Biocultural approaches to environmental management and monitoring: theory and practice from the cultural rainforests of Kitasoo/Xai'xais Territory

by

Bryant DeRoy
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Supervisory Committee:

Dr. Christopher Darimont (Department of Geography)
Supervisor

Dr. Christopher Bone (Department of Geography)
Committee Member

Dr. Iain McKechnie (Department of Anthropology)
Outside Member

Abstract

Biocultural approaches to Environmental Management (EM) and monitoring are an emerging strategy in sustainability planning. Unlike functional ecological approaches to EM, which exclude humans from ecological systems, biocultural EM approaches incorporate humans, communities and their values as integral part of ecological systems, and are grounded in collaborative processes that develop locally relevant management objectives and monitoring practices. Biocultural indicators are a key aspect of biocultural EM, providing links between worldviews, knowledge systems, agencies and institutions at various scales to guide and streamline implementation of management objectives. Although many Indigenous Peoples have been continually practicing biocultural approaches to EM for thousands of years, challenges exist in contemporary EM scenarios where multiple worldviews, political boundaries and knowledge systems collide. Some of the challenges or gaps in contemporary biocultural approaches are based in theory, and others are in practice. In Chapter One I highlight one of these gaps – the lack of guiding criteria to develop biocultural indicators in contemporary biocultural EM and monitoring. To address this gap, I propose a novel suite of six criteria (culturally salient, supportive of place-based relationships, inclusive, sensitive to impacts, perceptible, linked to human well-being) drawn from a case study in Kitasoo/Xai'xais Territory in the area now referred to as the North and Central Coast of British Columbia, Canada. In Chapter Two, I highlight a challenge in practice—the development of spatial models that incorporate a community-led approach. I show how this community-engaged approach benefitted the development and application of a landscape scale suitability model for culturally modified trees, a priority biocultural indicator. In conclusion, this theoretical and practical work identifies opportunities to amend existing Provincial and Federal legislation in order to support biocultural approaches to EM in Canada and shows how biocultural approaches may be applied in other social-ecological systems near and abroad.

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Introduction

The loss of biodiversity continues to increase despite efforts—from community or local projects to global initiatives—to curb or reverse this trend. Efforts to address anthropogenic impacts to biodiversity loss often take the form of Environmental Management (EM) and monitoring. EM broadly includes the development and implementation of policies and regulations that address issues such as biodiversity loss (Sutherland et al. 2004, Pullin et al. 2009). Monitoring is used to measure the success or failure of these policies and regulations to achieve desired EM outcomes over time (Sutherland et al. 2004, Stem et al. 2005, Pullin et al. 2009). In many cases, EM is directed at managing impacts from anthropogenic activities such as resource extraction. EM and monitoring initiatives have in many cases taken a 'functional ecological' approach to setting goals and targets, which seek to protect key ecosystem linkages and relationships that sustain ecological systems (De Groot et al. 2010). In general this approach to EM has not included humans as an integral part of ecological systems (Chan et al. 2012a, 2012b, Daniel et al. 2012).

The absence of social and cultural components in EM and monitoring has negatively impacted biodiversity conservation. Purely functional ecological approaches have specifically created barriers to EM implementation in systems where Indigenous Peoples have played an integral role in stewarding the environment for millennia (Sterling et al. 2017a, 2017b). For example, instead of recognizing and supporting existing Indigenous EM institutions and practices—which in many cases have been in place and supported ecosystem function for thousands of years—functional ecological EM is often guided by *ex situ* (provincial/state or federal) agencies and fails to align EM objectives with the values and knowledge of Indigenous Peoples and local communities (Caillon et al. 2017, Sterling et al. 2017a, 2017b). This can create cross-scale, cross-cultural and value-based conflicts that prevent EM outcomes (e.g., maintenance of biological diversity) from being achieved (Sterling et al. 2017a, Lyver et al. 2018).

One way to address issues of cross-cultural conflict in EM is to develop and support 'biocultural' approaches to EM and monitoring. Biocultural approaches to EM and monitoring acknowledge the role of people as participants in ecosystems by recognizing existing and interdependent linkages among social, cultural and ecological systems (Gavin et al. 2015, Lyver et al. 2018). Biocultural approaches can adapt to mobilize existing *in situ* or place-based methodologies, knowledge systems and values, which can facilitate socially and culturally just outcomes for EM (Maffi and Woodley 2010, Gavin et al. 2015). For example, in Aotearoa-New Zealand, major tenets of Maori biocultural stewardship are being incorporated into planning, current and future ecological restoration, EM and monitoring (Lyver et al. 2016).

Biocultural approaches can also draw from and employ existing methods and theory from functional ecological approaches to EM, such as the use of indicators. For example, ecological indicators ranging from the chemical contents of organic matter (Huang et al. 1979) to numerous invertebrate and vertebrate taxa have been used for decades as proxies for ecosystem function (Hilty et al. 2000, Carignan and Villard 2002, Niemi et al. 2004). In functional ecological EM and monitoring practice, these indicators can be managed for and monitored over time. The same concept of indicators can apply to biocultural EM. For example, biocultural indicators can include culturally significant species (such as food or medicinal plants) in addition to value- or perception-based indicators (such as well-being) (Cunningham et al. 2001, Garibaldi and Turner et al. 2004, Bennett et al. 2016, Sterling et al. 2017a). Indeed, many Indigenous Peoples utilize indicators as an integral part of their EM institutions and practice (Berkes et al. 2000).

Owing to the complexity of conflicting governance arrangements brought by colonialism in many parts of the world, it may be challenging to develop, implement and monitor biocultural indicators. For example, it can be difficult to reconcile the identification and prioritization of biocultural indicators where there are overlapping claims and worldviews from varying management agencies (e.g., First Nations governments, regional, state/provincial and federal governments). One potential challenge that may hinder the process of indicator development is a lack of guiding criteria that can facilitate the selection of indicators that are locally relevant, efficacious (capture other biologically and culturally significant species and relationships) and just (supportive of social and cultural priorities). The lack of guiding criteria is poses a greater challenge when diverse agencies, institutions, worldviews and priorities are engaged in the EM and monitoring process (Howitt et al. 2013, Sterling et al. 2017a).

Considering this challenge, in Chapter One I identify and address the lack of criteria for biocultural indicator development by using an inductive approach to highlight key themes that guided the development of biocultural indicators in a highly complex social-ecological system—the Great Bear Rainforest (GBR). I then conduct a literature review to assess if these themes have been used in other systems or EM approaches by drawing upon biocultural and functional ecological literatures to explore the utility of these themes outside of the GBR. From the GBR case study and literature review, I identify and describe six criteria (culturally salient, supportive of place-based relationships, inclusive, sensitive to impacts, perceptible, linked to human well-being) that can facilitate the development of biocultural indicators to support resilience in complex social-ecological systems.

In addition to the challenges influencing biocultural indicator development, barriers also exist in the implementation of EM. One major implementation challenge and opportunity facing biocultural EM initiatives is the development of locally informed spatial modelling resources that can effectively aid environmental managers and practitioners apply and monitor EM objectives on the ground (Angoletti and Rotherham 2015). In Chapter Two, I shift from theory to practice and show how diverse data sources and knowledge systems can be mobilized inform spatial resources as part of a partnership-based and community-led biocultural EM approach. Functional ecological EM has drawn upon spatial models for decades—for example, habitat suitability modelling has been widely used as a method to help environmental managers and planners visualize important habitat areas so that impacts to these areas can be mitigated (Rodríguez et al. 2007). Similar spatial modelling approaches are increasingly utilized as part of biocultural EM approaches (Scwartzman et al. 2000, Gaikwad et al. 2011, Gorenflo et al. 2012, Agnoletti and Rotherham 2015, Bond et al. 2019). However, it can be cost intensive to gather new locally-derived data and difficult to incorporate existing data and knowledge to inform biocultural spatial models.

In Chapter 2, I investigate these challenges and offer insight into how they are being addressed in biocultural EM approach led by the Kitasoo/Xai'xais First Nation, in what is now referred to as the Central and North Coast of British Columbia, Canada. I use an interdisciplinary and community-engaged spatial modelling approach to understand landscape patterns of an understudied biocultural indicator and archaeological feature —culturally modified trees (CMTs). This approach incorporates locally-developed survey methodologies, local and Indigenous Knowledge, high resolution remotely sensed data (Light Detection and Ranging [LiDAR]), as well as existing archaeological data from over 35 years of surveys. Culturally modified trees exemplify a biocultural indicator that achieves many of the guiding criteria described in Chapter One. CMTs are trees that show evidence of use in the form of characteristic scars from the harvest of bark, wood or sap by Indigenous Peoples (Garrick 1984, Moberly and Eldridge 1992, Mackie et al. 1993, Ostlund et al. 2003, Turner et al. 2009). CMTs are valued for their importance as living links between community members and their ancestors, in addition to being protected as archaeological sites in Canada (Turner et al. 2009). Kitasoo/Xai'xais community members selected CMTs as a priority biocultural indicator candidate for spatial modelling during community meetings that guide the objectives of the Kitasoo/Xai'xais Resource Stewardship Authority. The Kitasoo/Xai'xais First Nation is now implementing the CMT suitability models developed in Chapter Two as part of their spatial planning and survey approach to mitigate impacts cultural values in the context of commercial forestry.

This combined theoretical and practical work reveals how one approach to biocultural EM taken by the Kitasoo/Xai'xais — and themes that emerged from this work — can apply to other biocultural indicators in a diversity of social-ecological systems. This work and case study show how locally-derived spatial models developed with a community engaged approach can facilitate EM in complex geographies of coexistence (Howitt et al. 2013). The result of applying this approach and data in EM can provide enhanced protection of social and cultural values as well as biodiversity in the context of commercial resource extraction (e.g. commercial forestry).

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Chapter 1: Biocultural indicators to support locally-led environmental management and monitoring

ABSTRACT

Environmental management (EM) requires indicators to inform objectives and management action as well as monitor the impacts or efficacy of management practices. One common approach uses 'functional ecological' indicators which are typically species whose presence and/or abundance are tied to functional ecological processes, such as nutrient availability, trophic interactions and habitat connectivity. In contrast, and used for millennia by Indigenous Peoples, biocultural indicators are rooted in local values and enduring place-based relationships between nature and people. In many landscapes today where Indigenous Peoples are reasserting sovereignty and governance authority over natural resources, the functional ecological approach to indicator development does not capture fundamental values and ties to the natural world that have supported social-ecological systems over the long term. Accordingly, I argue that the development and use of biocultural indicators to shape, monitor and evaluate the success of EM projects will be critical to achieving ecological and social sustainability today. Herein I provide a framework comprised of criteria to be considered when selecting and applying meaningful and effective biocultural indicators in coastal temperate rainforests among the diverse array of potential species and values. I use a case study from a region now referred to as coastal British Columbia, Canada, to show how the suggested application of functional ecological indicators by the provincial government created barriers to the development of meaningful co-governance. I then explain how the Kitasoo/Xai'xais First Nation designed and implemented a biculturally-relevant suite of indicators in their own EM and monitoring processes. Drawing on my experiences working in service of the Kitasoo/Xai'xais Stewardship Authority and both the biocultural and functional ecological literature, I propose six generalizable criteria (culturally salient, inclusive, sensitive to impacts, socially just, perceptible, and linked to human well-being)

that can guide resource stewards in selecting locally relevant indicators to implement biocultural EM and monitor the performance of outcomes.

INTRODUCTION

Differences in indicators used in environmental management (EM) and monitoring often represent the interests and dominant knowledge sources associated with the governance system from which they emerge. Among many approaches to carrying out EM, which employ different indicators used in setting management thresholds and measuring performance or outcomes, one dominant approach is the *functional ecology approach.* Drawing on western science, and specifically population and community ecology, this reductionist approach seeks to maintain the linkages, processes and interactions that make up an ecological system (Dufrêne and Legendre 1997, Roberge and Anglestam 2004). Indicators used in functional ecology approaches are typically representative of certain bio-geo-climatic conditions, ecosystem characteristics or ecological processes (Caro and O'Doherty 1999, Gilby et al. 2017). Indicator species are often selected according to criteria that can evaluate the roles that a given species plays within its community (Siddig et al. 2016). For example, woodpeckers have been shown to be reliable indicators of local bird richness as well as overall health of forest ecosystems because they require large patches of mature forest and are habitat engineers for other species (Martin and Eadie 1999, Roberge and Anglestam 2006, Drever et al. 2008). However, there may be many different functional ecological indicator species for a given system.

In a functional ecology approach, a short list of indicator species is typically selected from the vast number of species that comprise a system to simplify management and monitoring efforts. 'Umbrella species' (Frankel and Soulé 1981, as cited in Roberge and Angelstam 2004) and groups of umbrella species are often used. 'Focal species' (Lambeck 1997) - also frequently used as indicators - are sensitive to environmental changes and have broad-scale or varied habitat requirements that encompass the needs of many other species and trophic interactions in the system (Lambeck 1997, Roberge and Anglestam 2004). For example, in the Canadian Rocky Mountains, carnivores have been proposed as useful focal species in determining ecological thresholds for EM because their individual niche characteristics and collective habitat requirements encompass those of many other co-occurring species (Carroll et al. 2001). These and other focal species comprise a form of functional ecological indicator that is commonly used to determine conservation goals and help set thresholds for minimum habitat requirements as well as to monitor and evaluate EM outcomes (Noss 1990, 1999, Roberge and Angelstam 2004). Criteria that are often used to determine the suitability of functional ecological indicators include endemism, impact of habitat area or patch-size on population viability (e.g. fecundity and survival), specificity of ecological processes that limit the distribution of individuals or populations, as well as their conservation status (Lambeck 1997, Caro and O'Doherty 1999, Carignan and Villard 2002, Coppolillo et al. 2004).

Although offering a potentially useful approach to measure impacts of land use, the focus on functional ecological criteria and indicators- particularly when driven by centralized state governments- has prevented local uptake and success in EM scenarios around the world. Functional ecological indicators are often developed by centralized management agencies (e.g. Provincial, State, or Federal governments) that have colonized or asserted decision-making authority over Indigenous territories, failed to accept guidance from local communities (Indigenous or non-Indigenous), and drawn on observational data that may not always be derived from the area affected by management decisions (Reed et al. 2006, Sterling et al. 2017*a*). Additionally, many consider that contemporary environmental management has not appropriately used scientific information in a way that fosters benefits to social, cultural and economic needs while also conserving biodiversity (Slocombe 1993, 1998, Sutherland et al. 2004, Turner et al. 2008, Cook et al. 2010, 2012, Chan et al. 2012*b*, Pullin et al. 2013, Artelle et al. 2014). This is partly due to the omission of social and cultural values during the development of ecological indicators used in management and monitoring (Turner et al. 2008, Chan et al. 2012*a,* 2012*b*, Sterling et al. 2017*a*).

In contrast to functional ecological approaches, biocultural approaches to EM projects - and their indicator development - typically start with values important to local governments, communities and stakeholders. Measures are developed based on *in situ* values (Garibaldi and Turner 2004, Maffi 2005, Maffi & Woodley 2012, Cuerrier et al. 2015, Gavin et al. 2015, 2018, Biedenweg et al. 2017, Sterling et al. 2017*a*, Artelle et al. 2018, Mcarter et al. 2018). For example, in the Western Province of the Solomon Islands a collaborative group of place-based researchers and community members are co-developing sustainability and well-being indicators based on values, perceptions and observations of community members resulting in indicators at various scales (e.g. habitat and species level) (McCarter et al. 2018). Biocultural approaches to EM foster human well-being and ecosystem integrity at the sub-regional scale, both major components of modern EM as well as global sustainability goals (Millennium Ecosystem Assessment 2005, Mascarenhas et al. 2010, Gavin et al. 2015, 2018, Hausmann et al. 2016, Bennett et al. 2017, Biedenweg et al. 2017, Sterling et al. 2017*a*, 2017*b*). Emphasizing their relevance and potential for governance resurgence by Indigenous Nations is the acknowledgement that biocultural approaches to EM have promoted social-ecological resilience for thousands of years (Berkes et al. 2000, Trosper 2002, Haggan et al. 2006, Atlas et al. 2017, Artelle et al. 2018). Despite this history, however, there is currently no overarching set of criteria to aid in the selection of biocultural indicators in today's complex resource management world in which scientific approaches and tools are available, and several governance arrangements often interact in managing the same landscapes (Indigenous, regional, state, federal, international).

Here I offer a practical and flexible suite of criteria drawn from my experience working in Indigenous-led EM and supported by literature on both biocultural and functional ecological approaches to indicator development that can steward desired outcomes in biocultural EM. I then show how biocultural indicators that exhibit these criteria can facilitate cross-cultural EM in complex geographies of coexistence (Howitt et al. 2013). As I detail below, these indicators can communicate in situ management priorities, as well as monitor and evaluate the implementation of EM. I develop this framework of criteria to help distil biocultural approaches to indicator development that foster socio-cultural resilience and well-being, while also promoting ecosystem integrity and biodiversity protection in the context of current resource extraction pressures. I illustrate these themes with a case study from Kitasoo/Xai'xais Territory in an area now also known as coastal British Columbia, Canada (Figure 2.1).

CASE STUDY

The nuances of environmental management implementation and co-governance in the Great Bear Rainforest

The Great Bear Rainforest (GBR) of British Columbia, Canada provides a model system to understand the value of applying biocultural approaches to promote socially, culturally and ecologically sustainable outcomes in EM. The Great Bear Rainforest is a complex geophysical landscape of archipelagos, fjords and mountainous terrain with a diversity of ecosystems from the hypermaritime rainforests to the drier interior montane forests. The terrestrial and freshwater environments have been enriched with marine derived nutrients dispersed by terrestrial and avian consumers as well as Indigenous Peoples from pulses of pacific salmon (*Oncorhynchus* spp.), pacific herring (*Clupea pallasii*) and eulachon (*Thaleichthys pacificus*) (among other marine species) for thousands of years (Gende et al. 2002, Brown and Brown 2009, Fox et al. 2014, Trant et al. 2016). The rich forested lands in the GBR are also considered to be a globally significant biodiversity refuge as well as a last remaining vestige of contiguous temperate old growth forest - valued for biodiversity, tourism, carbon storage and commercial logging (DellaSala et al. 2011, Lertzman and Mackinnon 2014).

