Integrating communities into ecosystem-based management

by

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Ethics Statement

The author, whose name appears on the title page of this work, has obtained, for the research described in this work, either:

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Abstract

Designing a system of ecosystem-based management (EBM) requires a context dependent understanding of landscape patterns across space and time. Hence for distinct social-ecological systems grappling with developing new policies to support EBM, researchers and planners need to think critically about the types of data sources and analytical approaches that are most appropriate for a specific situation. In this thesis, I describe my research in the Great Bear Rainforest on the coast of British Columbia, Canada, that involves collaborations with six different First Nation communities. I incorporate data with a historical or Indigenous context to assess and develop novel approaches for spatial analysis and EBM planning. This research was coproduced with Indigenous communities and aims to bring together disparate disciplines and knowledge systems. For example, first, I show that using species distribution models of western redcedar trees developed from combining field surveys and archaeological records can help predict the spatial extent and understand the past distribution of an important biocultural resource with rapidly shifting baseline conditions. Second, I show that using traditional ecological knowledge to refine categories of trees used by Indigenous carvers can change estimates of abundance and thus alter the resulting targets for an intergenerational stewardship strategy. Third, I show that forest harvesting on the central coast of BC, Canada has sequentially targeted the most productive and accessible components of the environment and that policy interventions can disrupt these trends. Fourth, I show that past spatial planning to design a system of landscape reserves significantly exceeded the associated conservation targets and that human and ecological factors affected the overall reserve design. Collectively, this research develops new approaches for using community and historical data in EBM planning and highlights the importance of collaborating with communities to address theoretical and applied research questions.

Keywords: Indigenous communities; ecosystem-based management; Great Bear Rainforest; spatial analysis; historical data, cultural resources

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I have spent over a decade at Simon Fraser University across my two graduate degrees, and a big part of why I hung around so long was due to the encouragement, guidance, and overall wisdom of my senior supervisor, Ken Lertzman. Ken and I have been on countless adventures together conducting field research, working in communities, and teaching, all of which have shaped my knowledge of ecology and forestry. I really want to thank Ken for his contributions to this thesis and for being such a great mentor. I also want to thank the rest of my supervisory committee for their distinct contributions to this thesis: Murray Rutherford ensured that my colloquial policy language and applied policy thinking was actually grounded in the policy sciences; Anders Knudby helped me develop and implement novel approaches to spatial analysis; and Meg Krawchuk provided valuable insights and comprehensive feedback that greatly improved all of my chapters. Having such a talented, thoughtful, and caring group of people to help steer my research makes me deeply appreciate the unique collaborative opportunities that come with doing a PhD.

More broadly, I want to thank my friends and colleagues in our Forest Ecology and Management Lab, especially Julie Nielsen who helped with all my research on monumental cedar. Thanks to the faculty, staff, and students in the School of Resource and Environmental Management for all the interesting discussions over the years and for generally trying to make the world a better place. I am also grateful to the many funders who supported this work, including the Social Sciences and Humanities Research Council of Canada (Joseph-Armand Bombardier Canada Graduate Scholarship), the Hakai Institute, the Tula Foundation, the Institute for Coastal Research, the Canadian Council on Ecological Areas, the Heiltsuk Integrated Resource Management Department, the Nanwakolas Council, and Simon Fraser University.

This work would not have been possible without the support and collaborations of my community research partners. I am very grateful to the Nanwakolas Council as well as the Heiltsuk, K'omoks, Wei Wai Kum, Tlowitsis, Mamalilikulla, and Da'naxda'xw / Awaetlala First Nations for partnering with me to carry out research in their communities and territories. In particular, I want to thank all the staff at the Nanwakolas Council and the Heiltsuk Integrated Resource Management Department, all the Guardians in the Hamayas Stewardship Network, all the carvers who shared their knowledge during our interviews, and the dozens of community members that I learned so much from over the past decade. Listing all the key individuals who contributed to this research would fill many pages, so instead, I will just say thank you, gila’kasla, and ğiáxsixa! However, I will specifically mention my friend and colleague, Mulidzas (Curtis Wilson) who was so instrumental in coordinating the Nanwakolas work on cultural cedar and who recently passed away.

Finally, I need to thank my family for supporting my academic journey over these many years. It doesn’t seem that long ago when my wife, one-year old son, and I drove into family housing at SFU to start graduate school with our truck loaded with all our worldly possessions. But given that my son is now a teenager and my daughter is ten years old, the epic length of my student life is glaringly apparent. It would have been impossible to maintain momentum over this time without all my family’s encouragement and love. To my parents, Jerry and Janet, to my children, Jonas and Karina, and to my wife, Valerye—thank you for coming with me on this amazing adventure!

Table of Contents

List of Tables

List of Figures

Chapter 1. Introduction

There is increasing evidence that human activities are degrading ecosystems and affecting a diverse range of ecological, economic, and cultural values (Dietz et al. 2003; Millennium Ecosystem Assessment 2005). How has society responded to this sustainability crisis? One solution has been to expand areas in which human use is restricted or altered (FAO 2010; Affolderbach et al. 2012), but designing conservation areas and policies for natural resource management is challenging and complex—particularly when these tasks are brought together under a framework of ecosystem-based management (EBM). In contrast to the more utilitarian and reductionist thinking that underpinned resource management of individual species and commodities during most of the 20th Century, EBM is rooted in systems thinking about ecological processes and structures. But EBM also takes a more nuanced view of the human dimensions of resource management by considering people as part of ecosystems and by accounting for cultural diversity across social systems (Grumbine 1994; Lertzman 2009).

EBM initiatives, therefore, need to undertake the very difficult task of accounting for context dependent conditions within a variety of distinct social-ecological systems (Acheson 2006). This includes assessing datasets at appropriate spatial and temporal scales, including those that are ultimately meaningful to local communities and resource users (McLain et al. 2013). Although an emerging body of interdisciplinary research supports the importance of integrating community perspectives and data into EBM, scholars continue to highlight the critical need for more empirical studies that report on methods, processes and outcomes from planning efforts in different jurisdictions (Ballard et al. 2008; Cheveau et al. 2008; McLain et al. 2013). In this thesis, I assess and develop innovative approaches to spatial analysis and planning that combine modern ecological datasets with those with a historical or Indigenous context. The overarching goal of this research is to predict and understand landscape patterns of conservation and resource use that can support the theory and practice of EBM.

Ecosystem-based management in the Great Bear Rainforest

My research focuses on the central coast of British Columbia, Canada—an area sometimes referred to as the Great Bear Rainforest (for more details about this study area, including maps, see Chapter 2-5). This region's EBM planning process has received substantial scholarly attention (Clapp 2004; Smith et al. 2007; Howlett et al. 2009; Price et al. 2009; Cullen et al. 2010; Pearson 2010; Clapp & Mortenson 2011; Dempsey 2011; Affolderbach et al. 2012; Bird 2012; Moore & Tjornbo 2012; Raitio & Saarikoski 2012; Saarikoski et al. 2013), partly because of its global conservation status as one of the largest undeveloped regions of coastal temperate rainforest (Allen 2005). The GBR has also gained notoriety around the world as a focal point in the "The War In The Woods" and, in response, the almost unprecedented policy shift towards 85% of its forests being designated for conservation (Price et al. 2009). In many ways, though, the GBR is being heralded for its transformative role in forest governance because of the novel solutions emerging from First Nations, conflicting stakeholders, and governments in collaborative planning processes (Cullen et al. 2010; Raitio & Saarikoski 2012).

The Great Bear Rainforest contains a unique set of conditions that makes it an interesting case study for evaluating the integration of communities and locally relevant datasets into EBM. For example, communities are explicitly accounted for in the conception of EBM that has been adopted, defined in the GBR as "*an adaptive approach to managing human activities that seeks to ensure the coexsistance of healthy, fully functioning ecosystems and human communities*". Unlike many frameworks developed for EBM that prioritize ecological goals (Grumbine 1994), human well being—which includes broad goals for forestry as well as specific objectives for Indigenous cultural values and resources—is placed on equal footing to ecosystems. This is particularly salient given that the region encompasses 29 First Nation territories and numerous forest-dependent communities with a high proportion of the residents of the region being Indigenous (Allen 2005). Shared decision-making agreements between First Nations and the provincial government provide an additional legal basis for thinking about EBM in a community context (Price et al. 2009). The authority of First Nations in governance of the region is supported by Supreme Court of Canada decisions affirming Aboriginal Title and Rights, and is manifest through government-to-government negotiations (Smith et al. 2007). This type of local influence over forest management comes after more than a century of top-down decisions by external stakeholders including the provincial government and corporate tenure holders (Price et al. 2009).

Collaborative research with Indigenous communities

To embed myself in this study system, I developed research partnerships and collaborated with Indigenous communities with territories that overlap the Great Bear Rainforest. These partners include the Heiltsuk First Nation and the five First Nations that are members of the Na̱nwaḵolas Council: Wei Wai Kum, K'omoks, Tlowitsis, Mamalilikulla, and Da'naxda'xw / Awaetlala. These First Nations have occupied coastal areas in the region for millennia—at least most of the Holocene (Cannon 2000; Fedje et al. 2018) and have thus accumulated rich place-based knowledge about their territories and environmental change over time. Similar to many Indigenous communities around the world (Larson et al. 2010), these Nations are considering strategies to orient forest planning and practices towards the needs and values of local people by asking questions often ignored within the industrial forest management paradigm. My ongoing discussions with community partners were thus instrumental in shaping the overall direction of each chapter to ensure that my thesis concurrently addresses applied questions and contributes to a broad scientific understanding.

In Chapter 2, for example, my research questions partly arise from the Heiltsuk First Nation's interest in better understanding the distribution of a keystone culturally important resource. Monumental western redcedar trees (*Thuja plicata*) are large, high quality trees suitable for special cultural purposes such as carving dug-out canoes, totem poles, and traditional houses, but their distribution in the GBR has been significantly altered due to industrial logging over the past century. Given this shifting baseline, I combine occurrence data from recent field surveys and archaeological records into species distribution models to predict both where these trees are currently growing as well as suitable sites where they may no longer exist. This approach allows me to demonstrate and discuss the utility of using archaeological data in species distribution modelling and EBM planning when the target species is associated with shifting environmental baselines, data limitations, and an important cultural resource.

In Chapter 3, I expand my research on monumental cedar to traditional territories in more southern portions of the GBR and build on some of the methods and recommendations from Chapter 2. I worked with the five First Nations of the Na̱nwaḵolas Council to predict the future abundance of monumental cedar (also referred to as Large Cultural Cedar in these territories), based on the traditional ecological knowledge of their carvers. I interviewed carvers to identify the distinct tree characteristics associated with different types of cultural carving practices and use the resulting criteria as a basis for carrying out field surveys. I examine how using traditional knowledge to refine interpretations of monumental cedar changes estimates of abundance, which ultimately affects intergenerational access to this cultural resource.

In Chapter 4, I investigate a long time series of forest harvesting data—an analysis catalyzed by First Nation concerns about dramatic changes to the forests within their territories. Specifically, I assess whether logging over time has disproportionately targeted the most productive and accessible areas of the landscape and progressively moved towards locations lower in economic value. I then examine how policy interventions affected these trends. Parallel to the concept of “fishing down the food chain”, common in fisheries management, I coin the phrase, “logging down the value chain” to describe this pattern of highgrading. Such shifting baseline conditions have direct implications for First Nation communities that are taking on increased management authority over local forests.

In Chapter 5, I examine the results of past planning efforts to design landscape reserves in the GBR. This spatial network of reserves was created based on pre-negotiated conservation targets between First Nations and the provincial government that were informed by stakeholders from industry and environmental groups, but my First Nation research partners did not know the extent to which these targets had been achieved. Therefore, I examine biophysical and human variables that affect patterns of reserve design and discuss implications for new approaches to EBM planning currently underway in the GBR, including using targets that place a cap on the amount of land that can be allocated to conservation. Finally, in Chapter 6, I develop conclusions based on the collective findings within this thesis.

I come to this research with a lifelong interest in place-based knowledge and engagement with coastal communities. I grew up in a small island community in the south coast of British Columbia and have been involved in forestry and forest ecology for most of my life as a woodlot operator, consultant, and researcher. This has provided a diverse career pathway beyond my academic background that allows me to view the topics discussed in this thesis through diverse lenses. I’ve worked as an operator of and consultant to small-scale forest woodlots, served as President of North Island Woodlot

Association, and studied community forestry for my master's degree. At various times, I worked as a paid contractor for both the Heiltsuk First Nation and the Na̲nwak̲olas Council, focused on projects that involved implementing EBM in a First Nation context. The substantial overlap in topics between my outside work and my PhD research is not a coincidence—many principles of EBM and community forestry resonate with me personally, align with my worldview, and reflect the broader choices I have made about my own lifestyle and community.

Research products and contributions

The chapters in this thesis are all thematically connected, but they were written as individual papers that have been or are intended to be submitted to different academic journals. Chapter 2, for example, has been published in the journal *Diversity & Distributions*, and I have plans to submit Chapters 3 through 5 to *Ecology and Society*, *PNAS*, and *Conservation Letters*, respectively. Therefore, aside from this introductory chapter and the concluding Chapter 6, which are written in the first-person singular form, my other chapters are based on collaborations with co-authors and are thus written in the first-person plural form. These co-authors, as well as my community partners, made significant contributions to this research throughout the entire process, but I am the primary author and led all work in terms of fleshing out the research questions, designing the studies, carrying out fieldwork, conducting analysis, and writing. I also contributed to substantial related research over the course of my PhD that is not reported in this thesis, including unpublished studies specifically developed for my community partners as well as two published journal articles for which I was a co-author.

Statement of Interdisciplinarity

The School of Resource and Environmental Management was founded based on the idea that interdisciplinary research is needed to effectively address sustainability problems. I agree with this view and have made attempts throughout this thesis to weave together different disciplines as well as alternative types of data and knowledge systems. All chapters have broad implications for both the natural and social sciences and have obvious foundations in ecology, planning and policy as well as some connections to ecological economics. My research also extends beyond these core disciplines by bringing together data rooted in traditional ecological knowledge with data more consistent with western science—an approach sometime referred to as transdisciplinarity.

Chapter 2. Combining data from field surveys and archaeological records to predict the distribution of culturally important trees

This chapter was previously published as a journal article with the same title, co-authored by J. Benner, A. Knudby, J. Nielsen, M. Krawchuk, and K. Lertzman in Diversity and Distributions and has been reprinted with permission from Wiley © 2019.

Abstract

Indigenous communities involved in conservation planning require spatial datasets depicting the distribution of culturally important species. However, accessing datasets on the location of these species can be challenging, particularly when the current distribution no longer reflects areas with the full range of suitable growing conditions because of past industrial activity. We test whether using occurrence data from community-based field surveys and archaeological records in species distribution models can help predict the current distribution and reconstruct the past distribution of monumental western redcedar trees (*Thuja plicata*). This species is critically important to Indigenous people of the Pacific Northwest of North America, but trees suitable for traditional carving and building are diminishing in abundance due to logging. Our analysis covers the spatial extent of the traditional territory of the Heiltsuk First Nation, which encompasses a portion of the Great Bear Rainforest in British Columbia, Canada. We built and compared species distribution models using the machine learning program, Maxent, based on occurrence data from field surveys and archaeological records of culturally modified trees. Our findings highlight similarities and differences between the predictions from these species distribution models. When validating these models against occurrences from an independent dataset, the archaeological record model performs better than the field survey model. These findings may arise because the independent dataset was collected on an unlogged island—an environment that aligns more closely with the historic forest conditions revealed by the archaeological records than the current distribution revealed by the field surveys. We demonstrate and discuss the utility of using archaeological data in species distribution modelling and conservation planning when the target species is associated with shifting environmental baselines, data limitations, and an important cultural resource.

Introduction

Indigenous people and communities are gaining enhanced rights and authority over their traditional lands, including the forest resources that are often deeply intertwined with their culture (Larson, Dahal, & Colfer, 2010). In such places, local knowledge and perspectives about culturally important plants and animals are often key factors in successful conservation and resource management initiatives (Berkes, Folke, & Gadgil, 1994; Charnley, Fischer, & Jones, 2007). What kind of data are applicable and meaningful to Indigenous communities in these contexts? Increasingly, data describing the locations of species occurrences are an integral part of spatial conservation planning because these data support prediction of spatially explicit species distributions (Elith & Leathwick, 2009; Franklin, 2009). But questions about how to effectively apply these methods arise when the target taxon is an important traditional resource used by Indigenous groups and is associated with a rapidly shifting distribution. In these situations, using occurrence data based on observations, knowledge, or physical evidence of local inhabitants is an underutilized approach that can provide insights into ecological communities, populations, and resource landscapes (Franklin, Potts, Fisher, Cowling, & Marean, 2015; Lopez-Arevalo, Gallina, Landgrave, Martinez-Meyer, & Munoz-Villers, 2011; Pesek et al., 2009; Ziembicki, Woinarski, & Mackey, 2013). Finding novel ways to bring together and compare alternative occurrence datasets, such as those based on field surveys and archaeological records, holds promise for spatially predicting past and current species distributions and more effectively meeting community objectives for conservation areas.

In the coastal temperate rainforests of north-western North America, western redcedar (*Thuja plicata* Donn ex D. Don; hereafter ‘cedar’) is important to coastal ecosystems, economies, and cultures (Klinka & Brisco 2009; Antos et al. 2016). Cedar is considered the “tree of life” to Indigenous people because of its prominent role across diverse aspects of traditional and contemporary life (Garibaldi & Turner 2004a; Zahn et al. 2018). For example, the emergence of cedar in these coastal forests during the Holocene is associated with rapid technological innovation stemming from its myriad uses in transportation, structural housing material, art, clothes, and spirituality (Hebda & Mathewes 1984; Stewart 1995). Evidence of these uses over past centuries are imprinted in coastal forests by way of culturally modified trees (Turner et al. 2009).

Furthermore, due to its great longevity (>1000 years) and potential sizes (> 3 m diameter and > 60 m tall), cedar is associated with many important ecological functions such as supporting wildlife habitat (Stevenson, Jull, & Rogers, 2006), diverse epiphytic communities (Price & Banner, 2017), soil stability and carbon storage (Klinka & Brisco 2009).

Although cedar is broadly distributed and relatively abundant in the temperate rainforest of British Columbia (BC), those trees of “monumental” quality (Figure 1.1)—large, high quality trees suitable for cultural purposes such as carving dug-out canoes, totem poles, and traditional houses—are rare (Sutherland et al. 2016). Monumental cedar is an inherently scarce resource in managed forests because of the unique developmental pathways required to reach monumental status and the long time periods, typically more than 250 years, required for its development (Sutherland et al. 2016). Accessible, large, high-quality cedar trees are also among the most profitable timber in coastal forests, making them a staple target of industrial logging (Nelson 2004). On the central and north coast of BC, for instance, analyses suggest that logging is occurring disproportionately within highly productive cedar stands (Green, 2007). The overharvesting of high value cedar can erode the natural capital of coastal temperate rainforests similar to high-grading of rare tree species in tropical forests (Schulze et al. 2008a) or large fish in the marine environment (Poos et al. 2010).

In addition to disproportionate harvesting, the current silviculture regime in BC, focused on principles of maximum sustained yield, does not provide sufficient time or possibly the growing conditions necessary for cedar to reach monumental status within the operational land base (LePage & Banner 2014; Sutherland et al. 2016). For example, conventional timber supply models calculate harvest rotation ages within managed forests (typically less than 100 years) that can be an order of magnitude shorter than the age of large monumental cedars (often more than 1000 years; Antos et al., 2016; MacKinnon, 2003; Waring & Franklin, 1979). Such divergences between managed and unmanaged forests are especially marked in the wetter portions of the coastal temperate rainforest, where large-scale disturbances, such as stand replacing fires, are exceedingly rare, leading to a natural disturbance regime characterized by small-scale gap dynamics and generating forests dominated by old growth (Daniels, 2003; Lertzman et al., 2002; Lertzman et al., 1996). Thus, the seral shifts produced by industrial silviculture dramatically decrease the number of large old trees present on the landscape. Such a change is salient in any forest around the world because large old trees disproportionately affect the structure, dynamics, and function in forests (Lutz et al. 2012; Stephenson et al. 2014), but when these stand elements are also a cultural keystone like monumental cedar (Garibaldi & Turner 2004), perturbations to their distribution influences the broader social-ecological system.

Figure 2.1 Pictures of monumental redcedar and culturally modified trees (photo credits: Ken Lertzman, Jordan Benner). Picture (a) is characteristic of an occurrence that met the field survey criteria for monumental cedar status: diameter at breast height greater than 1 m and bole length greater than 5 m with few knots or defects. Picture (b) shows a partially carved canoe that was likely abandoned over a century ago, which is characteristic of one type of aboriginally logged tree occurrence in the archaeological records (accompanying stump outside picture frame). Picture (c) shows a test hole in a redcedar tree—a technique traditionally used by Indigenous people to assess the proportion of heart rot in potential monumental cedar. Picture (d) shows a very large redcedar tree that does not meet the field definition for monumental cedar due to excessive rot and other tree defects.

In the Great Bear Rainforest (GBR) of coastal British Columbia, a regime of ecosystem-based management (EBM; Great Bear Rainforest Order, 2016; Price, Roburn, & MacKinnon, 2009) has been instituted which includes Cedar Stewardship Areas (CSAs), a land designation created to ensure an intergenerational supply of cedar. The Landscape Reserve Design process is the primary way in which areas are set aside from commercial logging based on targets and objectives for biodiversity, timber, and First Nation values (Great Bear Rainforest Order 2016). Although the specific characteristics of CSAs are vague in current planning documents, the concept is that in these areas certain limitations are placed on the commercial harvesting of cedar, but First Nations can access cedar for cultural purposes. According to the current EBM framework, planners must also incorporate First Nations' traditional forest resources and tree use into landscape reserve planning (Great Bear Rainforest Order 2016), thus creating a strong mandate for the conservation of monumental cedar. However, the large scale of spatial planning and the inherent rarity of monumental cedar, coupled with the lack of a comprehensive inventory in the region, makes it challenging to implement CSAs without more knowledge of the distribution of these trees across the landscape.

Species distribution models (SDMs) are widely discussed in the conservation literature and are increasingly being used to inform site selection for spatial planning around the world (Araujo & Williams, 2000; Ferrier, Watson, Pearce, & Drielsma, 2002; Franklin, 2009). Typically, SDMs are used to produce predictive maps that show the probability of species presence, or habitat suitability, based on the statistical relationship between observed species occurrences and environmental factors. One of the major challenges with SDMs in an applied context is that many empirical models are based on potentially biased field surveys (Phillips et al. 2009). Various methods have been developed to account for these issues, including creating target background data to reflect sampling effort (Phillips et al. 2009) and altering occurrence datasets to remove biases (Dudik et al. 2005). Although methods to address bias and uncertainty may improve predictions, they are difficult to apply in an objective manner because detailed information about sampling effort is often lacking (Elith & Leathwick, 2009; Phillips et al., 2009). Integrating multiple independent datasets is another approach to revealing biases and cross referencing knowledge of species distributions (e.g. Lopez-Arevalo et al., 2011). For instance, combining data from modern field surveys of monumental cedars with archaeological records of their past locations may fill the data gap arising because their modern distribution has been shaped by a century of industrial forest harvest. Bringing together such ecological and cultural data can also lead to a spatial conservation design that reflects the patterns and values of past traditional resource use.

In this research, we evaluate different data sources to predict the spatial distribution of monumental cedar in a portion of the GBR of BC, Canada. We examine SDMs derived independently from two types of monumental cedar occurrence data: community-based field surveys carried out by the Heiltsuk First Nation and archaeological records of traditional harvesting locations. We also create a third SDM that combines these two datasets. We first hypothesize that the distributions inferred from the field survey and archaeological datasets will have substantial overlap and exhibit similar relationships across environmental variables, and that both SDMs will be influenced by variables related to access, such as elevation and proximity to the ocean. Next, we compare the predictions based on these individual SDMs with a more recent, independent and systematic, dataset of monumental cedar from field transects, collected for this project. Our hypothesis is that the survey biases in the Heiltsuk field surveys and the traditional patterns of use in the archaeological records will limit congruence with this independent dataset. We also expect that the SDM based on pooled occurrences from the first two data sets will create a more accurate model of the spatial distribution of monumental cedar (as represented by our third data set), potentially via the most extreme biases in either dataset cancelling each other.

Methods

Study Area

Our study area encompasses the traditional territory of the Heiltsuk First Nation on the central coast of BC, Canada—an area that forms part of the GBR (Figure 2.2). This region's EBM regime has received substantial scholarly attention, partly because of its global conservation status as one of the largest undeveloped regions of coastal temperate rainforest (Allen 2005). The GBR has also gained notoriety around the world as a focal point in forestry conflicts (i.e. "the war in the woods") and, in relation, the almost unprecedented shift to 85% of its forests being off limits to logging (Great Bear Rainforest Order, 2016; Price et al., 2009). But this region also has a long history of forest management and stewardship that precedes the modern forest industry.

Indigenous people (referred to in BC as First Nations) have been occupying this territory for millennia, with archaeological records showing human settlement and use of the land dating back to over 10,000 years ago (Cannon 2000; McLaren et al. 2014). Although the Heiltsuk territory covers a large area of land (~15,000 km2), population densities are currently very low (~1 person/4 km2) and comprised of mostly First Nations. Currently, unemployment rates among First Nations are extremely high, with the majority of jobs provided by the local government and the natural resource sectors (Allen 2005). Accessing and distributing traditional food and other resources from the surrounding territory forms the basis of a significant subsistence economy and thus local people often make a clear and direct connection between ecological integrity and community well-being—two central pillars of EBM.

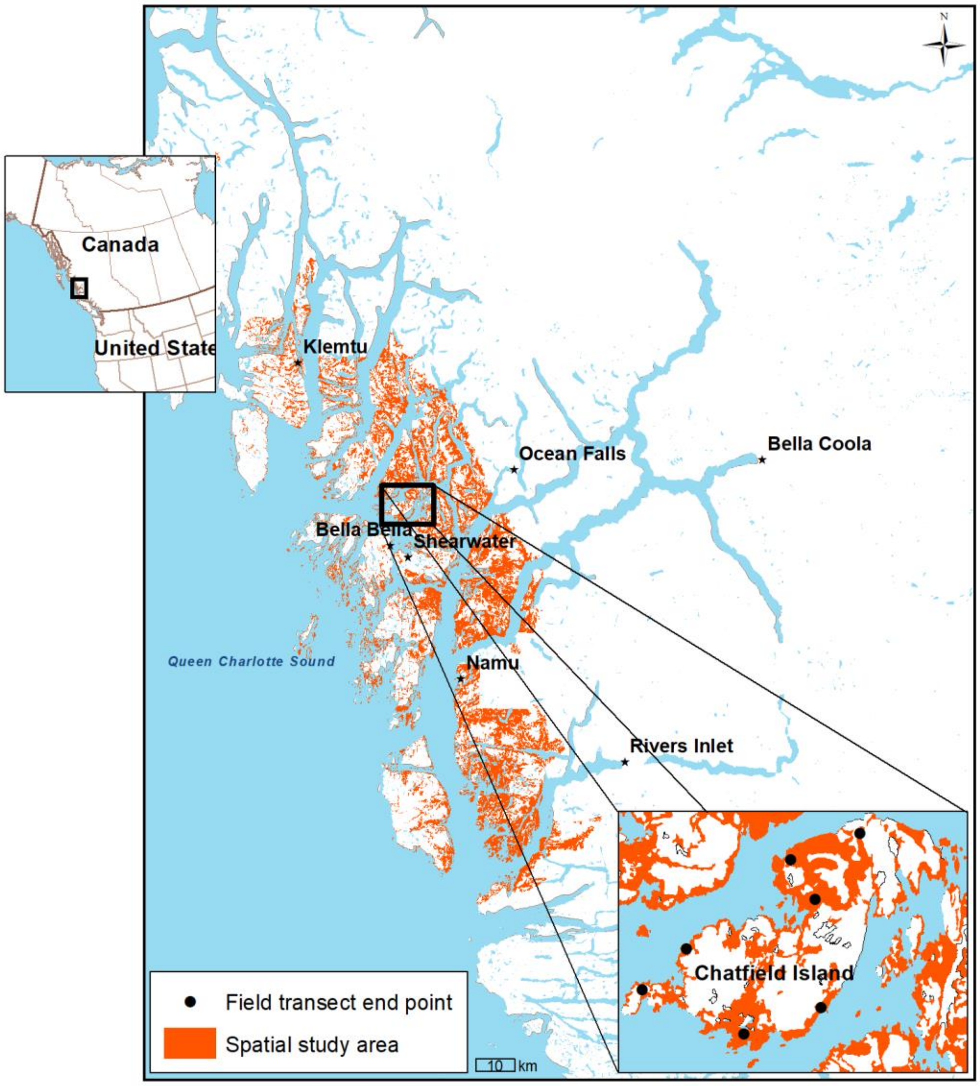

Figure 2.2 Map of the study area (shown in red) used for species distribution modelling. The SDMs calibrated on Heiltsuk field surveys and archaeological records across the entire study area were tested against the independent occurrences from field transects on Chatfield Island. Black dots on the inset map of Chatfield Island show the end points of each field transect.

