ADVANCING CUMULATIVE EFFECTS ASSESSMENT METHODOLOGY FOR RIVER SYSTEMS

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ABSTRACT

Increased land use intensity has adversely affected aquatic ecosystems within Canada. Activities that occur over the landscape are individually minor but collectively significant when added to other past, present, and reasonably foreseeable future actions, and are defined as cumulative effects. Existing approaches to cumulative effects assessment for river systems within Canada are ineffective. This thesis aims to improve the practice of cumulative effects assessment by evaluating current methodology for linking landscape change and river response over a large spatiotemporal scale. As part of this goal, I offer a framework for better incorporating science into current practices for cumulative effects assessment. The framework addresses the challenges involved in cumulative effects assessment, such as defining appropriate spatial and temporal scale, complex ecological and hydrologic pathways, predictive analysis, and monitoring. I then test the framework over a large spatiotemporal scale using a case study of the lower reaches of the Athabasca River Basin, Alberta. Three objectives are addressed: 1) changes in land use and land cover in the lower ARB for several census dates (1981, 1986, 1991, 1996, 2001) between 1976 (historic) and 2006 (current day) are identified; 2) linkages between landscape change and river water quality and quantity response are evaluated; and 3) results of the different methods used to link landscape stressors with stream responses are compared. Results show that the landscape has changed dramatically between 1976 and 2006, documented by increases in forest harvesting, oil sands developments, and agricultural intensity. Secondly, results suggest that linear regression tests combined with regression trees are useful for capturing the strongest associations between landscape stressors and river response variables. For instance, water abstraction and agricultural activities have a significant impact on solute concentrations. This suggests that water abstraction and agriculture are important indicators to consider when conducting a watershed cumulative effect assessment on a similar spatiotemporal scale. The thesis has strong implications for the need for improved water quality and quantity monitoring of Canada's rivers. The research provides a means of identifying appropriate tools for improved watershed cumulative effects assessment for scientists and land managers involved in the environmental impact assessment process and protection of Canada's watersheds.

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LIST OF ABBREVIATIONS

AENV	Alberta Environment
ARB	Athabasca River Basin
CANSIM	Canadian Socio-economic Information Management System
CEA	Cumulative Effects Assessment
Cl^-	Chloride
CWN	Canadian Water Network
EC	Environment Canada
EIA	Environmental Impact Assessment
GIS	Geographic Information Systems
HYDAT	Hydrometric Data
LOWESS	Locally weighted Scatter-plot Smoothing
LULC	Land Use and/or Land Cover
MAELs	Maximum Allowable Effects Levels
MODIS	MODerate resolution Imaging Spectroradiometer
Na^+	Sodium
NRBS	Northern River Basin Study
NRC	Natural Resources Canada
NTU	Nephelometric Turbidity Units
PAH	Polcyclic Aromatic Hydrocarbons
PFRA	Prairie Farm Rehabilitation Administration
RAMP	Regional Aquatics Monitoring Program
RMSE	Root Mean Square Error
SC	Specific Conductance
SEA	Strategic Environmental Assessment
	Sulphate
TDN	Total Dissolved Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
WCEA	Watershed Cumulative Effects Assessment
WRS	Worldwide Referencing System
WSC	Water Survey of Canada

Chapter 1- General Introduction

The progression of the industrial and green revolutions initiated a shift in land use practices within Canada. Increased intensity of land use practices such as agriculture and forestry, along with increased industrial development and urban growth have put substantial environmental pressure on Canadian river systems. Human manipulation of watersheds has raised concern for long term viability of aquatic ecosystems in regards to reduced water quality and depleted freshwater supply (Schindler, 2001). Furthermore, the accumulation of multiple stressors on the landscape exerts an even more complicated effect on riverine ecosystems because they interact in a way that is additive and synergistic over space and time (Spaling and Smit, 1993). To a degree, cumulative effects are considered under the *Canadian Environmental Assessment Act* in regards to proposed economic developments. However, it has recently been understood that Cumulative Effects Assessment (CEA), the systematic assessment of cumulative environmental effects, is ineffective (Duinker and Greig, 2006). This is because there are many challenges associated with conducting CEA.

Changes on the landscape can be a result of natural or anthropogenically driven events. The term “land use” is generally used to refer to human or economic activities on the land, while “land cover” refers primarily to the habitat or natural vegetation type present on the land, but can also refer to any type of feature present on the land, such as urban/municipal, cropland, forests or lakes (Lillesand and Kiefer, 2000; Gove at al., 2001; Turner et al., 2001). The landscape is typically made up of various combinations of land use and land cover (Ahearn et al., 2005). Although these terms are related, it is important to differentiate between them. Many studies refer to land use and land cover collectively, using terminology such as “land use/land cover” and “LULC” (LaGro and DeGloria, 1992; Ahearn et al., 2005). For the purposes of this thesis, the reference to natural land cover and human land use will be collectively referred to as LULC.

1.1 Problem Statement

LULC often takes the form of non-point land use practices which influence the health of river ecosystems in an indirect, secondary manner. For example, following the clearing of a single patch of forest, it is likely that there will be an increased rate of erosion which will affect the sediment load travelling to the stream or river, thereby increasing turbidity and subsequently altering water quality. This makes it difficult to measure and predict the numerous interactions occurring in upland areas of the landscape (MacDonald, 2000) and to determine their effects on river response in terms of biogeochemistry, quantity, and overall health. Watershed Cumulative Effects Assessment (WCEA) is commonly conducted at the local scale with a strong focus on local project developments. Local scale WCEA considers only a single upland development and its effects on the immediate riverine environment. Project-based WCEA efforts are thus futile because of the disregard for landscape influences occurring further away (Reid, 1998). CEA also raises the questions of responsibility for implementing CEA for river systems and ongoing monitoring once a project is in place. Project proponents and managers, governmental decision makers, and scientists alike all have an interest in carrying out effective CEA and river health monitoring, but their roles are less defined. The aforementioned challenges can be addressed by what is arguably the largest challenge associated with CEA; the lack of a structured, widely accepted scientific framework (Baxter et al., 2001). There is a need for the challenges associated with WCEA to be better addressed and managed. Instead, there are several differing paradigms for which CEA is considered. To date, there is little CEA literature and research that attempt to address cumulative effects of landscape stressors on the aquatic environment in a way that encompasses all recommendations for scientific measures, appropriate scoping, predictive analysis or ongoing monitoring. Thus, there is an opportunity for WCEA to be addressed on a comprehensive, holistic, scientifically-focused basis.

1.2 Over-arching thesis Objectives

The Canadian Water Network (CWN) is supporting a large Canadian research initiative aimed at addressing the issues of assessing, predicting, and managing cumulative effects of multiple stressors, and providing recommendations to significantly improve the practice of WCEA in Canada. In moving towards this goal, this thesis evaluates current methodology for linking landscape change and river response on a large spatiotemporal scale and proposes a new conceptual framework to better incorporate science into WCEA practice. Also, part of the framework is tested using a case study. Overall, this thesis can be used by the CWN CEA team to provide recommendations to shape WCEA practice in Canada.

1.3 Thesis Layout

The thesis is organized as a manuscript-style document. Chapter 2 provides a conceptual framework for better integrating science and practice for watershed CEA and river systems. The framework was further developed using a case example of the Athabasca River basin, one of the most adversely affected river systems in Canada. The framework offers recommendations for improved CEA in regards to monitoring, scoping and scale, physical hydrological and ecological influences, and impact prediction and scenario analysis. Chapter 3 provides a set of data, methods, and analyses that reflect the recommendations of the conceptual framework for improved watershed CEA presented in Chapter 2 based on the component involving identifying the effects of landscape change on river response. It involves the quantification of landscape change in the lower reaches of the Athabasca River basin, and attempts to link these changes to river response based on parameters included in a previous change analysis (see Squires et al., 2010) on river water quality and quantity. Chapter 4 offers a critical analysis of the conceptual framework presented in Chapter 2 using the findings from Chapter 3. Specifically, the framework is analyzed in regards to its robustness, utility, effectiveness, and performance. Recommended changes for future improvement to the framework and data requirements for performing WCEA are discussed.

Chapter 2- Bringing science into river systems cumulative effects assessment practice

2.1 Introduction

Anthropogenic disturbances on the landscape, combined with increasing water withdrawals and alterations to river systems are resulting in adverse effects to the sustainability of freshwater resources across the globe (see Schindler, 2001; Gleick et al., 2007). Environmental effects to river systems are largely cumulative in nature, caused by individually minor but collectively significant actions that accumulate over space and time. In a river systems context, cumulative effects result from changes to watershed processes, caused by the additive and synergistic interactions of multiple anthropogenic disturbances on the landscape (Reid, 1998). Almost all land use activities directly alter environmental parameters, including soil, topography, and vegetation, which, in turn modify the transport of water, sediment, organic matter, and pollutants that culminate in river systems (Johnson et al, 1997; Schindler, 2001). As such, river system health is largely a function of the types of interactions and processes that occur on the landscape within the boundary of the watershed.

Since the introduction of Environmental Impact Assessment (EIA) in Canada in the early 1970s, the assessment of cumulative environmental effects has been central to debate. Popularized in the early 1980s by the Canadian Environmental Assessment Research Council, Cumulative Effects Assessment (CEA) is now a requirement under the *Canadian Environmental Assessment Act,* and also under various provincial EIA laws and regulations. In principle, CEA demands that proponents examine the cumulative effects associated with their proposed development alongside relevant past, present and future projects. In a watershed context, there is a growing recognition of the need to assess the cumulative effects of development actions to river systems (e.g. Reid, 1998; Brismar, 2004; Schindler and Donahue, 2006). This need was reinforced at the 2010 Annual Meeting of the International Association for Impact Assessment in Geneva, where there were two sessions dedicated specifically to assessing cumulative impacts to watersheds. In practice, however, for various reasons (see Duinker and Greig, 2006; Schindler and Donahue, 2006; Gunn and Noble, 2009a, b), CEA in general is rarely done or rarely done well, which is problematic when intended for river system protection.

Arguably, the current state of CEA can be attributed, in large part, to the disconnect between the science of CEA and conventional EIA practice. Cumulative effects are frequently interactive or synergistic in nature and, as a result, the methodological complexity of CEA presents an overwhelming task to project-based EIA (Dubé, 2003). Further, according to Duinker and Greig (2007), the scientific community has done a poor job of developing the knowledge and tools needed for confident prediction of project cumulative effects, and the EIA community has done a poor job of approaching the cumulative effects of projects with a strong experimental design. Individually, both the science and the practice fall considerably short of effectively assessing and managing cumulative effects including those that affect watersheds and river systems, and a more integrated approach to CEA is needed (Cormier and Suter, 2008). The challenge is that there currently is not a single conceptual approach to CEA that is widely accepted by both scientists and managers (Squires et al., 2010), not to mention the number of challenges inherent to each the science and the practice that must be overcome in order to advance CEA science in EIA practice.

In response, this chapter presents a conceptual framework for better integrating science and practice for improved CEA in watersheds and river systems. First reviewed is the current state of EIA practice and CEA science, the key issues and challenges inherent to each, and the types of CEA science needed to improve current practice for Watershed Cumulative Effects Assessment (WCEA) – a concept based on Reid's (1993) notion of cumulative watershed effects. Then, the framework is developed using one of the most adversely affected river systems in Canada, the Athabasca River basin, as an illustrative case example and identify supporting methods for use in WCEA application that are complementary to both EIA practice and CEA science. The objective is to provide project proponents and land managers with a scientifically sound framework for the CEA of river systems.

2.2 State of CEA practice and science

In 2005, Agriculture and Agri-Food Canada conducted an EIA for a proposal to develop six irrigation sites to expand an existing irrigation system on the east shore of Lake Diefenbaker, Saskatchewan. The potential environmental effects of the irrigation developments were only vaguely described in the environmental impact statement. For instance, it was stated that some vegetation and soil disturbance as well as some erosion or re-distribution of sediment would result. It was also indicated that the use of heavy equipment could have a 'temporary' effect on air quality, while effects to wildlife were described as 'unlikely' (Agriculture and Agri-Food

Canada, 2005). In the cumulative effects section of the environmental impact statement, the report concluded that because mitigation measures would be conducted throughout the project, cumulative effects were considered 'not likely' to be significant. No other stressors to the aquatic system, on-site or off-site, previously assessed or not previously assessed, were considered in combination with the effects of the proposed development, and the impact predictions were not based on scientific design. The focus of the assessment was on mitigating the effects of the proposal to the point of acceptability and, in so doing, cumulative effects would somehow not occur.

As illustrated above, amongst the major shortcomings of the current approach to assessing the impacts of development on watersheds and river systems are that project-based EIA practices are limited in spatial and temporal scale, lack a sound scientific basis and, thus, do not fully encompass the interacting effects of multiple stressors over space and time (Baxter et al., 2001; Therivel and Ross, 2007). As a result, the cumulative effects of a project on broader watershed processes and river system condition go unchecked. The CEA process is conducted under EIA (Ball, 2011), arguably however, current practice EIA also lacks the proper science and quantitative methods for CEA particularly where projects are complex, or have complex interactions with the immediate environment and with other anthropogenic and natural disturbances (Kilgour et al., 2007). Such assessments are often inappropriately carried out on the basis of expert judgment or ad hoc lessons from elsewhere (Noble, 2008). For small projects, assessments are restricted in both time and resources to effectively integrate CEA science – if such projects are assessed at all. In the sections that follow I examine a number of key challenges to the current state of EIA practice and the science of CEA, and then attempt to bring together the practice and the science by using the Athabasca River basin to illustrate what is needed to move CEA forward for river systems.

2.2.1. Overcoming CEA to achieve project approval

Project proponents are required under the *Canadian Environmental Assessment Act*, and also under various provincial EIA laws and regulations, to include CEA in evaluating the environmental impacts of their projects; however, the ultimate goal for proponents is to obtain project approval. As such, CEA in practice frequently operates in such a way that meets the needs of project proponents in securing project approval, rather than assessing cumulative effects. For example, according to Kennett (2000), the province of Alberta formally excludes applications for individual oil and gas well sites from EIA under its *Environmental Protection and Enhancement Act*. As a result, oil and gas well development across the landscape, including the effects of service roads and trails required to install new well placements and maintain their operation, has become a significant ecological concern (Timoney and Lee, 2001). A similar case exists in southwest Saskatchewan, where a 1,940 km^2 land base is subject to the pressures of approximately 1,500 gas wells and more than 3,000 km of roads and trails (see Noble, 2008). Only 4 EIAs were completed for multi-well programs in the region, all of which concluded non-significant environmental impacts. While such an approach may serve the needs of development proponents, it fails to address the cumulative effects of development actions.

Under this sort of practice, scientific integrity is typically limited to the extent necessary to obtain project approval. While intentions for scientific integrity and quality CEA may be present, once a proposal enters the approval phase proponents and consultants may be content with providing assessments at levels that are considered to be simply ‘good enough’ (Warnback and Hilding-Rydevik, 2009). Consultants and impact analysts attempt to carry out CEA with scientific integrity but do not always follow through on this. For example, proponents rarely include details on matters for which little or no information is available in order to produce comprehensive and reliable assessments (Therivel and Ross, 2007). Ironically, this results in the exclusion of valuable and useful information that decision makers need in order to assess, monitor and manage cumulative effects on river ecosystems.

2.2.2 Differing ideals and other inconsistencies

Many of the challenges to CEA are due to the number of different CEA ideals and concepts. The CEA literature is somewhat convoluted, using various languages to define a single term or process, thus hiding commonalities and making the process of CEA seem needlessly more complex (Cormier and Suter, 2008). For example, one conceptualization of CEA divides it into stressor-based and effects-based approaches. Originally, the context for CEA was indeed stressor-based and focused on a proposed development, the predictions of project related effects using the current local environmental conditions as a baseline for comparison (Squires et al., 2010). However, while stressor-based CEA provides a predictive component for CEA, if a project is complex or produces complex effects, it would be difficult to predict true biological outcomes (Kilgour et al., 2007). Stressor-based CEA is also project specific and does not account for interacting effects on a broader spatial scale (Baxter et al., 2001); hence, the argument for effects-based CEA.

Effects-based methods take a more regional approach to CEA by focusing on the accumulated state of the existing environment and trying to identify unknown sources of stress and their interactions over a broad spatial scale (Culp et al., 2000). Amongst the main shortcomings to effects-based CEA, however, is that it lacks a predictive component in that the source of stress that has caused a particularly adverse effect is identifiable only *after* the effect has been measured (Dubé, 2003). Effects-based CEA involves heavy synthesis of field data, which can be expensive to collect in terms of time and money, and does not fit into the time restrictions of project assessments (Dubé and Munkittrick, 2001). Arguably, a solely effects-based CEA is a poor fit for project assessments because there would need to be a project in place before any effects could be measured.

Another conceptualization of CEA is Strategic Environmental Assessment (SEA). Under this approach, the division of stressor and effects-based CEA, or pitting one against the other, is seen as counterproductive to good-CEA (see Harriman and Noble, 2008). With EIA driven approaches, emphasis is on assessing the cumulative effects of individual and multiple developments over broad spatiotemporal scales, whereas SEA driven approaches emphasize the CEA of initiatives, plans, and opportunities and is objectives-driven and geared towards decision-making (Harriman and Noble, 2008). CEA is used in the decision-making process mainly to present and examine alternative outcomes in order to derive the most desirable course of action, and to facilitate a planning approach to address cumulative effects (Cooper, 2004; Harriman and Noble, 2008). However, Warnback and Hilding-Rydevik (2009) note that not all jurisdictions have SEA requirements and, where they do exist, there is not always a requirement that cumulative effects be addressed.

These different ideals and concepts introduce conflicting and diverging methodologies, which is counterproductive to ensuring good CEA. Such discipline-specific examinations of environmental responses commonly result in fragmented analyses that mask connections to overall environmental health (Griffiths et al., 1998). As a result, CEA literature is very diverse and lacks commonality, which can be frustrating and confusing for regulators, scientists, and proponents alike to gain a substantial understanding of how to effectively assess cumulative effects of projects in watersheds on river systems.

2.2.3 Multiple pathways and other exogenous factors

Much of the difficulty involved in CEA for river systems is attributed to the diversity of biological and physical processes and their interactions influencing land use impacts (Reid, 1998). Areas of disturbance on the landscape, such as agricultural production (i.e., pesticide and fertilizer input), wetland drainage, and impervious surface area may or may not be hydrologically connected to river bodies by surface pathways, but such disturbances do influence the movement and quality of water within a watershed. For example, following storm events, sediments, nutrients, and other pollutants are flushed from the soil and flow into river systems, magnifying the effects on water quality (Preston and Bedford, 1988).

In other words, environmental impacts are not only additive, in which individual projects contribute incremental levels of disturbance at the larger scale, but also synergistic in which the total effect of interactions is greater than the sum of effects of individual processes (Parker and Cocklin, 1993; Spaling and Smit, 1993; Piper, 2001; Seabrook, 2006; Gunn and Noble, 2009a,b). Many physical interactions occur simultaneously over time and space, which adds further complexity to understanding the science of CEA. For instance, deforestation alters ecosystem structure, function and composition, affects physical, chemical and biological ecosystem processes, energy flows, water flows, and nutrient and sediment transport in rivers (Boyle et al., 1997). Transport mechanisms, multiple linkages (pathways), source and sink phenomena, migratory patterns, and recovery rates, though challenging to the science of CEA, must also be understood by managers and integrated into CEA if cumulative effects to watersheds and river systems are to be understood and managed (Beanlands and Duinker, 1984).