The geopolitical and social/cultural landscape of EM is also extremely complex. First Nations have continually occupied and actively stewarded terrestrial and marine systems in the GBR for millennia (Trosper 2002, Haggan et al. 2006, Brown & Brown 2009). Prior to European colonization, which brought smallpox epidemics and deliberate cultural genocide resulting in rapid depopulation, hundreds of villages and camps (seasonal and permanent) existed along the coastline in the region (Cannon 2002). There is a rich diversity of language groups, with distinct cultural practices and lineages, comprised of different clans (Beck 2000). Indigenous laws, hereditary leadership and the potlatch system governed how, where and by whom resources were used and stewarded (Trosper 2002, Brown and Brown 2009). Deeply rooted social and cultural ties to places, species and interspecies relationships as well as traditional governance systems continue, despite the impacts of colonialism (Brown and Brown 2009, Artelle et al. 2018). Although never surrendering their titles to the Crown, First Nations' land and seascapes in this region also occur within the asserted boundaries of the Province of British Columbia and the nation of Canada. Ending decades of conflict regarding land use in these forests (namely commercial logging), recently legislated agreement exists among First Nations, the government of British Columbia, Environmental Non-Governmental Organizations (ENGOs) and private sector forest industry companies. However, the legal objectives established in the Great Bear Rainforest (Land Use) Order (hereafter LUO) (British Columbia Ministry of Forests, Lands, and Natural Resource Operations 2016) remain de-coupled from existing institutions and stewardship goals of some local communities.

The process by which indicator species were developed illustrates this disconnect. During negotiations that led to the GBR LUO (2016), the provincial government met with First Nations, industry groups, and ENGOs to select a suite of focal species that aimed to guide management targets to protect ecological integrity and biodiversity under the guiding principles of ecosystem-based management (EBM). The principles of EBM include maintaining viable populations of native species in their current range, including the diversity of native ecotypes in protected areas, maintaining the functioning of ecological processes across scales (e.g., nutrient cycles and disturbance regimes), setting management goals at appropriate time scales (longer term), and accounting for human use, interaction and occupancy of these areas (Grumbine 1994, Price et al. 2009). The resultant focal species habitat models were developed to set landscape-scale management targets. As I explain next, these targets, however, did not adequately represent the fundamental values and processes of in situ First Nations institutions (Price et al. 2009, Affolderbach et al. 2012).

Although the development of conservation targets in the GBR was officially a Government-to-Government negotiation between First Nations and the Province of BC, the balance of power was sometimes inequitable. For example, the Province of BC selected indicators that were outside of First Nations' Local Ecological Knowledge (LEK) and Indigenous Knowledge (IK). This put the two governments on very different playing fields in terms of what data they could contribute. Two such species were the marbled murrelet (*Brachyramphus marmoratus*) and the Northern goshawk (*Accipter gentilis*). Both are threatened species that are associated with mature forests and large trees, but are not commonly referenced in food, social or ceremonial practices or cultural beliefs among First Nations in the region (aside from the Haida) (Committee on the Status of Endangered Wildlife in Canada 2012). Although these threatened species are important ecologically, they did not resonate with First Nations' broader conservation priorities. Further discord occurred when Provincial biologists did not consider LEK or IK in the habitat suitability models for other focal species that *were* culturally significant (see Service et al. 2014 for an example with grizzly bear (*Ursus arctos*) distributions). Many of these plans were never fully implemented because of the imbalance of knowledge sets (between the Province and First Nations) created by the use of purely functional ecological indicators. The only focal species that has since been fully included in the land use planning process is the grizzly bear, which is a touchstone of cultural significance for many First Nations in the region. First Nations could supply their own datasets for critical (class A and B) grizzly bear habitat, which improved the existing habitat maps developed by the team of Provincial biologists and spatial analysts (Service et al., 2014). The opportunity for First Nations to provide their own datasets levelled the playing field in terms of information sharing, which facilitated co-governance.

Other disparities and disconnects between decision makers are a product of an EM scenario that emphasizes economic factors (market viability of yield and productivity for future tree harvest) rather than social or cultural factors. Although the GBR LUO allows for the opportunity to incorporate cultural values into land and resource management, the regulations that refer to schedules listing culturally significant values and species still place a higher legal protection for economic impacts. For example, 'aboriginal heritage features'- defined in the legislation as an "artefact, feature or site that is important to cultural practices, knowledge or heritage of a First Nation"- may be altered or removed if it is "required for road access, other infrastructure, or to address a safety concern and there is no practicable alternative" (British Columbia Ministry of Forests, Lands, and Natural Resource Operations 2016). More broadly, the chief forester at the Ministry of Forests, Lands and Natural Resource Operations (FLNRO) still sets the allowable annual cut (AAC- the maximum total volume of wood extracted yearly) for each forest district in the region without requirement for direct or transparent negotiation with First Nations (British Columbia Ministry of Forests, Lands, and Natural Resource Operations 2016). Although by definition and colonial law there is a power asymmetry, in the current political climate, some of these issues may be negotiated on a government-to-government basis. However, the bias in language used in the GBR LUO towards economic factors and the agendas of centralized colonial governments can contribute to 'knowledge disconnects' during implementation that prevent positive outcomes for sustainability and well-being in situ (Howitt et al. 2013, Sterling et al. 2017*a*, McCarter et al. 2018). Such disconnects have also been referred to as 'intercultural capacity deficits' - a lack in knowledge, understanding or acceptance of cross-cultural values, institutions and world views that prohibits intercultural communication and/or the performance of collaborative efforts (Allenby 2006, Turner et al. 2008, Howitt et al. 2013). Intercultural capacity deficits create a major barrier to success in co-governed EM arrangements globally (Howitt et al. 2013).

Insights from the Great Bear Rainforest can inform similar EM scenarios with overlapping governance structures and a diversity of world views. When the provincial agencies developed EM protocols, agendas and resources that did not incorporate Local or Indigenous Knowledge (Price et al. 2009, Service et al. 2014) or formalize support to protect long established ties between social and ecological systems, they disregarded existing EM structures that have been in place for thousands of years (e.g. the potlatch system Trosper, 2002, Turner et al. 2013, Artelle et al. 2018). Similar errors have occurred in many intercultural EM scenarios and are contributing to negative outcomes for biocultural resilience as well as biodiversity conservation globally (Maffi 2012, Howitt et al. 2013, Cuerrier et al. 2015, Sterling et al. 2017, McCarter et al. 2018).

DISCUSSION

Lessons from the Great Bear Rainforest and beyond

In BC and abroad, some First Nations have developed or are developing values-led management plans to implement EM and bridge these gaps. These management plans provide a tool that enables in situ agencies to communicate goals and priorities across cultures to bridge knowledge disconnects. For example, in both British Columbia and New Zealand, First Nations have developed written management plans that are guided by Indigenous law and cultural values that have always governed stewardship practices in those territories (Borrows 2005, Artelle et al. 2018, Kitasoo/Xai'xais 2018). In some cases, these management plans are considered living documents that allow for continual development. The resurgence of values-led approaches to EM is helping to reinforce critical components of social and cultural well-being, such as relationships or connections to place (Artelle et al. 2018).

The Kitasoo/Xai'xais First Nation (among others) are implementing their own strategy for applying EM. Their goals are being developed by the community as well as contributing to broader goals outlined in the GBR LUO. For example, the Kitasoo/Xai'xais are conducting cultural feature inventories to learn more about the distribution and abundance of culturally significant values on the landscape in efforts to measure and mitigate impacts from forestry. The Kitasoo/Xai'xais model their program after the Haida Nation, which negotiated a different agreement with the Province of BC (Council of the Haida Nation 2010), to implement a biocultural approach focused on ground-based efforts to identify cultural and heritage values before they become impacted by forestry. The Kitasoo/Xai'xais have developed inventory methods and standards (inspired by those developed by the Haida Nation) to survey for and identify culturally significant values and species in addition to threatened or endangered species and ecosystems. This process on which they are embarking provides an opportunity to implement biocultural EM in the context of modern forestry and realize locally-driven priorities on the ground. Drawing on my experience in this process and foundational concepts from literature on both functional ecological and bicultural indicators, I next offer insight into designing appropriate indicators for biocultural monitoring– an important starting point for locally-driven EM.

Emergent criteria to evaluate the utility of biocultural indicators

To implement management plans and actualize their benefits, appropriate indices are required to guide objectives and measure outcomes. Biocultural indicators can offer a culturally-relevant and comprehensive approach for communicating values and priorities across cultures to promote social, cultural, economic and ecological resilience (Gavin et al. 2015, Sterling et al. 2017a). However, the process of defining appropriate biocultural indicators has not been extensively developed (Sterling et al. 2017a). There are many methods for developing indicators that involve community input (e.g. surveys, interviews and focus groups); there are few resources available, however, that can help guide this selection process to filter appropriate indicators. These criteria provide a way to distil locally relevant indicators and empower local communities to protect in situ values before they are damaged. Although the selection of biocultural indicators is necessarily place-dependent, common themes can be derived from literature on biocultural diversity conservation. The set of criteria I propose in Figure 1.1 is intended to guide the selection process of indicators that represent local values, facilitate cross-scale linkages, and provide effective measures to evaluate biocultural EM outcomes.

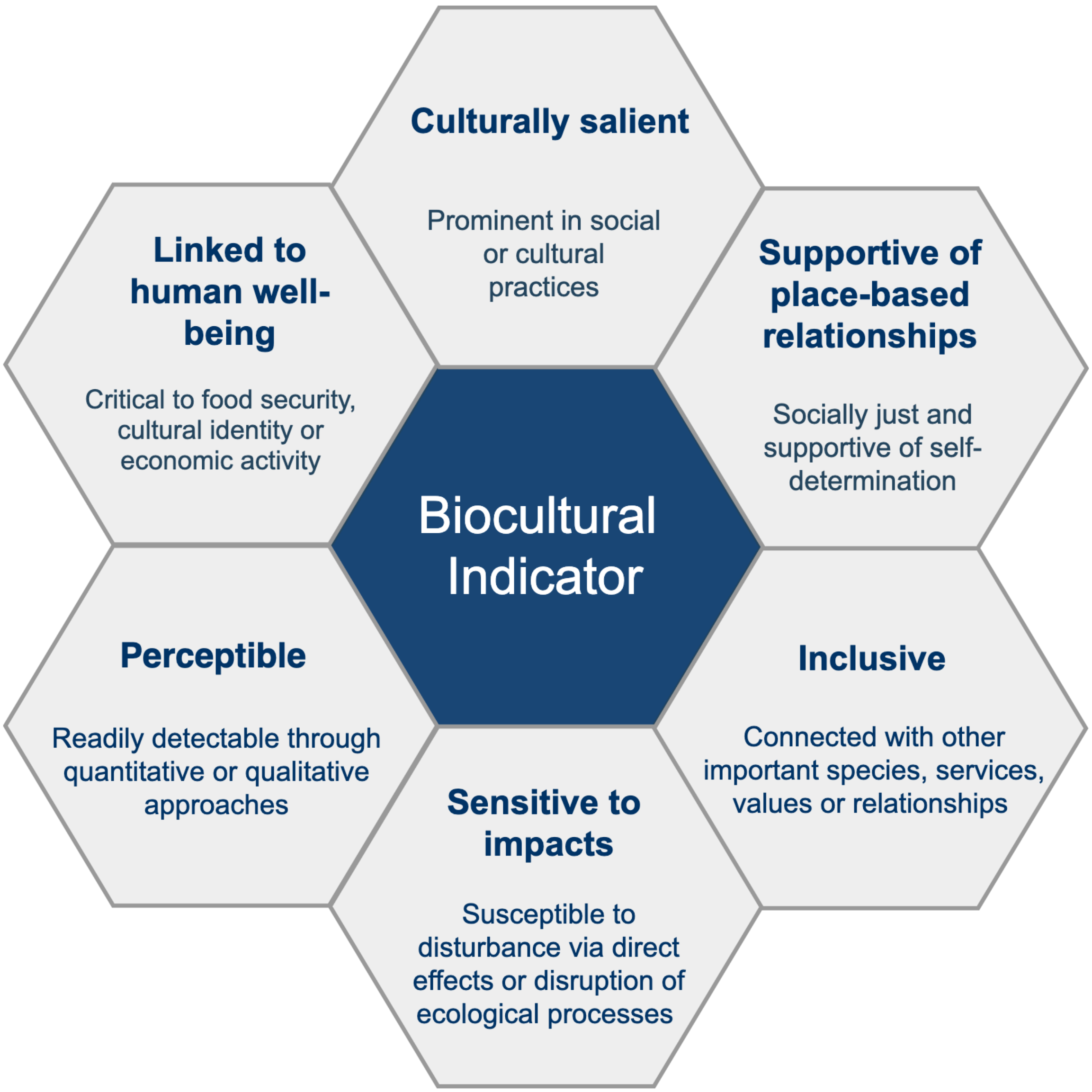

Figure 1.1: Six criteria proposed to guide development of bicultural indicators (culturally salient, sensitive to impacts, inclusive, perceptible, linked to human well-being, supportive of place-based relationships). These criteria are drawn from a case study in Kitasoo/Xai'xais territory in British Columbia, Canada and supported by literature in Canada and abroad.

To refine these themes into criteria I used an inductive approach to identify the criteria in Figure 1.1, which emerged as foundational concepts from my experience working in Indigenous-led – in this case, the Kitasoo/Xai'xais First Nation - biocultural stewardship. These criteria reflect key concepts that enabled progress in a highly contentious and complex governance process. I acknowledge that these criteria emerged from my experience working with one First Nation among thousands. I note that the process in other parts of the world where different governments have asserted competing claims on Indigenous Peoples' lands and waters will likely be different, perhaps radically, than that experienced by the Kitasoo/Xai'xais First Nation. Complementing my personal interactions while working in service of the Kitasoo/Xai'xais Stewardship Authority and written Kitasoo/Xai'xais management plans, I also searched the biocultural and functional ecological indicators literature to ascertain if—and if so, how—these concepts have been applied more broadly. I illustrate below how all of these criteria reflect core components of an Indigenous EM paradigm in which the Kitasoo/Xai'xais also mobilized western knowledge paradigms to implement biocultural EM.

Biocultural EM is place-based and therefore I offer these criteria as a synthesis of the concepts that enabled the implementation of biocultural EM in Kitasoo/Xai'xais Territory. However, I suggest their relevance as a tool to reduce conflict and guide collaborative conservation outcomes in other geographies of coexistence, particularly where western science approaches to stewardship historically or currently overlap spatially with Indigenous Knowledge-based approaches. I recognize that purely Indigenous Knowledge-based EM approaches may differ widely from the approach taken by the Kitasoo/Xai'xais. It is also critical to recognize that a fundamental component of this case study is that it was Indigenous-led. Knowledge integration has been and can be a damaging process when led by *ex situ* agencies due to the political nature of EM and power asymmetries (Nadasdy 1999, 2003). Ex situ agencies (e.g. colonial governments) have a history of ascribing unequal value to different knowledge sources or attempting to conform other knowledge sources to fit a western science lens, which can further remove decision-making power from in situ agencies (Nadasdy 1999, Bohensky and Maru 2011). Therefore, it is paramount that context, politics and decision-making power are deeply considered prior to developing biocultural indicators in line with these criteria (Nadasdy 2003, Howlett et al. 2009, Takeda and Røpke 2010, Bohensky and Maru, 2011, Moore and Tjornbo

2012). In Canada and other commonwealth nations, biocultural indicators developed in line with these criteria may offer an opportunity for engagement around reconciliation if colonial government involvement is permitted by Indigenous Nations and if the process is Indigenous-led.

In the GBR conflict existed from the start of negotiations because ex situ agencies (i.e., the Province of BC, industry, and ENGOs) led the process of developing indicators and conservation targets. Meanwhile, First Nations in the region were continuously monitoring other biocultural indicators (e.g. herring, salmon, eulachon and grizzly bears). The example of the grizzly bear as being the only indicator species to be co-implemented successfully by First Nations and the Province of BC illustrates how a) vital it was for conservation outcomes that the process was locally-led and b) how the following criteria could have guided the Province (and other agencies) to engage in a manner that supported local priorities and existing institutions. With this in mind, the following criteria offer a means to communicate priorities between in situ and ex situ actors as well as outline how the process of indicator development can and should be locally-led. The conceptual framework that these criteria create is not prescriptive or intended to be a step-by-step approach, but instead are offered to provide conceptual guidance while recognizing that processes will develop differently in different territories. Robust biocultural indicators may fully realize all six criteria; however, it is likely that all six criteria will not apply everywhere. Since the criteria emerged from a case study - where the Kitasoo/Xai'xais led a biocultural approach to EM that drew on both Indigenous Knowledge paradigms and western science paradigms - some criteria may not be applicable in purely IK-driven biocultural EM. I also acknowledge that I am not Indigenous and that my background is in western science, which might bias my understanding of this complex governance environment. The criteria presented in this paper do not speak for any Indigenous Nation but instead reflect themes that are clearly defined by the Kitasoo/Xai'xais as priority values that guide their approach to EM and have been shown to improve EM and enable co-governance when ex situ agencies are willing to accept them. Although this paper and the criteria were born out of collaboration with the Kitasoo/Xai'xais Stewardship Authority, I do not represent the Kitasoo/Xai'xais Nation or any other First Nation.

Cultural saliency

Cultural saliency provides a necessary starting point to biocultural indicators. Such saliency is a common theme in biocultural diversity projects (Maffi and Woodley 2012) and is central to the concepts of cultural keystone species (Garibaldi and Turner 2004) and cultural keystone places (Cuerrier et al. 2015). Garibaldi and Turner (2004) and Cuerrier et al. (2015) define elements of cultural influence that may form the best available definition of cultural saliency. Those elements include the extent of use in food, social, symbolic or ceremonial practices, use as a seasonal indicator, persistence in comparison to cultural change and the resistance to substitution or replacement (Garibaldi and Turner 2004). An example of a culturally salient species in the Great Bear Rainforest is the grizzly bear, a species that is entrenched in cultural practices for a number of coastal First Nations in British Columbia (Housty et al. 2014, Artelle et al. 2018). Practices such as stories and ceremony reinforce concepts of relatedness, respect and reciprocity between humans and grizzly bears in these systems (Housty et al. 2014, Artelle et al. 2018). These practices strengthen socio-ecological relationships and the resilience of human-bear systems (Clark and Slocombe 2009, Artelle et al. 2018).