Ecologically, the GBR region lies within the coastal temperate rainforest and is within the Coastal Western Hemlock (CWH) zone of BC's Biogeoclimatic Ecosystem Classification system (BEC; Meidinger & Pojar, 1991). The BEC system classifies ecosystems across nested scales: zones represent the broadest scale based on climate, site series represent the finest scale based on the local soil moisture and nutrient regimes (Meidinger & Pojar 1991). The GBR region is characterized by high annual rainfall (2000+ mm), moderate average monthly temperatures ranging from 4 to 16°C, and extensive coniferous forests. Heterogeneous physiography creates large regional variation in site productivity. Forests in the floodplains of large river systems can accumulate immense biomass above and below ground, whereas other areas that are severely limited by nutrients and water tables are characterized by bog ecosystems with markedly shorter forest canopies. Over a dozen tree species occupy these forests, the most common of which, depending on site conditions and disturbance histories, are western redcedar, western hemlock (*Tsuga heterophyla*), amabilis fir (*Abies amabilis*), Sitka spruce (*Picea sitchensis*), yellow-cedar (*Cupressus nootkatensis*), shore pine (*Pinus contorta var. contorta*) and red alder (*Alnus rubra*; *Alnus sitchensis* on the outer coast).

The spatial extent of our study area (350,000 ha) represents roughly 25% of the terrestrial area of the Heiltsuk territory. We selected this area based on availability of GIS data and in an attempt to exclude forests that are unlikely to yield monumental cedar because of short tree canopies, logging history, or unsuitable species composition. To identify a study area using these criteria, we queried the Vegetation Resource Inventory (VRI) spatial layer (Data BC; www.data.gov.bc) based on whether redcedar appeared in the species label (constituting at least 10% of forest canopy) of the stand, whether average stand height was equal to or greater than 20m, and whether average stand age was equal to or greater than 140 years. VRI data are derived from interpreted orthophotos and represent stand values averaged across broad areas (typically over 1 ha in size), thus masking fine resolution variability within stands. We created the final study area boundary by clipping to the Heiltsuk territory and by clipping to the Central Very Wet Hypermaritime BEC subzone and variant of the CWH zone (CWHvh2; Meidinger and Pojar, 1991) to distinguish among unique ecosystem types. We used the entire study area for model calibration and the portion of the study area that overlaps Chatfield Island for model validation (Figure 2.1).

Species occurrence data

Heiltsuk field surveys

Fieldwork conducted by Heiltsuk field crews across the Heiltsuk territory during the summers of 2013 and 2014 led to the identification of 68 monumental cedar trees within the study area. The Heiltsuk field crews recorded these data through targeted sampling focused on productive stands with redcedar or yellow-cedar as the leading species and through opportunistic sampling while completing other types of surveys, often around historic village sites. The criteria used by the field crews to identify monumental cedar in these surveys included a minimum trunk diameter of 1 m at breast height and at least 5 m of clear wood (i.e. free of large knots or branches and other tree defects such as excessive sweep or rot). There is no differentiation in this occurrence dataset between redcedar and yellow-cedar—two species that look very similar. However, given that the survey locations are mostly associated with redcedar stands and large yellow-cedar trees are much less common in the region, it is likely that almost all the occurrences are redcedar.

Archaeological records

The BC Archaeology Branch administers a database that contains archaeological features recorded by archaeologists within BC. In 2011, we accessed a GIS shapefile of archaeological features within Heiltsuk territory. Many of these features represent 10 m buffers around points or clusters of points, so we used the centroid of each polygon as our occurrence point. We used occurrence data based on the records describing culturally modified trees (CMTs). Monumental cedar is not listed specifically, but the CMT site type does contain records of 106 Aboriginally Logged Trees within the spatial study area, which identify where trees or portions of trees have been harvested in the past. Species information is not listed for every CMT, but western redcedar represents 84% of the populated fields and the generic term “cedar” represents the rest. We used these records of archaeological aboriginal logging as a proxy of historic monumental cedar occurrences, although further research is needed to quantify how frequently these occurrences would meet the monumental cedar criteria described for the modern Heiltsuk field surveys.

Chatfield Island transects

To create an independent validation dataset for testing models built from the above occurrence data, in July 2015 a team of four researchers from Simon Fraser University and two crewmembers from the Heiltsuk First Nation conducted field transects to identify monumental cedar. Unlike the Heiltsuk field surveys and archaeological records, which span the entire Heiltsuk territory, these transects were confined to Chatfield Island (Figure 2.1; validation study area = 1880 ha). We chose this island to conduct fieldwork based on a combination of the presence of extensive old growth forests with no logging history and logistical feasibility. Chatfield Island encompasses most of the range of ecosystems that are representative of the entire territory, but the island's landscape does not have large mountains or extremely productive floodplain forests that characterize some watersheds further inland. This sampling effort provided a more systematic and representative sample of available vegetation and environmental conditions than either of the other databases.

We sampled along seven transects, involving the team walking linear routes through the forest. We spread out and enumerated all monumental cedar trees within at least a 50 m wide belt transect, although various terrain obstacles limited our ability to travel in a straight line at all times. To anchor the endpoint of each, transect, we generated seven random points within the spatial study area on Chatfield Island using Random Point in ArcGIS 10.2. From the nearest accessible shore location, we travelled by foot to the random point guided by a compass and GPS (Eos Arrow 100 GNSS Receiver), searching for trees that met our criteria for monumental cedar. These criteria were based on the identification methods used during the Heiltsuk field surveys. We also searched on the return trip from each random point, usually on a parallel adjacent transect. The average distance travelled at each location, including the return trip, was 1.4 km (max = 2.1 km, min = 0.8 km). At each monumental cedar tree that we identified, we took photographs and recorded notes including GPS location, tree height using a hypsometer, tree diameter at breast height, and the length and number of clear faces. In total, we recorded 62 monumental cedar trees in our transects on Chatfield Island.

Species distribution models

Species distribution models (SDMs) are used to produce predictive maps that show the probability of species presence, or habitat suitability, based on the statistical relationship between observed species occurrences and environmental factors. To build SDMs, we used the machine learning program, Maxent (Phillips et al. 2006), supported through the 'dismo' Package in R (Hijmans, Phillips, Leathwick, & Elith, 2013; R Core Team, 2015). We developed an SDM for the Heiltsuk field surveys (hereafter *Heiltsuk model*), a second SDM for the archaeological record dataset (hereafter *Archaeo model*), and a third SDM that pools these datasets (hereafter *Combined model*). In building these models we used the default settings in Maxent with a final set of eight variables as environmental predictors (Table 2.1). We chose these variables because they are known to influence tree distributions and because we had access to corresponding spatial data. We examined correlations among variables and removed one potential predictor (Stand Volume) because it was highly correlated with Canopy Height (Pearson's R = 0.84), a variable that is more explicitly linked to the criteria for characterizing monumental cedar. We converted each variable to a raster file, based on the associated value in the cell centre, with 25 m grid cells, using the BC Albers projection. We chose Maxent to model our occurrence datasets because it is designed for presence-only data, provides robust results for small sample sizes compared to many other models, can integrate categorical data, and offers many options for model evaluation (Elith et al., 2006; Phillips et al., 2009).

Table 2.1 Environmental variables used as predictors in the species distribution models.

Variable Name	Source	Description
Elevation	TRIM[a]	Elevation (m) affects cedar growth by influencing temperature and the phase of precipitation (snow vs. rain). Survey intensity likely decreases with higher elevations due to the logistical challenges of accessing this type of terrain (i.e. access variable).
Ocean	TRIM	Euclidean Distance from ocean (m) affects cedar growth by influencing various aspects of microclimate. Survey intensity—and logging intensity—likely decreases with further distances from the ocean due to the logistical challenges of accessing this type of terrain from a boat (i.e. access variable).
Slope	TRIM	Slope (%) affects cedar growth by influencing the rate of precipitation runoff and light availability.
Solar	TRIM	Global solar radiation (WH/m^2; derived using the Area Solar Radiation tool in ArcGIS 10.3) affects cedar growth by influencing direct and diffuse light availability.
Site Index	VRI[b]	Site Index indicates site productivity. Site Index represents the potential height (m) of dominant trees at age 50, measured from breast height.
Canopy Height	VRI	Mean canopy height (m) affects the potential for trees to be characterized as monumental cedar because our criteria include tree size and the amount of clear wood.
Leading Species	VRI	Leading species within the canopy indicates whether or not cedar is the dominant species in the forest canopy.
Site Series	MFLNRO[c]	Site Series is an indicator of cedar growth because it codifies the soil nutrient and moisture regime. This variable is based on a combination of field verified terrestrial ecosystem mapping and modeled predictive ecosystem mapping.

[a]Terrain Resource Information Management (http://geobc.gov.bc.ca/base-mapping/atlas/trim/)
[b]Vegetation Resource Inventory (https://www.for.gov.bc.ca/hts/vri/)
[c]Ministry of Forests Lands and Natural Resource Operations

We created two scenarios for the *Heiltsuk* and *Archaeo* models to account for the potential influence of uneven survey intensity in the Heiltsuk field surveys as well as patterns of traditional use in the archaeological records. The first scenario included all variables (hereafter referred to as Including Access Variables or IAV), whereas the second scenario excluded the two variables most associated with access: proximity to Ocean, and Elevation (Excluding Access Variables or EAV). We developed this latter scenario to enable more explicit comparisons of the models and predictive maps, and so that decision-makers can more easily evaluate whether or not to incorporate access patterns in the design of Cedar Stewardship Areas.

We compared the *Heiltsuk* and *Archaeo* models and associated scenarios by examining three different statistical relationships: model fit, variable contributions to model performance, and probability distributions across the range of environmental values for each variable (i.e. Maxent marginal species response curves). We employed k-fold cross-validation (k=5) for each dataset so that model training used 80% of the occurrences and model testing used the remaining 20%. We used *Evaluate* and the *ROC* functions in R (Fielding & Bell 1997) to assess model fit through area under the receiver operating-characteristic curve (AUC) statistics (Hanley & McNeil 1982). With Maxent this metric represents sensitivity (correct positive predictions) plotted as a function of the proportional predicted area, where values of 0.5 represent random predictions, values above 0.5 indicate performance better than random, and values below 0.5 indicate performance worse than random (Phillips et al. 2006). Unlike models that integrate presence and absence records, the AUC scores in Maxent represent the probability that random presence sites will score higher than random background sites—in our case, 10,000 randomly generated points distributed across the spatial study area. Finally, we accessed Maxent outputs that describe variable importance as well as species response functions that show how the probability of presence varies with changing variable values, while keeping all other variables at their average value.

Predictive Maps

To compare the spatial distributions associated with the two models, we generated predictive maps based on 25 m raster cells. We used the *Predict* function in R to produce these maps (Hijmans et al., 2013). This function uses the statistical relationships within and across variables in the SDMs to interpolate in geographic space the probability of monumental cedar presence. The associated values, represented by coloured cells in the predictive maps, are relative measures of probability of presence, where typical presence localities have a value of around 0.5—a value that is likely higher than monumental cedar prevalence across the landscape. Maxent allows the default prevalence value to be changed, but estimating this parameter was beyond the scope of our study. We then extracted raster cells with values within the 90th percentile for the *Archaeo* and *Heiltsuk* models and calculated areas where these highly suitable areas overlap. We also assessed concordance between the two maps by using the *Istat* function within the 'SDM tools' package of R (VanDerWal et al. 2014). This function calculates the I similarity statistic following (Warren et al. 2008) where 0 represents no overlap and 1 represents complete overlap.

Model Validation

We quantified model performance by testing the associated predictions against our independent validation dataset derived from the Chatfield Island field transects. We used AUC to measure the extent to which the SDMs that were trained on the Heiltsuk field surveys and archaeological records correctly predict the 62 monumental cedar occurrences from the validation dataset. We also examined whether the *Combined* model increased the predictive ability of models relative to using the individual *Archaeo* and *Heiltsuk* models.

Results

Comparison of species distribution models

The *Heiltsuk* and *Archaeo* models show similar variables influencing predictions about the distribution of monumental cedar (Table 2.2). In the EAV scenario, both models show Canopy Height and Site Series among the top three most important predictors. There are also differences between these models: Slope is the most important variable in the *Archaeo* model, and Solar is the third most important variable in the *Heiltsuk* model. In the IAV scenario, the access variables, Elevation and Ocean, make large contributions to model fit and their inclusion increases the AUC relative to the EAV scenario. It is important to note that despite identical occurrence datasets, the response curves and variable contributions associated with the IAV and EAV scenarios are different because the additional variables in the former model alter the interactions among all variables.

Table 2.2 Comparison of model fit (AUC) and the three most important variables, including percentage model contributions, for both the Including Access Variables (IAV) and Excluding Access Variables (EAV) scenarios. Higher AUC values represent better predictive performance.

	Heiltsuk Model		***Archaeo Model***	
	IAV Scenario	**EAV Scenario**	**IAV Scenario**	**EAV Scenario**
AUC	0.940	0.881	0.912	0.850
Variable Contribution	Elevation: 42% Ocean: 19% Canopy Height: 18%	Canopy Height: 42% Site Series: 20% Solar: 18%	Ocean: 43% Elevation: 20% Site Series: 9%	Slope: 30% Site Series: 23% Canopy Height: 21%

When examining the three most important predictors in the EAV scenario, the similarities and dissimilarities between the *Heiltsuk* and *Archaeo* models are evident (Figure 2.3). For example, in both models the Canopy Height variable is associated with an increase in the probability of monumental cedar occurrence (i.e. predicted values) at taller canopy heights (canopy heights had to be higher than 20 m to form part of the study area), though the sparse data in the upper height range makes inferences about the response shape challenging. The modelled response to ecosystem types, as measured by Site Series, varies across the two SDMs. Site series classified as 13—associated with a “very rich” soil nutrient regime and a “very wet” soil moisture regime (Green & Klinka, 1994)—has the highest predicted values in the *Heiltsuk* model. In the *Archaeo* model, site series classified as 01 has the highest predicted values. The 01 “zonal” site series represents the average climatic conditions in the area and is associated with a very poor to medium soil nutrient regime and a moist to very-moist soil moisture regime (Green & Klinka, 1994). The Slope variable also responds differently in the two models. In the *Archaeo* model, where Slope is the most important factor, the response curve shows a slight increase in predicted values up to 20% slopes followed by a general decline in the probability of presence with increasing steepness. Finally, predicted values for Solar Radiation show a fairly consistent relationship between the models and, though hard to interpret in an applied sense, generally show areas with moderately high insolation to be most suitable.

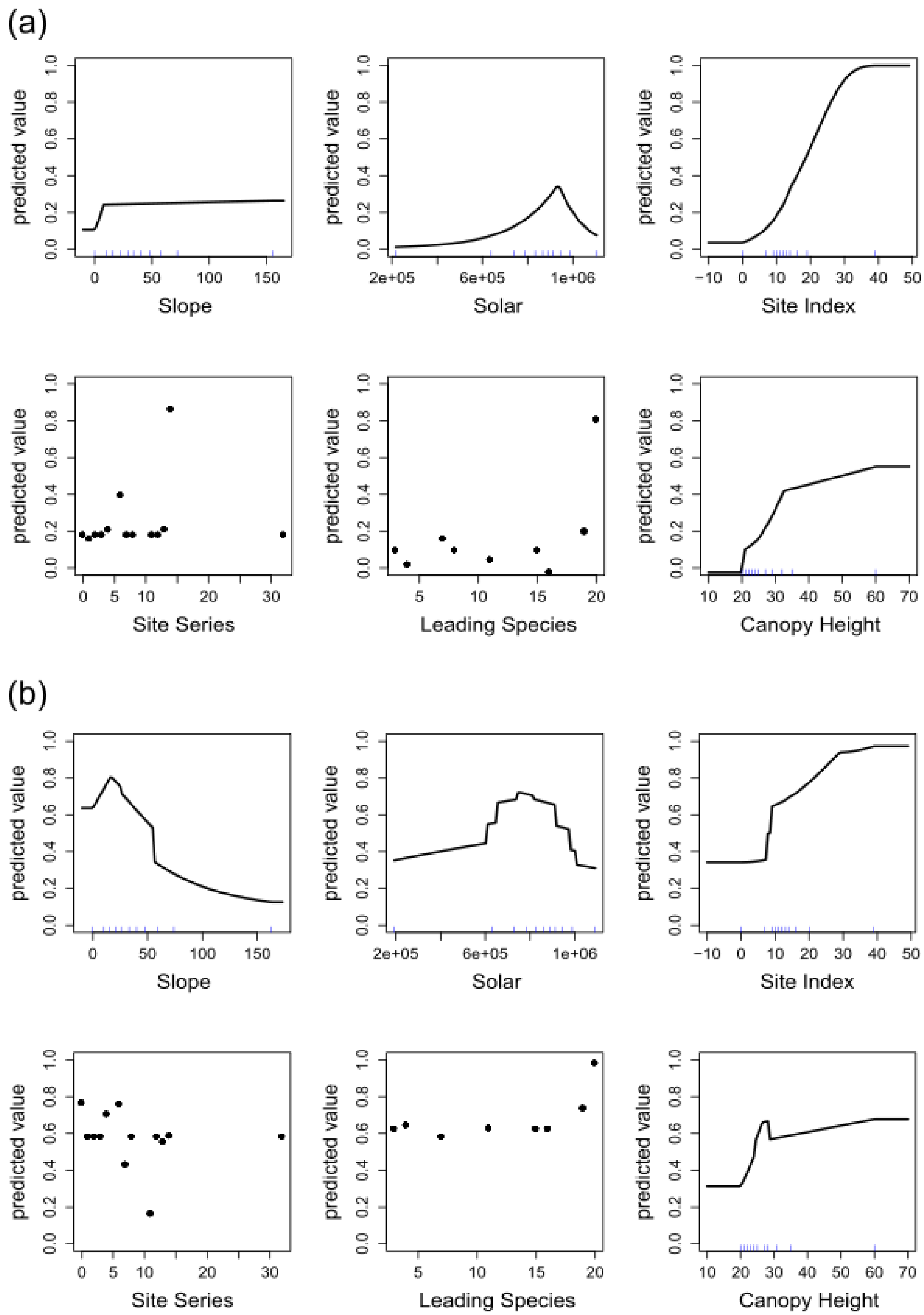

Figure 2.3 **Response curves (EAV Scenario) for variables in (a) the *Heiltsuk* model and (b) the *Archaeo* model. Higher values on the y-axis correspond to a greater probability of monumental cedar presence for given variable values on the x-axis. The variables Site Series and Leading Species are based on categorical values represented by codes (plotted as dots), whereas other variables are based on numeric values (plotted as lines).**

Comparison of predictive maps

Calculating overlap between the predictive maps using the I similarity statistic shows moderate spatial congruence in both the IAV scenario (I = 0.790) and the EAV scenario (I = 0.737). Correspondingly, a visual comparison of the two predictive maps qualitatively indicates distinct similarities and dissimilarities (Figure 2.4a,b). When focusing on areas that are predicted to have very suitable conditions for monumental cedar (e.g. 90th percentile of probability of presence values) there is only 24% overlap between these maps (Figure 2.4c).

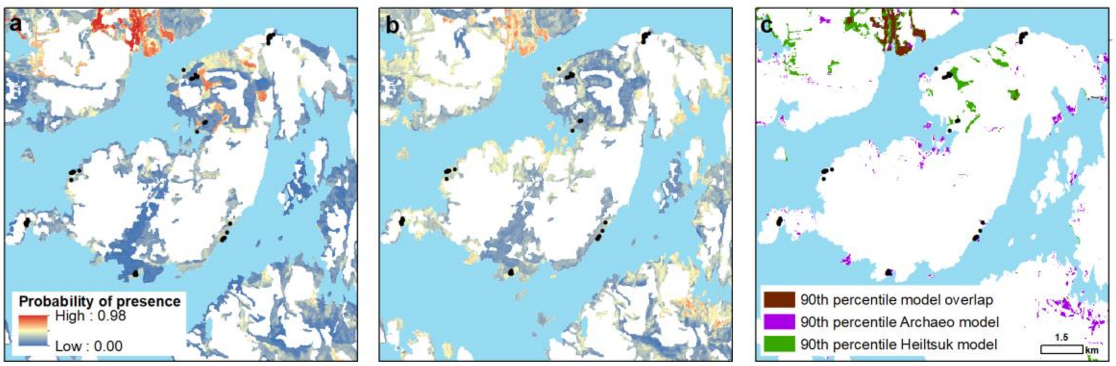

Figure 2.4 **Predictive maps for the spatial study areas around Chatfield Island showing the probability of monumental cedar presence from the (a) *Heiltsuk* model and (b) *Archaeo* model (EAV scenarios). In panel a and b, red indicates high probability of suitable conditions, yellow indicates conditions typical of where monumental cedar are found, and blue indicates low probability of suitable conditions. Panel c shows the 90th percentile of probability of presence values for each model, with areas of overlap in brown. The black dots in each panel represent the monumental cedar identified during field surveys.**

Model validation

Evaluating the SDMs against the independent validation dataset from occurrences on Chatfield Island (EAV scenario) shows predictive performance highest when using the *Combined* model (AUC = 0.751) or the *Archaeo* model (AUC = 0.745). The *Heiltsuk* model on its own performs poorly (AUC = 0.594).

Discussion

Integrating cultural occurrence data into species distribution models

Integrating occurrences from community-based field surveys and archaeological records into species distribution models provide data that can support predictions of the distribution of monumental cedar trees. When should Indigenous communities consider using archaeological occurrence datasets in species distribution modelling and conservation planning? If communities are only concerned with accurately mapping current presence distributions, then species inventories from large, rigorously designed surveys will probably be the most valuable data source for developing SDMs. In these situations, patterns of traditional harvesting sites across the landscape might be considered a "bias" that needs correcting. However, Indigenous communities involved in spatial planning processes can face unique challenges and often have a broader set of considerations, including shifting environmental baselines, data limitations, censored datasets because of past resource management activities, and distinct cultural objectives.

For example, triangulating results from field inventories with archaeological data has the benefit of extending the temporal resolution of species occurrences. The archaeological records of aboriginally logged trees used in our study spatially reference the location of traditional cedar harvesting sites over a long time period—likely up to several centuries, given the slow decay rate of cedar (Daniels 2003). This time scale is relatively recent, however, compared to other applications of archaeological data in SDM that involve hindcasting over millennia to different climatic conditions (Franklin et al., 2015). Incorporating predictor variables into SDM that reflect past climate is an important approach for reconstructing resource paleoscapes (Franklin et al., 2015), but is less critical in our study because the trees harvested in the aboriginally logged records survived in roughly the same climate as old monumental cedar trees still present on the landscape.

Including archaeological information in SDM is especially important for resources with rapidly shifting baseline conditions such as monumental cedar in the study area (Green, 2007) and other large old trees worldwide (Lindenmayer et al. 2012). This situation causes the modern distribution of remaining trees to be a censored dataset because they represent a non-random sample of the original distribution relative to various environmental gradients. The shift, due to industrial logging, in the locations where monumental cedar trees remain indicates that simply focusing on living occurrences will result in a biased sample of suitable growing conditions. If communities and planners are interested in the future recruitment of such trees, not just an inventory of existing ones, then using datasets with a historical context (Rhemtulla & Mladenoff 2007) such as archaeological records is important for understanding landscape patterns over time and for representing the broader distribution of potentially suitable areas. They should not rely solely on the current presence locations reflecting a century of distributional censoring by logging. This shifting baseline may have contributed to the *Archaeo* model having 8% poorer model fit, based on AUC scores, than the *Heiltsuk* model when tested against a partition of its own dataset, but 20% better predictive performance when these models were tested against the independent validation dataset. This latter dataset was derived from an area with almost no logging history, and thus should more closely align with the distribution revealed by the archaeological records.

Data limitations are another important reason to assess archaeological information in SDMs. Although Indigenous communities usually retain rich traditional ecological knowledge about culturally important plants and animals (Berkes et al., 1994), corresponding location data related to these species' distributions is typically less common. Where such limitations exist, using proxies such as archaeological records or traditional use data (Tobias 2009) can help fill this data gap. In our study, the *Archaeo* model has better predictive performance than the *Heiltsuk* model when tested against our independent validation dataset. The overall AUC is also marginally highest when pooling the occurrence data in the *Combined* model, perhaps by reducing the influence of the most extreme biases associated with either occurrence dataset. This model comparison suggests that, in the absence of robust field survey data, using archaeological records of culturally modified trees provides a good foundation for designing conservation areas.

Finally, Indigenous communities often have distinct cultural objectives in natural resource management and conservation planning. Culturally-based worldviews, for instance, may consider all land within traditional territories important to the survival of plants and animals—not just specifically designated zones—or may consider humans as an explicit part of the ecosystem (Berkes, 2007; O'Flaherty, Davidson-Hunt, & Manseau, 2008). In these cases, the use of resources might be viewed as an important coupling of ecological processes and socio-cultural behaviour that is essential to conservation design for social-ecological systems (Krebs et al. 2012; Polfus et al. 2014). In particular, integrating information about traditional patterns of Indigenous species use into conservation planning and resource management is critical for sustaining cultures because the associated harvesting locations are often intrinsically tied to Indigenous ways of life (Pesek et al. 2009). For example, in their study of the distribution of an important medicinal plant, Baumflek et al. (2015) used sociocultural variables, such as distance to roads, to constrain predicted suitable areas to locations considered accessible by Indigenous harvesters. Similarly, in our study's IAV scenario, more suitable conditions for monumental cedar are predicted closer to shore and at lower elevations, a trend that is better explained by the logistics of accessing monumental cedar than by the biophysical suitability for monumental cedar growth. Therefore, despite not necessarily revealing the true species distribution, SDMs based on such data are important if the objective is to create community-based conservation areas that reflect patterns of traditional use, or if future access for continued use is an important design criterion.

