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Assessing ecological condition of large landscapes with limited human impacts

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ASSESSING ECOLOGICAL CONDITION OF LARGE LANDSCAPES WITH LIMITED HUMAN IMPACTS

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Abstract: Our ability to measure global climate change has generated dire predictions in global ecosystem conditions. These predictions have inspired efforts to develop assessment metrics that examine alterations in ecological condition resulting from climate dynamics. As climate change drives future watershed- or regional-scale assessment model development, many questions will need to be addressed concerning potential tool constraints. Chief among these will be: to what degree is ecosystem condition affected by anthropogenic disturbance, climate-driven disturbance or natural variability, both individually and in combination? As a first step toward assessing impacts on ecological condition resulting from climate driven spatial or temporal disturbance gradients, an assessment methodology would need to be developed for ecosystems with limited direct human land use disturbance. In the first of three case studies, I propose such an assessment methodology for Glacier National Park (GNP), Montana. This approach combines theoretical elements of biological and ecosystem structural assessments with approaches developed for risk and landscape assessments to approach to assist GNP with prioritizing natural resource monitoring and management and with informing the public on the current condition of the park's ecosystem.

There has been increased accessibility to publicly available thematic maps derived from Landsat imagery that can be used to develop watershed or regional assessment tools in remote areas. Most remote sensing products have associated assessment of its error. However, the impacts of these uncertainties on landscape scale multi-metric management tools are poorly developed. In my second case study, I provide an approach that incorporates these errors into the assessment process.

Finally, dynamics of ecosystem are rarely incorporated into assessment tools as a means to distinguish natural variability from one perturbed by climate or anthropogenic disturbance. The Shifting Habitat Mosaic Concept addresses variability of floodplain habitat patch composition and provides a platform to develop potential assessment metrics for dynamics in floodplain habitat condition as climate shifts. In the third case study, I document the influence of multiple disturbance regimes across several geomorphic settings through a remotely sensed, multi-decadal whole-river census as a step towards developing effective metrics that measure perturbations in the variability of floodplain condition.

For my children Theodora and Elijah.
When I was your age, the U.S. Clean Water Act promised me water clean enough for fishing and swimming. I want to help deliver that promise to you.

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CHAPTER I: INTRODUCTION

Ecological monitoring and assessment is as old as the science of ecology as land, water, fish and wildlife resource managers applied ecological principles to their management needs. As the science gained theoretical and empirical advancement throughout the 20th century, assessment techniques and tools applied that knowledge to emergent environmental crises. After the establishment of a suite of United States environmental acts in the early 1970s, hundreds of qualitative and quantitative monitoring and assessment approaches have been developed predominantly to support resource managers at tribal, state, and federal regulatory agencies in the management of aquatic resources (Bartoldus 1999, Diaz et al. 2004, Fennessy et al. 2004, Böhringer and Jochem 2007a). Nationwide development and application of multi-metric indices have been encouraged by United States Environmental Protection Agency (USEPA), through its monitoring and assessment programs (USEPA 2006). As a consequence, multi-metric monitoring and assessment tools are well established within the United States and fairly well understood and accepted by resource and regulatory agencies.

In recent years advancements in remote sensing, geographic information systems, and computational power has increased our awareness of wide-spread environmental problems such as climate change resulting in an increase in the scale of management questions and solutions (Verdonschot 2000). Climate is one of the fundamental controls on ecological processes across the globe and recent climate change is causing unprecedented shifts in biodiversity (Staudinger et al. 2013), landscape pattern (Opdam and Wascher 2004), associated disturbance regimes (e.g., fire and flood (USEPA 2014a) and, importantly, ecosystem functions and services these attributes provide (Wrona et al. 2006, Grimm et al. 2013a, Nelson et al. 2013). These shifts in response to climate change are further amplified by anthropogenic stressors (e.g., environmental pollution, landscape fragmentation, and invasive species; Opdam and Wascher 2004, Grimm et al. 2013b, Staudt et al. 2013). As climate change and anthropogenic impacts continue and likely accelerate, there will be increasing impacts on ecosystem structure and function, and the ecological services to which humans and our complex economies depend. These various documented and accelerated changes have shifted biodiversity conservation efforts and natural resource management and policy approaches from climate change prevention to mitigation and adaptation (Stein et al. 2013). Effective ecosystem monitoring and assessment is essential to adaptive management efforts. However, as climate change drives future watershed- or regional-scale assessment model development, many questions will need to be addressed concerning potential tool constraints. Chief among these will be: to what degree is ecosystem condition affected by anthropogenic and climate-driven disturbances or simply natural variability, individually and in combination? This is, of course, a question that would take many careers to answer and well beyond the scope of a dissertation. However as a step toward addressing this central question, this dissertation will guided by three research questions: 1) how does one assess ecological condition in the absence of an anthropogenic disturbance gradient? 2) What is the effect of uncertainty endemic to remote sensing data on large scale assessments? 3) How can one distinguish natural dynamics from a system altered from climate-driven disturbances?

First, ecosystem responses to anthropogenic disturbance gradients are well established in the literature and monitoring and assessment tools have predominantly been developed to measure impacts across such gradients. To assess impacts on ecological condition resulting from spatial or temporal disturbance gradients dominated by climate change, an assessment methodology would need to be developed for ecosystems with limited direct human land use disturbance.

Second, providing an assessment approach that is well-accepted by managers requires a straightforward analysis of ecological data to facilitate for management application (Barbour et al. 1999). Assessment approaches in areas with limited human presence relies on remote sensing data; however these remote sensing products have known uncertainties and well documented errors, and the impacts of these uncertainties on landscape scale multi-metric management tools are poorly documented. The confidence that the end-user has in the assessment tool requires an understanding of the ramification of input error on assessment results. Third, ecosystem dynamics are well-studied in the literature, but poorly integrated into the field of assessment. In theory, the dynamics of ecosystem attributes are driven by natural disturbance dynamics thereby defining a natural range of variation of those attributes (Poff et al. 1997). Therefore, a perturbed system would be one whose dynamics extend beyond the natural range of variation. Determining the bounds of a system's range of variation may take tens or hundreds of years, if at all (Romme and Despain 1989). Although much work has been done on the effects of climate change on contemporary ecosystem dynamics in the scientific literature, little work has been done on creating assessment metrics that are applicable at a watershed scale (USEPA 2008).

1.1 Assessing Aquatic Ecosystems in Areas with Limited Land Use Impacts

Broad environmental problems necessitate an increase in the scale of monitoring and assessment of these problems (Verdonschot 2000). Over the last few decades three schools of aquatic assessment have increased the scope and scale of their approaches. Although their typology is not well established, they can be loosely categorized as 1) biological and structural assessment, 2) risk assessment, and 3) landscape assessment; all of which conduct some form of ecological assessment. The predominant goals of these ecological assessments have been to evaluate or predict the effects of human activities on natural resources and to provide analyses that can be translated into management actions. All assessment approaches require a blend of empirical data, best available science, and the judgments of experts to provide scientifically credible answers to policy-relevant questions (Grimm et al. 2013a).

Biological assessment began in the early 1900s, as a tool to measure the impacts of sewage on aquatic invertebrates (Kolkwitz and Marsson 1908). Originally these assessment approaches were mostly bio-chemical or uni-dimensional assessments of water quality (Cairns and Pratt 1993): however, these assessments were found to be inadequate as many human disturbances were found not to be restricted to being chemical or physical in origin (Verdonschot 2000). As human populations increased in the late 20th century, metrics that assess the condition of the structure of ecosystems was integrated with biological metrics to create tools that measured the multiple anthropogenic impacts such as changes in hydrological regime, sediment transport, habitat quality, and ecosystem function (Karr and Chu 1998, Barbour et al. 1999, Smith et al. 1995, Collins et al. 2008).

Biological and structural assessments have been developed throughout the world to provide politicians and decision makers with the ecological information on which to base their resource management decisions and communicate those decisions to the public (Turnhout et al. 2007, Dramstad 2009). This is often coupled to increased regulatory oversight (Holder and McGillivray 2007). Although many of the historical advancements of assessment are well documented in the scientific literature, much of its development and application occurred in management settings outside of academia (e.g., Adamus et al. 1987, Brinson et al. 1994, Smith et al. 1995, Hawkins et al. 2000, Wright et al. 2000, Hauer et al. 2002, Kleindl et al. 2009). Today, there are over 400 contemporary biological and structural assessment methods applied across a suite of environmental problems (Bartoldus 1999, Diaz et al. 2004, Fennessy et al. 2004, Böhringer and Jochem 2007a). The USEPA has organized these into a three-tiered approach to monitoring and assessment of aquatic resources. Level 1 assessment consists of habitat inventories and landscape-scale assessment, while Level 2 consists of rapid at-site assessment, and Level 3 consists of data-rich, often site-specific and generally intensive assessment (Kentula 2007). Recently, multi-metric index (MMI) tools, commonly developed for Level 2 rapid assessment, have been modified and applied to Level 1 approaches for watershed (Leibowitz et al. 1992, Abbruzzese and Leibowitz 1997, Brooks et al. 2004, Tiner 2004, Weller et al. 2007, Whigham et al. 2007, Meixler and Bain 2010, Rains et al. 2013), regionals (Reiss and Brown 2007, Collins et al. 2008), and compiled to provide continental-scale analysis (USEPA 2013). Level 1 landscape condition assessment tools have been developed and applied by local, state, and federal entities to address wetland programmatic efforts as well as broader efforts to inform conservation planning and prioritization efforts across large areas (e.g., Rains et al., 2013; Sutula et al., 2009; USDA, 2013; USEPA, 2014).

The increased regulatory oversight also inspired the development of tools to assess potential risks to the integrity of aquatic systems. Risk assessment in the United States grew in response to a series of environmental laws in the early 1970s (e.g., Clean Air Act of 1970, Federal Insecticide, Fungicide, and Rodenticide Act of 1972, Safe Drinking Water Act of 1974, Toxic Substances Control Act of 1976, and Clean Water Act of 1977; (Suter 2008). Risk assessment was well developed within the insurance industry as far back as the 1800s (Bernstein 1996) and uses analysis of past events, trends, mechanistic modeling, and professional judgment to estimate how proposed actions, events, and poorly defined trends will affect the future (Suter 2008). Mostly this assessment addressed the human health risk to identify potential chemical hazards, exposure to those hazards, and potential dose-response (NRC 1983). However in the 1980s, risk assessment expanded to address non-chemical stressors such as aquatic thermal regimes, sedimentation, and habitat loss. Thus began the field of ecological risk assessment and an increase in the spatial extent of its application (Landis and Wiegers 1997, Cormier and Suter 2008, Suter 2008, Schleier III et al. 2008).

Landscape assessment uses indicators designed to measure the extent and effects of anthropogenic land use impacts. These have been developed over the last several decades to provide qualitative descriptors or quantitative measures of landscape composition and configuration necessary to support system structure and function (Dale and Beyeler 2001, Bolliger et al. 2007). These indicators are used in many areas of research, resource management, policy development and decision making. Indicators intended for research fundamentally assess how pattern drives ecological processes and have been derived from fields such as geostatistics (Legendre and Fortin 1989), spectral and wavelet analysis (Natalie et al. 2003, Keitt and Urban 2005), and fractals and lacunarity (O'Neill et al. 1988, Plotnick et al. 1993). A subset of these measurements, such as patch diversity, dominance, size and aggregation, and parameter/area ratios, have been integrated into landscape assessment approaches (USEPA 1994). These measurements assess how pattern diverges from a reference state across the anthropogenic disturbance gradient (O'Neill et al. 1988, Riitters et al. 1995, Frohn 1997) following the long history of ecological assessment approaches to assist in resource management decisions.

Biological and structural assessment, risk assessment, and landscape assessment methods are beginning to approach the similar ecological problems albeit with their unique dogma. Lackey (1997) recognized that confusion and divisiveness occur as multiple assessment approaches are applied to similar ecological problems but conflate common ecological terms with their own unique definitions. No existing framework includes all types of environmental assessment approaches (Cormier and Suter 2008) but there is common ground to assist managers in navigating the multitude of assessment approaches (Stein et al. 2009a) to address management concerns or develop consensus among stakeholder values, goals, and priorities at all scales (Suter 2008).

This dissertation's first case study blends aspects of biological, structural, risk, and landscape approaches into a watershed-scale assessment of the ecological condition of areas with limited direct human disturbance. This assessment was developed for National Parks Service (NPS) for Glacier National Park (GNP), MT, to provide an analysis of ecological data in a straightforward manner to facilitate management applications, and communications with NPS, regulatory agencies, and park visitors (Kleindl et al. In Press). This dissertation chapter contains a digested version of that larger document and provides an assessment of four focal aspects important to the park: streams, large rivers, lakes, and salmonids. The assessment began with an evaluation of the spatial distribution of human activity within the park's watersheds; a contemporary assessment of spatial complexity of biotic and abiotic structural components associated with the four focal aspects; and the potential risk to some of these components resulting from changes in climate, cumulative air pollution, or exposure to invasive species. These are referred to as stressor, significance, and risk metrics, respectively. These assessments provide a range of watershed conditions within the limit of the park's boundaries as a prioritization tool to assist with finer scale monitoring or management decisions and to establish a baseline ecological assessment of Glacier National Park's watersheds for the purpose of monitoring future changes.

1.2 Incorporating Uncertainty into Large Scale Assessment

A gap exists between the science of ecology and the applied practice of management of ecological resources as they relate to the different goals of the two institutional cultures of academe and agencies (Turner et al. 2002). At large spatial scales, the science of ecology is concerned with the causes and ecological consequences of spatial pattern across landscapes (Turner et al. 2002). In contrast, a manager's goal is to interpret change for management action and to facilitate communication with stakeholders and policymakers (Barbour et al. 1999) and to maintain or alter natural resources to meet societal values (Turner et al. 2002). This gap has led to abundant criticism of index based approaches to assessment (Lackey 1997, Seegert 2000, Li and Wu 2004, Dramstad 2009, Green and Chapman 2011). As a regulatory, management, and communication tool, index based assessments exist in the difficult area between science and policy (Turnhout et al. 2007). Although many critiques of index-based assessment approaches are valid, not all can be implemented and still maintain the spirit of the tool from a regulatory perspective. For instance, assessment models need to be well calibrated (Seegert 2000), evaluated across environmental gradients (USEPA 2011), and users need to recognize that large amounts of information are lost when complexities of an ecosystem are summarized into one index value (May 1985, Green and Chapman 2011): however, approaches that may make an index scientifically robust may also make it less user friendly. Even if all the criticisms are accounted for and the best possible model is created, this does not determine its actual use in policy and management scenarios whose decisions may be more strategic than scientific (Turnhout et al. 2007). In an ideal process of ecological assessment tool development, the science team works with the policy and stakeholder team to create a model with clearly articulated objectives and limitations, and accounts for uncertainty in a manner that is easily understood by the end-user (Niemi and McDonald 2004, Turnhout et al. 2007).

Uncertainty can manifest in multi-metric assessment models at different locations in the development process (Walker et al. 2003, Refsgaard et al. 2007). Sources of uncertainty are associated with relationships between data inputs, defining the measurements (metrics) from the inputs, how these metrics relate to each other, and how they relate to outputs (Cressie et al. 2009). These include how the metrics are defined, the equations used, and assumptions that bound these models. The nature of these uncertainties can be both reducible epistemic uncertainty due to imperfect knowledge and non-reducible stochastic uncertainty due to inherent variability (Walker et al. 2003, Refsgaard et al. 2007). Tracking and reporting uncertainty is considered a best practice in most remote sensing efforts (Foody 2002). However, in a recent review of articles published in the journal Landscape Ecology between 2004 and 2008, 75% failed to provide assessment of uncertainty or error relating to image classification and mapping (Newton et al. 2009). Additionally, addressing classification accuracy and its influence on landscape indices has been largely ignored (Shao and Wu 2008). Equally, incorporating these known uncertainties into MMI tools in general is very limited (e.g., Fore et al., 1994, Whigham et al., 1999, Stein et al., 2009) and tend to be absent in the assessment implementation and reporting phase (Smith et al. 1995, Hauer et al. 2002, Klimas et al. 2004, Collins et al. 2008).

The challenge is to provide a pathway to incorporate known uncertainties from multiple data sources into an assessment tool used by planners, policy makers, lawyers, and scientists. In this dissertation's second case study, I address two questions as a step toward meeting this challenge: How sensitive is a landscape-scale multi-metric index to error from input data (specifically thematic land-cover misclassification)? What are the implications of this uncertainty to resource management decisions? I develop a simplified MMI with metrics derived from the 2006 National Land-cover Database thematic map (NLCD: MRLC 2013) to specifically address aspects of uncertainty that rise from a single source of data. I developed a multi-metric index that uses thematic Landsat data to provide an assessment of floodplain condition along 250 km of the Flathead River in northwestern Montana, USA. Typical of most multi-metric indices, our initial assessment does not account for misclassification errors within the thematic map and produces metric and index scores that are considered naive. I then provided an error simulation model to incorporate known map classification error into our multi-metric assessment tool by developing multiple potential map realizations based on classification probabilities and potential spatial correlations. I apply our MMI to each realization to establish a distribution of potential assessment scores and compare this distribution to the naive score to determine potential bias and the implications of that bias on management decisions.

1.3 Address Natural Variation in Large Scale Assessments

The science of ecology had high public profile in the late 1960s. Time Magazine called 1969 'The Year of Ecology' and Newsweek proclaimed 1970 as the 'Dawn of the Age of Ecology'. At the same period Eugene Odum published 'The Strategy of Ecosystem Development' (Odum 1969). In this paper, Odum argues that interaction of biotic and abiotic components brings an orderly evolution of ecosystems into a state of equilibrium and that repeated perturbations may not allow that system to reach a mature state of equilibrium. Odum was a very prominent and outspoken ecologist and in this time of increased awareness of the science, his and other similar contemporary concepts of equilibrium and disturbance had some influence on the formulation of the 1969 U.S. National Environmental Policy Act (NEPA) (Bosselman and Tarlock 1993). The NEPA and subsequent state environmental policy acts require that proposed projects funded by federal, state and local agencies assess potential direct, indirect, and cumulative environmental impacts. The NEPA, at the time of its conception, was intended to synthesize 'ecological science in action' and protect the 'balance-of-nature' by maintaining its internal resistance to change (Holder and McGillivray 2007). Since 1970, theoretical advances have provided new conceptual frameworks that define disturbance (White and Pickett 1985, Resh et al. 1988), address the extent and limit of ecosystem resilience to disturbance (Folke et al. 2004), define conditions where levels of disturbance may define or alter system equilibrium (Bormann and Likens 1979, Turner et al. 1993), or the potential of an ecosystem existing in permanent state of disequilibrium (Botkin 1990, Mori 2011). In spite of this, federal code has remained relatively static and for over forty years, the NEPA and other environmental protection regulations have required an assessment of the ramifications of human disturbance on ecosystem stability. This gap between ecological theory and application to environmental regulations is problematic (Suter 1981, Emery and Mattson 1986, Orians 1986).

Central to all regulatory monitoring and assessment tools is a means to measure departure from an expected condition of a stable, healthy ecosystem. The idea of ecological balance has long been explored in ecology (Cooper 1913, Clements 1916, Watt 1947, Whittaker 1953, Odum 1969, Bormann and Likens 1979, DeAngelis and Waterhouse 1987, Turner et al. 1993, Phillips 2004, Mori 2011, and many others) but over the last decade, three interrelated concepts have provided new insights into ecosystem equilibrium of floodplain systems.

<u>The Shifting Habitat Mosaic</u> (SHM: Stanford et al. 2005), was developed in recognition that the physical processes of cut-and-fill alluviation form a patchwork of geomorphic surfaces of different physical structure, age, and successional state within flood prone areas. The relative abundance of major floodplain abiotic and biotic habitat features (e.g., depositional bars, back channels, herb, shrub, and forest patches) in dynamic, free-flowing river systems remains relatively stable even as the mosaic of habitats changes in space over time (Arscott et al. 2002).

<u>The Range-of-Variation or Statistical Equilibrium</u> (Poff et al. 1997, Landres et al. 1999, White et al. 1999, Arscott et al. 2002) states that in unimpacted systems, the relative abundance of major floodplain biotic and abiotic habitat features exists in a limited dynamic range, but if disturbance frequency, magnitude, timing, or duration falls outside the parameters which the system has adapted to, then the system may shift out of that dynamic range and either become unstable or move through a change-in-state to a new quasi equilibrium (Poff et al. 1997).

<u>The Stable Trajectory Equilibrium</u> (White et al. 1999, Whited et al. 2007) states that elements of the disturbance regime change across time with dynamic climatic conditions and the range of variation that defines the SHM also trends with this change.

Thus floodplain conditions are shaped by a watershed's disturbance regime. In theory, describing the dynamic range of disturbance attributes is paramount in evaluation of ecological drivers of change and the construction of metrics and weighting parameters for regulatory assessment tools. Disturbance attributes are described in terms of spatial characteristics (extent, shape, and spatial distribution), temporal characteristics (frequency, and return interval), specificity (to species, size class, and successional state), magnitude (force, intensity, and severity) and synergisms (interactions among disturbances) (White et al. 1999). These disturbance attributes vary with climate, topography, substrate, and history and collectively define the disturbance regime. Variations in the disturbance regime produce a continuum of conditions (White et al. 2000). Intermediate disturbances maintain stream diversity (Connell 1978, Ward and Stanford 1983), but as the disturbance trends toward the extreme low or high range then biotic and abiotic diversity can become compromised (Poff et al. 1997).

From an assessment perspective, the concept of stable states may not be relevant. There is an alternative view that suggests it is impossible to define a natural disturbance regime and that a non-equilibrium regime exists (Mori 2011). Because of the variance and occasionally the vagaries of climate, there is no appropriate time period or spatial range that can be defined as the reference equilibrium state (Landres et al. 1999), and historical climate-driven disturbance can have legacy affects that inhibit the development of a true equilibrium state (Foster et al. 1998). Mori (2011) suggests that ecosystem management should recognize the non-equilibrium nature of ecosystems and landscapes and take into consideration unpredictability, instability and stochasticity as these lead to inevitable ecosystem changes. Mori (2011) further argues that studies of ecosystem resilience in non-equilibrium conditions are of paramount importance to ecosystem management and protection.

The increasing threat from climate change to ecosystem functions and services that support well-being of humanity (Boyd and Banzhaf 2007, Nelson et al. 2013) is beginning to drive assessment-tool development (Feld et al. 2009, Paetzold et al. 2010, Rounsevell et al. 2010), especially for remote areas with limited direct human disturbance elements. As Mori (2011) implies, it is the threat to the resilience of ecosystem structure and functions that support ecosystem services important to humans that will likely drive ecosystem management and protection (Carpenter et al. 2006). In this dissertation's third case study, I examine floodplain ecosystem dynamics using data from the early 1980s to 2013. This period marked a sharp rise in remote sensing products with the launch of Landsat Thematic mapper in 1984, climatic datasets such as Daymet beginning in 1980 (Thornton et al. 2012), and publicly available orthorectified imagery beginning in 1991 (USGS 2014a). Herein, I approach the development of assessment measures of floodplain dynamics in areas with limited direct human disturbance by assessing of contemporary and potential future variation in ecosystem attributes that diverge from a "resilience reference state" as defined by the conditions in the last decades of the Twentieth Century.

However, before these metrics can be developed, the dynamics of the floodplain and the disturbance regimes that drive them need to be quantified. Here I re-examine the Shifting Habitat Mosaic (SHM) concept of floodplain habitat patches which suggests that dynamics in space and time are influenced by hydrological disturbance driven by flood or flow pulses of sufficient power to initiate and maintain cut and fill alluviation and periodic avulsion of the channel and banks. However, floodplains are transitional zones between riverine and upland ecosystems and are subject to transitions of import restructuring from floodplain land use requiring an extension of SHM concept to capture the effects of the blending of hydrological and terrestrial disturbances on floodplain habitat patch composition. To examine the SHM, I investigated hierarchical relationships between hydrology, fire, anthropogenic disturbance, geomorphic position and floodplain habitat patch dynamics across space and time to test for factors that influence disturbance, disturbance/recovery pathways, and dynamic stability. I used graphical analysis to examine the locations and intensity of disturbance and recovery pathways of across floodplain transition zone throughout the 22 years which support the hypothesis that a blending of disturbance regimes and the resulting recovery pathways maintains the SHM across the floodplain area of this system.

1.4 Summary of Dissertation Chapter:

This dissertation is divided into 4 chapters. In this chapter (Chapter I), I have provided a conceptual foundation through a historical perspective of ecological assessment, the ecological theory that supports assessment, and regulatory directives that mandate its use.

In Chapter II, I present an assessment of the ecological condition of Glacier National Park as a case study of an assessment application in areas with limited human impacts. This chapter is a condensed version of a larger document developed for National Park Service (Kleindl et al. In Press). Herein it consists of 5 sections and an appendix. These sections cover an introduction to this effort (Section 1), an overview of the park (Section 2), a summary of methods used (Section 3), four individual multi-metric watershed-scale assessment models for the park's streams, lakes, large rivers, and salmonids (Section 4), and finally the results are presented within the context of the park's preexisting management boundaries to blend the model results with the park's management priorities (Section 5).

In Chapter III, I address large landscape assessment model uncertainty resulting from input error associated with map misclassifications that are endemic in remote sensing thematic products. The section modifies remote sensing error simulation models for application with multi-metric indices to examine potential bias of the metrics and compounded bias of the index. These results are placed into a management context. This section is a standalone manuscript (Kleindl et al., *In Review-a*).

In Chapter IV, I examine the role of terrestrial and aquatic disturbance regimes on floodplain habitat patch dynamics and extend the Shifting Habitat Mosaic concept of floodplain systems to include multiple disturbance vectors that operate at different temporal and spatial scales. This effort also approaches an establishment of reference conditions of a dynamic system which can be used to predict or monitor changes in ecological diversity as systems dynamics change from regional or global disturbances. This section is also a standalone manuscript (Kleindl et al., *In Review-b*).

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CHAPTER II: A MULTI-METRIC WATERSHED CONDITION MODEL FOR GLACIER NATIONAL PARK

1 Introduction

In response to increasing threats to the biological integrity of national parks, the U.S. Congress passed legislation in 2003 that instructed the National Park Service to assess environmental conditions in watersheds where park units are located. As a result of this legislation, the Water Resources Division of the National Park Service initiated a multi-year program to fund natural resource condition assessments for each of the 270 park units with significant natural resources. These natural resource condition assessments are intended to synthesize existing research and inventory and monitoring data into a knowledge base for use in park resource planning, decision making, monitoring prioritizations, accountability reporting, and partnership and education efforts. The assessments should provide a spatially explicit, multi-disciplinary synthesis of existing scientific data and knowledge, from multiple sources, that helps answer the question: What are the current conditions for important park natural resources? It is the intention of this chapter to provide an assessment of the current and potential future natural resource conditions to ultimately assist in prioritization of natural resources management actions and associated monitoring to address these conditions. Therefore it is a goal of this chapter, and associated GIS tools, to blend smoothly with existing management frameworks.

The park published a General Management Plan (GMP) in 1999 with the intention of influencing decisions for the following 20 years or more (Layman 1999). In that plan, the park articulates their overall management approach:

> The overall guiding philosophy is to manage most of the park for its wild character and for the integrity of Glacier's unique natural heritage, while traditional visitor services and facilities remain. Visitors would be able to enjoy the park from many vantage points. Visitor use would be managed to preserve resources, but a broad range of opportunities would be provided for people to experience, understand, study and enjoy the park. Cooperation with park neighbors would be emphasized in managing use and resources (Layman 1999).

Specifically, natural resources are managed in accordance with NPS policy "to understand natural processes and human-induced effects; mitigate potential and realized effects; monitor ongoing and future trends; protect existing natural organisms, species populations, communities, systems, and processes; and interpret these organisms, systems, and processes to the park visitor" (Layman 1999). The multi-metric condition assessment models in the following section were developed to assist with this NPS management guidance by keeping three primary audiences in mind: decision makers such as park superintendents, resource managers at the park, and scientists and technicians engaged to assist parks (e.g., Inventory and Monitoring Network ecologists and data managers). The assessment findings are designed to assist and inform these audiences for, among other things:

• Near-term strategic planning, to allocate limited staff and budget resources toward high priority (relatively more significant or vulnerable) park-managed watersheds and habitats;

• General Management Plan and Resource Stewardship Strategy development, which represent the planning process that formalizes park management zones, Desired Condition management objectives, and associated measurement indicators and targets;

• Park reporting to the Department of Interior's "land health goals" and to an Office of Management and Budget "resource condition scorecard";

• Park efforts to communicate and partner with other stakeholders, in order to address watershed- or landscape-scale resource management issues.

1.0 Chapter Goals

The specific objectives of this project were:

1. To provide park superintendents and managers with initial, science-based judgments about resource condition status of each watershed relative to other watersheds within the park, and to provide data, information, and recommendations that will be useful to park managers in their work to define the park's management zones and desired conditions.

2. To provide assessment statistics and summaries to allow park superintendents and managers to develop reports that meet Government Performance and Results Act and Office of Management and Budget reporting requirements.

3. To develop an assessment framework and process that can be repeated in the future and can serve as a template for resource assessments at other park units.

A main sign of success of this report will be the extent to which it provides park resource managers data and information that help them to see "the big picture" and relationships among critical issues, and to help place emerging issues within a local, regional, national, or global context.

1.1 Section Overview

The sections are organized as follows:

- Section 2 – Park Overview is a general history and description of the park and surrounding areas helpful for those not familiar with the park and its regional context.
- Section 3 –Describes the philosophical foundations of watershed-scale multi-metric assessment approaches and this unique application in an area with very limited human disturbance. The section also introduces general methods used to develop multi-metric indices that form the bases of this project and caveats that bound the application of these models.
- Section 4 –Details the four watershed-scale multi-metric indices use to assess condition of the park's ecological focal areas of concern. This section includes specific methods and results for each metric and assessment index.
- Section 5 – Provides these results in the context of the park's pre-existing management boundaries to blend the results for management prioritization.
- Appendix – Appendix A provides overview of the geographic information systems (GIS) models used in the analysis.

Because the assessment is broad and integrative, a strong emphasis was placed on conducting spatially-explicit analyses using GIS techniques. As a consequence, I developed numerous maps and visualizations of indicators and findings in this report, including a technical appendix, as well as a full suite of GIS datasets.

1.2 What This Chapter Does Provide:

This chapter provides a baseline ecological assessment of Glacier National Park's (GNP) HUC 10 watersheds from the perspective of four major focal areas selected in collaboration with the GNP resource management team:

- Streams: The condition of alpine, mid-elevation and lowland streams within the park's watersheds.
- Large Rivers: The condition of the North and Middle Forks of the Flathead River that form the western boundary to the park.
- Lakes: Potential risks to the condition of the park's lakes.
- Fish: The condition of native and non-native salmonid species within the park's watersheds.
- Mammals, birds, and vegetation are included in the complete document provided to the park.

This chapter provides Glacier National Park's resource management team with a watershed-scale assessment tool to define ecological condition within the park. The ecological condition of a watershed within the park is determined by a combination of metrics that measure ecological diversity, referred to as 'significance metrics', as they interact with metrics that measure the human threats referred to as 'stressor metrics', or risk of threats, referred to as 'risk metrics'. Collectively the study evaluates a subset of the biotic and abiotic structural components that describe ecological condition in areas of limited anthropogenic disturbance (See Section 2 for more details).

These assessment models provide metrics that assess the ecological significance, anthropogenic disturbance, and risk of future degradation of a park's watershed only relative to other watersheds within the park. Because the park itself is in outstanding condition relative to other watersheds and mountain ranges within the Rocky Mountains, this model was designed to provide a scaled index of ecological condition within GNP only (or those immediately adjacent to the park in the case of the North and Middle Forks of the Flathead River).

1.3 What This Chapter Does Not Provide:

This chapter uses existing data provided by the park. These data were extensively analyzed, but no additional data were collected. Where there were gaps in data and knowledge, I used expert opinion based on ecological theory to develop the models and in some cases to score or evaluate sub-index scores for indicators. This combination of both a quantitative and qualitative approach to metric development is common in multi-metric indices for management tools intended to provide scientists, managers and decisions makers with a prioritization approach for their various disciplines. *These indices and metrics are intended to indicate a range in the quality or "condition" of the system and its attributes not as a true measure of ecosystem complexity or cause-and-effect pathways.*

As an index that provides a single score that relates to the quality of the ecosystem, the multi-metric approach used herein is a simplification of ecosystem complexities as they integrate across multiple attributes and across wide topographic, aspect and distribution ranges. Thus, these index scores are intended as 'pointers' to areas of concern. The metrics within these indices provide finer detail of the potential drivers of ecosystem structure and function reflected in the index scores. *As a result, metrics and indices that make up this management tool do not provide* *information on actual quantitative thresholds beyond which ecological resilience is compromised.*

Additionally, there are disturbance vectors such as climate change and air pollution that impact the entire park in a variety of ways. These broad impacts are addressed in the expanded document provided to the park, but due to the complexity of these impacts across the varied topography of the park and the limits of existing data they were not integrated into all of the assessment indices. Rather, they were restricted to models that assess alpine areas and lakes. The human impacts metrics within the majority of the assessment models measure the extent of direct human land use impacts such as roads, trails, and facilities.

Finally, the following sections offer a watershed-scale, multi-metric assessment that focuses on the condition of biotic and abiotic ecosystem attributes within the park. The assessment addresses ranges in condition of natural attributes such as the amount of alpine community per watershed and the ranges of disturbance attributes such as the amount of trails and roads per watershed. However, there are disturbances in and around the park that occur at a scale larger than is appropriate for a multi-metric assessment approach, but do have an influence on the park conditions. Users of this section and the multi-metric indices herein will need to take into account this broad array of externalities to assess potential effects to various sub-indices and thereby impacts to the park.

2 Park Overview

Far away in northwestern Montana, hidden from view by clustering mountain peaks, lies an unmapped corner – the Crown of the Continent – slow-moving ice rivers still plow their deliberate ways, relics of mightier glaciers, the stiffened streams which in a past age fashioned the majestic scenery of today. ~ George Bird Grinnell, Century Magazine, September 1901

Glacier National Park in northwestern Montana, created by act of Congress in 1910, holds the geographic headwaters of a significant portion of the North American Continent. Within Glacier National Park resides the single spire, Triple Divide Peak, where three river systems of the continent converge at the intersection of the Continental and Hudson Divides. Water flowing to the west enters the Columbia River Basin (Pacific Ocean), waters flowing to the northeast flow into the Saskatchewan River Basin (Hudson Bay, Artic Ocean), and water flowing southeast enters the Missouri River Basin (Gulf of Mexico, Atlantic Ocean). Thus, the montane landscape and its headwaters quite literally form the water tower of the continent. The region containing Glacier National Park has been referred to as the Northern Continental Divide Ecosystem (Salwasser et al. 1987), the Northern Rocky Mountain Province (Bailey 1980), and the Crown of the Continent Ecosystem (Selkowitz et al. 2002). Although the first two names are most commonly used in scientific literature, they disregard the substantial portion of the contiguous montane system. Glacier National Park and its sister park in Canada, Waterton Lakes National Park, form the heart of the Crown of the Continent, which is more inclusive and representative of the importance of the region and is by far the earliest title given recognizing the regional hydrologic and geographic uniqueness of GNP and appeared in an article written by George Bird (Grinnell 1901) describing his travels in the region. Glacier National Park is characterized by high heterogeneity of watersheds and hydrology. To the east is the steppe of the Great Plains and the Rocky Mountain Front. Interior to GNP and to its west are the belt series mountain ranges dominated by sedimentary geologic formations of mountains and valleys with change in elevation exceeding 6000 feet [1]between the valley floors and along the mountain peaks.

In 1932, Glacier National Park and Waterton Lakes National Park in Canada were designated as Waterton-Glacier International Peace Park; the world's first of now many international peace parks distributed worldwide along international boundaries. Waterton-Glacier International Peace Park holds a United Nations designation as an International Biosphere Reserve and World Heritage Site. Central to this designation is the role of biodiversity and the quality and quantity of water as it interacts with the mountain-valley landscape. Indeed, the distribution and abundance of biota and the way people use the landscape are closely interconnected to the region's headwaters. Some of the best evidence for climatic change globally is found here. The glaciers of GNP have been shrinking rapidly since the founding of the park in 1910. A recent analysis estimated an ≈ 40% reduction in glacier volume since 1950 and simulation modeling has projected that the glaciers of GNP will be gone by 2050 (Hall and Fagre 2003). This and future changes will have a significant effect on headwater hydrologic regimes and the organisms that are dependent on continuity of flow in alpine running water habitats (Hauer et al. 1997).

The region around Glacier National Park is experiencing rapid growth in human population, particularly in the Flathead River Basin. Natural wildness, recreation and scenic attributes, epitomized by Glacier National Park, are the long-term primary drivers of economic growth for

[1] English units are preferred by the management team at GNP and are used throughout this chapter.

the region. Water quality, the support of aquatic organisms, and the integrity of aquatic and floodplain habitats are essential to maintaining the renewable goods and services that characterize the quality-of-life enjoyed by residents and visitors from around the world. Glacier National Park is critically important to global biodiversity. Indeed, GNP holds one of the highest accumulations of diversity of plants and animals in North America (Hauer et al. 2007), including the full array of native carnivores and ungulates. For example, valley bottoms and the river floodplains of GNP are critical habitat for most of the large animals of the ecoregion, including several species listed as sensitive, threatened or endangered, including bull trout, westslope cutthroat trout, grizzly bear, lynx, and wolverine.

2.1 Park Resource Setting/Stewardship Context

The following summary of the park and its management has been accumulated from information provided in several internal GNP management documents (Layman, 1999; NPS, 2004).

2.1.1 Background

Glacier National Park is located on the Canadian border in the northwestern section of Montana. The park is in the northern Rockies, and contains the rugged mountains of the Continental Divide. Together with Canada's Waterton Lakes National Park, it forms the Waterton-Glacier International Peace Park, a World Heritage Site (Figure 2.1). Glacier National Park's primary mission is the preservation of natural and cultural resources, ensuring that current and future generations have the opportunity to experience, enjoy, and understand the legacy of Waterton-Glacier International Peace Park. The purpose of Glacier National Park is distilled to three points:

- Preserve and protect natural and cultural resources unimpaired for future generations (1916 Organic Act);
- Provide opportunities to experience, understand, appreciate, and enjoy Glacier National Park consistent with the preservation of resources in a state of nature (1910 legislation establishing Glacier National Park); and
- Celebrate the on-going peace, friendship, and goodwill among nations, recognizing the need for cooperation in a world of shared resources (1932 International Peace Park legislation).

The park's distinctive qualities make it a significant resource regionally, nationally, and internationally (following bullets from NPS 2004).

- Glacier's scenery dramatically illustrates an exceptionally long geological history and the many geological processes associated with mountain building and glaciation;
    - Glacier has the finest assemblage of alpine glacial features in the contiguous 48 states, and it has relatively accessible, small active glaciers.

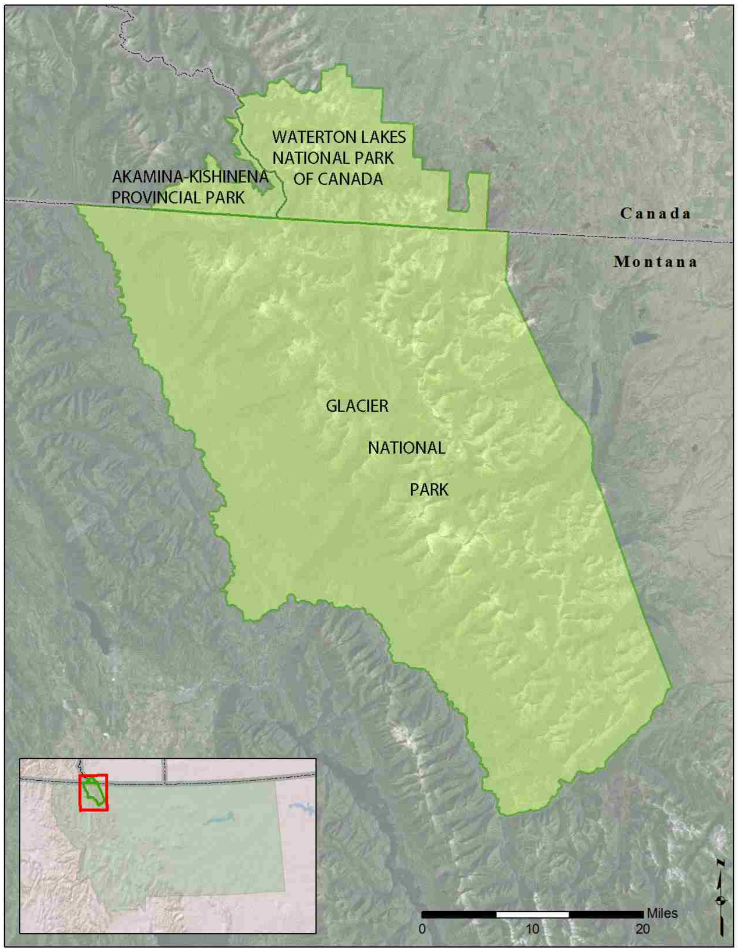

Figure 2.1. Location of Glacier National Park.

- Glacier provides an opportunity to see evidence of one of the largest and most visible overthrust faults in North America, exposing well-preserved Precambrian sedimentary rock formations.
- Glacier is at an apex of the continent and one of the few places in the world that has a triple divide. Water flows to Hudson Bay, and the Atlantic and Pacific Oceans.

▪ Glacier offers relatively accessible spectacular scenery and increasingly rare primitive wilderness experience;

- The Going-to-the-Sun Road, one of the most scenic roads in North America, is a National Historic Landmark.
- Glacier offers a challenging primitive wilderness experience and opportunities to listen to natural sounds.

▪ Glacier is at the core of the "Crown of the Continent" ecosystem, one of the most ecologically intact areas remaining in the temperate regions of the world;

- Due to wide variations in elevation, climate, and soil, four Floristic Provinces connect in Glacier and have produced diverse habitats that sustain plant and animal populations, including threatened and endangered, rare, and sensitive species.

    - Glacier is one of the few places in the contiguous 48 states that continue to support natural populations of all indigenous carnivores and most of their prey species.
    - Glacier provides an outstanding opportunity for ecological management and research in one of the largest areas where natural processes predominant. As a result, Waterton-Glacier International Peace Park has been designated as a world heritage site, and both parks have been designated as biosphere reserves.
- Glacier's cultural resources chronicle the history of human activities (prehistoric people, American Indians, early explorers, railroad development, and modern use and visitation) show that people have long placed high value on the area's natural features.
    - American Indians had a strong spiritual connection with the area long before its designation as a national park. From prehistoric times to the present American Indians have identified places in the area as important to their heritage.
    - The park's roads, chalets, and hotels symbolize early 20th century western park experiences. These historic structures are still in use today.
    - The majestic landscape has a spiritual value for all human beings - a place to nurture, replenish, and restore oneself.
- Waterton-Glacier is the world's first international peace park.
    - People of the world can be inspired by the cooperative management of natural and cultural resources that are shared by Canada and the United States.
    - Glacier National Park and Waterton Lakes National Park offer an opportunity for both countries to cooperate peacefully to resolve controversial natural resource issues that transcend international boundaries.

Glacier National Park is a cherished natural legacy to the American people and to other people throughout the world. The park provides unique experiences in the natural world and contains superb examples of pristine natural resources. However, Glacier National Park was rated the most threatened national park and natural area in the 1980 State of the Parks Report to Congress (NPS 2004). Surrounding land use, invasion of non-native species, air quality, changes in climate, international land management inconsistencies, inventory data gaps, funding, and visitor usage are cited as some of the main threats to Glacier National Park when it was placed on the National Park Conservation Association's Ten Most Endangered Parks list (NPCA 2011).

2.1.2 Description and Characterization of Park Natural Resources

The ecological communities of Glacier National Park are distinctly influenced by its location along the main range of the Rocky Mountains, and its geological history. Both of these factors drive climatic environments that dictate the establishment of vegetation types, producing patterns across the landscape that are remarkably predictable given variables such as elevation, aspect, slope and substrate.

Glacier is primarily a mountain park, with two north-south mountain formations, the Livingston and Lewis Ranges, making up most of the terrain (Figure 2.2). Uplifted geologic formations of the Belt Series (primarily) are the foundation of these ranges, mostly composed of sedimentary rock. Subsequent glacial action has carved and molded the deposits into sheer cliffs, broad cirques, hanging valleys and moraines.

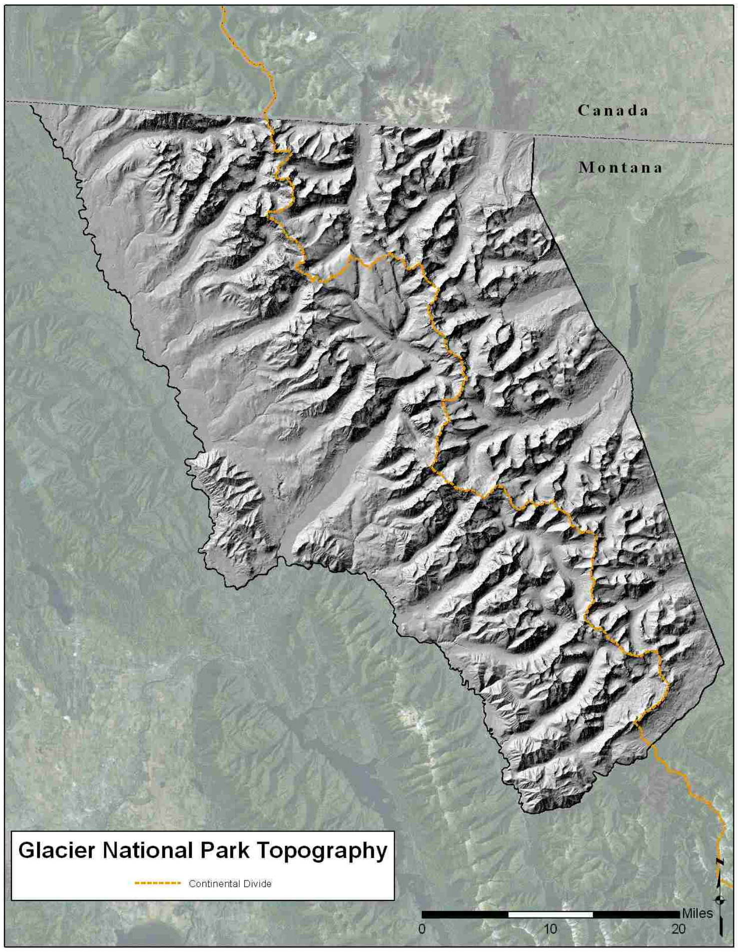

Figure 2.2. Topography of Glacier National Park.

The park lies midway along the north-south gradient of the Rocky Mountains, and species from four major floristic provinces converge here: the Cordilleran Floristic Province including the predominant Rocky Mountain subprovinces, as well as the Cascade Mountains subprovince with flora typical of the Pacific Northwest; the Great Plains Floristic Province represented on the eastern margins of the park; the Boreal Floristic Province with southern limits in the park; and the Arctic-alpine Floristic Province found above tree line (Lesica 2002). Also, the park is affected by two major climatic systems. The weather is alternately dominated by moist Pacific maritime and dry continental air masses, a mix that yields a broad range of temperatures, precipitation, and wind conditions. Add to that the extraordinary amount of topographic relief created by Pleistocene glaciers and ice sheets—a terrain so rugged that any given elevation offers an unusually broad range of exposures, soil conditions, moisture levels, and snow depths—in short, a multitude of microhabitats. Finally, the presence of both calcareous (calcium-rich, derived from limestone) and non-calcareous soils adds to the array of living spaces (McClelland 1970, Edwards 1957, Lesica 1996). The park has been termed a "continental biodiversity node," in other words, a natural mixing zone for biota of continental significance. The Continental Divide winds its way roughly through the center of the park, from the north boundary toward the southeast. On either side of the Divide alpine cushion plants are able to establish on sheltered sites with adequate moisture. Moving lower, where the climate becomes less physically harsh, alpine meadows develop and form a mosaic with shrubby krummholz vegetation. Lower still, subalpine woodlands develop on mountain slopes, becoming denser toward the bottom of deeply carved mountain valleys. Forests are replaced by shrubland and grassland vegetation along part of the park's western boundary, where soil is fertile and well-developed. On the park's eastern edge, coniferous forest is replaced by a mixture of aspen woodlands and grasslands (Figure 2.3).

