AI-2338 Final Technical Report

Nutrient Objectives for Small Streams in Agricultural Watersheds of Alberta

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November 15, 2019

Table of Contents

Table of Figures

List of Acronyms

AEHI	–	Aquatic ecosystem health impairment
AICc	–	Akaike's information criterion corrected for small sample sizes
APEX	–	Agricultural Policy Extender
BMP	–	Beneficial management practices
CART	–	Classification and regression trees
CEEOT	–	Comprehensive Economic and Environmental Optimization Tool
DEM	–	Digital elevation model
DO	–	Dissolved oxygen
ER	–	Ecosystem respiration
FWMC	–	Flow weighted mean concentration
GPP	–	Gross primary productivity
HPLC	–	High performance liquid chromatography
HRU	–	Hydrologic response unit
HUC	–	Hydrologic Unit Code
IWMP	–	Integrated watershed management plan
NAESI	–	National Agri-Environmental Standards Initiative
NEP	–	Net ecosystem productivity
NSE	–	Nash-Sutcliffe efficiency
PBIAS	–	Percent bias
PLR	–	Piecewise linear regression
QPLR	–	Quantile piecewise linear regression
RMSE	–	Root mean square error
SSO	–	Site-specific objective
SWAP	–	SWAT-APEX Interface Program
SWAT	–	Soil and Water Assessment Tool
TN	–	Total Nitrogen
TP	–	Total Phosphorus
WEDF	–	Weighted empirical distribution function
WPAC	–	Watershed planning and advisory councils

Executive Summary

Historic alteration of land cover in Canada's agricultural areas has increased the transport of nutrients, such as nitrogen and phosphorus, into surface water systems. Nutrient enrichment in surface water can lead to the undesirable growth of aquatic plants and algae. Excess plant growth, in turn, can compromise waterbodies for human uses through reductions in aesthetic quality, water conveyance capacity, and water quality for livestock watering or domestic consumption. Aquatic ecosystems are more readily affected by nutrient enrichment as excessive growth and decay of plants can alter foodwebs and depress dissolved oxygen levels. Changes to aquatic ecosystem structure can also modify the functional traits of aquatic ecosystems and impair ecosystem services provided by surface water systems, such as nutrient attenuation and fish habitat. Taken together, nutrient enrichment shifts aquatic foodwebs, changes the habitat quality of surface water systems, and degrades the ecosystem services and ultimate human uses of waterbodies. Thus, mitigating nutrient enrichment in surface water systems is important for preserving human and ecological uses of water systems.

Numeric guidelines for nutrients in surface waters serve a valuable role as benchmarks for water quality monitoring and watershed management programs. Numeric guidelines are beneficial as they allow for an interpretation of risk to water users based on water quality results, and provide management triggers and targets for water resource managers. Monitoring the progression of water quality toward management targets, in turn, allows for effective communication of watershed management programs to decision makers and the public. Nutrient objectives are also useful for agricultural sustainability initiatives by providing tangible values that can be used to demonstrate the effectiveness of land management for the preservation of water resources. However, despite the benefits of numeric guidelines for managing surface water, guidelines for nutrients do not currently exist for the majority of lotic systems in Alberta.

Efforts are being made to derive site-specific nutrient objectives at long-term water quality monitoring sites in rivers and lakes as part of statutory regional plans. These values, however, cannot adequately be applied to small streams given their inherently different hydrodynamics and ecology. The lack of nutrient guidelines, in turn, challenges assessments of water quality at the scale of watersheds typically designated for watershed management programs, which further challenges the ability of watershed managers to evaluate and communicate the successful progression of water quality improvements in managed systems.

A stressor-response study design was applied in this study to define numeric nutrient objectives for small streams draining agricultural watersheds. In this study, a variety of structural (e.g., algae concentrations) and functional (e.g., nutrient uptake) responses were measured together with in-stream nutrient concentrations. Relationships between aquatic ecosystem responses to in-stream nutrient concentrations were analyzed to determine threshold concentrations of nutrients that lead to discernable changes in the response variables. In total, 25 response variables were analyzed

in this study for their relationship with concentrations of total nitrogen (TN) and total phosphorus (TP) in a total of 56 streams across the Grassland and Parkland natural regions of Alberta. These natural regions encompass a large proportion of the privately-held agricultural zones of Alberta. The approach used here led to the derivation of generalized nutrient objectives that can be applied to all 3rd to 5th Strahler order streams in these regions.

Single-value numeric objectives for nutrients can lead to erroneous conclusions, given that aquatic ecosystem variables respond differentially to nutrient enrichment, and multiple environmental factors interact to influence aquatic ecosystem response to nutrients. In this study, uncertainty in the response of aquatic ecosystems to nutrient enrichment were considered through the integration of threshold responses from multiple ecosystem variables. This integration led to the development of categories of aquatic ecosystem health impairment (AEHI) risk, which are bound by numeric concentrations of TN and TP. In total, four categories of AEHI risk, from low to very high risk, were defined separately for the spring (April – May) and summer (June – August) seasons within the Grassland and Parkland natural regions. Separate AEHI risk ranges were determined according to season and natural region as it was estimated that the relationship between aquatic ecosystem responses and nutrient concentrations would change according to large-scale geographic and seasonal factors.

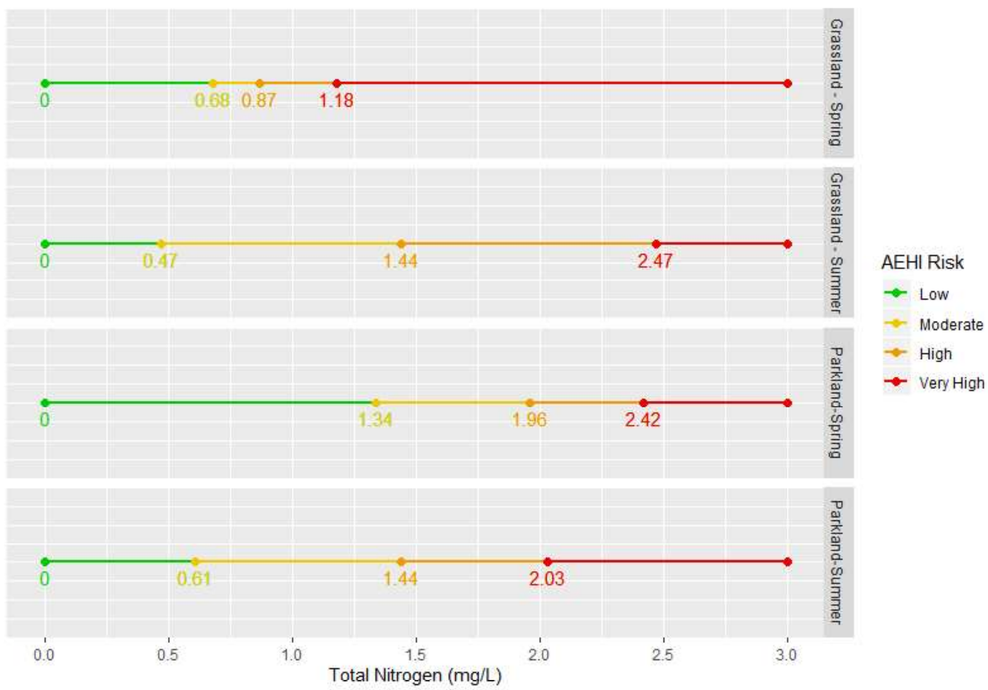

Figure E-1. Numeric nutrient objectives for total nitrogen (TN) that define the boundaries of aquatic ecosystem health impairment (AEHI) risk zones for each combination of natural region and season.

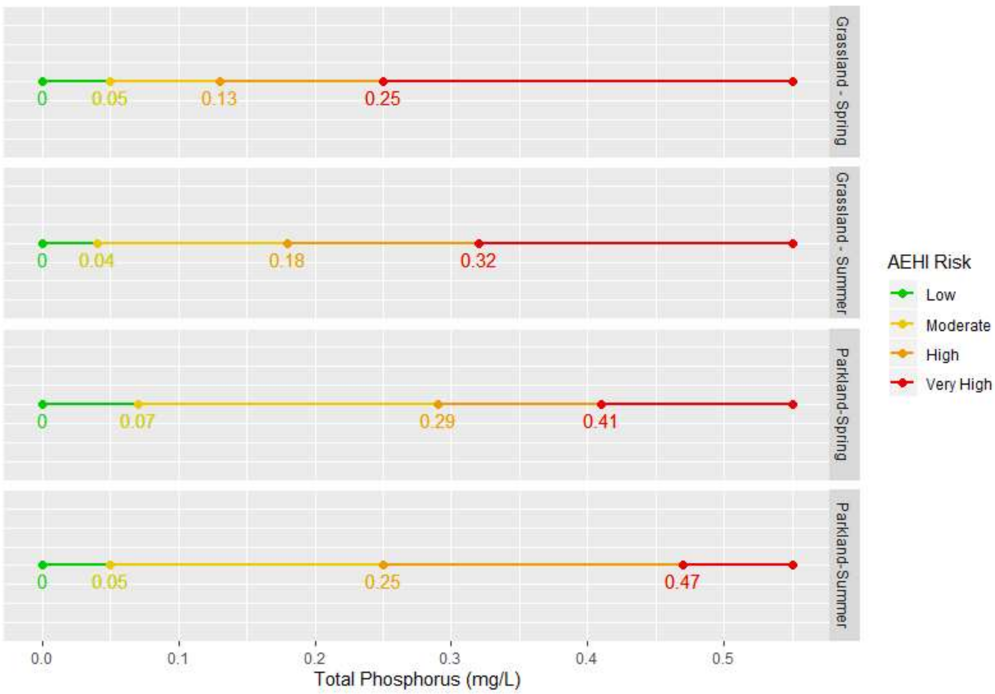

Figure E-2. Numeric nutrient objectives for total phosphorus (TP) that define the boundaries of aquatic ecosystem health impairment (AEHI) risk zones for each combination of natural region and season.

The numeric nutrient objectives, defined as boundaries to AEHI risk zones, derived through this study will allow for an improved assessment of water quality impairment owing to nutrient enrichment, as well as a phased-approach to watershed management in agricultural watersheds, where watershed managers can work toward continuous improvement of water quality. The derived objectives are also meaningfully protective of aquatic ecosystems. The transition from low-to-moderate AEHI risk tends to reflect commonly-used guideline values of TN and TP, and offers a sensitive indicator of change in minimally disturbed areas. However, these values may not be achievable in watersheds with intensive human land-use pressures. Values indicative of progressive deterioration, or conversely improvement, of aquatic ecosystems allow for a tiered, progressive approach to water quality management in agricultural watersheds.

As a result of the approach applied, the nutrient objectives derived here are applicable to all 3rd to 5th Strahler order streams in the Grassland and Parkland natural regions. Given the broad applicability of the objectives, a secondary objective of the study was to evaluate the relationship of the generalized objectives to site-specific objectives (SSOs) set for a specific stream system. A process modelling approach, using the QUAL2K in-stream water quality model, was used to set SSOs at two streams, one in each natural region, that have been historically studied and are known to have impaired water quality. The SSOs in this study defined the point at which concentrations of TN and TP crossed a threshold of pre-determined pH and dissolved oxygen (DO) thresholds, which are known to be protective of aquatic ecosystem health. Differential relationships between the pH and DO threshold responses were observed between the streams, indicating the response of aquatic ecosystems to nutrients depends on the stream environment. However, the SSOs defined in this way predominantly fell within the moderate or high AEHI risk categories of the generalized objectives. This indicates that the generalized objectives are both reasonably protective of aquatic ecosystems and are suitable benchmarks for improving aquatic ecosystem conditions at watersheds with high intensity of land-use pressures.

The achievability of the generalized nutrient objectives, as well as the SSOs, were assessed through watershed-scale simulations of changes to agricultural land management practices using the Comprehensive Economic and Environmental Optimization Tool (CEEOT) modelling platform. The CEEOT platform models the environmental benefits and farm economic impact of introducing beneficial management practices (BMPs) on the agricultural landscape. Modelling results demonstrated positive reductions of nutrient loading from the landscape into surface waters owing to the adoption of BMPs by agricultural producers. However, reductions in monthly flow-weighted mean concentrations (FWMC) were not associated with substantial improvements to water quality with respect to reductions in AEHI risk. The ability to compare the CEEOT model results to the AEHI risk boundaries and the SSOs for the study watersheds were challenged by: (i) the monthly FWMCs of nutrients were not calculated in a manner that reflects true in-stream concentrations; and (ii) the land-use scenarios included approximately 35% of the available land area, which was viewed as an attainable standard but may not be sufficient to affect in-stream nutrient concentrations. Additional work will be performed to integrate a more suitable in-stream water quality model with the CEEOT platform, and progressively larger land-areas will be simulated to determine the scale of land-use change needed to reduce AEHI risk.

The numeric nutrient objectives derived in this study can be broadly applied to small streams in the agricultural region of Alberta. The objectives were formed by a comprehensive evaluation of aquatic ecosystem response to gradients in nutrient concentrations, and appear to be protective of aquatic ecosystem health. Water and watershed managers can readily compare the values of water quality monitoring results to the standards to gauge the relative risk of aquatic ecosystem health impairment that a system may be confronting owing to nutrient enrichment. In addition, the values can be integrated as management targets or triggers in watershed management programs, as they appear to be reasonably reflective of changes in aquatic ecosystem parameters.

1. Introduction

1.1 Background

In Alberta, water management proceeds through a cumulative effects management framework that must be met by all economic sectors in the province. Beginning in 2003, the Water for Life (WfL) strategy set out goals to balance concurrent uses of safe secure drinking water, aquatic ecosystem health, and quality supplies to support a sustainable economy (AE, 2003). As part of the WfL strategy, Watershed Protection and Advisory Councils (WPAC) were created for each major river basin and empowered to report on the state of watershed and to prepare integrated watershed management plans (IWMPs). The IWMPs are voluntary, cooperative efforts aimed toward multi-stakeholder management of water resources. In 2008, principles of cumulative effects management were included in the Land-Use Framework (LUS, 2008), which is legislation that empowers statutory plans for environmental management. Under the LUF, seven land-use regions were created, each led by a multi-sector Regional Advisory Council mandated to manage the cumulative effects of land-use and resource extraction on land, air and water resources. Each regional plan is required to establish statutory plans for water resources, through approved Water Management Frameworks (WMF), that set out the goals and objectives of water quantity and quality within the region. The IWMPs are often used as advisory documents to the WMFs within the regional plans, but are not explicitly linked. Key to the success of both the IWMPs and regional plans are the consultation, involvement, and participation of all sectors within the watershed such that collaborative approaches are built to meet water management goals.

The area primarily suited for agriculture and settlement in Alberta, the White Area, accounts for 39% of the provincial land area and approximately 75% of it is privately held and predominantly used for agricultural activity (GoA, 2007). Hence, the agricultural sector has a relatively large geographic footprint in Alberta and non-point source pollution from agricultural activity is of concern to the management of water resources. Two federal-provincial studies conducted in Alberta [CAESA (1992 – 1997) and AESA (1999 – 2008)] revealed that an increase in the intensity of agricultural activity was associated with increasing concentrations of nutrients, in surface waters draining small agricultural watersheds (Anderson et al., 1998; Lorenz et al., 2008). Differences in nutrient export from agricultural land to surface water was also observed among different natural regions, where nutrient exports were highest in the Boreal and lowest in the Grassland, due to differences in landscape and climate (Lorenz et al., 2008).

Recognizing that agricultural activity can contribute to a decline in environmental quality, the agricultural sector has committed to principles of sustainable development, as evidenced by the growing adoption of sustainability frameworks. The *Unilever Sustainable Sourcing Program for Agricultural Raw Materials* and the *Sustainable Agriculture Initiative (SAI) Platform*, are global frameworks that make up the Farm Sustainability Readiness Tool that can be used by crop producers in Alberta (AFSE, 2018). In the livestock sector, Certified Sustainable Beef (CSB) and

Verified Beef Production Plus (VBP+) are verification frameworks that can be applied by livestock producers to market their products as being sustainably produced. Water management, particularly minimizing water use and protecting water quality through appropriate management of nutrients and manure, figure prominently within the crop and livestock sustainability frameworks. Thus the agriculture sector recognizes and is taking steps toward judicious management of water resources in a manner that is consistent with the principles of cumulative effects management.

Measuring water quality outcomes best occurs through the comparison of water quality parameters to established numeric values that are indicative of suitable water quality or aquatic ecosystem health. However, numeric guidelines for nutrients that can be used to interpret the success of agricultural land management practices on maintaining or improving water quality at the scale of small watersheds, where land management practices are intimately linked with surface water quality, do not exist in Alberta. In current practice, it remains challenging to interpret water quality results for small streams, plan watershed management activities for reducing nutrients, and measure the success of nutrient reduction practices in watersheds with predominantly agricultural land uses. While many studies define success as an overall reduction in nutrient loads, the resultant concentrations cannot be put into the context of aquatic ecosystem health, as aquatic ecosystems respond to concentrations instead of load effects.

Two general approaches exist for setting numeric values for nutrients (or other non-toxic chemical stressors): (i) the identification or estimation of background conditions either by assessing minimally disturbed watersheds or by modelling; or (ii) the identification of boundaries associated with ecological impairment, which identify the concentrations of N or P associated with ecologically meaningful changes in aquatic communities.(Dodds and Oakes, 2004; Chambers et al. 2012a). The extent of land conversion in the White Area precludes the definition of appropriate nutrient values through an assessment of control watersheds. In the National Agri-Environmental Standards Initiative (NAESI), a joint initiative by Environment Canada (EC) and Agriculture and Agri-Food Canada (AAFC) designed to establish non-regulatory environmental performance standards to guide agri-environmental decision-making in agricultural regions across Canada, provisional standards for nitrogen and phosphorus in surface waters were recommended for all major ecoregions of Canada (NAESI, 2009). However, the provisional standards recommended for the Prairie region were established using chemical-threshold approaches that are based on statistical distributions of monitoring data; the NAESI team identified a lack of biological assessment information paired with water quality data in the Prairie region that could meaningfully be used to define nutrient standards in this region (Chambers et al. 2012a,b).

The lack of numeric standards for concentrations of nutrients that reflect changes in aquatic ecosystem structure or function is a key knowledge gap for water resource management in Alberta. Their availability will aid cumulative effects management either through voluntary IMWPs, statutory regional plans, or industry-driven sustainability initiatives. The provision of numeric nutrient objectives will improve watershed management in Alberta by providing appropriate benchmarks for assessing the risk of aquatic ecosystem health impairment, prioritizing areas for expending scarce resources, and evaluating the success of watershed management programs.

1.2 Scope and Objectives

The goal of this study is to recommend numeric nutrient objectives for small streams in agricultural watersheds that can be used to guide watershed management activities on a provincial scale. The numeric objectives were established through a stressor-response study design (EPA, 2010; CCME, 2016), whereby structural and functional aquatic ecosystem components were measured together with nutrient concentrations in a range of representative streams. A stressor-response study design was chosen in recognition of the historic conversion of land to agricultural production, the intensity of agricultural production in the study watersheds, and the fact that agricultural land is unlikely to be taken out of production. The study outcomes are intended to inform water quality standards that can be realistically achieved in the socio-economic conditions of the White Area of Alberta.

In this study, distinct nutrient objectives established on a natural region basis were desired given that the concentrations, forms, and export of nutrients in agricultural streams were found to differ among the Grassland, Parkland and Boreal natural regions of Alberta (Lorenz et al., 2008). Distinct seasonal nutrient objectives were also desired as the response of aquatic ecosystems to nutrients is expected to change between spring and summer (USEPA, 2010). The project proceeded with three specific objectives:

1. Derive seasonal numeric nutrient objectives for total nitrogen and total phosphorus that can be generally applied to small streams (3rd to 5th Strahler order) in the Grassland and Parkland natural regions of Alberta;
2. Define site-specific nutrient objectives using an in-stream water quality model (QUAL2K) for two streams in high-intensity agricultural watersheds to assess the generalizability of the stressor-response approach;
3. Assess the achievability of the numeric and site-specific objectives through simulation of agricultural management practices in two agricultural watersheds.

In this report, nutrient objectives are defined as the numeric bounds of four categories of aquatic ecosystem health impairment (AEHI) risk. This approach was permitted through the concurrent evaluation of 25 aquatic ecosystem response parameters together with analysis of nutrient stressors. In this way, AEHI risk categories will facilitate assessments of aquatic ecosystem health, and the numeric bounds allow for flexible management of target streams. Objectives were only derived for total nutrients (TN and TP), given that rapid uptake of dissolved nutrients can make them unreliable indicators of eutrophication (CCME 2016).

This report presents methodological details (Section 2) and statistical approaches (Section 3) used to derive the generalized nutrient objectives. Section 4 assesses the representativeness of the generalized objectives through comparison to site-specific objectives derived using an in-stream water quality model. Section 5 assesses the achievability of the generalized and site-specific objectives through watershed-scale simulations of land management practices.

2. Application of Stressor-Response Study Design to Agricultural Streams

2.1 Stressor-Response Study Design

This study applied a stressor-response study design, which involves collecting concurrent information on stressor variables (e.g., nutrients) of increasing concentrations and ecosystem response variables (e.g., dissolved oxygen). These designs are recommended for setting numeric standards for non-toxic chemical stressors, such as nutrients, and where conditions do not permit comparison to control (i.e., undisturbed or minimally-disturbed) conditions (EPA, 2010; CCME, 2016). In the White Area of Alberta, most watersheds have a large degree of land conversion to agricultural, municipal, or other land-uses, so a control-comparison was not possible. In this study, a separate stressor-response design was applied to both the Grassland and Parkland natural regions of Alberta. These regions contain a high degree of agricultural activity, yet have been found to have differing concentrations and forms of nutrients. For instance, the Parkland natural region has been found to have higher nutrient concentrations, as well as a greater proportion of dissolved nutrients than the Grassland natural region (Lorenz et al., 2008).

Thirty streams of 3rd – 5th Strahler order were targeted within each of the Grassland and Parkland natural regions of Alberta for inclusion in the study in order to achieve a gradient in nutrient concentrations, and to ensure sufficient power to detect thresholds in aquatic ecosystem response to nutrients. Concentrations of total nitrogen (TN) and total phosphorus (TP) make up the stressor component of the study design. The objectives were set for total nutrients (i.e., inclusive of dissolved and particulate forms) given the variability in nutrient fractionation owing to dynamic biological and geochemical processes occurring in surface waters (Dodds, 2003).

Aquatic ecosystem components forming the response metrics include both structural and functional ecosystem components. Aquatic ecosystem structure was measured from algal communities both in suspended (phytoplankton) and attached (periphyton) forms. Phytoplankton algal communities were assessed only in terms of their total biomass, whereas both periphyton community biomass and composition were assessed, the latter both in term of pigment-derived and taxonomically-derived composition. Stream function responses included litter decomposition rates, nutrient uptake rates, and ecosystem metabolic rates of gross primary production and ecosystem respiration. All components were chosen in accordance with recommendation from an expert workshop held on best indicators of nutrient enrichment in streams (EPA, 2013).

2.2 Site Selection

Thirty streams of 3rd – 5th Strahler order were targeted within each of the Grassland and Parkland natural regions for inclusion in the study. Study streams were selected by the following criteria:

1) Historically monitored streams were preferentially included in the analysis, so that the available data could be used to inform secondary methods for setting nutrient objectives based on chemical thresholds, if required.
2) The remaining streams were selected within candidate Hydrologic Unit Code 8 (HUC8) watersheds using a stratified random sampling procedure. Stratifications were based on:
   a) runoff potential using the Water Erosion Potential map generated for Alberta's agricultural area (Jedrych et al. 2004); and
   b) agricultural intensity indices (Anderson et al. 1999);
3) Each stratification was separated into two zones defined by median quantiles of continuous data, yielding four combined categories of runoff potential and agricultural intensity;
   a) Up to five streams were randomly selected within each group in each natural region;
4) Monitoring sites in the selected streams were positioned as close as practicable to the watershed outlet. The following conditions were required at each monitoring site:
   a) Relatively open canopy so as not to introduce light limitations;
   b) Persistent flow conditions throughout the growing season (i.e., non-ephemeral);
   c) Easy access from roadway crossings;
   d) Landowner permission to access the stream through privately held land.

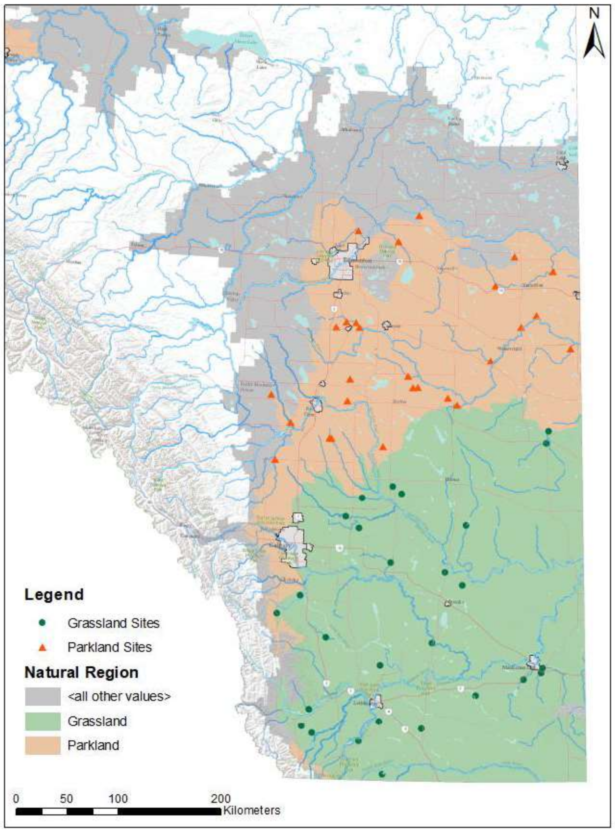

Figure 2-1. Location of study sites in the Parkland (orange) and Grassland (green) natural areas of Alberta. Portions of the agricultural zone located in the Boreal natural region are indicated in grey, but were not included in this phase of the project.

Using these criteria, 33 streams were selected in each natural region; however, only 27 and 28 streams were retained in the Grassland and Parkland natural regions, respectively. Difficulty in finding site access and/or amenable landowners that permitted access, or drying-out of selected streams were the primary causes for reducing overall site numbers. The retained sites had a relatively broad coverage over the study extent (Figure 2-1). Locations and characteristics of the study sites are presented in Tables A-1 and A-2.

2.3 Field Sampling Schedule

Ecosystem assessments were to be completed in both the spring (April – May) and summer (June – August) months of three years (2016 – 2018). This separation was desired to account for variable hydrology of small streams in the spring, which are driven by snowmelt dynamics and immature ecosystems, and the summer, which are typically lower flow and of mature ecosystem structure. Algal bioassessments were conducted in spring and summer of each year. However, logistic constraints with contracting and procurement at the onset of the study precluded the spring 2016 bioassessment. Aquatic ecosystem function was assessed variably between spring and summer. The whole-stream metabolism work proceeded in both spring and summer owing to the relative ease of the field methods. However, the litter decomposition and nutrient uptake experiments were only performed in the summer months owing to longer-term deployments and human resource constraints, respectively.

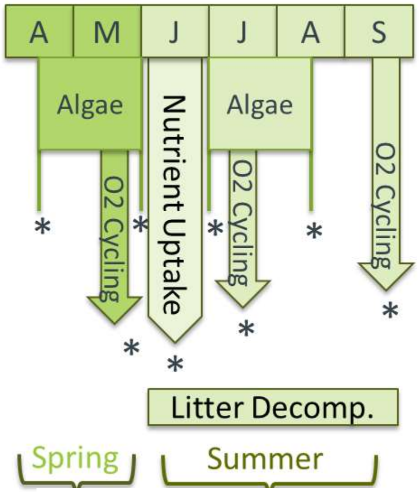

Figure 2-2. Generic schedule of aquatic ecosystem biological and functional assessments. Asterisks denote water sample collection and flow monitoring.

In total, algal bioassessments were completed on all streams included in the study and were paired with water sampling, flow measurement, and in situ water quality at each deployment and retrieval. However, due to the added complexity of the functional assessments, either through a need for increased equipment, materials, supplies or human resources, the function assessments could only be performed on a subset of the streams. Function assessments were performed on fifteen streams in 2017 and 2018; the focus was on the Grassland region in 2017 and the Parkland region in 2018. In 2017, 12 streams from the Grassland region and three streams from the Parkland region were assessed. In 2018, 12 streams (including the three streams from 2017) within the Parkland region and three streams (of the 12 monitored in 2017) were included in the assessments. The three-stream overlap between years was included to assess inter-annual variation in aquatic ecosystem function.

2.4 Water Sample Collection and Analysis

At each water sampling event, samples were collected at mid-channel, mid-depth through grab sampling, using either a telescoping sampling pole or by hand where channel and flow conditions permitted. All samples were stored on ice and submitted immediately to a contracted laboratory (ALS Environmental Ltd., Edmonton, AB) for analysis of water quality parameters outlined in Table A-3. At the time of sample collection, stream flow was measured through acoustic Doppler velocimetry, either using a handheld (SonTek FlowTracker, Xylem Inc.) or boat-mounted (SonTek StreamPro) device, and in situ water quality parameters were collected using a multi-parameter water quality sonde (In Situ SmarTROLL). Measured flow properties for the study sites are presented in Table A-4 and A-5, and in situ and laboratory water quality results are summarized in Table A-6 and A-7.

In stressor-response study designs, sites are intended to be selected to establish a gradient of the stressor variable. Here, the stressor variable consisted of concentrations of total nitrogen and total phosphorus. Evident from the collected data is that the Parkland natural region generally contains higher concentrations of total nitrogen and phosphorus than the Grassland region, but little general seasonal difference is apparent overall (Figure 2-3). The average concentrations of total nutrients collected from each stream (Figure 2-4) indicate that stressor gradients were established through the site selection procedure outlined in Section 2.2.

