2,500 iterations per trial was used to allow adequate iterations for PADDs to converge on a non-dominated solution set, or the solutions found by the multi-objective calibration algorithm that are not outperformed by any other solution. Non-dominated solutions in a

multi-objective calibration form what is called a pareto front, as there is no one best solution found but multiple optimizations that are not outperformed by other solutions.

In addition to manual parameter testing, a swing weight sensitivity analysis was performed for each modelling scenario to determine sensitive parameters for calibration. Parameters in each scenario were calibrated independently of other scenarios, with additional wetland parameters being added to the calibration in Scenarios 2 and 3. A total of 24 parameters were calibrated in Scenario 1, 39 parameters in Scenario 2, and 52 parameters in Scenario 3. Calibrated parameters for each modelling scenario, including their calibration ranges, are provided in Appendix D. Based on consistent data availability, a calibration period of nine years from 2010-01-03 to 2018-12-30 was selected. Based on data quality and degree of influence by regulated flows, six flow gauges were selected for calibration. Figure 4.16 shows the subset of flow gauges introduced in Section 4.2.3 used in calibration experiments, along with their upstream drainage areas.

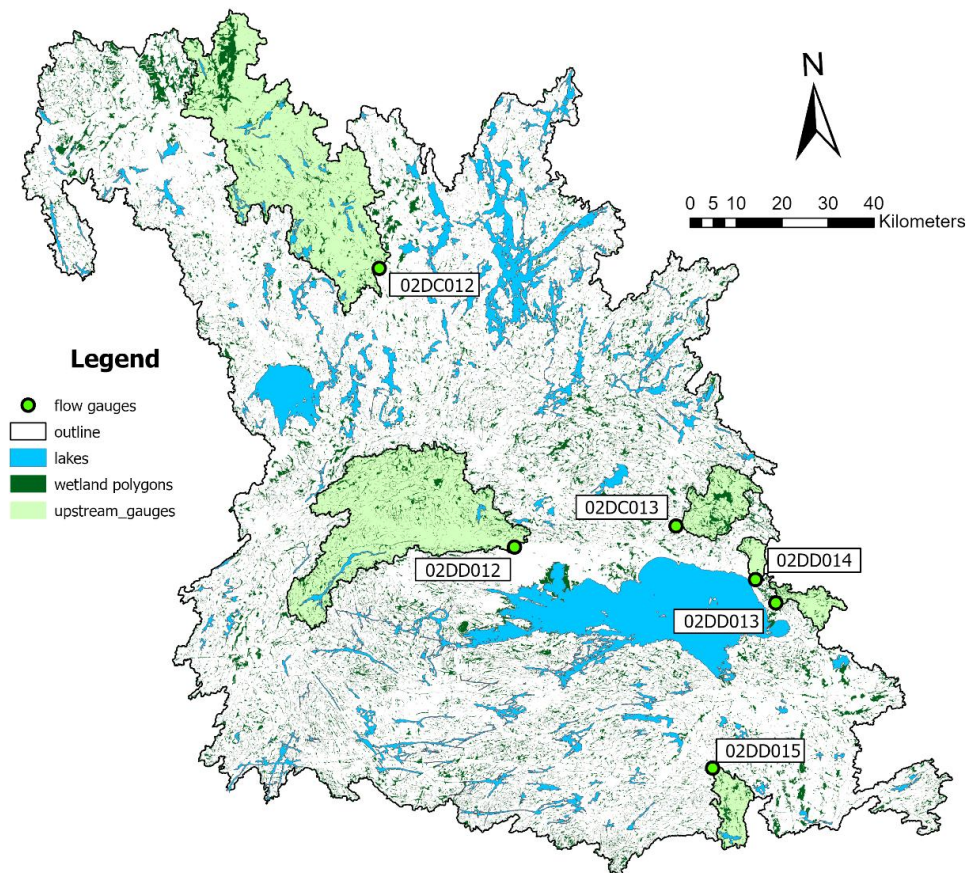


Figure 4.16. Flow gauges used in calibration experiments along with their upstream drainage areas and wetland polygon coverage.

Performance metrics are used in calibration as an objective function for the calibration algorithm to optimize. The calibration algorithm either minimizes or maximizes the objective function to solve for an optimal parameter set. Two performance metrics were optimized during the multi-objective calibration experiment: the median Kling Gupta Efficiency (KGE) and the median Nash Sutcliffe Efficiency (NSE) of logarithmically transformed flows (logNSE), both of which are maximized across all calibration flow gauges.

The NSE metric is widely used in hydrologic modelling and was first introduced in 1970 by Nash and Sutcliffe (Nash & Sutcliffe, 1970). The NSE is calculated using Equation 4.1 (Knoben et al., 2019; Nash & Sutcliffe, 1970):

$$NSE = 1 - \left[\frac{\sum_{i=1}^n (Q_i^{obs} - Q_i^{sim})^2}{\sum_{i=1}^n (Q_i^{obs} - \mu_{obs})^2} \right] \quad (4.1)$$

where Q_i^{obs} is the i^{th} observation value, Q_i^{sim} is the i^{th} simulated value, and μ_{obs} is the mean of observed data.

The NSE compares the difference between the residual data variance and the measured data variance using the ratio between the mean square error of the model and the variance of observed flows (Duc & Sawada, 2023; Santos et al., 2018; US Army Corps of Engineers, 2024). A perfect NSE score of 1.0 indicates a perfect model, and the worst NSE score is negative infinity. Using the mean of the observed timeseries in the NSE equation in place of simulated flows results in an NSE of 0. Thus, a negative NSE score indicates that using the mean observed time series value is a better match to observed data than model simulations (US Army Corps of Engineers, 2024). The first performance objective function was set to maximize the median logNSE of the six calibration flow gauges, which calculates the NSE using Equation 4.1 of the logarithmically transformed flow data. The logNSE evaluates model performance with an emphasis on low flows, therefore proving useful in models where capturing low flows is important (such as in wetland dominated regions) (Oudin et al., 2006). The logNSE was selected as a calibration metric in the Nipissing model to better evaluate the impact of low wetland flows on model performance.

The Kling Gupta Efficiency was proposed by Kling and Gupta in 2009 as an improved alternative to the NSE (Gupta et al., 2009). The KGE is calculated using Equation 4.2 (Gupta et al., 2009; Knoben et al., 2019):

$$KGE = \sqrt{(r - 1)^2 + \left(\frac{\sigma_{sim}}{\sigma_{obs}} - 1\right)^2 + \left(\frac{\mu_{sim}}{\mu_{obs}} - 1\right)^2} \quad (4.2)$$

where r is the linear correlation coefficient between observed and simulated values, σ_{sim} is the standard deviation in simulations, σ_{obs} is the standard deviation in observations, μ_{sim} is the mean of simulated values, and μ_{obs} is the mean of observed values.

The KGE functions similarly to the NSE, with the exception that the KGE was designed to address underestimation of variability and simultaneously assess the shape, timing, water balance, and variability of discharge data (Santos et al., 2018). However, one disadvantage of using the KGE is that scores are heavily penalized based on high errors (Santos et al., 2018). Similar to the NSE, the KGE ranges from negative infinity to 1.0, with a value of 1.0 representing a perfect match between simulated and observed values. A KGE of -0.41 indicates that the model performs the same as the mean of observed flows. Thus, to train the model to high flows and the general shape and timing of the observed hydrograph, the second objective function in the multi-objective calibration was set to maximize the median KGE of the six calibration flow gauges.

There is some variability in how “good” and “bad” NSE and KGE scores are defined throughout literature and in practice. Knoben et al. (2019) points out that NSE and KGE are inherently different performance metrics and are not directly comparable. Practitioners who use hydrologic models often define NSE and KGE thresholds that indicate useful model results. The most widely used threshold used to represent “not poor” NSE and KGE values is 0.5. NSE and KGE values of 0.5 and above are taken to indicate acceptable model performance (with additional checking of hydrographs and hydrology) and values below 0.5 indicate “poor” model performance. Similarly, a KGE and NSE value from 0.65 to 0.8 indicates strong performance, and 0.8 to 0.9 indicates excellent performance. For consistency with studies using NSE and KGE benchmarks of 0.5 to represent the cutoff between good and poor model performance (D. N. Moriasi et al., 2007; Fossey et al., 2015; Blanchette et al., 2019), the same threshold values were used in this study.

Percent bias (PBIAS) is an additional metric that was calculated to evaluate the performance at each calibration gauge. PBIAS is a performance metric used to measure the average tendency of simulated data to be larger or smaller than the observed data and can be used to indicate underestimations or overestimations of simulated flows compared to the observed (Gupta et al., 1999; Moriasi et al., 2015). PBIAS is calculated using Equation 4.3 (Moriasi et al., 2015):

$$PBIAS = \frac{\sum_{i=1}^n O_i - P_i}{\sum_{i=1}^n O_i} \times 100 \quad (4.3)$$

where n is the number of data points in the observed or simulated timeseries, O_i is the observed data point on day i of the simulation period, and P_i is the predicted data point on day i of the simulation period.

While PBIAS is useful in determining the average model simulation bias in the form of over or underprediction, it can also be deceiving in cases where the model overpredicts and underpredicts equally as often, resulting in a PBIAS close to zero despite the existence of model bias (Moriasi et al., 2015).

4.3.2 Validation

Validation was performed for each modelling scenario using all flow gauges shown in Figure 4.16 and evaluated using performance metrics. Additional validation was performed using flow duration curves, hydrographs, and the evaluation of level gauge performance at gauges 02DD022 and 02DD026 introduced in Section 4.2.3.

A validation period from 2005-01-01 to 2009-12-31 was chosen based on the availability of validation gauge data. Even still, gauges 02DC013 and 02DD012 did not contain full observation records during the validation period. Simulated data points at gauges that do not have corresponding observation data were not used in the calculation of model performance metrics.

Performance metrics used for model validation include the KGE, deviated KGE (KGE_D), NSE, logNSE, and PBIAS, with the addition of hydrographs and flow duration curves.

While the KGE was used to evaluate model performance at flow gauges in both calibration and validation periods, the deviated KGE was instead used to evaluate model performance at level gauges due to datum biases inherent in comparing simulated to observed level data. The deviated KGE is

calculated similarly to Equation 4.2, except that the bias term, or comparison of means between simulated and observed data, is omitted from the equation. This leaves the KGE deviation formula shown in Equation 4.4.

$$KGE_D = \sqrt{(r - 1)^2 + \left(\frac{\sigma_{sim}}{\sigma_{obs}} - 1\right)^2} \quad (4.4)$$

In addition to the described performance metrics, hydrographs and flow duration curves were generated across both the calibration and validation periods to further assess the model quality.

Chapter 5

Results

This Chapter presents calibration results based on the selection of a non-dominated calibration solution, in addition to validation results.

5.1 Calibration Results

The multi-objective calibration optimized to the median KGE and median logNSE of the selected calibration gauges produced Pareto trade-off curves for each random seed trial. Random seeds yield different trade-off curves due to their use of random initial parameter values that lead to different solutions. Each Pareto curve contains the non-dominated objective function solutions found during calibration.

To select the solution containing the “best” parameter values (the term “best” is subjective here as there is a trade-off between two objective functions), equal importance was given to both the median KGE and median logNSE objective functions. As a result, inflection points were selected as the best solution to maximize both high and low flows. The existence of Pareto uncertainty is recognized here, as any individual solution found along the Pareto curve contains unique parameter values that can lead to varying model results. However, for the purposes of this study in comparing the impact of model structure differences, the inflection point of the dominant Pareto curve was selected for analysis. The Pareto curves for each model scenario and trial are shown in Figure 5.1 along with the selected best solution.

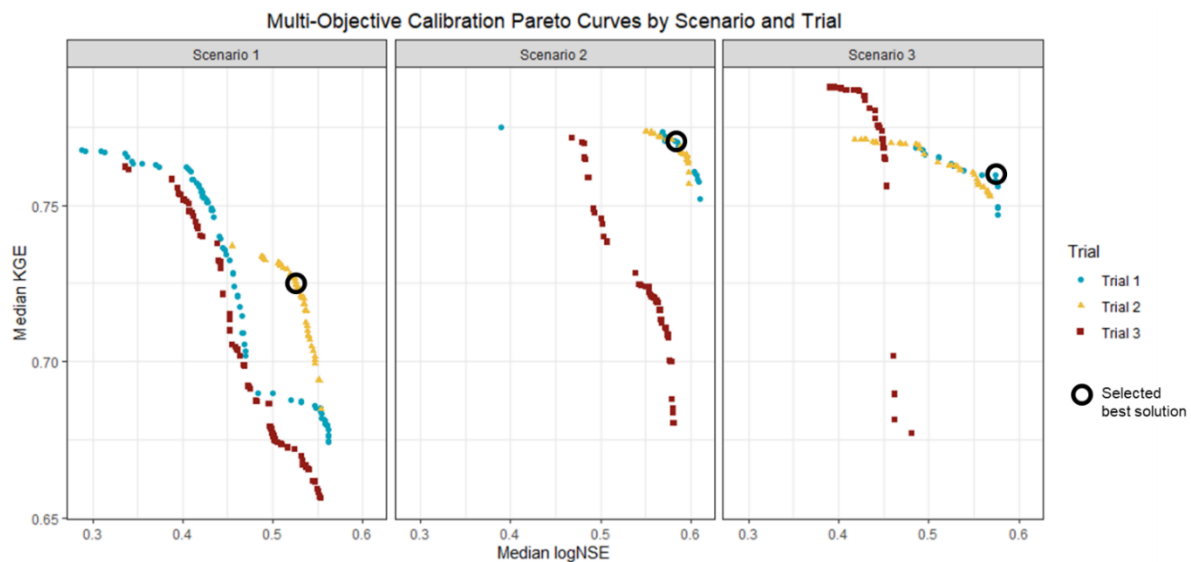


Figure 5.1 Multi-objective calibration pareto curves of non-dominated solutions for each model scenario and random seed trial

Figure 5.2 summarizes the calibrated objective function values reached for the best solution in each model scenario.

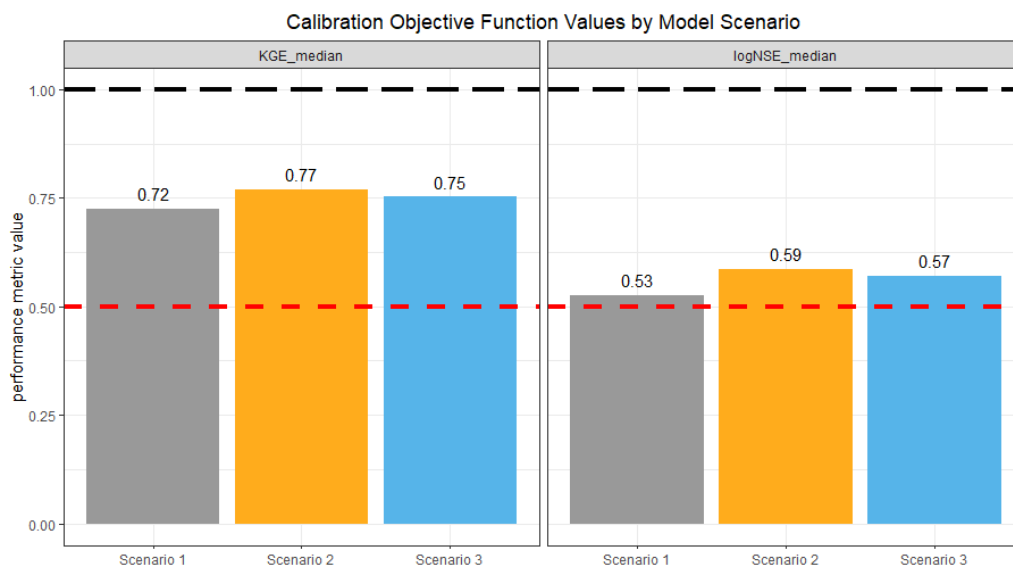


Figure 5.2 Calibration objective function values achieved during PADDs calibration for each model scenario best solution

From Figure 5.2, it is noted that in all model scenarios, an adequate median performance threshold of 0.5 for both the median KGE and logNSE is surpassed. Additionally, model Scenario 2 achieved the highest performing pair of objective function values, followed by Scenario 3 and then Scenario 1. However, it is worth noting that the differences between the objective function values is small and likely due to pareto solution uncertainty. To further understand differences in model performance, additional performance metrics were considered on a gauge-by-gauge basis.

Figure 5.3 shows the calculated performance metric values for each of the 6 calibration gauges.

