Water Use Metrics for the Determination of Environmental Impacts: Regional Assessment of Upstream Unconventional Oil and Gas

by

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Submitted by Kelsey Lane McAuliff in partial fulfillment of the requirements of

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ABSTRACT

Technology and innovation have increased the economic viability of horizontal drilling and multi-stage hydraulic fracturing, leading to the rapid increase in unconventional resource development in North America over the past fifty years. The quick development of the unconventional industry has been met with debate and criticism regarding industry methods/standards, volumes of water used, and impacts on the environment. In parallel, the field of water use metrics has also experienced a surge in popularity, most notably with the application of the *water footprinting* concept to evaluate the water use of businesses and countries alike. However, water use metrics evaluating water use impact have not been applied in the context of evaluating water use in unconventional oil and gas (UOG), which have instead focused on completing water use inventories. In this thesis, water use practices during UOG have been critically reviewed and analyzed to identify water sources and volume patterns. The review of water use practices in UOG is then used to develop criteria for evaluating common water use metrics to determine their applicability for inventorying and assessing the impacts of water use in UOG. A decision tree has been proposed and developed to facilitate the selection of water use inventory and impact metrics. Finally, a case study implements the selected Water Stress Index (WSI) framework to complete a regional water use inventory and midpoint impact assessment within the Montney unconventional play trend in British Columbia, Canada. Uncertainty analysis is performed under present and future scenarios to evaluate inherent parameter, model, and scenario uncertainties. While water use metrics do not replace site-specific assessment, they are important components of effective water management and can inform decision making, data collection and prioritization, and existing and future regional water stress conditions.

PREFACE

The contents of this thesis have been researched and developed by the author under the supervision of Drs. Rehan Sadiq and Kasun Hewage. A portion of this research has been submitted to a scientific journal for publication. All content and writing were completed by the author of this thesis.

- Parts of Chapter 2 and Chapter 3 of this thesis have been submitted as an article titled "Environmental Impacts of Water Use in Unconventional Oil and Gas Development: An Assessment on Method Evaluation and Selection" (McAuliff et al., 2017).

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LIST OF SYMBOLS AND ABBREVIATIONS

AB	Alberta
AD	Anderson-Darling goodness-of-fit test
BC	British Columbia
Bcf	Billion cubic feet
BCOGC	British Columbia Oil and Gas Commission
BHV	Bore hole volume
CAPP	Canadian Association of Petroleum Producers
CTA	Consumption-to-availability ratio
DALY	Disability adjusted life years
EMA	Environment Management Act
EUR	Estimated ultimate recovery
GIS	Geographic information systems
IRIS	Integrate Resource Information System
ISO	International Organization for Standardization
ISO 14040	ISO 14040 Environmental Management – Life Cycle Assessment – Principles and Framework
ISO 14044	ISO 14044 Environmental Management - Life cycle assessment - Requirements and guidelines
ISO 14046	ISO 14046 Environmental Management – Water footprint – Principles, Requirements, and Guidelines
KS	Kolmogorov-Smirnov goodness-of-fit test
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MCS	Monte Carlo simulation
Ministry	British Columbia Ministry of Natural Gas Development
NEB	National Energy Board
NEBC	Northeast British Columbia
NEWT	Northeast watershed assessment tool
NM	North Montney regional field
NPP	Net primary production
NWWT	Northwest watershed assessment tool
OGAA	Oil and Gas Activities Act
PDF	Potentially disappeared fraction of species
PNGA	Petroleum and Natural Gas Act
RH	Regional Heritage Regional Field
SETAC	Society of Environmental Toxicology and Chemistry
Tcf	Trillion cubic feet
UNEP	United Nations Environment Programme
UOG	Unconventional oil and gas
UUOGA	Upstream unconventional oil and gas activities
WA	Water availability
WFN	Water Footprint Network
WR	Water resources

WRPC	Water resources per capita
WSI	Water stress index
WSIn	Water stress indicator
WTA	Water-to-availability ratio
WU	Water use
WULCA	Working group on the assessment of freshwater use and consumption in life cycle assessment
WUPR	Water resources per resource

GLOSSARY

Blue water	Blue water is surface or ground water (Berger and Finkbeiner, 2013).
Consumptive water use	Consumptive water use results in a net loss of water to the originating source (watershed, drainage basin, etc.) (Jiang et al., 2013).
Conventional oil and gas	Conventional wells utilize a vertical wellbore to access oil and gas resources from high permeability reservoirs where oil and gas collect in pore spaces beneath an impermeable rock formation (Scanlon et al., 2014).
Decision Tree	A diagram which illustrates choices and outcomes of choices (Kurian, 2013).
Degradative water use	Degradative water use results in a “negative change in water quality” (International Organization for Standardization (ISO), 2015). Degradative water use is considered consumptive water use within certain metrics (e.g. Hoekstra et al. (2011)), and non-consumptive in others (e.g. Bayart et al. (2010)).
Direct water use	Direct water use refers to water utilized within the process under study (Jiang et al., 2013).
Estimated ultimate recovery (EUR)	The estimated amount of either oil or gas potentially recoverable from a well (Nicot and Scanlon, 2012).
Green water	Green water is bound within plants or soils and is otherwise unavailable for use (Bayart et al., 2010).
Gray water	In the context of water use metrics, gray water refers to a theoretical dilution volume necessary to assimilate wastewater to a pre-determined quality, typically based on water quality standards (Bayart et al., 2010).
Impact assessment	Per ISO 14040, impact assessment is “understanding and evaluating the magnitude and significance of the potential environmental impacts of a product system” (ISO, 2006b).
Indirect water use	Indirect water refers to the water volumes utilized within the supply chain feeding into the process under study (Jiang et al., 2013).

Inventory assessment	Per ISO 14040, inventory assessment is “compilation and quantification of inputs and outputs for a product” (ISO, 2006b). In water use metrics this may specifically entail compilation and quantification of water volume inputs and outputs.
Life cycle assessment	Per ISO 14046, lifecycle assessment is the “compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its lifecycle” (ISO, 2015)
Life cycle impact assessment	Per ISO 14040, lifecycle impact assessment is “aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system through the lifecycle” (ISO, 2006b)
Non-consumptive water use	Non-consumptive water use does not result in a net loss of water to the originating source (watershed, drainage basin, etc.), though it may, in whole or in part, decrease water quality (degradative water use) (Jiang et al., 2013).
Sensitivity analysis	Sensitivity analysis evaluates the contribution of each input to the results of the analysis (Helton et al., 2006).
Uncertainty analysis	Uncertainty analysis evaluates the uncertainty of input values on results values (Helton et al., 2006). Uncertainties can be differentiated into epistemic and aleatory uncertainties. Aleatory uncertainty (or type A or statistical uncertainty), is due to naturally occurring heterogeneity or randomness in data inputs. Epistemic uncertainty (or type B or subjective uncertainty) is due to systematic uncertainties due to a lack of knowledge or missing or incorrect data and models (Kreinovich et al., 2007)
Unconventional oil and gas	Unconventional wells target low permeability reservoirs (e.g. shale, limestone, siltstone, or sandstone) using the technologies of horizontal drilling and hydraulic fracturing to access oil and gas resources that are otherwise unavailable using conventional methods (International Risk Governance Council, 2013).
Water withdrawal	Per ISO 14046 “anthropogenic removal of water from any water body or from any drainage basin, either permanently or temporarily” (ISO, 2015). As such, water withdrawal does not differentiate between consumptive, non-consumptive, and degradative water uses.

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To my family, for their relentless encouragement

CHAPTER 1. INTRODUCTION

1.1. Research Motivation

Freshwater is a vital resource that interconnects the environment, humanity, and many economic activities (Scown et al., 2011). The energy sector accounts for 15% of global freshwater withdrawals, making it second behind agriculture, which accounts for 70% (i.e. municipal and household needs account for 10% and large industry accounts for 5%) (United Nations World Water Assessment Programme (WWAP), 2016). Globally, freshwater withdrawals have increased one percent annually since the 1980s and are forecasted to continue increasing over time.

Unconventional oil and gas (UOG) extraction relies on the processes of horizontal drilling and hydraulic fracturing to extract natural gas and/or oil from low permeability or "*tight*" reservoirs which were previously inaccessible with conventional methodologies. Technological advancements in horizontal drilling and hydraulic fracturing and the decline in conventional reserves have made UOG an economically viable alternative (Laurenzi and Jersey, 2013; Rivard et al., 2014). The combination of these two technologies has resulted in the rapid increase in the industry since the early 2000s, with natural gas being forecasted to account for almost half of North American natural gas by 2035 (Clark et al., 2013; Hultman et al., 2011; Scanlon et al., 2014; Vidic et al., 2013). However, the UOG industry places both consumption and pollution pressures on local water resources. As a result, regulator and industry decision makers must consider both resource production and the environmental impacts of energy production in conducting routine operations and planning for and mitigating the impacts of the industry into the future (Clark et al., 2013).

To this end, water use metrics, which have been developed in an effort to understand and support water conservation, sustainability efforts, and decision making, can be applied to quantify water use and estimate its associated impact (Cucek et al., 2012; Hoekstra et al., 2011). While these metrics have been developed to inform policy and decision making related to water management, they contain inherent uncertainties that must also be addressed and quantified.

1.2. Research Objectives

The volume of water used during upstream unconventional oil and gas activities (UUOGA) is dependent on a number of factors including geology, local water availability, applicable regulations, technology, and operator process. The impact of the quantity of water used is dependent on regional factors such as water availability, climate, population density, and competitive uses. The goal of this research is to quantify water use in UUOGA as a component of regional water use and evaluate regional water use through the application of a water use metric. To achieve the goal of this research, the following objectives are identified:

1. Identify patterns of water use during UUOGA and develop criteria to evaluate water inventory and water use impact metrics for their applicability to UUOGA
2. Evaluate water inventory and water use impact metrics and develop a decision tree for the selection of metrics applicable to UUOGA
3. Implement a selected metric (i.e. water stress index (WSI) midpoint metric) for the evaluation of water use within the Montney unconventional Play Trend case study region and identify and quantify the inherent uncertainties in the selected metric considering base-case and future scenarios

1.3. Organization of Thesis

This thesis contains five chapters, beginning with the first introduction chapter (Figure 1.1). Chapter 2 provides a literature review broken into two sections, the first on water use during UUOGA and the second related to water use metrics for inventory and impact assessment. Chapter 3 defines the water use metric evaluation process, develops a decision tree for water use metric selection, and finally prioritizes the Water Stress Index (WSI) framework for application in the case study performed in Chapter 4 (Pfister et al., 2009). Chapter 4 presents the implementation of the WSI framework through a case study evaluating water use within the unconventional Montney Play Trend in British Columbia, Canada. An uncertainty analysis is performed for the WSI framework and is also included in Chapter 4. Chapter 5 includes conclusions and recommendations for future research.

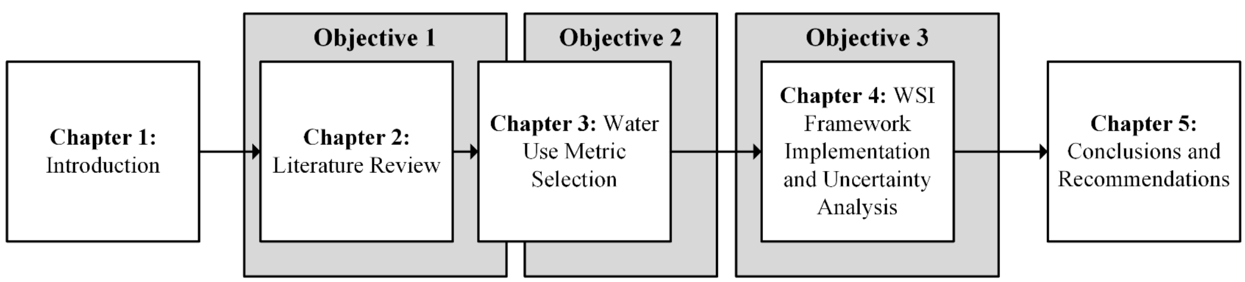

Figure 1.1 Thesis structure

CHAPTER 2. LITERATURE REVIEW

This chapter provides a review of literature relevant to UOG and water use metrics. A portion of this chapter has been submitted for publication as an article titled "Environmental Impacts of Water Use in Unconventional Oil and Gas Development: An Assessment on Method Evaluation and Selection" (McAuliff et al., 2017).

2.1. Unconventional Oil and Gas

Oil and natural gas collectively comprised 52.5% of global total primary energy supply in 2014 (International Energy Agency (IEA), 2016). Both conventional and unconventional technologies can be used to target oil and gas resources. Conventional wells differ from unconventional wells in that they are vertical as opposed to horizontal and access high permeability sandstone reservoirs. A conventional well bore targets the oil and gas which collects in the pore spaces beneath an impermeable rock formation (Scanlon et al., 2014). Unconventional wells utilize the technologies of horizontal drilling and multi-stage hydraulic fracturing to access low permeability reserves within shales, tight sands, or limestone.

Unconventional gas originating from shale (e.g., sedimentary rocks characterized by low permeability and clay-rich) can be specifically referred to as shale gas, while unconventional gas originating from sandstone, limestone, or siltstone (e.g. also characterized by low permeability) can be termed tight sand gas. It is of note that methane produced from coal seams (e.g. *coalbed methane*) can also be considered unconventional (International Risk Governance Council, 2013). Coalbed methane production involves de-watering coal to desorb methane bound to organic matter within the coal. Coalbed methane is outside of the scope of this thesis, which focuses on the development of unconventional resources (both oil and gas from shale and tight sands) using horizontal drilling and multi-stage hydraulic fracturing technologies.

Though individually the technologies of horizontal drilling and hydraulic fracturing are not new to the oil and gas industry, the combination of the two has facilitated the development of unconventional fields to the extent that unconventional development has been referred to as a *"revolution"* and a *"game changer"* in the energy market (Johnson and Johnson, 2012; Speight, 2013; U.S. Energy Information Administration (EIA), 2013). Figure 2.1 presents 2013 estimates of global technically recoverable shale oil (billion barrels) and shale gas resources (trillion cubic feet (Tcf)) from the United States Energy Information Agency's (EIA) assessment.

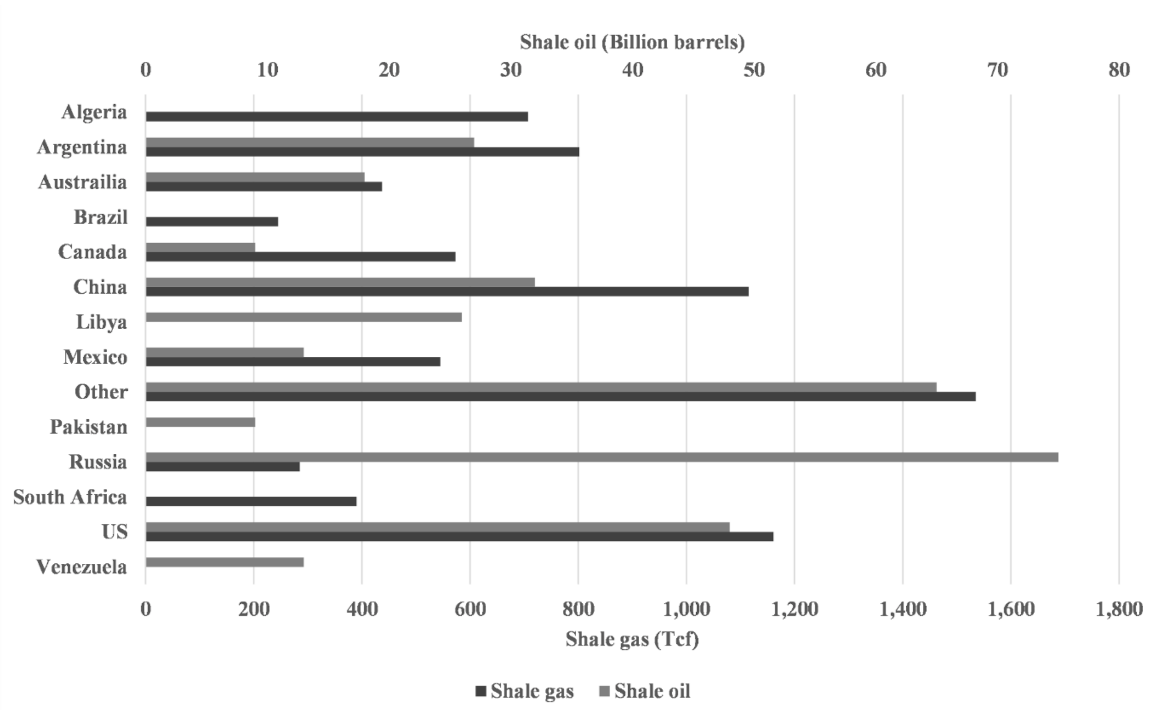

Figure 2.1 Technically recoverable shale oil (billion barrels) and gas (Tcf) reserves[1]

[1] Canadian shale gas resource estimate does not include unconventional gas produced out of the Montney and Doig Resource Plays in British Columbia and Alberta within Western Canada because these plays are classified as tight sand and siltstone. Figure adapted from (U.S. Energy Information Administration, 2013)

2.1.1. Unconventional Oil and Gas in North America

The unconventional "*revolution*" has primarily taken place in North America. Large-scale unconventional gas production began in the Barnett shale near Fort Worth, Texas when the combination of horizontal drilling and hydraulic fracturing made recovery of unconventional natural gas a commercial reality. In the United States (US) the major shale gas basins are the Bakken, Niobrara, Monterey-Tremblor, Permian, Eagle Ford, Barnett, Haynesville, Fayetteville, Woodford, and the Marcellus. Collectively, these plays produce both unconventional oil and gas and make the US the world leader in unconventional resource production (EIA, 2013).

To the North, Canada ranked fourth in global crude oil production, fifth in natural gas production, and fourth in natural gas export in 2015 (IEA, 2016). In Canada, unconventional resource development is occurring within shale/tight sand plays in British Columbia (BC) and Alberta (AB). While the BC and AB plays are the largest unconventional formations in Canada, smaller deposits are also found in Quebec, New Brunswick, Nova Scotia, and the Yukon and Northwest Territories.

Production rates from 2012 indicate that shale gas production in BC accounted for 25% of the annual total Canadian shale gas production. Within the four active unconventional plays in BC (i.e. the Horn River Basin, the Montney Play Trend, the Cordova Embayment, and the Liard Basin), the Montney Play Trend is the most active unconventional play with 89.4% of BC wells drilled within this play in 2014 (2013, 80%; 2012, 73%) (BC Oil and Gas Commission (BCOGC), 2014a; Rivard et al., 2014). The Montney Play Trend is estimated to contain 50% of the remaining recoverable raw gas reserves in BC (BCOGC, 2014a). As a result, UUOGA are forecasted to continue in the Montney Play Trend into the coming decades (BCOGC, 2013).

2.1.2. Montney Play Trend (British Columbia, Canada)

Physical characteristics within an unconventional play trend impact the demand for water resources during UUOGA. This section summarizes the Montney Play Trend and its physical characteristics within the province of BC (Figure 2.2).

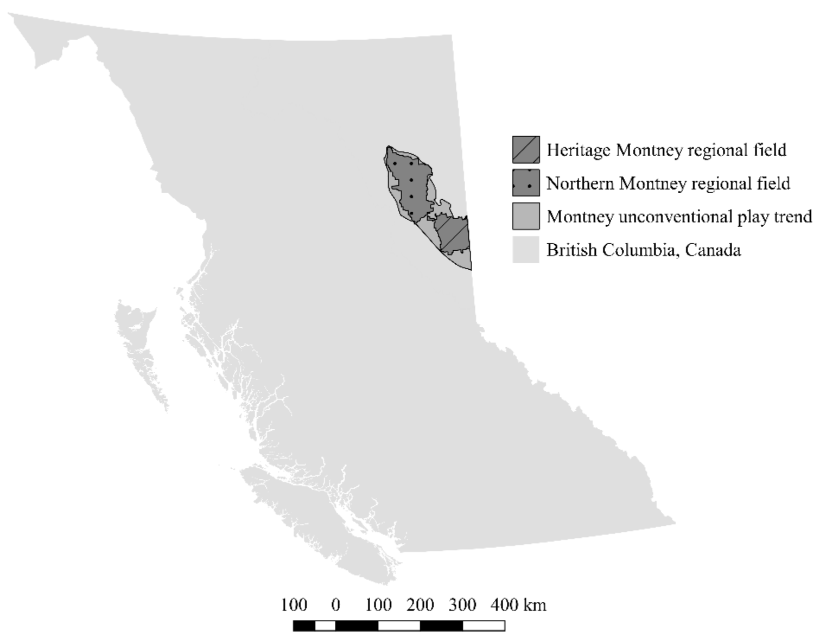

Figure 2.2 Montney play trend and regional fields[2]

The Montney Play Trend covers approximately 29,850 km^2 spanning northwest from the western AB border into BC. It is comprised of two regional fields, the Northern Montney (NM) (i.e. siltstone) and the Regional Heritage (RH) (i.e. dolomitic siltstone with shale) (Burke et al., 2011). Permeability throughout the Montney Play Trend is very low, requiring increased fracture stimulation to access oil and gas resources (Speight, 2013). Montney porosity varies from <10-

[2] Shape files for Figure 2.4 obtained from BCOGC Open Data Portal (BCOGC, 2016a, 2016c, 2016d)

35 % and has been found to have an overall clay content below 20% (Rivard et al., 2014). The thickness of the Montney ranges from 30-300 meters and increases from east to west, which has resulted in the development of oil zones in the east and dry gas zones in the west (Rivard et al., 2014). A summary of the physical characteristics of the Montney Play Trend and its NM and RH regional fields has been included within Table 2.1.

Table 2.1 Characteristics of the Montney Play Trend in British Columbia, Canada[3]

<table>
<tr><th>Parameter</th><th>NM</th><th>RH</th><th>Source(s)</th></tr>
<tr><td>Composition</td><td>Siltstone</td><td>Dolomitic siltstone with shale</td><td>(Burke et al., 2011)</td></tr>
<tr><td>Area (km^2)</td><td colspan="2">29,850 km</td><td rowspan="2">(BC Oil and Gas Commission, 2014b)</td></tr>
<tr><td>Thickness (m)</td><td colspan="2">30 - 300</td></tr>
<tr><td>Depth (m)</td><td>2,000 - 2,400</td><td>1,400 - 3,200</td><td>(BC Oil and Gas Commission, 2014; Rivard et al., 2014)</td></tr>
<tr><td>Total Organic Content (%)</td><td colspan="2">0.1 - 3.6</td><td rowspan="3">(Rivard et al., 2014)</td></tr>
<tr><td>Porosity (%)</td><td colspan="2">< 10 - 35</td></tr>
<tr><td>Clay Content (%)</td><td colspan="2">< 20</td></tr>
<tr><td>Water Saturation (%)</td><td colspan="2">25</td><td rowspan="5">(BC Oil and Gas Commission, 2014b)</td></tr>
<tr><td>Pressure (MPa)</td><td>14 - 86</td><td>14 - 53</td></tr>
<tr><td>Temperature (°C)</td><td>50 - 110</td><td>51 - 83</td></tr>
<tr><td>H_2S (%)</td><td>0 - 1.0</td><td>0 - 1.5</td></tr>
<tr><td>CO_2 (%)</td><td colspan="2">< 1</td></tr>
</table>

The Montney formation in the NM field ranges in depth between 2,000-2,400 meters, whereas the formation in the RH field varies between 1,400-3,200 meters. The percentage of hydrogen

[3] Table adapted from the BC Oil and Gas Commission (BCOCG, 2014b)

sulfide (H_2S) is minimally lower in the NM, ranging from 0-1.0% compared to 0-1.5 % in the RH (BCOGC, 2014a).