2.2.4 Problems of scale

Arguably, the largest and most persistent challenge to CEA is scale. Cumulative effects are the results of multiple activities in space that persist over time (MacDonald, 2000). Quite frequently, however, CEA is performed at the spatial scale of the individual project, which is characteristic of project-based EIA (Dubé, 2003). This scale is inappropriate for CEA because individual projects contribute only a small amount of stress to specific valued resources when considered next to the interacting processes that occur among multiple disturbances (Spaling and

Smit, 1993; Duinker and Greig, 2006). The challenge is that as the scale increases, some of the more local issues (e.g. project specific perturbations) may fall out and others (e.g. landscape-scale disturbances) are likely to become more important (Therivel and Ross, 2007). This is challenging to CEA in river systems as not all watershed processes play out at the same spatial (or temporal) scale. Geographic boundaries for assessments should thus be defined based on the processes that control the sources of stress, so as to ensure that the spatial scale is not specified as too large or too small (MacDonald, 2000). A multi-scale approach will help ensure that the same issues can be revisited, where needed, not only at different tiers but also at different spatial scales (see João, 2007).

Historical conditions and environmental trends in river systems are also important to CEA, as well as the consideration of predicted future conditions and scenarios (Baxter et al., 2001; Therivel and Ross, 2007). In order to make predictions and consider future scenarios, however, decision-makers need plans for future developments in a watershed, which, problematically, are quite difficult to obtain either because these plans are not available or because there is uncertainty as to the likelihood of implementation (Duinker and Greig, 2006). The proposed Cheviot Coal Mine in Alberta, Canada, serves as an example. CEA consultants for the mine had developed what was considered to be a very good vegetation survey and could have suggested which vegetation would have been affected by the mine, but plans for future forest harvest plans were not available to them, so, consequently, CEA lacked a predictive component of future effects (Therivel and Ross, 2007). Added to this uncertainty is that there are lengthy time lags between an action and effect (Reid, 1998) and, in some extreme cases, some changes may not be observable for decades (Rogers and DeFee, 2005).

2.3 CEA science for river systems: The Athabasca River basin

In the following sections I propose a conceptual framework for integrating CEA science into practice for river systems. Attention is focused on the fundamental components of CEA science that are central to any application, namely scoping and scale determination, baseline assessment, predicting cumulative impacts, and ongoing monitoring and evaluation. For comprehensive guidance on CEA frameworks at the regional scale I refer the reader to Harriman and Noble (2008). The principles are developed and discussed based on the case of the

Athabasca River basin, Alberta, though they are sufficiently broad to be applicable in other jurisdictions and river systems contexts.

The Athabasca River (Figure 2.1) originates in the Columbia Ice fields in the Rocky Mountains of Alberta, Canada, and drains into Lake Athabasca in northeast Alberta and northwest Saskatchewan, covering an area of approximately 157,000 km^2 (Culp et al., 2005). The basin encompasses a variety of land cover types such as sub-alpine, grassland, sub-boreal and boreal forest. In addition to urban settlement, the basin is exposed to a wide range of land use activities including large-scale agriculture, forestry, and petroleum extraction, making it amongst the most stressed river systems in Canada (Schindler and Donahue, 2006; Keepers of the Athabasca, 2008). Combined with point source sewage discharge from urban settlement, and non-point source urban and agricultural runoff, there are two bleached kraft pulp mill operations discharging to the river system (see Squires et al. 2009). Most notably, however, the basin is home to the Alberta oil sands, the second largest source of oil in world, with proven reserves of 170 billion barrels of oil sands bitumen and up to 315 billion barrels should favourable economic conditions prevail and new technologies become available (Government of Alberta 2010). Schindler et al. (2007) report that 2 to 4.5 barrels of water is required to produce 1 barrel of oil from current oil sands mining operations.

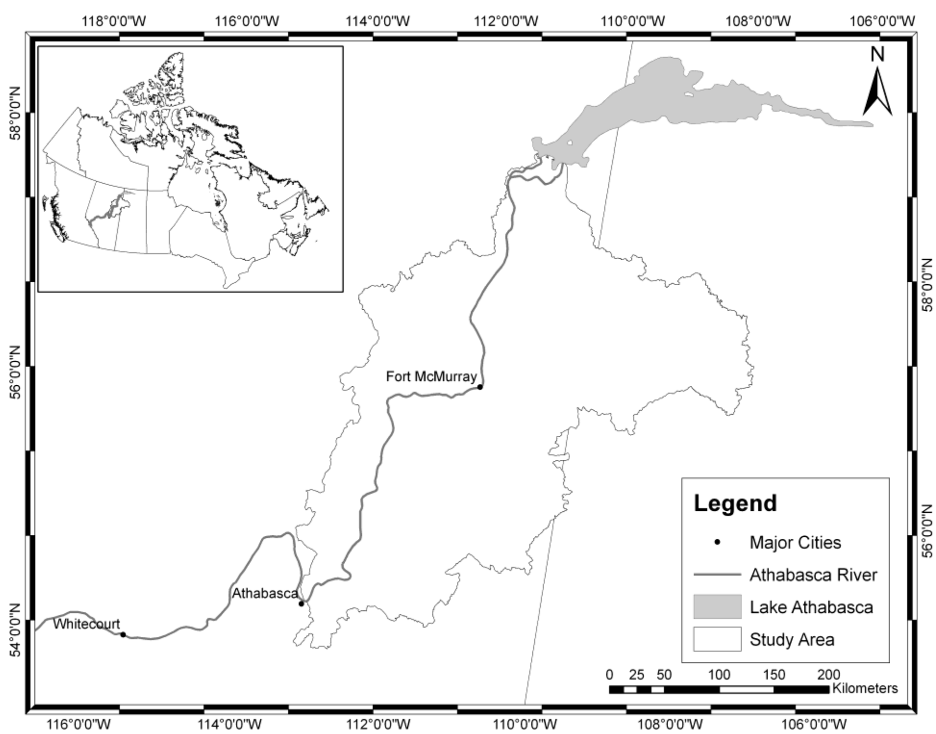


Figure 2.1. The reach of the Athabasca River under study, delineated because Squires et al. (2010) identified it as the reach that has experienced the greatest change in stream flows and water quality over the past 40 years. The length of the river reach within the study area is approximately 637 km. Inset shows the location of the Athabasca River within Canada.

2.3.1 CEA spatial and temporal scale

Defining an appropriate spatial scale for CEA for river systems depends on where active change is occurring in a watershed and what is the significance of those changes (see Lein, 2002). This helps to keep the assessment focused, ensuring that issues of concern are given priority (MacDonald, 2000). In the Athabasca River basin, the Regional Aquatic Monitoring Program (RAMP) was implemented in 1997 as a long-term monitoring program for the northern portion of the basin (i.e., north of Fort McMurray), where oil sands operations are concentrated. One of the main objectives of RAMP is to monitor aquatic environments in the oil sands area to detect and assess cumulative effects and regional trends in environmental parameters (Regional Aquatics Monitoring Program, 2002). Though a valuable monitoring tool, RAMP's spatial focus is solely on the area affected by oil sands operations. MacDonald (2000) stresses that it is important to define the spatial scale of an assessment in regards to the scale of the impacts, and because impacts to the Athabasca River system occur throughout the watershed, there is a need to conduct a CEA at a more regional or at least a sub-watershed scale.

2.3.1.1 CEA based on a river reach

Hydrologic processes occur over multiple scales and have different magnitude effects on river ecosystems. EIAs commonly define the spatial boundary of assessment at the individual project-scale, which is inappropriate for WCEA. Ideally CEA should occur at the regional and local scale as a collective (Dubé, 2003), but even what may be considered to be the regional scale is often defined at too small a boundary for good CEA. According to Squires et al., (2010), in order to account for accumulated changes in biophysical parameters of a river system, any CEA framework should consider change from river headwaters to mouth. However, the Athabasca River basin encompasses such a large spatial area that it would be impractical to define the entire watershed from headwaters to mouth as the spatial extent in a CEA for a development project. Instead, I argue that a river reach is a more appropriate spatial focus for WCEA so as to ensure that CEA science is manageable in the context of impact assessment practice. In the Athabasca system, the reach delineated as the study area in Figure 2.1 has been subject to development of a diverse range of land uses, thus I chose to focus on this reach for the case example. If CEA for a river system fails when applied to the area with the most landscape disturbance, or is not responsive to the range of stressors that exist in the most developed reach, it will make no difference if the rest of the river continuum is considered in CEA when projects are proposed. Thus, if a project were proposed for development in this area, I argue this spatial delineation to be appropriate for assessing potential cumulative effects.

Rivers are rarely hydrologically or hydrochemically connected to all land areas in their watersheds at all times, nor do all areas of the landscape contribute equally to the river or stream. Further, not all areas of a watershed are subject to anthropogenic land uses. Thus, it is important to identify areas of most probable influence on river biophysical properties. ‘Impact zones’ or ‘areas of influence’ have been explored as a way to define spatial extents of surrounding terrestrial influences to large rivers and thus may be an appropriate spatial unit to focus on when identifying and examining incremental and accumulating events responding from proposed developments (Johnson et al., 1997; Gove et al., 2001). Impact zone delineation depends on watershed size and the key hydrological processes and pathways that influence landscape-river interactions (Buttle, 2002). For example, Nadorozny (2009) showed that an appropriate spatial impact width for a large river, comparable in size to the Athabasca River, is up to 1-2 km on either side of the river. More generally, Gove et al. (2001) found that land use information from an intermediate (mesoscale) scale better predicted impacts to water quality than local riparian (microscale) and total basin scales (macroscale).

2.3.1.2 Temporal delineation

Definition of a temporal scale for CEA for river systems should consider how interactions from past, present, and future developments in the watershed might influence the effects from any proposed project. The common approach, however, is to define only the current environment as the basis for CEA (Dubé, 2003). But because the existing environment is a result of the influence of past actions, this approach assigns the effects of past and present actions to the current condition rather than to contributions to cumulative change (McCold and Saulsbury, 1996). CEA temporal scales must reflect how past actions and incremental changes on the landscape have influenced the present, and the resulting long- and short-term effects as well as future effects (Therivel and Ross, 2007). An appropriate CEA temporal scale is necessary so as to ensure that accumulating past, present and foreseeable future impacts are accounted for when assessing the cumulative impacts of project development on the river system (Spaling and Smit, 1995).

In terms of past effects, McCold and Saulsbury (1996) suggest that an appropriate baseline is the time in the past when the valued ecosystem component or indicator of concern was most abundant or least affected by human action. In the context of the Athabasca River basin, for example, oil sands were first developed intensively in the late 1960s (Timilsina et al., 2005). Agricultural presence in western Canada has been intensifying rapidly since the 1950s (Hobson et al., 2002), further exacerbated with the advent of the green revolution in the early 1970s. Population pressures have also increased considerably since the 1970s. As such, an appropriate temporal baseline for the Athabasca River basin, based on the intensity of land use and large-scale disturbances in the watershed, would appropriately extend to the late 1960s, prior to when the basin began to experience a large increase in the intensity of land use development. This past analysis is important in determining how the landscape has changed, to what degree these changes have impacted the current state of a river system, and to understanding the implications of future change under different assumptions about the current and future state of development. Of course, past analysis, specifically the selection of an appropriate baseline, is also influenced by data availability.

2.3.2. Constructing a CEA baseline

Only once the spatial and temporal scales are delineated can a CEA baseline be constructed. CEA for river systems is meant to evaluate environmental change as a basis for understanding environmental effects, which, according to Reid (1998), can only be measured and compared to a baseline of unchanged or ‘natural’ condition. The purpose of a baseline is thus to establish knowledge of the key assessment components and characteristics of the region that can be monitored over space and time for the purposes of change assessment, projected forward as part of a trends analysis or impact prediction, and used as the future conditions against which future scenarios can be assessed (Harriman and Noble, 2008). Constructing a CEA baseline for river systems then involves two fundamental components: the scientific understanding of the river system and identifying appropriate CEA indicators, and determining changes in landscape patterns and processes in the watershed that can be related to conditions of the aquatic environment. As an aside, data are likely to contain some degree of natural variability, and must be adjusted for or de-trended to ensure the baseline definition ensures the separation of natural and man-made variability.

2.3.2.1 CEA science and river system response

It is necessary to focus the CEA baseline on those components that are useful as scientific indicators of regional change, but at the same time ensuring that those indicators are useful for, or responsive to, project-level induced effects (Harriman and Noble, 2008). Numerous models and indices are used to assess cumulative effects to river systems, such as the index of biotic integrity to assess the condition of aquatic ecosystems, which rely on measures of faunal assemblages, mostly fish (Teels et al., 2006). These indices, while useful, are applied to a proposed development using expert judgment to populate the model and evaluate different criteria. This approach is problematic, however, in that indices are standardized for the most sensitive species, which may or may not be present in a given river system (Kilgour et al., 2007), not to mention that it is unlikely that more subtle cumulative effects would be identified using species and indices, specifically those effects characterized by space-time lags, path dependence and non-linear relationships (Therivel and Ross, 2007).

Amongst the main indicators used for CEA of river systems are biota presence and abundance (i.e. fish, algae/periphyton, benthos) and various measures of water quality and quantity that are used as a proxy to indicate likely impacts on biota – they are easier to measure and are useful in broader scientific methods such as experiments and models (Adams, 2003). In the Athabasca case, for example, the water quality parameters of concern include chloride, dissolved nitrogen, total phosphorus, polycyclic aromatic hydrocarbons, total organic carbon, turbidity, conductivity, dissolved sodium, trace metals and dissolved sulphate (Northern Rivers Basin Study, 2002; Keepers of the Athabasca, 2008; Squires et al., 2010). Reduced river flows are also of significant concern as a result of the bitumen extraction process in the oil sands region as, according to Squires et al. (2010), more water is being taken out of the river at the lower reaches of the basin than can be returned. Not only do lower flows threaten fish habitat and abundance, but concentrations of water quality parameters are also subject to increase, pending intensified development, further degrading water quality.

Collecting data for such indicators is, of course, costly in terms of time and financial resources. However, for CEAs conducted over large spatial scales, such as a river reach, much data are readily available from secondary sources, including various land use studies, project impact assessments, and government-funded monitoring programs. In the Athabasca, for example, real time and archived flow data are publically available from seven Water Survey of Canada HYDAT stations along the river.

2.3.2.2 CEA landscape inputs

Understanding changes on the landscape is central to understanding cumulative effects to river systems. The objective in WCEA is thus to identify spatial and temporal patterns and trends of disturbance in the watershed that can be projected forward under different assumptions about growth and development, and to relate these disturbances to responses in the river system. Landscape metrics are measurable units of landscape composition and act as a surrogate for change, thus allowing for the description and quantification of spatial patterns and ecological processes over time and space (Turner et al., 2001). Landscape metrics provide a measure of many different landscape activities such as amount of anthropogenic land conversion, riparian zone habitat, and historical land cover, to name a few (Johnson and Gage, 1997; Gergel et al., 2002).

An advantage to using landscape metrics for WCEA is that they act as indicators of responses by affected systems to cumulative change (Vos et al., 2001). These indicators can be used in regression and correlation analyses to provide an indication of cause-effect relationships between cumulative change and cumulative effects. In the Athabasca case, the basin is characterized by many different land uses. However, the landscape metrics that hold the most explanatory power in terms of various aquatic indicator responses would be those most useful to CEA. These metrics might include, for example, proportion of forested land, number of forested and non-forested patches, edge density, road-to-stream distance, impervious surfaces, and size of industry footprint. Multivariate analyses, in particular regression tree analysis, could prove useful to examine trends between various landscape metrics and aquatic response indicators, and to identify the variables with the most explanatory power for cumulative effects to the aquatic system (De’ath and Fabricus, 2000).

One caveat to landscape metrics, however, is that because natural systems are dynamic, diverse, unique entities, it is, arguably, impossible to determine cause-effect relationships (Johnson and Gage, 1997). Such analysis is thus indicative, but not descriptive, of cause-effect (Ahearn et al., 2005). That said, due to the unique nature of ecological systems, testing for cause-effect relationships using landscape metrics are considered to be the 'next best' method of quantifying change on the land that results in changes to rivers.

2.3.3 CEA impact prediction

CEA for river systems requires a predictive component to allow for proponents and land managers to identify and prepare for different future outcomes. In reality, however, natural systems are highly unpredictable, so a study design cannot be so rigid that it does not account for alternative outcomes (Boyle et al., 1997). In this regard, Duinker and Greig, (2007) argue not only for predictive components to be included in CEA, but to include the exploration for alternative scenarios.

2.3.3.1 Scenario-based analysis

In any single WCEA application, a range of alternative future scenarios and resource pressures in the watershed must be explored. Ideally, these scenarios should relate to both externally driven change, such as climate change and changes to economic conditions, and also to internally driven scenarios linked directly to land use and local drivers of change within the watershed. I focus my attention here on those internally driven scenarios. For example, within the lower reaches of the Athabasca River basin oil sands operations are proving to be problematic in terms of cumulative effects to water quantity (Woynillowcziz et al., 2005; Squires et al., 2010). Large amounts of water are withdrawn from the river on a daily basis for the bitumen extraction process, and less than 10% is returned to the river (Woynillowcziz et al., 2005). Given the increasing intensity of oil production in this region, CEA for any proposed development of the landscape must consider the scenario that flows will continue to decline based simply on existing trends. Other scenarios must consider the complex interactions between oil sands production and indirect effects to water quantity and quality. For example, the effects of increased oil production on the social environment translates to more jobs, and a subsequently larger human population in the watershed and increased demand for, and impacts to, water resources when workers and their families relocate to the region. Due also to economic spinoff opportunities and further increased demand for water resources, both water quantity and quality are of significant concern.

There is currently no scientifically agreed upon mechanism for CEA prediction for river systems. Due to the difficulties associated with time, cost and lack of capable personnel for executing ground-truthing, proponents typically have favoured small-scale experiments as the basis for impact prediction, in those cases where science is actually used in CEA (Schindler, 1998). However, the difficulty in performing a broader spatial scale assessment has recently become an easier task, as the development of digital data and digital map products have provided a means of conducting CEA at the regional scale in a cost-effective and timely manner (Smit and Spaling, 1995; MacDonald, 2000; Muller et al., 2007; Nitschke, 2008).

Geographic information systems (GIS) provide a valuable CEA tool for scenario-based analysis as it lends itself well for assessing spatial overlaps, spatial distributions of environmental change, and manipulation of 'what-if' scenarios to prepare for a number of potential alternate environmental consequences (Noble, 2008). In other words, GIS provides a means of quantifying and characterizing landscapes and landscape patterns in a watershed, which can be related to measurements of adverse environmental responses to yield statistical relationships between project developments and cumulative effects (Bolstad and Swank, 1997; Ahearn et al., 2005). Valuable data sets for use in CEA for river systems include physical landscape datasets such as hydrology, digital elevation models, forest fires, populated places, land use surveys, agriculture, and road density. Additionally, census data linked to attribute tables of GIS layers to yield non-ancillary information are particularly valuable, such as dollars spent on fertilizer for agricultural land as a landscape 'metric' or indicator of effects to water quality.