Uncertainty arising from spatial datasets

Across SDM studies worldwide, data quality, grain, and availability are often limiting factors in their application (Elith & Leathwick, 2009; Franklin, 2009). In contrast to many jurisdictions, our study area has extensive spatial data stemming from detailed planning and analyses associated with the Great Bear Rainforest over the past two decades (Price et al. 2009). Despite having such data, both SDMs in this study would benefit from more robust environmental predictors. Incorporating spatial data captured through high-resolution LiDAR sensors (Lefsky et al. 2002), for example, would provide a more accurate and fine scaled representation of Canopy Height and topographical variables such as Slope, Solar, and Elevation that are related to the distribution of monumental cedars. Potentially, combining LiDAR with hyperspectral data or other forms of remote sensing could even enable the mapping of all large cedar trees directly (Hyde et al. 2006), though tree defects and wood quality would still have to be assessed in the field to determine monumental status.

In addition to issues with the predictors, there are certain limitations and survey biases, beyond just access patterns, that underlie the cedar occurrence datasets in this study. In the *Heiltsuk* model, occurrences are partially based on intensive sampling effort around riparian areas and important cultural sites such as historical villages. This sampling bias potentially increases concordance with the *Archaeo* model because separate analysis (we only modelled biophysical variables in this study) suggests that proximity to village sites is an important predictor of aboriginally logged trees. Archaeological records of cultural cedars also have biases arising because the data are often collected in the context of archaeological impact assessments associated with forestry. Hence suitable conditions for logging, such as gentle terrain and productive forests (see Chapter 4), might help to explain the predictions for the Slope and Site Index variables (Table 2.1). Even the validation dataset, which is associated with the most random, independent survey design, is limited to one large island and thus does not cover the full range of environmental conditions across the larger territory. More research is needed to understand the extent to which these data limitations affect our findings.

Predicting the distribution of monumental cedar trees

The spatial predictions from the *Heiltsuk* and *Archaeo* models used in this study have moderate overlap. A higher degree of congruence between these predictive maps is not surprising given the survey biases and underlying differences in what the occurrence datasets represent. Each of the SDMs and scenarios in our study could be useful in conservation planning depending on the specific community objectives for monumental cedar. Local planners could prioritize the *Archaeo* model for its prediction of suitable conditions for recruitment, the *Heiltsuk* or *Combined* model for their prediction of the current distribution across the managed forestry land base, the IAV scenario for its prediction of traditional patterns of harvesting, or the EAV scenario for its prediction of suitable areas not constrained by access. If communities want to account for multiple data sources to address issues of survey bias, priority locations could be selected from portions of the landscape where highly suitable areas from different models overlap (Figure 2.4), or from a predictive map based on a single model that uses pooled occurrence data (e.g. *Combined* model).

To identify these highly suitable areas, a variety of methods could be used for selecting a specific threshold from the continuous Maxent predictions (Franklin, 2009). For instance, cells with values at the threshold of maximum sensitivity could be selected as priority sites for Cedar Stewardship Areas. At this threshold in the *Archaeo* model (sensitivity = 0.30), the top 30% of most suitable areas contain 82% of monumental cedar occurrences within the validation study area on Chatfield Island. These highly suitable areas could be translated directly into Cedar Stewardship Areas—ideally in combination with detailed field mapping—or they could be used as an input into conservation prioritization exercises, such as the Landscape Reserve Design process in the GBR, that account for multiple landscape values (Moilanen et al. 2011; Whitehead et al. 2014).

Understanding suitable growing conditions of monumental cedar trees

Our study primarily focuses on comparing alternative datasets to predict the distribution of monumental cedar, but do our results also help to explain these trees' distribution? In general, the Heiltsuk field surveys and transects on Chatfield Island suggest that monumental cedar trees are rare elements across the landscape. These findings are consistent with Sutherland et al. (2016), who found less than 3% of inventoried trees in their study suitable for large cultural carving practices. The response curves suggest that monumental cedar is relatively plastic and can grow in a range of conditions and that these trees tend to occupy sites that are relatively productive. This correlation with site productivity is, perhaps, not surprising given that these types of sites are typically associated with larger trees that more easily meet the criteria for monumental status. Overall it is challenging to determine the extent to which the modeled relationships among variables stem from ecological conditions and processes versus modelling and survey design limitations. For example, both models show a low probability of monumental cedar presence on flat terrain (0-5% slope). Follow-up research is needed to reveal whether this trend is driven by biological explanations, such as heavy competition from Sitka spruce in flatter floodplain ecosystems, or whether the trend is driven by issues such as a lack of surveys in these types of environments or the patterns of logging specific ecosystems over time (see Chapter 4).

In addition, a greater understanding of the growing conditions of monumental cedar would require a broader set of environmental variables across a hierarchy of scales. Some of our variables (e.g. Site Series) are appropriate for explanation because they describe underlying environmental conditions regardless of the trees growing on the site. Other variables, such as Canopy Height, are a manifestation of site characteristics and are thus more appropriate for predicting current distributions. The scale of these variables is also important to understanding distributions. Regional distributions are affected by broad climatic and topographic gradients, while distributions within a forest stand are more driven by micro site factors, such as fine-scale temperature, light, moisture, and nutrient regimes coupled through time with biotic interactions and disturbances (Mackey & Lindenmayer 2001; Pearson & Dawson 2003).

Recommendations for future research

Comparing and interpreting our SDMs and predictive maps provides useful insight into an important ecological and cultural keystone species, but more fundamental research on monumental cedar is clearly needed to predict and explain its distribution. Even the definition of “monumental” needs refining. Uncertainties that arose while planning and carrying out surveys suggest that there is a need to develop more nuanced field identification criteria based on cedar carvers’ preferences for desired wood characteristics (e.g. Sutherland et al. 2016). Then communities could carry out carefully designed surveys across traditional territories to capture more data on the presence, absence, and prevalence of these specific cultural tree grades. There are distinct trade-offs in approaches for designing these surveys. Targeting specific sites based on a priori ideas about suitable conditions will more quickly build an inventory of field validated locations, whereas a more randomized approach will create robust predictions across the full range of environmental gradients.

Like archaeological records, incorporating traditional or local knowledge into SDMs could support the identification of suitable monumental cedar growing sites that, due to logging, are no longer part of the current distribution. Knowledge holders could inductively rank the importance of environmental variables and describe the relationships across the range of variable values (Clevenger et al. 2002; Doswald et al. 2007; Fourcade et al. 2013; Polfus et al. 2014). Alternatively, local knowledge could be embedded in SDMs through interviews with traditional harvesters, whereby they record on a map, locations representing harvesting sites of specific carving materials. These presence locations could then be used in an empirical SDM, using methods similar to this study. But creating multiple predictive maps from traditional knowledge, archaeological records, or field surveys does not alleviate the difficult task of generating a single spatial layer that can be brought forward into planning processes. Because there is not perfect spatial concordance between these maps, the community could iteratively refine predictive maps based on traditional knowledge and local objectives for accessing monumental cedars over time (Pesek et al. 2009; Baumflek et al. 2015). Finally, to ensure an intergenerational supply of monumental cedar trees, information on predicted suitability or prevalence could then be cross-referenced with the anticipated community needs over time for canoes, totem poles, and traditional housing materials.

Conclusion

In this study we focus on novel ways to predict and conserve monumental cedar, but the framework and steps outlined here can extend to other important species and traditional resources globally, particularly given the ubiquity of culturally modified trees in forests around the world (Turner et al. 2009). Forest management regimes need to better protect these types of biocultural features due to their application in understanding cultural practices through time. Ultimately there is a suite of quantitative and qualitative methods available to examine the conditions that underpin distributions and prioritize locations for conservation areas. But the amount of resources and time required to pursue all this research can be an impediment to finalizing a conservation plan. So, whether it is the Heiltsuk Nation considering strategies to conserve and steward monumental cedar trees or other groups trying to map and manage resources, communities will need to decide what mix of methods and tools are most appropriate for their objectives. Whichever approach is chosen, it is important that communities consider and assess datasets that reveal past distributions and traditional patterns of resource use because industrial development is rapidly shifting the current distribution away from many suitable and culturally important locations across the landscape.

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Chapter 3. Using traditional ecological knowledge to understand the diversity and abundance of culturally important trees

Abstract

Combining Indigenous traditional ecological knowledge (TEK) with scientific research holds promise for more effectively meeting community objectives for the conservation of cultural forest resources. Our study focuses on predicting the abundance of western redcedar trees (*Thuja plicata*) within the traditional territories of five Indigenous Nations that are part of the Na̲nwak̲olas Council in British Columbia, Canada. Indigenous people in this region use western redcedar extensively for cultural practices such as carving dugout canoes, totem poles, and traditional buildings. However, after more than a century of industrial logging, the abundance of redcedar suitable for these types of practices is in decline and no longer reflects past baseline conditions. We assess how using TEK from interviews with Indigenous carvers refines predictions of resource abundance compared to using only conventional field surveys. Our findings reveal that western redcedar trees suitable for traditional carving are generally rare, and that some important growth forms, such as those associated with carving community canoes, are nearly extirpated from the landscape. Our study demonstrates a useful application of TEK in conservation planning and highlights concerns about the impact of industrial forestry on culturally important trees.

Introduction

Combining knowledge from different sources and epistemic systems is needed to understand system diversity, an important factor underpinning conservation and natural resource management strategies (Lertzman 2009; Salomon et al. 2018). Research, for example, suggests that accounting for diversity in genetics, behaviour, and functional groups may help to predict a system's ability to adapt and re-organize following perturbations (Folke et al. 2004). Understanding these finer levels of organisation can also reveal patterns and structures that are associated with important ecosystem services, such as the provisioning of fish, timber, and carbon (Schindler et al. 2010; Dymond et al. 2014; Dhar et al. 2016). However, conservation initiatives also benefit from an understanding of the broader social-ecological system, which includes biocultural knowledge about specific places and resources (Berkes et al. 1994; Acheson 2006). Indigenous communities may hold different perspectives and interpretations than those founded in western science about species and the environment that are based on distinct traditional uses, cultural connections, and language (O'Flaherty et al. 2008; Turner et al. 2009). The Inuit, for instance, are well known for their comprehensive vocabulary involving dozens of words to describe the many forms and uses of Arctic ice and snow (Krupnik 2011). Indigenous groups also have traditional names and knowledge bases that reflect local systems of categorizing biodiversity that can differ substantially from scientific taxonomy (Wilder et al. 2016). This biocultural lens can greatly enhance ideas grounded in ecological thinking and the natural sciences (Lepofsky 2009). It is also critical for understanding Indigenous resources around the world, such as specific growth forms of trees that are tied to different types of traditional uses (Turner et al. 2009; Blicharska & Mikusiński 2014; Chapter 2).

Diversity within traditional resources, including cultural classifications developed by local users, can only be integrated into conservation and natural resource management if researchers and planners work with local communities to understand traditional ecological knowledge (TEK) and other local knowledge. In their often cited definition, Berkes et al. (2000) describe TEK as "the cumulative body of knowledge, practice, and belief, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and with their environment". Combining TEK with western science can be a powerful tool in the search for sustainability (Kimmins 2008; Lepofsky 2009; Huntington et al. 2011) and has been used to inform resource management (Charnley et al. 2007), and understand relationships between environments and species (Polfus et al. 2014). TEK can also help to understand environmental change, including shifting baseline conditions, due to its association with specific places over long time periods (Savo et al. 2016). This temporal depth can help reveal species that no longer occupy a local environment, or species relationships and behaviours that have changed over time (Huntington et al. 2011; Wilder et al. 2016). The benefits of combining TEK with western science are becoming increasingly apparent and are advocated by many governments and institutions (Shawoo & Thornton 2019). However, bringing these knowledge systems together in a unified conservation initiative is challenging (O'Flaherty et al. 2008), and increasingly so because of the rapid loss of traditional languages, songs, and knowledge holders around the world (Turner & Turner 2008; Davis 2010; Fernández-Llamazares & Lepofsky 2019).

There is thus a profound need to understand biocultural perspectives and classifications of Indigenous resources through TEK when assessing the status and abundance of species and habitats. Excellent examples of such approaches are described in Turner et al. (2009), who provide a review of culturally modified trees around the world. These scholars highlight many situations where TEK formed the basis for understanding distinct tree forms, including bark and branches, that are associated with a wide range of traditional uses and practices. Another case study, led by researchers and Indigenous communities in the Amazon, shows that categories of medicinal plants and knowledge of suitable habitats developed by Q'eqchi' Maya healers helped researchers understand connections between species distributions and traditional patterns of use (Pesek et al. 2010). In China, Mao, Shen, & Deng (2018) used TEK to develop categories of traditionally used plants to demonstrate the importance of accounting for cultural resources that have more than one purpose and thus may be associated with different types of gathering methods. Without biocultural classifications of key resources, conservation initiatives and natural resource management will likely fail to account for the full range of locally meaningful, context-dependent diversity within Indigenous territories.

Here we blend qualitative and quantitative data to understand culturally significant, within-species diversity and the long-term sustainability of a keystone cultural resource: western redcedar (*Thuja plicata* Donn ex D. Don). We use a community-based research approach and a study area that covers the traditional territories of five Indigenous Nations that are part of the Na̲nwak̲olas Council in British Columbia, Canada. The Kwak'wala name for redcedar used by these Nations is "Auda". Indigenous people in this region, similar to many cultural groups in the Pacific Northwest of North America, use this species extensively for cultural practices related to clothing, transportation, housing, and spirituality—these myriad uses have led to redcedar being described as "the tree of life" (Stewart 1995; Garibaldi & Turner 2004). Western redcedar is a common species in some coastal ecosystems (Green & Klinka 1994), but the largest growth forms that are suitable for carving dugout canoes, totem poles, large ceremonial masks, and traditional buildings are rare (Chapter 2; Sutherland et al. 2016; Figure 3.1).

Figure 3.1 Pictures of (a) a Large Cultural Cedar tree and some traditional practices associated with its wood: (b) totem pole, (c) dug-out canoe (d) big house (photo credits: Mark Wunsch and Ken Lertzman).

This scarcity stems from industrial forestry practices that target these trees' highly valued timber (Green 2007) as well as the unique environmental conditions, including many centuries of growth (Daniels 2003), required for trees to develop the large sizes and other morphological characteristics suitable for carving. These distinct trees are often referred to as "Monumental Cedar" or "Large Cultural Cedar" (LCC) depending on the local context (see Chapter 2). To address First Nations' concerns about the long-term supply of LCC, including the current status of different culturally important growth forms, the Na̲nwak̲olas Council is developing a Large Cultural Cedar Strategy that aims to steward this important traditional resource for current and future generations. Community-based policies like this are relevant to scientific discourses because many people assert that TEK is important and useful, but it is ultimately hard to find concrete examples of its application in natural resource management.

Hence, our study contributes to both broad scientific scholarship and this applied LCC strategy. Specifically, our objectives are to 1) categorize different morphologies of western redcedar trees according to their traditional uses by Indigenous wood carvers, 2) assess how accounting for these culturally distinct growth forms refines our predictions of abundance across the traditional territories, 3) quantify the extent to which forests within these territories contain enough suitable trees to support cultural carving practices over the next three centuries. This overall research approach aims to address, in part, many of the gaps and recommendations identified in Chapter 2, including developing a more nuanced understanding of cultural redcedar, carrying out surveys across the range of environmental gradients within a region and cross referencing estimates of abundance with the cultural needs of the communities.

Methods

Study system

Our study area overlaps a subset of forests in the traditional territories of five Kwakwaka'wakw Indigenous groups on the Pacific coast of British Columbia, Canada (Figure 3.2). These First Nations (as they are referred to in this region of Canada) include the K'ómoks, Wei Wai Kum, Da'naxda'xw Awaetlala, Tlowitsis, and the Mamalilikulla, whose combined territories cover a terrestrial area of 21,604 km^2 spread over many islands and adjacent mainland regions. These First Nations assert legal

Aboriginal Rights, including title, over their unceded territories, and some portions of the territories are part of ongoing treaty negotiations. The territories include moderately sized towns of Indigenous and non-Indigenous people (population ~ 35,000) as well as other more remote areas with very low population densities. Archaeological evidence shows Indigenous people and communities occupying this region for over 10,000 years (Fedje et al. 2018). Collaboration among these Nations occurs through the Na̲nwak̲olas Council—a regional organisation that acts as a vehicle for member Nations to work together on land and marine planning (www.nanwakolas.com). Through various government-to-government agreements, the Na̲nwak̲olas member Nations undertake forest planning initiatives through a shared decision-making process with the provincial government of British Columbia (www.nanwakolas.com). The Na̲nwak̲olas Large Cultural Cedar strategy, which our study is informing, is one of these important initiatives (see Supporting Information).

Figure 3.2 **Map of study area, including the location of Large Cultural Cedar (LCC) surveys. The green study area, used for survey design and to assess LCC abundance, is based on potentially accessible and suitable LCC polygons across First Nations territories of the Nanwakolas Council. We could not access spatial data to identify these types of polygons for the most southern and northern portions of the territories, which is why the map does not show the full extent of the territories. The traditional carver knowledge that informs this study reflects a broader understanding of the territories that is not necessarily constrained to these green spatial polygons. The inset map shows the distribution of the coastal temperature rainforest (based on Wolf et al. 1995).**

These Nations' territories are covered by extensive coniferous forests that form part of the coastal temperate rainforest biome (Wolf et al. 1995) of the Pacific Northwest. Despite being part of this larger biome, natural disturbance regimes and average climatic conditions vary substantially across the study area (Meidinger & Pojar 1991). The south-east portions of the territories on Vancouver Island, for example, are characterized by a climate with average annual rainfall ~1200 mm, whereas more northern and continental areas around Knight Inlet receive more than twice this amount of precipitation and are cooler, with very steep mountainous topography containing permanent glaciers. Due to this environmental heterogeneity, as well as extensive industrial forest harvesting over the past century, forests in this region vary substantially in age structure (Meidinger & Pojar 1991; B.C. Ministry of Forests 2010). Productive ecosystems accessible to timber harvesting typically contain younger forests, whereas portions of ecological gradients that are less accessible and less productive—generally less profitable for harvesting timber—are associated with higher remaining proportions of structurally complex old growth forests (Chapter 4).

Carver interviews

To gain knowledge about tree characteristics that support different types of traditional carving practices, we conducted 13 semi-structured interviews in 2017 and 2018 with carvers from the Na̱nwaḵolas member Nations. This chapter is part of a broader set of studies using these interview data, which focus on topics related to the cultural value and historical use of cedar, influences and changes to carving practices over time, and cedar stewardship. Interviews typically took 2-3 hours and were conducted in the carvers' communities across Vancouver Island, BC. I was present and took part in seven of these interviews and Julie Nielsen (another PhD student) was present at all of them—we each led different portions of the interviews when we were both present.

We transcribed and performed thematic content analysis on interview data using the software program NVivo 12 for Mac (NVivo qualitative data analysis software 2019), which helped us to organise the carvers' knowledge of tree characteristics into general themes. We used these data to develop methods for field surveys that included an LCC identification manual listing the acceptable quantitative thresholds for eight different categories of tree morphological characteristics. These include specifications for tree diameter, length, knots, twist, sweep, rot, scars and seams, and shape (Table 3.1). Although western redcedar, as a species, is culturally important and used broadly by First Nations, the larger growth forms that contribute to LCC logs were of most concern to our community research partners because they were considered least available. We also delivered four different training sessions based on these methods to a total of 16 First Nation stewardship workers.

Table 3.1 Morphological characteristics for Large Cultural Cedar described by carvers. These thresholds roughly represent average values that were recorded across 13 interviews. However, the carvers sometimes had individual preferences that deviated from these standards, including more detailed information that reflects variations on the four listed LCC categories.

Characteristic	Threshold for identifying an LCC tree
Diameter (measured at 1.3m above ground level)	Totem Pole: Greater than 100 cm Chief Canoe: Greater than 120 cm Community Canoe: Greater than 150cm Big House Log: Greater than 100 cm
Length	Totem Pole: Greater than 5 m Chief Canoe: Greater than 7 m Community Canoe: Greater than 12 m Big House Log: Greater than 5 m
Knots	All LCC: one side (½ tree circumference) with knots less than 5 cm; opposite side can have larger knots
Twist	Totem Pole: Minimal twist Chief Canoe: Minimal twist Community Canoe: Minimal twist Big House Log: Less than 20 cm twist over 1 m length
Sweep	Totem Pole: Minimal sweep Chief Canoe: Less than 15% displacement of the diameter Community Canoe: Less than 15% displacement of the diameter Big House Log: Minimal sweep
Rot	Totem Pole: Less than 1/3 of the log diameter Chief Canoe: No rot Community Canoe: No rot Big House Log: Minimal rot (depends on log type)
Scars and Seams	All LCC: 2 or more quarters of the total circumference with scars or seams that are less than 10 cm deep
Shape	All LCC: One round side

Field surveys

We conducted field surveys across the study area to estimate the abundance of different types of LCC trees that are suitable for canoes, totem poles, and big house logs (Table 3.1). Small teams that included the First Nation stewardship workers that had completed the LCC training course carried out these surveys in 2017-2018. We used a survey design based on attributes listed in the provincial Vegetation Resource Inventory (VRI; British Columbia 2019) or from forest cover datasets provided by the major forestry tenure holders in the region. These data are based on orthophoto interpretation of forest stand attributes (British Columbia 2019). Where these two datasets overlapped, we used the tenure holder's data. We were not able to access forest inventory information for portions of a few management units and for some private land, meaning that our study area did not include the full range of stands where LCC might occur within the territories.

We delineated our study area based on forests that have reasonable accessibility for field sampling and future harvesting as well as potential for LCC occurrence based on the findings outlined in Chapter 2. In ArcGIS (ESRI 2019), we created a spatial subset of our forest inventory data where the forest cover information contained the following attributes: species composition in the main or upper canopy includes western redcedar, average stand height ≥ 25m, average stand age ≥ 140 years, and distance to road or ocean ≤ 500 m. The spatial extent of this query represents 69,863 ha—less than 2% of the total terrestrial land base in the Na̱nwaḵolas member territories. The study area would obviously capture more polygons and a larger proportion of the territories if we did not account for accessibility, but the First Nation research partners guiding this project wanted to understand LCC abundance in terms of the land base where trees can be harvested according to contemporary logging methods. Road networks built in the future will certainly expand the supply of accessible LCC, although the development of future roads is highly uncertain given changing markets and policy around logging old growth forests.

To ensure that survey effort was spread across the range of environmental gradients in the territories, we used the Biogeographic Ecosystem Classification (BEC) System to stratify the study area according to variants, which represent climatic and biophysical similarities (for more information about the BEC system, see Chapter 5 or Meidinger & Pojar 1991). We then used ArcGIS to create 10 random points within each variant. For each random point, we selected the overlapping forest cover polygon as a target location for a survey.

We navigated by boat, truck, and foot to each polygon using the GPS-enabled mapping application Avenza PDF on an Android tablet. Due to logistical challenges in the field (e.g. excessive snow or steep terrain) we did not survey every randomly selected polygon or every portion of each polygon. In total, we surveyed 403 ha across 28 polygons, ranging in size from 1 to 83 ha. We conducted multiple belt transects, which varied by location, within each polygon to cover as much area as was physically possible and safe. We determined the precise survey coverage by recording GPS tracks. During the transects, we assessed all potential LCC and recorded detailed information on tree morphological characteristics (Table 3.1) on a custom form within Avenza PDF. We used a Bluetooth GPS (EOS Arrow 100) to record coordinates and a combination of a Vertex Hypsometer, diameter tape, compass, and other field equipment to measure the tree and site attributes.

Data analysis

Our objective was to estimate the potential abundance of LCC across the territories by extrapolating the density of LCC occurrences found within our sample locations to our study area (green polygons in Figure 3.2). We divided the total number of LCCs located in the field, based on the LCC specifications, by the total surveyed area and then extrapolated this rate to our 69,863 ha study area. We did not assess variation in the spatial distribution of LCC (see Chapter 2 for an overview of this type of approach) due to reservations by our community research partners about explicitly showing or discussing these culturally sensitive locations. However, the random distribution of survey effort allowed us to roughly assume that the density calculated from our surveys characterizes abundance across the broader range of environmental gradients in the territories, though our inability to access some logistically challenging terrain may have introduced small biases.

This analysis helps illustrate the abundance of LCC as a broad category of traditional resources. But we also used the data from the interviews with carvers to further refine these estimates into subcategories representing more specific cultural uses: canoes, totem poles, and big houses. We used ArcGIS to match each LCC record with potential uses based on the trees' morphological characteristics. Many LCC trees with a set of specific characteristics can be used for multiple cultural uses (e.g. community canoes and certain big house logs such as large house beams both require large trees with few defects). This overlap in log specifications for different types of uses combined with differences in log specifications within an individual use category (e.g. small poles vs. large poles) makes it challenging to perfectly allocate trees to only a single use. Therefore, in addition to identifying this range of uses, we also allocated LCC trees to three aggregate categories with similar characteristics: Type 1, Type 2, and Type 3 (Table 3.2). We created a hierarchy for this allocation based on the rarity of the growth form, though the best use for a specific tree is inherently subjective and depends on individual perspectives among carvers and communities.

Table 3.2 Aggregate types of LCC based on similar size requirements. Redcedar trees meeting the definition of LCC (Table 3.1) are further refined based on log diameter and length thresholds.

Type	Cultural Use	Diameter	Length
Type 1	Community canoes, large totem poles, large big house logs	≥150 cm	12 m
Type 2	Chief canoe, medium totem poles, medium big house	120-149 cm	7 m
Type 3	Small totem poles, small big house logs	100-119 cm	5 m

To better understand whether the territories contain sufficient LCC for current and anticipated future First Nation use, we cross-referenced the abundance estimates of these LCC categories with estimates of the community and carver needs for these cultural products over time. In a previous assessment, the Nanwakolas Council estimated the LCC needs of their member Nations. This assessment report, in which we were not involved, was not published due to the culturally sensitive nature of the data. The analysis was based on discussions conducted by the Nanwakolas Council with the five communities and reflects the expected cultural needs over a 300-year planning horizon for totem poles, community canoes (larger structures), chief canoes (smaller structures), and different types of logs for building traditional big houses. It also includes assumptions to account for trees breaking during harvesting and defects such as rot and bark seams that are difficult to visually quantify while the tree is standing. Based on the listed specifications for log length and diameter, we allocated community canoes to Type 1 LCCs and chief canoes to Type 2 LCCs. The needs for totem poles were not refined according to size specifications or type, so we allocated 10% of the total needs across

Nations to Type 1 LCCs, 10% to Type 2 LCCs, and the remaining 80% to Type 3 LCCs. These allocations reflect our best estimate of relative wood use by carvers and their communities, but further engagement with these resource users is needed to understand whether this breakdown is appropriate. We allocated big house logs in the same manner, although these percentages more specifically reflect the listed diameter and length specifications in the assessment of cultural needs. To avoid sharing these culturally sensitive data outside the First Nation communities, we do not explicitly report the predictions of cultural needs for LCC. Instead, we combine these data with our predictions of abundance to quantify the extent to which the needs over time of the First Nations can be met in their territories by cross-referencing these two datasets (i.e. dividing the estimated needs by the estimated abundance).

Results

Applying knowledge from carver interviews

Although all interviewees provided some distinct perspectives and knowledge about cultural carving practices, responses from participants were highly consistent, allowing us to develop a field manual listing the tree characteristics suitable for LCC, including different categories of cultural uses of these trees (Table 3.1). These criteria generally capture the range of morphological tolerances for LCC expressed during the interviews, with specific thresholds approximately based on the average values reported by carvers. The interviewees also discussed many subtypes of carving products and uses of cedar, but in this study, we only focus on a few broad LCC categories for totem poles, dug-out canoes, and traditional housing logs. In addition to using these interviews to help address the question of, “what is Large Cultural Cedar?”, the Nanwakolas Council is using knowledge shared by carvers to support different components of their broader LCC Strategy (Table 3.3).