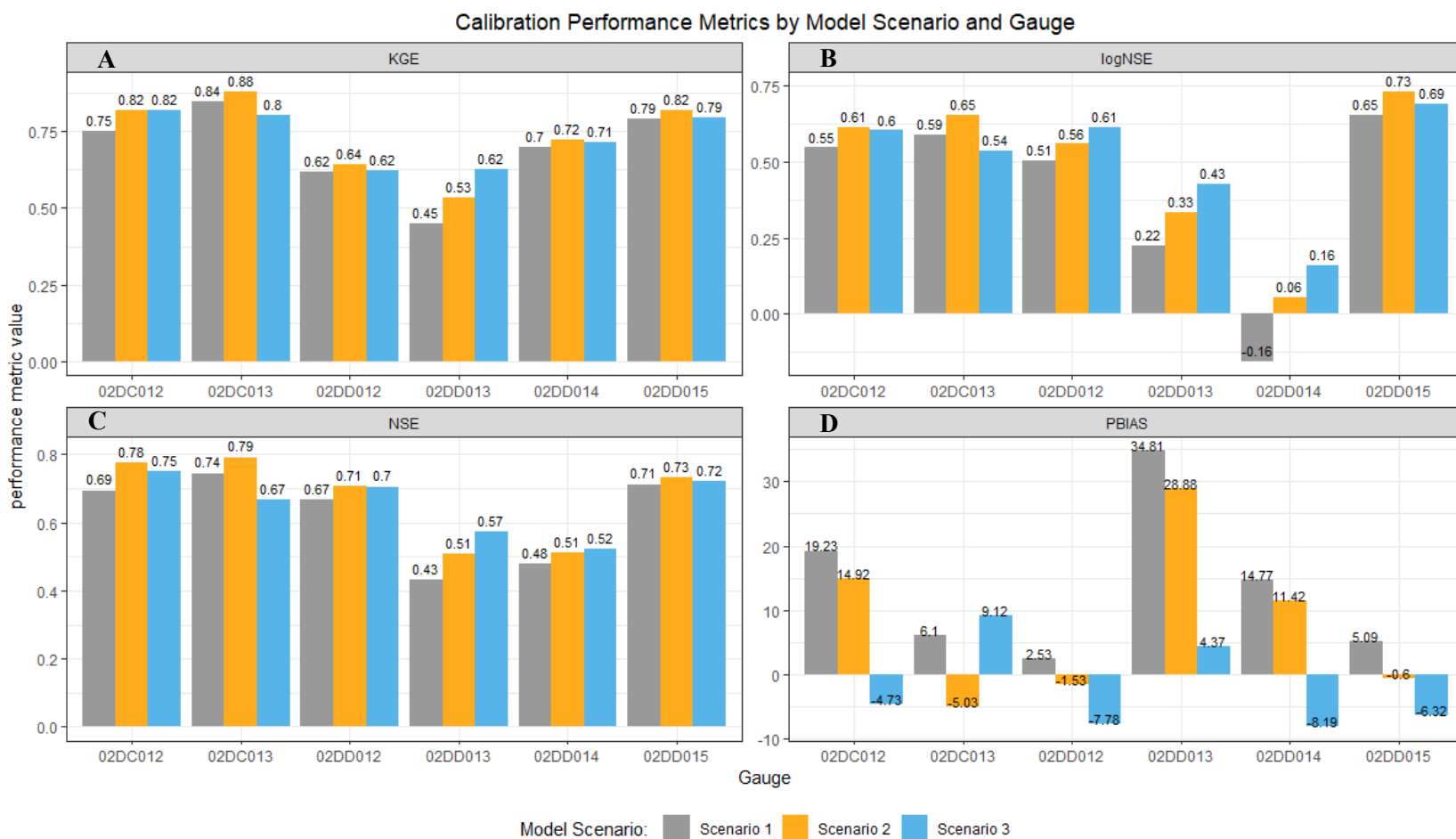


Figure 5.3 Model calibration performance metrics for each WSC calibration gauge. Panels A and C evaluate high flow simulations, panel B evaluates low flows, and panel D evaluates the model bias relative to observed data points.

In this analysis, panels A, B, and C of Figure 5.3 are interpreted by the step-like patterns of the bar plot between each scenario. An increasing staircase pattern (i.e., the performance metric value increases from Scenario 1 to Scenario 2 and from Scenario 2 to Scenario 3) indicates that adding wetland complexity increases the model resiliency based on the metric in question. A performance metric value increase from Scenario 1 to Scenario 2 and then decrease in Scenario 3 indicates that there is a trade-off between wetland complexity and model performance for that gauge. A step with a plateau (the performance metric value increases from Scenario 1 to Scenario 2 but stays the same between Scenario 2 and Scenario 3) indicates that adding wetlands to the model increases performance but adding wetland complexity does not. Lastly, a decreasing staircase pattern (the performance metric value decreases from Scenario 1 to Scenario 2 and from Scenario 2 to Scenario 3) indicates that adding wetland complexity degrades model resiliency and performance. Panel D is interpreted slightly differently, as zero is the best PBIAS value rather than 1.0 for KGE, NSE, and logNSE. As such, the larger the PBIAS bar, the worse the model accuracy. Panels A and C are discussed as high flow performance metrics, panel B is discussed as a low flow metric, and panel D is used to inform model over or underprediction.

Panel A in Figure 5.3 shows one instance of an increasing staircase pattern at gauge 02DD013, while gauges 02DC013, 02DD012, 02DD014, and 02DD015 show an increase from Scenario 1 to 2 and then decrease from Scenario 2 to 3, and 02DC012 shows a step plateau pattern.

Panel B shows three instances of increasing staircase pattern (02DD012, 02DD013, 02DD014) and three instances of an increase from Scenario 1 to 2 and then decrease from Scenario 2 to 3 (02DC012, 02DC013, and 02DD015).

Panel C shows two instances of increasing staircase pattern (02DD013 and 02DD014) and four instances of an increase from Scenario 1 to 2 and then decrease from Scenario 2 to 3 (02DC012, 02DC013, 02DD012, and 02DD015).

Lastly, panel D shows that at half of the gauges (02DC012, 02DD013 and 02DD014) Scenario 3 produces less bias than both Scenario 1 and Scenario 2. For the other half of gauges (02DC013, 02DD012, and 02DD015) Scenario 2 produces less bias than the other two scenarios, while Scenario 3 produces the most bias.

An interpretation of these results is presented in Chapter 6.

5.2 Validation Results

Similar to the calibration results, validation results are presented using a comparison of performance metrics between model scenarios and individual gauges. Figure 5.4 compares validation period performance metrics at each of the six flow gauges for each model scenario. Validation results are considered to be more reflective of actual model skill because they do not have artefacts of overfitting that calibration results will have.

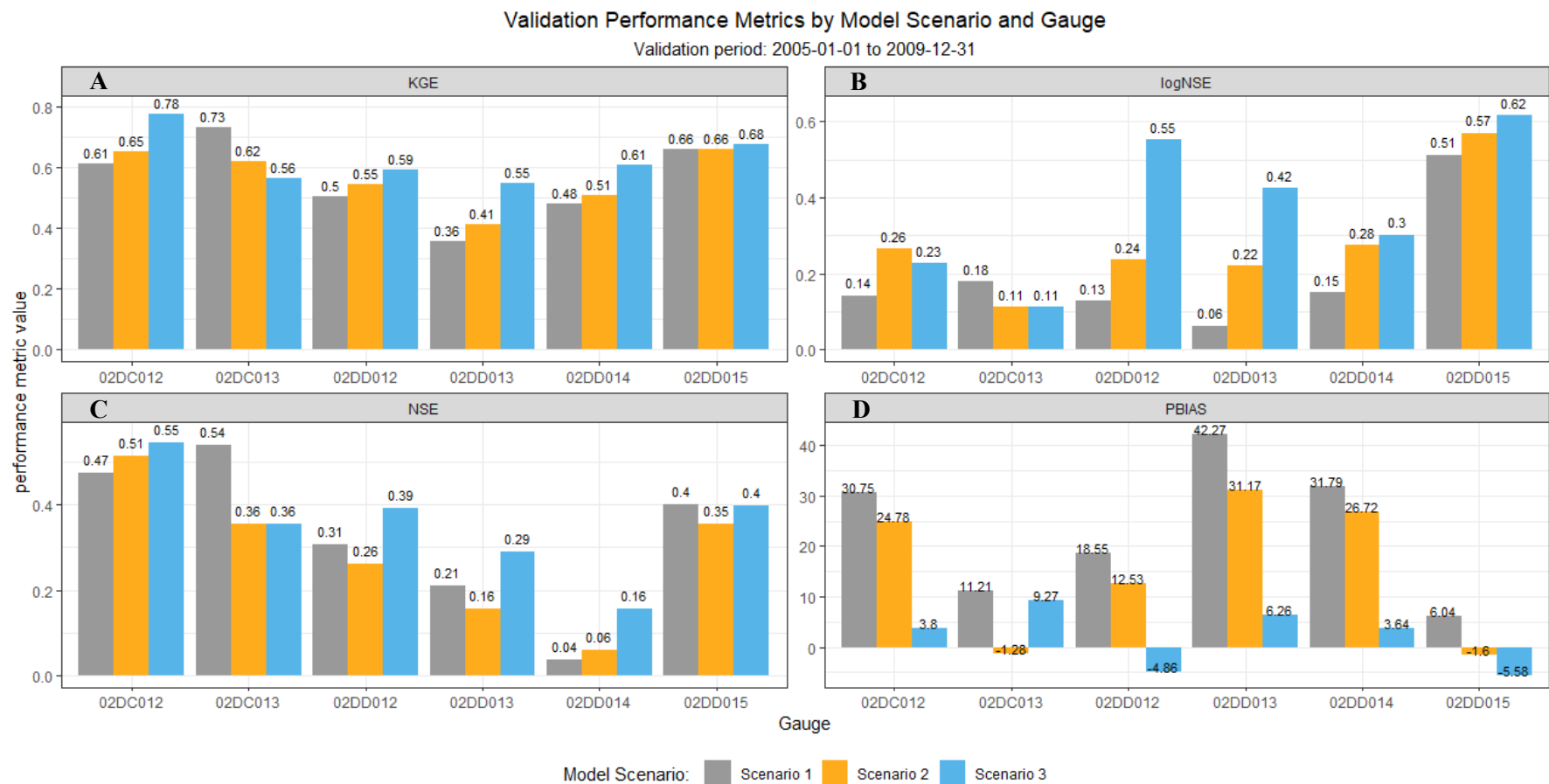


Figure 5.4 Model performance metrics for each WSC calibration gauge during the validation period (2005-2009). Panels A and C evaluate high flow simulations, panel B evaluates low flows, and panel D evaluates the model bias relative to observed data points.

Using the same step pattern interpretation as for Figure 5.3, Panel A in Figure 5.4 shows five out of six stream gauges with an increasing staircase pattern (02DC012, 02DD012, 02DD013, 02DD014, and 02DD015), while gauge 02DC013 shows a decreasing staircase pattern.

Panel B shows four instances of increasing staircase pattern (02DD012, 02DD013, 02DD014, and 02DD015), one instance of an increase from Scenario 1 to 2 and then decrease from Scenario 2 to 3 (02DC012), and one instance of a decreasing staircase pattern at 02DC013.

Panel C shows two instances of increasing staircase pattern (02DC012 and 02DD014), one instance of decreasing staircase pattern (02DC013), and three instances of an inconclusive pattern where performance decreases from Scenario 1 to Scenario 2 but increases from Scenario 2 to Scenario 3 (02DD012, 02DD013, and 02DD015).

Lastly, panel D shows that at four of the six gauges (02DC012, 02DD012, 02DD013, and 02DD014) Scenario 3 produces less bias than both Scenario 1 and Scenario 2. For the other two gauges (02DC013 and 02DD015) Scenario 2 produces the least amount of bias, while Scenario 1 produces the most bias.

5.2.1 Comparison of Calibration and Validation Results

Comparing calibration and validation results presented in Figure 5.3 and Figure 5.4 shows that overall, the performance metric values in panels A, B and C decrease from the calibration to the validation period. This was an expected result, as the model high and low flow simulation is trained to the calibration period and not the validation period. In general, between panels A, B and C in the calibration versus validation period, the validation period shows an increasing staircase pattern in almost double the number of gauges as in the calibration period (6 increasing staircase pattern in the calibration period versus 11 in the validation period).

Panel A during the calibration period shows that for the majority of gauges, Scenario 2 was the most accurate predictor of high flows while in the validation period, Scenario 3 was the most accurate high flow predictor for the majority of gauges.

Panel B during the calibration period shows an equal percentage of gauge low flows (50%) being best simulated by either Scenario 2 or Scenario 3. During the validation period, more than half of gauge low flows are best simulated by Scenario 3, with one gauge showing a better low flow simulation in Scenario 2 and one gauge showing decreasing low flow performance with wetland complexity.

Panel C during the calibration period shows that at more than half of gauges, Scenario 2 is a better predictor of high flows and Scenario 3 only one third of the time. During the validation period, more than half the time Scenario 3 was a better high flow predictor, however, with some gauges displaying a unique pattern of performance degradation from Scenario 1 to 2 and improvement in Scenario 3. One gauge in the validation also indicates no improvement in model performance with increasing wetland complexity.

Lastly, panel D during the calibration period shows that for half of the flow gauges, Scenario 2 produces the smallest magnitude of bias, with an underestimation of flows magnitudes and Scenario 3 produces the smallest bias magnitude for the other half with an overestimation of flows. The validation period sees Scenario 3 bias decrease on average and produces the smallest bias magnitude for more than half of the flow gauges.

Based on a gauge-by-gauge comparison, the calibration period was more accurately simulated at most gauges with Scenario 2, and the validation period was more accurately simulated with Scenario 3 at most gauges. Figure 5.5 displays performance statistics ranged over the 6 calibrated flow gauges in each model scenario.

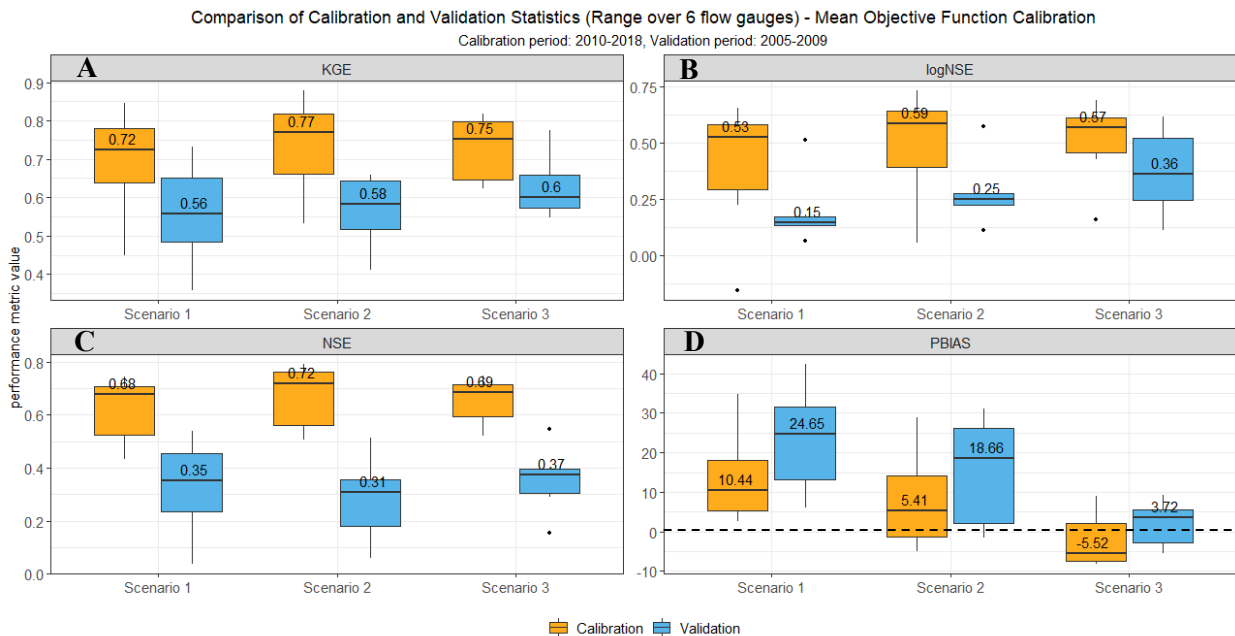


Figure 5.5 Calibration and validation period statistics comparison ranged over the 6 calibrated flow gauges

Again, the clearest observation is the degradation in median model performance from the calibration to validation period. There is one exception to this degradation – in panel D, the median PBIAS magnitude in Scenario 3 decreases from calibration to validation, indicating less flow simulation bias during the validation period. The median PBIAS in the validation period significantly decreases from Scenario 1 to Scenario 3, with a 6% decrease from Scenario 1 to 2 and a 15% decrease from Scenario 2 to 3. It is also noted that the spread in validation PBIAS across all gauges decreases from Scenario 1 to Scenario 3.

All panels show calibrated median performance metric values improving from Scenario 1 to Scenario 2 and degrading slightly from Scenario 2 to 3. In all cases, the calibrated performance metric values in Scenario 3 are improved from Scenario 1. In contrast, for all panels, the validation period median performance metric values improve from Scenario 1 to Scenario 3 in a staircase pattern (increasing for panels A, B and C and decreasing for panel D).