Supportive of place-based relationships

Biocultural indicators should also protect features that establish provenance of occupation or use and protect values that reinforce place-based relationships, such as archaeological sites (Sterling et al., 2017*a*). Much land in the Great Bear Rainforest, as well as other regions of the world, has never been formally ceded by Indigenous Nations. Frequently, if a Nation were to pursue land claims through the colonial courts system, they need extensive evidence of occupation or use. Archaeological sites such as culturally modified trees (e.g. Figure 1.2) (as well as ancient settlements, pictographs, petroglyphs or resource use sites etc.) are often the only remaining visible/tangible evidence of human presence on the landscape above the soil surface (Oliver 2007, Earnshaw 2017). These physical markers may also indicate sacred or important places and spiritual or social-ecological interactions, which can contribute significantly to local identity and sense of place (Östlund et al. 2002, Stedman 2003, Oliver 2007, Harwood and Ruuska 2013, Cuerrier et al. 2015). Less tangible markers such as place-based names and stories may indicate a strong connection to place, which can also be sensitive to impacts from management practices (Kaltenborn 1998, Williams and Stewart 1998, Hausmann et al. 2016).

Linked to human well-being

Indicators should also have a strong positive impact on human well-being, which can often be linked to ecosystem health and the functioning of ecological processes (Sterling et al. 2017*a*). A positive impact on human well-being can be realized in many different forms (Chan et al. 2012*a*). For example, some indicators may foster connections to place or in other cases bring about financial or nutritional benefit. Species and values that both contribute to economic opportunities and also are strongly linked to identity or cultural/spiritual fulfillment can be robust biocultural indicators. For example, seagrass has been considered to be a fundamental contributor to human wellbeing in coastal systems around the world because of the cultural services and ecosystem services various seagrass species provide (Cullen-Unsworth et al. 2014). By providing foundational habitat structure, seagrass meadows can be direct sources for food, medicine or economic security and are strongly linked to a lifestyle or culture that is coupled to spiritual fulfillment (Cullen-Unsworth et al. 2014).

Inclusive

Biocultural indicators should also be inclusive - representative of a multitude of ecological and cultural values/relationships. Inclusiveness in the biocultural indicator development context encourages the co-location of content (e.g. critical ecosystem and cultural system components) with context (e.g. human relationships to place as well as ecological connectivity/integrity/diversity; Dale et al. 2001, Lertzman and Mackinnon 2014, Ens et al. 2016). Through promoting overlapping or intertwining values, inclusive biocultural indicators may compound the effectiveness of stewardship objectives and present more holistic measures of success (Gavin et. al 2015, Sterling 2017b). This criterion not only builds on the idea of using umbrella species, but also incorporates fewer tangible components of social-ecological systems. An example from the GBR of a species that is strongly inclusive is the western redcedar (*Thuja plicata*), which has been considered a cultural keystone species of Pacific Northwest First Peoples (Garibaldi and Turner 2004). Cultural uses of western redcedar trees require specific morphological characteristics, which are associated with different stand structures. For example, stands of trees that contain characteristics such as a large diameter, straight grain and sound bole may be considered as 'monumental trees' that are suitable for cultural practices (e.g. canoe building, totem carving, or mask carving (Council of the Haida Nation 2010, British Columbia

Ministry of Forests, Lands, and Natural Resource Operations 2016, Sutherland et al. 2016, Benner et al. 2019). Due to the structural complexity in stands containing monumental cedar, these stands also provide valuable habitat for many other species as well as gene and carbon stores (DellaSalla et al. 2011, Lertzman and Mackinnon 2014, Sutherland et al. 2016). In some cases, these stands may also contain culturally modified trees that provide archaeological evidence of historic occupation and land use in addition to a direct connection between community members alive today and their ancestors (Cuerrier et al. 2015, Earnshaw 2017, Benner et al. 2019). These characteristics of western redcedar make for an indicator that exemplifies the pairing of content (habitat and valuable materials) and context (cultural ties to place).

Sensitive to impacts

Indicators should be sensitive to land management impacts. In other words, the biocultural indicator must be demonstrably affected by impacts to ecological processes that are directly linked to land management practices (Dale and Beyeler 2001, Lindemeyer et al. 2001, De Groot et al. 2010, Siddig et al. 2016). This requires that given biocultural indicators are strongly correlated or causal to specific ecological processes or they themselves are the targets of extractive industries. In a best-case scenario, there would be little lag time between anthropogenic disturbance (e.g. land use practice or pollution) and perceptible effect (Dale and Beyeler 2001). This requires the appropriate matching of temporal and spatial scale between the selected indicator, potential impacts and desired management goals (Niemi and McDonald 2004). A species that responds to impacts through a lagged effect on ecological processes due to a spatial or temporal mismatch may indicate a significant decline in ecosystem function only after it is too late to change management practices. For example, the marbled murrelet on the Central and North coast of British Columbia are not robust EM indicators because they may exhibit heterogeneous nesting ecology in these mountainous regions, not only relying on old growth trees (subject to logging) but also cliffs and talus slopes (not at risk) for nesting locations (Barbee et al., 2014). Additionally, murrelet habitat is partitioned between terrestrial nesting habitat and oceanic foraging habitat, which has made measuring impacts from terrestrial land management (e.g. nesting habitat loss and fragmentation) difficult to interpret - although it is clear that this species is impacted by cumulative effects (Committee on the Status of Endangered Wildlife in Canada 2012).

Perceptible

As the sociologist William Cameron (1963) reminded us, “Not everything that counts can be counted, and not everything that can be counted, counts.” Some cultural values may be impossible to measure through western science quantitative approaches, but this quality should not exclude them from monitoring efforts (Satterfield et al. 2013). Other approaches have long guided Indigenous Peoples, and there is little reason why such long-held practise should cease (Berkes et al. 2000, Moller et al. 2004, Haggan et al. 2006, Ens et al. 2015, 2016, Waller et al. 2018). Biocultural indicators can be measured through quantitative or qualitative approaches or a mixture of the two and draw from diverse methodologies. For instance, the collective long-term observations of the decline of Dungeness crab (*Cancer magister*) by First Nations fishers in coastal BC as well as Indigenous-led empirical studies and the application of Indigenous law led to closures of a commercial fishery and rapid recovery of Dungeness crab in the region (Frid et al. 2016). Biocultural indicators should be relatively easy to monitor continuously or routinely over long periods of time (Dale and Beyeler 2001, De Groot et al. 2010). Despite being frequently chosen as umbrella species, rare or cryptic species may not be readily measurable and thus do not make robust biocultural EM indicators—unless there are reliable methods that have been developed to efficiently and accurately detect and monitor them. For example, eDNA is an emerging technology that may make surveying for some rare and cryptic species more feasible and reliable (see Mächler et al. 2014 for an example of rare macroinvertebrates). Perceptibility or measurability should also not be confined to estimates of abundance. For instance, quality or similar characteristics of indicators can be assessed over time. Eckert et al. (2018) described how Indigenous fishers experienced declines in size of rockfish (*Sebastes* spp.) over several decades of commercial and recreational overexploitation in coastal British Columbia.

Challenges

As EM planning and policy continue to adapt and change to facilitate sustainable outcomes for social-ecological systems, there remain major challenges to the implementation of these policies. As a result, many EM projects get stalled in the implementation stage - arguably the most important component of the EM process. While each EM scenario is unique and requires a custom implementation plan, there are some common barriers that emerge in the literature and from my experience working in locally-led EM. I outline below some of these barriers as well as potential solutions to overcome these pressing challenges.

Urgency matters

The number of indicators required to adequately inform and monitor biocultural EM is place dependent as well as time dependent. The importance of urgency increases as systems become more degraded (Dahl, 2012). There is added urgency to develop and implement biocultural indicators, which are underrepresented in sustainability indicators globally (Dahl, 2012). In systems where management impacts have the potential to affect non-renewable values, such as cultural heritage, there is a need for indicators that can be assessed rapidly and accurately to inform EM (United Nations Educational, Scientific, Cultural Organization 2003, Cuerrier et al. 2015, McCarter et al. 2018, Sterling et al. 2017). For example, a long history of commercial logging in coastal BC that has targeted high quality cedar has precipitated a decline of both monumental cedar and Culturally Modified Trees (Guujaaw 1996, Turner et al. 2009, British Columbia Ministry of Natural Resource Operations and Ministry of Forests, Mines and Lands 2011, Earnshaw 2017, Benner et al. 2019).

Different scales of time, space and institutions matter

In addition to the urgency in many systems there is a need to understand better the local effects of longer-term ecological processes that are product of both local and global impacts. The impacts to some cultural values may be compounded by short-term and longer-term environmental change as well as local and global drivers of change (e.g. land use practices and pollution) (Sterling et al. 2017a, McCarter et al. 2018). For example, yellow-cedar (*Cupressus nootkatensis*) - another culturally significant tree species - is a target of commercial logging due to its high market value; it is also susceptible to declines from climate change (Krapek and Buma 2015, Oaks et al. 2015, Oaks 2018). Concordantly, there is need for indicators that provide information to inform EM as well as monitor long-term impacts. In some cases one indicator may be able to achieve both goals (Dale and Beyeler 2001). Growing evidence suggests that by fostering a locally-led or -guided approach, robust biocultural indicators may provide a hub upon which cross-scale linkages can be made and where in situ and ex situ agencies can engage to improve the outcomes of EM for social-ecological systems (Sterling et al. 2017a, McCarter et al. 2018).

Resources available for monitoring

Given that scale is a major factor in determining how different indicators may affect EM, there is value in categorizing issues that may be affected by changes at different rates. In other words, given that monitoring is expensive, triage might be necessary. Additionally, threatened biocultural indicators may require resources for monitoring more immediately. For example, pacific salmon are subject to cumulative effects from overharvesting, climate change, and terrestrial land use practices and have recently become a priority for increased monitoring resources in BC (Price et al. 2008, 2017, Department of Fisheries and Oceans Canada 2019). One strategy is to build processes to monitor indicators that are already in place and have good baseline data. It is also much easier for local agencies to monitor species that they are already engaged with (e.g. taking fin clips for genetic stock assessment while harvesting salmon). However, long term monitoring requires long-term financial and political commitments, which can be incongruous with short political cycles.

One solution to this wicked problem may be to look to and support Indigenous governance systems and institutions that have upheld long-term stewardship goals over millennia (Trosper 2002, Artelle et al. 2018). Systems such as hereditary leadership and the clan-based tenure system have long guided stewardship goals through the generations (Trosper 2002, Brown and Brown 2009, Salomon et al. 2015). Hereditary leaders are entrusted to place-based decision-making power for life to uphold Indigenous law and manage their allotted territories. This type of leadership is also evidence-based, whereby leaders have to publicly demonstrate the continued productivity of their lands and waters or risk losing their title or rank (Trosper 2002, Salomon et al. 2015). This accountability to the public reinforces the values-driven management of resources for sustainable harvest (Trosper 2002, Brown and Brown 2009, Salomon et al. 2015). In the context of co-governance, Gavin et al. (2015) also advocate for a focus on relationship building and partnerships to increase institutional diversity and capacity where the rights of all parties are fully understood, upheld and respected. I suggest that if these approaches are locally-led, then institutional diversity may also increase the resilience in these systems to waxing and waning support from short term colonial ('democratic') cycles of elections and governments.

How many indicators?

The number of indicators required to achieve desired outcomes in biocultural EM will vary as a function of system dynamics, scale and project scope. All indicators that meet the above criteria should be considered in biocultural EM. However, fewer indicators that are highly inclusive and can capture the fundamental interactions- from drivers of change to impacts on biocultural values- may be more effective than many indicators with narrow scope (Landres et al. 1988, Levett 1998, Hagan and Whitman 2006, Tallis et al. 2010). Fewer are generally better; every new indicator brings the need to measure and monitor, which can be costly and time consuming. Prohibitive complexity can become an implementation issue if there are too many indicators to monitor (Tallis et al. 2010).

Considering that fewer highly-inclusive indicators may facilitate desired outcomes in EM better than many less-inclusive indicators, it could be argued - especially if monitoring resources are constraining - that indicators with strong links to cascading effects should be given priority. Cascading effects not only alter complex trophic interactions but also deeply impact local communities. For example, both the initial collapse of sea otter (*Enhydra lutris*) due to overharvest from the fur trade on the pacific coast *and* the subsequent reintroduction of sea otter (in conjunction with colonial mismanagement of shellfish fisheries) into areas where they were extirpated have caused negative impacts to Indigenous communities (Estes and Palmisano 1974, Salomon et al. 2015). Such linkages and indirect effects can be extremely difficult to manage. However, shifting towards precautionary management practices guided by biocultural indicators in industries that have known impacts on socially and culturally significant species and relationships (e.g. the high percentage of bycatch of eulachon in the shrimp (*Pandalus* spp.) trawl fishery) could be one strategy to mitigate these less visible impacts. Supporting resurgent management practices - in some cases that are based on reciprocity - could be another strategy to build resilience in complex social-ecological systems (Salomon et al. 2015, Artelle et al. 2018).

Sense of place: a robust biocultural indicator

One example of a biocultural indicator that can satisfy all six of the criteria is sense of place. Sense of place can be a vital component of cultural identity and well-being as well as a holistic indicator of social or cultural resilience for place-based cultures (Cheng et al. 2003, Bennett et al. 2016). Sense of place is built on the perceptions of in situ peoples and therefore numerous methodologies exist to evaluate those perceptions. For example, qualitative methods and Indigenous methodologies offer approaches to incorporate such seemingly intangible indicators in EM. Where the standard functional ecological toolkit may guide managers to focus on quantitative measures of biomass or a rare ecotype to set management targets, the evaluation of local peoples' perceptions can accurately assess the impacts that a given activity has on peoples' sense of place (Stedman 2003, Rogan et al. 2005, Fry et al. 2009, Bennett et al. 2016, Caillon et al. 2017). In a co-governance or Indigenous-led land use decision-making process, evaluating and respecting the perceptions of local people can be a method to prevent desecration of culturally salient spaces and place-based relationships as well as provide an opportunity to strengthen relationships among communities, governments and industry operators (Sterling et al. 2017*a*). Sense of place can be a highly inclusive indicator - encapsulating both social and cultural values/relationships as well as habitat and ecological relationships (Cheng et al. 2003). Considering how sense of place performs when evaluated against the six criteria (culturally salient, supportive of place-based relationships, inclusive, sensitive to impacts, perceptible and linked to human well-being) it has the potential to be a robust biocultural indicator.

In addition to likely encapsulating the six criteria, sense of place can also illustrate how the criteria may be flexible to a variety of methodologies and ways of knowing. Perceptions – with a special focus on how they can monitor change over time as a function of proposed or tangible changes to the environment – can identify precautionary thresholds for sense of place (as well as other biocultural indicators) or inform management action in relation to longer term processes for which other data sources are lacking (e.g. climate change; Johannes 1998, Moller et al. 2004, Leonard et al. 2013, Sterling et al. 2017a). Apart from the perceptions of local peoples, there are other methods of monitoring change or impacts to sense of place. For example, analytical tools and observational data may help identify, define, reinforce and communicate the boundaries of finer scale culturally significant areas that may or may not be currently known to local communities. For example, if historical evidence of occupation and resource use is a value important to a First Nations community, then culturally modified trees and groups of culturally modified trees may themselves constitute culturally salient spaces (e.g. Figure 1.2) (Harwood and Ruuska 2013, Cuerrier et al. 2015, Earnshaw 2017, 2019). Other methods such as *Values Mapping* can communicate existing relationships between people and place (Mclain et al. 2013). Participatory approaches to values mapping have been used to identify and define 'special places' for consideration in resource management decision-making on the Olympic Peninsula in Washington State (Cerveny et al. 2017). These examples show a variety of approaches to monitoring sense of place; I suggest that a key factor in the success of these approaches and others is that the process is locally-led.

Figure 1.2: Two large culturally modified western redcedar trees with healing rectangular bark strip scars in Kitasoo/Xai'xais territory. Photo: Bryant DeRoy

In addition to accommodating different methodologies and ways of knowing, sense of place can further illustrate how challenges of scale can be considered and ideally overcome. For example, in cultural landscapes there may be cultural keystone places at a variety of scales, (e.g. important mountains, valleys, bays, estuaries, caves, stands/groves, village sites, individual petroglyphs or trees). In some cases these places can be cumulative - the sum of nearby places as well as the space in-between can contribute to a sense of place (Cuerrier et al. 2015, Lepofsky et al. 2017). For example, the Whanganui River in New Zealand plays an important role in the culture of various Māori groups, and recently gained legal status to personhood by the parliamentary government of New Zealand (New Zealand Government 2016). The legal designation attributes personhood not only to the river itself, but also as “an indivisible and living whole, comprising the Whanganui River from the mountains to the sea, incorporating all its physical and metaphysical elements” (New Zealand government 2016). Sense of place can be multifaceted - incorporating a variety of different typologies of values (e.g. biological, recreational, historical, future, spiritual, aesthetic, economic and cultural among others) or defined by one key value (Brown and Reed 2000, Cerveny et al. 2017). This flexibility enables a custom and local approach to address the relevant priorities of communities in biocultural EM.

Limitations

The goal of this synthesis paper is to bring together key themes that have emerged during while working in locally-led and Indigenous-led EM as part of the Kitasoo/Xai’xai cultural feature inventory program. I used an inductive approach because many of the six criteria presented in this paper have been clearly defined and used by some First Nations in the area now referred to as the Great Bear Rainforest. These criteria took shape through Indigenous laws as well as EM and monitoring practice (e.g. management plans and applied stewardship). Although the criteria presented in this paper emerged from my experience working with one First Nation among thousands globally, I have demonstrated how these criteria are flexible enough to apply outside of the Great Bear Rainforest. Indeed, the criteria I offer also align with those detected in my informal review of related biocultural and functional ecology EM literature. I note, however, that each EM scenario should be treated as unique and other locally-led or Indigenous-led approaches may seek to develop their own criteria and indicators based on their own terms.