Table 3.3 Examples of traditional ecological knowledge and perspectives shared by carvers that informed the Nanwakolas Large Cultural Cedar (LCC) stewardship strategy.

Carver Knowledge	Example Quote	Connection to Nanwakolas LCC Strategy
Importance of LCC	"Cedar gave everything from clothing to transportation to housing"	Develop a comprehensive and intergenerational stewardship strategy
Declining supply of suitable LCC	"[There] probably [won't be LCC] even 20 years from now, the way they are going...scorching the earth for the last cedar they can find"	Immediately implement new policies to conserve LCC and develop a recruitment strategy
Cultural needs of the Nations over time	"If we had the option to re-build and re-create all the things that were taken away, and all the things that were burned and demolished and destroyed, we would need a whole lot of logs"	Cross reference LCC abundance estimates with the long-term needs of the Nations
Overlap in morphological characteristics between LCC and the trees targeted by the forest sector for timber	"Our perfect tree is their perfect tree as well"	Balance cultural and broad socio-economic interests by allowing some Type 2 and Type 3 LCCs to be harvested for commercial timber
Relationship between LCC trees and the surrounding forest	"Let's worry about protecting the land, then the trees will come with it"	Implement retention buffers around LCC during forestry operations and conserve important landscapes

Predicting the abundance of Large Cultural Cedar trees

Across 403 ha of surveyed forests in the study area, we frequently observed western redcedar trees. This is to be expected, given that we targeted forest cover polygons that explicitly listed this species as present. However, only 337 of the thousands of redcedar trees encountered met our criteria for LCC (0.84 LCCs / ha). While we recorded some site characteristics at each LCC, our analysis examining associations with specific biophysical characteristics is beyond the scope of this study. Within this broad LCC category, the density of trees suitable for specific cultural uses varies depending on the acceptable thresholds for each morphological characteristic. For example, while most LCC met the minimum specifications for smaller types of building materials for a big house, only a few trees contained larger logs suitable for main house beams. Similarly, the rarest type of LCCs encountered during the surveys were trees suitable for carving community canoes as only 2 out of the 337 LCC trees from the entire sampled area matched the criteria for this cultural use. Extrapolating our observed density of LCC from field transects to the entire study area shows very large differences in the predicted abundance of Type 1, 2, and 3 LCC (Figure 7).

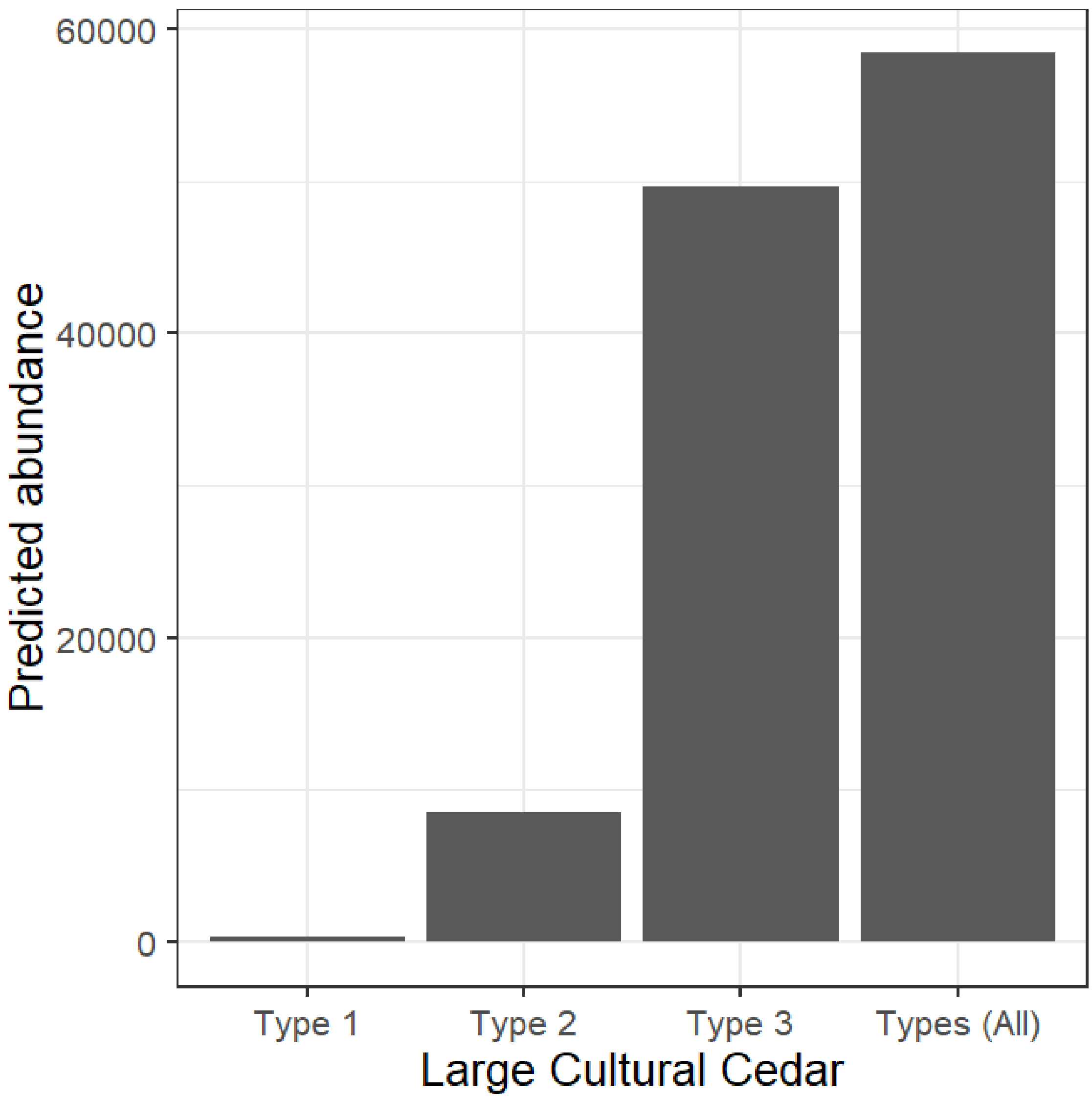

Figure 3.3 **Bar plot showing the predicted abundance of Large Cultural Cedar (LCC) within the study area. Categories on the x-axis represent different types of LCC (see morphological characteristics in Table 3.1 and 3.2).**

Developing community-based policies to support intergenerational stewardship

The staff and member First Nations of the Nanwakolas Council are currently developing a full suite of policies and guidelines that will contribute to an overall LCC strategy for the territories of the member First Nations. Based partly on the methods and findings from our study, the member First Nations are formally adopting, by way of a Declaration under traditional law, a new LCC operational protocol for forestry tenure holders (see Supporting Information). This operational protocol contains many new policies that tenure holders must adopt when applying for permits to harvest timber and that they must adhere to when carrying out forestry activities in the territories. These include requirements for pre-harvest assessments based on the LCC identification criteria (Table 3.1) as well as management rules such as maintaining no-harvest buffers around LCC trees and stands (see Supporting Information). The operational protocol also contains LCC retention targets based on the relationship between LCC abundance and community needs (Figure 3.4). For example, when cross-referencing abundance estimates with predicted First Nations needs for LCC over 300 years, our results highlight that some LCC categories are rarer than others. While predictions for Type 2 and 3 LCCs show the territories containing more trees than the anticipated needs, predictions for Type 1 LCCs show that community needs are dramatically above the current stock of LCCs (Figure 3.4). Although the member First Nations did not solely rely on this analysis to reach decisions about retention targets, the policies generally reflect the main quantitative results: Type 1 LCCs require 100% retention, Type 2 LCCs require 50% retention, and Type 3 LCCs require 25% retention (see Supporting Information).

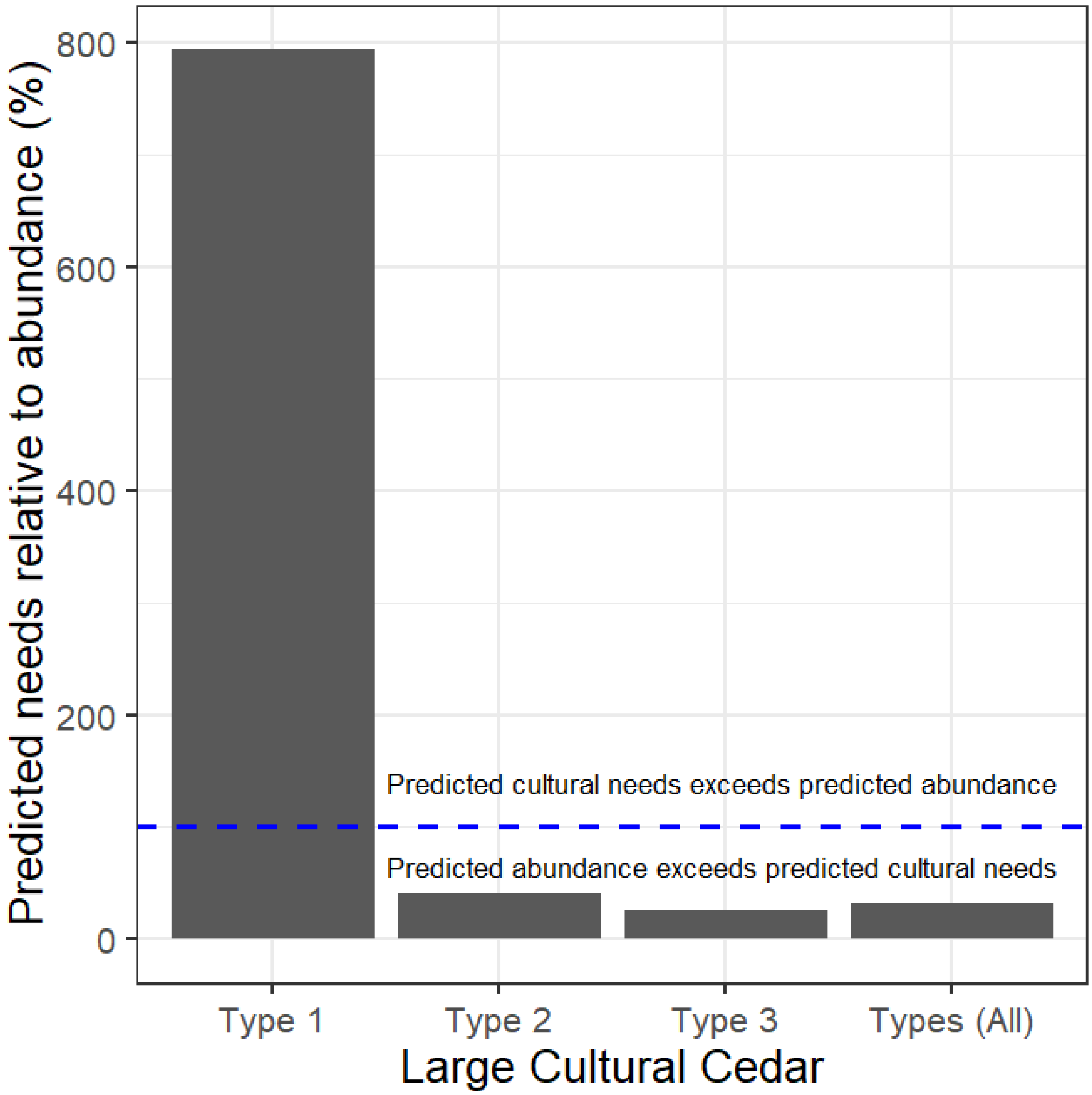

Figure 3.4 **Bar plot showing the extent to which Indigenous communities can meet their cultural needs over time for Large Cultural Cedar (LCC) within their territories. Values on the y-axis represent the predicted cultural needs for LCC for the next 300 years divided by the predicted abundance in the study area, expressed as a percentage. Categories on the x-axis represent different types of LCC (see morphological characteristics in Table 3.1 and 3.2). Percentages on the y-axis above 100% (blue dashed line) show where the predicted cultural needs for LCC over the next 300 years exceed the predicted abundance of LCC across the territories. In contrast, percentages under 100% show where the predicted abundance of LCC trees currently exceed the predicted cultural needs.**

Discussion

Trees suitable for specific types of Indigenous carving practices are rare

Large Cultural Cedar trees are an important cultural resource that our results suggest are rare within the Indigenous territories of our study area. When traditional ecological knowledge of Indigenous carvers is used to refine this broad LCC category based on distinct uses, our abundance estimates reveal that certain types of trees, such as those associated with the specifications for carving canoes for community use, are nearly extirpated from the land base. For instance, only 2 of 337 LCC's within our sampled locations and an estimated 337 LCCs within the study area meet the criteria for these types of trees—and it is uncertain how many trees will recruit into this category over the next 300 years. For such long-lived and rare growth forms, the predicted abundance of trees will not meet the cultural needs of carvers and their communities into the future.

Our predictions of LCC abundance generally align with knowledge shared during our interviews, which highlighted sustainability concerns including intergenerational access to this resource. As one carver exclaimed, "It is coming to that point where the logs are getting smaller, they are getting knottier, twisty looking…that's all there is left!" These predictions are also consistent with other studies of LCC in coastal BC. For example, although not explicitly reported in Chapter 2, the predicted density for LCC derived from that study's field validation data is 1.25 trees/hectare, slightly higher than the 0.84 trees/hectare reported here. The lower density observed in the present study is consistent with the longer and more intense history of industrial exploitation in this more southern region. Sutherland et al. (2016), who examined cultural ecosystem services in a generally wetter region on the west coast of Vancouver Island that is nearby our Chapter 3 study area, did not report density estimates in their research, but generally found that monumental cedar trees are more common in riparian ecosystems than nearby upland forests. These studies use different identification methods for LCC/monumental cedar and focus on different parts of BCs coast—regions that may contain proportionally different levels of old growth cedar stands across the landscape compared to the Nanwakolas member Nations' territories. Therefore, despite similar results within potentially suitable old growth forests, the overall extent of stands where

LCC are likely to occur is highly variable across the Na̲nwak̲olas Nations' territories and coastal BC more broadly. Such differences arise due to heterogeneity in the natural distribution of ecosystem types in the coastal temperate rainforest (Meidinger & Pojar 1991; Wolf et al. 1995) coupled with legacies of intense harvest history in particular landscapes (Chapter 4).

Like most predictions of species abundance, our analysis contains inherent uncertainties and unknowns that are difficult to quantify. For example, due to logistical challenges and limited resources, we were not able to assess every randomly selected polygon from our initial survey design, so some environmental gradients may not be fully captured in our dataset relative to their distribution across the study area. Data issues such as inaccurate attributes within forest cover maps also result in uncertainties about the status of the current land base. Temporal factors, such as impacts from logging (e.g. Chapter 4) and climate change (e.g. Hennon et al. 2012), make projections over centuries into the future even more uncertain. Moreover, uncertainties also extend to projections of community needs over time for cultural resources because these are partly contingent on the continuity of traditional practices and assumptions concerning population growth. Though maintaining species with such broad cultural connections as LCC will likely be important to Indigenous communities regardless of specific quantitative use.

An industrial forestry paradigm hinders stewardship of long-lived cultural resources

The depletion of LCC in our study area reflects a global trend of diminishing supplies of large old trees and the ecosystems that support them, including many species that are culturally important (Chapter 4; Albert & Schoen, 2013; Lindenmayer, Laurance, & Franklin, 2012; Moga et al., 2016; Schulze, Grogan, Landis, & Vidal, 2008). One way to better understand this type of environmental change is to use the approach outlined in Chapter 2, which compares predictions of the distribution of monumental cedar based on field surveys to predictions based on historical occurrence data such as archaeological records of traditional harvest locations. But even without such retrospective assessments, it is obvious that an industrial forestry paradigm focused on prioritizing timber economic values will not generate the types of stands required to provide cultural resources that are dependent on old growth forests.

This is especially the case for culturally important growth forms such as LCC trees that require several centuries of growth to achieve suitable sizes. Many carvers in our study also suggested that specific biophysical conditions such as shade are needed to produce dense wood grain that is free of large knots. Clearcut openings and subsequent young forests managed on short rotations are common silvicultural practices in managed forests in coastal BC (B.C. Ministry of Forests 2010) and these types of morphological characteristics are unlikely to develop under those conditions. Relative to the time scales of conventional industrial harvest rotations (Binkley 1987; Mathey et al. 2009), these types of old trees cannot meaningfully be considered a renewable resource. The failure to adequately account for the importance of cultural resources often stems from thinking about species and the environment as homogenous, substitutable commodities, instead of considering their diverse uses and the broader ecosystem and cultural services they provide that may not be reflected in economic markets (Turner et al. 2009; Chan et al. 2012; Blicharska & Mikusiński 2014).

Traditional ecological knowledge helps define and interpret biocultural diversity

The loss of many of the world's Indigenous knowledge systems creates a gap during conservation planning because of its important role in defining and interpreting cultural resources (Berkes et al. 1994; Davis 2010). Our interviews with traditional carvers of cedar revealed a broad range of conditions that make a specific tree suitable for a specific purpose. As one carver said, "there isn't just one family member of cedar, there's six or seven family members". Accurately representing the nuances shared by carvers is difficult and codifying this knowledge into quantitative thresholds to support the identification of LCC was necessarily reductionist and simplistic. This type of interpretation did not account for the rich qualitative context and relationships between biophysical processes and cultural practices that will be explored in forthcoming studies. The standardized approach to identifying LCC in the field manual, however, was necessary to support efficient and effective LCC data collection by community members with varying levels of field experience and traditional knowledge. These quantitative thresholds were also useful in representing essential elements of TEK in a formal resource planning context because they enabled the Na̲nwak̲olas Council to develop specific LCC survey and management protocols based on categories for canoes, totem poles, and big house logs.

Our findings support the idea that finer resolution assessments of diversity can inform conservation policies that are more directly connected to local ecosystems and their services. Schindler et al. (2010), for instance, show that accounting for the temporal and spatial heterogeneity of sockeye salmon in Bristol Bay, Alaska helps predict resilience in salmon populations, which in turn supports a more economically viable fishery. Similarly, in forests, stands with more diverse tree species and structures, and management strategies to promote these characteristics, are generally more resilient to impacts from mountain pine beetle outbreaks and are better able to provide timber and carbon sequestration services over the long term (Dymond et al. 2014; Dhar et al. 2016). This nuanced understanding of system diversity is especially salient when addressing rare biocultural resources and their societal connections. For instance, the inability to locate many tree forms suitable for carving community canoes during our field surveys was of deep concern to our research partners because this cultural resource was key to Indigenous people travelling along North America's coast over millennia and remains intimately tied to traditional and contemporary Indigenous culture.

Hence developing stewardship strategies based solely on aggregate categories, such as redcedar as a species, large redcedar trees, or even LCC as a broad category of traditional use would not adequately focus on, identify, and conserve the specific wood forms that are vital for maintaining cultural traditions and connections across generations. If, for instance, the Nanwakolas Council had simply built their stewardship strategy on abundance estimates across all LCC categories, which show abundance exceeding needs, then the rarest types of LCC might still be available for commercial timber harvesting. Instead, they developed policies based on a refined understanding of cultural uses, which supported the decision to conserve all Type 1 LCCs for First Nation cultural use (i.e. 100% retention target in the LCC Operational Protocol Agreement; Supporting Information).

We therefore echo other scholars who highlight the importance of combining TEK with western science when taxa are associated with cultural uses and practices by Indigenous groups. Whether the cultural resource is medicinal plants in the Amazon (Pesek et al. 2010), marine invertebrates in Alaska (Salomon et al. 2007), large mammals in Canada (O'Flaherty et al. 2008; Polfus et al. 2014), cultural processes such as the use of fire (Lake et al. 2017), or cultural trees around the world (Turner et al. 2009), TEK has proven valuable in understanding species and management systems that can account for local contexts. When thinking about the management of long-lived species with rapidly shifting baseline conditions, such as LCC, the knowledge base underpinning stewardship strategies should have a temporal resolution that reflects the life history of the resource. As one carver stated in an interview, “I know this tree was standing somewhere 500 years ago and here I am carving [it], and I always think to myself, this pole was already there 500 years ago… and I just sort of shape it out, give it its final shape.”

Community-based research supports applied conservation goals

This study has theoretical and applied implications for community-based conservation. Our overall approach for understanding and predicting the abundance of traditional resources to support conservation can be implemented in a diverse range of social and ecological contexts. The elements of this approach span many different aspects of the research process that includes substantial work in communities partnering and building relationships between researchers and Indigenous groups, jointly developing research questions, and understanding culturally important resources through interviews with knowledge holders. It also includes work in the field carrying out surveys with community members and, finally, analytical work that addresses the anticipated cultural needs over time of communities and their resource users. This collaboration and co-production of science also emphasized respectful data sharing and capacity building within the communities. Many of these elements have been used or recommended in other studies involving community-based research with Indigenous groups (Huntington et al. 2011; Chan et al. 2012; Wilder et al. 2016; Salomon et al. 2018), but rarely are they all blended into a single project.

Our research is also distinct in that the key findings, which emerged through gathering and applying TEK, are being directly used by Indigenous communities to develop new forestry policies within the study area (see Supporting Information). Such applied uses of TEK are broadly relevant to scholarship on this topic, including translational ecology more broadly (Enquist et al. 2017), because of the paucity of concrete examples where this epistemic system is put into practice and policy. Developing effective applied policies for conservation and natural resource management based on an academic study is much more likely when Indigenous communities and their knowledge holders are full partners in all aspects of research collaborations.

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Supporting Information

The following version of the Nanwakolas Operational Protocol for Large Cultural Cedar is a draft and, therefore, all sections are subject to potential changes. Upon completion, the final version will be available on the Nanwakolas Council website (www.nanwakolas.com).

Draft Nanwakolas Operational Protocol for Large Cultural Cedar

The lands and resources of our territories have sustained our culture, way of life, spirituality, economy and society for countless generations. From child to Elder, a vital part of who we are as peoples is identified and lived on the land. This includes our forests and trees, which have always provided our peoples with bounty and well-being.

Maintaining the health of our forests and trees is a responsibility and trust that each generation of our peoples carries to those who came before, and those who will come after. Through our laws and protocols, we care for the land, and ensure it is able to maintain us for all time to come.

Today, as our ancestors have always done, we continue to apply our laws and protocols on the land. We do this guided by the principles of sustainability and balance that are integral to our cultures and teachings. We also do so as part of ensuring respect, recognition and implementation of our title and rights, and as part of upholding the standards of the *United Nations Declaration on the Rights of Indigenous Peoples*.

This Protocol is one aspect of our approach to stewarding our essential resource of Large Cultural Cedar (or LCC) adopted under our laws and jurisdiction. Specifically, this Protocol provides detailed operational guidance and standards to be followed when forestry activity may be planned or proceeding in areas where there are LCC.

To be clear, adhering to this Protocol does not mean that there is any consent by us to any forestry approvals, decisions, or activities in our territories. Decisions regarding consent are dealt with by other of our laws and protocols, as well as agreements and arrangements with the Crown and companies. No forestry activity in our territories should take place without first securing that consent and ensuring that our title and rights are fully respected. However, where forestry activity is proceeding, we expect all companies in our territories to adhere to this Protocol.

Background

1. **Importance of LCC's**: Cedar is a central and important cultural, social, spiritual, and economic resource to member First Nations of the Na̱nwaḵolas Council and has a prominent role across diverse aspects of traditional and contemporary life. Although western redcedar (*Thuja plicata*) and yellow cedar (*Xanthocyparis nootkatensis*, also known as cypress) is relatively abundant across certain ecosystems in these Nations' territories, large, high quality trees suitable, including for carving and building canoes, totem poles, and big houses, are rare.

2. **Scarcity**: Large Cultural Cedar is a scarce resource in managed forests because of the unique developmental pathways required to grow these trees and the long time periods (typically > 300 years) required for their development. LCC are also scarce because colonialism, unchecked resource extraction, infringements of our title and rights, and failure to seek or secure our consent, has depleted our forest resources, including LCC. Despite this, because these types of trees also provide the most profitable timber in coastal forests, they are an on-going target of the forestry sector.

3. **Strategy for an Intergenerational Supply of LCC's**: To ensure an intergenerational supply of LCC we are applying our laws and protocols through

a contemporary LCC stewardship strategy. The following operational protocols are one component of this overall strategy.

LCC Surveys

4. **Requirement for LCC Surveys**: Large Cultural Cedar surveys are required:

    a. When requested by one of our First Nations; or

    b. Where high potential LCC (i.e. any redcedar or yellow cedar tree greater than 100cm diameter at breast height) are identified during operational planning, unless the First Nations consent that a LCC survey is not required.

5. **Conduct of LCC Surveys**: First Nation members that have completed the Nanwakolas LCC training course will conduct all LCC surveys. If the First Nation does not have one of these surveyors available to conduct an LCC survey, the Nation(s) and the forestry licensee will discuss using an alternate qualified individual. To promote logistical efficiencies, LCC surveys may overlap in timing with a preliminary field reconnaissance or an archaeological impact assessment, although conducting these two types of surveys concurrently is optional at the discretion of the First Nations.

6. **Defining and Identifying LCC's**: Criteria for identifying LCCs are stated in the Nanwakolas Council Large Cultural Cedar Identification Manual. First Nation knowledge-keepers were interviewed to understand tree characteristics that make a LCC suitable for different traditional uses.

7. **Carrying Out of Surveys**: LCC Surveys will be carried out using survey methods consistent with the Archaeological and CMT Inventory Handbook developed by the BC Resource Inventory Standards Committee (RISC). This approach will involve systematic transects with 100% survey coverage of all stands containing potential LCC within development areas as well as in adjacent forests where LCC management may influence the design of cutblocks and roads. In practice, LCC surveys will need to occur in most old growth forests where redcedar or yellow cedar is present.

LCC Operational Management

8. **Development of Retention Requirements**: Stand level retention requirements for LCCs have been developed, partly based on predictions of the abundance of different types of LCC trees (based on cultural use/purpose) across Nanwakolas First Nations territories relative to predicted First Nation needs for cultural logs over the next 300 years.

9. **Definition of Retention**: The word "retention" is used to describe both when LCC trees are retained during harvesting as well as when LCC trees are harvested for current cultural use by First Nation(s). There is inherent uncertainty in these numbers and many factors influence how these estimates relate to retention targets. The Nanwakolas Council is conducting ongoing monitoring and research about LCC. The findings from this work will support potential refinements to the retention requirements in this Protocol over time.

   The minimum retention requirements are based on the standard that rarer types of LCC are associated with higher levels of protection.

10. **Minimum Retention Requirements**: There are three different minimum retention requirements for LCC. Trees meeting the broader definition of LCC (see 6) are further refined into Type 1, Type 2, or Type 3 categories based on log diameter and length thresholds.

Table 3.4 Overview of the minimum retention targets for LCC. Rarer types of LCC are associated with higher levels of protection. Although carvers consider a broad range of tree characteristics when determining the suitability of a tree for a specific cultural use, these general Type 1, Type 2, and Type 3 categories are based only on log diameter and length thresholds within trees meeting the definition of LCC (see 6).