The low permeability of the Montney Play siltstone/shale makes it well suited for unconventional technologies. However, the unconventional industry is no exception to the fact that energy production is inextricably linked to water use. (IEA, 2015). Presently, the energy sector accounts for 15% of global water withdrawals and that demand is expected to increase with the projected increase in energy demand (IEA, 2015; WWAP, 2016). In the Montney play trend, water demand for unconventional activities is forecasted to continue in pace with the industry's development in the coming decades (BCOGC, 2013). Regulation of unconventional operations in BC falls within the jurisdiction of the BC Oil and Gas Commission (BCOGC). The regulatory framework for unconventional activities in BC is presented within Appendix B of this thesis.

While groups such as the Canadian Association of Petroleum Producers (CAPP) and the Montney Water Project are actively engaged in initiatives to reduce freshwater demands within the Montney Play Trend unconventional industry, water remains a critical component of the processes involved in the life cycle of an unconventional well (Canadian Association for Petroleum Producers, 2012; Geoscience BC, 2011). The processes which occur during the upstream unconventional well life cycle and the water inputs required during these processes are defined in Section 2.2.

2.2. Water Use in Unconventional Well Processes

The life cycle of an unconventional well can be divided into upstream (i.e. pre-production) and downstream (i.e. post-production) components. Upstream process steps include well site investigation, well pad preparation, well drilling, well completion, gas production, and well closure. Downstream steps include gas processing, gas transmission, gas distribution, and finally combustion (Figure 2.3). Within each of these process steps, water can be used either directly or indirectly. Direct water use refers to water volumes utilized within the process under study, while indirect water refers to the water volumes utilized within the supply chain feeding into the process under study.

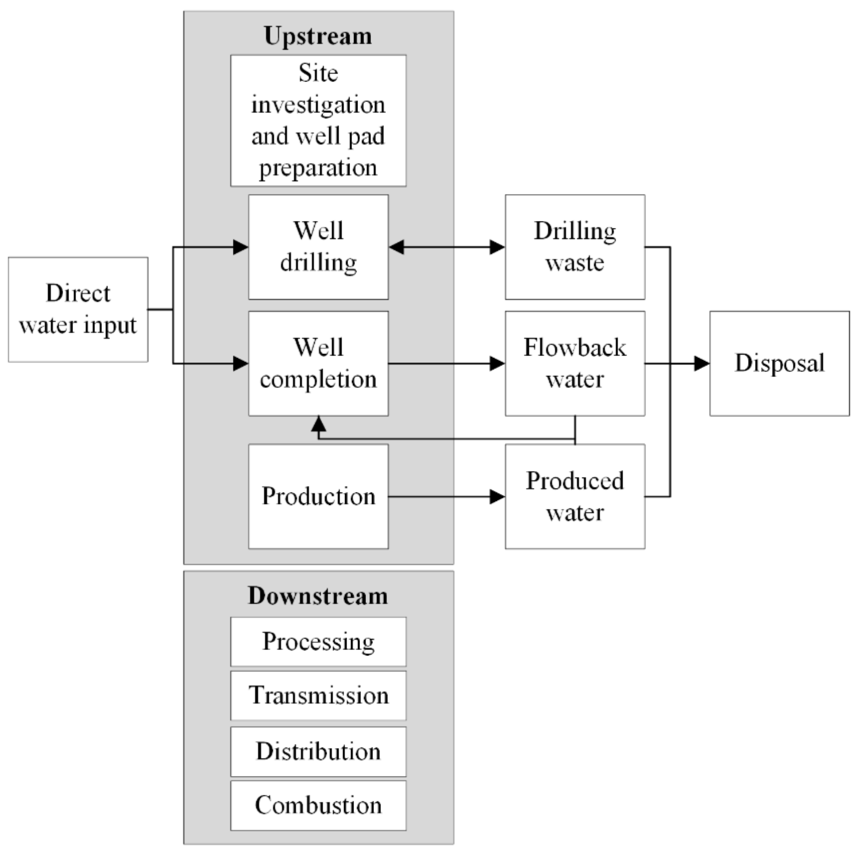

Figure 2.3 Direct water input in the upstream unconventional well life cycle

Site investigation refers to the process of evaluating a given location for its natural gas extraction potential and involves surveying for hydrocarbon reserves and a feasibility determination for constructing a well pad. Spatial and geological characteristics vary across formations which have the potential to impact production rates, technology requirements, and the economic feasibility of a given formation. Furthermore, variations within the same formation can result in differences in productivity between wells (Speight, 2013). Unconventional well pads can support several wells at a single well pad. Well pad preparation involves site grading, well pad construction, and access road construction. A well pad consists of well heads, fluid and chemical storage, machinery and vehicle parking, and in some cases lined storage ponds or above ground storage tanks for wastewater collection. During well pad preparation, water is consumed directly in the construction of the well pad, access roads, and maintenance/cleaning of equipment. Water is consumed indirectly through the production of supplies such as the diesel used to operate machinery.

Once the well pad has been prepared, well drilling occurs followed by the cementing of steel casing(s) within the borehole. Horizontal drilling allows an unconventional well to access more of the target formation from a single wellhead (Council of Canadian Academies, 2014). During well drilling, drilling mud is circulated within the well bore to remove cuttings, dissipate heat, and stabilize pressure between the well bore and the target formation. Drilling mud is typically water-based, but increasingly operators are using alternatives, such as oil-based muds, to drill the horizontal section of unconventional wells (Scanlon et al., 2014). Water-based drilling muds require direct water inputs to ensure adequate mud composition (Gregory et al., 2011). As drilling is completed, steel casing is cemented in place within the borehole to isolate the borehole from the surrounding formation. Water is used directly during cementing to produce cement.

Casings are cemented sequentially by decreasing casing diameter, beginning with the surface casing and followed with intermediary casing(s), before finally the production casing is cemented in place.

Though water is used to facilitate many of the upstream processes, hydraulic fracturing, which occurs during well completion, is the most water-intensive process out of all upstream activities (BCOGC, 2014b; Clark et al., 2013b; Jiang et al., 2013). Hydraulic fracturing involves the injection of water, proppant material (e.g. sand), and chemical additives under high pressure to expand fractures (created during perforation) within the target formation (Nicot and Scanlon, 2012). Proppant refers to granular material available in multiple sizes, either with or without coating. Proppant is injected with fracturing fluid to ensure opened fractures do not close during production which enhances permeability and improves the productivity of wells.

Proppant comprises approximately 8% of the overall make-up of fracturing fluid and chemical additives comprise about 1%. Operator practices determine how various proppant sizes are injected, either collectively or in sequence, to optimize the flow of hydrocarbons from the formation (Bortolan and Kotousov, 2013). Though the chemical composition of fracturing fluid is specifically selected by operators to optimize well production, the main constituents of fracturing fluid (not including water) which make up the largest percentage of chemical additives are acids (0.11%), surfactants (0.08%), friction reducers (0.08%), clay stabilizers (0.05%), and gelling agents (0.05%) (Jiang et al., 2011). Water is used indirectly during the hydraulic fracturing process in the mining, manufacture, and transport of proppant and chemical additives. Time to complete drilling and hydraulic fracturing can range from weeks to months and varies between operators and between wells, however, these processes necessitate that the required volume of water is available at once (J.-P. Nicot et al., 2011).

Once a well has been fractured, fluids, known as flowback water (FW), return to the surface. FW can consist of previously injected fracturing fluid (i.e. water, chemicals, and proppant) and water originating from the formation. FW can have a high salt content and contain metals from contact with the formation. FW specifically refers to the wastewater that initially returns to the surface during the first 10-14 days following hydraulic fracturing. After the flowback period, the fluids returning to the surface are termed produced water (PW) (Jiang et al., 2013). PW can contain both injected and formation water and is often characterized by high salinity, metals content, and may also include naturally occurring radioactive materials (NORMs) (Carnegie Mellon Scott Institute for Energy Innovation, 2013; Jiang et al., 2013). The highest rate of fluid returning to the surface occurs initially after fracturing and declines over time, with the rate of decline being dependent on well specific factors such as geology, technology, and operator techniques (Kondash and Vengosh, 2015; Nicot and Scanlon, 2012).

Disposal practices and regulations differ between jurisdictions, however, FW and PW are often not discharged back into the environment and must be either treated and/or reused or sent for disposal, thus upstream water use is considered consumptive. FW and PW volumes that are not immediately transported offsite for disposal or reuse are stored temporarily within lined storage pits or above ground storage tanks onsite and/or offsite. Figure 2.4 illustrates a schematic of FW and PW generation and contribution to inputs for subsequent wells. The volume of FW and PW available for reuse is dependent on the volume of FW and PW generated from a well, the composition of FW and PW, the total fluid demand for a subsequent well, and the infrastructure available to store and transport wastewater between wells. Operator technologies can maximize the volume of FW and PW available for recycling/reuse. For example, pipelines and storage tanks allow an operator to collect volumes of FW/PW so that it can be transported to subsequent well locations.

The benefits of reuse/recycle of produced water are two-fold. First, an operator can reduce the volume of freshwater and second, the operator can reduce the volume of wastewater requiring disposal. While the reuse of waste fluids is an emerging practice and offers the benefits of reducing firewater and waste disposal demands, there are inherent risks to the process. For example, reusing/recycling waste fluid requires that the fluids be transported and stored in the appropriate locations to be available for future fracturing operations.

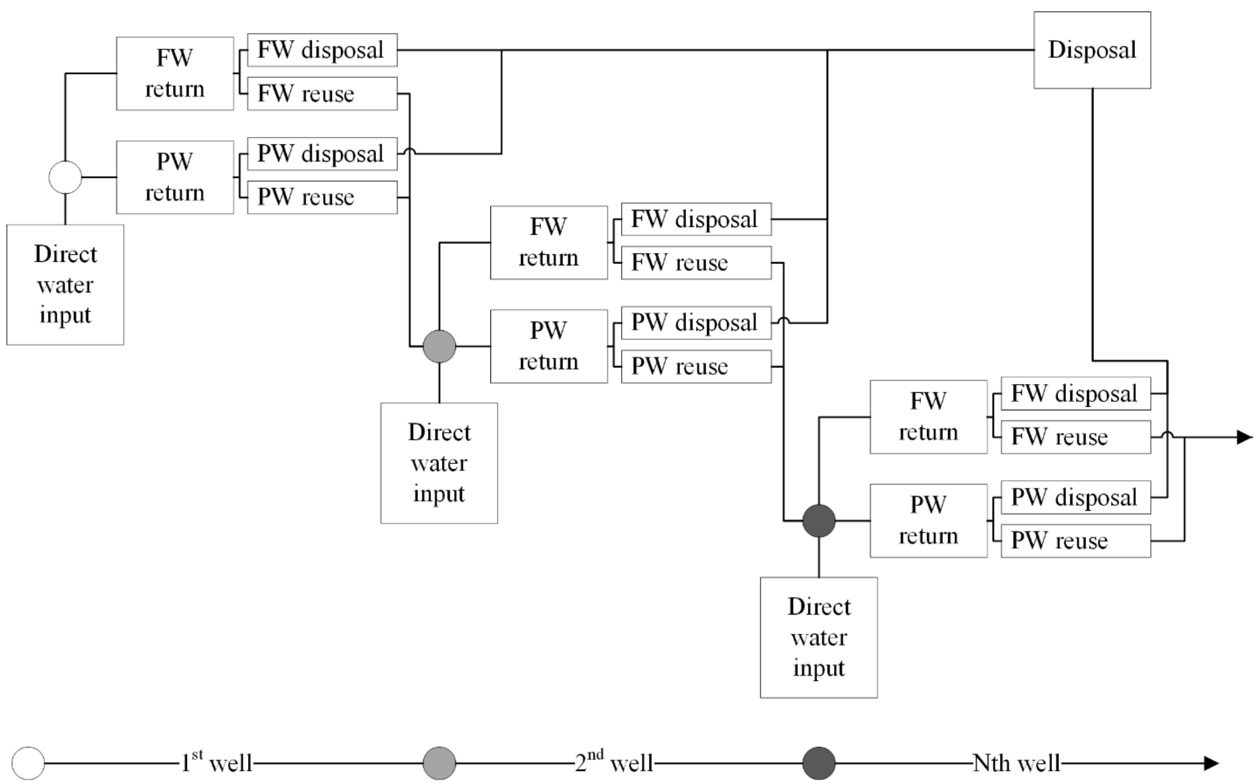

Figure 2.4 Direct water use, flowback, and produced water reuse cycle in UUOGA

Increasing the transportation and storage requirements of waste fluids may also increase the risk of unintended release of fluids during transportation and storage (Laurenzi and Jersey, 2013).

The simultaneous increase in UOG development and awareness of global water concerns has led to criticism of the unconventional industry regarding water quantity, water contamination, and management of wastewater (Nicot et al., 2014; Rivard et al., 2014; Vidic et al., 2013). Water use metrics, defined in the following section, provide techniques for quantifying water use and its impact, as evaluated through assessment of environmental indicators.

2.3. Water Use Metrics

Water scarcity, referring to the lack of water available to meet daily needs, is a reality currently faced by approximately a third of the world's population. The major drivers resulting in water scarcity include local water availability and the rate of human use. Factors such as climate change and seasonal variability in the hydrologic cycle can exacerbate water scarcity, while technology and innovation, with respect to wastewater recycling/reuse and storage capacity, reduces local water scarcity (Berger and Finkbeiner, 2010). As demand increases, freshwater resources are increasingly under pressure in terms of both consumption and pollution (Hoekstra and Mekonnen, 2012; Kounina et al., 2012). Global estimates for water demand forecast that demand will continue to increase due to rising population and industrialization (Hoekstra and Mekonnen, 2012; WWAP, 2016). The increasing global focus on issues of freshwater scarcity and degradation has influenced the development of water quantification and impact metrics which aim to synthesize relevant environmental information through the calculation of a single metric capable of informing decision makers interested in reducing adverse environmental outcomes (Ridoutt et al., 2015).

2.3.1. Emerging Field

Water use metrics have been developed in an effort to enhance policy and decision making in relation to regional and global water management. Water use metrics can be completed as a component of life cycle inventory (LCI) and life cycle impact assessments (LCIA), which aim to evaluate environmental impact along a supply chain. However, a growing emphasis on water use and its regional and global impacts has inspired water quantification and impact metrics to be completed as stand-alone analyses. This field experienced a surge in popularity in the early 2000s and has been applied on scales from individual businesses and organizations to countries with the intent to support policy and decision making. These studies have fostered an understanding of the regional and global nature of local freshwater consumption and global consequences to problems such as water scarcity and pollution (Hoekstra and Mekonnen, 2012).

The virtual water concept, which refers to evaluating the volume of water embedded within a product, process, or supply chain, was initially introduced to provide policy and decision makers with data and information regarding water consumption (Hoekstra and Mekonnen, 2012). This provided opportunities to initiate water reductions or externalize water use thereby conserving local water resources (Berger, 2014; Hoekstra and Mekonnen, 2012). The original concept of virtual water has since been expanded upon, as practitioners have developed and refined methods and indicators to address the unique challenges presented by water use quantification and impact assessment.

Water use analysis presents unique challenges that set it apart from other life cycle studies such as carbon footprints. For example, carbon footprints combine the point source emissions of major greenhouse gasses and express results in units of carbon dioxide equivalents to understand the contribution of the emissions on global climate change (Berger and Finkbeiner, 2010). Water footprints, however, require a much more localized assessment as the water availability experienced within a given area of study and is not necessarily indicative of the water availability within the surrounding region (Brown and Matlock, 2011). Water resources are subject to consumption and degradation pressures around the globe and their availability is determined in part by the quality and in part by the hydrological cycle, which is subject to both spatial and temporal variability. Technologies such as water treatment, water storage, and effective water management can buffer the impacts of variable quantity and quality (WWAP, 2016).

Many published water use metrics are available today and have been applied to a wide range of industries. The different methodologies within each of the metrics allow a practitioner to select the most appropriate metric(s) for the intended scope and objectives of a study. Water use metrics can be generally classified as inventory, midpoint, or endpoint assessments (Berger and Finkbeiner, 2010). Water use inventories are a summation of water inputs that occur within a defined scope of assessment. Midpoint metrics evaluate water use impact in terms of indicators that occur in the middle of a cause-and-effect chain. Finally, endpoint metrics evaluate damages due to water use on ecosystem, resource, or human health areas of protection.

Inventories report volumetric results but can also report results as a ratio of water used/consumed to product produced or costs. Midpoint and endpoint metrics report results either as volumes or as impact indices which each have their merits as well as disadvantages. Volumetric indices are generally thought to be accessible to a broader audience, but can also result in misconception without an appropriate discussion about the interpretation of results. For example, within the field of water use metrics, gray water represents a theoretical dilution volume, not a physical volume of waste water. Conversely, though the results of impact indices can be more comprehensive in their ability to capture aspects of water use impact such as water scarcity, the results may be more challenging to interpret (e.g. results reported in terms of disability adjusted life years) (Bayart et al., 2014; Berger and Finkbeiner, 2013). Water use metrics can be completed as stand-alone analyses or as a component within a broader LCI or LCIA (Kounina et al., 2012).

2.3.2. Water Use Metric Guidelines and Standards

Inconsistency in both terminology and applied methodology across studies implementing water use metrics has made comparison prohibitive, and at times has generated contradictory results, creating concern that a lack of consistency could jeopardize the impact of future studies (Kounina et al., 2012; Ridoutt et al., 2015; Ridoutt and Pfister, 2010). In an effort to standardize the terminology and methodological implementation, the Water Footprint Network (WFN) and the United Nations Environment Programme (UNEP)/Society of Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative's working group on the Assessment of Freshwater Use and Consumption in Life Cycle Assessment (WULCA) developed guidance documents to support practitioners in their analysis. In addition to guidance documents, the International Organization for Standardization (ISO) developed standard 14046 (ISO 14046) *Environmental Management – Water footprint – Principles, Requirements and Guidelines* which details the effective practice of water footprint analysis (Berger & Finkbeiner, 2010; Hoekstra et al., 2011; International Organization for Standardization (ISO), 2015; Kounina et al., 2012; Ridoutt & Pfister, 2010).

The 2011 WFN published the *Water Footprint Assessment Manual: Setting the Global Standard* (hereafter referred to as the Manual) guideline document for the completion of a water footprint per Hoekstra's method (hereafter referred to as Hoekstra's water footprint). This document is loosely structured around the four phases of Hoekstra's water footprint which include goal and scope definition, water footprint accounting, sustainability assessment, and response formulation. In addition, the Manual includes chapters on terminology (Chapter 3), limitations (Chapter 6), and challenges (Chapter 7). Chapter 6 *Limitations* defines a "non-exhaustive" list of limitations of the water footprint as an analysis methodology and its application limitations for companies and governments. The Manual speaks to the limited lens through which Hoekstra's water footprint evaluates environmental impacts. This limitation is not exclusive to Hoekstra's water footprint and even the most comprehensive water use metric, or combination of water use metrics, evaluates environmental impact through the lens of water resources and as such has not considered all other relevant environmental issues. As such, the Manual recommends the completion of Hoekstra's water footprint with other analytical methods. The glossary within the Manual contains terms and definitions specific to Hoekstra's water footprint but is not necessarily consistent with those defined in ISO 14046. For example, where ISO 14046 suggests qualifiers to define the impact category of water footprints, the Manual defines qualifiers based on the scope of study (e.g. water footprint of national consumption or water footprint of a business).

ISO 14046 (2015) specifies key terminology, principles, methodological frameworks, reporting requirements, and critical review guidelines applicable to completing a water footprint assessment. Notably, ISO 14046 (2015) reserves the use of the term *water footprint* for studies which comprehensively quantify potential environmental impacts relating to water. ISO 14046

(2015) also includes provisions for the use of qualified water footprint terminology (i.e. water scarcity footprint) for studies which do not meet its principles of comprehensiveness. This thesis will generally refer to these metrics as *water use metrics* because comprehensiveness must be determined by a practitioner based on study specific parameters.

The procedural components of ISO 14046 refer heavily to ISO standard 14044 (ISO 14044) *Environmental Management – Life cycle assessment – Requirements and Guidelines*, which defines the methodological and reporting requirements and terminology applicable to life cycle analysis in its application as an environmental management technique. Building off of ISO 14044 (and ISO standard 14040 (ISO 14040) *Environmental management – Life cycle assessment – Principles and framework),* ISO 14046 defines terminology, data input, characterization, comparison and limitation considerations, and requirements specific to the evaluation and interpretation of water use metrics[4] (ISO, 2006a). The life cycle approach presented in both ISO 14044 and ISO 14046 considers the use of resources and releases to the environment throughout the life cycle of a product, process, or organization from a predetermined scope (e.g. cradle-to-grave). For both LCIA and water impact assessments, this is accomplished through the completion of four phases; goal and scope definition, inventory analysis, impact assessment, and results interpretation (defined in Section 4.2 *Phases of an LCA* and Section 5.1 *General Requirements* within ISO 14040 and 14046, respectively).

The limitations of water use metrics are discussed throughout ISO 14046 (2015). Comparative limitations between studies are addressed in Sections 6.3 *Comparative Studies* and Section 7 *Critical Review* of ISO 14046 (2015), which establish baseline considerations for ensuring

[4] Referred to as water footprints in text.

consistencies between functional units and methodologies for comparative studies, while also defining the importance and applicability of critical review (ISO, 2015). Section 5.6 of ISO 14046 (2015), *Limitations of water footprint*, identifies limitations caused by inherent uncertainties and the impact that uncertainty and variability can have on water use metric results.

2.3.3. Uncertainties in Water Use Metrics

There are many, sometimes conflicting, definitions for the types of uncertainty (Sadiq and Tesfamariam, 2009). This thesis will adopt the general classification of uncertainty into two groups: aleatory and epistemic uncertainty. Aleatory uncertainty (also known as type A or statistical uncertainty), refers to the uncertainty caused by naturally occurring heterogeneity and variation in data inputs, while epistemic uncertainty (also known as type B or subjective uncertainty) generally refers to systematic uncertainties due to missing or incorrect data and models as a result of lack of knowledge (Kreinovich et al., 2007). Theoretically, epistemic uncertainty can be reduced through technological advancements in monitoring and measurement or improved modeling capabilities, however, the same does not apply to aleatory uncertainty which is an inherent reality for complex and variable processes (Beven, 2015).

Sources of uncertainty can be categorized into three groups; model, parameter, and scenario uncertainty. Model uncertainty can result due to variation in the methodologies between models (Lloyd and Ries, 2007). Water use metrics can provide an initial understanding regarding the quantity of freshwater use and its associated impacts on ecosystem quality, however, each of the different metrics which evaluate the ecosystem quality (all midpoint metrics referenced within Chapter CHAPTER 3) employ different mathematical relationships to evaluate cause and effect chains (Kounina et al., 2012). The difference in capabilities of each of these metrics and the impact that these differences impart on metric results reflects model uncertainty. ISO 14046 (2015) alludes to the inherent model uncertainties associated with water use metrics in its recommendation for completion of multiple metrics in parallel. Furthermore, when considered in the context of overall environmental decision making, water use metrics exclusively evaluate water-related impacts and as such must not be considered as comprehensive environmental impact indicators. While cited as a limitation of water use metrics within ISO 14046 (2015), the model uncertainty resulting from the inclusion of water use metric(s) within a broader comprehensive assessment of environmental impact is also of note.