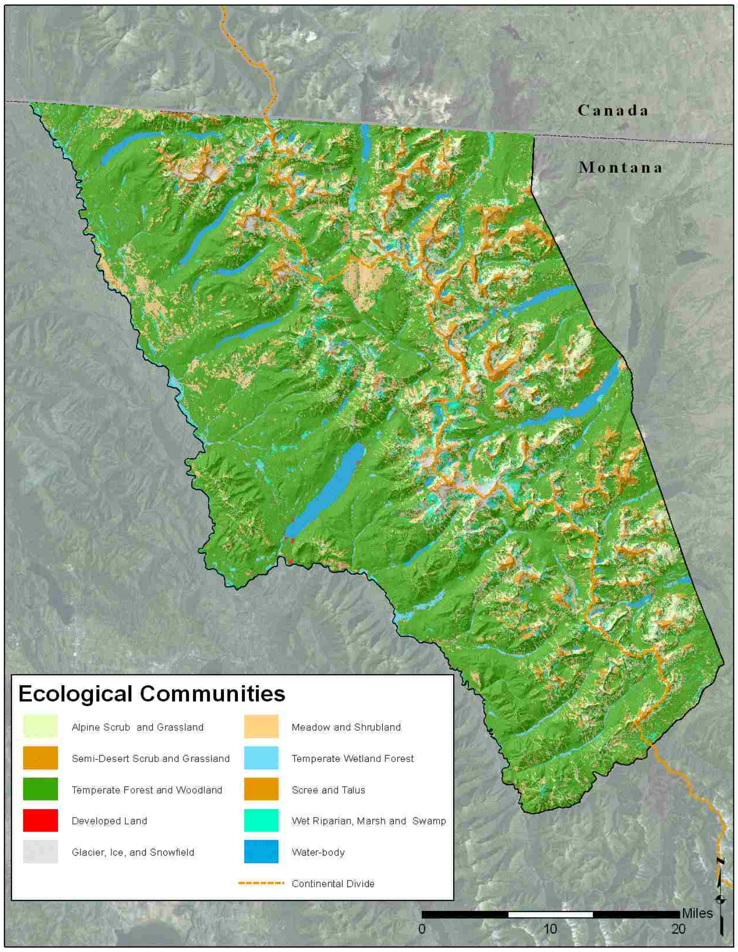

Figure 2.3. Ecological communities of Glacier National Park

3 Multi-Metric Assessment Approach and Background

3.1 Glacier National Park Management Framework

It is a goal of this section, and associated GIS tools, to blend smoothly with existing GNP management frameworks. The park's General Management Plan (GMP) presents a strategy to guide future decisions based on six geographic management areas. Each of these areas is made up of management zones. These areas and the zones within have different management priorities based on the land and visitor uses that are appropriate to the development and activities are described for those zones. The six geographic areas include; 1) Many Glacier, 2) Goat Haunt-Belly River, 3) Going-to-the-Sun Road Corridor, 4) Two Medicine, 5) Middle Fork, and 6) North Fork (Table 3.1 and Figure 3.1). The geographic areas vary in the amount of infrastructure and visitor access and as a result vary in the intensity of park management. Additionally, each area is made up of four types of management zones that guide specific management approaches (Figure 3.2). These zones are the: 1) Visitor Service Zone, 2) the Day Use Zone, 3) Rustic Zone, and 4) Backcountry Zone. The Visitor Service and Rustic Zones are currently used in park planning, while the Day Use and Backcountry zones are in working draft. The Visitor Service Zone includes developed areas, paved roads, and campgrounds with potable water and sanitation facilities. The Rustic Zone will include primitive facilities and campgrounds representative of early western national park development and traditional visitor experiences in them. The Day Use Zone, currently in working draft, includes selected areas generally with specific destinations that visitors can reach easily within a day from visitor use zones. Finally the Backcountry Zone, also currently in working draft, is an area where natural resource management is focused on protection and (when necessary) restoration of resources and natural processes.

Table 3.1. Areas of General Management Zones within the Management Areas

General Management Areas	Area (Acres)	Visitor Services (acres)	Rustic Zone (acres)	Backcountry (acres)*	Day Use (acres)*
Goat Haunt	165,472	1,202	0	163,467	1,216
Going-to-the-Sun Road	183,855	15,017	26	165,386	5,205
Many Glacier	65,935	1,908	0	60,651	4,455
Middle Fork	225,769	195	0	225,577	0
North Fork	286,111	18	571	285,259	0
Two Medicine	80,830	533	17	78,459	2,442
Total	1,007,972	18,873	614	978,799	13,318

*These management zones are preliminary and areas are estimated.

Figure 3.1. General management areas within Glacier National Park.

Because most natural processes are predominantly bound by watersheds, this section will present the analyses and summaries in a watershed context rather than the existing management zone context. For this approach I summarized each indicator at the Hydrologic Unit Code, level 10 (HUC 10) - for consistency and comparability. The HUC-10 watershed scale was selected over the finer scale HUC-12 during initial meetings with GNP resource staff. The finer scale HUC-12 watershed assignments would create approximately 60 assessment watersheds. The spatial analysis tools in the following section have been created and delivered to the park's GIS team for this project and can be applied to these 60 HUC-12 watersheds with minor modifications. For example, elevation specific metrics should not be applied in sub-basins that do not include that elevation range. However, the park's resource team expressed an interest in reporting only on the condition of the HUC-10 assessment watersheds for this effort. Therefore, the HUC 10 watersheds used for the remainder of the section, with slight modifications, resulting in 13 different assessment watersheds providing a finer resolution than the existing the management zones and can be used in unison as the management needs arise (Table 3.2 and Figure 3.3).

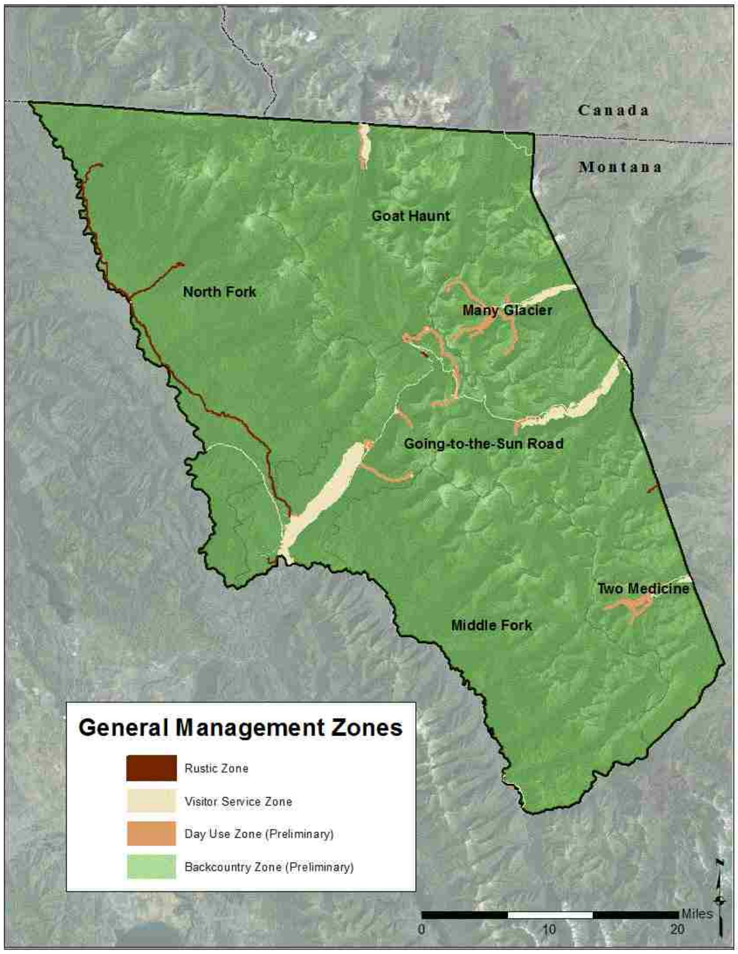

Figure 3.2. General management zones within the management areas of Glacier National Park.

Table 3.2. HUC-10 Watershed Assessments Areas

Hydrologic Unit Code - Level 10 Watershed Name	Watershed Area (Acres)
Belly	57,109
Camas	66,103
Coal/Ole	123,833
Cut Bank	27,283
Kennedy	26,608
Kintla/Bowman	132,186
Lake McDonald	119,523
Nyack	105,842
Quartz/Logging	87,828
Saint Mary	93,953
Swiftcurrent	53,780
Upper Two Medicine	53,776
Waterton	60,434
GNP Total	1,008,256

Figure 3.3. Watershed assessment areas used for this study.

3.2 Analysis of Condition

This section provides a baseline assessment of the ecosystem condition of Glacier National Park's watersheds, specifically, the condition of a subset of biotic and abiotic structures necessary to maintain the health of specific ecological focal areas. This watershed assessment focuses on four major ecological focal areas that were identified during initial meetings with GNP resource staff as important ecological components to the GNP monitoring and assessment program. The four focal areas include: 1) GNP stream systems, 2) North and Middle Fork of the Flathead River, 3) lake systems, 4) fish populations. These focal Elements, summarized in Table 3.3, are further described in Section 5. The condition of each focal element and the average ecological condition across all focal elements are provided in Section 6.

To provide GNP resource personnel with a prioritization tool for future monitoring and management, an approach to distinguish areas of higher and lower ecological significance in a landscape with very limited human impacts was necessary. Most ecosystem function assessments (e.g., HGM: Smith et al. 1995), integrity assessments (e.g., IBI: Karr and Chu 1998), or condition assessments (CRAM: Collins et al. 2006) commonly adopt a Reference Condition Approach (RCA: Bailey et al. 2007) where a site-of-interest is compared to a gradient of similar ecosystems that range from relatively unexposed to severely altered by stressors. If such a gradient were applied to GNP, nearly all sites would exist at the non-impacted range of an RCA. An assessment that provides a result in which all sites are in near-perfect condition would be uninformative to GNP's resource managers. As detailed in the section below, a new approach to assess a range of ecological conditions in an ecologically intact system was developed for this project to assist resource staff in monitoring and management prioritization.

Table 3.3. Description of the Assessment Focal Elements.

Focal Assessment Elements	Overview of Assessments
1. Steams	The condition of alpine, mid-elevation and lowland streams, and rivers, within the park's watersheds
2. Flathead River	The condition of the North and Middle Forks of the Flathead River adjacent to the park
3. Lakes	The condition of the lakes with the park's watersheds.
4. Fish	The condition of native and non-native salmonid species within the park's watersheds and the North and Middle Forks of the Flathead River.

This section uses the term 'ecological condition' as a means to capture three aspects of the park's relatively unimpacted ecological systems: ecological diversity, existing anthropogenic impacts that may affects that diversity, and potential future risks to that diversity to provide GNP with a watershed-scale prioritization tool to assist with finer scale monitoring or management decisions. Ecological diversity is measured by the contemporary natural range of biotic and abiotic ecosystem attributes that occur in the park. Ecological diversity provides increased resilience to potential ecosystem changes due to global impacts such as climate variability, regional impacts due to air quality, or local impacts due to increased human interaction (Chapin et al. 2000, Elmqvist et al. 2003). Assuming that watersheds with more diverse ecosystems and habitats are more significant, I applied a suite of metrics to capture this contemporary range of ecological significance within the park. Second, this assessment captures the range of human land use impacts on the natural ecosystem. Human impacts within the park are limited, but there are areas of higher and lower concentrations providing a human disturbance gradient similar to those commonly used in other RCA-based assessments approaches. Third, this assessment captures potential risks to the parks ecological significance. These risks assess potential future impacts from stressors such as air pollution or climate change. However, these risks are ameliorated by the system's ability to buffer against those potential impacts. Ecological condition, as measured here, is a combination of metrics that measure the natural range of ecological significance as they interact with metrics that measure human threats or risk within the park.

Preliminary Model Caveats

There are caveats throughout this section specific to individual models, but there are elements that the end-user should be aware of during the application of all models. This chapter reports on a pilot approach to an ecological assessment of natural resource conditions in and immediately adjacent to GNP. The broad project objective was to evaluate the conditions for a subset of important park natural resources - that is, a set of ecological attributes and resource condition indicators most relevant to GNP. The report relied on evaluation and synthesis of existing scientific data and information from multiple sources, combined with best professional judgment from an interdisciplinary team of specialists. *To the extent possible, I made use of quantitative data and analyses, but, especially where there are gaps in data and knowledge, the report also recognizes the practical need to use expert opinion for many of the indicators.*

This assessment approach uses ecosystem attributes that can be derived from existing data to measure, only within or immediately adjacent to the park's boundaries, the range of the park's ecological significance as well the range of human land use impacts that threaten that significance and potential threats (risk) to that significance. These are referred to as 'risk metrics' (e.g., presence of boat ramps and *potential* invasion of non-native or drastic expansion of individual native aquatic species). *Clearly the distinction between known and potential stress is* *not definitive, it is the responsibility of the end-user to make this distinction in the management decision.* These attributes are simplified to metrics (explained below) that range from 0 to 1 which, through expert opinion, indicate the quality of that attribute. These metrics are further simplified as they are combined into a multi-metric index. *These indices are intended to indicate a range in the quality of the system and its attributes not a true measure of ecosystem complexity or cause-and-effect pathways.* It is simply a management tool to provide scientists, managers and decisions makers with a prioritization approach for their various disciplines.

As stated above, HUC-10 watershed assignments create 13 assessment watersheds within the park. As the assessment indices are applied to these watersheds there is yet another simplification of ecosystem complexities as these indices integrate across wide topographic, aspect and distribution ranges. Again these index scores are intended as 'pointers' to areas of concern. The metrics within these indices provide finer detail of the potential drivers of the index score and the raw data used in the metric scoring provide further detail. As stated in the text below, an area may receive a low index score due to simplified patch complexity (e.g., reduced habitat structure in the confined reaches of the Flathead River) or due to proximity of human infrastructure or both. *It is incumbent upon the end-user to examine the details within this chapter and supporting data to support their ultimate management or monitoring decisions.*

Finally, there are disturbance vectors such as climate change and air pollution that impact the entire park in variety of ways. These broader impacts are addressed in Section 3, but due to the complexity of these impacts across the varied topography of the park and the limit of existing data they have not been integrated in all of the assessment indices. Rather, they are limited to models that assess alpine areas and lakes. The human impacts metrics within the majority of the assessment models measure the extent of direct human land use impacts such as roads, trails and facilities.

3.3 Assessment Models and Condition Indices

Assessing ecological condition of relatively unimpacted areas is not commonly done and there is no clear guidance to conduct such an assessment. Therefore a new approach was developed to address the park's unique ecological aspects. During initial meetings with NPS staff, it was agreed that the assessment of condition would use a multi-metric index approach. This approach was influenced by existing efforts to assess ecological condition in large landscapes (Schweiger et al. 2002, Tiner 2004, White and Maurice 2004, Whigham et al. 2007, Jacobs et al. 2010) and ecological risk of large landscapes (Landis and Wiegers 1997, Cormier and Suter 2008, Schleier III et al. 2008).

Index-based models have been developed throughout the world to provide politicians and decision makers with the ecological information on which they base their resource management decisions and communicate those decisions to the public (Turnhout et al. 2007, Dramstad 2009). Index-based models used to assess the ecosystem condition in the United States generally provide a quantitative measure describing where a system lies on a disturbance continuum ranging from least impacted condition to highly impaired (Fennessy et al. 2004). In reviews of contemporary aquatic assessment tools, an index was the predominant approach (Bartoldus 1999, Diaz et al. 2004, Fennessy et al. 2004, Böhringer and Jochem 2007b).

There are a total of seventeen metrics used in development of the assessment models in this effort. Table 3.4 provides the focal area and key ecological attribute where the metrics are applied, the metric name and symbol, which of the three metric types it is, and a definition of the metric.

Typical of multi-metric indices, each of these metrics represents a variable in the models and consists of four components (Schneider 1994). These include: (a) metric name and symbol; (b) methods to measure the metric via a procedural statement for quantifying or qualifying the measure directly or calculating it from other measurements; (c) range of values (i.e., numbers, categories, or numerical estimates; (Leibowitz and Hyman 1997) that are generated by applying the procedural statement, and (d) a scheme to provide a sub-index score for each metric. Table 3.5 provides examples of these components.

Attributes chosen for this assessment may be presented as ratios, percentages, or description. The next step in tool development is to normalize these various results by applying a sub-index to each metric which standardizes the variables by transforming them to dimensionless scores that use the same scale. Table 3.6 provides an example of a scored metric. The scores range from 0 to 1 and provide a means of qualifying the conditions related to the metric; 0 being poor and 1 being excellent. Some of these metrics may be further weighted based their relative influence on the focal ecological element as determined by literature, theory, and expert opinion. These assigned weighting multipliers are qualifiers used to disperse the results and assist in the index development only and are not intended to actually quantify the true contributions of each metric to the ecosystem support.

Finally, the metrics are combined into a series of multi-metric assessment indices that integrate information across a suite of ecological attributes. The assessment models for this section are expressed as a simple formula that combines metrics in certain ways to yield an estimate of the watershed condition relative to the focal area of interest. The condition index is best expressed as a percentage of total possible points. The design of the indices allows additional attributes and threats to be added in the future as more monitoring data becomes available.

Table 3.4. Relationship of Metrics to Focal Areas for Glacier National Park. Metric Type: S = significance metric. P = stressor (perturbation) metric. R = risk metric.

Focal Area	Key Ecological Attribute	Metric Name	Metric Symbol	Metric Type	Metric Definition
Streams	Stream Type Diversity	Alpine Stream	$V_{APLINESTR}$[1]	S	Stream type diversity composed of multiple streams subsections associated source, adjacent vegetation or landscape (e.g., wet meadows, forest, or confined valley)
		Subalpine Stream	$V_{SUBALPINESTR}$	S	
		Valley Bottom Stream	$V_{VLYBTMSTR}$	S	
	Stream Stressors	Alpine Stream Buffer	$V_{ALPSTRBUF}$	P	Proximity of stream buffer to human disturbance from park infrastructure.
		Subalpine Stream Buffer	$V_{SUBSTRBUF}$	P	
		Valley Bottom Stream Buffer	$V_{VLYVALBUF}$	P	
Flathead River	Floodplain and Buffer Connectivity	Floodplain Connectivity	$V_{FPCONNECT}$	S	Cover of native vegetation patches in the river's floodplain and buffer
	Floodplain and Buffer Stressor	Buffer Development	$V_{BUFFCDN}$	P	Extent and type of anthropogenic land cover in in the river's floodplain and buffer.
		Floodplain Development	V_{FPCDN}	P	
		Buffer Road Density	$V_{BUFFROAD}$	P	Extent of road density in in the river's floodplain and buffer.
		Floodplain Road Density	V_{FPROAD}	P	
Lake	Potential Lake Risk	Potential Acid Sensitivity	$V_{ACID\text{-}SEN}$	R	The ability of a lake's sub-watershed to buffer acidic atmospheric inputs.
		Potential Aquatic Nuisance	V_{EXOTIC}	R	Potential exposure to nuisance aquatic species from boat access.
		Potential Nutrient Sensitivity	V_{ALGAE}	R	Potential sensitivity of a lake to increased nutrient input.
Fish	Salmonid Distribution	Flathead Salmonid Distribution	$V_{FLTHDFISH}$	S	Metrics derived from ratios of native salmonid and non-native salmonids in GNP lakes and streams as well as the Flathead system.
		Lake Salmonid Distribution	$V_{LAKEFISH}$	S	
		Stream Salmonid Distribution	$V_{STRFISH}$	S	

1. The symbol consists of a 'V' for variable and a descriptive title.

Table 3.5. Components of a metric.

Metric Name and Symbol	Measures Descriptions	Range of Values	Scoring Scheme
River Buffer Development ($V_{BUFFCDN}$)	Characteristic plant communities. No grazing, or development beyond walking trails, horse paths, and bike trails. LULC Codes 41, 42, 43, 52, 71, 90, and 95	Descriptive	Categorical (e.g., 0, 0.5, 1.0)
Human Disturbance ($V_{HUMANDIST}$)	Percent of watershed human disturbance measured by proximity of raster cells to roads, facilities, campground, and trails.	0 to 100%	Continuous (e.g., 0-1)

Table 3.6. Post-fire habitat metric scoring.

Metric Criteria:	Metric Score
Watershed contains greater than 15,000 acres of post fire habitat	1.00
Watershed contains between 300 – 15,000 acres of post fire habitat	0.50
Watershed contains less than 300 acres of post fire habitat	0.10

Below is a scale for interpreting a condition index. This scale provides a coarse "snapshot" of the ecological condition of each focal area and allows comparison to previous or future assessments. It is a simply management tool to provide scientist, managers and decisions makers with a prioritization approach for their various disciplines and the qualifiers of condition help point the specialists toward the problems within their interest. It is important, however, that these specialists examine the individual metric scores in order to identify specific ecosystem attributes that may be imperiled. The indices are designed to provide a score between 0 and 1; for example, a score of 0.66 has a condition of 66% of its potential and is considered "fair" (Table 3.7).

Table 3.7. Relationship of Index Score and Condition.

Index Score	Percent of Optimum	Interpreted Condition
<0.50	Less than 50%	Critically Compromised
0.50 – 0.59	50% to 59%	Poor
0.60 – 0.69	61% to 69%	Fair
0.70 – 0.79	71% to 79%	Good
0.81 – 0.89	81% to 89%	Very Good
≥0.90	90% and Greater	Excellent

In this section, the authors and GNP resource management have identified four ecosystem focal elements (see Table 3.1) relevant to Glacier National Park. Seventeen metrics were assessed. In Section 5, the metrics and the subsequent indices are described in detail.

3.4 Spatial Data Overview

3.4.1 Data Layers

All spatial analyses were completed using ESRI ArcGIS (GIS) Version 10.0 with all data projected in the NAD-1983, UTM Zone 12N. Numerous GIS data layers were applied to the spatial analysis necessary to develop the metrics and the associated maps. Several of the layers were provided by the GNP GIS team or the State of Montana's geographic information clearinghouse (http://nris.mt.gov/gis/). Other layers were created specifically for this project and are available through the GNP-GIS office. The most critical GIS data layers are listed below (Table 3.8). Because of the numerous GIS layers used for this project, the layer name is included parenthetically only at the first mention of its use in the metric specific methods in later sections. For example: "the digital elevation model (DEM10.grd) was used to establish shaded relief in the background of each GNP image. The DEM was also used to establish elevation specific habitat zones."

The GNP land cover data layer, created for the USGS-NPS Vegetation Mapping Program (VMP) (Hop et al. 2007), plays a particularly import role in this analysis. The goals of the park's VMP were to (1) adequately describe and map plant communities and other land cover of the park and (2) provide useable baseline vegetation information to scientists and NPS resource managers. The project, initiated in 1998 and completed in 2009, resulted in the production of a list of plant communities (a plant community classification), their ecological description and a map showing their distribution produced in UTM coordinates (NAD 83) with a 1:24,000 scale and a minimum mapping unit of 0.5 hectares. The project reported an overall accuracy of 87.9% above the acceptable minimum total accuracy for land cover classification of 85% (Anderson et al. 1976).

Table 3.8. Listing of datasets, attributes, and scale of data used to access metrics.

Content	Layer Acquired	Source
Park Boundary	Boundary2003	Glacier National Park GIS Department
Background Aerial Image	World_Imagery.lyr	ESRI ArcGIS Online
2001 Land Use and Cover Class	NLCD, National Land Cover Database – 2001	USGS Seamless Data
Assessment Watersheds	GNP_HUC10-2011	USGS NHD Hydrological Database
Streams/Waterbodies	Major Lakes and Streams - 1:100,000 scale	NRIS Montana State GIS Data
	Streams	Glacier National Park GIS
Roads	Transportation network – Date unknown	ESRI Geodatabase
Topography	National Elevation Dataset	Glacier National Park GIS
Wetland Coverage	National Wetland Inventory – 2005	US Fish and Wildlife Service
GNP Land Cover	GLAC_VegMap	Glacier National Park GIS

The VMP project is useful in summarizing current conditions of Glacier's vegetation because it allows quantitative analysis of the relative abundance of developed and markedly disturbed areas. The map and associated metadata provide a broad overview of current vegetation, broken into units that indicate and define land cover at relatively coarse resolution (1.25 acres). The detail and extent of the VMP for the Park provided the basis for many of the vegetation based metrics used throughout this condition assessment.

More complete information about these and other GIS data layers can be found in the methods section of each focal area description, Appendix A (GIS Models) and the metadata associated with the GIS geodatabase.

4 Watershed-Scale Condition Assessment

In this section I provide condition assessment models for each of the four focal areas: 1) stream systems, 2) North and Middle Fork of the Flathead River, 3) lake systems, 4) fish populations.

4.1 Focal Area – Streams Systems

In response to degraded water quality, loss of potable water supplies, loss of fishable waters and conversion of wetlands, the U.S. federal government developed and passed the 1972 Federal Water Pollution Control Act (33 U.S.C. 1344). The purpose of this act, which later became known as the Clean Water Act (CWA), was to "….restore and maintain the chemical, physical, and biological integrity of the waters of the United States." Since 1972, the United States has invested millions of dollars in the development of approaches to conducting environmental assessment of the nation's waters (see Barbour et al. 1996, Stevenson and Hauer 2002 for reviews). Recent attempts to develop guidelines for stream and lake assessment have the advantage of building upon experiences from a long history of aquatic ecosystem assessment as well as recent innovations. This assessment addressed streams and tributaries within the interior of the park, and are divided into 3 overarching types, Alpine Streams, Subalpine Streams, and Valley Bottom Streams.

4.1.1 Alpine Streams

Alpine streams throughout the world have varied hydrologic and biogeochemical characteristics, as well as variation in biota. Despite the worldwide distribution of alpine stream systems, studies of their biota and biogeochemistry are limited (Ward 1994). There are three main types of alpine streams developed from descriptions in the European literature of the Alps; kryal, krenal and rhithral, each with distinct biotic and abiotic characteristics (Illies and Botosaneanu 1963). Kryal streams are fed by year-round melt water directly from snowfields, icefields and glaciers and are characterized by high heterogeneity within and between streams of this type. Krenal streams arise as springbrooks, hydrologically maintained by groundwater. Krenal streams generally have relatively stable chemical, hydrological and thermal conditions. Rhithral streams are characterized by seasonal snowmelt, and have wide temperature fluctuations, as well as diverse biota. Krenal streams transition into rhithral streams as distance from the groundwater sources increase and waters coalesce from the spring source. Alpine streams often have high gradients with waters flowing over bedrock and cobble-boulder substrate, high dissolved oxygen levels, high variation in temperature regimes due to open canopies and summer solar radiation, and low nutrient concentrations.

Alpine streams of GNP occur abundantly along the continental divide (Ward 1994). Waters of these alpine springbrooks generally are supplied by permanent snowfields or small icefields isolated behind mounds of colluvium. Stream temperatures remain at 32-33ºF (0-0.5ºC) at the springhead and vary less than 4ºF (2ºC) within 0-32 feet (0-10 m) of the source. However, solar radiation in mid-summer can quickly elevate the temperature of these streams. Mid-afternoon temperatures as high as 70-73ºF (21-23ºC) in alpine streams can occur within only a few hundred meters of their source. Although these streams can become quite warm during the day, night temperatures are often 32-37ºF (0-3ºC). Thus, diel temperature flux in the alpine, at distances of a few hundred meters from the source, can vary >64ºF (18ºC).

Fauna of the alpine streams of GNP is dominated by aquatic insects. Generally within 330 feet (100 m) of their source, krenal streams are dominated by several species of Simuliidae (black flies) and Heptageniidae (mayflies) (Hauer et al. 2001). The endemic stonefly, *Lednia tumana*, inhabits a narrow stream-type and spatial distribution, restricted to short sections (about 1,650 feet or 500 m) of cold, krenal alpine streams directly below glaciers, permanent snowfields, and springs (Muhlfeld et al. 2011). Simulation models suggest that climate change threatens the potential future distribution of these sensitive habitats and the persistence of *L. tumana* through the loss of glaciers and snowfields. The caddisfly, *Allomyia bifosa*, is found exclusively near the springhead of permanently flowing krenal alpine springs fed by snow or ice fields and associated with wet meadows (Hauer et al. 2007). Alpine aquatic invertebrates are ideal early warning indicators of climate warming in mountain ecosystems as the habitat that supports their life histories become increasingly reduced in distribution and abundance (Muhlfeld et al. 2011).

4.1.2 Subalpine Streams

The subalpine streams of Glacier National Park are highly variable, but tend to have similar unifying characteristics. Hydrologically, these streams receive most of their flow from rain and snow deposited at high elevation of the alpine and within the subalpine zone of the mountain slopes. Abundant groundwater enters these streams following discharge into small springs along the toe of side slopes. Stream discharges in GNP subalpine streams closely follow that of a snowmelt regime (Poff and Ward 1989). Hauer et al. (2001 and 2003) in a study of McDonald Creek in Glacier National Park observed inter-annual variation in the magnitude and timing of maximum discharge, but this occurred each year of an 8-year study between mid-May and mid-June. Discharge typically increased >10 times the autumn base flow. Over 90% of the total nitrogen flux from the McDonald Creek basin occurred as NO_3 with maximum concentrations approaching 450µg/L, but minimum concentrations less than 100µg/L. These low concentrations predominant throughout the fall and winter base flow period and increase very rapidly at the onset of spring runoff. The rate of increase in NO_3 concentrations is significantly greater than the rate of increase in spring discharge. This suggests that nitrate is accumulated and concentrated in the groundwater over the winter near the valley floor where the first snow melt that initiates the flood period occurs in the spring and discharges high NO_3 water from side slope aquifers into the stream. Nitrogen concentration decreases after the initial pulse in the early spring; and although discharge increases, primarily driven by high elevation snowmelt as the spring warming progresses, nitrogen concentration decreases. This is most likely the result of dilution of the groundwater by melting snows from high elevation. Although I have no direct evidence, I strongly suspect that the high concentration of Alder (*Alnus spp*.) in avalanche chutes and high slope wetlands may play a significant role in the loading of NO_3 to subalpine shallow aquifers. Many studies have shown that soils directly surrounding stands of Alder are rich in nitrogen allowing for increased production by neighboring species. Postgate (1978) showed how Alder communities can increase soil nitrogen as much as 100kg N/hectare/year through the mineralization of leaf litter alone. On a floodplain in the Alaskan interior, Alder communities are believed to have increased total soil nitrogen accumulation by a factor of four over a twenty year span (Walker 1989).

In the pristine forest streams of Glacier National Park, Hauer et al. (2001) observed very predictable temperature regimes closely correlated with elevation. This has a direct effect on the distribution of stream organisms including benthic macroinvertebrates and fish. Hauer et al. (2001) collected over 100 species of the three dominant orders of aquatic insects occurring commonly in GNP subalpine streams (i.e., Ephemeroptera, mayflies; Plecoptera, stoneflies; and Trichoptera, caddisflies). Taxa within the same order and possessing similar trophic relations had abundance patterns and predictable distributions along the elevation and temperature gradient.

4.1.3 Valley Bottom Streams and Rivers

Valleys of Glacier National Park were modified by Pleistocene alpine glaciers that carved through the landscape. Valley bottom, alluvial streams and rivers are characterized by broad and active alluvial floodplains, with highly complex physical and biological interactions between stream channels, surficial backwaters, springbrooks, and buried paleo-channel networks (Stanford and Ward 1993, Hauer et al. 2003, Stanford et al. 2005). These complex interactions within and between habitats are driven by strong lateral and vertical flux of water and materials including flood-caused cut and fill alluviation, routing of river water and nutrients above and below ground, channel avulsion, and dynamics of large wood. The strong forces are driven by the river hydrologic regime and sediment dynamics to form and maintain a complex, dynamic distribution of resource patches and associated biota: the shifting habitat mosaic (SHM: Stanford et al. 2005). These characteristics are critically important in maintaining water quality, bioproduction, and biodiversity of the stream and river systems of the valley floors.

Floodplains composed of coarse sediments engaged in the processes embodied by the SHM are penetrated by river waters creating complex three-dimensional mosaics of surface and subsurface habitats (Brunke and Gonser 1997, Poole et al. 2002). Ground water – surface water interactions are critical characteristics of these streams and their floodplain corridors. Alluvial aquifer water returning to the surface is generally higher in NO_3 and PO_4 than surrounding surface flows, resulting in patches of high algal productivity (Bansak 1998, Wyatt et al. 2008). In these valley bottom tributary streams with broad floodplain reaches, hyporheic return flow also results in increased macroinvertebrates growth and productivity (Pepin and Hauer 2002) and growth rates of floodplain vegetation (Harner and Stanford 2003). Native species of fish, particularly the salmonids (bull trout, westslope cutthroat trout, mountain whitefish) focus on the complexity of floodplains and spawn in habitats dominated by extensive groundwater – surface water interaction (Baxter and Hauer 2000).

The floodplains of montane alluvial rivers are extremely ecologically diverse. The valley floodplains of GNP have high biodiversity, from floodplain plant species and aquatic food webs (Stanford et al. 2005) to large carnivores (Demarchi et al. 2003). The continuity of these highly diverse components of the GNP landscape is very dependent on hydrologic linkages and the high water quality associated with the geology as well as the park's pristine character (Stanford and Ellis 2002).

4.1.4 Methods: Aquatic Resources as Indicators of Streams and Rivers

The following is a summary of each variable used in the stream and rivers assessment models. Each variable provides the variable code, name, definition, the rationale for selecting and scaling the variable, and the scaled variable in table form or a description of the formulas and methods used to scale the variable. As with all models in the assessment it provides a score for a watershed derived from the diversity of the habitat and proximity to human activity that is relative to other watersheds in the park only.

The streams and rivers condition index is made up of three metrics: alpine streams ($V_{ALPINESTR}$), subalpine streams ($V_{SUBALPINESTR}$), and valley bottom streams ($V_{VLYBTMSTR}$). These three stream types were derived for each watershed from the park's available elevation and topological data from the digital elevation model (DEM; dem10.grd). The Flathead River index is addressed separately.

4.1.4.1 Alpine Stream ($V_{ALPINESTR}$)

Alpine streams were defined as any GNP streams (streams_clip.shp) located above 6,500 feet on the DEM and within alpine vegetation cover types on the GNP vegetation classification data (glac_vegmap.shp). An alpine layer was created (Alpine_trueveg_wtrshd.shp) which was intersected with GNP study watersheds (GNP_HUC10_2011.shp) to create the final alpine streams GIS layer (alpine_streams.shp).

Streams located within this alpine area were divided into six classes based on their hydrogeomorphic position. Five of these classes were based on the stream outlets proximal to: 1) wet meadows, 2) snowfields, 3) shrub wetlands, 4) glaciers, and 5) lakes as determined by the polygons within the GNP vegetation classification dataset. Total stream length for each class was determined within a consistent buffer distant of 300 feet (INFISH 1995) down slope of each of the five polygon classes. The remaining class includes all streams within the alpine areas that are outside of the outlet buffers. A weighted multiplier was established for each outlet class based on sources of carbon and outlet stream temperature likely to occur at the outfalls of each class type. These assigned weighting multipliers are derived from literature, theory, and expert opinion and are qualifiers used to disperse the results and assist in the index development only and are not intended to actually quantify the true contributions of biotic support per class. Multiplies, buffers, and data sources listed in Table 4.1.

Table 4.1. Data source, buffer distance, and weighted multiplier for each alpine class.

Land Cover	Data Source	Buffer distance (ft)	Weighted Multiplier
Wet Meadow	glac_vegmap.shp	300	6
Shrub Wetlands	glac_vegmap.shp	300	5
Snowfields	glac_vegmap.shp	300	4
Glaciers	1998_glaciers.shp	300	3
Lakes	lakes_clip.shp	300	2
Other	glac_vegmap.shp	N/A	1

The following describes the steps necessary derive the final Alpine Stream Metric score from these 6 stream classes:

1. Proportion of alpine stream associated with an outlet class was determined within a watershed (stream outlet class length divided by total alpine stream length; e.g., sum of all wet meadow outlet stream lengths divided by the total alpine stream length).
2. A weighting multiplier was then applied to each class relative to their general quality in term of their contribution to stream biotic support (see Table 4.1). For example, shrub meadows generally provide more fine particulate organic matter to support downstream biota then the outlets of lakes. As above, these assigned weighting multipliers are qualifiers used to disperse the results and assist in the index development only and are not intended to actually quantify the true contributions of biotic support per class. The formula for the weighted habitat diversity is:

   Weighted habitat diversity = ((6*wetmeadow + 5*shrubwetland + 4*snowfield + 3*glacier + 2*lake + otherstreams)/21)
3. The watershed's weighted habitat diversity scores are then divided by the maximum score across all watersheds to acquire relative habitat diversity score relative to maximum habitat diversity in the park.
4. Adjusting the score for unimpacted conditions: These stream metrics measures the diversity of naturally occurring ecosystem attributes within the park and will later be joined with a

metric that measures the extent of human disturbance in the park. A choice was made to lift the diversity scores to be between 0.8 and 1.0 to balance the effects of natural system diversity and human perturbation on the overall condition index. This allows the stream ecological significance score to be further scaled toward the unimpacted range of the spectrum. The logic is that streams in the park's alpine habitat will show a range of conditions, with diverse alpine systems scoring higher then less diverse systems. However if an alpine system exists in the park in the 2011 assessment and is not impacted by human interaction, it will not score lower than 0.80. The formula for the final metric score is:

$$V_{ALPINESTR} = (\text{Weighted habitat diversity} + 4)/5.$$

For future assessments, this final score is multiplied by the total current alpine area divided by the 2011 total alpine area. For this 2011 assessment, the multiplier is equal to 1.0, but for future assessments this may be less than 1 due to alpine habitat loss resulting from such impacts as climate change which will result in a lower overall condition score compared to the 2011 results. Additionally, the overall metric score may change if there are future changes in the land cover that makes up the six different alpine stream classes.

4.1.4.2 Subalpine Streams ($V_{SUBALPINESTR}$)

Subalpine streams were defined as any GNP streams located below the alpine habitat and above the valley bottom systems as defined below. These streams were intersected with the GNP study watersheds creating a new subalpine streams GIS layer (subalpine_streams.shp).

Subalpine streams were divided into 4 classes based on their hydrogeomorphic position or vegetation cover within each watershed's subalpine area. These classes were defined by; 1) proximity to a lake outlet, 2) location within avalanche chutes, 3) location within forest cover, or 4) all other streams in the subalpine area as determined by the polygons within the GNP vegetation classification dataset. Stream length was determined for lake land cover class using the buffer distances and data sources listed in Table 4.2. The stream buffers were chosen based on the relative area of channel lengths in the elevation zone. The following describes the steps necessary derive the final Subalpine Stream Metric score from these 4 stream classes:

1. Stream outlet class length relative to the maximum outlet density in the park: The relative length was determined for each stream outlet class by dividing the stream class length by the total subalpine stream length within a watershed. This gave a score between 0 and 1 that provided a comparison of subalpine stream habitat diversity within the park.
2. Weighted subalpine habitat diversity: As above, these assigned weighting multipliers are qualifiers used to disperse the results and assist in the index development only and are not intended to actually quantify the true contributions of biotic support per class. The formula for the weighted diversity is:

   Weighted Habitat Diversity = ((2*Avalanche Chute + 2*Lake outlet + Forested Streams + Non-Forested Streams)/7)
3. Habitat diversity relative to maximum outlet habitat diversity in the park: The watershed's weighted diversity score is then divided by the maximum across all watersheds to acquire a relative diversity score.
4. Adjusting the score for unimpacted conditions: These stream metrics measures the diversity of naturally occurring ecosystem attributes within the park and will later be joined with a metric that measures the extent of human disturbance in the park. A choice was made to lift the diversity scores to be between 0.8 and 1.0 to balance the effects of

natural system diversity and human perturbation on the overall condition index. This allows the stream ecological significance score to be further scaled toward the unimpacted range of the spectrum. The logic is that streams in the park's subalpine habitat will show a range of conditions, with diverse subalpine systems scoring higher then less diverse systems. However, if a subalpine system exists in the park in the 2011 assessment and is not impacted by human interaction, it will not score lower than 0.80.The formula for the final metric score is:

$V_{SUBALPINESTR}$ = (Weighted habitat diversity + 4)/5.

Table 4.2. Data source, buffer distance, and weighted multiplier for each subalpine class.

Land Cover	Data Source	Buffer distance (ft)	Weighted Multiplier
Avalanche Chutes	Glac_vegmap.shp	N/A	2
Lakes Outlet	lakes_clip.shp	600	2
Forested Streams	Glac_vegmap.shp	N/A	1
Other	Glac_vegmap.shp	N/A	1

4.1.4.3 Valley Bottom Streams ($V_{VLYBTMSTR}$)

To define the park's valley bottom streams, the floodplain of streams within the park were digitized using the GNP stream layer, the digital elevation map, and visual assistance from oblique views within Google Earth's 3-D models (http://www.google.com/earth/index.html). The digitized floodplain areas were saved as a new GIS layer (valley_unconfined.shp). The furthest upstream floodplain area in each stream defined the upstream extent of the valley bottom system. All stream sections below that upstream extent and were not in a floodplain, were defined as confined reaches and were saved separate GIS layer (valley_confined_streams.shp).

Valley bottom streams were divided into 3 classes based on their hydrogeomorphic position within each watershed. These classes were based on; 1) proximal to a lake outlet, 2) stream reaches within an unconfined floodplain, and 3) stream reaches within a confined valley bottom (lacking defined floodplain). Stream length was determined for each class using the buffer distances and data sources listed in Table 4.3. The stream buffers were chosen based on the relative area of channel lengths in the elevation zone. The following describes the steps necessary derive the final Valley Bottom Stream Metric score from these stream classes:

1. Stream outlet class length relative to the maximum outlet density in the park: The relative length was determined for each stream outlet class by dividing the stream class length by the total valley bottom stream length within a watershed. This gave a score between 0 and 1 that provided a comparison of valley bottom stream habitat diversity within the park.
2. Weighted subalpine habitat diversity: As above, a weighting multiplier was then applied to each class relative to their general contribution to stream biotic support. The formula for the weighted diversity is:

   Weighted diversity = ((5*Lake Outlet + 2* Unconfined Valley Bottom+ Confined Valley Bottom)/8)
3. Habitat diversity relative to maximum habitat diversity in the park: The watershed's weighted diversity score is then divided by the maximum across all watersheds to acquire a relative diversity score.
5. Adjusting the score for unimpacted conditions: These stream metrics measures the diversity of naturally occurring ecosystem attributes within the park and will later be joined with a metric that measures the extent of human disturbance in the park. A choice

was made to lift the diversity scores to be between 0.8 and 1.0 to balance the effects of natural system diversity and human perturbation on the overall condition index. This allows the stream ecological significance score to be further scaled toward the unimpacted range of the spectrum by applying this formula: (relative diversity score + 4)/5. The logic is that streams in the park's alpine habitat will show a range of conditions, with diverse alpine systems scoring higher then less diverse systems. However valley bottom system not impacted by human interaction will not score lower than 0.80. The formula for the final metric score is:

$$V_{VLYBTMSTR} = (\text{Weighted habitat diversity} + 4)/5.$$

Table 4.3. Data source, buffer distance, and weighted multiplier for each valley bottom class.

Land Cover	Data Source	Buffer distance (ft)	Weighted Multiplier
Lakes Outlet	valley_lake1500outlets.shp	1500	5
Unconfined valley bottom	valley_unconfined.shp	N/A	2
Confined valley bottom	valley_confined_streams.shp	N/A	1

4.1.4.4 Stream Buffer Disturbance ($V_{ALPSTRBUF}$, $V_{SUBSTRBUF}$, $V_{VALSTRBUF}$)

Potential impacts of the park's streams and stream buffers are likely to occur through interactions with the park's roads, trails, and camping and facilities infrastructure. Floodplain areas in the park are variable and generally there are insufficient scientific data to support the use of specific buffer width the will attenuate all human disturbance (Palik et al. 2000, Todd 2000). Recommended buffers to protect fish and aquatic habitat are wide ranging. For example recent literature review conducted by Montana Department of Environmental Quality show recommendation ranging from 40 to 300 feet (Ellis 2008). For the purposes of this condition assessment, a buffer of 300 feet was selected following the recommendations from (INFISH 1995).

A raster based assessment was developed using five raster datasets (or shape files converted to raster) available from the park: park's roads geo-database, railways from the BNSF geo-database, trails (Trails_20050501.shp), buildings (building2006.shp), and campsites (campsites.shp). For consistency, all rasters developed for this assessment consisted of a 30-meter grid (about 100-feet). A potential impact buffer was established for each dataset based on 100-foot increments. These increments were assigned graduated scores on potential impacts to buffer condition (Table 4.4) based on expert opinion only. If the cell was less than or equal to the Euclidean buffer distance and greater than the previous buffer distance the cells received the assigned cell score. If greater than the max distance then the cell score was 1.0. The layers were then superimposed and the lowest of each layer's cell score was assigned to the final cell layer and all cells outside these buffers have a score of 1.0. Finally, the park-wide raster was clipped to the approximate 300-foot (90-meter) stream buffers and the cell score were averaged determine the park's buffer condition for the alpine, sub-alpine and valley bottom streams. The cell scores were then averaged for each watershed.

The potential impact buffers applied to the infrastructure made a few assumptions (Table 4.4). All roads in the park are not the same, but for this assessment they are treated as large highways assuming that a culvert is placed under the paved areas, shoulders and trapezoidal fill that would be 100 feet from the center line and where a stream buffer would score a '0'. The next 100 feet would be cleared of vegetation and the stream buffer would score a '0.1'. The next 100 feet would be disturbed forest and the stream buffer would score '0.5' and beyond that the stream buffer would be unimpacted and score a '1.0'. The same logic was applied to railroads. I assumed for building and campsites that the adjacent areas would be thinned to a distance of 100 feet where a stream buffer would score ‘0.5’. Human interaction would gradually diminish to a distance of 300 feet where the stream buffer would score increase to ‘0.9’. Trails would have an impact to stream buffers to a distance of 200 feet from potential human interaction. These assumptions are an oversimplification of the parks interactions with stream buffers, but because the interaction with stream buffers in the park are limited, the approach allows for an increased signal in the metric scoring.

Table 4.4. Human disturbance ($V_{HUMANDIST}$) metric scoring assigned to the appropriate raster cells.

			Raster Cell Score Assigned to Buffer Distances									
Raster	**Approximate Buffer size (ft)**	**Total Buffer size (m)**	**0**	**0.1**	**0.2**	**0.3**	**0.4**	**0.5**	**0.6**	**0.7**	**0.8**	**0.9**
Road 1	300	90	100	200	-	-	-	300	-	-	-	-
Railroad	300	90	100	200	-	-	-	300	-	-	-	-
Building and Campsites	300	90	-	-	-	-	-	100	-	200	-	300
Trail	200	60	-	-	-	-	-	100	-	-	-	200

4.1.4.5 GNP Stream Condition Index

To acquire an overall condition score for each watershed, I applied following index. For each elevation zone, a significance metric that captures the relative stream type diversity is multiplied by a stressor metric that measures the degree that stream buffers interact with human infrastructure. In areas where there is little interaction within the buffer, this multiplication and square root element of the model has a similar effect on the condition score as averaging. However where the buffer impacts are greater in this contemporary assessment or if human infrastructure increases in the future of the park this multiplicative and square root element of the model will drive the scores much lower. An additional constant was added to this model to account for change in alpine area due to future loss of alpine area from climate change or potential disturbances.

Stream Condition Index = (Square Root (($V_{ALPINESTR}$ * AlpChnge) * $V_{ALPSTRBUF}$) + Square Root ($V_{SUBALPINESTR}$ * $V_{SUBSTRBUF}$) + Square Root ($V_{VLYBTMSTR}$ * $V_{VALSTRBUF}$))/3

AlpChnge = current alpine area/2007 alpine area (derived Hop et al 2007). For this 2012 assessment, Alpchnge = 1.

4.1.5 Stream Condition Results

4.1.5.1 Subalpine Streams ($V_{ALPINESTR}$)

Table 4.5 provides the proportions (see methods above) of alpine stream classes within each of the park’s watersheds. For example, no wet meadows are mapped in the alpine area of Coal/Ole and Kintla/Bowman watersheds and therefore those streams have a 0.0 for that outlet class.

Four other watersheds show 2% wet meadow outlet class and are the highest found in the park. These outlet proportions were applied to the weighted sub-index provided in the methods above to derive the alpine stream condition metric score. All streams scored very high with this assessment, with Belly and Nyack scoring the highest and the less diverse Cut Bank and Kennedy scoring the lowest (Figure 4.1).

Table 4.5. Relative density of watershed alpine stream classes and the resulting metric score.