5.2.2 Lake Level Gauge Validation

An additional check was performed during the validation period to assess each model scenario's ability to replicate lake level observations. Lake levels at gauges 02DD022 and 02DD026 were simulated and model performance was measured using the KGE deviation. Both 02DD022 and 02DD026 are located on regulated lakes, with 02DD026 existing on the French River downstream of Lake Nipissing, Chaudière Dam, and Portage Dam and 02DD022 existing on Dollars Lake. Despite being regulated, validation of level gauges is useful identifying the impact of model structure on lake processes. Validation results for 02DD022 and 02DD026 using KGE deviation are presented in Figure 5.6.

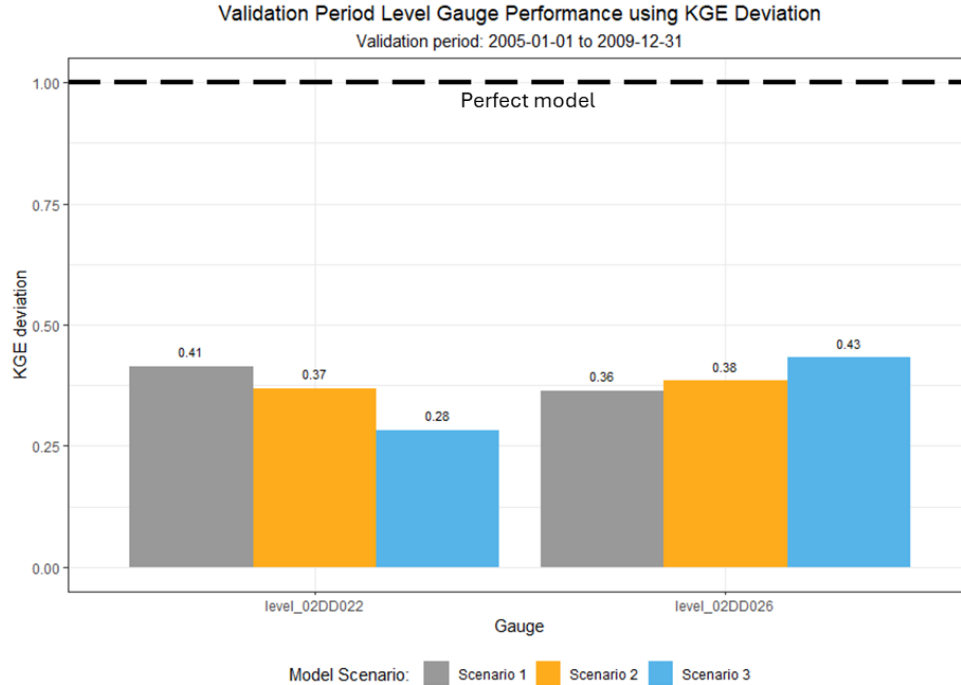


Figure 5.6 Validation period simulation of level gauges using KGE deviation

It is first observed that the KGE deviation for the simulated level gauges is much lower (below 0.5) than the calibrated and validated flow gauge KGE values. This result is expected as the model is not able to capture the effects of regulation on water levels. Gauge 02DD022 sees a decreasing staircase pattern with Scenario 1 scoring the highest KGE_D , and Scenario 3 scoring the lowest. Conversely, gauge 02DD026 experiences the opposite effect, with an increasing staircase pattern indicating that the highest KGE_D value is achieved by Scenario 3 and lowest value is achieved by Scenario 1.

5.3 Flow Duration Curves

Flow duration curves (FDCs) are used to display the percentage of time (or probability) of streamflow magnitudes being equaled or exceeded. The most accurate streamflow magnitude simulation is one that closely follows the observed FDC line, while simulated FDCs that deviate from the observed line are less capable of simulating streamflow magnitudes accurately. For each stream gauge and model scenario, simulated FDCs were plotted over both calibration and validation periods and compared to the observed FDC, as shown in Figure 5.7. The FDCs were plotted using both the calibration and validation period simulated data because FDCs require as much data as possible to be useful for

analysis. Since the validation period only contained 5 years of simulated flows, the calibration period was added to span a total of 14 years (2005 to 2018, inclusive).

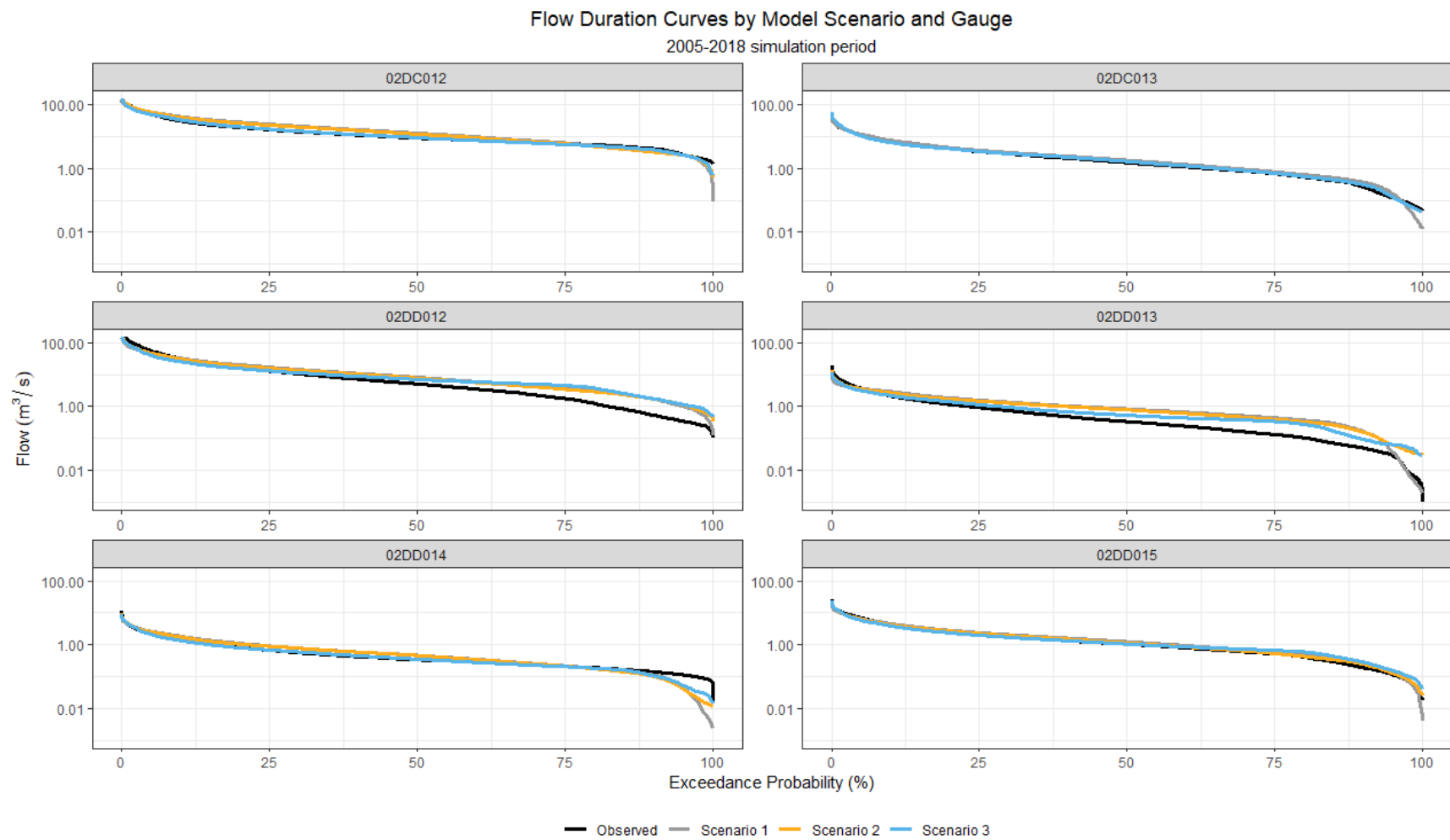


Figure 5.7 Flow duration curves plotted across both calibration and validation periods by model scenario and flow gauge

The flow duration curves are interpreted here from left to right, with the lowest probability of exceedance indicating the highest flows and the highest probability of exceedance indicating the lowest flows. Model scenario curves that fall below the observed line tend to underpredict flows at a particular magnitude, and scenario curves that lie above the observed line tend to overpredict flows at that magnitude.

For most gauges, high flows (< 25% exceedance probability) are fairly well simulated by the matching of each scenario FDC to the observed line. High flows at gauges 02DC012, 02DC013, 02DD013, and 02DD014 are overestimated by all scenarios. In these cases, Scenario 3 is seen to more closely follow the observed line while Scenarios 1 and 2 deviate faster. Underestimation of high flows is seen at gauges 02DD012 and 02DD015, with roughly similar results for all scenarios.

Low flows (> 75% exceedance probability) are less well represented than the high flows. Gauge 02DC012 and 02DD014 see all scenarios underestimate low flows, with Scenario 3 being the closest match to observed flows, then Scenario 2, then Scenario 1. Gauges 02DD012 and 02DD013 see all scenarios overestimate flows, with Scenario 1 being the closest match to observed flows, then Scenarios 2 and 3 (which show similar behaviour to each other). Gauge 02DD015 low flows are best estimated by Scenario 2, with underpredictions by Scenario 1 and overpredictions by Scenario 3. Lastly, 02DC013 low flows are well simulated by Scenarios 2 and 3, with Scenario 3 providing a close match to the observed. Scenario 1 overestimates at an exceedance probability of about 90% and 100%.

The most notable differences between observed and simulated FDCs occurs at gauges 02DD012, 02DD013, and 02DD014. These differences are further discussed in Chapter 6.

5.4 Hydrographs

Hydrographs are a timeseries plot of flow over the model simulation period used to assess the timing, peak flows, low flows, and general hydrology of simulation results compared to the observed data. Hydrographs are presented for each modelled flow gauge over the validation and calibration periods for all modelling scenarios in Figure 5.8 to Figure 5.13.