Type	Cultural Use	Status	Diameter	Length	Retention
Type 1	Community canoes, large totem poles, large big house logs	Very Rare	≥150 cm	12 m	100%
Type 2	Chief canoe, medium totem poles, medium big house	Rare	120-149 cm	7 m	50%
Type 3	Small totem poles, small big house logs	Moderately Rare	100-119 cm	5 m	25%

11. **Calculating the Number of Trees to be Retained**: In applying the minimum retention requirements in section 10, the following principles will be used to calculate the number of trees to retain:

   a. The retention requirements represent the minimum percentage of LCCs that must be retained or provided to First Nation(s) for current cultural use within each discrete development area (i.e. individual cutblock or road right of way).

   b. If present in the development area, a minimum of one tree of each LCC type must be retained.

   c. LCC retention percentages should be rounded up to the next highest whole number. For example, if 5 Type 3 LCCs are present in a development area, then 2 of these trees would need to be retained.

   d. Similar to the accounting rules used for Wildlife Tree Retention Areas, LCCs contribute towards meeting the retention targets if they are either in the development area, along the development area boundary, or within one tree length of the boundary (see 15 for the landscape context for applying targets).

   e. To ensure that LCC retained within development areas are accessible and do not conflict with other stewardship objectives of First Nations, individual LCC trees only count when calculating the number of trees that can be harvested for commercial timber if they are not associated with the following areas or features:

      i. Operationally inaccessible areas (based on professional and/or First Nation judgment in the field);

      ii. Culturally Modified Trees;

      iii. Bear dens; or

iv. Riparian management zones or riparian reserve zones (including buffers associated with High Value Fish Habitat and Non-High Value Fish Habitat in the Great Bear Rainforest Order).

For example, if 4 Type 2 LCCs are present in a development area, one of which is a culturally modified tree and another that is located in a riparian reserve zone, then only 1 LCC (50% of the 2 applicable LCCs) can be harvested for commercial timber. Note that LCCs may still be harvested in RMZs, but these trees are not accounted for when calculating the percentages of trees that can be harvested for commercial timber.

12. **Landscape Context for Application Minimum Retention Requirements**: These minimum retention requirements apply to all development areas within the First Nations territories. Although the specific number of LCCs available for cultural use in a watershed will vary depending on the broader matrix of conservation areas vs. managed forest, the specific landscape context will not influence retention targets in development areas unless the following occurs:

    a. The applicable First Nation(s) agree to consider adjusting LCC targets based on the landscape context; and

    b. A complete LCC survey of the applicable watershed is carried out (the spatial extent of the applicable watershed area should be determined based on First Nation(s) guidance).

13. **Stewardship and Recruitment of LCC's**: To ensure that LCC trees persist in stands over time, the following management strategies will be used to mitigate windthrow risk, recruit future LCC, and maintain the ecological conditions around the LCC:

    a. Buffer LCC with a minimum 1 tree length reserve zone and a ½ tree length management zone (based on the height of the LCC);

    b. Apply buffer from the polygon created by connecting trees when at least 3 LCCs or CMTs are within 30m or each other; and

c. Retain previously identified LCC trees and retention areas (i.e. through sequential harvest rotations).

14. **Making Retention Decisions**: This Protocol allows for some harvesting of Type 2 and Type 3 LCC trees to address commercial timber objectives. Applying the retention targets thus creates choices around retaining versus harvesting specific trees. In general, many of these decisions can be made in the field based on operational logistics (e.g. safety, access, etc.) by forestry engineers and planners, but First Nation stewardship workers and their First Nation(s), via Information Sharing, may choose to prioritize specific LCCs for either long-term retention or current cultural use.

15. **First Nation Access to LCC Logs**: This Protocol is intended to support current and future First Nation tree use. Therefore, all, some, or none of the LCC associated with the retention targets may be allocated towards meeting current cultural needs of the First Nation(s). First Nations are currently developing cultural wood programs that will provide more details about the flow of LCC to their communities and carvers. In the interim, forestry licensees and First Nation(s) will discuss access to LCC in development areas as part of the Information Sharing Protocol.

16. **Monitoring**: To ensure that implementation of this LCC protocol is consistent with First Nations' objectives for LCC, a robust monitoring framework is being developed by Na̲nwak̲olas. This monitoring will assess both compliance and effectiveness of LCC management.

17. **Review**: This Protocol will be reviewed periodically, and changes to it may be made by the First Nations. In conducting reviews, the First Nations will consider information regarding the state of the forests in the territories, feedback from communities, members, and companies, as well as progress, challenges, and topics in the implementation of the Protocol. If the Protocol is changed, updated versions will be made public, including to all companies.

Chapter 4. Logging down the value chain: Decline in the productivity and accessibility of forests harvested over a half century

Abstract

Many industrial economic models of natural resource management incentivize the sequential harvesting of resources based on profitability, disproportionately targeting the higher value elements of the environment. In fisheries, this issue is highlighted by research that identifies and investigates the problem of “fishing down the food chain”. Harvesting that focuses on highgrading the most productive and accessible environmental gradients is also thought to occur in the forestry sector, though such a paradigm is incongruent with a stewardship ethic, entrenched in the forestry literature, that aims to maintain and enhance forest condition over time. How have these conflicting objectives arising from profit-focused economics and stewardship ethics affected patterns of forest harvesting over time, and how have conservation-oriented policies influenced harvesting patterns? We use harvest data over a 47-year period as well as aggregated time series data that span over a century on the central coast of BC, Canada to assess temporal changes in how logging is distributed among various classes of site productivity and terrain accessibility. We show a distinct trend over time towards decreasing site productivity of logged forests and some evidence of a decline in the accessibility of the forests being logged. Policy changes enacted in BC in the mid-1990s appear to have strongly affected these trends, illustrating both a tendency to harvest down the value chain when choices are unconstrained and the potential of policy choices to impose a greater stewardship ethic on harvesting behavior. Logging down the value chain has led to a reduced state of forest value on the modern landscape, with implications for communities that rely on, and are assuming increased management authority over, local forests.

Introduction

Humans are drastically affecting ecosystems and stocks of resources worldwide in both aquatic and terrestrial systems. For example, in the marine environment much attention has been given to the issue of “fishing down the food chain”—data on fisheries landings over time show that, having exhausted stocks of preferred, larger, high trophic level species, harvesting has targeted increasingly lower trophic levels (Pauly et al. 1998). Preferential harvesting of large land animals for human consumption is also imperiling megafauna in ecosystems around the world (Ripple et al. 2019). Similarly, scarcity is causing global energy production to undergo a transition towards higher cost methods of extraction and alternative sources after depleting conventional sources with higher value and lower production costs (Jaccard 2009). The systematic erosion of high value components of the environment raises concerns about sustainability and intergenerational access to natural resources. Disproportionately altering one portion of an environmental gradient can also cause cascading effects across ecosystems (Ripple et al. 2014) and can affect ecosystem services and functions, which are maintained through a portfolio of diverse ecological structures and functions (Schindler et al. 2010). Historical highgrading of large old trees, for instance, is thought to have restructured forest demography, gap dynamics, and fire behavior in dry conifer ecosystems such as ponderosa pine forests (Kaufmann et al. 2000). These types of ecological changes can also catalyze shifts in natural resource policy—a prominent example being the overharvesting of old growth Douglas-fir ecosystems in the Pacific Northwest of the United States resulting in the loss of habitat for the threatened northern spotted owl and the subsequent development of the conservation-based Northwest Forest Plan (Franklin & Norman Johnson 2014).

Many factors influence harvesting patterns of resources. These can reflect complex land use histories based on societal views shifting through time that alter ecosystem services and convert regional land-cover (Rhemtulla et al. 2007, 2009). In forestry, the traditional economic paradigm associated with the industrial forest sector prioritizes timber stands that maximize profits, usually linked to accessible areas, such as flat terrain or short transportation distances to log markets, as well as productive locations that promote fast tree growth and stands with large trees. Analyses of past harvesting activity in Alaska and coastal BC demonstrate a forest sector preference for logging valley bottoms—a classic example of accessible and productive high-value forests (Pearson 2010; Albert & Schoen 2013). There is also evidence that valuable, rare tree species such as mahogany in tropical rainforests (Schulze et al. 2008) and large old, high quality Douglas-fir and western redcedar in temperate rainforests (Kaufmann et al. 2000; Green 2007; Chapter 2) have been disproportionately logged over recent decades. Given that trees can grow for centuries in some biomes (Antos et al. 2016), there is a large opportunity cost of waiting to log stands at rates that allow such large, old trees to persist across the landscape. This partly stems from the influence of economic discount rates over such long time horizons. Discount rates have the reverse effect of compound interest, creating a situation where the assessed economic value of future benefits and costs is lower than the assessed economic value of otherwise equivalent present benefits and costs. The downward pressure on net present value resulting from discount rates can affect harvesting trade-offs related to letting trees grow and gain value in the forest versus reallocating the profits from harvested trees into other types of capital with greater economic returns on investment (Brukas et al. 2001).

In contrast, and sometimes in conflict, with these traditional economic models of timber production used by the industrial forest sector, forestry and foresters also have a long tradition of stewardship ethics (Leopold 1949; Smith & Kelty 2018). A broad range of factors, including tenure rights, drive behavior and outcomes in resource management (Ostrom & Nagendra 2006; Benner et al. 2014), and hence the views espoused by prominent historical figures in forestry, such as Aldo Leopold, might be associated with distinct social-ecological conditions. For example, localized rights and responsibilities through small woodlots and community forests are associated with certain types of local benefits, but many scholars caution that no single form of tenure will be a panacea for addressing the many dimensions of sustainability (Charnley & Poe 2007; Ostrom et al. 2007; Benner et al. 2014).

Although the tradition of forest stewardship ethics has prioritized ecological, social, and economic objectives to varying degrees over time, it generally emphasizes that different types of forest values, including those based on timber, should be maintained on the landscape to provide benefits across generations (Hammond 1992; McQuillan 1993; Bengston 1994); the net effects of management should not decrease the average value of the forest estate. One important idea that arises from this ethic is that, over time, forest operations should "log the profile" of the forest—harvesting ecosystem types, species mixes, and productivity classes roughly in proportion to their presence on the landscape to maintain timber and non-timber values and associated ecosystem services. This is contrasted with the notion of "highgrading", where the higher valued components of a stand or landscape are targeted preferentially, which would decrease the residual value over time. Many of the economic incentives for modern large corporate forest entities, including the form of tenure rights through which they hold interests in the forest, often act against practicing a stewardship ethic for socioeconomic and ecological values (Charnley & Poe 2007; Larson et al. 2010; Benner et al. 2014).

Changes in forest policy can also influence the decision making of resource managers as they navigate trade-offs between these types of conflicting objectives. Often catalyzed by shifting public values about stewardship, economics, and the allocation of benefits from harvesting, policy changes over recent decades in jurisdictions around the world have addressed many issues that affect the spatial and temporal patterns of harvesting (Fenger 1996; Song et al. 2004; Ostrom & Nagendra 2006; Power 2006). Prominent examples of such issues in the Pacific Northwest include the size and distribution of cutblocks (Fenger 1996; Franklin & Norman Johnson 2014), whether clearcut logging should be replaced by alternatives such as retention silvicultural systems (Lindenmayer et al. 2012), and protection for ecological features such as riparian areas, unstable steep slopes, and wildlife habitat, particularly for threatened and endangered species (DellaSala et al. 2015).

Each of the paradigms described above—short-term profits vs. stewardship-driven—has clear implications for how logging will be distributed over time (Figure 4.1). If the industrial economic paradigm of maximizing short-term profits is primarily dictating timber-harvesting patterns, logging should initially focus on accessible, high-value stands and be allocated over time to sequentially less valuable, productive and accessible stands. Consequently, if the highest value components of the environment are targeted first, over time the areas being logged should increasingly represent the productivity and accessibility of the overall landscape (i.e. lower values) until the time when average conditions are reached. Alternatively, if the stewardship paradigm of maintaining forest condition is primarily dictating timber-harvesting patterns, over time logging will be occurring in stands that are broadly representative of the productivity and accessibility of the overall landscape and that do not trend downwards with respect to these variables.

We investigate the influence of economics, stewardship, and government policy on harvesting behavior by assessing the spatial distribution of logging over a half century on the central coast of British Columbia (BC), Canada in relation to environmental gradients associated with productivity and accessibility. Specifically, we assess which paradigm best predicts spatial harvesting patterns, and whether this has changed over time in response to policy changes. Although excellent research has examined historical harvesting patterns in forests (e.g. Albert and Schoen, 2013; Pearson, 2010), to our knowledge few studies have analyzed time series data at the scale necessary to describe annual trends that span different policy regimes of environmental variables related to forest productivity and accessibility.

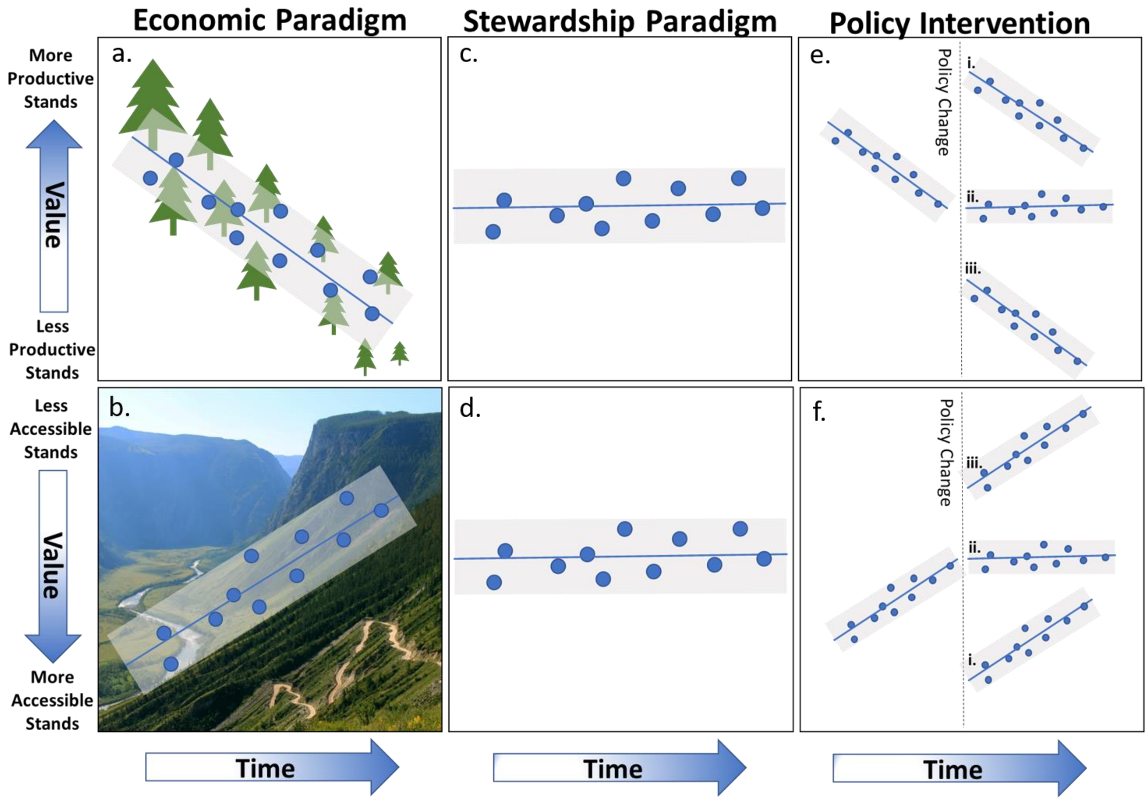

Figure 4.1 **Conceptual hypotheses related to the environmental gradients of harvested areas over time. A traditional economic paradigm associated with industrial forestry would suggest that the most accessible and productive components of the landscape are sequentially targeted (a, b), whereas a stewardship paradigm would suggest that the types of forests being logged are similar over time (c, d). A policy intervention focused on increasing stewardship might cause the trend line to level-off if harvesting becomes more representative of the productivity and accessibility of the overall land base (e.ii, f.ii), shift towards lower average variable values if productive and accessible parts of the landscape (e.g. riparian areas) become unavailable to harvest (e.iii, f.iii), or shift towards higher average variable values if unproductive and inaccessible parts of the landscape (e.g. steep, unstable terrain) become unavailable to harvest (e.i, f.i).**

Methods

Study Area

We focus our analysis on a portion of the central coast of BC (Figure 4.2). This region encompasses diverse landscapes that are home to iconic wildlife such as grizzly bears (*Ursus arctos* horribilis), productive salmon runs, and large, long-lived trees such as western redcedar (*Thuja plicata* Donn ex D. Don; Daniels et al., 1995; Hocking and Reynolds, 2011; McAllister et al., 1997; Chapter 2 and 3). Due to the wet and heterogenous biophysical environment of the central coast, the forests vary greatly, from small stature, low productivity hyper-maritime bog-forests to tall, massive, highly productive floodplain forests (Meidinger & Pojar 1991; Green & Klinka 1994). Indigenous populations have occupied the area continuously for over 10,000 years (Cannon 2000; McLaren et al. 2014), including the Heiltsuk First Nation, whose traditional territories cover much of our study area (http://www.heiltsuknation.ca/about-2/territory/). Our spatial analysis focuses only on the subset of this area that is forested (i.e. the green areas in Figure 4.2, representing 855,133 ha).

The central coast of BC forms a major part of the global distribution of coastal temperate rainforest and is often referred to as the Great Bear Rainforest (GBR; Price et al., 2009). While there is a long history Indigenous use of the land, the level of forest harvesting has increased dramatically over the past one and a half centuries due to the growth of industrial logging, mostly by non-Indigenous corporate entities, focusing on species such as western redcedar, Sitka spruce (*Picea sitchensis*), western hemlock (*Tsuga heterophyla*), amablis fir (*Abies amabilis*), and Douglas-fir (*Pseudotsuga menziesii*) in the more southern portion of the region (Rajala 2006). In the 1990s and 2000s, these forestry activities were the catalyst for internationally significant land-use disputes, leading to new agreements, regulations and policies to address sustainability concerns (Fenger 1996; Hayter 2003; Price et al. 2009).

Because roughly 95% of BC's forests occur on public land, with rights and responsibilities conferred to private interests through a tenure system, provincial policies have a strong influence on behaviour in the forest industry. A major policy shift towards stronger stewardship objectives occurred in 1995 with the introduction of the Forest Practices Code (FPC; Forest Practices Code Act, 1994). The highly prescriptive regulatory environment of the FPC influenced many dimensions of forest management that previously had received little attention in regulations, such as new rules for cutblock configuration and size, adjacency criteria, and increased protection around streams navigable by salmonid populations. Subsequently, in the GBR specifically, a new regime of ecosystem-based management (EBM) was implemented in stages through various agreements and regulations between 2000-2016 (Central and North Coast Order, 2006, 2009, 2016). These changes led to new objectives for conservation and cultural heritage at the stand and landscape level that constrained logging in many parts of the GBR, including over entire watersheds. Negotiations around EBM also catalyzed a new working arrangement between the forest industry and environmental organizations and a new government-to-government relationship between Indigenous and provincial governments (Price et al. 2009).

Figure 4.2 **Map of study area (green) and forest harvesting within the central coast of British Columbia, Canada. Due to extensive landscapes of relatively unproductive soil (e.g. outer islands in the study area), forests that are economically feasible to log represent a much smaller portion of the land base compared to forests in more southern parts of the province.**

Forest harvest datasets

To determine the spatial extent of logging in each year between 1970-2016, we accessed the GIS layer Harvested Areas of BC from the BC Provincial government (Harvested Areas of BC, 2018). This layer combines harvest data from government sources, reporting from forestry licensees, and a gap analysis to estimate unreported harvesting based on remote sensing. The resulting spatial polygons represent the outer boundary of each cutblock, and typically encompass in-stand reserved area (provincial policies typically require a minimum of 7% stand level retention) if forest tenure holders used silvicultural systems other than clearcutting. We used polygons representing the area harvested each year—based on the Harvest Completion Date field. We noticed some missing spatial cutblock information in the 2018 dataset (verified using satellite imagery), which had appeared in a Harvest Areas of BC file that we previously downloaded in 2015, so we merged these two spatial datasets to create a final layer for analysis.

To understand harvesting over a longer time period between 1860-1969, we also analyzed historical cutblock data using an earlier analysis of historical orthophotos in the study area by Pearson et al. (2009), which categorized forest disturbances based on several categories, including logging (see Pearson et al. 2009 for more details). Because this layer did not list information about the harvest completion date, we instead used current stand age as a proxy to estimate harvest date. We obtained current stand age information from overlapping forest cover data found in the Vegetation Resource Inventory (https://www.for.gov.bc.ca/hts/vri/), a spatial data layer that characterizes stands according to various attributes of landcover and vegetation based on orthophoto interpretation. We categorized harvesting according to the data source (historical orthophoto covering 1860-1969 vs. modern provincial dataset covering 1970-2016) and the type of forest being harvested (second growth forests harvested for the second time vs. old growth forests harvested for the first time). We then combined these categories to create three datasets for further analysis: 1) cutblocks representing historical old growth logging, 2) cutblocks representing modern old growth logging, and 3) cutblocks representing modern second growth logging. We determined the locations of second growth logging by assessing areas of overlap between the historical and modern logging datasets and determined old growth logging by areas that did not overlap. All three cutblock datasets were then converted to raster files for further analysis (25m grid cells, BC Albers projection). Our analyses of year-to-year trends focus only on the modern old growth logging dataset because of the higher temporal uncertainty in the historical logging dataset and the small sample size in the modern second growth logging dataset. We therefore only report results for these latter two datasets aggregated across all applicable years.

Data Analysis

We assessed factors associated with logging patterns to examine the extent to which harvesting is targeting specific environmental gradients. We first calculated the mean values of three variables within harvested areas that are associated with site productivity and terrain accessibility: Slope, Site Index, Distance to Large Stream; Table 4.1), using the Zonal Statistics tool in ArcGIS 10.5 (ESRI 2018, Redlands CA). To understand trends at the scale of individual cutblocks, we derived these values for each of our logging datasets by setting the zone of analysis to the year the area was harvested, resulting in mean variable values aggregated across raster cells for each discrete area logged (i.e. cutblock) in each year. We then calculated the mean variable values of all logging in each year, weighted by area using the *plyr* package (Wickham, 2011) in the statistical program R (R Core Team 2018) and created graphs using the *ggplot* package. Finally, for each variable we used a linear regression model to examine statistical differences between the mean variable values for areas overlapping old growth logging in the pre-FPC period (1970-1994) and the mean variable values for areas overlapping old growth logging in the post-FPC period (1995-2016).

Table 4.1 Environmental variables related to site productivity and terrain accessibility used as predictors in our model. In general, more productive and accessible forests are associated with more profitable harvesting.

Variable Name	Source	Description
Slope	TRIM[a]	Slope (%) is related to terrain accessibility. We interpret increasing values as decreasing accessibility.
Site Index	VRI[b]	Site Index is a measure of site productivity. Site Index represents the potential height (m) of dominant trees at age 50, measured from breast height. We interpret increasing values as increasing site productivity.
Distance to Large Stream	MFLNRO[c]	Distance to Large Stream is related to site productivity and accessibility. Increasing values represent further distances from rivers of major watersheds, which we interpret as decreasing site productivity and decreasing accessibility at a landscape scale (not necessarily reflecting operational scale policies for riparian buffers).

a. Terrain Resource Information Management (http://geobc.gov.bc.ca/base-mapping/atlas/trim/)
b. Vegetation Resource Inventory (https://www.for.gov.bc.ca/hts/vri/)
c. Ministry of Forests Lands and Natural Resource Operations High Value Fish Habitat spatial layer (query based on selecting large stream reaches that list S4-S7 attribute values—not to be confused with the S1-S6 stream classifications used in BC)

We also predicted the distribution over time of modern old growth logging to examine the extent to which harvesting occurred disproportionately along environmental gradients reflecting productivity and accessibility. We used the machine learning algorithm Maxent (Phillips et al. 2006), accessed through the *dismo* Package (Hijmans, R. J. et al. 2013) in R. Maxent is a robust method for predicting species distributions that quantifies relationships between occurrence data and environmental predictors (Franklin 2009). To understand trends over time, we created separate spatial models for each year of harvesting. To characterize raster cells representing harvested forests (occurrences, or “presence” locations in the model), we extracted variable values for Slope, Site Index, and Distance to Large Stream from 1000 random cells in each year of logging, with a minimum distance of 50 m between random points to avoid duplicate selections. The spatial extent of logging in some years led to fewer than 1000 random points fitting within these constraints, so the total number of presence points across all 47 models was 32,140. To characterize raster cells representing unharvested forests (“absence” locations in the model), we used these same modelling rules within the unlogged portion of the study area. We used the default settings in Maxent and k-fold cross-validation (k=5) for each dataset so that model training used 80% of the occurrences and model testing used the remaining 20%.

To quantify model fit, we calculated the Area Under the Receiver Operator Curve (AUC) statistic (Hanley & McNeil 1982), which shows how well a model can discriminate between logged areas and unlogged areas. An AUC above 0.5 would suggest that logged areas are predictable based on their environmental conditions because they are sufficiently different from a random sample of conditions in the study area. We also evaluated statistical relationships by examining variable contributions to model performance as well as individual variable response curves with covariates held at their mean value. This overall approach allowed us to understand factors influencing historical harvesting patterns based on the presence and absence of logging in the study area through time.

Results

Trends in productivity and accessibility over time

The total area harvested each year generally increased through the 1970s and 1980s until the mid-1990s, at which time there was a clear transition to a long-term decreasing trend (Figure 4.3). The time of this transition roughly corresponds with the introduction and implementation of major changes in provincial regulatory policy for forest stewardship in BC under a new Forest Practices Code (discussed above). The area logged in the study area over the past 156 years is 56,811 ha, approximately 7% of the total forested area. Of this logged area, historical logging (prior to 1970) accounts for 12%, modern old growth logging (since 1970) accounts for 87% (Figure 4.4), and modern second growth logging accounts for less than 1%. The mean cutblock size for modern old growth logging is 5.0 ha (range = 0.1-433 ha, although these numbers are somewhat uncertain because our harvest data may not precisely match the discrete cutblock boundaries that were engineered in the field in a given year). Mapping all three logging datasets over time shows clear changes in the spatial distribution of cutblocks over the different time periods (Figure 4.5). Logging activity, for instance, begins closer to the ocean and moves up valley bottoms through the 1970s and 1980s. In the 1990s and 2000s, logging occurs in much smaller discrete areas that are dispersed more broadly across the landscape. Starting in 2008, there was a major forest sector downturn tied to a global recession that, combined with agreements and policies to implement EBM, led to reduced harvesting activity in BC (see Benner et al. 2014)—73% less area was logged in the study area during the period 2008-2016, inclusive, relative to the previous nine-year period.

