Systematic conservation planning in human-dominated landscapes: maximizing efficiency in biodiversity conservation via carbon sequestration and land management

by

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Abstract

Investments in habitat and biodiversity conservation are critically needed given mounting effects of global climate change and unprecedented rates of ecological degradation and species extinction. However, as more regions of the world are converted to human use, we are also experiencing a shift in the traditional targets of conservation from protecting 'ecologically intact' landscapes to restoring degraded habitat by prioritizing conservation investments under triage. The overall goal of this thesis was to evaluate alternative ways of funding conservation initiatives. To reach that goal, I first used 1,770 avian point counts in a 2,520 km2 study area, remote-sensed data and models incorporating imperfect detectability to predict habitat occupancy in 47 widely-distributed native birds, which were also classified by experts according to their habitat association. Forest and Savannah association scores for these species were then used as weights in a composite distribution map of species communities. My results showed that composite maps of widespread indicators improve site prioritization by incorporating the behavioural and demographic responses of a diverse range of indicators to variation in patch size, configuration and adjacent human land use. Using these composite maps, I asked how the sale of forest carbon credits could reduce land acquisition costs, and how the alternate goals of maximizing α or β-diversity in focal communities could affect the prioritization of parcels for acquisition. My results indicate that carbon sales have the potential to enhance conservation outcomes in human-dominated landscapes by reducing the net acquisition costs of land conservation. Maximizing β versus α-diversity may further reduce costs by reducing the total area required to meet conservation targets and enhancing landscape heterogeneity. In cases where land purchase is not an option, private land conservation covenants can provide an alternative; although serious questions exist about long-term monitoring and enforcement costs of covenants given the risk that owners might violate or challenge them in court. My findings suggest that violation or dispute rate can substantially affect long-term costs of covenants and potentially surpass the cost of land purchase. Overall, I tested several ways to successfully fund conservation investments and highlight potential benefits and shortfalls of each.

Preface

Though I took the lead in study design, collecting and analyzing the data, formulating the conclusions and writing for each chapter, this thesis benefitted greatly from generous advice and data sharing by my colleagues. The following contributions by co-authors and other colleagues are gratefully acknowledged:

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Chapter 1: Introduction

Background

Investments in habitat and biodiversity conservation are critically needed given mounting effects of global climate change and unprecedented rates of ecological degradation and species extinction recorded during the 'Anthropocene' (Ehrlich and Pringle 2008, Butchart et al. 2010, Bayon and Jenkins 2010, Estes et al. 2011). However, as more regions of the world are converted to human use (Luck 2007, Ellis 2011), we are also experiencing a shift in the traditional targets of conservation area design from protecting 'ecologically intact' landscapes (Noss and Cooperrider 1994, Possingham et al. 2000, Moore et al. 2004, Nicholson et al. 2006) to restoring degraded habitat by prioritizing conservation investments under triage (Sinclair et al. 1995, Lamb et al. 2005, Bowen et al. 2007, Chazdon 2008, Vellend et al. 2008, Wilson et al. 2011). This shift in emphasis raises many questions about the efficacy of alternative approaches and likelihood of funding each. The overall goal of this thesis was to evaluate alternative ways of funding conservation initiatives. To reach that goal, the definition of what is a successful investment is needed, along with assessments of alternative mechanisms for funding conservation goals.

Habitat loss and fragmentation are widely recognized as contributing to native species decline (Wilcox and Murphy 1985, Murcia 1995, Vitousek et al. 1997, Gonzalez et al. 1998), and it often accelerates biodiversity loss by facilitating human commensal species that drive changes in vegetation structure and plant and bird species richness and abundance (Andrén 1994, Heikkinen et al. 2004, Allombert et al. 2005, Martin et al. 2011). Forest-reliant birds are a particular concern in fragmented landscapes because they decline rapidly in richness and abundance as forest patch size is reduced or human development increased

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(Blair 1996, Boulinier et al. 2001, Groom and Grubb 2002), due in part to the effects of exotic competitors associated with human-dominated landscapes (Andrén 1994, Jewell and Arcese 2008, DeWan et al. 2009). For example, in the densely populated Georgia Basin of British Columbia (BC), Canada, ≥60% of the Coastal Douglas-fir (CDF) ecozone has been converted to exclusive human use (Austin et al. 2008) and less than 0.3% of historical old growth forest cover still exists (>250 years; Madrone Environmental Services, 2008). In regions like this, identifying, conserving and restoring mature forest stands (>80 years) may represent the only viable path to conserving endemic forest plant and animal communities outside of intensively managed reserves or botanical gardens in future, and it thus remains a key objective of conservation research (Bowen et al. 2007, Gardner et al. 2009).