Hydrograph over Calibration and Validation Periods

Gauge 02DC012

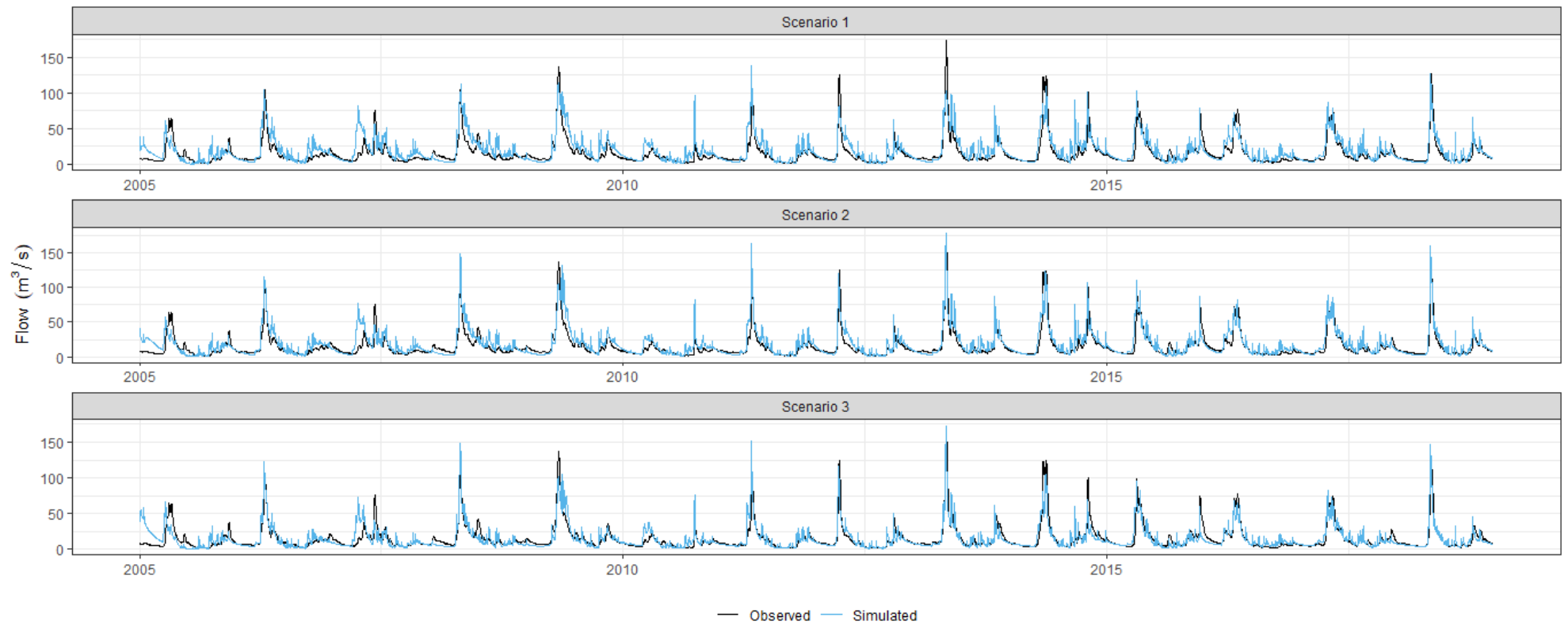


Figure 5.8 Flow gauge 02DC012 hydrograph for all model scenarios

Hydrograph over Calibration and Validation Periods
Gauge 02DC013

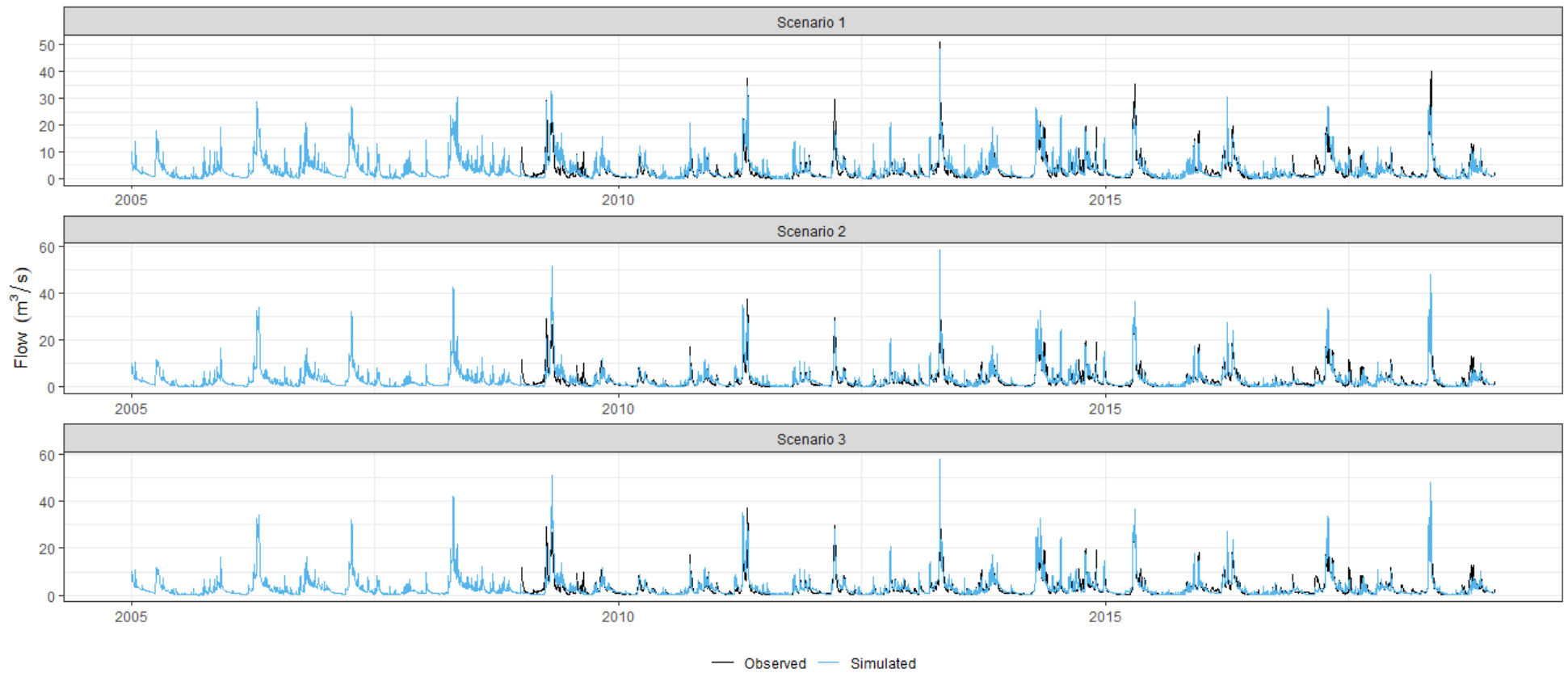


Figure 5.9 Flow gauge 02DC013 hydrograph for all model scenarios

Hydrograph over Calibration and Validation Periods

Gauge 02DD012

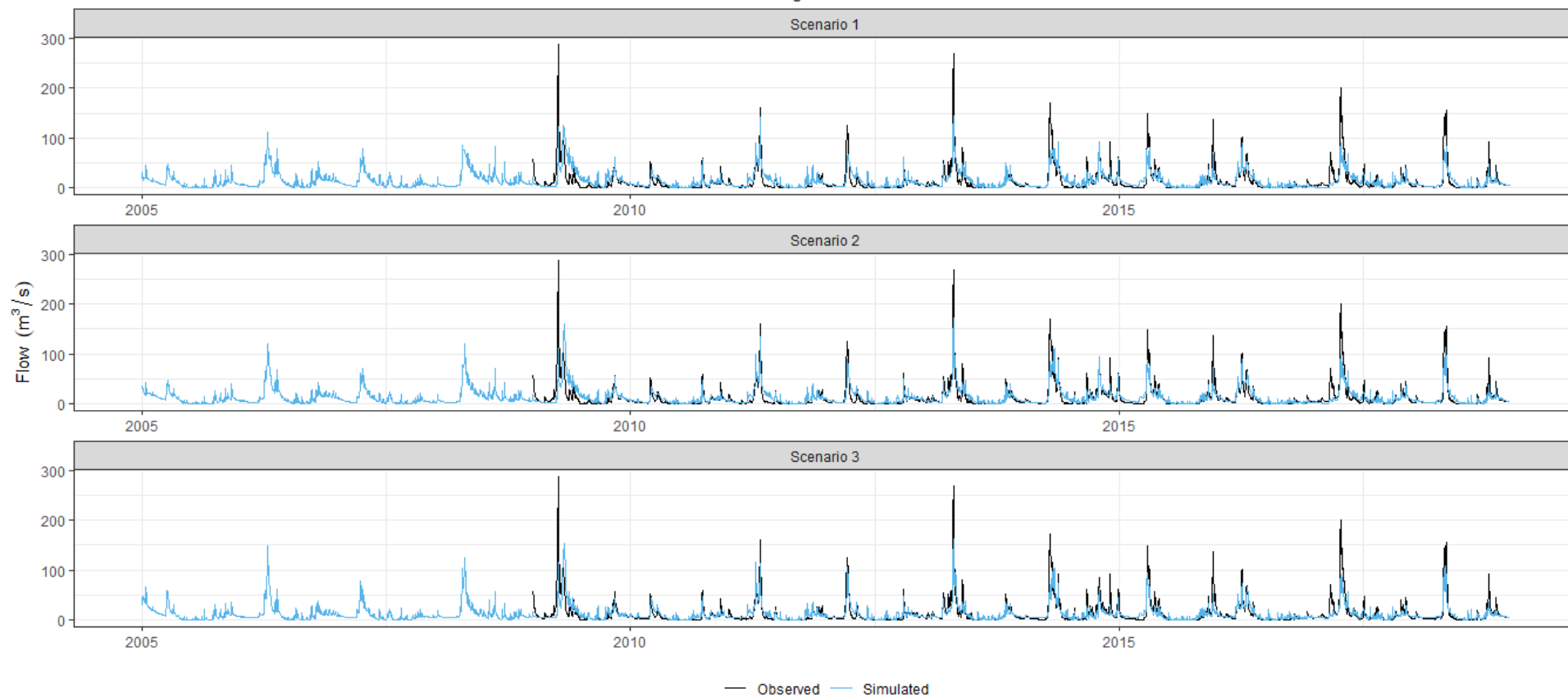


Figure 5.10 Flow gauge 02DD012 hydrograph for all model scenarios

Hydrograph over Calibration and Validation Periods

Gauge 02DD013

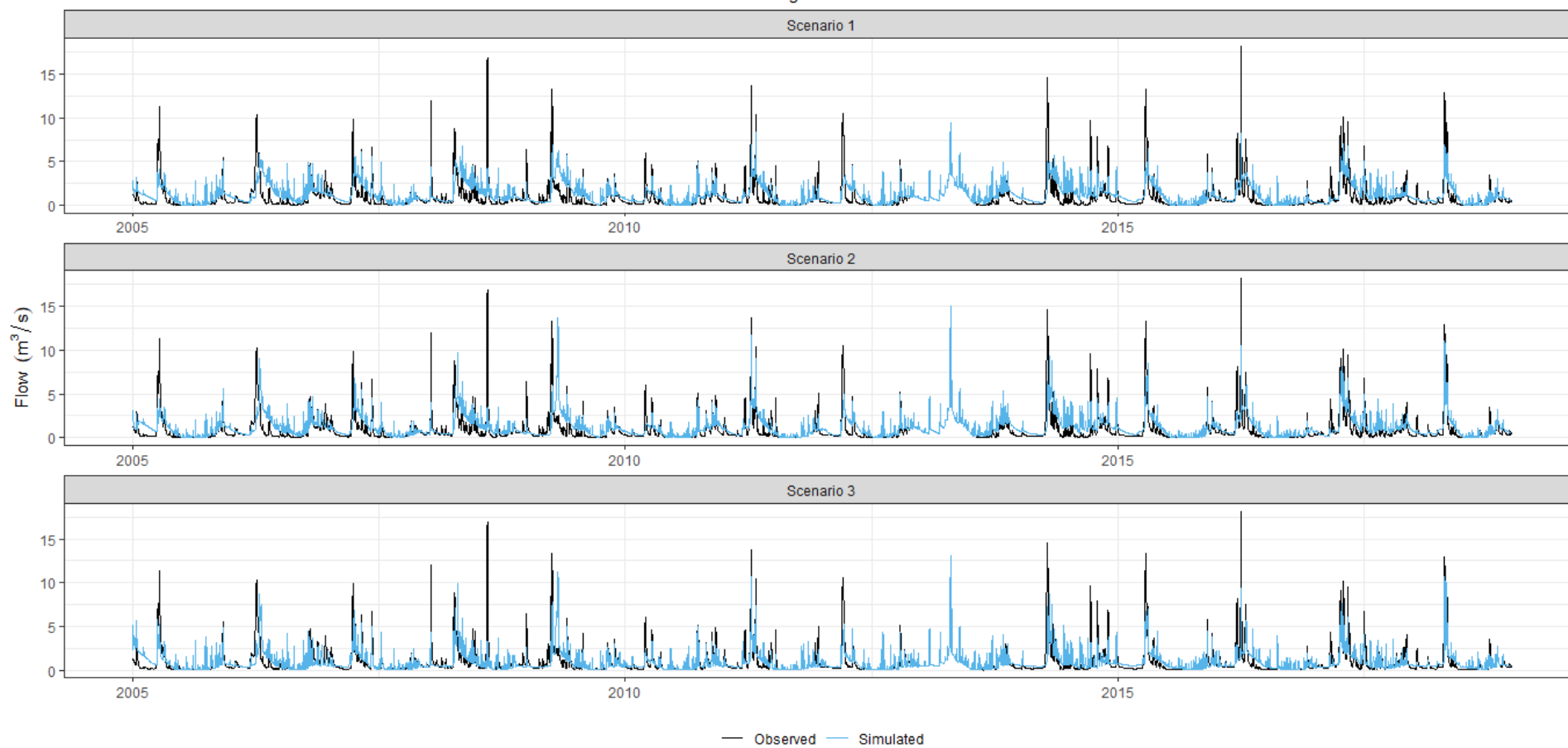


Figure 5.11 Flow gauge 02DD013 hydrograph for all model scenarios

Hydrograph over Calibration and Validation Periods

Gauge 02DD014

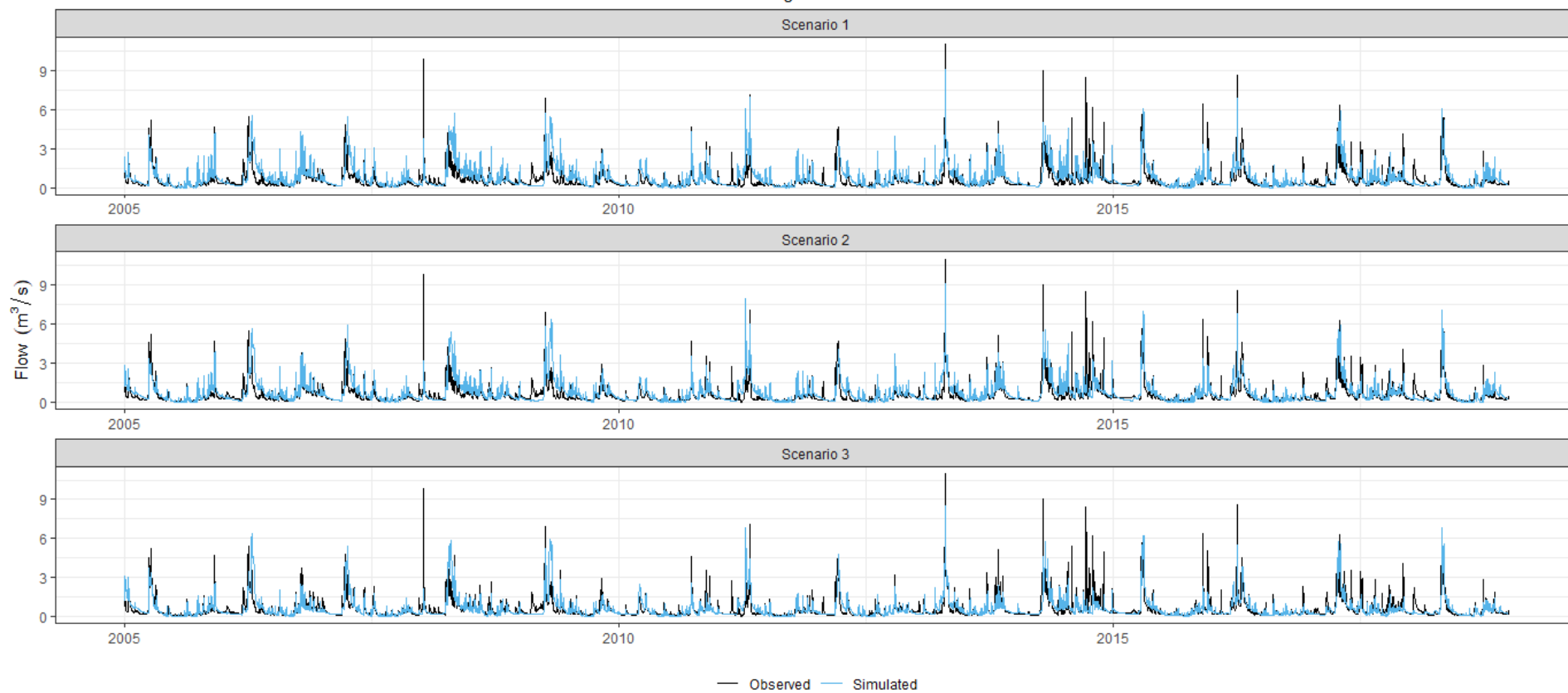


Figure 5.12 Flow gauge 02DD014 hydrograph for all model scenarios

Hydrograph over Calibration and Validation Periods

Gauge 02DD015

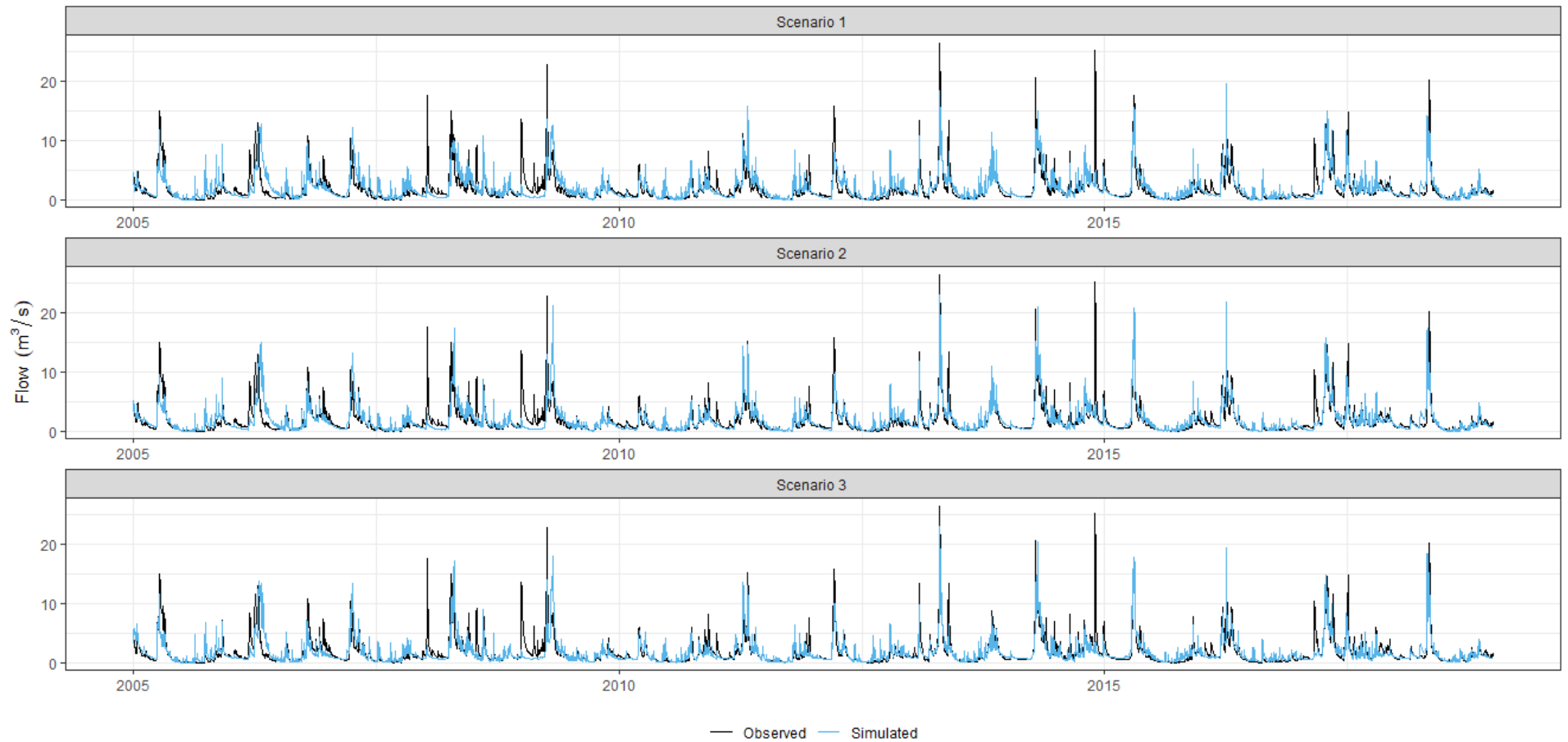


Figure 5.13 Flow gauge 02DD015 hydrograph for all model scenarios

Many of the simulated hydrographs (between all model scenarios) fail to accurately represent the magnitude of several high peaks shown in the observation data, such as the several missed peaks at 02DD013, 02DD014, and 02DD015. Results consistent across all model scenarios are identified as an artifact of both model structure and input data uncertainty, as these shortcomings are consistently seen across any of the three model simulations regardless of the presence or absence of wetlands. Hydrograph timing is another example of a hydrograph feature that appears consistent across model scenarios.

However, the magnitudes of some peaks do change between scenarios. One example is the peak flow reached right before 2010 at gauge 02DC013, where the peak magnitude is better captured by Scenario 1 than the two wetland scenarios. Conversely, the magnitude of the large peak right before 2010 at gauge 02DD013 is better simulated by Scenarios 2 and 3 rather than Scenario 1. Across all plots, there is no clear pattern of improved peak flow simulation between any given scenario.

The simulation of low flows also varies between scenarios. Across all gauges, the simulation of low flows improves with added wetland complexity. The most noticeable examples occur at gauge 02DD012 and 02DD013 immediately after the last large peak of 2009. In these cases, the low flow values following the peak are better captured by the wetland scenarios and in particular, Scenario 3. When comparing the details of each hydrograph, a similar trend of low flow simulation is observed.

Looking at the hydrograph shape also reveals some differences between model scenarios. In some areas, such as at gauge 02DC012 in about 2013-2014 and at gauge 02DD013 in 2015, the hydrograph shape improves with increasing model complexity. The opposite can be said about much fewer cases, if any at all, where Scenario 1 hydrograph shape is better than Scenario 2 and 3 hydrograph shape. Smaller differences between the hydrograph shape are likely present between all scenarios, with examples of each scenario dominating the others in terms of performance. However, over the long simulation period, these instances even out and do not contribute significantly to overall performance at each gauge.

Chapter 6

Discussion

6.1 Insights from Calibration and Validation

Based on calibration results and Figure 5.3, most often a trade-off between wetland complexity and performance was observed by Scenario 2 receiving the best-performing scores for multiple metrics. As a result, both high and low flows during the calibration period were generally better simulated by Scenario 2. This is in part attributed to pareto solution uncertainty. Depending on the choice of pareto solution (i.e., any solution other than the chosen inflection point), individual gauge performance varies as one objective function is prioritized over the other at any given solution. On the other hand, assuming pareto solution uncertainty is minimal, instances where Scenario 2 outperforms the other two scenarios at individual gauges indicates a trade-off between adding wetland complexity and achieving high model performance. In other words, model performance increases from Scenario 1 to Scenario 2 with the addition of wetlands and then decreases in Scenario 3 when wetland complexity increases. This result also likely depends on the study area and site-specific properties of each simulated drainage area, as the complexity of wetland simulation required for an acceptable model result will vary with drainage area size, wetland types, and soil type.