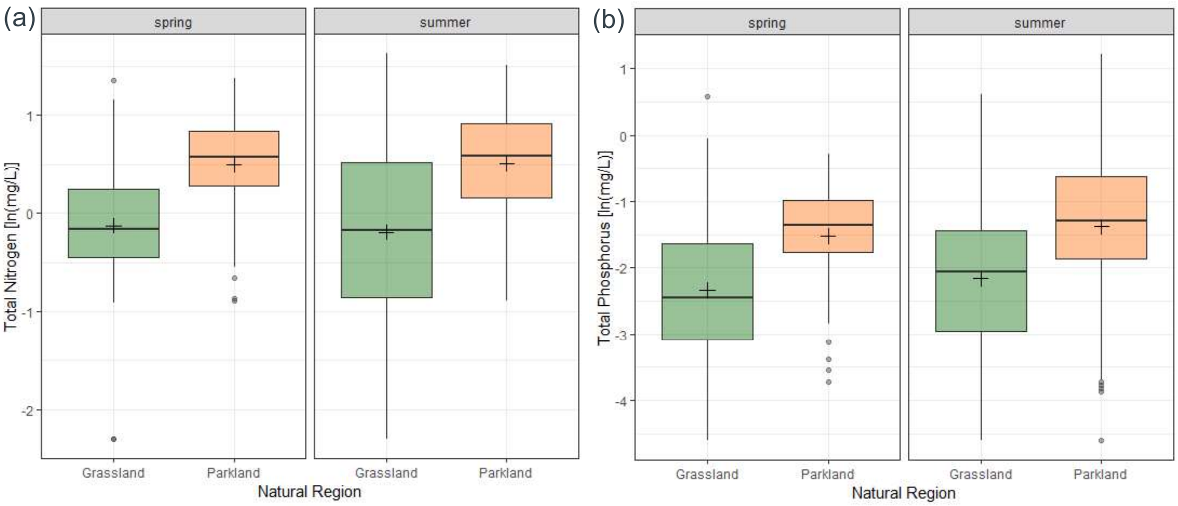

Figure 2-3. Seasonal distribution of total nitrogen (left) and total phosphorus (right) in the Grassland (green) and Parkland (orange) natural regions. Concentrations are presented in natural logarithm units. Box boundaries indicate the lower (25^{th}), middle (50^{th}) and upper (75^{th}) quartiles, stick ends denote 5^{th} and 95^{th} percentiles, circles indicate outliers, and the (+) is the mean value.

2.5 Aquatic Ecosystem Structural Responses

Stream ecosystems are complex and include many groups of organisms whose community structure responds to nutrient enrichment. In this study, algal communities were chosen for assessment of structural responses given that they are often considered to show more sensitive responses to nutrient concentrations than higher-level organisms, being primary producers that directly use inorganic nutrients (EPA 2013). Algal bioassessments were conducted through the use of floating algal samplers containing glass microscope slides (periphytometers), which were chosen to standardize the attachment matrix and reduce the confounding effect of substrate type on algal community development (Tarkowska Mieczan, 2012). The periphytometers were installed by tethering to a cinder block with 1 – 2 m of airline cable and placing them mid-channel. A temperature and light pendant logger (HOBO UA-64, Onset Corp.) was attached to each periphytometer. The periphytometers were retrieved after approximately four weeks following deployment (Biggs, 1998). Upon collection, the microscope slides were placed into LockMailer™ (Simport Scientific Inc.) microscope slide tubes. One half of the slides were transported in insulated coolers for taxonomic identification (MB Laboratories, Sydney, BC) and the other half of the slides were transported on ice for pigment and biomass analysis (Vinebrooke lab, University of Alberta, Edmonton, AB). Taxonomic cell counts were performed following protocols presented in Strickland and Parsons (1972), using a Rafter cell counter. Pigment analysis was performed using high performance liquid chromatography (HPLC), and biomass of the entire slide biofilm (comprised largely of periphyton but also of other microorganisms) was measured as ash-free dry mass (APHA, 1999).

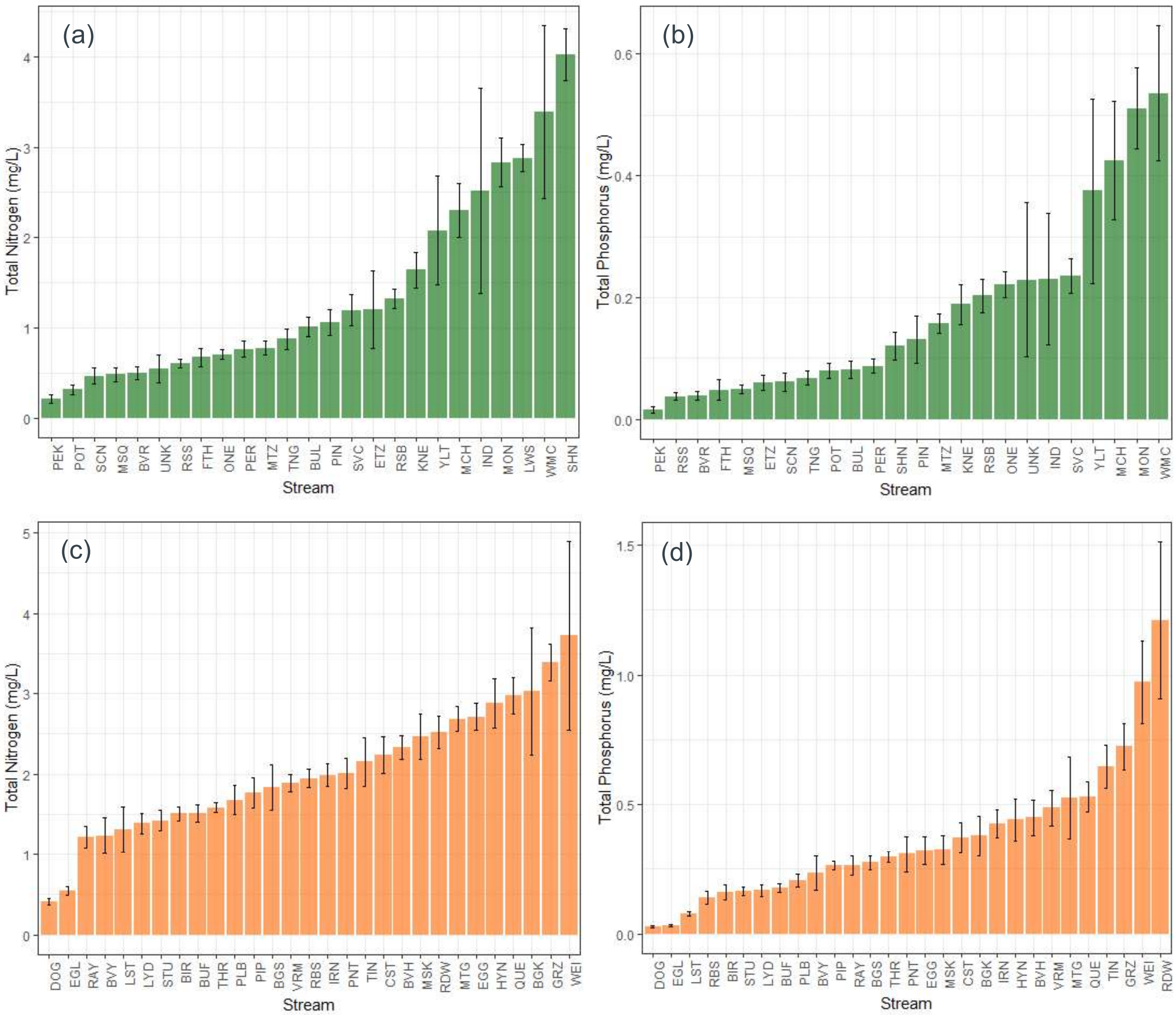

Figure 2-4. Mean concentrations of total nitrogen (a and c) and total phosphorus (b and d) in the Grassland (green) and Parkland (orange) natural regions, in ascending order of concentration. Error bars represent ± standard error of the mean.

Despite the concentrations of nutrients being evidently higher in the Parkland natural region in comparison to the Grassland natural region, coarse metrics of algal concentration, biomass and communities were not substantially different between the natural regions (Figure 2-5). However, seasonality in growth yielded some noticeable trends on the aggregate. In the Parkland, phytoplankton Chlorophyll a (Chl a) concentrations were observed to be higher in spring than in summer, but were higher in summer over spring in the Grassland region. Higher concentrations of periphyton Chl a were observed in spring over summer in the Parkland and Grassland natural region, although the central tendencies (means and medians) were not markedly different between the seasons with natural regions. Generic richness, and diversity to a lesser extent, of algal taxa also tended to be higher in spring than summer for both natural regions.

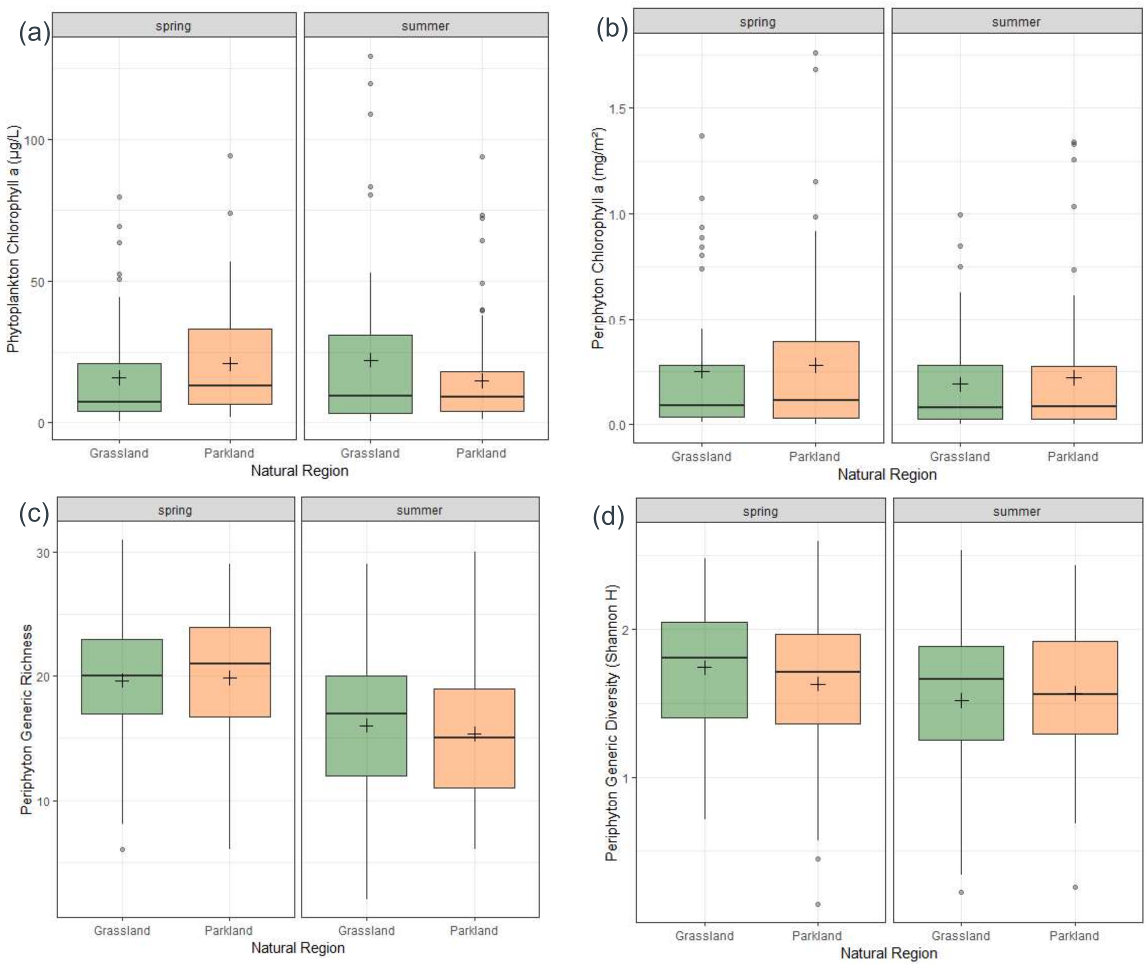

Figure 2-5. Metrics of algal community responses to in the Grassland (green) and Parkland (orange) natural regions, including Chlorophyll a in phytoplankton (a) and periphyton (b), periphyton richness (c), and Shannon Diversity Indices (d). Algal metrics are separated according to seasonal assessment periods.

2.6 Aquatic Ecosystem Functional Responses

2.6.1 Whole Stream Metabolism

Whole-stream metabolism includes metrics of gross primary productivity (GPP), ecosystem respiration (ER), and net ecosystem productivity (NEP). While these metabolic rates intended to inform the rates of production and decomposition of carbon compounds, respectively, their calculation is based upon the production and consumption of oxygen (Table A-8). Oxygen measurements are a suitable proxy for carbon in this context, as it can be readily and continuously monitored with electronic sensors and is theoretically linked with carbon production and decomposition (Young et al. 2008). Measurements of dissolved oxygen and water temperature were made with multi-parameter water quality sondes (YSI EXO2, Xylem Inc., Yellow Springs, OH, USA) that were deployed over 3 – 5 day periods. Photosynthetically active radiation (PAR) was measured concurrently using Odyssey™ deployable waterproof loggers (Dataflow Systems Ltd., Christchurch, New Zealand) attached to the sonde housing. Flow measurements were collected at both the deployment and retrieval of the sondes, and water samples were collected at either the start or end of the deployment. Metabolic rates of GPP, ER, and NEP were calculated for each deployment using the streamMetabolizer package (Appling et al. 2018) in R (v. 3.5.2, 2018) using maximum likelihood estimation (MLE) methods. Summaries of the stream metabolic rates for each stream is presented in Table A-9.

With respect to nutrient enrichment, GPP and NEP are expected to be more responsive as they are directly linked to algal growth and production, whereas ER can be influenced by the heterotrophic processing of allochthonous (externally-derived) organic material from adjacent landscapes (Young et al., 2008). However, NEP is the balance between GPP and ER. From the data collected, seasonal trends in GPP are evident, where GPP is highest in the summer and lowest in the fall (Figure 2-6). The Grassland region appeared to have slightly higher mean and median concentrations of GPP in all seasons. Mean and median NEP was also positive in Grassland streams, indicating that GPP is greater than ER, and hence autotrophic producers are more prolific than heterotrophic consumers in this region.

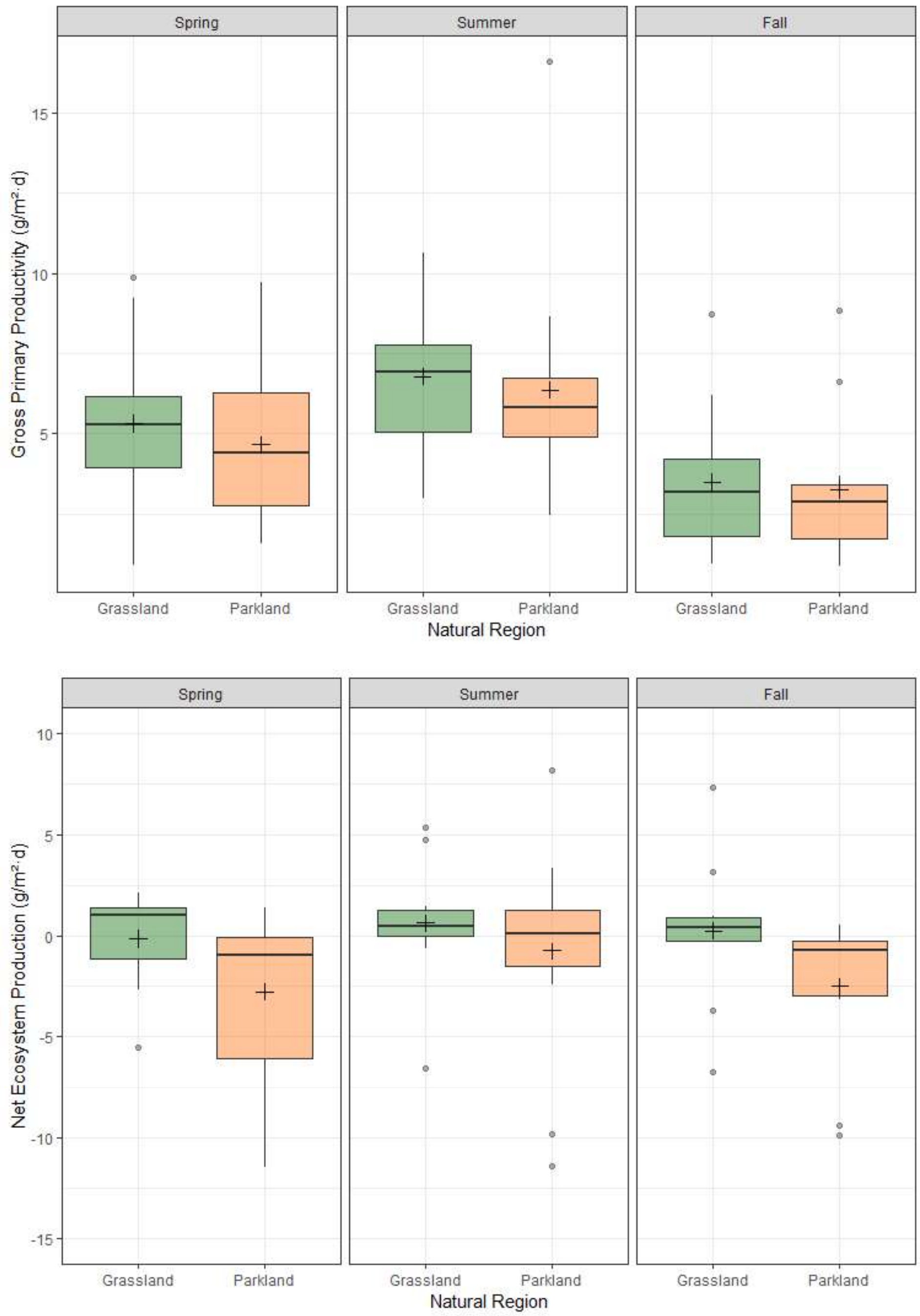

Figure 2-6. Gross primary productivity (a) and net ecosystem production (b) in the Grassland (green) and Parkland (orange) natural regions. Seasonal categories include spring (April - May), summer (June - August) and Fall (September - October). Box boundaries indicate the lower (25^{th}), middle (50^{th}) and upper (75^{th}) quartiles, stick ends denote 5^{th} and 95^{th} percentiles, circles indicate outliers, and the (+) is the mean value.

2.6.2 Litter Decomposition

Litter decomposition is an important ecosystem function as it is responsible the integration of allochthonous (from terrestrial sources) organic matter into the aquatic food chain. In this study, both microbial and total litter decomposition rates were measured using, respectively, fine-mesh (0.2 mm) and coarse-mesh (2 mm) litterbags filled with dried reed grass (*Phalaris arundinacea*) harvested from a common source, and constructed with Nitex mesh. For each site, twelve fine-mesh and twelve coarse-mesh bags were strung onto a 20' airline cable, which was then tethered to the stream bank, below the water level. At the time of installation, two of each fine- and coarse-mesh bags were removed from the tethered line (i.e., were installed into the water), and processed as the Day 0 samples. The remaining samples were collected in pairs of fine- and coarse-mesh bags at each of five collection events at approximately 2 – 3 week intervals. At the time of retrieval, all bags (including the Day 0 bags) were rinsed in stream water by inserting, lifting and draining the bags ten times; this allowed for the removal of surface debris collected on the litterbag. The litterbags were stored in clear plastic bags and placed in an ice-filled cooler for transport to the laboratory, and then stored at fridge temperature until processing. Processing consisted of rinsing the litterbags in deionized water to remove silt, drying the biomass at 50°C until it reached a constant mass, and measuring total biomass of the sample.

Litter decomposition rates are expected to increase under moderate nutrient enrichment, as the aquatic ecosystem becomes more productive and heterotrophic organisms proliferate, but are expected to decline with high nutrient enrichment as higher-level organisms become stressed (Young et al., 2008). In this study, total and microbial decomposition rates were observed to be relatively similar between the natural regions, although the range of values tended to be higher in the Grassland region (Figure 2.7). Summaries of decomposition rates calculated for each stream are presented in Table A-10.

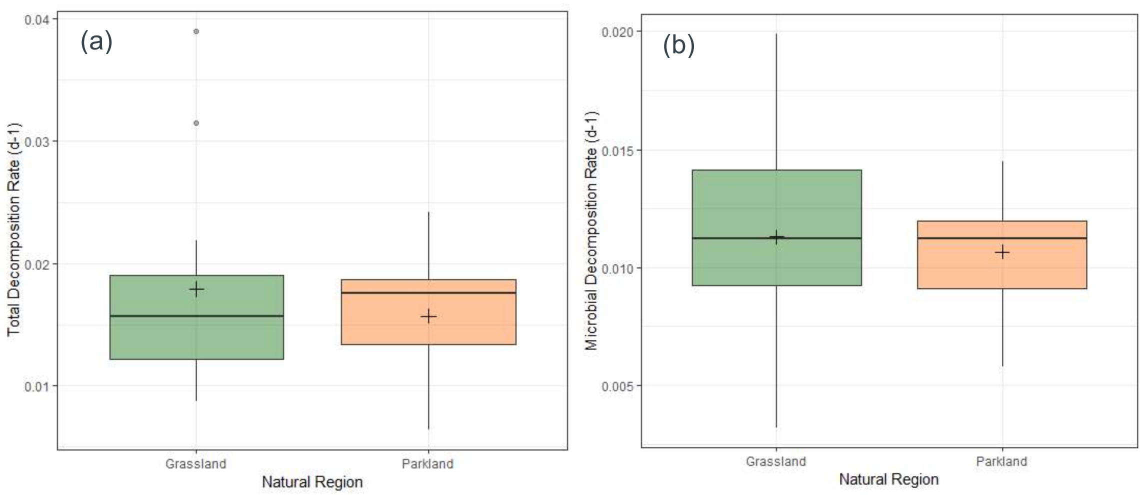

Figure 2-7. Total (a) and microbial (b) decomposition rates of grass litter in the Grassland (green) and Parkland (orange) natural regions.

2.6.3 Nutrient Uptake Rates

The attenuation of nutrients in small streams is an important ecosystem function that preserves the health of downstream receiving environments, such as lakes, reservoirs and rivers. Healthy aquatic ecosystems have some capacity to retain nutrients generated from land-use activity. However, nutrient retention capacity can be saturated under conditions of high nutrient enrichment, potentially leading to greater downstream impacts (Bernot and Dodds, 2005). Nutrient uptake rates were measured through solute tracing studies, according to protocols presented in the Stream Solute Workshop (1990). Briefly, in each study stream a concentrated stock solution of chloride, ammonia, and phosphate was prepared and injected at a constant rate into the stream to generate a continuous plume. Longitudinal changes in concentrations were measured at five locations downstream of the injection site. Downstream sampling stations were determined on-site during the day of the tracer injection in order to adapt the sampling protocol to the concurrent flow conditions. The length of the mixing zone was calculated using the method supplied in the *Code of Practice for Hydrologic Tracing Analysis Studies* (Alberta Environment, 1998). The first sampling station was located approximately at 10% or 20 m beyond the calculated mixing zone length, whichever was greater. The fifth and terminal sampling location was estimated from the average velocity of the stream to accommodate a 4-hour continuous injection. The second through fourth sampling location were placed approximately equidistant between the first and final sampling locations. Pre-injection samples were collected at each site to characterize the ambient concentrations of the tracers. Once the injection was started, the first and fourth monitoring sites were monitored for changes to specific conductance using deployable multi-parameter water quality sondes (YSI EXO2, Xylem Inc.). Sampling commenced once a plateau was reached at the fourth station, and proceeded in order from the first to fifth stations. All samples were held on ice and submitted to ALS Environmental (Edmonton, AB) for analysis of total and dissolved nutrients (Table A-3). Nutrient uptake parameters calculated for each stream are presented in Table A-10. Nutrient uptake velocities were selected as the most appropriate uptake rate to represent differences between streams, as they are a measure of nutrient spiraling, or the distance travelled by inorganic solutes before being taken up, standardized for stream flow (Figure 2-8). Nutrient uptake velocities were observed to be higher in the Grassland natural region, particularly for ammonia-nitrogen (Figure 2.8).

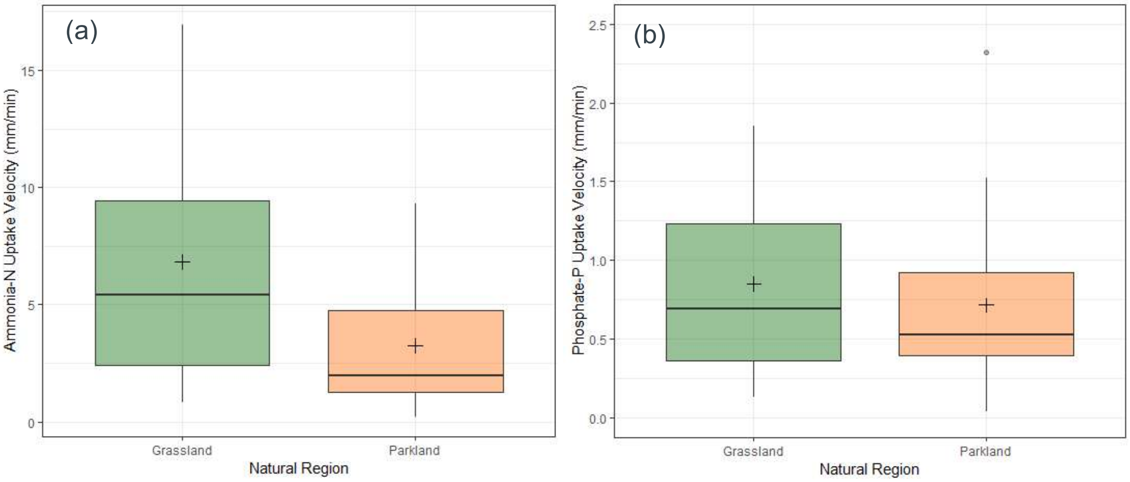

Figure 2-8. Uptake velocities of ammonia (a) and phosphate (b) in the Grassland (green) and Parkland (orange) natural regions. Box boundaries indicate the lower (25th), middle (50th) and upper (75th) quartiles, stick ends denote 5th and 95th percentiles, circles indicate outliers, and the (+) is the mean value.

2.7 Summary

In total, 25 and 29 streams were monitored in the Grassland and Parkland natural regions of Alberta in an effort to gain necessary information for calculating numeric nutrient objectives using a stressor-response study design. Applying separate objectives for each natural region appears to be appropriate given the observed differences in ambient nutrient concentrations between the regions, and the mixed differences in ecosystem response variables. Evaluation of threshold responses of the varying response values to increasing concentrations of nutrients are explored in Section 3.

3. Deriving Nutrient Objectives from Stressor-Response Data

3.1 Rationale and Overview of Approach

The stressor-response approach chosen for the development of numeric nutrient objectives in this study is commonly used to define levels of water quality parameters that are protective of a designated use (e.g. aquatic ecosystem health). For mid-size streams in the agricultural region of Alberta, however, target levels of nutrient concentrations or other metrics representing aquatic ecosystem health have not been recommended. For this reason it was decided to investigate relationships between TN and TP and a variety of response metrics related to aquatic ecosystem health, and to empirically identify thresholds in these relationships that could be used to guide the suggestion of nutrient objectives.

Ecological thresholds are often interpreted as transitions between ecological states in which a return to a less impacted state is difficult once a threshold is crossed (Groffman et al., 2006). This concept makes ecological thresholds attractive candidates for management objectives, as evidenced by the use of empirical thresholds to suggest nutrient management standards in a variety of jurisdictions (e.g. Black et al. 2011; Chambers et al. 2012; Evans-White et al. 2009; Qian et al. 2003; Richardson et al. 2007; Smith et al. 2013).

In stressor-response relationships, estimates of threshold values in stressor variables -are typically different depending on the response metric being examined and the statistical methods used to detect the threshold (Black et al. 2011; Brenden et al. 2008; Dodds et al. 2010; Qian 2014). Therefore, instead of relying on a single response metric and statistical method, 25 response metrics (Section 3.2) and three threshold-detection methods (Section 3.3) were used to establish a range of potential ecological thresholds applicable to aquatic ecosystems in each natural region. Different statistical methods were applied for threshold analyses in acknowledgement that stressor-response relationships may not all be best-represented by the same model, yet uncertainty exists in which may be the best model in a given relationship (Qian, 2014). Assessments of the strength of each identified threshold were conducted to prevent the inclusion of spurious results, as detailed in Section 3.4.

The range of ecological thresholds obtained from the above analysis was used to define numeric boundaries of aquatic ecosystem health impairment (AEHI) risk categories (low, medium, high, and very high) that can be used flexibly to set nutrient objectives in any given scenario (Section 3.5.3). In this study, the numeric bounds of the AEHI risk categories were established using the 10^{th}, 50^{th}, and 80^{th} percentiles of weighted empirical cumulative distribution functions of threshold values (Section 3.5.1).

3.2 Response Metrics and Datasets Used in Threshold Analyses

A variety of univariate metrics, calculated from data collected as per Section 2, that are known or hypothesized to respond to nutrient enrichment in aquatic ecosystems were selected for threshold analysis (Table A-11). A total of 25 metrics were selected, and correlation analysis confirmed that redundancy of metrics was not a concern as strong correlations among the metrics was very rare (data not shown). Prior to threshold analysis, variables were transformed to reduce skewness and to help meet the assumptions of the parametric piecewise linear regression method (Section 3.3) (Qian 2014; Qian and Cuffney 2012). Total nitrogen and phosphorus, which were both right-skewed, were natural-log transformed. Transformations used for response metrics are listed in Table A-10. Before transformation and further analyses, below-detection-limit data (generally less than 5% of a given variable) were set at half the respective detection limit. Due to the low frequency of total nutrient concentrations being below detection, a proportional approach was used to impute values due to its simplicity, presumption that significant bias of the results would occur, and the lack of statistical threshold methods that are designed for use with censored data.

Threshold analysis of response metrics derived from periphyton data were performed separately for each natural region and season. The periphyton community metrics at each site were related to the average of the two water samples collected at the time of deployment and retrieval of the periphytometers. For periphyton Chl *a*, data from 2016 were excluded as the analytical lab and method were switched and an analysis of the data indicated that there were likely methodological differences in the estimation of Chl *a* that could confound statistical relationships to nutrients. In the analysis of water column Chl *a* both Chl *a* and nutrient concentrations comprised average values from the (generally two) samples collected in a given season in a given year, as for nutrients in periphyton analyses. For periphyton response metrics, two years of data (2017 and 2018) were available for spring threshold analyses, whereas three years (2016, 2017, 2018) were available for summer, with the exception of periphyton Chl *a*.

For the analysis of functional metrics (litter decomposition, nutrient uptake, and whole-stream metabolism), data from both natural regions were combined to obtain a sufficient sample size for threshold analyses. This was because fewer streams were studied and many of these were only studied for one year with a specific focus on the Grassland or Parkland natural regions between the years (Section 2.3). Litter decomposition and nutrient uptake metrics were only applied to the derivation of summer AEHI risk categories, and whole stream metabolism was included in the derivation of both spring and summer seasons. As litter decomposition indicators represented decay coefficients derived from mass loss throughout the summer, they were related to the seasonal average of the nutrient concentrations measured at each site. Nutrient uptake metrics were related to the average of all nutrient concentrations measured at each site up until the time of the solute tracer study. Stream metabolic rates were assessed against the average nutrient concentrations occurring within the season of the metabolic assessments.