Parameter uncertainty is the uncertainty generated from input values due to inherent variation, imprecision in measurements, or unavailable values (Lloyd and Ries, 2007). In the context of water use metrics, parameter uncertainty can arise due to missing data or erroneous data estimates. Finally, scenario uncertainty is the variability in scenario characteristics, such as spatiotemporal change, that can result from differences across defined scenarios (Lloyd and Ries, 2007). During scope definition of a water use metric assessment, a specific scenario under which water use will be evaluated is defined. The assumptions made in scenario definition can have a significant impact on the reliability and overall uncertainty of results.

Since water use metrics involve inherent uncertainty, methodological and data limitations must be addressed with adequate assumptions and discussion. Transparent communication of assessment scope, functional units, limitations, and interpretation of water use metrics is critical and hence should be included to avoid misleading results.

The next section provides a review and comparison of studies which have evaluated water use during UUOGA. These studies can be generally classified as water inventories and midpoint metrics.

2.3.4. Water Use Studies for UUOGA

Water use studies have been completed for the major unconventional plays in the US. The majority of the studies have been water use inventories considering either direct, or, direct and indirect water use during the life cycle of an unconventional gas well. These studies have applied inventory and midpoint water use metrics to establish state-of-the-practice water use results for individual shale gas plays or to compare results. Kondash and Vengosh (2015) presented a water use inventory, reporting volumes of water use over the total volume of gas produced as water use intensity values for the major shale gas plays in the US. State-of-the-practice water use studies have been completed for the Marcellus Play in Pennsylvania, US (Jiang et al., 2013; Laurenzi and Jersey, 2013) and for the Bakken Play in North Dakota/Montana, US (Scanlon et al., 2016). Both Marcellus studies considered direct and indirect water use over the life cycle of a well, considering life cycle stages of well pad preparation to gas processing and transmission (Jiang et al., 2013) and well pad preparation to electricity generation (Laurenzi and Jersey, 2013).

There are three general types of comparative analyses completed within unconventional gas water use studies. First, water use inventories, including water use and wastewater generation, are compared between shale gas plays (Kondash and Vengosh, 2015; Nicot and Scanlon, 2012). Secondly, water use inventory results are compared across multiple fuel types (Clark et al., 2013; Scanlon et al., 2014; Scown et al., 2011). Finally, water use inventories and midpoint assessments are completed within defined scenarios to compare water use (Clark et al., 2013;

Jiang et al., 2013; Scanlon et al., 2016). These scenarios vary by time and compare past, present, and future water use and wastewater recycling rates.

Published literature evaluating water use in UOG operations in North America generally considers three processes when evaluating the direct water contribution during upstream processes; well drilling, cementing, and hydraulic fracturing (Clark et al., 2013; Nicot et al., 2014; Nicot and Scanlon, 2012; Scanlon et al., 2014). ISO 14044 (2006a) and ISO 14046 (2015) define cut-off criteria as the "…specification of the amount of material or energy flow or the level of environmental significance associated with unit processes or product system to be excluded from a study." As a result, and depending on the scope and objectives of each study, variation in the parameters included in a water use inventory exists between studies. Table 2.2 compares the direct water use parameters considered in five published studies evaluating water use in the upstream portion of UOG activities.

Table 2.2 UUOGA input parameters (m^3)

Well Drilling	Cementing	Hydraulic Fracturing	References
300 - 380[5]	-	3,500 – 26,000[6]	(Jiang et al., 2013)
259	-	17,468[7]	(Laurenzi and Jersey, 2013)
640 – 1,080[8]	70 - 140	6,000 - 33,400[9]	(Clark et al., 2013)
1,370[10]	75	7,570 – 18,549[11]	(Scanlon et al., 2014)
500	-	2,900 - 20,000[12]	(Nicot and Scanlon, 2012)

[5] Range of values in a triangle distribution with a median value of 320 m^3/well.
[6] Range of values in normal distribution with a mean of 15,000 m^3 for Marcellus unconventional shale wells
[7] Mean value for Marcellus unconventional shale wells
[8] Range of estimated volumes for Barnett, Fayetteville, Haynesville, and Marcellus unconventional shale wells.
[9] Range of estimated volumes for Barnett, Fayetteville, Haynesville, and Marcellus unconventional shale wells in m^3/job. Jobs/well ranged between 1 and 3 per well. Under the maximum jobs/well scenario, the range of water required increases to between 18,000 and 100,200 $m^{3/}$well.
[10] Estimated value based on well design specifications.
[11] Hydraulic fracturing water demand for unconventional oil and gas wells within the Eagle Ford and Bakken plays.
[12] 2010 range of water demand for hydraulic fracturing in horizontal wells within the Barnett play. Median value reported of 10,600 m^3/well.

Fracfocus.ca and fracfocus.org in Canada and the US, respectively, provides operational data points for water used for hydraulic fracturing. However, operators are not required to report water use for well drilling or cementing (i.e. process steps which occur outside of hydraulic fracturing) (Nicot and Scanlon, 2012). As a result, studies must either estimate direct water demands for well drilling and cementing processes or obtain operational data directly from operators.

Laurenzi and Jersey (2013) and Jiang et al., (2013) reported analysis considering both upstream and downstream portions of a well life cycle, but did not report direct water use during cementing. Laurenzi and Jersey (2013) reported well cement requirements as an input to their assumed well design (with well dimensions assumed from Rach (2009)), however, water use results defined within Tables S8 and S9 (in Laurenzi and Jersey (2013)) for line item '*cement manufacture*' amount to 7.19 m^3 per well, which is much lower than the values cited for direct water use for cementing in other studies. It is possible that this volume reflects the indirect water requirement for cement manufacture (Clark et al., 2013; Laurenzi and Jersey, 2013; Nicot et al., 2014).

The input values for well drilling presented by Jiang et al (2013) and Laurenzi and Jersey (2013) were derived from literature values and operational data for the Marcellus Play, respectively. Clark et al. (2013) estimated drilling volumes for the Barnett, Fayetteville, Haynesville, and Marcellus unconventional shale wells based on well dimensions, estimating that the water demand for drilling is approximately six times the borehole volume (BHV), and reporting ranges of drilling water demands for each formation. Cementing estimations were also based on well design specifications and were reported as either single values or as a range of values for each play. The drilling and cementing estimates presented by Clark et al. (2013) were also assumed by Kondash and Vengosh (2015) who presented direct water use for UUOGA within the Bakken, Barnett, Eagle Ford, Fayetteville, Haynesville, Marcellus, Monterey-Tremblor, Niobrara, Permian, and Woodford plays. Like Clark et al. (2013), Scanlon et al. (2014) also approximated the water demand for well drilling as six times BHV and presented the calculated mean water drilling demands/well for horizontal wells within the Eagle Ford Play. Scanlon et al. (2014) estimated cementing water demands from reported cement volumes within their dataset. Nicot and Scanlon (2012) proposed a mean value of 500 m^3 for well drilling based on wells fractured in Texas, US in 2008, yet also highlighted the variation in reported drilling volumes due to naturally occurring variation between fields as result of operator practices, formation characteristics, and rapid changes in available technologies.

These studies all provide context into the volume of water used during UUOGA and the ways in which variation in operator technique and spatial and temporal distribution influences water use. Within these studies, evaluation of the impact of UUOGA water use is identified as an area requiring future research. Chapter CHAPTER 3 evaluates published water use metrics for their applicability to impact assessment with respect to water use during UUOGA.

CHAPTER 3. SELECTION OF SUITABLE WATER USE METRICS

In this chapter, data input categories and assessment mechanisms relevant to UUOGA are identified from the general water use patterns for UUOGA reviewed in Chapter 2. The data input categories and assessment mechanisms are then considered as the criteria used to evaluate published water use metrics to determine their application to quantifying water use and its impact on environmental indicators in regions with UUOGA. Identified water use metrics are critically reviewed to establish a water use metric selection decision tree as a tool to facilitate metric selection. Finally, the water stress index (WSI) midpoint metric is selected via the decision tree and defined for application to the case study analysis within this thesis.

A portion of this chapter has been submitted for publication as an article titled "Environmental Impacts of Water Use in Unconventional Oil and Gas Development: An Assessment on Method Evaluation and Selection" (McAuliff et al., 2017).

3.1. Method Selection Criteria

Water use metric selection is a subjective process that depends on project-specific information and the capabilities of the metric, however, general input requirements applicable to UUOGA can be surmised. Five input categories were identified based on water use patterns and processes in UUOGA. First, given that water for use in UUOGA originates from a variety of surface and ground water sources, metrics should be capable of differentiating water use by type of water source, or elementary flow. Second, water use during UUOGA can be either consumptive or non-consumptive and metrics should be capable of accounting for one of these water use types. Third, to best capture the location-specific nature of water use and availability, water use metrics must evaluate water use at a regional scale or finer resolution. Similarly, the variation in water availability over time and the requirements of UUOGA to obtain water sources in a relatively short time frame requires that metrics are capable of evaluating water use at an annual or seasonal resolution. Finally, the assessment mechanism used to evaluate water stress within midpoint metrics should evaluate either water withdrawals (withdrawal-to-availability) or water consumption (consumption to availability). Collectively, the five categories of elementary flows, water use type, spatial resolution, temporal resolution, and water stress constitute the baseline components that must be included in water use metrics evaluating UUOGA. The five input categories and their respective parameters are defined within Table 3.1.

Table 3.1 Five input categories and their respective parameters

Elementary Flows	**Water Use Type**	**Spatial Resolution**	**Temporal Resolution**	**Water stress**
Surface water (river)	Consumptive	Region	Annual	Withdrawal-to-availability
Surface water (lake)	Non-consumptive (degradative)	Watershed	Seasonal	Consumption-to-availability
Groundwater (fossil)		Map cell		
Groundwater (renewable)				

Figure 3.1 includes the input categories and parameters (identified in Table 3.1), defined within the assessment framework for water use in unconventional gas. Elementary flows refer to the movement of untransformed material or energy between the environment and the space-time system boundary. In the UUOGA scenario, water inputs commonly include a mix of flow (i.e. stream or lake with a high regeneration rate), fund (i.e. renewable groundwater with a low regeneration rate), or stock (i.e. fossil groundwater with a negligible regeneration rate)

elementary flows. Though ocean/sea water may become increasingly relevant to coastal UUOGA operators pursuing alternatives to freshwater for their process inputs, it is not commonly included within water use metrics. An exception is the Boulay et al. (2011) inventory metric, which includes sea/ocean water by characterizing it as “poor quality” surface water.

Within the study system, water resource uses can be classified as either consumptive use or non-consumptive use. Consumptive use, sometimes referred to as off-stream or evaporative use, refers to water resources extracted and not returned to the original river basin (Bayart et al., 2010; Berger and Finkbeiner, 2010; A.-M. Boulay et al., 2011; Koehler, 2008). Consumptive water use has the potential to reduce local water availability impacting quantities available for competing environmental and anthropogenic users. Non-consumptive water use, also termed as on-stream or non-evaporative use, refers to water resources extracted and returned to the original river basin after use (Hoekstra et al., 2011; Jiang et al., 2013). Without treatment, degradative use reduces the quality of water resources for competing users (Figure 3.1) (Boulay et al., 2011; Ridoutt and Pfister, 2013). This definition of degradative use is consistent with Boulay et al. (2011), however, differs from Hoekstra et al. (2011) wherein degradative use is considered as consumptive use.

The volume of fresh water available for human use is determined by its availability, demand, quality, and the presence of compensation mechanisms such as reservoirs (Bayart et al., 2010). Across the globe, there is high spatial and temporal variability in freshwater resources in terms of both availability and demand (Bayart et al., 2010; WWAP, 2016).

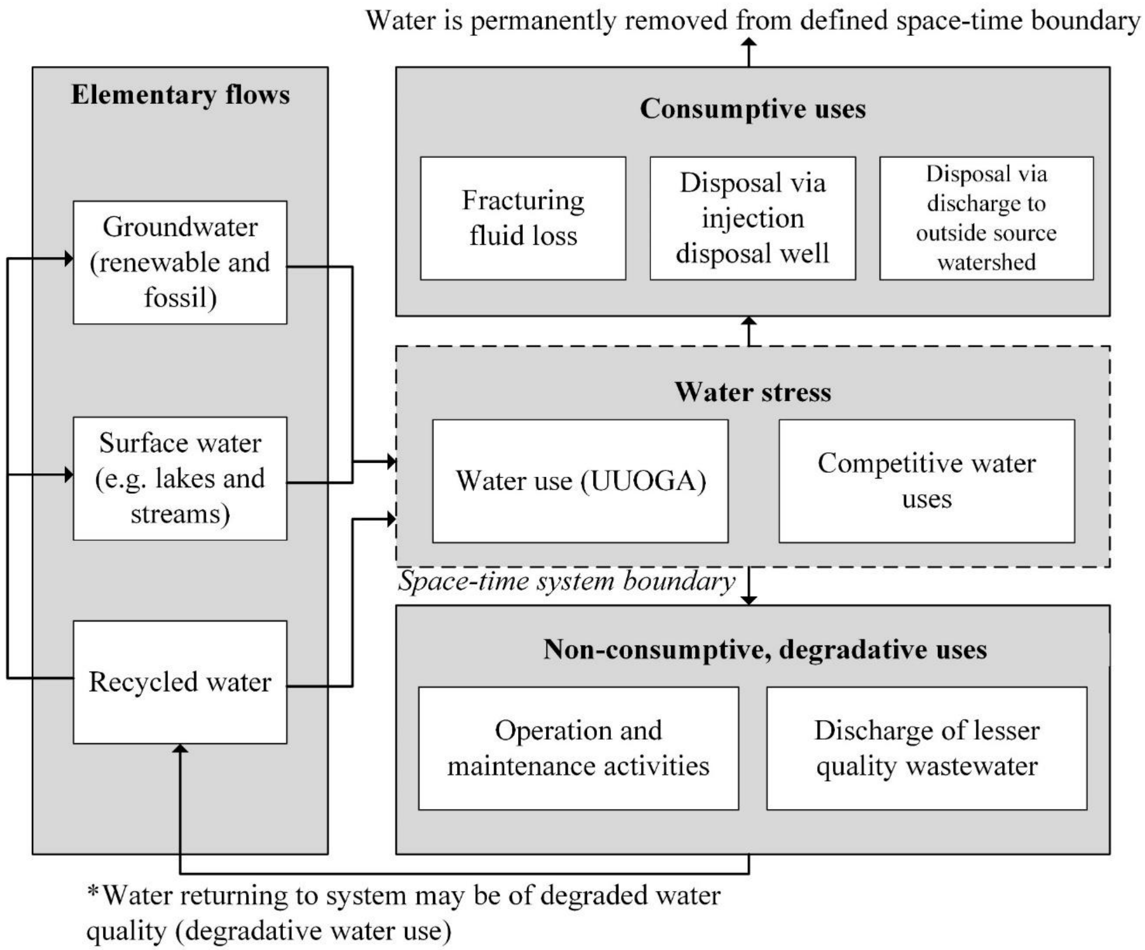

Figure 3.1 Water use in UUOGA impact assessment framework

Furthermore, the environmental impact of water use is generally lesser in areas with greater water resource availability compared to the impact of the same volume of water use in areas with lesser water availability (Pfister et al., 2009; Scown et al., 2011). North American unconventional gas plays occur in areas with varying degrees of water scarcity, necessitating a regional assessment of water use.

Spatial and temporal resolution are both data and methodology dependent and must be defined during the scope definition portion of the analysis. UUOGA processes occur within a relatively short timeframe (i.e. weeks to months) making temporal variation in water availability especially relevant to the UUOGA context. Spatial and temporal resolutions within water use metrics are often limited by data availability, which in turn define the space-time system boundary (Figure 3.1).

The water stress component (Figure 3.1) represents the local environmental context within which water use impact will be considered. Water stress is typically represented in terms of water scarcity, determined by water use and water availability values. Water scarcity can be considered either through a withdrawal-to-availability (WTA) ratio or a consumption-to-availability (CTA) ratio. WTA and CTA differ in the sense that volumes of water considered in a WTA can include both consumptive and non-consumptive volumes, while CTA considers only consumptive use (Boulay et al., 2015).

Recent reviews of water use metrics have included over 25 metrics developed for water use inventory, midpoint, or endpoint assessments (Boulay et al., 2015; Kounina et al., 2012). For use in the UUOGA scenario, common water use metrics were evaluated for their inclusion of the five input categories defined within Table 3.1.

Table 3.2 Comparison of metric input categories and parameters

		Inventory Metrics			**Midpoint Metrics**					**Endpoint Metrics**	
Input Category	**Parameters**	(Boulay et al., 2011)	(Milà i Canals et al., 2009)	(Hoekstra et al., 2011)	(Milà i Canals et al., 2009)	(Pfister et al., 2009)	(Ridoutt and Pfister, 2010)	(Hoekstra et al., 2011)	(Swiss Ecological Scarcity, 2013)	(Pfister et al., 2009)	(Hanafiah et al., 2011)
Elementary flows	Renewable ground water	X	X	X	X	X	X	X	X	X	X
	Fossil ground water	X	X	X	X	X	X	X	X	-	-
	Surface water (river)	X	X	X	X	X	X	X	X	X	X
	Surface water (lake)	X	X	X	X	X	X	X	X	X	X
Water use type	Consumptive	X	X	X	X	X	X	X	X	X	X
	Degradative	X	X	X	-	-	X	X	-	-	-
Spatial resolution	Country	DD	DD	DD	X	X	X	X	X	-	-
	Region	DD	DD	DD	-		-	-	-	-	-
	River basin	DD	DD	DD	X	X	X	X	X	X	X
	Map cell	DD	DD	DD	-	X	X	-	-	X	-
Temporal resolution	Annual	DD	DD	DD	-	X	X	-	-	X	X
	Seasonal	DD	DD	DD	-	-	X	-	-	-	-
Water stress	WTA	N/A	N/A	N/A	-	X	X	-	X	N/A	N/A
	CTA	N/A	N/A	N/A	-	-	X	X	-	N/A	N/A
	Other	N/A	N/A	N/A	X	-	-	-	-	N/A	N/A

DD: Parameter is data dependent.
NA: Not applicable. Water stress characterization is applicable to midpoint metrics only

This has resulted in three inventory, five midpoint, and two endpoint water use metrics which included provisions for addressing each of the five input criteria. The inclusion of input categories and parameters by each of the eight metrics is described in the following Sections 0, 3.3, and 3.4, as well as within Table 3.2.

3.2. Inventory Metrics

The three inventory metrics propose either input classification or characterization during inventory assessment or subsequent inclusion into either midpoint or endpoint metrics. Boulay et al. (2011) proposed a functionality-based regionalized inventory metric that considers both elementary flows and quality of ground and surface water. Functionality can decrease for users both through consumption, where water becomes unavailable for use in a location and through degradation, where water quality decreases beyond what is acceptable for a given purpose. User groups of domestic, agricultural, and industrial are differentiated per their respective water quality requirements. The Boulay et al. (2011) metric proposed the use of water quality parameters previously established by government agencies around the world for further characterization of water input quality. This resulted in eight water quality categories with functional distinction between users sensitive (i.e. low, medium, or high sensitivity) to microbial or toxicant contamination. The Boulay et al. (2011) inventory method allows for the differentiation of elementary flows in such a way that reuse/recycled wastewater or saline water used for UUOGA would be considered independently from fresh surface and ground water sources.

The Milà i Canals et al. (2009) inventory metric was developed to complement the proposed impact pathway that considers direct water use that results in changes to freshwater availability for ecosystems, which in turn affects ecosystem quality. In this metric, elementary flows are classified as blue (i.e. fresh surface and ground water), green (i.e. water bound as soil water or in biotic matter), and gray water. In the context of water use metrics, the gray water refers to the "volume of freshwater needed to assimilate emissions to freshwater" (Hoekstra and Mekonnen, 2012; Kounina et al., 2012; Milà I Canals et al., 2009; Ridoutt and Pfister, 2010). Milà i Canals et al. (2009) adopted the *virtual water content* inventory assessment methodology proposed by Chapagain and Orr (2008). In this methodology, virtual water refers to the volume required to produce a product or embodied within a product. Non-consumptive (i.e. evaporative in text) and consumptive (i.e. non-evaporative in text) water use of elementary flows are considered, though unlike the inventory method proposed by Boulay et al. (2011), the method does not account for degradative use of elementary flows. The majority of water use in UUOGA would fall within the blue water category as a volumetric value. However, this inventory method does not provide a mechanism to consider recycled/reused wastewater differently than fresh surface water.

Finally, Hokestra's water footprint inventory metric proposed by Hoekstra et al. (2011) has been readily applied by industries and organizations. Hoekstra's water footprint categorizes blue, green, and gray water consumption (Hoekstra et al., 2011). Gray water is also considered to be a midpoint indicator of degradative freshwater use. As with the Milà i Canals et al. (2009) inventory method, the Hoekstra et al. (2011) inventory method of UUOGA would provide a volumetric value within the blue water category and does not differentiate between freshwater and recycled/reuse water.

3.3. Midpoint Metrics

Four midpoint metrics assess water use against various environmental indicators. First, Milà i Canals et al. (2009) proposed the water resources per capita (WRPC), water use per resource (WUPR), and the water stress indicator (WSIn) as indicators for assessing freshwater ecosystem impact as midpoint metrics. WSIn is identified as the preferred and most comprehensive approach, which incorporates anthropogenic water use (WU), water resources (WR), and environmental water requirements (EWR) in its calculation. However, EWR estimations must be supported by sufficient spatial data to be extended to the assessment scope defined by an analysis. Evaluating environmental flow needs is an emerging field and large uncertainties in the assessment of EWR and ecological response to WU must be considered in the application of this metric (Ministry of Environment, 2016). The WRPC indicator considers renewable WR and population. Results can be categorized into three groups consisting of water stress, water scarcity, or absolute water scarcity. Results of less than 1,677 m^3 per capita per year fall within the water stress category, less than 1,000 m^3 per capita per year fall within the water scarcity category (also referred to as chronic water scarcity) , and less than 500 m^3 per capita per year fall within the absolute water scarcity category (Milà I Canals et al., 2009; WWAP, 2016). The WUPR indicator is the ratio of WU to WR, with WR representing the volume of water withdrawn from natural resources.

For characterizing UUOGA water use, the Milà i Canals et al. (2009) framework provides three indicators, each employing a slightly different lens with which to evaluate water use. UUOGA water use for operations occurring in locations with significant domestic water demand may find the WRPC indicator informative to evaluate anthropogenic water use, whereas UUOGA operations occurring in more remote locations may be better represented by the WSIn or WUPR indicators for evaluating anthropogenic water use. While each of the midpoint metrics within the Milà i Canals et al. (2009) framework provides context to quantify UUOGA water use as a portion of anthropogenic water use and its potential to contribute to water scarcity, the completion of all three metrics would provide water managers with the most comprehensive set of information with which to evaluate water use in UUOGA.

The Pfister et al. (2009) midpoint metric, also known as the water stress index (WSI), evaluates the midpoint indicator of water deprivation. WSI is calculated using a modified WTA, calculated at the river basin level, which considers water withdrawals by industry, agriculture, and domestic user groups. The WSI is a logistic function, reporting water stress values between 0.01 and 1. A WSI of 0.5 corresponds to a WTA of 0.4 which indicates the transition between moderate and severe water stress. Pfister et al. (2009) propose the WSI for use in screening assessment to identify geographic locations of water stress or as a characterization factor to support existing LCIA methodologies, specifically within the damage assessment framework of Ecoindicator-99. The Pfister et al. (2009) WSI is similar to the Milà i Canals et al. (2009) WUPR indicator in that it evaluates water use with respect to water resource availability, however, the Pfister et al. (2009) WSI also includes a variation factor in its calculation, providing additional context to UUOGA water use for operations in areas with variability in water resources. The variation factor considers inputs of monthly and annual precipitation variation and whether or not surface water is regulated through retention technologies such as dams to better illustrate the ways in which variation in water availability and technology influence the impacts of water use.