Here I define restoration as the assisted regeneration of degraded habitat (Chazdon 2008). Prior to European colonization the CDF occurred as uneven-aged forest (often >300 years old) dissected by shallow and deep-soil meadow and woodland communities (Meidinger and Pojar 1991, Mosseler et al. 2003) maintained in part by aboriginal land management practices to enhance hunting opportunities and root and fruit harvests (MacDougall et al. 2004, Dunwiddie and Bakker 2011, McCune et al. 2013, Turner 2014). The goal of restoration of forests for this thesis is to speed up forest regeneration after a history of logging in ways that emulate natural processes, but allow us to expedite regeneration into old growth forest states (Lamb et al. 2005, Chazdon 2008, Wilson et al. 2011). For with historic shallow and deep-soil meadow and woodland habitats the goal of restoration here is to actively prevent forests from encroaching into meadow habitats and emulate aboriginal land management practices (Devine et al. 2007, Dunwiddie and Bakker 2011).

In the absence of detailed data on species and habitat distribution, conservation planners often rely on course-scale targets and 'ad hoc' criteria for conservation prioritization, due to the cost or difficulty of collecting distribution data for rare species or communities that are the targets of conservation, but experience shortcomings as a consequence (Ando et al. 1998, Arponen et al. 2008, Wiersma and Nudds 2009, Fuller et al. 2010). An alternative approach is to develop predictive distribution maps for representative taxa that, based on known habitat affinities, ease of detection and abundance, provide cost effective estimates of native species richness that can be used to prioritize areas of conservation interest (review in Branton & Richardson 2011). For example, many vagile butterflies, birds and bats are highly adept at crossing wide gaps after disturbances (Lees and Peres 2009) and select habitats linked to forest successional stage (Chazdon et al. 2009), thus making them ideal indicators of forest structure and habitat quality. Following this approach, De Wan et al. (2009) used land cover data and counts of 'forest interior birds' to identify forest patches likely to support old growth forest bird species communities. These and other results suggest that predictive distribution maps of indicator species could become valuable tools for identifying potentially relatively intact communities of native species in human dominated landscapes (Scott et al. 1993, Müller et al. 2003, Hijmans and Graham 2006). Occurrence data have been used in these mapping efforts (Kavanagh and Stanton 2005, Jewell et al. 2007), but due to their inability to detect all species during sampling, these methods are negatively biased in their predictive abilities, which is why new methods have been developed that specifically account for imperfect detectability (Mackenzie et al. 2002, 2006).

Recently, it has been suggested that an emphasis on species richness (α-diversity) might not be sufficient to maximize community resilience in the face of current drastic changes, and an inclusion of community dissimilarity (β-diversity) models has been proposed to create more effective conservation networks in the future (Arponen et al. 2008, Mokany et al. 2011). Therefore, in highly heterogeneous systems conservation attempts should ideally not only strive to maximize native species richness of e.g., herbaceous, woodland or forest communities (α-diversity), but also seek to incorporate dissimilarities in community composition between sites (β-diversity) to represent the entire set of species communities of a system (Ferrier et al. 2007, Marsh et al. 2010, Mokany et al. 2011). To date there are only three studies published that have, to some extent, modeled both α- and β-diversity in relation to conservation planning (Fairbanks 2001, Wiersma and Urban 2005, Marsh et al. 2010); an additional study used simulated data in a theoretical approach (Arponen et al. 2008). These studies have suggested that a focus on β-diversity is likely to enhance long term persistence in diverse species assemblages and reserve networks. Mokany et al. (2011) have recently proposed a framework for combining α- and β-diversity models, but to the best of my knowledge no studies have evaluated that proposed framework or applied these findings and suggestions to systematic conservation planning to investigate the potential advatages of using these summary metrics for meeting conservation targets.

The conservation of biodiversity is further challenged by competing societal demands for land and public funds, particularly in human-dominated landscapes wherein the rapid conversion of native landscapes by humans results in the fragmentation and loss of natural habitats, isolation of formerly connected populations, and dramatic reductions in the abundance of historically wide-spread species (Bennett and Arcese 2013, Arcese et al.

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2014). In such landscapes, the costs of purchasing privately-held land or compensating owners for lost opportunities incurred as a result of conservation can be substantial (Naidoo et al. 2006, Wunder 2007). One way of making conservation more affordable is therefore by offsetting purchase and convenient costs via payments for ecosystem services. Ecosystem services, defined as the direct and indirect provision of benefits by ecosystems to people, represent one framework for characterizing human benefits of ecosystem conservation via food production, water purification, and carbon sequestration (Millenium Ecosystem Assessment 2005). Advocates of this approach hope that this concept will expand conservation activities and contribute to its funding (Goldman et al. 2008), while also benefitting people (Daily et al. 2000, Chan et al. 2007).