Watershed	Relative Habitat Density: Wet Meadow Outlet	Shrub Alpine Outlets	Snowfield Outlet	Glacier Outlet	Lake Outlet	Other Alpine Streams	Adjusted Metric Score
Belly	0.00	0.06	0.27	0.05	0.02	0.61	1.00
Camas	0.02	0.00	0.00	0.00	0.00	0.98	0.90
Coal/Ole	0.00	0.00	0.27	0.00	0.01	0.72	0.97
Cut Bank	0.01	0.02	0.12	0.00	0.02	0.82	0.94
Kennedy	0.00	0.04	0.09	0.00	0.03	0.84	0.94
Kintla/Bowman	0.00	0.00	0.32	0.01	0.02	0.65	0.99
Lake McDonald	0.00	0.01	0.28	0.00	0.02	0.68	0.98
Nyack	0.02	0.00	0.31	0.06	0.02	0.60	1.00
Quartz/Logging	0.01	0.02	0.28	0.02	0.02	0.66	0.99
Saint Mary	0.02	0.03	0.27	0.02	0.01	0.64	0.99
Swiftcurrent	0.02	0.00	0.20	0.01	0.02	0.74	0.96
Upper Two Medicine	0.00	0.04	0.14	0.00	0.00	0.82	0.95
Waterton	0.01	0.01	0.30	0.02	0.05	0.61	1.00

4.1.5.2 Subalpine Streams ($V_{SUBALPINESTR}$)

Table 4.6 provides the proportions (see methods above) of subalpine stream classes within each of the park's watersheds. For example, Swiftcurrent and Cut Bank have the highest proportion of stream length associated with lake outlet class and Kennedy has the highest proportion of stream length in forested area and watersheds with the highest diversity of stream classes, relative to the weighted categories, scored the highest for this condition metric (Figure 4.2).

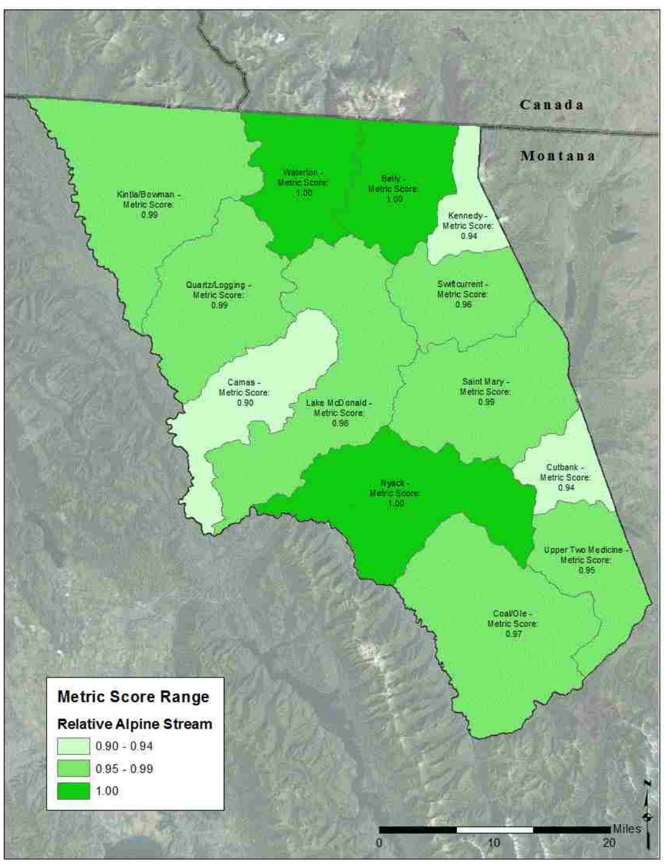

Figure 4.1. Metric measuring alpine stream condition in GNP.

Table 4.6. Relative length of watershed subalpine streams classes and the resulting metric score.

	Adjusted Relative Stream Length				
Watershed	**Lakes Outlet**	**Streams in Avalanche Chutes**	**Forested Streams**	**Non-forested Streams**	**Metric Score**
Belly	0.02	0.17	0.63	0.19	0.98
Camas	0.00	0.14	0.79	0.07	0.98
Coal/Ole	0.01	0.21	0.68	0.09	0.99
Cut Bank	0.03	0.08	0.71	0.18	0.97
Kennedy	0.02	0.04	0.83	0.11	0.96
Kintla/Bowman	0.01	0.10	0.81	0.09	0.97
Lake McDonald	0.01	0.19	0.63	0.17	0.98
Nyack	0.01	0.27	0.61	0.11	1.00
Quartz/Logging	0.01	0.10	0.77	0.12	0.97
Saint Mary	0.02	0.13	0.64	0.21	0.98
Swiftcurrent	0.05	0.16	0.54	0.25	0.99
Upper Two Medicine	0.02	0.10	0.68	0.20	0.97
Waterton	0.02	0.28	0.57	0.13	1.00

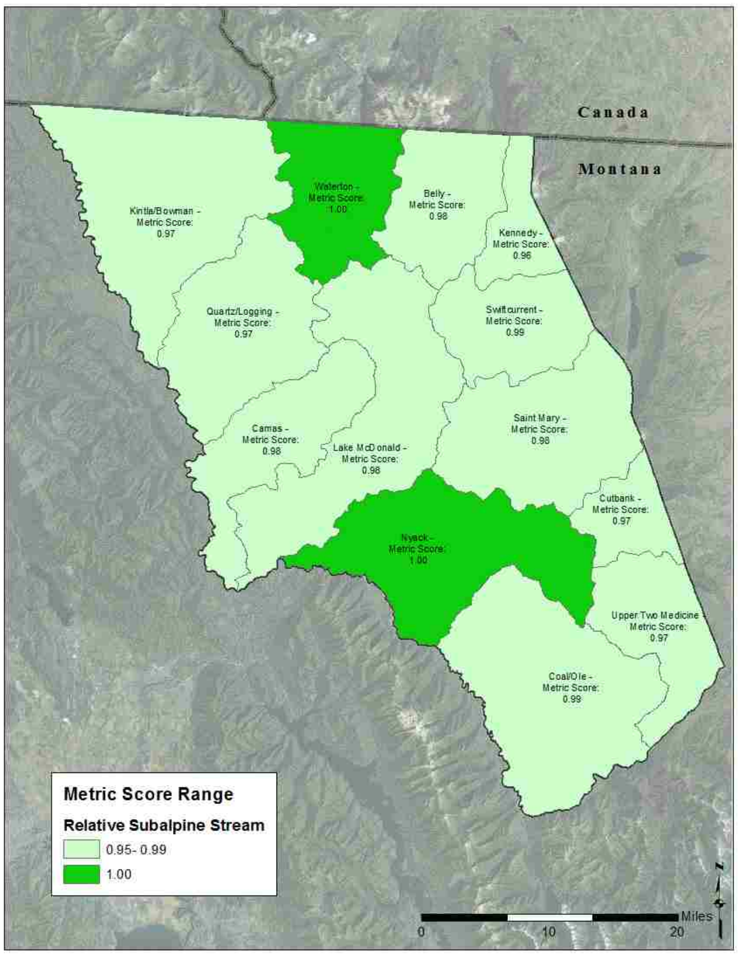

Figure 4.2. Metric measuring subalpine stream conditions in GNP.

4.1.5.3 Valley Bottom Streams ($V_{VLYBTMSTR}$)

Table 4.7 provides the proportional length (see methods above) of valley bottom stream classes within each of the park's watersheds. For example, because Upper Two Medicine has most (22) lakes associated with valley bottom streams and therefore 18% of the streams length associated with lake outlets however, although Lake McDonald has a very large lake, it only has (3) valley bottom lakes and only 3% of the streams length associated with lake outlets. Watersheds with the highest diversity of these stream classes, relative to the weighted categories, scored the highest for this condition metric (Figure 4.3).

Table 4.7. Relative length of watershed valley bottom streams classes and the resulting metric score.

Watershed	Adjusted Relative Stream Length			Metric Score
	Lake Outlet	Unconfined Valley	Confined Valley	
Belly	0.15	0.58	0.26	0.99
Camas	0.06	0.68	0.25	0.97
Coal/Ole	0.02	0.69	0.29	0.96
Cut Bank	0.05	0.60	0.35	0.96
Kennedy	0.10	0.71	0.19	0.98
Kintla/Bowman	0.04	0.36	0.60	0.93
Lake McDonald	0.03	0.52	0.45	0.94
Nyack	0.02	0.63	0.35	0.95
Quartz/Logging	0.06	0.42	0.52	0.95
Saint Mary	0.07	0.54	0.39	0.96
Swiftcurrent	0.09	0.83	0.08	0.99
Upper Two Medicine	0.18	0.56	0.25	1.00
Waterton	0.09	0.52	0.40	0.96

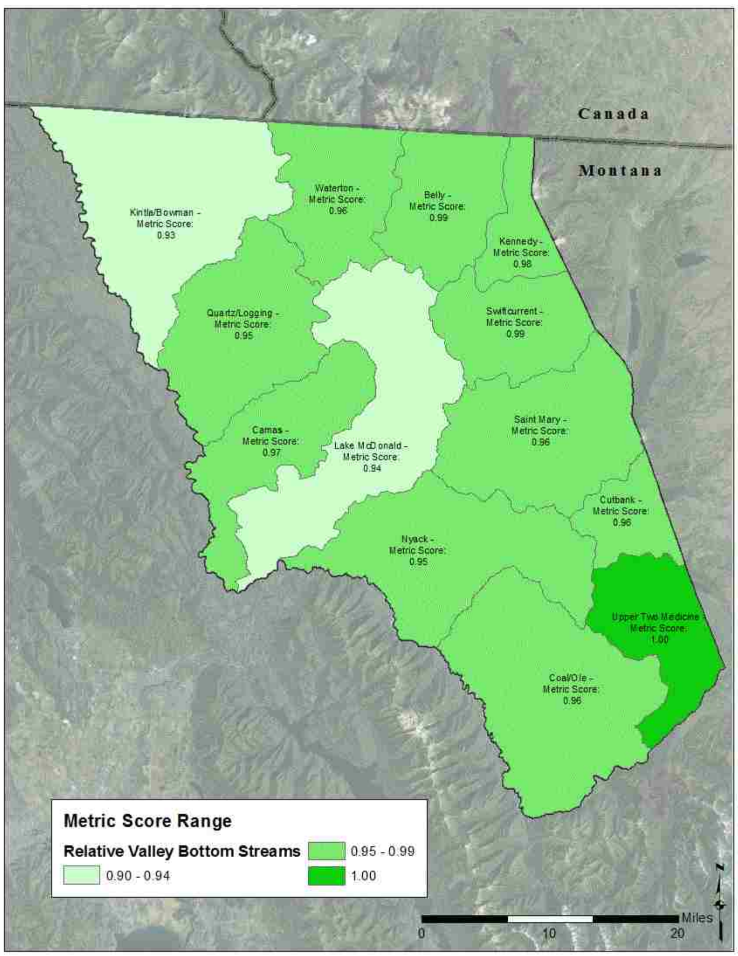

Figure 4.3. Metric measuring valley bottom stream conditions in GNP.

4.1.5.4 Stream Buffer Integrity (V_{ALPBUF}, V_{SUBBUF}, V_{VALBUF})

As stated above, potential impacts of the park's streams and their buffers are likely to occur through interactions with the park's roads, trails, and camping and facilities infrastructure. It is clear that degree of interaction between infrastructure and streams will vary from site-to-site. These buffer integrity metrics are an oversimplified measurement of these interactions intended to provide park managers to with information of where these impacts are likely to be greatest. Table 4.8 shows the degree of interactions of park infrastructure and stream buffers in alpine, subalpine and valley bottom streams. In the alpine and subalpine areas, all watersheds scored very high. The lowest score was in Lake McDonald watershed do the Going-to-the-Sun Road and its nearby trails. In the valley bottom areas, Lake McDonald watershed again scored the lowest due to the road networks and facilities (Figure 4.4).

Table 4.8. Stream buffer condition as measured by its proximity to park infrastructure.

	Metric Scores		
Watershed	**Alpine Buffer**	**Subalpine Buffer**	**Valley Bottom Buffer**
Belly	1.00	0.98	0.93
Camas	1.00	0.97	0.93
Coal/Ole	1.00	0.97	0.95
Cut Bank	0.98	0.98	0.91
Kennedy	1.00	0.98	0.91
Kintla/Bowman	0.99	0.98	0.94
Lake McDonald	0.94	0.94	0.74
Nyack	0.99	0.98	0.93
Quartz/Logging	1.00	0.99	0.96
Saint Mary	0.99	0.96	0.91
Swiftcurrent	0.99	0.96	0.91
Upper Two Medicine	1.00	0.96	0.91
Waterton	0.99	0.98	0.94

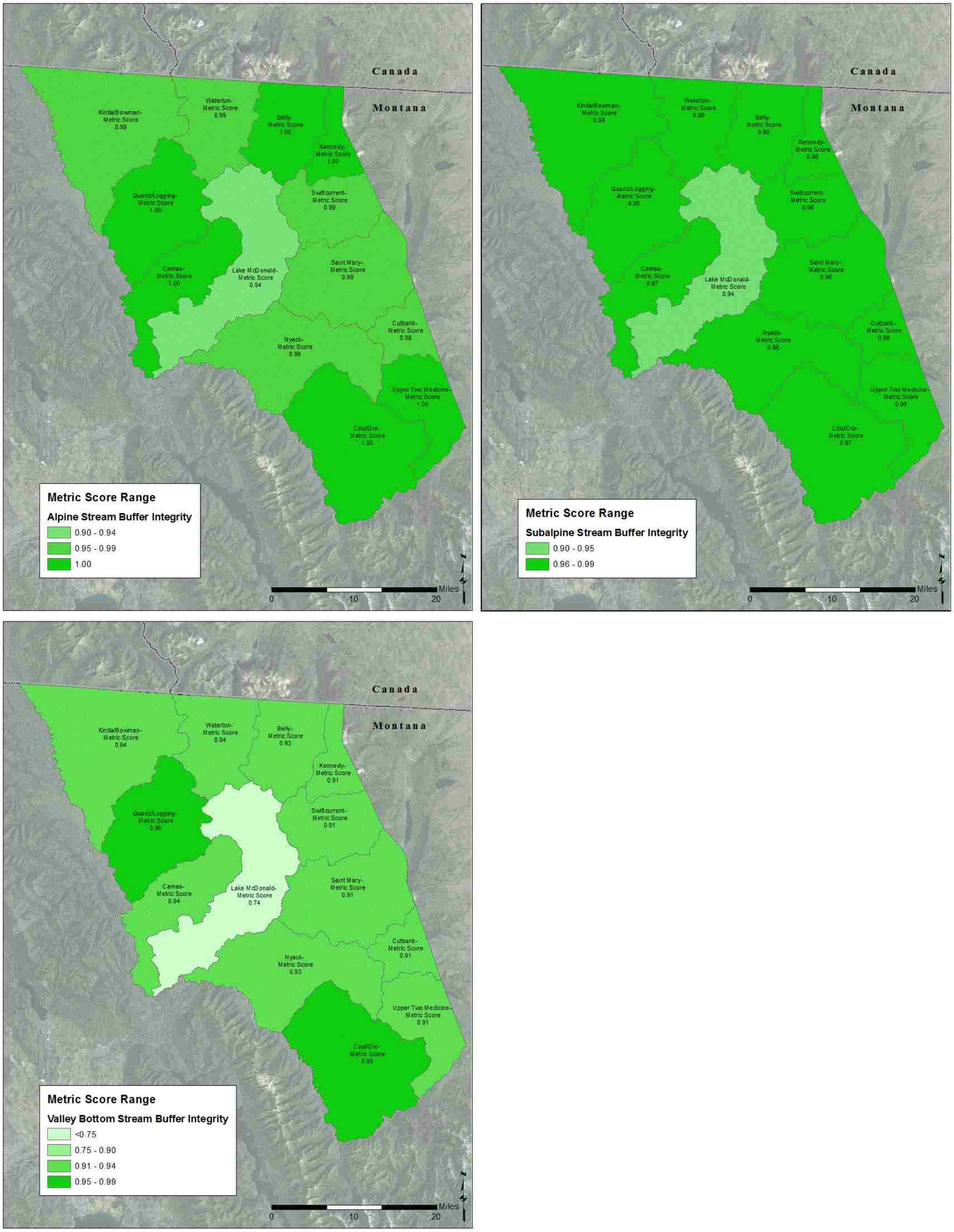

Figure 4.4. Metric measuring Alpine stream condition in GNP.

4.1.5.5 GNP Stream Condition Index

Six metrics: alpine, subalpine and valley bottom stream significance, and three buffer stressor metrics for each zone were combined to provide an assessment index of stream condition in the park's watersheds (Table 4.9). The significance metrics measure diversity ecosystems that support streams for each watershed. Additionally, the stressor metrics measure likely impacts to stream buffer due to proximity to park infrastructure. Here, Lake McDonald watershed scored the lowest (index score of 0.92) due predominantly to buffer impacts in each zone (Figure 4.5).

Table 4.9. Stream condition score for all watersheds with GNP.

	Metric Scores						
Watershed	Alpine Stream	Alpine Buffer	Subalpine Stream	Subalpine Buffer	Valley Bottom Stream	Valley Bottom Buffer	Index Score
Belly	1.00	1.00	0.98	0.98	0.99	0.93	0.98
Camas	0.90	1.00	0.98	0.97	0.97	0.93	0.96
Coal/Ole	0.97	1.00	0.99	0.97	0.96	0.95	0.97
Cut Bank	0.94	0.98	0.97	0.98	0.96	0.91	0.96
Kennedy	0.94	1.00	0.96	0.98	0.98	0.91	0.96
Kintla/Bowman	0.99	0.99	0.97	0.98	0.93	0.94	0.97
Lake McDonald	0.98	0.94	0.98	0.94	0.94	0.74	0.92
Nyack	1.00	0.99	1.00	0.98	0.95	0.93	0.97
Quartz/Logging	0.99	1.00	0.97	0.99	0.95	0.96	0.98
Saint Mary	0.99	0.99	0.98	0.96	0.96	0.91	0.96
Swiftcurrent	0.96	0.99	0.99	0.96	0.99	0.91	0.97
Upper Two Medicine	0.95	1.00	0.97	0.96	1.00	0.91	0.96
Waterton	1.00	0.99	1.00	0.98	0.96	0.94	0.98

4.1.6 Assessment of availability and gaps in monitoring data

Alpine shrub and emergent marsh wetlands are an important component in the alpine stream condition wetland assessment (Mitch and Gosselink, 2000). Hop et al. (2007) reported producer accuracy (omission) of 87% for wet meadows in their land cover map. This is above the acceptable minimum total accuracy for land cover classification of greater than 85% (Anderson et al. 1976). However, it is the opinion of the author that there is an error of omission in the classification of the alpine wetlands (e.g., wet meadows) greater than what is accounted for in the GNP vegetation classification accuracy assessment. This opinion is based on extensive field experience within the park. Higher detail in the mapping of wetlands within the park would not only increase the robustness of the alpine stream condition metric, but would allow an assessment of the park's wetland aquatic resources as well.

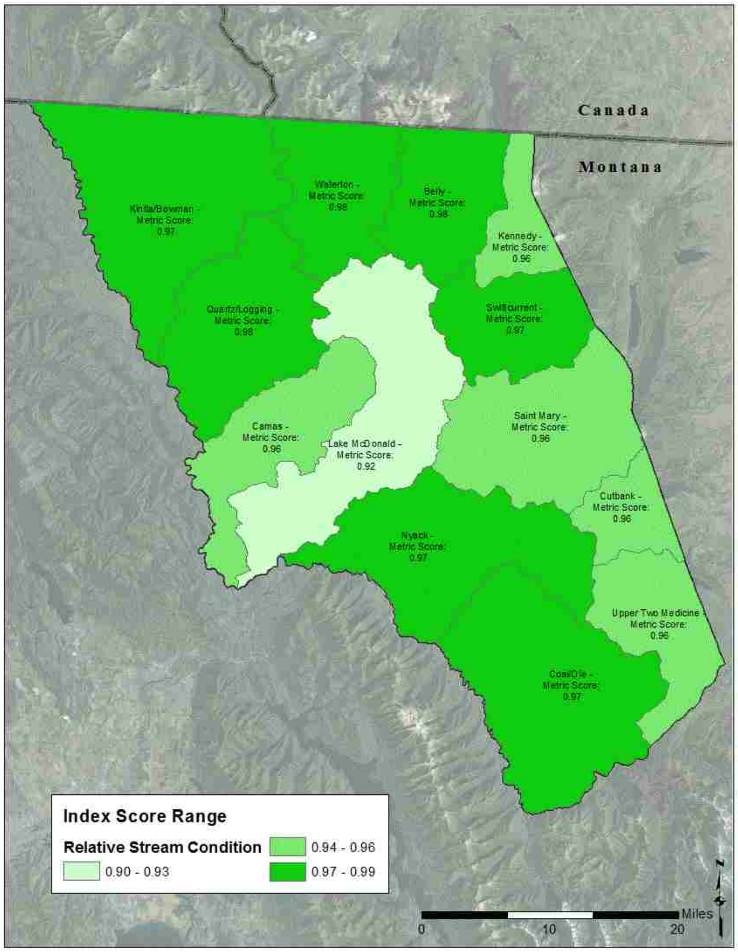

Figure 4.5. Index measuring overall stream conditions in GNP.

4.2 Focal Area - North Fork and Middle Fork of the Flathead River

River drainage networks throughout the Northern Rocky Mountains are an integral part of the landscape mosaic that forms regional patterns of topography, geochemistry, vegetation, and the bio-physical processes that provide the template for ordering biological systems; including the distribution and forms of wetlands on floodplain surfaces. Physical, chemical and biological patterns and processes in river networks are structurally and functionally linked and operate across a hierarchy of spatio-temporal scales. At the landscape scale the river network is intimately linked to longitudinal gradients, floodplain vegetation and processes in and around wetlands, and surface-subsurface water exchange (Stanford and Ward 1993, Jones and Mulholland 2000). The later has a profound effect on floodplain water flux.

The North Fork and Middle Fork of the Flathead River make up the western and southwestern boundary of GNP. These large fifth-order rivers are among a very small suite of large rivers in the conterminous 48 U.S. states that are completely unregulated by dams or diversions. The National Wild and Scenic Rivers System was created by Congress in 1968 (Public Law 90-542; 16 U.S.C. 1271 et seq.) to preserve certain rivers with outstanding natural, cultural, and recreational values in a free-flowing condition for the enjoyment of present and future generations. Both the North Fork and Middle Fork of the Flathead River were added to this designation in 1976 and are part of the National Wild and Scenic Rivers System. The Middle Fork has it headwaters in the Bob Marshall Wilderness Area. As it flows north-by-northwest to its confluence with the North Fork, the Middle Fork emerges from the wilderness complex and encounters the U.S. Highway 2 and Burlington Northern Santa Fe Railroad transportation corridor at Bear Creek at the southwestern tip of GNP. Along the Middle Fork's length where it forms the southwestern border of the GNP to its confluence with the North Fork, it passes through a series of confined and unconfined reaches within a narrow valley. There are only two major floodplain reaches along this section of the Middle Fork; between the town of West Glacier and the confluence of the North Fork and the other at what is known as the Nyack Floodplain.

The North Fork has its headwaters in southeastern British Columbia (BC), Canada. As it flows south-by-southeast to its confluence with the Middle Fork near West Glacier, the North Fork flows through a broad U-shaped valley with expansive alluvial floodplains. The North Fork valley is a major contributor to the biodiversity of GNP and is regarded as one of the wildest rivers in America. However, unlike the Middle Fork with headwaters in wilderness designated area, the North Fork in British Columbia has a 30-40 year history of proposed industrial development in the form of coal mining and coal bed methane (CBM) extraction, oil and gas leases, gold mine prospecting, and phosphate mine prospecting. Recently, these have been part of a Transboundary negotiation and subsequent memorandum of understanding between BC and Montana to ban all mining, CBM or other gas and oil development in the North Fork; BC and Canada have passed protection, however, the negotiations and bills by Montana and the USA are still in process as this report is being developed. Nonetheless, until the headwaters of the North Fork are placed in a permanent protected status with international recognition, the threat to the ecological integrity of the North Fork will remain at significant risk. Please refer to Hauer and Sexton (2010), and Hauer and Muhlfeld (2010) for greater detail into the potential effects of coal mining in Canada on the ecological integrity of the Transboundary North Fork.

The nature and scope of the river-floodplain corridors often changes dramatically from high gradient headwaters to braided middle reaches to meandering lowland sections (Lorang and Hauer 2006). At the landscape spatial scale, the natural state of the North Fork and Middle Fork alluvial river systems are characterized by alternating confined and unconfined valley segments occurring in series along the longitudinal gradient. Confined valley segments are generally characterized by narrow valley walls, near-surface bedrock, absence of a floodplain, and relatively high stream gradient. In unconfined alluvial segments, these rivers flow across deposits of gravel and cobble associated with alluvial floodplains. These reaches commonly have a vertical dimension of groundwater-surface water interaction extending tens of meters into the alluvium and a lateral dimension under the floodplain for hundreds of meters (Stanford et al. 2005).

A fundamental driver of physical, chemical and biological patterns and processes of the river network of these two rivers is the spatial and temporal dimension of flooding and the role of floodplain and floodplain wetlands in the ecological functions along their riverine-corridor ecosystem. The interaction of climate, geomorphology, hydrologic conditions, vegetation, wetlands, river channel complexity and floodplain connectivity affect the intensity, predictability, and duration of floods. In the North and Middle Forks of the Flathead River, the annual hydrograph is dominated by the spring snowmelt period that extends from late March or early April through June.

Ecologically, streams and rivers reflect the legacy of their catchments, their geomorphology, hydrologic and climatic drivers, biogeochemistry, and the complexity of their habitat development. Inorganic and organic materials are transported downstream from erosional zones characterized by confined stream reaches and high gradients to depositional zones characterized by unconfined reaches and relatively low gradients. Thus, the materials are deposited on expansive geomorphic landforms (i.e., floodplains) that have filled the valley with alluvium. As stated by Stanford (1998), "The process of cut (erode) and fill (deposit) alluviation creates the physical features and characteristics of the river corridor." This process, which results in the transport and deposition of bed-sediments, is also critical to maintaining the zones of preferential flow between surface waters and hyporheic groundwaters. The floodplain landforms of the North Fork and Middle Fork river-corridors are viewed correctly when placed in the context of a dynamic mosaic of habitats that transition between saturated and unsaturated conditions in both time and space and act as interconnected patches on the floodplain surface and below ground. Many of these features can be easily recognized on the surface of the floodplain using aerial photographs.

4.2.1 Flathead River Methods

The assessment here is for use in floodplain-wetland complexes where the river is unconfined and has a broad floodplain. These floodplain-wetland complexes are ecologically diverse. Overbank flows scour and deposit sediments and create a shifting mosaic of complex hydrologic habitats such as bars exposed at low flows, secondary channels, sloughs, and backwater ponds. These areas are referred to as the parafluvial. Other areas are inundated less frequently, become stable, and come to be dominated by advanced-stage plant communities. These areas are referred to as the orthofluvial, and are divided into the active orthofluvial, i.e., the area that is annually inundated by overbank flows, and the passive orthofluvial, i.e., the area that is rarely inundated by overbank flows (Hauer and Lamberti 2011). These areas provide a complex environment, resulting in floodplains that consist of integrated wetland/upland complexes with many surface habitats that are ecologically linked to the functioning floodplain wetlands.

To prepare for the analysis of the North and Middle Forks of the Flathead River adjacent to the park, the floodplain was digitized using the GNP stream layer, the digital elevation map (slope threshold greater than 5%), background USDA National Agriculture Imagery Program (NAIP: seamless.usgs.gov), and visual assistance from oblique views within Google Earth's 3-D models. Each assessment area was selected based on continuous floodplain reaches separated by geomorphic constrictions on the valley resulting in thirteen assessment areas (northfork.shp and middlfork.shp). A 0.62-mile (1-km) buffer was applied each assessment area (Figure 4.6) to assess anthropogenic influences adjacent to the floodplain (e.g., roads, ex-urban development, agriculture etc.). This buffer was established solely on best profession judgment. Buffers on both sides of the floodplain assessment site were joined and treated as one assessment area resulting in thirteen buffer assessment sites.

The following is a summary of each variable used in the Flathead River assessment model. Each variable provides the variable code, name, definition, the rationale for selecting and scaling the variable, and the scaled variable in table form.

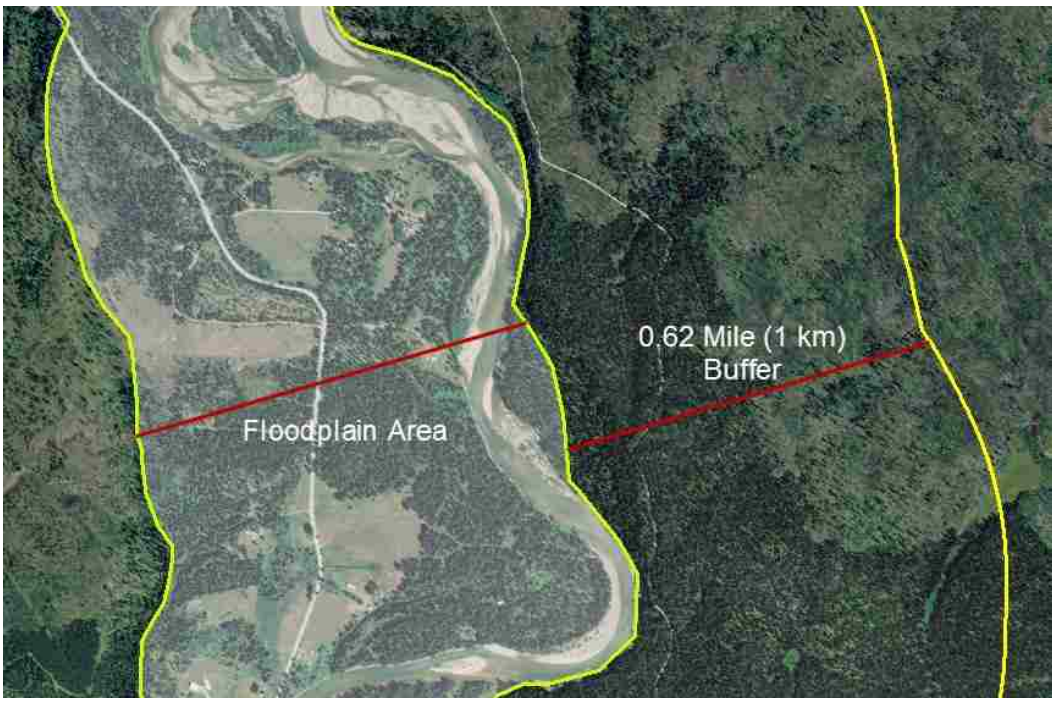

Figure 4.6. Example of selected floodplain area and 0.62 mile buffer on the Flathead River (North Fork).

Several of the metrics for the Flathead River assessment relied on the U.S. Geological Survey's land cover and land use (LULC) remote sensing interpretation (Fry et al. 2009) to provide spatially appropriate data on LULC. For this project I used the 2001 National Land Cover Database (NLCD) from USGS Seamless Data (seamless.usgs.gov), in a 30-m cell size ARCGRID. The raster was converted to polygon for the metric calculations. NLCD was chosen to for ease of comparison to later efforts of Multi-Resolution Land Characteristics Consortium (mrlc.gov) to classify national Landsat coverage. Table 4.10 lists the LULC codes are found within the study area and used in the following assessment variables.

Table 4.10. List of USGS' 2001cover types prevalent among the floodplain-wetland complexes of alluvial gravel-bed rivers of the Southern Rocky Mountains.

USGS LULC Type	Description
11	Open water
21	Developed open space area with <20 percent
22	Developed area with >20-49 percent
23 or 24	Developed areas. >50% impervious areas.
31	Exposed cobble riverbed and secondary channels during base flow and inundated during most annual high flows. (Caution LULC Code 31 also includes gravel pits)
41	Deciduous forest >5 meters tall greater than 20% cover
42	Evergreen forest >5 meters tall greater than 20% cover
43	Mixed deciduous forest and evergreen forest >5 meters tall greater than 20% cover
52	Shrub dominated over 20% shrub cover.
71	Dry herbaceous dominated (shrub cover less than 20%).
81 and 82	Agricultural field, may be a meadow or plowed, often planted and hayed, may have origin as a forested surface, but now logged, or may have been a natural meadow.
90	Woody wetlands shrub or forest greater than 20% cover
95	Moist herbaceous dominated in linear depressions (paleo channels) (shrub cover less than 20%).

4.2.1.1 Buffer ($V_{BUFFROAD}$) and Floodplain (V_{FPROAD}) Roads Density

To evaluate road density, data from Montana transportation data (NRIS 2011) were assessed within the thirteen buffer and floodplain assessment areas. A road density (Total Site Road and Railroad Length (ft)) / (Site Area (acre)) was determined for each buffer assessment area and sub-index scores were derived based on a site's density relative to the highest road density in all the river and buffer assessment areas. The highest road density is 0.0026 feet of road for every acre of assessment area (Site Middle Fork Buffer 1). To create a sub-index score between 0 and 1, the following formula was used:

$V_{BUFFROAD}$ Sub-Index Score = (Maximum Road Density (linear mile/sq. mile) – Assessment Site Road Density (linear mile/sq. mile))/ Maximum Road Density (linear mile/sq. mile)

4.2.1.2 Buffer and Floodplain Development ($V_{BUFFCDN}$ and V_{FPCDN})

The extent of human-altered land cover polygons within an assessment area serves as an indicator of the site's overall anthropogenic stressors. Table 4.9 presents a series of approximate ranges of losses of habitat in the buffer and the floodplain. To calculate the metric score, the relative areas of each polygon within the assessment areas were calculated and multiplied by the weighted sub-score in Table 4.11. As with all metrics in these models, these categorical breaks are not based on actual ecological thresholds. Rather they are best professional judgment of the relative anthropogenic stress from each of the LULC types. The scores were then totaled for each assessment area to obtain the assessment site metric score that ranged between 0 and 1.

4.2.1.3 Floodplain Habitat Connectivity ($V_{FPCONNNECT}$)

Connectivity of floodplain habitat decreases with human disturbance, (e.g., grazing/land clearing, agriculture, and urbanization), and this influences the ability of wide-ranging wildlife to locate, access, utilize, and disperse from a variety of habitat types. In the disturbed conditions, mixed conifer, cottonwood forest, and shrub community cover is significantly reduced and replaced by pasture or domestic or commercial development. V_{FPCON} assesses the amount of woody cover, wetlands and exposed cobble in the floodplain area and was scaled using best professional judgment. The total areas for the appropriate LULC codes within the assessment areas were derived and the scaled scores were applied. The total areas for the 31, 41, 42, 43, 52, 90 and 95 LULC codes (see Table 4.11) within the assessment areas were derived and this total area was divided by the total area of the assessment area which providing a continuous score:

V_{FPCON} Sub-Index Score = Total area of polygons with LULC code of 31, 41, 42, 43, 52, 71, 90, and 95 (acre) / Total Area of Assessment Site (acre)

Table 4.11. Description of land cover and the weighted sub-score assigned to Land Use and Land Cover (LULC) polygons.

Buffer and Floodplain Land Use Criteria	**Weighted Sub-score**
Characteristic plant communities. No grazing, or development beyond walking trails, horse paths, and bike trails. LULC Codes 41, 42, 43, 52, 71, 90, and 95 (11- Open water in Buffer, excluded in Floodplain)	1.0
Characteristic plant communities. May have very light grazing by domesticated animals (e.g., cattle, horses). Minor departure from the characteristic plant community, undisturbed condition across more than 90% of the area of the buffer. Minor departures include LULC codes 31	0.8
Moderate departure from characteristic plant coverage. May have moderate levels of grazing by domesticated animals (e.g., cattle, horses). Undisturbed condition across more than 50% of the area of the buffer. Moderate departures include LULC codes 81 and 82.	0.4
Significant departure from characteristic plant coverage over 75% of the buffer area. May have heavy grazing by domesticated animals (e.g., cattle, horses). May include low density domiciles. Significant departures include LULC codes 21 and 22.	0.3
Highly significant departure from characteristic plant coverage over >75% (most) of the buffer area. LULC codes 23.	0.2
Highly significant departure from characteristic plant coverage over >75% (most) of the buffer area. May include paved parking lots or other major disturbances and concentrations of anthropogenic activities. LULC codes 24.	0.0

4.2.1.4 Flathead River Condition Index

To acquire an overall condition score for each study area on the Flathead River, the following index was applied the condition metrics. The significance metric that measures the condition of the buffer 1 kilometer outside of the floodplain is considered twice as important as the stressor metric that measures road density in the buffer based on professional opinion. Collectively these metrics provide an assessment of the buffer considered as important as the two remaining significance metrics that measure floodplain connectivity and land cover and the remaining stressor metrics that measure the floodplain road density. As with all models in the assessment it provides a score for a watershed derived from the diversity of the habitat and proximity to human activity that is relative to other watersheds in the park only.

$$\text{Index} = (((((V_{BUFFCDN}*2) + V_{BUFFROAD})/3) + V_{FPROAD} + V_{FPCDN} + V_{FPCONNECT})/4)$$

4.2.2 Flathead River Results

A total of 13 reaches were assessed along the approximately 100 river miles of the North and Middle Forks of the Flathead River adjacent to the park. The reaches were selected based on changes in physiographic conditions, predominantly at points where the floodplain areas are confined by the adjacent upland slopes. The linear distance of these reaches ranged from about 4.0 to 15.0 miles in length and averaged about 7.5 miles in length (Table 4.12 and Figure 4.7).

Table 4.12. North and Middle Fork Flathead River assessment reaches and acreages.

Reach Name	Floodplain Site Acreage	Buffer Site Acreage	Site Length (Miles)
Middle Fork 1	826	4565	13
Middle Fork 2	178	3536	10
Middle Fork 3	3892	8009	21
Middle Fork 4	161	2802	8
Middle Fork 5	460	5276	17
Middle Fork 6	330	3540	11
Middle Fork 7	188	3399	10
North Fork 1	1401	5403	9
North Fork 2	538	11129	13
North Fork 3	1077	10975	13
North Fork 4	5225	4674	30
North Fork 5	4618	4264	31
North Fork 6	2331	3368	16

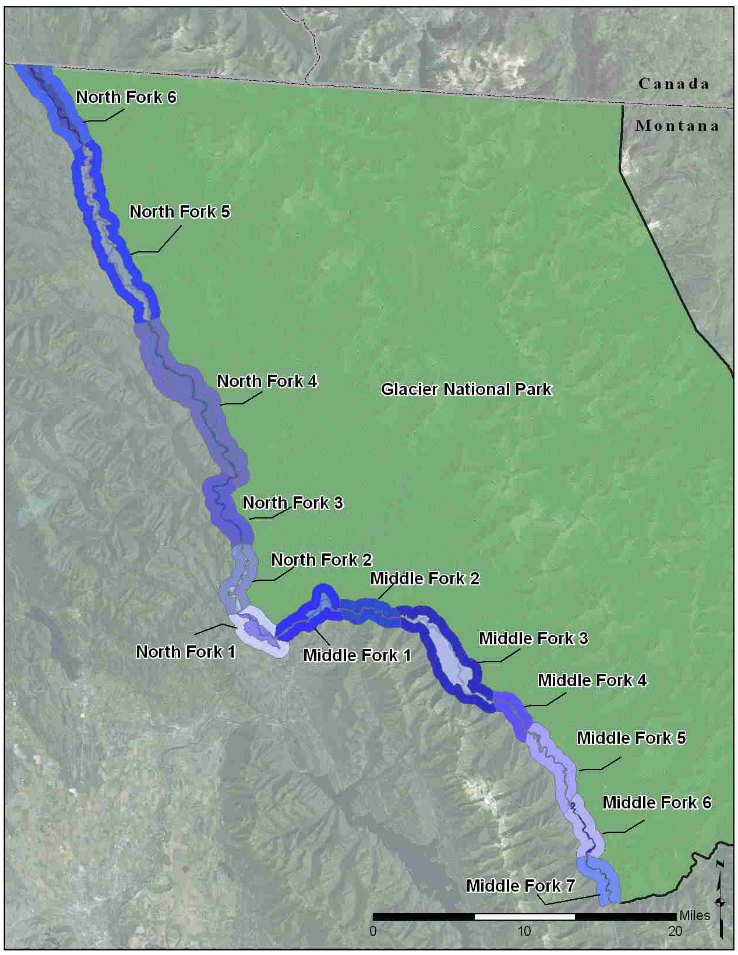

Figure 4.7. Locations of Flathead River reaches and local features.

4.2.2.1 Reach Road Density ($V_{BUFFROAD}$ and V_{FPROAD})

All assessment areas contain a combination of major paved roads, secondary paved and unpaved roads, and/or railroad. These predominantly occur in the areas outside the park. Road density is an indirect measure of the degree of human interaction within the assessment reach (Trombulak and Frissell 2000, Theobald 2003, Nielsen et al. 2004). The lowest score in the buffer assessment area that contains the town of West Glacier (Middle Fork Reach 1) and the lowest score in the floodplain area is the reach upstream from West Glacier (Middle Fork Reach 2). This assessment area contains a confined reach where the Burlington Northern and Santa Fe railroad is directly adjacent to the river (Table 4.13, Figure 4.8, and 5.9).

Table 4.13. Measurements of road density and related metric scores for $V_{BUFFROAD}$ and V_{FPROAD}.

Buffer Name	Buffer Road Density (linear mile/sq. mile)	Buffer Road Density Metric Score	Floodplain Road Density (linear mile/sq. mile)	Floodplain Road Density Metric Score
Middle Fork 1	2.6	0.00	1.5	0.43
Middle Fork 2	1.5	0.43	1.6	0.39
Middle Fork 3	0.6	0.77	0.9	0.65
Middle Fork 4	1.6	0.38	0.0	0.99
Middle Fork 5	1.6	0.37	0.0	1.00
Middle Fork 6	1.6	0.38	0.8	0.70
Middle Fork 7	1.3	0.52	0.5	0.82
North Fork 1	1.2	0.54	0.5	0.83
North Fork 2	1.4	0.45	0.3	0.88
North Fork 3	1.6	0.40	1.2	0.55
North Fork 4	1.2	0.55	0.6	0.78
North Fork 5	1.2	0.55	1.5	0.42
North Fork 6	1.8	0.33	0.4	0.83

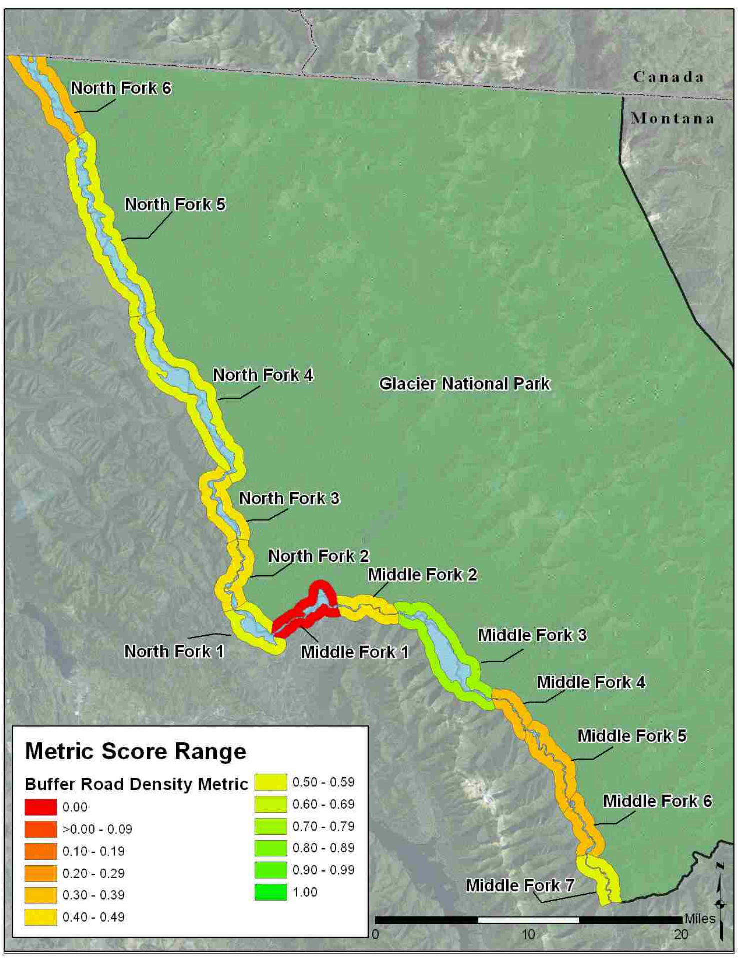

Figure 4.8. Flathead River buffer Road Density metric range.

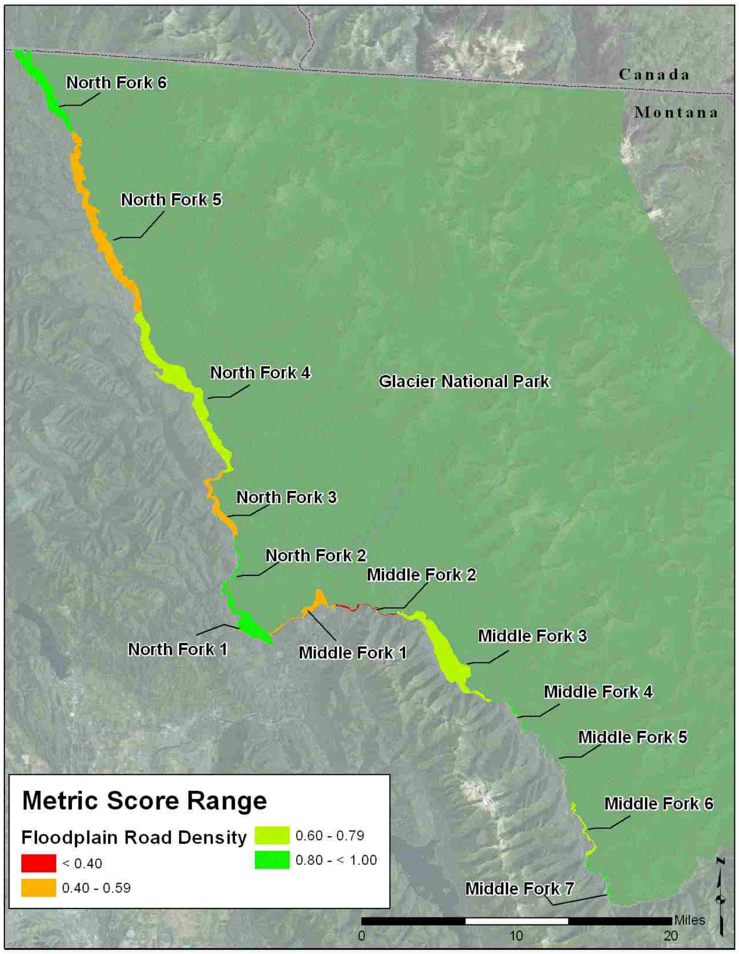

Figure 4.9. Flathead River floodplain road density metric score range.

4.2.2.2 Buffer Land Use Condition ($V_{BUFFCDN}$)

The overall land use in the assessment area is rural and forested with some agriculture use. The buffer area assesses both sides of the North and Middle Forks. Although one half of the buffer area in each assessment reach is within the park, the buffer was assessed as a whole for each reach. In general, the buffer in the study area is fairly intact. However, there are varied land uses in the assessment areas, such as agriculture, housing, and golf courses, which diverge from the native conditions. The lowest score is in the assessment areas that contains the town of West Glacier (Middle Fork Reach 1) (Table 4.14 and Figure 4.10).

Table 4.14. Flathead River buffer land use metric scores with in the study area.

Riverine Reach Name	Buffer Land Use Metric Score
Middle Fork 1	0.94
Middle Fork 2	0.95
Middle Fork 3	0.98
Middle Fork 4	0.96
Middle Fork 5	0.97
Middle Fork 6	0.97
Middle Fork 7	0.97
North Fork 1	0.97
North Fork 2	1.00
North Fork 3	1.00
North Fork 4	0.99
North Fork 5	0.95
North Fork 6	1.00

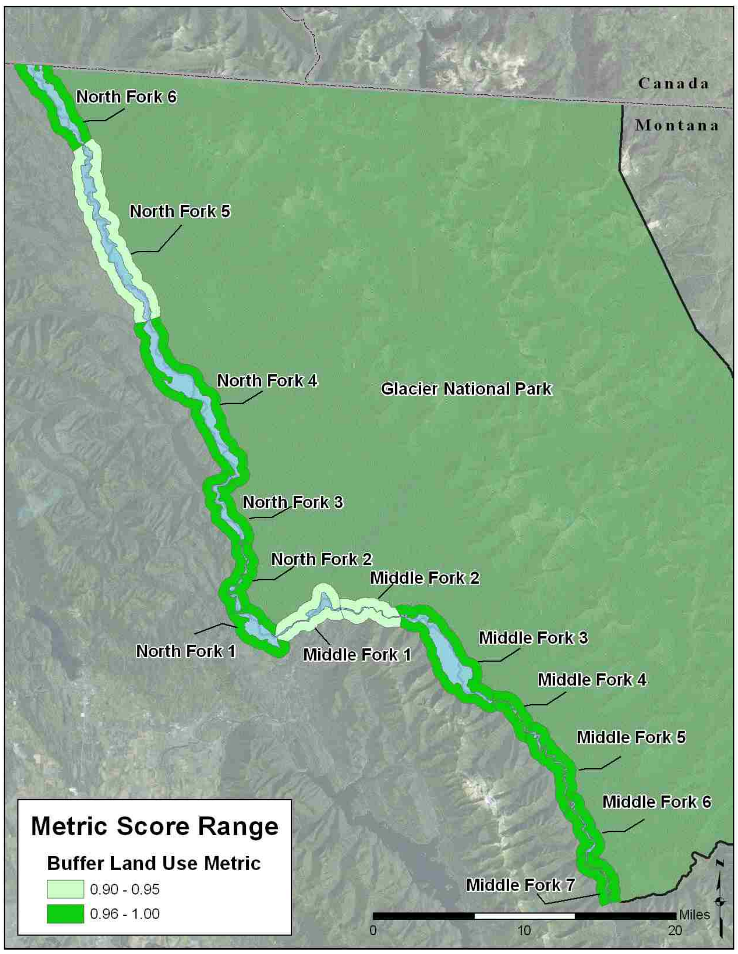

Figure 4.10. Flathead River buffer land Use metric score range.