CONCLUSION

Increasing global resource demands create new challenges in designing management approaches that can provide sustainable and socially just outcomes in EM. One aspect that can address some problems that are common in co-governance or resurgent EM arrangements is to support the development of indicators that acknowledge, represent and uphold local values and relationships to the land. Applying a biocultural approach to indicator selection in cross-cultural EM provides a more inclusive option for stewarding both biological and socio-cultural diversity in comparison to purely ecological indicators and can bridge knowledge gaps in intercultural EM systems to facilitate co-governance (Maffi 2005, Howitt et al. 2013, Service et al. 2014, Sterling et al. 2017a, Sterling et al. 2017b). By proposing a suite of six potential criteria that emerged from this case study and supported by a diverse body of literature, I suggest characteristics that contribute to the utility and effectiveness of potential indicators in highly complex social-ecological systems. The set of criteria developed here may offer a tool to distil and communicate local priorities and promote stewardship outcomes that support biocultural resilience. These criteria should be considered a starting point for discourse around developing locally relevant indicators in a wide range of cross-cultural governance scenarios.

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Chapter 2: Indigenous Knowledge and Remotely-Sensed Data to model landscape suitability of a biocultural indicator: culturally modified trees

ABSTRACT

Environmental Management (EM) and monitoring initiatives must reconcile the need to incorporate social and cultural components with objectives for biodiversity protection and stewardship. Community- or Indigenous-led approaches to EM and monitoring enable local values—including social and cultural ties to the environment—to guide the development and implementation of EM objectives. These approaches can draw upon local and Indigenous knowledge and existing stewardship practice to carry out place-based approaches to EM and monitoring issues. In many cases existing Indigenous EM and monitoring approaches are already engaged in addressing the same issues that other agencies and governments are also attempting to manage. Conflict can arise when different agencies assert different EM and monitoring approaches or goals that apply to the same spatial area without collaborating with or supporting existing local or Indigenous EM goals and institutions. In Canada and abroad, some Indigenous Nations are in the process of developing resurgent EM institutions and practices—that may have been disrupted—to address contemporary EM issues. Resurgent EM approaches may vary in the types of knowledge or values that guide them. The Kitasoo/Xai'xais, among other First Nations in what is now referred to as the Central Coast of British Columbia, are developing and practicing resurgent EM in their territory. As part of this resurgent EM strategy, the Kitasoo/Xai'xais Stewardship Authority is conducting biocultual diversity surveys to understand more about the distributions and relative abundance of culturally significant values in their territory. A key component of this resurgent EM regime is the incorporation of biocultural indicators—culturally significant values—used as management targets that can be measured and monitored via diverse methodologies (e.g. culturally modified trees (CMTs)). Working with a community-based team, I draw on diverse data sources and knowledge systems to develop and refine a locally informed suitability model of unexamined areas to aid in the stewardship of this culturally significant and legally protected value. I find that the incorporation of cultural predictor variables—informed by local and Indigenous Knowledge—are among the

most influential in predicting the suitability of known CMT occurrence. This research highlights how a community-engaged and community-led approach to EM combined with survey and geospatial data can improve the development of management tools—specifically spatial suitability models—and the outcomes for EM.

INTRODUCTION

Biological diversity is strongly linked to social, cultural and economic resilience. Increasingly there is a need to combine these dimensions in environmental management (EM) and monitoring projects to achieve sustainability goals, reverse global declines in biological diversity, and promote resilient cultures and communities (Lertzman 2009). Biocultural approaches to EM offer an efficacious method to guide EM and monitoring objectives in a way that helps achieve these outcomes (Pungetti 2013, Gavin et al. 2015, Sterling et al. 2017*a*). Biocultural approaches seek to protect and steward linkages between humans and ecological systems by supporting locally-led EM objectives and by adopting a partnership-based approach to communicate these linkages across spatial and institutional scales (Gavin et al. 2018, Sterling et al. 2017a, Caillon et al. 2017, Wali et al. 2017).

Despite this promise, EM and monitoring often involves complex planning processes that develop targets in the context of legislation or policy at various spatial and institutional scales. These targets can be met by applying monitoring processes that focus on locally-important values, species and linkages among them (Gallagher 2008, Verschuuren 2012, Artelle et al. 2018). Success or failure in achieving targets is often measured through monitoring processes that can employ diverse methodologies, ontologies and knowledge systems (e.g. perception-based approaches or empirical science) (Newing 2010, Bennett et al. 2016, Caillon et al. 2017). One challenge facing cross-cultural or inter-governmental biocultural EM projects is designing planning processes that are compatible with the local social-ecological environment and explicitly incorporate humans and their values as part of ecological systems (Artelle et al. 2018).

A range of innovative approaches to EM and monitoring that are explicitly adopting values that reflect local peoples are emerging. For example, researchers in Australia have developed a management framework in collaboration with two Indigenous groups in the Kimberly region—

the Bardi Jawi and Nyul Nyul—that is guided by key Indigenous wetland stewardship principles (Pyke et al. 2018). This emerging framework has helped to transcend inter-cultural and inter-agency barriers to promote a guiding stewardship planning framework in which "wetlands need people" (Pyke et al. 2018). Similarly, the Omora Ethnobotanical Park in South America has incorporated ten principles based on Yaghan Indigenous Knowledge and stewardship practice to guide development of the Park's mission and to align the objectives among institutions involved, including the Yaghan and Chilean government (Rozzi et al. 2006).

Beyond guiding principles and establishing frameworks, spatial modelling offers a powerful tool as part of interdisciplinary, cross-cultural approaches to implementing biocultural EM. Spatial modelling has been a major part of implementing 'functional ecological' approaches to EM (i.e., those that do not include biocultural indicators) for decades (Franklin 2010, Elith and Leathwick 2009, Guisan et al. 2013). However, spatial modelling has had mixed success in achieving desired EM outcomes on the ground. Failures of spatial modelling in EM can often be attributed to error propagation in the models. This can be caused by use of *ex situ* occurrence data—observational data gathered outside the management area—or by inclusion of environmental predictor variables that do not strongly contribute to or limit the given species' habitat (Store and Kangas 2001, Franklin 2010). Therefore, spatial modelling in EM is more likely to help achieve desired management targets when occurrence data used to evaluate the model or predictor performance are locally-derived, and/or the model predictor variables are locally informed (Polfus et al. 2014), providing new opportunities to link local values with local data.

These linkages between locally-derived data and values can inform the selection of specific spatial modelling approaches. Many methods are available for EM application. Some methods employ complex algorithms and machine learning that often require intense data collection to parameterize models correctly. Other approaches are more flexible, allowing for the incorporation of local or expert knowledge where gaps exist in empirical data from the field or literature (Store and Kangas 2001). One method that is particularly well suited for the incorporation of local and expert knowledge in a suitability modelling framework is multi-criteria evaluation (MCE). MCE is a modelling process by which a suite of potential criteria or predictor variables that contribute to the suitability of a given outcome are empirically compared against one another to determine the relative influence of each predictor on the expected outcome (i.e., location of an important resource) (Eastman, Store and Jokimaki 2003). MCE methods have been used to incorporate local and expert knowledge as part of spatial modelling frameworks in a diverse array of applications, including decision-making models and habitat suitability modelling (Store and Kangas 2001, Polfus et al. 2014, 2016). MCE can serve as a useful and directly applicable suitability modelling approach in regions where prior empirical study is limited (Store and Kangas 2001).

The area now referred to as the Great Bear Rainforest (GBR) of British Columbia, Canada, is a region in which a shortage of empirical study exists for many species of EM concern. This region is comprised of complex socio-cultural and terrestrial biogeoclimatic systems that have been heavily influenced by thousands of years of stewardship practice by First Nations people. Indeed, First Nations in this region have been engaged in EM with many species of concern for over 14,000 years (Berkes et al. 2000, Mackie et al. 2018). Recent Provincial legislation that applies to the GBR (‘Great Bear Rainforest Land Use Objectives Order’ (GBR LUO 2016)) sets specific management targets for a suite of species and ecological and cultural values in this region. While some species and values that have conservation targets in the GBR LUO (2016) have been studied empirically in this region, others have not. Culturally modified trees (CMTs) are one example of a biocultural resource, a heritage feature within the GBR LUO, that is understudied in the region. CMTs may also act as biocultural indicators—providing indication of culturally significant areas that may also be biologically rich (Sutherland et al. 2016, Benner et al. 2019). CMTs fulfill many of the six criteria for biocultural indicators identified in Chapter 1. For example they are supportive of place-based relationships, perceptible through field survey, sensitive to impacts from commercial forestry, and their bark provides a material central to current and historical Kitasoo/Xai’xais cultural practice (Kitasoo/Xai’xais 2018).

Understanding the distribution of CMTs in the context of EM is a stewardship priority for the Kitasoo/Xai’xais, a sovereign First Nation in the area that is implementing resurgent EM guided by their Indigenous laws and values. The Kitasoo/Xai’xais is conducting territory-wide cultural feature inventory surveys with the intent to develop predictive models for the distributions of several culturally significant species and values (e.g. CMTs). These models will support the

Kitasoo/Xai'xais in directly managing these resources in the context of industrial forestry and other extractive industries in their territory. Although fieldwork for this research surveyed a number of species and values (Kitasoo/Xai'xais 2018), the analysis in this Chapter focuses on bark stripped CMTs, a priority indicator for EM based on community input gathered prior to this research. CMTs were chosen because of their value as markers of Indigenous heritage, occupation and stewardship with the potential to provide key evidence in rights and title claims, and their legal protection under colonial law (e.g., the *Heritage Conservation Act* 1996, Oliver 2007). CMTs may serve as a biocultural stewardship umbrella—similar to an umbrella species—to signal biocultural richness and protect less measurable or visible biocultural diversity (Garibaldi and Turner 2004, Turner et al. 2009). Further, despite ~35 years of culturally modified tree surveys in the region, there has been little effort to mobilize these data in support of EM (Benner et al. 2019).

Given this focus, I developed a partnership with the Kitsaoo/Xai'xais Stewardship Authority to apply spatial modeling approaches that link local stewardship values and data. The Kitasoo/Xai'xais Stewardship Authority is explicitly guided by community values through the process of open house community meetings, where community members contribute to and vote on the priorities and management objectives of the Kitasoo/Xai'xais Stewardship Authority. Specific research goals as identified by the Kitasoo/Xai'xais Stewardship Authority (with community input) for this project included: a) to evaluate potential landscape, ecological and cultural predictor variables that may contribute to the site suitability and distribution of past and present bark arboriculture; and b) to provide useful information to Kitsaoo/Xai'xais Lands Managers about where culturally modified trees are likely to occur, particularly in the context of commercial logging. The resulting suitability models will serve in a resurgent management regime that is revitalizing Indigenous law, mobilizing Indigenous and local Knowledge systems and interacting Western science approaches to guide EM decision-making and practice.

METHODS

Summary

I used a spatial multi-criteria evaluation (MCE) modelling framework that uses sensitivity analysis to assess the relative influence of predictor variables in predicting the suitability of recorded CMT occurrence. CMT data were derived from recent field surveys in addition to georeferenced archived archaeological surveys. Predictor variables were derived from highly processed Light Detection and Ranging (LiDAR) data and Indigenous Knowledge (i.e. expert opinion of cultural leaders), among other knowledge types and data sources. I chose the MCE modelling approach because it offered a straightforward method that could be shared with community research partners (with varying degrees of GIS experience) in a transparent manner. This approach enabled a deeper level of community engagement because it allowed me to receive feedback quickly during the model development process and also built trust in the final model among Kitasoo/Xai'xais Stewardship Authority staff (those who are implementing the model in EM).

Study Area

The study area is located on what is now referred to as the Central and North Coast of British Columbia, Canada, within the boundaries of the region recently designated as the Great Bear Rainforest. The area covers ~118,845 hectares (Figure 2.1) and is encompassed within a portion of Kitasoo/Xai'xais territory, spanning the temperate rainforests of the coastal mainland and a marine archipelago. Driven by this variation in geography, there are two major biogeoclimatic subzones within the study area including the very wet hypermaritime coastal western hemlock (CWHvh2) variant and the mild maritime coastal western hemlock (CWHvm1) variant (BC VRI 2018). Nested within these two major biogeoclimatic subzones, there are many different site series groups at the scale of a forest stand. Much of this region remains roadless, with the exception of isolated networks of forestry roads. The study area was delineated by the availability of LiDAR data in the region, which was used to calculate several of the predictor variables in the model. The LiDAR extent covers the majority of the spatial area where commercial forestry can take place in the region (i.e., outside of established Conservancies).

Figure 2.1 The study area is within Kitasoo/Xai'xais Territory, located on the Central and North Coast of British Columbia, Canada

Biocultural diversity field survey methods

I conducted a stratified sampling protocol, which sought to achieve equal survey effort across major biogeoclimatic gradients within the study area. I stratified our survey effort across both major subzones (CWHvh2 and CWHvm1) and ecological site series group types in the region (Figure 2.2). The surveys were also stratified at varying elevations, distances and orientations from ocean shorelines from 0 to 4.5 kilometers. Despite this stratification, roughly 95% of the study area is within 4 kilometers of the ocean, owing to the abundance of shoreline and our reliance on boats for transportation to survey starting points. In addition to boats, surveyors reached survey starting points via helicopters, mountain bikes, and hiking. Our survey crews spent 10 survey days at two different base camps accessed via helicopter in order to increase survey coverage in difficult to access inland and elevated areas. Once surveyors reached survey starting points, we conducted surveys by hiking predetermined transect routes.

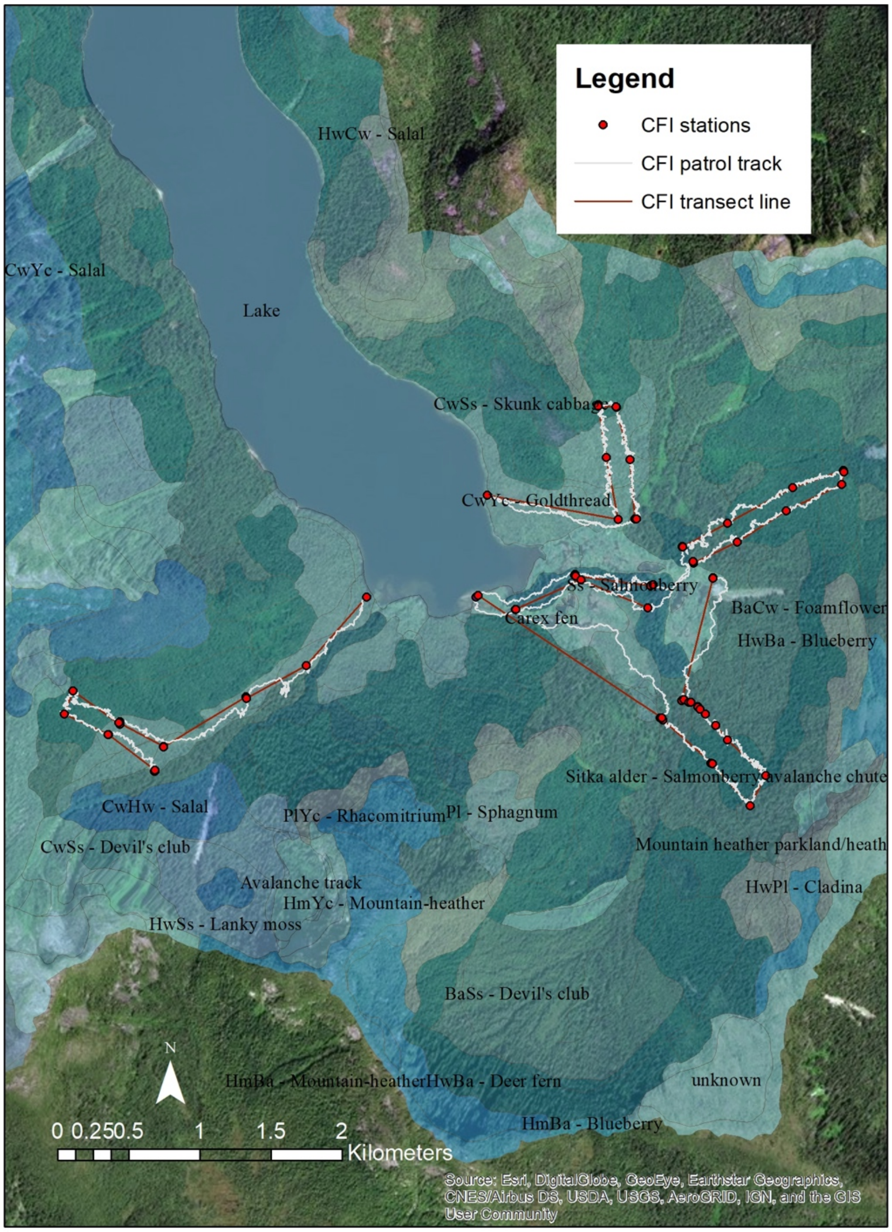

Figure 2.2 Cultural feature inventory (CFI) transects conducted in Kitasoo/Xai'xais Territory at different elevations, aspects, slopes and across site series groups.

The survey protocols followed those described in the Kitasoo/Xai'xais Cultural Feature Identification Standards Protocol (CFI SP)(Kitasoo/Xai'xais 2018). In brief, a trained crew of surveyors hiked transects recording CMT type, relevant measurements, location data, and photos of all encountered culturally modified trees (See Appendix B for example Datasheet). However, the transect design for the 2018 field season differed from those in the CFI SP because these surveys were not conducted in engineered cutblocks. Instead, to maximize coverage of different biogeoclimatic zones and site series groups, the transects were typically “out and back” surveys resembling a U shape, typically 500m-1500m out, 200m across, and then 500m to 1500m back (Figure 2.2). I did not use random start locations due to the cost and time required to access the sites. Therefore, I drew transects prior to surveying in order to maximize the coverage in different ecotypes, as well as different slope angles, aspects, elevations and distances from ocean, lake and major river shorelines. Our field surveys covered roughly 110 kilometers of straight-line distance and ~9600 meters in elevation.