2.3.3.2 Determining the significance of cumulative effects

A fundamental question in WCEA is when do effects to the river system indicate an irreversible level of impairment? Thresholds are commonly used to classify effects as 'acceptable' or 'unacceptable' (Ziemer, 1994; Kilgour *et al*., 2007). For WCEA, it is important to define what a river system is most sensitive to, and to ensure that the effects of landscape development do not irreversibly impact the river system. This task does not come easy, as thresholds, especially for rivers, are difficult to determine – scientifically and socially (Duinker and Greig, 2006; Squires et al., 2010). While there is limited guidance for appropriate thresholds determination in a river system, thresholds should at least be defined based on ecologically relevant spatiotemporal scales. In many instances, thresholds may be set based on maximum allowable effects levels (MAELs), which serve as benchmarks against which environmental effects, compliance, performance, and baseline change can be evaluated. Without setting thresholds for cumulative effects for river systems, assumptions about impacts from future developments may be underestimated (Dubé, 2003). Such thresholds and MAELs are not likely to pre-exist and will need to be determined as part of the scoping process based on consultation with regulatory agencies and on levels of 'socially acceptable' or 'ecologically tolerable' change as identified by stakeholders and the scientific community.

Recent catchment classification efforts, which focus on defining, understanding and predicting watershed function (see McDonnell, 2007; Tetzlaff et al., 2008), hold some promise for determining when effects to the river system indicate an irreversible level of impairment. Although not intended for this purpose, catchment classification could be useful to WCEA in that it is at least partially directed toward detailing important watershed functional traits. In general, if the sensitivity of a river to a suite of specific aquatic response indicators were known *a priori,* and trends between the aquatic response indicators and specific landscape metrics were already established through quasi cause-effect relationships, project proponents could simply evaluate how their proposed activities affect the chosen landscape metrics and thus would infer effects on the river system. Once refined, catchment classification could be used to easily identify which specific aquatic response indicators are most sensitive to change in response to the additive and synergistic interactions of multiple anthropogenic disturbances for a given landscape.

2.3.4 CEA monitoring

Long-term monitoring, feedback and learning are essential to CEA for river systems. In the Athabasca basin, the Northern River Basin Study (NRBS) was developed by the governments of Canada, Alberta, and the Northwest Territories in 1991 and continued to 1996 (Northern River Basins Study, 2002). The overall objective of this study was to provide an understanding of how anthropogenic developments impacted the ecology of the Peace, Athabasca, and Slave Rivers. The NRBS used a CEA approach that was developed based on measurements of fish response to stressor exposure (Culp et al., 2000). The study generated a multitude of information and recommendations for improved environmental assessment, which were subsequently used to inform EIA and impact management practice for one pulp and paper industry in the area. Specifically, the proposed Alberta-Pacific pulp mill near the town of Athabasca, Alberta, raised concern about low dissolved oxygen levels in the Athabasca River. This prompted the recommendation that all pulp mills in the area commit to monitoring dissolved oxygen in the river, and collectively take action if levels became too low (Wrona et al., 2000; Therivel and Ross, 2007). This example shows how CEA science can be included in project impact assessment practice. However, despite the outcomes and provisions of data from the NRBS, it was limited to a 5-year window and, as such, has provided only a limited contribution to understandings of incremental changes in the Athabasca basin to date.

Needed then, are long-term commitments to ensure successful CEA monitoring. However, this raises the question of who is responsible for monitoring in a watershed. Therivel and Ross (2007) argue that project proponents are responsible for monitoring the effect of their developments, but not the effects of others. Similarly, Harriman and Noble (2008) suggest that project proponents should be made responsible for monitoring, but not beyond the scale of their associated development. Therivel and Ross (2007) further argue that cumulative effects require cumulative solutions that involve the combined action of multiple authorities. There is an opportunity for the collective efforts of proponents and government to protect river systems. Arguably, proponents should be made responsible for monitoring the aquatic system to which their development immediately affects in regards to water quality and quantity standards and frequency of sampling, under the direction of government through standardized terms of reference for project EIAs and monitoring protocols. In response, governments, then, should use the information generated from the science to enhance and enforce new environmental policy and regulation. Further, governments should provide appropriate spatial data about landuse to scientists who can use them to generate landscape metrics and models to link with aquatic systems for improved WCEA understanding, and help to reduce the disconnect between CEA science and EIA practice.

2.4. Conclusions

River systems are sensitive to changes on the landscape. River system responses to landscape change are further exemplified by the interactions of surrounding changes on the landscape that accumulate over time and space. Watershed cumulative effects assessment examines the interactions between landscape changes that accumulate over time and space and river system response, and examines the outcomes of these interactions under different futures of growth and development in the watershed. Currently however, CEA for river systems is proving to be ineffective due, in large part, to the disconnect between science and practice. Overcoming challenges set forth by scaling issues, diverging views, policy and legislation, and complex ecological pathways is in itself the main challenge for those who try to carry out effective CEA. The proposed conceptual framework provides a potential way forward, and a point of discussion, to help address the disconnect between CEA science and EIA practice for river systems.

My recommendations for improved WCEA outline that the practice must be broadened to include appropriate quantitative methods, and that the science must be flexible enough to ensure that CEA is carried out without delay. A concise description of the type of scientific information needed and the approaches that should be considered in project and watershed CEA undertakings are crucial to the decision-making process and undoubtedly valuable components for improved WCEA. Ideally, implementation of WCEA rests on a multi-stakeholder approach. I suggest that governments must assume leadership in WCEA: establishing objectives and thresholds based on sound scientific guidance; ensuring that point-specific project-based EIAs are relevant to evaluating and monitoring cumulative effects at the broader watershed scale; and providing direction to project specific EIAs through terms of reference set based on knowledge gained from broader WCEA programs. Project proponents may also have to bear an additional cost, meeting not only their EIA obligations but also being engaged in broader cumulative effects monitoring programs. Scientists need to do a better job of providing useful metrics and tools for both assessing and predicting impacts within time frames that suit CEA. Lastly, co-operation between scientists, proponents, and regulators is needed in order to properly incorporate the science into CEA practice for ensuring the sustainability of watersheds and river systems.

Chapter 3- Landscape change and its effects on water quality and quantity response

3.1 Introduction

The health of a river is influenced by activities that occur within the boundary of its watershed. Anthropogenically driven landscape alteration can have adverse effects on water quality and quantity. Expansion and increased intensity of land use practices such as agriculture, forestry, urban growth, and industrial development have been shown to adversely affect stream health (Xian et al., 2007; Scrimgeour et al., 2008; Kelly et al., 2010). The sources of these pollutants can be point or diffuse in nature and occur over broad spatiotemporal scales, acting in a manner that is cumulative with other stressors on the surrounding landscape.

Cumulative effects assessment (CEA) in Canada is outlined under the *Canadian Environmental Assessment Act*. However, current approaches to CEA for rivers are ineffective due to actions such as poor scaling definitions and inadequate quantitative methods to assess landscape change and their effects on river response (Preston and Bedford, 1988; Duinker and Greig, 2006). It has been acknowledged in the literature that watershed cumulative effects assessment (WCEA; sensu Reid, 1993) should be conducted over broad spatiotemporal scales, shifting away from traditional project-based methods, and include strong scientifically based methods. Chapter 2 outlines the necessary measures to be taken to improve WCEA including suggestions for how to link landscape change to river response. Testing of various methods for linking landscape change to river response is needed across large spatiotemporal scales.

There are a multitude of activities that interact over the landscape throughout time, and have the potential to indirectly or directly affect water quality and quantity. It is important to utilize appropriate methods to identify the key landscape stressors and understand their associations with observed river response. Many quantitative methods are available for identifying key landscape drivers and their change over space and time, but must be chosen depending on the study design and the questions being asked. Large-scale landscape change studies for the purposes of addressing WCEA have recently been made more viable due to the development of powerful and affordable GIS and remote sensing technologies (Griffith, 2002; Lein, 2002; Gergel, 2007). The spatially and temporally expansive natures of cumulative effects are effectively handled by GIS given its capacity to handle large volumes of spatially referenced data for which there are records over long periods of time (Parker and Cocklin, 1993). Landscape mapping, often accomplished with GIS, is further complemented by the use of remote sensing technologies, as it provides a means of studying changes in the physical environment over space and time. Appropriate scientific methods for linking landscape change to river response are important to WCEA in order to understand land-water relationships. Linking landscape change to river response often involves the use of statistics. Simple linear, step-wise, and multiple regressions are used in ecological studies to describe the effects of landscape stressors on river response. More recently, regression trees have increasingly been used by ecologists to explain the variation of a single response variable by one or more explanatory variables (De'ath and Fabricus, 2000). Regression trees could prove to be valuable because of their predictive component, which allows the analyst to identify which landscape practices will have the strongest impact on river response, as it is not feasible to measure all variables evenly across the landscape. Other methods for linking landscape change to river response include principle components analysis (Fitzpatrick et al., 2007) and predictive modeling (Matheussen et al., 2000; Jorgensen et al., 2009). These methods have been extensively applied in studies examining the effects of land cover change on river response and should not be ruled out from such problems due to their proven value in identifying landscape- river linkages.

The Athabasca River Basin, Alberta, Canada is exposed to a multitude of land use activities, making it one of the most adversely affected watersheds in Canada (Schindler and Donahue, 2006; Keepers of the Athabasca, 2008). Squires et al. (2010) conducted a change analysis of water quality and quantity across the Athabasca River, from headwaters to mouth between historic (1966-1976) and current day (1996-2006) time periods. They found discharge had decreased by up to 30% for the lowest reaches of the river. They also showed that dissolved sodium, sulphate, chloride and total phosphorous concentrations were significantly higher in the current time period along most reaches. It was hypothesized that the cause of these changes was increased water abstraction. It is common in the literature to see change assessments conducted on various water quality or quantity variables, and to then assume changes are the result of landscape change. However, most studies fail to rigorously test such hypotheses, despite the numerous methods available for doing so.

The goal of this chapter is to evaluate the framework presented in Chapter 2 for linking LULC change and river response on a large spatiotemporal scale using a test basin – the lower reaches of the Athabasca River, Alberta. To meet this goal, three objectives are addressed: 1) changes in LULC in the lower ARB for several census dates (1981, 1986, 1991, 1996, 2001) between 1976 (historic) and 2006 (current day) are identified; 2) linkages between LULC change to river water quality and quantity response are evaluated for the same variables Squires et al. (2010) identified as most affected; and 3) results of the different methods used to link landscape stressors with stream responses are compared.

3.2 Methods

3.2.1 Study area

Discussed briefly in Chapter 2, the lower reaches of the Athabasca River have been subject to increased economic exploitation over the past 30 years (Culp et al., 2005), thus have been chosen as the study area. The lower reaches of the Athabasca River belong to the Great Slave Lake drainage area (coded as 07) and are further divided into two sub drainage areas, the Central lower (coded as 07C), and the Lower (coded as 07D), based on hydrologic connectivity (Natural Resources Canada, 2003). The watershed is an ~88 000 km^2 area located north of the town of Athabasca, Alberta and south of the Athabasca Delta and Lake Athabasca (Figure 3.1). Most of the watershed area falls within the province of Alberta, but a small portion lies in the province of Saskatchewan. The lowermost areas, near Fort McMurray and the Athabasca Delta, are located primarily in the boreal forest, while the uppermost areas, near the town of Athabasca, are located in grassland-boreal transitional forest area (Keepers of the Athabasca, 2008). Thus, the study area has land cover types such as grassland, sub-boreal, and boreal forest (Culp et al., 2005). The major cities in the lower reaches of the ARB are Athabasca and Fort McMurray, which had populations of 2 575 and 47 705, respectively, in the 2006 census (Statistics Canada, 2006). The lower reaches of the ARB are exposed to a wide range of land use practices; predominantly advanced resource extraction such as natural gas production, oil sands developments, and forest harvesting (Culp et al., 2005; Government of Alberta, 2009). Other land uses include agriculture (cropland and livestock), and coal mining, as well as an expanding urban population. Fires occur regularly in forested areas.

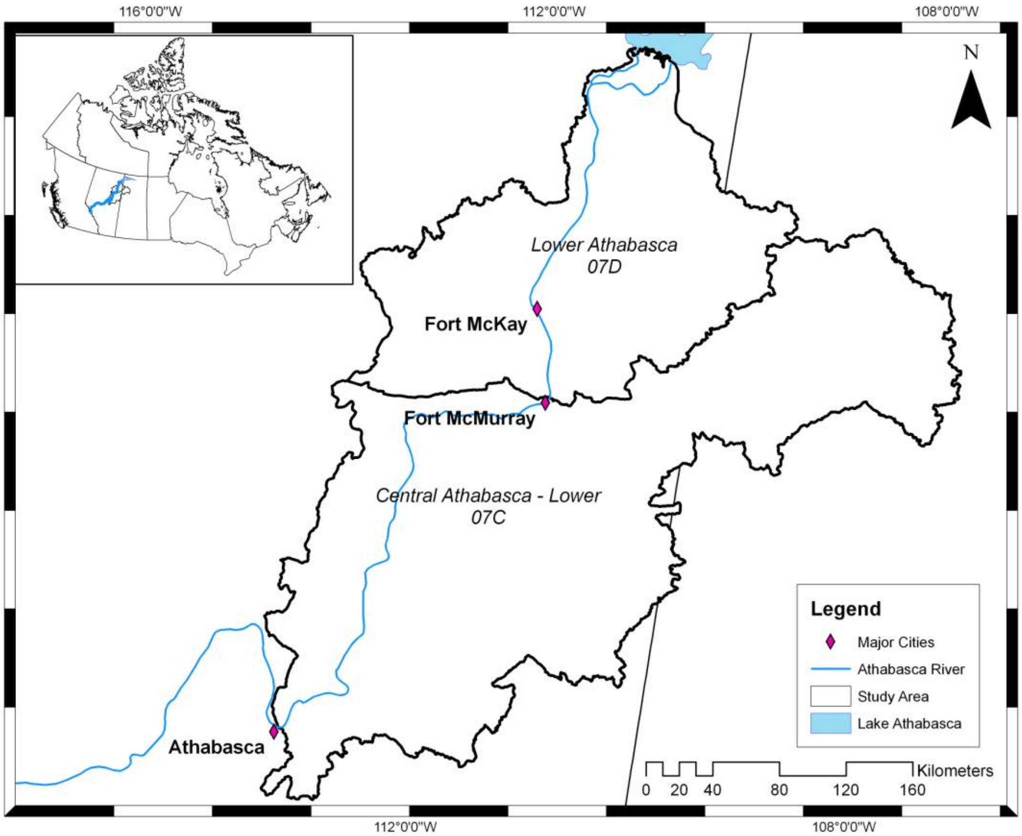


Figure 3.1. The lower reaches of the Athabasca River basin, which have experienced the greatest change in LULC from 1976-2006.

3.2.2 Landscape change assessment

Satellite imagery was used to evaluate how the landscape has changed between 1976 and 2006. Landsat was chosen because it is the only sensor that was in operation in the early 1970s, and has a high spatial resolution, ranging from 30 to 60 m (Jensen, 2005). Furthermore, as of 2009, the entire Landsat archive is freely available to the public. Because Landsat has a high spatial resolution, the swath width, or footprint, is relatively small, at 185 km. Thus, multiple images were required to cover the entire study area for each date. The Landsat Worldwide Referencing System was used to determine which Landsat frames covered the study area (see Wulder and Seemann, 2001). Nine Landsat Multispectral Scanner (Landsat MSS) images were used for 1976, and eight Landsat Thematic Mapper (Landsat TM) images were used for 2006 (Table 3.1). Images were chosen for both time periods based on the criteria that they were at least 90% cloud free and were as close to the vegetation growing season as possible. However, concessions were made due to the presence of unavoidable cloud and haze cover, and sensor noise (i.e. line streaking). According to Gillanders et al. (2008), it is preferable to use imagery from an off-year rather than off-season due to phenological differences between seasons. For the 1976 time period, the number of images that were acquired in an off-year was seven. For the 2006 time period, the number of images that were acquired in an off-year was five.

Table 3.1. Dates and Landsat frame path and row numbers for 1976 (based on the Worldwide Referencing System one [WRS-1]) and 2006 (based on the Worldwide Referencing System two [WRS-2]).

WRS-1		Circa 1976	WRS-2		Circa 2006
Path	Row	Date (mm/dd/yyyy)	Path	Row	Date (mm/dd/yyyy)
42	20	9/21/1974	40	20	9/24/2006
44	20	5/25/1973	41	20	8/27/2005
44	21	8/16/1976	41	21	8/27/2005
44	22	9/21/1976	42	21	5/30/2005
46	19	8/2/1974	42	22	8/21/2006
46	20	8/20/1974	43	19	8/25/2005
46	21	7/20/1973	43	20	6/25/2006
46	22	9/26/1979	43	21	8/25/2005
47	20	8/3/1974			

Using PCI Geomatica version 10.3, images were georeferenced to a digital topographic map with the projection NAD 83 UTM zone 12. The root mean square errors of all registration points were <1 pixel. Radiometric correction for sensor error was only done for the 1976 images, as the 2006 images were of good quality and had little to no sensor noise. Sensor correction for 1976 images involved a series of moving window filters (3x3 and 5x5 pixel windows) to smooth the sensor noise and produce a streak-free image. Radiometric correction for atmospheric error was done on each image for both dates. Raw digital numbers were converted to radiance and then to reflectance following the procedures outlined in Chander et al. (2009).

A supervised classification was conducted on each individual image for each time period (17 image classifications in total) and included the following land cover classes: agriculture, coniferous forest, mixed forest, deciduous forest, clear cuts, industrial, urban/ impervious, water, no data (i.e. cloud, shadow), and burn scar. Land cover classification from advanced very high resolution radiometer (AVHRR) land cover data of Canada, circa 1992, produced by Natural Resources Canada was used to guide each classification, as well as the GIS datasets used in the thesis, such as forestry, digital topographic maps, and fires, that provided LULC information for 1976 and 2006 (described below). Before the classifications for each respective time period were mosaicked, an accuracy assessment was conducted on each individual image classification. The average accuracies for the 1976 imagery classifications and 2006 imagery classifications were 70% and 71% respectively. It is typically recommended that classification accuracies not fall below 85% (Foody, 2002). However, studies that meet this standard are often conducted over small spatiotemporal scales for which physical ground-truthing is possible and validation information (i.e. air photos) is abundant. Franklin et al. (2002a) explain that an accuracy of 75% is often achieved with classifications of large areas with the dominant classes being vegetation ones, such is the case with this study area. Classification accuracy also depends largely on the quality and availability of validation data (Franklin et al., 2002b). Because of the large spatiotemporal scale investigated here, as well as the spatially expansive vegetative cover, 70% accuracy is reasonable. Adjacent images had overlapping areas that slightly differed in their classifications. Therefore, the overlapping area was given the classification of the image with the highest classification accuracy. Last, the classification mosaics were clipped to the study area boundary. An image algebra change detection was performed by subtracting the 1976 classification from the 2006 classification in ArcMap using the 'raster calculator' function. This produced a 'difference' raster which shows areas that have experienced the greatest or least amount of change between 1976 and 2006.

LULC classifications were performed to demonstrate landscape composition in 1976, and how it had changed as of 2006. Each class in each time period is composed of a particular number of pixels. The spatial resolution (pixel size) of the Landsat MSS imagery (circa 1976) is 60 m. The scenes were re-sampled to a resolution of 30 m prior to being mosaicked and classified in order to ensure consistency with the Landsat TM (circa 2006) imagery which has a pixel size of 30 m. A 30 m x 30 m pixel thus represents approximately 9×10^{-4} km^2. Therefore, the number of pixels in a class was multiplied by 9×10^{-4} to determine the approximate land area in km^2 of a class.