4.2.2.3 Floodplain Land Use Development (V_{FPCDN})

As with the buffer land use assessment above, the floodplain land use of the Flathead River is relatively intact. There are minor departures from the forested, shrub, and herbaceous communities typically found in healthy floodplain areas. These departures include agricultural fields, roads and urbanized areas. As with the buffer land use, the lowest floodplain land use score is in the assessment area that contains the town of West Glacier (Middle Fork Reach 1) (Table 4.15 and Figure 4.11).

Table 4.15. Flathead River floodplain land use metric scores with in the study area.

Riverine Name	Floodplain Land Use Metric Score
Middle Fork 1	0.77
Middle Fork 2	0.90
Middle Fork 3	0.86
Middle Fork 4	1.00
Middle Fork 5	0.97
Middle Fork 6	0.98
Middle Fork 7	0.96
North Fork 1	0.93
North Fork 2	1.00
North Fork 3	0.96
North Fork 4	0.94
North Fork 5	0.81
North Fork 6	1.00

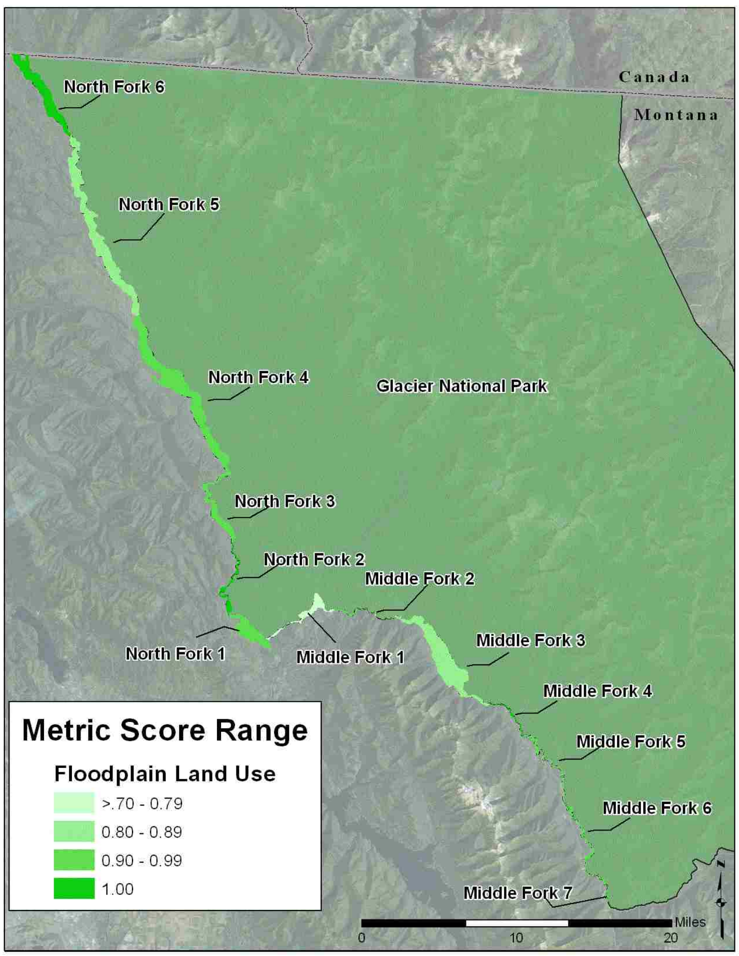

Figure 4.11. Flathead River floodplain land use metric score range.

4.2.2.4 Floodplain Connectivity ($V_{FPCONNECT}$)

The contiguity of habitat patches serves as an indicator of the reach's capacity to function as habitat for wide-ranging wildlife. Unlike the land use condition metric, this metric assesses the composition of remaining native land cover. Within the assessment areas there are departures from a continuous land cover made up of mixed conifer, cottonwood forests, and shrub communities along with forested wetlands and open cobble bar. There are several assessment reaches where these cover class have been replaced with pasture or domestic or commercial development. As with other metrics in this assessment, the lowest floodplain connectivity score is in the assessment area that contains the town of West Glacier (Middle Fork Reach 1) (Table 4.16 and Figure 4.12).

Table 4.16. Measurement or Condition for $V_{FPCONNECT}$

Riverine Name	Percent Cover of 2001 LULC Codes 31-71 and 90	Floodplain Connectivity Metric Score
Middle Fork 1	50%	0.50
Middle Fork 2	77%	0.77
Middle Fork 3	72%	0.72
Middle Fork 4	81%	0.81
Middle Fork 5	81%	0.81
Middle Fork 6	87%	0.87
Middle Fork 7	89%	0.89
North Fork 1	78%	0.78
North Fork 2	80%	0.80
North Fork 3	75%	0.75
North Fork 4	82%	0.82
North Fork 5	59%	0.59
North Fork 6	95%	0.95

4.2.2.5 Flathead River Condition Score

The Flathead River condition scores represent a combination of significance metrics that measure the range of natural vegetation patch connectivity and four stressor metrics that measure human alterations within the floodplain area and buffers of the Middle and North Forks of the Flathead River. For instance, Middle Fork Reach 2 is a confined channel with a limited floodplain and thereby limited habitat diversity. Middle Fork Reach 1 has a broad floodplain but it has been impacted by the urban activities of the town of West Glacier. Both of the scores of these two reaches represent the departure from floodplain conditions that provide a diverse native habitat. In the North Fork, Reaches 2 and 3 scored the lowest due high road densities in the buffer and floodplain and low wetland density in both reaches. All scores represent a departure from an unaltered floodplain condition resulting from concentrated human use (Table 4.17 and Figure 4.13).

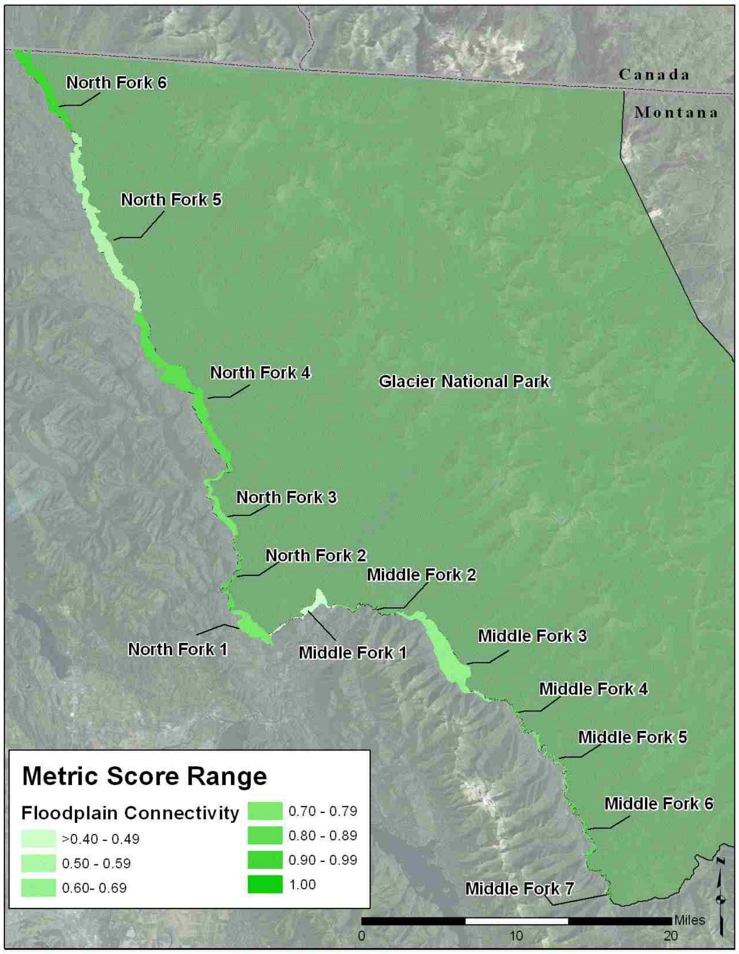

Figure 4.12. Flathead River floodplain connectivity metric score range.

Table 4.17. The metric and index scores for each floodplain assessment area in Glacier National Park.

Floodplain Reach Name	Buffer		Floodplain			
	land Use	Road Density	Land Use	Connectivity	Road Density	Index
Middle Fork 1	0.94	0.00	0.77	0.50	0.43	0.58
Middle Fork 2	0.95	0.43	0.90	0.77	0.39	0.71
Middle Fork 3	0.98	0.77	0.86	0.72	0.65	0.79
Middle Fork 4	0.96	0.38	1.00	0.81	0.99	0.89
Middle Fork 5	0.97	0.37	0.97	0.81	1.00	0.89
Middle Fork 6	0.97	0.38	0.98	0.87	0.70	0.83
Middle Fork 7	0.97	0.52	0.96	0.89	0.82	0.87
North Fork 1	0.97	0.54	0.93	0.78	0.83	0.84
North Fork 2	1.00	0.45	1.00	0.80	0.88	0.87
North Fork 3	1.00	0.40	0.96	0.75	0.55	0.77
North Fork 4	0.99	0.55	0.94	0.82	0.78	0.85
North Fork 5	0.95	0.55	0.81	0.59	0.42	0.66
North Fork 6	1.00	0.33	1.00	0.95	0.83	0.89

4.2.3 Assessment of availability and gaps in monitoring data

Landsat based thematic land cover products and other data spatially appropriate for assessments at this scale generally follow standardized reporting guidelines that articulate known uncertainties inherent in their efforts (U.S. Bureau of Budgets 1947, Anderson et al. 1976). The NLCD map accuracy for the Rocky Mountain Region is 79% for 2001 for Anderson Level II classification (Wickham et al. 2010). Uncertainty is a known degree of unreliability of knowledge ranging from certainty (determinism) to total ignorance or a lack of awareness that knowledge is wrong or imperfect. The position along this range translates into a state-of-confidence (Walker et al. 2003). Further research is needed to determine the impact on multi-metric index scores from error propagation resulting from known uncertainties such as the accuracy of the 2001 NLCD.

Wetlands were not addressed in the section because of the limited data. National Wetland Inventory (NWI) data was used as an important metric in the Flathead River assessment; however, like other nationwide surveys of natural resources it is important to be cautious of the continuity and extent of this coverage. It is generally recognized that NWI has varying accuracy (Stolt and Baker 1995). Evaluations of NWI have reported various error rates in studies performed around the country, for example very low, less than 5%, omission and commission error rates were reported in Massachusetts (Swartwout 1982) and Michigan (Kudray and Gale 2000). However, in other studies omission error rates were found to be much higher; about 50% omission was found in a Nebraskan study (Kuzila et al. 1991), and greater than 85% in Virginia (Stolt and Baker 1995). In general, omission tends to be a common bias with NWI data (Tiner 1997).

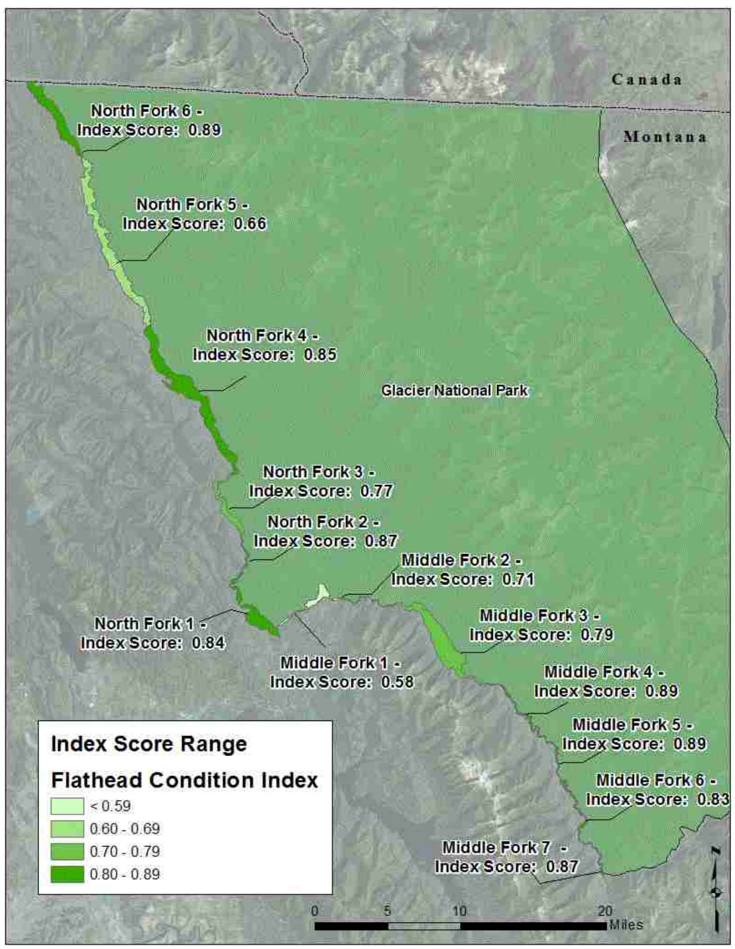

Figure 4.13 Flathead River condition index score range.

4.3 Focal Area – Glacier's Lakes

The lakes of the park were formed through glacial activity. At its maximum, 10,000–20,000 years BP, the last Wisconsin glaciation covered about ¼ of the world's land area. Glacial lobes from the Cordilleran Ice Sheet extended down the Rocky Mountain Trench (a tectonic fault block basin) covering the western region of GNP with subsequent alpine glaciers reaching into the eastern region of the park. Advancing glaciers scraped the land, pushing rock and earth, scouring deep basins and depositing terminal moraines that eventually became glacial lakes. As the Cordilleran Ice Sheet retreated, further reworking of the landscape occurred leaving behind layers of stones and fine particles of variable thickness on top of the underlying rock (glacial till). Soils today developed from the physical and biological/chemical weathering of the drift layer and the subsequent incorporation of organic matter. In regions underlain by extremely hard rock and little overlying glacial drift, the glaciers scraped the bedrock, creating shallow basins now occupied by lakes, ponds and wetlands. Through these glacial and tectonic processes a variety of lake types (Hutchinson 1957) were formed.

High-elevation lakes are ice covered in winter (some are ice free for only a few months) and are hydrologically dominated by snow melt and some by both glacial and snow melt. Located in drainage basins with low average air temperatures, minimal vegetation and poorly developed soils, most high-elevation lakes (particularly alpine lakes) tend to be nutrient poor. However, the alpine and subalpine lakes of GNP vary in their chemistry, productivity and biotic communities. Differences in the extent of vegetation and soil development, bedrock chemistry, climate, ratio of drainage area to lake volume and biotic history all play a role in present day lake chemistry and productivity.

Most of the valley bottom lakes are a result of glacial erosion and subsequent deposition of lateral and terminal moraines in deepened and widened tectonic fault-block valleys. In general, valley bottom lakes are more productive than higher elevation lakes due to the larger ratio of drainage area to lake volume, greater soil and vegetation development, higher supply rates of all major and minor nutrients (longer contact time between water and soils), higher temperatures and longer growing season.

In order to establish a water quality baseline for select lakes in GNP, the National Park Service (NPS) and the Flathead Lake Biological Station (FLBS) documented the annual variability in water chemistry, physical characteristics and plankton communities of a subset of the park's lakes from 1984 to 1990 (Ellis et al. 1992, 2002). Five valley bottom lakes and eight alpine and subalpine lakes were monitored. The majority of lakes sampled in this study were strongly phosphorus (P) limited. That is, there is a paucity of P relative to nitrogen (N) and production of phytoplankton must be limited by the input of P (Wetzel 2001). An increase in the atmospheric deposition of P could cause an immediate stimulus of autotrophic productivity (Ellis and Stanford 1988a, 1988b) and secondarily alter the food web of these lakes through the process of eutrophication.

Bergstrom (2010) showed that the dissolved inorganic nitrogen (DIN) to total P ratio was a better indicator than the ratio of total N (TN) to total P (TP) for determining N and P limitation of phytoplankton. Ratios of DIN:TP for several of the high elevation lakes and one low elevation lake studied by (Ellis et al. 1992, 2002) were borderline between P limitation and N limitation. It is possible that these lakes may be co-limited by both N and P. Studies have reported co-limitation of phytoplankton growth by P and N in nearby Flathead Lake (Spencer and Ellis 1990). The increase in atmospheric deposition of ammonium and nitrate in the northwest U.S.

(Lehmann et al. 2005) has the potential to increase algal production in some of the high-elevation lakes limited or co-limited by N. Several of the high-elevation lakes and one valley bottom lake were very soft-water systems (low conductivity) reflecting the lack of bicarbonate-rich limestone formations within the Belt Series geology of GNP. While they are not the most dilute lakes in the world (Eilers et al. 1990), they clearly have very little buffering capacity and would be more sensitive to acidic precipitation than lakes influenced by more carbonate-rich facies of the Belt Series. All of the lakes were oligotrophic or ultra-oligotrophic.

Since the early limnological studies of GNP lakes, an analysis of the concentrations and biological effects of airborne contaminants in air, snow, water, sediments, lichens, conifer needles and fish in two watersheds in GNP has been conducted (Landers et al. 2008; seven other national parks in the western U.S. were included in this study). Semi-volatile organic compounds (SOCs) and heavy metals (e.g., Hg) were the primary focus of the study. The sediment of both lakes contained SOCs, polycyclic aromatic hydrocarbons, Pb, Cd and Hg. Sediment profiles indicated SOCs have not decreased since use ceased, but most of the other contaminants decreased about the time reductions in emissions were required by the Clean Air Act. Numerous pesticides and Hg were detected in fish from the lakes. These contaminants are of major concern for the health of the lakes, particularly the biota.

The geologic and elevation gradients probably had the greatest natural influence on biotic assemblages and the introduction of fish in alpine lakes has likely produced the most measurable effects in relation to other potential pollutants (Ellis et al. 2002). Detrended correspondence analysis showed that the environmental gradient in geology among the lake watersheds exhibited the greatest strength in accounting for the variation in the phytoplankton community. However, upon examination of taxonomic groups of both phytoplankton and zooplankton, significant differences were observed in lakes with fish versus lakes without fish. The biomass of phytoplankton, grouped by class, was significantly different in lakes with fish. The biomass of Cryptophyceae, Xanthophyceae and Bacillariophyceae were all significantly lower in fishless lakes than in lakes with fish. These differences may reflect variable grazing of the phytoplankton community due to differences in zooplankton species present in lakes with fish versus fishless lakes. Lakes containing fish did not have many large zooplankton (i.e., copepods and cladocera) and the community was usually dominated by the smaller rotifers. Large zooplankton, particularly the red-bodied *Hesperodiaptomus shoshone*, were always present in the fishless lakes. Clearly, grazing by fish has an effect on the pelagic food web.

From the time of the parks establishment through the early 1970s, large numbers of non-native fish were planted across the park, and in some cases established self-sustaining reproducing populations. Introductions or invasions of nonnative organisms can result in major changes in the trophic structure of aquatic ecosystems, often altering the abundance, biomass or productivity of a population, community or trophic level across more than one link in the food web (Carpenter et al. 1985). The purposeful introduction of 20 vertebrate and invertebrate species to nearby Flathead Lake (a large downstream lake in the Flathead watershed) over the last century resulted in a trophic cascade affecting the phytoplankton and zooplankton communities, planktivorous fishes, piscivorous fishes and even terrestrial bald eagles (Ellis et al. 2011). This resulting alteration of the entire food web of Flathead Lake extended into GNP resulting in the loss of nonnative kokanee salmon from their primary spawning grounds (McDonald Creek), the dispersal of the large fall congregation of bald eagles that fed on the kokanee, and more importantly, the dramatic increase in an additional nonnative top predator (i.e., lake trout) which is invading numerous lakes and streams within GNP.

4.3.1 Methods: Glacier Lakes

4.3.1.1 Acid Sensitivity ($V_{ACID\text{-}SEN}$)

The most sensitive measure of water's ability to buffer acidic atmospheric inputs is acid neutralizing capacity (ANC). Nanus et al. (2009) conducted an assessment of the sensitivity of lakes to acidic aerosol deposition in five Rocky Mountain national parks, including GNP. Utilizing lake basin characteristics and ANC measurements, they calibrated statistical models to predict which lakes had a high probability for sensitivity to acidic deposition. Thirty-three lakes were sampled in GNP and three had ANC concentrations < 50 ueq L, three within a range of 100−200 ueq L and the remaining lakes were > 200 ueq L (see Fig. 2, Nanus et al. 2009). Utilizing data from all five national park lakes (Nanus et al. 2009) found that lakes most likely to be sensitive to acidic deposition are located in basins with elevations >3000 m, with >80% of the catchment bedrock having low buffering capacity and with <30% of the catchment having a northeast aspect.

Alkalinity also approximates the ability of surface waters to neutralize acidity. The mean alkalinity for five valley bottom lakes and eight alpine and subalpine lakes in GNP from annual collections for the period 1984−1990 (Ellis et al. 1992, 2002) in relation to the percent of a lake's contributing watershed containing argillite plus quartzite is shown in Figure 4.14. This figure suggests that as the percent of argillite plus quartzite in a lake's contributing watershed increases, the alkalinity of the lake decreases and thereby the lake's ability to buffer acidic aerosol deposition also decreases.

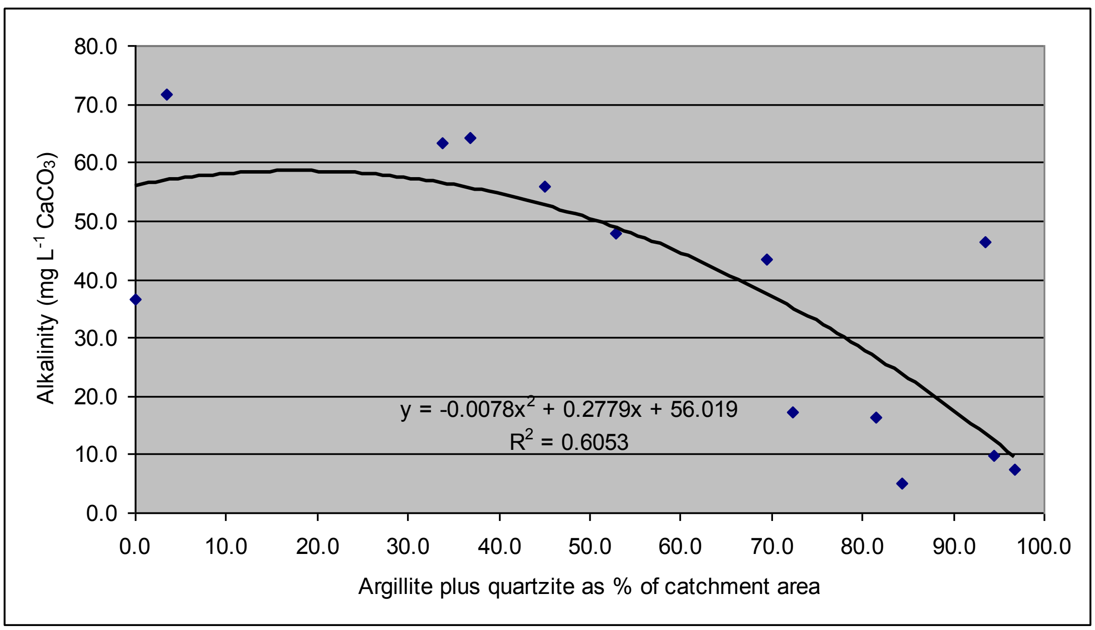

Figure 4.14. Approximate relation between the percent of contributing watershed comprised of argillite plus quartzite and its relative acid buffering capacity.

The percent of argillite plus quartzite in a lake's contributing watershed can be calculated with the park's existing data. The lake basin areas were calculated by joining all catchments above all lakes from the National Hydrological Database (NHD) catchment shape files (catchments.shp). This layer was then intersected with the soils layer (soils.shp) to express the percent coverage of quartzite plus argillite bedrock within each lake basin. A union of these basins with the lakes spatial layer determined the percent argillite plus quartzite for each lake's contributing watershed.

This percentage was used as a metric for the sensitivity to potential acid risks in the park's lakes. Lakes with ANC < 100 ueq L are considered to be very sensitive to acidic deposition (Williams and Tonnessen 2000). Lakes with ANC values less than 100 ueq/L or alkalinity values less than 5 mg/L CaCO3 (about 85% argillite plus quartzite in the contributing basin) tend to be more susceptible to acidification. The metric was scored according to this percentage of the contributing watershed that contained argillite and quartzite associated with levels of alkalinity (Figure 4.14 and Table 4.18).

Table 4.18. Acid Sensitivity ($V_{ACID\text{-}SEN}$) metric scoring.

Acid Sensitivity Metric Criteria:	Metric Score
Percent of contributing watershed comprising argillite plus quartzite < 85 %	1.00
Percent argillite plus quartzite 85% - < 100 %	0.50

4.3.1.2 Enhanced Algal Production (V_{ALGAE})

As stated above, the dissolved inorganic nitrogen (DIN) to total phosphorus (TP) ratio is a good indicator for determining whether N or P (or both N and P) would stimulate algal growth, thus trending toward more productive conditions. These data are only available for a few of the park's lakes and are non-representative of the entire park, but because this is spatially explicit data that can be augmented through increased monitoring, it was used as a metric. The scaling (Table 4.19) was based on Bergström (2010), which states that lakes with a DIN:TP ratios greater 3.4 have a high probability of P limitation, while a ratio below 1.5 indicates N limitation. The paper also states that phytoplankton shift from N to P limitation when DIN:TP ratios increase from 1.5 to 3.4. It is feasible that some lakes within that shifting range may be co-limited by both N and P, that is, both nutrients would stimulate algal growth.

Although lakes at both ends of the DIN:TP spectrum are at risk of increased algal production, the metric scores (Table 4.19) were based upon the potential for increased algal growth from additional N inputs due to the well documented increase in atmospheric deposition of ammonium and nitrate in the northwest U.S. (Lehmann et al. 2005). However, the metric could also be designed for increasing P inputs (i.e., DIN:TP ratios >3.4 at high risk), should such a trend eventuate. Clearly, most of the GNP lakes would be degraded by additional inputs of P (see Table 4.21). The metric scores in Table 4.21 were assigned to each lake watershed. If no data were available for the DIN:TP ratio, the lake received a score of 1.0 for the time-being until more data is available. Each lake area was then multiplied by its assigned sub-metric score and this figure was totaled for each watershed and divided by the total area of all lakes in the watershed resulting in a final metric score between 0 and 1.

Table 4.19. Enhanced Algal Production (V_{ALGEA}) metric scoring.

Metric Criteria:	Sub-Metric Score
Dissolved inorganic nitrogen to total phosphorus ratio >= 3.4	1.00
Dissolved inorganic nitrogen to total phosphorus ratio 2.5 - <3.4	0.80
Dissolved inorganic nitrogen to total phosphorus ratio 1.5 - <2.5	0.50
Dissolved inorganic nitrogen to total phosphorus ratio <1.5	0.01

4.3.1.3 Risk of Invasive Diatoms, Mollusks and Aquatic Macrophytes (V_{EXOTIC})

Boats are a primary potential source of invasive aquatic species to the lakes of the northern Rocky Mountains. These invasive species include plants; Eurasian water milfoil (*Myriophyllum* *spicatum*) and purple loosestrife (*Lythrum salicaria*), invasive invertebrates; zebra mussel (*Dreissena polymorpha*), Quagga mussel (*Dreissena bugensis*), New Zealand mud snail (*Potamopyrgus antipodarum*); and the invasive diatom didymo (*Didymosphenia geminata*). This metric addresses the potential risk of exposure of the park's lakes to nuisance aquatic species, the extent of non-native salmonid fish species that are already in the park's lakes are addressed separately in the fish section below. Proximity of the park's lakes to boat ramps, paved roads and unpaved roads was used as a measure of the potential risk and is scaled in qualitative categories in Table 4.20. This score represents potential risk of invasion and expansion of these species, not actual measurements of such. As with the metric above, the metric scores in Table 4.20 were assigned to each lake watershed. Each lake area was then multiplied by its assigned sub-metric score and this figure was totaled for each watershed and divided by the total area of all lakes in the watershed resulting in a final metric score between 0 and 1.

Table 4.20. Invasive mollusks and aquatic macrophytes metric scoring.

Metric Criteria:	Metric Score
No road within 300 feet of lake.	1.00
Unpaved Road adjacent to lake with no formal boat ramp	0.90
Paved Road adjacent to lake with no formal boat ramp	0.80
Unpaved Road adjacent to lake with formal boat ramp	0.70
Paved Road adjacent to lake with formal boat ramp	0.60
Nuisance aquatic species are present	0.01

4.3.1.4 Calculation of Total Lake Condition Score

To acquire an overall lake condition score for each watershed, the following index was applied the condition metrics. This model assesses the risk to GNP lakes that could potentially degrade their ecological condition. It is constructed to measure the risk to water chemistry equally and these collectively have an influence on lakes, but the multiplicative aspect of the model indicates that exposure to nuisance aquatic species will severally degrade a lake's conditions. Because the lakes in GNP are not currently acidified, have excessive algae growth or invasive species, this index measures only the risk of these degradations occurring.

$$\text{Lake Condition Score} = (((V_{ACID} + V_{ALGAE})/2) * V_{EXOTIC})$$

4.3.2 Lake Condition Results

4.3.2.1 Enhanced Algal Production (V_{ALGAE})

Data on the dissolved inorganic nitrogen to total phosphorus ratio is only available for 13 lakes in the park. Four of these 13 lakes have a ratio that put these lakes at risk of enhanced algal production with increasing nitrogen deposition (Table 4.21). As a result the watersheds containing these lakes scored lower than others in the park (Table 4.22 and Figure 4.15). If data was not available it is assumed that the lake has a DIN:TP ratio >3.4 until further data is available.

Table 4.21. DIN:TP ratios for GNP lakes where data was available (Ellis et al. 1992).

Lake	Watershed	DIN:TP Ratio
Upper Dutch	Camas	2.0
Cobalt	Upper Two Medicine	2.2
Beaver Woman	Coal/Ole	2.4
Stoney Indian	Waterton	2.5
Two Medicine	Upper Two Medicine	2.6
Medicine Grizzly	Cut Bank	2.9
Gyrfalcon	Quartz/Logging	5.80
Gunsight	Saint Mary	5.80
Snyder Lakes 2	Lake McDonald	6.50
Swiftcurrent	Swiftcurrent	8.80
St. Mary	St. Mary	23.80
Waterton	Waterton	36.90
McDonald	Lake McDonald	47.30

Table 4.22. Watersheds within GNP that contain lakes sensitive to increasing nitrogen which would result in enhanced algal production.

	The 4 categories of DIN:TP ratios[1] found in sample lake contributing basins and the percent of lakes in those categories.				
Watershed	**<1.5**	**1.5 - <2.0**	**2.0 - <3.5**	**>3.4**	**Metric Score Range**
Belly	0%	0%	0%	100%	1.00
Camas	0%	0%	100%	0%	0.99
Coal/Ole	0%	0%	100%	0%	0.98
Cut Bank	0%	0%	100%	0%	0.96
Kennedy	0%	0%	0%	100%	1.00
Kintla/Bowman	0%	0%	0%	100%	1.00
Lake McDonald	0%	0%	0%	100%	1.00
Nyack	0%	0%	0%	100%	1.00
Quartz/Logging	0%	0%	0%	100%	1.00
Saint Mary	0%	0%	0%	100%	1.00
Swiftcurrent	0%	0%	0%	100%	1.00
Upper Two Medicine	0%	0%	100%	0%	0.92
Waterton	0%	0%	50%	50%	1.00

1. If the DIN:TP data were not available for a lake, it is assumed that the ratio was >3.4 until detailed data can be provided.

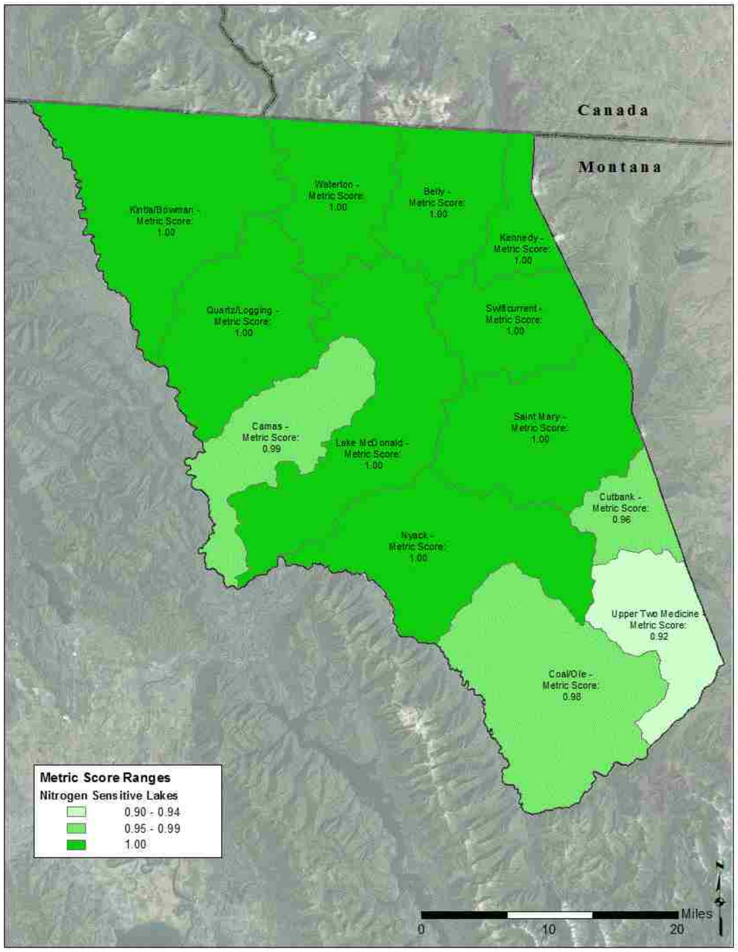

Figure 4.15. Metric for lakes in GNP that are potentially at risk of enhanced algal production.

4.3.2.2 Acid Sensitivity ($V_{ACID\text{-}SEN}$)

The percent of argillite plus quartzite in each lake's contributing watershed was determined a sub-index score was assigned to each lake above and below 85% (see Table 4.18). The area of each lake was multiplied by the sub-index score, totaled for the watershed and divided by the total lake area in that watershed. The subsequent weighted average provided the watershed scale metric score. From these lake assessments, it was found that lake systems in the Cut Bank and Camus watersheds have a higher sensitivity to acid deposition then other watersheds in the park (Table 4.23 and Figure 4.16).

4.3.2.3 Risk of Invasive Diatoms, Mollusks and Aquatic Macrophytes (V_{EXOTIC})

Several of the larger lakes within the park either have a boat ramp on the lake or are adjacent to a paved road. As a result Saint Mary and Lake McDonald watersheds have the highest risk of invasion by nuisance aquatic species (Table 4.24 and Figure 4.17).

Table 4.23. Percent watershed that is comprised of argillite and quartzite.

Watershed	Greater Than or Equal To 85%	Less Than 85 %	Metric Score Range
Belly	0%	100%	0.98
Camas	7%	93%	0.74
Coal/Ole	13%	87%	0.89
Cut Bank	58%	42%	0.74
Kennedy	29%	71%	0.84
Kintla/Bowman	1%	99%	0.99
Lake McDonald	0%	100%	0.99
Nyack	6%	94%	0.84
Quartz/Logging	0%	100%	1.00
Saint Mary	2%	98%	0.99
Swiftcurrent	2%	98%	0.98
Upper Two Medicine	8%	92%	0.89
Waterton	1%	99%	0.99

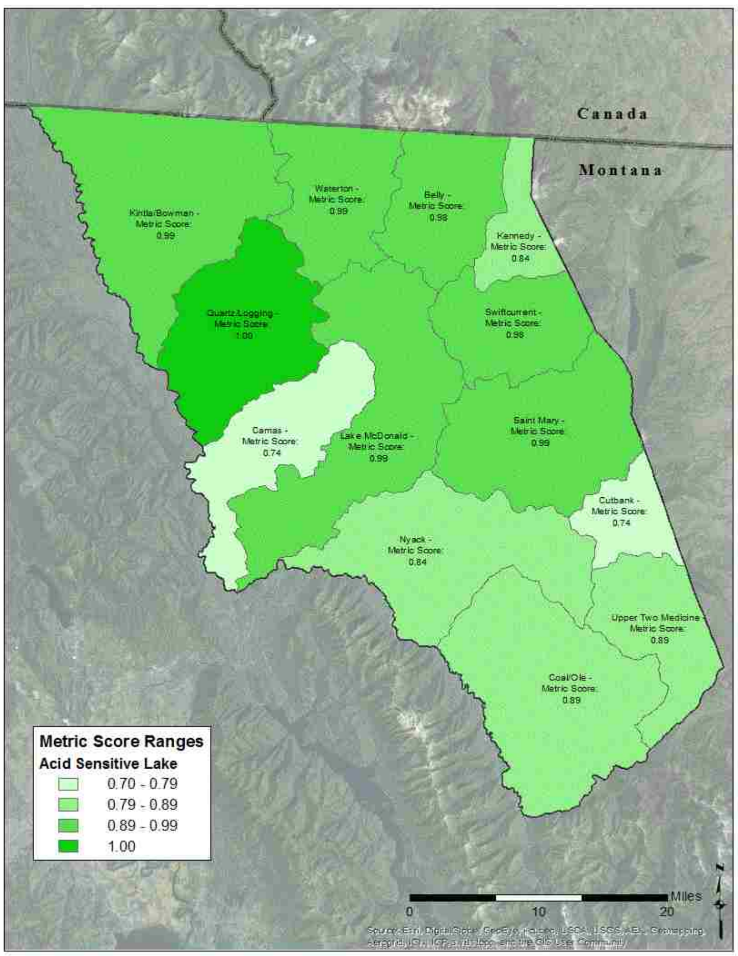

Figure 4.16. Metric for lakes in GNP that are potentially threatened by acidification.

Table 4.24. Percent of lake area with various types of boat access.

Watershed	Paved Road w/ Ramps	Unpaved Road w/ Ramps	Paved Road w/no Ramps	Unpaved Road w/no Ramps	No Road	Metric Score
Belly	0%	0%	0%	0%	100%	1.00
Camas	0%	0%	0%	0%	100%	1.00
Coal/Ole	0%	0%	0%	0%	100%	1.00
Cut Bank	0%	0%	0%	0%	100%	1.00
Kennedy	0%	0%	0%	0%	100%	1.00
Kintla/Bowman	0%	80%	0%	1%	19%	0.76
Lake McDonald	94%	0%	0%	0%	6%	0.63
Nyack	0%	0%	0%	0%	100%	1.00
Quartz/Logging	0%	0%	0%	0%	100%	1.00
Saint Mary	87%	0%	0%	0%	13%	0.65
Swiftcurrent	0%	5%	65%	0%	31%	0.86
Upper Two Medicine	60%	0%	1%	0%	40%	0.76
Waterton	56%	0%	0%	0%	44%	0.78

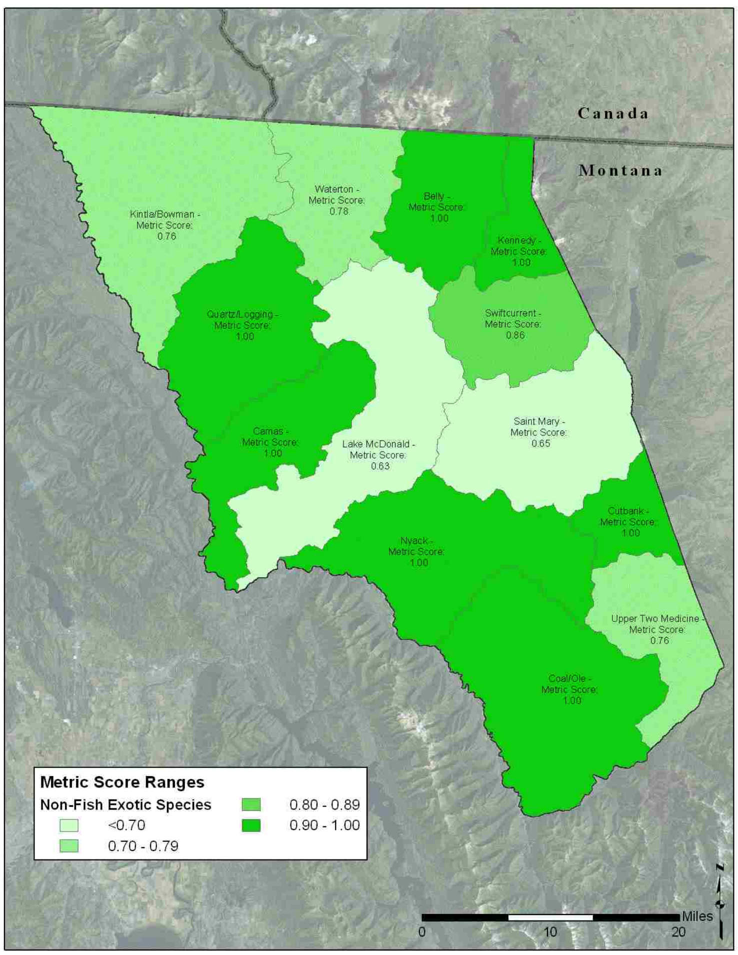

Figure 4.17. Metric for lakes in GNP that are potentially threatened by exotic species.

4.3.3 GNP Lake Condition Score

The GNP lake condition (risk) scores represent a combination of risk metrics that measure the range of natural variability of attributes that buffer lakes from potential acidification or increased eutrophication and potential human degradation resulting from the introduction of aquatic nuisance species. Lake McDonald, Saint Mary and Upper Two Medicine watersheds scored the lowest (i.e., at highest risk) primarily because of potential exposure to non-native aquatic species (Table 4.25 and Figure 4.18).

Table 4.25. The lake metric and index scores in each watershed in Glacier National Park.

Watershed	Acid Sensitive Lakes	Nitrogen Sensitive Lake	Exotic Species (Non-Fish) Risk	Lake Condition Index
Belly	0.98	1.00	1.00	0.99
Camas	0.74	0.99	1.00	0.87
Coal/Ole	0.89	0.98	1.00	0.94
Cut Bank	0.74	0.96	1.00	0.85
Kennedy	0.84	1.00	1.00	0.92
Kintla/Bowman	0.99	1.00	0.76	0.76
Lake McDonald	0.99	1.00	0.63	0.63
Nyack	0.84	1.00	1.00	0.92
Quartz/Logging	1.00	1.00	1.00	1.00
Saint Mary	0.99	1.00	0.65	0.65
Swiftcurrent	0.98	1.00	0.86	0.85
Upper Two Medicine	0.89	0.92	0.76	0.69
Waterton	0.99	1.00	0.78	0.78

4.3.4 Assessment of availability and gaps in monitoring data

Data is limited on the DIN:TP ratio and the alkalinity (or preferably acid neutralizing capacity) and other basic water chemistry of GNP lakes. Increased monitoring of the DIN: TP ratio would provide the data necessary to refine the enhanced algal production metric. Determination of acid neutralizing capacity of high and low elevation GNP lakes within each watershed would provide a more precise measurement of the sensitivity of lakes to acidic precipitation. The additional data from Nanus et al. (2009) should be incorporated into the acid sensitivity metric; however at the time of this production, those data were not available. Analysis of the lake condition parameters $V_{ACID\text{-}SEN}$ and V_{ALGAE} could be improved by assessing risk within subwatersheds as the larger lake watersheds transect varying parent material, soils, forest cover and other inherent characteristics that influence those parameters. Additionally, a finer resolution of lake catchment information would refine the acid sensitivity metric.

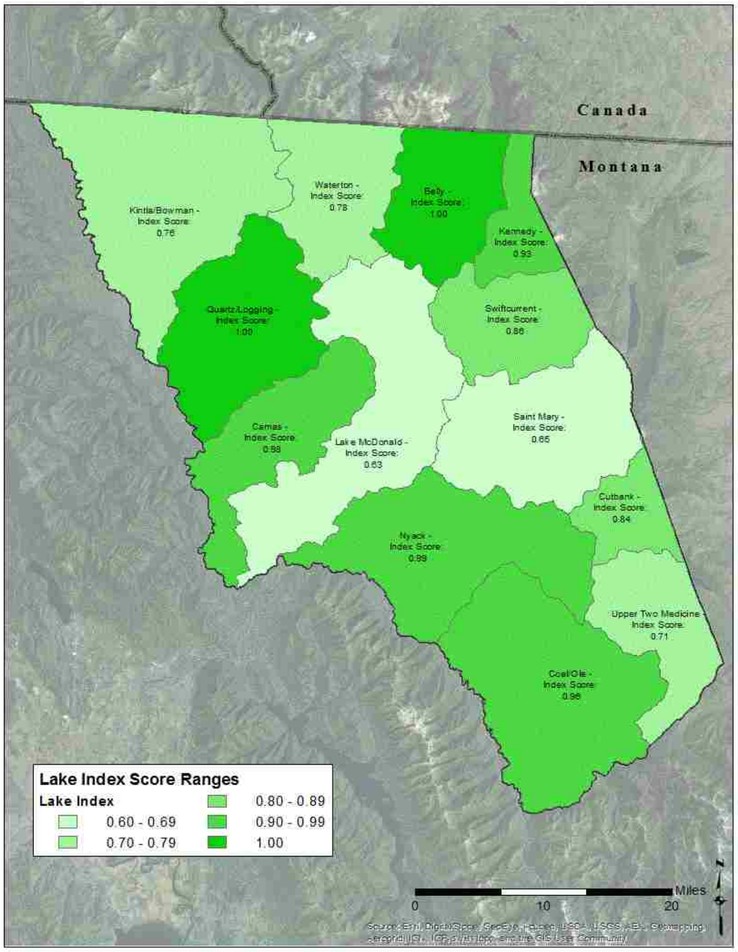

Figure 4.18. Index for lake condition in GNP.

4.4 Focal Area – Native and Invasive Fish Populations

Glacier National Park supports 713 lakes ranging in size from fractions of acres up to Lake McDonald covering about 6,900 acres, and greater than 250 miles of stream habitat for aquatic species (GNP GIS data). A diversity of native and introduced fish species occur in park waters (Table 4.26). However, there is limited historic (Read et al. 1982, Weaver et al. 1983) data to base precise native fish distributions or abundance estimates. Most of the effort in GNP has been focused in the North and Middle Forks of the Flathead River and their tributaries in the park. More recently, significant effort has been focused on describing the distribution of bull trout in the St. Mary River drainage on the east side of the park. In the Flathead River Basin (Columbia Drainage) Flathead Lake and the Flathead River upstream of the lake, including the connected and accessible headwater streams and lakes (i.e., in the park and the Bob Marshall and Great Bear Wilderness), historically functioned as an interconnected watershed for migratory fish. Early in the 20th century, much of the interest in fishery resources related to "improving" the existing fishery by introducing native and non-native fish to historically fishless lakes (e.g., Yellowstone cutthroat trout to Hidden Lake) and introducing non-native species to lakes well populated with fish, but considered to have too few species in the community (e.g., lake trout, lake whitefish, kokanee, yellow perch, etc. introductions into Flathead Lake). Herein I focus the assessment analysis on two genera, *Salvelinus* (char) and *Oncorhynchus* (trout) and the compromised ecological integrity of GNP native populations due to either competitive exclusion or hybridization of the native species by introduced non-natives. See Appendix B for distribution maps of all salmonid species.

Table 4.26. Native (Nat) and introduced (Intro) salmonids in the three Drainages of Glacier National Park (modified from Downs et al. 2011).