Recording culturally modified trees from field surveys

Culturally modified trees come in many forms depending on what kind of resource was being extracted from the tree (e.g. bark) or the type of marker, symbol or spiritual meaning imparted. Culturally modified trees are classified into types and subtypes by archaeologists to aid in their identification and to help interpret their meaning on the landscape. CMT type include ‘bark-stripped’, ‘aboriginally logged’ and ‘other’. Subtypes can include, but are not limited to, ‘tapered bark strip’, ‘rectangular bark strip’, ‘planked’, ‘tested’, ‘notched’, ‘felled’, ‘sectioned’, ‘kindling’, and ‘arborglyph’ among others (British Columbia Archaeology Branch 2001). Each subtype of CMT has a variety of forms. The wide variety of CMTs can make them difficult to identify during field surveys, and some CMT subtypes may be less visible to the surveyor than others. For example, bark stripped CMTs that have had a portion of the bark removed are often striking features that still bear toolmarks from the harvester who removed the bark, depending on the date of modification (Earnshaw 2016, 2019). In some cases, very old bark strip scars may be visible on standing dead snags or the tree may have completely healed over the scar. Further, some CMT features such as tapered bark strips may resemble natural scars from wind damage among other causes (Eldridge et al. 1988, Garrick 1998, British Columbia Archaeology Branch 2001).

Figure 2.3 Two different types of healing bark stripped culturally modified trees on western redcedar (*Thuja plicata*), a tapered bark strip scar (left) and a rectangular bark strip scar (right) with surveyors Sarein Basi-Primeau and Stephen Neasloss.

Given the high degree of complexity in CMT identification, Lead Surveyors based all of the recorded observations on key principles for CMT identification. These principles were informed by the British Columbia Archaeology Branch (2001) *Handbook to the Identification and Recording of Culturally Modified Trees,* and through hands-on field training. Survey crews participated in fourteen days of field training in CMT identification led by two different experts—an archaeologist and a registered professional forester both deeply familiar with CMTs. From this training and use of the CMT Guidebook, I co-developed a tailored survey protocol that helped surveyors independently and consistently designate a CMT (or not) on trees that bore scars that did not exhibit all the key signatures of a CMT, such as stone or metal tool marks or scar crust. Part of this process included recording the context in which a recorded CMT was observed; for example, if there were other archaeological features nearby or there were observed conditions that may have caused natural scarring (see example Datasheet in Appendix B). Our survey crews also measured and recorded all of the feature characteristics (e.g. scar length and healing lobe thickness) following the guidelines outlined by the British Columbia Archaeology Branch (2001). All of these observations contributed to the confidence surveyors had in their observations, which was recorded as Low (possible), Medium (probable) and High (confident). I also co-developed a list of characteristics that would make the scar in question highly unlikely to be of cultural origins to reduce the risk of false identification. For example, if there is old outer bark on the scar face it is likely that the scar was of natural origin because bark stripping removes the innermost layer of bark. In addition, if the scar had numerous knots near the base it is less likely that the scar was of cultural origin because it can be extremely difficult to peel bark completely off the tree when there are many branches in the strip (Personal communication Nick Reynolds 2018). For this analysis I tested the models using only CMT observations that were bark stripped. These primarily occur on western redcedar (hereafter redcedar, *Thuja plicata*) and yellow-cedar (*Cupressus nootkatensis*). Bark stripped CMT subtypes include 'tapered bark strip' scars and 'rectangular bark strip' scars.

Georeferencing CMTs from previous surveys

In addition to the 2018 field surveys, I also incorporated bark strip CMT data (n = 511) from the Provincial archaeological database from the study area into the analysis. However, in order to utilize these data as point features I had to georeference 126 individual archaeological siteform maps. I assembled scanned copies of siteform maps from the Archaeological Branch of British Columbia's Rapid Access to Archaeological Data (RAAD) website and georeferenced them using Arcmap version 10.6. I used a minimum of 4 registration points for each map using a projected basemap with NAD83 BC Environment Albers projection and datum. A small number of hand drawn siteform maps that contained CMT features (n=6) were not included in this analysis due the high level of spatial imprecision. I added registration points until the map lined up with clearly defined landscape features. The scale of the georeferenced map was within +/-5m of the basemap scale. I then added points for all CMTs that were depicted on the sitemaps. I used RAAD to access the CMT tables associated with each siteform map and to identify the type and subtype and species (if available) of each CMT point.

Sample size and summary of occurrence data

2018 field survey data

Our survey leads recorded a total of 277 bark stripped CMTs during the 2018 cultural feature inventory surveys. Of those CMTs, there were 184 taper-stripped CMTs and 93 rectangular-stripped CMTs (Table 2.1). There were 185 bark stripped CMTs recorded with high confidence, 51 recorded with medium confidence and 41 recorded with low confidence. I incorporated all of the recorded bark stripped CMTs in the analysis because of the conservative nature of survey protocols and the precautionary approach taken during the analysis. The field recorded bark stripped CMT observations are primarily redcedar (n = 254) with a smaller proportion recorded on yellow-cedar trees (n=17). There were only six recorded observations of bark stripped CMTs on either western hemlock (*Tsuga heterophylla*) or sitka spruce (*Picea sitchensis*), which were not included in the analysis. All rectangular bark strips used in the analysis were found on western redcedar while taper strips occurred on both redcedar and yellow-cedar.

Table 2.1 CMT occurrences from both field surveys and georeferenced data (RAAD) included in the suitability modelling analysis.

CMT subtype	Field Survey (n)	RAAD (n)
Taper Strip	184	339
Rectangular Strip	93	172
Total	277	511

Georeferenced archaeological data

The CMTs that were georeferenced from the siteform maps accessed through the RAAD system included 339 tapered bark strip scars and 172 rectangular bark strip scars. The archaeological database in the study area shows a similar pattern among CMT species with ~318 redcedar bark stripped CMTs, ~16 yellow-cedar bark stripped CMTs and 3 bark stripped CMTs from other species (e.g. western hemlock). Not all observations in the RAAD system had information on the host tree species. 232 of the CMTs from RAAD fell within cutblock boundaries where commercial logging operations had either removed the recorded CMTs (via a site alteration permit) or left a small buffer around the recorded CMTs. I removed observations that fell within cutblock boundaries because I used a tree canopy height model as part of the analysis and the canopy height for these areas was altered by the removal of trees.

Final CMT occurrence data

Although there was a total of 788 bark stripped CMTs documented in the study area, I used the 556 bark stripped CMTs that occurred outside of cutblock boundaries—including the survey data and those georeferenced from RAAD due to the fact that these CMTs occur in unaltered forest stands. I then converted these data to a raster surface using the Point to Raster tool in ArcMap. The resulting raster surface containing 413 cells ($25m^2$) became the observation surface used to evaluate the suitability models. I assumed that our 254 observations from the 2018 cultural feature inventory surveys in areas that have not been commercially logged made up for any potential loss of spatial patterning from removing recorded CMTs where the host stand had been significantly altered.

Remote sensing

Remote sensing products were used during the development of seven out of the eight predictor variables that were incorporated in the models. I used Light Detection and Ranging (LiDAR) data in developing five out of the eight predictors and satellite imagery in three of the eight predictors. These data were collected, processed and shared via data-use agreement by Western Forest Products. Briefly, their methods used a full waveform sensor (model Riegl LMS-Q780) mounted on a fixed wing aircraft using a beam divergence of <0.25 mrad and a scan angle of +/-30° and 55-60% swath overlap with an absolute horizontal accuracy of 15cm (95% confidence) and an absolute vertical accuracy of 10cm (95% confidence). Raw LiDAR points were then reclassified into both a bare earth Digital Elevation Model (DEM) and a Crown Height Model (CHM). The DEM was derived from the mean value of all ground points within a one metre squared and the CHM was derived from the highest point value within each one metre squared.

In addition to the LiDAR data, satellite imagery data were for the development of the two predictor variables for the percent canopy cover of yellow-cedar and western redcedar. The Forest Inventory Branch of the Province of BC created the satellite imagery derived Vegetation and Resource Inventory geodatabase using photo interpretation methods with regional ground truthing (BC VRI 2018).

Predictor variables of CMT occurrence

I assembled a suite of eight predictor variables (elevation, a cost distance submodel, distance from known habitation sites, a crown height model, slope, aspect, and the percent canopy cover of yellow-cedar and redcedar) based on existing literature as well as local knowledge. These predictors were also selected based on the quality of data available, which includes LiDAR, satellite imagery and local and Indigenous knowledge (Table 2.2). Functions were selected to reflect the relative rate at which these layers become more or less suitable on the reclassified scale of 1-10 (see Appendix A). For example, tapered bark stripped trees (depicted in Figure 2.3) can be made on relatively small-diameter trees (between 30 and 40cms in diameter) with enough room to pull a strip of bark near two handwidths apart while leaving ~75% of the bark intact near the base of the scar (Garrick 1984, Arcas 1999, Brown and Brown 2009, Personal communication Simon Mason 2018). Rectangular bark strips are typically harvested from larger trees, which would be unlikely to occur in forests where the canopy height is under 10m (Moberly and Eldridge 1992). Since bark stripped trees can continue to grow for hundreds of years, I assume that even the tallest cedar can bear bark harvest scars (Earnshaw 2019). The suitability of the reclassified canopy height raster was rescaled with a logistic growth function using the Rescale by Function tool in ArcMap so that cells with a median canopy height under 10m were not suitable but trees with a canopy height over 10m became increasingly more suitable.

Table 2.2 Names, sources, and references of each layer that was used as predictor variables in the sensitivity analysis as well as a description of transformations that were used.

Layer Name	Source	Description of Layer Transformation	References
Cost Distance Submodel	Local and Indigenous Knowledge, LiDAR	Submodel made using canoe landing beaches as source cells and slope as cost surface	Gustas et al. 2017, Personal communication Vernon Brown 2018,
DEM	LiDAR	Elevation reclassified in 100m intervals, lower more suitable than higher	Arcas 1999
Habitation	Local and Indigenous Knowledge, Archaeological Record	Euclidean distance from known habitation sites in 1km intervals	Personal communication Vernon Brown 2018, Personal communication Blake Evans 2018
CHM	LiDAR	Rescaled by logistic growth function	Arcas 1999, Brown and Brown 2009
Slope	LiDAR	Rescaled by exponential function so that more suitable slopes are <60 degrees	Benner 2019, Arcas 1999, Tobler 1993
Aspect	LiDAR	Rescaled by symmetric linear function to favour north facing aspects	Stryd and Eldridge 1993, Personal communication Vernon Brown 2018
% canopy cover red cedar	BC VRI 2018 (Satellite Imagery)	% canopy cover of western redcedar- rescaled by logistic growth function. (higher % canopy cover = higher suitability)	Stafford and Maxwell 2008, Stewart 1984, Turner 2009
% Canopy cover yellow-cedar	BC VRI 2018 (Satellite Imagery)	% canopy cover of yellow-cedar- rescaled by logistic growth function- (higher % canopy cover = higher suitability)	Stafford and Maxwell 2008, Stewart 1984, Turner 2009

Slope

Slope was incorporated because it is a dominant landscape feature in the study area and it has been suggested in previous empirical study to have influence on the ability of harvesters to access cultural cedar. Slope was derived from the Light Detection and Ranging Data (LiDAR, methods below) data using the Slope tool in ArcMap. Slope was reclassified to a 1-10 scale in 10 degree increments where lower slopes were deemed more suitable than higher slopes. This layer was then rescaled using an exponential growth function based on Tobler's (1993) *hiking* function, which plots walking pace (or effort) in relation to slope angle from empirical evidence. In this case, slopes become gradually less suitable from 0 to 40 degrees and highly unsuitable above 50 degrees. This one metre resolution slope layer was then aggregated into 25m^2 cell size using the mean slope value.

Cost distance submodel

We hypothesised that other factors—in addition to slope—might impact cedar accessibility to harvesters and would play some role in the site suitability of CMT features. Specifically, local Indigenous Knowledge holders from the Kitasoo/Xai'xais Nation hypothesized that the distribution of currently visible CMTs (those made within the last ~250 years) were more likely to be in areas that were accessible from resource harvesting and travel corridors involving a combination of canoe and foot travel (Personal communication Vernon Brown 2018). Trail networks and other infrastructure (e.g. lake canoes) would have likely improved the ease of travel to inland harvesting and stewardship areas (Personal communication Vernon Brown 2018). This information, supplemented with historical text references from nearby First Nations Territories (Chittenden 1884), formed the conceptual framework for our harvester accessibility submodel. This conceptual submodel was used to parametrize a cost distance surface that reflected local and Indigenous knowledge as well as this historical evidence of pervasive infrastructure and inland landscape utilization by First Nations People (Equation 1a and 1b).

To create a cost distance surface that represented areas that were accessible via canoe I reclassified shorezone data from the Canadian Hydrographic Service (accessed 2018) to identify shorezones that could support canoe access and those that could not. Due to the large amount of variation in shoreline topography, we assumed that most shoreline types would accommodate some version of canoe-based access, the exception being cliffs. This classification includes rock shelves, boulders, cobble beaches, mud flats, sandy beaches as shorezone that could support canoe-based drop-off or landing. The cells containing these shorezone types were then used as source cells in our cost-distance submodel. I used the LiDAR derived slope predictor as the cost surface. Slope is a permanent and prominent landscape feature in the in the study area due to the mountainous topography. In order to simulate canoe travel on large lakes (>10ha) I used freshwater data from the British Columbia Freshwater Atlas to reclassify all freshwater bodies in the study area that are greater than 10ha to 0% slope (very low cost). Finally, the multi-directional cost-distance surface was calculated by using equations for both diagonal and perpendicular cost accumulation from cell to cell (Equation 1a and 1b).

Equation 1a: the formula used to calculate accumulative cost diagonally from the starting cell (cost1) to the next cell (cost2).

$$Accumulative\ cost = a1 + 1.414214(cost2 + cost3)\ /\ 2$$

$$Where:$$
$$a1 = (cost1 + cost2)\ /\ 2$$

Equation 1b: The formula used to calculate accumulative cost perpendicularly from the starting cell (cost1) to the next cell (cost2).

$$Accumulative\ cost = a1 + (cost2 + cost3)\ /\ 2$$

Percentage of canopy cover for western redcedar and yellow-cedar

To determine the relative availability of both western redcedar and yellow-cedar within a stand I derived a layer for each species that estimates their percent cover among all canopy tree species. These predictors were included because I assumed that stands with more available cedar would contain more suitable bark harvesting trees and would also be more visible to harvesters in the area (Personal communication Vernon Brown 2018, Benner et al 2019). These two layers were created by selecting all polygons from the Vegetation and Resource Inventory (VRI) database (BC VRI 2018) with either yellow-cedar or redcedar listed as one of the canopy species. The attribute table for this layer also contains percent cover of each species in each of the different canopy strata. I then summed the percent cover for both yellow-cedar and western redcedar in all canopy strata to obtain a total percent coverage value. The polygons in the VRI were derived by photointerpretation and have been ground-truthed regionally (BC VRI 2018). However, there are likely uncertainty in these data because of the province-wide scale at which the data have been extrapolated.

Cultural activity hubs

The distance from known habitation sites was included as a predictor variable because we hypothesized that known habitation sites, camps and long-term subsistence archaeological features indicate extensive cultural activity. This predictor pairs local and Indigenous Knowledge of landscape use with existing archaeological records. I combined spatial data from traditional use interviews with community members (conducted prior to the study in 2001) with recorded archaeological sites in the provincial archaeological database that had at least one indicator of habitation or sustained cultural activity (e.g. fish traps, midden, canoe runs or house platforms). Considering the deep history of human habitation in the Central Coast region—greater than 14,000 years—it is likely that habitation sites would have shifted, perhaps greatly, with the advancing and receding of ice as well as the corresponding rising and falling of sea level (McLaren 2008, McLaren et al. 2015, Mackie et al. 2018). However, given that this analysis is concerned with culturally modified trees that are visible today, the majority of our observations likely represent cultural use of these trees in the last ~250 years.

Many of the habitation sites likely have a much deeper history than 250 years. Indigenous knowledge and also midden excavations have showed that some sites have been used continually (to present) for thousands of years (personal communication Vernon Brown 2018, McLaren et al. 2015). Further, since many of the habitation sites in the existing archaeological records from the territory were recorded during boat based or shorezone surveys we can infer that these centres of cultural activity (especially those that have not been dated) would have coincided with modern shorelines and would be more likely to have been hubs of cultural activity, in many cases continuous until present. Therefore, we assumed that known habitation sites have been, and continue to be, active cultural activity hubs over the last ~250 years – the time period from which the bulk of our CMT observations are likely to have occurred (personal communication Vernon Brown 2018, Earnshaw 2019). I then used the Euclidean Distance tool in Arcmap (10.6) to generate a distance surface that could be used to evaluate how the proximity to known cultural activity centres might influence the suitability of bark harvesting areas. This distance layer was reclassified using one-kilometer intervals in radii forming a continuous surface from cultural activity hubs. One-kilometer intervals were chosen to reflect distances that would be traveled in a day. This layer was then extracted by and snapped to the study area raster.

Aspect

I assumed that south-facing aspects would be more vulnerable to damage or stress from prevailing southerly winds during the winter months, and also exposed to greater solar radiation (personal communication Vernon Brown). Other literature or in situ information regarding the influence of aspect on the suitability of bark harvesting trees is sparse. However, Stryd and

Eldridge (1993) found evidence during their inventory of Meares Island, BC that showed the highest density of CMTs were found on North and East facing slopes. Therefore, I examined whether there was a detectable influence of Aspect in the study area. The aspect layer was derived from the LiDAR data for the region. This layer was reclassified to a 1-10 scale in 36-degree increments. The layer was rescaled using a symmetric linear function so that north-facing aspects were more suitable than south facing aspects.

Canopy Height
A common rule of thumb for pulling taper strips of bark is that the tree should be no larger than can be hugged (Brown and Brown 2009). With the ideal size much smaller around 30-40 centimetres in diameter – where a strip roughly two handwidths could be pulled while leaving enough bark in-tact on the tree for the tree to survive (Garrick 1998, Brown and Brown 2009, Arcas 1999, personal communication Simon Mason 2018,). Rectangular bark strips are often taken from larger diameter trees because they typically remove a wider strip of bark than taper strips (Moberly and Eldridge 1992, Turner et al 2009, Province of BC 2001). This layer was derived from the LiDAR data, reclassified to a 1-10 scale in 9 metre increments. Then it was rescaled using a logarithmic growth function so that trees >10m became increasingly more suitable for cultural use.