However, although individual ecosystem services have been characterized and valued, e.g., carbon sequestration as a means of climate regulation (Arroja et al. 2006), very little is known about their compatibility with biodiversity conservation. Recent research suggests that areas with high levels of biodiversity are not necessarily the areas that would be prioritized for ecosystem services (Chan et al. 2006, Naidoo et al. 2008, Egoh et al. 2009, Thomas et al. 2012). Because ecosystem services can also be estimated as co-benefits or opportunity costs of conservation such as forgone timber harvest or agriculture revenue (Naidoo et al. 2006, Egoh et al. 2007), there is an urgent need for research that explores alternative frameworks for incorporating potential costs and benefits into conservation planning (Chan et al. 2011). Chan et al. (2011:2) have identified two key questions that need to be addressed: "(1) how well do biodiversity and ecosystem services correlate across space? And, given imperfect correlations, (2) how can we use existing planning tools to most effectively prioritize for a range of management considerations within a particular

landscape and on a constrained budget?" Some of those problems might be avoided by matching ecosystem service values to specific elements of interest (like growing forests).

The sale of carbon credits has the potential to increase the effectiveness of conservation if forests exhibiting high carbon sequestration or stock overlap areas of high biodiversity (Nelson et al. 2009, Chan et al. 2011). To the degree that carbon and biodiversity values overlap, carbon credit sales could be used to protect forests that would otherwise be logged (Chan et al. 2011, Douglass et al. 2011) or to restore those that still support valued old forest communities (Schuster and Arcese 2013). Payments for carbon may offset timber values sufficiently that property owners elect to maintain forest for its amenity and carbon values. In addition, it is likely that protecting forests ensures the provision of additional ecosystem services, such as the water supply or storage capacity of forest ecosystems, as illustrated at the Panama Canal, where protecting forest lands insures water flow to the canal (Condit et al. 2001).

Global concern for the consequences of climate change has motivated 35 nations and 13 sub-national jurisdictions to put a price on carbon (Climate Commission 2013). For example, Douglass et al. (2011) suggested that managers can conserve more biodiversity, capture more carbon and reduce the cost of conserving savannah ecosystems by integrating these goals as conservation plans are developed. Douglass et al. (2011) also showed that priorities for land stewardship payments that account for an emerging carbon market are likely to deliver greater benefits at lower cost than if carbon markets are ignored. However, several outstanding issues arise when considering the role of carbon markets in forest restoration. One issue is that biodiversity values may be lower in stands with the highest return from carbon sequestration sales because sequestration typically peaks in stands of

intermediate age (e.g., Pregitzer and Euskirchen, 2004), although old forests act as carbon sinks and continue to accumulate carbon over time, but may do so at slower rates (Luyssaert et al. 2008), resulting in the highest return on standing carbon credit sales. Thus, it remains unclear whether protecting forests with high standing carbon or high carbon uptake rates might yield different results for conservation efforts.

Systematic conservation planning attempts to achieve a given conservation target (e.g., reserve at least 10% of every species range) for least cost (Margules and Pressey 2000, Naidoo et al. 2006). Because information on land acquisition costs is often hard to come by, most conservation planning tools use simplistic proxies for economic cost, such as the total area reserved (Dobson et al. 1997, Araújo et al. 2002). Several studies have suggested that incorporating spatially explicit information about economic costs, e.g., on land purchase costs, into conservation planning can be as or more important than incorporating spatial heterogeneity of biodiversity features (Ando et al. 1998, Polasky et al. 2001, Ferraro 2003, Naidoo et al. 2006), and that inaccurate cost data may undermine efficient conservation planning (Ferraro 2003, Perhans et al. 2008). As a result, the role of economic cost in spatial prioritization has recently been emphasized (Naidoo et al. 2006, Carwardine et al. 2008, Bode et al. 2008b, Moilanen and Arponen 2011). Several studies demonstrate efficiency gains as a consequence of including spatially explicit economic cost data into conservation plans (Naidoo et al. 2006, Bode et al. 2008b, Chan et al. 2011), despite the possibility that real purchase costs may differ from those predicted, such as when the price landholders are willing to accept differs from assessed values (Carwardine et al. 2010, Knight et al. 2011). This suggests that the inclusion of spatially explicit land purchase cost estimates has the potential to significantly influence conservation area design results.