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Arctic grayling (*Thymallus arcticus*)	Introduced	--	Introduced
Brook trout (*Salvelinus fontinalis*)	Introduced	Introduced	Introduced
Bull trout (*Salvelinus confluentus*)	Native	--	Native
Lake trout (*Salvelinus namaycush*)	Introduced	--	Native
Lake whitefish (*Coregonus clupeaformis*)	Introduced	--	Native
Mountain whitefish (*Prosopium williamsoni*)	Native	Native	Native
Pygmy whitefish (*Prosopium coulteri*)	Native	--	Native
Kokanee (*Oncorhynchus nerka*)	Introduced	--	Introduced
Rainbow trout (*Oncorhynchus mykiss*)	Introduced	Introduced	Introduced
Westslope cutthroat trout (*Oncorhynchus clarkii lewisi*)	Native	Native	Native
Yellowstone cutthroat trout (*Oncorhynchus clarkii bouvieri*)	Introduced	Introduced	Introduced

4.4.1 Salvelinus (char)

Bull trout (*Salvelinus confluentus*) are native in GNP watersheds located west of the Continental Divide and east of the Continental Divide north of the Hudson Bay Divide. In addition, GNP supports both native (Hudson Bay drainage) and introduced (Columbia River drainage) populations of lake trout (*Salvelinus namaycush*), which are found principally occupying lake habitats. When lake trout are either introduced or are an invasive species into areas that have been historically occupied by bull trout, lake trout tend to out-compete the native bull trout in lake habitats leading to significantly reduced bull trout abundances in locations that were previously known to be strong populations (Fredenberg et al. 2007). Although lakes in GNP have experienced introductions and invasions of nonnative fishes, extirpations of native species as a direct result of the establishment of nonnative species has not been observed. It is believed that most recent invasions of lakes has occurred as a result of out-migration from an expanding lake trout population in Flathead Lake as a result of food web changes with cascading impact to both Flathead Lake and the upper Flathead River Basin (Ellis et al. 2011). Meeuwig et al. (2008) used a landscape ecological approach to examine the influence of landscape characteristics and heterogeneity on native fish species richness among lakes in Glacier National Park in the North and Middle Fork of the Flathead drainages. They found that nonnative species, particularly lake trout (*Salvelinus namaycush*), have become widespread throughout many of the tributaries of the North and Middle Forks of the Flathead River in GNP, but they also showed that the upstream extent of lake trout distribution was limited by the presence of barriers to fish dispersal.

Lake trout invasion of lakes in the North and Middle Forks of the Flathead River from Flathead Lake has been the focus of recent research culminating in an action plan to conserve bull trout (Fredeberg et al. 2007). They grouped 17 lakes assessed for lake trout invasion into three threat categories: 1) secure lakes; Upper Kintla, Trout, Arrow, Isabel, and Upper Isabel, 2) vulnerable lakes; Akokala and Cerulean, and 3) compromised lakes; Lower Kintla, Bowman, Upper Quartz, Middle Quartz, Lower Quartz, Logging, Rogers, Harrison, McDonald, and Lincoln. Secure lakes were small backcountry lakes with fish passage barriers in the drainage downstream of the lake. As a result, they considered these lakes to have the most secure populations of bull trout. Vulnerable lakes were grouped together because they believed there is a high likelihood that they could become compromised by lake trout because of the absence of physical structures that would preclude fish passage in the drainages downstream thus giving access to invasive fishes moving upstream. Compromised lakes were defined as containing lake trout or brook trout (*Salvalinus fontinalis*). The status of lake trout invasion and corresponding status of bull trout populations in each lake was determined to be variable. These invasions illustrate there are no physical barriers downstream of these lakes to preclude ongoing lake trout movement or future invasions of other species from other waters in the interconnected Flathead Basin. See Appendix C for distribution maps of all char species.

4.4.2 Oncorhynchus (trout)

Westslope cutthroat trout (*Oncorhynchus clarkii lewisi*) are native in GNP waters throughout the park in each of the watersheds of this assessment. Westslope cutthroat trout is a "Species of Concern" in Montana (Natural Heritage Program and American Fisheries Society) and as such is managed as a species of concern by the NPS. This unique subspecies of inland native trout has been petitioned for listing under the Endangered Species Act (ESA). However, in 2000, the U.S. Fish and Wildlife Service determined that listing was not warranted. GNP is considered a range-wide stronghold for westslope cutthroat trout; the long term persistence of non-hybridized populations is threatened by recent spread of nonnative rainbow trout introgression (Hitt et al. 2003, Boyer et al. 2008).

Recent work in the North Fork and Middle Fork Flathead Rivers has shown a rapid increase in hybridization spreading in an upstream direction, threatening potentially pure westslope cutthroat trout populations in GNP. For example, using molecular DNA techniques, (Hitt et al. 2003) found new hybridization in 8 of 11 (73%) reaches that were determined to be non-hybridized in 1988. Boyer et al. (2008) and Taylor et al. (2009) found that hybridization is spreading among reaches in an upstream direction from source streams in the lower river, and may not be constrained by environmental factors (Muhlfeld et al. 2009). These data suggest that hybridization is spreading upstream, threatening non-hybridized (i.e., genetically pure) westslope cutthroat trout stocks in GNP.

Westslope cutthroat trout have long been considered an integral component of biodiversity, culture and economy in GNP. They are part of a historic fishery that is a fundamental part of the biodiversity of the park. As such, protecting native fish resources is a high priority for conservation and management programs in GNP. Invading nonnative species, rainbow and lake trout, could overwhelm and replace native westslope cutthroat trout due to hybridization, competition, and predation. Westslope cutthroat trout are particularly susceptible to hybridization with nonnative rainbow trout in situations in which anthropogenic habitat disturbances increase water temperature and degrade stream habitats. Habitat degradation and fragmentation have been identified as leading factors in the decline and extirpation of westslope cutthroat trout populations throughout their range (Liknes and Graham 1988). Muhlfeld et al. (2009) showed that hybridization is likely to spread further up the North Fork and Middle Fork of the Flathead River and basin tributaries, causing additional westslope cutthroat trout populations to be lost, unless populations with high amounts of rainbow trout admixture are suppressed or eliminated. They also showed that protection of hybridized populations facilitates the expansion of hybridization. To preserve non-hybridized westslope cutthroat trout populations, managers should consider eradicating hybridized populations with high levels of rainbow trout admixture and restoring streams characterized by warm temperatures and high levels of disturbance. See Appendix C for distribution maps of all trout species.

4.4.3 Methods GNP Fish

4.4.3.1 Stream Native/Non-Native Fish ($V_{STREAMFISH}$) and Flathead Fish ($V_{FLTHDFISH}$)

The spatial distributions of native and non-native salmonid species (see Table 4.26) within the park were provided by Montana Fish, Wildlife, and Parks (fwp.mt.gov/fishing/mFish/). These distributions were generated from a combination of actual sampled distribution and best professional judgment. The distribution layer was applied to all permanent streams within each watershed. If a species was present, the mapped portion of the stream within the species distribution received a numerical assignment of 1.0 representing 100% occupancy for that species. If the species is absent, the stream portion received a score of 0.0 representing 0% occupancy. All See Appendix C for distribution maps of all salmonid species.

A similar approach was used for the distribution of cutthroat-x-rainbow trout hybrid. However, information from Boyer et al. (2008) and Muhlfeld et al. (2009) suggests that there is a concentration of hybrids at the confluence of the Middle Fork and North Fork that dissipates up the North Fork to the town of Polebridge, Montana. Above that point the cutthroat-x-rainbow trout hybrid have limited to no occurrence. To capture this decreasing distribution the North Fork was divided into 20 even segments from the confluence to Polebridge. These segments were scored as 1.0 (100% occupancy) at the confluence section to 0.0 (0% occupancy) at Polebridge with decreases of 5% at each segment to capture the decreasing concentrations of the cutthroat-x-rainbow trout hybrid. For the purpose of this assessment, it was assumed that the cutthroat-x-rainbow trout hybrid decreasing concentrations occur in a radius from the North Fork/Middle Fork confluence. Therefore, this segmented scoring was also applied to the Middle Fork upstream from the confluence with the same stream segment length and occupancy assignments. This approach was also applied to tributaries from the park that intersects with the North and Middle Fork of the Flathead River within this radius. For example, a park stream has its confluence on a North Fork segment with 60% occupancy; the joining stream would be assigned 55% occupancy and would decrease in occupancy in same segments lengths used in the North Fork/Middle Fork and continue upstream until it reaches 0% occupancy.

The following describes the steps necessary to derive the final Stream Native-Non-native Metric score:

1. The sum of total permanent stream length presumed to be occupied by each salmonid species, including cutthroat-x-rainbow hybrids, was derived for each watershed. These lengths were divided by the total length of the permanent streams as mapped by GIS stream shape files provided by GNP (stream.shp) within each watershed to obtain a percent occupancy for each species.
2. The sum of the percent occupancy of all native species and all non-native species for each watershed was determined. Because this is a cumulative percent occupancy for each watershed, the sum occupancy may be larger than 100%.
3. A ratio of native to non-native salmonid occupancy was acquired with a final score ranging from 0 to 1 using the following formula:

   $V_{STREAMFISH}$ (or $V_{FLTHDFISH}$) = Native stream occupancy / sum of native and non-native stream occupancy

4.4.3.2 Lake Native/Non-native Fish ($V_{LAKEFISH}$)

The spatial distribution of native and non-native salmonid species in the park's lakes (See Table 4.26) were also provided by Montana Fish, Wildlife, and Parks and the distribution layer was applied to all lakes within each watershed. As with the streams, the distributions were generated from a combination of actual sampled distribution and best professional judgment. If the distribution maps indicated that a species was present in a lake, the lake received a numerical assignment of 1.0 representing 100% occupancy for that species. If the species is absent, the lake received a score of 0.0 representing 0% occupancy. All See Appendix C for distribution maps of all salmonid species.

The distributions of the cutthroat-x-rainbow trout hybrid for lakes were mapped as an extension of the hybrid mapping for streams. If a lake was associated with a stream containing hybrids, then the lake received an occupancy assignment relative to the stream segment occupancy, however, the lake was treated as one segment of the hybrid distribution and received one occupancy assignment despite its size. For example, a stream at the outlet of a lake has hybrid occupancy of 70% then the lake received an occupancy score of 60% and stream segments entering the lake received an occupancy assignment of 50%.

The following describes the steps necessary derive the final Lake Native-Non-native Metric score:

1. The sum of total lake area occupied by each salmonid species, including cutthroat-x-rainbow hybrids, was derived for each watershed. The area of lakes occupied by a species was divided by the total lake area from GIS lake shape files provided by GNP (lake.shp) within each watershed to obtain a percent occupancy for each species.
2. The sum of the percent occupancy of all native species and all non-native species for each watershed was determined. Because this is a cumulative percent occupancy for each watershed, the sum occupancy may be larger than 100%.
3. A ratio of native to non-native salmonid occupancy was acquired with a final score ranging from 0 to 1 using the following formula:

$V_{LAKEFISH}$ = native lake occupancy / sum of native and non-native lake occupancy

4.4.3.3 Calculation of GNP Fish Condition Score

To acquire an overall fish condition score for the park, the following index was applied the fish distribution metrics. The model is simply an average of the native / non-native ratios of GNP's streams and lakes. The fish condition of the Flathead River's North and Middle Forks were provided as a separate index below.

Fish Condition Index Score = $((V_{STREAMFISH} + V_{LAKEFISH})/2)$

4.4.4 Spatial Analysis Results

Please note that the fish distribution provided by Fish Wildlife and Parks were generated from a combination of actual sampled distribution and best professional judgment, therefore the following data are the best available estimate of distributions. Additional sampling would be required. These results use the best available data to provide a prioritization of future monitoring efforts. Appendix C provides maps of specific fish distributions in the park's lakes and streams.

4.4.4.1 Stream Native/Non-native fish ($V_{STRFISH}$)

The range of native and non-native occupancy in GNP stream are provided in Tables 5.27a and 5.27b and the condition scores are provided in Table 4.27c. Cut Bank has no native salmonids mapped in its streams but a high occupancy of non-native salmonids. As a result this watershed scored the lowest in the park for stream fish condition. Alternatively Kintla/Bowman has no mapped non-native salmonids and therefore scored the highest for this metric (Figure 4.19).

Table 4.27a. The percent occupancy of native salmonids in GNP streams.

	Percent Presence of Native Stream Fish				
Watershed	**Bull Trout**	**Lake Whitefish**	**Mountain Whitefish**	**Cutthroat Trout**	**Lake Trout**
Belly	22	0	28	13	9
Camas	14	0	32	39	0
Coal/Ole	29	0	33	43	0
Cut Bank	0	0	0	0	0
Kennedy	32	0	21	25	0
Kintla/Bowman	45	0	27	55	0
Lake McDonald	2	0	18	44	0
Nyack	21	0	22	25	0
Quartz/Logging	11	0	23	39	0
Saint Mary	15	0	18	24	0
Swiftcurrent	20	0	6	29	0
Upper Two Medicine	0	0	12	1	0
Waterton	0	19	0	10	19

Table 4.27b. The percent occupancy of non-native salmonids in GNP streams.

	Percent Presence of Non-Native Stream Fish				
Watershed	**Brook Trout**	**Rainbow Trout**	**Yellowstone Cutthroat Trout**	**Cutthroat-x-rainbow Hybrid**	**Arctic Grayling**
Belly	31	33	0	0	16
Camas	1	1	0	11	0
Coal/Ole	16	0	0	1	0
Cut Bank	50	37	29	0	0
Kennedy	6	0	0	20	0
Kintla/Bowman	0	0	0	0	0
Lake McDonald	22	14	3	17	0
Nyack	25	0	0	9	0
Quartz/Logging	0	0	0	11	0
Saint Mary	2	24	0	5	0
Swiftcurrent	30	0	0	7	0
Upper Two Medicine	25	7	8	0	0
Waterton	51	27	0	0	0

Table 4.27c. The stream native/non-native salmonids occupancy and metric score in GNP streams.

Watershed	**Stream Native Sum**	**Stream Non-Native Sum**	**Stream Fish Metric Score**
Belly	72	80	0.47
Camas	85	13	0.87
Coal/Ole	105	17	0.86
Cut Bank	0	116	0.00
Kennedy	78	26	0.75
Kintla/Bowman	127	0	1.00
Lake McDonald	64	56	0.53
Nyack	68	34	0.67
Quartz/Logging	73	11	0.87
Saint Mary	57	31	0.65
Swiftcurrent	55	37	0.60
Upper Two Medicine	13	40	0.25
Waterton	48	78	0.38

4.4.4.2 Lake Native/Non-native fish ($V_{LAKEFISH}$)

The range of native and non-native occupancy in GNP lakes is provided in Tables 4.28a and 4.28b and the condition scores are provided in Table 4.28c. Cut Bank and Upper Two Medicine have no native salmonids mapped in its lakes but a high occupancy of non-native salmonids. As a result these watersheds scored the lowest in the park for lake fish condition. Lake McDonald has the highest occupancy of native fish relative to the non-native fish occupancy and therefore scored the highest for this metric (Figure 4.20).

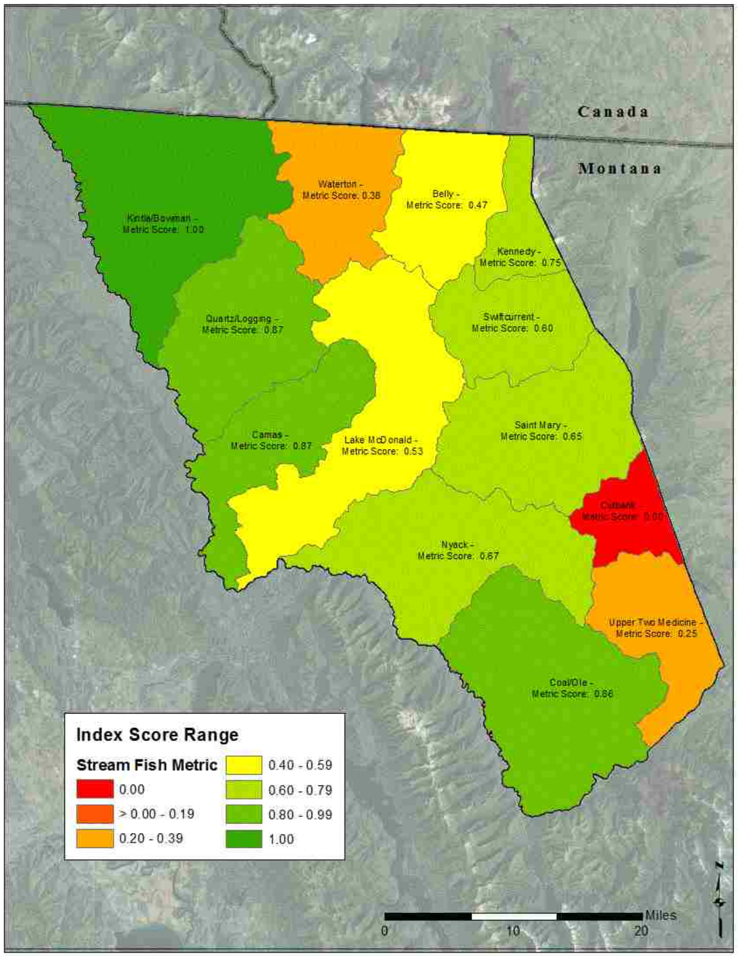

Figure 4.19. Metric for fish condition in GNP streams.

Table 4.28a. The percent occupancy of native salmonids in GNP lakes.

	Percent Presence of Native Fish					
Watershed	**Bull Trout**	**Lake Whitefish**	**Mountain Whitefish**	**Pygmy Whitefish**	**Cutthroat Trout**	**Lake Trout**
Belly	0	0	39	0	6	39
Camas	65	0	26	0	65	0
Coal/Ole	14	0	0	0	15	0
Cut Bank	0	0	0	0	0	0
Kennedy	30	0	0	0	0	0
Kintla/Bowman	92	0	81	0	81	0
Lake McDonald	94	94	94	94	94	0
Nyack	59	0	59	0	59	0
Quartz/Logging	90	0	90	0	90	0
Saint Mary	0	87	3	0	87	0
Swiftcurrent	2	0	65	0	0	0
Upper Two Medicine	0	0	0	0	0	0
Waterton	0	56	0	0	2	56

Table 4.28b. The percent occupancy of non-native salmonids in GNP lakes.

Watershed	Percent Presence of Non-Native Fish						
	Brook Trout	Lake Trout	Kokanee	Rainbow Trout	Yellowstone Cutthroat Trout	Cutthroat-x-rainbow Hybrid	Arctic Grayling
Belly	3	0	0	55	0	6	16
Camas	0	15	0	15	27	65	0
Coal/Ole	0	0	0	0	0	15	0
Cut Bank	0	0	0	18	9	0	0
Kennedy	0	0	0	0	21	0	0
Kintla/Bowman	0	80	80	0	0	81	0
Lake McDonald	0	94	94	94	4	24	0
Nyack	87	59	54	54	0	36	0
Quartz/Logging	0	86	0	0	3	90	0
Saint Mary	93	87	0	92	6	87	0
Swiftcurrent	82	0	76	65	0	0	0
Upper Two Medicine	75	0	0	55	4	0	0
Waterton	57	0	0	65	0	2	0

Table 4.28c. The native/non-native salmonids cumulative percent occupancy and metric score in GNP Lakes.

Watershed	Native Sum Cumulative Percent	Non-Native Sum Cumulative Percent	Lake Fish Metric Score
Belly	84	79	0.52
Camas	156	123	0.56
Coal/Ole	30	15	0.66
Cut Bank	0	27	0.00
Kennedy	30	21	0.59
Kintla/Bowman	254	242	0.51
Lake McDonald	468	308	0.60
Nyack	177	290	0.38
Quartz/Logging	270	180	0.60
Saint Mary	177	365	0.33
Swiftcurrent	67	223	0.23
Upper Two Medicine	0	135	0.00
Waterton	113	124	0.48

4.4.4.3 Fish Condition Index

The distributions of salmonid populations are used to calculate a ration of native to non-native salmonids in GNP lakes and streams. Metrics derived from these ratios are averaged to provide an overall index of salmonid condition for each watershed (Table 4.29). Cut Bank has no native salmonids in either its lakes or streams and therefore scored a 0.00 for condition. Upper Two Medicine has no native salmonids mapped in its lakes and therefore scored low. Lake McDonald and the Belly watersheds have the highest ratio of native fish relative to the non-native fish occupancy in its lakes and Kintla/Bowman in its streams and therefore scored the highest for this index. However all watersheds have been impacted by non-native fish species (Figure 4.21).

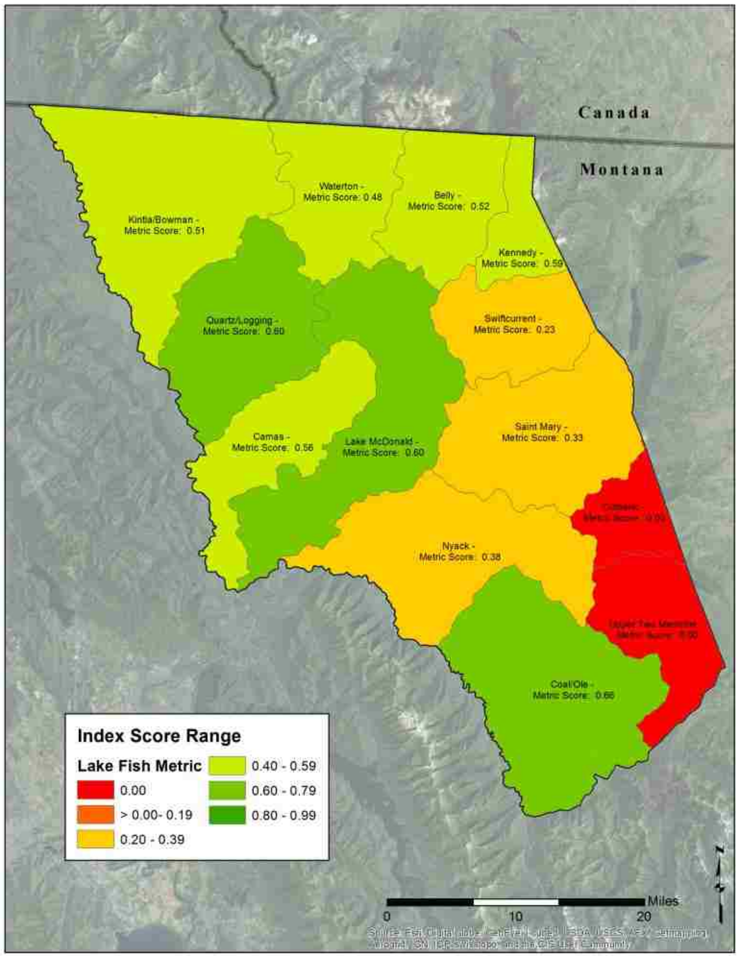

Figure 4.20. Metric for fish condition in GNP lakes.

Table 4.29. The fish metric and index scores in each watershed in Glacier National Park.

Watershed	Stream Fish Metric	Lake Fish Metric	GNP Fish index
Belly	0.47	0.52	0.49
Camas	0.87	0.56	0.72
Coal/Ole	0.86	0.66	0.76
Cut Bank	0.00	0.00	0.00
Kennedy	0.75	0.59	0.67
Kintla/Bowman	1.00	0.51	0.76
Lake McDonald	0.53	0.60	0.57
Nyack	0.67	0.38	0.52
Quartz/Logging	0.87	0.60	0.74
Saint Mary	0.65	0.33	0.49
Swiftcurrent	0.60	0.23	0.41
Upper Two Medicine	0.25	0.00	0.13
Waterton	0.38	0.48	0.43

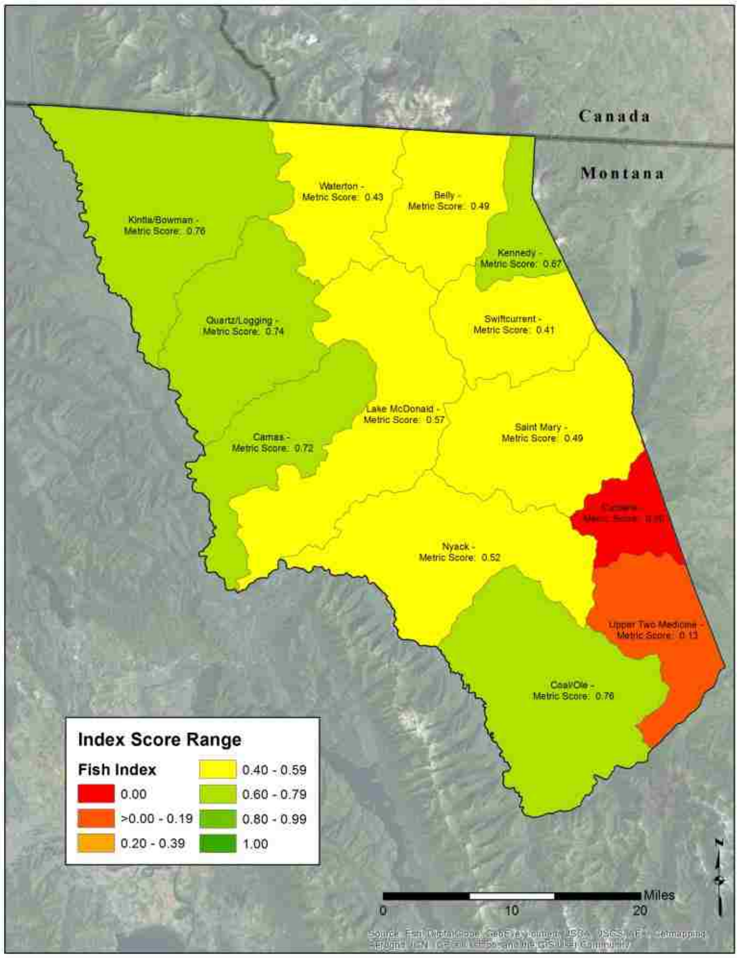

Figure 4.21. Index for fish condition in GNP lakes and streams.

4.4.4.4 Flathead Fish ($V_{FLTHDFISH}$)

The fish condition assessment of Flathead River is comprised of a stand-alone metric. This system is different in its size and extent of human disturbance than other stream systems in the park. The cumulative percent occupancy of all native and non-native in North and Middle Forks Flathead River and the resulting condition scores are provided in Tables 5.30a and 5.30b and the condition scores are provided in Table 4.30c. Lake trout are present throughout the study area; however rainbow and cutthroat-x-rainbow hybrid do not extend to the upstream assessment areas of the North Fork. Therefore the assessment areas closer to the confluence generally have lower index scores than those further upstream (Figure 4.22).

Table 4.30a. The percent occupancy of native salmonids in the North and Middle Forks of the Flathead River.

Flathead River Assessment Area	Percent Presence of Native Fish		
	Bull Trout	Mountain Whitefish	Cutthroat Trout
Middle Fork 1	100	100	100
Middle Fork 2	100	100	100
Middle Fork 3	100	100	100
Middle Fork 4	100	100	100
Middle Fork 5	100	100	100
Middle Fork 6	100	100	100
Middle Fork 7	100	100	100
North Fork 1	100	100	100
North Fork 2	100	100	100
North Fork 3	100	100	100
North Fork 4	100	100	100
North Fork 5	100	100	100
North Fork 6	100	100	100

Table 4.30b. The percent occupancy of non-native salmonids in the North and Middle Forks of the Flathead River.

Flathead River Assessment Area	Percent Presence of Non-Native Fish			
	Arctic Grayling	Rainbow Trout	Lake Trout	Cutthroat-x-Rainbow Hybrid
Middle Fork 1	0	100	100	88
Middle Fork 2	0	100	100	70
Middle Fork 3	0	100	100	48
Middle Fork 4	0	100	100	27
Middle Fork 5	0	100	100	9
Middle Fork 6	0	100	100	0
Middle Fork 7	0	100	100	0
North Fork 1	0	100	100	92
North Fork 2	0	100	100	77
North Fork 3	0	100	100	56
North Fork 4	93	6	100	21
North Fork 5	100	0	100	0
North Fork 6	100	0	100	0

Table 4.30c. The percent occupancy of non-native salmonids in the North and Middle Forks of the Flathead River.

Flathead River Assessment Area	Native Sum	Non-Native Sum	Flathead Fish Index Score
Middle Fork 1	300	288	0.51
Middle Fork 2	300	270	0.53
Middle Fork 3	300	248	0.55
Middle Fork 4	300	227	0.57
Middle Fork 5	300	209	0.59
Middle Fork 6	300	200	0.60
Middle Fork 7	300	200	0.60
North Fork 1	300	292	0.51
North Fork 2	300	277	0.52
North Fork 3	300	256	0.54
North Fork 4	300	220	0.58
North Fork 5	300	200	0.60
North Fork 6	300	200	0.60

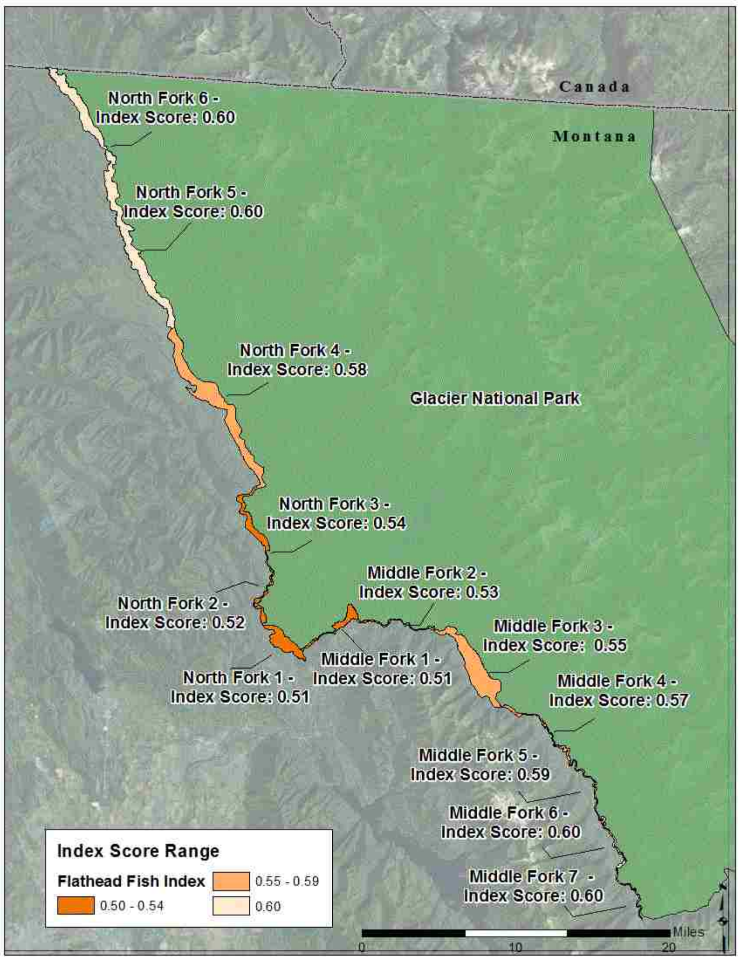

Figure 4.22. Flathead River fish condition index score range.

5 Condition of Existing Management Zones

5.1 Glacier National Park Management Framework

The park is managed in accordance with NPS policy "to understand natural processes and human-induced effects; mitigate potential and realized effects; monitor ongoing and future trends; protect existing natural organisms, species populations, communities, systems, and processes; and interpret these organisms, systems, and processes to the park visitor" (Layman 1999). To facilitate these management directives, the park published a General Management Plan (GMP: Layman 1999). The park's GMP presents a strategy to guide future decisions based on six geographic management areas rather than the 13 watersheds I presented in the above section. Currently, these areas have different management priorities based on the land and visitor uses that are appropriate to the development and activities described for those zones. The six geographic areas include; 1) Many Glacier, 2) Goat Haunt-Belly River, 3) Going-to-the-Sun Road Corridor, 4) Two Medicine, 5) Middle Fork, and 6) North Fork (see Figure 3.2). The geographic areas vary in the amount of infrastructure and visitor access and as a result vary in the intensity of park management.

In an attempt to assist the park's resource management team meet the mandate articulated by NPS and the park's GMP, I converted the watershed scale results presented Section 5 into the current GMP management zone scale.

5.2 Methods

To develop the GMP management zone condition scores, the percent of each management zone that is comprised of an assessment watershed was determined by intersecting the GMP spatial layer (gmp_area.shp) with the assessment watershed layer (GNP_HUC10-2011.shp). The relative area of each assessment area was then multiplied by the condition index score for the respective assessment model and summed for each management zone presented below. The final map represents the overall average of all assessment indices at a management zone scale.

5.3 Results

Management zone scores are comprised of the spatially averaged index scores of portions of the watersheds within each zone (Table 5.1). Because the watershed scores are spatially averaged to create the management zone score (Table 5.2), a larger watershed a lower condition score due to limited habitat, increased human use, or non-native species will strongly influence the condition score of the management zone. The average ecological condition of all management zones (Table 5.2) is a coarse assessment comprised of multiple watersheds and multiple assessment models. It is analogous to a grade point average on an academic transcript. It provides an overview of the conditions, but requires a close look at the management zone condition scores and the condition scores of the assessment watersheds contained within the management zones. The assessment of the Flathead River is not included because it is adjacent to the park; please see Section 5 for details.

Table 5.1. Percent of general management zones that are comprised of each assessment watershed.

	Percent of General Management Areas Composed of Assessment Watersheds					
Assessment Watersheds	**Goat Haunt**	**Going-to-the-Sun Road**	**Many Glacier**	**Middle Fork**	**North Fork**	**Two Medicine**
Belly	34.50	-	0.02	-	-	-
Camas	0.01	0.06	-	-	23.06	-
Coal/Ole	-	-	-	54.82	-	0.07
Cut Bank	-	-	-	-	-	33.74
Kennedy	8.74	-	18.43	-	-	-
Kintla/Bowman	0.02	-	-	-	46.18	-
Lake McDonald	20.18	46.72	0.04	0.03	0.05	-
Nyack		2.13	-	45.13	-	0.05
Quartz/Logging	0.02	-	-	-	30.68	-
Saint Mary	-	51.04	0.08	0.01	-	0.05
Swiftcurrent	-	0.04	81.44	-	-	-
Upper Two Medicine	-	-	-	0.01	-	66.09
Waterton	36.49	-	-	-	0.02	-

Table 5.2. Percent of general management zones that are comprised of each assessment watershed.

	Condition Indices			
General Management Areas	**Stream**	**Lake**	**Salmonid**	**Average**
Goat Haunt	0.97	0.83	0.50	0.77
Going-to-the-Sun Road	0.94	0.65	0.53	0.71
Many Glacier	0.97	0.87	0.46	0.77
Middle Fork	0.97	0.93	0.65	0.85
North Fork	0.97	0.86	0.75	0.86
Two Medicine	0.96	0.74	0.09	0.60

As data is compiled into coarser scales (from 13 watersheds to six management zones), information is potentially lost. As the park's resource managers use the maps provided below to prioritize their mitigation, protection and monitoring activities, I recommend that they examine the various ecological conditions of the assessment watersheds within each management zone. Parsing the assessment watersheds may help further prioritize activities within each management zone. To assist in that effort, I provided two maps for each of three focal assessment models (Figures 6.1a and 6.1b through Figures 6.6a and 6.6b). The first is the focal assessment index watershed scores overlain with the management zone boundaries, and the second are the management zone scores. The focal condition scores of all aquatic areas are averaged to provide an overview of the aquatic conditions by zone (Figure 5.7). Because the assessment of the North and Middle Forks of the Flathead River are adjacent to the park and not within a management zone, the indices for the Flathead River are not included in this section.

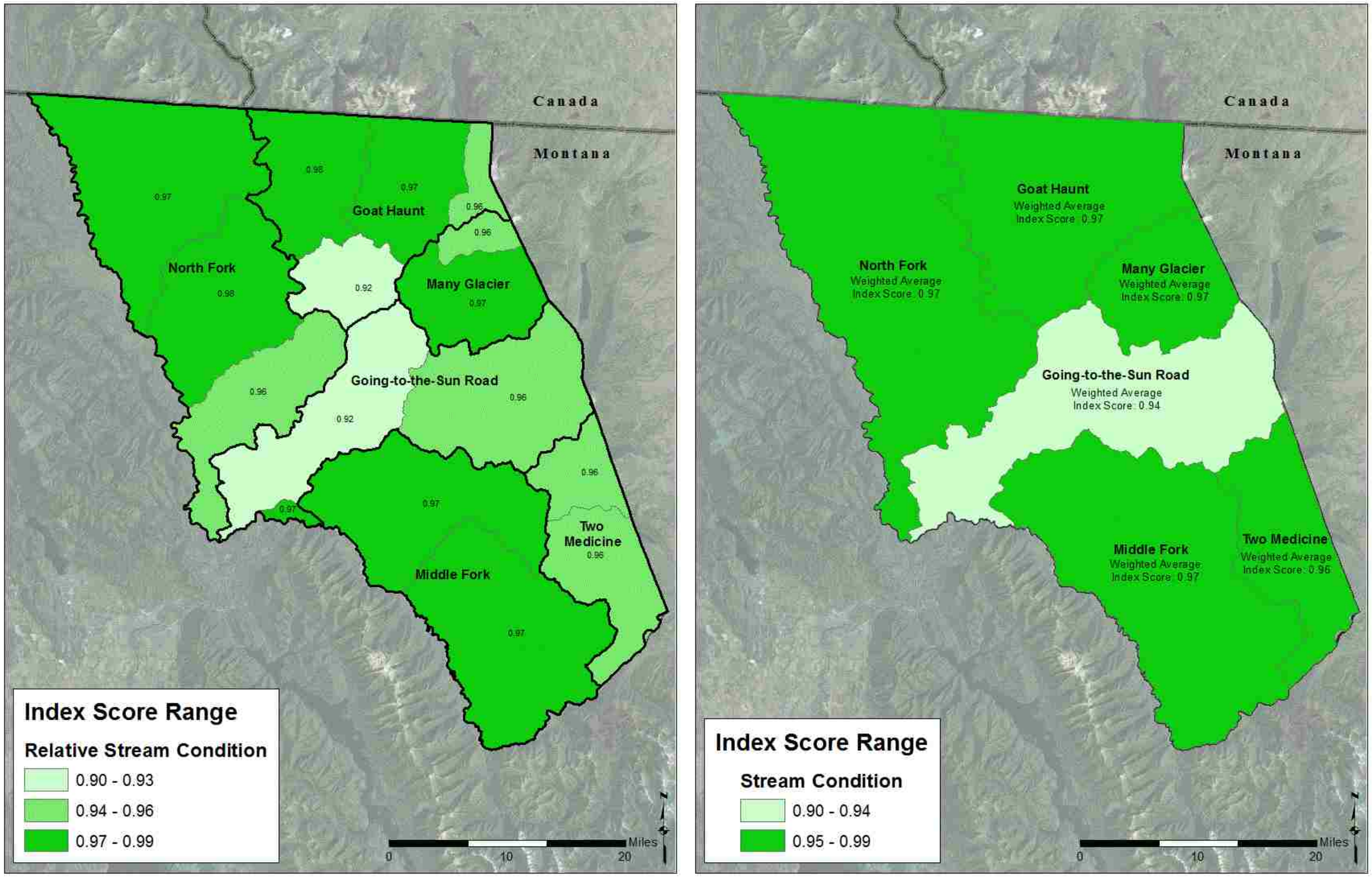

Figure 5.1a and b. Figure a. watershed stream ecological condition scores within each management zone. Figure b. stream condition score for each management zone.

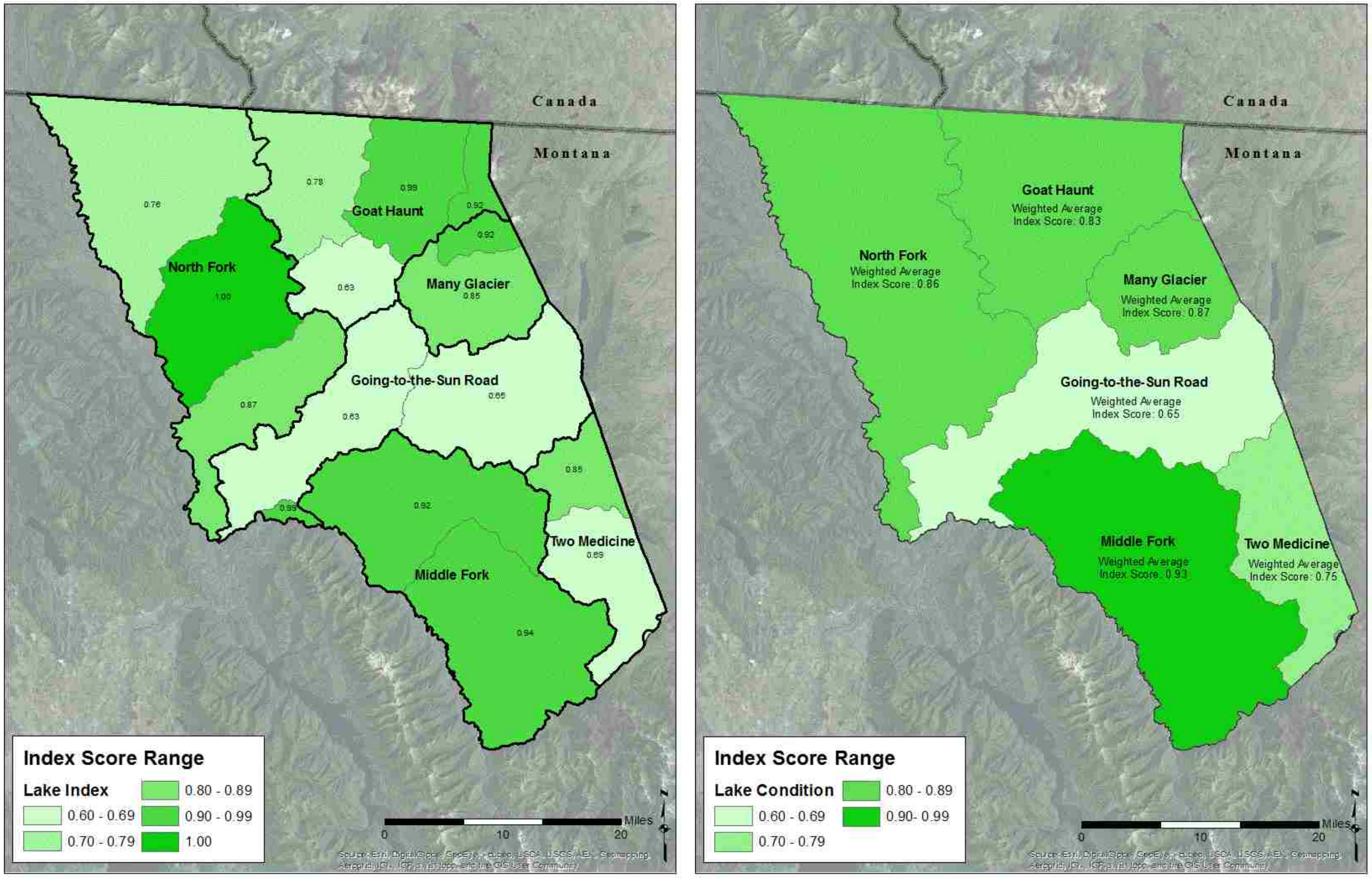

Figure 5.2a and b. Figure a. watershed lake ecological condition scores within each management zone. Figure b. lake condition score for each management zone.

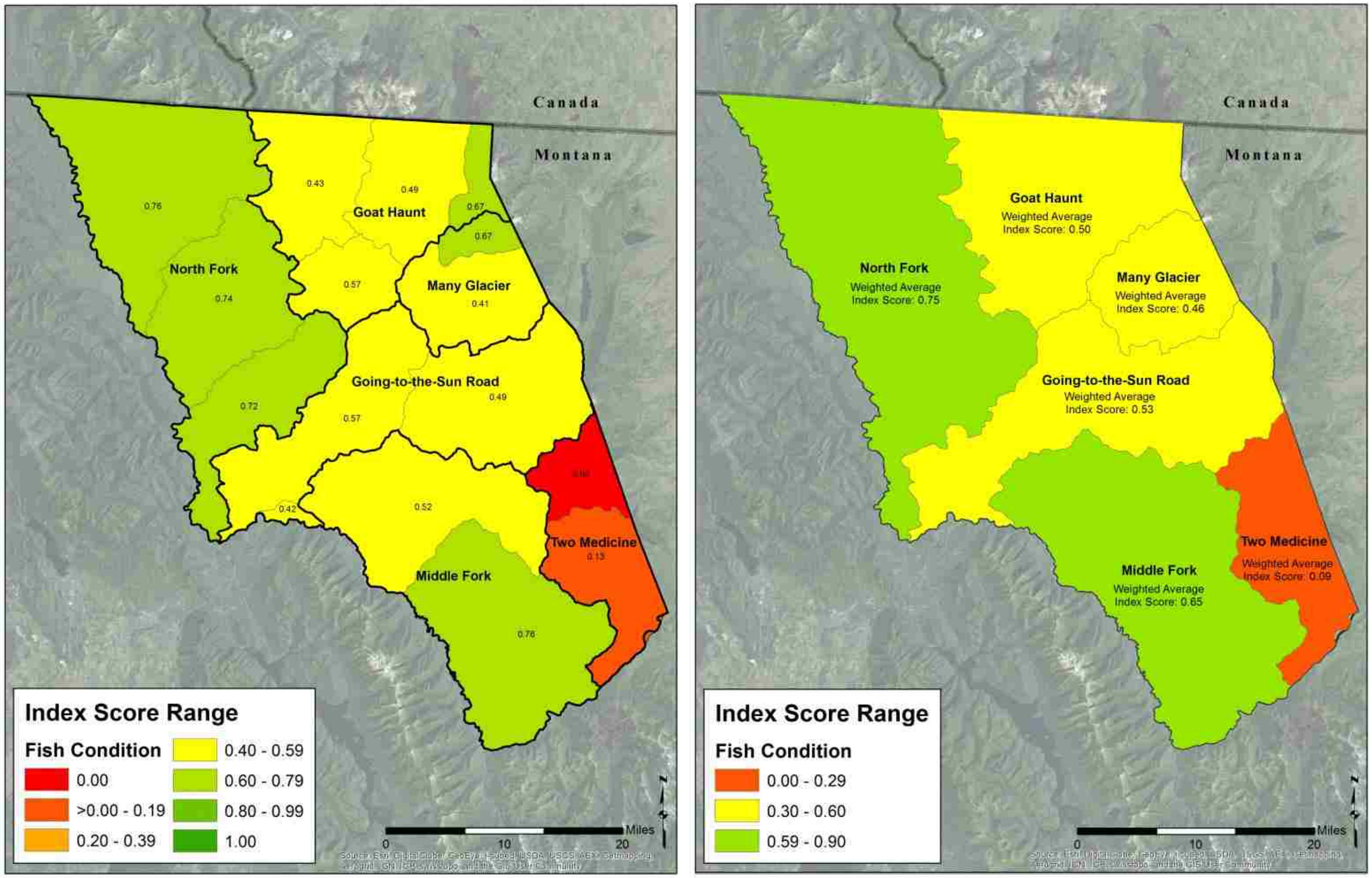

Figure 5.3a and b. Figure a. watershed fish ecological condition scores within each management zone. Figure b. fish condition score for each management zone.

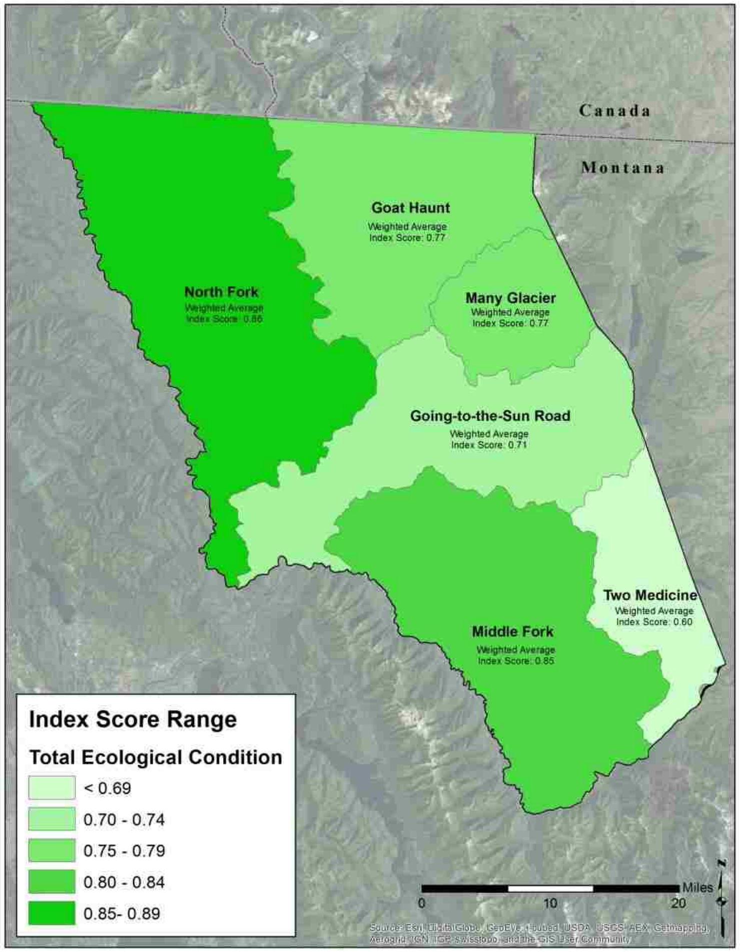

Figure 5.7. Ecological condition of each management zones comprised of an average of all assessment models.