Other LULC GIS datasets used in the thesis include forest harvest and forest fire spatial extent and water allocation license locations. Each LULC dataset was divided into the Central Lower or Lower Athabasca sub-basins to ensure consistency with the census of agriculture and census of population datasets (described below). Forest harvest data from 1976 to 2006 were obtained in tabular form from Alberta Sustainable Resource Development. The Saskatchewan portion of the study area falls into unmanaged forest, and thus there is no commercial harvest. Alberta harvest data were sorted according to timber year, which starts May 1st of one year, and ends April 30th of the next year. The dataset contains harvest records divided amongst townships and ranges. Thus, a township and range grid GIS dataset was obtained from the GIS services library at the University of Calgary, Calgary, Alberta. I imported the harvest table into ArcMap (ESRI ArcGIS version 9.3) and joined the dataset to the township and range layer using the 'joins and relates' attribute function in ArcMap. This enabled me to reduce the township and range layer to respective townships and ranges included only in the harvest table, as well as to show the spatial location and extent of forest harvest activities from 1976 to 2006. The harvest dataset provides a detailed description of different harvesting activities, including blow-downs, clear cuts, partial cuts, mineable oil sands areas, salvage cuts, commercial thins, etc. Next, the forest GIS layer was divided into the Central Lower and Lower Athabasca sub-basins. The total area of removed forest was determined for each timber year for the Central Lower ARB and Lower ARB from 1976 to 2006.

Forest fire GIS layers were obtained for all of Alberta and Saskatchewan from Alberta Sustainable Resource Development and the Saskatchewan Ministry of Environment, respectively. Individual burns were outlined as polygons that showed the locations and spatial extent of forest fires (> 10 km^2) from 1976 to 2006. Because the layer represented forest fires for all of Alberta and Saskatchewan, the layer was clipped first to the study area boundary, and then the sub-basin (Central Lower and Lower Athabasca) boundaries, resulting in fragmented polygons. This made it necessary to re-calculate the resulting fragmented polygon areas using the 'calculate geometry' function in ArcMap. The total area burned was determined for each year for the Central Lower ARB and Lower ARB from 1976 to 2006, excluding 1992, 1996, and 1997 for which there are no fire records.

Water allocation and consumption data (collectively referred to as water abstraction), were provided by Alberta Environment in tabular form, and were also explored as indicators of change in river response. The dataset includes active licenses from the main stem of the Lower Athabasca River and are organized according to the date the license holder began diverting water from the Athabasca River. These data are available from 1976 to 2006 and provide a quantification of the number of active licenses, allowable use (m^3/s) and actual consumptive use (m^3/s) per year for the Central Lower ARB and Lower ARB. Two different datasets were obtained: i) all active allocation licenses for the study area excluding those pertaining to oil sands operations; and ii) those owned by oil sands operators. The dataset included latitude and longitude co-ordinates for each license location, which allowed me to transform the data into a point file in ArcMap using the 'add xy data' tool. Licenses were assigned to the Central Lower or Lower Athabasca sub-basins, based on location. The number of water allocation licenses and consumptive use (m^3/s) were determined for each sub-basin.

Agriculture and population data were provided by CANSIM, Statistics Canada's socioeconomic database, via the Interpolated Census of Agriculture and Interpolated Census of Population, respectively. Data were available by major drainage area, and met the boundaries of the Central Lower and Lower Athabasca sub-basins shown in Figure 3.1. There is no agricultural activity in the Lower Athabasca sub-basin. Agricultural variables used in this thesis include number of farm units, agricultural land area (km^2), average farm unit size (km^2), cropland area (km^2), improved pasture area (km^2), cattle density ($\#/km^2$), chemical product expenses ($), and fertilizer expenses ($). These data were only available for each census year, i.e, in five-year intervals starting in 1976. Similarly, population statistics for the Central Lower and Lower Athabasca sub-basins are only available for census years. Variables include total population (#), total population density ($\#/km^2$), urban population (#), rural population (#), and total private dwelling density ($\#/km^2$).

Originally, there was a much larger list of variables, especially from the agriculture and urban land use classes. A correlation matrix was constructed on the full list of variables using Pearson correlation coefficients (Table 3.2). The results of the correlations were used to inform the selection of landscape variables for further use in order to reduce the original list to variables that are relatively independent of one another (Turner et al., 2001). There is much correlation between landscape variables that represent a general land use type, such as cattle density and fertilizer expenses which represent the general land use agriculture. This is expected because if a land use increases in intensity, the landscape variables that represent the land use will show increased intensity as well. Further, there are landscape variables that are very similar in terms of what they describe about the general land use. In this situation I chose the landscape variable that I thought best represented the activity out of the two variables, as no guide describing how to narrow variables exists in the literature. For example, agricultural activity pertaining to livestock can be described by the number of cattle and cattle density. To reduce redundancy, I chose to only include cattle density in my analyses because it represents the number of animals per unit area ($\#/ km^2$). As well, the maximum annual allowable surface water diversion and consumptive use both describe water abstraction. I chose to include consumptive use only in my analyses because it is a measure of the volume of water actually consumed, while the allowable amount only represents the volume of water that *could* be consumed. This reduced the list of variables from 33 to 18.

Table 3.2. Pearson correlation coefficients (r) for all landscape variables collected. * and ** indicate significant correlations at p<0.05 and p<0.01, respectively. N/A indicates insufficient data to calculate correlations between variable pair. Numbers on columns correspond to numbered metrics in the first column. Final variables used in analyses are bolded.

Landscape variable number/ name	1	2	3	4	5	6	7	8	9	10
1 Number of farm units										
2 Agricultural land area	-.840*									
3 Average farm unit size	-.965**	.948**								
4 Agricultural land as a share of total area	-.839*	1.000**	.947**							
5 Cropland area	-.906**	.898**	.934**	.896**						
6 Improved pasture area	-.927**	.899**	.954**	.899**	.797*					
7 Number of cattle	-.915**	.903**	.940**	.904**	.811*	.967**				
8 Cattle density	-.914**	.902**	.939**	.903**	.809*	.966**	1.000**			
9 Chemical product expenses	-.955**	.921**	.958**	.921**	.966**	.908*	.951**	.949**		
10 Chemical product expenses per total area	-.955**	.921**	.958**	.921**	.966**	.908*	.951**	.949**	1.000**	
11 Fertilizer expenses	-.906*	.923**	.927**	.923**	.975**	.856*	.916*	.915*	.990**	.991**
12 Fertilized land area	-.853*	.753	.799	.755	.827*	.799	.831*	.831*	.914*	.914*
13 Fertilizer expenses per total area	-.906*	.923**	.927**	.923**	.975**	.856*	.916*	.915*	.991**	.991**
14 Total population	-.563	.251	.483	.247	.391	.444	.290	.288	.120	.120
15 Total population density	-.539	.237	.464	.232	.363	.431	.274	.272	.088	.088
16 Urban population	-.480	.148	.384	.143	.329	.333	.164	.163	-.035	-.035
17 Urban population as a share of total population	.152	-.357	-.254	-.359	-.083	-.350	-.485	-.485	-.639	-.639
18 Rural population	-.683	.473	.657	.469	.485	.654	.560	.558	.393	.392
19 Rural population as a share of total population	-.152	.357	.254	.359	.083	.350	.485	.485	.639	.639
20 Total private dwellings	-.728	.562	.722	.558	.591	.683	.561	.559	.442	.442
21 Private dwelling density	-.725	.567	.723	.564	.579	.696	.569	.567	.439	.439
22 Population per private dwelling	.791*	-.712	-.818*	-.709	-.750	-.719	-.654	-.653	-.640	-.640
23 Urban private dwellings	-.700	.509	.680	.504	.577	.624	.492	.491	.390	.390
24 Rural private dwellings	-.744	.629	.762*	.625	.589	.755*	.655	.653	.508	.508
25 Population per urban private dwelling	.818*	-.770*	-.855*	-.767*	-.818*	-.733	-.694	-.694	-.732	-.732
26 Population per rural private dwelling	.789*	-.712	-.822*	-.709	-.722	-.752	-.660	-.658	-.600	-.599
27 Water allocations	-.907**	.856*	.933**	.854*	.899**	.824*	.842*	.842*	.873*	.873*
28 Maximum Annual Surface Water Diversion	-.872*	.823*	.896**	.821*	.847*	.799*	.857*	.857*	.824*	.824*
29 Consumptive use	-.580	.770*	.668	.770*	.802*	.544	.546	.545	.702	.702
30 Area burned	.133	-.692	-.419	-.693	-.446	-.351	-.478	-.473	-.973*	-.973*
31 Area Harvested	-.727	.639	.722	.638	.695	.615	.737	.739	.661	.661
32 Maximum Annual Surface Water Diversion: Oil Sands	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
33 Consumptive use: Oil Sands	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

	11	12	13	14	15	16	17	18	19
11 **Fertilizer expenses**									
12 Fertilized land area	.914*								
13 Fertilizer expenses per total area	1.000**	.914*							
14 **Total population**	-.008	-.050	-.008						
15 **Total population density**	-.039	-.086	-.039	.649*					
16 **Urban population**	-.160	-.138	-.160	.938**	.865**				
17 Urban population as a share of total population	-.616	-.345	-.616	-.782**	-.085	-.532			
18 **Rural population**	.279	.123	.279	.860**	.180	.631*	-.970**		
19 Rural population as a share of total population	.616	.345	.616	.782**	.085	.532	-1.000**	.970**	
20 Total private dwellings	.340	.184	.340	.960**	.643*	.901**	-.711**	.825**	.711**
21 **Private dwelling density**	.337	.192	.337	.696**	.950**	.864**	-.164	.287	.164
22 Population per private dwelling	-.582	-.342	-.582	-.709**	-.694**	-.758**	.324	-.473	-.324
23 Urban private dwellings	.285	.149	.285	.917**	.829**	.960**	-.518	.642*	.518
24 Rural private dwellings	.411	.231	.411	.854**	.236	.646*	-.885**	.964**	.885**
25 Population per urban private dwelling	-.695	-.435	-.695	-.729**	-.688**	-.767**	.378	-.506	-.378
26 Population per rural private dwelling	-.526	-.308	-.526	-.734**	-.564*	-.710**	.527	-.601*	-.527
27 **Water allocations**	.847*	.623	.847*	.572	.505	.647*	-.241	.370	.241
28 Maximum Annual Surface Water Diversion	.797	.558	.797	-.242	.857**	.194	.647*	-.586*	-.647*
29 **Consumptive use**	.775	.736	.775	-.325	.815**	.124	.690*	-.662*	-.690*
30 **Area burned**	-.995**	-.751	-.995**	-.174	-.126	-.136	.266	-.198	-.266
31 **Area Harvested**	.645	.408	.645	.528	.084	.390	-.463	.533	.463
32 Maximum Annual Surface Water Diversion: Oil Sands	N/A	N/A	N/A	-.031	-.037	-.039	.142	.067	-.142
33 **Consumptive use: Oil Sands**	N/A	N/A	N/A	.649	.645	.640	-.612	.716	.612

	20	21	22	23	24	25	26	27
20 Total private dwellings								
21 Private dwelling density	.764**							
22 Population per private dwelling	-.850**	-.859**						
23 Urban private dwellings	.956**	.911**	-.898**					
24 Rural private dwellings	.889**	.396	-.622*	.716**				
25 Population per urban private dwelling	-.855**	-.847**	.992**	-.898**	-.633*			
26 Population per rural private dwelling	-.799**	-.687**	.724**	-.791**	-.667**	.732**		
27 Water allocations	.762**	.754**	-.929**	.831**	.541	-.956**	-.890**	
28 Maximum Annual Surface Water Diversion	-.049	.747**	-.408	.288	-.444	-.397	-.250	.429
29 Consumptive use	-.183	.648*	-.231	.165	-.567	-.221	-.114	.245
30 Area burned	-.189	-.142	.119	-.162	-.203	.148	.635*	-.022
31 Area Harvested	.649*	.349	-.668*	.564	.633*	-.711**	-.661*	.820**
32 Maximum Annual Surface Water Diversion: Oil Sands	.152	.162	-.514	.148	.190	-.514	.184	.749
33 Consumptive use: Oil Sands	.749	.754	-.885*	.747	.749	-.885*	-.545	.811

	28	29	30	31	32	33
28 Maximum Annual Surface Water Diversion						
29 Consumptive use	.962**					
30 Area burned	.052	.089				
31 Area Harvested	.067	-.154	-.113			
32 Maximum Annual Surface Water Diversion: Oil Sands	.688	.621	.545	.483		
33 Consumptive use: Oil Sands	.772	.757	-.353	.558	.656	

3.2.3 River Response data

The responses of river water quality and quantity were examined for the lower reaches of the Athabasca River from 1976 to 2006. The water quality parameters used were the same ones used by Squires et al. (2010) and include concentrations of total organic carbon (TOC), total dissolved nitrogen (TDN), total phosphorus (TP), chloride (Cl^-), sodium (Na^+), sulphate ($^-$), turbidity, and specific conductance (SC). Squires et al. (2010) used these variables because availability met their study's spatial and temporal requirements. However, they only analyzed changes in these parameters between 1966-1976 and 1996-2006. Additional data were analyzed in this thesis. Specifically, values for the aforementioned parameters were obtained for years between 1976 and 2006 from Environment Canada and Natural Resources Canada.

The outlet at the mouth of the ARB was used to represent water quality and quantity for the lower reaches of the ARB. Water quantity data were obtained from one of the Water Survey of Canada's (WSC) national archive for hydrometric data (HYDAT) gauging stations (station AB07DD007: Athabasca River Above Jackfish Creek), which is located near the mouth of the Athabasca River (Figure 3.2). This station was chosen because it is the only HYDAT station that has recorded data continuously from 1976 to the present. However, station AB07DD007 only records stream stage (m). Since no rating curve exists, stage could not be converted to discharge. No stations near the mouth of the ARB continuously recorded discharge from 1976-2006 (Table 3.3). Averages were computed for annual stream stage (m), winter stream stage (m), and summer stream stage (m) from daily records between 1976 and 2006.

Solute concentrations, turbidity and SC were obtained from two water quality stations near the mouth of the ARB (Environment Canada station AL07DD0001; Natural Resources Canada Station AB07DD0360; Figure 3.2). Not all parameters were available for each year at each station; therefore there are significant missing data (Table 3.4). Specifically, there are no annual water quality data available for 1985, 1986, 1987, and 1988. Annual averages were calculated for the solute variables (mg/L), SC (μS/cm) and turbidity (NTU).

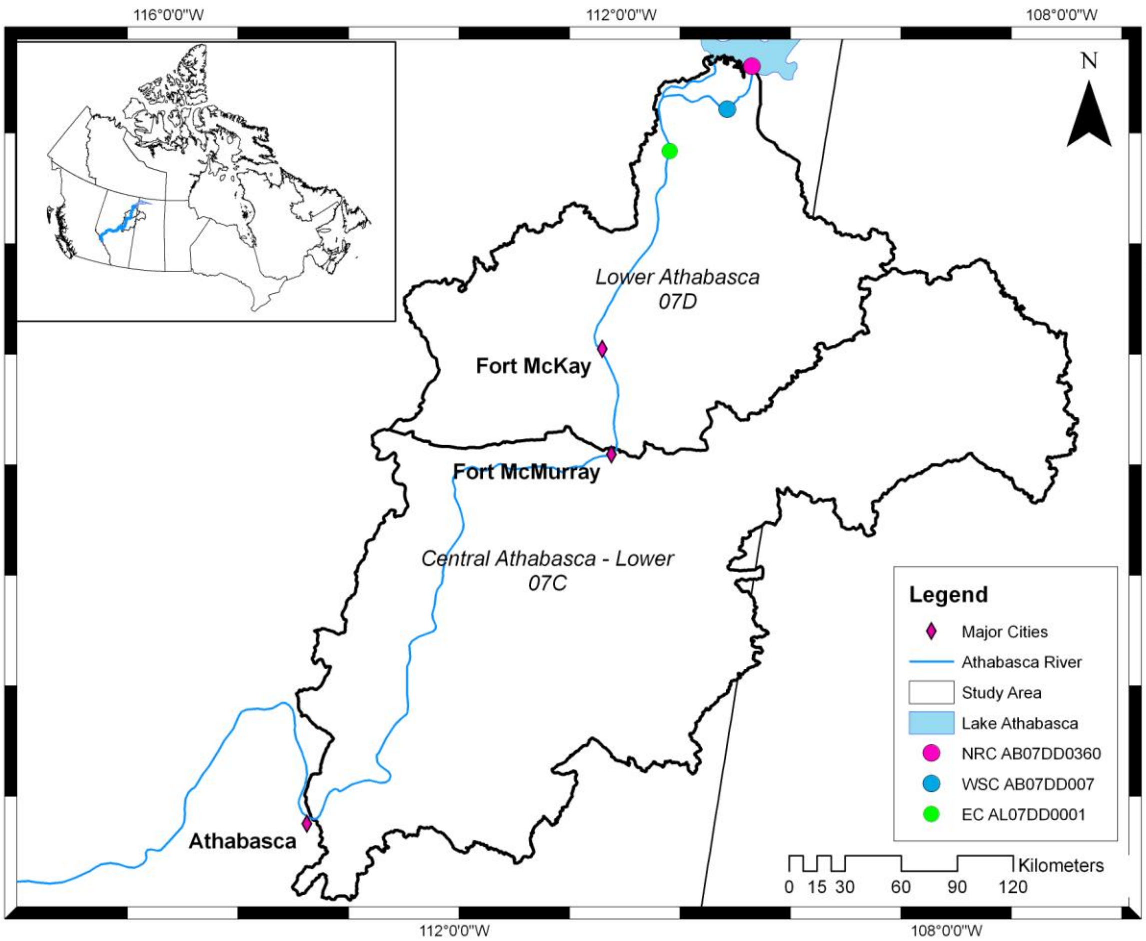


Figure 3.2. Locations of the Federal and Provincial water quality and quantity monitoring stations from which data were extracted for use in this thesis. Stations operators include Natural Resources Canada (NRC), Water Survey of Canada (WSC) and Environment Canada (EC).

Table 3.3. Water quantity station locations and data availability, starting at the central lower reach and continuing to the mouth of the lower reach. Asterisks represent the station from which data was extracted for use in the analyses. See Figure 3.2 for locations of stations from which data was used.

Reach	Station ID	Name	Source	Discharge	River Stage
Central Lower	07BE001	At Athabasca	HYDAT	1913-2009	-
	07DA001	Below McMurray	HYDAT	1957-2009	-
Lower	S24	-	RAMP	2001, 2006	-
	07DD001	At Embarras Airport	HYDAT	1971-1984	-
	07DD011	Near Old Fort	HYDAT	-	1975-2009
	07DD007*	Above Jackfish Creek	HYDAT	-	1971-2009

Table 3.4. Water quality parameters and dates used to link landscape change to river response. The table is organized by station location, starting at the town of Athabasca and continuing to the mouth at the Athabasca Delta. The table shows the years for which water quality data was available at each station. *Represents stations from which data were used. See Figure 3.2 for station locations. Data sources: Alberta Environment (AENV), Environment Canada (EC), and Natural Resources Canada (NRC).