The cost of land purchase and opportunity costs of conservation can be substantial in human-dominated landscapes where private ownership prevails (Naidoo et al. 2006, Wunder 2007). In these cases, one cost-effective route to conservation may include private land conservation easements or covenants, which are registered on title to prohibit future land use change likely to reduce conservation values in exchange for monetary or other compensation (Merenlender et al. 2004, Knight et al. 2011). Conservation covenants may cost less in the short- and long-term (Pence et al. 2003) but are riskier with respect to the maintenance of conservation goals in the future. Nevertheless, several studies indicate that in areas where land owners are unwilling to sell, they may be interested in committing to some form of conservation covenant or voluntary conservation agreement (Langholz and Lassoie 2001, Winter et al. 2005, Knight et al. 2010). Given the growing importance of private land conservation initiatives (e.g., Fishburn et al. 2009), more work is needed to understand what benefits might be achieved by offsetting the opportunity costs of land development, rather than engaging solely in land purchase. To date, however, very few studies have addressed the potential complexities of such investments to ask if covenants or land purchase are most likely to maximize the conservation of high biodiversity value areas in a region (Armsworth and Sanchirico 2008, Fishburn et al. 2009). In particular, serious questions exist about the long-term costs of monitoring and enforcing covenants given the risk that property owners violate the covenant or challenge it legally (Knight et al. 2010).

Where land is mostly held privately, land values are high, and there is consistent or increasing development pressure, alternatives to outright acquisition may represent the most efficient route to conservation in the short to medium term. Because covenant violations and related legal disputes appear to be increasing and may become more frequent after repeat

sales (Rissman & Butsic 2011), the long-term costs of covenants due to monitoring, legal or remediation costs to covenant holders could be substantial, although no study has empirically investigated this yet (Byers et al. 2005). Violations may also reduce conservation values via land conversion or other effects. If these costs prove substantial the value of covenants as conservation tool may be over-estimated.

To answer the outstanding questions raised so far and reach the overall thesis goal, I conducted assessments of alternative mechanisms for funding conservation goals. I used a case study from the Coastal Douglas-fir Biogeoclimatic zone (CDF) of northwestern North America to explore cost-efficient routes to restore critically endangered Old Forest and Savannah habitats and associated bird communities. The CDF provided an outstanding example of an area to explore my research questions, as regional, provincial and federal authorities own <20% of the area and only ~9% is managed for conservation.

The Coastal Douglas-fir Biogeoclimatic zone

The Coastal Douglas-fir (CDF) ecozone of the Georgia Basin (British Columbia, Canada) is a classic example of an endangered but extraordinarily diverse region that has been rapidly converted to exclusive human use ($\geq 60\%$) (Austin et al. 2008) and, as a result, retains $\leq 0.3\%$ of historic old forests (>250 years) (MES 2008) and $\leq 10\%$ of oak woodland and savannah (Lea 2006), both of which provide habitat for 117 species at risk of extirpation (Austin et al. 2008). Because regional, provincial and federal authorities own <20% of the region and only ~9% is already conserved, cost-efficient routes to conservation are urgently needed.

Prior to European colonization the CDF occurred as uneven-aged forest (often >300 years old) dissected by shallow and deep-soil meadow and woodland communities (Meidinger and Pojar 1991, Mosseler et al. 2003) maintained in part by aboriginal land management practices to enhance hunting opportunities and root and fruit harvests (MacDougall et al. 2004, Dunwiddie and Bakker 2011, McCune et al. 2013, Turner 2014). Oak woodland and savannah community distributions are predicted to shift under future climate conditions, and only a small fraction of the current protected areas have the potential to accommodate this shift (Pellatt et al. 2012). The resulting land use heterogeneity within the region and potential for humans to directly or indirectly affect native species richness (Gonzales and Arcese 2008, Martin et al. 2011, Bennett et al. 2012, Schuster and Arcese 2013) make this system ideal for studying trade-offs involved when attempting to maximize α- versus β-diversity in conservation plans, while simultaneously maximizing ecosystem service values represented as total carbon stored or sequestered, as well as contrasting land purchase and conservation covenant scenarios.

Thesis chapters

My thesis develops and describes a novel approach to predict bird species distributions in the CDF, uses a systematic approach to conservation prioritization of high biodiversity areas incorporating forest carbon sales, and explores the long term cost effectiveness of conservation covenants. My overall goal in this work was to provide guidance and examples of rigorous, cost-effective approaches to use for conserving lands in human-dominated landscapes. Below I outline the content and rationale for three chapters, designed to identify high biodiversity areas and evaluate alternative ways of successfully funding conservation initiatives.