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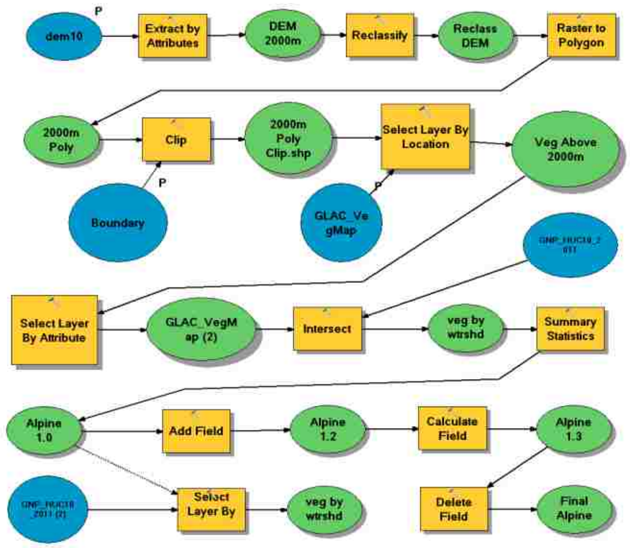
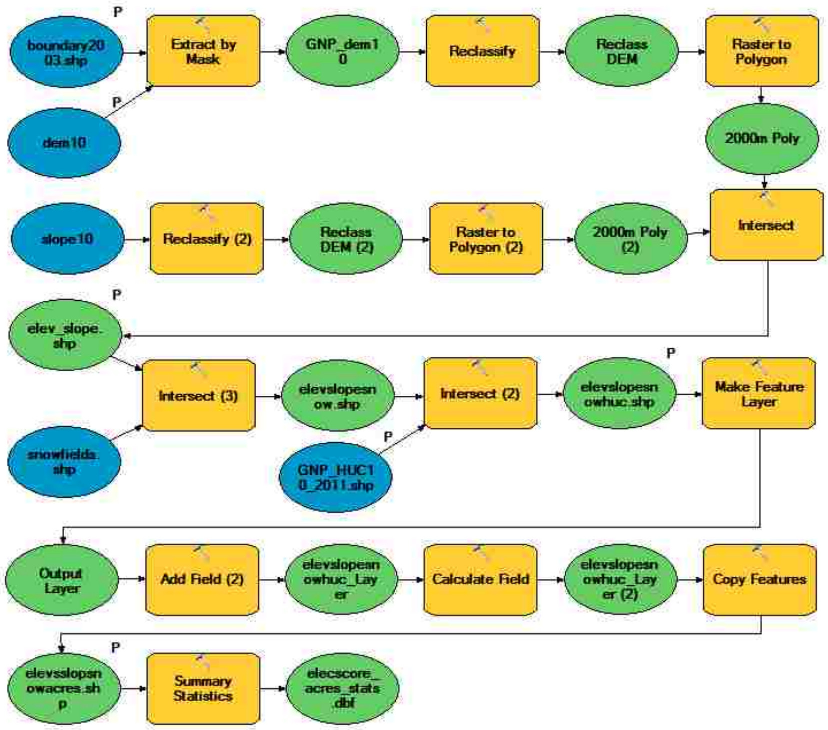
1 Appendix A: GIS Models

Alpine Habitat Metric



Elevation



A. Instructions for computing road density per watershed using ArcGIS
1) Two types of data are required in coverage, shapefile or Geodatabase format.
a. Roads layer
b. Watershed/HUC layer
2) Create a Personal Geodatabase called RoadsHUC (ArcCatalog > right click in folder > New > Personal Geodatabase)
3) Import roads layer and HUC layer into the Geodatabase (ArcCatalog > right click on layer > Export > Coverage to Geodatabase)
4) Intersect the roads and HUC layer (ArcMAP > Tools > Geoprocessing Wizard > Intersect)
5) Export the intersection attribute table to an Excel spreadsheet. (ArcMap > Add intersection layer > Open attribute table > Options > Export > Export to .dbf file outside of the Geodatabase)
6) Open .dbf file in Excel worksheet
7) Remove all extraneous data from spreadsheet and format cells
8) Put data in a PivotTable (Data > PivotTable > Wizard – HUC number goes in Row and Shape_Length goes into Data)
9) Add HUC areas to the spreadsheet and order sequentially (same as pivot table).
10) Compute density (Length/1000)/(Area/10000

CHAPTER III: EFFECT OF THEMATIC MAP MISCLASSIFICATION ON LANDSCAPE MULTI-METRIC ASSESSMENT

1 Abstract

Advancements in remote sensing and computational tools have increased our awareness of broader environmental problems thereby creating a need for monitoring, assessment and management at these scales. Over the last decade several watershed and regional multi-metric indices have been developed to assist decision makers with planning actions these scales. Most of these indices assume their assessment results are free from error, however these tools use remote sensing products that are subject to land-cover misclassification and these errors are rarely incorporated in the assessment results. Using a Monti Carlo error simulation model, I found that land-use intensity and fragmentation metrics, both commonly used in landscape assessment, have different sensitivities to map misclassification and that these sensitivities change depending on the land cover of the assessment site. These sensitivities result in a bias between the metric scores that do not account for error (naive scores) and simulated metric scores that incorporate potential error and this bias varies in magnitude and direction across different land cover compositions. When combined into a multi-metric index, the bias indicates that our naive assessment model may over-estimate the habitat condition of sites with limited human impacts and to lesser extent either over or under-estimates the habitat condition of sites with mixed land-use.

2 Introduction

Advances in ecological assessment tools designed to assist in the management of aquatic systems at broad spatial scales have paralleled increased access to remote sensing products and advances in geographic information processing. Remote sensing products such as thematic maps from Landsat data or orthorectified imagery provide the necessary baseline data to link alterations in landscape structure to perturbations in ecosystem functions at these large scales. These remote sensing data have known errors that should be, and generally are, clearly articulated in metadata or associated accuracy reports. However efforts to incorporate these errors into ancillary products such as assessment tools are currently very limited (Shao and Wu 2008). Ignoring the implications of these known errors on the results of assessment models potentially affects the level of confidence that resource managers have in the information the tools provide, which potentially determines the extent of their use.

Indicator-based ecological assessment models have been developed to provide decision and policy makers with the ecological information on which to base resource management decisions, communicate those decisions to the public, and develop rules to protect resources (Turnhout et al. 2007; Dramstad 2009). In reviews of contemporary aquatic assessment models, the multi-metric index (MMI) was the predominant indicator-based approach (Diaz et al. 2004; Fennessy et al. 2004; Böhringer and Jochem 2007). MMI tools developed for assessments for watersheds (Brooks et al. 2004; Tiner 2004; Weller et al. 2007; Meixler and Bain 2010), regions (e.g., Reiss and Brown, 2007; Collins et al., 2008), and compiled to provide continental assessments (USEPA 2013) commonly use remotely sensed data and imagery to develop scale appropriate metrics (Fennessy et al. 2007). While cartographic data generally follow standardized reporting guidelines that articulate known uncertainties inherent in the product (Foody 2002), incorporating these known uncertainties into MMI tools is rare (Fore et al. 1994; Whigham et al. 1999; Stein et al. 2009) and tend to be absent in the assessment implementation and reporting phase (Smith et al. 1995; Hauer et al. 2002; Klimas et al. 2004; Collins et al. 2008).

Ideally, a well-constructed ecological MMI is designed to facilitate resource decisions by providing straightforward analyses of ecological data to enable translation to management applications (Barbour et al. 1999), yet addressing the implications of uncertainty in such tools can be complex. The challenge is to provide a pathway to incorporate known uncertainties from multiple data sources into an assessment tool used by planners, policy makers, lawyers, and scientists. In this paper I address two questions as a step toward meeting this challenge: How sensitive is a landscape-scale multi-metric index to error from input data (specifically thematic land-cover misclassification)? What are the implications of this uncertainty to resource management decisions?

3 Methods

To answer these questions I developed a multi-metric index that uses thematic Landsat data to provide an assessment of floodplain condition along 250 km of the Flathead River in northwestern Montana, USA. Typical of most multi-metric indices, our initial assessment does not account for misclassification errors within the thematic map and produces metric and index scores that are considered naive. I then provided an error simulation model to incorporate known map classification error into our multi-metric assessment tool by developing multiple potential map realizations based on classification probabilities and potential spatial correlations. I apply our MMI to each realization to establish a distribution of potential assessment scores and compare this distribution to the naive score to determine potential bias and the implications of that bias on management decisions.

3.1 Study Area and Site Selection

Our assessment model is centered on the Flathead River system above Flathead Lake within northwestern Montana, USA and includes portions of the North Fork, Middle Fork and main stem of the Flathead River (Figure 1). The study area consists of land-use and land-cover (LULC) typical in the floodplains of larger rivers in the Northern and Canadian Rocky Mountains (Figure 2). The North Fork of the Flathead River has its headwaters in southeastern British Columbia, Canada and enters the study area as it crosses the U.S. border. Within the study area, the river flows 93 km south-by-southeast along the northwest boundary of Glacier National Park (GNP) through a broad U-shaped valley with expansive low-gradient montane alluvial floodplains predominantly covered with forest and grassland (simply called unmanaged lands here) and occasional pasture, urban and exurban development (called managed lands here). The Middle Fork has its headwaters in the Bob Marshall Wilderness Area and enters the study area as it emerges from the wilderness complex and meets the southwest boundary of GNP. Within the study area the Middle Fork flows 70 km through a series of confined and unconfined reaches within a narrow valley that also contains U.S. Highway 2, the Burlington Northern Santa Fe Railroad transportation corridor and the small town of West Glacier, MT at the southwestern tip of GNP. The main Flathead River channel begins at the North and Middle Forks confluence and flows about 86 km southerly leaving the study area as it enters the 480 km^2 Flathead Lake. Along the way this sixth-order river leaves the confined forested slopes and enters a broad piedmont valley floodplain consisting of agricultural, urban and exurban development interspersed with floodplain forest.

Nineteen assessment areas were selected based on continuous floodplain reaches separated by geomorphic constrictions on the river valley (Figure 1): nine sites on the North Fork (numbered N-1 through N-9 from downstream to upstream), three sites on the Middle Fork (M-1 through M-3) and seven on the main stem of the Flathead River (F-1 through F-7). These sites consist of both broad alluvial depositional areas typically associated with floodplain ecosystems and confined reaches with limited floodplain. Local biological diversity of river and floodplain systems is strongly influenced by land-use at several scales including local buffers (Morley and Karr 2002; Allan 2004; Pennington et al. 2010). To account for local land-use impacts adjacent to floodplain habitats I established a 1-km buffer to the entire floodplain area and delineated 19 buffer assessment sites perpendicular to the outer edge of each floodplain assessment site.

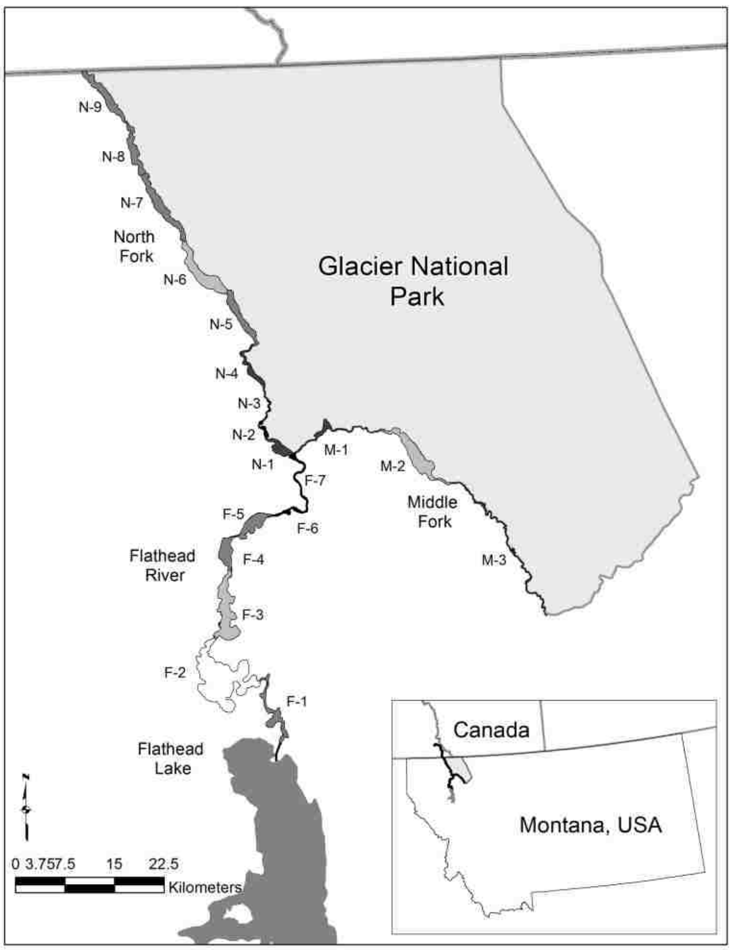


Figure 1. Location of study area and the 19 floodplain assessment sites. N-1 through N-9 are on the North Fork of the Flathead, M 1 through M-3 are on the Middle Fork, and F-1 through F-7 are sites on the Flathead River.

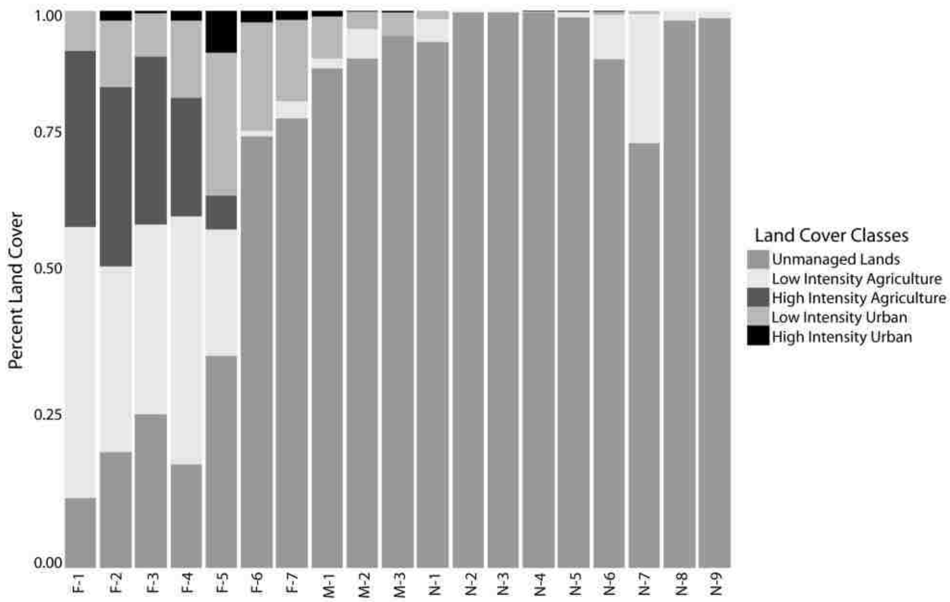


Figure 2. Percent cover of land-cover classes in each assessment site (floodplain and buffer area combined).

I digitized the nineteen assessment sites in ArcGIS 10.0 (ESRI 2011) with the assistance of 2005 background orthoimagery from the USDA National Agriculture Imagery Program (NAIP: (USGS 2014), overlain with a 30-m digital elevation map (USGS 2013), visual assistance from oblique views within Google Earth's 3-D models (Google Earth 2013), oblique imagery from aerial reconnaissance, and multiple site visits. Unless otherwise stated, all data collection, organization and subsequent analyses were conducted in ArcGIS 10.0 (ESRI 2011) and the R system for statistical computing (R Core Team 2013).

3.2 Multi-Metric Index Case Study

Multi-metric indices are composed of metrics that provide qualitative measure of the condition of biotic and abiotic structural attributes that in combination support ecological function or maintain ecosystem integrity. To create metrics, a score is assigned (e.g., 0-1, 1-100) to the attribute where the low range represents a heavily disturbed condition and the high range represents the best condition (Karr and Chu 1998). To create an index, these metrics are combined in a manner that best describes the attribute's relative contribution to system function or integrity based on reference data, literature and the expert opinion of the model developers. MMI's are established, tested and refined within a reference domain (e.g., bioregion, physiographic region or political boundary). Ultimately the MMI provides an assessment score that represents overall condition of a site-of-interest relative to the range of conditions in the model's domain (Smith et al. 1995; Barbour et al. 1999; Stoddard et al. 2008). In practice, a robust landscape scale floodplain assessment tool would incorporate attributes from multiple spatial datasets such as road densities, wetland inventories, soils databases, elevation, slope, and human population density. However I developed a simplified MMI with metrics derived from a single thematic map, the 2006 National Land-cover Database (NLCD) to specifically address aspects of uncertainty that arise from a single source of data.

NLCD thematic classified maps were developed for the conterminous United States by a coalition of U.S. agencies (MRLC 2013) using Landsat Thematic Mapper (TM) data for the 1992 map (Vogelmann et al. 2001) and Landsat Enhanced Thematic Mapper+ (ETM+) data for maps from years 2001 (Homer et al. 2007), 2006 (Fry et al. 2011) and 2011 (Jin et al. 2013). From 2001 on, NLCD used a decision-tree-based supervised classification approach to create a land-cover classification scheme at a spatial resolution of 30 m followed by aggregation of pixels to achieve a minimum mapping unit of about 0.40 ha to assign pixels to one of sixteen classes (Homer et al. 2004, 2007). The supporting NLCD literature also provide accuracy assessments in the form of a confusion matrix containing overall, producer, and user accuracy calculations that clearly articulate map classification error (MRLC 2013). These products do not require the map user to collect or process additional data; therefore, I apply the same limitation and do not collect additional site-specific accuracy data for this study beyond what is supplied with the NLCD product. Here I use the 2006 NLCD classified map from Path 41 and Row 26 (MRLC 2013) clipped to our floodplain and buffer polygons for each of the 19 assessment areas.

3.2.1 Landscape Metrics

For our landscape-scale MMI, I derived two metrics from the 2006 NLCD data: 1) a perturbation metric that assessed land-use intensity, and 2) a fragmentation metric that measured land-cover configuration. Each metric was calculated for the buffer and floodplain areas then combined into the assessment index.

Perturbation Metric for Buffer and Floodplain Areas (Met_{BP} and Met_{FP}): The aerial extent of human altered land-cover within an assessment site is a commonly used indicator of the site's overall anthropogenic stressors (O'Neill et al. 1999; Tiner 2004; Brown and Vivas 2005). To extract this information from the NLCD categorical maps, 16 land-cover classes from the original map were binned into five major land-use groups that best represent the anthropogenic land-use disturbance gradient found within the study area: 1) unmanaged lands, 2) low-intensity agriculture, 3) high-intensity agriculture, 4) low-intensity urban, and 5) high-intensity urban.

Each of the five land-cover groupings was subjectively weighted to best represent the degree of divergence from land-cover characteristic of undisturbed conditions typical of Rocky Mountain valleys (Table 1). Within each assessment area, the buffer (Met_{BP}) and floodplain (Met_{FP}) areas were scored separately using Equation 1:

$$Met = \frac{\sum_{x=1}^{X}(\sum C_{Lx} * w_{Lx})}{N} \qquad \text{(Equation 1)}$$

Where the metric score (Met) for the buffer or floodplain assessment area is equal to the total raster cells per cover class (C_{Lx}) multiplied by the sub-score for that class (w_{Lx}) from Table 1, summed across all classes (x) then divided by the total cell count (N) of the assessment area to obtain a score that ranges between 0.0 and 1.0. The closer the metric score is to 1.0, the more likely the area has a land-cover characteristic of an undisturbed system. A score closer to 0.5 represents agricultural land-cover and 0.0 represents an area dominated by urban land-cover.

Table 1. NCLD cover types binned to reflect a gradient of major land-use categories and the weighted sub-score assigned to each category reflecting the gradient of land-use intensity used in the perturbation metric.

Buffer and Floodplain Land-use Criteria	**Weighted Sub-score**
Unmanaged Land-cover: Land-cover characteristic of Rocky Mountain floodplain systems, which include open water, forest, shrub, herbaceous and wetlands cover classes. NCLD Codes 11, 12, 41, 42, 43, 52, 71, 90, and 95.	1.0
Low Intensity Agriculture: Herbaceous areas used for pasture and hay. NCLD code 81.	0.8
High Intensity Agriculture: Cultivated row crops. NCLD code 82.	0.5
Low Intensity Urban: Developed open space and low intensity developed lands. NCLD codes 21 and 22.	0.2
High Intensity Urban: Barren ground (predominantly gravel mines but also includes to a much lesser extent cobble) as well as medium and high intensity developed lands. NCLD codes 23, 24, and 31.	0.0

Habitat Fragmentation Metric for Buffer and Floodplain Areas (Met_{BF} and Met_{FF}): Perturbation metrics above assess the extent of human alteration; however two sites with the same relative abundance of unmanaged land could provide different levels of structural support for native biota depending on the degree of fragmentation (Vogt et al. 2007). Our fragmentation metric measures the degree of continuity within landscape patterns (Gustafson 1998; O'Neill et al. 1999). I used a morphological spatial pattern analysis (MSPA) GIS tool (Joint Research Station 2014) to identify the extent of contiguous and isolated patches, perforations within those patches by agriculture and urban area, and the amount of edge between these managed and unmanaged lands. The MSPA consists of cover types in Table 1 binned to create a binary map of unmanaged and managed lands collectively found within the buffer and floodplain (Table 2). The output of the MSPA tool is a map containing a mutually exclusive set of seven patch and edge structural classes (Vogt et al. 2007; Soille and Vogt 2009; Suarez-Rubio et al. 2012): 1) core areas, 2) patch edges, 3) loops, 4) bridges, 5) branches, 6) islets, and 7) managed lands. Each structural class was subjectively assigned a weighted sub-score that represents the degree of fragmentation or edge (Table 3). The structural class assignments were then clipped to each buffer and floodplain assessment site.

Table 2. NCLD cover types collapsed into a land-use binary map made up characteristic and non-characteristic land-cover.

Aggregated Land-use groups	NLCD Classification Code
Unmanaged Lands	11, 12, 41, 42,43, 52, 71, 90, 65
Managed Lands	31, 81, 82, 21, 22, 23, 24

The fragmentation metric score for both the buffer (Met_{BF}) and floodplain (Met_{FF}) was calculated using Equation 1 where total raster cells per MSPA structural class (C_{Lx}) at each site was determined and multiplied by the (w_{Lx}) from Table 3. The closer the metric score is to 1.0, the more the likely the area has contiguous land-cover characteristic of an undisturbed system and the closer to 0.0, the more likely the area has a contiguous cover of managed land.

Table 3. Description of structure categories of the fragmented landscape and the weighted sub-score assigned to each category reflecting the gradient of habitat quality used in the fragmentation metric.

Fragmentation Structure	Weighted Sub-score
Core Areas – pixels of unmanaged lands inside of a defined 90-meter (3 pixels) wide patch width (pixel value from a post MSPA map are 17, 117)	1.0
Patch Edge – pixels of unmanaged lands that are comprised of patch edge adjacent to managed land-cover type (MSPA pixel value 3, 5, 35, 67, 103, 105, 135, 167)	0.8
Loop – pixels that connect one patch of core unmanaged lands to the same core area and are completely made up of edge (MSPA pixel value 65, 69, 165, 169)	0.6
Bridge –pixels that connect one patch of core unmanaged lands to another core area and are completely made up of edge (MSPA pixel value 33, 37, 133, 137)	0.6
Branch – pixels that emanate from core, bridge, or loops into managed lands and are completely made up of edge (MSPA pixel value 1, 101)	0.4
Islet – pixels of unmanaged lands within a patch of managed lands that is completely made up of edge (MSPA pixel value 9, 109)	0.2
Managed Lands – all remaining pixels (MSPA pixel value 0, 100)	0.0

3.2.2 Flathead River Floodplain Condition Index

Finally I applied the index model (Equation 2) to calculate the Flathead River floodplain habitat condition based on land-use intensity and habitat fragmentation:

$$\text{Index} = (((\text{Met}_{BP} + \text{Met}_{BF})/2) + \text{Met}_{FP} + \text{Met}_{FF})/3 \qquad \text{(Equation 2)}$$

The condition of the buffer influences the condition of the floodplain (Allan 2004); therefore I first averaged the buffer metrics (Met_{BP} and Met_{BF}) to determine its condition. I then add that product to the floodplain metrics (Met_{FP} and Met_{FF}) and averaged the final product to provide a score between 0-1. Scores closer to 0.0 represent a disturbed landscape and scores closer to 1.0 represent an intact ecosystem in excellent condition. This MMI provides a naive estimate of ecological condition and is, in essence, the data collection component of the methods. The following data analysis methods address the impact of input map error on these results.

3.3 Data Analysis

I address map misclassification effects on the MMI results by first reducing map error from the original NLCD 2006 map (MRLC 2013) where possible without additional data collection. I then incorporate the remaining unavoidable error into the metrics and index. Finally I test bias of the naive MMI results when I incorporated the remaining error.

3.3.1 Reducing Uncertainty

Two maps were created for the study area: 1) a land-use map used to assess the two perturbation metrics, and 2) a binary map used to assess the two fragmentation metrics. Each map was created by aggregating thematic classes from the original data and thereby decreasing the thematic resolution of the original land cover classification. I also aggregated the confusion matrix from the original accuracy assessment to create new confusion matrices for each new map and, calculated the associated accuracy indices (Congalton and Green 2008), and compared these to the original 2006 NLCD accuracy indices (Fry et al. 2011) to determine the effects of changing thematic resolution on error.

3.3.2 Error Simulation Model

Simulation models that use available confusion matrix information to account for misclassification error were developed in the 1990s (Fisher 1994; Hess and Bay 1997; Wickham et al. 1997). These models convert confusion matrix user's or producer's accuracy information to a matrix of probabilities that inform the likelihood that an individual pixel is misclassified (Hess and Bay 1997). To meet the needs of potential resource managers, I created a matrix of probabilities based on user's accuracy. This "User Probability Matrix" (UPM) is the proportion of locations classified in the map as k_i (mapped pixels in class (k) found across all reference columns i through n) in a confusion matrix. For example, a hypothetical accuracy assessment is conducted on 100 randomly selected pixels mapped as forest (k). These mapped pixels are checked against ground reference data and 90 pixels are actually forest (k_1) and the remaining 10 are grassland (k_2). From these hypothetical accuracy data, our UPM would assume that there is a 90% probability that any forested pixel in our map is actually forest and a 10% probability that it is actually grassland. Following this, I created UPMs for all thematic classes from the confusion matrices of both the perturbation land-cover and binary fragmentation input maps (Tables 4 and 5).

Table 4. User probability matrix represents the likelihood that a pixel on the perturbation map is actually one of several ground reference pixels. UPM is used to support the perturbation metric simulation.

	Reference				
Map	Unmanaged Lands	Low Intensity Agriculture	High Intensity Agriculture	Low Intensity Urban	High Intensity Urban
Unmanaged Lands	93.10	3.24	1.68	1.78	0.20
Low Intensity Agriculture	16.32	77.29	1.25	4.88	0.26
High Intensity Agriculture	4.02	5.50	88.05	2.40	0.03
Low Intensity Urban	19.96	5.10	5.14	65.40	4.40
High Intensity Urban	18.32	0.81	0.27	8.31	72.29

Table 5. User probability matrix represents the likelihood that a pixel on the fragmentation map is actually one of several ground reference pixels. UPM is used to support the fragmentation metric simulations.

	Reference	
Map	Unmanaged Lands	Managed Lands
Unmanaged Lands	93.10	6.90
Managed Lands	10.39	89.61

In geographic studies, it is accepted that 'nearby things are more similar than distant things' (Tobler 1970) and is the basis of most spatial autocorrelation studies and tools (Goodchild 2004). Because I did not collect additional data, I could not assess the spatial structure of the error. Therefore, in the second step of our simulation model, I incorporated an autocorrelation filter proposed by Wickham et al. (1997) which assumes an overall 10 percent difference in the classification error between the edge and interior pixels of a land-cover patch resulting from the influence of correlation between classified pixels (Congalton 1988). Applying a 10% spatial autocorrelation filter decreases the likelihood classification error within patches (salt and pepper errors) and increases the likelihood of misclassifications near patch boundaries that are generally associated with errors resulting from with mixed pixels and spatial misregistration. I applied a 3x3 moving window to locate the interior and edge of patches in the two metric input maps, and created filters that decreased the effects of the UPM by 5% for the interior pixels and increased the UPM by 5% at the patch edge. Finally, I tested the Wickham et al. (1997) 10% autocorrelation modification against a 20% gradient to determine the sensitivity of the simulated index results to these modifications.

Finally, to account for the remaining classification error I applied a confusion frequency simulation Monte Carlo model (CFS) that takes advantage of the *a priori* error probabilities in the UPMs to create stochastic realizations of our perturbation and fragmentation input maps

(Fisher 1994; Wickham et al. 1997). For each simulation, the CFS: 1) identified cover class k assigned to an individual map pixel, 2) drew a random variable from a uniform (0, 1) distribution, 3) adjusted the random variable with the autocorrelation filter, 4) looked up the probabilities associated with all reference classes ($k_1 - k_n$) in the UPM, 5) assigned reference class k_i to the output simulation for that cell, based on the modified random value and user probability, and 6) repeated the process for all remaining classes to create a single simulated realization of the map. The CFS was conducted under the assumptions that 1) each pixel was eligible for selection, and 2) each pixel was classified independently (Hess and Bay 1997). With this process, 1000 Monte Carlo simulations were created for each map. For the fragmentation map, the MSPA tool was applied to each simulated output.

3.3.3 Metric and Index Error Assessment

Following each simulation I calculated a buffer and floodplain score for each metric (Equation 1) and total index score (Equation 2) generating a distribution of 1000 potential metrics and condition scores. It was assumed that each Monte Carlo simulation was an independent sample of that classification error and that the distribution of simulated metric and index scores represented a raw stochastic sample of the error model behavior. I did not make assumptions about the structure of the simulated distributions; therefore I chose a Wilcoxon signed rank test to test for differences between simulated site results. Additionally, to give an estimate of the potential variability in metric and index scores due to misclassification, 95% confidence intervals around the mean simulated score were derived from the 2.5th and 97.5th percentile of the metric and index scores distribution. The mean was chosen over the median as a conservative estimate of that distribution. Finally, the difference between original naive and simulated scores determined the bias of naive assessment.

4 Results

4.1 Naive Multi-Metric Index Results

Typical of most MMIs, the initial results of this model are reported assuming that the input data are free from error. The final naive perturbation and fragmentation metric and index scores (Table 6) articulated in the synoptic map (Figure 3), closely matched the land-use / land-cover gradient across the study area (Figure 2). Areas with intact, unmanaged lands scored in the upper index range (>0.90), areas with a mix of low intensity agriculture and unmanaged lands scored in the middle range (~0.70 – 0.80), and areas with a mix of high and low intensity residential, agriculture and unmanaged lands scored toward the lower end of the range (0.50- 0.70).

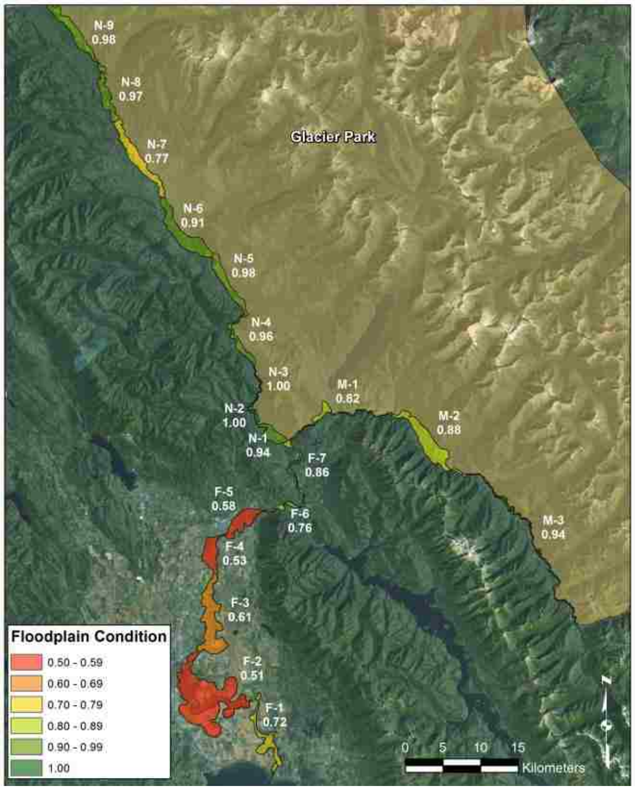


Figure 3. Synoptic map of Flathead River MMI scores.

Table 6. Metric and index results for the naive assessment of the case study.

	Perturbation Metrics		Fragmentation Metrics		Index
Site	Buffer	Floodplain	Buffer	Floodplain	Score
F-1	0.69	0.93	0.17	0.79	0.72
F-2	0.58	0.79	0.07	0.41	0.51
F-3	0.64	0.88	0.09	0.57	0.61
F-4	0.62	0.84	0.06	0.42	0.53
F-5	0.59	0.79	0.26	0.53	0.58
F-6	0.83	0.85	0.75	0.72	0.79
F-7	0.86	0.93	0.76	0.83	0.86
M-1	0.94	0.86	0.90	0.67	0.82
M-2	0.98	0.92	0.98	0.73	0.88
M-3	0.96	0.95	0.95	0.92	0.94
N-1	0.98	0.98	0.96	0.87	0.94
N-2	1.00	1.00	1.00	1.00	1.00
N-3	1.00	1.00	1.00	1.00	1.00
N-4	1.00	0.99	0.99	0.90	0.96
N-5	1.00	0.99	0.99	0.97	0.98
N-6	0.99	0.97	0.95	0.80	0.91
N-7	0.97	0.91	0.85	0.48	0.77
N-8	1.00	0.99	1.00	0.93	0.97
N-9	0.88	1.00	1.00	1.00	0.98

4.2 Map Classification Resolution

Aggregating land-cover groups lowered the resolution of thematic classifications in the original dataset from 16 classes to 5 classes for the perturbation map (Table 1) and 2 classes for the fragmentation map (Table 2). The 2006 NLCD map reported, at a national scale, an overall map accuracy of 78% for maps classified into their standard 16 Level 2 land-cover classes (Wickham et al. 2013). For the perturbation metrics, the original 16 x 16 confusion matrix collapsed into a 5 x 5 matrix, increasing the overall accuracy to 90% (Table 7). For the fragmentation metrics, a 2 x 2 confusion matrix summarized the binary cover classes with an overall accuracy of 92% (Table 8).

Table 7. Confusion matrix of five land-use classes of the perturbation input map with supporting statistics.

	Reference						
Map	Unmanaged Lands	Low Intensity Agriculture	High Intensity Agriculture	Low Intensity Urban	High Intensity Urban	Total	**User Accuracy**
Unmanaged Lands	65.61	2.28	1.19	1.25	0.14	70.47	**93.10%**
Low Intensity Agriculture	1.08	5.13	0.08	0.32	0.02	6.64	**77.29%**
High Intensity Agriculture	0.64	0.88	14.08	0.38	0.01	15.99	**88.05%**
Low Intensity Urban	0.94	0.24	0.24	3.06	0.21	4.69	**65.40%**
High Intensity Urban	0.41	0.02	0.01	0.19	1.61	2.23	**72.29%**
Total	68.67	8.55	15.60	5.21	1.98	100.00	
Producer Accuracy	**95.53%**	**60.02%**	**90.28%**	**58.87%**	**81.15%**		
Total Accuracy	**90%**						

Table 8. Confusion matrix of binary land-use classes of the fragmentation input map with supporting statistics.

	Reference			
Map	Unmanaged Lands	Managed Lands	Total	**User Accuracy**
Unmanaged Lands	65.61	4.86	70.47	**93.10%**
Managed Lands	3.07	26.47	29.54	**89.61%**
Total	68.67	31.33	100.01	
Producer Accuracy	**95.53%**	**84.49%**		
Total Accuracy	**92 %**			

4.3 Confusion Frequency Simulation Results

For the error simulation model, user probability matrices (Tables 4 and 5) and autocorrelation filters were used in the confusion frequency simulations to provide a distribution of metrics and index scores, with a 95% confidence intervals (Table 9 and Figure 4)[2]. The simulated and naive results are very similar and closely match the LULC gradient across the study area (Figure 2). A pairwise Wilcoxon signed rank test was applied to all simulated index sites using both the 10% and 20% autocorrelation filter under the null hypothesis that there are no differences between the simulated sites. For sites N.2 and N.3 there was very strong evidence that they have the same mean index score (p-value equal to 1.0) using the 10% filter but there is strong evidence that all sites are different (p-value <2.2e-16) using the 20% filter. Sites N.2 and N.3 both had naive score of 1.0 and all other naive scores were different. All remaining sites failed to support the null hypothesis showing strong evidence of a difference between sites (p-value < 2.2e-16) for both filters.

4.4 Sensitivity of the simulated results to land-Cover

Information from two sites with very different land-covers (N-3 and F-4) provides a graphical example of assessment metrics and index response to map misclassification. Site N-3 is located adjacent to Glacier National Park and is classified in the original NLCD map as 99.7%

[2] Mean and 95% confidence interval for the metric scores are available in appendix A.

unmanaged lands and 0.3% low intensity agriculture (Figure 5 and Table 10). Site F-4 is located in the Kalispell Valley and contains a portion of the town of Columbia Falls, MT and nearby agriculture activities. The original land-use intensity classified this site as 22.6% characteristic lands, 42.2% and 19.8% low and high intensity agriculture respectively and 13.6% and 1.8% low and high intensity urban respectively (Figure 6 and Table 10). The landscape pattern structural classes in the two sites (Table 11) also reflect the land use distributions. Site N-3 received metric and index scores of 1.0 for the naive assessment consistent with its nearly contiguous cover of unmanaged lands (Table 12). Site F-4 scored 0.61 for the naive index score consistent with its urban and agricultural land-use mixed with patchy unmanaged land cover.

Table 9. Distribution of index results and associated confidence intervals from the 1000 Monte Carlo confusion frequency simulations using the 10% autocorrelation filter and naive index results.

Site	Simulated Index Score			Naive Index Results
	2.50%	50%	97.50%	
F.1	0.695	0.698	0.701	0.716
F.2	0.509	0.510	0.511	0.510
F.3	0.591	0.593	0.595	0.605
F.4	0.528	0.531	0.534	0.534
F.5	0.583	0.586	0.589	0.583
F.6	0.767	0.773	0.779	0.788
F.7	0.828	0.833	0.838	0.858
M.1	0.793	0.797	0.800	0.816
M.2	0.851	0.853	0.855	0.878
M.3	0.913	0.916	0.920	0.944
N.1	0.903	0.907	0.911	0.938
N.2	0.963	0.970	0.977	0.999
N.3	0.963	0.971	0.977	0.999
N.4	0.926	0.930	0.933	0.962
N.5	0.951	0.954	0.957	0.985
N.6	0.876	0.878	0.881	0.911
N.7	0.745	0.747	0.749	0.768
N.8	0.941	0.944	0.946	0.974
N.9	0.949	0.951	0.954	0.980

For illustrative purposes, a single simulation was performed using the UPM from Tables 4 and 5, and the 10% autocorrelation filter to create the map realizations in Figures 5 and 6. The simulated realization reflects potential errors along patch edges and salt and pepper errors within patches (Figures 5 and 6: Panels B and D). These simulated errors decreased the overall cover of unmanaged lands in Site N-3 by about 2.4% as these pixels are reassigned to low intensity agriculture and urban land-cover (Table 10). These reassigned pixels are peppered across the landscape (Figure 5: B and D) and changed the composition of the landscape pattern structural classes (Table 11). These map changes resulted in a slight decrease in the buffer and floodplain perturbation metrics of 0.005 and 0.004 respectively, a larger decrease in the buffer and floodplain fragmentation metric scores of 0.057 and 0.053 respectively and an overall decrease in the index from 1.0 to 0.97 (Table 12).

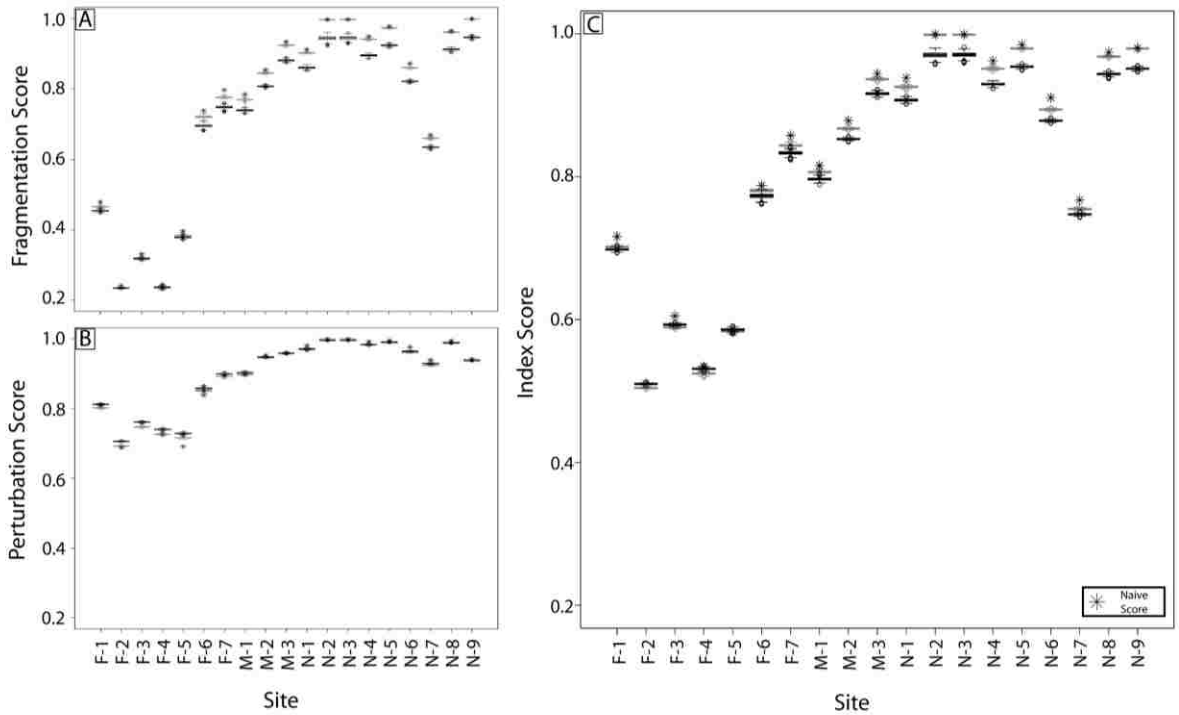


Figure 4. Naive data (stars) and distribution boxplots of simulated fragmentation (A), perturbation (B) scores averaged from the buffer and floodplain results, and index (C) scores with the 10% autocorrelation filters (black) and 20% autocorrelation filters (gray).

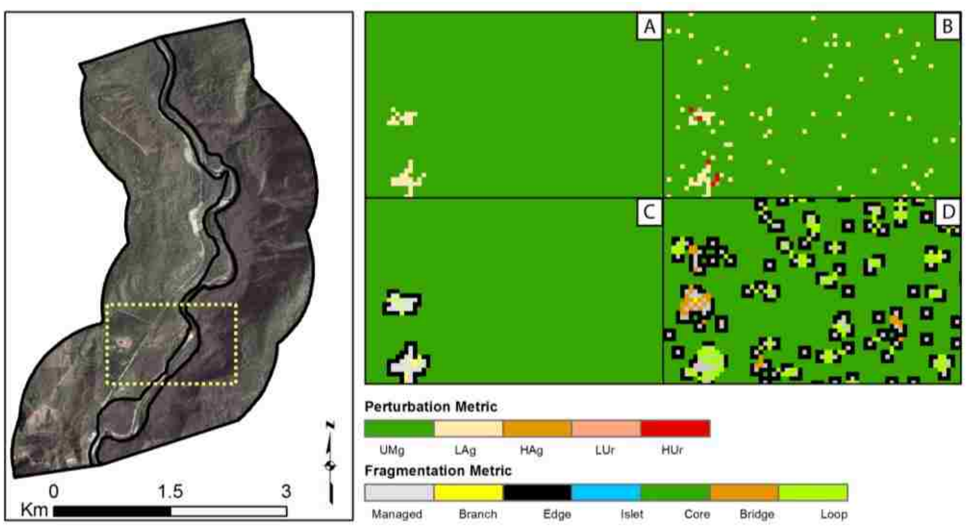


Figure 5. Naive and simulated perturbation maps (A & B respectively) and fragmentation maps (C & D respectively) for Site N.3.

The pixels reassigned in the Site F-4 simulation increased the percent cover of unmanaged lands from about 23% to 32% mostly from former agriculture and low intensity urban sites. There was also slight increase the high intensity urban cover from about 2% to 4%. All these changes are along patch edges and peppered within the patches (Figure 6: B and D). Although there was an increase in the cover of unmanaged lands, there was a decrease in the continuous patch cover in these lands. The buffer areas have higher urban and agriculture cover and the redistribution of pixel classes resulted in an increase of both the mean perturbation and fragmentation metric scores in the buffer by 0.030 and 0.011 respectively. However the floodplain originally had higher cover of unmanaged lands and the redistribution of pixel classes in the simulated map decreased the cover of unmanaged lands which decreased both the mean perturbation and fragmentation metric scores in the buffer by 0.009 and 0.018 respectively. After the calculation of the index, the changes in the metric scores were essentially eliminated and with no change between the naive and mean simulated index that both scored 0.53 after rounding (0.534 and 0.531 respectively: Table 12).

Table 10. Percent of land-cover classes from the original and simulated maps for Sites N-3 and F-4.

		Percent cover of Perturbation Classes				
		Unmanaged Lands	Low Intensity Agriculture	High Intensity Agriculture	Low Intensity Urban	High Intensity Urban
Site F-4	Original	22.67	42.23	19.76	13.56	1.79
	Simulation	31.51	35.25	18.61	10.25	4.39
Site N-3	Original	99.73	0.27	-	-	-
	Simulation	97.58	2.37	-	0.01	0.04

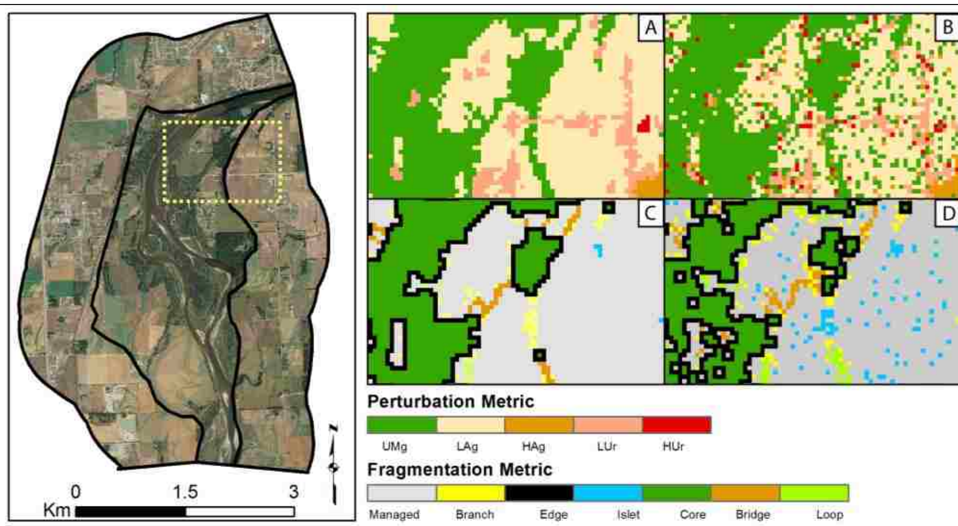


Figure 6. Naive and simulated perturbation maps (A & B respectively) and fragmentation maps (C & D respectively) for Site F.4.

Table 11. Percent of landscape pattern structural classes from the original and simulated maps for Sites N-3 and F-4.

		Percent cover of Landscape Pattern Structures Classes						
		Core	Edge	Loop	Bridge	Branch	Islet	Managed Lands
Site F-4	Original	11.66	6.63	0.47	0.53	2.04	1.34	77.33
	Simulation	7.89	7.28	1.14	1.90	2.82	4.55	74.43
Site N-3	Original	99.52	0.42	0.06	-	-	-	-
	Simulation	74.85	17.43	4.10	0.24	0.05	-	3.34

Table 12. Metric and index results for naive and simulated distribution for Sites N-3 and F-4 including resulting bias.