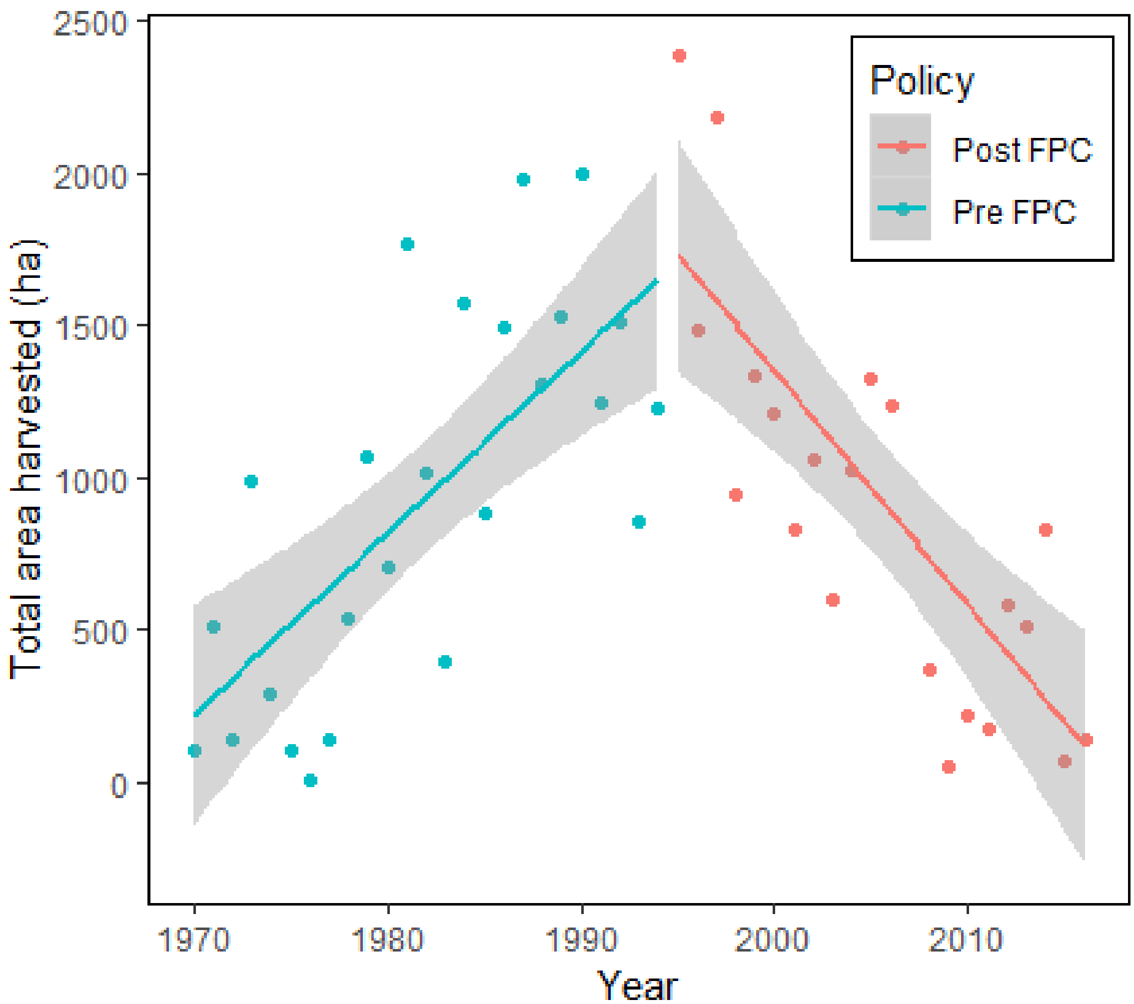

Figure 4.3 Total area harvested annually within our study area between 1970-2016. The peak of logging activity occurs in the mid-1990s, a period associated with major forestry policy changes resulting from the implementation of the Forest Practices Code (FPC). The blue and red lines represent best-fit regression lines and the shaded bands represent 95% confidence intervals.

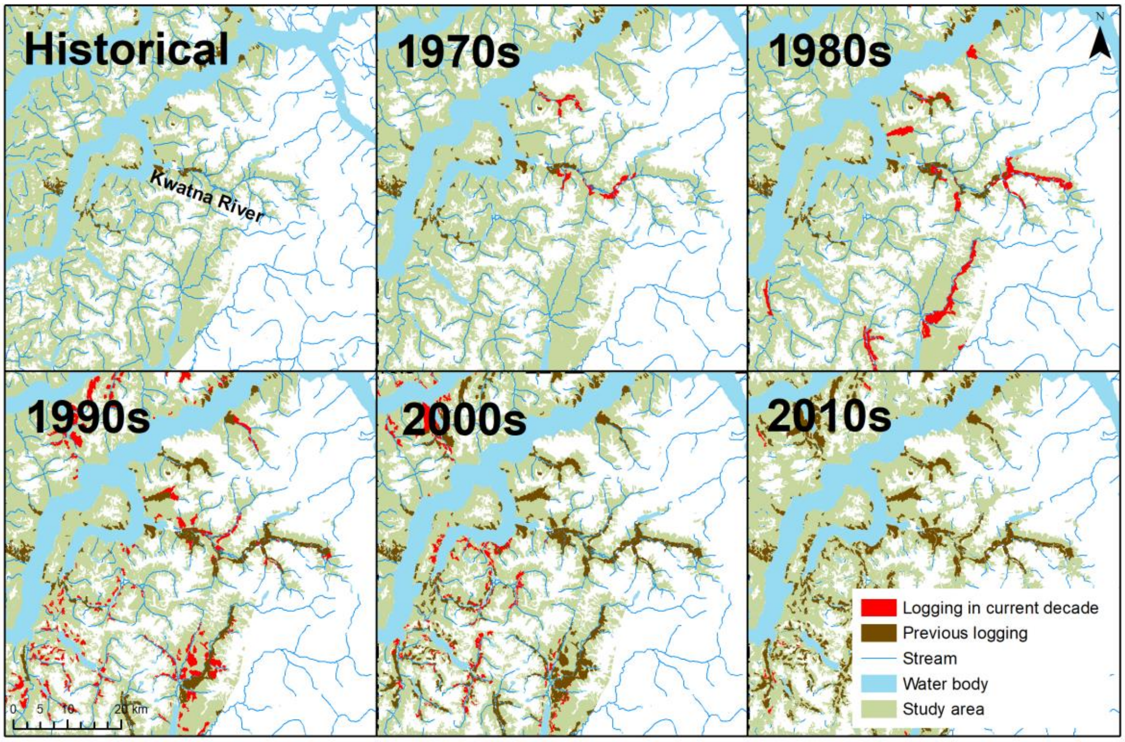

Figure 4.4 **Map showing the spatial distribution of cutblocks in different time periods. The highest rate of logging occurred in the 1980s-2000s. These panels are zoomed into the landscape around the Kwatna River watershed, which represent less than 10% of the overall study area.**

Notwithstanding variability and non-linear relationships, logging clearly began in accessible, productive locations and over time shifted to forests that were progressively less productive and accessible. Specifically, between 1970 and 2016, there is a downward trend in the mean Site Index of old growth logging (Figure 4.5a), a slight upward trend in the mean Slope (Figure 4.5b), and an upward trend in the Distance to Large Stream (Figure 4.5c). Over the most recent nine years in the sample there is greater variability among years in mean variable values, corresponding to the period with a downturn in the forestry sector and less harvesting in the study area. The mean values for all historical logging are similar to the mean values for second growth logging (historical: Site Index = 19.1, Slope = 42.5%, Distance to Large Stream = 4017m; second growth: Site Index = 20.7, Slope = 30.1%, Distance to Large Stream = 3938m). Since the second growth logging is taking place entirely in stands that were logged historically, this consistency makes sense.

For the variables Site Index, Slope, Distance to Large Stream, and Annual Area Harvested, there is a change in the regression line in the mid-1990s, a period associated with the FPC policy changes (Figure 4.3 and 4.5). However, the trends in variable values exhibit different types of changes around this time: the regression line for Site Index shifts toward lower productivity and becomes relatively steeper, the regression line for Slope shifts toward better accessibility and become relatively flatter, and the regression line for Distance to Large Stream does not shift and become relatively flatter. In other words, the trend showing a decline in the productivity of harvested forests becomes more pronounced after this policy change and the trend showing a decline in the accessibility of harvested forests becomes less pronounced after this period (although still trending towards decreasing accessibility overall). When examining the mean coefficient values from the linear regression model for all years in the post-FPC period, Site Index is 82% of the pre-FPC period ($p > 0.005$), Slope is essentially unchanged relative to the pre-FPC period ($p > 0.695$), and Distance to Large Stream is 126% of the pre-FPC period ($p > 0.005$). Although the differences across policy regimes are partly explained by the overall directional trends across the entire study period, these results suggest that the FPC is correlated with changes to the areas selected for harvest. The dramatic change in the annual area harvested (Figure 4.3) around this period also strengthens the hypothesis that the FPC strongly influenced harvesting activity.

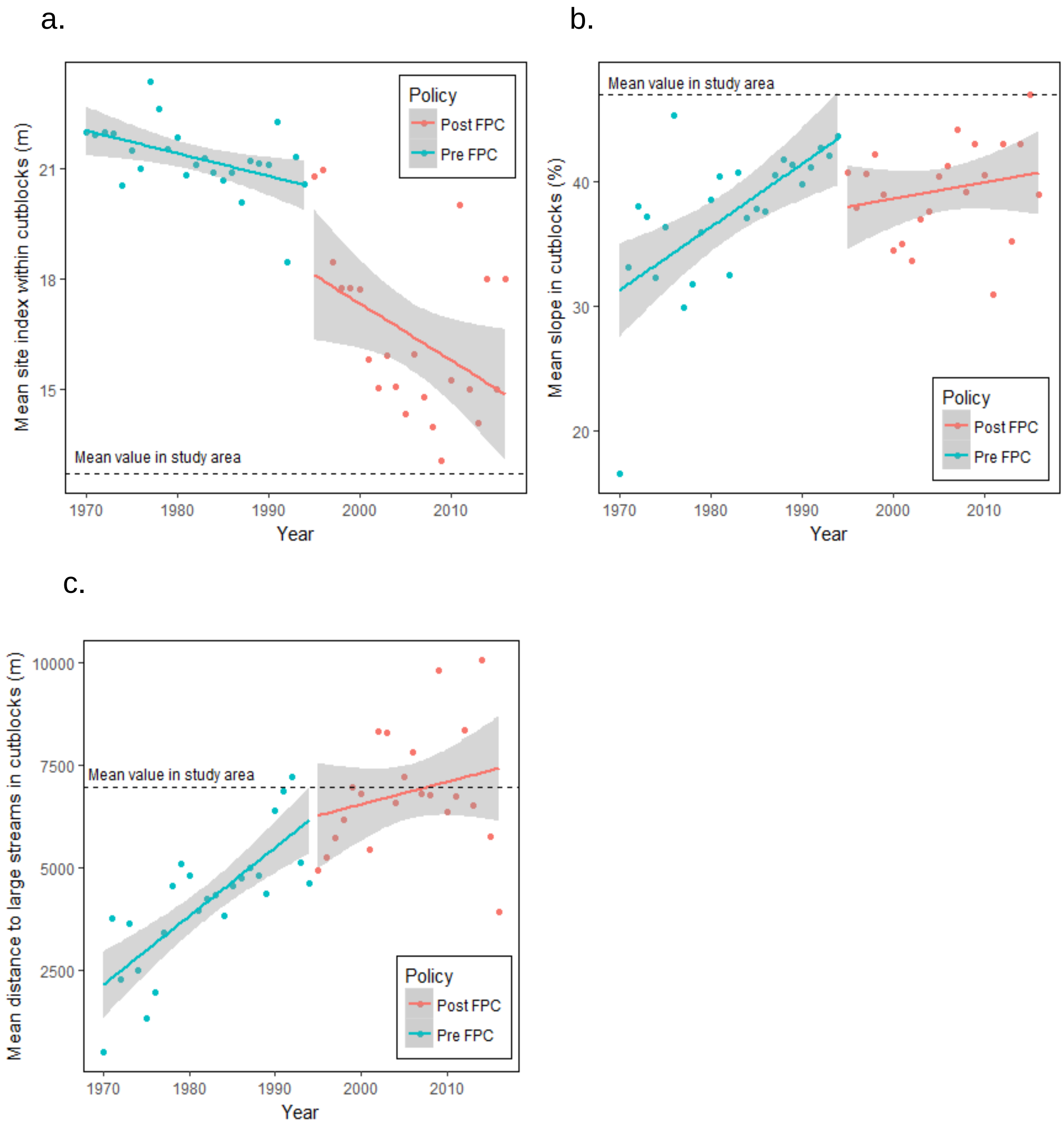

Figure 4.5 **Average (a) Site Index, (b) Slope, and (c) Distance to Large Stream of logged areas between 1970-2016. The productivity and accessibility of the areas being logged declines over the study period and the policy changes associated with the FPC are correlated with distinct changes to harvesting patterns. The shaded bands represent 95% confidence intervals.**

Concordance between logged areas and broader landscape

Over the course of the study period, logged areas became more representative of the broader environmental conditions across the study area. For example, comparing models representing each year of modern old growth logging shows a temporal trend of declining AUC values, which suggests that our model has progressively less ability to discriminate locations with cutblocks from locations without cutblocks, particularly when data for the period after the forestry downturn between 2008-2016 is aggregated (Figure 4.6). When using all data for modern old growth logging between 1970-2016, the model performs well (AUC = 0.87), indicating that, over most of the period, logging is not well distributed across the range of environmental conditions existing within the study area. Across all years (all models), Site Index is the most important variable responsible for this relationship, contributing to 85% of the model's predictive performance, although in any individual year the relative importance of the three variables and their specific interactions changes. Thus, overall, logged areas are biased towards higher productivity locations on the landscape. Consistent with the trend highlighted in the AUC analysis, all three of our variables have their average values within cutblocks moving towards, reaching, or moving past their respective mean study area values over time (Figure 4.5). Overall, the response curves show that forests with gentler slopes, higher productivity, and closer to large streams have a higher probability of logging activity compared to unlogged forests.

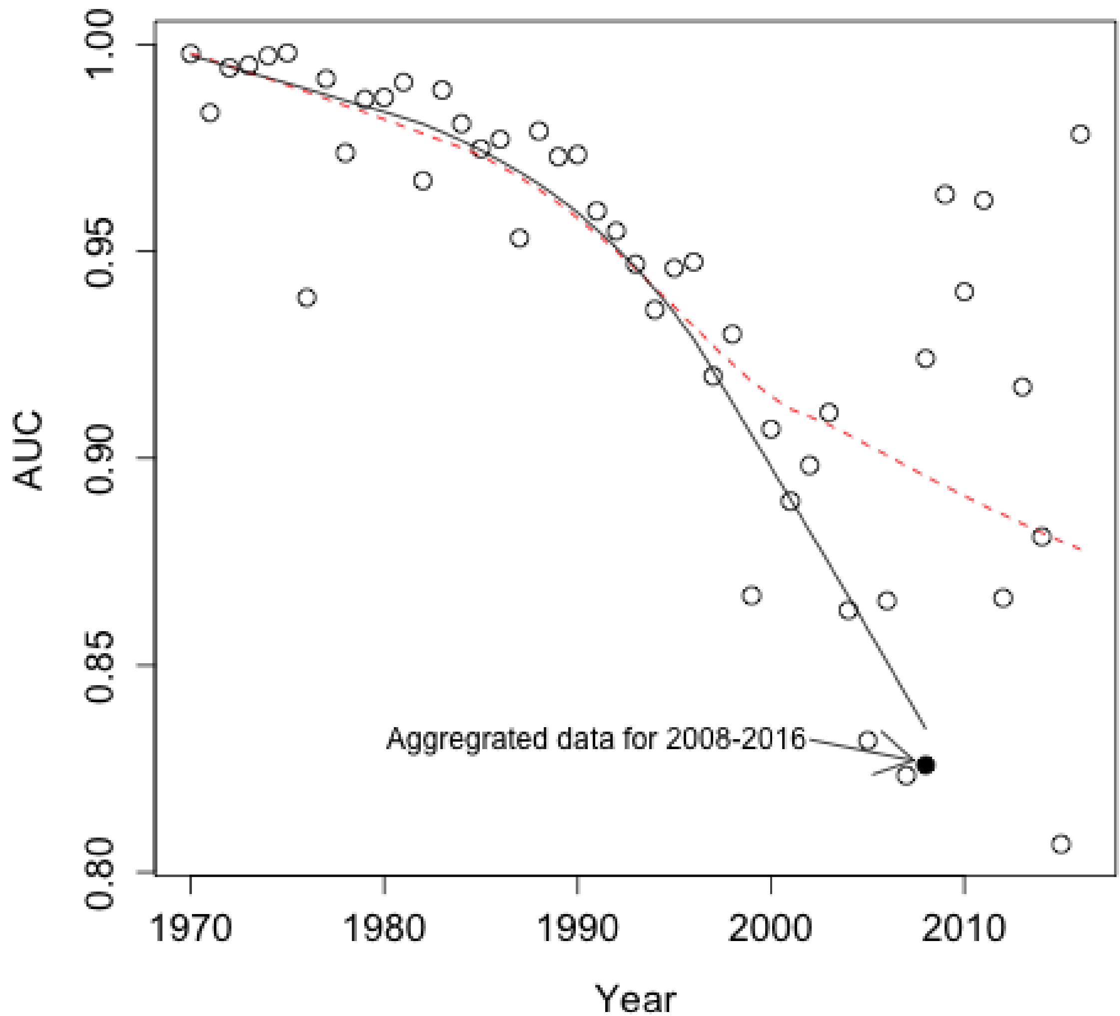

Figure 4.6 **Scatterplots showing area under the reciever operator curve (AUC) by year. Over time the model has less of an ability to discrimate between logged and unlogged forests, suggesting that harvesting is becoming more representative of the broader environmental conditions of the landscape (average conditions: AUC = 0.5). The AUC values between 2008 and 2016 show much more variability than previous years, largely because the economic downturn led to less logging in those years (about 25% of pre-2008 logging rates) and thus a paucity of data. The overall trend using all of the data is approximated by the dashed red lowess smoother. The solid circle represents data for for a model that aggregrates data across 2008-2016 to account for the reduced area logged in the study area during this period. The overall trend using data aggregrated for this latter period is approximated by the solid black lowess smoother.**

Discussion

A changing landscape

Our findings reveal a long-term trend of logging down the value chain. Specifically, harvesting over the past half century in our study area on the central coast of BC generally started in higher productivity and more accessible forests and proceeded to less productive and accessible forests over time. This pattern has led over time to the forests harvested being more representative of the environmental conditions of the broader landscape because forestry companies targeted the most profitable components first and were forced into increasingly average conditions over time. Over the study period, it becomes more difficult to discriminate between the underlying environmental conditions of logged areas vs. unlogged areas. This convergence over time is mostly driven by the relationship between the spatial distribution of harvesting and Site Index, a metric of site productivity and the most significant variable in our model. Such a trend suggests that as time progressed across the study period, the forestry sector had fewer opportunities to harvest the types of productive stands that produce the largest trees, which are relatively rare in the regional context. In addition to this general trend, conservation-oriented regulations and policies implemented over the second half of the study period appear to have also altered harvesting behavior by constraining the land base available for logging. Our quantitative analysis brings empiricism to a long-standing narrative in the forestry sector about high grading and the influence of policies, and is consistent with other assessments of harvesting patterns in the coastal temperate rainforest, which show disproportionate logging activity in the most productive landforms in Alaska (Albert & Schoen 2013) and in valley bottoms on the central coast of BC (Pearson 2010). Our study also reinforces the benefits, espoused by other scholars (e.g. Rhemtulla & Mladenoff 2007), of using historical datasets to understand land use and landscape change over time.

Competing paradigms

We posed two alternative paradigms that have shaped the development of forestry practice: traditional forest economics, and forest stewardship. When the overall time period of industrial development is considered, the trends we observe here align more closely with the expectations of the industrial economic paradigm focused on prioritizing profitability, than with those of the forestry stewardship paradigm. This result is not surprising given that over the study period large corporations have held most forestry tenures on the central coast of BC and government priorities in the early study period mostly focused on exploiting the timber resource and maximizing short-term revenues and economic development (Rajala 2006). The modern global forest industry grapples with diverse corporate responsibilities that include new regimes of forest certification and also diverse perspectives among foresters who integrate stewardship principles into various decisions, but maximizing profits to shareholders typically remains the chief motive for company decisions (Li & Toppinen 2011).

There are also some practical and regulatory factors underpinning this economic paradigm. For example, the operational logistics of accessing forests across the landscape affect patterns of logging because locations such as towns, highways, or river mouths, which may be spatially correlated with gradients of productivity and accessibility, are logical entry points for developing road networks across watersheds. Additionally, despite the provincial government advocating, during assessments of timber supply, that companies log the forest profile (Snetsinger 2011), there is not an associated law or policy in BC that requires harvested forests to be representative of the broader land base to achieve social or ecological goals. Maintaining this type of flexibility in forest management decisions is consistent with the performance-based regulatory environment currently being used in the province (Hoberg & Malkinson 2013). Large forest companies are unlikely to unilaterally stop logging down the value chain, so changes in government policies may be required to shift the behaviour of the forest industry.

Policy interventions can make a difference

Profit or stewardship-driven decisions by forestry companies and foresters do not take place in isolation of other factors, such as the mediating effects of policy, technology, and log markets. Examining our time series data and spatial trends over time shows the implementation of major changes in provincial regulations and policies in the mid-1990s correlating strongly with changes to the productivity and accessibility of harvested forests, though the specific statistical changes to our three indicators show mixed results and do not consistently align with simple a priori expectations. Because regulations and policies associated with the Forest Practices Code and the Great Bear

Rainforest limited timber harvesting in portions of the land base and placed constraints on the size and distribution of cutblocks to address biodiversity objectives and negative public perceptions about industrial forestry (Fenger 1996), the harvesting regime transitioned from large, concentrated clear-cuts that progressed up major watershed drainages to smaller cut blocks dispersed across the broader landscape. In addition to government policies, incremental technological innovations such as grapple yarders and helicopters likely changed harvesting patterns over time as well, by enabling access to more isolated and logistically challenging terrain outside of the historically harvested valley bottoms. Another spatial driver of logging activity not captured by our study is forest composition relative to log markets. While we did not integrate tree species data into our model because the pre-harvest stand composition was unknown, other research suggests that a particular species—western redcedar—is being disproportionately targeted in the study area over recent decades due to strong markets for this type of wood (Green, 2007), leading to a shift in the distribution of large, old redcedar trees (Chapter 2). This suite of diverse and interacting factors makes it challenging to attribute spatial-temporal changes in harvesting patterns to any single cause, but government policy clearly was a perturbation to the overall system.

An alternative state of forest value

The forest harvesting patterns shown in our study area are consistent with an economic paradigm focused on maximizing profits, which over the long term has implications for intergenerational access to timber resources. Because forestry is considered renewable and intended to span multiple harvest rotations, second growth logging provides an opportunity to target the same environmental gradients as historical logging of old growth. Indeed, the average variable values in our study area for second growth and historical logging are similar. But despite representing the same land base in terms of productivity and accessibility, second growth stands often contain lower timber volumes and lower market value, depending on demand for particular forest products, because the trees are younger, smaller, and have different wood characteristics (Barbour et al. 2002). These second growth forests also lack many of the ecological values and ecosystem services provided by the pre-disturbance forest, such as high carbon stocks, species diversity, and complex structure that supports wildlife (Fredeen et al. 2005; LePage & Banner 2014). At the scale of human generations, this shift to a less valuable resource on the same area of ground could be thought of as a persistent alternative state of low value given the centuries needed to re-grow old growth forests in places like the coastal temperate rainforest, and the persistence of such old growth forests for millennia in the absence of major natural disturbances (Lertzman et al. 2002).

Increasing attention is being paid globally to the ecological and cultural roles of large old trees and the services they provide (Lindenmayer et al. 2012; Blicharska & Mikusiński 2014; Lindenmayer & Laurance 2017). Though we only focus on trends in site productivity, the landscape trajectory we describe is one where the large, old landscape elements were preferentially targeted by early resource development, producing a land base depauperate of key historical components. The landscape legacies of this manifest, for instance, in a scarcity of culturally important species such as monumental cedar trees that are critical to Indigenous carvers (Chapter 2 and 3), a condition that will take many hundreds of years to redress. To borrow another idea from the fisheries literature, this also represents a perceived shifting baseline (Pinnegar & Engelhard 2008), in that our modern conception of the forest land base may have normalized a condition of degraded value, economically, culturally, and ecologically.

Intergenerational access to forest value

Around the world jurisdictions are increasingly shifting the authority to make governance decisions about land and forest tenures from state and corporate actors to communities and Indigenous groups (Agrawal 2001; Larson et al. 2010). What is the effect of an alternative state of reduced forest value on such communities? As our study shows for the Great Bear Rainforest, the land base allocated to communities often follows decades or centuries of extraction of the most profitable resources—described by Anderson et al. (2015) as “managing the leftovers”. If the profits from originally extracting this natural capital were directly invested locally in other forms of human and physical capital (e.g. schools, hospitals, water treatment), then the community might perceive past land use decisions that resulted in the alternative state as a more reasonable trade-off.

In many cases, however, this type of devolution of governance to the local level is occurring in isolated regions inhabited by Indigenous populations where much of the accumulated profits from resource extraction have already been distributed outside local communities (Calfucura 2018). In regions like our study area, for example, the drawdown of natural capital from logging over the past century has been concentrated locally but its conversion into physical and human capital has often been dispersed outside the local area (Rajala 2006). As Solow (1993) argues, “the cardinal sin is not [extracting]; it is consuming the rents from [extracting]”. In the case of remote communities, inhabitants may think that sending rents outside the local economy is functionally analogous to consuming those rents. This means that capital substitution resulting from harvesting the most productive and accessible forests might be thought of as a societal benefit at a provincial scale but may not have necessarily maintained or increased well-being over time at a community scale, particularly when local market externalities, such as degraded ecosystem services in community watersheds, get factored into the costs and benefits of past management decisions. Given these scale-dependent trade-offs, the emergence of community-based management may well lead to different choices concerning the distribution of forestry activity than those made historically by corporate or state actors (e.g. Benner et al., 2014; Ostrom and Nagendra, 2006).

Will ecosystem-based management change the forestry paradigm?

In addition to potential bottom-up changes emerging from increased community authority, new government policies at the state (or provincial) level can also transform harvesting patterns, as our data on the FPC show. In the future, the ongoing implementation of a new regime of EBM in the GBR and its strong emphasis on the principle of ecological representation in reserve planning (Price et al. 2009) should lead to a greater range of environmental gradients being conserved (see Chapter 5). Creating reserve designs based on principles of ecological representation will help to limit the ability of forest operations to log down the value chain at a landscape scale, but trends consistent with targeting the most productive and accessible stands may persist within the newly constrained managed forest because landscapes are inherently heterogenous; there will always be some forests that are relatively more profitable (and thus more desirable for timber) than others. It is important to note differences between the underlying objectives of EBM compared to the classic stewardship objective to log the profile. This latter ethic is mostly rooted in the social goal to provide equal or greater harvesting opportunities for future generations, whereas ecological representation is rooted in the goal to conserve forest biodiversity. Therefore, ensuring ecological representation will ultimately reduce harvesting opportunities for future generations relative to past harvesting patterns.

Conclusion

Overall, it is clear from this research that forest management has resulted in logging down the value chain over the past century and that more recent conservation-oriented policies have affected land use in complex ways. These patterns of harvesting are similar to some of the trends in the ocean made famous by Pauly et al. (1998). In both systems, the most valuable components of the environment have been targeted first, with less and less valuable components being harvested sequentially over time. Although our terrestrial study did not focus on trophic levels, the broad implications for ecological integrity and sustainable access to resources create a strong thematic connection with this foundational marine research.

In our study system, the choices made by resource managers initially reflected the dominance of an industrial economic paradigm focused on short-term profitability, enabled by the widespread availability of high value forests and the allocation of tenure rights to large industrial forestry companies. This was eventually modified by constraints on those choices imposed by a history of highgrading the most productive and accessible parts of the land base. Finally, explicit changes in the policy environment intended to reflect broader goals of stewardship interact with historical constraints to produce a modern allocation of harvesting that differs substantially from that in the early study period. This illustrates both a tendency to harvest down the value chain when choices are unconstrained and the ability of policy choices to impose a greater stewardship ethic on harvesting behavior. Although harvesting patterns have altered the state of resource value, new opportunities are emerging through devolved decision-making to communities and through ecosystem-based management to more equitably balance intergenerational access to resources and the environment.