Reach	Station ID	Data Source	TOC	TDN	TP	Cl^-	Na^+		SC	Turbidity
Central Lower	07BE0001	AENV	1987-2003	1987-1998	1987-2009	1987-2009	1987-2009	1987-2009	1987-2009	1987-2009
	07BE0001	EC	1969-87	1978-1987	1960-1987	1960-1987	1960-1987	1960-1987	1960-1987	1960-1987
	07CC0030	AENV	-	-	1987-2009	1987-2009	1987-2009	1987-2009	1987-2009	1987-2009
	07DA0001	AENV	1976	1976	1976	1976	1976	1976	1976	1976
	07DA0001	EC	1971-1978	-	1967-1978	1967-1978	1967-1978	1967-1978	1967-1978	1967-1978
Lower	07DA1470	NRC	1976	1976	1976	1976	1976	1976	1976	1976
	07DA1500	NRC	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1981
	07DA1520	NRC	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1977
	07DA1540	NRC	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1981
	07DA1550	NRC	1976-1983	1976-1983	1976-1983	1976-1983	1976-1983	1976-1983	1976-1983	1976-1981
	07DD0001*	EC	1989-2006	1989-2006	1989-2006	1989-2006	1989-2006	1989-2006	1989-2006	1989-2006
	07DD0010	AENV	-	-	1987-2009	1987-2009	1987-2009	1987-2009	1987-2009	1987-2009
	07DD0105	AENV	1997-2003	-	1997-2003	1997-2003	1997-2003	1997-2003	1997-2003	1997-2003
	07DD0360*	NRC	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1984	1976-1981

3.2.4 Linking landscape change to river response

In order to test for linkages between all LULC variables (i.e. datasets on an annual basis [~30 years] and datasets by census year [seven dates]) and river response, four different datasets were built based on frequency of data collection: i) annual water quality; ii) annual water quantity; iii) census year water quality; and iv) census year water quantity. Data from respective census years were extracted from the forestry, fires, and water abstraction datasets (1976, 1981, 1986, 1991, 1996, 2001, 2006) for inclusion in the census year water quality and census year water quantity datasets. Conversely, because the variables from the agriculture and population datasets are only available by census year, the annual water quality and annual water quantity datasets only contained forestry, fires, and water abstraction variables. Separate analyses were performed on each of the four datasets. Due to poor spatial data availability, individual years were used in this thesis as the replicates. Thus, there were two values for each landscape variable in the annual and census year datasets -- one comprising data from the Central Lower and one from the Lower ARB. Exceptions were for the agriculture variables where data were only available for the Central Lower ARB.

Using temporal replicates undoubtedly raises concern for autocorrelation. However, this approach is common in landscape ecology studies that aim to quantify an ecosystem, such as a watershed or drainage basin, at such large spatiotemporal scales (Hargrove and Pickering, 1992). It was necessary to first correct for potential temporal autocorrelation before data could be used in linking landscape change to river response. This was done by first regressing each variable (water quality, water quantity, landscape) against time (Table 3.5and Table 3.6). The variables that showed non-significant regressions ($p>0.05$) were log-transformed for use in the subsequent analyses, whilst the variables that showed significant regressions ($p<0.05$) were transformed using the LOWESS (locally weighted scatter plot smoothing) technique. LOWESS involves fitting weighted least squares regressions to produce a ‘smoothed’ curve through the raw data (see Cleveland and Grosse [1991] for details on the technique). Briefly, the resulting values of the smoothed curve are subtracted from the original raw values, producing a series of residuals that are used in subsequent analyses. The LOWESS technique is commonly used in hydrologic studies to remove trends due to discharge and produce a de-trended-flow-adjusted residual time series (Westbrook and McEachern, 2002), or to de-trend for seasonality (Esterby, 1996). However, LOWESS has also been used in landscape studies as a way to de-trend confounding factors in datasets, and in my case time, in order to correct for pseudoreplication. The LOWESS technique was conducted using Revolution R Enterprise version 3.1.1 software, using the ‘LOWESS’ command in the ‘stats’ package.

Table 3.5. Correlations between year and river response variables. * and ** indicate significance at $p<0.05$ and $p<0.01$ respectively.

	Annual	Winter	Summer	TOC	Cl^-	TDN	TP	Na^+	SC		Turbidity
Annual (r)	-0.509**	-0.545**	-0.302*	-0.001	0.460**	-0.178	-0.126	0.549**	0.624**	0.731(**)	0.05
Census (r)	-0.53	-0.676**	-0.144	-0.147	0.601*	-0.641*	-0.371	0.734**	0.779**	0.805(**)	-0.804(*)

Table 3.6. Correlations between year and landscape effect variables. * and ** indicate significance at p<0.05 and p<0.01 respectively. Recall that agricultural and population variables are only available on a census year basis, thus cannot be correlated with non-census years (--).

	Annual (r)	**Census (r)**
Area burned	-0.127	-0.284
Area harvested	0.553**	0.494
Consumption	0.405	0.481
Oil sands Consumption	0.686**	0.793
Water allocations	--	0.819**
Number of farm units	--	-0.966**
Agricultural land area	--	0.837*
Average farm unit size	--	0.970**
Cropland area	--	0.850*
Improved pasture area	--	0.917*
Cattle density	--	0.946**
Chemical product expenses	--	0.833
Fertilizer expenses	--	0.709
Total population	--	0.542
Total population density	--	0.787**
Urban population	--	0.688*
Rural population	--	0.182
Private dwelling density	--	0.878**

Pearson correlation coefficients were then computed for the LOWESS residuals and log-transformed values in the annual-quality, annual-quantity, census-quality, and census-quantity datasets. Plots are presented for variables with significant correlation coefficients. Step-wise regressions were performed on each data set for which significant correlations between water quality or quantity variables and landscape variables existed in order to identify the relative importance of each stressor in driving the river response. Multiple regressions were also explored on landscape stressors and river response, but were strongly limited due to insufficient data pairs between landscape variables and river response variables for each replicate.

Last, regression trees were explored as a means to describe relationships between landscape stressors and river response. Regression trees are being increasingly used by ecologists in order to explain variation of a single response variable by one or more explanatory variables (De'ath and Fabricus, 2000). In landscape analyses, regression trees are used to identify main variables from a number of landscape metrics (Pan et al., 1999). Regression trees involve splitting data along axes of the explanatory variables, which are divided into binary classes based on a threshold value (Breiman et al., 1984). Each class is characteristic of more binary explanatory classes, thus the tree keeps splitting along subsequent axes, resulting in many nodes. At each node, the split that explains the largest amount of deviance is used. This approach has the capacity to capture relationships between variables that would be expected due to their ecological nature, but are otherwise not readily apparent with other statistical linear methods, such as regression (Urban, 2002). Regression trees were constructed with Revolution R Enterprise version 3.1.1 software, using the ‘rpart’ command in the ‘rpart’ module on the LOWESS and log-transformed data within the annual-quality and annual-quantity datasets. There were not enough data to perform regression tree analysis on the census-quality and census-quantity datasets. From the annual-quality dataset, each water quality variable (TOC, Cl^-, TDN, TP, Na^+, $^-$, SC, turbidity) was used to produce a regression tree with each landscape variable (area burned, area harvested, consumptive use, consumptive use [oil sands]) within the dataset. Similarly, each water quantity variable (average annual stream stage, average winter stream stage, average summer stream stage) within the annual-quantity dataset were used to produce a regression tree with all four landscape variables in the dataset.

3.3 Results

3.3.1 Quantifying landscape change

The lower ARB watershed has changed a great deal over the past three decades as a result of land conversion. It is evident from the LULC classifications that the proportion of forest and grassland cover has decreased while the proportion of area classed as anthropogenic activities has increased. Deciduous, coniferous and mixed forest decreased by 7% (5841 km^2) between 1976 and 2006 (Table 3.7). Most of the forest area was lost to human activities. Forestry, urban, industrial and agriculture classes collectively increased by 9616 km^2, or 11%, over the study period. Forest harvesting was greatly intensified over the study period, with the area of cut-blocks increasing from virtually zero in 1976 to 5386 km^2 in 2006. Industrial activity, predominantly characterized by the oil sands development near Fort MacKay, increased by 889 km^2 over the study period.

The spatial distribution of LULC also has changed between 1976 and 2006 as evidenced from the image algebra change detection image (Figure 3.3). There is a large crescent shaped forest fire scar in the centre of the image as well as a smaller forest fire scar on the Alberta-Saskatchewan border. The oil sands region, near Fort McKay, shows dark blue pixels, indicating a high degree of landscape change between 1976 and 2006. Figure 3.4 shows that the oil sands developments on the east side of the river and north of Fort McKay would have occurred sometime after 1976, as these portions of the oil sands developments are not present in Figure 3.4a. Further, the portion of the oil sands developments that are present in Figure 3.4a have increased in size since 1976, as they appear larger in 2006 as shown by Figure 3.4b.

The area of each land cover class depicted in Table 3.7 represents approximate values (see section 3.2.2 for details on calculation). The GIS layers offer more accurate values, but are congruent with the nature of the land covers shown in the land cover classification in terms of growth or decline. For example, the forestry GIS dataset shows that from 1976 to 1979 the total amount of forest removed was approximately 3 km^2. During these years, the operators were removing timber from one forestry category; clear cuts. During the 1980s, the total amount of forest removed was approximately 6.2 km^2. There was a drastic change throughout the 1990s with the total amount of forest removal increasing to 474 km^2 (Figure 3.5). The number of forestry categories increased as well to encompass clear cuts, blow down (snag) clearing, clear cuts after blow down, liquidation cuts, removal in mineable oil sands areas, partial cuts, salvage cuts after wildfire with reforestation responsibility, and salvage cut after wildfire without reforestation responsibility. Forestry operations seemed to peak in the 1990s, as 2000-2006 saw a slight decrease in forest removal (430 km^2). Notably, forest harvest activities are limited to the Alberta portion of the study area, as the Saskatchewan portion falls outside the commercial forest boundary (Mark Doyle, Forest Service, Saskatchewan Ministry of Environment, pers. comm.).

Table 3.7. Land cover classification results in number of pixels and total area (approximate; see section 3.2.2 for details on calculation). Cloud/no data and water classes were not included in the table. Classes in the % change (total area) column are made up of the proportion of pixels making up a given class in 1976 subtracted from that in 2006 (70% overall classification accuracy).

Class	**1976**			**2006**			**% Change**
	No. pixels	**km^2**	**% of total**	**No. pixels**	**km^2**	**% of total**	**(total area)**
Deciduous	10814071	9867	11.22	9025774	8692	9.89	-1.34
Coniferous	29068046	26522	30.17	27872720	26841	30.53	0.36
Mixed	34404893	31391	35.71	27421793	26406	30.04	-5.67
Barren	5538389	5053	5.75	6578745	6335	7.21	1.46
Urban	378631	345	0.39	2996214	2885	3.28	2.89
Industrial	342645	313	0.36	1248276	1202	1.37	1.01
Fire Scar	3547383	3237	3.68	4152483	3999	4.55	0.87
Agriculture	1479814	1350	1.54	2232781	2150	2.45	0.91
Cut-blocks	~0	~0	-	5593480	5386	6.13	6.13
TOTAL	96340756	87902	-	91281981	87902	-	-

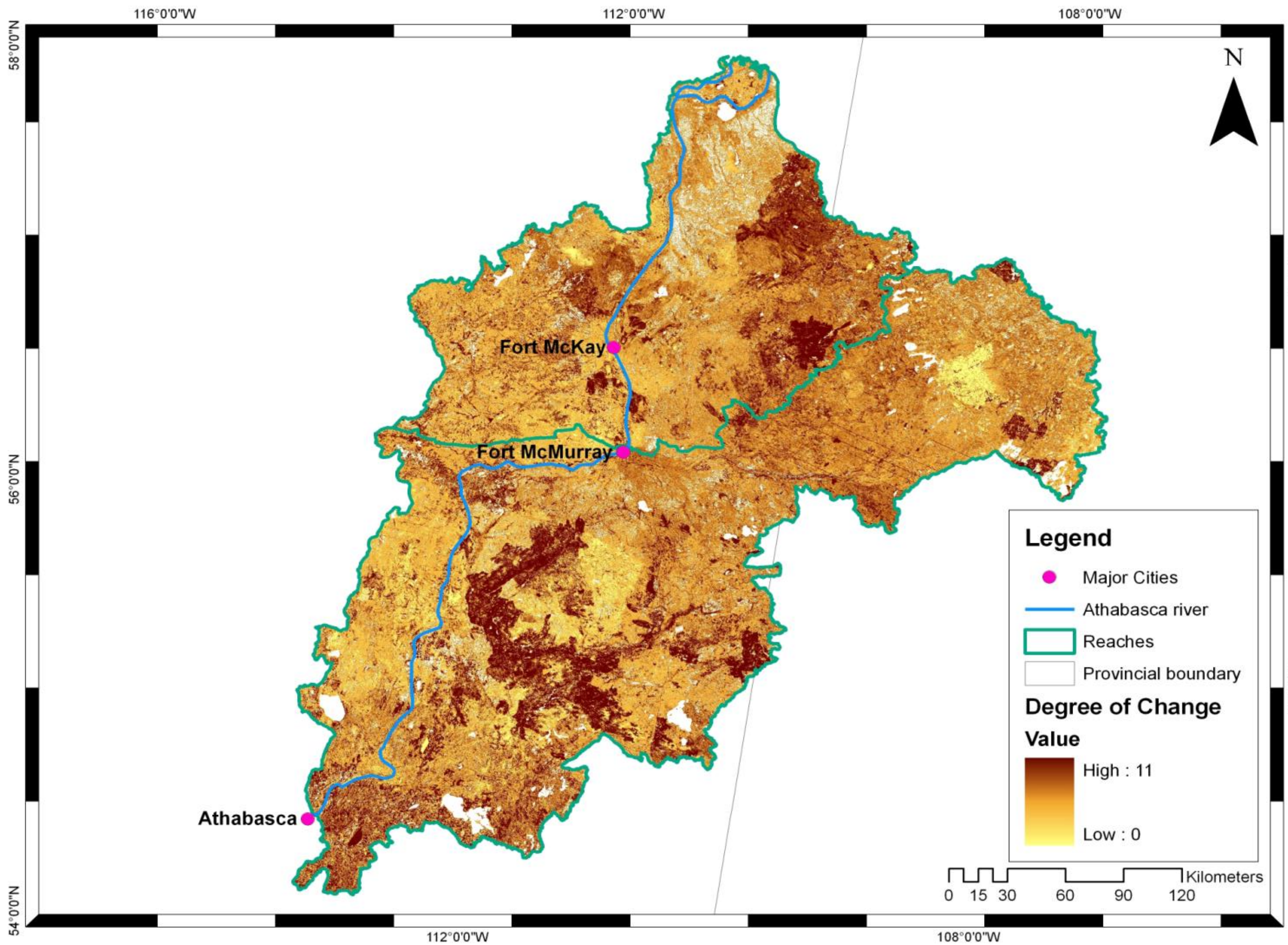


Figure 3.3. Image algebra change raster, calculated from subtracting the 2006 classification from the 1976 classification. Lighter colors represent areas of little (yellow) to no (white) change, while darker colors represent areas of moderate (orange) to extreme (dark brown) change. Areas of minimal change represent vegetation classes, large bodies of water, and barren land (i.e. bedrock outcrops, sediment [Athabasca Delta]).

The forest fire GIS layer shows that from 1976 to 1979 there were 46 individual fires that collectively covered 600 km^2. However, in the 1980s, area burned was considerably higher. From 1980 to 1989 there were 186 fires that burned 16 864 km^2. These fires were mostly contained to the northeastern portion of the study area, which falls in Saskatchewan (Figure 3.6). Fires in the 1990s also mostly occurred in Saskatchewan. This area is characterized mainly by coniferous boreal forest. There was however a large fire that occurred in 1995 in the centre of the study area, south of Ft. McMurray. The number of fires during the 1990s numbered 388, collectively occurring over ~7925 km^2. From 2000 to 2006 the pattern of forest fire occurrence was similar to those in the 1990s in that the major fires occurred primarily in Saskatchewan, as well as the centre of the basin, as per the aforementioned image algebra change detection image. There were 785 fires between 2000 and 2006, covering a total area of ~6162 km^2.

In 2006, there were 14 water allocation licenses withdrawing water from the Central Lower reach (Table 3.8). Water was withdrawn mainly for urban, agricultural, and industrial purposes. The remaining 13 licenses were found in the Lower reach. Here, water is mainly withdrawn for industrial purposes, specifically oil and gas activities, as well as for oil sands open pit mining projects. In the Central Lower reach, consumptive use increased 170% between 1976 and 2006 (Table 3.8). Similarly, the number of licenses increased from 2 in 1976 to 14 in 2006. In the Lower reach, consumptive use increased 337% between 1986 and 2006 (Table 3.8). The number of licenses also increased from 3 to 13. Table 3.8 does not include withdrawals for the specific purpose of oil sands open pit mining projects, which occur exclusively in the Lower reach; these are instead reported in Table 3.9. There was a marked increase in oil sands activities in the early 1980s, and as a result, increased water use.

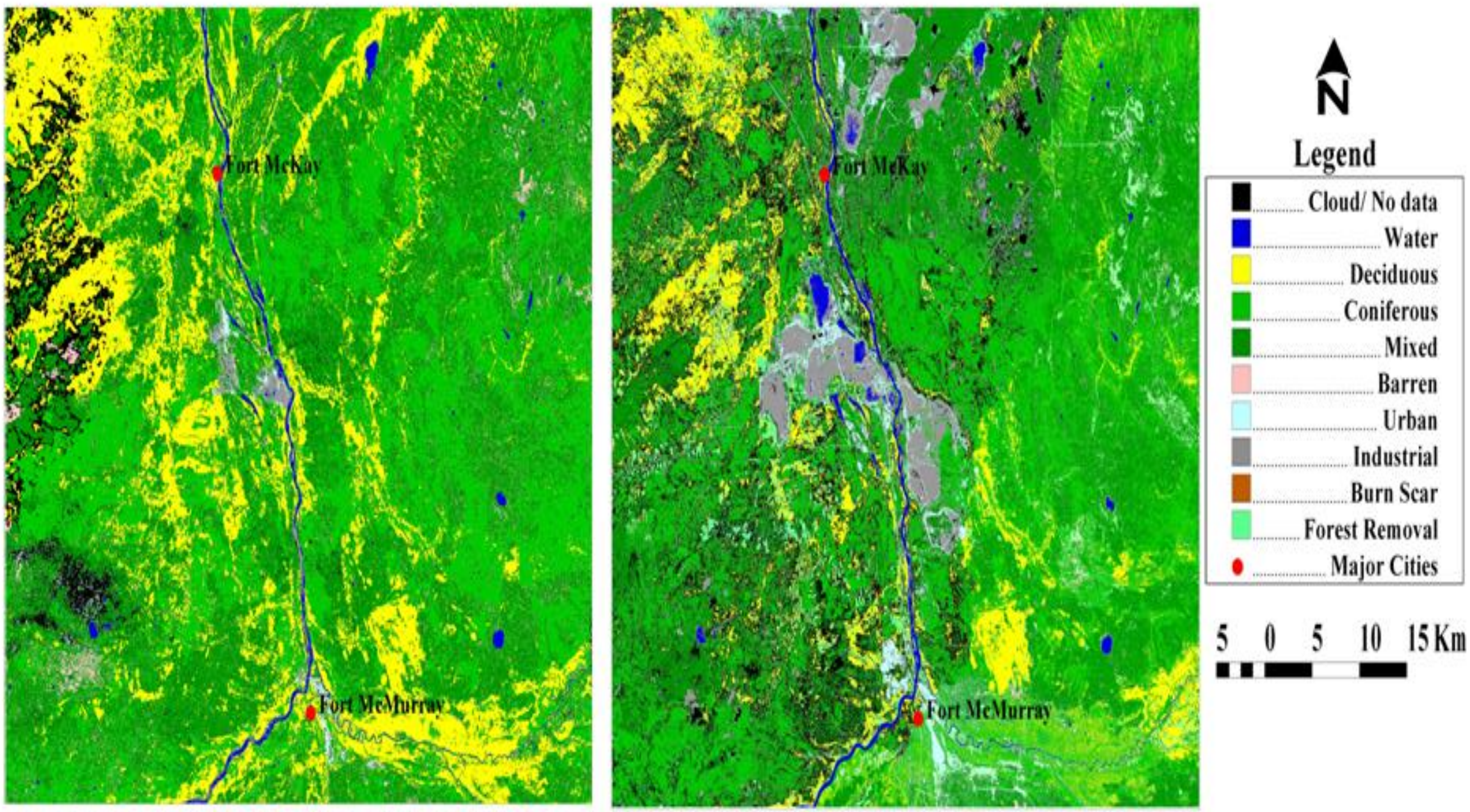


Figure 3.4. LULC classification zoomed into the Oil Sands region north of Fort McMurray, Alberta in a) 1976 and b) 2006. Increased spatial extent of the industrial (oil sands mining), forestry (cut-blocks), and urban (city of Ft McMurray, increased road density) classes are much more apparent in the 2006 classification (70 % overall classification accuracy).