3.3 Threshold Analysis

Thresholds in the response of indicators of aquatic ecosystem health to concentrations of TN and to TP were detected using classification and regression trees (CART), piecewise linear regression (PLR), and quantile piecewise linear regression (QPLR) statistical methods. Of the methods selected, each assumes a different relationship between stressor and response metrics (Figure 3-1). CART analysis (Breiman et al. 1984), as implemented here, models a step-change in the mean of the response metric, with the threshold value representing a point along the stressor gradient where the mean abruptly changes (Figure 3-1). CART has been used previously to locate thresholds in the response of aquatic ecosystem health to nutrients (Chambers et al. 2012; Evans-White et al. 2009; Miltner 2010; Qian et al. 2003; Smith et al. 2013). CART was implemented in R (R Development Core Team 2018) using the 'anova' method of the function 'rpart' in the package 'rpart' (Therneau and Atkinson 2018). PLR methods differ in that they model situations where the linear slope of the mean of the response metric changes along the stressor variable, with the threshold representing the point at which the slope changes (Figure 3-1). It has been used previously to model thresholds in the response of algal metrics to nutrient concentrations (Black et al. 2011), among other stressor-response applications (Brenden et al. 2008; Toms and Lesperance 2003). PLR was performed in SAS version 9.4 (SAS Institute 2012) using 'PROC NLIN' following the guidance of Ryan and Porth (2007). QPLR models a conditional quantile of a response metric distribution rather than the mean. QPLR can also portray a step relationship, as it has no requirement of continuity or linear connectivity at the threshold value. In our analysis, we used the 75th quantile to emulate "limiting factor" relationships in which nutrients control the *upper* limit of response metrics, but at any given site response metrics may be depressed owing to other limiting factors (Figure 3-1) (Brenden et al. 2008; Cade and Noon 2003). The 75th quantile was chosen to approximate an upper limit given that we had small datasets prone to outliers, negating the use of higher quantiles that are more affected by outliers (Brenden et al. 2008). QPLR was

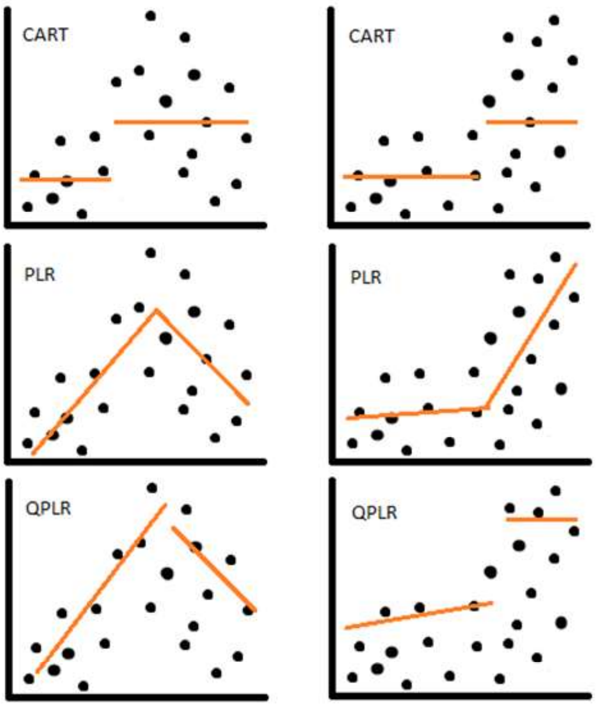

Figure 3-1. Illustration of stressor-response models fit by CART, PLR, and QPLR for two stressor response relationships.

performed in the software GUIDE (v. 29.6, Loh 2018). In all analyses, models were constrained to a single threshold with a minimum of four data points on either side of the threshold.

3.4 Threshold Model Evaluation

Prior to being included in the development of nutrient objectives, the identified thresholds were assessed according to three conditions aimed at ensuring that retained values represented likely ecological thresholds and were not spurious results. The first condition was that simpler (non-threshold) models were not a better fit to the data (Black et al. 2011; Qian and Cuffney 2012). This condition was evaluated by calculating Akaike's information criterion for small sample sizes (AICc, Sugiura 1978) for the threshold model, an equivalent simple linear model, and an intercept-only model. Threshold models were retained where the AICc value of the threshold model was at least two units lower than both the linear regression and intercept model (Burnham and Anderson 2002). The second condition required was that the relationship between the aquatic ecosystem metric and increasing concentrations of nutrients followed expectations that were hypothesized according to strong evidence in the literature. This condition ensured that aquatic ecosystem impairment resulted from the crossing of a threshold value (i.e., moving from low- to high-concentrations), such that unexpected contrary relationships (i.e,. improvements) were not included in measure of AEHI risk (Black et al. 2011). The third condition was that the estimated threshold value was reliable, as evidenced by examining bootstrap distributions (1000 replicates) of threshold values. Cases in which the most common threshold value estimated in the original dataset was not one of the most common values estimated in the bootstrap distribution, these estimates were interpreted as being spurious and were not included. Bootstrap threshold distributions for CART analyses were obtained using the boot function in the 'boot' package (Canty and Ripley 2017; Davison and Hinkley 1997) in R. The 'boot' package in R was also used to obtain QPLR threshold distributions by creating a shell function in R that ran QPLR analysis in GUIDE (Loh 2018). For PLR, threshold distributions were obtained using PROC NLIN's built-in option for obtaining bootstrap estimates of parameters. Finally, the threshold values were assessed to determine whether threshold values estimated by each of the three methodologies (CART, PLR, and QPLR) were redundant (i.e., detected the same aspect of the stressor-response relationship). The thresholds values were averaged where the methods were deemed redundant, but were kept separate where the methods detected different positions of the stressor-response relationship. This evaluation was performed to reduce bias associated with multiple inclusions of thresholds that reflected the same response along a stressor gradient.

3.4.1 Threshold Values Retained for Derivation of Nutrient Objectives

In total, 438 threshold analyses were performed; however, only 116 (26%) of these analyses met the necessary conditions for inclusion in the derivation process for the AEHI risk boundaries (Tables A-12). Of the 25 response metrics assessed, 20 were retained in at least one nutrient-region-season combination. The variables that most commonly met these requirements were those related to nutrient uptake, broad algae classifications (e.g., relative abundance of green algae), phytoplankton abundance, and the generic diatom index. Response metrics less often retained, if ever, were those related to stream metabolism (including litter decomposition and excepting net ecosystem productivity), variables related to algal community diversity, and areal uptake rates of phosphorus. The abundance of periphyton and relative abundance of pollution sensitive diatoms were retained with intermediate frequency. Of the three threshold methodologies used, thresholds derived by CART analysis most often met the conditions for inclusion (65 analyses) followed by QPLR (29) and PLR (22). This is likely because stressor-response relationships were generally weak with high residual variation, and the addition of parameters (penalized in AICc) for modelling slopes on either side of threshold values did not improve model fit sufficiently to warrant additional model complexity.

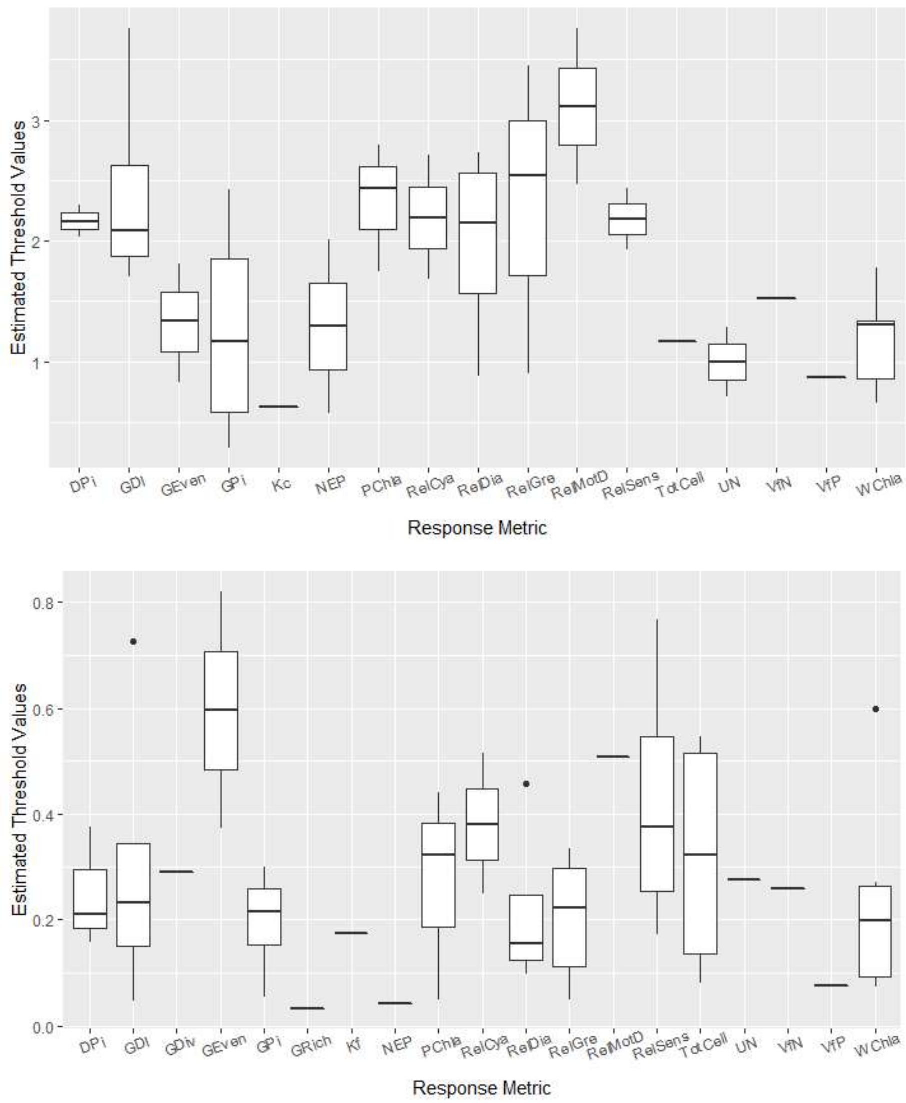

Figure 3-2. Threshold values retained for AEHI risk category derivation, by response metric. Boxplots represent the distribution of threshold values for total nitrogen (a) and total phosphorus (b) for given metrics observed across combinations of natural, season, and threshold detection methodology. See Table A-10 for explanation of metric codes.

Some observations can also be made regarding which parameters tended to have higher or lower threshold values (Figure 3-2). For TN, parameters related to periphyton community composition (except for the green algae pigment index and total cell count) tended to have higher threshold values than those related to ecosystem function, phytoplankton abundance, and generic evenness. The same pattern was not observed as strongly for TP. Thresholds related to ecosystem function and phytoplankton Chl *a* still tended to be on the lower end of the distribution of threshold values, but those related to algal communities were more variable.

3.5 Numeric Nutrient Objectives Suggested for Agricultural Watersheds in Alberta

3.5.1 Derivation of AEHI Risk Boundaries Based on Distribution and Weight of Threshold Responses

The distribution of retained threshold values from each combination of nutrient, natural region and season (Section 3.4.1) was used to develop the numeric boundaries of aquatic ecosystem health impairment (AEHI) risk categories. These numeric boundaries, in turn, are intended to be used flexibly to set nutrient objectives for a given watershed management scenario (Section 3.5.3). Boundaries between AEHI risk states were established using the 10^{th}, 50^{th}, and 80^{th} percentiles of weighted empirical distribution functions (WEDFs) of retained thresholds (Figure 3-3). Thresholds were weighted in order to give preferential influence to aquatic ecosystem metrics that are deemed more valuable. The weight of each metric was measured on a scale of 0 to 10 and was determined based on a survey of water resource management professionals in the province, with the scores for each metric being averaged across professionals into a proportional weight. Concentrations below the 10^{th} percentile were designated as representing low risk of AEHI owing to nutrient enrichment, those between the 10^{th} and 50^{th} percentile were deemed moderate risk. Concentrations between the 50^{th} and 80^{th} percentile were labelled as having high risk of AEHI, with those over the 80^{th} percentile were considered to have very high risk. Threshold values included in the derivation of AEHI risk categories, and their weights, for TN and TP are listed in Table A-13 and A-14, respectively.

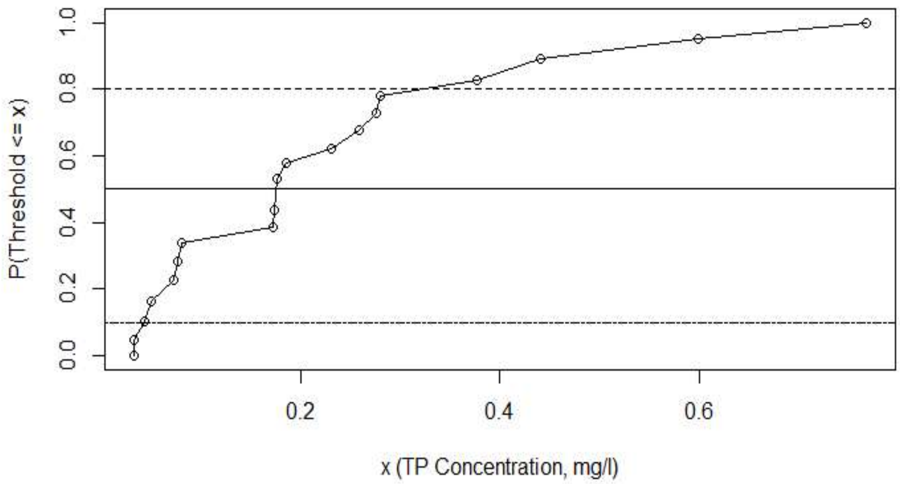

Figure 3-3. Exemplar of a weighted cumulative empirical distribution function of threshold values (for TP in the Grassland natural region, in summer). Intercepting lines represent the 10th (dash-dot), 50th (solid), and 80th (dash) percentiles of the weighted cumulative distribution function.

3.5.2 Summary of AEHI Risk Categories

Categories representing the risk of AEHI based on nutrient enrichment, and the nutrient concentrations defining these categories, are displayed in Table 3-1. For TP, the boundary between low and medium AEHI risk (0.04 – 0.07 mg/L TP) was consistent across seasons and natural regions and tended to align with TP values suggestive of the transition between trophic states in lakes and/or major rivers. The boundary between medium and high AEHI risk (the 50th percentiles of threshold WEDFs) and between the high and very high AEHI risk (80th percentile) were slightly higher for the Parkland NR than the Grassland NR, with seasonal differences being relatively negligible. For TN, less consistency was observed in AEHI risk category boundaries, with less noticeable patterns between NRs and/or seasons. At this point it is difficult to say whether differences in risk category boundaries between seasons and/or NRs are meaningful, as different response metrics were ultimately included in the development of AEHI risk categories for different combinations of natural region and season. In future refinements of AEHI risk categories, it will be considered whether data from seasons and/or natural regions should be merged, or whether the requirements for threshold retention outlined in Section 3.4 should be relaxed in certain cases in order to include a standard set of response metrics in risk category development.

Table 3-1. Aquatic ecosystem impairment risk categories for nutrient-natural region-season combinations. The maximum and minimum concentrations measured are reported for reference.

AEI Risk:	**Low**		**Medium**	**High**	**Very High**
	Lowest measured concentration (mg/L)	10^{th} Percentile Thresh WEDF (mg/L)	50^{th} Percentile Thresh WEDF (mg/L)	80^{th} Percentile Thresh WEDF (mg/L)	Highest measured concentration (mg/L)
TN					
Grassland					
Spring	<0.20	0.68	0.87	1.18	4.82
Summer	<0.20	0.47	1.44	2.47	12.4
Parkland					
Spring	<0.20	1.34	1.96	2.42	13.1
Summer	0.25	0.61	1.44	2.03	15.3
TP					
Grassland					
Spring	<0.02	0.05	0.13	0.25	1.78
Summer	<0.02	0.04	0.18	0.32	1.84
Parkland					
Spring	<0.02	0.07	0.29	0.41	1.57
Summer	<0.02	0.05	0.25	0.47	3.35

3.5.3 Suggested Use

The AEHI risk categories defined here are intended to have two primary applications: (i) the interpretation of aquatic ecosystem health owing to measured concentrations of nitrogen and phosphorus in surface waters; and (ii) the setting of nutrient management objectives in voluntary watershed management programs. There is no current intention for numeric concentrations bounding AEHI risk categories to be recognized as nutrient objectives as understood in regulatory frameworks, in which exceedances of triggers, limits, etc., instigate specific management responses. For a one-time assessment in a given stream, the concentrations of TN and TP averaged from a minimum of two samples collected within a given season must be performed in order to be consistent with the way in which data were aggregated in the threshold analysis (Section 3.2). When interpreting nutrient concentrations and the AEHI risk they represent, it is emphasized that the stressor-response relationships and thresholds used to derive risk categories were inherently variable, so nutrient concentrations corresponding with a given risk of AEHI should not be used to definitively conclude that a system is impaired, but as a general indication of impairment probability.

For setting management objectives, the intent here is that AEHI risk categories could be used flexibly by those undertaking watershed management activities. For example, if a stream were to have levels of nutrients that indicated a very high AEHI risk, watershed managers may choose a phased approach to progressively bring water quality into a high, then moderate risk category, and so on. If a stream were to be assessed to have a low AEHI risk, a “maintain or improve” approach may be taken to simply prevent nutrient concentrations from increasing in a manner that would amplify AEHI risk. If watershed managers were to develop a program that included management targets and triggers for a given stream, it is recommended that targets could be set in the middle of a desired risk category and that the upper boundary of that category could represent a trigger for management action. That being said, these are only a few examples of how watershed managers may choose to use the AEHI risk categories developed here (and in future refinements). A comprehensive guide outlining the recommended usage of the AEHI risk categories is outside of the scope of the current document, but will be developed more fully during the expansion of the current project to include the Boreal natural region, and before public release of the research.

3.5.4 Comparison to Nutrient Objectives Recommended for Similar Jurisdictions

For agricultural streams in Alberta, as part of the National Agri-Environmental Standards Initiative (NAESI), Chambers et al. (2012) used the average value of three percentile approaches (derived from historical nutrient concentration data in Alberta streams) to recommend provisional nutrient standards for Alberta of 0.98 mg/L TN and 0.11 mg/L TP. These concentrations are similar in magnitude to concentrations occurring in the medium AEHI risk categories developed here. Chambers et al. (2012) also recommended nutrient standards for the Manitoba Prairies and transition to Boreal Plains, which were inclusive of biological assessment data, of 0.102 mg/L TP and 0.39 mg/L TN. These values correspond to medium AEHI risk for TP, but low AEHI risk for TN. Numeric nutrient criteria have also been developed for some US States, although the only region comparable to Alberta with established criteria is the Northwestern Glaciated Plains of Montana. Here, criteria based on percentile and biological stressor-response approaches are 1.3 mg/L TN and 0.11 mg/L TP. Again, these values generally correspond to medium risk of AEHI as defined here. A subset of “level III” ecoregions within the Montana Northwestern Glaciated Plains – occurring in the foothills and upland areas – has lower criteria of 0.08 mg/L TP and 0.56 mg/L TN, which are more similar to the low AEHI risk categories developed here. Overall, the interpretation of ecological impairment as it relates to nutrient concentrations is in good agreement between these three cases, giving confidence that the use of the AEHI risk categories for assessment and/or watershed management purposes is consistent with the understanding of nutrient-related aquatic ecosystem impairment gained through other efforts in similar regions.

4. Assessing the Representativeness and Achievability of Nutrient Objectives

4.1 Modelling Purpose and Approaches

The nutrient objectives derived in this study, and reflected as numeric boundaries to AEHI risk categories, are intended to be generalized objectives that can be applied to all 3rd to 5th Strahler order streams draining agricultural watersheds in the Grassland and Parkland natural regions of Alberta. As described previously, the AEHI risk categories were derived from a representative set of streams along a gradient in agricultural land uses and runoff potential. The applicability of the AEHI risk categories to the management of specific streams should be assessed in order to determine whether the ranges of nutrient objective ranges are representative and/or achievable in the context of site-specific watershed management programs.

The representativeness of the AEHI risk categories to site-specific contexts was assessed by relating the generalized risk zones to site-specific thresholds of aquatic ecosystem impairment defined through the application of an in-stream water quality and ecological model to specific streams. Specifically, site-specific thresholds were derived for two watercourses, one in the Grassland region and one in the Parkland, using the QUAL2K in-stream water quality model. QUAL2K is a one-dimensional advection-dispersion-reaction (ADR) model that simulates eutrophication state variables, such as dissolved oxygen (DO), pH, benthic algae, phytoplankton, etc., in addition to water quality state variables on a longitudinal basis with daily kinetics (Chapra et al. 2010). Because QUAL2K is a one-dimensional steady-state model, the data requirements for calibration and derivation of site-specific thresholds are lower by comparison to dynamic models, and allow for the simulation of chronic conditions that are not affected by point source plumes, episodic inputs or significant temporal variation in flow or loadings (Flynn et al. 2015). QUAL2K has been used previously for setting site-specific nutrient objectives for rivers (Nielsen et al. 2012; Suplee et al. 2015). In order to facilitate calibration, the QUAL2Kw model was used in this study as it has an automated calibration function based upon genetic algorithms (Pelletier et al., 2006). The outcome of the analysis was intended to compare the congruence between numeric nutrient objectives generalized across natural regions with site-specific ecological responses to nutrient enrichment.

While generalized and/or site-specific nutrient objectives add value to water quality interpretations, their utility as management targets ideally would be verified through watershed-scale simulation of land management practices. The Comprehensive Economic and Environmental Optimization Tool (CEEOT) framework was selected to test the effectiveness of the modification of practices for reducing nutrient loading to streams from agricultural operations, and subsequently for reducing in-stream concentrations of nutrients. The CEEOT simulations provide estimates of monthly flow-weighted mean concentrations (FWMC) as a quotient between monthly loads and flow volumes. The CEEOT framework is an integrated modelling system that enables interfacing among three separate computer models; SWAT, APEX, and FEM (Farm Economic Model; Osei et al., 2000a). The SWAT and APEX models are integrated into the SWAT-APEX Interface Program (SWAPP) module that enables the evaluation of environmental effects of agricultural beneficial management practices (BMP). Based on prior experience in other watersheds, the SWAPP system is expected to yield improvements over the SWAT model alone, as spatially-explicit simulation of agricultural BMPs in the APEX model improves model prediction performance (Jedrych et al. 2014a, Saleh and Gallego 2007; Saleh et al. 2007; Osei et al. 2008a). The main advantages of CEEOT is that it can predict long-term environmental and economic effects under a variety of BMP scenarios at the field, farm, sub-watershed and watershed scales. The framework takes advantage of detailed simulation of field processes through APEX, and of the large watershed routing capabilities available in SWAT (Saleh and Gallego, 2007). Previous applications of CEEOT in Alberta (Jedrych et al. 2014a; Jedrych et al. 2014b) showed good performance in Alberta conditions. SWAT has also been used in the NAESI program to assess the feasibility of meeting ideal performance standards and/or to establish achievable performance standards (i.e., phased management targets) in degraded watersheds (Yang et al. 2011, 2012). In this study, the CEEOT model was used to assess the achievability of the generalized and site-specific nutrient objectives through modelling of watershed-scale implementation of agricultural BMPs in watersheds with high in-stream nutrient concentrations.

4.2 Watershed Descriptions

The QUAL2Kw and CEEOT models were applied to two watersheds with predominantly agricultural land uses that have been found to have elevated concentrations of nutrients as identified in historic studies: Indianfarm Creek, west of Lethbridge, and Threehills Creek, near Red Deer (Figure 4-1).

The Indianfarm Creek watershed has mean-annual precipitation of approximately 630 mm, with snowfall accounting for about 27% of the total (Environment Canada 2013). The average monthly minimum and maximum temperatures range from -10^{o}C in January to 23^{o}C in July. The estimated annual runoff depth (50% probability of exceedance) is 100 mm (Cole 2013). Soils belong to the Black Chernozemic soil-zone and have clay (5%), silty clay (33%), clay loam (37%), and loam (25%) textures. The soils are vulnerable to wind and water erosion, which has influenced land-use practices in the area, such as the use of zero tillage. The major land crop cover includes barley (40%), pasture (40%), hay (16%), and natural and residential areas (4%) (AARD 2014).

The Threehills Creek watershed is located near Red Deer, Alberta. Mean-annual precipitation is approximately 450 mm, and snowfall accounts for about 14% of the precipitation (Environment Canada 2013). The average monthly minimum and maximum temperatures range from -16^{o}C in January to 23^{o}C in July. The estimated annual runoff depth (50% probability of exceedance) is 10 mm (Cole 2013). Soils belong to the Black Chernozemic soil-zone and have loam (96%) and clay loam (4%) textures. The major land crop-cover includes barley (20%), canola (22%), spring wheat

(12%), pasture (26%), hay (6%), and natural and residential areas (14%) (Seitz Vermeer, N. et al. 2019).

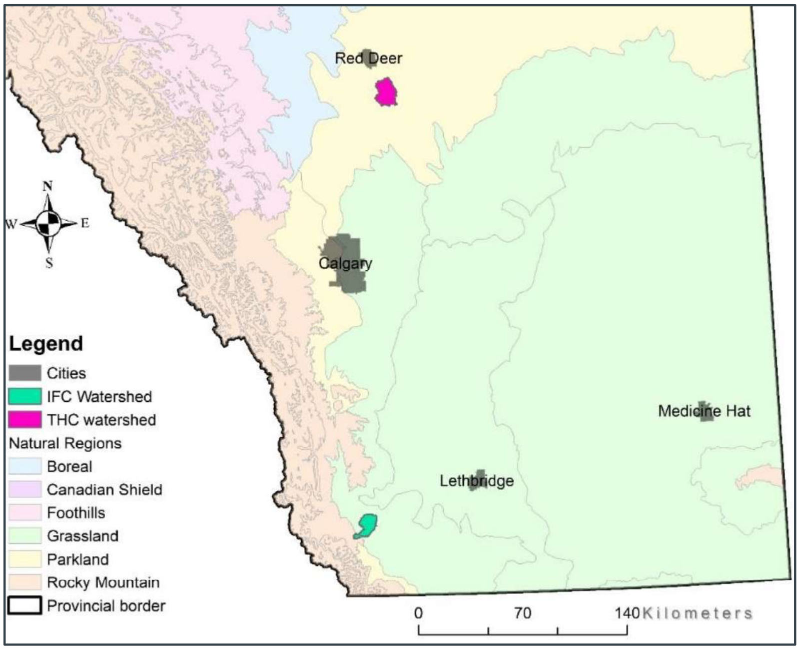

Figure 4-1. Location of Indianfarm Creek (IFC) and Threehills Creek (THC) watersheds in Alberta.

4.3 QUAL2Kw Model Application

At a minimum, setting site-specific nutrient objectives for a stream system using the QUAL2Kw platform requires the identification of a study reach, the delineation of a headwater and downstream reach to the study reach, and an identification of major inputs or abstractions to or from the study reach (Nielsen et al. 2012). However, in many cases, particularly along major rivers, multiple reaches are delineated for inclusion in the QUAL2Kw model, and separate objectives are identified for each reach or a number of individual reaches (Flynn et al., 2015). In this study, site-specific objectives were established for a specific reach segment corresponding to monitoring sites set-up as part of historic studies at these watersheds, and which correspond to the monitoring site used for collecting stressor-response data as part of this investigation. Additional reaches were delineated as the following: one reach downstream of the sites-specific objective (SSO) reach, one reach upstream of the SSO reach, a headwater reach further upstream, and at least one tributary that intersects the delineated reaches (Figure 4-2).

Defining a critical flow condition is also necessary for setting site-specific objectives. For evaluating nutrient thresholds, a design flow of 14-day low-flows every five years (14Q5) was chosen, as per recommendations by Suplee et al. (2015). Historic flow data collected from the SSO reach was used to calculate the 14Q5 flows for Indianfarm and Threehills Creeks.

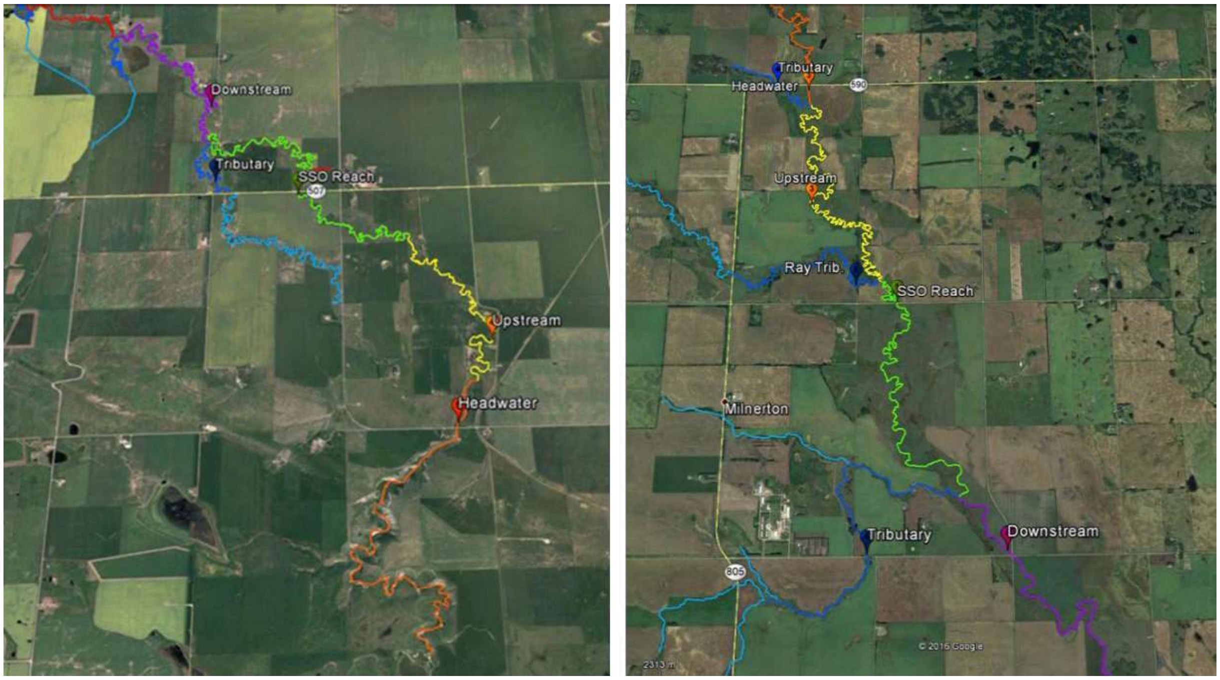

Figure 4-2. Reaches included in QUAL2K models of Indianfarm Creek (left) and Threehills Creek (right). Site-specific objectives (SSO) were established for the SSO reach (green). The SSO reach was chosen as the location of historic water quality and flow monitoring at these watersheds.