Based on validation results shown in Figure 5.4 and Figure 5.5, Scenario 3 performs better for median high and low flows and performance metrics for both high and low flows at individual gauges more often outperforming the other two scenarios. Physically, this means that the addition of wetland complexity, especially accounting for geographically isolated wetland contributing area, improves flow simulations through increased water retention in wetlands (increased depression storage), slower moving water through wetland systems, and groundwater interaction. The routing of water to depression storage increases from Scenario 1 to Scenario 2 by the inclusion of wetlands, and from Scenario 2 to Scenario 3 by the lateral diversion of water based on isolated wetland contributing area. This increased water retention in the wetland scenarios could explain the behaviour seen in the validation period at outlier gauges where Scenario 1 outperforms both Scenario 2 and Scenario 3. One example of a validation outlier gauge seen in Figure 5.4 occurs in panel A at gauge 02DC013, where the Scenario 1 KGE is 0.73 and deteriorates to 0.62 in Scenario 2 and even further to 0.56 in Scenario 3. It is also observed that across all gauges shown in Figure 5.4, panel A shows all Scenario 3 KGE

values above the adequate performance threshold of 0.5, while Scenario 1 and Scenario 2 occasionally dip below the threshold.

One additional observation can be drawn from Panel D of Figure 5.5, where the spread in PBIAS across model scenarios in both the calibration and validation period decreases from Scenario 1 to Scenario 3. Specifically, the validation period experiences a significant drop (-15%) in median PBIAS from Scenario 2 to Scenario 3 and large reduction in PBIAS spread. In addition, the PBIAS spread and median both increase substantially from the calibration to the validation period in Scenario 1 and Scenario 2; however, the opposite is observed in Scenario 3. The median PBIAS and PBIAS spread from calibration to validation in Scenario 3 decreases in magnitude and switches from an underprediction to an overprediction of observed flows. The overall decrease in median and spread of PBIAS from Scenario 2 to Scenario 3 is indicative of low flows being better captured by Scenario 3 through lateral diversion of isolated wetland contributing area to depression storage.

Flow duration curves are more inconclusive of model performance, as all scenarios exhibit instances of over and underestimation of flows. In general, high flows are well represented by all modelling scenarios and low flows are more poorly represented. At gauges 02DC012, 02DC013, and 02DD014 low flows are best represented by Scenario 3 due to wetland presence and increased water retention compared to the other modelling scenarios. The observed flow duration curve at gauge 02DD013 displays divergent behaviour from the wetland scenarios, as Scenarios 2 and 3 are not able to capture the low flow magnitudes. In this case, Scenario 1 with no wetland simulation best captures the low flow magnitudes. Large discrepancies between the wetland scenarios and observed data may be due to site specific properties. For example, at the time of this thesis being written, a site visit of gauges 02DD013 and 02DD014 revealed very slow moving (almost stagnant) water being measured at 02DD013 and very shallow water measured at 02DD014 through an urban park in the city of North Bay. Site photos of the measured streams and gauge measurement equipment boxes are shown in Figure 6.1.



Figure 6.1 Photographs of flow gauge locations near North Bay, Ontario taken by Madeline Tucker on July 11, 2025. Panel (a) shows the flow measurement area for gauge 02DD013, (b) shows the measurement equipment box for gauge 02DD013, (c) shows the creek immediately upstream of the flow measurement area for gauge 02DD014, and (d) shows the measurement equipment box for gauge 02DD014.

Instances of wetland scenarios overpredicting low flow magnitudes, such as at gauges 02DD013 and 02DD012, may indicate groundwater influence that is simulated by the wetland representation but not picked up by the surface water gauges. This would result in a bias between the observed and simulated scenarios, with a greater impact on wetland scenarios due to the addition of wetland baseflow and interflow parameters. Instances of wetland scenarios underpredicting low flow magnitudes, such as at gauge 02DD014, indicates that wetland processes in the model retain more water than the actual wetlands within the drainage area due to the lateral diversion of water from isolated wetland contributing areas to depression storage being calculated with a basin-wide diversion percentage not specific to any gauge drainage area.

The hydrographs presented in Figure 5.8 to Figure 5.13 indicate across all gauges the improved simulation of low flows and overall hydrograph shape with increasing wetland complexity (Scenario 3). This is again a result of increased water retention by wetland storage and increased groundwater interaction with wetland systems. Some uncommon instances of flow peaks being better represented by Scenario 1 than the wetland scenarios are likely an artefact of pareto solution uncertainty.

Hydrographs from additional pareto solutions for each modelling scenario were compared and showed variability in model scenario ability to capture the peak flow magnitudes. It is also noted that all modelling scenarios are observed to have an oversensitivity to summer rainfall events, indicating a model structural issue that was not corrected by calibration.

Level gauge simulations were not thoroughly analyzed in this study due to a lack of lake level gauges appropriate for validation given their period of record and presence of water regulation. As shown in Figure 5.6, simulated lake level performance varies with modelling scenario and gauge, resulting in the inability to make physically based conclusions. However, the use of lake level gauges in validation does provide insight into model performance with respect to water level regulation. In all modelling scenarios and for both lake level gauges used for validation, the KGE_D value remains below the established acceptable model performance threshold of 0.5. This failure to meet an acceptable performance threshold is indicative of the model's inability to simulate lake levels of regulated water bodies, as expected.

The Nipissing watershed used in this study is a regional scale watershed that is large in size (over 10,000 km²). The size of each gauge drainage area is small relative to the size of the entire basin, which may have an impact on simulation results. For example, gauges where inconclusive flow duration curve and performance metric results occur (02DD013, 02DD014, and 02DD015) contain the smallest drainage areas out of those used for calibration and validation. Further to this, the percentage of wetlands present by area in each gauge drainage area may also impact model simulation results and model hydrology. A summary of each gauge drainage area and the corresponding modelled wetland percentage calculated by area is provided in Table 6.1.

Table 6.1 Percentage of flow gauge drainage areas containing modelled wetlands and their respective drainage areas

Gauge	Drainage Area (km²)	Wetland % Area by Gauge Drainage Area
02DC012	1,205	10%
02DC013	162	24%
02DD012	927	12%
02DD013	69	15%
02DD014	36	8%
02DD015	103	14%

Explaining the behaviour behind instances of divergence between observed and simulated wetland flow duration curves using gauge drainage areas and wetland coverage by area remains inconclusive and requires further study. Gauge 02DD013, where the largest difference between observed and wetland scenario flow duration curves was experienced, has one of the smallest drainage areas out of the six flow gauges and the second highest percentage of wetlands by area. This could indicate that groundwater connection between wetlands and the stream network are not captured by the gauge only measuring surface flows, thus leading to less observed lateral movement of water. Conversely, gauge 02DC013 contains the largest percentage of wetlands by area in a medium sized drainage area and is a location where the Scenario 3 flow duration curve matches the observed very well. Thus, in the context of the Nipissing model and gauges used for calibration and validation, no obvious relationship between model performance, gauge drainage area size, and percent wetland coverage by area was observed. This relationship is likely influenced by the types of wetlands within each basin which vary across the watershed, and site-specific features (e.g., dams) not accounted for in the model. These results would be expected to change when conducting similar studies on other watersheds.

6.2 Wetland Mapping Products

As discussed in Section 4.2.4, multiple wetland mapping products were considered in this thesis, and a comparison was performed to aid in selecting a product to be used in the Nipissing model. This section discusses the implementation of a wetland mapping product comparison in support of thesis objective #3.

Between the four wetland mapping products considered (Ontario Landcover dataset, MNRF dataset, FRI dataset, and NTEMS dataset), three main discrepancies were observed. The first and most notable difference between mapping products is the number of wetlands included in each product. Wetland coverage in mapping products varies by the resolution at which wetlands are identified from satellite imagery and the criterion used for defining and identifying wetlands. Depending on how wetlands are defined, there may be disagreement between data layers on what is classified as a wetland and what is classified as a “pond” or other water body and thus do not include these polygons as wetlands.

A second difference between wetland mapping products is the spatial variability of wetland extents. Wetlands are naturally dynamic systems, with surface and subsurface extents changing yearly, seasonally, or even diurnally. Their true extents are also often difficult to observe with satellite imagery, especially in areas with forested wetlands where tree cover can conceal the wetland. This can create blurry areas between mapping products and their estimated wetland extents. Some wetland mapping products may include either more wetland data or more accurate extents using ground truthing methods such as visual inspection during field visits. However, it is not feasible for all wetlands across the domain of the wetland mapping products (often either on a provincial or national scale) to be visited and have their extents verified.

A third noticeable difference between wetland mapping products is the classification attributes included with the wetland polygons. Classification systems can either disagree completely about the wetland type (e.g., one product classifies a polygon as a marsh while another classifies the same polygon as a fen), or the classification terminology varies between sources. Some wetland mapping products go as far as to distinguish the ecological wetland classifications in addition to treed versus non-treed types while others may classify wetlands based on ecological classification alone or include no classification scheme at all. Classifications are not only subjective due to limitations inherent to

using satellite imagery to estimate wetland type, but also in the way mapping product developers define different wetland types.

Considering these three main discrepancies between wetland mapping products, the four wetland mapping products considered in this study were compared with respect to the number of wetlands included in each product, spatial variability of wetland extents, and classification of the wetland polygons. To assess wetland abundance in each product, the total area of all wetland polygons contained within the Nipissing watershed was compared for each wetland layer. This was performed instead of relying on wetland polygon count because depending on the data type of each layer (i.e., rasterized versus polygon shapefile) the wetland polygon count may not be representative of the true number of polygons contained within the dataset. For instance, datasets that are initially raster files and then converted to shapefiles produce broken polygons that may be part of a bigger polygon with the same wetland attributes, inflating the number of wetlands represented. To further illustrate this point, the dataset with the greatest number of wetland polygons was the NTEMS dataset, with 1,418,033 polygons, followed by the MNRF dataset with 48,399 polygons, FRI dataset with 42,384 polygons, and lastly the Ontario landcover dataset with 15,772 polygons. The total area of wetland polygons across the Nipissing watershed was then calculated and revealed that the MNRF layer contained the greatest wetland area at 2,171 km², followed by the NTEMS layer with 1,906 km², FRI layer with 1,387 km², and Ontario landcover layer with 558 km². Based on the initial number of polygons counted, the NTEMS seemed to include the greatest number of wetlands but by area the MNRF dataset contains a larger wetland footprint.

Spatial variability was compared by intersecting each dataset in ArcGIS and viewing the areas of overlap and non-overlap. The result of this intersection operation is shown in Figure 6.2. The Ontario landcover dataset was excluded from this analysis, as its wetland area coverage is at least half of the wetland area contained within the other datasets.

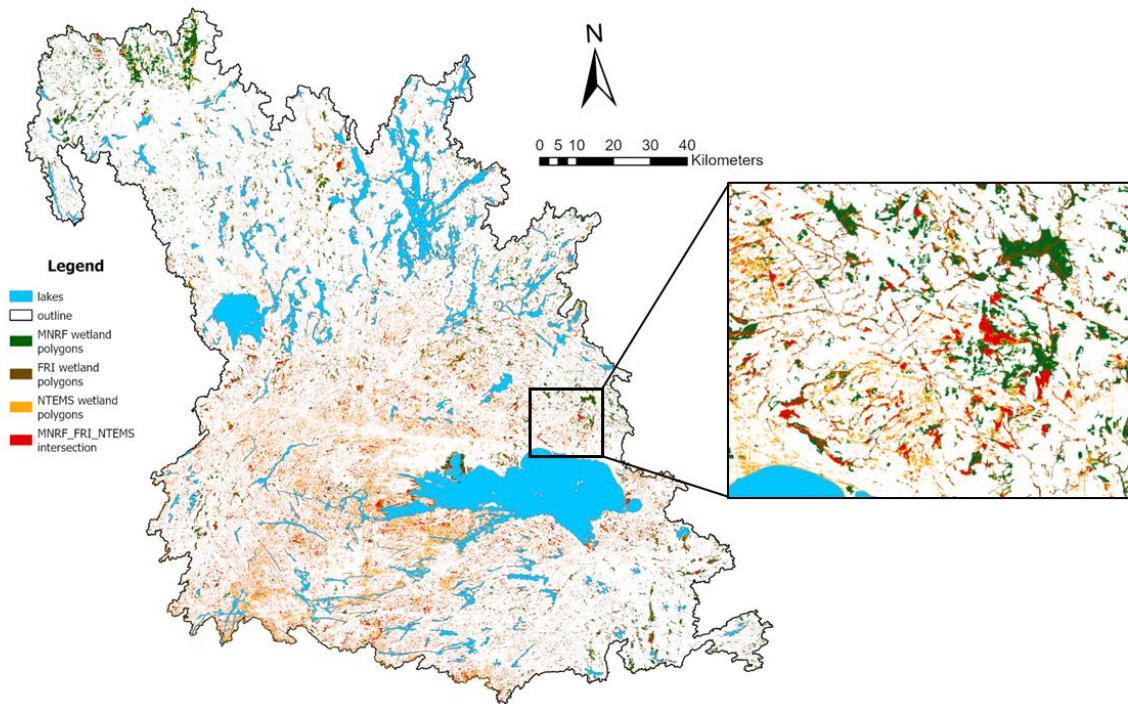


Figure 6.2 Wetland mapping data product overlap for the Nipissing watershed

The FRI dataset contains many polygons overlapping with the MNRF dataset, as the latter was partly informed by the FRI dataset. The MNRF dataset however, unlike the FRI product, is a wetland-specific dataset and is a more comprehensive product for wetland mapping by including more wetland coverage by area. The NTEMS wetland dataset, which is a national scale product, contains different spatial coverage than the other datasets in that wetlands are heavily concentrated around Lake Nipissing and the middle of the basin and contain less wetlands towards the edges of the watershed. This is inconsistent with the MNRF and FRI datasets, as both include more wetland coverage basin wide. In the Nipissing basin, the two most spatially consistent datasets include the FRI and MNRF product as they are both Ontario datasets created by the Provincial government.

Wetland classifications between each product were also compared. The Ontario landcover dataset contains the highest level of classification detail, with 9 distinct wetland types based on ecological type and forest cover. These classes include marshes, swamps, bogs, and fens, and further classification based on forested presence in bogs and fens (treed or open), forest type in swamps (deciduous or coniferous) and the degree of connection to tides in marshes (intertidal, supertidal, or

inland). The MNRF dataset reports all ecological class types (bog, fen, swamp, marsh, and open water) along with an extra “unknown” category. The NTEMS dataset uses a binary wetland classification scheme to identify treed versus non-treed wetlands and does not include ecological types (Hermosilla et al., 2016; Wulder et al., 2018). Lastly, as the FRI dataset is a forestry dataset and not a wetland-specific product, no wetland classifications are provided.

The comparison described here informed the selection of the MNRF dataset for use in the Nipissing model study, as out of the four products considered it is the most comprehensive wetland product at a Provincial scale, contains the greatest wetland footprint by area, contains overlap and areas of agreeance between the other compared products, and contains wetland classifications by ecological types that were not retained in the version of the Nipissing model presented in this thesis, but could easily be retained for further study. In this example, testing the impact of wetland inclusion on hydrologic models like Raven was the purpose for modelling, thus requiring a dataset with as much wetland data as possible.

As demonstrated by this example wetland mapping product comparison, wetland datasets contain a fair level of uncertainty with respect to number of wetlands, wetland boundaries, and classification of individual polygons. The comparison of wetland mapping products outlined here aims to improve user understanding of each considered product and the sources of uncertainty between each dataset. Ultimately, the selection of one dataset over another can vary based on both knowledge about the quality of the dataset after performing a product comparison, and the purpose of the model in which the wetland mapping is to be used. To summarize, a recommended comparison procedure of different wetland mapping products includes the following:

1. Compare wetland coverage between datasets by calculating wetland polygon coverage by area.
2. Compare spatial variability between datasets by intersecting wetland data layers and visually inspecting areas of overlap and non-overlap.
3. Compare data attributes, particularly wetland classifications if provided and identify classification discrepancies.

After performing the described dataset comparisons, modellers should select a wetland dataset that best suits their modelling purpose and spatial scale. A thorough wetland product comparison should

allow the modeller to feel confident in their choice of mapping product because they understand its limitations, uncertainties, and its appropriateness for their modelling needs.

Chapter 7

Conclusions and Recommendations

7.1 Conclusions

In this thesis, three research objectives were proposed to investigate the impact of wetland representation on flow estimation accuracy in hydrologic modelling. The first objective was to inventory existing wetland modelling methods and develop a catalogue of conceptual-numerical wetland modelling methods in hydrology. The second objective was to implement select conceptual-numerical modelling methods in a case study hydrologic model and evaluate model performance to determine the impact of wetland representation. The third objective was to examine how available wetland mapping products can inform hydrological modelling of wetlands.

The first objective was accomplished by conducting a thorough literature review of wetland classification systems and existing wetland modelling methods. The developed catalogue of conceptual-numerical models was intended to help modellers better understand the complex hydrology of different types of wetlands and identify numerical water balance equations based on wetland representations in literature. The conceptual-numerical models identified include riparian wetlands, lake fringe wetlands, geographically isolated wetlands, separated fill-and-spill wetlands, ombrogenous wetlands, and seasonally flooded wetlands. Each wetland conceptual-numerical model can also be forested or non-forested, and permafrost-underlain or not.