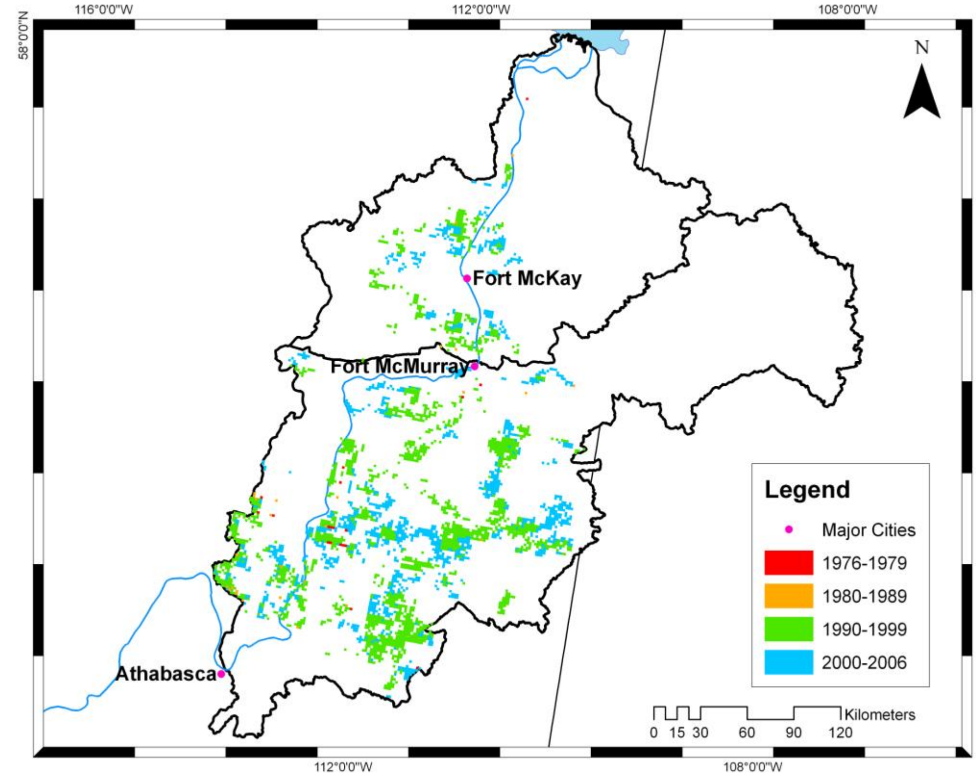


Figure 3.5. Locations of forest removal operations in the Athabasca River basin by decade.

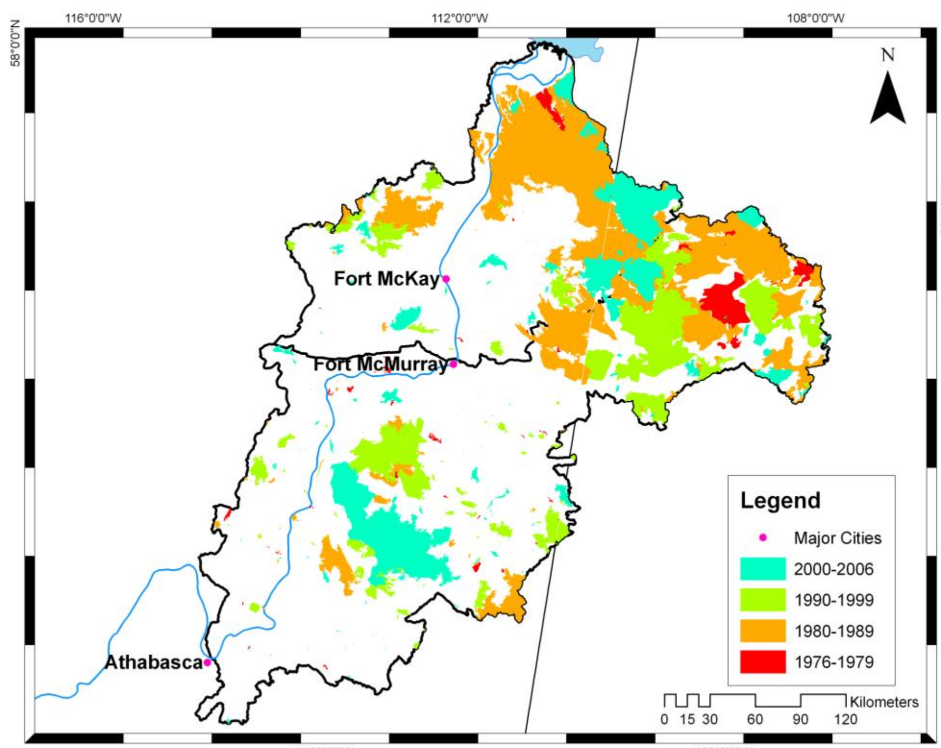


Figure 3.6. Locations of major forest fires in the Athabasca River basin by decade.

Table 3.8. Water abstractions for each census year in the Central Lower and Lower Athabasca sub-basins. Allowable use and consumptive use include water abstraction for purposes not pertaining to the oil sands operations.

	Central Lower reach			**Lower reach**		
Census Year	No. Active Licenses	Allowable Use (x 10^6 m^3)	Consumptive Use (x 10^6 m^3)	No. Active Licenses	Allowable Use (x 10^6 m^3)	Consumptive Use (x10^6 m^3)
1976	2	1.51	3.89	-	-	-
1981	4	2.32	4.88	-	-	-
1986	4	2.32	4.88	3	61.71	61.51
1991	9	27.27	27.24	5	121.53	82.68
1996	12	63.75	10.42	6	121.53	82.68
2001	13	65.04	10.422	6	121.53	82.68
2006	14	65.13	10.51	13	306.85	267.99

Table 3.9. Water abstraction data for each census year (beginning in 1981) for oil sands operations.

	Lower Reach - Oil Sands	
Census Year	Diversion (x 10^7 m^3)	Net use (x 10^7 m^3)
1981	9.74	4.45
1986	6.75	4.04
1991	7.15	3.85
1996	7.97	4.33
2001	9.34	6.95
2006	9.34	7.86

The census of agriculture indicates that there has been an increase in agricultural intensity in the study area. This is shown by a 42% increase in cattle density (number/ km^2) between 1976 and 2006 (Table 3.10). Improved pasture area, which involves maintaining sufficient vegetation food sources for grazing in pastures, increased 71% between 1976 and 2006. Increased agricultural intensity is also evident from farm expenses. Chemical product expenses increased 92% and fertilizer expenses increased 95% between 1981 and 2006.

Table 3.10. Agricultural activities from Canada's Interpolated Census of Agriculture for each census year between and including 1976 and 2006 in the Central Lower (07C) reach. (Chemical product and fertilizer expenses are in 1992 dollars.)

Year	**Number of farm units (#)**	**Agricultural land area (km^2)**	**Average farm unit size (km^2)**	**Cropland area (km2)**	**Improved pasture area (km^2)**	**Cattle density (#/km^2)**	**Chemical product expenses ($ x 10^5)**	**Fertilizer expenses ($x 10^5)**
1976	882.60	2123	2.41	778.70	224.20	0.78	--	--
1981	813.6	2011	2.47	820	219.6	0.71	6.1	11.9
1986	918.8	2012	2.19	778.4	126.6	0.48	1.4	3.9
1991	772.90	2326	3.01	986.50	288.10	0.90	11.7	26.3
1996	767.50	2234	2.91	941.70	266.00	0.97	10.8	23.7
2001	719.90	2311	3.21	1009.60	318.20	0.98	12.4	25.4
2006	712.90	2361	3.31	949.70	384.70	1.11	11.7	23.2

The census of population describes changes in both spatial development and number of people. The number of private dwellings and private dwelling density indicate spatial change in terms of housing, which also has an impact on the number of roads and sidewalks that must be built to accommodate such infrastructure. The number of people is indicated by variables such as total population, which is made up from the urban and rural populations. Within the study area, the total population increased 153% between 1976 and 2006 (Table 3.11). Most of these people lived in urban areas in both 1976 (70%) and 2006 (80%). The number of people living in urban areas increased 190% between the two time periods. Also observed is a concomitant increase in dwelling density; 377% between 1976 and 2006.

Table 3.11. Population statistics from Canada's Interpolated Census of Population for the Central Lower and Lower reaches for census years between and including 1976 and 2006.

	Year	**Total population (#)**	**Total population density (#/km^2)**	**Urban population (#)**	**Rural population (#)**	**Private dwelling density (#/km^2)**
Central Lower Reach	1976	22566	0.4	15118	7448	0.11
	1981	32627	0.57	24023	8604	0.17
	1986	30515	0.54	22230	8285	0.17
	1991	27018	0.47	19653	7365	0.16
	1996	26955	0.47	18525	8430	0.16
	2001	35508	0.62	24757	10751	0.24
	2006	36320	0.64	25235	11085	0.28
Lower Reach	1976	2330	0.08	2260	70	0.02
	1981	9088	0.3	8984	104	0.09
	1986	15944	0.53	15272	672	0.15
	1991	18115	0.61	17602	513	0.18
	1996	17745	0.59	17164	581	0.19
	2001	17310	0.58	16711	599	0.19
	2006	26762	0.89	25243	1519	0.34

3.3.2 Linkages between landscape change and river response

3.3.2.1 Simple and step-wise regressions

The census-quality dataset yielded five significant simple regression models (Table 3.12) while the annual-quality dataset yielded four significant models (Table 3.13). There were no significant regression models between water quantity variables and landscape metrics. Simple regression analyses show water abstraction is closely associated with TP and Na^+ concentrations, as well as SC. Oil sands consumptive use was positively correlated with both TP ($p< 0.05$) and Na^+ ($p= 0.002$). Consumptive use for activities other than those related to the oil sands was positively correlated with SC ($p=0.048$). Area burned was negatively correlated with TOC ($p=0.020$). One agriculture variable, the number of farm units, was negatively correlated with Cl^- ($p= 0.016$), SC (p= 0.004), Na $^{+}$ (p=0.003), and $^{-}$ (p=0.006). TDN was positively correlated with fertilizer expenses and (p=0.009) and consumptive use (non-oil sands activities) (p=0.048). Fertilizer expenses and consumptive use (non-oil sands activities) were run as a step-wise regression against TDN. The step-wise regression showed that the model that best explained the variance included fertilizer expenses only (p=0.009).

Table 3.12. Pearson correlation coefficients (r) for the census-quality dataset. Variables that were LOWESS smoothed are indicted by "LS". * and ** (bold) indicate significant correlations (p<0.05 and p<0.01, respectively).

Variable		TOC	Cl- (LS)	TDN (LS)	TP	Na^{+} (LS)	SC (LS)	$^{-}$ (LS)	Turbidity (LS)
Number of Farms (LS)	r	.791	**-.895***	.251	.396	**-.957****	**-.946****	**-.935****	.552
	P	.061	.016	.631	.437	.003	.004	.006	.448
Agricultural Land (LS)	r	.204	.231	.750	.296	.314	-.130	-.181	.132
	P	.698	.660	.086	.569	.544	.807	.731	.868
Average Farm Size (LS)	r	-.073	.541	.551	.164	.627	.226	.168	-.117
	P	.890	.268	.257	.757	.183	.667	.750	.883
Cropland Area (LS)	r	-.134	.575	.498	.171	.703	.341	.312	-.246
	P	.800	.232	.315	.745	.119	.509	.547	.754
Improved Pasture Area (LS)	r	-.074	.531	.511	.082	.546	.145	.048	.020
	P	.890	.279	.300	.877	.262	.784	.927	.980
Number of Cattle (LS)	r	.652	-.332	.794	.475	-.313	-.695	-.741	.888
	P	.161	.521	.059	.341	.546	.126	.092	.112
Chemical Expenses (LS)	r	.163	.135	.053	-.296	-.126	-.319	-.470	.866
	P	.794	.829	.932	.628	.840	.601	.425	.134
Fertilizer Expenses	r	.541	-.246	**.960****	.679	-.060	-.454	-.427	-.105
	P	.346	.690	.009	.208	.923	.442	.473	.895
Total Population (LS)	r	-.232	.434	-.293	-.267	.342	.461	.418	-.598
	P	.469	.159	.356	.401	.277	.131	.176	.117
Total Population Density (LS)	r	-.315	.561	-.283	-.391	.360	.462	.375	-.175
	P	.318	.058	.373	.208	.251	.131	.230	.679
Urban Population (LS)	r	-.247	.155	.202	-.370	.094	.129	.103	-.283
	P	.438	.631	.529	.237	.771	.689	.750	.496
Rural Population (LS)	r	-.087	.040	.106	-.212	-.027	-.016	-.051	-.056
	P	.787	.903	.742	.508	.933	.961	.874	.895
Private Dwelling Density (LS)	r	-.314	.543	-.269	-.430	.348	.440	.347	-.450
	P	.320	.068	.398	.163	.268	.152	.269	.264
Number of Water Allocation Licenses (LS)	r	.020	-.059	-.046	-.068	-.012	.004	.016	-.414
	P	.956	.871	.899	.851	.974	.991	.965	.307
Surface Water Consumption	r	-.004	-.114	**.636***	-.140	-.105	-.286	-.287	.455
	P	.990	.754	.048	.700	.772	.423	.421	.257
Area Burned	r	-.241	-.249	-.226	-.484	-.536	-.255	-.314	.075
	P	.532	.518	.558	.187	.137	.507	.411	.905
Area Harvested	r	.126	-.319	.454	-.083	-.233	-.309	-.269	-.457
	P	.747	.403	.219	.831	.547	.419	.483	.303
Surface Water Consumption Oil Sands	r	-.068	-.062	.117	.519	.238	.326	.477	.216
	P	.932	.938	.883	.481	.762	.674	.523	.862

Table 3.13. Pearson correlation coefficients (r) for the annual-quality dataset. Variables that were LOWESS smoothed are indicted by “LS”. * and ** (bold) indicate significant correlations ($p<0.05$ and $p<0.01$, respectively).

		TOC	Cl^- (LS)	TDN	TP	Na^+ (LS)	SC (LS)	$^-$ (LS)	Turbidity
Area Burned	**r**	**-.354***	.037	-.204	-.002	-.141	.076	.124	-.215
	p	.020	.812	.190	.992	.367	.628	.428	.222
Area Harvested (LS)	**r**	.166	-.036	.056	-.023	-.022	-.073	-.011	.219
	p	.282	.816	.718	.882	.888	.637	.944	.186
Surface Water Consumption	**r**	-.496	.532	-.274	.478	.420	**.580***	.446	-.178
	p	.101	.075	.389	.116	.174	.048	.146	.601
Surface water Consumption Oil Sands (LS)	**r**	.036	.181	.114	**.655****	**.587****	.115	.127	.243
	p	.863	.377	.580	.000	.002	.575	.536	.276

3.2.2.2 Multiple Regression

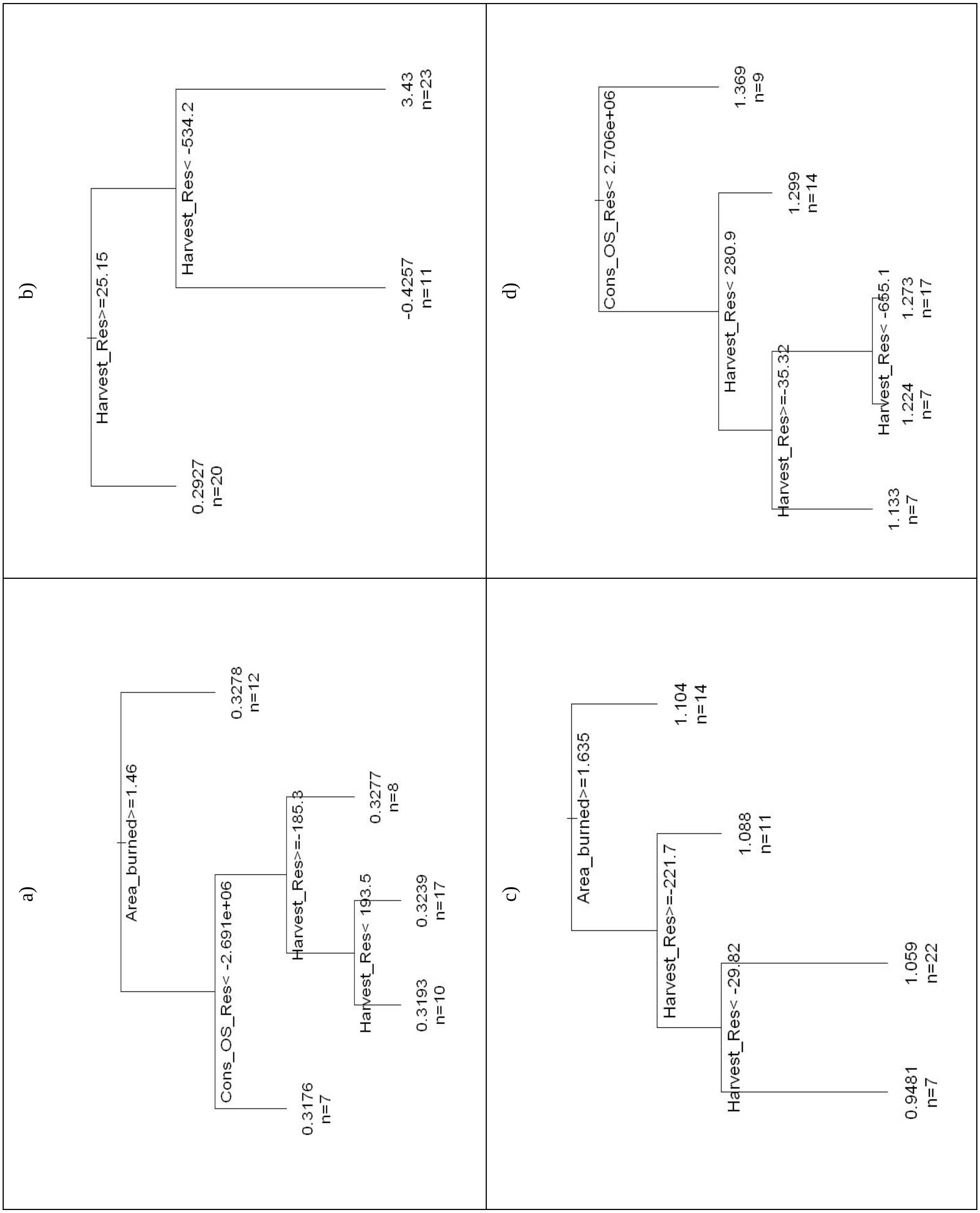
Multiple regressions were performed on all datasets. This was limited however, by the nature of the data. For instance, is it recommended that there be a ratio of *m*+2 replicates for a multiple regression test, where *m* is the number of independent, or stressor, variables (Zar, 1999). This recommendation was followed as close as possible. Further, it was impossible to produce statistically viable multiple regression models that included variables for which significant data were missing. Thus, these variables were not included in the analyses. It was necessary to discern the use of independent variables on the basis of different grouping characteristics. For instance, some groups were based on land use type as these typically had similar patterns of data availability. As well, some groups were built based on data availability alone. Significant models were produced only for the water quality datasets. For the annual-quality dataset, the only landscape variables that had sufficient data to perform multiple regressions were area burned, area harvested, and water consumption by oil sands. There was only one river response variable (Na^+) for which the model was significant ($p=0.025$). The model showed that consumption by oil sands was the best predictor among the remaining independent variables ($p=0.012$). The census-quality dataset also produced only one significant model. SC was regressed against number of farm units, cropland area, improved pasture area, and cattle density ($p=0.003$). The model showed that improved pasture area was the best predictor among the remaining independent variables ($p=0.008$).