4.3.1 Model Calibration and Validation

Field monitoring data was collected from each of the study reaches in July 2017 and 2018 for model calibration and validation, respectively. July sampling events were targeted as they represent the period in which 14Q5 flow conditions are most likely to occur. Separate QUAL2Kw models were parameterized for each stream using field-collected and desktop data. Automated calibration was performed to minimize the root mean squared error (RMSE) between the predicted values of water column Chlorophyll a, which is a proxy for phytoplanktonic algae and an important state variable representing eutrophication processes, and the measured values. Chl a was chosen as it is driven by biological processes, and drives responses in pH and dissolved oxygen (DO), which were the state variables used to assess threshold responses to increasing nutrients. Validation data collected in 2018 were used to assess the performance of the calibrated model. The RMSE values from the validation dataset were used later to prescribe uncertainty around the threshold values of DO and pH – two key indicators of ecosystem impairment described in Section 4.3.2. In addition to RMSE, model calibration and validation were assessed using the Nash-Sutcliffe Efficiency (NSE) and percentage bias (PBIAS), which estimate the fit of the model outcomes relative to measured values and the average percentage of under- or over-estimation (- or + PBIAS, respectively) of the model outputs. Model fit statistics in the calibration and validation data are presented in Table A-15.

4.3.2 Site-Specific Objective Derivation Process using QUAL2Kw

Setting site-specific objectives using QUAL2K specifically, or more generally in-stream water quality models capable of simulating biological processes, is typically performed by simulating the response of state variables through gradual nutrient enrichment, and in which values indicative of ecosystem impairment are known. In following guidance by Suplee et al. (2015), site-specific objectives for each stream were established as the threshold concentration of TN or TP at which DO and pH exceed pre-established values that are protective of aquatic life: minimum daily dissolved oxygen of 5.0 mg/L and maximum daily pH of 9.0. In this study, following guidance from Nielsen et al. (2012) and Suplee et al. (2015), site-specific nutrient objectives were established independently for total nitrogen and phosphorus by setting one nutrient as non-limiting (i.e., at relatively high concentrations) and gradually increasing the concentration of the target nutrient in successive iterations until benchmark values of DO and/or pH were met. In this study, TN was held at 3000 µg/L for all simulations with increasing concentrations of TP, and TP was held at 300 µg/L for all simulations with increasing concentrations of TN.

4.3.3 Site-Specific Objectives for Indianfarm and Threehills Creeks

Simulations of increasing concentrations of TN and TP yielded expected increases in daily maximum pH (Figures 4-3 and 4-5) and decreases in daily minimum DO (Figures 4-4 and 4-6). Threshold responses to TN and TP for the daily maximum pH of 9.0 and daily minimum DO of 5.0 mg/L were observed in both streams. However, the relative concentrations of the threshold responses differed between the two streams, likely in response to differential stimulation of autotrophic (plants and algae) versus heterotrophic (decomposers) processes between the sites. Threshold response values, along with the model error (defined here as ± RMSE values) are presented in the respective figures.

The TN thresholds identified in the pH and DO simulations were relatively comparable at Indianfarm Creek. At Threehills Creek, threshold levels of daily minimum DO concentration occurred at lower concentrations of TN occurred at lower concentrations than threshold levels of daily maximum pH (Figures 4-3 and 4-4). At Indianfarm creek, threshold concentrations of TP were established for daily maximum pH at lower concentrations than for thresholds determined at daily maximum pH. The reverse was observed for Threehills Creek, where threshold responses for DO were at lower concentrations of TP than for threshold response of pH (Figures 4-5 and 4-6). These simulated results suggest that increasing concentrations of TN and TP at Indianfarm Creek likely augmented autotrophic production, or the growth of algae and plants, as increasing photosynthesis in the water column leads to elevated pH. Conversely, increases in TN and TP appear to stimulate heterotrophic (or organic matter respiration) over autotrophic activity at Threehills Creek, as evidenced by the relative sensitivity of DO response over increases in pH. These results provide evidence that aquatic ecosystem responses to nutrient increases are inherently site-specific. The simulation results also highlight that aquatic ecosystem impairment due to nutrient enrichment can manifest differently according to which ecosystem metric is being evaluated, underscoring why, ideally, multiple metrics would be evaluated and/or incorporated into any assessment of aquatic ecosystem impairment resulting from nutrient enrichment.

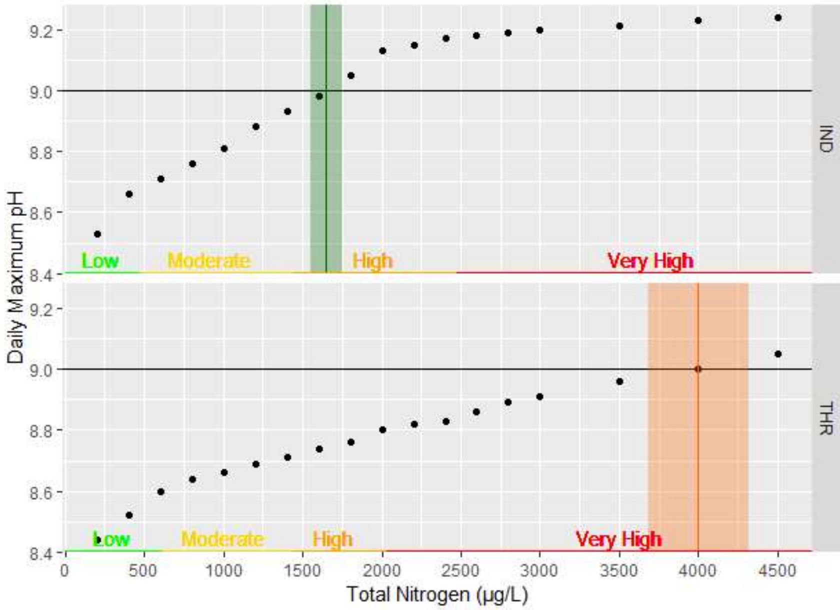

Figure 4-3. Threshold response (vertical line) and model error (coloured box) of simulated daily maximum pH as responding to increasing concentrations of total nitrogen at Indianfarm Creek (IND) and Threehills Creek (THR). Maximum permissible daily pH is 9.0, presented as a black line. Zones of aquatic ecosystem health impairment (AEHI) risk for total nitrogen are presented at the bottom.

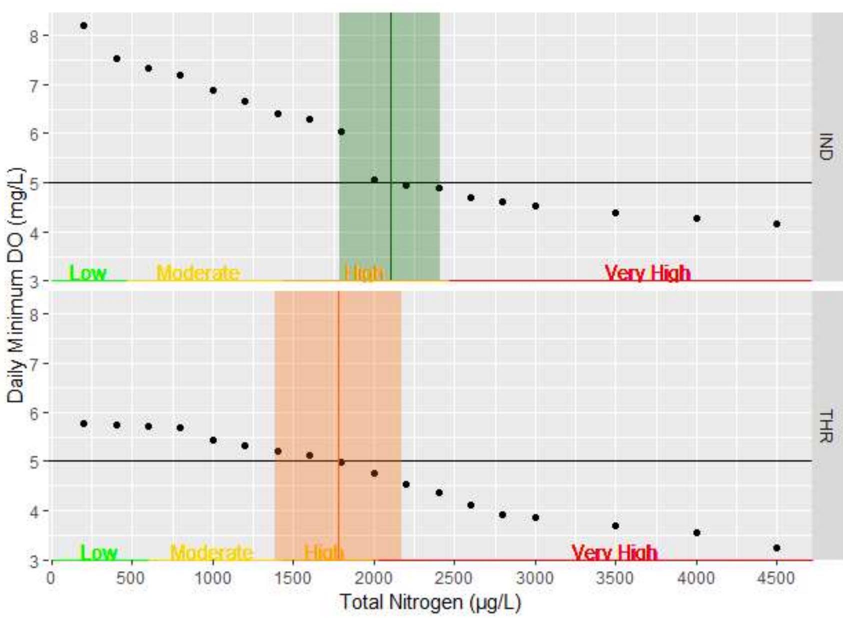

Figure 4-4. Threshold response (vertical line) and model error (coloured box) of simulated daily minimum dissolved oxygen (DO) as responding to increasing concentrations of total nitrogen at Indianfarm Creek (IND) and Threehills Creek (THR). Minimum permissible daily DO is 5.0, presented as a black line. Zones of aquatic ecosystem health impairment (AEHI) risk for total nitrogen are presented at the bottom.

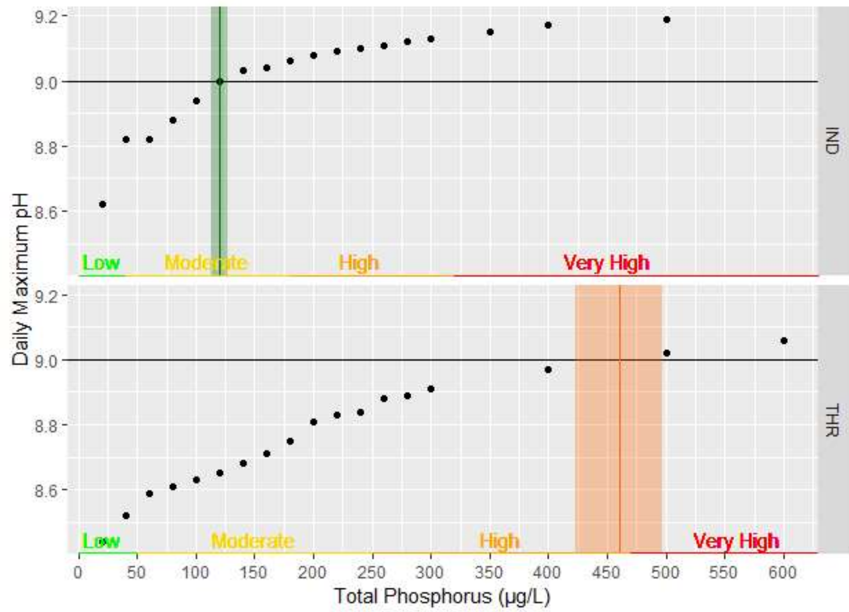

Figure 4-5. Threshold response (vertical line) and model error (coloured box) of simulated daily maximum pH as responding to increasing concentrations of total phosphorus at Indianfarm Creek (IND) and Threehills Creek (THR). Maximum permissible daily pH is 9.0, presented as a black line. Zones of aquatic ecosystem health impairment (AEHI) risk for total phosphorus are presented at the bottom.

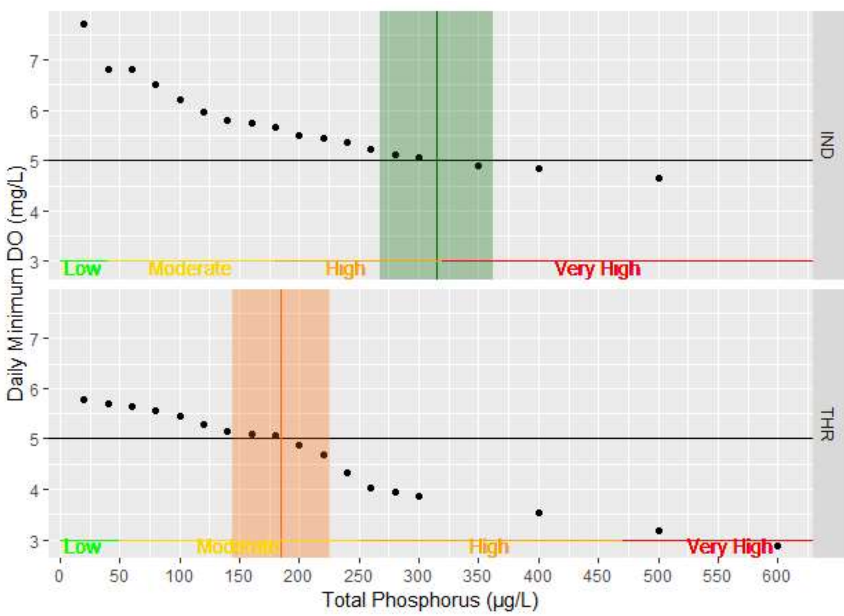

Figure 4-6. Threshold response (vertical line) and model error (coloured box) of simulated daily minimum dissolved oxygen (DO) as responding to increasing concentrations of total phosphorus at Indianfarm Creek (IND) and Threehills Creek (THR). Minimum permissible daily DO is 5.0, presented as a black line. Zones of aquatic ecosystem health impairment (AEHI) risk for total phosphorus are presented at the bottom.

4.3.4 Comparison of Site-Specific Objectives to Generalized Regional Objectives

The use of models for establishing site-specific nutrient objectives often leads to the reporting of a single-value threshold as a management target for nutrients. However, if multiple nutrient thresholds values are calculated from different ecosystem metrics, the model results can be used to define numeric boundaries of AEHI risk, akin to the generalized values reported in Section 3. In this study, by using the threshold values of TN and TP determined previously, three AEHI risk zones (low, moderate, and high) can be defined and bounded at positions along the concentration gradient of TN and TP. However, watershed managers may wish to use single values as the nutrient objective, in which the threshold response of the most sensitive parameter would be selected. The site-specific objectives determined for Indianfarm and Threehills creeks, along with their respective AEHI risk and SSO risk ratings, are presented in Table 4-1.

Table 4-1. Summary of site-specific objectives of TN and TP calculated from daily maximum pH and minimum DO simulations in Indianfarm Creek and Threehills Creek and their comparison to AEHI risk ratings for the Grassland and Parkland natural regions, respectively.

		Indianfarm Creek		Threehills Creek	
		TN (mg/L)	TP (mg/L)	TN (mg/L)	TP (mg/L)
Daily Max. pH	Threshold	1.65	0.120	4.00	0.460
	Range (±RMSE)	(1.55 - 1.75)	(0.113 - 0.127)	(3.68 - 4.32)	(0.423 - 0.497)
	AEHI Risk Rating*	H	M	VH	H
	SSO Risk Boundary**	L - M	L - M	M - H	M - H
Daily Min. DO	Threshold	2.10	0.315	1.78	0.185
	Range (±RMSE)	(1.79 - 2.42)	(0.268 - 0.362)	(1.39 - 2.17)	(0.144 - 0.226)
	AEHI Risk Rating*	H	H	H	M
	SSO Risk Boundary**	M - H	M - H	L - M	L - M

*L = low, M = moderate, H = High, and VH = very high.

**L-M: transition from low-to-moderate risk; M-H: transition from moderate-to-high risk

4.4 CEEOT Model Application

4.4.1 Input Data

In CEEOT, watershed configuration is determined by using topographic, land use, and soils data to delineate watersheds, sub-watersheds, and hydrologic response units (HRUs), the latter which represents areas within a watershed that have unique land use, soil, and slope characteristics and are the base element for watershed-scale simulations through SWAT. For Indianfarm Creek, topography was obtained using a 4 m digital elevation model (DEM), resulting in a delineated watershed with an overall area of 14,145 ha (Figure 4-7). In combining topographical data and soils and land use data (obtained as above) 154 different HRUs were defined (Jedrych, et al.

2014a). The topography of Threehills Creek watershed was derived from available 5 m DEM data, which delineated a 19,808 ha watershed (Figure 4-7); 474 different HRUs were defined.

Climate input files included daily precipitation, maximum and minimum temperatures, solar radiation, wind speed, and relative humidity. These data were derived from twenty-six years of daily climate data (1986 to 2011 for Indianfarm Creek and 1992 to 2017 for Threehills Creek), obtained for all townships in each watershed using an extrapolation procedure developed by ACIS (2014). The Agricultural Region of Alberta Soil Inventory Database (AGRASID) was used to define soil information in each watershed (Alberta Soil Information Centre 2001). The Indianfarm and Threehills Creek watersheds were characterized by 97 and 43 soil polygons and 12 and 5 soil series, respectively. In this analysis, each soil polygon was represented by its dominant soil series. Land management data comprised information on crop practices and rotations, fertilizer and manure management, and livestock numbers and locations. At Indianfarm Creek, watershed data were obtained from a previous study and were based on producer interviews and field observations conducted in 2008 (Olson and Kalischuk 2011). Land management practices for Threehills Creek were developed through consultation with local crop specialists, a 2013 field survey completed for a concurrent study (Seitz Vermeer et al. 2019), and Natural Resources Conservation Board livestock license data.

Water quantity and quality data from previous studies (Olson and Kalischuk, 2011; Seitz Vermeer et al. 2019) were used to calculate loads of TN, TP, and total suspended solids (TSS) for model calibration and validation (Section 4.4.2). Five (2007 to 2011) and four (2014 to 2017) years of flow data and TSS, TN, and TP concentration data were available for the Indianfarm and Threehills Creek watershed outlets, respectively. Flow data comprised a continuous record from flow monitoring stations, while water quality data represented intermittent samples taken using a flow-biased sampling regime, where water sample collection frequency was high during spring melt (twice per week) and tapered to a lower frequency (bi-weekly) during low-flow conditions. In total, 119 samples from Indianfarm Creek and 72 from Threehills Creek were used to inform the CEEOT model. To calculate loads of TSS, TN, and TP, measured concentrations from each sample were assigned to corresponding flow periods and loads were calculated for each period by multiplying the concentrations with the corresponding flow volumes. Monthly loads were then calculated by adding the discrete loads estimated within each month.

4.4.2 Calibration and validation procedure.

SWAPP calibration and validation procedures were conducted by running the model on a daily basis from 2002 to 2011 and from 2009 to 2017 for the Indianfarm Creek and Threehills Creek watersheds, respectively. Five year warm-up, or equilibration, periods were used to: (i) enable recalculation of default input values characterizing initial soil and land use conditions, (ii) reduce the effect of these initial parameter-values on model outputs, and (iii) achieve relatively stable prediction of the management practices being simulated. Accordingly, the first 5 years of model output were not included in model evaluations. In the Indianfarm Creek simulations, four years

(2007 to 2010) of model output were selected for model calibration and 1 year (2011) for model validation. In the Threehills Creek watershed, 3 years (2014 to 2016) were used for model calibration and 1 year (2017) was used for model validation. The calibration periods include a wide range of monthly flow rates that were comparable with the flow rates that were included in the validation period.

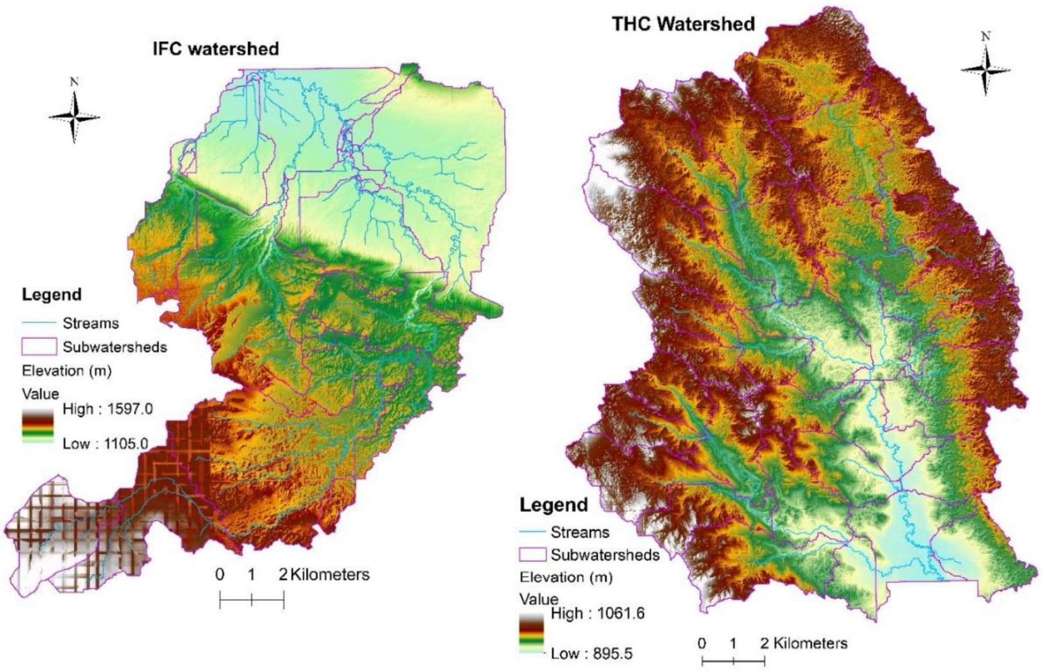

Figure 4-7. Distribution of sub-watershed borders, streams, and elevation ranges within Indianfarm Creek and Threehills Creek watersheds.

The selection of SWAPP initial parameters for calibration were based on the previous calibration and sensitivity analyses conducted by Jedrych et al. (2014a,b). During the calibration process, the initial values of selected parameters from SWAT and APEX were adjusted to produce simulated flow, TSS, TN, and TP values that were in predefined range of the measured data. The values of selected calibration parameters were allowed to vary while the other SWAT and APEX parameters not being calibrated were held constant. After a number of model iterations, the final parameter values were estimated for each watershed (Table A-17 and A-18); these calibrated values were used for model validation and BMP scenario evaluation.

During calibration, the mean monthly SWAPP simulated flow, TSS, TN and TP loads were compared to mean monthly measured values. The coefficient of determination (R^2), Nash and Sutcliffe efficiency (*NSE*), and percent bias (*PBIAS*) (Moriasi et al., 2007) were used to evaluate the predictive performance of the SWAPP module. In general, model simulations are considered “satisfactory” when $R^2 > 0.6$ and $NSE > 0.50$, and if *PBIAS* is ± 25% for streamflow, 55% for sediment, and 70% for nutrients (Moriasi, et al. 2007). R^2, *NSE*, and *PBIAS* values show satisfactory model prediction for all environmental indicators (Table A-19). Similarly, during the validation period, the majority of R^2, *NSE*, and *PBIAS* values indicated satisfactory model performance. The only exception was under prediction of flow in Indianfarm Creek where the calculated *PBIAS* of 36% exceeded the recommend ± 25% value range.

4.5 Watershed-Scale Simulations of Agricultural Beneficial Management Practices

4.5.1 Selection of BMPs and Scenarios

Seven agricultural BMP options were considered for use in the modelled watersheds using guidance from Jedrych et al. (2014a,b), and included commonly applied manure, livestock, and soil conservation practices (Table A-20). The BMPs were applied to selected fields (sub-areas) based primarily on the concentration of TP in runoff from those fields under baseline scenarios. The concentration criteria for application of a BMP varied by land-use type and watershed. For example, the modelled concentration of TP in runoff from pasture fields was higher in Indianfarm Creek than in Threehills Creek owing to a larger number of grazing cattle in this watershed. Subsequently, the concentration values used for selection of stream fencing BMP locations was higher in the Indianfarm Creek watershed (TP > 1.6 mg/L) than in Threehills Creek (TP > 0.8 mg/L) watershed (Table A-20). Ultimately, the percentage of watershed area selected for BMP applications was different in each watershed due to differences in farming practice between the areas. In Indianfarm Creek, filter strips, stream fencing, rotational grazing, and STP limit BMPs accounted for the largest area, whereas in Threehills Creek, only filter strips and STP limit BMPs had substantial application (Table A-20).

In total, six BMP simulation scenarios were created from the seven chosen BMPs (Table A-21). The baseline scenario represented status quo land management practices, wherein land management practices reflect the current land-uses occurring on the farm fields. Subsequent BMP simulation scenarios were progressively nested, where one additional BMP was added per successive scenario. However, for Scenario 3 both grassed waterways and wetland restoration were added as these BMPs accounted for a small area in both watersheds (Table A-21).

All scenario simulations were conducted for a 26-yr period using climate data from 1986 to 2011 for Indianfarm Creek and from 1992 to 2017 for Threehills Creek. The first year of simulation was considered to be a warm-up period and therefore only 25-yrs of model output were used for BMP scenario evaluation.

4.5.2 Discussion of scenario simulation results

The 25-yr SWAPP-predicted monthly runoff depth and TSS, TN, and TP load values were used to evaluate environmental effects of proposed scenarios by calculating the monthly mean, 50th, 25th, and 75th percentile values for all scenarios and comparing them to those derived from the baseline scenario (Tables A-21 to A-24). Scenario 6 simulation results are not included in this report because SWAPP prediction of soil test phosphorus (STP) limit effects were not consistent among sub-watersheds and need additional verification. That said, the STP results will be added to future project communications.

Simulation of the BMP scenarios demonstrated a progressive reduction in TN loads in both watersheds, with mean and median load reductions reaching 25% in both watersheds (Figure 4-8). However, substantial variation in the simulated efficacy of agricultural BMPs is noted, as over 25% of the simulated results lead to load increases in watersheds and a substantial portion of the simulated months exceeded 50% load reductions. The greatest reductions in TN loads were observed in Scenario 1 (+ filter strips) and Scenario 4 (+stream fencing) in Indianfarm Creek, and Scenario 2 (+dugouts) in Threehills Creek (Figure 4-8). The percentage change of monthly FWMC of TN was also reduced through simulation of the BMP scenarios (Figure 4-9). However, the mean and median monthly FWMC for TN in Indianfarm Creek were reduced by approximately 0 – 5% in for all scenarios, although the range of reductions in over the simulation period tended to get higher as additional BMPs were added. In Threehills Creek, monthly FWMC of TN were reduced by approximately 25% on average in Scenarios 2 through 5 (Figure 4-9).

With respect to TP loads, the simulated BMP scenarios were associated with approximately 10% load reduction on average in Indianfarm Creek and negligible reductions on average in Threehills Creek (Figure 4-10). In Indianfarm Creek, TP load reductions were most associated with Scenario 1 (+filter strips), and to a lesser extent Scenario 4 (+stream fencing). Despite reductions in TP load in Indianfarm Creek, the monthly FWMC of TP tended not to deviate from the baseline scenario (i.e., 0% change) on average, and percentage change tended to deviate in large part between 15 – 25% reductions or gains over the simulation period (Figure 4-11). The negligible reductions of TP loading in Threehills Creek predictably led to negligible changes to the monthly FMWC for TP over the simulation period (Figure 4-11). Thus, it appears that despite positive removal performance of TN occurring from the simulated BMP scenarios, little benefit to TP load or concentration reductions can be attributed to the BMP scenarios as currently defined.

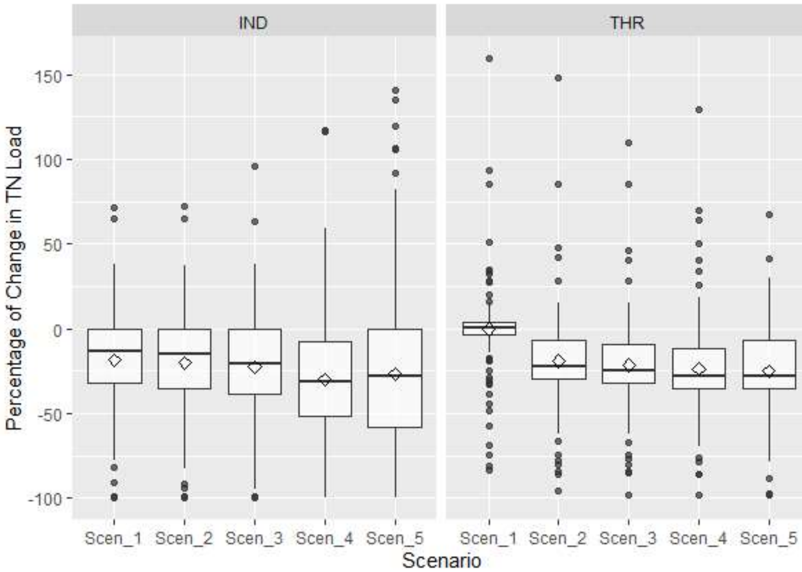

Figure 4-8. Percentage change in monthly total nitrogen (TN) load relative to the baseline scenario over a 25-year simulation period in five alternate land-management scenarios at Indianfarm (IND) and Threehills (THR) Creeks. Negative values indicate load reductions and positive values indicate load increases. Central diamonds reflect mean values.

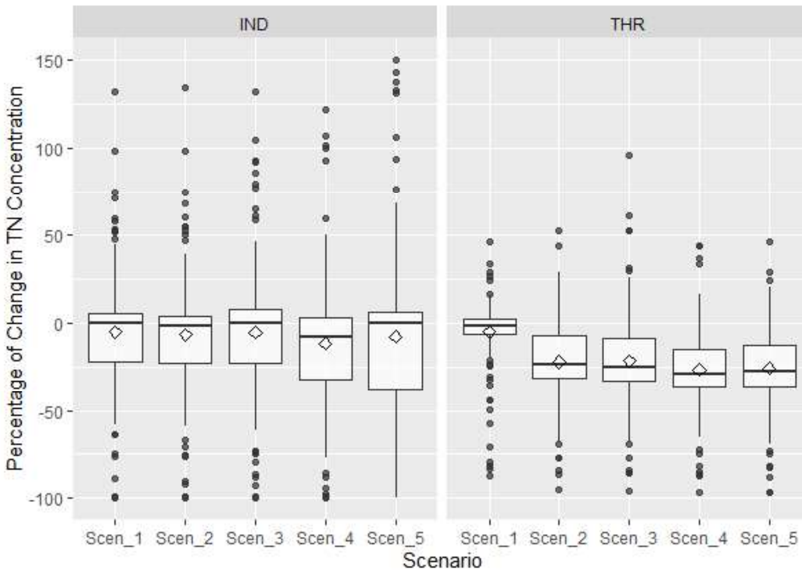

Figure 4-9. Percentage change in monthly flow-weighted mean concentrations of total nitrogen (TN) relative to the baseline scenario over a 25-year simulation period in five alternate land-management scenarios at Indianfarm (IND) and Threehills (THR) Creeks. Negative values indicate concentration reductions and positive values indicate concentration increases. Central diamonds reflect mean values.