Acknowledgements

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Chapter 5. The influence of ecological and human factors on the spatial design of conservation areas

Abstract

Setting quantitative targets is an important step in conservation planning but achieving these targets can be complex and challenging. To better understand variability in the implementation of regulations that specify conservation targets, we examined the design of 90 networks of landscape reserves developed between 2009 and 2012, in the Great Bear Rainforest, British Columbia, Canada. Although this reserve design process was based on a broad set of standardized objectives for ecological integrity and human well-being, the spatial configuration of reserves was primarily driven by ecological representation targets set by the provincial government, focused on conserving percentages of different types of old growth forest ecosystems. We used a geographic information system and a series of generalized mixed effects models to compare the actual reserve systems implemented through this complex land-use planning process with the targets for ecological representation established in regulations a priori. There was a great deal of variability in how targets were met. Overall, the area within implemented reserves exceeded ecological representation targets by 59%, partly stemming from the diverse suite of conservation objectives that the reserve network was attempting to address in addition to the representation targets. Human and ecological factors influenced the degree to which reserves deviated from targets. Less productive forests, less ambitious representation targets, and planning units associated with more pre-existing conservation area were associated with greater levels of deviation above targets. The individual professional leading the design in any particular landscape planning process also influenced the highly variable patterns of reserve design across the study area. These findings highlight the challenges posed by heterogenous social-ecological systems when attempting to achieve conservation targets and underscore the need for planning initiatives to clearly define and interpret whether these thresholds represent minimums, maximums, or targets that must be precisely met.

Introduction

Setting quantitative targets is a common approach toward achieving specific outcomes in natural resource management and conservation (Butchart et al. 2010; Hagerman & Pelai 2016). Such targets usually focus on allocating a spatial or volumetric percentage of a resource or the environment towards an economic, social, cultural, or environmental objective. For example, targets are widely used in mitigation strategies for reducing greenhouse gas emissions (United Nations 2015), biodiversity conservation and in natural resource management disciplines such as fisheries (Poos et al. 2010), and forestry (Lindenmayer et al. 2012). However, setting a target does not mean that the target will be met, with outcomes often depending on a complex range of interacting factors (Hagerman & Pelai 2016). Understanding factors that affect the implementation of planning activities designed to meet a priori targets is a key part of tackling some of the world's foremost sustainability challenges.

Conservation initiatives that involve designing a system of reserves in marine or terrestrial environments often use quantitative targets to systematically prioritize areas based on goals to support biodiversity and minimize socioeconomic costs (Sarkar et al. 2006; Klein et al. 2008). There is a large body of literature addressing applied conservation tools that focus on selecting sites to support reserve design based on formal targets (Gonzales et al. 2003; Sarkar et al. 2006; Fraschetti et al. 2009). But further research is needed on evaluating factors that affect the implementation of regulations or policies to achieve conservation targets, either in terms of the design of spatial conservation areas or in terms of management outcomes for biodiversity and socioeconomic opportunities (Leslie 2005; Knight et al. 2006). Analyses have been conducted of the outcomes of implementation for some high profile conservation initiatives such as the Northwest Forest Plan in the United States (Power 2006; Spies & Martin 2006; DellaSala et al. 2015) and the Great Barrier Reef Marine Protected Area in Australia (McCook et al. 2010). In both of these cases, comprehensive assessments of the implemented plans revealed extensive biodiversity benefits resulting from the established reserve networks, but mixed outcomes in terms of socio-economic impacts. These types of assessments are important, but to fully understand the underlying reasons for an observed effect on the ground, more knowledge about the original design of the reserve network is required. In particular, data are needed about the human and ecological factors that affect the translation of policy targets into a spatial reserve network.

Some approaches to reserve design use targets based on the principle of ecological representation, which aims to protect sites along ecological and environmental gradients proportionally to their natural distribution across the landscape or seascape (Pressey et al. 1993; Klein et al. 2008; Andrew et al. 2014). For instance, conservation initiatives have employed targets focused on conserving percentages of ecological surrogates such as distinct types of shoreline substrates (Banks & Skilleter 2007) and old growth forests (Price et al. 2009). The hypothesis that representing ecosystems in conservation areas relative to their broader distribution will achieve biodiversity objectives stems from the recognition that we have insufficient knowledge about the dynamics and habitat requirements for the majority of species (Pressey et al. 1993; Schwartz 1999). This would suggest that we need to address the needs broadly of many poorly known taxa and their ecological interactions. In contrast to the more ad hoc and simplistic approaches historically used to create many parks around the world (Sellars 1997; Branquart et al. 2008), establishing and achieving specific targets for ecological representation in the spatial design of conservation areas often requires, through principles of systematic conservation planning, detailed datasets, complex spatial models, and a highly technical planning process (Schwartz 1999).

Given the huge effort involved and the costs incurred in spatial conservation planning, it is obviously prudent to compare the completed plans with the original objectives. Such assessments should determine whether or not the pre-determined targets were achieved in the resulting spatial design and test potential factors that could have influenced any deviations from the representation targets (Margules & Pressey 2000). Learning from implementation data is an important aspect of many ecosystem-based management processes—especially those that advocate for adopting principles of adaptive management (Grumbine 1994; Price et al. 2009; McCook et al. 2010). In some jurisdictions a performance-based regulatory regime requires ongoing monitoring and evaluation (Hoberg & Malkinson 2013). However, in practice these types of evaluations are rarely completed and, when they are completed, they rarely address variables rooted in the human dimensions of planning, which are salient given their strong influence on outcomes in complex social-ecological systems (Folke et al. 2005; Leslie 2005; Hagerman & Pelai 2016).

In this study, we examine both biophysical and human factors that we hypothesized would affect the spatial design of conservation areas in the Great Bear Rainforest (GBR), British Columbia, Canada. This conservation planning process, nested within an overall framework of EBM, provides a dynamic and globally relevant example of implementing policies with ecological representation targets. Unlike large jurisdictions with a single conservation target for reducing greenhouse gas emissions or creating protected areas, the landscape reserve design (LRD) process for the GBR examined here included provincially set ecological representation targets that applied to 90 separately developed LRDs, creating a novel opportunity for robust analysis. We assess the extent to which the ecological components in each reserve design deviate from the a priori representation targets and examine how four variables influence these deviations: Representation Target, Existing Conservation, Site Productivity, and Lead Professional (Table 5.1). Finally, we discuss implications of this research for new approaches to conservation planning and natural resource management that involve multiple targets, including those that place upper caps on conservation to maintain specific proportions of the land base for economic and other opportunities.

Table 5.1 Overview of potential variables affecting landscape reserve designs (LRDs).

Variable	Description	Hypothesis	Rationale
Representation Target (Predictor)	The percent of an ecosystem type that must be integrated into LRDs according to policy guidance.	The implemented LRDs will exceed policy targets to a greater extent for planning units that require lower levels of conservation.	It is challenging to achieve diverse conservation objectives at lower levels of conservation.
Existing Conservation (Predictor)	The percent of an ecosystem type that is under a pre-existing conservation designation.	The implemented LRDs will exceed policy targets to a greater extent for planning units that have higher levels of pre-existing conservation.	Portions of ecosystems that were committed to conservation through previous processes did not necessarily have this status accounted for during target formulation, so some targets may be partially or entirely met prior to any design work being carried out.
Site Productivity (Predictor)	The site productivity class as expressed by site index, representing the potential height (m) of dominant trees in stands at age 50, measured at breast height.	The implemented LRDs will exceed policy targets to a greater extent for planning units that are associated with less productive ecosystems.	Less productive ecosystems are generally associated with less valuable timber, so allocating these types of stands to reserves has less impact on forestry opportunities.
Lead Professional (Predictor)	The analyst leading the development of LRDs in a landscape unit.	We do not pose a specific hypothesis for this predictor but expect that the individual lead professional will influence variability across LRDs.	Designers have diverse perspectives and values about prioritizing areas across the landscape and make distinct tradeoffs among competing objectives.
Deviation from Target (Response)	The percentage difference between the area required in the policy target and the area contained within the completed LRDs.	The completed LRDs will deviate above aggregate policy targets.	Biophysical and human factors make it challenging to achieve diverse conservation objectives based on precise *a priori* targets, especially when factors are not accounted for during target formulation, and policies in the GBR characterized targets as minimum thresholds.

Methods

Study system—The Great Bear Rainforest

Background and regulatory framework

Our study area covers the GBR, an ecologically and culturally diverse region that extends from the south-central coast of British Columbia to southern Alaska and forms a major part of the global distribution of coastal temperate rainforest (Fig 5.1; for more information on the biophysical and socio-economic characteristics of study area, see Chapter 2-3 or Price et al. 2009). The GBR is a landmark conservation initiative intended to conserve 70% of most ecosystem types that are present in the region under a newly developed framework of EBM. The initiative relies heavily on explicit conservation targets to be achieved through a formal landscape reserve design (LRD) process, based on a series of regulations and policies established in 2007, 2009, 2013, and 2016 (Government of British Columbia 2007, 2009, 2013, 2016).

Many details related to formulating and implementing conservation targets in the GBR have evolved over time (see Supporting Information), but a core element of the GBR initiative includes an objective to capture within the LRD process a percentage of each different type of old growth forest ecosystem (see, Price et al., 2009 for more information about this approach). Substantial legal and policy guidance has emerged as part of EBM implementation, but at the time of writing, a final set of LRDs in the GBR is not yet complete after more than a decade of complex and contentious spatial planning by First Nations, the provincial government, forestry licensees, and environmental groups. This planning process has been punctuated by occasional changes in the governing regulations and policies as the system was refined. Such a slow pace of implementation is a common issue with conservation planning around the world (Sarkar et al. 2006). However, technical planning teams in the GBR completed an interim set of 90 LRDs between 2009-2012 that provide important lessons for spatial planning in the GBR as well as for conservation initiatives in other jurisdictions around the world. Our study focuses on these LRDs.

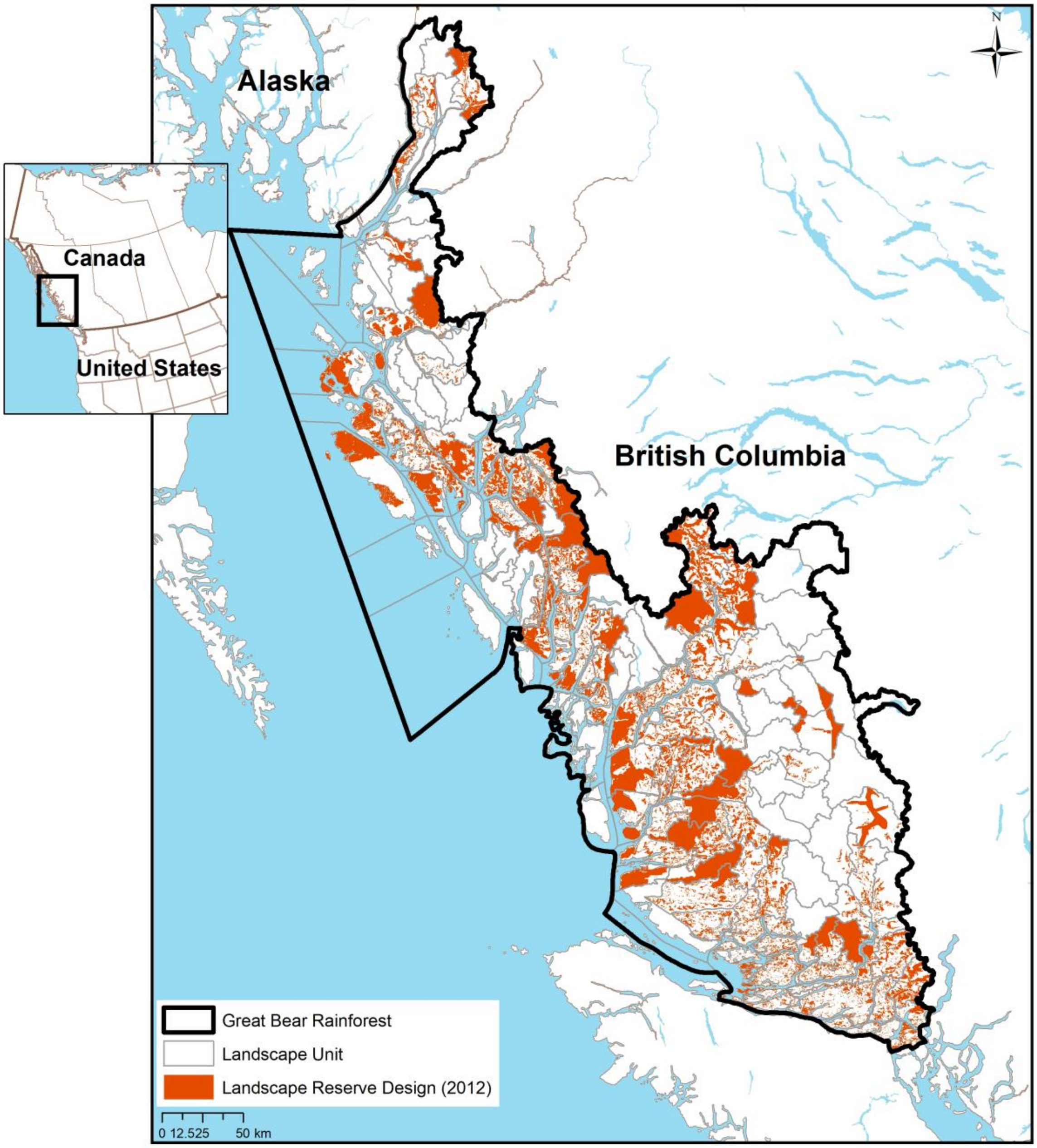

Figure 5.1 Map of study area showing the Great Bear Rainforest and the Landscape Reserves Designs (LRDs; red polygons) that were completed in different Landscape Units between 2009-2012.

The initial land use objectives for the GBR, including representation targets set for LRDs, were established in 2007 under BC's Land Act through land use orders. These were amended in 2009 (the "2009 Order") and 2013, and then subsequently replaced in 2016 by the Great Bear Rainforest Order, which combined northern and southern sections of the GBR (Government of British Columbia 2007, 2009, 2013, 2016). Over this period, the GBR regulations and policies were nested within a "performance-based regulatory approach" that was used more broadly for forest management in BC under a framework established by the Forest and Range Practices Act (FRPA), which came into effect in 2004. Hoberg and Malkinson (2013) note that the land use objectives in the GBR "are so specific as to be arguably inconsistent with a performance-based approach". LRDs, for example, have prescriptive aspects, such as the quantitative targets and the specific steps expressed in the LRD methodology, but they also allow for some flexibility in design and professional discretion. In contrast to some of the systematic planning tools that are widely used in conservation initiatives globally (Margules & Pressey 2000), the LRD methodology in the GBR outlines an approach to prioritizing reserve areas that includes iterative spatial analysis based on key spatial datasets, government-to-government decision making between the Province of BC and Indigenous groups, and stakeholder engagement (Price et al. 2009; Province of British Columbia 2016).

The LRDs analyzed in this study were finalized following the 2009 Order, which did not include explicit regulatory restrictions on impacts to timber supply. Such flexibility represents a distinct contrast to restrictions established at a provincial scale for other forestry policies, such as the quantitative 6% threshold for impacts to timber supply set under the previous regulatory regime of the Forest Practices Code (British Columbia 1994) or the qualitative phrase, "without unduly reducing the supply of timber" that underpins many objectives established under FRPA (British Columbia 2004). The new set of objectives and targets in the 2016 GBR Order, however, aimed to address concerns raised by the forestry sector by adopting Managed Forest targets, which essentially reflect a legal upper cap on the amount of conservation area that can be built into LRD.

Spatial planning for ecological representation is being implemented in the GBR through a hierarchical process, with targets for protecting representative old growth ecosystems at various scales. For example, the 2009 Order set an aggregate conservation target of 50% for the entire GBR, but individual Landscape Units (LUs)—administrative units which function as discrete land bases for each individual LRD planning process (Figure 5.1)— could have targets set at 30%, 50%, 70% or 100%. This range of targets across LUs reflects the conservation priority of the associated areas based on negotiations between provincial and First Nation governments as well as stakeholder input. Unlike many EBM planning initiatives, these ecological representation targets were not simply based on conserving a fixed percentage of the overall land base. Instead, they reflect an approach to representation that aims to conserve old growth forests according to the range of natural variability within which these ecosystems were historically present across the landscape (RONV; for more information, see Price et al. 2009). At the time of the 2009 Order, about one-third of the GBR was already designated as conserved areas through a network of parks, conservancies, and various other conservation areas in which there were no overlapping forestry tenures or harvesting rights (Province of British Columbia 2009). But this level of protection fell short of the 70% “low risk” benchmark for the conservation of old growth ecosystems recommended by the Coast Information Team—the scientific advisory body tasked with defining and laying the groundwork for EBM in the GBR—or even the interim 50% target established by the province in the 2009 Order (Province of British Columbia 2009). Therefore, to fill this deficit in the conservation budget, the governments and stakeholders collaborated on spatial LRD planning, though most of the design work was led by foresters and biologists that worked for the major forestry tenure holders and government agencies. This process included the types of fixed designated conservation areas mentioned above as well as an additional layer of “soft reserves” that can move across the landscape over time based on changing forest dynamics as long as the ecological representation targets for old growth ecosystems are met.

Planning units for ecological representation

Each LRD was developed for a Landscape Unit based on ecological planning units at the stand scale that reflect concepts and definitions within the province’s Biogeographic Ecosystem Classification (BEC) system (Meidinger & Pojar 1991). This system classifies ecosystems hierarchically over different spatial scales based on climate, soil, and vegetation (zone>subzone>variant>site series). Representation requirements could be set at various levels along this spectrum. For instance, prior to the GBR Orders, Wells et al. (2003) examined ecosystem representation in the GBR’s protected areas to the level of BEC variant and Gonzales et al. (2003) used a similar approach in their evaluation of systematic reserve designs. Dropping one level lower on the BEC system’s hierarchy is site series (SS), which is the finest resolution of ecosystem mapping based on the site level soil moisture and nutrient regime, and a substantially more detailed and data intensive way to represent ecosystem variability.

Many areas of the GBR, however, had not been mapped to site series prior to the 2009 Order, making surrogate approaches more practical over this large geographical region. The first iteration of attempts to implement ecological representation in LRD (i.e. those associated with this study) used planning units based on Site Series Surrogates (SSS). As the name suggests, SSS are surrogates for site series identified using two variables: the leading species within stands, and site productivity classes based on site index—representing the average height of dominant trees at 50 years of age, measured at breast height (hereafter, we use the term "ecosystem types" when discussing SSS). However, now that most of the GBR's land base is mapped to site series, new LRDs associated with implementing the 2016 Order are using site series groups (SSG) as the planning units for ecological representation. SSG are aggregates of similar site series, meaning that ecological representation under the 2016 Order is on a coarser scale than individual SS on their own.

Data Analysis

Each LRD process produces spatial areas within a LU where commercial forestry is prohibited and other areas where forestry can occur. We accessed spatial datasets and associated reports for 90 LRDs that were completed between 2009-2012, representing a total area of reserves of almost 1.7 million ha. In ArcGIS (ESRI 2019), we created a dataset containing those areas where LRDs overlap with a spatial layer characterizing the distinct ecosystem types. We then joined this dataset, now containing information about the amount and types of ecosystems captured in LRDs, to a spreadsheet listing the targets for ecological representation that were previously established under the 2009 Order (Province of British Columbia 2009). Of the 90 LRD processes originally accessed, we eliminated from our analysis three LRDs for LUs where the specified values for representation targets were inconsistent between the LRD reports and the policy guidance spreadsheet (indicating perhaps that the targets had been misinterpreted by the planners). This final dataset allowed us to assess the extent to which the implemented LRDs were congruent with policy targets and to quantify percentage deviation from targets for each ecosystem type in each LU. Deviation is the response variable for the 2061 unique targets for the different ecosystem types that were established in policy across LUs in our study area.

In addition to examining total deviations from targets, we also investigated how different factors affect patterns of deviation. We populated new fields in the above dataset with values for four predictor variables (Table 5.1). We selected these variables due to their importance to reserve design based on a review of the literature and on our collective experience over the past three decades in research and forest planning in the GBR and other regions implementing EBM. The values for our predictor Representation Target—describing the percent of an ecosystem type that must be integrated into LRD according to policy guidance—came directly from the policy guidance spreadsheet associated with the 2009 Order and its Schedules (Province of British Columbia 2009). Similarly, we obtained the values for our predictor Site Productivity— describing the site productivity class (poor, medium, good)—from the policy guidance spreadsheet because this variable was explicitly used to characterize ecosystem types. To derive values for our predictor Existing Conservation—describing the percent of an ecosystem type that is under a pre-existing conservation designation—we conducted overlays in ArcGIS that calculated how much area with pre-existing conservation designations was captured in each ecosystem type in each LRD. Finally, we cross-referenced the applicable Landscape Unit with the LRD reports to derive values for our predictor Lead Professional—describing the person leading LRD work in each LU.

To examine the relationship between Deviation and our four predictor variables, we built generalized linear mixed effects models and performed model diagnostics in R (R Core Team 2018). We used the *nlme* package (Pinheiro et al. 2019) to build and test four separate models based on these predictors, with the LU name as the random effect to account for the nested structure of the data. This type of error structure occurred because the targets for each ecosystem type were nested at the LU scale, thus creating significant autocorrelation across the 87 LRDs assessed in our study. We generated summary results for all categorical classes within each variable using the function *intervals* to calculate coefficients and confidence intervals.

We also tested our models and calculated p-values based on only two classes for each predictor variable, generally representing low vs. high values. For Representation Target, we compared observations in areas that required more than half of the area to be conserved to those observations in areas that required less than half the area to be conserved (i.e. more ambitious targets vs. less ambitious targets). For Existing Conservation, we compared observations with more than half of their area under pre-existing conservation designations to those observations with less than half of their area under pre-existing conservation. For Site Productivity, we compared observations listing the productivity class as ‘good’ to those observations listing ‘poor’. We did not statistically test the effect on target deviation from Lead Professional and instead just examined overall variability across LUs.

We also built a multivariate mixed effects model containing all four variables to assess the relative importance of these variables for explaining patterns of deviation from targets. Although the variables may interact with each other in our study system, we built the model assuming no interactions to simplify interpretations about the relative influence of individual factors. We used the *dredge* function, accessed through the *MuMIn* package (Barton & R Core Team 2019) to determine the strength of evidence for these models via maximum likelihood and AICc rankings (Burnham & Anderson 2002). Finally, we accessed the *importance* function to assess the relative variable importance (RVI) of each predictor.

Results

Deviation from representation targets

The implementation of landscape reserve planning under the 2009 Order for this population of 87 reserve designs in the GBR resulted in deviation above ecological representation targets by a mean of 59% (SD: +/- 68%) and a median of 39%—a substantial over-representation of targeted attributes (Figure 5.2). Some of the LUs have LRDs with targets for ecological representation deviating by an average of over 300% of the required target value. Only one of the 87 LUs analyzed in this study deviates under the target values for the LU, and by only a small amount, but the report for this LU states that the design achieved targets, suggesting that small mapping discrepancies may be responsible for the apparent failure to reach policy targets.

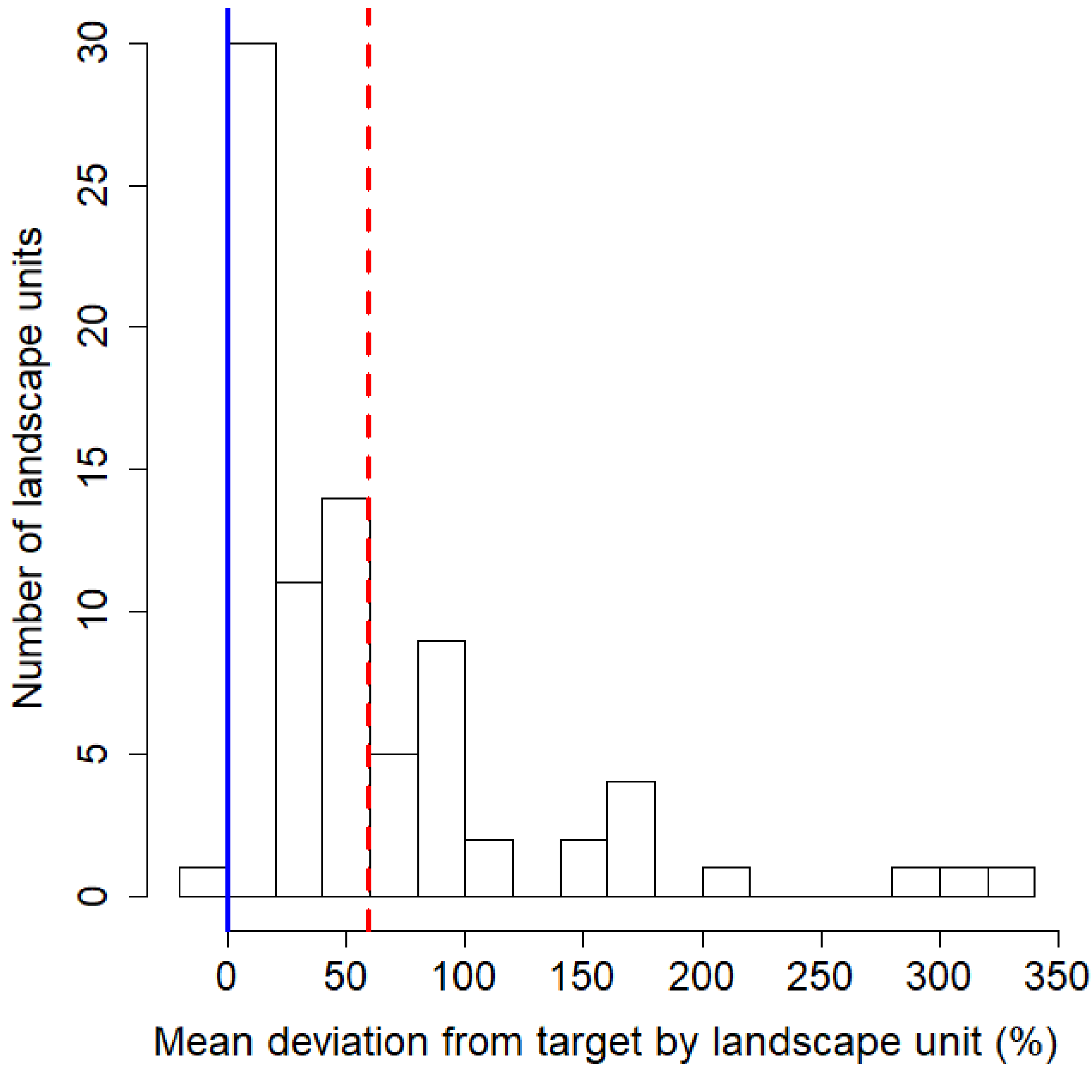

Figure 5.2 **Histogram of percentage deviation between representation targets according to policy guidance and the completed spatial Landscape Reserve Designs (LRDs). Deviations of 0% (blue solid line) represent the implemented LRDs perfectly achieving policy targets, while positive values represent the implemented LRDs exceeding targets. Across all landscape units, the mean deviation is 59% above target (red dashed line).**

Factors influencing the implementation of representation targets

All four predictor variables—Existing Conservation, Representation Target, Site Productivity, and Lead Professional—show distinct trends in the patterns of deviation between the ecological representation targets and the implemented LRDs (Figure 5.3). When the planning units for ecosystem types are associated with less productive forests, less ambitious representation targets, and targets with more area in pre-existing conservation areas, the completed design is more likely to exceed the *a priori* target by a greater amount. Planning units for ecosystem types requiring less than half of an ecosystem's area to be conserved (i.e. less ambitious targets) exceed their targets by an average of 34% more than those requiring more than half of an ecosystem's area to be conserved ($p < 0.01$; Figure 5.3a). Planning units associated with ecosystem types that have more than half of their area under pre-existing conservation constraints exceed their targets by an average of 27% more than those with less than half of their area under pre-existing conservation ($p < 0.01$; Figure 5.3b). Planning units associated with ecosystem types with low productivity exceed their targets by an average of 10% more than those with good productivity ($p < 0.01$; Figure 5.3c). There is also a high level of variability in target deviation across LUs depending on the professional leading the design process, with some individuals creating LRDs that on average deviate from targets twice the amount of other people (Figure 5.3d).