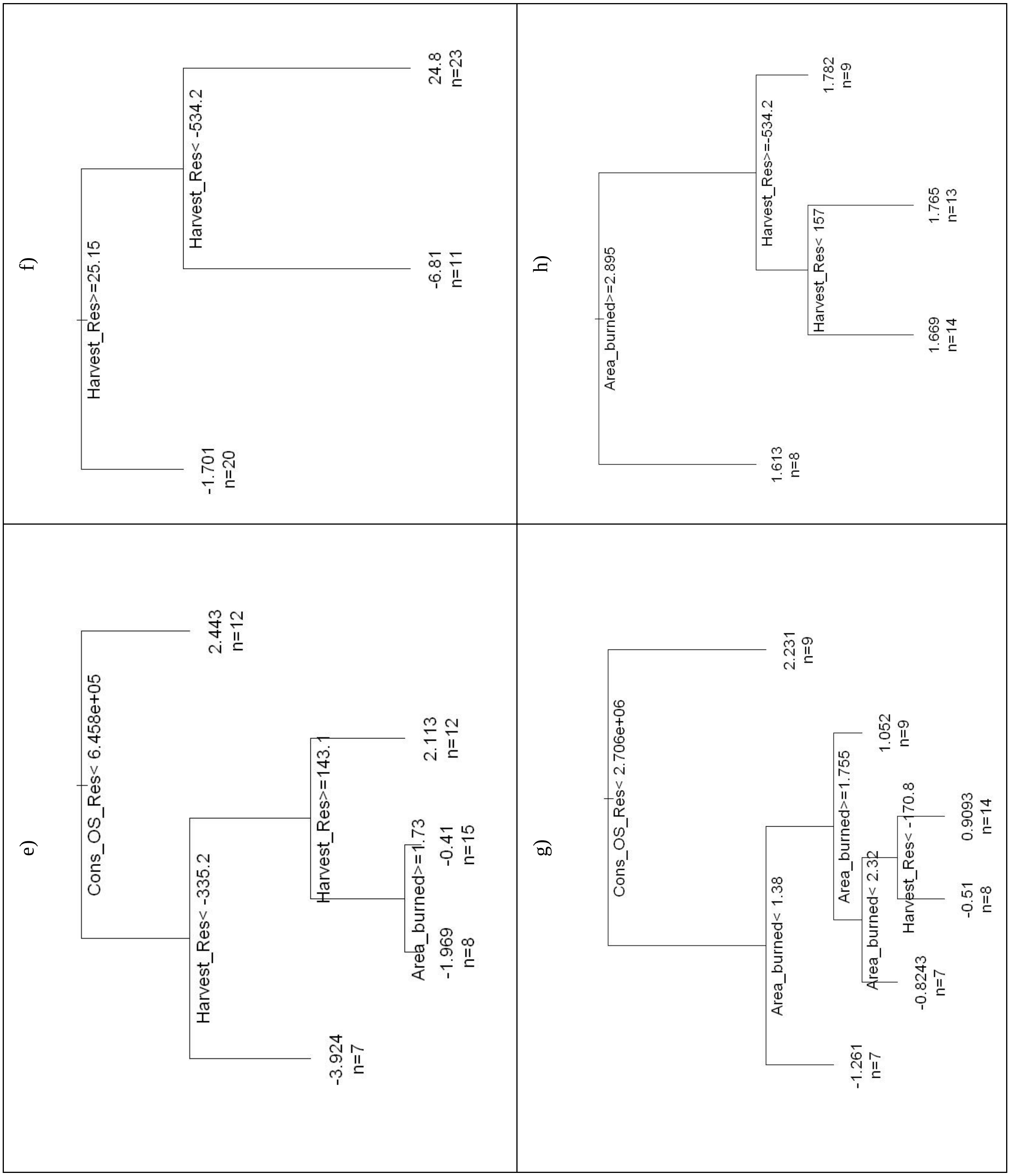
3.2.2.3 Regression Trees

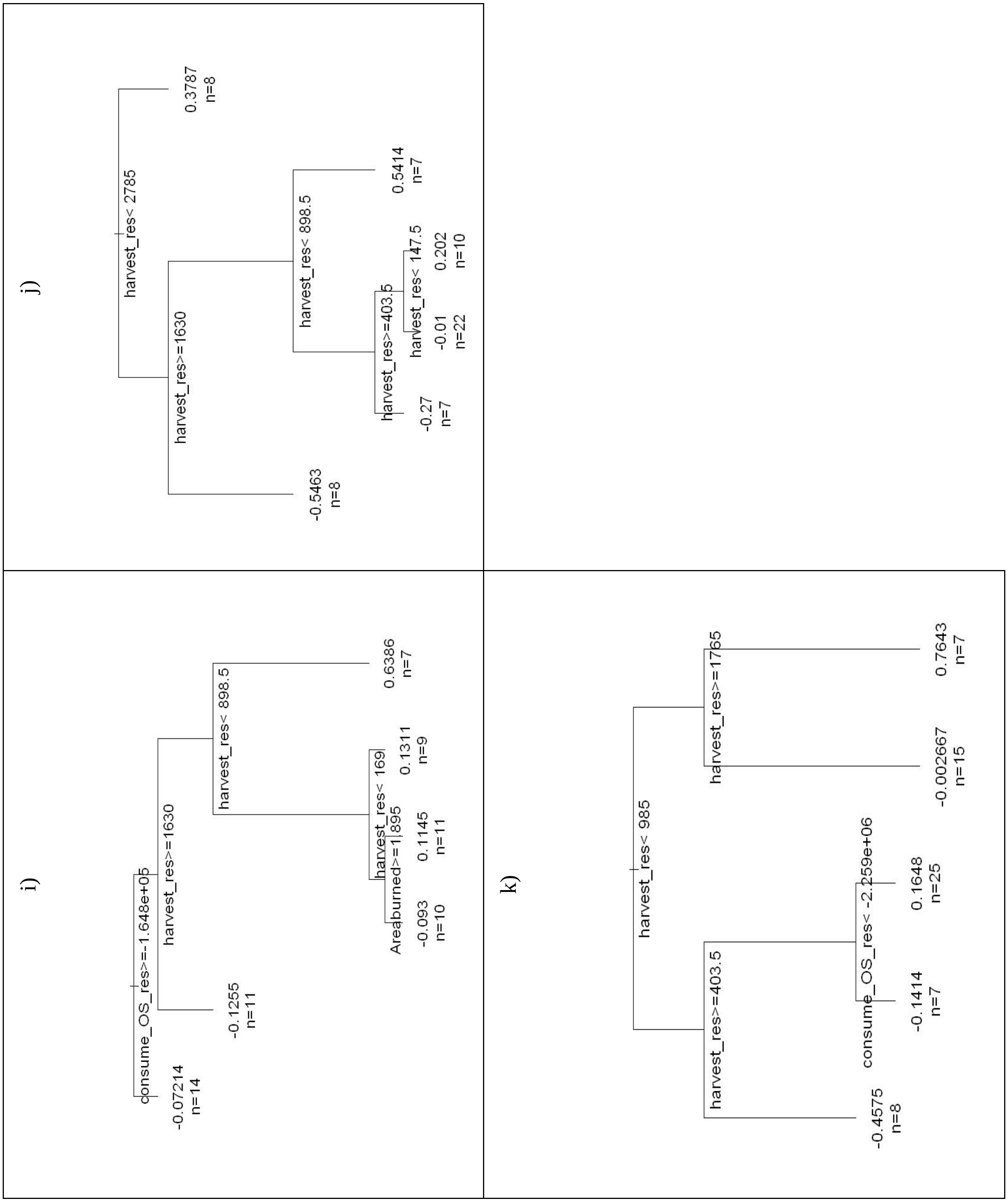
Regression trees were used to identify important landscape variables in relation to response variables among the annual-quality dataset. The first characteristic to appear in each regression tree was considered to be the landscape stressor that best explained the variance for the river response variable for which the regression tree was built. Three regression trees showed area burned as the first characteristic in the tree, or as explaining the most variance (TOC, TDN, turbidity). Three more trees showed consumption by oil sands as explaining the most variance (Na^+, TP, $^-$), while the last two trees showed that area harvested explained the most variance (Cl^-, SC) (Figure 3.7). The regression trees for TOC, TP, and Na^+ showed that the landscape variables that appear as the first characteristic in their respective regression trees were the same variables that showed significant relations for their respective simple regression models.

Regression trees were also performed on the annual-quantity dataset. Harvested area best explained the variance for average winter stream stage and average summer stream stage (Figures 3.7j, k). Average annual stream stage was best explained by consumption by oil sands operators (Figure 3.7i).

There were 11 river response variables in total (eight quality, three quantity) for which regression trees were built. Area burned was the first characteristic to appear in three of the regression trees (Figures 3.7a, c, h). Area harvested appeared as the first characteristic in four regression trees (Figures 3.7 b, f, j, k) Water consumption by oil sands operators appeared as the first characteristic in the remaining four regression trees (Figures 3.7d, e, g, i).


a)
Area_burned>=1.46
Cons_OS_Res< -2.691e+06
Harvest_Res>=-185.3
Harvest_Res< 193.5
0.3176
n=7
0.3193
n=10
0.3239
n=17
0.3277
n=8
0.3278
n=12
b)
Harvest_Res>=25.15
Harvest_Res< -534.2
0.2927
n=20
-0.4257
n=11
3.43
n=23
c)
Area_burned>=1.635
Harvest_Res>=-221.7
Harvest_Res< -29.82
0.9481
n=7
1.059
n=22
1.088
n=11
1.104
n=14
d)
Cons_OS_Res< 2.706e+06
Harvest_Res< 280.9
Harvest_Res>=-35.32
Harvest_Res< -655.1
1.133
n=7
1.224
n=7
1.273
n=17
1.299
n=14
1.369
n=9


e)
Cons_OS_Res< 6.458e+05
Harvest_Res< -335.2
Harvest_Res>=143.1
Area_burned>=1.73
-3.924
n=7
-1.969
n=8
-0.41
n=15
2.113
n=12
2.443
n=12
f)
Harvest_Res>=25.15
Harvest_Res< -534.2
-1.701
n=20
-6.81
n=11
24.8
n=23
g)
Cons_OS_Res< 2.706e+06
Area_burned< 1.38
Area_burned>=1.755
Area_burned< 2.32
Harvest_Res< -170.8
-1.261
n=7
-0.8243
n=7
-0.51
n=8
0.9093
n=14
1.052
n=9
2.231
n=9
h)
Area_burned>=2.895
Harvest_Res>=-534.2
Harvest_Res< 157
1.613
n=8
1.669
n=14
1.765
n=13
1.782
n=9



i)
consume_OS_res>=-1.648e+05
-0.07214
n=14
harvest_res>=1630
-0.1255
n=11
harvest_res< 898.5
0.6386
n=7
harvest_res< 169
0.1311
n=9
Areaburned>=1.895
-0.093
n=10
0.1145
n=11
j)
harvest_res< 2785
0.3787
n=8
harvest_res>=1630
-0.5463
n=8
harvest_res< 898.5
0.5414
n=7
harvest_res>=403.5
-0.27
n=7
harvest_res< 147.5
-0.01
n=22
0.202
n=10
k)
harvest_res< 985
harvest_res>=403.5
-0.4575
n=8
consume_OS_res< -2.259e+06
-0.1414
n=7
0.1648
n=25
harvest_res>=1765
-0.002667
n=15
0.7643
n=7

Figure 3.7. Regression trees for a) TDN, b) Cl^-, c) TOC, d) TP, e) Na^+, f) SC, g) $^-$, h) Turbidity, i) Average annual stream stage, j) Average winter stream stage, and k) Average summer stream stage. The trees represent the respective water quality or quantity dependent variables. Interpretation for each tree is described using Figure 3.7a (TDN) as an example: the tree is defined by four landscape variables and five terminal nodes ('leafs') in a hierarchical manner. Each landscape variable is split by a threshold rule, while each node is a grouping defined by the mean value and total number of observations that meet the threshold rule. The first landscape variable divisor is area burned. The second and third landscape variables isolate effects of water consumption by oil sands and area harvested, respectively, based on LOWESS smoothed values. The last landscape variable further isolates effects of area harvested based on LOWESS smoothed values.

3.4 Discussion

3.4.1 Landscape changes

The remote sensing imagery and GIS datasets provide a clear indication that LULC has changed dramatically over the lower ARB since the 1970s, highlighting potential threats to river water quality and quantity. It was found that areas of vegetative classes (deciduous, coniferous, and mixed forest) decreased, while areas of those classes resulting from anthropogenic actions (forest removal, agriculture, oil sands mining) increased. While there were no classifications done on years between 1976 and 2006, it is evident that this area of the ARB has been exploited for economic purposes. Specifically, there were large increases in forest removal and water abstraction. Other studies, conducted on much smaller spatial scales within the study area have found similar results. For example, Latifovic et al. (2005) found similar patterns of vegetation decrease (coniferous, deciduous, mixed) and anthropogenic activity increase (cut-blocks, oil sands mining) between 1992 and 2001 in their remote sensing land cover change assessment of the oil sands region near Fort McMurray, Alberta. They also showed that the majority of recently purchased (2000-2006) water allocation licenses belonged to oil sands operators. Schindler et al. (2007) came to a similar conclusion in their 2007 report on oil sands development and water use in the ARB. Large-scale river basins the world over are subjected to rapid economic development, the effects of which in regards to river response have been studied more recently. Generally, the larger the basin, the more opportunity there is for economic activities to occur across the landscape. A similar situation exists for the ~14 000 km^2 expanse of the Nile River delta, Egypt, where water quality has been threatened by rapid agricultural expansion and urban growth since the early 1980s (Abdulaziz et al., 2009). As well, in their study of land use change in the Tocantins River of central Brazil, a region approximately 175 000 km^2 in area, Costa et al.

(2003) found that the large scale deforestation and conversion of forested land to agriculture, along with an increased urban population since the mid-1950s, has resulted in reduced infiltration, increased runoff and a subsequent increase in river discharge.

The use of remote sensing in conjunction with GIS is a common practice in LULC assessment studies (Gove et al., 2001; Ahearn et al., 2005). While GIS and remote sensing seemed to accurately show the largest changes over time, there were several challenges encountered as a result of performing a landscape change over such a large spatiotemporal scale. It was necessary to collect multiple satellite images for each time period to cover the entire study area. The difficulty in using multiple images is the low probability of collecting a set of cloud-free, error-free images in the same season for a given year. Despite the fact that these images can be corrected to a certain extent, there still remains some uncertainty regarding accuracy of information obtained from the images. As well, it takes considerable time and meticulous effort to process each image, which becomes more cumbersome with the more images that are used. For example, a general rule of thumb for georeferencing an image is to use a first-order linear polynomial transformation and a root mean square error (RMSE) threshold of < 1 pixel (Jensen, 2005). This often requires the initial collection of at least 20 tie points and the subsequent deletion of tie points that have a large amount of individual error (i.e. greatly exceed the RMSE threshold) until the total RMSE for the remaining tie points are less than the specified RMSE threshold. Therefore, if 20 tie points are initially collected for each image and there are 17 images (9 in 1976, 8 in 2006), there would be 20 x 17 (340) tie points created in total, as was the case with this study. Further, georeferencing an image is only the first step in preparing an image for data extraction; radiometric correction and supervised classification must then be carried out before LULC classes can be analysed. It may be preferable to use satellite imagery with a larger spatial footprint or pixel size, such as MODIS (MODerate resolution Imaging Spectroradiometer), but this would greatly reduce spatial resolution, subsequently decreasing classification accuracy (O’Neill et al., 1996).

GIS datasets for some landscape features were also not available. For example, wetland drainage and impervious surface area density (urban and rural road networks) were hypothesized to be drivers of landscape change in the lower ARB because of their potential to contribute sediment and ions to the river ecosystem from urban areas (Trombulak and Frissel, 2000; Xian et al., 2007). However, they were excluded from analysis due to a lack of data. While it is possible to manually digitize a road network to a GIS from remotely sensed data, at such a large scale considered in this study it would be very costly in terms of time. This raises questions about its feasibility of use in a time restricted WCEA if not already available in a digital, low cost format. Arguably, poor data availability is not uncommon for such large spatiotemporal scales in terms of landscape, water quality, and water quantity (Reid, 1993; Seabrook et al., 2006).

3.4.2 Links between landscape change and river water quality and quantity

The simple, step-wise, and multiple regression tests produced strong, significant associations between water abstraction and water quality as well as agriculture and water quality variables. Water abstraction was described by water consumption for oil sands purposes and consumption for non- oil sands purposes. The simple regressions found that water abstraction was the major driver of TP and Na^+ concentrations (oil sands), as well as SC (non-oil sands), which all show significant positive associations with water consumption. Regression trees that showed water consumption by oil sands as the first node were of the river response variables TP, Na^+, $^-$, and average annual stream stage. The association between Na^+ and water consumption (oil sands) was further supported by a multiple regression test, which showed a positive association. The oil sands mining operations in the lower ARB is perhaps presently the largest threat to river water quality and quantity. The oil sands operations have increased in area and intensity since the late 1990s, with two large mining companies (Albian Sands and Canadian Natural Resources) beginning oil production operations in addition to the two major oil companies that had been operating mining projects since the late 1960s (Suncor and Syncrude). This has resulted in an increased number of licenses for water withdrawals, and subsequent removal of increased volumes of water from the lower ARB needed for the bitumen extraction process, the majority of which is not returned to the river (Schindler et al., 2007). This explains why water abstraction is associated with average annual stream stage, as shown by the regression tree (Figure 3.7i). As well, the removal of water likely results in an increased concentration of various pollutants. Recently, there has been work done which suggests the effects of water abstraction is more of a concern for water quality rather than water quantity in the lower Athabasca River, aside from winter low flow conditions (Page et al., 2010; John Pomeroy, pers. comm.). It may be postulated that there would have been an association observed between water abstraction and average winter river discharge, had adequate discharge data been available.

It is crucial for provincial and federal governments to implement flow monitoring at several points throughout this lower region in order for changes in discharge to be discernable. Production of a rating curve will permit conversion of stage to stream flows. The rates of change of stream flow are important to determine when the collective actions of oil sands operations have exceeded critical thresholds for stream health and integrity. The need for improved water monitoring specifically in the lower Athabasca is further emphasized by the projected increase in oil sands activities over the next decade (Timoney and Lee, 2009). Currently, it is difficult to determine if oil sand projects have had an effect on river discharge (and water quality) because data collection between RAMP and federal and provincial governments is spatiotemporally sparse and thus is unable to detect any measureable change (Dirk deBoer, pers. comm.; Office of the Auditor General of Canada, 2010; Schindler, 2010; Main, 2011). It's likely that the inverse condition is true, but the data does not show (or support) true conditions. Furthermore, project proponents need access to good, complete datasets in order to perform accurate CEAs. The use of poor datasets subjects the CEA to uncertainty and could lead to erroneous conclusions about the conditions of river water quality and quantity, rendering the entire CEA ineffective.