Ridoutt and Pfister (2010) expand on the Pfister et al. (2009) WSI midpoint metric with the inclusion of the gray water indicator for the assessment of degradative water use. In addition to a gray water volumetric value, the Ridoutt and Pfister (2010) metric also incorporated a volumetric value for the impact of land use on blue water resources. The resulting metric considers blue water (surface or groundwater) consumption, gray water requirement, impact of land use to blue water, and the WSI from Pfister et al. (2009) resulting in a volumetric "*stress-weighted water footprint*". When applied to UUOGA, this metric would reflect the differences in water use between UUOGA operations occurring in different locations with varying water availability.

Hoekstra's water footprint midpoint metric was developed to consider both consumptive water use of blue and green water, and degradative use through gray water. Hoekstra's water footprint reports volumetric results. In the context of UUOGA, an industrial process, green water has less application than it would in an agricultural process. Additionally, as UUOGA practice favors injection well disposal of waste fluids resulting from UUOGA, under an ideal scenario, the components found in wastewater would not be released into the environment, limiting the application of the gray water volume. Despite these limitations, Hoekstra's water footprint is a standalone method that has gained popularity in both industry and academia (Hoekstra et al., 2011).

The final midpoint metric evaluated, the Swiss Ecological Scarcity metric (2013), uses the distance-to-target principle, which is the counterpart to the functionality principle (utilized in the Boulay et al. (2011) inventory assessment methodology discussed above). With the distance-to-target approach, water quality is assessed by determining the effort (dilution or treatment) necessary to process a water source to a predetermined final quality (Klinglmair et al., 2014). In the Ecological Scarcity metric, the distance-to-target principle is applied through the deduction of critical flows, based on existing legislative guidelines or standards, and current flows which represent the current state. Critical and current flow values become components of the proposed equation for calculating the eco-factor midpoint indicator. The eco-factor equation also incorporates normalization and weighting terms, which provide local and temporal specificity. The results of the eco-factor equation are output in eco-points which quantify the impact of emissions and/or resource extraction (Frischknecht et al., 2006; Frischknecht and Sybille, 2013). The Swiss Ecological Scarcity methods' (2013) distance-to-target method, through consideration of local water quality regulations, provides an alternative method for evaluating UUOGA water use at a local level. Instead of a characterization factor determined through the WSI as in Ridoutt and Pfister (2010), the Swiss Ecological Scarcity (2013) method utilizes characterization factors derived from local water quality and quantity regulations.

3.4. Endpoint Metrics

Pfister et al. (2009) also proposed an endpoint indicator metric for evaluating the damage area of protection, ecosystem quality, through the use of the Ecoindicator-99-method that assesses the "effect of freshwater consumption on terrestrial ecosystem quality." This metric considers the vascular plant growth constraint indices developed by previous research, to identify net primary production (NPP) limited by water availability (Pfister et al., 2009). The proposed calculation considers annual precipitation at the cell level and aggregates them for application to a river basin, which facilitates regional assessment. Endpoint metrics are data intensive, however, with sufficient data the Pfister et al. (2009) endpoint metric provides a biologically based indicator for evaluating the environmental impact of water use in UUOGA.

Finally, Hanafiah et al. (2011) proposed an endpoint metric for evaluating the impact of anthropogenic consumptive freshwater use on the potentially disappeared fraction of fish species. In the water consumption portion of this metric, a characterization factor is calculated using a species-discharge relationship to determine freshwater species richness. Another characterization factor is calculated for water consumption at the river basin level, to assess the impact of anthropogenic water use on freshwater fish species richness in terms of PDFs. Normalization factors were also calculated for direct water consumption in terms of PDF, which considers river basin specific water consumption, water consumption characterization factors, and river basin population. Water consumption values consider water use from the four user groups consisting of households, irrigation, industry, and livestock (Hanafiah et al., 2011). The Hanafiah et al. (2011) metric excluded river basins over 42 degrees latitude because these river basins had not reached their maximum species richness potential due to the relatively recent glaciation. This is determined by Hanafiah et al. (Hanafiah et al., 2011) through the observation of weak species-discharge relationship in high latitude basins. As a result, this metric may not be applicable to UUOGA occurring in Canadian river basins above 42 degrees latitude, thus limiting the application of this metric to international comparisons of unconventional operations in North America (Hanafiah et al., 2011).

3.5. Discussion

While all of the selected metrics in this chapter and within Table 3.2 consider multiple types of elementary flows, not all considered the impact of degradative outputs on the environment (Boulay et al., 2011; Ridoutt and Pfister, 2010). All metrics are also capable of reporting results at the river basin level, indicating the importance of spatial specificity within the field of water use metrics. Though largely dependent on input data, the majority of metrics returned results with an annual temporal resolution, which may not provide sufficient transparency into monthly or seasonal variations in water use or water availability. This is of concern especially for areas experiencing seasonal variation in water availability (e.g. dry vs. monsoon seasons).

Often, water use studies are constrained by data availability. In the UUOGA context, water use metric practitioners must ultimately select the metric that incorporates input categories and data resolution most relevant to study and location of operations. UUOGA operations occurring in environments with high seasonal variation in water availability may find that reporting the environmental impact of annual water use does not sufficiently capture the impact of water use during dry periods. Similarly, evaluating water use impact at a country resolution in countries which have highly variable water availability, as is the case across North America, may not be sufficient. In these examples, finer spatial and temporal data would be necessary to ensure environmental relevance. While regional specificity is essential to understanding the environmental impacts of water use, assessment of highly localized impacts, such as the impact on river flow rate within a given river basin due to local water use, is outside the scope of water use metrics and better suited for an alternative assessment framework (e.g. environmental impact assessment) (Milà I Canals et al., 2009).

Impact assessment can be accomplished either with midpoint or endpoint metrics based on volumetric, CTA, WTA, and biodiversity/flow rate assessment mechanisms. Within each assessment mechanism, published metrics evaluate one or more impact indicators. The differences in impact indicators are perhaps best represented by the result units generated from each of the metrics. The Hoekstra et al. (2009) metric provides volumetric results for blue, green, and gray water. The Milà I Canals et al. (2009) metrics of WRPC and WUWR midpoint indicators are per capita and unit-less scores of impact of water use via the CTA mechanism. The Swiss Ecological Scarcity (2013) metric provides scored results in units of Eco-points. Milà I Canals et al. (2009) and Pfister et al. (2009) midpoint metrics report volumetric indices based on CTA and WTA mechanisms, respectively. The Pfister et al. (2009) and Hanafiah et al. (2011) endpoint metrics evaluate damage to ecosystem quality through impact indicators of change to green water availability and change in water discharge. The Pfister et al. (2009) metric results in units of NPP and is most applicable to studies quantifying the impact of water use on vascular plant biodiversity. Alternatively, Hanafiah et al. (2011) evaluates damage to ecosystem quality by determining PDF, which provides a wildlife based representation of water use impact.

In Figure 3.2, a decision tree for metric selection of water use metrics applicable to UUOGA is displayed, which serves as a tool for metric selection to support analysis within this thesis and for future study. In this figure, the water use metrics evaluated in Table 3.2 are presented along decision pathways differentiated by study scope, assessment mechanisms, and reporting units.

For the purpose of this research, which aims to quantify water use and its impact, an index based midpoint metric was identified as the preferred metric, given the research objectives, data availability, and the target audience being readers of this thesis. Of the three available index based midpoint metrics, the Pfister et al. (2009) midpoint metric (WSI Framework) was selected for implementation in this research due to the consideration of seasonal and annual precipitation variation impact in its WSI.

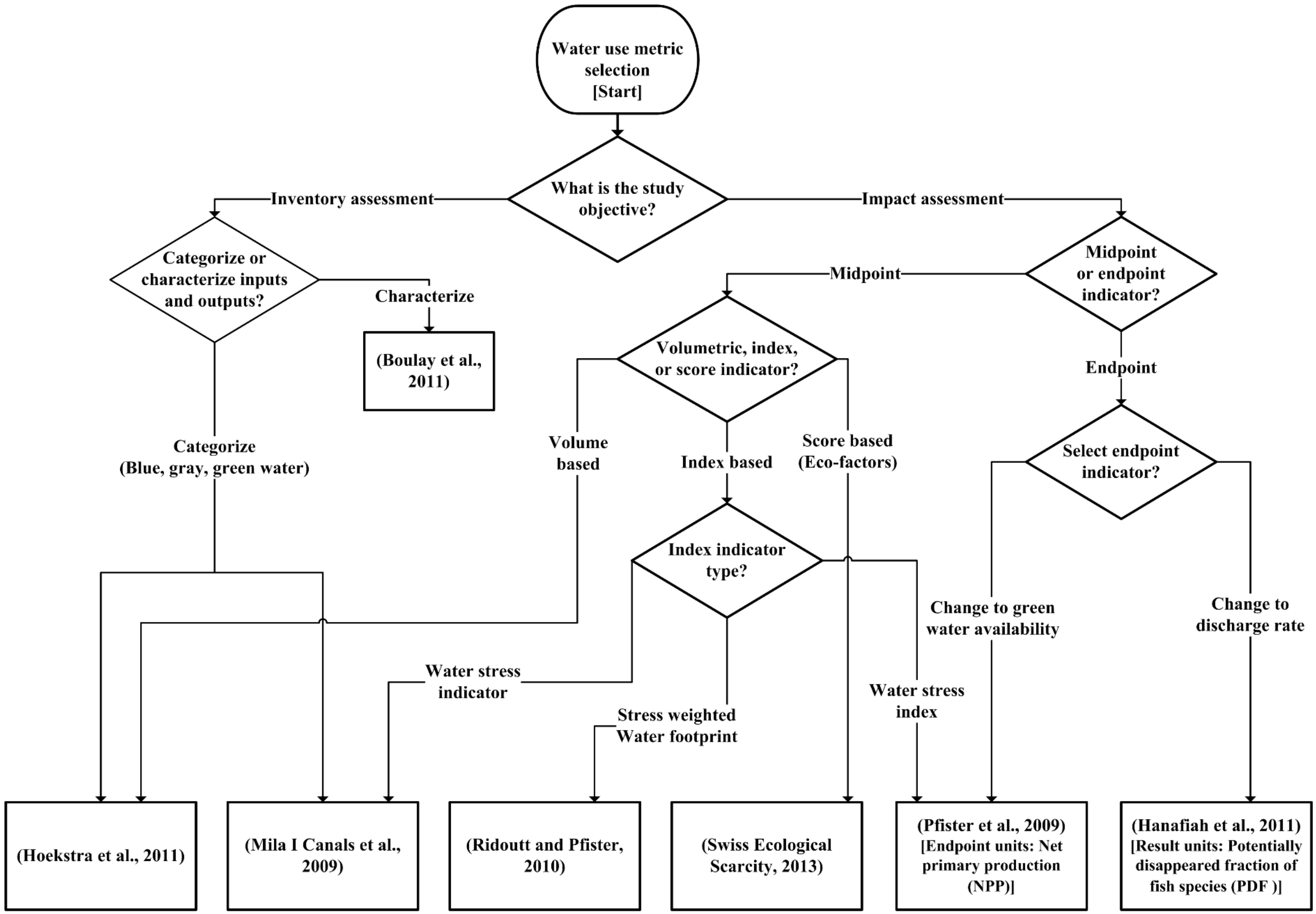

Figure 3.2 Decision tree for metric selection

Seasonal and annual precipitation variation and the impact of variation on water availability is a relevant concern to water users within northeast BC (NEBC) (i.e. the region identified for analysis in this research). The input parameters and equations which comprise the WSI Framework are defined in the following section.

3.6. WSI Framework

Per Pfister et al., (2009), water stress is frequently characterized by a WTA ratio. Equation 3.1 defines the WTA equation in which the WU factor considers water withdrawals from multiple user groups *'j'*, within a spatial unit *'i'* and the WA factor considers blue water, or fresh surface and ground water, availability within a spatial unit *'i'*.

$$\mathrm{WTA_i} = \frac{\sum_j \mathrm{WU_{ij}}}{\mathrm{WA_i}} \qquad \text{Equation 3.1}$$

The WSI Framework expands upon WTA (Equation 3.1) with a modified WTA (WTA*, Equation 3.2) which incorporates a variation factor (VF) to better capture stress associated with variation in monthly (s*month) and annual (s*year) precipitation. The impact of variation can be mitigated to some extent by the use of water storage technologies. To factor both precipitation variation and storage capabilities into the WTA* equation, slightly different WTA* equations are available for river basins with strongly regulated flows (SRF) and without SRF. Equation 3.2 is defined within Pfister et al. (2009) for SRF river basins. The equation for non-SRF river basins considers the unaltered VF factor (i.e. not the square root of VF).

$$\mathrm{WTA^*} = \sqrt{\mathrm{VF}} \times \mathrm{WTA} \qquad \text{Equation 3.2}$$

The VF value is a measurement of, considering an aggregation of the s*month and s*year factors. The s*month and s*year factors which serve as inputs to the calculation of $VF_{i,}$ (Equation 3.3) represent the geometric standard deviations of monthly and annual precipitation, respectively.

$$VF_i = e^{\sqrt{\ln(S^*month)^2 + \ln(S^*year)^2}} \qquad \text{Equation 3.3}$$

VF_i values for individual spatial units can be aggregated to a larger spatial unit (VFws) (i.e. watershed) via Equation 3.4, which weighs VF_i values by mean annual precipitation (P_i) within the spatial unit *'i'*.

$$VF_{ws} = \frac{1}{\sum P_i} \sum_{i=1}^{n} VF_i \cdot P_i \qquad \text{Equation 3.4}$$

WSI (Equation 3.5) considers the WTA* factor and outputs values ranging between 0.01 and 1.

$$WSI = \frac{1}{1 + e^{-6.4 \times WTA^*} \left(\frac{1}{0.01} - 1\right)} \qquad \text{Equation 3.5}$$

The WSI framework is presented in Figure 3.3 as a hierarchy of data inputs (WU, WA, s*month, s*year, P_i), intermediate calculations (WTA, VF_i, VFws, and WTA*), and the final result of WSI.

Thresholds for water stress determined through WTA are based on expert opinion and have been defined as moderate water stress occurring at a WTA of 20% and severe water stress occurring at a WTA of 40-60%. WSI results correspond to the above WTA threshold values with water stress occurring at a WSI of 0.09 and severe water stress occurring at WSI of 0.5 (Pfister et al., 2009).

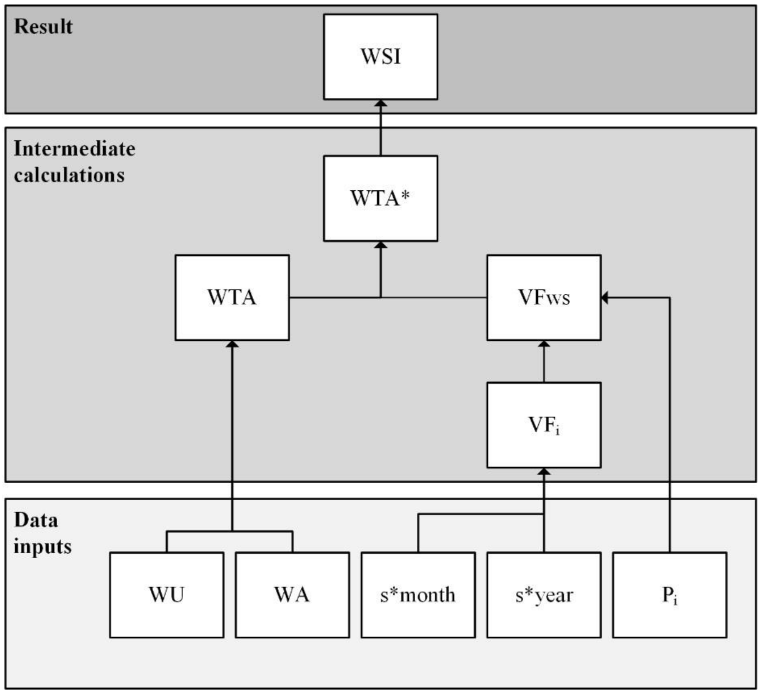

Figure 3.3 WSI framework

The following chapter applies the WSI framework in a regional case study of water use in the Montney Play Trend region of BC, Canada. This region has recently experienced rapid development of UOG and also supports water demands of other sectors such as forestry, agriculture, and mining. Data collection, analysis, and results from the WSI framework analysis are included within Chapter 4.

CHAPTER 4. IMPLEMENTATION OF WSI FRAMEWORK

The WSI Framework analysis of the Montney Play Trend region involves two components, the first is a water use inventory and the second is a water use impact analysis (Figure 4.1). The water use inventory will quantify the volume of water withdrawn for use during UUOGA, the regional water use (withdrawal), and regional water resource availability. The results from the UUOGA water use inventory represent a portion of the larger regional water use inventory considered as an input for the midpoint analysis component of this research.

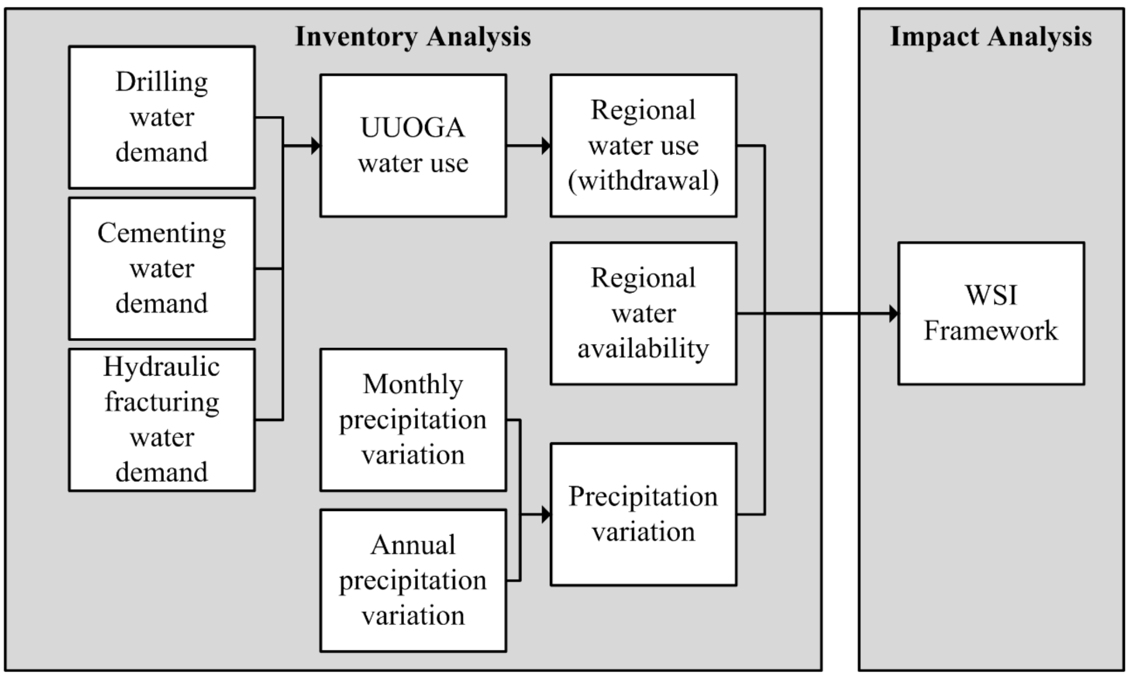

Figure 4.1 WSI analysis framework

The data inputs and their respective sources for the Montney study region are presented in Table 4.1. Precipitation data was collected from 1961-1990 to establish a *climate normal base scenario*. This is consistent with the inputs and assumptions defined within Pfister et al. (2009).

Table 4.1 Inventory analysis data sources

Input Parameters	**Source**
Hydraulic fracturing water demand	(BC Oil and Gas Commission, 2015b)
Cementing water demand	(BC Oil and Gas Commission, 2015c)
Drilling water demand	
Regional water use (withdrawal)	(Gassert et al., 2014)
Regional water availability	
Precipitation variation	(Harris et al., 2014)

4.1. Data Collection and UUOGA Water Use Inventory

Data collection for the UUOGA water inventory was made possible by the BCOGC who granted access to Fracfocus and its *Integrated Resource Information System (IRIS)* databases, which contain information on oil and gas operations within BC (BC Oil and Gas Commission, 2015c). The province of BC required mandatory public disclosure (published on www.fracfocus.ca) of hydraulic fracturing fluid volume and chemical composition beginning on January 1, 2012. Consequently, the dataset contains direct water input volumes for hydraulic fracturing for wells fractured between January 1, 2012, and September 2015 (when the database was accessed) (BCOGC, 2015b). The BCOGC's *IRIS* database and *IHS AccuMap®* database also included additional operational data such as casing size and well depth (IHS Accumap, 2016).

Data was collected specifically for horizontal wells (i.e. vertical and deviated/directional wells were excluded and represented 1.0% of the sample well population) located within the Montney field (as identified by the associated regulatory codes of 9022 and 9021 for NM Field and RH Fields, respectively). Wells with hydraulic fracturing water volumes less than 378 m^3 were also eliminated from the dataset. This is considered to be a conservative lower bound estimate consistent with those found in similar studies (Scanlon et al., 2014). The resulting dataset includes 1,242 unique wells. Data entries were not complete for each well parameter across databases and as such, the different analyses completed within this thesis utilized different subsets of the total study well population based on data availability.

Reported hydraulic fracturing water demands were available for 1,242 wells within the study dataset (582 wells within the NM and 660 wells within the RH). Overall wells within the Montney play trend required an average of 10,881 m^3 of water for hydraulic fracturing. In the NH, the average water use for hydraulic fracturing was 12,840 m^3/well and in the RH average water use for hydraulic fracturing was 9,153 m^3/well. Additional qualitative analysis regarding well design and operational characteristics as they pertain to reported water use during hydraulic fracturing is presented in detail in Appendix A.

Scanlon et al. (2014) reported that when considered together, water demand for drilling and cementing processes represent about 8% of the water demand for hydraulic fracturing. While these processes generally contribute a relatively small volume of water to the overall water demand during upstream operations, well drilling and cementing volumes were estimated to gain a better understanding of the water demands for these processes within the Montney Play. Well drilling and cementing volumes were calculated from well dimensions and cement quantities reported within the BCOGC's *IRIS database*.

To estimate well drilling water demands, a ratio of 6:1 drilling water to BHV employed within Clark et al. (Clark et al., 2013) and Scanlon et al. (2014) is assumed. Casing size and casing shoe depth dimensions were available for 1,217 wells within the study dataset and were used to estimate BHV for Montney unconventional wells. Within this subset of 1,217 wells, 575 wells

(52%) are within the NM field and 642 wells (48%) are within the RH field. The largest casing shoe depth value per well from the IRIS dataset was compared to the total depth values from *IHS AccuMap®* dataset and on average a 0.2% difference between the values was identified across both datasets. As a result, it was assumed that the largest depth reported for casing shoe depth for a well was equivalent to the total depth of a well. Twenty wells with greater than a 5% difference between largest casing shoe depth and total depth have been eliminated from the dataset because casing dimensions could not be used to determine the total BHV of the well. Average drilling water demand in the Montney is found to be 1,204 m^3/well (averages of 1,209 m^3 for NM wells and 1,199 m^3 for RH wells).