		Perturbation		Fragmentation		Index
		Buffer	Floodplain	Buffer	Floodplain	
Site F-4	Original	0.62	0.84	0.06	0.42	0.53
	Simulation	0.650 (+/- 0.004)	0.831 (+/- 0.004)	0.071 (+/- 0.003)	0.402 (+/- 0.007)	0.531 (+/- 0.003)
	Bias	-0.030	0.009	-0.011	0.018	-0.001
Site N-3	Original	1.00	1.00	1.00	1.00	1.00
	Simulation	0.995 (+/- 0.001)	0.996 (+/- 0.002)	0.943 (+/- 0.006)	0.947 (+/- 0.020)	0.971 (+/- 0.007)
	Bias	0.005	0.004	0.057	0.053	0.029

4.5 Metric and Index Bias

Bias between the naive index score and the simulated results was determined using the 10% autocorrelation filter. The difference between the naive score and total distribution of simulated scores indicated a bias in the estimation of the index and metrics resulting from misclassification (Figure 7). The fragmentation metric showed a greater bias in sites dominated by unmanaged lands (Figure 7A). Within the perturbation metric, sites with heterogeneous land-use had a negative bias between the naive and simulated results (Figure 7B). Collectively, there was a positive bias between most naive and simulated index results with the highest bias in sites dominated by unmanaged lands (Figure 7C).

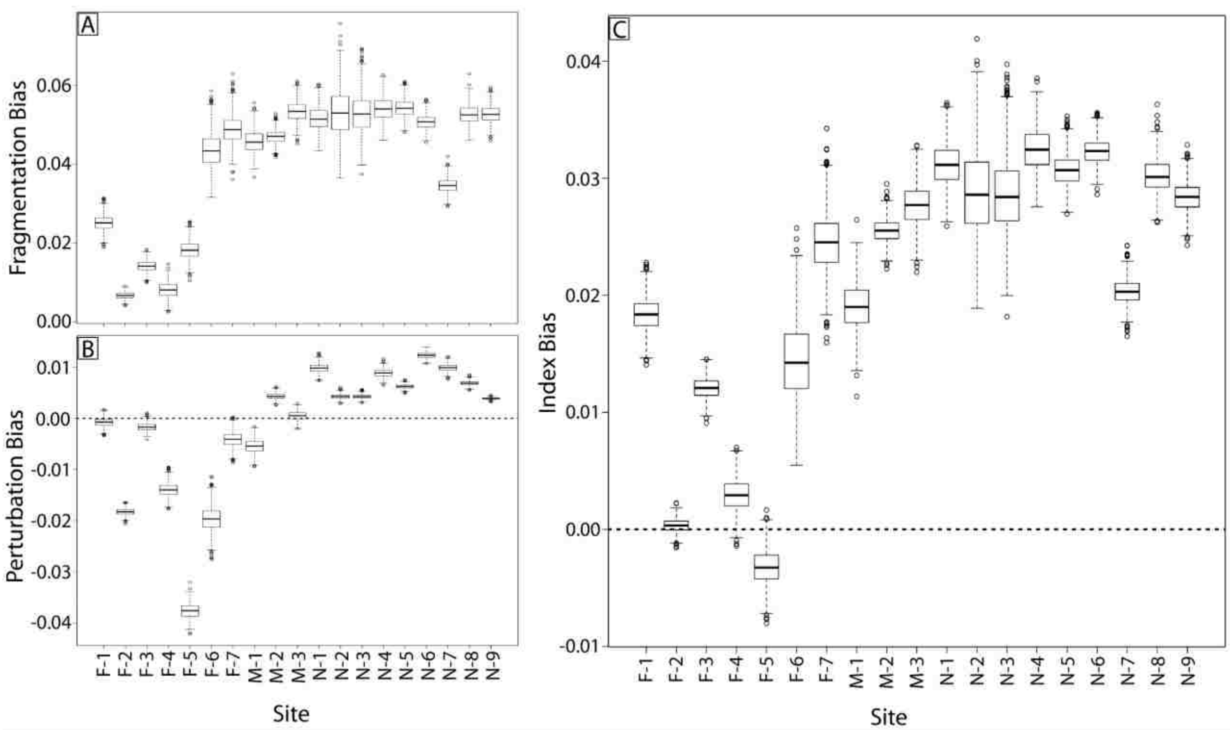


Figure 7. Distribution boxplots of bias of fragmentation (A), perturbation (B) scores averaged from the buffer and floodplain results, and index scores (C) for each assessment site.

5 Discussion

The confusion frequency simulation error model used here reveals that classification error affects assessment results in four important ways. First, naive results common to many large landscape assessment and monitoring efforts provide a biased estimate of habitat condition compared to results that include error. Second, depending on the land-cover composition of the assessment site, the magnitude and direction of this bias changes (Figure 7 and Tables 10-12). Third, the magnitude and direction of the bias is independent for each metric (Figure 7). Finally, when these metrics are combined into an index, this bias is partially attenuated (Figure 7 and Table 12).

All maps contain errors, and accuracy assessments provide insight into the extent and nature of misclassifications that are present. The confusion matrix is a foundation of classification accuracy assessment (Foody 2002). The NLCD 2006 map used here provides a confusion matrix associated with an accuracy assessment conducted at continental scale only (Wickham et al. 2013). Fang et al. (2006) found that confusion matrices developed closer to the site of interest have much different error rates than regional or continental matrices. However, the map user will be limited to the data provided unless they conduct their own accuracy assessment effort. At any scale, the confusion matrix also has its own suite of inherent uncertainties. For instance, collection of reference data can also contain unmeasured sources of error (Foody 2002); ground accuracy assessment teams may be inconsistent in the classification of mixed land-cover in the assessment area or stratified random reference samples that may not capture spatially specific classification error (e.g., near patch edges). Additionally, although a confusion matrix is excellent at capturing thematic errors of omission and commission, it cannot capture all the non-thematic error that affects classification such as misregistration of the image with ground data (Stehman 1997). Ultimately, obtaining a reliable confusion matrix and associated Kappa indices can be problematic (Pontius and Millones 2011); however it currently remains the core accuracy assessment tool (Foody 2002).

Confusion frequency simulation error models developed for categorical thematic maps use available information from the confusion matrix to account for error resulting from misclassification (Fisher 1994; Hess and Bay 1997; Wickham et al. 1997; Langford et al. 2006). In the simulated realizations used here, pixels within the homogeneous unmanaged land-cover are reclassified according to the user probability matrix resulting in increased land-use heterogeneity and thereby lower assessment metric and index scores (Tables 9). In contrast sites with heterogeneous land uses are remixed to an alternative version of heterogeneity resulting in a simulated map that may have higher or lower assessment scores depending on the ratio and spatial composition of managed to unmanaged lands in the original map (Figures 2 and 4).

Because I intentionally did not collect site-specific map accuracy data, I remain ignorant of the spatial structure of the map error. However, I recognize that spatial autocorrelation affects the extent of misclassification within and between land-cover patches (Congalton 1988). When applied here, the 10% spatial autocorrelation filter decreases the randomly located misclassifications within patches (salt and pepper error) and increases the misclassifications near patch boundaries. However when applying the 20% autocorrelation filter this effect is exaggerated resulting in simulated results that trend toward the naive results and an overall decrease in bias between the naive and simulated index scores (Figure 4C). Without collecting the required local reference data to test the true relationships with autocorrelations, I felt it was best to be conservative in the face of uncertainty (Armstrong 2001) and applied the 10% autocorrelation filter to the CFS error model. Ultimately, without an estimate of the structure of the spatial error, our simulation will likely contain its own misclassifications. However, our simulated values of ecological condition provide a more conservative estimate than our naive model results.

A remote sensing product such as the NLCD (MRLC 2013) is an appealing source of information for regional ecosystem assessment and monitoring. The NLCD provides thematic land-cover information and accuracy assessments that do not require the end-user to conduct the expensive and time consuming (Foody 2002; Fang et al. 2006) necessary steps to process and analyze raw Landsat imagery or to collect additional accuracy assessment data (Homer et al. 2004, 2007). The above approach is not intended as an assessment of the quality of the NLCD product; rather it is intended as a straight-forward approach that could be used with any number of land-cover products. As the ease of access to classified Landsat products increase and assessment tools expand to watershed or regional scales, the number of landscape metrics will likely expand as well, each with their unique sensitivity to classification error. Incorporating error sensitivity into the assessment model building process can help determine the level of classification errors that can be tolerated for existing and new landscape metrics and subsequent indices (Shao and Wu 2008). For instance, several authors have found that some landscape metrics are more sensitive to classification error than others (Hess and Bay 1997; Wickham et al. 1997; Shao et al. 2001; Langford et al. 2006). As our work has shown, metrics also respond differently to classification error across disturbance gradients associated with changes in LULC in each assessment site. Wickham et al. (1997) found that the actual differences in LULC composition needed to be at least 5% larger than the misclassification rate to be confident that differences in landscape metrics were not due merely to classification errors.

5.1 Implications of Land-cover Misclassification to Resource Decisions

Millions of dollars are spent annually in the U.S. on ecological monitoring, assessment, and restoration (Lovett et al. 2007; USEPA 2012). Landscape metrics and indices assist decision-makers with allocating limited funds by prioritizing monitoring, protection, and restoration efforts (Hyman and Leibowitz 2000; Lausch and Herzog 2002; Steel et al. 2004; Hierl et al. 2008). Landscape metrics and indices are also frequently used to refine or test finer-scale monitoring and assessment tools (Stein et al. 2009; Rains et al. 2013). Also, quality thresholds are frequently used to trigger management actions and addressing the effects of classification error on assessment metric and index scores can assist decision makers in determining which sites are above or below a those threshold. However, classification accuracy influence on landscape indices has been largely ignored (Shao and Wu 2008). Without error assessment, applications of large landscape models for conservation decisions or finer scale model development may be flawed.

Critical examinations of index-based approaches in the scientific literature (May 1985; Seegert 2000; Green and Chapman 2011) have addressed the short-comings of metrics and indices in terms of sensitivity, calibration, and information loss. What are not seen in the literature are the criticisms from the intended end-users of such tools. Even if the scientific criticisms are accounted for, these tools may fall into disuse when passed from scientist to end-user due to the overall lack of confidence in the assessment tool resulting from uncertainty in its input data, the metrics it uses and the output it creates. Tracking and reporting uncertainty is considered best practice in most remote sensing and quantitative efforts. Although scientists have a general operational definition of uncertainty based on a model's statistical properties, when applied to resource management uncertainty in scientific outcomes potentially translates into a state-of-confidence that the decision maker has in its application. Policy makers view these uncertainties in association with their management goals and priorities (Walker et al. 2003).

Reducing error where possible is a first step to address uncertainty. The initial dataset provided an overall accuracy of 78% for the 2006 NLCD continental-scale accuracy assessment but to create our assessment tool it was necessary to aggregate several of the land-cover categories into land-use groups thereby lowering map classification resolution resulting in an increase in overall accuracy to 90% for the perturbation map and 92% for the binary map (Tables 6 and 7). Although there are measured differences between the naive and simulated results of both the metrics and the index that imply caution in the use of naive results alone, there are no radical departures between the two results (Figure 4) likely because of the input maps' higher accuracies. However, merely providing information on error within the model results does not necessarily assist the end-user in their ability to absorb that uncertainty into their decision. Interpretation tools such as fuzzy sets and fuzzy operational rules make it possible to formalize the knowledge of experts to provide information to assist the tool end-user in areas where numerical data many be limited (Uricchio et al. 2004). Still, applying well-established approaches to characterize and interpret the degrees of uncertainty within data (e.g., rough sets, fuzzy sets, probability density functions) do not guarantee the assessment tool will be used. As a tool, index-based assessments exist in the difficult area between science and policy (Turnhout et al. 2007), and scientists and model builders are not necessarily involved in the ultimate use of their product as a decision tool. Ideally, during assessment tool development process the science team works with the policy and stakeholders team to create a product that accounts for uncertainty and clearly articulated limitations of the tool in a manner that is easily understood by the end-user so that the degrees and types of uncertainty in the tool output can be reasonably absorbed into their decision process in a straightforward manner (Niemi and McDonald 2004; Turnhout et al. 2007).

6 Conclusion

Our results elucidate the potential bias between the more common naive approach to ecological assessment and an approach that includes error. I show an increase in overall map accuracy as the 16 land-cover categories in the original NLCD thematic map was aggregated into the 5 land-use groups for the perturbation map and the 2 land-cover groups for our fragmentation map. The resulting assessment metrics within our multi-metric index respond in different ways to map error depending on the land-cover pattern of each assessment site. When combined into an index, it appears that naive scores slightly over-estimate ecological quality within sites comprised of contagious unmanaged lands associated with higher quality floodplains, and potentially under-estimate the quality in more disturbed sites comprised of heterogeneous land uses. Naive approaches are easier to implement but at a minimum recognizing that using such an approach is biased may help with the end-user's state-of-confidence.

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Appendix

Confusion Frequency Simulation Metric Results: Using the confusion frequency simulation each pixel retained its class assignment or was reassigned according to an outcome of a uniform random draw between 0 and 1 that was adjusted by the spatial autocorrelation filter. One thousand simulations of each metric were performed and an index score was calculated per iteration. These simulations provide a distribution of index scores and a 95% confidence interval given the probabilities of class assignment (Tables A-1 and A-2). The simulated data in Tables A-1 and A-2 are provided in three significant digits to demonstrate the limitations of the confidence intervals. The naive results are provided for comparison purposes and are reported in two significant digits which is a general precision standard for most 0-1 MMI results.

Table A-1. Perturbation metric results and confidence intervals from the 1000 Monte Carlo confusion frequency simulations and naive results for comparison.

	Buffer Perturbation			Naive Score	Floodplain Perturbation			Naive Score
Site	2.50%	50%	97.50%		2.50%	50%	97.50%	
F.1	0.705	0.707	0.709	0.69	0.914	0.917	0.920	0.93
F.2	0.620	0.622	0.624	0.58	0.788	0.790	0.791	0.79
F.3	0.658	0.660	0.662	0.64	0.861	0.863	0.865	0.88
F.4	0.646	0.650	0.653	0.62	0.827	0.831	0.835	0.84
F.5	0.652	0.655	0.659	0.59	0.798	0.802	0.806	0.79
F.6	0.856	0.859	0.862	0.83	0.847	0.857	0.865	0.85
F.7	0.873	0.875	0.877	0.86	0.919	0.925	0.930	0.93
M.1	0.939	0.941	0.942	0.94	0.860	0.865	0.870	0.86
M.2	0.981	0.982	0.982	0.98	0.912	0.914	0.917	0.92
M.3	0.965	0.966	0.967	0.96	0.948	0.951	0.954	0.95
N.1	0.977	0.979	0.980	0.98	0.960	0.963	0.966	0.98
N.2	0.994	0.995	0.996	1.00	0.994	0.996	0.997	1.00
N.3	0.994	0.995	0.995	1.00	0.994	0.996	0.998	1.00
N.4	0.994	0.994	0.995	1.00	0.969	0.972	0.975	0.99
N.5	0.992	0.992	0.993	1.00	0.985	0.987	0.988	0.99
N.6	0.979	0.979	0.980	0.99	0.947	0.949	0.951	0.97
N.7	0.961	0.962	0.963	0.97	0.894	0.897	0.899	0.91
N.8	0.995	0.995	0.996	1.00	0.979	0.981	0.982	0.99
N.9	0.880	0.881	0.881	0.88	0.993	0.994	0.994	1.00

Table A-2. Fragmentation metric results and confidence intervals from the 1000 Monte Carlo confusion frequency simulations and naive results for comparison.

	Buffer Fragmentation			Naive Score	Floodplain Fragmentation			Naive Score
Site	2.50%	50%	97.50%		2.50%	50%	97.50%	
F.1	0.165	0.168	0.170	0.17	0.733	0.740	0.747	0.79
F.2	0.077	0.079	0.081	0.07	0.387	0.390	0.392	0.41
F.3	0.092	0.094	0.097	0.09	0.535	0.540	0.545	0.57
F.4	0.068	0.071	0.074	0.06	0.395	0.402	0.409	0.42
F.5	0.251	0.255	0.260	0.26	0.494	0.501	0.508	0.53
F.6	0.709	0.715	0.722	0.75	0.657	0.676	0.692	0.72
F.7	0.713	0.717	0.722	0.76	0.766	0.780	0.792	0.83
M.1	0.845	0.849	0.852	0.90	0.620	0.631	0.640	0.67
M.2	0.920	0.924	0.928	0.98	0.685	0.691	0.697	0.73
M.3	0.893	0.896	0.899	0.95	0.857	0.866	0.876	0.92
N.1	0.898	0.905	0.912	0.96	0.807	0.817	0.826	0.87
N.2	0.932	0.942	0.952	1.00	0.925	0.947	0.967	1.00
N.3	0.937	0.943	0.949	1.00	0.926	0.947	0.965	1.00
N.4	0.937	0.942	0.947	0.99	0.838	0.849	0.859	0.90
N.5	0.933	0.938	0.942	0.99	0.903	0.911	0.918	0.97
N.6	0.890	0.895	0.899	0.95	0.743	0.749	0.754	0.80
N.7	0.804	0.809	0.813	0.85	0.454	0.460	0.465	0.48
N.8	0.939	0.945	0.949	1.00	0.873	0.881	0.889	0.93
N.9	0.941	0.946	0.951	1.00	0.940	0.947	0.954	1.00

CHAPTER IV: FIRE AND FLOOD EXPANDING THE SHIFTING HABITAT MOSAIC CONCEPT

1 Abstract

The floodplain Shifting Habitat Mosaic (SHM) Concept suggests that habitat patch dynamics in space and time are influenced by hydrologic disturbance driven by flood pulses of sufficient power to initiate incipient motion of the substratum and maintain cut and fill alluviation of the channel and banks. However, where the floodplain ends the upland begins and along with it are other important disturbance regimes that frequently function at landscape spatial scales. In the Rocky Mountains of both the U.S. and Canadian fire is an important terrestrial disturbance that directly effects floodplain habitat patch composition. I examined the intersection of hydrologic and terrestrial disturbances on floodplain habitat patch composition across the aquatic - terrestrial ecotone and its resultant extension of the SHM concept. I sampled the floodplains along the North Fork of the Flathead River; a free-flowing river in southeastern British Columbia, Canada and flowing into northwestern Montana, USA. I used remotely sensed imagery, meteorological inputs, empirical and modeled rainfall-runoff data, fire location and frequency data and anthropogenic land-use data over a 22 year period (1991-2013) to examine hierarchical relationships between hydrology, fire, anthropogenic disturbance, geomorphic position and floodplain habitat patch dynamics. These factors, across space and time, influence disturbance, disturbance/recovery pathways, and stability of floodplain habitats and their spatial and temporal dynamics. I used path analysis (i.e., a form of multiple regression) to reveal that fire had the strongest direct effect on floodplain habitat patch composition (0.32 – 0.43 across all years) and that stream power and geomorphic position having a moderate direct effect on floodplain habitat patch mosaics (0.02 – 0.25, and 0.07 – 0.23 respectively across all years). Collectively, these three factors explained 13% – 26% (across all years) of the variance in floodplain habitat patch composition. Graphical analysis was used to examine the locations and intensity of disturbance and recovery pathways across riparian transition zones throughout the 22 years of the study period. Path analysis and graphical approaches support the hypothesis that a blending of aquatic and terrestrial disturbance regimes and their resulting recovery pathways maintain the SHM across the floodplain area of this system.

2 Introduction

The hydrologic disturbances that dominate free-flowing rivers play important roles in shaping the floodplain surface and maintaining floodplain habitat patch dynamics (Tockner et al. 2000, Stanford et al. 2005). However, riverscapes are a subset of larger landscapes and are subject to terrestrial disturbances that act at a different spatial and temporal scale than hydrologic disturbance regimes. Because floodplains are transition zones between aquatic and terrestrial systems, they are exposed to both hydrologic and terrestrial disturbances that collectively influence floodplain habitat patch composition and dynamics. In this chapter, I explore the dominant disturbance factors, both hydrologic and terrestrial, that influence floodplain habitat composition of a large free-flowing Transboundary Rocky Mountain (Canada - United States) river by observing 22 years of disturbance and recovery pathways. With this effort, I expand the Shifting

Habitat Mosaic concept (Stanford et al. 2005) to capture the effects of both hydrologic and terrestrial disturbance regimes on floodplain habitat patch dynamics.

From a hydrogeomorphic perspective, floodplains are characterized within the framework of landscape position, dominant water sources, and hydrodynamics (Brinson 1993, Montgomery 1999). The dominant water sources for floodplains are the rivers themselves from either surface or subsurface pathways; however, these waters also may have complex interactions with other local water sources (Mertes 1997, McGlynn and McDonnell 2003). The hydrodynamics of floodplains are influenced by flood and flow pulses (Junk et al. 1989, Tockner et al. 2000, Lorang and Hauer 2006), bank storage dynamics where surface and groundwater exchange occur between the channels and the underlying or adjacent deposits (Cooper and Rorabaugh 1963, Intaraprasong and Zhan 2009), and perirheic flows where river water interacts with regional and/or local waters from upslope surface and/or ground water sources (Mertes 1997, McGlynn and McDonnell 2003). The varied hydrodynamics result in multiple complex inundation and recession pathways (Hughes 1980, Lewin and Hughes 1980), with diverse energy gradients across the floodplain surface (Tockner et al. 2000, Tockner and Stanford 2002, Lorang and Hauer 2003). High energy flow pulses act at annual or sub-annual scales as the channel expands and contracts within the bankfull boundaries creating in-channel and near-channel erosional and depositional features (Tockner et al. 2000, Tockner and Stanford 2002, Lorang and Hauer 2006). Beyond the banks, high energy erosive flooding occurs at annual and supra-annual scales, which can result in channel avulsion and cut-and-fill alluviation (Ward et al. 2002, Stanford et al. 2005, Hauer and Lorang 2004, Slingerland and Smith 2004). The flood's erosive energy dissipates as it extends beyond the bankfull perimeter and disperses across the expansive surface of the floodplain (Ward et al. 2002) creating backwater flooding or inundation as it interacts with rising hyporheic and/or perirheic waters (Mertes 1997, McGlynn and McDonnell 2003). On large rivers, these energy and floodplain inundation gradients vary longitudinally both across large depositional floodplains and constrained reaches with limited floodplain surfaces, collectively determining the formation and maintenance of floodplain geomorphic features (Miall 1985) as well as the degree of connectivity across the river-floodplain transition (Tockner et al. 2000).

Riverscapes are a subset of the larger landscapes that surround them (Allan 2004) and are subject to terrestrial disturbance regimes. Fire has played a critical role in shaping the forests of the Western North America (Arno 1980, Keane et al. 2002) including floodplain forests (Dwire and Kauffman 2003, Pettit and Naiman 2007, Poff et al. 2011). Fire operates at different spatial and temporal scales and severities than the hydrologic disturbances that have been the focus of much of the floodplain literature. Fires in the Rocky Mountains have a spatially explicit return interval of about 30 to 100 years (Arno et al. 2000) and are generally mixed-severity events (Arno et al. 2000, Perry et al. 2011) composed of 'stand replacement burn areas', 'nonlethal burn areas', and intermediate aspects of both (Brown 1995). Ultimately these leave patchy, erratic patterns of mortality and survivorship (Arno et al. 2000). Mortality patterns produce diverse forest communities composed of mixed species/age-class mosaics with patch sizes ranging from a few square meters to tens or hundreds of hectares (Perry et al. 2011). Although mixed severity fires are poorly understood and poorly documented (Perry et al. 2011). It is estimated that they account for up to 50% of the composition of major forest types in the

Rocky Mountains (Schoennagel et al. 2004). Given the return intervals, these patches shift both spatially and temporally creating a landscape-scale terrestrial shifting habitat mosaic as described by Bormann and Likens (1979).

Fire regimes are influenced by top-down and bottom-up biogeoclimatic forces (Turner and Romme 1994, Perry et al. 2011). The combination of fuel abundance, species composition, micro-climate, and fuel and soil moisture gradients are major mechanisms of producing the mixed fire regime (Bekker and Taylor 2001, Perry et al. 2011). As described above, floodplains of gravel-bed rivers in the Rocky Mountains are characterized by transitions from the river's edge to terrestrial ecosystems within which are gradients of these elements that enhance or dissuade fire (Dwire and Kauffman 2003, Pettit and Naiman 2007, Poff et al. 2011). As the floodplain is exposed to fire, it may exhibit fire severity that matches the biogeoclimatic gradient. Although likely scale dependent, the lateral floodplain fire disturbance gradient may be conceptualized as approximately inverse to the hydrologic disturbance gradient, with lower severities closer the river's edge and higher severity fires likely to occur at the terrestrial/floodplain ecotone (Figure 1).

Herein I propose; 1) that both erosive flooding and fire are primary factors that shape floodplain patch composition, 2) that the floodplain habitat patch composition resulting from these disturbances and subsequent recovery vectors shifts in space and time, and 3) that these disturbance/recovery vectors are closely associated with geomorphic position during contemporary events. Collectively, by including these multiple disturbance vectors and subsequent recovery pathways, I expand the SHM concept of floodplains from one singularly driven by major hydrologic events to one blending hydrologic and terrestrial disturbances.

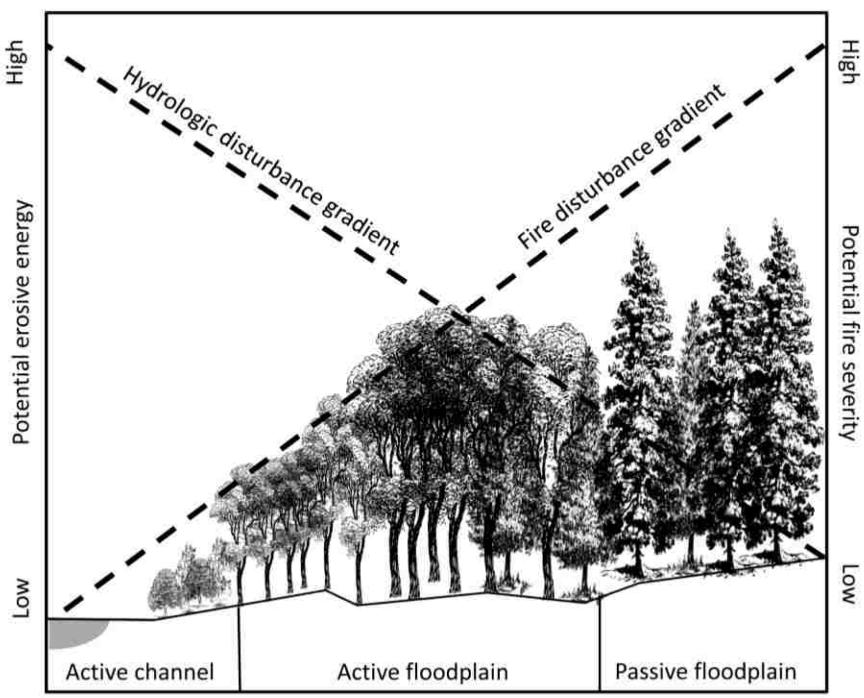

Figure 1. Conceptualized inverse lateral disturbance gradients across the river-to-upland floodplain transition zone.

3 Methods

Study Area - The study area is the North Fork of the Flathead River, a cobble dominated, free-flowing, snowmelt system in southeastern British Columbia, Canada and northwestern Montana, USA (Figure 2). This trans-boundary watershed is 4,057 km^2 with the river flowing from north to south in the first valley west of the continental divide. In British Columbia the lands within the watershed are predominantly managed by the BC Ministry of Environment, Lands and Parks and private land owners. In Montana, lands are managed by a mixture of National Park Service, U.S. Forest Service, State of Montana, Flathead County, and private land owners. In Montana, the North Fork comprises the western boundary of Glacier National Park. The mean annual precipitation is 560 mm falling predominantly as snow, and the mean annual temperature is 4.0°C, with monthly averages ranging from −14.1° C in January to 26.6° C in July (WRCC 2014). The highest elevation in this montane watershed is 3,078 meters above sea level at Kintla Peak in Glacier National Park.

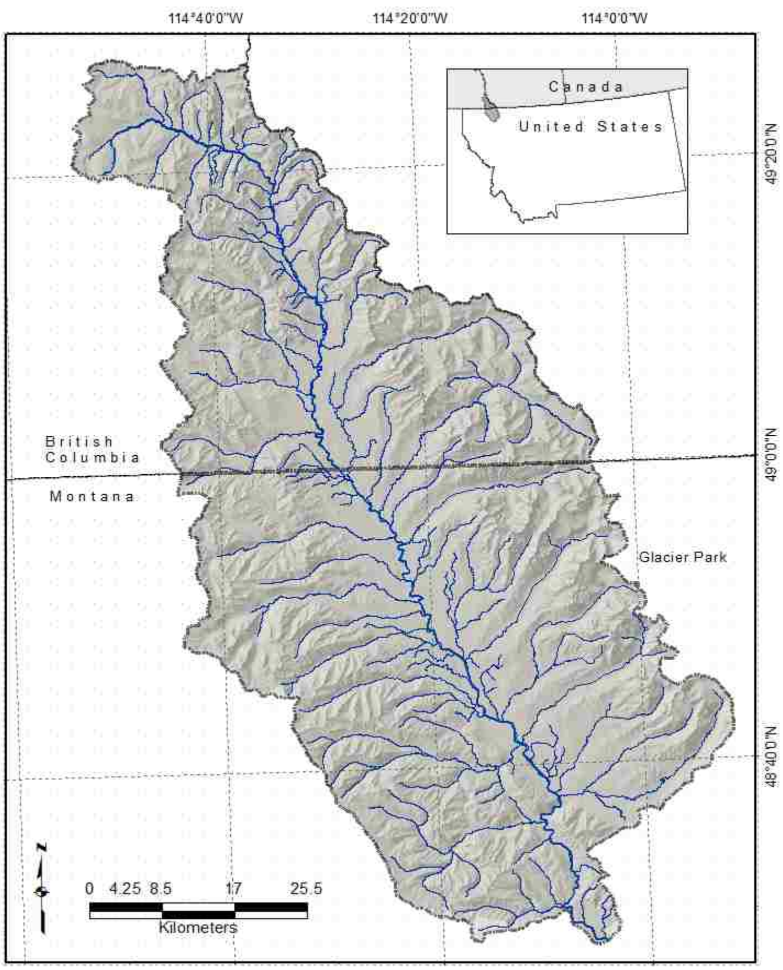

Figure 2. Location of North Fork Watershed and it major contributing streams.

Along its 160 km course, the North Fork ranges in elevation from 1,543 meters to 948 meters with a mean slope of 0.003 percent. At the U.S. Geological Survey (USGS) gage station near the bottom of the study area (USGS gage station #12355500), the North Fork had a mean annual discharge during the time span of this study (1980 to 2013) of 83 cms and a maximum discharge of 1,062 cms recorded on 6/8/1995. The greatest discharge for the period of record extending to 1911 was 1,957 cms on June 9, 1964. The 2-year return interval is ~ 585 cms, the 10-year return is ~ 850 cms and the 50-year flood is ~ 1120 cms (Omang 1992).

The study area floodplains within the North Fork's riverine influenced valley bottom are defined by the lateral extent of three generalized geomorphic zones: the active channel (river and parafluvial areas), the active floodplain (active accretion orthofluvial) and the passive floodplain (passive accretion orthofluvial) (Stanford et al. 2005). Within the bankfull boundaries, flow pulses maintain active channel features characterized by open water and its varied channel bed, depositional cobble surfaces and, at the ecotone between active channel and active floodplain, older cobble dominated bars with deciduous seedlings and forbs. The active floodplain generally contains erosive and depositional features from high energy floods including natural dikes, paleo-channels, backwater ponds, interspersed with islands of poorly developed soils dominated by herbaceous forbs and grasses, deciduous shrubs and trees including alders (*Alnus*), willows (*Salix*), and cottonwood (*Populus*), and patches of conifers (*Picea, Abies, Pseudotsuga*). The passive floodplain generally consists of benches with well-developed soils dominated by late successional riparian deciduous and/or coniferous gallery forest that interact, albeit rarely, with low energy floods and occasionally erode and reengage with the river via channel alluviation at the interface between the active channel and passive floodplain. Within these features, and the adjacent uplands, there is also evidence of recent and historic fires as well as logging, and sporadic agriculture and exurban developments.

Spatial extent of the study area was mapped in ArcGIS 10.0 (ESRI 2011) with the assistance of background imagery from USDA National Agriculture Imagery Program (NAIP: USGS 2014), from year 2005, overlain with a 30-meter digital elevation map (USGS 2013), visual assistance from oblique views within Google Earth's 3-D models (Google Earth 2013), oblique imagery from aerial reconnaissance, and multiple site visits.

3.1 Data collection

To fulfill the objectives of this study, four types of data were collected across the 1980-2013 study period: 1) fire extent and anthropogenic disturbance within the study area and period, 2) floodplain habitat cover types, 3) major geomorphic features within the floodplain, and 4) daily stream power as a surrogate measure of the potential erosive force of flooding at each inter-confluence reach via modeled discharge and slope. The 1980 to 2013 study period coincides with the rise of widely available, large scale data such as products derived from Landsat thematic imagery (e.g., fire extent and severity), high resolution orthorectified aerial imagery, and gridded meteorological datasets (e.g., DayMet: Thornton et al. 2012). Unless otherwise stated, all data collection and subsequent analysis was organized in ArcGIS 10.0 (ESRI 2011) and the R system for statistical computing (R Core Team 2013).

Fire extent – The mapped fire extent in the U.S. is provided by the Monitoring Trends in Burn Severity (MTBS) project which maps the severity and perimeters of large fires across all lands in the United States using Landsat TM satellite imagery from 1984 to 2012 (MTBS 2014). MTBS uses pre- and post-fire satellite imagery to establish fire perimeters and uses a Normalized Burn Ratio (NBR) to index the severity of the burn within the perimeter (Escuin et al. 2008). However, MTBS maps a large extent annually with limited plot-based confirmation of burn severity within each fire polygon (MTBS 2014). As a result uncertainties exist regarding specific locations of severity within each map. Therefore, I chose the discrete fire perimeter for the subsequent effects analysis and relied on my own sampled data to determine specific locations and disturbance response from fire within my study period. British Columbia government provides data of fire perimeter only (DataBC 2014) however all fires in the floodplain in BC happened prior to 1940 allowing for sufficient recovery of the forest, therefore, BC fires were not included in subsequent analysis. Table 1 provides a summary of the spatial extent of historic fires within the floodplain area.

Table 1. Years of fires within the study area and percent of floodplain area burned.

Location	1919	1929	1931	1936	1988	2001	2003
USA	-	-	-	-	11.9%	14.5%	13.1%
Canada	3.1%	20.2%	5.4%	55.3%	-	-	-

Source: US: (MTBS 2014), Can: (DataBC 2014)

Habitat classification – I used publicly available high-resolution imagery (1991, 2003, 2005, 2009, 2011, and 2013; Table 2) to classify floodplain habitat types within each inter-confluence reach for each year of the image data. All imagery was georectified by the provider and spatial accuracy was confirmed by comparing multiple points across all years. All points were within 5 meters, with the majority of locations within 2-3 meters across the entire image series with the exception of one 2003 quad which required a 100 meter correction.

Table 2. Publicly available aerial image sources used in the analysis.

Year	Image Source	Scale	Date of Image	Image Type	Availability
1991	DOQ	1 m	Late summer to early fall	Black and white	U.S.
2003	DOQ	1 m	Late summer to early fall	Black and white	U.S.
2005	NAIP	1 m	Mid to late summer	3-band natural color images	U.S.
2005	BC Imagery	0.5 m	Mid to late summer	3-band natural color images	Canada
2009	NAIP	1 m	Mid to late summer	3-band natural color images	U.S.
2011	NAIP	1 m	Mid to late summer	3-band natural color images	U.S.
2013	NAIP	1 m	Mid to late summer	3-band natural color images	U.S.

Note: NAIP imagery refers to the National Agriculture Imagery Program. DOQ imagery refers to digital orthophoto quadrangle, U.S. imagery available from The National Map Viewer ((USGS 2014b), B.C. imagery available at DataBC (DataBC 2014).

The aerial imagery was used to conduct analysis of multiple floodplain reaches on the North Fork. The upstream and downstream limits of these reaches were delineated at the confluences of the main stem and each major contributing stream. Major streams were defined as those whose bed-and-bank could readily be observed from the available aerial imagery. A few contributing major streams were nearly adjacent and those were combined resulting in 37 inter-confluence (IC) reaches, which range in size from 53 ha to 812 ha with a total area of 10,165 ha. The IC-reaches are numbered from (IC-1) at the bottom of the watershed to (IC-37) at the top of the watershed. The IC-reaches consisted of both broad alluvial depositional areas typically associated with floodplain ecosystems and confined reaches with limited floodplain. Aerial imagery for the entire watershed was available for 2005 only. However, reaches IC-1 through IC-22 had consistent imagery across all years covering 93 km of river and 6,320 hectares of floodplain. Site IC-22 begins at confluence of Sage Creek and the North Fork, 1.2 km downstream from the Canadian Border.

I used the generalized random tessellation stratified (GRTS) sampling function within the 'Spsurvey' R-package (Kincaid and Olsen 2013) to randomly select 100 un-stratified sample points within each IC-reach. These points remained fixed in space and were used to classify the floodplain habitat types across each of the 6 sample years. Through several site visits and previous studies (Hauer et al. 2002), I identified twelve floodplain habitat types that could be identified by aerial imagery (Table 3). The sample points were imported into ArcGIS and transformed into cross-hatched circles with a 5 meters diameter. Within ArcGIS, the aerial image was visually examined under each point from a fixed scale of 1:1500. The dominant cover within the cross-hatched circle was assigned to one of the twelve floodplain habitat types in Table 3. The black and white 1991 image was captured in the fall and provided a clear distinction between deciduous and coniferous mature trees and informed sample points within stable patches in other years where the distinction was less clear. The sampling protocol was iteratively refined by site visits. One hundred sample points stratified across all floodplain habitat types were verified during a site visit and any errors found in the field were either corrected for similar points across all years, or led to a refinement of the floodplain habitat type definitions. A frequency of occurrence table for each floodplain habitat type was produced for each IC-reach for each year. Managed lands (i.e., agriculture and logging: cover type 10) and exurban development (cover type 11) were among the cover types classified from the imagery. Their relative percent cover were calculated for each IC-reach across all years and removed from the floodplain habitat matrix. The remaining floodplain habitat matrix contained only floodplain habitat types 1-9 and 12 and was used in the following data analysis as the response variable. Floodplain cover types 10-11 were assessed as potential explanatory variable. All floodplain habitat types and their transitions across years were used in the graphical analysis.

Geomorphic composition – Three floodplain geomorphic positions were mapped for each sample year: the active channel (i.e., channel and parafluvial areas), the active floodplain (i.e., active accretion orthofluvial) and the passive floodplain (i.e., passive accretion orthofluvial). Each geomorphic feature was delineated using heads-up digitizing (i.e., manually drawing polygons around each feature) in ArcGIS with a minimum mapping unit of approximately 25 m^2. The polygons were identified with the assistance of background imagery, visual assistance from oblique views within Google Earth's 3-D models (Google Earth 2013), oblique imagery from aerial reconnaissance, and multiple site visits. Relative ratios of each of the three geomorphic features were calculated for each IC-reach for each year.

Table 3. List of cover types prevalent among the North Fork floodplain (Modified from Hauer et al. 2002).

Habitat ID	Description
1	Mature conifer
2	Mature deciduous
3	Immature deciduous 2-6 m in height
4	Cottonwood, willow, or alder seedlings and early seral stages up to 2 m in height interspersed with open cobble area
5	Filled or partially filled abandoned channel dominated by mix of willows, alder, shrubs, and interspersed herbaceous cover and post-fire herbaceous dominated interspersion of fire scared snags or fire stressed trees
6	Herbaceous vegetation dominated bench, may have interspersions of an occasional shrub. Includes post-fire herbaceous dominated with interspersion of fire scared snags or fire stressed trees.
7	Exposed cobble riverbed
8	Main-channel
9	Off main channel surface water
10	Managed lands including agriculture meadows and plowed fields that are often planted and hayed, fallow fields that are proximal to Cover Type 11and recently logged lands and tree farms
11	Domestic or commercially developed lands including homes, buildings, gravel pits, and transportation corridors
12	Early succession forest: immature woody species predominantly composed of conifer or shrubs 2-6 m in height and < 10 cm dbh. Interspersed with fire scared snag

Discharge of contributing basins – To estimate stream power for each of the 37 IC-reaches, a continuous daily hydrograph was modeled for the cumulative watershed area above each major contributing stream confluence using the HBV-EC model, a variant of the conceptual hydrological model, HBV-96 (Lindström et al. 1997) modified by Environment Canada for application in Northern and Canadian Rocky Mountain systems (Jost et al. 2012, Mahat and Anderson 2013). HBV-EC has been incorporated by Environment Canada into a desktop hydrologic modelling environment known as Green Kenue (Canadian Hydraulics Centre 2010) and was used for initial model setup after which an executable version of HBV-EC was used for iterative parameter optimization in MATLAB version 7.10.0 (MathWorks 2010).

Model setup consisted of defining alpine and sub-alpine climatic zones above and below 1,980 meters of elevation, respectively, each of which is associated with climatic data as well as four land-cover types: open, forest, glacier and water; creating a unique parameter set for each zone (Canadian Hydraulics Centre 2010). Land cover was fixed across the modeling period for climatic zones with open areas, lakes and glaciers classified as such and the remaining area classified as forest. Required climate inputs were mean temperature and evaporation-rate as monthly time steps, and mean temperature, rainfall, and snowfall as daily time steps for the time period being simulated. The monthly evaporation rate was acquired from the closest pan evaporation data (WRCC 2013) and was applied to each zone. All daily meteorological inputs were obtained from DAYMET, a 1-km gridded metrological dataset with daily data from 1980 to 2012 (Thornton et al.

2012). The required daily and monthly data were spatially averaged for each climate zone across each of the 37 cumulative sub-watershed areas.

The modelling strategy was to develop and calibrate a hydrological model with a specific spatial structure and model parameter set for the entire North Fork watershed using daily discharge from USGS gauge station (#12355500). A parameter set for a calibrated HBV-EC model from the nearby Mica Watershed (Jost et al. 2012), located approximately 150 km north of the North Fork Watershed was used for model setup in Green Kenue. Through ten iterations, model parameters were optimized against increasing Nash-Sutcliffe (1970) efficiency objective function thresholds (0.2 to 0.6). Initial iterations consisted on 100K model runs decreasing to 12K as the model became more efficient. Ultimately the MATLAB's Monte-Carlo Analysis Toolbox (Wagener et al. 2001), established the top-ten optimal parameter sets. These were applied to the cumulative watershed above the British Columbia gauge station (#08NP001: Environment Canada, 2013) located on the North Fork at the U.S./Canadian border for validation and final parameter set selection. This final optimal parameter set was then used to model discharge of the 37 cumulative watershed areas upstream from each of the 37 IC-reaches to establish a continuous daily discharge for each reach. Slope was also necessary for the stream power calculation and was obtained from the Green Kenue slope tool which uses elevation at each confluence node and stream length of the reach from the 30 meters digital elevation model.

Stream power represents the potential amount of energy in Watts that a stream can exert on its bed and bank as a product of the density of water (1000 kg/m^3), acceleration due to gravity (9.8 m/s^2), slope of the reach in m/m from the Green Kenue slope tool, and discharge within that reach in m^3/s from the modeled results (Bull 1979). Collectively these data provide daily stream power as a surrogate of erosive power of the large flows for each IC-reach from 1980 to 2012.

3.2 Hypotheses Testing and Data Analysis

I was interested in three questions to be applied to the data: 1) which combination of environmental factors best explain the floodplain habitat patch composition across the IC-reaches at various points in time? 2) If and where did floodplain habitat patch composition changed over the sample period? 3) What are the disturbance and recovery pathways that best describe those changes on the landscape? I hypothesized that the combined interaction of erosive flooding and fire events influence floodplain habitat patch composition and that geomorphic position influenced where each disturbance vector had the greatest effect. I choose path analysis, an extension of multiple regression, to disentangle my hypothesized causal interactions by using quantitative correlational interrelationships. From these hypothesized interactions, I proposed an a priori path model that incorporated these explanatory and response attributes of the system. The path analytic method estimated the magnitude and strength of effects within the hypothesized causal system (Stage et al. 2004).

Mantel tests and path analysis – Mantel tests, path analysis and graphical interpretation were used to examine these hypotheses. The strategy used to refine this analysis was to first find measurements of the explanatory variables that best correlate with patch composition. Of the 6 sample years, only 2005 include both the Canadian and U.S. portions of the watershed and provided a survey of all 37 reaches (2005-37 dataset).

These data were used to explore explanatory variables and select an optimal set for the path analysis model, after which, path models of subsequent datasets were analyzed using these optimal variables.

I used the 2005-37 data to refine the path model by testing and eliminating insignificant or redundant potential explanatory variables. The potential explanatory variables consisted of percent of anthropogenic floodplain habitat types (e.g., agricultural or logging: Type 10, or exurban: Type 11), various measures of stream power (e.g., calculated from mean or upper 75th quartile of modeled discharge mean, log mean or sum of stream power), other measures of discharge (e.g., days above bankfull), various measures of geomorphic composition (e.g., percent composition or ratios of passive floodplain to active floodplain or channel) and other physical measure of each IC-reach (e.g., slope, distance, area, width). I first created a Bray-Curtis similarity matrix of the floodplain habitat patch composition and a suite of Euclidian distance matrices obtained from potential explanatory variables. Mantel tests (Mantel 1967) were then conducted to directly compare the multivariate habitat patch similarity matrix with potential independent explanatory variables (Legendre and Legendre 2012). Finally, through iterative Mantel tests, a subset of environmental parameters was selected that best correlate with the Bray-Curtis floodplain habitat patch similarity matrix across all sampled IC-reaches (Sokal et al. 1995, Strohbach et al. 2009). The iterative Mantel tests allowed us to refine my modeled system by selecting the best measures of the environmental variables, to make me aware of other potential explanatory variables that were significant, and allowed us to remove or be aware of redundant or collinear variables.

Once I acquired a subset of environmental variables, I then conducted partial Mantel tests to inform both the direction and significance of the paths in the model (Legendre and Legendre 2012). Partial Mantel tests estimate the strength of the correlation between two distance matrices after the effect of one or more matrices had been eliminated (Mantel 1967, Smouse et al. 1986). As in the Mantel test, significance was assessed by repeated permutations that provide a reference distribution for the computed statistic (Smouse et al. 1986).

With path model finalized, I used simple Mantel correlation coefficients to calculate path coefficients, coefficient of determination, and remaining error in the path analysis (Sokal et al. 1995, Natel and Neumann 1992, Grace 2006). Path coefficients, analogous to regression weights or partial correlation coefficients, range from 0 to 1 and describe the strength of each pathway. The relative sizes of path coefficients tell us if my hypothesized causal relationship is supported by the data and to provide a means to calculate direct, indirect, and total effects that each explanatory variable has upon the response variable (Castillo-Monroy et al. 2011). Mantel and partial Mantel tests were performed on all distance matrices with the 'vegan' package in R (Oksanen et al. 2013) and all path statistics and direct, indirect, and total effects were calculated using the 'sem' package in R (Fox et al. 2013).

Turnover – Path analysis alone does not provide insight into the locations of floodplain habitat patches or degree of shift of these patches across time. Initially, I examined turnover in the habitat composition by conducting pairwise Mantel tests of the Bray-Curtis similarity habitat matrices across all years to assess degree of dissimilarity (i.e., turnover) of the floodplain habitat patch composition across the sampling period. I then examined the specific shifts in structure by conducting graphical analysis.

Graphical analysis – Aerial imagery was publicly available for sites in the USA portion of the watershed (IC-1 through IC-22) for 6 years across the 1991-2013 study period and only 1 year in the Canadian portion of the watershed (Table 2). To examine all changes at IC-1 through IC-22 across this period, I developed a transition table between one sample year and the following sample year. These inter-annual transition tables were developed into an alluvial diagram to visually examine the type and intensity of disturbance and recovery pathways of habitat shift between the series of sampling events. The alluvial graphical tool is currently in development in R (https://github.com/mbojan/alluvial).

The shifting pathways were also summarized in a final transition table across all 22 years (1991-2013) to examine two elements of change: 1) percent change by geomorphic position and 2) inter-related networks of habitat shift across time. Change associated with disturbance or recovery pathways across all years was graphed by where they occur on floodplain geomorphic positions. A network graph was created containing dense sub-graphs, or 'communities' of shift, determined by using a random walk-trap approach. Random walks define sub-graphs where most changes related to one another are occurring based on the idea that random transitions through a network graph tend to get "trapped" into densely connected parts and that these connected parts correspond to 'communities', or in this case densely related habitat shifts within the floodplain (Pons and Latapy 2005).