Suitability modelling approach

I incorporated the predictor variables that were derived through interdisciplinary methods and drew from different knowledge sources in a multi-criteria evaluation framework (Figure 2.4). This framework uses sensitivity analysis to compare the relative influence of each predictor variable (DEM, cost distance submodel, CHM, distance from known habitation sites, slope, aspect, percent canopy cover for yellow-cedar and redcedar) on the response variable (recorded bark stripped CMT locations). I then used the analytical hierarchy process (AHP) to assign a rank to each predictor based on the sensitivity analysis results and derive a model with best-fit weights (Saatyy 1980, Store and Kangas 2001). I use this best-fit model to gain insight into the distribution of different suitability score classes and the potential impacts from commercial timber harvest.

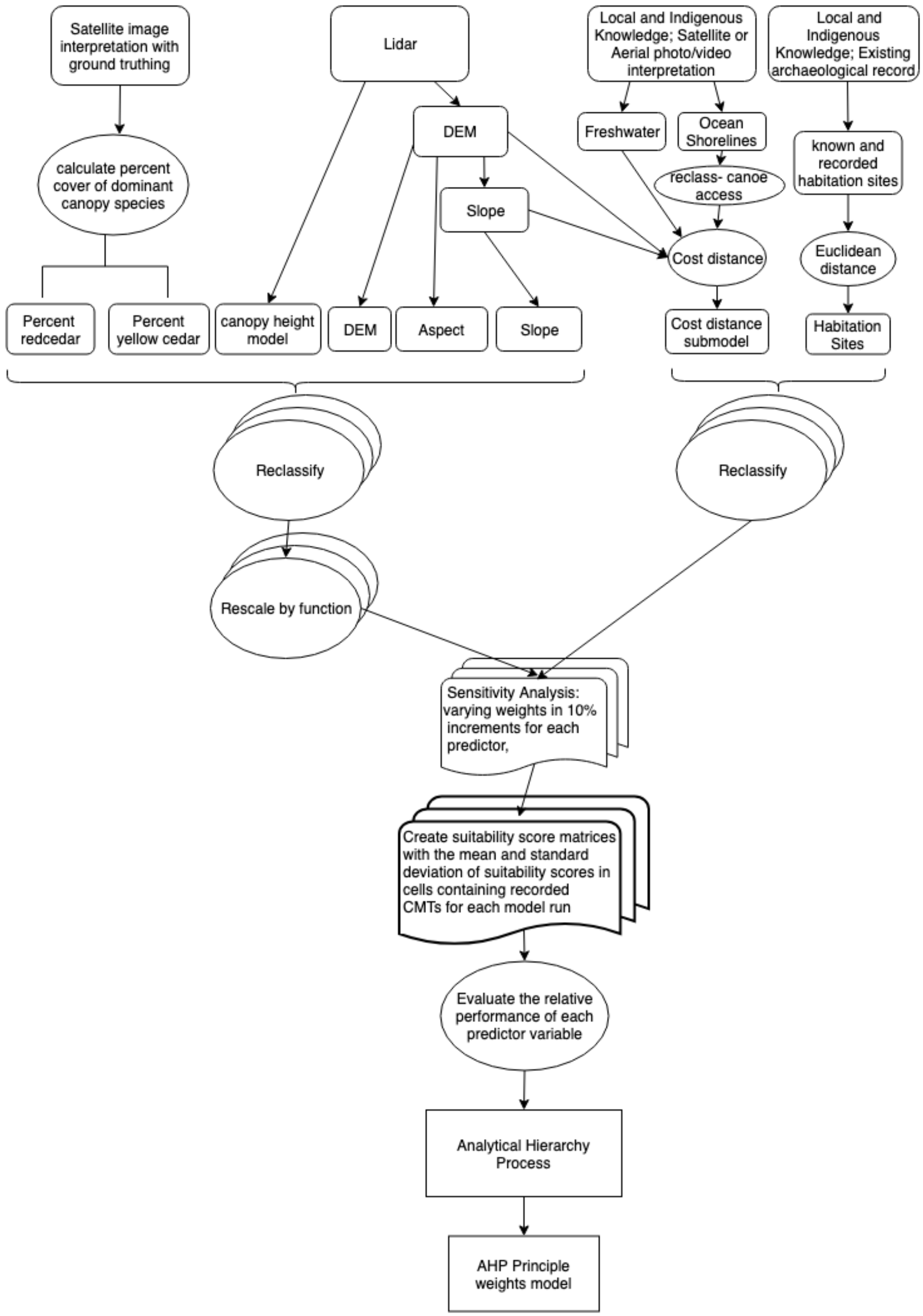

Figure 2.4 The framework that shows how I conducted a multi-criteria evaluation by using sensitivity analysis to incorporate data from different data sources (remote sensing [LiDAR and satellite imagery], Indigenous Knowledge, local knowledge, field surveys, and existing

archaeological data) to model the suitability of a biocultural indicator - culturally modified trees.

Multi-criteria Evaluation using sensitivity analysis

I chose the multi-criteria evaluation (MCE) method because it is a flexible and simple technique that allowed us to incorporate local and expert knowledge and further develop an understanding of the relative influence of ecological, landscape and cultural predictors in modelling the suitability for cultural cedar bark harvest (Store and Kangas 2001, Store and Jokimaki 2003). The multi-criteria evaluation process consisted of assembling remotely sensed data and reclassify predictor variables to a common suitability scale of 1-10, 1 being less suitable and 10 being more suitable. I reclassified the predictors based on local and Indigenous Knowledge of the potential influence of different predictors as well as existing literature and empirical study from within the study area and elsewhere in British Columbia (Table 2.2). In some cases, the reclassified layers were then rescaled by a function to more accurately reflect non-linear suitability of some predictors (Table 2.2; see Appendix A). The non-linear predictors included aspect, slope, and canopy height and percent canopy cover of redcedar and yellow-cedar. I developed 56 unique multi-criteria models with varied weights by using the Weighted Sum tool in Model Builder in ArcMap. The resulting multi-criteria models were evaluated in a sensitivity analysis (SA) framework where I evaluated the relative importance of each predictor variable and the ability of these predictors to accurately model the suitability of recorded CMT locations.

CMT sensitivity analysis

I used the weighted sum tool in model builder to create the 56 unique models that used the Weighted Sum tool in ArcMap to apply weights that were varied using the one-at-a-time (OAT) method (Daniel 1958, Daniel 1973 as cited in Chen et al. 2010). The weights were varied in 10% increments from 20% to 80% as well as one model run with equal weights of 12.5% for each criterion. One criterion in each model run was weighted according to the 10% increments and the remaining criteria were weighted equally so that the combined weights for each model were equal to one. The cell values in the resulting raster suitability surface are the sum of the suitability score of each input factor multiplied by the assigned weights. The sensitivity of each of these model results was then evaluated using our observational data for recorded locations of bark stripped CMTs. I used the 'Extract By Mask' in tool in ArcMap to extract the resulting suitability scores in cells that overlapped with CMT observations. The resulting suitability scores were then assembled in a table showing the mean suitability score extracted in cells containing CMT observations for each of the 56 model runs with varied weights (Table 2.3).

Null sensitivity analysis and comparison

In order to determine if these criteria and models were significantly better at predicting the known locations of bark stripped CMTs than random locations, I developed a null sensitivity analysis that I could use to test the significance of the CMT-SA. I created a random 'occurrence surface' using the Create Random Points tool in ArcMap (version 10.6). The random points were generated for the same extent as the study area – excluding water bodies and harvested cutblocks. I used the Point to Raster tool to convert the random points to a raster surface that was snapped to the study area raster. I used this null occurrence raster to extract model suitability scores from the same weighted sum models used in the CMT-SA. I then assembled the mean suitability scores from the null sensitivity analysis results in a heatmap table and compared the difference in means between the CMT-SA table and the null-SA table.

Overlay analysis

In addition to the 56 models with varied weights I also developed a base model with equal weights for each layer to further evaluate the relative importance of each layer. I used the base model to conduct a simple overlay analysis, where one input factor was removed each model run in order to clarify how each of the layers influences the mean suitability score and the standard deviation around the mean (Table 2.4). To gain further insight into the spatial distribution of influence of the top two predictor variables in the CMT sensitivity analysis I also removed both of these layers for one model run.

Analytical hierarchy process principle weights model:

In order to apply the results from the sensitivity analysis, I used the Analytical Hierarchy Process (AHP) to determine a reproducible principle weighting scheme that assigns best-fit weights. Usually AHP is used to assign best-fit weights based on subjective ranks determined by expert opinion. I adapted these methods to instead use the relative importance rank that was determined in the CMT-SA to inform the AHP ranks (Table 2.4). I assigned a rank to each input factor based on Saaty's (1977) scale of importance (see Appendix A), a 1 being least important and 9 exceedingly important. I used the AHP (v2.0) extension for Arcmap, which implements Saaty's (1980) standard pairwise comparison matrix (see methods used in a similar context in Chen et al. 2010). Principle weights were then derived from the matrix comprised of normalized pairwise rank comparisons and their reciprocals (see Appendix A).

AHP model comparison with the timber harvesting landbase

I then used the resulting AHP principle weights model to compare how the predicted suitability of bark arboriculture may be impacted by commercial forestry in the study area. I conducted a simple overlay analysis to derive the relative proportion of high (>7.5), moderate (5.5-7.5) and low (1.3-5.5) suitability areas that are both inside the timber harvesting landbase (THLB) and outside the THLB. The THLB refers to the spatial extent of areas that have been zoned for commercial timber harvest and is represented by polygons derived from the VRI (2018) geodatabase. These polygons were developed by the Forest Inventory Branch (BC VRI 2018) based on the economic viability of the timber and area-based protections defined in the GBR LUO (2016).

RESULTS

Multi-criteria evaluation using sensitivity analysis

The multi-criteria evaluation sensitivity analysis showed that the suitability predictions of known CMT locations have a strong relationship with elevation and cultural predictor variables (the cost distance submodel and distance from known habitation sites). The digital elevation model (DEM) has the highest mean suitability scores for each weighted scenario (6.98-8.93) and the percent canopy cover of yellow-cedar factor has the lowest range of suitability scores (6.28-2.58; Table 2.3). I found that crown height, distance from known habitation sites (6.88-8.02), the cost distance submodel (6.98-8.88) and the DEM all display an increase in suitability score with increased weight, while slope, aspect and percent canopy cover of yellow-cedar and red cedar display decreasing suitability scores with increased weights. With increased weights, the suitability scores for slope (6.66-6.05), aspect (6.69-6.26) and percent canopy cover of red (6.59-

5.42) and yellow-cedar (6.28-2.58) all fall below the equal weights suitability score of 6.74, which suggest that they do not meaningfully influence the accurate prediction of known CMT locations.

Table 2.3 Mean suitability scores for each model run with weights varied in 10% increments for each model run. Suitability scores are the sum of all predictors multiplied by their weights in each model run. Higher suitability scores (red) signify a prediction of higher suitability in cells containing recorded CMTs by the predictor with an increased weight. Lower suitability scores (green and blue) signify reduced ability of predictors to consistently predict high suitability in cells containing CMTs. Results were rescaled from a scale of 0-1 to a scale of 1-10 for ease of interpretation.

Mean suitability scores for cells containing bark strip CMT observations

	% Canopy cover yellow-cedar	% Canopy cover red cedar	Aspect	Slope	CHM	Distance from known habitation sites	Cost Distance submodel	Digital Elevation Model
equal weights	6.74	6.74	6.74	6.74	6.74	6.74	6.74	6.74
0.20	6.28	6.59	6.69	6.66	6.86	6.88	6.98	6.98
0.30	5.66	6.40	6.61	6.56	7.01	7.07	7.30	7.31
0.40	5.05	6.20	6.54	6.46	7.16	7.26	7.61	7.63
0.50	4.43	6.01	6.47	6.36	7.32	7.45	7.93	7.96
0.60	3.81	5.81	6.40	6.26	7.47	7.64	8.25	8.28
0.70	3.20	5.61	6.33	6.16	7.62	7.83	8.57	8.60
0.80	2.58	5.42	6.26	6.05	7.78	8.02	8.88	8.93

I found that the standard deviations for both the cost distance submodel (.59-.62), the DEM (.58-.70) and the percent canopy yellow-cedar (.58-.69) predictors had both the lowest and most stable range of standard deviations (Table 2.4). The cost distance submodel had the lowest and most stable range of standard deviations compared to all other predictors. This suggests that at a higher weight (>50%) the cost distance submodel is marginally more consistent at predicting known bark strip CMT locations than the DEM. The worst performing predictor - percent canopy cover of yellow-cedar - shows a similar range of standard deviation to the best performing predictor, which is the DEM. The low amount of variation around the mean for higher percent canopy cover of yellow-cedar suggests this predictor is a consistently a poor predictor of recorded bark strip CMT locations. In contrast, the percent canopy cover of redcedar has the second highest range of standard deviations (.69-2.05), which suggests that this predictor is inconsistently influential in predicting the suitability of occurrence for recorded bark strip CMTs.

The distance from known habitation sites predictor has a slightly lower range of standard deviations (input range) than the crown height model (input range), however the variability of both of these predictors increases with increased weights in the models. A similar pattern—although with a greater spread of values—occurs in the standard deviation values for the aspect, slope and percent redcedar canopy cover predictors. Therefore, these predictors become more variable in their ability to predict the suitability of known bark strip CMTs as the weights increase in the models.

Table 2.4 Standard deviation around the mean suitability score for each model run with varied weights. Table cells correspond to the means in the previous table (2.4).

Standard deviation of suitability scores in cells with bark stripped CMT observations

	% Canopy cover yellow-cedar	% Canopy cover red cedar	Aspect	Slope	CHM	Distance from known habitation sites	Cost Distance Submodel	Digital Elevation Model
equal weights	0.62	0.62	0.62	0.62	0.62	0.62	0.62	0.62
0.20	0.58	0.69	0.68	0.67	0.63	0.65	0.59	0.58
0.30	0.54	0.85	0.85	0.80	0.72	0.74	0.56	0.56
0.40	0.53	1.06	1.06	0.98	0.88	0.86	0.54	0.55
0.50	0.54	1.29	1.29	1.18	1.07	1.00	0.54	0.56
0.60	0.57	1.54	1.54	1.40	1.28	1.16	0.56	0.59
0.70	0.63	1.79	1.80	1.63	1.50	1.33	0.58	0.64
0.80	0.69	2.05	2.06	1.86	1.72	1.50	0.62	0.70

Null model

To test if the models are substantially better at predicting the occurrence of recorded CMT locations compared to random cell locations I conducted a null sensitivity analysis that used random cell locations to evaluate the model results (supporting information). Although the rank of predictor sensitivity between the CMT-SA and the null-SA (see Appendix A) are fairly similar, the range of mean suitability scores is much lower in the null-SA, with the highest suitability score of 6.85 (see appendix A). Mean suitability scores between the CMT sensitivity analysis (Table 2.3) and the null sensitivity analysis (see Appendix A) differed significantly [t = 7.95, df = 140.01, $\alpha = .05$, p=5.487e^{-10}]. Therefore, utilizing and weighting the input predictors that were selected and developed with support from local and Indigenous Knowledge, literature, and historical evidence allowed us to generate a suitability surface that can consistently and accurately predict higher suitability in areas with known CMT occurrences.

Overlay analysis

The results of the overlay analysis confirmed that the ability of the models to predict the suitability of known CMT locations is highly dependent on the predictors most closely linked to elevation (Table 2.5). However, I also observe that both the DEM and the cost distance submodel increase the mean suitability scores in different areas. The mean suitability scores fall when both the DEM and Cost distance submodel are removed compared to when just one of these two predictors were removed. This result may also indicate that each factor contributes to raising the suitability scores of the model in different spatial areas. Conversely, the suitability scores are increased most when the percent canopy cover of yellow-cedar and redcedar layers are removed. This coincides with the sensitivity analysis results (Table 2.3) that shows that the percent canopy cover layers for yellow and red cedar do not positively contribute to the ability of the models to predict the high suitability of cells containing recorded bark stripped CMTs. These results also confirm the importance of both landscape and cultural drivers in the models.

Table 2.5 The mean suitability score and standard deviation of model results for an overlay analysis where all predictor variables were weighted equally and where one factor was removed during each model run. During one model run both cost distance and the DEM were removed.

Name of layer removed from model	Mean suitability score	Standard Deviation
Cost distance submodel and DEM	5.80	0.83
DEM	6.34	0.71
Cost distance submodel	6.35	0.73
Habitation	6.51	0.66
CHM	6.57	0.73
Aspect	6.85	0.65
Slope	6.87	0.66
% canopy cover redcedar	6.98	0.65
% canopy cover yellow-cedar	7.52	0.73
All layers included with equal weights	6.74	0.62

Analytical Hierarchy Process based on MCE rank

The principle weighting scheme calculated from the pairwise comparison ranks (See Appendix A) with a consistency ratio of resulted in the lowest weight assigned to the factor for percent canopy cover of yellow-cedar (2.36%) and the highest weights were assigned to the predictors for distance to habitation sites, the cost distance submodel and elevation (18.52%) (Table 2.6). The resulting model shows a mean suitability score of 6.8 and the mean suitability score of the cells containing known CMT observations is 7.9. This model shows a significantly higher mean suitability score in the cells containing CMTs than the overall model average [$t = 48.8$, $df = 340$, $\alpha = .05$, $p < 0.001$]. Within the entire study area, high suitability cultural cedar stands predicted using the AHP principle weights model cover ~12,088.5 hectares (10.17%), moderate suitability stands cover ~76,555ha (64.41%) and low suitability stands cover 30,607ha (25.5%). These results indicate widespread cultural use throughout the study area for *only one* CMT subtype (bark-stripped CMTs) among may other archaeological site types and biocultural indicators, which may lend insight into the potential for intensive landscape utilization in the study area (Lepofsky and Armstrong 2018). Further, this model is based on only those CMT observations that are recorded (and currently visible to surveyors), which likely represents a small fraction of the CMTs that actually exist in the study area.

Table 2.6 The predictor variables included in the analytical hierarchy process, their associated rank, and the derived principle weights.