Scanlon et al. (2014) estimated water demands for cementing assuming a water to cement ratio of 5.5 gallons of water/sack cement (0.49 by weight) from reported dry cement values. Within the Montney study dataset, cement values (in tonnes) are reported for 1,237 wells (580 wells within the NM and 657 within the RH), and assuming the same 0.49 water to cement ratio, the average water demand in the Montney for cementing is found to be 51 m^3/well (58.6 m^3/well in the NM field and 43.5 m^3/well in the RH field).

Table 4.2 presents observed average and summed total average values for water use during well drilling, hydraulic fracturing, and cementing activities for wells within the Montney Play Trend, NM field, and RH Fields. Additional qualitative analysis regarding well design and operational characteristics of NM and RH wells is presented within Appendix A while the regulatory framework governing Montney unconventional wells is described in Appendix B.

Table 4.2 Observed averages and total of UUOGA water inventory parameters (m^3)

Parameter	Montney Play Trend	NM Field	RH Field	Source
Hydraulic fracturing	10,881	12,840	9,153	(BC Oil and Gas Commission, 2015b)
Well drilling	1,204	1,209	1,199	(BC Oil and Gas Commission, 2015c)
Cementing	51	59	43	
Total	12,136	14,108	10,395	

4.2. Regional Water Use and Water Availability Inventory

Typically, water is collected from sources in close proximity to a well site (Jiang et al., 2013). For this analysis, it was assumed that water used for upstream unconventional operations is sourced within the defined study area (Figure 4.2). Montney unconventional well locations were obtained from the BCOGC's FracFocus.ca database. Unconventional well locations were mapped against river basin delimitations provided within the World Resources Institute's *Aqueduct Global Maps 2.1* dataset (*Aqueduct 2.1*; accessed at www.wri.org/resources/data-sets/aqueduct-global-maps-21-data) using *QGIS* (version 2.16.2), an open source geographic information system (GIS) software. The unconventional wells fell within four hydrological catchments (referred hereafter as river basins) identified within *Aqueduct 2.1* by the five digit numeric identifiers of 11863, 12175, 12187, and 12208 (Figure 4.2). The *Aqueduct 2.1* data set defines river basins based on the boundaries defined within the *Global Drainage Basin Database* (Gassert et al., 2014).

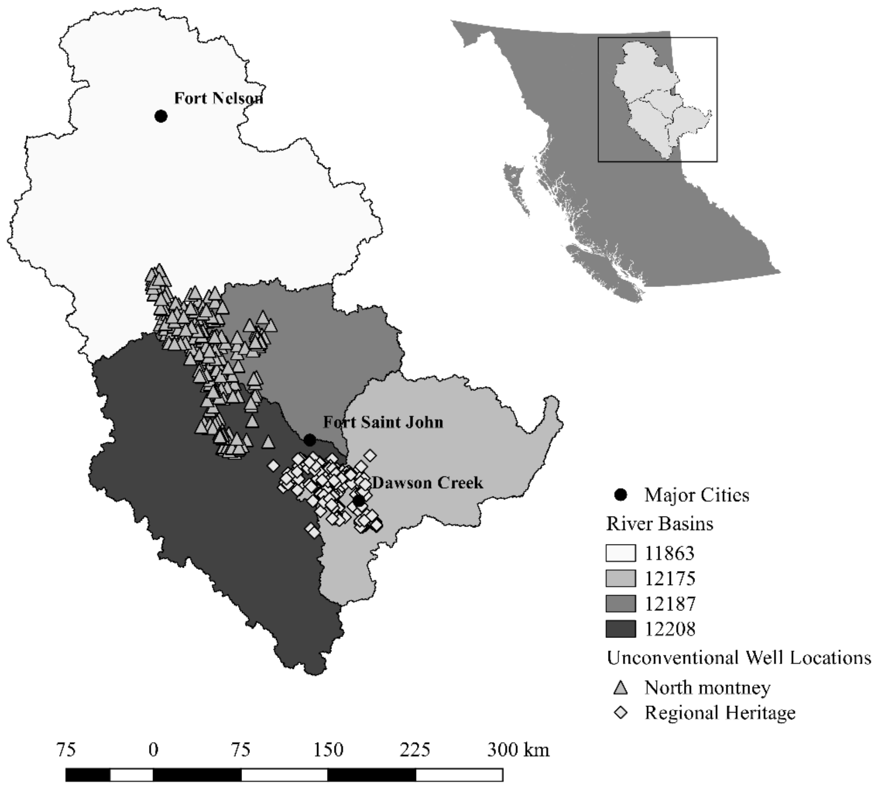

Figure 4.2 Study river basins, Montney unconventional well locations, and major cities[13]

The BCOGC, in its role issuing water licenses and short term Section 8 approvals of water use for oil and gas purposes, considers water management basins which fall within the bounds of the *Aqueduct 2.1* river basins. The boundaries of the BCOGC water management basins are shown in Figure 4.3, within the boundaries of the *Aqueduct 2.1* river basins (BCOGC, 2016b; Gassert et al., 2014). The vertical boundary east of Dawson Creek within Figure 4.3 represents the

[13] Shape/data files for Figure 4.2 collected from BCOGC (2015b, 2016a) and Gassert et al., (2014). City location coordinates collected from Google Earth (2016)

provincial border between BC and AB. Table 4.3 specifies BCOGC water management basins falling within each of the *Aqueduct 2.1* river basin boundaries considered in this analysis.

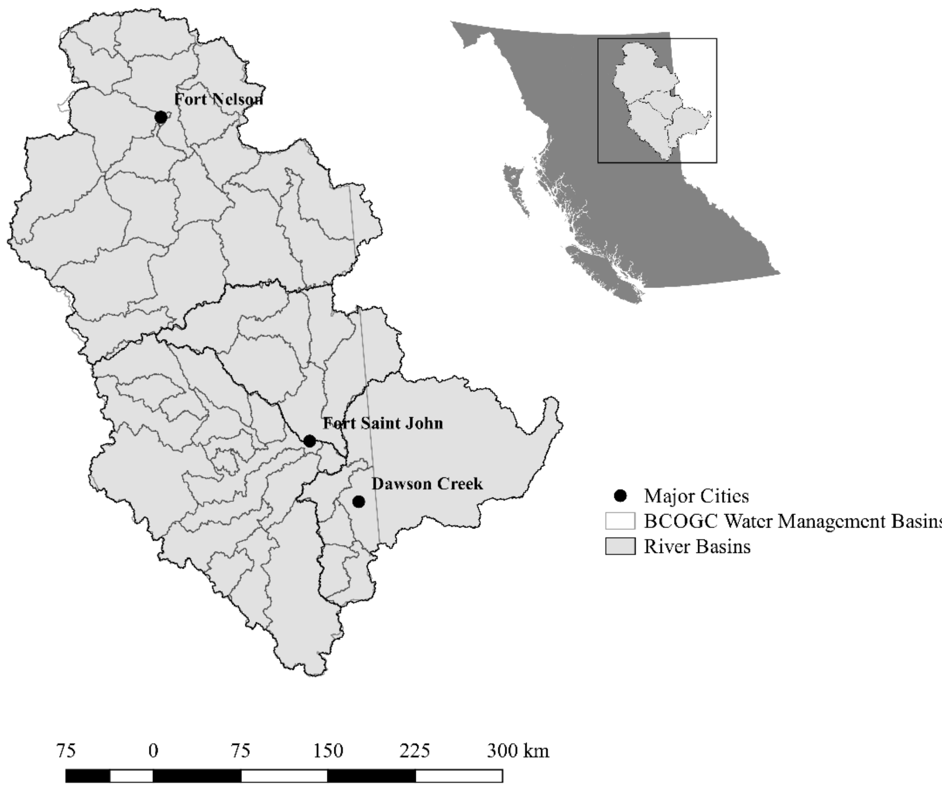

Figure 4.3 WRI River basins and BCOGC water management basins[14]

[14] Shape/data files for Figure 4.3 were collected from BCOGC (2016e), BCOGC (2016a), and Gassert et al., (2014). City location coordinates collected from Google Earth (2016).

Table 4.3 BCOGC water management basins falling within study river basins[15]

WRI River Basins			
11863	**12208**	**12175**	**12187**
Capot Blanc Creek	Upper Halfway River	Lower Peace River	Upper Beatton River
Kiwigana River	Chowade River	Lower Kiskatinaw River	Middle Beatton River
Lower Fort Nelson River	Cameron River	Pouce Coupe River	Milligan Creek
Middle Fort Nelson River	Lower Halfway River	Middle Kiskatinaw River	Blueberry River
Sahtaneh River	Peace Arm	West Kiskatinaw River	Lower Beatton River
Lower Muskwa River	Farrell Creek	East Kiskatinaw River	Doig River
Snake River	Cache Creek		
Middle Muskwa River	Lynx Creek		
Lower Prophet River	Upper Peace River		
Upper Fort Nelson River	Moberly River		
Upper Muskwa River	Upper Pine River		
Middle Prophet River	Lower Pine River		
Klua Creek	Lower Peace River		
Fontas River	Burnt River		
Upper Prophet River	Sukunka River		
Middle Sikanni Chief River	Murray River		
Lower Sikanni Chief River			
Kahntah River			
Upper Sikanni Chief River			

[15] BCOGC water management basin names collected from BCOGC (2016b) and WRI river basins were collected from Gassert et al., (2014). BCOGC water management basins falling within WRI River basin boundaries were determined using Q GIS.

The water management basins depicted within Figure 4.3 fall within the Mackenzie River Basin, the largest river basin in the province of BC, which drains runoff from portions of Saskatchewan, AB, BC, the Northwest Territories, and Yukon Territory to the Mackenzie River and ultimately to the Arctic Ocean. In NEBC, the Peace River, Liard River, and Hay River drain surface water from the region to the Mackenzie River. It is estimated that the mean annual flow in the NEBC region of the Mackenzie River is 120.6 billion m^3/year (The Province of British Columbia, 2015).

In the broader context of the province of BC, multiple sectors utilize water resources in addition to oil and gas. Surface water allocations (i.e. consumptive and non-consumptive water uses) by sector in NEBC are displayed in Figure 4.4. The oil and gas sector is responsible for 0.02% (20.0 M m^3/year) of total annual water allocations by volume.

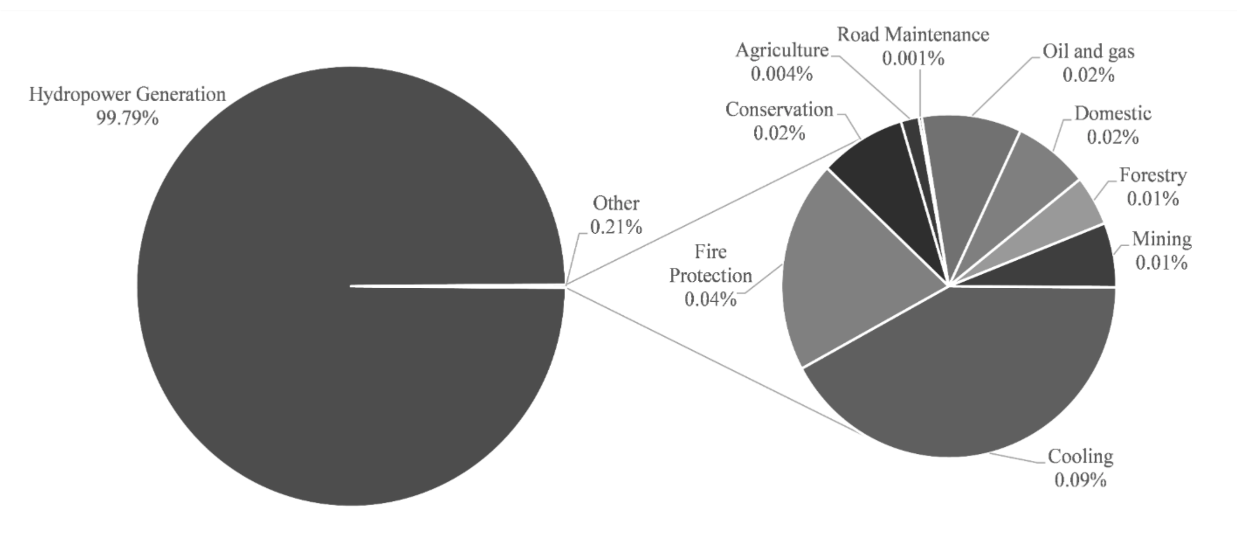

Figure 4.4 Surface water allocations by sector in British Columbia[16]

[16] Figure adapted from (The Province of British Columbia, 2015)

Water allocations represent an allowable maximum volume of water for a given user and, in general, the volume of water used is less than the volume allocated. (The Province of British Columbia, 2015). Water allocations and can be further differentiated into consumptive and non-consumptive water uses. The majority of surface water allocations are for non-consumptive use (i.e. hydropower generation is a non-consumptive use), and the regions consumptive use allocation is estimated to be 0.05% of mean annual surface water runoff (The Province of British Columbia, 2015).

Figure 4.5 illustrates the portion of consumptive water uses by sector in NEBC. While the oil and gas sector accounts for 0.02% of surface water allocations in NEBC, it accounts for 32% of consumptive surface water allocations within the same region.

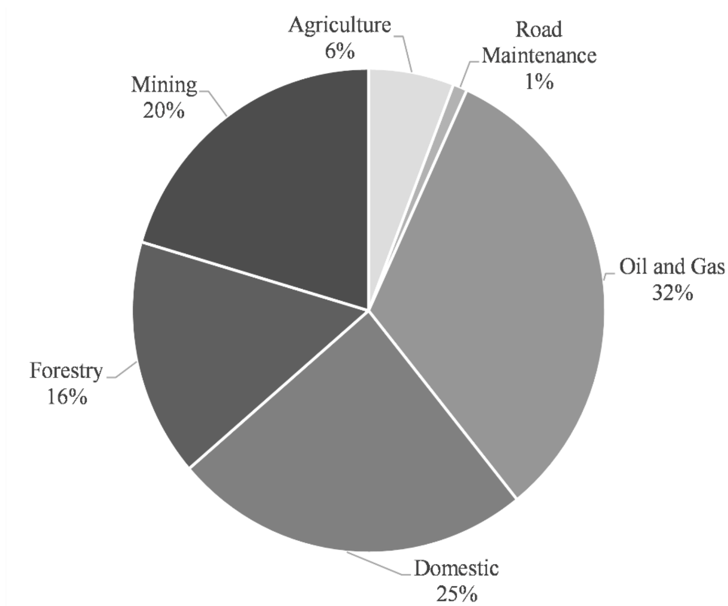

Figure 4.5 Consumptive water allocations in BC (2003-2006)[17].

[17] Figure adapted from (British Columbia, 2006)

The oil and gas industry is the exception to the fact that BC sectors (e.g. agriculture and forestry) have not been required to report how much of the allocated surface water volume is actually used. However, the 2016 *Water Sustainability Act* includes provisions which require monitoring and reporting of actual water use for all large users (e.g. forestry, mining, agriculture) (The Province of British Columbia, 2015). For the oil and gas industry, beginning in January of 2014, companies holding either a Section 8 short-term water license or a water license have been required to report actual water use. The BCOGC publishes an annual report titled *Water Use for Oil and Gas Activity*, which defines both allocated and withdrawn volume (BCOGC, 2014c, 2015d). Implementation of the *Water Sustainability Act* began in 2016 and should improve water accounting practices into the future. While water licenses and short-term Section 8 approvals capture the majority of water use within the oil and gas sector, companies also source water from water source wells, which contributed 14% and 13% of total water withdrawals in 2014 and 2015, respectively (BCOGC, 2014b, 2015). A final source of water includes wastewater generated from oil and gas activities or municipalities (BCOGC, 2015; The Province of British Columbia, 2015).

Due to the lack of provincial water withdrawal data across sectors, *Aqueduct 2.1* estimates for WU (withdrawal) and WA parameters have been considered for each of the four study river basins. *Aqueduct 2.1* WU values collected are themselves a summation of agricultural, industrial (which includes UUOGA), and domestic user group withdrawals within a river basin. The water availability (WA) values collected from the WRI represent *blue water availability* or freshwater availability from surface and groundwater resources. WU and WA values were collected for each of the four study river basins (Gassert et al., 2014).

Table 4.4 Regional withdrawal and availability values (m^3).

River Basin ID	Water Use (WU)	Water Availability (WA)
11863	8.207E+07	5.923E+09
12175	3.404E+08	2.733E+10
12187	1.221E+08	9.553E+08
12208	1.499E+08	2.493E+10

4.3. Precipitation Variation

Precipitation data for the study region was collected from the *CRU dataset* (TS 3.10 version 3.24) (Harris et al., 2014). The *CRU dataset* is a global dataset containing historic temperature and precipitation data for 0.5-degree grid cells. Precipitation data from the 87 grid cells which encompass the study region were collected for the years between 1961 and 1990 to capture a *climate normal scenario*. A map showing the overlay of the 87 grid cells and the study river basins is presented in Figure 4.6 along with major city locations. Again, the right boundary of the grid cells represents the provincial border between BC and AB.

The water use, water availability, and precipitation data referenced within this section are used to complete the water use inventory and water use impact analysis defined in Section 4.4 below. The water use inventories completed for UUOGA, regional water use across all water users, precipitation variation, and regional water availability constitutes the necessary inputs to the assessment framework in support of the WSI calculation which is presented within the Section 4.4 below.

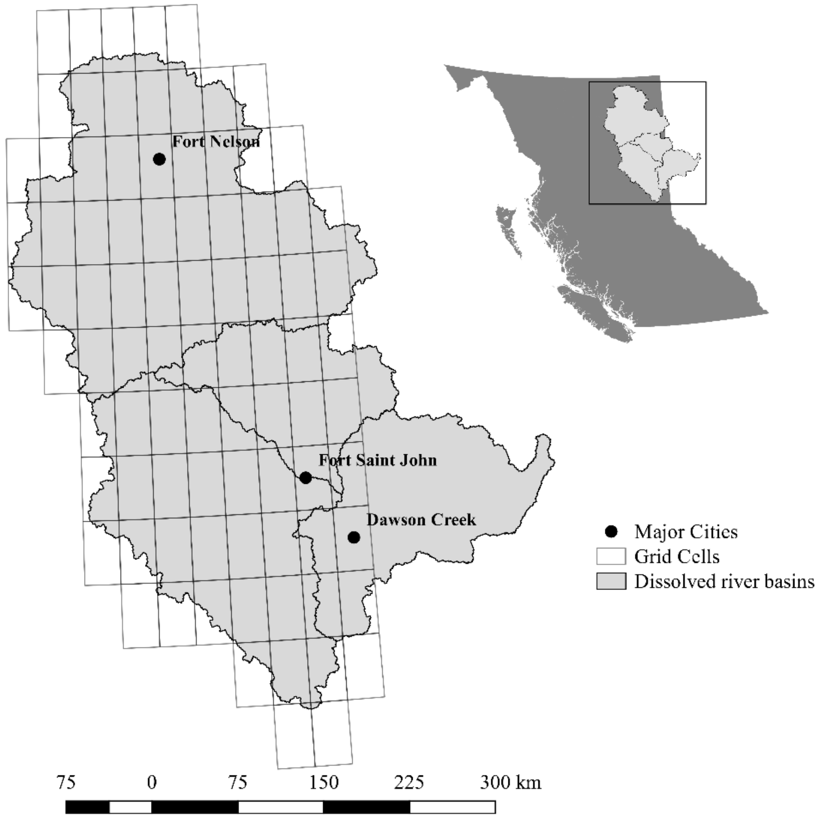

Figure 4.6 Study river basins and *CRU dataset* grid cells[18].

4.4. WSI Calculation: Regional Validation

The WSI framework was validated within the Montney study region to confirm the methodology and verify the reproducibility of results in this regional assessment. Per the Nilsson et al. (2005) distinctions for SRF and non-SRF assumed within Pfister et al. (2009), the river basins evaluated within this study are considered to have SRF. The intermediate VF_i calculation represents the variation in monthly and annual precipitation. Due to precipitation variation being of concern in

[18] Shape/data files for Figure 4.6 were collected from Harris et al., (2014), Gassert et al., (2014), and BCOGC (2016a). City coordinate locations were collected from Google Earth (2016).

the study region, the intermediate VF_i results are also highlighted within Figure 4.7 for each grid cell. Based on precipitation data collected between 1961 and 1990, VF_i values ranging between 1.7 and 2.5 were observed for grid cells evaluated in this analysis.

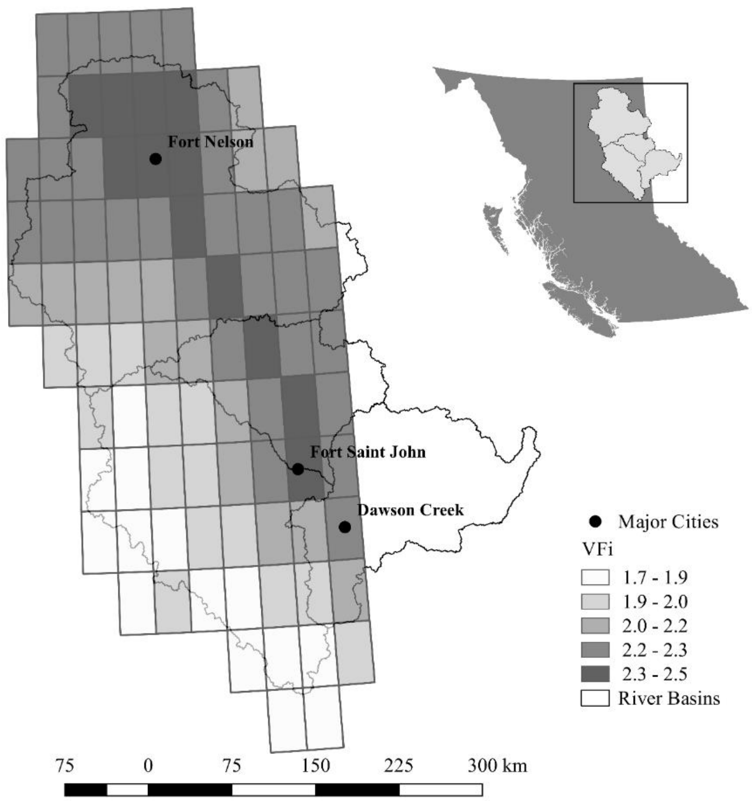

Figure 4.7 VF_i results[19]

In the global maps presented within Pfister et al. (2009), VFws values (Equation 3.4) range between less than 1.2 to greater than 11. While in a global context, the VF_i subsequent VFws values within the study area are low, precipitation variation is considered challenge in NEBC for existing and future oil and gas activities (Fraser Basin Council, 2015). All grid cells falling

[19] Shapefiles for figure 4.7 were collected from Harris et al., (2014), Gassert et al., (2014), and BCOGC (2016a). City location coordinates were collected from Google Earth (2016).

partially or completely within river basin boundaries were considered in the VFws calculation (Equation 3.4). The WSI values calculated for each river basin (Equation 3.5) consider the determined VFws values. The results of the WSI calculation and all its intermediate calculations are presented in Table 4.5 for each river basin alongside the results reported within Pfister et al. (2009). Pfister et al, (2009) presents calculated VFws, WTA*, and WSI results graphically in global figures which display grid cell results according to color gradients which require that results be interpreted visually. Results from Pfister et al. (2009) are included within Table 4.5 as a range for VFws and WTA* values to capture the range of results presented within the region evaluated in this analysis (VF_i and WTA are not presented which is why they have been omitted from the results within Table 4.5).