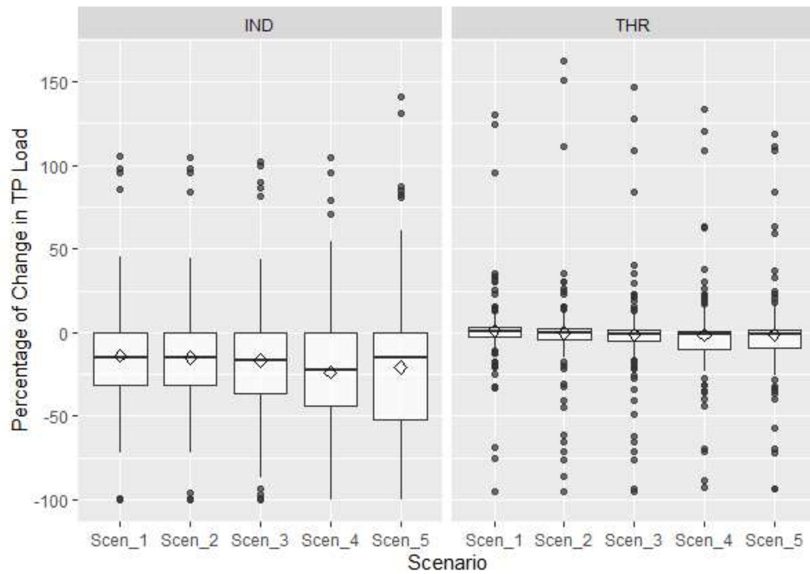

Figure 4-10. Percentage change in monthly total phosphorus (TP) load relative to the baseline scenario over a 25-year simulation period in five alternate land-management scenarios at Indianfarm (IND) and Threehills (THR) Creeks. Negative values indicate load reductions and positive values indicate load increases. Central diamonds reflect mean values.

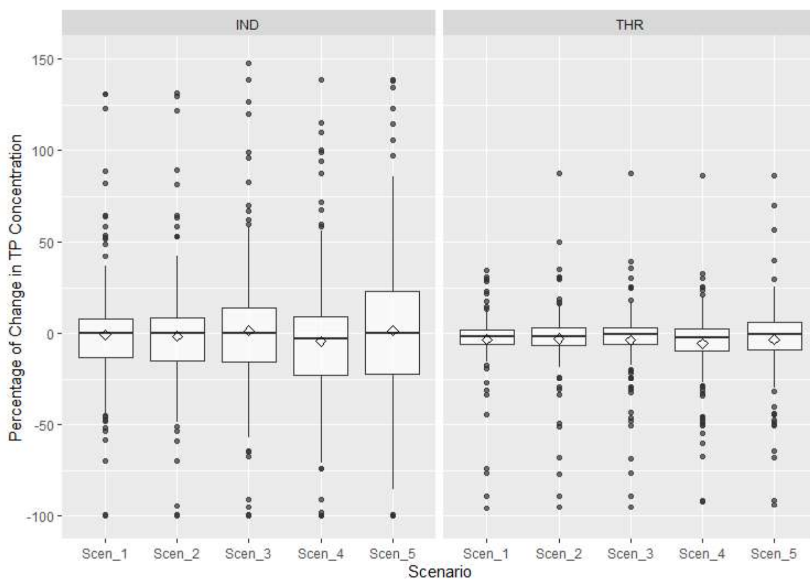

Figure 4-11. Percentage change in monthly flow-weighted mean concentrations of total phosphorus (TP) relative to the baseline scenario over a 25-year simulation period in five alternate land-management scenarios at Indianfarm (IND) and Threehills (THR) Creeks. Negative values indicate concentration reductions and positive values indicate concentration increases. Central diamonds reflect mean values.

4.6 Nutrient Objective Representativeness and Achievability

The monthly FWMC values for TN and TP estimated over the 25-year simulation period were used to evaluate the potential achievability of the proposed nutrients objectives through agricultural BMP implementation in both watersheds. Through the SWAPP modelling framework, FWMC values were estimated manually by dividing the predicted TN and TP monthly loads (kg) by corresponding monthly runoff volumes (m^3). The monthly FWMC of TN and TP were then compared to the AEHI risk categories and the SSO risk categories calculated individually for each watershed. The FWMC results for Indianfarm Creek were compared to the AEHI risk categories for the Grassland natural region, and the FWMC results for Threehills Creek were compared to the AEHI risk categories for the Parkland natural region. The proportion of simulated monthly FWMC values for TN and TP falling within the respective AEHI and SSO risk categories are presented in Figures 4-12 to 4-15. In addition to the simulated FWMC for each BMP scenario, the results from the water samples collected at Indianfarm and Threehills Creeks as part of this study were compared to the respective AEHI and SSO risk categories to compare the relative distribution of risks between the simulated monthly FWMC and measured concentrations.

Figure 4-12. Proportion of monthly flow-weighted mean concentrations of total nitrogen (TN) falling within the site-specific objective (SSO) risk categories derived for Indianfarm (IND) and Threehills (THR) creeks. Scenarios refer to the CEEOT simulation scenarios and the Study refers to the number of samples collected during the 2016 – 2018 field sampling matched against the SSO risk categories.

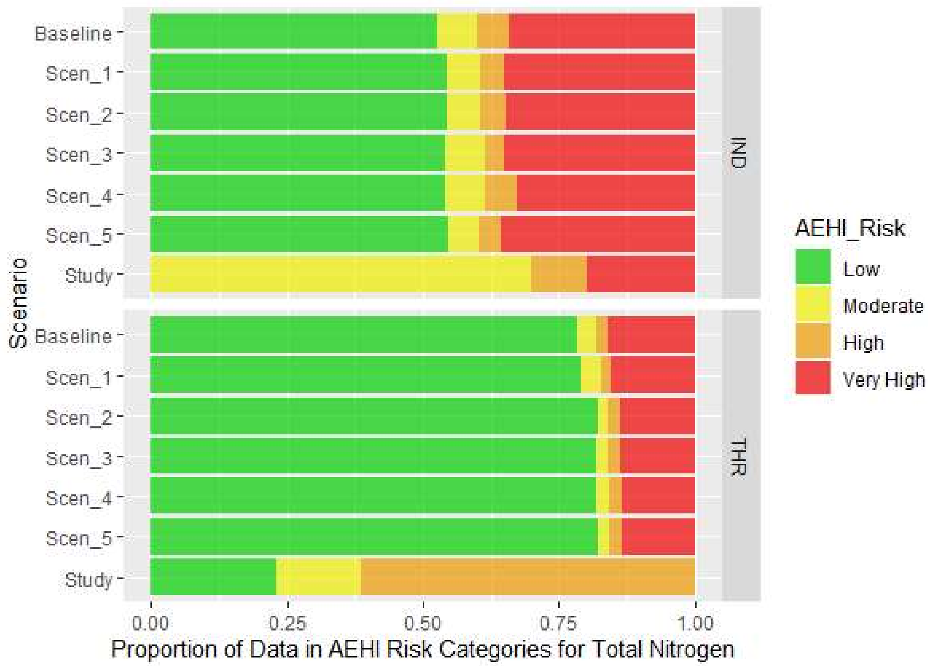

Figure 4-13. Proportion of monthly flow-weighted mean concentrations of total nitrogen (TN) falling within the aquatic ecosystem health impairment (AEHI) risk categories derived for Indianfarm (IND) and Threehills (THR) creeks. Scenarios refer to the CEEOT simulation scenarios and the Study refers to the number of samples collected during the 2016 – 2018 field sampling matched against the AEHI risk categories.

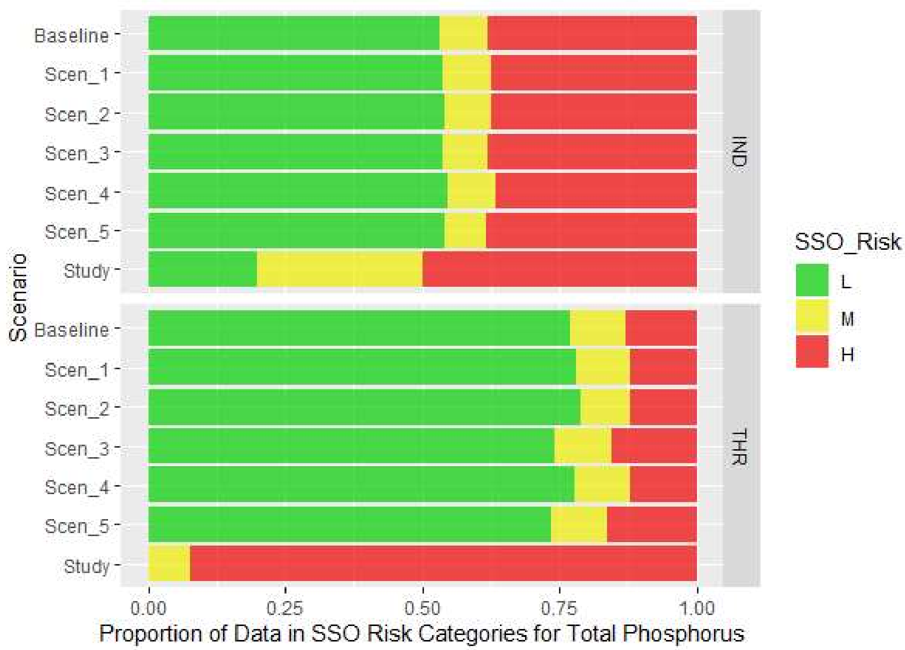

Figure 4-14. Proportion of monthly flow-weighted mean concentrations of total phosphorus (TP) falling within the site-specific objective (SSO) risk categories derived for Indianfarm (IND) and Threehills (THR) creeks. Scenarios refer to the CEEOT simulation scenarios and the Study refers to the number of samples collected during the 2016 – 2018 field sampling matched against the SSO risk categories.

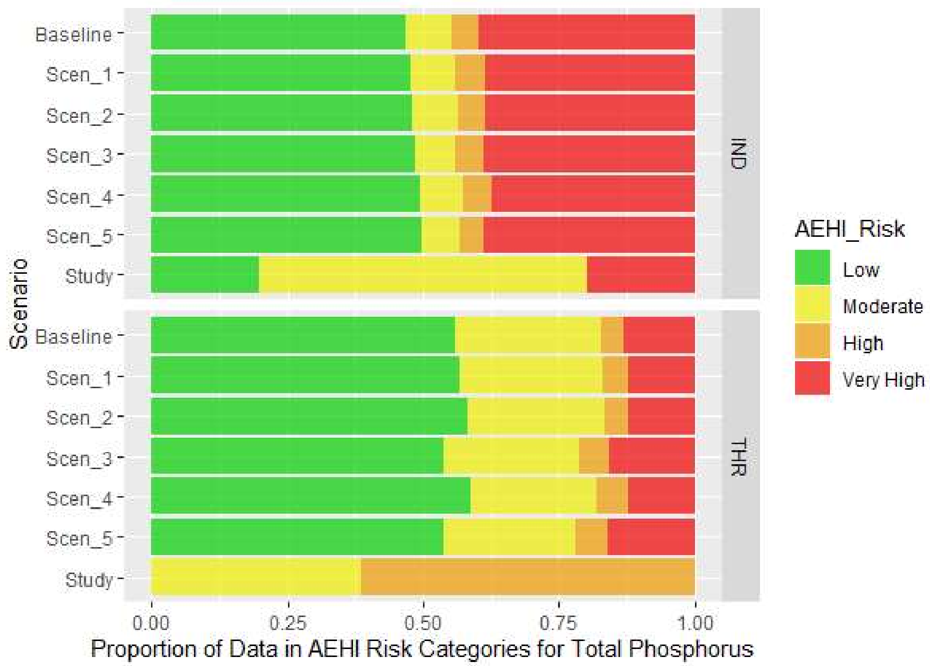

Figure 4-15. Proportion of monthly flow-weighted mean concentrations of total phosphorus (TP) falling within the aquatic ecosystem health impairment (AEHI) risk categories derived for Indianfarm (IND) and Threehills (THR) creeks. Scenarios refer to the CEEOT simulation scenarios and the Study refers to the number of samples collected during the 2016 – 2018 field sampling matched against the AEHI risk categories.

Evident from the results is that the monthly FWMC for TN appeared to reflect predominantly low risks of aquatic ecosystem impairment, where approximately 65% and 80% of simulated monthly FWMC values were in the low-risk zones for Indianfarm Creek and Threehills Creek, respectively (Figures 4-12 and 4-13). However, less than 25% of the measured concentrations of TN in this study samples were attributable to low-risk of aquatic ecosystem impairment when compared to either the AEHI or SSO objectives. Also evident is that the simulated BMP scenarios led to marginal decreases in the risk of aquatic ecosystem impairment, as illustrated by marginal increases to the proportion of FWMC in the low- or moderate-risk categories and lowering of the proportion of FWMC in the high-risk categories. Overall, the proportions of monthly FWMC of TN in low, moderate, and high (or very high) AEHI and SSO risk categories tended to be similar. However, the AEHI risk categories appeared to be slightly more conservative as illustrated by the slightly higher proportion of values in the very-high risk zone and lower proportion of values in the low risk zone in comparison to the high and low risk zones of the SSO risk categories, respectively.

The comparison of simulated monthly FWMC for TP to the AEHI and SSO risk categories suggest that the study watersheds have an elevated risk of aquatic ecosystem impairment owing to TP concentrations, where approximately 45% - 50% and 55% - 75% of the simulated monthly FWMC of TP for Indianfarm and Threehills Creeks, respectively, are characterized as low-risk (Figures 4-14 and 4-15). These modelling results also show that the simulated BMP scenarios are associated with marginal improvements to water quality as it relates to increasing the proportion of concentrations within low- or moderate-risk zones and reducing the proportion of concentrations within high-risk zones. Similar to the TN results, the measured values of TP from both watersheds tended to be rated to a greater extent in moderate, high, or very-high risk categories than the simulated monthly FWMC values. In addition, the AEHI risk categories also appeared to be more conservative than the SSO risk zones, for the simulated monthly FWMC values, but were less conservative than the SSO risk zones for the measured values of TP.

5.Conclusions and Future Work

5.1 Conclusions

The goal of the project was to define numeric nutrient concentration objectives for small streams in agricultural areas that are regionally-applicable to Alberta's agricultural landscape, and achievable through judicious land management practices. The stressor-response approach applied in this study resulted in numeric nutrient objectives that reflect varying degrees of aquatic ecosystem impairment owing to nutrient concentrations. The decision to produce separate nutrient objectives for disparate natural regions remains valid, owing to the degree of differences observed in the distribution of nutrient concentrations and aquatic ecosystem variables between the two natural regions studied here. Future work should include agricultural areas in the Boreal natural region, to gain complete coverage of Alberta's agricultural area.

The approach to integrate a number of ecosystem threshold estimates within the AEHI risk categories was a novel approach applied in this study. However, the approach is scientifically sound and has been validated by the stakeholder panel associated with this project. Adopting a range of concentrations that border categories of risk allows for flexible approaches to water and watershed management to be applied. For example, in areas where current conditions suggest very high risk of aquatic ecosystem impairment, progressive management can work toward meeting progressively lower numeric concentrations of nutrients to demonstrate gradual improvement in aquatic ecosystem health. Conversely, historic and cross-jurisdictional nutrient guidelines tend to be single values that reflect transitions from low impact-to-impacted states that may be difficult to achieve in highly impacted systems.

Based on the results of the simulations conducted in Section 4, it appears that the AEHI risk categories that are generalized to the Grassland and Parkland regions tend to be reasonable proxies for assessing aquatic ecosystem impairment in the absence of detailed site-specific investigations. The risk zones were reasonably comparable to site-specific objectives defined by simulating threshold responses of aquatic ecosystem parameters (DO and pH) to increasing nutrient concentrations, and the simulated and measured concentrations of nutrients tended to yield similar proportions of samples within low-to-high zones when comparing the AEHI and site-specific risk categories.

The utility of either the generalized AEHI risk boundaries or those derived on a site-specific basis as agricultural watershed management targets remains uncertain. The results of the watershed-scale modelling performed in this study yielded only marginal reductions to the risk of aquatic ecosystem impairment from agricultural BMPs. However, the modelling platform used in this study is thought to be better able to simulate load reductions from the landscape to water systems over in-stream concentrations of nutrients. In addition, the BMP scenarios defined here only covered less than 40% of the agricultural land-base. This coverage area was thought to be a realistic coverage and comparable to BMP uptake in watershed management programs, but may not be enough to sufficiently affect in-stream water quality such that nutrient objectives that are protective of aquatic ecosystem health can be realized. Additional modelling efforts that integrate a better in-stream water quality model, such as QUAL2K, and progressively simulate larger land areas are needed to better assess the theoretical achievability and appropriateness of the AEHI risk boundaries as nutrient management targets.

Going forward, the use of numeric objectives for small streams as defined in this study, could be readily applied to voluntary watershed management plans. The results defined here will provide information with which water quality monitoring data can be compared so as to gain a better appreciation of the status of aquatic ecosystem health within various watersheds. Further consultation with Government of Alberta departments and affected stakeholders would be required prior to the study results being included in statutory regional plans, given the legal and regulatory implications in this circumstance. Initial conversations with Alberta Environment and Parks concluded that such consultations should take place when final study results are available for all natural regions in the province, as per the first recommendation for future work presented in Section 5.2. Continued refinements to watershed-scale simulations models will take place in the meantime so as to gain a better appreciation of the achievability of the derived objectives and to address concerns by agricultural stakeholder groups prior to these values gaining statutory status.

5.2 Future Work

1) **Conduct an equivalent stressor-response study in the agricultural area of the Boreal natural region.**
   Substantial proportions of the Athabasca, Peace and North Saskatchewan river basins contain land being used for agricultural production that are contained within the Boreal natural region. The results of this study cannot be extrapolated to these regions. Performing the same scope of work within the Boreal agricultural area will allow for the complete assignment of nutrient objectives for small streams in agricultural regions across the province.

2) **Improve the achievability assessment of the nutrient objectives derived for Alberta's agricultural area.**
   Limitations in the modelling framework used precluded a confident assessment of the achievability of the derived nutrient objectives. Application of modelling platforms that are better suited for modelling in-stream nutrient concentrations, as well as an expanded definition of agricultural management scenarios are needed to better assess the theoretical achievability of the nutrient objectives derived here. Specific recommendations are to:
   a) Make use of a better in-stream water quality model for estimating nutrient concentrations that are inherently linked to biogeochemical processing, such as by replacing the QUAL2E module with a calibrated QUAL2K model in SWAT.

b) Expand upon the types of zone of coverage of agricultural BMPs included in the CEEOT framework.
c) Make use of more sophisticated modelling platforms that are capable of simulating the effect of anthropogenic impacts other than agriculture, such as residential inputs, linear features (e.g., roads).

3) **Engage with stakeholder groups to define an approach for uptake and use of the project results.**
Assumptions were made by the project team on how the values presented here as appropriate nutrient objectives for small streams would be applied and interpreted in water quality monitoring and watershed management programs. Engagement sessions with pertinent stakeholders, such as agricultural sustainability and watershed management groups, are needed to better understand how the results should be communicated to audiences such that the values derived will be used in management programs in Alberta.

4) **Use study results to create a provincial map of priority watersheds affected by surface water nutrient enrichment.**
No provincial-scale map of surface water quality risks for non-point source pollution currently exist, which challenges the ability of watershed and agricultural program managers to prioritize scarce funding resources to high-risk areas. A comparison of historic water quality monitoring data from small streams across the province, and potentially estimates of average nutrient concentrations for small watersheds across the province using geospatial modelling techniques (e.g., SPARROW model), can be compared to the generalized AEHI risk categories to map the risk of aquatic ecosystem impairment across the province. In this way, funding resources and management programs aimed at improving water quality can be prioritized to high risk areas.

6. Project Communications and Extension Plan

6.1 Communication Activities Completed

6.1.1 Stakeholder Committee Meetings

Completed

- Project initiation meeting was held in April 2016
- 2016/17 stakeholder update was held on April 13, 2017
- Technical meeting with AEP/UA was held on February 23, 2018
- 2017/18 stakeholder committee meeting was held on April 27, 2018
- Technical meeting with AEP/UA was held on February 21, 2019

Planned

- Stakeholder meeting to discuss project outcomes, project extension and next steps planned for Fall 2019

6.1.2 Publications

Planned

- Three Master's of Science thesis. Expected completion dates are: one in September 2019, and two in September 2020.
- At a minimum, five publications in academic journals are expected from the M.Sc. theses, one paper on the statistical derivation method, and one paper on the overall project outcomes

6.1.3 Extension

Completed

- An article was prepared for the Intensive Livestock Working Group (ILWG) newsletter
- A project webpage was activated on Alberta Agriculture and Forestry Ropin' the Web site
- Project-related presentations were delivered to the following organizations:
    - Bow River Phosphorus Management Plan Committee – March 2, 2017
    - Alberta Agriculture and Forestry, Land-Use Unit – July 12, 2017
    - Southeast Area Watershed Alliance (SEAWA) – November 2, 2017
    - Alberta Environment Limnology Congress – November 29, 2017
    - Intensive Livestock Working Group – December 12, 2019
    - Crop Sector Working Group – November 22, 2019

Planned

- Final detailed technical report will be uploaded onto Government of Alberta website
- Factsheets will be prepared for the project outcomes, detailing the objectives and their intended use

6.1.4 Presentations: Conferences and Workshops

Completed

- Canadian Aquatic Biomonitoring Information Network (CABIN) Science Forum 2017
  - March 1, 2017 – Edmonton, AB, Canada
- North Saskatchewan River Watershed Alliance Water Quality Forum
  - April 6, 2017 – Edmonton, AB, Canada
- Society for Canadian Limnologists, Canadian Conference for Fisheries Research
  - January 4, 2018 – Edmonton, AB, Canada
- Society for Freshwater Scientists, SFS Annual Meeting
  - May 19 - 23, 2019 – Salt Lake City, Utah, USA
- Manure Management Update
  - January 22, 2019 – Lethbridge, AB
- North Saskatchewan River Modelling Workshop
  - February 13, 2019 – Edmonton, AB

Planned

- Conference presentations are expected to occur more frequently as the M.Sc. candidates progress in their work, and communications around the key messages are approved by the GoA
- Presentations to WPACs and commodity groups will be performed to present project outcomes

7. Key Performance Metrics

Metric		Target	Actual	Planned
HQP Training	Undergraduate	6	5	
	Graduate (Master's)	2	3	
Presentations	Conferences	9	6	3
	Stakeholder groups	12	6	6
Publications	Theses	2	0	3
	Peer-reviewed	5	0	5
	Newsletters/Industry Communications	3	2	3
Policies Informed	Guidelines/Surface water quality management frameworks	1	0	1
	Agricultural funding programs	1	0	1
	Watershed management plans	5	0	5

8.References

Alberta Farm Sustainability Extension (AFSE) Working Group. 2008. Farm Sustainability Readiness Tool. Online; Accessed April 17, 2019. https://www.farmsustainability.ca/en

American Public Health Association (APHA). 1999. Part 10300 – Periphyton. In: Standard methods for the examination of water and wastewater, 20th Edition.

Appling, A.P., Hall, R.O., Arroita, M. and Yackulic, C.B. 2018. streamMetabolizer: Models for estimating aquatic photosynthesis and respiration. R package version 0.10.9. https://github.com/USGS-R/streamMetabolizer.

Bahls, L. L. 1993. Periphyton bioassessment for Montana streams. Water Quality Bureau, Department of Health and Environmental Sciences, Helena, MT.

Baker, M. E. and King, R. S. 2010. A new method for detecting and interpreting biodiversity and ecological community thresholds. Methods in Ecology and Evolution 1(1):25-37.

Basu, B. K. and Pick, F. R. 1996. Factors regulating phytoplankton and zooplankton biomass in temperate rivers. Limnology and Oceanography 41(7):1572-1577.

Bernot, M. J. and Dodds, W. K. 2005. Nitrogen retention, removal, and saturation in lotic ecosystems. Ecosystems 8(4):442-453.

Bernot, M. J., Tank, J. L., Royer, T. V. and David, M. B. 2006. Nutrient uptake in streams draining agricultural catchments of the midwestern United States. Freshwater Biology 51(3):499-509.

Bilkovic, D. M., Roggero, M., Hershner, C. H. and Havens, K. H. 2006. Influence of land use on macrobenthic communities in nearshore estuarine habitats. Estuaries and Coasts 29(6):1185-1195.

Black, R. W., Moran, P. W. and Frankforter, J. D. 2011. Response of algal metrics to nutrients and physical factors and identification of nutrient thresholds in agricultural streams. Environmental Monitoring and Assessment 175(1):397-417.

Borchardt, M. A. 1996. Nutrients. Pages 183-227 *in* R. J. Stevenson, M. L. Bothwell, R. L. Lowe, eds. Algal ecology: Freshwater benthic ecosystems. Academic Press, San Diego, CA.

Breiman, L., Freidman, J. H., Olshen, R. A. and Stone, C. J. 1984. Classification and regression trees. Chapman and Hall/Wadsworth, Belmont, CA.

Brenden, T. O., Wang, L. and Su, Z. 2008. Quantitative Identification of Disturbance Thresholds in Support of Aquatic Resource Management. Environmental Management 42(5):821-832.

Burnham, K. P. and Anderson, D. R. 2002. Model selection and multimodel inference: a practical information-theoretic approach, 2nd edn. Springer-Verlag, New York, NY.

Cade, B. S. and Noon, B. R. 2003. A gentle introduction to quantile regression for ecologists. Frontiers in Ecology and the Environment 1(8):412-420.

Canty, A. and Ripley, B. 2017. boot: Bootstrap R (S-Plus) Functions. R package version 1.3-20.

CCME. 2016. Guidance manual for developing nutrient guidelines for rivers and streams. Canadian Council of Ministers of the Environment, Canada.

Chambers, P. A., McGoldrick, D. J., Brua, R. B., Vis, C., Culp, J. M. and Benoy, G. A. 2012a. Development of environmental thresholds for nitrogen and phosphorus in streams. Journal of Environmental Quality 41(1):7-20.

Chambers, P.A., Culp, J.M., Roberts, E.S. and Bowerman, M. 2012b Development of environmental thresholds for streams in agricultural watersheds. Journal of Environmental Quality 41, 1-6.

Chételat, J., Pick, F. R., Morin, A. and Hamilton, P. B. 1999. Periphyton biomass and community composition in rivers of different nutrient status. Canadian Journal of Fisheries and Aquatic Sciences 56(4):560-569.

Claustre, H., Hooker, S. B., Van Heukelem, L., Berthon, J. F., Barlow, R., Ras, J., Sessions, H., Targa, C., Thomas, C. S., van der Linde, D. and others. 2004. An intercomparison of HPLC phytoplankton pigment methods using in situ samples: application to remote sensing and database activities. Marine Chemistry 85(1):41-61.

Covino, T., McGlynn, B. and McNamara, R. 2012. Land use/land cover and scale influences on in-stream nitrogen uptake kinetics. Journal of Geophysical Research: Biogeosciences 117(G2):G02006.

Davison, A. C. and Hinkley, D. V. 1997. Bootstrap methods and their applications. Cambridge University Press, Cambridge.

Dodds, W. K. 2006. Eutrophication and trophic state in rivers and streams. Limnology and Oceanography 51(1, part 2):671-680.

Dodds, W. K., Clements, W. H., Gido, K., Hilderbrand, R. H. and King, R. S. 2010. Thresholds, breakpoints, and nonlinearity in freshwaters as related to management. Journal of the North American Benthological Society 29(3):988-997.

Dodds, W. K. and Smith, V. H. 2016. Nitrogen, phosphorus, and eutrophication in streams. Inland Waters 6(2):155-164.

Dodds, W. K., Smith, V. H. and Lohman, K. 2002. Nitrogen and phosphorus relationships to benthic algal biomass in temperate streams. Canadian Journal of Fisheries and Aquatic Sciences 59(5):865-874.

Evans-White, M. A., Dodds, W. K., Huggins, D. G. and Baker, D. S. 2009. Thresholds in macroinvertebrate biodiversity and stoichiometry across water-quality gradients in Central Plains (USA) streams. Journal of the North American Benthological Society 28(4):855-868.

Gessner, M. O. and Chauvet, E. 2002. A case for using litter breakdown to assess functional stream integrity. Ecological Applications 12(2):498-510.

Government of Alberta (GoA) 2007. Understanding land use in Alberta. Online. Accessed April 17, 2018. 1http://www.assembly.ab.ca/lao/library/egovdocs/2007/alsrd/159513.pdf

Grattan, R. M. and Suberkropp, K. 2001. Effects of nutrient enrichment on yellow poplar leaf decomposition and fungal activity in streams. Journal of the North American Benthological Society 20(1):33-43.
Groendahl, S. and Fink, P. 2017. Consumer species richness and nutrients interact in determining producer diversity. Scientific reports 7:44869-44869.
Groffman, P., Baron, J., Blett, T., Gold, A., Goodman, I., Gunderson, L. H., Levinson, B. M., Palmer, M. A., Paerl, H. W., Peterson, G. D. and others. 2006. Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems 9:1-13.
Gulis, V., Ferreira, V. and Graça, M. A. S. 2006. Stimulation of leaf litter decomposition and associated fungi and invertebrates by moderate eutrophication: implications for stream assessment. Freshwater Biology 51(9):1655-1669.
Hering, D., Johnson, R. K., Kramm, S., Schmutz, S., Szoszkiewicz, K. and Verdonschot, P. F. M. 2006. Assessment of European streams with diatoms, macrophytes, macroinvertebrates and fish: a comparative metric-based analysis of organism response to stress. Freshwater Biology 51(9):1757-1785.
Kelly, M. G., Penny, C. J. and Whitton, B. A. 1995. Comparative performance of benthic diatom indices used to assess river water quality. Hydrobiologia 302:179-188.
Koenker, R. 2018. quantreg: Quantile Regression. R package version 5.36. https://CRAN.R-roject.org/package=quantreg.
Lange-Bertalot, H. 1979. Pollution tolerance of diatoms as a criterion for water quality estimation. Nova Hedwigia 64:285-305.
Lange, K., Liess, A., Piggott, J. J., Townsend, C. R. and Matthaei, C. D. 2011. Light, nutrients and grazing interact to determine stream diatom community composition and functional group structure. Freshwater Biology 56(2):264-278.
Lecerf, A., Usseglio-Polatera, P., Charcosset, J. Y., Lambrigot, D., Bracht, B. and Chauvet, E. 2006. Assessment of functional integrity of eutrophic streams using litter breakdown and benthic macroinvertebrates. Archiv für Hydrobiologie 165(1):105-126.
Liess, A., Lange, K., Schulz, F., Piggott, J. J., Matthaei, C. D. and Townsend, C. R. 2009. Light, nutrients and grazing interact to determine diatom species richness via changes to productivity, nutrient state and grazer activity. Journal of Ecology 97(2):326-336.
Loh, W.-Y. 2018. GUIDE User Manual (updated for version 29.6). University of Wisconsin, Madison. Available at: www.stat.wisc.edu/~loh/guide.html.
McCauley, E., Downing, J. A. and Watson, S. B. 1989. Sigmoid relationships between nutrients and chlorophyll among lakes. Canadian Journal of Fisheries and Aquatic Sciences 46(7):1171-1175.
Miltner, R. J. 2010. A method and rationale for deriving nutrient criteria for small rivers and streams in Ohio. Environ Manage 45(4):842-55.
Mulholland, P. J., Marzolf, E. R., Hendricks, S. P., Wilkerson, R. V. and Baybayan, A. K. 1995. Longitudinal patterns of nutrient cycling and periphyton characteristics in streams: A test of upstream-downstream linkage. Journal of the North American Benthological Society 14(3):357-370.
Niyogi, D. K., Simon, K. S. and Townsend, C. R. 2003. Breakdown of tussock grass in streams along a gradient of agricultural development in New Zealand. Freshwater Biology 48(9):1698-1708.
Odum, E. P., Finn, J. T. and Franz, E. H. 1979. Perturbation theory and the subsidy-stress gradient. BioScience 29:349-352.
Pan, Y. and Lowe, R. L. 1994. Independent and interactive effects of nutrients and grazers on benthic algal community structure. Hydrobiologia 291:201-209.
Passy, S. I. 2007. Diatom ecological guilds display distinct and predictable behavior along nutrient and disturbance gradients in running waters. Aquatic Botany 86(2):171-178.
Peterson, C. and Grimm, N. 1992. Temporal variation in enrichment effects during periphyton succession in a nitrogen-limited desert stream ecosystem. Journal of the North American Benthological Society 11(1):20-36.
Piggott, J. J., Niyogi, D. K., Townsend, C. R. and Matthaei, C. D. 2015. Multiple stressors and stream ecosystem functioning: climate warming and agricultural stressors interact to affect processing of organic matter. Journal of Applied Ecology 52(5):1126-1134.
Potapova, M. and Charles, D. F. 2007. Diatom metrics for monitoring eutrophication in rivers of the United States. Ecological Indicators 7(1):48-70.
Pringle, C. M. 1990. Nutrient spatial heterogeneity: Effects on community structure, physiognomy, and diversity of stream algae. Ecology 71(3):905-920.
Qian, S. S. 2014. Ecological threshold and environmental management: A note on statistical methods for detecting thresholds. Ecological Indicators 38:192-197.
Qian, S. S. and Cuffney, T. F. 2012. To threshold or not to threshold? That's the question. Ecological Indicators 15(1):1-9.
Qian, S. S., King, R. S. and Richardson, C. J. 2003. Two statistical methods for the detection of environmental thresholds. Ecological Modelling 166(1–2):87-97.
R Development Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
Richardson, C. J., King, R. S., Qian, S. S., Vaithiyanathan, P., Qualls, R. G. and Stow, C. A. 2007. Estimating ecological thresholds for phosphorus in the everglades. Environmental Science & Technology 41(23):8084-8091.
Rumeau, A. and Coste, M. 1988. Initiation à la systématique des diatomées d'eau douce pour l'utilisation pratique d'un indice diatomique générique. Bulletin Français de la Pêche et de la Pisciculture 309:1-69.