The second objective was accomplished by the building of a semi-distributed hydrologic model of the Nipissing watershed in Ontario using the Raven Hydrologic Modelling Framework and testing three model scenarios: the first scenario (Scenario 1) was a base model with no wetland representation, the second scenario (Scenario 2) contained spatially-explicit wetland representation of a single wetland conceptual-numerical model type using a wetland mapping product provided by the Ontario Ministry of Natural Resources and Forestry (MNRF), and a third scenario (Scenario 3) contained three spatially-explicit wetland conceptual-numerical model types (riparian wetlands, lake fringe wetlands, and geographically isolated wetlands) using the same MNRF wetland mapping product to identify wetland locations. Wetlands within the MNRF dataset were classified based on spatial overlap of the wetland polygons with the modelled stream network, lake polygons, or neither. Additionally, Scenario 3 included treatment of geographically isolated wetland contributing areas by diverting a

percentage (proportional to the percent coverage of modelled wetlands across the flow gauge drainage areas), of water in non-wetland hydrologic response units (HRUs) to isolated wetland HRUs.

Calibration and validation were performed for each modelling scenario and results were compared.

Results from the multi-objective calibration of each modelling scenario to both high and low flow metrics (median KGE and median logNSE of flow gauges) indicated that the Scenario 2 model structure (with spatial representation of one wetland type) outperformed the other two scenarios at most calibration flow gauges due to both pareto solution uncertainty and trade-off between wetland complexity and model performance being impacted by site-specific properties. All median KGE and logNSE objective function values achieved during the calibration period exceeded an adequate performance threshold of 0.5 in all modelling scenarios.

Model validation indicated that the Scenario 3 model structure outperformed the other two scenarios at the majority of gauges and across multiple performance metrics measuring both high and low flows. Low flows were especially handled well by Scenario 3 compared to Scenario 1 and 2 when comparing performance metrics and hydrographs, as the most water was stored in Scenario 3 wetland depression storage by abstraction, lateral diversion processes, and baseflow and interflow parameters set up for each simulated wetland type. The validation period median percent bias experienced a significant decrease of 15% from Scenario 2 to Scenario 3, and an additional decrease in the spread of results over all calibrated flow gauges. The significant improvement in percent bias achieved in Scenario 3 highlights the importance of low flow accuracy to hydrologic model performance.

Flow duration curves and hydrographs plotted by flow gauge demonstrated that site-specific properties of the entire study area and individual gauge drainage areas can impact simulation results. For instance, groundwater flow modelled by Raven was not measured and accounted for at observation gauges, resulting in a discrepancy between some observed and simulated results. There was no relationship found between gauge drainage area, wetland coverage percent by area, and model performance at individual gauges but this would likely change if a similar study was conducted on a different watershed.

The third thesis objective was achieved by comparing four different wetland mapping products and assessing their suitability for use in the Nipissing study. Wetland mapping products analyzed include the Ontario GeoHub Wetlands dataset (Ontario Ministry of Natural Resources and Forestry, 2010a), Ontario Provincial Land Cover Dataset (Ontario Ministry of Natural Resources and Forestry, 2000b),

Forest Resource Inventory dataset Version 2 (Ontario Ministry of Natural Resources and Forestry, 2020), and the National Terrestrial Ecosystem Monitoring System for Canada (Hermosilla et al., 2016; Wulder et al., 2018, 2025). It was recommended that the assessment of wetland mapping product suitability for a given study area be performed by initially comparing wetland coverage between datasets using the wetland polygon coverage by area, then comparing spatial variability between datasets by inspecting areas of overlap and non-overlap, and finally comparing data attributes, particularly wetland classifications and any discrepancies between dataset attributes.

7.2 Recommendations

The Nipissing watershed was chosen as a Raven model study site based on its simplicity in terms of landcover classes (containing mostly forest and lake coverage and negligible urban and agricultural areas), in addition to high wetland area coverage. This allowed for simple land-water HRUs to be created for the base model scenario without wetlands and increase in complexity through the addition of wetlands with each new scenario. However, a few limitations of the chosen study area became apparent during the project. First, the Nipissing watershed is highly regulated, with many hydroelectric dams existing at Lake Nipissing and in the French River delta. The water regulation downstream of Lake Nipissing resulted in flow and level gauges at the watershed outlet being ineligible for use in calibration and as a model performance benchmark. In addition, the flow data at some gauges in the watershed contained warnings about data quality and had to be removed from consideration, and the remaining flow data contained flow records with data period gaps, requiring the model simulation period to be shortened. Another limitation of using the Nipissing watershed in this study was the lack of available climate data. As noted earlier in this thesis, climate station records in and around the Nipissing watershed are sparse and inconsistent. This lack of station data required the use of gridded climate data, introducing additional model uncertainty.

An ideal site for a wetland study similar to that presented in this thesis would include the following characteristics:

- Forest, lake, and wetland landcover classes with no or negligible urban, agricultural, etc. land classes (similar to the Nipissing watershed).
- No water regulation influences. All flow gauges would measure natural flows, and no dams would exist in the watershed. This would allow the use of flow gauges across the watershed and especially at the outlet to be used in model calibration and validation.

- Good quality flow data would exist throughout the watershed and for consistent time periods, allowing for a model simulation period of 20 to 30 years.
- Good quality climate station data would exist throughout the watershed and for consistent time periods, allowing the use of gauge climate data rather than gridded for a 20-to-30-year modelling period.
- Availability of water level data not influenced by regulation.
- Flow gauge drainage areas would contain a diversity of wetland coverage percentages.
- In a perfect world where the modeller had access to the flow measuring equipment or measurement of groundwater data, adding groundwater probes or monitoring wells near the flow gauges would provide insight into any substantial groundwater gains or losses not captured by surface water gauges.

The addition of flow gauges at the watershed outlet for model calibration and validation would provide improved model evaluation and more insight into how each wetland scenario compares to the base model scenario. The use of meteorological gauge data rather than gridded inputs would reduce data input uncertainty, although there is still inherent uncertainty associated with gauge data. Lastly, including flow gauges with drainage areas containing a diversity of wetland coverage would allow comparison of modelling scenarios in basins with little to no wetland coverage and basins with dense wetland coverage.

If the opportunity for a similar study was available, it is recommended that the ideal study conditions listed above be considered and a site other than the Nipissing watershed be chosen. In addition, the Nipissing study was an example of a regional hydrologic model, meaning that it is both semi-distributed and large (over 10,000 km² in area). Thus, the results of this experiment are only valid for the Nipissing regional model. To further test the impacts of the same type of wetland modelling performed in the Nipissing study, it is recommended that a smaller (non-regional) watershed be chosen for a similar modelling study.

The wetland modelling presented in this study focused on wetland routing classifications (i.e., wetland connection in the watershed) as a basis for determining if the presence of explicitly represented wetlands impacts model simulation accuracy. Future work could further test increased wetland conceptual-numerical model complexity by further classifying wetlands based on their ecological types. For instance, the MNRF wetland dataset includes ecological classifications, which

were not retained in this study. However, a future study could retain the ecological classifications along with the wetland connectivity classifications and incorporate parameters/processes depending on the wetland conceptual-numerical model type. For example, a wetland classified as a palustrine swamp could be categorized as a forested geographically isolated wetland with unique vegetation types and additional forested wetland properties. Ultimately, the testing of wetland conceptual-numerical model complexity would shed light on how much complexity in a given model is worthwhile in terms of wetland representation.

It is also recommended to test the effect of wetland conceptual-numerical models on different sized watersheds and watersheds with varying wetland composition by area. The Nipissing watershed model presented in this thesis is a large watershed (19,000 km²) with 11% wetland coverage and is the only regional model case study to the author's knowledge that tests the implementation of multiple wetland types. The Nipissing model study should be repeated on smaller watersheds with more and less wetland presence to determine how wetland routing implementation impacts model performance at different spatial scales. These future models should then implement wetland schemes that increase in complexity and test the different wetland conceptual-numerical models presented in this thesis.

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Appendix A
Meteorological Data Quality Comparison

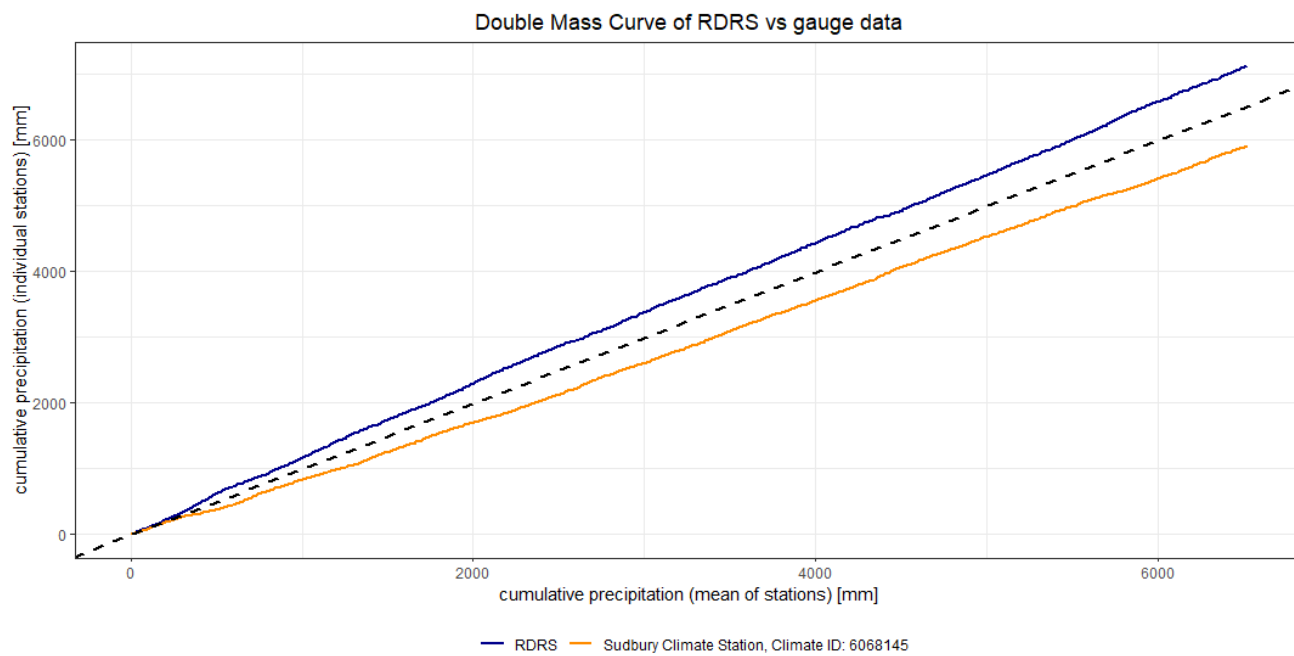


Figure A.1 Double mass curve plotted for RDRS v2.1 gridded product and Sudbury Climate Station (Climate ID 6068145)

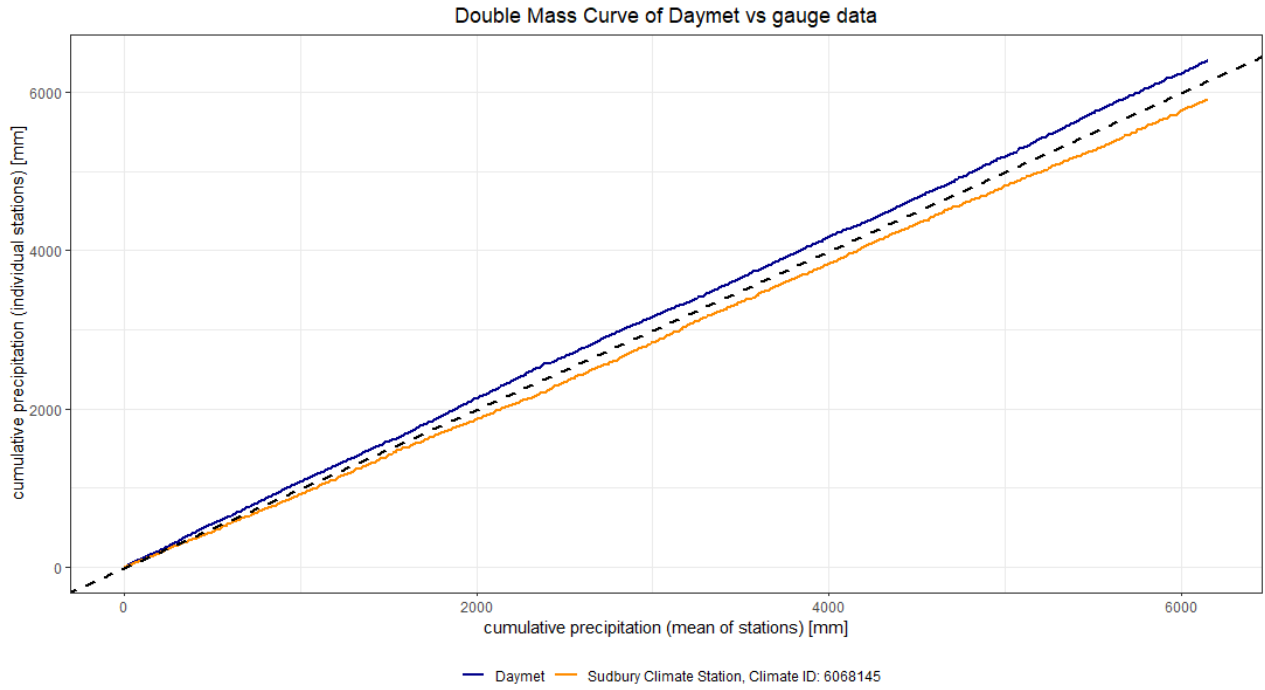


Figure A.2 Double mass curve plotted for Daymet gridded product and Sudbury Climate Station (Climate ID 6068145)

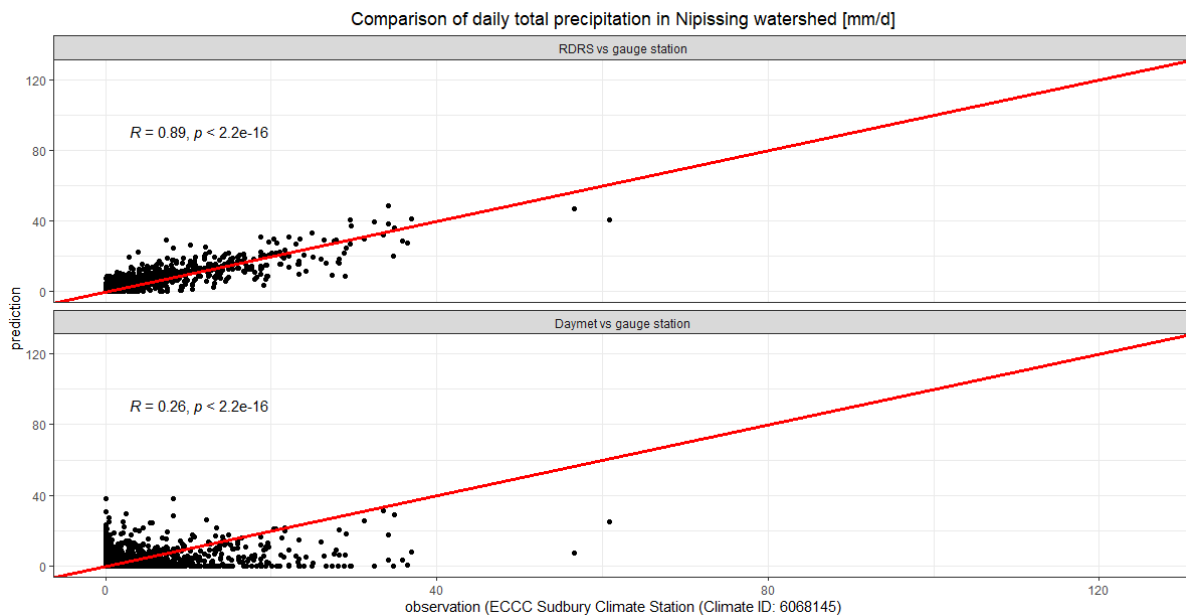


Figure A.3 Comparison of daily total precipitation between gridded products and Sudbury Climate Station (Climate ID 6068145)