Table 4.5 WSI Framework validation results and comparison

Parameter	**River Basin**				**(Pfister et al., 2009)**
	11863	**12187**	**12208**	**12175**	
VF	1.89-2.46	0.97-1.14	1.05-1.56	1.05-1.35	-
VFws	2.21	2.19	1.93	2.04	1.2-1.9
WTA	0.01	0.12	0.01	0.01	-
WTA*(%)	2.06	18.9%	0.84%	1.78%	<1-2.9%
WSI	0.01	0.03	0.01	0.01	<0.1

WSI values calculated within Pfister et al. (2009) and in this analysis, are less than 0.1. The differences between the VFws results calculated in this study and those calculated within Pfister et al. (2009) are small (VFws values of greater than 11 were reported within Pfister et al. (2009)). WTA* values were also very similar across studies with the exception of the WTA* value calculated in this research for river basin 12187 (Table 4.5). This variation in the 12187 WTA* is likely due to the relatively high WTA value. The WTA ratios in Pfister et al. (2009) are assumed from the *WaterGap2* model while the WTA ratios in this research have been assumed from the *Aqueduct 2.1 dataset*. The preferred selection of the *Aqueduct 2.1 dataset* allowed for analysis within defined river basins. Additionally, some variation between WTA values (which impact WTA, WTA*, and WSI calculations) could be a result of Pfister et al. (2009) considering data published in 2003 and this research considering data published in 2015 (Gassert et al., 2014). The WTA* values reported in Pfister et al. (2009) range between <1 and >500%. Considering this wide range of possible values, the variation observed is considered to be acceptable. Overall, the difference in the results of intermediate calculations between Pfister et al. (2009) and in this research did not affect the consistency in WSI results across studies.

4.5. Results and Discussions

The WSI calculations determined within this thesis fall below the threshold of moderate water stress, which occurs at a WSI of 0.09. Severe water stress occurs above a WSI of 0.5. While these values are based on WTA based water stress values, thresholds of water stress within a region may occur lower on the WSI gradient than is presented within the literature (Pfister et al., 2009). Figure 4.8 graphically displays the WSI gradient which evaluates water stress as determined through evaluation of the water deprivation indicator.

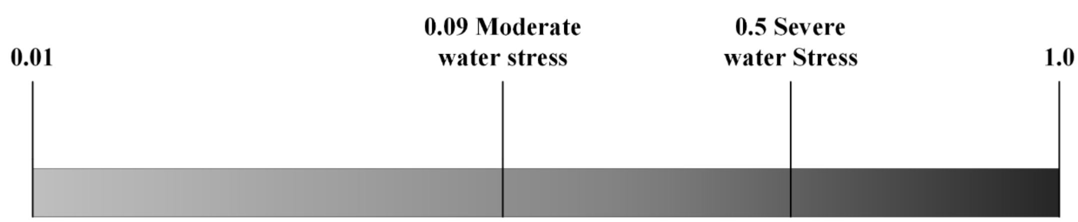

Figure 4.8 WSI result gradient

In this framework, water stress refers to water use which deprives subsequent users of freshwater, specifically with respect to quantity (Pfister et al., 2009).

Despite the determined regional WSI results for NEBC which occur below the thresholds for moderate water stress (0.09), short-term water licenses have been suspended three times in recent history in NEBC for the oil and gas industry (2010, 2012, and 2014) due to low flow conditions (BCOGC, 2016d). The directive that announced the suspension in 2014, Directive 2014-01, indicates that suspensions were due to “escalat[ing] concerns for impacts to fish and aquatic resources and community water supply” (BCOGC, 2014a).

The WSI framework considers precipitation data that is representative of the naturally occurring variation over a 30-year reference period (i.e. 1961-1990), however, understanding the impact of extreme (i.e. low and high) values for input parameters (i.e. annual precipitation or water withdrawal) is important to understanding the full range of water stress values which could occur. In addition to natural variability in inputs, inherent uncertainty exists in the input data and in the WSI framework itself. To understand the variability and uncertainty associated with the input data and the WSI framework, a probabilistic analysis using Monte Carlo simulations (MCSs) is completed within the WSI framework.

4.6. Uncertainty Analysis

Water use metrics provide an approach for modeling complex environmental processes to generate results to inform decision making. As a result, water use metrics, as is the case with all models, contain inherent uncertainties. Quantitative and/or qualitative assessment of inherent uncertainties can increase the reliability of metrics and their results as an input to decision making (ISO, 2015). This section presents the results of qualitative and quantitative uncertainty analysis completed to evaluate uncertainties within the WSI framework. Additional information regarding quantitative uncertainty and sensitivity analysis is presented in Appendix C.

Model, parameter, and scenario uncertainties are evaluated in the context of the WSI framework in the subsequent sections and within Table 4.6. In following with the ISO 1404 (2015) recommendation for completing quantitative and/or qualitative analyses of uncertainty, model uncertainty has been addressed via a qualitative assessment, and parameter and scenario uncertainties have been assessed via a quantitative assessment. Quantitative assessment is completed using Monte Carlo simulations (MCSs), which are frequently used to conduct probabilistic assessment in uncertainty analysis (Sankaran, 1977; Shahriar et al., 2014).

Table 4.6 Characterization of WSI framework uncertainty and applied analysis technique

Source of Uncertainty	Uncertainty Type	Source Category	Analysis Method
Imprecise precipitation measurements	Aleatory	Parameter	Quantitative-MCS
Imprecise estimation of WU and WA	Aleatory	Parameter	Quantitative-MCS
Inaccurate model for estimating WU and WA	Epistemic	Model	Qualitative
WSI Framework	Epistemic	Model	Qualitative
Estimation of future results based on base case data	Aleatory	Scenario	Quantitative-MCS

The software application Oracle ® *Crystal Ball* was used to complete all quantitative assessment. Oracle ® *Crystal Ball* analysis software is an excel application capable of stochastic forecasting and simulation within Microsoft Excel spreadsheets (Oracle, 2012).

4.6.1. Model Uncertainty

The WSI midpoint metric is based on the WTA ratio to evaluate the impact of water deprivation on an ecosystem quality indicator. The values at which the impact of water deprivation on ecosystem quality is experienced is determined through expert judgment, which may not be representative across all ecosystems. Localized assessment of water use impact via the completion of an alternative framework, such as environmental impact assessment, would provide further context to the ability for the WSI metric and its inputs to determine water use impacts (Kounina et al., 2012).

The analysis within the WSI framework presented in this thesis has considered WU and WA estimations as input parameters to the WSI framework. The estimation methodology for determination of these values at a river basin level employs mathematical models to estimate river basin water use based on national and regional use and availability data. Physical measurement of WU and WA values is theoretically possible using robust monitoring and measurement practices, however, due to lack of available data estimations (such as the WU and WA values in this research) are frequently used. As a result, uncertainty resulting as a result of WU and WA estimation models must be acknowledged. Parameter and scenario uncertainties associated with WU and WA values are also discussed within the following sections.

4.6.2. Parameter Uncertainty

To capture the effect of parameter uncertainty on WSI values, not addressed within the deterministic WSI framework presented within Section 0 and implemented within Chapter

CHAPTER 4, a probabilistic assessment using MCS was performed. This analysis provides context to the range of WSI that could be expected within the study region.

To establish a model within Oracle® *Crystal Ball* a user must define "*assumption*" distributions for input parameters and designate an equation (or equations) as a "*forecast*." Oracle ® *Crystal Ball* runs MCS for a specified number of trials for all "*assumptions*" to generate an output distribution of "*forecast*" values (Oracle, 2012). All MCSs completed as a part of both parameter and scenario uncertainty analysis were run with 1,000 trials.

Distributions were fit for input parameters using Oracle's ® Crystal Ball analysis software "*assumption*" definition tool, which can be achieved using either deterministic descriptive values (e.g. mean and standard deviation) or by defining an assumption from a dataset (Appendix C). The WSI framework, based on lognormal distributions of precipitation and availability data determined using the Kolmogorov-Smirnov (KS) goodness of fit test and expert opinion, respectively. However, for this probabilistic analysis, distributions were not fit to precipitation data, rather distributions were being fit to geometric standard deviations of collected precipitation data (s*month and s*year parameters). The Anderson-Darling (AD) goodness of fit test, is generally preferred to the KS test because it is better able to capture deviation in the tails of a distribution (Anderson and Darling, 1954). As a result, the AD test was utilized to select either a normal or a lognormal distribution for all distribution fitting performed for this research.

Distributions were fit to s*month, s*year, and P_i calculations from each of the grid cells within the study region. s*month inputs favored the normal distribution fit while the s*year inputs favored the lognormal distribution. The fit distribution, mean and standard deviation of input parameters is presented in Table 4.7.

WU and WA data collected for this research represent an aggregated measure of water use and availability based on average conditions (Gassert et al., 2014). As a result, only one value is provided per river basin for each WU and WA parameter. To establish a distribution, values 25 and 50 percent above and below each of the four collected WU and WA values were considered to establish regional mean and standard deviation values. Lognormal regional WU and WA distributions were defined based on static mean and standard deviations. This is consistent with Pfister et al. (2009) where a lognormal distribution for WA was also assumed (see Table 4.7).

Table 4.7 WSI input parameter distributions

Input Parameters	**Fit**	**Mean**	**Standard Deviation**
s*month	Normal	2.10	0.19
s*year	Lognormal	1.15	0.01
P_i (m)	Lognormal	0.55	0.15
Withdrawal (WU) (m^3)	Lognormal	1.75E+08	1.30E+08
Availability (WA) (m^3)	Lognormal	1.48E+10	1.36E+10

An MCS was run to determine distributions of each of the intermediate calculations and the final WSI calculation, defined in section 0. To capture the effect of aggregating VF_i values in the VF_{ws} calculation (Equation 3.4), twenty-seven, or the average number of grid cells falling within the perimeter of the four study river basins, values from the VF_i and P_i distributions are considered for each run of the MCS to generate the VF_{ws} distribution. The results of all MCSs completed for each of the intermediate calculations within the WSI framework are presented in Table 4.8. Additionally, the final WSI distribution determined during parameter uncertainty analysis is also presented in Table 4.8.

Table 4.8 WSI framework results resulting from parameter uncertainty analysis

Variables	**Distribution Fit**	**Range**	**Mean**	**Median**
VF_i	Lognormal	1.45-2.5	2.12	2.12
VFws	Normal	1.99-2.24	2.12	2.12
WTA	Lognormal	0.0-0.31	0.02	0.01
WTA*	Lognormal	0.0-0.45	0.03	0.02
WSI	Lognormal	0.01-0.5	0.01	0.01

4.6.3. Sensitivity Analysis

Sensitivity analysis was conducted within the WSI framework parameter uncertainty analysis using *Oracle ® Crystal Ball* software. Sensitivity analysis evaluates the contribution of each input to the results of the analysis (Helton et al., 2006). To capture all inputs (e.g. s*month, s*year, P_i, WU, and WA) to the WSI equation, the WTA* equation (Equation 3.2) is contained within the WTA* factor for analysis. The resulting equation for WSI is defined within Equation 4.1. The results of the sensitivity analysis indicate that the WTA* factor had a 99.5% contribution to variance with the VF_{ws} contributing 0.5% to variance in WSI results.

$$\mathrm{WSI} = \frac{1}{1 + e^{(-6.4)\times(\sqrt{\mathrm{VF}}\times \mathrm{WTA})}\left(\frac{1}{0.01} - 1\right)} \qquad \text{Equation 4.1}$$

While the WSI Framework presents a modified method to capture monthly and annual variation, the impacts of VF_i and VF_{ws} on the variation in WSI calculation was observed to be minimal in this assessment, with WTA* inputs contributing most significantly to the variance in WSI results.

4.6.4. Scenario Uncertainty

To determine the potential impact that changes to water demand and precipitation patterns could have on regional WSI, four scenarios (scenarios A, B, C, and D) have been defined based on expected future change. In these scenarios, s*month and s*year remain unchanged from present day values which are assumed to be the best available representation of the inter and intra-annual variation that could be expected for the region. However, it is estimated that the region will likely experience an increase in variation of precipitation in the coming decades (Fraser Basin Council, 2015).

In NEBC, precipitation is forecast to increase between 4 and 17% by 2080 (Pacific Climate Impacts Consortium, 2012). WA for Scenarios A and C is determined by increasing the present-day availability parameter for each river basin by the minimum forecast increase (4%) (Availability low). For scenarios B and D, the present-day WA parameter has been increased by the maximum percentage increase forecast for the region (17%) (Availability high).

Provincial estimates indicate that changes in the sectors of agriculture, oil and gas, forestry, and mining will create increased water demands in the coming decades (The Province of British Columbia, 2015). Regional population projections are also forecast to increase by approximately 26% by 2050 increasing domestic water demands (The Province of British Columbia, 2016). Similarly, in the Agriculture sector, present day water demand for irrigation is projected to increase between 13% and 45% by 2050 in NEBC, in response to warmer temperatures and a longer growing season (Kerr Wodd Leidal, 2016). In the oil and gas sector, the BCOGC forecasts that freshwater use for unconventional activities will increase between 2014 and 2019 and decline through 2025. The NEBC Resource Municipalities Coalition's report *Comprehensive*

Economic Analysis and Projections for NEBC (2015) projects that between 130 and 936 wells will be drilled per year between the years of 2020 and 2045 (NEBC Resource Municipalities Coalition, 2015).

To capture the potential future demands for sectors that have not reported future water demand projections and to capture future demand that may not have been accounted for in existing projections, scenarios A and B will assume a WU 25% greater than present day (Withdrawal low), and scenarios C and D will assume WU 50% greater than present day (Withdrawal high). It is assumed that these values encompass future water demand increases from all sectors currently operating in NEBC. The WU high scenarios are considered worst case and may be in excess of can be expected for WU for the region in the coming decades. Nevertheless, the availability high scenarios provide insight into the types of water stress which could be encountered under high water use conditions. A scenario description matrix is presented in Table 4.9 and defines the withdrawal and availability values considered under of the four scenarios.

Table 4.9 Scenario definition matrix

	WITHDRAWAL LOW	**WITHDRAWAL HIGH**
AVAILABILITY LOW	Scenario A	Scenario C
AVAILABILITY HIGH	Scenario B	Scenario D

All scenario WSI distributions best fit a lognormal curve. Collectively, values across the four scenarios ranged between 0.01 and 0.75, with mean values of either 0.01or 0.02. All minimum, mean, and median values fell below the threshold of moderate water stress (0.09), however, maximum values under all scenarios exceeded the threshold for moderate water stress.

Maximum values reported under scenario C also exceeded the threshold for severe water stress (0.5). The results of the WSI simulations for the four scenarios are displayed within Table 4.10.

Table 4.10 Scenario uncertainty analysis WSI results for scenarios A, B, C, and D

Scenario	Minimum	Maximum	Mean	Median
A (A_{low} W_{low})	0.01	0.13	0.01	0.01
B (A_{high} W_{low})	0.01	0.36	0.01	0.01
C (A_{low} W_{high})	0.01	0.75	0.02	0.01
D (A_{low} W_{high})	0.01	0.15	0.02	0.01

The results of the scenario analysis indicate the potential for regional water stress in the extent of the determined maximum values. However, determined mean values are low, below the threshold for water stress. From directives (in 2010, 2012, and 2014) suspending short term water approvals due to low flow conditions in NEBC, it can be surmised that further refinement in the WSI framework through increased data collection would be necessary to capture seasonal and annual variability in the experienced regional water stress with water stress characterizations determined through the WSI framework.

Sensitivity analyses were also completed for the four scenario assessments. Sensitivity analyses for each of the four scenarios indicate that like the sensitivity analysis completed under present day conditions in section 4.6.3, WTA* contributes between 99 and 100% to the variance in WSI result with VF_{ws} contributing the remaining difference.

CHAPTER 5. CONCLUSIONS

Development and extraction of unconventional resources is a water-intensive process and quantifying water use and the impact of water use in regions experiencing rapid development of unconventional resources is a critical component of effective water management. This thesis performs a review of water use practices during UUOGA to determine the sources and volumes of water use. Based on the review of water use practices, criteria are established against which common water use metrics are evaluated to determine their suitability for assessing the impacts of water use in UUOGA. A decision tree that differentiates between selected water use metrics based on the study scope (inventory or impact assessment), assessment mechanism, impact indicator assessed, and result units, is developed as a tool to facilitate future selection of water use metrics evaluating UUOGA. A case study implements the selected WSI Framework to complete a regional water use inventory and assessment of water stress within the case study region of the Montney unconventional play trend in NEBC. Finally, an uncertainty analysis is completed within the WSI framework under present day and future scenarios to evaluate parameter, model, and scenario uncertainties.

Water use metrics, such as the WSI, which characterize water stress, can provide valuable regional information to decision makers in absence of robust site-specific data. The results of the WSI framework analysis within this thesis evaluate present day and future scenarios which indicate the environmental impact of water use, as evaluated through the lens of WSI, is likely to be below the thresholds of water stress in NEBC (Pfister et al., 2009). However, NEBC is home to an increasing population of people, aquatic and terrestrial wildlife, and also supports industry such as timber, mining, and agriculture in addition to oil and gas. Water management is critical to ensuring that regional impacts of water use do not increase with increasing future demands for water resources.

5.1. Significance of Work

The recent popularity surge for water use metrics has resulted in the widespread application of the *water footprinting* concept with the goal of providing concise information to decision makers wishing to implement sustainability-conscious practices. Water use metrics provide a methodology for evaluating the impact of regional water use, however, despite the criticism received by the UOG industry, these metrics have not been applied within UOG water use studies. Selection criteria for water use metrics defined within this thesis yielded water use metrics appropriate for assessing the types of water use (consumptive, degradative) relevant to UUOGA and evaluating the regional environment impact of water use. The critical review of the selected water use metrics and their capabilities and limitations in providing important information regarding data requirements, available methodologies, and limitations for decision makers striving to effectively manage water resources. The decision tree has been developed within this thesis to support future analysis applying water use metrics to evaluate UOG/UUOGA water use against environmental indicators. The WSI framework analysis and corresponding uncertainty analysis on water use in NEBC evaluates the range in potential present day and future scenario WSI results, highlighting its benefits and limitations as a decision-making tool.

While water use metrics provide a method for evaluating water use, there are inherent limitations in these metrics and their application to decision making. These limitations do not nullify its results, rather provide more context for decision makers to evaluate and understand the factors resulting in quantity and quality changes in water resources as a result of water use. The following section discusses the limitations of water use metrics in parallel with recommendations for future study.

5.2. Limitations and Recommendations

The WSI midpoint metric framework provides a high-level synopsis of water use, considering variation in precipitation, and can be completed at spatial scales ranging from regional to global. The WTA water stress thresholds, which the WSI framework is largely based on, are defined by expert opinion and based on global observations. These thresholds should be validated with regional monitoring to determine localized thresholds at which water stress occurs given local conditions, such as ecosystem demands and evapotranspiration rates. Defining regional thresholds would decrease uncertainty around the water stress thresholds defined within the WSI framework, or other metrics based on WTA or modified WTA thresholds.

Water use type (consumptive, non-consumptive, degradative) each affect the impact of water use on the environment differently. In BC, water withdrawals are reported for the oil and gas industry as water diverted from a source. However, this value does not differentiate between consumptive and degradative uses of withdrawn volumes. Outside of the oil and gas sector in BC, water withdrawals have not been reported prior to the 2016 implementation of the *Water Sustainability Act*. Rather, consumptive allocation volumes constitute the estimated volumes of water withdrawn, which often exceeds actual water withdrawal. As a result, actual water withdrawal volumes, consumptive water use, and degradative water use volumes are not well characterized. The Pfister et al. (2009) WSI metric in combination with global datasets estimating water withdrawal is a suitable alternative for regions without explicit data on water use type. In the case study presented in this thesis, a global dataset provides an estimate for water withdrawals within the four river basins evaluated. Thus, all water use types are considered to have an equal effect which can over, or under, estimate the actual impact of water use depending on water use types and the region in which they occur. As data collected per the implementation of the 2016 *Water Sustainability Act* becomes available, repeat analysis of the WSI metric, or an alternative water use metric identified within the method selection decision tree, could provide further insight into the environmental context of water use in NEBC.

In this thesis, the WSI metric could not be applied to the BCOGC water management basins (Figure 4.3) due to limited data availability. Increasing the regional specificity of WSI assessment, and all water use metrics, is data intensive and oftentimes the necessary data cannot be obtained. In this research, one data point was available for WU and WA for each of the four study river basins and the WU and WA distributions may not capture the true range in these values. More robust water use accounting, such as WU and WA values for each BCOGC water management basin, would increase the certainty around the WU parameter. Additionally, better water use accounting could facilitate efforts towards sector-specific water demand forecasting.

Increasing spatial and temporal specificity of water use data collection is an area deserving of future research due to the importance of this information to present and future water management. Increased data collection and monitoring which evaluates sector-specific water demand forecasting into the future would give context to the suitability of existing storage structures and water management practices in meeting future water demands. Increased water accounting with respect to the inputs required within the WSI framework could provide increased the temporal and spatial resolution of results which could also illustrate inter-regional trends in water stress over time. Observation of these trends is important for present day and future water management, especially in NEBC river basins whose boundary crosses provincial borders where the responsibility of water management falls to multiple provincial jurisdictions.

REFERENCES

Anderson, T. W., & Darling, D. A. (1954). A Test of Goodness of Fit. *Journal of the American Statistical Association*, *49*(268), 765–769. https://doi.org/10.1080/01621459.1954.10501232

Asgarpour, S. (2013). The modern practices of hydraulic fracturing: A focus on Canadian resources. *Journal of Canadian Petroleum Technology*, *52*(2), 83–85.

Aven, T. (2011). On Different Types of Uncertainties in the Context of the Precautionary Principle. *Risk Analysis*, *31*(10), 1515–1525. https://doi.org/10.1111/j.1539-6924.2011.01612.x

Bayart, J. B., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., & Koehler, A. (2010). A framework for assessing off-stream freshwater use in LCA. *International Journal of Life Cycle Assessment*, *15*(5), 439–453. https://doi.org/10.1007/s11367-010-0172-7

Bayart, J. B., Worbe, S., Grimaud, J., & Aoustin, E. (2014). The Water Impact Index: A simplified single-indicator approach for water footprinting. *International Journal of Life Cycle Assessment*, *19*(6), 1336–1344. https://doi.org/10.1007/s11367-014-0732-3

BC Oil and Gas Commission. (2013). Lng Forecast Scenario. Retrieved from https://www.bcogc.ca/node/11277/download

BC Oil and Gas Commission. (2014a). Directive 2014-01 Suspension of Short-Term Water Withdrawals (Peace River). Retrieved November 27, 2016, from http://www.bcogc.ca/node/11274/download

BC Oil and Gas Commission. (2014b). Hydrocarbon and By-Product Reserves in British Columbia. Retrieved from http://www.bcogc.ca/node/11111/download

BC Oil and Gas Commission. (2014c). *Water Use for Oil and Gas Activity 2014 Annual Report*. Retrieved from https://www.bcogc.ca/node/11263/download

BC Oil and Gas Commission. (2015a). Compliance & Enforcement Manual (Version 2.2), (June). Retrieved from https://www.bcogc.ca/node/6096/download

BC Oil and Gas Commission. (2015b). Fracfocus.ca. Retrieved from fracfocus.ca

BC Oil and Gas Commission. (2015c). IRIS Database.

BC Oil and Gas Commission. (2015d). *Water Management for Oil and Gas Activity 2015 Annual Report*.