Chapter 2: Conservation often focuses on 'ecologically intact' habitats with little human influence. But, where all such habitats have been lost or modified, identifying promising conservation and/or restoration targets is a key goal. I describe a direct approach to identify high conservation value targets using predictive distribution maps of taxa that, based on habitat affinity, ease of detection and abundance can be used to infer native species richness and prioritize conservation investment. For this analysis I used 1169 avian point counts in a 1,560 km^2 study area, remote-sensed data and models incorporating imperfect detectability to predict habitat occupancy for 18 widely-distributed native birds, 12 of which were determined by experts to be positive indicators of old-forest conditions. For Chapters 3 and 4 I extended the study area to 2,520 km^2 by adding 602 avian points counts for a total of 1770. Forest-association scores for these 12 species were then used as weights in a composite distribution map of the probability of community occurrence, highlighting potentially high quality old forest stands. My results indicate that the use of composite maps of widespread indicators improve site prioritization by incorporating the behavioural and demographic responses of a diverse range of bird species to patch size, configuration and adjacent human land use.

Chapter 3: Conservation initiatives to protect and restore valued species and communities in human-dominated landscapes face huge challenges linked to the cost of acquiring habitat. This challenge is particularly severe in human-dominated landscapes, where private ownership prevails and the cost of purchasing properties or compensating land holders for lost opportunity incurred as a result of conservation can be substantial. One way of making conservation more affordable is by offsetting land purchase costs via payments for ecosystem services. In this chapter I ask how the sale of forest carbon offsets could

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reduce land acquisition costs. A second issue is whether to develop conservation plans that maximize species richness (α-diversity) within habitats or maximize dissimilarities in community composition (β-diversity) to accommodate landscape complexity and species that utilize multiple habitats. Under climate change, it has been suggested that an emphasis on community dissimilarity (β-diversity) may deliver more robust conservation plans than those based on species richness. Here I examine the potential value of carbon credit sales to offset land acquisition costs by developing conservation area designs that maximize β or α-diversity in native old forest and savannah bird communities in relation to forest structure and human land use. Specifically, I ask how protecting forests with high carbon storage versus high carbon uptake is likely to affect conservation outcomes given alternative carbon markets, and in relation to increasing targets for the total area conserved.

Chapter 4: Conservation initiatives to protect and restore valued species and communities in human-dominated landscapes face huge challenges linked to the cost of protecting habitat. One cost-effective route to conservation may include private land conservation covenants; although serious questions exist about long-term monitoring and enforcement costs of covenants given the risk that owners might violate or challenge them in court. I explore the long term (100 years) cost-effectiveness of conservation covenants, defined as the fraction of high-biodiversity landscape protected given the total conservation network cost, as compared to a strategy of land purchase. To do so I ask the following questions: 1) To what extent does dispute rate reduce the cost-effectiveness of conservation covenants as compared to purchasing conservation land? 2) Assuming that covenant violations reduce the area of high-biodiversity landscapes protected, what is the area of the

high-biodiversity landscape that is protected after 100 years, given the alternate tactics of financing and managing conservation covenants versus purchasing conservation land?

In Chapter 5, I highlight how my thesis research has contributed to reaching the goal of evaluating alternative ways of successfully funding conservation initiatives by identifying high biodiversity areas (Chapter 2), examining the role of offsets via carbon sequestration ecosystem service payments (Chapter 3) and conservation covenants (Chapter 4). I further generalize the findings to make them more tangible and applicable for practical application and highlight overall conclusions that can be drawn from this study. I point out some of the limitations of this study related to biodiversity, land value and conservation covenant data and discuss adjustments needed before my theoretical analysis can facilitate land management in practice. I examine the implications this work could have for management practices and summarize ongoing applications of my work at the local, regional and provincial scale. Finally I provide recommendations for future work, such as incorporating changes to species distributions due to climate and land use change and ways to improve the conservation covenant cost-effectiveness analysis.

Chapter 2: Using bird species community occurrence to prioritize forests for old growth restoration[1]

Introduction

Conservation area design often focuses on protecting 'ecologically intact' landscapes subject to minimal human influence in marine, terrestrial and mixed settings (Nicholson et al. 2006, Hazlitt et al. 2010), but different approaches are required where most or all intact habitat has already been lost (e.g., Vellend et al. 2008, Wilson et al. 2011). In these cases, habitat restoration is often the focus of conservation investment (Bowen et al. 2007, Chazdon 2008), ideally using a prioritization framework that accounts for the costs and benefits of restoration and is spatially and temporally explicit (Wilson et al. 2011). For example, Bowen et al. (2009) suggested that conserving older forests where regeneration occurs in response to small or large scale disturbances could represent a cost effective way to restore old-growth conditions and communities (see also Jönsson et al. 2009). I present a method to identify regenerating forest patches most likely to support rich communities of forest-associated birds at the landscape scale.