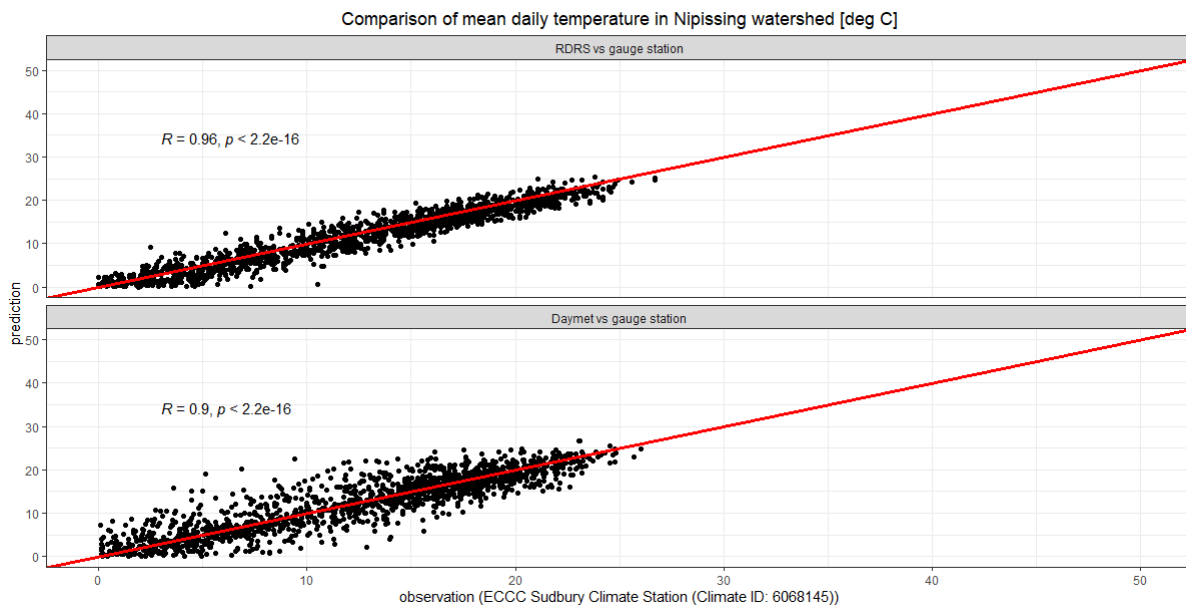


Figure A.4 Comparison of mean daily temperature between gridded products and Sudbury Climate Station (Climate ID 6068145)

Appendix B

Wetland Dataset Comparison

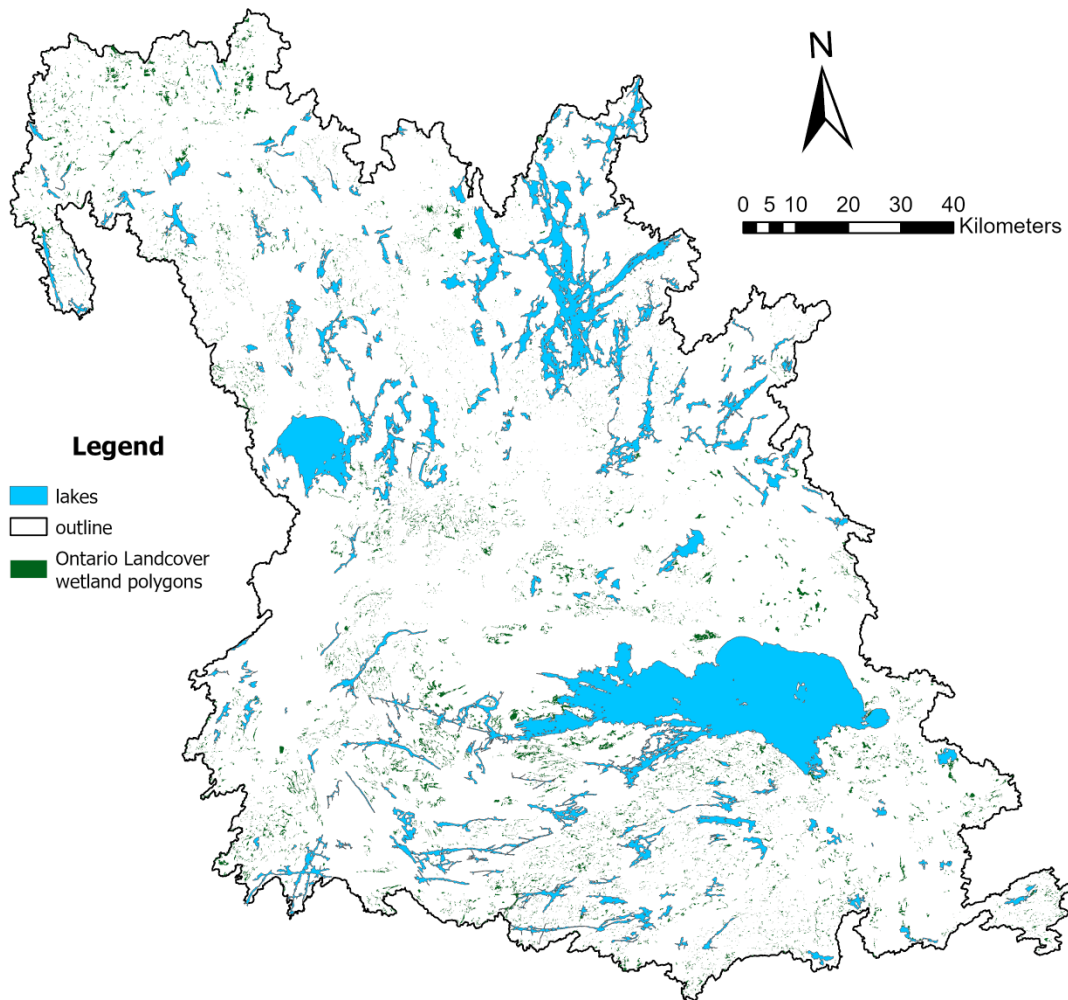


Figure B.1 Ontario Landcover Dataset (2000) wetland polygons plotted in the Nipissing watershed (Ontario Ministry of Natural Resources and Forestry, 2000b)

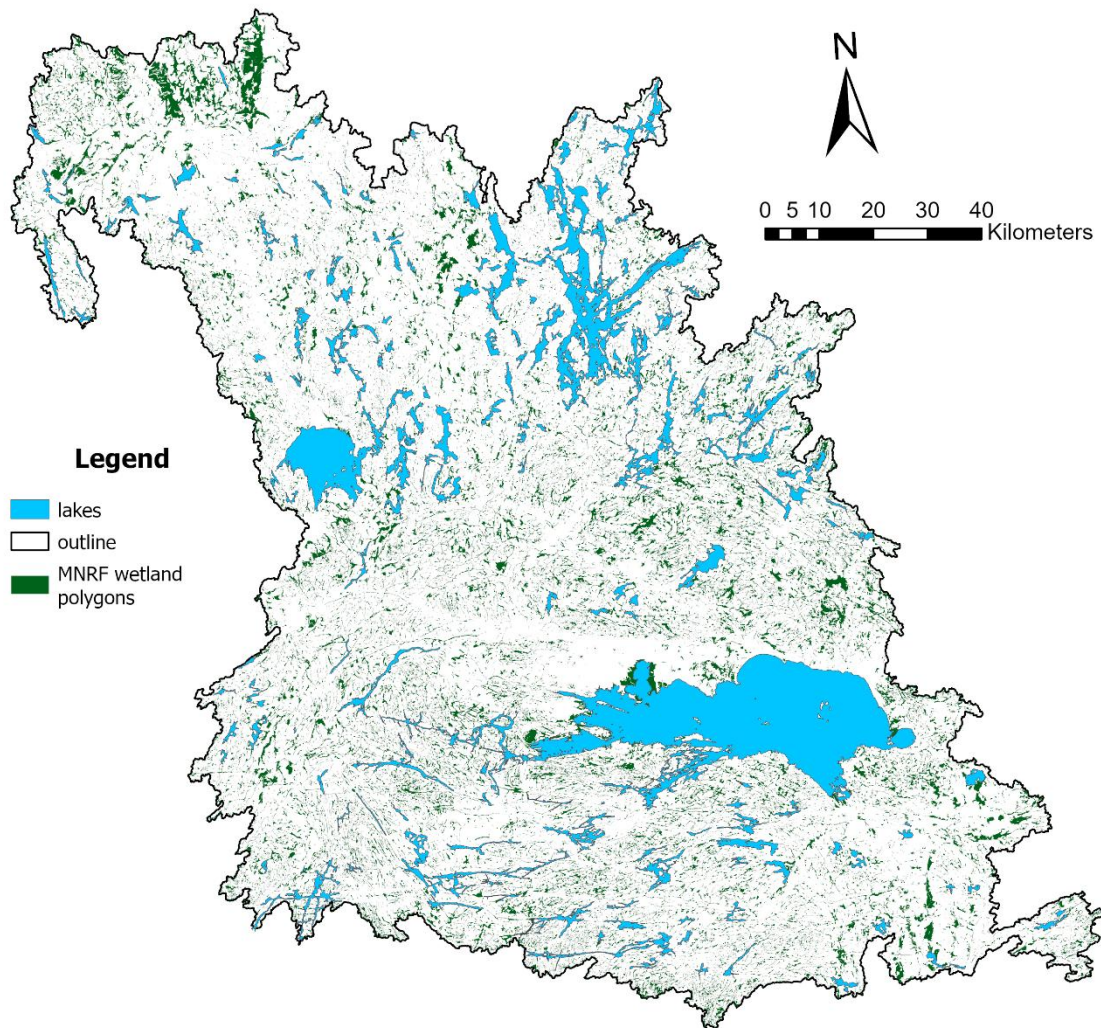


Figure B.2 Ontario Ministry of Natural Resources and Forestry (MNR) wetland polygons plotted in the Nipissing watershed (Ontario Ministry of Natural Resources and Forestry, 2010a)

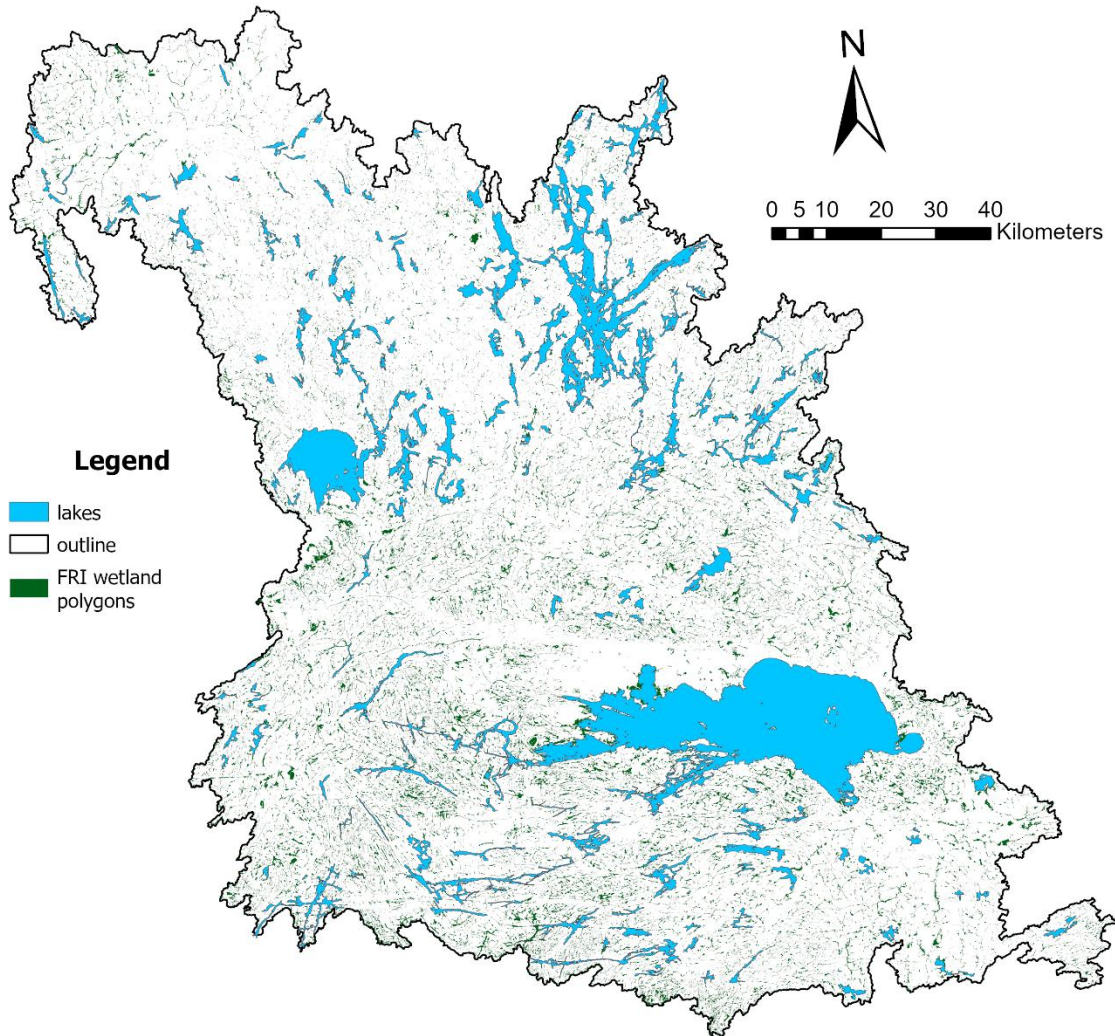


Figure B.3 Ontario Forest Resources Inventory (FRI) Version 2 wetland polygons plotted in the Nipissing watershed (Ontario Ministry of Natural Resources and Forestry, 2020)

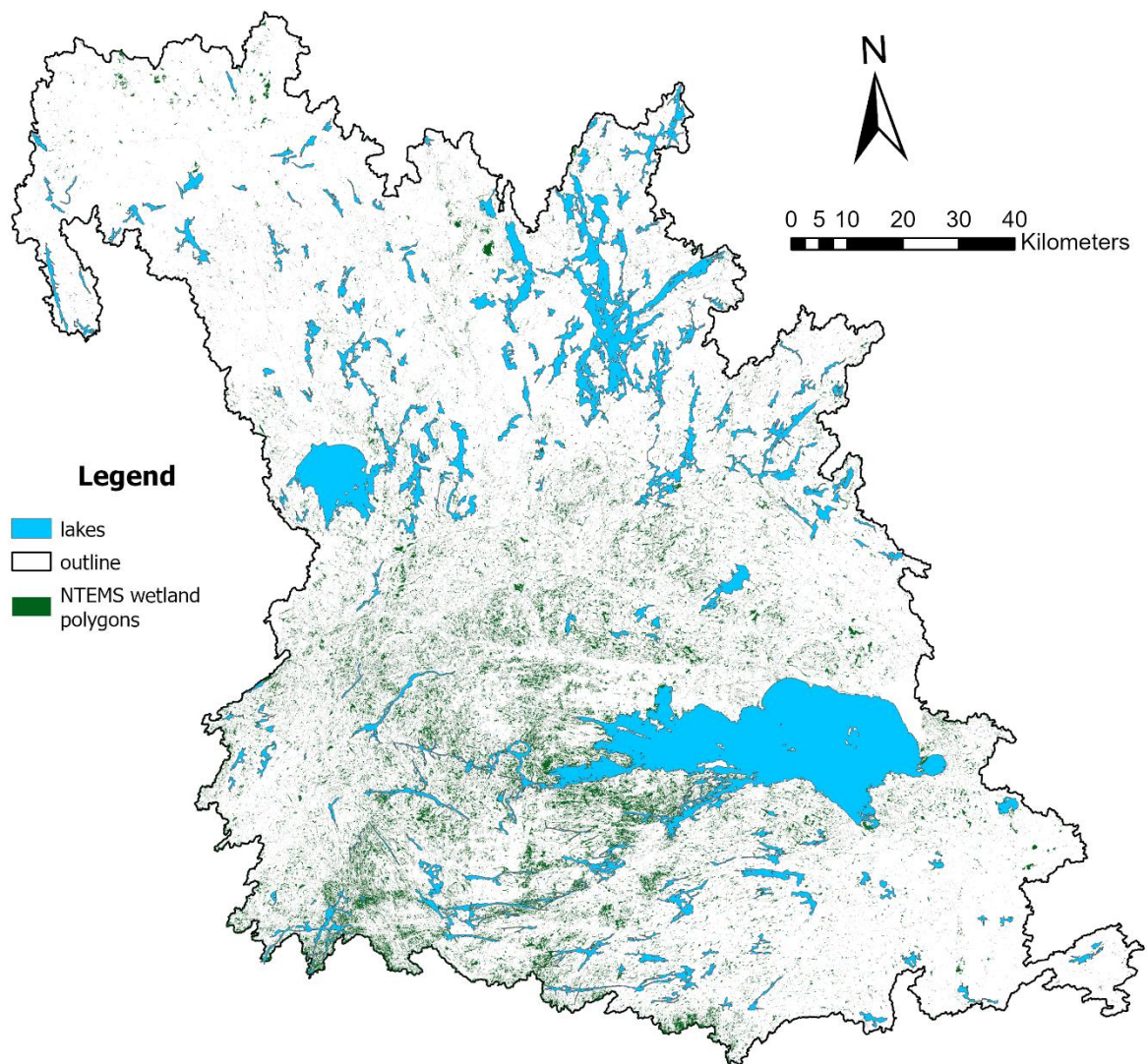


Figure B.4 National Terrestrial Ecosystem Monitoring System for Canada (NTEMS) wetland polygons plotted in the Nipissing watershed (Hermosilla et al., 2016; Wulder et al., 2018, 2025)