BC Oil and Gas Commission. (2016a). Administrative Zones. Retrieved from http://data.bcogc.opendata.arcgis.com/datasets/032cac78a0264d23b7461ba2f8e1a8d7_1

BC Oil and Gas Commission. (2016b). Northeast Water Tool. Retrieved from https://water.bcogc.ca/newt

BC Oil and Gas Commission. (2016c). Regional Fields. Retrieved from http://data.bcogc.opendata.arcgis.com/datasets/2e34e8d9065a46929a9dfdbf97ad3838_1

BC Oil and Gas Commission. (2016d). Unconventional Play Trends. Retrieved from http://data.bcogc.opendata.arcgis.com/datasets/1fa73f091aac4ff1b8b0deee2fa6ff39_1

BC Oil and Gas Commission. (2016e). Water Management Basins. Retrieved from http://data.bcogc.opendata.arcgis.com/datasets/a8b2c47ecf0c4c9c8d3406dd3a06d4e0_0

BC Oil and Gas Commission. (2016f). What is being done to ensure water supplies are conserved? Retrieved November 27, 2016, from http://www.bcogc.ca/what-being-done-ensure-water-supplies-are-conserved

Berger, M. (2014). *Water Footprint – Assessing Impacts of Water Use along Product Life Cycles*. Technische Universität Berlin.

Berger, M., & Finkbeiner, M. (2010). Water footprinting: How to address water use in life cycle assessment? *Sustainability*, *2*(4), 919–944. https://doi.org/10.3390/su2040919

Berger, M., & Finkbeiner, M. (2013). Methodological Challenges in Volumetric and Impact-Oriented Water Footprints. *Journal of Industrial Ecology*, *17*(1), 79–89. https://doi.org/10.1111/j.1530-9290.2012.00495.x

Beven, K. (2015). Facets of uncertainty: epistemic uncertainty, non-stationarity, likelihood, hypothesis testing, and communication. *Hydrological Sciences Journal*, *6667*(April), 150527103244004. https://doi.org/10.1080/02626667.2015.1031761

Bortolan, L., & Kotousov, A. (2013). International Journal of Rock Mechanics & Mining Sciences Residual opening of hydraulic fractures filled with compressible proppant. *International Journal of Rock Mechanics and Mining Sciences*, *61*, 223–230. https://doi.org/10.1016/j.ijrmms.2013.02.012

Boulay, A.-M., Bouchard, C., Bulle, C., Deschênes, L., & Margni, M. (2011). Categorizing water for LCA inventory. *International Journal of Life Cycle Assessment*, *16*(7), 639–651. https://doi.org/10.1007/s11367-011-0300-z

Boulay, A.-M., Motoshita, M., Pfister, S., Bulle, C., Muñoz, I., Franceschini, H., & Margni, M. (2015). Analysis of water use impact assessment methods (part A): evaluation of modeling choices based on a quantitative comparison of scarcity and human health indicators. *International Journal of Life Cycle Assessment*, *20*(1), 139–160. https://doi.org/10.1007/s11367-014-0814-2

Boulay, A., Bulle, C., Deschênes, L., & Margni, M. (2011). LCA Characterisation of Freshwater Use on Human Health and Through Compensation. *Towards Life Cycle Sustainability Management*, *Submitted*(Part 3), 193–204. https://doi.org/10.1021/es1030883

British Columbia. (2006). Surface Water Allocated for Consumptive and Non-consumptive Use by Sector, (March 2003), 2006. Retrieved from http://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/water-rights/surface_allocation_volume_purpose.pdf

Brown, A., & Matlock, M. D. (2011). A Review of Water Scarcity Indices and Methodologies. *The Sustainability Consortium*, 19. https://doi.org/White Paper #106

Burke, L. H., Energy, R. P. S., Nevison, G. W., & Peters, W. E. (2011). SPE 149344 Improved Unconventional Gas Recovery With Energized Fracturing Fluids : Montney Example, 1–10.

Canadian Association for Petroleum Producers. (2012). CAPP Hydraulic Fracturing Operating Practice : Water sourcing, measurement, and reuse.

Carnegie Mellon Scott Institute for Energy Innovation. (2013). *Shale Gas and the Environment: Critical Need for a Government- University-Industry Research Initiative*. Retrieved from http://www.cmu.edu/energy/public-policy/policymaker-guide.pdf%5Cnpapers3://publication/uuid/7C62C8F4-A0F6-4B90-A1BC-7155A87F641B

Chapagain, A. K., & Orr, S. (2008). An improved water footprint methodology linking global consumption to local water resources: A case of Spanish tomatoes. *Journal of Environmental Management*, *90*(2), 1219–1228. https://doi.org/10.1016/j.jenvman.2008.06.006

Clark, C. E., Horner, R. M., & Harto, C. B. (2013). Life Cycle Water Consumption for Shale Gas and Conventional Natural Gas, 1–9.

Council of Canadian Academies. (2014). *Environmental Impacts of Shale Gas Extraction in Canada*.

Cucek, L., Klemes, J. J., & Kravanja, Z. (2012). A review of footprint analysis tools for monitoring impacts on sustainability. *Journal of Cleaner Production*, *34*, 9–20. https://doi.org/10.1016/j.jclepro.2012.02.036

De Rocquigny, E., Devictor, N., & Tarantola, S. (2008). Sensitivity analysis methods (pp. 259–284). John Wiley & Sons, Ltd.

Ernst and Young. (2015). *Review of British Columbia's hydraulic fracturing regulatory framework.*

Forbes, C., Evans, M., Hastings, N., & Peacock, B. (2011). *Statistical Distributions Fourth Edition*. Wiley.

Fraser Basin Council. (2015). Climate Risk Assessment for the Oil & Gas Sector, (May).

Frischknecht, R., Steiner, R., Braunschweig, A., Egli, N., & Hildesheimer, G. (2006). Swiss ecological scarcity method: the new version 2006.

Frischknecht, R., & Sybille, B. K. (2013). Swiss Eco-Factors 2013 according to the Ecological Scarcity Method. Federal Office for the Environment (FOEN). Retrieved from http://www.bafu.admin.ch/publikationen/publikation/01750/index.html?lang=en

Gassert, F., Landis, M., Luck, M., Reig, P., & Shiao, T. (2014). Aqueduct Global Maps 2.1: Constructing decision-relevant global water risk indicators. *World Resources Institute*, (April), 31.

Geoscience BC. (2011). Collaborative Geoscience for Resource Development and Resource Management in the Montney Gas Play, Northeast British Columbia Montney Water Montney Water Project Supports the Kiskatinaw River Watershed Research Project, (June). Retrieved from http://www.geosciencebc.com/i/pdf/MWPbrochure.pdf

Google Earth. (2016). Google Earth City Location.

Gregory, K. B., Vidic, R. D., & Dzombak, D. A. (2011). Water Management Challenges Associated with the Production of Shale Gas by Hydraulic Fracturing. *Elements*, *7*(3), 181–186. https://doi.org/10.2113/gselements.7.3.181

Hanafiah, M., Xenopoulos, M., Pfister, S., Leuven, R., & Huijbregts, M. (2011). Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. *Environmental Science & Technology*, *45*(12), 5272–8. https://doi.org/10.1021/es1039634

Harris, I., Jones, P. D., Osborn, T. J., & Lister, D. H. (2014). Updated High Resolution Grids of Monthly Climatic Observations-The CRU TS 3.10 Dataset. International Journal of Climatology. https://doi.org/10.1002/joc.3711

Helton, J. C., Johnson, J. D., Sallaberry, C. J., & Storlie, C. B. (2006). Survey of sampling-based methods for uncertainty and sensitivity analysis. *Reliability Engineering and System Safety*, *91*(10–11), 1175–1209. https://doi.org/10.1016/j.ress.2005.11.017

Hoekstra, A., Chapagain, A., Aldaya, M., & Mekonnen, M. (2011). *The Water Footprint Assessment Manual*. *Febrero 2011*. https://doi.org/978-1-84971-279-8

Hoekstra, A. Y., & Mekonnen, M. M. (2012). The water footprint of humanity. *Proceedings of the National Academy of Sciences*, *109*(9), 3232–3237. https://doi.org/10.1073/pnas.1109936109

Hultman, N., Rebois, D., Scholten, M., & Ramig, C. (2011). The greenhouse impact of unconventional gas for electricity generation. *Environmental Research Letters*. https://doi.org/10.1088/1748-9326/6/4/049504

IEA, I. E. A. (2015). World Energy Outlook. https://doi.org/http://www.iea.org/publications/freepublications/publication/WEB_WorldEnergyOutlook2015ExecutiveSummaryEnglishFinal.pdf

IHS Accumap. (2016). IHS Accumap Dataset. Retrieved from https://www.ihs.com/products/oil-gas-tools-accumap.html

International Energy Agency. (2016). Energy Statistics. *Statistics*. Retrieved from http://www.iea.org/statistics/statisticssearch/

International Organization for Standardization. (2006a). International Standard ISO 14044 Environmental Management - Life cycle assessment - Requirements and guidelines.

International Organization for Standardization. (2006b). ISO 14040 Environmental management — Life cycle assessment — Principles and framework. *ISO 14040*. https://doi.org/10.1136/bmj.332.7550.1107

International Organization for Standardization. (2015). ISO 14046 Environmental management - Water footprint - Principles , requirements and guidelines.

International Risk Governance Council. (2013). Risk Governance Guidelines for Unconventional Gas Development. Retrieved from http://www.irgc.org/wp-content/uploads/2013/12/IRGC-Report-Unconventional-Gas-Development-2013.pdf

Jiang, M., Hendrickson, C. T., & Vanbriesen, J. M. (2013). Life cycle water consumption and wastewater generation impacts of a Marcellus shale gas well. *Environmental Science and Technology*, *48*(3), 1911–1920. https://doi.org/10.1021/es4047654

Jiang, M., Michael Griffin, W., Hendrickson, C., Jaramillo, P., VanBriesen, J., & Venkatesh, A. (2011). Life Cycle Analysis on Greenhouse Gas (GHG) Emissions of Marcellus Shale Gas. *Environmental Research Letters*, 1–20. Retrieved from http://iopscience.iop.org/1748-9326/6/3/034014/fulltext/

Johnson, E. G., & Johnson, L. a. (2012). Hydraulic fracture water usage in northeast British Columbia : locations, volumes, and trends. *Geoscience Reports*, 41–63.

Kerr Wodd Leidal. (2016). *Evaluation of Irrigation Potential in the BC Peace Region*. Retrieved from http://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/northeast-water-strategy/latest-information/evaluation-irrigation-potential-peace-summary.pdf

Klinglmair, M., Sala, S., & Brandão, M. (2014). Assessing resource depletion in LCA: A review of methods and methodological issues. *International Journal of Life Cycle Assessment*, *19*(3), 580–592. https://doi.org/10.1007/s11367-013-0650-9

Koehler, A. (2008). Water use in LCA: Managing the planet's freshwater resources. *International Journal of Life Cycle Assessment*, *13*(6), 451–455. https://doi.org/10.1007/s11367-008-0028-6

Kondash, A., & Vengosh, A. (2015). Water Footprint of Hydraulic Fracturing. *Environmental Science & Technology Letters*, 150918090943001. https://doi.org/10.1021/acs.estlett.5b00211

Kounina, A., Margni, M., Bayart, J.-B., Boulay, A.-M., Berger, M., Bulle, C., … Humbert, S.

(2012). Review of methods addressing freshwater use in life cycle inventory and impact assessment. *The International Journal of Life Cycle Assessment*, 707–721. https://doi.org/10.1007/s11367-012-0519-3

Kreinovich, V., Beck, J., Ferregut, C., Sanchez, A., Keller, G. R., Averill, M., & Starks, S. A. (2007). Monte-Carlo-type techniques for processing interval uncertainty, and their potential engineering applications. *Reliable Computing*, *13*(1), 25–69. https://doi.org/10.1007/s11155-006-9021-6

Kurian, G. T. (2013). *The AMA Dictionary of Business and Management*. AMACOM.

Laurenzi, I. J., & Jersey, G. R. (2013). Life Cycle Greenhouse Gas Emissions and Freshwater Consumption of Marcellus Shale Gas. *Environmental Science & Technology*, *47*(9), 4896–903. https://doi.org/10.1021/es305162w

Lloyd, S. M., & Ries, R. (2007). Characterizing, propagating, and analyzing uncertainty in life-cycle assessment. *Journal of Industrial Ecology*, *11*(1), 161–181. https://doi.org/10.1162/jiec.2007.1136

Loucks, D. P., van Beek, E., Stedinger, J. R., Dijkman, J. P. M., & Villars, M. T. (2005). *Model Sensitivity and Uncertainty Analysis*. *Water Resources Systems Planning and Management: An Introduction to Methods, Models, and Applications*. https://doi.org/ISBN: 92-3-103998-9

McAuliff, K., Sadiq, R., & Hewage, K. (n.d.). Environmental Impacts of Water Use in Unconventional Oil and Gas Development: An Assessment on Method Evaluation and Selection.

Milà I Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., & Clift, R. (2009). Assessing freshwater use impacts in LCA: Part I - Inventory modeling and characterization factors for the main impact pathways. *International Journal of Life Cycle Assessment*, *14*(1), 28–42. https://doi.org/10.1007/s11367-008-0030-z

Ministry of Environment. (2016). Environmental Flow Needs Policy. Retrieved from http://www2.gov.bc.ca/assets/gov/environment/air-land-water/water/water-rights/efn_policy_mar-2016_signed.pdf

National Energy Board. (2013). *Canada's energy future 2013 - Supply and demand projections to 2035*. Ottawa, ON.

NEBC Resource Municipalities Coalition. (2015). *Comprehensive Economic Analysis and Projections for NEBC*. Retrieved from http://nebccoalition.com/images/Coalition_Reports/Preliminary Projections Report - 2015-Aug 4 DISTRUIBUTED.pdf

Nicot, J.-P., Hebel, A. K., Ritter, S. M., Walden, S., Baier, R., Galusky, P., … Breton, C. (2011). *Current and Projected Water Use in the Texas Mining and Oil and Gas Industry*. Retrieved from http://www.twdb.texas.gov/publications/reports/contracted_reports/doc/0904830939_MiningWaterUse.pdf

Nicot, J.-P., Scanlon, B., Reedy, R., & Costly, R. (2014). Source and Fate of Hydraulic Fracturing Water in the Barnett Shale: A Historical perspective. *Environmental Science and Technology*, (48), 2464–2471.

Nicot, J. P., & Scanlon, B. R. (2012). Water use for shale gas production in Texas, U.S. *U.S. Environmental Science and Technology*, *46*, 3580–3586. https://doi.org/10.1021/es204602t

Nilsson, C., Reidy, C. A., Dynesius, M., & Revenga, C. (2005). Fragmentation and Flow Regulation of the World's Large River Systems. *Science*, *8*, 405408.

NIST/SEMATECH. (2013). Exploratory Data Analysis. *E-Handbook of Statistical Methods*. Retrieved from http://www.itl.nist.gov/div898/handbook/

Norton Rose, Fullbright & Jaworski. (2013). Trade secrets and the regulation of hydraulic fracturing. Retrieved from http://www.nortonrosefulbright.com/files/us/images/publications/20130410TradeSecretsAndTheRegulationOfHydraulicFracturing.pdf

Oracle. (2012). Oracle Crystal Ball User's Guide (Release 11.1.2). Retrieved January 9, 2016, from http://docs.oracle.com/cd/E17236_01/epm.1112/cb_user/frameset.htm?ch01s04.html

Pacific Climate Impacts Consortium. (2012). Summary of Climate Change for British Columbia in the 2080s.

Pfister, S., Koehler, A., & Hellweg, S. (2009). Assessing the environmental impacts of freshwater consumption in LCA. *Environmental Science and Technology*, *43*(11), 4098–4104. https://doi.org/10.1021/es802423e

Rach, N. M. (2009). CNX Gas drills record Marcellus shale well. *Oil Gas J*, *107*(3), 51–55.

Ridoutt, B., Fantke, P., Pfister, S., Bare, J., Boulay, A.-M., Cherubini, F., … Wiedmann, T. (2015). Making Sense of the Minefield of Footprint Indicators. *Environmental Science & Technology*, 150212130530000. https://doi.org/10.1021/acs.est.5b00163

Ridoutt, B. G., & Pfister, S. (2010). A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. *Global Environmental Change*, *20*(1), 113–120. https://doi.org/10.1016/j.gloenvcha.2009.08.003

Ridoutt, B. G., & Pfister, S. (2013). A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. *International Journal of Life Cycle Assessment*, *18*(1), 204–207. https://doi.org/10.1007/s11367-012-0458-z

Rivard, C., Lavoie, D., Lefebvre, R., Séjourné, S., Lamontagne, C., & Duchesne, M. (2014). An overview of Canadian shale gas production and environmental concerns. *International*

Journal of Coal Geology, *126*, 64–76. https://doi.org/10.1016/j.coal.2013.12.004

Sadiq, R. (2001). *Drilling waste discharges in the marine environment: A risk-based decision methodology*. The Memorial University of Newfoundland.

Sadiq, R., & Tesfamariam, S. (2009). Environmental decision-making under uncertainty using intuitionistic fuzzy analytic hierarchy process (IF-AHP). *Stochastic Environmental Research and Risk Assessment*, *23*(1), 75–91. https://doi.org/10.1007/s00477-007-0197-z

Sankaran, M. (1977). *Reliability-Based Mechanical Design*. (T. A. Cruse, Ed.). Marcel Dekker, Inc.

Scanlon, B. R., Reedy, R. C., Male, F., & Hove, M. (2016). Managing the Increasing Water Footprint of Hydraulic Fracturing in the Bakken Play, United States. *Environmental Science & Technology*, acs.est.6b01375. https://doi.org/10.1021/acs.est.6b01375

Scanlon, B. R., Reedy, R. C., & Nicot, J. P. (2014). Comparison of water use for hydraulic fracturing for unconventional oil and gas versus conventional oil. *Environmental Science & Technology*, *48*(20), 12386–93. https://doi.org/10.1021/es502506v

Scown, C. D., Horvath, A., & Mckone, T. E. (2011). Water Footprint of U . S . Transportation Fuels, 2541–2553.

Shahriar, A., Sadiq, R., & Tesfamariam, S. (2014). Life cycle greenhouse gas footprint of shale gas: a probabilistic approach. *Stochastic Environmental Research and Risk Assessment*, *28*(8), 2185–2204. https://doi.org/10.1007/s00477-014-0874-7

Speight, J. G. (2013). *Shale Gas Production Processes*. Elsevier Science. Retrieved from http://www.sciencedirect.com/science/book/9780124045712

The Province of British Columbia. (2015). Northeast Water Strategy: Ensuring the responsible use and management of Northeast British Columbia's water resources, 1–40. Retrieved from http://www2.gov.bc.ca/gov/DownloadAsset?assetId=036CAF38B8DC439492702F9C11C7A3FB&filename=2015-northeast-water-strategy.pdf%5Cnpapers2://publication/uuid/D1CC8124-A73C-4960-9389-8C8BA5F85199

The Province of British Columbia. (2016). Population Projections. Retrieved from http://www.bcstats.gov.bc.ca/StatisticsBySubject/Demography/PopulationProjections.aspx

U.S. Energy Information Administration. (2013). *Technically Recoverable Shale Oil and Shale Gas Resources : An Assessment of 137 Shale Formations in 41 Countries Outside the United States* (Vol. 2013). Retrieved from https://www.eia.gov/analysis/studies/worldshalegas/archive/2013/pdf/fullreport_2013.pdf

United Nations World Water Assessment Programme (WWAP). (2016). *The United Nations World Water Development Report 2016: Water and Jobs*. Paris: United Nations Educational, Scientific, and Cultural Organisation (UNESCO). Retrieved from http://unesdoc.unesco.org/images/0024/002439/243938e.pdf#244432

Vidic, R. D., Brantley, S. L., Vandenbossche, J. M., Yoxtheimer, D., & Abad, J. D. (2013). Impact of shale gas development on regional water quality. *Science (New York, N.Y.)*, *340*(6134), 1235009. https://doi.org/10.1126/science.1235009

APPENDICES

APPENDIX A

Unconventional Oil and Gas Water Demand Drivers

All data for the analysis presented in this appendix was collected from the BCOGC *IRIS* and *Fracfocus* databases (2015b, 2015c). While production rates vary between wells, a typical estimated ultimate recovery (EUR) for a Montney well is 4.8 billion cubic feet (bcf) of natural gas (Shahriar et al., 2014). Mean water demand for hydraulic fracturing to access the hydrocarbon reserves by year is presented in Figure A.1. The study dataset contained only 40 wells from 2015 (data collected was accessed in September 2015), so values depicted within Figure A.1 for 2015 were obtained from the BCOGC report, *Water Management for Oil and Gas Activities* (BCOGC, 2015). Since 2012, both the number of wells drilled per year in the NM and RH and the mean volume of water used for hydraulic fracturing for each well has increased. In the NM, average water use per well increased by 9.4% between 2012 and 2013 (2012:11,634 m^3/well; 2013:12,729 m^3/well), 4.0% between 2013 and 2014 (2014:13,251 m^3/well), and 22.7% between 2014 and 2015 (2015:16,258 m^3/well). A greater number of wells were drilled per year in the NM in 2013 and 2015, while a greater number of wells were drilled in the RH in 2012 and 2014. In general, between 2012 and 2015, RH wells use less water for hydraulic fracturing than NM wells. In the RH, average water use per well increased by 16.5% between 2012 and 2013 (2012:7,224 m^3/well; 2013:8,414 m^3/well), 4.0% between 2013 and 2014 (2014:10,895 m^3/well), and 22.7% between 2014 and 2015 (2015: 12,225 m^3/well).

Well design specifications can impact the volume of water use required for a well, and the volume of hydrocarbons produced. The number of fracture stages, length of lateral, and type of treatment fluid used for fracturing stages can have an impact on the volume of water required to complete hydraulic fracturing. Well design parameters by year for NM and RH fields are presented in graphs *a-d* within Figure A.2.