Habitat loss and fragmentation are widely recognized as contributing to native species decline (Murcia 1995, Gonzalez et al. 1998), in part by facilitating human commensal species that reduce the abundance of native plant and bird populations (Andrén 1994, Allombert et al. 2005, Martin et al. 2011). Forest-associated birds are a particular concern in fragmented landscapes because they decline rapidly in richness and abundance as

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forest patch size is reduced or human development increases (Boulinier et al. 2001, Groom and Grubb 2002), due in part to effects of competitors in human-dominated landscapes (Andrén 1994, Jewell et al. 2007, DeWan et al. 2009). In the densely populated Georgia Basin of British Columbia (BC), Canada, ≥60% of the Coastal Douglas-fir (CDF) ecozone (Meidinger and Pojar 1991) has been converted to exclusive human use (Austin et al. 2008) and just 0.3% of historical old growth forest cover still exists (>250 years; Madrone Environmental Services 2008). In regions like this, identifying, conserving and restoring mature forest may represent the only viable path to conserving endemic plant and animal communities outside of intensively managed reserves (Bowen et al. 2007, Gardner et al. 2009). My goal here is to test if predictive occupancy maps of forest-associated bird species occurrence can be used to identify forest stands that could form core old growth reserves in future.

The cost and effort required to reliably map the distribution of rare species and communities targeted for conservation often results in the adoption of coarse-scale targets and 'ad hoc' criteria in site prioritization plans, but frequently at the cost of ecosystem representation and species diversity (e.g., Arponen et al. 2008, Fuller et al. 2010). An alternative approach is to develop predictive distribution maps for representative taxa that, based on known habitat affinities, ease of detection and abundance, effectively estimate native species richness and thus can be used to prioritize areas of conservation interest (e.g., Branton and Richardson 2011). For example, many butterflies, birds and bats respond to gaps after disturbances (Lees and Peres 2009) or select habitats linked to forest successional stage in ways that make them ideal indicators of forest structure and quality (Chazdon et al. 2009). Following this approach, DeWan et al. (2009) also used land cover and pooled

observations of 'forest interior birds' to map habitat likely to support late successional birds but did not validate their approach or correct for uneven sampling. Nevertheless, these and other authors suggest that predictive distribution maps of common indicators could help planners identify habitat patches likely to support valued communities of native species (Müller et al. 2003, Hijmans and Graham 2006) if imperfect detection (Mackenzie et al. 2002), non-representative sampling issues and validation are addressed. Here, I extend and validate an approach to create composite maps of forest-associated bird communities using aerial photographs of the distribution of mature and older forest habitat.

Specifically, I asked professional ornithologists to rank 18 forest bird species by their association with old forest characteristics (e.g., large diameter trees, well-developed understory, light gaps and coarse woody debris) in my 1,560 km^2 Georgia Basin study area (Figure 2.1). For each species identified as a potential indicator of older, coastal Douglas-fir stands, I estimated its probability of occurrence in my study area using avian point counts and a wide range of remotely-sensed variables linked to coarse and fine scale attributes of stand structure and land cover. I next created a spatially-explicit, composite map by weighting each species' occurrence map by its association with old forest stands as determined by expert opinion. By doing so, I was able to predict the occurrence of forest-associated bird communities and to identify particularly species-rich old forest patches for potential recruitment into old growth conditions. Last, I compared my predictions to an independent map of mature and older forest (>80 years) distribution to ask if my map might improve habitat prioritization based solely on forest age and patch size.

Methods

Study area and sampling methods

I focused on a 1,560 km^2 portion of the Coastal Douglas-fir zone of BC (2,520 km^2) that includes >2000 islands from 0.0003 - 32,000 km^2 (e.g., Vancouver Island). Roughly 40% of the CDF region still occurs as uneven-aged forest, interspersed with shallow soil balds, deep soil meadows and woodland/savannah habitat. Mature CDF forests support long-lived conifers and a mainly deciduous tree and shrub sub-canopy subject to disturbance and, as a result, are structurally complex (Meidinger and Pojar 1991, Mosseler et al. 2003). As part of a larger program on avian conservation (e.g., Jewell et al. 2007, Martin et al. 2011), trained observers conducted point counts on 45 islands from 30 Apr – 11 Jul, 2005 - 2010 (except 2006, Figure 2.1). Trained observers recorded all birds detected in a 50m radius in a 10 min period between 5 AM – 12 PM, at 629 sample locations (mean distance between all locations = 19 km). A very small percentage (0.04%) of location pairs were identified in the lab as <100m apart, due in part to GPS error in forested habitats, because field observers were careful to local sample points >100m apart using maps. Total visits to each location ranged from 1-12 (mean = 1.86), with each location recorded by handheld GPS (GPS60, Garmin Ltd, Kansas, USA).