Several community algorithms were examined and the random walk provided the highest modularity index score for my network. Modularity is a measure of community structure within networks that compares the number of connections within the community groups with expected numbers from a random distribution (Lancichinetti et al. 2008), providing a score that ranges from -0.5 (no community) to 1.0 (isolated community). The community graph was created in the 'iGraph' package in R (Csardi and Nepusz 2006), and contain vertices (habitat types) and edges (transitions) that connect a habitat type that had shifted to another type across that time period. If a large number of sample points classified as cover type *x* had shifted to type *y*, then the edges were weighted to reflect that intensity.

4 Results

Mantel Tests and Path Analysis – Three parameters provided the highest correlations to floodplain habitat patch composition: percent of the IC-reach that was burned (Fire: r_M = 0.46, p = 0.001); stream power calculated from the upper 75th quartile of modeled discharge (calculated from the beginning of the modeling period (1/1980) to June of the sample year[3] (StrPow: r_M = 0.22, p = 0.002); and the ratio of passive floodplain benches to the remaining valley bottom (GeoPos: r_M = 0.13, p = 0.039) (Table 4). Several other attributes had high Mantel correlations but were either collinear with the above variables (e.g., slope had an r_M = 0.13 and a p = 0.046 but is included in the stream power

[3] Except for 2013, climatic data was only available until the end 2012 therefore the modeled discharge ended in 2012. For sample year 2013, stream power was calculated for the 75th quartile of modeled discharge from 1/1980 to 12/2012

calculations) or redundant (e.g., stream power calculated from mean modeled discharge had an r_M = 0.21 and a p = 0.002) and were eliminated from further analysis. Other insignificant correlations were eliminated (e.g., spatial distance: r_M = 0.05, p = 0.191).

Partial Mantel (r_{pM}) tests between stream power or fire extent against floodplain habitat patch composition after removing the effects of other explanatory variables was strong for both comparisons. However, the relationship between geomorphic position and habitat structure is weak when the effects of stream power and fire are removed (lower portion of Table 4).

Table 4. Simple (above break) and partial (below break) Mantel test Pearson correlation results for three spatial factors against habitat cover for the 2005-37 IC-reaches. Correlations were tested between stream power (StrPow), fire (Fire), geomorphic position (GeoPos) and habitat composition (Hab).

	StrPow:Hab	Fire:Hab	GeoPos:Hab	StrPow:Fire	GeoPos:StrPow	GeoPos:Fire
	r_M= 0.22	r_M = 0.46	r_M = 0.13	r_M= 0.27	r_M= 0.06	r_M= -0.01
	p = 0.002	p = 0.001	p = 0.039	p = 0.001	p = 0.117	p = 0.486
StrPow (R)	-	r_{pM} = 0.40*	r_{pM} = 0.11	-	-	r_{pM} = -0.02
Fire (R)	r_{pM} = 0.15*	-	r_{pM} = 0.14	-	r_{pM} = 0.07	-
GeoPos (R)	r_{pM} = 0.24*	r_{pM} = 0.45*	-	r_{pM} = 0.27*	-	-

All partial Mantel tests that remain significant at the Bonferroni corrected level (0.05/3 = 0.0167) for an overall significance of p = 0.05 (Miller 1966), after removing a particular (R) spatial environmental factor effect, are marked by an asterisk in the lower section.

The final path model (Figure 3) is schematically represented in a path diagram, where the arrows depict the relationships of stream power, geomorphic position, and fire extent, all of which interact either directly or indirectly to effect floodplain habitat patch composition. The solid lines indicate significant relationships between elements, dashed lines recognize insignificant relationships, and the curved line signifies recognized collinear relationships. There is a direct relationship between pairing of all variables in my final path model, therefore it was considered saturated and precludes an overall test-of-fit (Castillo-Monroy et al. 2011).

After the path model was finalized, I used simple Mantel coefficients for all datasets to calculate the model's path coefficients, total error and coefficient of determination for each year (Sokal et al. 1995). The path model explains 24% of the variance in floodplain habitat patch composition for the 2005 whole river dataset (2005-37) and explained between 13% and 26% of the variance in floodplain habitat patch composition across all years (Table 5). Fire (path coefficient p_{43}) had the greatest effect on the floodplain habitat patch composition across all years (r ranging from 0.32 to 0.45).

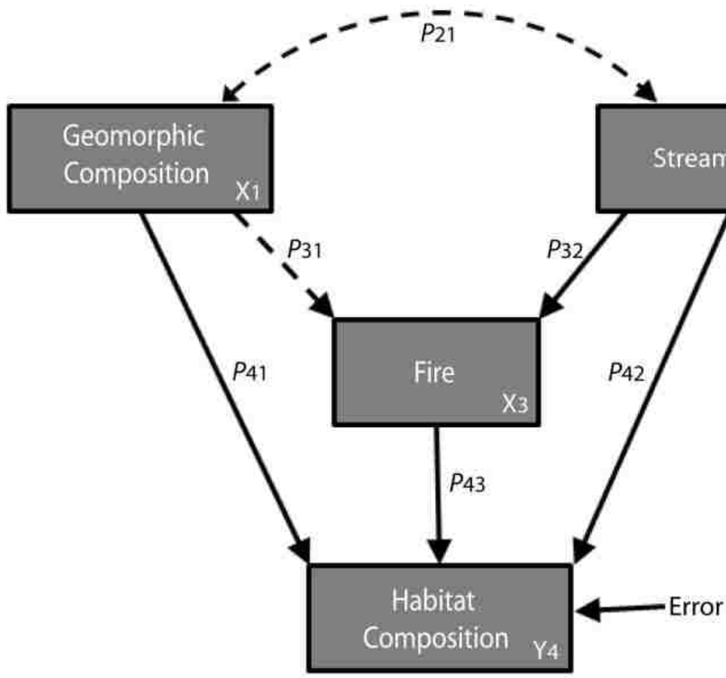

Figure 3. Final path analysis depicting the relationships between the explanatory variables X1-X3 (i.e., geomorphic composition, stream power and fire) and the response variable Y4 (floodplain habitat patch composition). Arrows indicate the direction of influence, with the associated path identification number. Double sided arched line indicates recognized collinearity, and solid lines indicate statistically significant relationships.

Table 5. Path coefficients, total error, and coefficient of determination for all sample years. Path coefficient labels (e.g., p21, p31, etc.) relate to the path labels in Figure 3.

Year	p_{21}	p_{31}	p_{32}	p_{41}	p_{42}	p_{43}	Error	R^2
2005.37	0.06	-0.02	0.27	0.13	0.09	0.43	0.76	0.24
1991	-0.05	0.15	-0.18	0.17	0.17	0.42	0.78	0.22
2003	-0.06	-0.06	-0.01	0.14	0.11	0.32	0.87	0.13
2005	-0.06	-0.06	-0.01	0.09	0.02	0.39	0.84	0.16
2009	-0.06	-0.06	-0.01	0.11	0.25	0.43	0.75	0.25
2011	-0.07	-0.06	-0.01	0.13	0.25	0.38	0.79	0.21
2013	-0.07	-0.06	-0.01	0.11	0.24	0.45	0.74	0.26

From the above path coefficients, the direct, indirect and total effects of an explanatory variable on the response variable can be determined. Fire also had the greatest total effect on the floodplain habitat patch composition across all years (Table 6). The total effects of geomorphic composition and stream power on floodplain habitat patch composition varied widely across the sample years (ranging from 0.07 to 0.23 and 0.01 to 0.25 respectively).

Table 6. Direct, indirect and total effects of explanatory factors on the floodplain habitat patch composition.

Environmental Variable	Effect	2005-37	1991	2003	2005	2009	2011	2013
Geomorphic Composition	Indirect	-0.01	0.06	-0.02	-0.02	-0.02	-0.03	-0.03
	Direct	0.13	0.16	0.14	0.09	0.10	0.13	0.11
	Total	0.12	0.23	0.12	0.07	0.08	0.10	0.08
Stream Power	Indirect	0.12	-0.08	<-0.01	-0.01	-0.01	-0.01	<-0.01
	Direct	0.09	0.17	0.11	0.02	0.25	0.25	0.24
	Total	0.21	0.09	0.10	0.01	0.25	0.24	0.24
Fire	Indirect	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	Direct	0.43	0.42	0.32	0.39	0.43	0.38	0.45
	Total	0.43	0.42	0.32	0.39	0.43	0.38	0.45

Turnover – Although the path model alludes to either dynamics in the explanatory variables or composition of the response variable, it does not directly address those changes. Simple Mantel tests compared the Bray-Curtis similarity of the floodplain habitat patch composition across all year in pairwise fashion. Sites further away in years they are more dissimilar (R^2 = 0.93, p < 0.001) indicating turnover in the patch composition (Figure 4).

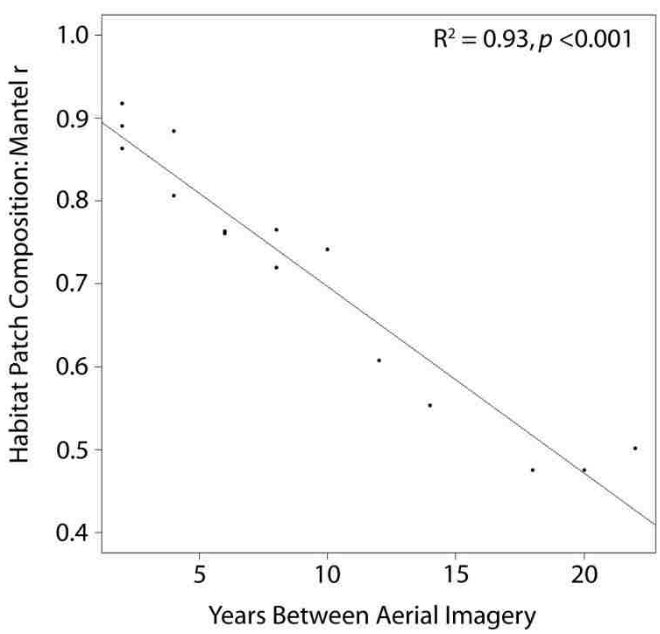

Figure 4. Relationship between pair-wise Mantel r of Bray-Curtis similarity matrices of habitat patch composition across all years of aerial imagery.

Type and Location of Transitions – An alluvial graph was developed through the observations in IC-1 through IC-22 reaches of the 2,200 points that changed in habitat composition and 11,000 points that remained static across the 22 years (n = 13,200) (Figure 5). From the results of the path analysis and field observations, many cover class transitions could be assigned to disturbance or recovery vectors. For instance, between 1991 and 2003, many sample points that were classified as mature conifer in 1991 were clearly burned and those points were converted to herbaceous fields with snags and fallen timber in 2003. In the alluvial graph, these transitions occurred between habitat type 1 and habitat type 6 (see Table 3), and were attributed to fire driven disturbance (orange in Figure 5). Following the same approach, the remaining disturbance and recovery pathways, flooding, anthropogenic and succession, were assigned.

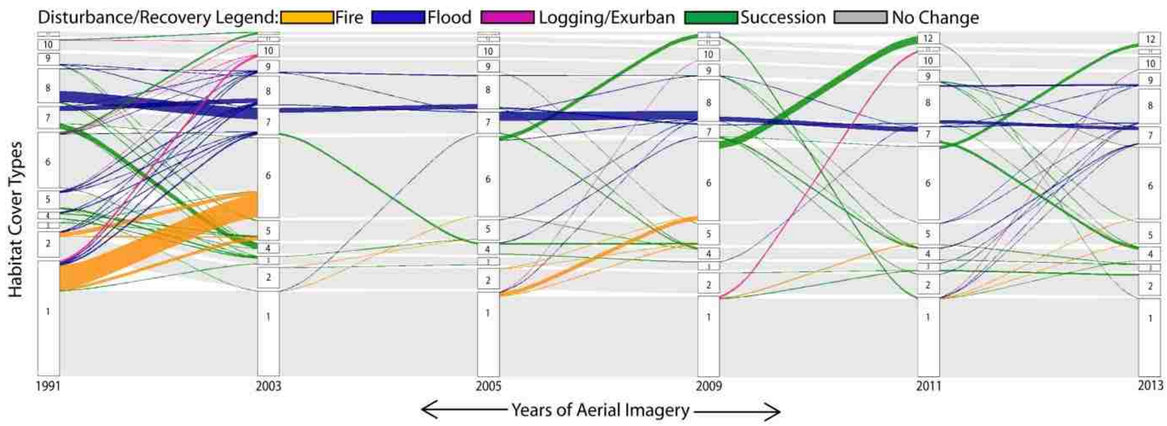

Figure 5. Alluvial graph of habitat patch composition transitions of cover types 1-12 (see Table 3) between each year of available aerial imagery, with changes due to fire (orange), flooding (blue), anthropogenic influence (pink), succession (green), and no change (gray).

To graphically examine where the drivers of turnover are occurring on the riverscape, I organized matrices of habitat cover by geomorphic position across all years and assigned these transitions based on the results of the path analysis and the alluvial graph (Figure 6). Transitions associated with fire appear to have equal occurrence on the passive and active floodplain, while hydrology shows a strong occurrence within the active channel with lesser occurrence latterly across the floodplain. Logging and exurban development appear to occur predominantly in the passive floodplain. Recovery occurs mostly within the active channel where most of the hydrologic disturbance is occurring then in the passive floodplain where the fire and logging are occurring.

Shifting Habitat Mosaic – Transitions across all 22 years were combined into a single summary table (Table 7) and when applied graphically each cover type is considered a 'node' and the transitions between types are considered 'edges' (Figure 7). Arrows signify the direction of the edge and the numbers of transitions inform the weight of the edge. Collectively, these cover types and their interactive transitions are displayed as a network. Within this network are dense sub-graphs that represent closely related transitions or disturbance/recovery communities. Using a random walk algorithm (walktrap within iGraph in R: Pons and Latapy 2005), optimal disturbance/recovery communities were established (Modularity = 0.419) (Figure 7). The factors found to be significant in the path analysis (i.e., hydrologic and fire disturbance), as well as the non-significant, but important drivers implied in the alluvial graph (i.e., anthropogenic disturbance), informed the assignment of the drivers to shifting mosaics and the cover types they influence as well as the influence of human actions on cover type transitions.

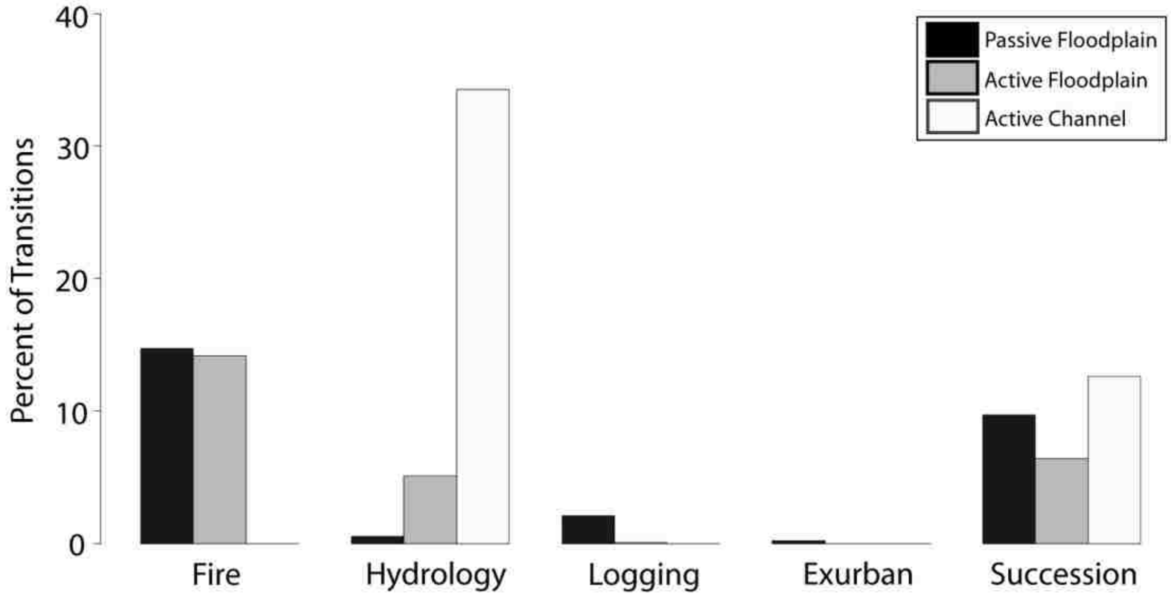

Figure 6. The percent of habitat patch transitions across the study period by the dominant disturbance/recovery vectors that are occurring each floodplain geomorphic feature; active channel (white) active floodplain (gray) and passive floodplain (black)

5 Discussion

I expand the floodplain SHM Concept by including fire as a significant driver on floodplain habitat patch composition in Northern and Canadian Rocky Mountain rivers (Table 6). Because floodplains are transitional zones between the aquatic environments of the river and terrestrial environments of the uplands, they are subject to a suite of aquatic and terrestrial disturbance regimes and it is the combination and interaction of these disturbances that determine the composition and dynamics of floodplain habitats (Figures 5 and 7). The SHM Concept recognizes the dynamics of floodplain habitat (Arscott et al. 2002, Stanford et al. 2005); however, because river hydrodynamics is an important driver of the floodplain geomorphic composition, the floodplain SHM literature has focused on this primary disturbance factor while ignoring other important floodplain disturbance drivers such as fire. This may be a result of the limited recent fire occurrence on research floodplains used to develop the floodplain SHM Concept (e.g., Nyack Floodplain, MT-USA; and Tagliamento River, Italy).

Our path models indicated a strong direct effect of fire on floodplain habitat patch composition (0.43) and that stream power and geomorphic position have a strong to moderate direct effect on floodplain habitat patch mosaic (0.21, and 0.13 respectively; 2005-37 dataset) (Table 6). One cannot infer causality from path analysis (Everitt and Dunn 2001) only the magnitude of the relationship between variables (Stage et al. 2004) with limitations. For instance the model shows strong effect of fire across all years, yet it does not explain the variation in the effects of explanatory variables between years, nor across scale (e.g., 2005-37 whole river dataset versus the 2005-22 dataset that includes site below the Canadian border, Table 6). Although each year's models explained a modest portion of the variance in the floodplain habitat patch composition, unexplained variance of 0.74-0.87 is present (Table 4). Therefore it is likely that 1) other factors not included in my model may have a significant effect on patch composition that was not captured at the scale of this multi-reach assessment, and/or 2) uncertainty stemming from the date set (e.g., gridded meteorological data, modeled discharge, slope from the DEM, inadequate delineation of ground features, or misidentification of cover types) may influence model results. Despite these limitations, the path analysis allows us to compare the magnitude of the relationship between variables, which is one element to support the plausibility of my a priori causal hypotheses. The graphical analysis provides the second element of supports for the plausibility of my a priori causal hypotheses as well as insights to the dynamics pathways of disturbance and recovery vectors in contemporary time.

Table 7. Floodplain habitat patch transitions across the 22 year study period.

	Cover Type	Transition To: 1	2	3	4	5	6	7	8	9	10	11	12
Transition From:	1	2879	-	2	1	29	199	11	10	4	18	-	3
	2	-	718	2	-	6	26	3	2	3	-	-	-
	3	-	8	251	-	3	1	1	1	-	-	-	-
	4	-	-	7	322	8	-	12	8	3	-	-	-
	5	1	-	9	1	684	-	2	8	4	1	-	-
	6	-	-	2	1	-	2504	1	2	1	1	1	106
	7	-	-	2	74	-	2	507	124	7	-	-	-
	8	-	-	-	8	3	-	136	1092	12	-	-	-
	9	-	-	-	4	4	1	8	8	410	-	-	-
	10	-	-	-	-	-	-	-	-	-	425	1	4
	11	-	-	-	-	-	-	-	-	-	-	148	-
	12	12	-	-	-	-	-	-	-	-	-	-	138

The SHM concept recognizes that the relative abundance of floodplain habitat patches remains relatively stable across ecological time scales (Arscott et al. 2002, Ward et al. 2002, Stanford et al. 2005, Latterell et al. 2006, Whited et al. 2007). Ecological time is defined as the period necessary to establish the range-of-variation making up the floodplain habitat mosaic dynamics (Slobodkin 1961, Landres et al. 1999, Poff et al. 1997, White et al. 1999, Arscott et al. 2002). Over ecological time, high energy flood and flow pulses drive the physical processes such as sediment deposition, cut-and-fill alluviation and channel avulsion (Ward et al. 2002, Stanford et al. 2005, Hauer and Lorang 2004, Slingerland and Smith 2004) which in turn form a patchwork of geomorphic surfaces of different physical structure and age, with associated vegetation communities and varied successional states (Hauer et al. 2003, Stanford et al. 2005). The geomorphic makeup, vegetation composition, and flooding inundation and recession pathways also create shifting gradients of microclimates, and soil and fuel moisture content that determine fire susceptibility and intensity (Dwire and Kauffman 2003, Pettit and Naiman 2007, Poff et al. 2011). Therefore, both flood and fire continuously interact at an ecologic time scale to affect floodplain habitat patch dynamics in the Rocky Mountains (Dwire and Kauffman 2003, Pettit and Naiman 2007, Poff et al. 2011).

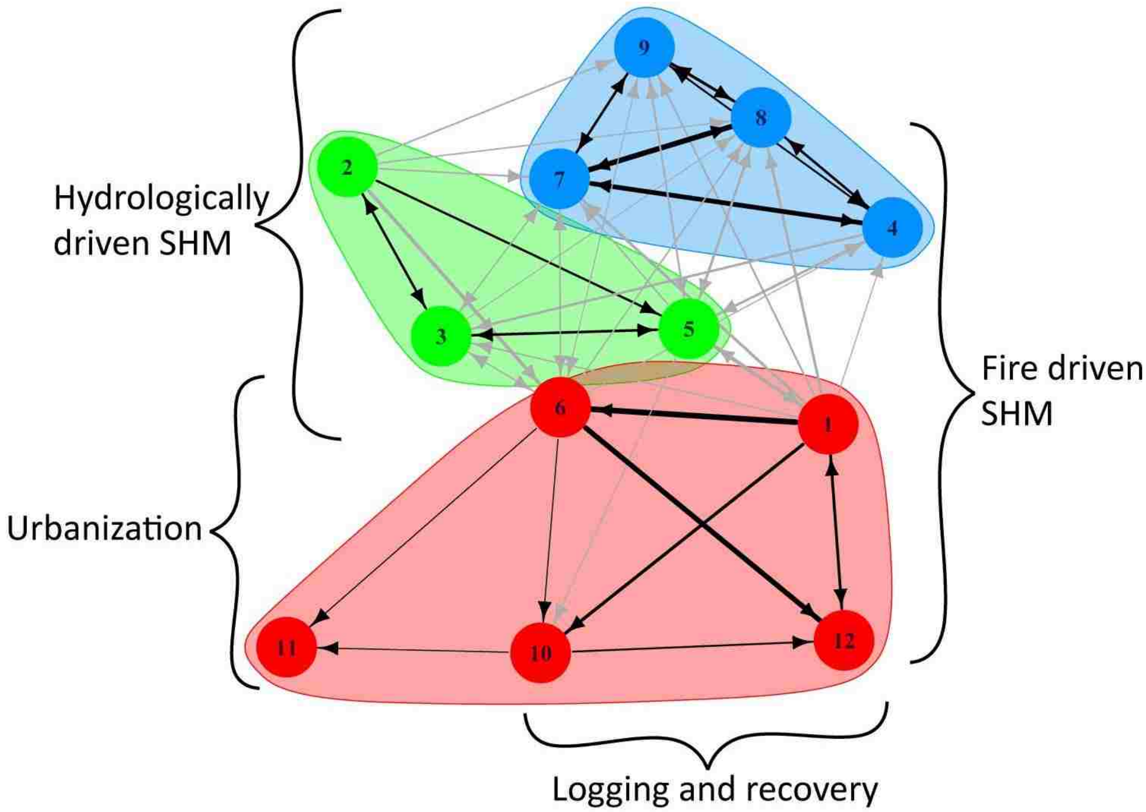

Figure 7. Network graph of habitat patch transitions of cover types 1-12 (see Table 3) across the study period. Dense sub-graphs of optimal disturbance/recovery communities, demarked by colored clouds, are derived from a random walk algorithm denote (Modularity 0.419). Arrows show the direction of the transitions between cover types with heaver lines indicating larger number of transitions, black lines indicting transitions captured within the random walk algorithm and red lines are transitions among dense sub-graphs. Transitions within the blue cloud occur predominantly below bankfull, the green cloud occurs predominantly in the active floodplain, and the red cloud occurs predominantly in the passive floodplain.

Currently, there are limited data to truly capture the spatial and temporal components necessary to establish a range-of-variation that defines a steady-state at an ecological scale (Turner et al. 1993) for entire river systems. But, over the last several decades, increasing access to remote sensing products allow for assessment of disturbance and recovery dynamics at a contemporary time scale. Although, only a portion of the total floodplain SHM dynamics can be examined at this time scale, it reveals the individual mechanisms at large spatial scales that drive aspects of the shifting mosaics, albeit with some limitations. For instance, I proposed that during contemporary disturbance events, fire and flooding would have inverse effects on the landscape given the transitional aspects of the biotic, abiotic, and microclimate elements across the floodplain (Figure 1). Although a transitional influence of flooding may be observed (Figure 6), it is less clear that the effects of fire appear to be equally present on both the passive and active floodplain surfaces. A gradient of fire intensity from the xeric conifer communities on the passive floodplain to the mesic/aquic communities in the active floodplain or channel assumes the presence of a gradient of moisture or microclimatic conditions that would enhance or deter fire. However, during dry periods these moisture and microclimatic gradients no longer exist and the dry abundant fuel load in the flooded areas increases potential floodplain fire intensity (Pettit and Naiman 2007). Either such conditions were present during the 1998, 2001, and 2003 fires to create fires that burned across the floodplain geomorphic features and/or there was a lack of fine scale resolution in my mapping of the passive and active floodplain resulting from interpretations through close canopies, intra-year image rectification or other issues common to photogrammetry.

Despite the various limitations, this study shows that increasing density of high resolution imagery allows us to monitor and study changes in floodplain composition and associated flooding, fire and anthropogenic stresses at a contemporary, whole river scale. For instance, the alluvial graph (Figure 5) shows that between 1991 and 2003, several points classified as mature conifers and mature cottonwood were burned by the 1998, 2001 or 2003 fires (Table 1). Over the subsequent years, several other classification points show that other trees in the burned areas die, likely from fire stress, and sites transitioned from forested to herbaceous cover types. Similarly, between 1991 and 2003, any cover originating from or transitioning to aquatic cover types comprising of river (type 8), cobble (type 7), and side-channels (type 9) were attributed to hydrologic disturbance. Many smaller events where vegetated surfaces (types 1, 3, 4, 5, or 6) transition to aquatic cover types were likely a result of alluviation while conversion between aquatic types were assumed to be the result of flow pulses and occur below bankfull in the active channel. Anthropogenic change was not considered to be significant in the path analysis likely due to the limited human footprint in the North Fork floodplain. However some transitions were observed where mature conifer was logged (type 10) or converted to homes (type 11).

Floodplains are among the most threated ecosystems worldwide (Tockner and Stanford 2002). Changing disturbance dynamics associated with climate change will affect floodplain patch composition dynamics and can be counted among the many threats to floodplains, including grazing, dams, invasive species, flood control, irrigation appropriations, and urbanization (Poff et al. 2011). Examining a 60-year time series of aerial imagery, Whited et al. (2007) showed relationship between changes in floodplain habitats of the Nyack Floodplain (MT, USA) and annual flood magnitude associated with the cooling and warming phases of the Pacific Decadal Oscillation (PDO). They found that increased flooding associated with the cooling phase of the PDO led to extensive restructuring of the floodplain, while the limited flooding associated with the PDO warming phase led to decreased flooding and a floodplain associated with later successional vegetation stages. Other authors have found that many of Rocky Mountain rivers have historic drying trends over the last century (Rood et al. 2005) after taking the variability associated with the PDO and other oceanic oscillations into account (St Jacques et al. 2010a, 2013). Lower total discharges are further degraded by anthropogenic withdrawals (St Jacques et al. 2010b). Many of these rivers are predicted to have further decrease in flow over the next century (St Jacques et al. 2013) with lower snow-melt driven peak floods occurring earlier in the spring as well as lower summer base flows (Rood et al. 2005, 2007), which are further compounded by appropriations and extraction on water quantity. Collectively decreased flood peaks, earlier runoff, and lower summer flows will likely lead to overall maturation of floodplain forests with restricted cottonwood recruitment and stressed adult cottonwood (Rood et al. 2008) that will likely create conditions sufficiently xeric to result in increased fire prone floodplain communities. At the same time climate change will also affect fire intensity and frequency (Dale et al. 2001, McKenzie et al. 2004) as the number of days of high fire danger are projected to increase in the future from increased drying in the Western U.S. (Brown et al. 2004). Ultimately fire will likely be a dominant factor in the shifting mosaic of floodplain habitat patch composition in drier periods and flooding in the wetter periods.

The value of contemporary assessments provided by increased remote sensing data density over the last 40 years allows us to monitor changes in floodplain ecosystem integrity and its associated functions and related ecological services. Under the climate driven changes described above, it is likely that the disturbance/recovery paths (Figure 5) and community interaction (Figure 7) will reflect such change. However, flow and flood pulses will interact with the same patch classes albeit to a lesser extent. Flow pulses below bankfull will continue interact with cobble (type 7), earlier successional seedlings and samplings (type 4), redistribute the locations of the river and backwaters (types 8 and 9), and that flood pulses continue to effect patches within active floodplains (types 2, 3, and 5) and passive floodplains in alluviation events (types 1 and 6). Equally so, increased fire driven disturbance/recovery vectors would increase the interactions the same patch classes with increased fire occurrence in mature conifer and cottonwoods (types 1 and 2) creating more herbaceous fields (types 5 and 6) or immature woody species (types 3 and 12).

Although the network graph provides a visualization of the transition communities and the important drivers of the SHM, it would not likely change in configuration as the system changes in future climatic scenarios. The network graph would change if the transitions between patches in a river were drastically altered by episodic, large scale events such as massive flooding, fire/flood interactions such as excessive sediment transport from contributing basins, or if the transitions were cut off by flood control structures, fire repression, increased agriculture or urbanization or invasion of exotic species. Networks developed before and after such events could be used to detect change in floodplain dynamics. Future work is required to develop this interpretative power of the alluvial and network graphs into a quantitative assessment metric.

6 Conclusion

Floodplains are transition zones between aquatic and terrestrial systems that are exposed to both hydrologic and terrestrial disturbances that collectively influence floodplain habitat patch composition and dynamics. These disturbances continuously interact over ecological time scales to create a shifting mosaic of floodplain habitat patches occupied by associated endemic species adapted to the composition and dynamics of these habitats. I found through path analysis that the magnitude of fire's direct effect on floodplain habitat patch composition was greater than that of flooding or geomorphic position. These results were supported by graphical analyses that indicate fire and its legacy drives a large portion of floodplain disturbance and recovery dynamics. Therefore, floodplain SHM Concept should include not only flooding but also fire as a significant driver of floodplain habitat patch composition in Rocky Mountain river floodplains. These disturbance events occur at different frequencies and locations on the floodplain. Availability of remote sensing data and its ancillary products provide a venue to assess whole-river dynamics and increase my understanding of natural processes of large river floodplain transition zones. These also allow us to assess changes in the ecosystem integrity as well as the associated alterations of ecosystem functions and services that these threatened systems provide.

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CHAPTER V: CONCLUSIONS

1 Overview

Ecological monitoring and assessment tools are constantly being refined or altered to meet specific regulatory, management, or community needs. As these tools change, they have the potential of getting mired in political or professional conflicts as definitions become confused, or applications exceed intended use (e.g. Stein et al. 2009, Kleindl et al. 2010). Many of these conflicts arise because ecological assessments rely on indicators that situated in the "*fuzzy area between science and policy and between the production and the use of scientific knowledge*" (Turnhout et al. 2007). The elements of an ideal indicator have been refined over the last several decades and generally should have the following properties (modified from UNESCO 2006 and Rees et al. 2008):

1. Informative - convey information that is responsive and meaningful to decision-making;
2. Responsive - linked to a conceptual stressor–response framework;
3. Sensitive - capable of measuring change or its absence with confidence (robust to influences of confounding environmental factors);
4. Anticipatory - early warning of potential problems;
5. Readily Measureable - cost-effective assessment metrics;
6. Interpretable - easy to understand and communicate to a as wide a range of stakeholders; and
7. Grounded on scientific theory: Indicators should be based on well-accepted scientific theory, rather than on inadequately defined or poorly validated theoretical links

As with most assessment approaches, the above indicator properties are well-suited for assessment of ecosystems that are exposed to clear anthropogenic disturbance gradients. However, as I stated in the introduction of this dissertation, future development of assessment models intended to measure the effects of global climate change on watershed or regional ecological integrity will require indicators that address perturbations which extend beyond direct anthropogenic land use alterations. As these indicators are developed, a central question must be addressed: to what degree is ecosystem condition affected by anthropogenic and climate-driven disturbances or simply natural variability, individually and in combination? This central question would require multiple careers to address and is beyond the scope of a single dissertation, however it guided the dissertation's three research questions. 1) How does one assess ecological condition in the absence of an anthropogenic disturbance gradient? 2) What is the effect of uncertainty endemic to remote sensing data on large scale assessments? 3) How can one distinguish natural dynamics from a system altered from climate-driven disturbances?

In the following chapter, I use the above indicator properties to provide a brief critical examination of the above case studies to see where these efforts satisfy these criteria, where and why they may fall short of these properties and whether these properties apply to tools intended to measure the effects of global change on remote landscapes. Additionally, I will address what are the logical next steps for each of these efforts to facilitate their integration into management settings or to continue in the advancement of the necessary scientific foundations. Finally, I will provide a few concluding statements on the three research questions.

2 A Multi-Metric Watershed Condition Model for Glacier National Park

The Glacier National Park condition assessment case study presented in Chapter 2, was a digested version of a larger effort provided to the park in Kleindl et al. (*In Press*). The original document was developed as part of a multi-year program from the Water Resources Division of the National Park Service (NPS) to fund natural resource condition assessments (NRCA) for 270 park units with significant natural resources. These NRCA are intended to synthesize existing research, and inventory and monitoring data into a knowledge base for use in park resource planning, decision making, monitoring prioritizations, accountability reporting, and partnership and education efforts. Therefore a goal of this document was to provide a tool that would blend smoothly with the park's existing management frameworks that center on the NPS policy "to understand natural processes and human-induced effects; mitigate potential and realized effects; monitor ongoing and future trends; protect existing natural organisms, species populations, communities, systems, and processes; and interpret these organisms, systems, and processes to the park visitor" (Layman 1999).

The final product used available data supplied by the park does provide a tool that meets the park's management goals and satisfies many of the criteria articulated by UNESCO (2006) and Rees et al. (2008). The metrics and indices convey information that is meaningful to decision-makers, provide early warning of potential problems, are easy to measure, the final products and maps are easy to interpret, and the approach is grounded in scientific theory. In most applications, ecological assessment tools measure response along a stress gradient and are sensitive to changes in stress at a site without being influenced by confounding environmental factors. Glacier National Park has a very limited stress gradient from anthropogenic disturbance and the metrics within my assessment tool that measured that disturbance meet all of the UNESCO (2006) and Rees et al. (2008) indicator criteria. It is clear that the park is at risk of losing resources from global change vectors like climate change. However, given the limitations of the available data, a stress gradient that can be attributed to climate change within the park is either very limited or essentially non-existent.

To overcome the lack of a disturbance gradient measureable with the available data, I replace it with a diversity gradient under the assumption that more diverse watersheds are more resilient to global disturbance. By doing so, I combine contemporary impacts with potential future risks to specific ecological elements that are of interest to the park. As a result, I began the project by developing a contemporary assessment of ecological condition, as NPS wished, but ended with a modified risk assessment of current and potential future change.

In an ideal model development process, the tool should be (UNESCO 2006):

1. Relevant to management objectives,
2. Clearly linked to the outcome being monitored,
3. Developed with all those involved in management, and
4. Part of the management process and not an end in themselves.

To develop a model that meets the end-users needs, these steps should be observed. This requires frequent meetings and updates with the end-user throughout the model building process. Without that close communication, the tool may not satisfy the end-user's needs. For instance, NPS provided funding for several pilot programs develop a methodology to conduct NRCAs that could be applied to all 270 parks. Although my final product provided Glacier National Park with a utilitarian model that meets its management needs, it did not meet NPS needs for a model framework that could be applied to several parks with a wide range different land uses. It would be very difficult to develop such model without a close relationship with NPS' NRCA team from the inception and throughout of the project. I would also argue that a dissertation driven effort may not be the best place to develop such a model due to the vagaries of graduate school that create delays and creative diversions all within a pavilion of limited funding.

Throughout the development of the assessment tool, it became clear that biological and structural assessment, risk assessment, and landscape assessment methods are coalescing at these watershed scales to address the common ecological problem of global change. However it is also clear that these approaches are entering the same arena with their own unique dogma. Lackey (1997) recognized that confusion and divisiveness occur as multiple assessment approaches are applied to similar ecological problems but conflate common ecological terms with their own unique definitions. Cormier and Suter (2008) recognized that there are no existing frameworks that include all types of environmental assessment approaches and provide a conceptual approach were these frameworks could coexist. As a next step to the problems found within this dissertation, I will use one of the models developed for the GNP assessment approach to test the Cormier and Suter (2008) conceptual framework with the intention of clarify the common language and advance the approaches to assessing impacts of global change on remote areas.

Finally, how does one assess ecological condition in the absence of an anthropogenic disturbance gradient? I would argue now that the question may not have been stated correctly. Ecological condition of areas with no direct human impact should be considered "excellent" by the generally accepted definition of the term 'condition', when examined statically as it was in the GNP assessment. In the dissertation's first case study, the term "conditions" may be too limited. Ultimately the final product is a predominantly a measure of risk to that condition. However it implies that across time unimpacted system will lose some elements that are important to ecological integrity, function, or condition when impacted by global stressors. Assessment approaches address change across time may capture that loss. This approach was the focus of the third case study of the dissertation.

3 Effect of thematic map misclassification on landscape multi-metric assessment

Typical of most assessment models, the first case study did not account for error endemic in the model input data, primarily because these data were either not available or simply reported and not included in the assessment model. As the first case study was underway, the second research question was developed. What is the effect of uncertainty endemic to remote sensing data on large scale assessments? This question was beyond the scope of the GNP case study and was developed as a stand-alone effort. I felt that any effort to incorporate these known uncertainties into an assessment tool used by planners, policy makers, lawyers, and scientists would have to be as straightforward as possible to stay in-line with the intent of the indicator criteria presented by UNESCO (2006) and Rees et al. (2008). The methodology proposed in this case study provides metrics and indices that convey information that is arguably more meaningful to decision-makers than naive scores, the simulated approaches are still responsive to stressor gradients, the final products and maps are relatively easy to interpret, and the approach is grounded in scientific theory. However, the distributions of simulated assessment scores are less sensitive to subtle changes in land use composition than the naive scores or conversely, the naive scores may be artificially sensitive to such changes. Above all, the error simulation method chosen for the second case study does not meet the 'readily measurable' criteria of indicators despite attempts to choose most straightforward of approaches available in the literature.

The U.S. Environmental Protection Agency (USEPA) has recognized that there are many approaches to monitoring and assessment with varying degrees of effort and scale of applications. They proposed a three-tiered approach to monitoring and assessment of aquatic resources. Their Level 1 assessments consist of habitat inventories and landscape-scale assessment, while their Level 2 consists of rapid at-site assessment, and Level 3 consists of intensive assessment (Kentula 2007). It is implied by USEPA that Level 1 assessments rely entirely on GIS data to provide a coarse gauge of ecosystem condition within a watershed (USEPA 2015). Level 1 assessments could be developed by specialized teams that are expected to have the technical skills necessary to provide an estimation of uncertainty such as I presented in the second case study, however these tools could also be developed by teams that do not have those specialized skills. The approach presented in the second case study requires an understanding of Monti Carlo techniques to generate multiple realizations of raster maps and could be beyond the ken of most practitioners.

Additionally, it was the intent of the second case study to provide an approach that could implemented without collecting of additional detailed data. However, without collecting those additional data, it is not possible to account for the local spatial structure of the error. Given the presumed limitations of the practitioners and the limitations of modeling error without measuring the spatial structure, a logical next step in my effort would be to provide a broad analysis error response to multiple landscape metrics (e.g., McGarigal et al. 2002) as a service to future users. Such an effort would have an assessment domain comprised of multiple sites across a complete disturbance gradient from remote forested areas to urban areas using use a thematic map with hyper-local error assessment in the form of a confusion matrix. This hyper-local error assessment would also have the ability to provide the structure of the error within that assessment domain. Then apply an assessment approach similar to the second case study that includes naive assessment, a confusion frequency simulation approach, and a more detailed approach such as sequential indicator simulation (SIS) that incorporate *a posteriori* probabilities from hard data collected at-site to produce indicator covariances or variograms (Kyriakidis and Dungan 2001, de Bruin et al. 2004, Boucher and Kyriakidis 2006) to compare the range of responses based on level of effort. The family of error models that include SIS and indicator cokriging may be required for some applications but may exceed the degree of detail required by a USEPA Level 1 assessment and should be discussed by the model and resource decision team.

Another comment suggested that my assessment of spatial fragmentation is very different than how the NLCD product is created. NLCD has a minimum mapping unit within the classification algorithm so that lone pixels are essentially regrouped to help reduce misclassification and to make the map appear smooth and acceptable to users less familiar with remote sensing. My work was not intended as a criticism of the NLCD efforts, rather it is intended to address future use of those products. All maps are wrong and cartographer cannot foresee the usage of their products after they are created but they do attempt to provide all as much information as possible to enlighten the map user of the extent of that error. CFS approach I applied created another artificial representation of reality but one that is based on the probabilities provided by NLCD.

Finally, what is the effect of uncertainty endemic to remote sensing data on large scale assessments? Our results elucidate the potential bias between the more common naive approach to ecological assessment and an approach that includes error. I show an increase in overall map accuracy as the 16 land-cover categories in the original NLCD thematic map was aggregated into the 5 land-use groups for the perturbation map and the 2 land-cover groups for our fragmentation map. The resulting assessment metrics within our multi-metric index respond in different ways to map error depending on the land-cover pattern of each assessment site. When combined into an index, it appears that naive scores slightly over-estimate ecological quality within sites comprised of contagious unmanaged lands associated with higher quality floodplains, and potentially under-estimate the quality in more disturbed sites comprised of heterogeneous land uses. Naive approaches are easier to implement but at a minimum recognizing that using such an approach is biased may help with the end-user's state-of-confidence. Baseline response to error should be conducted on new landscape scale indicators and to facilitate this process, perhaps, remote sensing products should include information of spatial structure of their error.

4 Fire and Flood Expanding the Shifting Habitat Mosaic Concept

The first dissertation case study proposed an approach to address the assessment of ecosystems with limited human disturbance. During that process it became clear that existing ecological indicators designed to assess the effects of human impacts in ecosystem condition are not as affective when modified to address watershed-scale perturbations from subtle global scale climate or pollution impacts. To measure the subtle effects global change on otherwise unimpacted ecosystems, a static contemporary assessment may be insufficient. Chapter IV was as a step toward developing metrics that can measure ecosystem dynamics and distinguish perturbations to those dynamics. This chapter provided a hypothesis driven study that accounts for a wide range of disturbance vectors that shape a floodplain community.

The SHM Concept recognizes the dynamics of floodplain habitat (Arscott et al. 2002, Stanford et al. 2005); however, because river hydrodynamics is an important driver of the floodplain geomorphic composition, the floodplain SHM literature has focused on this primary disturbance factor while ignoring other important floodplain disturbance drivers such as fire. This may be a result of the limited recent fire occurrence on research floodplains used to develop the floodplain SHM Concept (e.g., Nyack Floodplain, MT-USA; and Tagliamento River, Italy). My study provided a near-census analysis of the North Fork of the Flathead (North Fork) over nearly 25 years. This expanded examination of floodplain dynamics captured a range of overbank flooding events and several floodplain fires. One criticism of the work stated that the timing of the fires within the study period would affect the results of the study. I would argue that this is a valid criticism of the path analysis results relating to the importance of fire on the North Fork floodplain habitat composition. The frequencies of events from riverscape and landscape disturbance regimes differ and the relative importance to the floodplain habitat dynamics depend on the magnitude, intensity and recovery time between each event (Turner et al. 2003). The path analysis provided insight of the importance of fire for each sample year with relatively little change even up to a decade after the last fire event. However the path analysis did not adequately capture the total disturbance/recovery dynamics because it did not include some disturbance elements that were not significant (e.g., exurbanizaiton and logging) nor did it include the influence of recovery pathways.

The graphical analysis provides a nice visualization of the floodplain dynamics across the study period and supported the hypothesis that fire and flood are important disturbances elements across the different major geomorphic features. But I do not provide a quantitative metric of the system dynamics. I inferred that network analytics could provide a quantitative measure that could be readily converted into an ecological indicator. A logical next step would be to use the expansive field of social network analysis to derive measure of ecologically dynamics. Social network metrics include:

- Distance - number of steps from one node to another.
- Size - Diameter as measured by the longest distances between any two nodes in a network with connected nodes have distance 1, and average path length as measured by the average distance between all pairs of nodes.

- Density - The proportion of edges that actually exist relative to the edges that could exist in principle.
- Centrality, Centralization, Point Centrality – The most important vertices within a graph measured as a degree, eigenvector, closeness or betweenness.
- Components and cliques - A clique is a sub-set of nodes where all possible pairs of nodes are directly connected demonstrate homophily: those with similar attributes tend to form ties with one another.

Clearly there are several challenges in applying social network metrics to ecosystems. The most important of which is selecting metrics designed to measure dynamics of social networks that are ecologically meaningful. Another challenge is to test the strength of social network metrics that are robust in very large data sets but may not be when applied to assessing floodplain dynamics from relatively limited data sets. Assuming those challenges can be overcome, then one must determine if the metrics are sensitive enough to detect changes in the floodplain dynamics network across time or space.

Finally, how can one distinguish natural floodplain dynamics from a system altered from climate-driven disturbances? Floodplains are transition zones between aquatic and terrestrial systems that are exposed to both hydrologic and terrestrial disturbances that collectively influence floodplain habitat patch composition and dynamics. These disturbances continuously interact over ecological time scales to create a shifting mosaic of floodplain habitat patches occupied by associated endemic species adapted to the composition and dynamics of these habitats. I found through path analysis that the magnitude of fire's direct effect on floodplain habitat patch composition was greater than that of flooding or geomorphic position. These results were supported by graphical analyses that indicate fire and its legacy drives a large portion of floodplain disturbance and recovery dynamics. Therefore, floodplain SHM Concept should include not only flooding but also fire as a significant driver of floodplain habitat patch composition in Rocky Mountain river floodplains. These disturbance events occur at different frequencies and locations on the floodplain. Availability of remote sensing data and its ancillary products provide a venue to assess whole-river dynamics and increase my understanding of natural processes of large river floodplain transition zones. These also allow us to assess changes in the ecosystem integrity as well as the associated alterations of ecosystem functions and services that these threatened systems provide.

5 Ecological Indicators for Areas with Limited Human Impacts

The intent of this dissertation was to provide a step towards developing an assessment tool that will inform decision makers and the public of the impacts from recent global change on remote areas. I attempted to adhere to the ideal properties of ecological indicator listed by UNESCO (2006) and Rees et al. (2008) but, given the complexities of such an assessment, it is difficult to meet indicator criteria that are designed to measure the more overt anthropogenic disturbance regime. There is great utility in assessment of coarse scale disturbance resulting from climate change (Melillo et al. 2014, World Health Organization 2014) and perhaps a new list of ideal properties of ecological indicators appropriate for such measure should be made. Given my finding in this dissertation, the following UNESCO (2006) and Rees et al. (2008) criteria should remain the same:

1. Informative - convey information that is responsive and meaningful to decision-making;
2. Interpretable - easy to understand and communicate to a as wide a range of stakeholders; and

3. Grounded on scientific theory: Indicators should be based on well-accepted scientific theory, rather than on inadequately defined or poorly validated theoretical links

However, the following criteria should be refined to include:

4. Responsive - linked to a conceptual stressor–response framework,
    a. Must distinguish between climate driven changes, direct human impacts, and natural variability,
5. Sensitive - capable of measuring change or its absence with confidence (robust to influences of confounding environmental factors)
    a. Requires long term studies or historical information to include measure of actual change,
6. Anticipatory - early warning of potential problems;
    a. Include measure of risk of systems that are susceptible to future change;

Finally, the criteria that ecological indicators should be readily measureable (cost-effective assessment metrics) should eliminated, at least for now. Further scientific study is necessary to establish reliable, watershed-scale measures of climate change that would meet the above indicator criteria.

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