Appendix C

Raven Input Files

Raven RVI file for Nipissing model Scenario 1

```
# -----
:StartDate      2005-01-01 00:00:00
:EndDate        2018-12-30 00:00:00
:TimeStep       1.0

:Method          ORDERED_SERIES
:InterpolationMethod NEAREST_NEIGHBOR
:SoilModel        SOIL_MULTILAYER 3
:Routing          ROUTE_DIFFUSIVE_WAVE
:CatchmentRoute   ROUTE_TRI_CONVOLUTION
:Evaporation      PET_HARGREAVES_1985
:OW_Evaporation   PET_HARGREAVES_1985
:SWCanopyCorrect  SW_CANOPY_CORR_STATIC
:RainSnowFraction RAINSNOW_DINGMAN
:PotentialMeltMethod POTMELT_HBV
:PrecipIceptFract PRECIP_ICEPT_LAI
:LakeFreeze       LFREEZE_BASIC
:MonthlyInterpolationMethod MONTHINT_LINEAR_MID

# Define HRU Groups
:DefineHRUGroups Forest Lake_HRUs NonLake Water_simple

# --Hydrologic Processes-----
:HydrologicProcesses
    :SnowRefreeze      FREEZE_DEGREE_DAY  SNOW_LIQ      SNOW
    :Precipitation      PRECIP_RAVEN       ATMOS_PRECIP  MULTIPLE
    :CanopyEvaporation  CANEVP_MAXIMUM     CANOPY        ATMOSPHERE
    :CanopySnowEvap     CANEVP_MAXIMUM     CANOPY_SNOW   ATMOSPHERE
    :CanopyDrip          CANDRIP_RUTTER     CANOPY        PONDED_WATER
    :SnowBalance         SNOBAL_HBV         MULTIPLE      MULTIPLE
    :OpenWaterEvaporation OPEN_WATER_EVAP  DEPRESSION    ATMOSPHERE
    :Infiltration        INF_HBV            PONDED_WATER  MULTIPLE
    :Baseflow            BASE_POWER_LAW     SOIL[1]       SURFACE_WATER
```

```

:Baseflow          BASE_POWER_LAW      SOIL[2]          SURFACE_WATER
:Interflow         INTERFLOW_PRMS      SOIL[0]          SURFACE_WATER
:Percolation       PERC_GAWSER         SOIL[0]          SOIL[1]
:Percolation       PERC_GAWSER         SOIL[1]          SOIL[2]
:SoilEvaporation   SOILEVAP_HBV        SOIL[0]          ATMOSPHERE

# Flush extra water in Lake Soil0 layer from LakeFreeze process to surface
water
:Flush             RAVEN_DEFAULT        SOIL[0]          SURFACE_WATER
:-->Conditional    HRU_TYPE IS LAKE
# Flush extra water in Lake ponded water to surface water
:Flush             RAVEN_DEFAULT        PONDED_WATER    SURFACE_WATER
:-->Conditional    HRU_TYPE IS LAKE
:EndHydrologicProcesses

```

Scenario 1 RVC file

```

:UniformInitialConditions SOIL[0] 400

```

Raven RVI file for Nipissing model Scenario 2

```
# -----
:StartDate      2005-01-01 00:00:00
:EndDate        2018-12-30 00:00:00
:TimeStep       1.0

:Method          ORDERED_SERIES
:InterpolationMethod NEAREST_NEIGHBOR
:SoilModel        SOIL_MULTILAYER  3
:Routing          ROUTE_DIFFUSIVE_WAVE
:CatchmentRoute   ROUTE_TRI_CONVOLUTION
:Evaporation      PET_HARGREAVES_1985
:OW_Evaporation   PET_HARGREAVES_1985
:SWCanopyCorrect  SW_CANOPY_CORR_STATIC
:RainSnowFraction RAINSNOW_DINGMAN
:PotentialMeltMethod POTMELT_HBV
:PrecipIceptFract PRECIP_ICEPT_LAI
:LakeFreeze       LFREEZE_BASIC
:MonthlyInterpolationMethod MONTHINT_LINEAR_MID

# Define HRU Groups
:DefineHRUGroups Wetland_Connected Forest Lake_HRUs NonLake Water_simple

# --Hydrologic Processes-----
:HydrologicProcesses
    :SnowRefreeze      FREEZE_DEGREE_DAY  SNOW_LIQ      SNOW
    :Precipitation      PRECIP_RAVEN        ATMOS_PRECIP    MULTIPLE
    :CanopyEvaporation  CANEVP_MAXIMUM      CANOPY          ATMOSPHERE
    :CanopySnowEvap     CANEVP_MAXIMUM      CANOPY_SNOW     ATMOSPHERE
    :CanopyDrip          CANDRIP_RUTTER      CANOPY          PONDED_WATER
    :SnowBalance        SNOBAL_HBV          MULTIPLE        MULTIPLE
    :OpenWaterEvaporation OPEN_WATER_EVAP    DEPRESSION      ATMOSPHERE
    :Infiltration        INF_HBV            PONDED_WATER    MULTIPLE
    :Baseflow           BASE_POWER_LAW      SOIL[1]         SURFACE_WATER
```

```

:Baseflow          BASE_POWER_LAW      SOIL[2]          SURFACE_WATER
:Interflow         INTERFLOW_PRMS      SOIL[0]          SURFACE_WATER
:Percolation       PERC_GAWSER         SOIL[0]          SOIL[1]
:Percolation       PERC_GAWSER         SOIL[1]          SOIL[2]
:SoilEvaporation   SOILEVAP_HBV        SOIL[0]          ATMOSPHERE

# Flush extra water in Lake Soil0 layer from LakeFreeze process to surface
water
:Flush             RAVEN_DEFAULT        SOIL[0]          SURFACE_WATER
:-->Conditional    HRU_TYPE IS LAKE
# Flush extra water in Lake ponded water to surface water
:Flush             RAVEN_DEFAULT        PONDED_WATER    SURFACE_WATER
:-->Conditional    HRU_TYPE IS LAKE

# Wetland processes
# melted snow needs to be flushed to depression
:Flush             RAVEN_DEFAULT        PONDED_WATER    DEPRESSION
:-->Conditional    HRU_GROUP IS Wetland_Connected
:DepressionOverflow DFLOW_THRESHPOW     DEPRESSION        SURFACE_WATER
:Seepage           SEEP_LINEAR          DEPRESSION        SOIL[0]
:EndHydrologicProcesses

```

Scenario 2 RVC file

```

:UniformInitialConditions SOIL[0] 400

```

Raven RVI file for Nipissing model Scenario 3

```
# -----
:StartDate      2005-01-01 00:00:00
:EndDate        2018-12-30 00:00:00
:TimeStep       1.0

:Method          ORDERED_SERIES
:InterpolationMethod NEAREST_NEIGHBOR
:SoilModel        SOIL_MULTILAYER 3
:Routing          ROUTE_DIFFUSIVE_WAVE
:CatchmentRoute   ROUTE_TRI_CONVOLUTION
:Evaporation      PET_HARGREAVES_1985
:OW_Evaporation   PET_HARGREAVES_1985
:SWCanopyCorrect  SW_CANOPY_CORR_STATIC
:RainSnowFraction RAINSNOW_DINGMAN
:PotentialMeltMethod POTMELT_HBV
:PrecipIceptFract PRECIP_ICEPT_LAI
:LakeFreeze       LFREEZE_BASIC
:MonthlyInterpolationMethod MONTHINT_LINEAR_MID

# Define HRU Groups
:DefineHRUGroups Wetland_Connected Wetland_Isolated Wetland_lake_adj
Wetland_HRUs Forest Lake_HRUs NonLake Water_simple

# --Hydrologic Processes-----
:HydrologicProcesses
    :SnowRefreeze      FREEZE_DEGREE_DAY  SNOW_LIQ      SNOW
    :Precipitation      PRECIP_RAVEN       ATMOS_PRECIP   MULTIPLE
    :CanopyEvaporation  CANEVP_MAXIMUM     CANOPY         ATMOSPHERE
    :CanopySnowEvap     CANEVP_MAXIMUM     CANOPY_SNOW    ATMOSPHERE
    :CanopyDrip         CANDRIP_RUTTER     CANOPY         PONDED_WATER
    :SnowBalance        SNOBAL_HBV         MULTIPLE       MULTIPLE
```

```

# Lateral divert command to account for isolated wetland contributing
areas

:Flush          RAVEN_DEFAULT PONDED_WATER TRUNK 4.047000E-01
  :-->Conditional  HRU_GROUP IS Forest

:LateralFlush    RAVEN_DEFAULT Forest TRUNK To Wetland_Isolated
DEPRESSION

:OpenWaterEvaporation OPEN_WATER_EVAP    DEPRESSION    ATMOSPHERE
:Infiltration    INF_HBV                  PONDED_WATER    MULTIPLE
:Baseflow        BASE_POWER_LAW           SOIL[1]          SURFACE_WATER
:Baseflow        BASE_POWER_LAW           SOIL[2]          SURFACE_WATER
:Interflow       INTERFLOW_PRMS           SOIL[0]          SURFACE_WATER
:Percolation     PERC_GAWSER              SOIL[0]          SOIL[1]
:Percolation     PERC_GAWSER              SOIL[1]          SOIL[2]
:SoilEvaporation SOILEVAP_HBV            SOIL[0]          ATMOSPHERE

# Flush extra water in Lake Soil0 layer from LakeFreeze process to surface
water

:Flush          RAVEN_DEFAULT    SOIL[0]          SURFACE_WATER
  :-->Conditional  HRU_TYPE IS LAKE

# Flush extra water in Lake ponded water to surface water

:Flush          RAVEN_DEFAULT    PONDED_WATER    SURFACE_WATER
  :-->Conditional  HRU_TYPE IS LAKE

# Wetland processes
# melted snow needs to be flushed to depression

:Flush          RAVEN_DEFAULT    PONDED_WATER    DEPRESSION
  :-->Conditional  HRU_GROUP IS Wetland_HRUs

:DepressionOverflow DFLOW_THRESHPOW    DEPRESSION    SURFACE_WATER
:Seepage          SEEP_LINEAR          DEPRESSION    SOIL[0]

:EndHydrologicProcesses

Scenario 3 RVC file

:UniformInitialConditions SOIL[0] 400

```

Appendix D

Nipissing Model Parameter Ranges for Calibration

Figure D.1 Model Scenario 1 Calibration Parameters and Ranges

Raven Parameter	Description	Nipissing Model Parameter Name	Min Value	Max Value
Soil thickness	Soil layer thickness [m]	TOP_BED, MED_NON_BED	0.01	8
BASEFLOW_COEFF	Linear soil coefficient for baseflow storage/routing [1/d]	MED_BED, MED_NON_BED	0.0001	0.5
BASEFLOW_N	VIC/ARNO baseflow exponent [-]	MED_BED, MED_NON_BED	0.5	10
MAX_INTERFLOW_RATE	PRMS maximum interflow rate [mm/d]	TOP_BED, TOP_NON_BED	0.01	10

FIELD_CAPACITY	Field capacity saturation of soil [-]	TOP_BED, TOP_NON_BED	0.15	0.5
MAX_PERC_RATE	VIC/ARNO/GAWSER percolation rate [mm/d]	TOP_BED, TOP_NON_BED	0.01	10
PET_CORRECTION	Correction factor for potential evapotranspiration [-]	TOP_BED, TOP_NON_BED	0	1
MELT_FACTOR	Maximum snow melt factor used in degree day models [mm/d/°C]	LAKE, WATER_SIMPLE, FOREST	1	10
DD_MELT_TEMP	Melt temperature [°C]	LAKE, WATER_SIMPLE, FOREST	-5	3
OW_PET_CORR	Fraction of PET applied to open water evaporation [-]	LAKE, WATER_SIMPLE, FOREST	0.2	1.2

RESERVOIR_CREST_WIDTH	Multiplier for lake weir outlets [-]	LAKE	0.01	1
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Figure D.2 Model Scenario 2 Calibration Parameters and Ranges

Raven Parameter	Description	Nipissing Model Parameter Name	Min Value	Max Value
Soil thickness	Soil layer thickness [m]	TOP_BED, TOP_NON_BED, TOP_WETLAND_CON	0.01	8
BASEFLOW_COEFF	Linear soil coefficient for baseflow storage/routing [1/d]	MED_BED, DEEP_BED, MED_NON_BED, DEEP_NON_BED, MED_WETLAND_CON, DEEP_WETLAND_CON	0.0001	0.5

BASEFLOW_N	VIC/ARNO baseflow exponent [-]	MED_BED, MED_NON_BED, MED_WETLAND_CON,	0.5	10
MAX_INTERFLOW_RATE	PRMS maximum interflow rate [mm/d]	TOP_BED, TOP_NON_BED, TOP_WETLAND_CON	0.01	10
FIELD_CAPACITY	Field capacity saturation of soil [-]	TOP_BED, TOP_NON_BED, TOP_WETLAND_CON	0.15	0.5
MAX_PERC_RATE	VIC/ARNO/GAWSER percolation rate [mm/d]	TOP_BED, TOP_NON_BED, TOP_WETLAND_CON	0.01	10
PET_CORRECTION	Correction factor for potential evapotranspiration [-]	TOP_BED, TOP_NON_BED, TOP_WETLAND_CON	0	1
MELT_FACTOR	Maximum snow melt factor used in degree day models [mm/d/°C]	LAKE, FOREST, WATER_SIMPLE, WETLAND_CON	1	10

DD_MELT_TEMP	Melt temperature [°C]	LAKE, FOREST, WATER_SIMPLE, WETLAND_CON	-5	3
DEP_MAX	Maximum total average depression storage in each HRU [mm]	WETLAND_CON	0	200
OW_PET_CORR	Fraction of PET applied to open water evaporation [-]	WETLAND_CON	0.2	1.2
DEP_MAX_FLOW	Outflow rate with full depression storage [mm/d]	WETLAND_CON	0	10
DEP_THRESHOLD	Threshold storage at which flow commences [mm]	WETLAND_CON	0	200
DEP_N	Power law coefficient for depression outflow [1/d]	WETLAND_CON	0.5	3
DEP_SEEP_K	Depression linear seepage constant [1/d]	WETLAND_CON	0.001	0.5

RESERVOIR_CREST_WIDTH	Multiplier for lake weir outlets [-]	LAKE	0.01	1
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Figure D.3 Model Scenario 3 Calibration Parameters and Ranges

Raven Parameter	Description	Nipissing Model Parameter Name	Min Value	Max Value
Soil thickness	Soil layer thickness [m]	TOP_BED, TOP_NON_BED, MED_BED, MED_NON_BED, TOP_WETLAND_CON, TOP_WETLAND_ISO	0.01	8
BASEFLOW_COEFF	Linear soil coefficient for baseflow storage/routing [1/d]	MED_BED, MED_NON_BED, MED_WETLAND_CON, MED_WETLAND_ISO	0.0001	0.5

BASEFLOW_N	VIC/ARNO baseflow exponent [-]	MED_BED, MED_NON_BED, MED_WETLAND_CON, MED_WETLAND_ISO	0.5	10
MAX_INTERFLOW_RATE	PRMS maximum interflow rate [mm/d]	TOP_BED, TOP_NON_BED	0.01	10
FIELD_CAPACITY	Field capacity saturation of soil [-]	TOP_BED, TOP_NON_BED, TOP_WETLAND_CON, TOP_WETLAND_ISO	0.15	0.5
MAX_PERC_RATE	VIC/ARNO/GAWSER percolation rate [mm/d]	TOP_BED, TOP_NON_BED, TOP_WETLAND_CON, TOP_WETLAND_ISO	0.01	10
PET_CORRECTION	Correction factor for potential evapotranspiration [-]	TOP_BED, TOP_NON_BED, TOP_WETLAND_ISO	0	1
MELT_FACTOR	Maximum snow melt factor used in degree day models [mm/d/°C]	LAKE, FOREST, WATER_SIMPLE, WETLAND_CON,	1	10

		WETLAND_ISO, WETLAND_LAKE_ADJ		
DD_MELT_TEMP	Melt temperature [°C]	LAKE, FOREST, WATER_SIMPLE, WETLAND_CON, WETLAND_ISO, WETLAND_LAKE_ADJ	-5	3
DEP_MAX	Maximum total average depression storage in each HRU [mm]	WETLAND_CON, WETLAND_ISO, WETLAND_LAKE_ADJ	0	200
OW_PET_CORR	Fraction of PET applied to open water evaporation [-]	WETLAND_CON, WETLAND_ISO, WETLAND_LAKE_ADJ	0.2	1.2
DEP_THRESHOLD	Threshold storage at which flow commences [mm]	WETLAND_CON, WETLAND_ISO, WETLAND_LAKE_ADJ	0	200
DEP_SEEP_K	Depression linear seepage constant [1/d]	WETLAND_CON, WETLAND_ISO, WETLAND_LAKE_ADJ	0.001	0.5

RESERVOIR_CREST_WIDTH	Multiplier for lake weir outlets [-]	LAKE	0.01	1
Lateral Flush Fraction	Fraction of water moved from forest to isolated wetland HRUs to account for contributing areas [-]	WETLAND_ISO	0.28	0.48