Expert rankings

I asked 12 professional ornithologists with >5 years of experience with local birds to estimate the degree of association of 18 candidate species (Table 2.1) to old forest and woodland CDF habitats. Specifically, experts ranked species' according to their expected association (low = -1, medium = 0 or highly associated = 1) with each of 6 successional habitats in present-day CDF forests: herbaceous, shrub/herb, pole/sapling, young forest,

Figure 2.1: Location of the study area and point count locations.
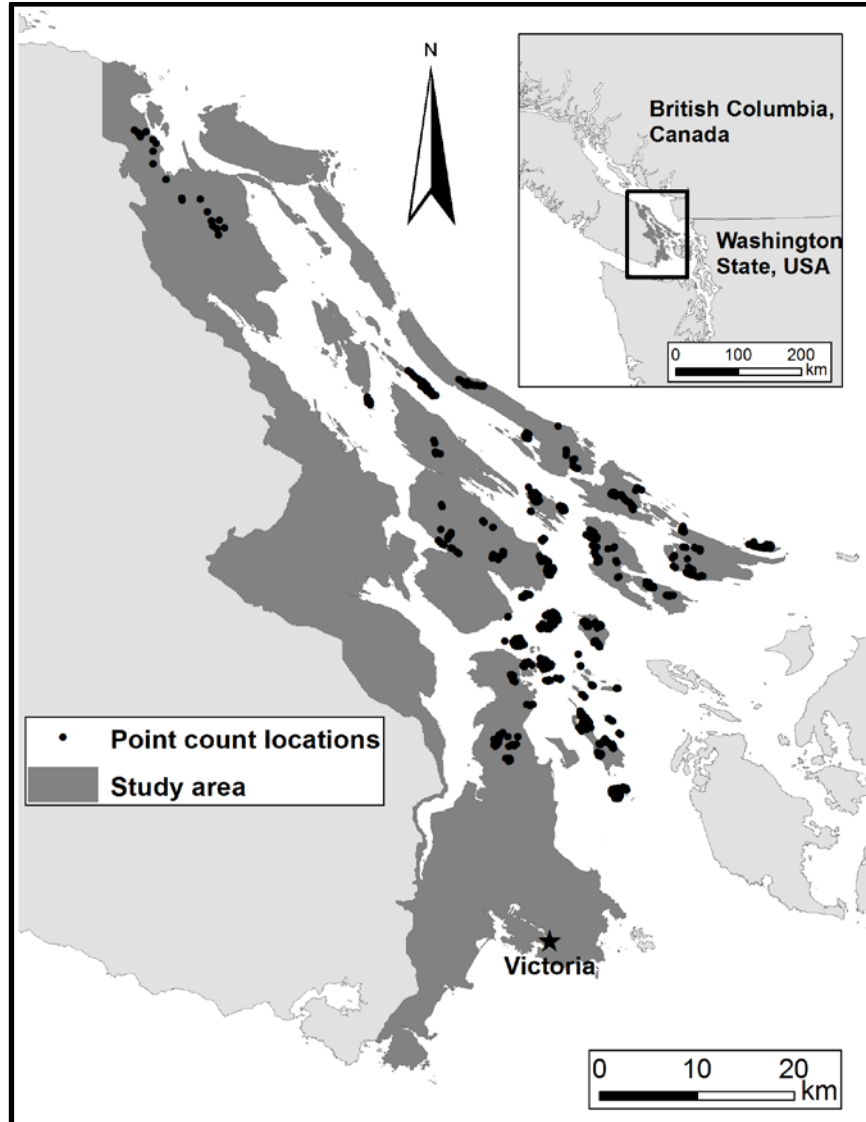
mature forest and old forest (for details see Appendix 1). Expert ranks were then averaged

for each species and habitat type. Variation between experts was 0.14 on average, indicating

stable habitat association estimates between experts. Old forest association scores for each

species were then calculated by summing a species' rank in each habitat, multiplied their

weights. I tested a total of 5 weighing schemes, but because results did not vary considerably

I used the following weights emphasising older forest: herbaceous (-2), shrub/herb (-1),

pole/sapling (-0.5), young forest (+0.5), mature forest (+1) and old forest (+2). Doing so

resulted in a score for each species that ranged from a minimum of -7, indicating no

association to old forest structure, to a maximum of 7, indicating a strong association to old

forest structure. This score was then standardized to fall between -1 and 1 by dividing by the

maximum value possible (7). All birds with positive forest association scores were therefore

considered to be members of the CDF old forest bird community, with those species having

higher forest association scores contributing most to composite maps (see below).

Table 2.1: Old forest scores for the 18 bird species I elicitated expert opinions for. For each
species that had a positive score the species' weight used in the community score map is
shown. The higher a species' old forest score the higher its community score map weight.
Weights sum to 1.