Ryan, S. E. and Porth, L. S. 2007. A tutorial on the piecewise regression approach applied to bedload transport data. General Technical Report RMRS-GTR-189. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado.
Sánchez, L. B., Lachos, V. H. and Labra, F. V. 2013. Likelihood based inference for quantile regression using the asymmetric laplace distribution. Journal of Statistical Computation and Simulation 81:1565–1578.
Sandin, L. and Solimini, A. G. 2009. Freshwater ecosystem structure–function relationships: from theory to application. Freshwater Biology 54(10):2017-2024.
SAS Institute. 2012. SAS® 9.4. SAS Institute, Cary, NC.
SAS Institute Inc. 2008. SAS/STAT® 9.2 User's Guide. SAS Institute Inc., Cary, NC.
Scheffer, M., Hosper, S. H., Meijer, M. L., Moss, B. and Jeppesen, E. 1993. Alternative equilibria in shallow lakes. Trends in Ecology & Evolution 8(8):275-279.
Smith, A. J., Thomas, R. L., Nolan, J. K., Velinsky, D. J., Klein, S. and Duffy, B. T. 2013. Regional nutrient thresholds in wadeable streams of New York State protective of aquatic life. Ecological Indicators 29(0):455-467.
Stelzer, R. S. and Lamberti, G. A. 2001. Effects of N:P ratio and total nutrient concentration on stream periphyton community structure, biomass, and elemental composition. Limnology and Oceanography 46(2):356-367.
Stevenson, J., Pan, Y. and Van Dam, H. 2010. Assessing environmental conditions in rivers and streams with diatoms. *in* J. Smol, E. Stoermer, eds. The diatoms: Applications for the environmental and earth sciences, 2nd Edition. Cambridge University Press, Cambridge.
Stevenson, R. J., Hill, B. H., Herlihy, A. T., Yuan, L. L. and Norton, S. B. 2008. Algae–P relationships, thresholds, and frequency distributions guide nutrient criterion development. Journal of the North American Benthological Society 27(3):783-799.
Strickland, J.D. and Parsons, T.R. 1972. A practical handbook of seawater analysis, 2nd Edition. Fisheries Research Board of Canada. Ottawa, ON.
Sugiura, N. 1978. Further analysis of the data by Akaike's information criterion and the finite corrections. Communications in Statistics, Theory and Methods A7:13–26.
Therneau, T. and Atkinson, B. 2018. rpart: Recursive Partitioning and Regression Trees. R package version 4.1-13. https://CRAN.R-project.org/package=rpart.
Tomal, J. H. and Ciborowski, J. J. H. 2017. Statistical methods for ecological breakpoints and prediction intervals. Submitted to Ecological Modelling.
Toms, J. D. and Lesperance, M. L. 2003. Piecewise regression: a tool for identifying ecological thresholds. Ecology 84(8):2034-2041.
USEPA. 2000. Nutrient Criteria Technical Guidance Manual: Rivers and Streams. EPA-822-B-00-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.
USEPA. 2010. Using stressor-response relationships to derive numeric nutrient criteria. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, Washington, D.C.
Van Nieuwenhuyse, E. E. and Jones, J. R. 1996. Phosphorus-chlorophyll relationship in temperate streams and its variation with stream catchment area. Canadian Journal of Fisheries and Aquatic Sciences 53:99-105.
Wagenhoff, A., Lange, K., Townsend, C. R. and Matthaei, C. D. 2013. Patterns of benthic algae and cyanobacteria along twin-stressor gradients of nutrients and fine sediment: a stream mesocosm experiment. Freshwater Biology 58(9):1849-1863.
Wagenhoff, A., Townsend, C. R., Phillips, N. and Matthaei, C. D. 2011. Subsidy-stress and multiple-stressor effects along gradients of deposited fine sediment and dissolved nutrients in a regional set of streams and rivers. Freshwater Biology 56(9):1916-1936.
Whitton, B. A. 1970. Biology of Cladophora in freshwaters. Water Research 4:457-476.
Yates, A. G., Brua, R. B., Culp, J. M. and Chambers, P. A. 2013. Multi-scaled drivers of rural prairie stream metabolism along human activity gradients. Freshwater Biology 58(4):675-689.
Young, R. G., Matthaei, C. D. and Townsend, C. R. 2008. Organic matter breakdown and ecosystem metabolism: functional indicators for assessing river ecosystem health. Journal of the North American Benthological Society 27(3):605-625.

Appendix A: Tables

Table A-1. Location and characteristics of the streams monitored in the Grassland natural region.

Stream Name	Stream Code	Latitude	Longitude	Major Basin	Watershed Characteristics									
					Area	Avg. Land Slope	Avg. Runoff	Cropland	Pasture	Grassland	Forest + Shrub	Residential	Wetland	Water
					(km2)	(%)	(mm)	%	%	%	%	%	%	%
Bullshead Creek	BUL	49.9605	-110.6058	South Sask.	505	4.52	15.2	12	4.4	74	6.6	1.2	1.3	0.2
Beaver Creek	BVR	49.6135	-113.7497	Oldman	264	13.97	27.5	11	1.8	54.1	31	1	0.7	0
Etzikom Tributary	ETZ	49.4687	-112.2259	Milk	678	2.75	37.3	57.8	3.6	28.3	1.7	2.3	2.3	2.9
Foothill Creek	FTH	49.4071	-113.7021	Oldman	136	3.77	286.4	23.4	14.8	43.6	13.6	2.1	1.3	0.7
Indianfarm Creek	IND	49.4655	-113.8433	Oldman	67	4.02	230.1	26.9	16.7	36.7	16.4	1.8	0.8	0.4
Kneehills Creek	KNE	51.4805	-113.1099	Red Deer	2239	1.68	21.3	79.2	5.2	7.7	2.4	2.9	1.8	0.6
Lower Sounding Creek	LWS	52.0849	-110.4798	Sounding Ck.	4937	1.29	3.7	24.3	13	47	4.3	1.3	8.6	1.2
Michichi Creek	MCH	51.5267	-112.5572	Red Deer	290	2.97	6.9	70	3.9	14.7	2.1	2.5	5.8	0.2
Monitor Creek	MON	51.9823	-110.5147	Sounding Ck.	1470	1.23	4.4	27.8	13.7	43.5	5.8	1.6	6.7	0.8
Mosquito Creek	MSQ	50.2517	-113.5538	Oldman	956	4.65	146.4	39.7	10	38.9	7.4	2.2	1.3	0.3
Matzhiwin Creek	MTZ	50.8418	-111.9314	Red Deer	2146	1.07	11.4	43.2	5.4	40.3	0.6	1.7	6	2.3
Onetree Creek	ONE	50.734	-111.6894	Red Deer	901	0.55	6.4	27.4	8	49.5	0.8	3.5	6.9	2.8
Pekisko Creek	PEK	50.4453	-114.2387	Bow	199	17.4	305.8	7.5	1.1	27	57.8	0.3	3.7	0.1
Seven Persons Creek	PER	49.9022	-110.8456	South Sask.	1791	2.13	17.7	42.2	2.7	49.7	1.6	1.4	1	0.8
Pinepound Creek	PIN	49.347	-113.0697	Oldman	222	3.6	65.2	48.6	5.2	40.6	2.2	1.2	1.6	0.5
Pothole Creek	POT	49.5241	-112.7986	Oldman	563	3.44	49.4	41.1	3	41.6	5.8	2.7	4.3	1.2
Rosebud Creek	RSB	51.3185	-113.3335	Red Deer	1711	1.55	24.2	75.1	7.6	7.3	1.9	3.4	3.4	1
Ross Creek	RSS	50.005	-110.585	South Sask.	914	3.14	14.9	25.9	7.2	54.9	7.3	2	1.6	0.4
Scandia Creek	SCN	50.2224	-112.0973	Bow	321	0.29	17.2	59	8.4	25.1	0.3	2.2	4.2	0.5
Shanks Creek	SHN	49.0581	-112.7377	Milk	75	2.27	50.2	48	7.1	41.5	0.7	1.8	0.5	0
Serviceberry Creek	SVC	51.2177	-113.1456	Red Deer	1219	1.03	29.2	68.3	8.7	9.8	1.1	4	5.6	2
Tongue Creek	TNG	50.6111	-113.9285	Bow	239	5.07	264.5	38.3	9.9	33	9.7	2.5	6.4	0.1
Unknown Creek	UNK	50.0186	-112.7538	Oldman	56	1.05	16.1	91.1	1.3	3.7	0.3	2.5	0.5	0.1
West Michichi Creek	WMC	51.5985	-112.701	Red Deer	597	1.67	13.7	55.9	2.7	22.8	7.6	2.3	6.9	1.4
Yellow Lake Tributary	YLT	49.7624	-111.497	South Sask.	114	0.45	14.5	93.5	0.3	1.8	0.1	3	0.4	0.2

Table A-2. Location and characteristics of the streams monitored in the Parkland natural region.

Stream Name	Stream Code	Latitude	Longitude	Major Basin	Watershed Characteristics									
					Area	Avg. Land Slope	Avg. Runoff	Cropland	Pasture	Grassland	Forest + Shrub	Residential	Wetland	Water
					(km2)	(%)	(mm)	%	%	%	%	%	%	%
Bigknife Creek	BGK	52.4918	-112.3639	Battle	344	0.64	38	31.6	25.5	18.6	8.1	2.5	11.5	1.8
Bigstone Creek	BGS	53.0292	-113.4138	Battle	748	0.7	32.5	58.8	23.8	1.6	8.1	3.5	2.7	1.3
Birch Creek	BIR	53.366	-111.2457	North Sask.	1287	1.65	18.8	59	16.3	7	6.6	2.7	3.7	3.5
Buffalo Creek	BUF	53.0075	-110.8685	Battle	972	1.6	24.4	62.8	21.4	4	6	2.3	2.3	0.7
Beaverhill Creek	BVH	53.7452	-112.6821	North Sask.	2486	0.73	27.7	44.7	12.8	7.2	15.8	3.8	6.7	8.8
Big Valley Creek	BVY	51.9403	-112.8386	Red Deer	487	2.27	23.1	21	13.7	30.8	22.8	2	7.4	2
Castor Creek	CST	52.3146	-111.7904	Battle	219	1.03	24.7	29.2	21.7	26.6	8.8	3.7	8.6	0.5
Dogpound Creek	DOG	51.7942	-114.3613	Red Deer	1004	3.51	68.2	56.2	10.1	13.8	15.6	2.6	1.4	0.1
Egg Creek	EGG	53.9732	-112.3814	North Sask.	689	0.58	18.4	68.8	14	3.6	3.8	3.3	4.9	1.2
Eagle Creek	EGL	51.9449	-114.4269	Red Deer	109	2.32	130	48.6	11.1	12.3	23.7	2.8	1.1	0.1
Grizzlybear Creek	GRZ	53.1066	-110.6441	Battle	568	3.25	25.5	52.9	19.4	10.4	8.7	2.4	3.6	1.8
Haynes Creek	HYN	52.332	-113.3635	Red Deer	172	1.83	18.5	82.3	7.9	2.4	3.2	2.3	1.7	0.2
Iron Creek	IRN	52.708	-111.3103	Battle	3564	1.06	18.3	71.1	10.3	6.4	4.6	2.8	3.2	1.2
Lasthill Creek	LST	52.3614	-114.4574	Red Deer	796	1.83	59.6	32.3	24.5	10.3	21.9	2.8	7.4	0.7
Lloyd Creek	LYD	52.7406	-114.1403	Red Deer	247	2.2	71.4	28.3	29.8	1.9	25.3	2.1	11.2	1.1
Maskwa Creek	MSK	52.9804	-113.5595	Battle	439	0.7	34.1	56.9	25.4	1.1	8	3.7	3	1.7
Meeting Creek	MTG	52.5601	-112.4996	Battle	740	1.75	31.4	62.9	11.1	7.9	7.1	2.7	3.6	4.2
Pipestone Creek	PIP	53.027	-113.2698	Battle	1107	0.88	30.7	53.4	27.2	1.9	9.4	3.9	2.8	1.3
Parlby Creek	PLB	52.5212	-113.3345	Red Deer	174	2.35	33.9	52.7	25	3.2	11.1	2.5	3.7	1.8
Paintearth Creek	PNT	52.3747	-111.9243	Battle	372	1.97	37	28.1	28.6	23.6	8.4	3.3	5.9	0.4
Queenie Creek	QUE	53.6216	-110.9648	North Sask.	353	2.3	34.8	63.4	15.8	6.5	6	2.5	4.5	1
Ray Creek	RAY	52.001	-113.5999	Red Deer	43	3.68	39.1	81.9	5.9	2	6.7	2.4	1	0
Ribstone Creek	RBS	52.8136	-110.1552	Battle	4326	1.96	10.5	30	16.4	26	17.2	1.8	6.3	1.1
Redwillow Creek	RDW	52.433	-112.4475	Battle	705	0.88	27.8	46.8	14.5	13.9	7.9	3.9	8	4.3
Sturgeon River	STU	53.8326	-113.2828	North Sask.	3390	1.35	41.4	45.7	19.2	0.6	13	12	4.4	4.5
Threehills Creek	THR	51.9973	-113.5687	Red Deer	198	2.92	31.6	69	11.3	5.3	9.1	2.4	1.8	0.9
Tindastoll Creek	TIN	52.1222	-114.1606	Red Deer	147	1.03	61.4	82.8	7.6	0.6	5.1	2.7	1	0.2
Vermilion River	VRM	53.4917	-110.3986	North Sask.	6913	1.52	22.8	62.3	15.6	7.2	6	3.2	3.3	1.9
Weiller Creek	WEI	52.9855	-113.2197	Battle	287	0.56	36.4	73.8	7.1	1.1	5.1	9.2	2.5	1

Table A-3. Analytical methods used for chemical and biological analyses.

Parameter	Abbreviation	Units	Method Reference	Description
Total Nitrogen (TN)	TN	mg/L	APHA 4500 N-Calculated	Total Nitrogen is calculated parameter from = Total Kjeldahl Nitrogen + [Nitrate and Nitrite (as N)]
Total Dissolved Nitrogen (TDN)	TDN	mg/L	APHA 4500 N-Calculated	Total Dissolved Nitrogen is calculated from = Dissolved Kjeldahl Nitrogen + [Nitrate and Nitrite (as N)].
Total Kjeldahl Nitrogen (TKN)	TKN	mg/L	APHA 4500-Nitrogen (Organic)	Total Kjeldahl Nitrogen is determined by sample digestion at 380°C with analysis using an automated colourimetric finish.
Dissolved Kjeldahl Nitrogen (DKN)	DKN	mg/L	APHA 4500-Nitrogen (Organic)	Same as TKN, but on sample filtered through a 0.45 micron membrane filter.
Ammonia in Water by Colour	NH_3-N	mg/L	APHA 4500 NH3-NITROGEN (AMMONIA)	Ammonia is determined using the automated phenate colourimetric method.
Nitrite and Nitrate in Water by IC (NO_2-N and NO_3-N)	NO_2-N, NO_3-N	mg/L	EPA 300.1 (mod)	Inorganic anions are analyzed by Ion Chromatography with conductivity and/or UV detection.
Total Phosphorus (TP)	TP	mg/L	APHA 4500-P PHOSPHORUS	Total Phosphorus is determined colourimetrically after persulphate digestion of the sample.
Total Dissolved Phosphorus (TDP)	TDP	mg/L	APHA 4500-P PHOSPHORUS	Total Dissolved Phosphorus is determined colourimetrically after persulphate digestion of a sample that has been lab or field filtered through a 0.45 micron membrane filter.
Dissolved. Orthophosphate (PO_4-P)	PO_4-P	mg/L	APHA 4500-P PHOSPHORUS	Dissolved Orthophosphate is determined colourimetrically on a sample that has been lab or field filtered through a 0.45 micron membrane filter.
Dissolved Organic Carbon (DOC)	DOC	mg/L	APHA 5310 B-WP	Filtered (0.45 um) sample is acidified and purged to remove inorganic carbon, then injected into a heated reaction chamber where organic carbon is oxidized to CO_2 which is then transported in the carrier gas stream and measured via a non-dispersive infrared analyzer.
Total Suspended Solids	TSS	mg/L	APHA 2540 D-Gravimetric	Gravimetric determination of solids in waters by filtration and drying filter at 104°C.
Chlorophyll *a*	Chl-*a*	µg/L	EPA 445.0 ACET	Chlorophyll *a* is determined by a 90% acetone extraction followed with analysis by fluorometry using the non-acidification procedure. This method is not subject to interferences from chlorophyll *b*.
Pheophytin *a*	Pheo-*a*	µg/L	EPA 445.0 ACET	Pheopigments present are determined collectively as Pheophytin *a* by a 90% (v/v) acetone extraction followed with analysis by fluorometry using the acidification procedure.

Table A-4. Hydrological properties of the streams monitored in the Grassland natural region measured from April through September in 2016 to 2018.

Stream Code	Strahler Stream Order	Width (m)			Depth (m)			Velocity (m/s)			Discharge (m^3/s)		
		Avg.	Min.	Max.	Avg.	Min.	Max.	Avg.	Min.	Max.	Avg.	Min.	Max.
BUL	5	6.9	5.7	7.5	0.38	0.2	0.51	0.297	0.05	0.627	0.814	0.085	1.914
BVR	5	2.7	0.4	4.9	0.14	0.03	0.3	0.139	0.002	0.63	0.049	0.001	0.196
ETZ	3	3	2.4	3.9	0.13	0.1	0.19	0.115	0.092	0.153	0.05	0.024	0.071
FTH	5	4.3	1	7	0.16	0.06	0.32	0.144	0	0.467	0.124	0	0.608
IND	5	2.3	1.8	3	0.33	0.12	0.58	0.066	0.016	0.333	0.102	0.013	0.518
KNE	6	9.9	9.2	10.6	0.48	0.27	0.89	0.074	0.006	0.342	0.444	0.021	3.213
LWS	6	3.2	2.5	4.2	0.16	0.07	0.34	0.031	0	0.07	0.014	0	0.041
MCH	6	3.9	3.6	4.5	0.37	0.28	0.66	0.094	0.003	0.41	0.199	0.004	1.075
MON	5	5.7	5.2	6.7	0.54	0.36	1.11	0.007	0.003	0.019	0.023	0.005	0.062
MSQ	6	9.4	8.4	10.5	0.57	0.42	0.73	0.161	0	0.272	0.865	0	1.522
MTZ	4	5.7	2.4	7.7	0.39	0.14	0.84	0.285	0.07	0.574	0.484	0.105	0.876
ONE	4	6.7	5.2	7.5	0.65	0.46	0.79	0.214	0.066	0.342	0.978	0.208	1.698
PEK	5	9.9	2.1	16	0.3	0.11	0.49	0.329	0.062	0.755	1.002	0.115	3.552
PER	6	3.8	2.9	4.4	0.48	0.2	0.77	0.161	0.027	0.318	0.321	0.021	0.567
PIN	2	4.7	3.4	6.3	0.68	0.5	0.98	0.013	0.011	0.063	0.041	0.038	0.208
POT	5	5.5	4.9	6.2	0.38	0.28	0.62	0.137	0.1	0.26	0.304	0.179	0.74
RSB	6	5.1	4.6	6	0.34	0.22	0.64	0.358	0.062	0.685	0.686	0.072	2.539
RSS	5	5.2	4.2	6.1	0.65	0.33	0.83	0.037	0.002	0.162	0.102	0.008	0.287
SCN	3	6	5.2	6.8	0.64	0.46	0.83	0.253	0.014	0.403	0.998	0.038	1.738
SHN	4	11.3	9.2	13	0.49	0.32	0.74	0.011	0	0.043	0.05	0.003	0.154
SVC	5	5.4	5.1	5.6	0.63	0.52	0.9	0.32	0.232	0.521	1.113	0.716	2.533
TNG	5	4.7	0.9	7.9	0.31	0.12	0.46	0.039	0.001	0.119	0.053	0.001	0.171
UNK	3	3.4	1.4	5.5	0.48	0.29	0.66	0.459	0.069	1.223	0.677	0.079	1.358
WMC	4	2.6	2	3	0.55	0.12	0.83	0.046	0.002	0.281	0.103	0.002	0.704
YLT	3	2.2	1.2	3.5	0.34	0.14	0.55	-0.005	0.015	0.009	0.007	0.003	0.015

Table A-5. Hydrological properties of the streams monitored in the Parkland natural region measured from April through September in 2016 to 2018.

Stream Code	Strahler Stream Order	Width (m)			Depth (m)			Velocity (m/s)			Discharge (m^3/s)		
		Avg.	Min.	Max.	Avg.	Min.	Max.	Avg.	Min.	Max.	Avg.	Min.	Max.
BGK	4	6	4.4	7.2	0.8	0.42	1.13	0.027	0.001	0.118	0.168	0.002	0.668
BGS	5	5.1	2.7	9.1	0.45	0.18	0.99	0.269	0.003	1.03	0.943	0.006	5.33
BIR	4	4.6	3.2	5.6	0.59	0.25	0.86	0.112	0.005	0.332	0.324	0.005	0.917
BUF	4	4.5	3.2	5.6	0.47	0.04	0.71	0.135	0.003	0.354	0.308	0.005	0.976
BVH	5	6	3.9	6.9	1.02	0.36	1.76	0.054	0.008	0.394	0.263	0.041	2.05
BVY	4	6.8	2.1	17.5	0.49	0.2	1.18	0.021	0.001	0.079	0.027	0.004	0.066
CST	5	5.7	4.1	8.9	0.73	0.37	1.47	0.03	0.006	0.213	0.164	0	1.25
DOG	5	8	7	10.2	0.42	0.23	0.69	0.273	0.077	0.565	0.888	0.254	2.714
EGG	5	7.4	3.4	12.3	0.69	0.36	1.29	0.185	0.017	0.468	1.209	0.153	5.74
EGL	4	4.4	3.2	6.6	0.45	0.22	0.83	0.03	0.022	0.249	0.046	0.038	0.279
GRZ	5	3.1	2.5	3.8	0.5	0.33	0.82	0.117	0.022	0.377	0.212	0.036	0.692
HYN	4	5.5	4.4	6.4	1.11	0.89	1.2	0.017	0.005	0.049	0.125	0.027	0.355
IRN	5	10	7.5	13.6	0.55	0.35	0.72	0.103	0.003	0.244	0.57	0.028	1.621
LST	6	12.4	9.8	15.1	0.7	0.14	1.57	0.134	0.014	0.441	1.865	0.095	8.86
LYD	5	8.8	7.6	15.1	0.39	0.07	1.52	0.126	0.001	0.462	0.935	0.003	8.86
MSK	4	6.3	4.1	9.4	0.59	0.37	0.83	0.086	0	0.233	0.57	0.001	1.88
MTG	4	5.7	2.6	8.7	0.6	0.12	1.05	0.094	0	0.385	0.435	0.001	2.47
PIP	5	7	4.4	10	0.6	0.36	1.11	0.163	0.002	0.597	1.076	0.004	6.59
PLB	3	4.6	3.1	7.2	0.7	0.5	1.01	0.064	0.007	0.192	0.258	0.02	1.17
PNT	5	4.7	3.7	5.5	0.99	0.77	1.28	0.037	0.013	0.314	0.264	0.028	2.19
QUE	5	7.9	5.9	15.8	0.74	0.34	1.33	0.04	0.003	0.168	0.476	0.007	3.53
RAY	5	1.2	0.9	1.6	0.42	0.17	0.72	0.082	0.003	0.162	0.04	0.003	0.133
RBS	5	6.4	5.5	7	0.44	0.28	0.65	0.421	0.115	0.683	1.325	0.203	3.086
RDW	5	4.7	2.8	5.9	0.48	0.29	0.83	0.077	0	0.306	0.274	0	1.23
STU	6	18	13.2	22.6	0.78	0.3	1.64	0.222	0.033	0.748	4.583	0.137	22.5
THR	5	2	1	3.1	0.66	0.39	0.96	0.047	0	0.196	0.07	0.001	0.348
TIN	4	2.5	1.8	3.2	0.25	0.1	0.42	0.118	0.122	0.462	0.107	0.045	0.584
VRM	6	16.7	3.8	26.4	0.65	0.26	1.52	0.297	0.624	0.963	8.033	1.36	38.5
WEI	4	3.3	2.4	4.2	0.75	0.46	1.03	0.098	0.001	0.457	0.297	0.002	1.78

Table A-6. Descriptive summary of water quality parameters measured for each stream measured in the Grassland natural region.

Site Code	n	pH		Dissolved Oxygen (mg/L)		Specific Conductance (µS/cm)		TN (mg/L)		NH_3-N (mg/L)		NO_3-N (mg/L)		TP (mg/L)		TDP (mg/L		PO4-P (mg/L)		DOC (mg/L)		Chl a (µg/L)	
		Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev
BUL	15	8.7	0.4	9.60	2.13	777	699	1.01	0.40	0.051	0.027	0.019	0.015	0.081	0.055	0.046	0.038	0.022	0.032	9.5	5.0	19.98	16.21
BVR	15	8.5	0.2	9.77	1.06	624	85	0.50	0.30	0.030	0.013	0.016	0.013	0.039	0.028	0.023	0.020	0.005	0.002	5.5	0.9	12.54	30.93
ETZ	6	8.2	1.2	10.63	1.47	1205	821	1.20	1.04	0.025	0.000	0.203	0.390	0.059	0.030	0.028	0.020	0.005	0.002	10.7	7.7	17.85	20.41
FTH	10	8.3	0.3	7.30	3.15	702	117	0.67	0.32	0.065	0.052	0.010	0.000	0.048	0.051	0.026	0.019	0.007	0.005	7.8	1.8	21.10	38.97
IND	10	8.4	0.1	9.13	4.93	654	121	2.51	3.60	0.086	0.111	0.041	0.053	0.231	0.341	0.034	0.028	0.016	0.025	11.0	3.9	127.20	291.75
KNE	18	8.6	0.4	10.62	2.05	1532	319	1.64	0.84	0.040	0.031	0.042	0.095	0.189	0.139	0.102	0.080	0.044	0.050	23.5	24.0	29.72	37.57
LWS	9	9.1	0.6	14.98	5.32	2802	356	2.88	0.46	0.035	0.026	0.021	0.013	1.289	0.401	1.253	0.422	1.081	0.377	32.2	4.7	30.74	33.90
MCH	10	8.6	0.1	9.28	2.13	1960	482	2.29	0.94	0.058	0.046	0.028	0.024	0.426	0.310	0.282	0.300	0.155	0.143	26.1	5.1	32.74	21.28
MON	9	8.9	0.5	9.57	2.78	3030	784	2.83	0.83	0.148	0.267	0.021	0.013	0.511	0.200	0.413	0.147	0.258	0.203	29.0	11.0	41.21	26.10
MSQ	15	8.6	0.2	9.94	0.84	547	259	0.48	0.31	0.038	0.046	0.013	0.012	0.050	0.028	0.024	0.016	0.008	0.009	5.6	3.2	7.58	6.76
MTZ	15	8.6	0.1	10.28	1.65	1049	364	0.78	0.31	0.028	0.009	0.020	0.018	0.157	0.062	0.124	0.052	0.098	0.059	9.2	3.4	7.94	12.19
ONE	15	8.7	0.4	9.69	1.42	818	308	0.71	0.23	0.044	0.041	0.020	0.024	0.221	0.083	0.184	0.097	0.136	0.089	7.3	1.3	13.41	14.00
PEK	10	7.5	2.6	9.68	0.92	362	55	0.21	0.15	0.049	0.075	0.019	0.013	0.015	0.016	0.014	0.013	0.004	0.001	3.5	1.1	1.59	1.58
PER	14	8.4	0.2	9.12	1.20	543	302	0.76	0.33	0.055	0.040	0.021	0.017	0.087	0.044	0.061	0.027	0.046	0.025	9.1	4.7	6.60	7.01
PIN	10	8.2	0.3	8.58	2.66	1993	374	1.06	0.46	0.025	0.000	0.016	0.013	0.130	0.124	0.076	0.115	0.050	0.103	13.1	5.6	10.09	15.17
POT	16	8.6	0.2	9.76	0.66	720	480	0.32	0.21	0.031	0.014	0.017	0.016	0.079	0.050	0.028	0.017	0.007	0.006	4.7	2.4	6.39	3.66
RSB	17	8.6	0.5	9.99	1.65	1210	300	1.32	0.44	0.044	0.020	0.016	0.015	0.203	0.112	0.125	0.086	0.084	0.081	21.2	20.9	27.24	21.28
RSS	11	8.2	0.2	8.19	2.76	1609	169	0.61	0.15	0.030	0.012	0.021	0.019	0.038	0.020	0.027	0.015	0.008	0.005	9.0	2.3	3.45	4.68
SCN	10	8.5	0.2	9.69	0.38	498	49	0.47	0.28	0.032	0.022	0.014	0.013	0.061	0.048	0.095	0.179	0.026	0.028	5.5	2.7	5.35	6.97
SHN	10	8.7	0.4	7.73	3.71	765	143	4.03	0.91	0.131	0.102	1.985	1.233	0.121	0.072	0.049	0.046	0.008	0.005	12.3	5.8	32.60	38.42
SVC	10	8.5	0.2	8.60	1.21	1142	721	1.19	0.55	0.072	0.072	0.063	0.068	0.235	0.090	0.080	0.028	0.056	0.030	10.7	6.4	28.32	20.69
TNG	10	8.5	0.4	8.71	2.08	780	121	0.87	0.35	0.042	0.031	0.016	0.012	0.067	0.036	0.040	0.028	0.017	0.022	11.3	2.9	8.81	9.88
UNK	15	8.5	0.1	9.18	1.20	468	328	0.55	0.59	0.039	0.033	0.015	0.012	0.229	0.491	0.153	0.443	0.140	0.457	5.7	4.2	12.01	21.49
WMC	10	8.4	0.2	9.52	4.86	1171	394	3.39	3.03	0.200	0.454	0.817	2.524	0.536	0.352	0.333	0.276	0.230	0.236	24.7	4.7	58.76	57.04
YLT	6	8.4	0.3	9.23	1.35	1763	474	2.07	1.48	0.358	0.651	0.040	0.042	0.375	0.372	0.270	0.293	0.239	0.284	15.6	4.2	12.21	6.33

Table A-7. Descriptive summary of water quality parameters measured for each stream measured in the Parkland natural region.