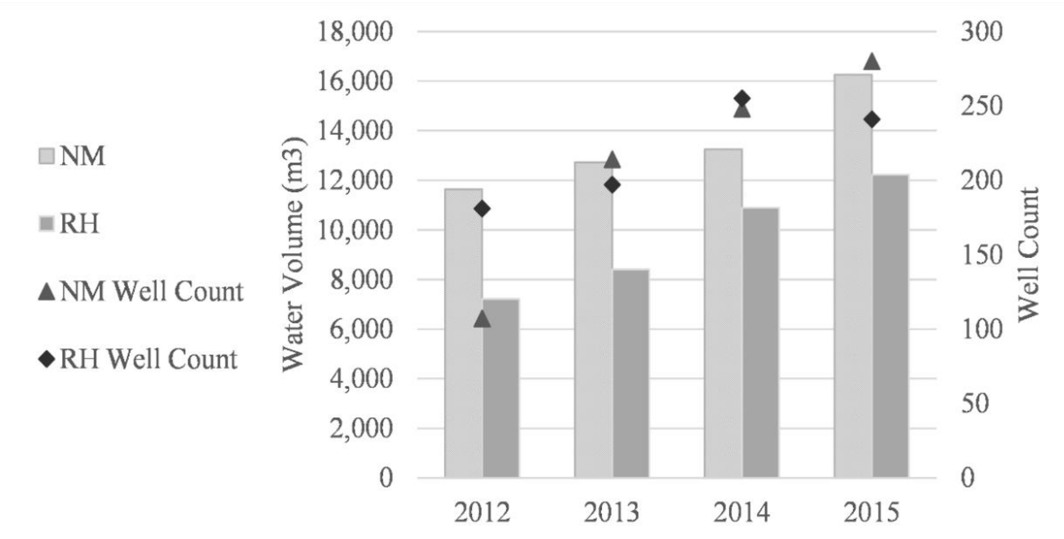

Figure A.1 Average hydraulic fracturing water demand (m^3/well) and well counts

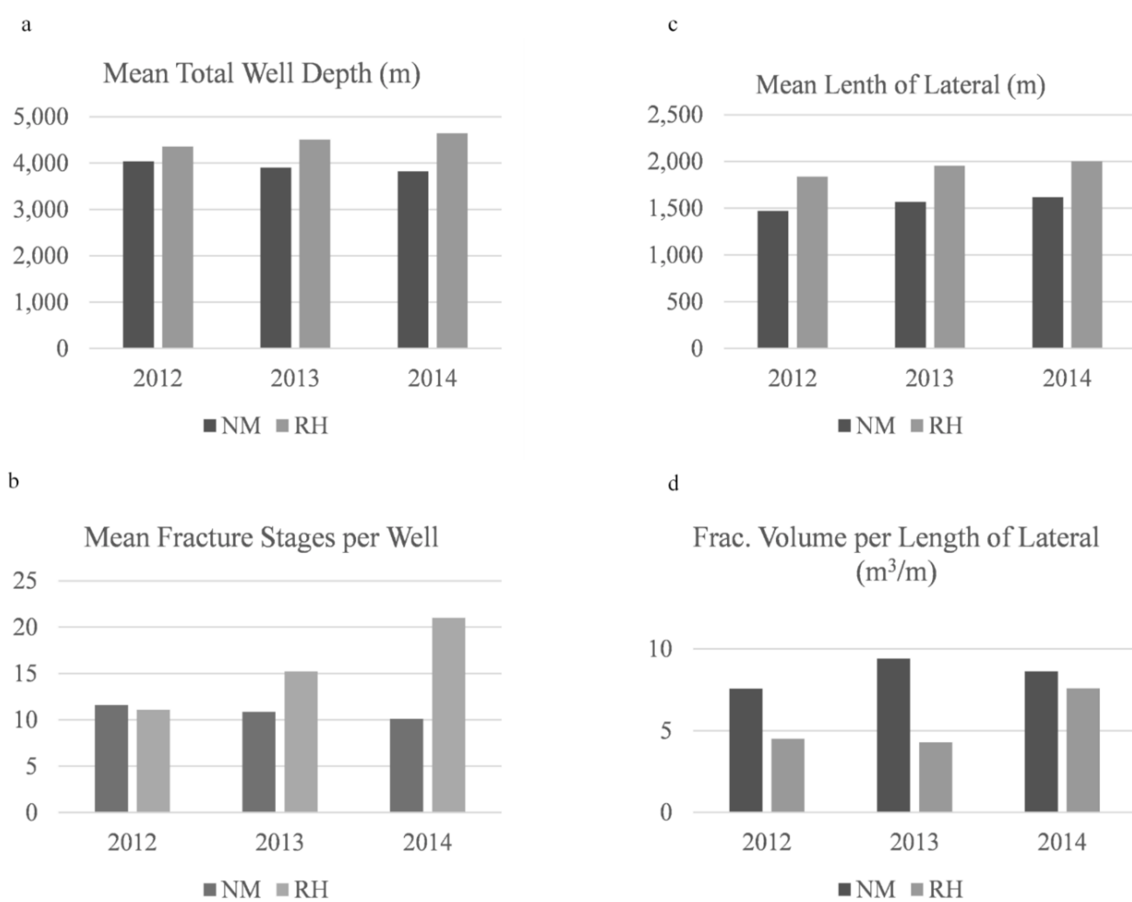

Figure A.2 Well characteristics by year for NM and RH wells (2012-2014)

Mean total well depths (m) in graph *a* of Figure A.2 decreased slightly, 3% per year, in the NM between 2012 and 2014 (2012: 4,042; 2013: 3,902; 2014: 3,823) and increased (also by 3% per year) in the RH (2012: 4,356; 2013:4,507; 2014: 4,644).

The same general trend was observed for the average number of fracture stages per well. Fracture stages refer to the intervals along the lateral portion of an unconventional well bore that are isolated and hydraulically fractured in sequence (Scanlon et al., 2016). In the NM, an average decrease of 7% per year in average number of fracture stages per well is observed, while in the RH the average number of fracture stages per well increased by an average of 37.5% per year (Figure A.2: graph *b*). The length of the lateral portion of each unconventional well was calculated as the maximum reported completion bottom depth less the minimum reported completion top depth. Completion bottom depth and completion top depth values were obtained from the *IRIS* dataset. Within graph *c* (Figure A.2), the mean length of lateral (m) increased in both the NM and RH by an average of 5% per year (NM 2012:1,471; 2013: 1,568; 2014:1,621; and RH 2012:1,837; 2013: 1,953; 2014: 2,003). Finally, fracturing fluid volume per length of lateral (m^3/m) is presented in graph *d* (Figure A.2). Unlike the other graphs within Figure A.2, fracturing volume per length of lateral does not consistently increase or decrease. Rather the NM fracturing volume/length of lateral increases between 2012 and 2013 and decreases between 2013 and 2014. In the RH, fracturing volume/length of lateral decreases between 2012 and 2013 and increases between 2013 and 2014.

Freshwater from surface water sources constitutes a large (64%) portion of the fluid used for hydraulic fracturing, but operators can also use alternative fluids, including saline water obtained from deep source water wells, wastewater obtained from oil and gas activities or local municipalities, and water obtained through private acquisition (e.g. local landowners). Figure A.3 displays BCOGC's estimate of the portion of source water types used for hydraulic fracturing activity throughout NEBC (BCOGC, 2015d).

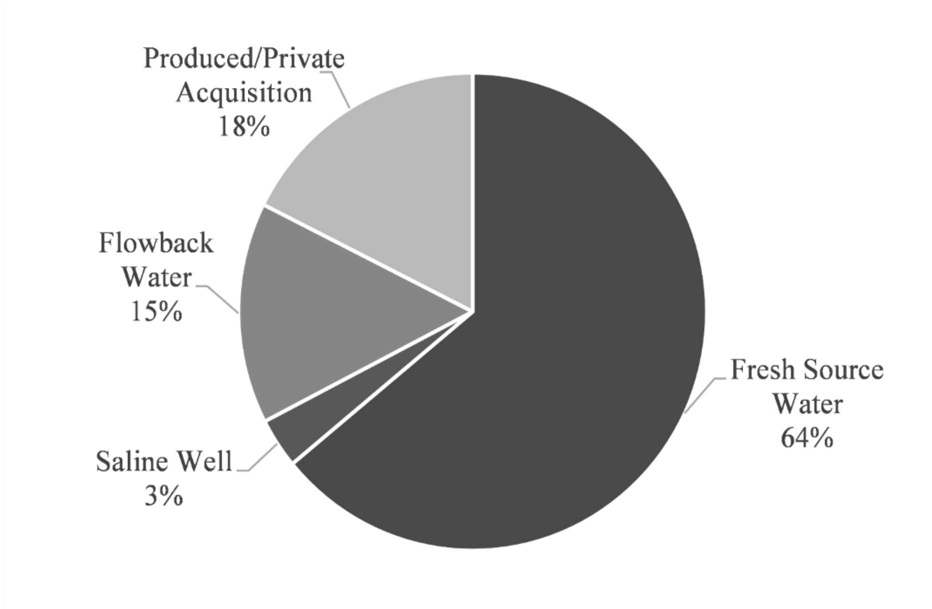

Figure A.3 Water sources of BC fracturing fluid[20]

The type of base fluid(s) used to complete each fracture stage is reported in Figure A.4. The majority of stages were completed using freshwater, and the percentage of stages completed using freshwater increased between 2012 and 2014. As the percentage of fracture stages completed using freshwater increased, the percentage of fracture stages completed with saline water decreased.

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[20] Figure A.3 adapted from (BCOGC, 2014c)

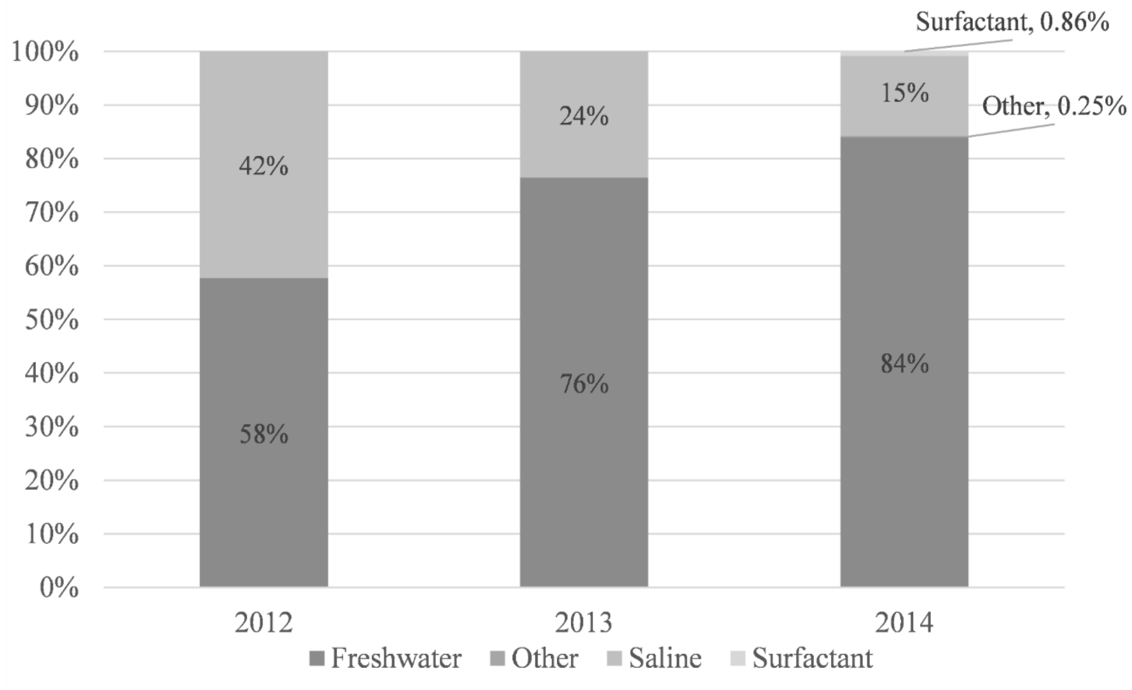

Figure A.4 Percentage of fracture stages completed by base fluid

Wells within the study population which report fluid treatment type within the *IRIS* database reported three general fluid types: slick water, crosslinked, and linear (Figure A.5). Slickwater remains the most common fluid type used by industry in BC and is a mixture of water, proppant, friction reducers, and other chemicals (Rivard et al., 2014). Linear and crosslink gelling agents are also used to a lesser extent in the Montney Play. Crosslinked and linear gel fracturing fluids have a higher viscosity than slickwater fluids (which are largely comprised of water) which results in an increased ability to transport proppant throughout the formation fractures. Linear gels are formed when powdered polymers (e.g. guar gum), are hydrated during their combination with a fracturing fluid solution (saline or freshwater) (Johnson and Johnson, 2012).

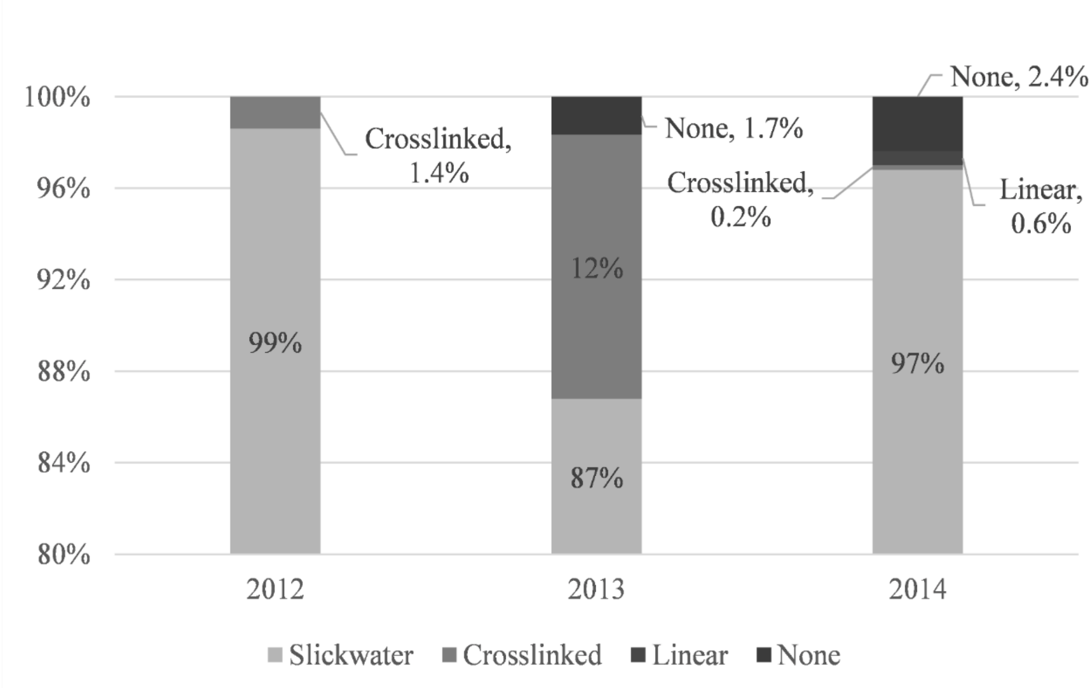

Figure A.5 Percentage of fluid treatment type by fracture stage

Crosslinked gels are named for the links between metallic salt molecules and hydrated polymers which form with increasing temperatures experienced down the well bore and increases the viscosity of the fracturing fluids (Asgarpour, 2013). The most appropriate type of fracturing fluid for a given well is determined by an operator based geology, well production costs, and hydrocarbon production (Johnson and Johnson, 2012). Evaluating changes in well characteristics over time can shed light on the changes on water use and hydrocarbon production efficiency. As data is collected over time within the BCOGC databases, long-term trends can be applied to better predict water demands into the future.

APPENDIX B

BC Unconventional Oil and Gas Regulatory Framework

The regulatory framework which governs unconventional gas operations across North America exists on multiple levels, with federal governments providing authority for states, provinces, and territories to regulate operations within their jurisdiction. Acts define legal requirements for the oil and gas industry, which includes unconventional operations. Regulations exist under each Act to describe the applicable expectations, implementation details, and compliance and enforcement tools.

At the federal level, the National Energy Board (NEB) is responsible for promoting safety, environmental protection, and efficiency in energy production at the federal level. In this role, the NEB regulates the import and export of energy products, the construction and operation of international and interprovincial pipelines, and the construction and operation of international and certain interprovincial power lines (National Energy Board, 2013). The legal requirements which govern BC's oil and gas industry are defined within Acts originally passed by the BC legislature. The BC Ministry of Natural Gas Development (the Ministry) is responsible for guiding development and economic benefits of natural gas resource development, which includes issuing subsurface petroleum and natural gas drilling licenses under the Petroleum and Natural Gas Act (PNGA). The Ministry is also responsible for the BC Oil and Gas Commission (BCOGC), a crown corporation and the independent provincial regulator responsible for the oversight of oil and gas activities in BC (National Energy Board, 2013).

The BCOGC is the main agency responsible for the Oil and Gas Activities Act (OGAA), though it has been structured to ensure single-window regulatory functionality and in addition to OGAA also maintains the authority to administer certain requirements within other provincial acts such as the Water Sustainability Act, Environmental Management Act, Forest Act, and Heritage Conservation Act (Ernst & Young, 2015). Under each Act, regulations identify compliance expectations for industry and the regulator itself, such as data collection and monitoring and reporting requirements. For example, the Drilling and Production Regulation establishes design specifications for boreholes to protect surface fresh groundwater resources and the Environmental Protection and Management Regulation establishes the limitations to impacting surrounding water quality during oil and gas activities.

In addition to Acts and Regulations, the BCOGC provides oversight, guidance, and information to the unconventional industry through permitting, compliance and enforcement, and information documents and programs. The BCOGC's permitting process allows for the regulator to establish conditions or required actions during the permit application process and issue permit cancellations as necessary. This provides flexibility for permits to be reviewed and addressed on a case-by-case basis. The BCOGC's compliance and enforcement process entails the performance of inspections, investigations, and audits to ensure compliance with provisions contained both within regulatory provisions and permit conditions (BCOGC, 2015a). Information content provided by the BCOGC includes guidance documents, non-regulatory standards, letters, and industry support tools. For example, the BCOGC letter (OGC 09-07) entitled Storage of Fluid Returns from Hydraulic Fracturing Operations includes the requirements for containment, storage, and disposal of fluids which return to the surface following hydraulic fracturing activities.

The BCOGC publicly maintains a number of web-based informational programs such as its NorthEast (and NorthWest) Watershed Assessment Tool (NEWT and NWWT, respectively) which are GIS-based hydrologic mapping tools used by the BCOGC in its review of short-term water use applications (BCOGC, 2014; Ernst & Young, 2015). The FracFocus database (available at FracFocus.ca) which contains information pertaining to the hydraulic fracturing process such as well location, fracturing fluid quantity, depth, and composition of fracturing fluid, is also maintained by the BCOGC. This information provides regulators, industry, and the public with information regarding the quantity and quality of water used for hydraulic fracturing (Norton et al., 2013). Disclosure of fracturing information to the FracFocus registry became a requirement in 2012, and the registry contains data from approximately 1,500 wells fractured between 2012 and 2014 (BCOGC, 2014). An exception to the disclosure of chemicals within hydraulic fracturing fluid is trade secret chemicals that meet the requirements for a claim of exemption through the Hazardous Materials Information Review Act (Norton et al., 2013).

Combined with its role as the regulator for the BC oil and gas industry, the BCOGC is also responsible for the authorization and regulation of water use from both surface and subsurface sources for the oil and gas industry in BC. As such, the BCOGC is tasked with ensuring that water use for the oil and gas industry does not inhibit environmental and community water needs.

APPENDIX C

Uncertainty and Sensitivity Analyses

Monte Carlo simulations (MCSs) are designed to consider a distribution (e.g. range and probability) of input variables to determine a resulting distribution which communicates the range of outputs and the associated probabilities at which they occur (Sankaran, 1977). MCSs have been widely applied in probabilistic quantitative assessment of both uncertainty and sensitivity (Helton et al., 2006). Uncertainty analysis is completed to determine and quantify uncertainty in the inputs (i.e. parameters), models, or scenarios, while sensitivity analysis is performed to determine the contribution of inputs to the variation of results. As a model is always a simplification of reality, the additional context provided by uncertainty and sensitivity analyses is useful when considering model results to inform decision making (Loucks et al., 2005).

In MCSs, a probability distribution is considered as an alternative to a single value for variables that contain inherent uncertainty. A different set of values from the probability distributions input to the model are considered during each trial or run. A typical MCS will complete many trials (often 1,000 or 10,000) to generate a probability distribution of outcomes (Aven, 2011). The first step of uncertainty and sensitivity analysis via MCSs is the characterization of uncertainty via the definition of probability distributions.

Probability distributions communicate the probability associated with a set of continuous variables (e.g., those which can occur at an infinite number of points within an interval) or discrete variables (e.g., those which occur at specified values within an interval) (Forbes et al., 2011). Probability distribution functions for continuous variables are referred to as probability density functions (PDFs) and discrete probability distributions functions are referred to as probability mass functions (NIST/SEMATECH, 2013). Discrete and continuous probability functions convey the nature of data in their shape which indicates the minimum and maximum values and the presence of skewness in data. All data considered in this research were continuous. Common probability functions for continuous distributions are presented in Figure C.1 (Sadiq, 2001).

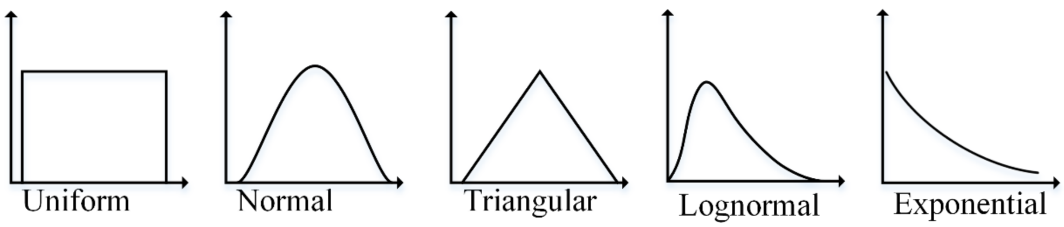

Figure C.1 Common PDFs of statistical distributions (continuous)

In each of the distributions found within Figure C.1, the *y* (i.e. vertical axis) represents probability and the *x* (i.e. horizontal axis) represents the data interval.

Uniform distribution defines an equal probability for all data within the range of a continuous distribution. Normal (or Gaussian) distribution is the most widely used and defines the highest probability around the mean value occurring in the center of the distribution. Triangular distribution defines the highest probability around a *most likely* value occurring within the extents (not necessarily in the center) of the range created by minimum and maximum values. Lognormal distribution defines values which are asymmetrical and positively skewed, with most values found near the lower extent of the distribution. Finally, exponential distribution is defined by the highest probability at the lower extent of the distribution and lowest probability at the highest extent of the distribution. Exponential distributions are often best fit to time-related data (e.g. failure rates) (NIST/SEMATECH, 2013; Oracle, 2012). While the distributions presented represent the most commonly used types of statistical distributions, numerous others have been defined and applied throughout literature for different applications.

Determination of the most appropriate statistical distribution for sample data can be accomplished using a goodness-of-fit test. Generally, goodness-of-fit tests measure how well sample data fits a given probability function by testing the hypothesis:

- H_0: The model fits
- H_A: the model does not fit

Despite testing the same general hypothesis, goodness-of-fit tests each have varying capabilities. Modern software, such as *Oracle ® Crystal Ball* used in this thesis, provide means for completing distribution fitting by calculating a fit statistic to rank multiple probability functions. Table C.1 defines and compares three common goodness-of-fit tests (NIST/SEMATECH, 2013).

Table C.1 Goodness-of-fit test comparison

Goodness-of-Fit Test	**Comments**	**Applicable Distributions**
Anderson-Darling (A-D)	A-D is an alternative to the C-S and K-S tests and is generally more sensitive to data at the tails of a distribution than the K-S test. The A-D test applies only to continuous distributions.	Normal Lognormal Weibull Exponential Logistic
Chi-Square (C-S)	Considers binned data and can be used to test both discrete and continuous distributions.	Normal Lognormal Exponential Logistic Binomial
Kolmogorov-Smirnov (K-S)	The K-S test tests the maximum distance between the sample and an expected curve. However, the K-S statistic applies only to continuous distributions, is most sensitive at the center of the distribution, and sample distribution characteristics (i.e., mean, standard deviation) must be defined independently of sample data.	Normal Lognormal Weibull Exponential Logistic

MCS sampling strategies are available and include random sampling, importance sampling, and Latin hypercube sampling (LHS). Random sampling, as the name implies, includes the sampling of values randomly from within a defined distribution. Importance sampling assigns different weights to low probability/high consequence input distributions to avoid large sample sizes. Finally, LHS is the generally preferred MCS sampling strategy for very complex models due to its efficiency in comparison with random and importance strategies (Helton et al., 2006; Sadiq, 2001). In this thesis, random sampling has been applied throughout.

Interpretation of the range in output values and their associated probabilities presented within resulting MCS probability distributions provides quantitative information regarding epistemic uncertainties. Sensitivity analysis completed via MCSs evaluates variance of model results due to the contributions of input probability distributions. While this type of analysis is somewhat limited by the number of input probability distributions which it can consider, it provides a suitable method for models without significant complexity (De Rocquigny et al., 2008). MCSs and sensitivity and uncertainty analyses can be completed by software programs, such as *Oracle ® Crystal Ball*, which was used within this thesis (Oracle, 2012).