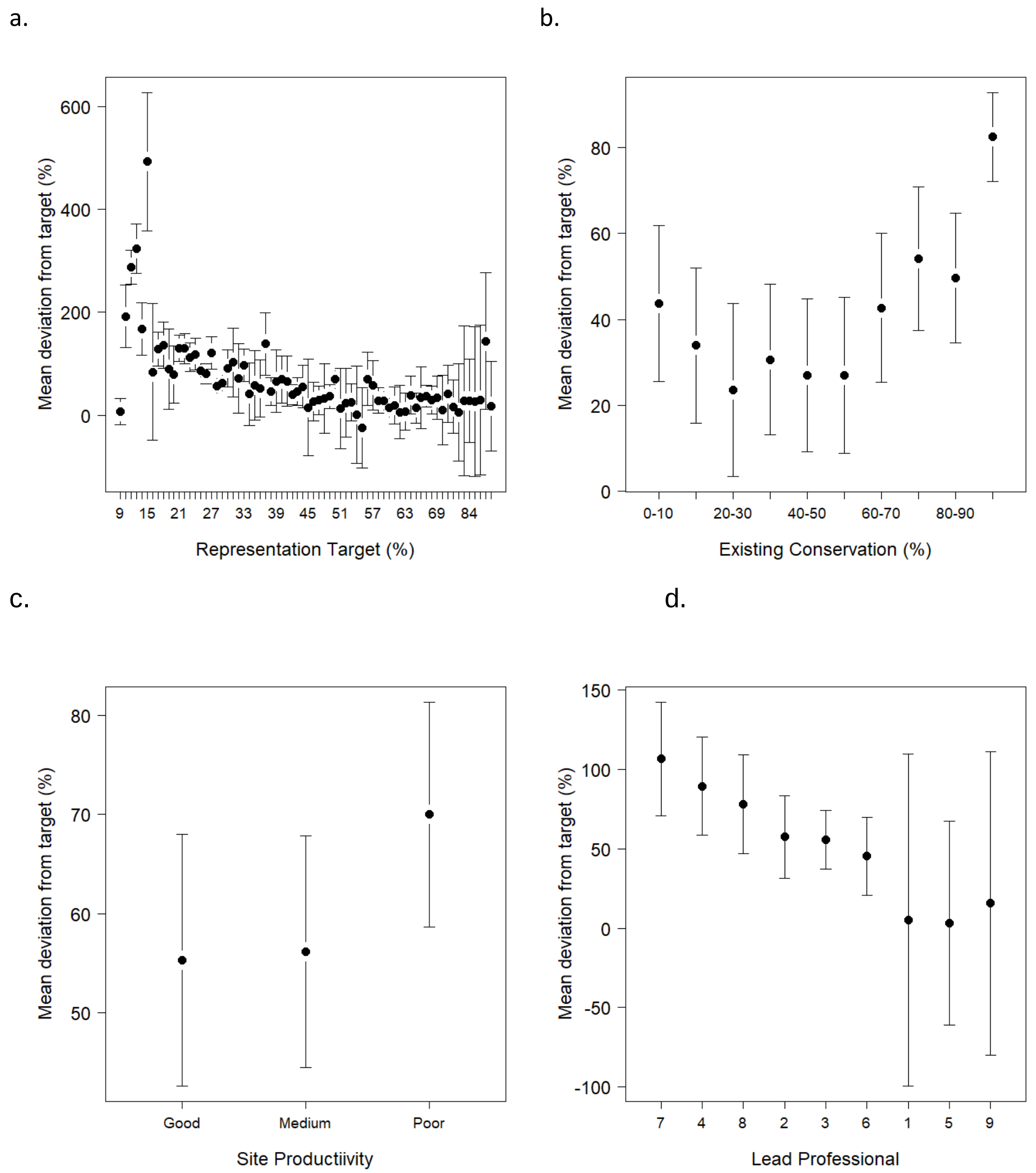

Figure 5.3 **Relationships between the mean deviation from the planning unit target and four predictor variables (a) Representation Target (%), (b) Existing Conservation (%), (c) Site Productivity, and (d) Lead Professional (confidence intervals shown around mean; see Table 5.1 for variable descriptions). Deviations of 0 on the y-axis represents the implemented LRD perfectly achieving policy targets, while positive values represent the implemented LRD exceeding targets.**

Our multivariate analysis suggests that all four predictor variables are associated with Deviation from Target (Table 5.2, Relative Variable Importance: Representation Target = 1; Existing Conservation =1; Lead Professional = 0.99; Site Productivity = 0.35). Specifically, the two models with strongest support contain either all candidate variables or all variables except Site Productivity. These two models perform equally well because of the very small differences in delta values, suggesting that Site Productivity is not necessarily needed to predict Target Deviation if the other three predictors are accounted for in the model. Our models with only individual predictor variables provide relatively weak evidence for predicting Target Deviation compared to these top performing multivariate models. When assessing just models with single predictors, Representation Target has the strongest support for predicting Target Deviation (delta is 56 points lower than the second-best model with a single variable).

Table 5.2 Strength of evidence for alternative models that test the effect of Representation Target (RT), Existing Conservation (EC), Lead Professional (LP), and Site Productivity (SP) on Deviation from Target. Each row represents an alternative candidate model, with the model structure characterized based on whether or not a plus (+) symbol is listed for a given predictor variable (i.e. a "+" indicates a predictor is included in the model).

RT	EC	LP	SP	adjR^2	df	logLik	AICc	delta	weight
+	+	+		0.34	52	-10738.9	21584.8	0.0	0.64
+	+	+	+	0.34	54	-10737.4	21586.0	1.2	0.35
+	+			0.33	44	-10752.2	21594.5	9.8	0.00
+	+		+	0.33	46	-10750.5	21595.3	10.5	0.00
+		+	+	0.29	45	-10800.7	21693.7	108.9	0.00
+		+		0.29	43	-10807.7	21703.5	118.8	0.00
+			+	0.28	37	-10815.8	21707.2	122.4	0.00
+				0.27	35	-10822.8	21717.0	132.2	0.00
	+	+		0.24	20	-10865.0	21770.4	185.6	0.00
	+			0.23	12	-10874.2	21772.5	187.7	0.00
	+	+	+	0.24	22	-10864.1	21772.8	188.0	0.00
	+		+	0.24	14	-10873.3	21774.7	189.9	0.00
		+	+	0.19	13	-10928.5	21883.3	298.5	0.00
			+	0.18	5	-10937.9	21885.7	301.0	0.00
		+		0.18	11	-10936.6	21895.4	310.6	0.00
				0.08	2	-11046.5	22096.9	512.1	0.00

Discussion

Variability in social-ecological systems influences the design of conservation areas

We found that completed spatial plans for conserving old growth forests in the Great Bear Rainforest surpassed the previously established policy targets. The average area captured across 87 landscape reserve designs exceeded the targets by 59%. Both human and ecological factors influenced this outcome. The variables Representation Target, Existing Conservation, Site Productivity, and Lead Professional all affected spatial planning to some extent, but Representation Target contributed most to the observed patterns of deviation from the a priori target. For each of these factors, the overall statistical relationships aligned with our hypotheses and collectively show the importance of accounting for different dimensions of social-ecological systems when predicting or assessing outcomes of conservation initiatives.

The challenge of meeting multiple objectives

Completed LRDs exceeded targets to a greater extent when the planning units for ecosystem types were associated with less ambitious thresholds for conserving old growth forests. This trend likely exists because the LRD process, similar to other conservation initiatives with broad sustainability goals (Leslie 2005), addressed objectives beyond just representation targets and forestry opportunities. Notable among these other objectives were conservation biology principles such as reserve connectivity and habitats for focal species, as well as Indigenous interests and cultural sites (Province of British Columbia 2009). There is obviously greater flexibility to integrate these types of objectives without exceeding targets when the targets require the capture of a larger proportion of each ecosystem type in LRD. As an extreme example, it would be much easier to achieve multiple conservation objectives and not exceed targets in a planning process where 99% of an ecosystem is targeted for reserve compared to a situation with a 1% target. Conservation targets associated with EBM regimes, such as the Great Bear Rainforest (Province of British Columbia 2016), Clayoquot Sound (CSSP, 1995), and the Northwest Forest Plan (Spies & Martin 2006) typically range from 30-70%—far higher than prominent national and international aspirational targets such as the Brundtland's Report's 12% (WCEED, 1987) or the Aichi Biodiversity Targets' 17%

threshold (CBD/COP5 2000) but not out of line with the “Nature Needs Half” movement that advocates for a 50% conservation target (Dinerstein et al. 2017). Our results thus raise concerns about whether it is reasonable to expect to concurrently meet multiple planning objectives and precise conservation targets at lower thresholds, especially in situations where targets are viewed as caps that should not be surpassed.

The effect of baseline conditions

Completed LRDs exceeded targets to a greater extent when the planning units for ecosystem types overlapped forested areas with more pre-existing conservation. This relationship is logical because areas that were previously committed to conservation had to be included in LRD as part of the design process, despite not being accounted for during target formulation. Some targets were already met or exceeded through existing land designations, especially those associated with ecosystems in landscape units with large Parks and Conservancies, so meeting or exceeding the targets in LRD was a necessary consequence. Even if each ecosystem’s conservation status had been accounted for when setting targets, protected areas established in the past were rarely protected with the same kind of ecological representation goals as were expressed in the GBR planning process (Lertzman & MacKinnon 2013). Hence existing protected areas might contribute to the statistics on total area protected without necessarily adding to the representation quota.

Accounting for the baseline condition of existing conservation areas is not just an important consideration in the GBR—it forms an explicit stage in the wider practice of systematic conservation planning (Margules & Pressey 2000). Some landscapes, regions, or jurisdictions, can achieve long-term policy targets and biodiversity objectives by maintaining the status quo, while others need drastic policy changes to achieve the same targets (Green et al. 2014). For example, achieving climate change targets is easier in places or during periods with low economic growth because less resources and energy are being consumed (Obani & Gupta, 2016). Meeting third-party standards for forest certification is also easier in places that already have strong government policies for stewardship because the standards might already be legally required (Suzuki & Olson 2008). Understanding potential tensions arising from heterogenous baseline conditions as early as possible in conservation planning, including during the setting of quantitative targets, will lead to fewer surprises during subsequent stages of design and implementation.

Accounting for economic opportunities

Completed LRDs exceeded targets to a greater extent when the planning units for ecosystem types were associated with less productive forests. The GBR's goals for human well-being include forestry opportunities and, in general, forestry prioritizes stands with higher site productivity due to this variable's relationship with timber yield (Chapter 4). Thus, it is not surprising that LRD planners limited the extent to which more productive components of the landscape were built into LRD relative to less productive areas. Avoiding LRD allocations in stands valued by the forestry sector is consistent with other planning processes that optimize reserve design by balancing biodiversity conservation and economic opportunity costs (Naidoo et al. 2006; Klein et al. 2008). It is also similar to the environmental gradients of many parks around the world, which were often created to reflect aesthetic factors and subject to constraints to minimize impacts on development opportunities, resulting in site selection geographically biased towards rock, ice, and other areas of low productivity (Pressey et al. 1993; Lertzman & MacKinnon 2013). However, emphasizing such socio-economic goals may conflict with ecological goals if representation targets, like those in the 2009 Order, are explicitly based on productivity classes. In these cases, the goal to conserve the continuum of site productivity classes present on the landscape is at odds with the goal to maintain the most productive sites for natural resource management. Using representation targets that are less tied to productivity and more directly connected to ecological structures and processes may help to balance, but not eliminate, trade-offs arising from competing ecological and socioeconomic objectives.

The influence of people in planning

The extent to which completed LRDs deviated from their targets varied with the individual leading the design process. The observed differences across planning processes may stem from interactions and correlations with other variables in our study, such as the geographic distribution of forestry company tenure holdings, which are inherently connected to specific lead professionals. But the human dimensions of EBM planning are also known to affect outcomes because practitioners hold a broad range of perspectives and make distinct trade-offs about how different social, ecological, and economic values should be prioritized across the landscape (McQuillan 1993; Rauscher 1999). In this study, we did not investigate how different social factors affected reserve planning, but scholarship in the social sciences suggests that variables such as the educational backgrounds of practitioners (e.g. foresters vs. biologists) or the institutional arrangements in which they work (e.g. government vs. private sector) might help to explain variation in outcomes (Agrawal 2001; Price & Leviston 2014). Although this broader mix of factors can influence the implementation of regulations to achieve targets, such human dimensions often receive less attention during planning processes than the more high profile quantitative targets for conservation areas (Hagerman & Pelai 2016).

What is a target? Implications of an upper cap on conservation

We examined the extent to which completed conservation plans aligned with policy targets, but it is also important to distinguish whether potential deviations from targets are good, bad, or simply context dependent. On the one hand, trends like those in our study, which show targets consistently exceeded, could be viewed as an implementation failure if regulations or policies explicitly considered targets as fixed thresholds that reflect upper caps on conservation. For example, though the concept has evolved over time to better account for economics and ecological processes, setting targets for harvest rates in forestry, fishing, and hunting based on maximum sustained yield originally reflected an idea that exceeding certain conservation thresholds may have negative consequences for the resource biomass over time (Binkley 1987; Mace 2001; Jenks et al. 2002). On the other hand, exceeding targets could be viewed as an implementation success if such thresholds were intended to function as a floor, or lower limit, for conservation. Targets to address climate change through reductions in greenhouse gas emissions are a good example of targets that jurisdictions are hoping to surpass (United Nations 2015). But this type of contextual policy analysis concerning targets was vague in the LRD process and is often lacking in conservation planning more broadly, leaving the determination of success or failure largely subjective. As Weaver (2009) notes, when policies allow flexibility, it is important to understand how much compliance is “good enough”— in our case, how much target deviation is acceptable. In our study, however, it is impossible to answer this qualitative question with only our quantitative data.

The idea that conservation targets should function as an upper cap on the amount of land set aside for conservation was built into more recent GBR regulations through the creation of Managed Forest targets (Province of British Columbia 2016; see Supporting Information). Targets such as these create a situation where reserve design cannot exceed representation targets and still comply with regulations. Formally allocating a dedicated percentage of the land base towards forestry objectives has been advocated by many actors in the forest industry for decades, argued from the perspective that these types of targets are widely used for achieving conservation objectives (Mathey et al. 2008). The ability to meet targets that function as caps on conservation while achieving other planning objectives is challenging, particularly when targets are generated without datasets and design principles that are critical inputs to the final spatial design. For example, though the LRD targets were originally established to reflect specific conservation priorities, the ultimate goals for these plans reflect a constellation of factors and diverse cultural and ecosystem services, so it is perhaps unrealistic to expect all these factors to be achieved with purely ecologically based thresholds. Employing conservation and forestry-based targets that sum to 100%, therefore, creates a dilemma between integrating these types of important datasets/principles and achieving targets—a tension that can greatly impede progress towards reserve implementation. In the cases we examined, applying targets for a managed forest may have precluded the professionals involved in LRD planning from solving the problems to which they had been assigned.

The reality of conservation planning across diverse social-ecological systems is that most decisions require difficult trade-offs, and outcomes from one stage of planning can have cascading effects across other stages. Hence, following the principles of adaptive management (Folke et al. 2005), it is important that initiatives consider conservation planning as an iterative and nonlinear process that ideally has the capacity to absorb new knowledge, such as the type of findings revealed in this study, without the entire process falling apart. Promoting this kind of adaptive resilience by creating flexible and dynamic planning processes while also remaining committed to pragmatic progress towards targets and on-the-ground implementation is challenging. But developing a clear understanding early in planning about the nature of targets, including how targets are intended to be interpreted and why they should be interpreted in this way, will create a more transparent and efficient process with more clarity about what constitutes a successful planning outcome. For regions like the GBR, where conservation activities are nested within an ecosystem-based management framework that includes equal standing for ecological integrity and human well-being, including forestry opportunities, such interpretations are particularly important because individuals and groups can have fundamentally different perspectives about these goals. Despite the many challenges arising from the use of conservation targets in our study area and criticisms of this approach more broadly (Hiers et al. 2016), the ongoing promotion of these benchmarks by scholars and governments (Dinerstein et al. 2019) suggests that they will remain a prominent tool for addressing many of the planet’s environmental challenges.

Acknowledgements

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Supporting Information

An evolving approach to landscape reserve design

Major changes to EBM in the GBR occurred in 2016 with the adoption of new provincial regulations for the region (Province of British Columbia 2016). This change caused the LRDs analyzed in our study to be discarded because they no longer met the new land use objectives. For example, these new regulations designated new large conservation areas that LRDs must incorporate, changed the unit of analysis for ecological representation, and introduced a suite of methodological changes to the LRD process. Perhaps most influential on LRDs, the new regulation also created three new sets of quantitative targets at the scale of the GBR and individual LUs that regulate spatial allocations for ecological representation, minimum levels of old growth, and managed forest. This broad suite of changes brings to the fore an important question: was the early LRD work worth the huge investment in time and effort given that the implemented spatial design only influenced forestry and conservation for a few years and now must be redesigned? The answer to this question depends in part on the extent to which the work already completed on these LRDs can inform the development of new LRDs that meet the 2016 requirements. The answer also depends on whether analyses of past planning processes such as these are used as a guide to improve EBM policies and on-the-ground management over time. As mentioned earlier in this chapter, evaluations like the one carried out in our study are rarely completed (Leslie 2005; Hagerman & Pelai 2016)

The variables assessed in our evaluation of the 2009 GBR regulations may still be relevant under the new 2016 regulations, but the context in which these variables operate has changed. First, the 2016 regulations have less variability in ecological representation targets across Landscape Units in the GBR, but landscapes in the southern plan area, where forestry opportunities are greatest, generally still have lower overall conservation targets. Thus, our results suggest the greatest challenge of meeting targets will be in these areas. Second, the amount of pre-existing conservation is also more explicitly accounted for in the three new legal targets, meaning that deviations could more closely relate to design decisions than the baseline conditions of pre-existing conservation vs. managed forest. Despite this new approach to targets, landscapes with more area under existing conservation may have less flexibility to integrate other types of EBM objectives because the representation target could already be met, and the legal managed forest target would not allow more area of that ecosystem to be added to LRD. Third, the unit of analysis for representation targets has changed from one based on productivity classes and stand species (i.e. Site Series Surrogates) to one more fundamentally focused on distinct BEC units (Site Series Groups). Hence the range of productivity classes or specific tree species may not be captured in LRD to the same degree as before, and the new LRDs have more flexibility to emphasize less productive sites and lower value species to enhance forestry opportunities. Finally, the methodology and framework for LRD is also more detailed than before and involves a broader group of participants, including more direct representation from Indigenous groups. In principle, these more heterogenous LRD teams for each LU could dampen the variability arising from the views of a single lead professional, though the diversity of groups across the GBR plan area will likely still be a factor that affects design.

Chapter 6. Conclusion

Implementing community-based research

The value and benefits of working with local communities to carry out academic research has become increasingly apparent in the last few decades (e.g. Salomon et al. 2018). Collaborating with place-based Indigenous communities, in particular, allows researchers to gain context-dependent perspectives and knowledge that is not readily available within academic sources such as the *Web of Science*. The spatial and temporal scale of this knowledge base is unique because it is often rooted in oral histories about a specific environment over centuries and millennia. Such detailed knowledge presents a huge opportunity for researchers to learn about social-ecological relationships, generate hypotheses for further exploration, and triangulate findings among different types of data—all of which can broaden our understanding of scientific theories. But designing studies based on research questions that are relevant and important to communities also helps to address practical, applied problems for those communities and bring forward local perspectives about sustainability that are critical for successful conservation outcomes (Dietz et al. 2003; Acheson 2006b; Enquist et al. 2017). For example, the findings from my research with the Na̲nwak̲olas Council on Large Cultural Cedar (Chapter 3) formed the basis of an operational protocol agreement with forestry tenure holders, which will help steward this important cultural resource over many generations.

This is not to say that community-based research is easy. Indeed, working with communities can take substantially more time than traditional approaches to academic research—for me, building relationships and trust with a community required an extended period of interaction and communication. My research over the past eight years involved dozens of community meetings, interviews with many knowledge holders, hiring local community members to assist in fieldwork, training of those workers, and discussions with decision makers. However, there can be a disconnect between the relatively slow pace of academic research and the relatively fast pace of real-world planning processes, because decision-makers need timely access to data and results. The full scope of data and findings from research with Indigenous communities may also not be appropriate for open or even academic access, especially when researchers collect and analyze culturally sensitive data. Two of my thesis chapters (Chapter 2 and 3), for instance, contain only a subset of the results of the companion reports and presentations developed for internal community use. In addition, I carried out a research project over the course of two years that focused on using cultural data in conservation planning, which, because of concerns about confidentiality of data, ultimately proved too difficult to translate into a thesis chapter that could be made available to external audiences.

Despite these challenges, the experience and insights that I gained working with Indigenous communities during my PhD were invaluable and should be viewed as a testament to the power of community-based research. I come away from my years at Simon Fraser University, not only with strong scholarly contributions that have been or will be published in academic journals, but also with a rich, multi-layered appreciation about the study systems in which I worked. If I had not embedded myself in these communities, I would never know, for example, that the BC coast has conservation areas delineated based on the habitat of Sasquatch or that certain watersheds remain unharvested because decades ago community members scared off loggers by hanging bloody goat heads from trees. I also would not have had the opportunity to carry out fieldwork in some of the most amazing ecosystems in the world or to cultivate so many friendships within these communities. For me, the richness of this learning experience has been multi-dimensional and goes well beyond the scholarly aspects of the work.

Furthermore, the reciprocity of this research has led to a variety of concrete, practical benefits for the communities with which I worked. These include spatial datasets that can support conservation planning, new polices and field manuals to steward cultural resources, and expanded capacity among First Nation members to carry out field surveys. I thus encourage more academics to collaborate with communities to gain these types of experiences and to ensure that solutions to theoretical and applied problems are meaningful to the actual people that must directly live with the outcomes.

Bringing together novel spatial methods and data sources

Understanding social-ecological relationships across space and time is key to effective forest management, whether this knowledge is used to support traditional

Indigenous systems or more western science-based conceptions of EBM (Lertzman 2009). In places like the GBR, which contain dynamic forested ecosystems and human communities that have co-existed over very long time periods, assessing datasets at appropriate temporal and spatial scales is particularly important. Throughout this thesis, I show that it is not only possible, but indeed often hugely beneficial, to incorporate different methods and types of spatial-temporal data, especially when they span distinct disciplines and sources of knowledge.

For example, in Chapter 2, I incorporate into species distribution models spatial occurrences of monumental cedar that were derived from both ecological field surveys and from archaeological records that have a longer temporal resolution. Bringing these disparate data sources together helps to predict the distribution of a cultural keystone species and shows how using traditional patterns of harvesting can help reconstruct distributions of long-lived species with rapidly shifting baselines. Of the research gaps identified in this chapter, perhaps none was more salient than the need to develop a refined understanding of cultural resources: in this case, I needed to talk with Indigenous carvers and other community members to develop a better definition for monumental cedar trees. Therefore, in Chapter 3, I worked with research partners to interview 13 traditional cedar carvers, the results of which I then coded into a set of morphological characteristics to inform the field identification of trees with different cultural uses. The findings from this study show that, although monumental cedar trees are generally rare across First Nation territories, certain growth forms such as those suitable for carving community canoes, are nearly extirpated from the land base. Although diversity within biological systems is widely recognized as key to supporting ecological resilience, this study highlights the importance of using data based in traditional knowledge to describe biocultural diversity.

Chapters 4 and 5 are not as explicitly rooted in community resource use and knowledge as Chapters 2 and 3, but these latter studies use datasets with a historical context that have broad implications for communities involved in EBM. In Chapter 4, for instance, I examine changes to the environmental conditions of harvested forests by building 47 separate species distribution models to test a uniquely long time series dataset. I show that harvesting over time is targeting forests with sequentially less productivity and accessibility and that policy interventions can disrupt these trends. Understanding changing baseline conditions, including trends that show a pattern of ‘logging down the value chain’, can affect how communities view opportunities for economic development and conservation. Chapter 5 extends some of these ideas about trade-offs among competing EBM objectives by assessing the implementation of landscape reserves to achieve conservation targets. I show that these reserves consistently exceeded their conservation targets and that both human and ecological factors affected reserve design. In that chapter, I also discuss broader ideas about the nature of targets, which are highly relevant to other disciplines and global initiatives that are attempting to solve sustainability problems with these types of thresholds. My thesis suggests that modern scientific approaches to EBM planning and analysis can be enriched and improved by incorporating data about landscape patterns with a historical or Indigenous context. It collectively demonstrates the strength of epistemological pluralism.

Supporting the theory and practice of ecosystem-based management

Implementing EBM is hard. Not long ago—within the industrial paradigm, at least—it was assumed that species and resources could be managed individually without thinking about connections among them or with the broader environment (Folke et al. 2004; Rajala 2006). It was assumed that planning could be aggregated and simplified across administrative units that cover entire regions without thinking about spatial scales like Indigenous territories, local communities, watersheds, or distinct types of ecosystems (Margules & Pressey 2000; Price et al. 2009). It was assumed that management is best undertaken by centralized governments and corporate stakeholders without worrying much about the values, perspectives, and interests of local people living within the landscapes (Agrawal et al. 2008; Larson et al. 2010). It was assumed that locations for conservation areas should only overlap geographical areas with low economic value and not consider dynamic social-ecological inputs into planning like the range or natural variability or traditional patterns of Indigenous resource use (Schwartz 1999; Branquart et al. 2008; McLain et al. 2013). A lot of things were assumed.

But now that scholars and practitioners are confronted with finding ways to support EBM by bringing together these types of complex ideas, planning often dramatically slows down (Rauscher 1999). Indeed, in places like the GBR, developing the framework and implementation details for EBM can take decades, often leading to significant burnout for governments, communities, and planners involved in the process. Is such a pace necessarily bad? Although it is certainly nice to achieve planning objectives in a timely fashion, given the significant departure from past eras of natural resource management, perhaps adequate time is simply needed to effectively transition to a new EBM paradigm. Furthermore, given that principles of adaptive management underpin EBM, people should expect planning and policies to change based on new knowledge. For these reasons, initiatives should probably avoid using terms like "Full Implementation of EBM" which was associated with planning milestones in the 2016 GBR regulations, because it signals that planning has a clear end date. Instead, EBM should embrace and operationalize ideas of adaptive management by planning for ongoing research, monitoring and iterative planning to address the truly dynamic nature of social-ecological systems.

Overall, my thesis provides a framework and empirical findings that can support this type of adaptive approach to management. I demonstrate the need for community-based research and the utility of using novel methods and datasets to highlight trends across meaningful spatial and temporal scales. My research shows the importance of using archaeological data to reconstruct past distributions of cultural resources for use in conservation planning; the importance of using traditional knowledge to categorize and estimate the abundance of cultural resources so that vague policy terms like "sufficient quantity of monumental cedar" can be translated into quantitative operational protocols; the importance of extending the temporal resolution of forest harvesting to understand how baseline conditions have shifted as a function of forest sector behaviour and policy interventions; and the importance of bringing lessons from past spatial planning into new processes. Having spent over a decade working as an EBM researcher and practitioner, however, I know from firsthand experience that science alone is not the sole force shaping the direction of EBM. Decisions are often made based on value judgements, including those that may not appear that different from past resource eras that reduced discourses to binary choices between the environment and logging. But as forestry stumbles towards sustainability, more research like that carried out in this thesis will be needed to provide pathways for shifts to a more dynamic model of EBM. The power of this kind of work is in its ability to help us understand the intersection among values, culture, history, and the modern environment.

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