Site Code	n	pH		Dissolved Oxygen (mg/L)		Specific Conductance (µS/cm)		TN (mg/L)		NH_3-N (mg/L)		NO_3-N (mg/L)		TP (mg/L)		TDP (mg/L		PO4-P (mg/L)		DOC (mg/L)		Chl a (µg/L)	
		Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev	Avg	Std.Dev
BGK	10	8.0	0.6	8.00	2.67	633	135	3.03	2.49	0.200	0.348	0.583	1.798	0.380	0.240	0.283	0.207	0.204	0.183	24.4	4.1	24.08	17.15
BGS	10	8.0	0.2	9.01	1.76	672	215	1.83	0.90	0.116	0.111	0.092	0.121	0.275	0.088	0.217	0.092	0.166	0.104	19.1	5.2	16.46	14.31
BIR	10	7.9	0.2	5.95	3.77	935	110	1.51	0.27	0.065	0.058	0.035	0.031	0.161	0.094	0.118	0.064	0.087	0.051	26.4	4.3	5.09	2.90
BUF	16	8.3	0.1	8.35	2.27	1353	242	1.52	0.44	0.057	0.092	0.014	0.014	0.177	0.061	0.156	0.060	0.112	0.053	20.6	2.9	5.08	2.89
BVH	16	7.6	0.3	6.75	3.20	471	139	2.33	0.58	0.061	0.078	0.026	0.032	0.450	0.275	0.337	0.241	0.231	0.143	28.5	5.7	21.82	14.51
BVY	10	8.3	0.3	9.39	3.22	1477	241	1.24	0.71	0.031	0.013	0.015	0.013	0.234	0.212	0.184	0.182	0.150	0.170	19.1	9.9	6.81	6.10
CST	10	7.8	0.3	6.71	2.63	878	221	2.24	0.72	0.034	0.023	0.014	0.012	0.371	0.188	0.113	0.046	0.070	0.031	28.2	5.9	38.16	19.28
DOG	20	8.7	0.3	11.32	1.65	504	113	0.41	0.16	0.035	0.038	0.016	0.021	0.026	0.018	0.023	0.016	0.010	0.015	7.1	1.8	2.40	2.21
EGG	10	8.1	0.5	7.62	2.66	774	148	2.71	0.53	0.122	0.117	0.032	0.045	0.322	0.165	0.274	0.164	0.216	0.149	35.5	8.4	9.85	10.14
EGL	16	8.2	0.1	9.96	1.76	653	65	0.54	0.22	0.065	0.128	0.011	0.004	0.031	0.017	0.028	0.035	0.006	0.004	10.0	4.1	5.06	3.23
GRZ	17	8.6	0.2	8.44	1.69	2110	544	3.40	0.94	0.047	0.033	0.014	0.005	0.723	0.366	0.607	0.336	0.488	0.301	40.7	8.8	19.43	10.56
HYN	10	8.1	0.4	4.89	3.96	1347	274	2.88	0.97	0.092	0.083	0.010	0.000	0.441	0.256	0.363	0.156	0.303	0.126	39.1	11.9	54.19	102.12
IRN	10	8.5	0.2	10.13	2.25	1203	235	1.99	0.44	0.038	0.029	0.014	0.012	0.426	0.168	0.342	0.151	0.285	0.142	23.4	5.1	20.72	17.33
LST	16	8.2	0.2	10.26	3.85	540	114	1.32	1.14	0.065	0.095	0.024	0.020	0.079	0.032	0.043	0.020	0.024	0.014	14.9	7.0	7.30	4.37
LYD	15	8.2	0.4	9.78	2.56	434	81	1.39	0.50	0.038	0.025	0.022	0.022	0.167	0.086	0.133	0.066	0.102	0.049	19.4	5.9	12.97	14.89
MSK	10	7.8	0.5	6.83	2.96	493	98	2.46	0.89	0.221	0.276	0.070	0.132	0.326	0.178	0.252	0.187	0.201	0.136	24.0	3.5	25.35	31.61
MTG	10	8.2	0.3	7.64	3.68	878	148	2.69	0.47	0.101	0.142	0.022	0.025	0.525	0.493	0.259	0.243	0.204	0.236	30.5	4.9	27.48	23.32
PIP	16	7.9	0.2	7.78	2.34	692	171	1.77	0.75	0.108	0.120	0.073	0.092	0.264	0.071	0.181	0.089	0.119	0.061	18.8	3.7	16.83	10.48
PLB	10	8.0	0.2	8.83	2.67	723	173	1.68	0.57	0.083	0.051	0.442	0.296	0.205	0.080	0.126	0.063	0.266	0.539	13.4	4.3	22.10	18.56
PNT	10	7.7	0.3	6.65	2.69	737	334	2.02	0.60	0.080	0.090	0.035	0.049	0.309	0.217	0.054	0.028	0.025	0.026	23.7	7.0	40.14	28.93
QUE	10	8.0	0.3	6.48	2.39	1337	352	2.97	0.72	0.096	0.085	0.018	0.016	0.530	0.189	0.424	0.174	0.344	0.150	33.9	13.9	17.02	15.92
RAY	13	8.1	0.2	9.07	3.17	994	78	1.22	0.50	0.040	0.018	0.013	0.008	0.265	0.132	0.239	0.113	0.188	0.094	17.8	5.7	10.16	12.76
RBS	10	8.4	0.1	8.87	0.81	1040	195	1.95	0.38	0.028	0.010	0.010	0.000	0.140	0.075	0.086	0.040	0.055	0.038	29.4	6.6	4.78	4.47
RDW	10	8.5	0.6	9.04	4.35	1526	406	2.52	0.64	0.069	0.068	0.014	0.008	1.209	0.959	1.025	0.798	0.905	0.651	31.9	6.8	12.90	14.71
STU	20	8.1	0.4	9.38	1.71	685	165	1.43	0.58	0.093	0.091	0.087	0.157	0.165	0.078	0.094	0.042	0.066	0.034	15.6	4.7	13.11	14.04
THR	20	8.2	0.2	8.04	2.88	993	143	1.58	0.27	0.041	0.021	0.027	0.065	0.297	0.094	0.242	0.103	0.199	0.082	20.6	3.4	19.67	24.86
TIN	10	8.3	0.3	10.71	3.69	872	147	2.15	0.94	0.047	0.036	0.105	0.300	0.644	0.264	0.509	0.276	0.451	0.225	24.3	6.2	27.19	39.54
VRM	10	8.2	0.2	7.79	1.85	1019	242	1.89	0.33	0.047	0.030	0.016	0.011	0.486	0.219	0.421	0.221	0.362	0.217	23.7	4.8	10.01	7.13
WEI	16	7.9	0.3	8.27	2.13	1005	216	3.72	4.68	1.648	3.794	0.565	0.801	0.971	0.643	0.800	0.656	0.709	0.686	15.3	2.5	35.10	25.16

Table A-8. Descriptions and abbreviations of aquatic ecosystem functional metrics calculated in this study.

Parameter	Abbreviation	Unit	Description
Gross primary productivity	GPP	g $O_2/m^2/d$	Amount of primary production, or the synthesis of organic compounds, within a stream system per unit area per day, as expressed through the generation of dissolved oxygen.
Ecosystem respiration	ER	g $O_2/m^2/d$	Amount of organic compounds degraded through heterotrophic processes in a stream per unit area per day, as expressed through the consumption of dissolved oxygen.
Net ecosystem production	NEP	g $O_2/m^2/d$	Difference between GPP and ER, representing the total amount of organic carbon generated per unit area per day in an ecosystem available for storage, export or utilization.
Aeration constant	K600	h^{-1}	Rate of atmospheric oxygen being dissolved in water at the air-water interface.
Spiraling length of Nitrogen	Sw-N	m	Average distance traveled before inorganic nutrients are removed from solution through biological assimilation or physico-chemical sorption processes.
Spiraling length of Phosphorus	Sw-P	m	
Uptake velocity of Nitrogen	VfN	mm min^{-1}	Rate at which nutrients moves from the water to the streambed relative to concentration. Calculated by multiplying the inverse of uptake length by the depth and velocity of the stream reach.
Uptake velocity of Phosphorus	VfP	mm min^{-1}	
Areal uptake rate of Nitrogen	UN	mm m^{-2} min^{-1}	Measure of nutrient flux per area of streambed. Calculated as the product of uptake velocity and ambient concentration of nutrients in the stream water
Areal uptake rate of Phosphorus	UP	mm m^{-2} min^{-1}	
Total decomposition rate	Kc	d^{-1}	Rate of decay of litter using coarse mesh bags (2 mm size in this study) that permit entry of macroinverbrate decomposers combined with microbial decomposers.
Microbial decomposition rate	Kf	d^{-1}	Rate of decay of litter occurring by microorganisms, as determined by using fine mesh bags (0.2 mm size) that exclude macroinvertebrates.
Ratio of microbial-to-total decomposition rates	Kf_Kc	-	Proportion of microbial decomposition to total decomposition rates. Indicator of the relative degree of decomposition attributable to microbial to macroinvertebrate processes.

Table A-9. Aquatic ecosystem metabolic rates calculated for each stream in the Grassland and Parkland natural regions.

Creek	Year	Spring					Summer					Fall				
		Days*	GPP	ER	NEP	K600	Days*	GPP	ER	NEP	K600	Days*	GPP	ER	NEP	K600
Grassland																
BUL	2017	2	3.96	-6.6	-2.64	15.49	2	4.23	-4.1	0.13	1.09	2	4.18	-1	3.18	10.24
BVR	2017	2	-	-	-	-	2	2.99	-2	1	19.31	2	0.92	-0.52	0.41	7.74
KNE	2017	2	4.39	-2.61	1.78	7.43	2	7.29	-7.33	-0.04	3.89	2	3.35	-3.16	0.19	1.13
KNE	2018	3	7.02	-5.61	1.42	6.56	3	7.85	-6.4	1.44	4.77	2	6.21	-5.3	0.91	1.99
LWS	2017	3	9.86	-10.33	-0.47	5.17	-	-	-	-	-	-	-	-	-	-
MON	2017	3	2.13	-3.29	-1.16	1.92	-	-	-	-	-	-	-	-	-	-
MSQ	2017	2	5.96	-3.83	2.13	9.63	2	10.63	-5.27	5.36	4.31	2	4.35	-3.76	0.59	0.41
MTZ	2017	2	9.22	-8.18	1.04	8.57	2	6.58	-6.54	0.03	13.73	2	1.95	-2.26	-0.31	7.24
ONE	2017	2	5.16	-6.18	-1.02	7.15	2	10.22	-5.47	4.75	5.79	2	4.15	-7.86	-3.72	5.14
PER	2017	2	5.3	-7.05	-1.75	6.17	2	7.76	-8.35	-0.59	3.68	2	1.2	-0.73	0.47	3.84
POT	2017	1	5.96	-4.79	1.18	18.02	2	7.43	-6.19	1.24	20.59	2	8.72	-1.37	7.36	22.97
RSB	2017	2	3.12	-1.87	1.25	6.02	2	6.54	-6	0.53	4.26	2	3.03	-3.28	-0.25	3.17
RSB	2018	3	6.16	-4.11	2.05	7.11	3	5.14	-4.64	0.5	4.78	2	2.39	-1.39	1	6.69
UNK	2017	2	0.88	-6.38	-5.51	16.35	2	4.82	-11.38	-6.56	25.26	2	1.34	-8.1	-6.76	26.03
Parkland																
BUF	2018	2	4.41	-11.17	-6.76	6.09	2	6.58	-16.38	-9.8	6.14	1	8.85	-43.02	-34.17	15.26
BVH	2018	5	5.83	-17.31	-11.47	6	1	4.82	-1.48	3.34	1.44	2	1.62	-10.7	-9.38	3.04
DOG	2017	4	6.15	-4.77	1.38	4.41	2	6.81	-5.92	0.89	8.83	3	2.18	-2.88	-0.7	9.83
DOG	2018	3	1.57	-2.64	-1.08	4.37	3	16.61	-8.41	8.2	11.64	4	1.78	-1.55	0.23	11.98
EGL	2018	2	2.3	-2.88	-0.58	1.2	2	6.21	-4.52	1.69	1.61	3	0.83	-1.23	-0.4	2.69
GRZ	2018	2	3.3	-4.26	-0.96	1.5	4	5.15	-7.58	-2.43	3.98	2	3.67	-3.14	0.53	4.07
LST	2018	3	2.71	-3.06	-0.34	2.49	2	5.5	-4.23	1.27	2.89	2	1.04	-2.3	-1.26	6.04
LYD	2018	3	7.55	-7.33	0.22	7.96	2	6.13	-4.82	1.31	6.25	2	3.29	-6.45	-3.17	6.83
PIP	2018	2	2.06	-3.75	-1.68	3.38	2	2.46	-3.9	-1.44	1.37	3	3.32	-6.11	-2.79	6.23
RAY	2018	4	3.31	-8.65	-5.34	11.33	-	-	-	-	-	-	-	-	-	-
STU	2017	1	2.79	-9.95	-7.16	6.57	2	8.64	-8.22	0.41	3.39	2	6.62	-7.27	-0.66	4.07
STU	2018	2	5	-13.34	-8.35	3.09	2	4.43	-4.65	-0.22	3.32	2	2.82	-2.9	-0.08	2.41
THR	2017	4	9.74	-10.5	-0.76	5.99	1	3.52	-4.31	-0.79	1.63	-	-	-	-	-
THR	2018	4	6.41	-5.34	1.07	3.19	2	6.97	-18.38	-11.41	4.59	4	2.95	-12.84	-9.9	7.31
WEI	2018	2	6.75	-6.51	0.25	0.97	2	5.31	-6.84	-1.52	2.18	-	-	-	-	-

*Refers to the number of complete days that yield valid calculations of GPP, ER, NEP during the assessment period. Values presented are the average of the valid days.

Table A-10. Nutrient spiraling and litter decomposition rates calculated for each stream in the Grassland and Parkland natural regions.

Site	Years	SwN* m	SwP m	VfN mm min^{-1}	VfP mm min^{-1}	UN mm m^{-2} min^{-1}	UP mm m^{-2} min^{-1}	Kc d^{-1}	Kf d^{-1}	Kf_Kc
Parkland										
BUL	2017	568	5004	16.32	1.85	847	11.57	-0.016	-0.012	0.72
BVR	2017	152	288	10.85	4.85	212.7	33.38	-0.009	-0.005	0.61
KNE	2017	1101	3870	2.93	0.83	32.22	97.12	-0.016	-0.01	0.64
KNE	2018	800	12325	1.94	0.13	77.95	0.53			
LWS	2017							-0.013	-0.003	0.25
MON	2017							-0.015	-0.014	0.9
MSQ	2017	3915	10781	1.89	0.69	119.48	9.17	-0.011	-0.009	0.8
MTZ	2017	1308	12186	2.58	0.28	105.36	54.45	-0.032	-0.02	0.63
ONE	2017	1487	11567	7.78	1	274.32	95.66	-0.039	-0.014	0.37
PER	2017	678	3793	3.59	0.64	263.85	32.06	-0.024	-0.012	0.48
POT	2017	5786	21117	0.84	0.23	19.07	3.89	-0.006	-0.006	0.91
RSB	2017	364	6024	7.21	0.44	134.11	93.52			
RSB	2018	339	3170	16.95	1.81	575.4	189.88	-0.018	-0.013	0.7
UNK	2017	2940	18050	8.95	1.46	538.17	90.12	-0.016	-0.016	1.02
Grassland										
BUF	2018	1312	9683	1.69	0.23	86.75	33.91	-0.021	-0.015	0.69
BVH	2018							-0.019	-0.012	0.64
DOG	2017	251	1006	9.3	2.32	159.38	23.58	-0.014	-0.012	0.83
DOG	2018	1943	4154	3.25	1.52	56.3	6.81	-0.014	-0.008	0.52
EGL	2018	1296	888	0.51	0.74	12.87	5.87	-0.014	-0.011	0.84
GRZ	2018	589	1063	1.09	0.6	50.76	257.16	-0.019	-0.014	0.74
IRN	2018	449	20975	1.85	0.04	59.36	7.23			
LST	2018	4922	986	0.33	1.18	8.17	13.36	-0.009	-0.008	0.99
LYD	2018	28	235	4.29	0.51	149.08	84.7	-0.018	-0.011	0.61
PIP	2018	638	7073	1.83	0.17	58.27	18.03	-0.009	-0.009	1.08
RAY	2018	276	1044	2.07	0.55	93.34	72.5	-0.022	-0.011	0.51
STU	2017	506	4442	8.56	0.98	272.95	94.9	-0.018	-0.011	0.62
STU	2018			5.82	0.47	120	24.6	-0.011	-0.008	0.74
THR	2018	97	1046	4.9	0.46	75.53	58.61	-0.013	-0.01	0.74
WEI	2018	3952	2126	0.2	0.37	7.4	79.73	-0.019	-0.011	0.6

Table A-11. Ecosystem response metrics assessed for threshold responses to TN and TP, their predicted response to increased nutrients, and the transformation used to improve distributional properties for statistical analysis.

Metric Code	Metric Description	Unit	Transformation	Expected Response to Nutrients[1]	Response Indicating AEHI	References
Phytoplankton: Abundance						
WChla	Water column Chlorophyll *a*	mg/L	Natural logarithm	Increase	Increase	(Van Nieuwenhuyse and Jones 1996) (Basu and Pick 1996) (Chambers et al. 2012)
Periphyton: Abundance and Pigments						
TotCell	Algal cell density	#/cm^2	Natural logarithm	Increase or Humped	Increase or Decrease[2]	(Wagenhoff et al. 2013) (Pan and Lowe 1994) (Wagenhoff et al. 2011)
PChla	Periphyton Chlorophyll *a*	µg/cm^2	Natural logarithm	Increase	Increase	(Dodds 2006; Dodds et al. 2002) (Dodds and Smith 2016) (Chambers et al. 2012)
DPi	Diatom Pigment Index: Ratio of diagnostic pigments of diatoms to Chlorophyll a	-	Natural logarithm	Increase or Humped	Decrease	Proposed
GPi	Green Algae Pigment Index: Ratio of diagnostic pigments of green algae to Chlorophyll a	-	Natural logarithm	Increase or Humped	Increase or Decrease[2]	Proposed
PPCi	Photoprotective Carotenoid Index: Ratio of photoprotective pigments to all pigments	-	Logit	Decrease	Decrease	(Claustre et al. 2004)
Periphyton: Diversity Metrics based on Taxonomic Cell Counts						
GRich	Richness (genus level)	#	None	Humped	Decrease	(Pringle 1990) (Stevenson et al. 2008) (Wagenhoff et al. 2011) (Liess et al. 2009)
GEven	Pielou's Evenness (genus level)	-	None	Humped	Decrease	(Stevenson et al. 2008) (Wagenhoff et al. 2013) (Liess et al. 2009)
GDiv	Shannon Diversity (genus level)	-	None	Humped	Decrease	(Groendahl and Fink 2017)
Periphyton: Community Composition Metrics based on Taxonomic Cell Counts						
RelCya	Relative abundance of cyanobacteria	-	Logit	Decrease	Increase	(Mulholland et al. 1995) (Peterson and Grimm 1992) (Black et al. 2011) (Bahls 1993)
RelGre	Relative abundance of green algae	-	Logit	Increase or Humped	Increase or Decrease[2]	(Borchardt 1996) (Chételat et al. 1999) (Whitton 1970) (Stelzer and Lamberti 2001) (Black et al. 2011) (Bahls 1993)(Groendahl and Fink 2017)
RelDia	Relative abundance of diatoms	-	Logit	Increase or Humped	Decrease	(Mulholland et al. 1995) (Black et al. 2011; Peterson and Grimm 1992) (Black et al. 2011)
RelMotD	Relative abundance of motile diatoms	-	Logit	Increase	Increase	(Pringle 1990) (Lange et al. 2011) (Passy 2007)
RelSens	Relative abundance of pollution-sensitive diatom taxa	-	Logit	Decrease	Decrease	(Lange-Bertalot 1979) (Bahls 1993) (Black et al. 2011)
GDI	Generic Diatom Index	-	None	Decrease	Decrease	(Rumeau and Coste 1988) (Kelly et al. 1995) (Hering et al. 2006)

Metric Code	Metric Description	Unit	Transformation	Expected Response to Nutrients[1]	Response Indicating AEHI	References
Ecosystem Function: Nutrient Uptake						
VfN	Nitrogen uptake velocity	m/s	Natural logarithm	Humped or Decrease	Decrease	(Bernot and Dodds 2005) (Bernot et al. 2006) (Covino et al. 2012)
VfP	Phosphorus uptake velocity	m/s	Natural logarithm	Humped	Decrease	(Bernot and Dodds 2005) (Bernot et al. 2006) (Covino et al. 2012)
UN	Areal uptake of Nitrogen	$mg/m^2/s$	Natural logarithm	Humped	Decrease	(Bernot and Dodds 2005) (Bernot et al. 2006) (Covino et al. 2012)
UP	Areal uptake of Phosphorus	$mg/m^2/s$	Natural logarithm	Humped	Decrease	(Bernot and Dodds 2005) (Bernot et al. 2006) (Covino et al. 2012)
Ecosystem Function: Litter Decomposition						
Kf	Microbial litter breakdown rate	mg/d	None	Increase or Humped	Increase	(Grattan and Suberkropp 2001; Gulis et al. 2006; Niyogi et al. 2003; Piggott et al. 2015)
Kc	Total litter breakdown rate	mg/d	None	Decrease or Humped	Decrease	(Gessner and Chauvet 2002) (Lecerf et al. 2006)
KfKc	Ratio of Kf to Kc (proportion of breakdown owing to microbial action)	-	None	Increase	Increase	(Gessner and Chauvet 2002) (Lecerf et al. 2006)
Ecosystem Function: Whole-Stream Metabolism						
GPP	Gross primary productivity	g $O_2/m^2/d$	None	Increase	Increase	(Yates et al. 2013) (Young et al. 2008)
ER	Ecosystem respiration	g $O_2/m^2/d$	None	Increase	Increase	(Yates et al. 2013) (Young et al. 2008)
NEP	Net ecosystem productivity	g $O_2/m^2/d$	None	Decrease	Decrease	(Yates et al. 2013) (Young et al. 2008)

[1] Expected responses represent the response across a global range of nutrient concentrations. In the particular streams studied the observed relationship could therefore be increasing, decreasing, or humped depending on the range of nutrients observed. In the case of humped relationships, effort has been made to suggest which of the potential relationships were most likely to be observed.

[2] For some humped relationships an increase or decrease could represent impairment. For example, for green algae, an increase in abundance could indicate increasing dominance of filamentous green algae, a nuisance in aquatic systems, while a decrease could be representative of impairment if coupled with increased Cyanobacteria.

Table A-12. Total number of threshold analyses performed and total number and proportion of threshold values retained for each response metric.

Response Metric	Number of Threshold Analyses Performed	Number of Threshold Values Retained	Proportion of Threshold Values Retained
VfP	6	5	0.83
UN	6	4	0.67
VfN	6	4	0.67
GDI	24	13	0.54
WChla	24	13	0.54
RelGre	24	12	0.5
GPi	24	10	0.42
NEP	12	4	0.33
RelDia	24	8	0.33
DPi	24	7	0.29
PChla	24	6	0.25
RelSens	24	6	0.25
TotCell	24	6	0.25
GEven	24	5	0.21
RelCya	24	5	0.21
Kc	6	1	0.17
Kf	6	1	0.17
RelMotD	24	4	0.17
GDiv	24	1	0.04
GRich	24	1	0.04
ER	12	0	0
GPP	12	0	0
KfKc	6	0	0
PPCi	24	0	0
UP	6	0	0
Total	438	116	0.26

Table A-13. TN threshold values (in mg/L) and response metric weights included in derivation of AEHI risk categories via analysis of weighted cumulative distributions functions of threshold values. Thresholds shaded in dark grey were averaged and included as one value in the WEDF for that NR-Season combination.

Metrics*	Grassland						Parkland						Weight
	Spring			Summer			Spring			Summer			
	PLR	CART	QPLR	PLR	CART	QPLR	PLR	CART	QPLR	PLR	CART	QPLR	
WChla			0.86			0.65		1.34	1.78			1.3	0.98
TotCell											1.17		0.88
PChla						2.79			2.44		1.74		0.99
DPi				1.78	2.32						2.29		0.74
GPi		0.68			2.42	0.29					1.66	1.66	0.74
PPCi													0.48
GRich													0.78
GEven		1.34	0.83					1.81					0.71
GDiv													0.72
RelCya								1.68	1.68		2.7		0.85
RelGre	0.94	0.85	0.9		3.45		2.4	2.68					0.74
RelDia		0.85				2.73		2.51			1.79		0.74
RelMotD					2.47						3.75		0.64
RelSens					1.93				2.44				0.76
GDI				1.93				2.25			3.75	1.71	0.7
VfN					1.87	1.24					1.87	1.24	0.87
VfP		N/A			0.9	0.84		N/A			0.9	0.84	0.87
UN				0.64	1.28	0.78				0.64	1.28	0.78	0.81
UP													0.83
Kf													0.77
Kc		N/A			0.63			N/A			0.63		0.74
KfKc													0.75
GPP													0.89
ER													0.89
NEP		2.01			0.58			2.01			0.58		0.83

*Metric codes are described in Table A-11.

Table A-14. TP threshold values (in mg/L) and response metric weights included in derivation of AEHI risk categories via analysis of weighted cumulative distributions functions of threshold values. Thresholds shaded in dark grey were averaged and included as one value in the WEDF for that NR-Season combination.

Metrics*	Grassland						Parkland						Weight
	Spring			Summer			Spring			Summer			
	PLR	CART	QPLR	PLR	CART	QPLR	PLR	CART	QPLR	PLR	CART	QPLR	
WChla	0.07			0.6		0.07	0.27	0.15	0.15		0.25	0.25	0.98
TotCell		0.32			0.08			0.55		0.13	0.51		0.88
PChla		0.32		0.44	0.05								0.99
DPi		0.21		0.28	0.41						0.16		0.74
GPi				0.18				0.3		0.06	0.18	0.34	0.74
PPCi													0.48
GRich					0.03								0.78
GEven								0.37			0.82		0.71
GDiv								0.29					0.72
RelCya								0.25			0.51		0.85
RelGre	0.09	0.05			0.18			0.3			0.34	0.27	0.74
RelDia		0.13			0.18			0.1			0.46		0.74
RelMotD							0.47	0.55					0.64
RelSens				0.17	0.77	0.28	0.47						0.76
GDI			0.15	0.27	0.33	0.14	0.05	0.35		0.67	0.95	0.6	0.7
VfN				0.24	0.28					0.24	0.28		0.87
VfP		N/A		0.13	0.04	0.08		N/A		0.13	0.04	0.08	0.87
UN					0.28						0.28		0.81
UP													0.83
Kf						0.17						0.17	0.77
Kc		N/A						N/A					0.74
KfKc													0.75
GPP													0.89
ER													0.89
NEP					0.04	0.04					0.04	0.04	0.83

*Metric codes are described in Table A-11.

Table A-17. Calibrated SWAT parameters for Indianfarm Creek and Threehills Creek watersheds.

Parameter name in SWAT*	Description of selected parameters	Calibrated Values	
		Indianfarm	Threehills
SFTMP.bsn	Snowfall temperature (ºC).	0.6	-1
SMTMP.bsn	Snowmelt base temperature (ºC).	2.72	5
SMFMX.bsn	Melt factor for snow on June 21 (mm water/ºC-day).	4.95	6.5
SMFMN.bsn	Melt factor for snow on December 21 (mm water/ºC-day).	2.48	0.35
TIMP.bsn	Snow pack temperature lag factor.	0.07	0.54
SNOCOVMX.bsn	Minimum snow water content that corresponds to 100% snow cover, SNO100, (mm water).	37	30
SNO50COV.bsn	Fraction of snow volume represented by SNOCOVMX that corresponds to 50% snow cover.	0.16	0.6
IPET	Potential evapotranspiration method	2	2
ESCO.bsn	Soil evaporation compensation factor.	0.67	0.85
EPCO.bsn	Plant uptake compensation factor.	0.89	1
SURLAG.bsn	Surface runoff lag coefficient.	1	6

* Abbreviations for SWAT input files extensions: .bsn = basin.

Table A-18. Calibrated parameters for the APEX model for the Indianfarm Creek and Threehills Creek watersheds.