According to the regression trees for average winter stream stage and average summer stream stage, the landscape indicator that best explained these variables was area harvested, shown by its presence in the first node the respective regression trees. Forest removal activities are associated with increased stream flow for several reasons. First, forest removal via clear cutting subjects these areas to increased snowpack during winter. This snowpack is lost to spring freshet and may also be influenced by early melt due to the increased albedo of such open spaces, resulting in higher peak and baseflows (Buttle et al., 2000; Brooks et al., 2003; Burke et al., 2005). As well, large machinery is required to remove trees from stands, an activity that results in soil compaction which reduces infiltration, resulting in larger volumes of runoff (Pomeroy et al., 1997). This may also describe why SC and Cl^- are associated with area harvested, as seen in the first node of their regression trees. Large volumes of runoff have the capacity to increase SC with the introduction of dissolved solids and ions into the river, including Cl^- (Brooks et al., 2003; Squires et al., 2010).

One of the simple regression tests showed a negative association between TOC and area burned. Forest fires were also shown to explain the most variance for TOC shown by its presence in the first node of the regression tree for TOC. It is important to consider the effects of forest fires on river response because of their capacity to release ions and other nutrients from forest litter and soils, as well as hydrophobic soil conditions that often follow a burn, which alter runoff and flow regime (Brooks et al., 2003). Fires can be anthropogenic (prescribed burns, accidental) or natural (lightning) in source. There are several possible explanations for this negative association. First off, forest fires have the capacity to burn up a substantial amount of organic detritus material on the forest floor, so that rather than leaching large amounts of carbon into a stream or river, carbon is emitted into the atmosphere (Harden et al., 2000). As well, some studies argue that given adequate precipitation, as burned areas begin to revegetate in the months and years following a fire, nutrient uptake resumes, which reduces nutrient losses to surrounding streams and rivers (Lathrop, 1994). Trees, particularly aspen, have been shown to re-establish very quickly in Alberta's boreal forest (Chen and Popadiouk, 2002). The majority of the landscape in the lower ARB consists of coniferous and deciduous forest. Putz et al. (2003) suggest that large, hot fires can burn tree roots, promoting rapid preferential flow, or infiltration through large macropores, subsequently decreasing runoff and reducing the amount of contaminants reaching the stream as overland flow. Also, the forested areas of the lower ARB are comprised of large volumes of valuable commercial timber. Fire suppression occurs over much of the study area (Alberta portion), and these efforts are documented to have improved or increased since the early 1980s (Armstrong, 1999; Johnson et al., 2001; Cumming, 2005). Thus, fire suppression efforts to reduce or eliminate forest fire severity may have affected carbon release. In general, a few of the studies conducted in coniferous forest settings in boreal plain and shield sites have found little evidence of the effects of forest fire on stream or lake water TOC concentrations (Richter et al., 1982; Bayley et al., 1992; Mast and Clow, 2008). Area burned appeared in the first node of the regression trees for TDN and turbidity. Forest fires are associated with high volumes of runoff due to the development of hydrophobic soil conditions following a burn (Burke et al., 2005). This surface runoff promotes nutrient and sediment export resulting in increased turbidity and nutrient concentration in receiving streams (Mast and Clow, 2008).

Results of the regression tests also indicate that agriculture has a strong impact on river water quality. Simple regressions resulted in strong negative relationships between the number of farm units and Cl^-, Na^+, SC, and $^-$. There has been a steady decline in the number of farm units over the years. However, the total farmland area in Alberta has remained in static while the average farm unit size has experienced a steady increase over the years (Statistics Canada, 2008). While there were no significant relationships between average farm unit size and water quality variables, it may be suggested that given increasing farm unit size, the decrease in number of farm units acts as a surrogate for increased agricultural intensity, hence why strong negative associations are observed between the number of farm units and water quality variables Cl^-, Na^+, SC, and $^{-}$.

The multiple regression test conducted on SC showed that the variable that had the most explanatory power was improved pasture area (km^2). Improved pasture area is defined as pasture field areas that have been 'improved' by means of seeding (not for crop purposes), irrigation, fertilizing, or controlling weed growth, which according to Agriculture and Agri-Food Canada (1997), results in low risk of soil degradation. An increase in SC may be the result of reduced runoff. An increase in seeded area likely results in reduced soil erosion, thereby enhancing soil infiltration and decreasing runoff (Gray et al., 1985; van der Kamp et al, 2003). A decrease in runoff then likely decreases dilution effects and results in increased river SC.

The step-wise regression between TDN and the amount spent on fertilizer and water consumption (non-oil sands) shows that the amount spent on fertilizer best explains the variance in TDN. The purchase of fertilizer has been used as a surrogate for increased agricultural intensity in LULC studies because it represents one of the major material inputs within agricultural areas (Kerr and Cihlar, 2003). The most common chemical constituents found in fertilizers are nitrogen and phosphorus (Carpenter et al., 1998). According to Schindler et al. (2006), the production of nitrogen fertilizer in Canada has increased rapidly since the 1950s, indicating an increased demand for fertilizer. This corresponds to the increase in fertilizer purchases documented from each census year since 1986.

3.4.3 Method evaluation

Simple, step-wise, and multiple regressions in conjunction with regression trees were useful in this study for capturing the strongest associations between landscape stressors and river response variables. In general, regression trees produced results congruent with those of the simple and multiple regressions conducted on the annual-quality dataset in that the landscape stressors that showed significant associations with river response variables in the regressions appeared first in the regression trees as explaining the most variance. The landscape variables that appeared in the first node in the regression trees were considered to have the most effect on the respective river response variable. TOC, TP, Na^+, and SC were the only water quality variables from the annual-quality dataset which showed significant associations from regression analysis with any of the landscape variables and in the cases of TOC, TP, Na^+, the landscape variables that produced significant associations were emulated in the first node of their respective regression trees (area burned, consumption by oil sands, and consumption by oil sands respectively). Consumption by oil sands and area harvested appeared in the first node in four regression trees, while area burned appeared first in three regression trees.

Goetz and Fiske (2008) used the same statistical tools to examine relationships between the built environment and the biota of surrounding streams. Their approach involved testing the capacity of predictive models, which included regression trees to identify key land cover variables in relation to stream response variables. They found that the results of multiple regressions and regression trees were comparable based on variance explained by each, and stated that regression tree results were a useful addition to multiple regression for identifying landscape variables that are not easily incorporated into multiple regression analyses. Comparatively, Vayssières et al. (2000) examined the performance of predictive logistic regression models against predictive regression tree models in their study aimed at demonstrating methodology for predicting distribution of plant species. Their findings suggest that the regression tree models performed better than the logistic regression models in the majority of tests performed, indicating the strong predictive ability of regression trees for identifying key independent variables. Use of regression trees benefitted the study because of the capacity to handle missing values, ease of interpretation, and overall data exploration capability.

Regression trees were also performed on all remaining water quality and quantity response variables in the annual datasets, despite the lack of significant associations from resulting regression tests, because they provide a prediction of what landscape stressors are the most likely to exhibit an effect on river response. The regression trees for the remaining water quality variables, $^-$, Cl^-, TDN, and turbidity, showed that these variables were best explained by consumption by oil sands, area harvested, area burned, and area burned, respectively. Although there were no significant regressions between landscape variables and stream stage, the regression tree analyses offer a means for identifying potential key landscape indicators associated with water quantity. For example, average annual stream stage, average winter stream stage, and average summer stream stage were best explained by water consumption by oil sands, area harvested, and area harvested, respectively. Regression trees alone provide a method for determining which landscape stressors are likely to exhibit an effect on river response and proved to be a valuable tool for linking landscape change to river response in this study, as simple regressions did not show significant relations between stream stage and landscape variables.

Studies that have been successful in applying regression trees for predicting landscape effects on species or river response have typically utilized large datasets that include large numbers of replicates and numerous landscape variables (De'ath and Fabricius, 2000; Vayssières et al., 2000; Goetz and Fiske 2008). This indicates that the regression tree method performs better with larger sample sizes. In general, regression trees do not explain relationships between landscape stressors and river response as succinctly as regression tests, but because of their predictive capacity and ease of interpretation, they are easily used as a tool to determine major drivers of landscape change and their impacts on river response, especially over such large watershed scales where landscape inputs can be quite variable over space and time. When supported by regression tests, this provides a powerful method for identifying probable landscape sources of river stress. The regression trees produced in this study show that the landscape variables for water abstraction, agriculture and forest disturbance had the most effect on river response variables. Additionally, associations between landscape stressors and river response indicators that show consistency between regression tests and regression trees should be examined because of the potential to identify key landscape indicators for WCEA. Because consumption by oil sands also was significantly correlated with water quality variables in the regression tests, it can be suggested that water consumption by oil sands is one of the key landscape stressors exerting an effect on river water quality in the lower reaches of the Athabasca River.

There were several challenges associated with using the lower reaches of the Athabasca River as the case study area. First, because there were collectively only three water quality and quantity monitoring stations from which data could be collected, replicates were restricted to year (temporal) instead of location (spatial). Also, the lack of appropriate discharge data made the comparison between landscape change and river flow (volume) not possible. Discharge or flow (m^3/s) is commonly used in hydrological studies to describe river water quantity (Putz et al., 2003; Rood et al., 2008). Unfortunately, stream stage, or level (m), was the only variable from the WSC gauging stations in the lower ARB with data available over the defined time period, and thus was used as the measure of water quantity. Similarly, inclusion of certain water quality variables was limited to those that met the spatiotemporal criteria. There are several water quality indicators known to be associated with non-point sources in the lower ARB which could not be included in this study due to intermittent data collection such as trace metals, polycyclic aromatic hydrocarbons (PAHs), and other major ions (calcium, magnesium, potassium) (Headley et al., 2005; Keepers of the Athabasca, 2008; Kelly et al., 2010). The scientific rigour of this study was reduced by the small number of variables that could be included in the analyses.

Measuring cumulative effects of land use change is a difficult task because the interaction between multiple landscape stressors and their effects on the surrounding river system can be confounded by time lags. Time lags provide a possible explanation for why relations between stressors and effects are not readily observed. The amount of time it takes for an observable effect to occur in a stream or river, as a result of one or more activities on the landscape, is site-specific and depends on different ecological responses (Scrimgeour et al., 2008). In fact, according to Reid (1993), time lags can mask actual reasons for why a response is observed in the first place, such as anthropogenic activities that have occurred decades or centuries in the past. Future study of time lags in the lower reaches of the Athabasca is warranted.

3.5 Conclusions

Much of the WCEA research conducted in the past has been done on local, project- based scales, and has lacked a poor scientific foundation. This chapter evaluated some of the methods proposed in Chapter 2 for linking landscape change to river response. Remote sensing and GIS were found to be valuable tools for assessing large-scale landscape change. Simple, step-wise, and multiple regression analysis, as well as regression trees were conducted to explore associations among landscape change and river response. Collectively, it was found that water abstraction had the strongest effect on water quality, followed by agriculture, and finally the combination of forest disturbances, as evidenced by harvesting activities and fires. It was further demonstrated that by exploring associations among landscape change and river response through regression analysis in conjunction with regression trees provided a useful method for identifying key landscape variables in relation to river response over a large spatiotemporal scale. The identification and prediction of landscape variables that exhibit an effect on river water quality and quantity, demonstrated using the Athabasca River, Alberta as an example, is crucial for watershed cumulative effects assessment. Proper scientific methods guided by proper CEA practice, is key to moving towards improved CEA for watersheds and subsequent river protection.

Chapter Four- General Conclusions

4.1 Framework Analysis

Sound scientific methods are needed for identifying the major drivers of landscape change and their associated influences on river water quality and quantity. Chapter 2 offered a conceptual framework for improving WCEA. Argued was that identifying the landscape metrics or indicators of landscape change that hold the most explanatory power, or have the most effect, on river response indicators are those most valuable to WCEA. Chapter 3 evaluated results of different methods for linking landscape change to river response, specifically correlations, stepwise and multiple regression, and regression trees. These methods showed some strong associations between landscape stressors and river effects, and results were relatively consistent among linear regressions, stepwise regressions and regression trees. This was despite a lack of good spatiotemporal data. One of the major findings was that the Athabasca oil sands operations is the largest development (threat) to water quality in the study area based on associated impacts of water abstraction. Page et al. (2010) also concluded that water quality is more prevalent as an issue in regards to the oil sands industry, rather than water quantity. Agricultural activities represented by census data were also recognized as important drivers of landscape change because of their effects on water quality, as has been shown by others working at smaller spatial scales (Carpenter et al., 1998). As well, the combination of forest disturbances, explained by forest harvesting and forest fires, is also important when considering effects on the river system, shown in the regression trees.

The statistical methods evaluated in this thesis represent not an exhaustive list, but rather an appropriate list of techniques commonly used to link landscape change to river response over the large spatiotemporal scale of the lower Athabasca River. Other methods not explored in this thesis but could be useful in linking landscape stress to river response in other large watersheds includes multivariate regression, statistical modeling and trend analysis. Method choice must be based on the amount of data available, and what is appropriate for the study and be specific to the questions being asked.

Another important point made in Chapter 2 was the need for spatiotemporally expansive data for effective CEA. The irregular spatial and temporal monitoring over the lower reaches of the Athabasca River produced poor datasets which significantly restricted the types of methods that could be compared in Chapter 3. Poor data availability will weaken any evaluation through reducing scientific and statistical rigour.

The framework proposed herein highlights the need for defining an appropriate spatiotemporal scale upon which to conduct CEA. It was suggested that the spatial scale for WCEA be based on a river reach for large watersheds, rather than the scale of an individual project, or local activities. Even by narrowing the spatial scale of interest to a river reach, in this case the lower reaches of the Athabasca River (~88 000 km^2), large areas provide many challenges. Specifically, finding ample landscape data over such a scale proved to be problematic for several reasons. One, some of the landscape characteristics I intended to explore were unavailable digitally, such as road density network. While roads could be digitized into a GIS from topographic maps, logistically doing so for such a large area is unrealistic because of the significant amount of human resources required to do so – an option not available to proponents performing WCEA on a budget and deadline. Two, government datasets that were available contained many missing sample points due to mechanical failure or poor record keeping; a problem not uncommon over such a long time period. Three, missing data can also exist in remote sensing products in that obtaining cloud-free and error-free imagery for the desired season of a particular year is not a guarantee. Four, imagery collected in the 1970s has much lower spatial resolution than its technologically improved counterparts of the 1990s and 2000s. Arguably, there are algorithms available for image enhancement, but they are associated with a certain degree of information loss. A lack of data will introduce constraints to any good CEA efforts, but CEA must be done despite sparse data. CEAs must be designed so that the most obvious parameters pertaining to the landscape and receiving environment in question are considered and focused on.

It has been recognized that there is a need to scale up from local scale CEAs, but the spatial scale used is dependent on many factors such as data availability, the goal of the study, and the questions being asked. Studies that have been successful in identifying linkages between landscape change and river response at the regional scale do not attempt to quantify the landscape by means explored above due to difficulties involved. Rather, these studies resort to utilizing pre-existing datasets, defining study boundaries based on census delineations, subdividing the landscape into smaller spatial units, conducting a study over a maximum of three seasons, or using hydrological modeling (Richards et al., 1996; Costa et al., 2003; Meador and

Goldstein, 2003; Siriwardena et al., 2006). CEA must be based on a scale larger than the local boundary, but the larger the scale, the less is understood about interactions and cumulative effects (Reid, 1998). Attempting to link landscape change to river response over an entire river or even a reach as seen here can be not only problematic, but superfluous. I recommend that the spatial scale of analysis be based on a more intermediate (i.e. a few thousand square kilometers) scale. If data are still sparse, approaching CEA at the meso-scale makes it much more feasible in regards to time and cost for land managers and proponents to manually extract landscape data from satellite imagery and digital maps to meet the goal of their WCEA. Further, it may also be useful to explore landscape change based on data that exists *a priori* as a first step, and collect subsequent information as necessary. Finally, I recommend that proponents consider WCEA in terms of the unique characteristics of the watershed. The interacting effects of anthropogenic activities with natural landscape features (topology, climate, etc.) may not be readily apparent, which is why it is important to consider the makeup of the physical landscape (Rothwell et al., 2010).

The difficulties in obtaining landscape data were similar to those encountered when collecting river response data. The few monitoring stations in the lower ARB that have been established since the 1970s have considerable missing data. In the case of some WSC water quantity monitoring stations, there is inconsistency in data collection. For example, many WSC stations were able to provide flow (m^3/sec) data, but in later years only provided data for stream stage (m). The lack of appropriate rating curves for conversion of stream stage to discharge values makes this data virtually useless. The lack of adequate water quality and quantity data, which are federally and provincially managed and publicly available, is quite distressing and disgraceful.

Very recently, the Canadian federal government came forward and confessed the same. On 7 December 2010, the Commissioner of the Environment and Sustainable Development released a report to the House of Commons outlining Environment Canada's freshwater quality and quantity monitoring. He reported that monitoring programs were "not well managed to adequately monitor and report on the quality and quantity of Canada's surface fresh water resources". Further, in regards to the effects of oil sands development on river water quality and quantity, the report states that the monitoring program has "no…long-term data to track changes in water quality and aquatic ecosystem health in the river associated with oil sands development" and that the Department "has not determined whether it currently has an adequate number of stations to monitor flow related to oil sands development" (Office of the Auditor General of Canada, 2010). Canada is just one example of a wider spread problem with availability of suitable data for conducted large spatiotemporal scale CEA. According to a 2006 water monitoring report released by the United Nations, there exists several international water monitoring initiatives, in the developed world no less, that lack regular updating, affecting data reporting capacity (Faures, 2006). This raises a rather troubling question; if data collected are inadequate, inaccurate, or incomplete, how can linkages between stressors and effects be tested accurately in Canada? A more troubling question exists: is the move toward WCEA in Canada (Duinker and Greig, 2006) an exercise in futility? Lack of data itself therefore poses a potential threat to protection of freshwater resources.

It has been recognized for quite some time that CEA practice needs improvement. Only recently have efforts been made to address the need for improved CEA and provide recommendations. This thesis is part of one of those efforts as it is part of a large, Canadian research initiative, financially supported by the Canadian Water Network (CWN), aimed at addressing the issues of assessment, prediction, and management of the cumulative effects of multiple landscape developments for the purposes of improved cumulative effects assessment for river systems. The goal of this thesis was not to conduct a WCEA but to offer an example of choosing appropriate data and methods and how to apply them for the purposes of quantifying landscape change and identifying linkages between landscape and river response to proponents tasked with conducting WCEA. Identifying landscape effects on water quality and quantity is a key component of any WCEA. What comes after is an understanding of the effects of the landscape on river response, and how a proposed development will contribute to that landscape composition. The major contributions of this work to the CWN project is the identification of major landscape stressors, water abstraction and agriculture, in the lower ARB, one of Canada's most threatened, rigorously studied, yet least understood rivers. While the framework presented herein offers a Canadian example for improved cumulative effects assessment, the recommendations and need for improved WCEA in other countries makes it applicable for protection and sustainability of watersheds world-wide.

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