Layer Name	AHP Rank	Principle Weight (%)
Cost Distance Submodel	8	18.52
DEM	8	18.52
Distance from known habitation sites	8	18.52
CHM	7	16.40
Slope	4	9.32
Aspect	4	9.23
% Canopy cover red cedar	3	7.04
% Canopy cover yellow-cedar	1	2.36

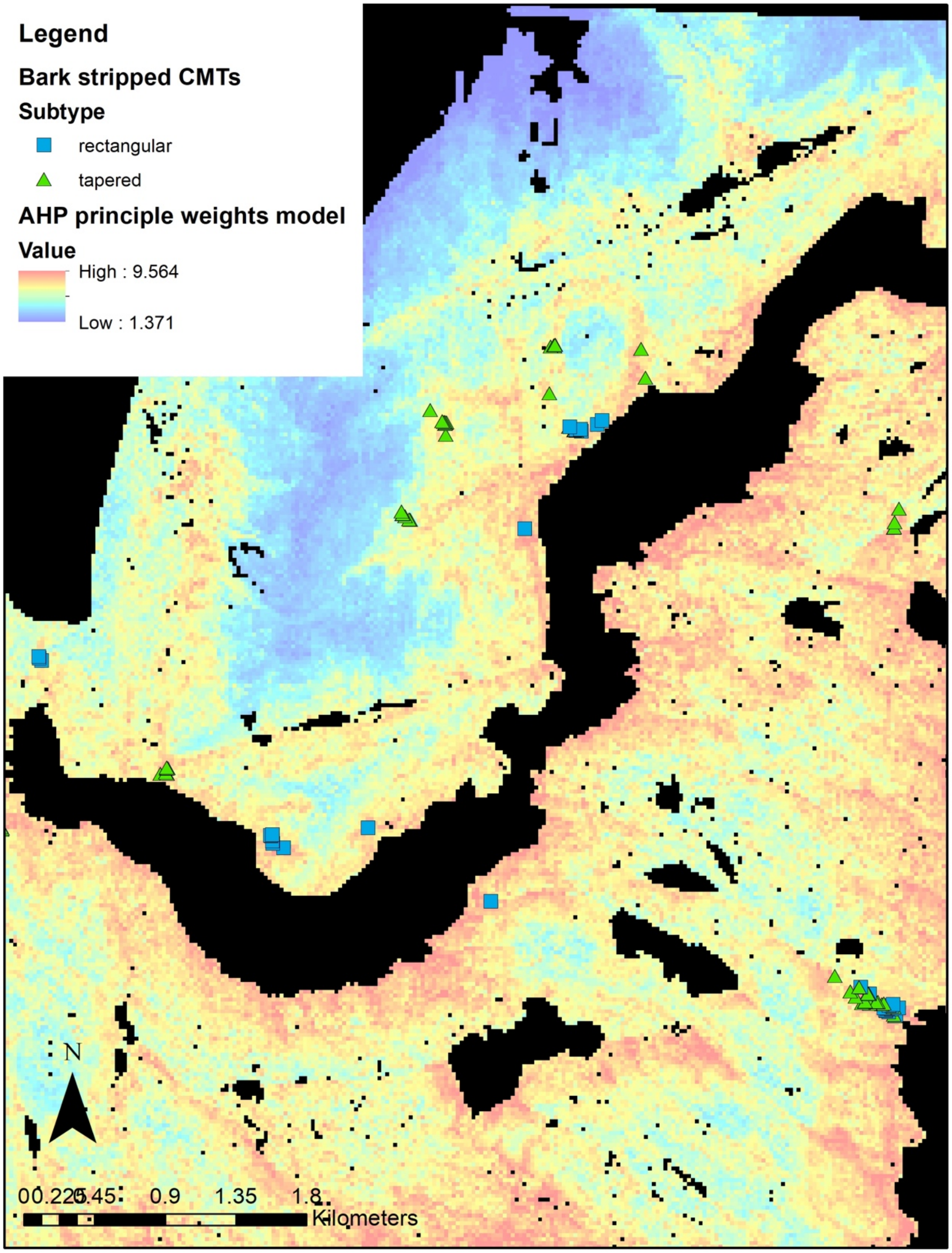

Figure 2.5 A portion of the of the suitability surface that was generated by using the principle weighting scheme that was derived via the Analytical Hierarchy Process based on the sensitivity analysis ranks. Due to the sensitivity of this data and as per my research agreement with the Kitasoo/Xai'xais Authority, I am not able to make the full extent of the suitability model publically available.

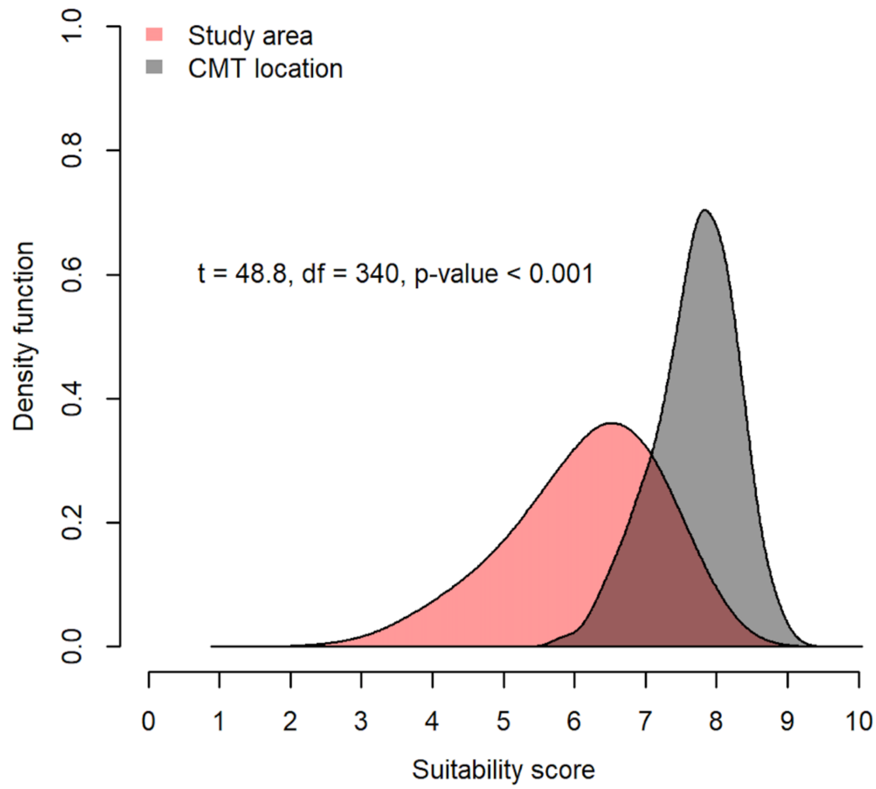

Figure 2.6 Density functions for the frequency of suitability scores for the entire study area (red) compared to the density function of cells (n = 413) that contain known CMT observations (grey).

Cultural cedar and the timber harvesting landbase

The timber harvesting landbase (THLB) polygons comprise ~26.6% of the study area. I found that given the relative proportion of the total number of cells inside the THLB and outside the THLB, high suitability cells make up 13% of the cells inside the THLB and only 8.9% of the cells outside of the THLB (Table 2.7). Moderate suitability cultural cedar stands make up 74.2% of the area inside the THLB and 60.79% of the area outside the THLB. These results represent one biocultural indicator and one archaeological site type among many. Therefore it is likely that a much higher percentage of the area within the THLB contains other site types or biocultural features. These results should be interpreted only for its value in comparing the relative distribution of bark-stripped CMT suitability is far more likely to occur inside the THLB compared to outside the THLB.

Table 2.7 Descriptive statistics for the raster surface that was produced using the Analytical Hierarchy Process principle weights model. The model values were reclassified into high, moderate and low suitability classes.

Suitability score range	Suitability class	# Cells	Proportion of Study Area	Hectares	Relative proportion inside THLB	Relative proportion outside THLB
>7.5158	High (>75%)	193416	10.17%	12088.5	13.42%	8.93%
5.4675 – 7.5158	Moderate (50%-75%)	1224880	64.41%	76555	74.26%	60.79%
1.3700 – 5.4675	Low (<50%)	489717	25.75%	30607.3	12.3%	30.27%

DISCUSSION

Cultural, landscape and ecological predictors and the role of incorporating local and Indigenous Knowledge

I show that elevation and culturally-related dimensions of interaction with CMTs perform best among predictors of bark-stripped CMT distribution in Kitasoo/Xai'xais Territory. In particular, the cost distance surface performs much better than the criteria for slope alone, which indicates that cumulative impediments (costs) in the form of slope are more likely to affect the suitability of a given area for bark harvesting compared to specific site (cell) level conditions (Table 2.3). Although the cost distance submodel revealed a slightly lower range of means in the CMT sensitivity analysis (SA) (Table 2.3) than the digital elevation model, the cost distance submodel has a lower standard deviation (Table 2.4). Accordingly, it is likely more consistent and accurate in predicting the suitability of CMT locations compared to DEM. The distance from habitation input factor is the third most important factor in the CMT-SA, making both of the cultural predictors—derived from local and Indigenous Knowledge and LiDAR data—among the top three predictors.

Although I expected that all of the ecological variables (percent canopy cover of yellow-cedar and redcedar and the crown height model) have predictive utility, percent canopy cover of both yellow-cedar and redcedar performed poorly in predicting suitability. I attribute this surprising result to the low sample size of yellow-cedar CMTs (n = 33) used in the CMT-SA evaluation layer. Further, the consistency in the decreasing suitability scores for percent canopy cover (low standard deviation, Table 2.4) suggests that the low (~20%) mean canopy cover of yellow-cedar could also have limited the influence of this factor. It is also possible that the spatial resolution of the source data (derived from satellite imagery) did not accurately represent the relative canopy composition at the stand-level scale. These findings inhibit the ability of the models to predict the suitability for yellow-cedar CMTs with sufficient accuracy and consistency.

One out of the three ecological predictors did perform relatively well. The crown height model is the fourth most sensitive predictor in the CMT-SA, which indicates that as tree height increases from ~10m there is increasing suitability for bark stripped CMTS (Table 2.3). However, the more variable nature of this ecological input factor indicates that a variety of tree heights can be suitable (Table 2.4). In taller stands canopy and sub-canopy stand structure could play a role in creating ideal conditions for bark harvesting trees. For example, canopy closure and multiple canopy layers can create ideal growing conditions for trees with fewer branches below the crown —offering a surface unobstructed by limbs from which to pull bark (Sutherland et al. 2016).

In addition to the ecological predictors, the remaining landscape predictors other than elevation (slope and aspect) that were derived from LiDAR data were not sensitive to changes in their weights; the mean suitability scores were stable in the CMT-SA (Table 2.3). Further, the standard deviations of slope and aspect were among the highest in the models. This inconsistency suggests that the site- or cell-level conditions for recorded CMTs were highly variable and do not correspond to any particular slope angle or aspect.

Key insights from the AHP principle weights model

The AHP principle weights model (Figure 2.5, 2.6) shows that the predictor variables included are able to consistently and accurately predict higher suitability in cells containing known bark stripped CMT locations when best-fit weights are applied. This model can therefore provide useful information about the suitability of forest stands for past, present and future cultural bark harvesting in unsurveyed regions of the study area. For example, the model (Figure 2.5) suggests that there are clustered high suitability areas at various spatial scales, from small groups of high suitability cells to subregional 'hot spots'. Also, roughly 10% of the landbase is within the high suitability class (>7.51, Table 2.7), which indicates that although bark arboriculture was widespread across the Territory most of the landbase does not provide suitable conditions. It is important to also note that this model is based on relatively recent bark-stripped CMT observations, many of which were made during a time when Kitasoo/Xai'xais People experiencing the genocidal impacts of colonialism (Cannon 2002). Therefore, it is likely that deeper-time culturally modified trees and stands (made prior to colonisation) extend spatially beyond those visible to present day surveyors (Earnshaw 2019).

Given the clustered nature of highly suitable areas for bark arboriculture, the overlay analysis identifies important spatial patterns in the context of commercial logging. I also found that the relative proportion of high suitability cells within the THLB (13%) is almost twice the amount of high suitability area outside the THLB (8%; Table 2.7). This overlap between high suitability forest stands and the THLB illustrates the utility of this model (Figure 2.5) in mitigating the impacts to CMTs from commercial timber harvest. It is also important to note that this model and other predictive models do not replace the need for ground-based survey. The undocumented removal of CMTs can occur even if archaeological surveys were conducted within a cutblock prior to cutting (Earnshaw 2016, 2019). For example, healed over bark stripped CMTs can be invisible to surveyors today and are often co-located with currently visible bark-stripped CMTs—indicating sustained cultural stewardship in those areas (Earnshaw 2016, 2019). If either currently visible CMTs or deeper-time healed over bark stripped CMTs are removed without documentation, this erasure of cultural heritage could also have negative implications for First Nations in rights and title cases tried in colonial courts (Earnshaw 2017, 2019).

Commercial timber harvesting in Canada has impacted cultural forests for nearly 200 years. CMTs were not protected by Canadian legislation until the *Heritage Act* (1996)–even then, the legislation only protects CMTs dated prior to 1846 (Turner et al. 2009, Earnshaw 2019). The models from Kitasoo/Xai'xais Territory (Figure 2.5) and other recent models of different CMT subtypes in this region (see Benner et al. 2019) suggest that the cumulative impacts from commercial timber harvest prior to the protection of CMTs in 1985 is likely considerable. Further, in other regions it has been suggested that roughly half of all bark stripped CMTs are not currently visible (healed over) to surveyors, suggesting continued undocumented removal of CMTs even after their legislated protection (Earnshaw 2017, 2019, Government of Canada

1996). Given that current harvest rates cover roughly 2,800Km^2 of forest per year nationally, the undocumented removal of CMTs is likely considerable (Turner et al. 2009, Earnshaw 2017, 2019, Government of Canada, 2019). I contend that the requirement for tangible evidence of First Nation occupation and use on forested lands – in addition to providing proof of exclusivity of occupation – may be unjust requirements in rights and title claims cases considering potential cumulative impacts to cultural sites (including CMTs).

Applying spatial models for biocultural indicators

Territory Scale

Spatial models do not replace the need to conduct ground-based surveys in proposed extraction or development areas, but may offer insight into how impacts to culturally significant areas can be minimized. Spatial models—such as the AHP principle weights model—can inform early planning processes to reduce the likelihood that on-site surveys of proposed cutblocks will encounter culturally significant and legally protected values. However, it is still likely—given the relatively low survey effort for the large area modeled, the use of only bark-stripped CMTs and the limited visibility of older CMTs—that CMTs and other archaeological features and biocultural indicators can be encountered in regions with relatively low suitability. Therefore a precautionary approach should be (and is being used by the Kitasoo/Xai'xais Stewardship Authority) taken when implementing this model. One potential method for applying the suitability model would be to overlay proposed cutblock polygons to evaluate the mean, maximum and range of suitability scores within the boundaries. This could give Lands Managers and the communities they represent valuable information about the relative risks of developing an area before plans proceed, and investments made into engineering, etc. At the very least, if development proceeds in areas of high suitability, proposed blocks can be surveyed intensively prior to development and followed by a post harvest assessment for healed over bark-strip scars (Earnshaw 2016, 2019).

Biocultural spatial models can also be implemented in contexts of non-commercial resource harvesting. Specifically, the AHP principle weights model and future suitability models of other biocultural indicators and culturally significant species or values could also be incorporated as part of community harvest management plans, protected area management or comprehensive assessments for non-extractive industries such as tourism (e.g. Kitasoo/Xai'xais 2018). Scaling up, multiple biocultural spatial models could also be combined in a cultural ecosystem services modelling framework (Nelson et al. 2009, Klain et al. 2012, Pert et al. 2015). In this way, Lands Managers could identify areas that have a high degree of overlap for a diversity of cultural ecosystem services (e.g. marine and terrestrial resources). Lastly, the observed influence of known habitation sites or archaeological subsistence features on the suitability of recorded bark stripped CMTs may suggest that the AHP principle weights model could also be used as an archaeological prospecting tool (Gallagher and Josephs 2008, Hesse 2013).

Limitations

The development of the Kitasoo/Xai'xai cultural feature inventory project—of which this study is a part—included several trade-offs. One involved study design. I employed a non-random sampling strategy, owing to logistical constraints and the difficulty in conducting grid based or census-based methods. A transect-based study design still allowed us the opportunity to detect all of the target species (Kitasoo/Xai/xais 2018)—each with a diversity of habitat or site condition requirements. Further, sampling a greater total area was a higher priority than conducting more intensive surveys in a series of smaller census plots because some of the surveys were also incorporated as part of preliminary biocultural field reconnaissance in the forestry planning process. Clearly, when financial and time resources are plentiful, census-based surveys provide a better approach for many spatial modelling approaches (Muir and Moon 2000, Store and Jokimaki 2013).

Additionally, although this approach sought to utilize both existing archaeological data to collect new spatial information on culturally modified trees, the accuracy of the georeferenced points could be highly variable. Uncertainty also exists in the spatial data used in the predictor variables; in this case this occurs primarily in the two layers that were derived from satellite imagery (percent canopy cover of redcedar and yellow-cedar). I assumed that the 25m^2 cell resolution captured much of this spatial uncertainty, however this uncertainty is difficult to quantify. Confidence in the models could be improved by ground-truthing the spatial accuracy of a subset of the georeferenced points across the study area. Further, considering that Cultural Feature Inventory surveys are ongoing, there is also a need to incorporate new observational data to reflect any changes that they might contribute. In this sense the findings in Chapter Two could be utilized to inform the development of a more easily updateable or “living model”. The final suitability surface should always be interpreted with caution that recognizes the uncertainty in the observational and remotely sensed data. Lastly, another aspect that could improve future modelling of culturally modified trees and other biocultural indicators would be incorporating community-based interviews in a way that more explicitly informs the models. For example, models may be able to incorporate harvester preferences such as certain qualities or conditions that make a given species or value more suitable for food, social or ceremonial purposes.

CONCLUSION

Biocultural EM and monitoring is an emerging domain that couples local values and scientific approaches to protect linkages between people and nature. Biocultural indicators, such as culturally modified trees, offer a focal point where multiple knowledge sources, institutions and agencies can engage to effectively to design and implement EM and monitoring. This case study shows how a community-engaged spatial modelling approach guided and enabled the incorporation of locally-informed input predictor variables. These culturally relevant predictors of CMT distribution were among the most influential in accurately and consistently predicting the landscape suitability for cedar bark arboriculture. As this research confirms, local and Indigenous values can direct both the development and application of science as well as the implementation of biocultural EM decisions. This modelling approach and other community-led and locally-informed modelling approaches can facilitate the implementation and monitoring of biocultural EM in diverse social-ecological systems.