Parameter description		Calibrated values	
		Indianfarm	Threehills
APEX APEXCONT.dat file			
Variable(n)			
IET	Potential evapotranspiration equation code	4	4
ITYP	Peak rate estimate code	-1	-1
NVCN	Non-varying CN-CN2 used	4	4
GWSO	Maximum groundwater storage	50	180
RFTO	Groundwater residence time in days	10	90
RFPO	Return flow/(Return flow + Deep percolation)	0.2	0.85
DRV	Equation for water erosion	3	3
APEX PARM0604.dat file			
PARM (n)			
4	Water storage N leaching	0.5	1
7	N fixation	0.5	0.6
8	Soluble phosphorus runoff coefficient. (0.1 m^3 Mg^{-1})	20	18
14	Nitrate leaching ratio	0.1	0.25
15	Runoff CN weighting factor	0	0.01
17	Soil evaporation - plant cover factor	0.5	0.35
18	Sediment routing exponent	1	1.1
19	Sediment routing coefficient (Mg m^{-3})	0.01	0.03
20	Runoff curve number initial abstraction	0.2	0.15
22	Runoff CN retention parameter for frozen soil	0.2	0.05
23	Hargreaves PET equation coefficient	0.0026	0.0028
29	Biological mixing efficiencies	0.5	0.45
30	Soluble phosphorus runoff exponent	1	1.4
32	Organic N and P sediment transport exponent	1	1.2
34	Hargreaves PET equation exponent	0.52	0.525
37	Crop residue runoff	1	0.4
38	Water stress weighting coefficient	1	0.7
40	Groundwater storage threshold	0.2	0.2
42	SCS curve number index coefficient	1	1.4
45	Sediment routing travel time coefficient	10	0.5
46	RUSLE C - factor coefficient	1.5	0.5
47	RUSLE C - factor coefficient	0.5	1.5
49	Maximum rainfall interception by plant canopy (mm)	4	3.5
59	P upward movement by evaporation	5	3
62	Manure erosion equation coefficient	0.1	0.18
68	Manure erosion exponent	0.7	0.1
69	Manure erosion coefficient	1	1.4
74	Nitrate leaching ratio for lateral return flow	0.02	0.01
78	Soil water value to delay tillage	0.7	1
80	Soil radiation threshold for snowmelt	12	20
82	N upward movement by evaporation coefficient	0.1	1

Table A-19. Summary of SWAPP model fit statistics for flow, total suspended solids (TSS), total nitrogen (TN), and total phosphorus (TP) for Indianfarm Creek and Threehills Creek watersheds.

		Calibration			Validation	
Parameter*	R^2	NSE	PBIAS	R^2	NSE	PBIAS
Indianfarm Creek		*— 2007 - 2010 —*			*— 2011 —*	
Flow	0.9	0.9	-22.1	0.74	0.65	36.3
TSS	0.89	0.83	-12.3	0.77	0.77	4.4
TN	0.89	0.89	-26.8	0.97	0.65	51.1
TP	0.9	0.8	6.8	0.76	0.75	16.5
Threehills Creek		*— 2014 - 2016 —*			*— 2017 —*	
Flow	0.86	0.68	13.9	0.75	0.72	6.1
TSS	0.96	0.8	5.6	0.93	0.9	2.4
TN	0.86	0.66	20.9	0.92	0.78	33.5
TP	0.92	0.8	12.3	0.96	0.89	25.6

* R^2 (correlation of determination), *NSE* (Nash-Sutcliffe efficiency), and *PBIAS* (percent bias)

Table A-20. List of beneficial management practices (BMP) and criteria used for sub-area selection.

BMP type	Watershed	Number of fields	Watershed affected area (%)	Fields selection criteria for BMP application	BMP Description
Filter Strip	Indianfarm	19	23.2	TP > 1.6 mg l^{-1}, Cultivated	10m filter strips or vegetated areas were applied at the edge of cultivated fields to remove larger particles including sediment and nutrients from the runoff water
	Threehills	106	31.5	TP > 2 mg l^{-1}, Cultivated	
Dugout (Pond)	Indianfarm	17	9.3	TP > 2.0 mg l^{-1}, Area > 20 ha, Pasture	Dugouts were applied on pasture fields to provide water for grazing cattle and to reduce amount of surface runoff. The capacity of dugouts were assumed 5000 m^3
	Threehills	13	7.2	TP > 0.8 mg l^{-1}, Area > 30 ha, Pasture	
Grassed Waterway	Indianfarm	1	0.5	TSS load > 2 Mg ha^{-1} yr^{-1}, Cultivated	10 m wide grassed waterways were added to concentrated flow paths in fields meeting the TSS load requirement
	Threehills	0	0	TSS load > 2 Mg ha^{-1} yr^{-1}, Cultivated	
Wetland Restoration	Indianfarm	2	5.6	Area drained during 1991-2010 period	The extent of wetland restoration was estimated based on the difference of wetland area that was recorded by the 1990 and 2010 Landsat imagery data.
	Threehills	3	3.3	Drain during 1991-2000 period	
Stream Fencing	Indianfarm	46	29.8	TP > 1.6 mg l^{-1}, Pasture	Stream fences were applicable to pasture fields. In simulations, it was assumed that fences were located 5 m away from streams and restricted cattle access to streams all year around.
	Threehills	24	8.3	TP > 0.8 mg l^{-1}, Pasture	
Rotational Grazing	Indianfarm	57	39.4	TP > 0.8 mg l^{-1}, Area > 9 ha	Pasture fields were divided into three equal-size subareas, and rotational grazing started on June 08 and ended on September 08 of each year. In addition, cattle were simulated to move from pasture to pasture with 7 days of grazing at a time and 20 days of field-resting time between grazing.
	Threehills	24	8.3	TP > 0.8 mg l^{-1}, Area > 9 ha	
Soil P	Indianfarm	17	17.4	STP > 200 ppm, Manured	CEEOT checked the STP level on January 01 of each year; if the STP level exceeded 200 ppm, no manure was added. Where STP levels were below 200 ppm, 17 Mg ha^{-1} of beef manure was automatically applied.
	Threehills	134	37.1	STP > 200 ppm, Manured	

Table A-21. List of scenarios and associated BMPs used in CEEOT simulations at Indianfarm Creek and Threehills Creek watersheds.

Scenarios	Beneficial management practices						
	Filter Strip	Dugout (Pond)	Grassed Waterway	Wetlands restoration	Stream Fencing	Rotational Grazing	Soil P
Baseline	Status quo land management practices						
Scenario 1	X						
Scenario 2	X	X					
Scenario 3	X	X	X	X			
Scenario 4	X	X	X	X	X		
Scenario 5	X	X	X	X	X	X	
Scenario 6	X	X	X	X	X	X	X

[y] AOPA = Alberta Agricultural Operation Practices Act.

Table A-22 .SWAPP 25-yrs simulated monthly total nitrogen (TN) load and flow weighted mean concentrations (FWMC) values of mean, 50th, 25th, and 75th percentile, and the estimated cumulative impacts (Change %) of simulated scenarios relative to baseline scenario at the outlet of Indianfarm Creek watershed.

Scenario	Month	TN Load (kg ha^{-1})				TN Load Change (%)				TN FWMC (mg l^{-1})				TN FWMC Change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Baseline	March	0.416	0.232	0.085	0.503					5.64	3.61	2.01	6.71				
	April	0.648	0.362	0.083	0.773					3	2.43	0.87	3.65				
	May	0.767	0.116	0.067	0.793					4.33	2.43	0.71	3.57				
	June	1.374	0.326	0.033	1.936					3.61	3.56	0.85	5.47				
	July	0.743	0.134	0.007	0.34					5.32	3.68	0.64	8.03				
	August	0.258	0.002	0	0.146					3.78	0.57	0	6.52				
	September	0.135	0	0	0.004					1.51	0	0	1.27				
Scenario 1	March	0.36	0.242	0.085	0.481	-14	-9	-24	-3	5.28	3.52	1.81	6.1	3	-2	-11	24
	April	0.552	0.218	0.1	0.679	-15	-13	-34	0	2.9	1.76	1.15	4.13	7	2	-13	18
	May	0.665	0.1	0.044	0.553	-18	-17	-30	-2	3.87	1.77	0.84	5.14	-1	0	-17	11
	June	1.081	0.661	0.028	1.504	41	-22	-37	1	3.61	3.55	1.05	4.88	75	-15	-23	14
	July	0.512	0.068	0.004	0.227	-31	-31	-38	-15	3.6	2.17	0.62	6.05	-16	-17	-33	8
	August	0.187	0.001	0	0.071	-27	-21	-32	0	2.83	0.25	0	3.24	-20	-2	-29	0
	September	0.126	0	0	0.004	-4	0	-3	0	2.02	0	0	1.24	1	0	0	0
Scenario 2	March	0.359	0.242	0.085	0.478	-14	-9	-24	-4	5.28	3.52	1.79	6.06	3	-1	-11	24
	April	0.548	0.207	0.098	0.675	-16	-14	-34	-1	2.89	1.75	1.12	4.12	6	0	-13	17
	May	0.661	0.097	0.041	0.55	-19	-17	-35	-3	3.85	1.76	0.84	5.12	-2	-2	-22	11
	June	1.069	0.654	0.024	1.495	39	-22	-38	0	3.58	3.46	0.98	4.83	73	-15	-26	13
	July	0.506	0.067	0.004	0.22	-33	-31	-39	-16	3.56	2.15	0.58	5.96	-17	-17	-33	2
	August	0.185	0.001	0	0.068	-28	-21	-33	0	2.81	0.21	0	3.19	-21	-5	-29	0
	September	0.124	0	0	0.004	-8	0	-4	0	2	0	0	1.19	-3	0	0	0
Scenario 3	March	0.351	0.211	0.079	0.457	-17	-9	-22	-7	5.33	3.78	1.9	6.13	4	-1	-9	28
	April	0.531	0.181	0.093	0.645	-22	-23	-37	-3	3.01	1.78	1.13	4.37	6	6	-9	19
	May	0.663	0.099	0.042	0.499	-20	-24	-36	-6	4.06	1.76	0.87	4.96	5	5	-13	10
	June	1.054	0.659	0.025	1.474	41	-27	-39	-1	3.69	3.49	1.12	4.91	86	-11	-24	10
	July	0.499	0.066	0.004	0.209	-38	-35	-59	-23	3.64	2.16	0.61	6	-20	-21	-51	2
	August	0.181	0.001	0	0.07	-27	-15	-36	0	2.86	0.19	0	3.23	-18	-1	-29	1
	September	0.119	0	0	0.004	-11	0	-6	0	2.08	0	0	1.26	-3	0	0	0

Table A-22 .SWAPP 25-yrs simulated monthly total nitrogen (TN) load and flow weighted mean concentrations (FWMC) values of mean, 50th, 25th, and 75th percentile, and the estimated cumulative impacts (Change %) of simulated scenarios relative to baseline scenario at the outlet of Indianfarm Creek watershed.

Scenario	Month	TN Load (kg ha^{-1})				TN Load Change (%)				TN FWMC (mg l^{-1})				TN FWMC Change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Scenario 4	March	0.334	0.144	0.048	0.468	-31	-31	-50	-9	4.94	3.03	1.91	7.19	-5	-5	-28	7
	April	0.49	0.187	0.062	0.575	-30	-26	-49	-14	2.98	1.84	1.01	4.39	7	5	-13	21
	May	0.564	0.101	0.03	0.559	-32	-32	-53	-16	3.94	1.58	0.89	4.29	-2	-7	-22	16
	June	0.942	0.615	0.021	1.44	35	-37	-42	-25	3.71	3.46	0.87	4.96	92	-19	-30	3
	July	0.428	0.056	0.004	0.199	-47	-45	-64	-37	3.24	1.93	0.49	4.93	-31	-31	-54	-14
	August	0.15	0.001	0	0.065	-38	-41	-54	-10	2.6	0.21	0	3.74	-29	-29	-48	0
	September	0.102	0	0	0.002	-12	0	-13	0	1.66	0	0	0.72	-5	0	0	0
Scenario 5	March	0.365	0.194	0.047	0.428	-25	-30	-47	-8	5.44	3.78	1.81	5.99	0	0	-18	18
	April	0.504	0.196	0.079	0.635	-10	-23	-43	-10	3.39	2.9	1.26	4.29	45	11	-13	47
	May	0.59	0.118	0.037	0.596	-19	-18	-46	4	4.18	2.45	0.81	5.02	19	9	-4	29
	June	1.029	0.639	0.019	1.069	59	-35	-53	-5	3.8	3.57	0.92	5.22	130	-10	-23	5
	July	0.326	0.049	0.003	0.199	-36	-49	-62	-41	3.29	2.27	0.53	4.78	-14	-31	-54	-8
	August	0.092	0.001	0	0.061	95	-58	-83	0	3.04	0.2	0	2.58	168	-36	-69	0
	September	0.131	0	0	0.001	-13	0	-6	0	2.47	0	0	0.39	-2	0	0	0

Table A-23.SWAPP 25-yrs simulated monthly total phosphorous (TP) load and flow weighted mean concentrations (FWMC) values of mean, 50th, 25th, and 75th percentile, and the estimated cumulative impacts (Change %) of simulated scenarios relative to baseline scenario at the outlet of Indianfarm Creek watershed.

Scenario	Month	TP Load (kg ha^{-1})				TP Load change (%)				TP FWMC (mg l^{-1})				TP FWMC Change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Baseline	March	0.08	0.05	0.02	0.12					1.07	0.8	0.43	1.04				
	April	0.12	0.06	0.02	0.12					0.55	0.35	0.18	0.72				
	May	0.12	0.02	0.01	0.1					0.52	0.34	0.14	0.72				
	June	0.3	0.13	0.01	0.29					0.79	0.65	0.23	1.2				
	July	0.13	0.01	0	0.07					0.8	0.5	0.12	1.18				
	August	0.05	0	0	0.02					0.65	0.11	0	0.94				
	September	0.07	0	0	0					0.41	0	0	0.47				
Scenario 1	March	0.07	0.05	0.02	0.14	-9	-7	-19	0	1.12	0.8	0.43	1.28	6	1	-5	18
	April	0.11	0.06	0.02	0.13	-8	-15	-31	6	0.59	0.39	0.18	0.81	14	1	-7	27
	May	0.12	0.01	0.01	0.12	-11	-16	-31	1	0.57	0.37	0.12	1.03	6	0	-12	13
	June	0.27	0.05	0.01	0.35	-7	-16	-28	5	0.76	0.61	0.26	1.2	11	-5	-12	14
	July	0.1	0.01	0	0.05	-26	-29	-38	-17	0.68	0.43	0.11	0.94	-10	-11	-27	4
	August	0.04	0	0	0.02	-27	-18	-52	0	0.5	0.05	0	0.75	-20	-10	-31	0
	September	0.07	0	0	0	-4	0	0	0	0.41	0	0	0.51	1	0	0	0
Scenario 2	March	0.07	0.05	0.02	0.14	-9	-7	-19	0	1.12	0.8	0.43	1.28	6	1	-5	18
	April	0.11	0.06	0.02	0.13	-8	-13	-31	4	0.59	0.39	0.18	0.81	14	2	-7	28
	May	0.12	0.01	0.01	0.12	-11	-14	-30	0	0.57	0.36	0.13	1.03	7	0	-10	13
	June	0.26	0.05	0.01	0.35	-7	-16	-29	5	0.76	0.61	0.26	1.2	11	-6	-13	14
	July	0.1	0.01	0	0.05	-26	-30	-38	-17	0.68	0.43	0.1	0.94	-9	-11	-28	4
	August	0.04	0	0	0.02	-27	-19	-53	0	0.49	0.05	0	0.75	-20	-10	-32	0
	September	0.07	0	0	0	-9	0	-2	0	0.4	0	0	0.51	-5	0	0	0
Scenario 3	March	0.07	0.05	0.02	0.14	-12	-9	-22	0	1.13	0.82	0.45	1.3	8	4	-3	20
	April	0.11	0.05	0.02	0.12	-14	-17	-32	7	0.62	0.41	0.19	0.83	17	7	-3	36
	May	0.12	0.01	0.01	0.11	-11	-12	-32	0	0.61	0.39	0.14	1.08	17	12	-6	28
	June	0.26	0.05	0.01	0.35	-7	-18	-29	1	0.78	0.65	0.27	1.23	17	0	-8	18
	July	0.1	0.01	0	0.05	-31	-30	-43	-20	0.69	0.44	0.13	0.97	-11	-16	-29	-1
	August	0.04	0	0	0.02	-27	-17	-45	0	0.51	0.06	0	0.74	-17	-8	-33	0
	September	0.07	0	0	0	-12	0	-5	0	0.39	0	0	0.34	-5	0	0	0

Table A-23.SWAPP 25-yrs simulated monthly total phosphorous (TP) load and flow weighted mean concentrations (FWMC) values of mean, 50th, 25th, and 75th percentile, and the estimated cumulative impacts (Change %) of simulated scenarios relative to baseline scenario at the outlet of Indianfarm Creek watershed.

Scenario	Month	TP Load (kg ha^{-1})				TP Load change (%)				TP FWMC (mg l^{-1})				TP FWMC Change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Scenario 4	March	0.07	0.04	0.02	0.13	-23	-17	-40	-12	1.05	0.83	0.35	1.28	3	2	-13	13
	April	0.1	0.06	0.01	0.11	-18	-20	-34	-6	0.64	0.49	0.22	0.91	26	17	1	34
	May	0.11	0.01	0	0.11	-21	-21	-52	0	0.59	0.36	0.13	1.04	11	0	-17	34
	June	0.25	0.05	0	0.3	-18	-27	-43	-12	0.75	0.61	0.18	1.14	8	-8	-25	4
	July	0.09	0.01	0	0.05	-37	-37	-49	-25	0.66	0.41	0.14	0.91	-19	-19	-35	-8
	August	0.03	0	0	0.02	-36	-31	-62	-3	0.48	0.04	0	0.77	-26	-19	-50	0
	September	0.06	0	0	0	-13	0	-7	0	0.37	0	0	0.17	-7	0	0	0
Scenario 5	March	0.07	0.05	0.01	0.14	-18	-14	-38	1	1.14	0.97	0.41	1.33	8	11	-6	24
	April	0.11	0.05	0.01	0.12	-7	-8	-32	3	0.71	0.51	0.32	0.91	47	29	2	81
	May	0.12	0.02	0.01	0.13	-8	-2	-31	19	0.67	0.52	0.15	1.16	32	29	-5	61
	June	0.27	0.07	0	0.3	-9	-25	-49	1	0.85	0.68	0.18	1.27	24	3	-22	24
	July	0.07	0.01	0	0.05	-30	-39	-49	-16	0.72	0.48	0.14	1.13	-3	-12	-29	7
	August	0.02	0	0	0.01	3	-53	-72	0	0.53	0.04	0	0.79	43	-11	-59	0
	September	0.07	0	0	0	-13	0	0	0	0.45	0	0	0.07	-6	0	0	0

Table A-24 .SWAPP simulated monthly total nitrogen (TN) load and flow weighted mean concentrations (FWMC) of mean, 50th, 25th, and 75th percentile values, and the estimated cumulative impacts (change %) of simulated scenarios relative to baseline scenario at the outlet of Threehills Creek watershed.

Scenario	Month	TN Load (kg ha^{-1})				TN Load change (%)				TN FWMC (mg l^{-1})				TN FWMC change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Baseline	March	0.176	0	0	0.321					1.25	0.04	0	2.79				
	April	0.546	0.23	0	0.887					2.05	2.43	0.26	3.06				
	May	0.041	0.004	0.002	0.006					3.04	1.05	0.19	2.4				
	June	0.024	0.003	0.001	0.011					0.64	0.17	0.12	0.66				
	July	0.022	0.001	0	0.002					0.21	0.07	0.02	0.29				
	August	0.001	0.001	0	0.001					0.08	0.04	0.02	0.09				
	September	0	0	0	0.001					0.07	0.03	0.02	0.13				
Scenario 1	March	0.173	0	0	0.32	3	0	-4	0	1.24	0.04	0.01	2.68	3	0	-3	0
	April	0.54	0.236	0	0.882	-2	1	-7	4	1.99	2.4	0.2	3.13	-3	-1	-8	2
	May	0.047	0.004	0.002	0.006	6	0	-3	5	2.92	1.13	0.23	2.7	7	-2	-5	2
	June	0.025	0.002	0.001	0.019	25	-1	-9	5	0.64	0.25	0.11	0.68	69	-2	-7	6
	July	0.023	0.001	0	0.003	36	1	-1	4	0.24	0.07	0.03	0.26	26	2	-1	5
	August	0.001	0.001	0	0.001	1	1	-1	1	0.08	0.04	0.03	0.09	0	0	-2	2
	September	0	0	0	0.001	1	0	-1	1	0.07	0.03	0.02	0.14	0	0	-3	1
Scenario 2	March	0.174	0	0	0.325	-2	0	-7	0	1.26	0.01	0	2.72	-2	0	-7	1
	April	0.538	0.238	0	0.861	1	1	-6	6	1.97	2.4	0.16	3.09	-7	-1	-7	1
	May	0.047	0.003	0.001	0.006	-3	-12	-16	-4	2.66	0.97	0.21	2.45	-3	-11	-18	-3
	June	0.025	0.002	0.001	0.019	15	-17	-28	-9	0.61	0.25	0.1	0.68	66	-16	-22	-7
	July	0.022	0.001	0	0.002	6	-13	-17	-5	0.19	0.06	0.03	0.25	4	-12	-16	-5
	August	0.001	0	0	0.001	-17	-18	-22	-14	0.07	0.03	0.03	0.07	-17	-18	-21	-15
	September	0	0	0	0	-17	-19	-23	-16	0.06	0.02	0.02	0.1	-18	-19	-23	-15
Scenario 3	March	0.174	0	0	0.324	-3	-2	-6	0	1.31	0.01	0	2.84	1	0	-2	4
	April	0.537	0.235	0.002	0.859	309	2	-3	9	2.02	2.32	0.21	3.17	19	0	-2	11
	May	0.047	0.003	0.002	0.006	1	-16	-18	-4	2.64	0.98	0.19	2.44	-3	-15	-18	-5
	June	0.024	0.002	0.001	0.019	12	-20	-30	-11	0.62	0.26	0.1	0.69	67	-17	-25	-7
	July	0.022	0.001	0	0.002	1	-19	-23	-10	0.19	0.06	0.03	0.26	2	-14	-18	-8
	August	0.001	0	0	0.001	-23	-23	-27	-19	0.07	0.03	0.02	0.07	-20	-21	-24	-17
	September	0	0	0	0	-20	-22	-27	-17	0.06	0.02	0.02	0.1	-19	-21	-24	-15

Table A-24 .SWAPP simulated monthly total nitrogen (TN) load and flow weighted mean concentrations (FWMC) of mean, 50th, 25th, and 75th percentile values, and the estimated cumulative impacts (change %) of simulated scenarios relative to baseline scenario at the outlet of Threehills Creek watershed.

Scenario	Month	TN Load (kg ha^{-1})				TN Load change (%)				TN FWMC (mg l^{-1})				TN FWMC change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Scenario 4	March	0.176	0	0	0.318	-5	-2	-17	0	1.31	0.01	0	2.78	-2	0	-6	3
	April	0.532	0.214	0.001	0.869	129	2	-3	9	2.07	2.49	0.18	3.19	-4	0	-12	6
	May	0.04	0.003	0.001	0.005	-14	-17	-21	-13	2.55	0.85	0.14	2.06	-19	-17	-27	-13
	June	0.025	0.002	0.001	0.036	33	-23	-36	6	0.68	0.34	0.09	0.83	89	-20	-28	4
	July	0.022	0.001	0	0.002	-21	-24	-31	-9	0.19	0.05	0.02	0.18	-18	-20	-28	-9
	August	0.001	0	0	0.001	-27	-26	-34	-23	0.07	0.03	0.02	0.07	-25	-25	-29	-21
	September	0	0	0	0	-22	-25	-27	-20	0.06	0.02	0.02	0.1	-22	-23	-26	-19
Scenario 5	March	0.175	0	0	0.327	-6	-3	-17	0	1.32	0.01	0	2.73	-2	0	-6	3
	April	0.53	0.23	0.001	0.788	129	-1	-6	7	2.01	2.44	0.19	2.99	-2	0	-11	4
	May	0.046	0.003	0.001	0.005	18	-16	-21	-11	2.72	0.97	0.16	2.52	22	-16	-25	-10
	June	0.023	0.002	0.001	0.011	-16	-25	-35	-11	0.58	0.13	0.09	0.43	-14	-23	-28	-4
	July	0.023	0.001	0	0.001	-16	-25	-30	-13	0.2	0.05	0.02	0.35	-16	-20	-24	-15
	August	0.001	0	0	0.001	-28	-28	-34	-24	0.07	0.03	0.02	0.07	-21	-21	-25	-16
	September	0	0	0	0	-24	-26	-29	-21	0.06	0.03	0.02	0.1	-17	-20	-24	-15

Table A-25. SWAPP 25-yrs simulated monthly total phosphorous (TP) load and flow weighted mean concentrations (FWMC) values of mean, 50th, 25th, and 75th percentile, and the estimated cumulative impacts (change %) of simulated scenarios relative to baseline scenario at the outlet of Threehills Creek watershed.

Scenario	Month	TP Load ($kg\ ha^{-1}$)				TP Load change (%)				TP FWMC ($mg\ l^{-1}$)				TP FWMC change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Baseline	March	0.046	0.002	0	0.077					0.4	0.22	0.1	0.56				
	April	0.116	0.055	0	0.223					0.47	0.46	0.07	0.75				
	May	0.005	0	0	0.001					0.16	0.09	0.02	0.21				
	June	0.013	0.002	0	0.006					0.14	0.12	0.04	0.19				
	July	0.02	0.001	0	0.003					0.15	0.05	0.01	0.17				
	August	0.001	0	0	0.001					0.08	0.02	0.01	0.07				
	September	0	0	0	0					0.01	0	0	0.02				
Scenario 1	March	0.045	0.002	0	0.075	-1	0	0	0	0.4	0.22	0.1	0.58	0	0	0	0
	April	0.115	0.055	0	0.228	-2	0	-4	3	0.46	0.46	0.07	0.68	-3	-1	-4	0
	May	0.006	0	0	0.001	-2	0	-3	2	0.15	0.09	0.02	0.21	-4	-1	-6	3
	June	0.013	0.001	0	0.006	8	0	-5	1	0.14	0.12	0.04	0.18	204	-1	-4	2
	July	0.02	0.001	0	0.003	22	0	-1	4	0.16	0.05	0.02	0.17	11	0	-1	5
	August	0.001	0	0	0.001	2	0	0	1	0.08	0.02	0.01	0.07	2	0	-1	1
	September	0	0	0	0	4	0	0	1	0.01	0	0	0.02	3	0	-1	1
Scenario 2	March	0.046	0.002	0	0.077	0	0	0	0	0.4	0.22	0.1	0.57	0	0	0	0
	April	0.114	0.056	0	0.228	4	1	-4	4	0.46	0.45	0.05	0.7	-5	-1	-4	1
	May	0.006	0	0	0.001	-2	0	-1	5	0.15	0.09	0.02	0.21	-4	-1	-6	4
	June	0.013	0.001	0	0.006	5	-1	-6	1	0.14	0.12	0.04	0.18	248	0	-4	3
	July	0.02	0.001	0	0.003	13	0	-2	3	0.16	0.05	0.02	0.17	8	1	-1	5
	August	0.001	0	0	0.001	1	0	-1	1	0.08	0.02	0.01	0.07	2	0	-2	1
	September	0	0	0	0	2	0	-1	0	0.01	0	0	0.02	2	0	-1	1
Scenario 3	March	0.045	0.002	0	0.075	-1	0	-1	0	0.42	0.22	0.1	0.61	4	0	0	5
	April	0.114	0.054	0	0.225	143	0	-2	12	0.46	0.46	0.04	0.71	-3	0	-5	2
	May	0.006	0	0	0.001	5	-2	-4	1	0.16	0.09	0.02	0.22	-3	0	-5	4
	June	0.013	0.001	0	0.006	3	-1	-7	2	0.14	0.12	0.04	0.19	249	0	-4	6
	July	0.02	0.001	0	0.003	9	-2	-3	1	0.16	0.05	0.02	0.18	8	2	-1	4
	August	0.001	0	0	0.001	-2	-2	-5	1	0.08	0.02	0.01	0.07	2	2	-1	6
	September	0	0	0	0	0	-1	-2	0	0.01	0	0	0.02	2	1	-2	2

Table A-25. SWAPP 25-yrs simulated monthly total phosphorous (TP) load and flow weighted mean concentrations (FWMC) values of mean, 50th, 25th, and 75th percentile, and the estimated cumulative impacts (change %) of simulated scenarios relative to baseline scenario at the outlet of Threehills Creek watershed.

Scenario	Month	TP Load (kg ha^{-1})				TP Load change (%)				TP FWMC (mg l^{-1})				TP FWMC change (%)			
		Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th	Mean	50th	25th	75th
Scenario 4	March	0.045	0.002	0	0.071	-1	0	-2	0	0.41	0.22	0.1	0.59	4	0	0	3
	April	0.114	0.052	0	0.226	80	0	-4	20	0.47	0.45	0.03	0.76	-10	-1	-5	2
	May	0.005	0	0	0.001	-1	-2	-11	0	0.14	0.09	0.02	0.22	-8	0	-12	2
	June	0.013	0.001	0	0.006	10	-2	-7	2	0.15	0.12	0.04	0.2	254	1	-6	5
	July	0.02	0.001	0	0.003	1	-2	-5	4	0.16	0.06	0.01	0.18	7	2	-2	6
	August	0.001	0	0	0.001	-3	-2	-6	1	0.08	0.02	0.01	0.07	0	0	-1	5
	September	0	0	0	0	-1	-1	-3	-1	0.01	0	0	0.02	-1	1	-3	2
Scenario 5	March	0.046	0.002	0	0.077	0	0	0	0	0.42	0.22	0.1	0.62	5	0	0	6
	April	0.113	0.056	0	0.235	82	-2	-5	26	0.47	0.45	0.03	0.72	-8	-3	-6	3
	May	0.005	0	0	0.001	2	-2	-11	3	0.15	0.09	0.02	0.22	-5	0	-15	4
	June	0.013	0.001	0	0.006	4	-3	-7	2	0.15	0.12	0.04	0.19	3	1	-5	7
	July	0.021	0.001	0	0.003	10	-2	-5	0	0.16	0.05	0.02	0.18	7	2	-2	7
	August	0.001	0	0	0.001	-6	-4	-8	-1	0.08	0.02	0.01	0.08	4	5	1	9
	September	0	0	0	0	-5	-4	-7	-2	0.01	0	0	0.02	4	4	-1	10