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Conclusion

Contemporary biocultural approaches to EM that integrate analytical tools are still in the early stages of development, in Canada and globally. Chapters One and Two of this thesis address two major challenges facing biocultural EM projects both in the Great Bear Rainforest and abroad. The first challenge addressed was theoretical—a lack of criteria to guide communities and EM agencies in developing effective and just biocultural indicators that can be used to set management goals and monitor effectiveness of EM implementation. Chapter One identified six criteria (culturally salient, supportive of place-based relationships, inclusive, sensitive to impacts, perceptible, and linked to human well-being) that emerged from the Kitasoo/Xai'xais-led approach to biocultural EM, supported with literature and examples from British Columbia and beyond. The second challenge—addressed in Chapter Two—was in practice. This challenge pertains to the difficulty in mobilizing existing data and in situ or place-based knowledge, in addition to efficiently gathering appropriate new data, to inform the development of spatial biocultural EM resources that can be used during implementation of EM targets.

Given the goals and scope of this thesis, some methodological and analytical limitations exist that could present opportunities for future research and biocultural EM projects. One limitation, described in Chapter One, is that this case study is embedded in one First Nation's—the Kitasoo/Xai'xais—approach to biocultural EM. I recognize that this approach may differ greatly from the direction that other Indigenous Nations are taking in this context. Further, it was possible to summarize these themes into criteria because prior to this research there had already been many years of community meetings and community-led policy development on which I was able to draw. Therefore, it is possible that other criteria or themes may emerge from *new* biocultural EM projects that are at different stages of development. Further, the Kitasoo/Xai'xais have developed a biocultural EM strategy that mobilizes different knowledge systems (local knowledge, Indigenous Knowledge and western science) to plan, implement and monitor EM. Other Indigenous Nations or local communities may not wish to adopt this approach, especially if a purely local or Indigenous Knowledge approach is desired. Therefore, not all of the criteria offered in Chapter One may always apply, or new criteria may need to be developed to guide the selection of biocultural indicators in line with in situ values and worldviews. More generalizable criteria may emerge with a greater diversity of case studies. Regardless, the underlying foundation of all biocultural EM projects is that they are locally-led or –guided. This means that other communities or Indigenous Nations may wish to develop their own process and criteria for selecting indicators and those processes should be recognized and supported as integral to the success of EM outcomes if ex situ agencies are also involved.

One analytical limitation to the goals and scope of Chapter Two was that I did not conduct new interviews to inform the spatial models, but instead incorporated existing interview-based data. I made this decision to use existing data —based on input from community members—because numerous traditional use studies had already been conducted in the community, and those data had not yet been incorporated directly as part of spatial EM planning resources in the context of commercial forestry. Although the suitability modelling approach taken in Chapter Two includes some aspects of local and Indigenous knowledge, other aspects such as harvesting preferences or thresholds for cultural resource quality could also be incorporated into suitability models for culturally significant species and indicators (e.g. Polfus et al. 2014). Specific harvesting information, such locations of higher-quality bark, could help tailor suitability models to identify exceptional or key biocultural spaces and harvesting places (Plieninger et al. 2013). Interviews can be a valuable method to improve locally-informed spatial models if a collaborative approach is taken and community members suggest that it would benefit the models.

In general, in partnership-based or co-managed projects, trust is perhaps the most valuable and challenging aspect to achieve. For example, the political landscape in which biocultural EM initiatives are developed is often extremely complex and it can be difficult to overcome historical conflicts and power asymmetries to build trust among in situ and ex situ cultures and agencies (Low and Shaw 2011). Recognizing and addressing past conflicts, wrongdoings and power asymmetries is one part of overcoming barriers to trust and collaboration at all levels of inter-cultural or inter-agency governance (Adams et al. 2014). One potential avenue where this can be (and is being) addressed in Canada as well as other so-called Commonwealth Nations and "geographies of coexistence" (Howitt et al. 2013) is through a process of reconciliation—a process involving the implementation of calls to action related to renewed nation-to-nation relationships with Indigenous Peoples, the recognition of the rights of Indigenous Peoples and a building a foundation of respect and cooperation among many other potential aspects (Truth and Reconciliation Commission, 2015). For example, reconciliation could also play a role in revitalizing existing Indigenous EM institutions if Indigenous Nations want to be involved in the reconciliation process and also if that process is Indigenous-led or -guided. There are also specific policies that need to be reconsidered, given some of the findings in this thesis.

POLICY IMPLICATIONS

To illustrate the application and opportunities presented by this work, I conclude by offering recommendations that apply to existing policies at various institutional and spatial scales related to the study area. The GBR exemplifies a geography of coexistence in that there are different types of government (Indigenous and colonial) whose jurisdiction are considered by each party to apply at various and overlapping scales (e.g. Indigenous Territories, Canada, the Province of British Columbia and the Central and North Coast regional district). There are four major pieces of legislation that impact how terrestrial-based biocultural values and indicators—such as culturally modified trees—are managed for in the study area and throughout British Columbia and Canada. I highlight, by region, existing colonial legislation and opportunities therein to address the findings in this thesis.

Regional

The Great Bear Rainforest Land Use Objectives Order GBR LUO (2016) applies to the smallest spatial area, a region now referred to as the central and north coast of British Columbia, Canada – where the case studies in Chapter One and Two took place. The GBR LUO specifically includes culturally modified trees as cultural heritage features of management concern, but does not apply specific management targets. This legislation is also unique in that much of the targets for commercial forestry management apply to primary old growth forest that has not been commercially harvested. I found roughly 10% of the study area is highly suitable for bark arboriculture and that 13% of the area within the timber harvesting landbase is also highly suitable. This suitability reflects only one subtype of CMT among many other archaeological site types and other critically important biocultural indicators. It is likely that commercial resource extraction impacts these other biocultural features in different spatial areas from the model presented in Chapter Two. The GBR LUO could improve the protection and documentation of all CMT subtypes and other archaeological features by requiring post harvest CMT surveys in addition to mandatory pre harvest surveys (Earnshaw 2019). Further, the Province of British Columbia should offer financial support to First Nations in the region to enable training and implementation of similar cultural feature inventory surveys as those carried out in Kitasoo/Xai'xais Territory.

Provincial

Moving beyond the GBR, roughly 95% of timber supply in British Columbia is considered publicly owned 'crown land' by the Province of BC, even though much of this timber supply overlaps with sovereign First Nations Territories. The *Forests and Ranges Practices Act* (FRPA)(2002) regulates the harvest of timber on all land considered 'crown land' by the Province of BC. A new category for biocultural indicators could be added to FRPA section 149 to assist in the process of communicating bicultural EM priorities between in situ and ex situ agencies. Although there is a category for 'cultural heritage features' in section 149, this makes reference to only those features protected under the federal *Heritage Conservation Act* (1996).

The *Heritage Conservation Act* (1996) only protects 'historic' (those dated prior to 1846) culturally modified trees as archaeological site types (Earnshaw 2019). The *Heritage Conservation Act* (1996) could be amended to recognize that all culturally modified trees are the living heritage of Indigenous Peoples. Given the amount of commercial forestry that has occurred since the mid 1800's in British Columbia, it is likely that many 'historic' and 'contemporary' (in the *Heritage Conservation Act* – those trees that were culturally modified after 1846) CMTs have been removed without documentation (Turner et al. 2009, Earnshaw 2019). Canadian federal courts should also recognize this erasure of tangible proof of continued use and occupation of the land in regions where commercial timber harvest, landscape alteration and wildfires have significantly altered the forest.

Federal

Aside from increasing protections of CMTs, there are also opportunities to incorporate other biocultural indicators and approaches existing federal EM legislation. Although the *Species at Risk Act* (2002) requires collaborative approaches to inform the recovery of endangered species;

in many cases, however, Canadian government agencies have yet to fully adopt and carry out collaborative approaches (Polfus et al. 2014). One way to do so would be to streamline objectives from agencies at various scales (e.g. federal and provincial ministries) to support the development and implementation of biocultural indicators (Polfus et al. 2014, Sterling et al. 2017). For example, biocultural indicators can be applied during threatened and endangered species recovery planning and implementation. In some cases the habitat for endangered species may be overlapped with important biocultural indicators or some biocultural indicators may also be endangered or threatened species (e.g. Polfus et al. 2014). Another way this issue can be overcome both in Canada and abroad could be for federal and provincial/state agencies to support local or Indigenous EM institutions and commit funding to develop or co-develop locally informed biocultural spatial models if financial support and/or collaboration are desired by Indigenous institutions (Polfus et al. 2014).

In conclusion, biocultural indicators can serve as a means for local communities and Indigenous Nations to communicate EM priorities across political and institutional boundaries and spatial scales. Although there are numerous challenges facing the development and implementation of biocultural EM projects, I show how the Kitasoo/Xai'xais First Nation is overcoming some of these common barriers – both in theory and in practice. I also highlight opportunities to incorporate biocultural approaches to EM (including biocultural indicators) in existing pieces of legislation that apply to various spatial scales in Canada and beyond.

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Appendices

Appendix A: Supplementary Figures

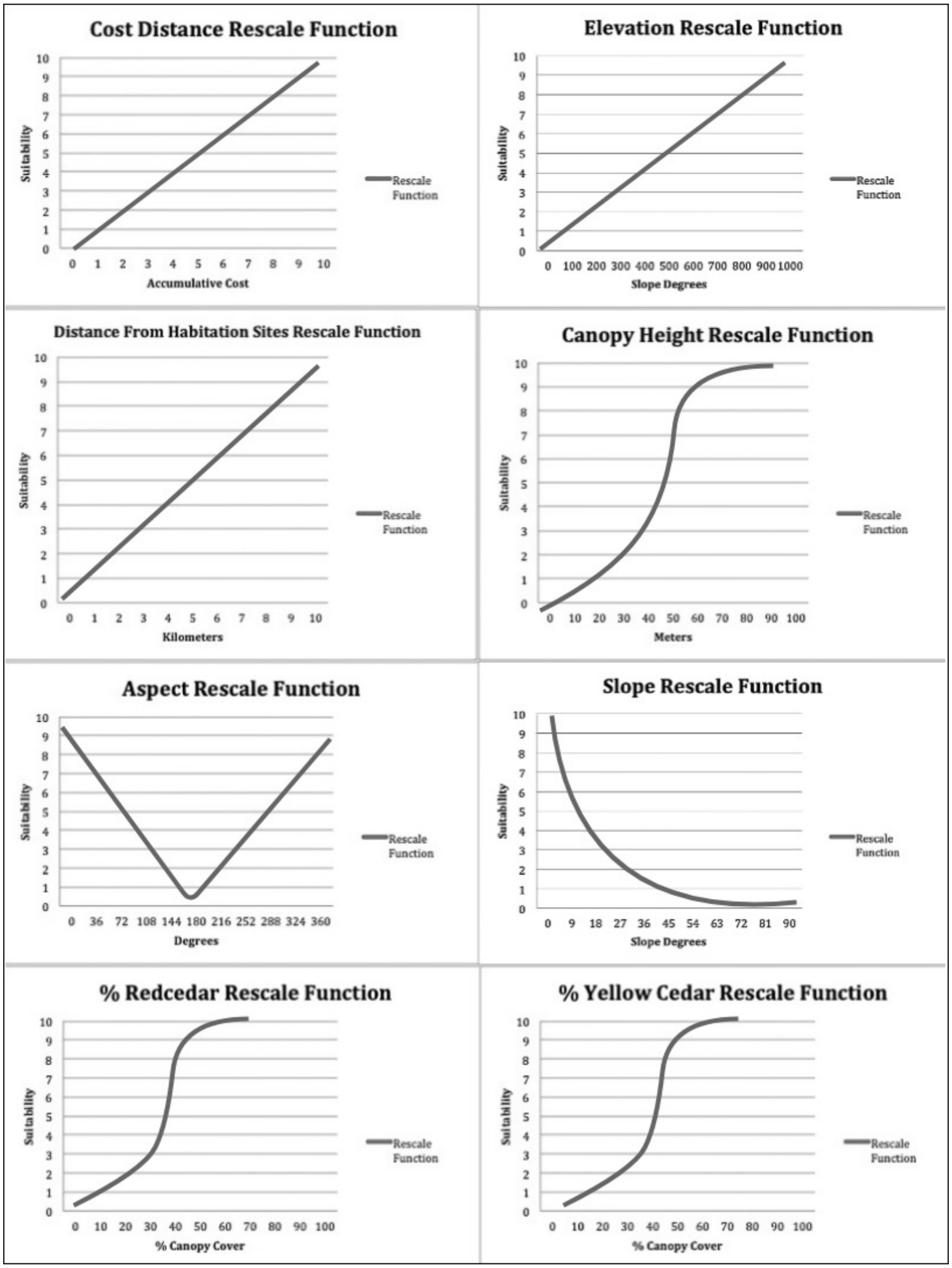

Figure S1: Rescaling functions used for each predictor variable during the parameterization of the sensitivity analysis.

Table S1: Results for the null models where each model run was tested using a random points mask instead of the CMT observations mask. There were N=511 random cells derived from the random points layer.

Mean suitability scores for random cells								
	% Canopy cover yellow-cedar	% Canopy cover red cedar	Aspect	Slope	CHM	Distance from known habitation sites	Cost Distance Submodel	DEM
equal weights	5.49	5.49	5.49	5.49	5.49	5.49	5.49	5.49
0.20	5.19	5.33	5.55	5.46	5.48	5.64	5.63	5.64
0.25	4.99	5.22	5.59	5.45	5.47	5.74	5.72	5.74
0.30	4.79	5.11	5.63	5.43	5.46	5.84	5.82	5.84
0.35	4.59	4.99	5.45	5.41	5.45	5.94	5.91	5.94
0.40	4.39	4.89	5.71	5.39	5.44	6.04	6.01	6.04
0.50	4.00	4.67	5.79	5.36	5.42	6.25	6.20	6.25
0.60	3.60	4.45	5.87	5.32	5.40	6.45	6.38	6.45
0.70	3.20	4.23	5.95	5.29	5.38	6.65	6.57	6.65
0.80	2.80	4.01	6.03	5.25	5.36	6.85	6.76	6.85

Table S2 – The analytical hierarchy process scale of relative importance used to rank each predictor variable (adapted from Saaty 1977; 1980).

Intensity of importance	Definition
1	Equal importance
2	Weak
3	Moderate importance
4	Moderate plus
5	Strong importance
6	Strong plus
7	Very strong or demonstrated importance
8	Very, very strong
9	Extreme importance

Table S3: Pairwise comparison weights matrix based on ranks assigned in the analytical hierarchy process implemented in ArcMap.

Eigenvector of weights based on rank from the sensitivity analysis results

	% Canopy cover yellow-cedar	% Canopy cover red cedar	Aspect	Slope	CHM	Distance from known habitation sites	Cost Distance Submodel	DEM
% Canopy cover yellow-cedar	1.00	0.33	0.25	0.25	0.14	0.13	0.13	0.13
% Canopy cover red cedar	3.03	1.00	0.75	0.75	0.43	0.38	0.38	0.38
Aspect	4.00	1.33	1.00	1.00	0.57	0.50	0.50	0.50
Slope	4.00	1.33	1.00	1.00	0.57	0.50	0.50	0.50
CHM	7.14	2.33	1.75	1.75	1.00	0.88	0.88	0.88
Distance from known habitation sites	7.69	2.63	2.00	2.00	1.14	1.00	1.00	1.00
Cost Distance Submodel	7.69	2.63	2.00	2.00	1.14	1.00	1.00	1.00
DEM	7.69	2.63	2.00	2.00	1.14	1.00	1.00	1.00

Appendix B: Cultural Feature ID Data Sheet

Cultural Feature ID Data Sheet

Survey Area: ______________________________ Transect #: ____________________

Crew Leader: _________ Crew: ______________________________

POC: ____________________ (E), ______________________ (N) Bearing Out: ___________

Traverse Bearing: ____________ Bearing Return: ____________

Date: ______________________ Start Time: _________________

End Time: _________________ Survey Effort: ________ Total Dist. Covered: ___________

Weather/comments:____

Monumental Cedar: Flow chart, species, DBH, log lengths, GPS coordinates
CMT: Species, subtype, DBH, HAG, healing lobe (left and right), scar width, scar length, confidence level, GPS coords, accuracy, diagnostic features, comments
AHF: Subtype, GPS coordinates, comments
AFR: Species, stem count, GPS coordinates, FSC patch?
Yew Tree (greater than 3m tall): Stem count (exact number of stems), height, GPS coordinates, high quality? (**HQ**) How many if more than? (i.e. 1x HQ)
Species Codes: Red Cedar (**Cw**), Yellow-cedar (**Yc**), Amabalis Fir (**Ba**), Mountain Hemlock (**Hm**), Western Hemlock (**Hw**), Pine (**Pl**), Spruce (**Ss**), Yew (**Tw**), Alder (**Dr**), Hellebore (**HELL**), Devils Club (**DECL**), High bush Cranberry (**HICR**)

Feat. #	Feature Type						SPECIES CODE	SUBTYPE CODE For CMTs and AHFs (look in orange notebook)	*STEM COUNT* *1-4* *5-10* *11-20* *21-30* *31-50* *>50* *Exact # of stems for Yew*	DBH (cm)	HAG (cm)	Healing lobe thickness (cm)		Aspect of Scar Face	Slope (%)	Scar Width (cm)	Approx. Scar Length (cm)	Confidence Level (L, M, H)	Log Lengths (m)	GPS Coords		GPS Accuracy (m)	Diagnostic Features Present: -Toolmarks (TM) -Scar Crust (SC) -HAG Rotten (HR) -Kinked Lobes (KL) -Other features Nearby (OF) Features that decrease Confidence: -Windfall (WF) -Very Rotten (VR) -Healed Over (HO) -Hand Logging (HL) FSC patch? (FSC) High quality yew? (HQ)
	AHF	H-CMT	C-CMT	AFR	Mon	Yew						L	R							Easting	Northing		
																				_ _ _ _ _ _ _	_ _ _ _ _ _ _		