Common name	scientific name	old forest score	weight
brown creeper	*Certhia americana*	0.81	0.109528
chestnut-backed chickadee	*Poecile rufescens*	0.75	0.101475
dark-eyed junco	*Junco hyemalis*	-0.03	---
golden-crowned kinglet	*Regulus satrapa*	0.76	0.103085
olive-sided flycatcher	*Contopus cooperi*	0.45	0.0615
orange-crowned warbler	*Vermivora celata*	-0.05	---
pacific wren	*Troglodytes pacificus*	0.63	0.085
pacific-slope flycatcher	*Empidonax difficilis*	0.63	0.085
pine siskin	*Carduelis pinus*	0.69	0.093
purple finch	*Carpodacus purpureus*	0.58	0.078
red-breasted nuthatch	*Sitta canadensis*	0.79	0.106
rufous hummingbird	*Selasphorus rufus*	-0.07	---
song sparrow	*Melospiza melodia*	-0.36	---
spotted towhee	*Pipilo maculatus*	-0.03	---
Townsend's warbler	*Dendroica townsendi*	0.69	0.093
white-crowned sparrow	*Zonotrichia leucophrys*	-0.66	---
Wilson's warbler	*Wilsonia pusilla*	0.06	0.008
yellow-rumped warbler	*Dendroica coronata*	0.56	0.076

Landscape covariates

Because birds respond to many fine and coarse scale habitat features (e.g., Lawler and Edwards 2006) I developed covariate descriptors of landscape condition and context using coarse (1km) and fine (100m) scale features to advance early work conducted only at coarse scales (e.g., DeWan et al. 2009). For modelling species detection and occurrence, I chose candidate predictors based on their proven ability to predict species occurrence at site and landscape levels in similar exercises or regions (e.g., Guisan and Thuiller 2005, Jewell et al. 2007). All covariate names appear in Table 2.2 and were derived from the following sources: (i) Terrain Resource Information Management (TRIM, http://archive.ilmb.gov.bc.ca/crgb/pba/trim/specs/specs20.pdf, date accessed: 2011-10-10); (ii) Sensitive Ecosystems Inventory (SEI): East Vancouver Island and the Gulf Islands (http://www.env.gov.bc.ca/sei/, date accessed: 2011-10-10); (iii) Earth Observation for Sustainable Development Landcover (EOSD LC 2000, Wulder et al. 2008); (iv) aerial photographs to calculate the islands sizes; and (v) Terrestrial Ecosystem Mapping (TEM) of the CDF Zone (MES 2008). To reduce the potential number of covariates considered overall, I consolidated TEM ecosystem units using rules described in Appendix 2. TRIM data is continuously updated and assessed for quality control. SEI data represent a more opportunistic data source and potentially lacks some of the true occurrences of some attributes, but still represents a valuable and often used data source. EOSD data stems from remotely sensed satellite information and due to its nature of training and predicting data sets the reliability of this data source is related to the accuracy and predictive powers of the training data set, which is about 77% in this case (Wulder et al. 2008). TEM data is based on manual interpretation of air photos, hence the data quality varies with airphoto interpreter,

but generally manual image interpretation is still superior to automated approach. This 0indicates that TEM data would be the most reliable data source available in the study region.

My dataset comprised 29 predictor covariates of site and landscape condition with correlations less than 0.7, plus 5 spatial covariates (Table 2.2), derived at each of 629 avian point count locations. The data source satellite and aerial photography imagery was collected between 2002 and 2004, which was in part supplemented by ground work until 2008. All covariates were created using Hawth's Tools (Beyer 2004), ArcGIS 9.2 (ESRI 2009) and Geospatial Modelling Environment (Beyer 2010) in conjunction with ArcGIS 10 (ESRI 2010) and R v. 2.12.2 (R Development Core Team 2010). Due to their widely varying scales, all covariates were standardized about their mean value, to ensure that importance was not driven by measurement scale (White and Burnham 1999).

Table 2.2: Detectability and occupancy model covariate description including data source and covariate abbreviation used here. The data source satellite and aerial photography imagery was collected between 2002 and 2004, which was in part supplemented by ground work until 2008.

Source	Covariate decription	Abbreviation
TRIM	Total amount of unpaved road length within a 1km buffer	rdl_up_1k
	Total amount of unpaved road length within a 100 buffer	rdl_up_100
	Total amount of paved road length within a 1km buffer	rdl_p_1k
	Total amount of paved road length within a 100 buffer	rdl_p_100
	Nearest road	near_road
	Nearest freshwater source	near_frshw
	Nearest shoreline	near_saltw
Aerial photographs (small islands) and TRIM	Island size	Is_size
Terrestrial Ecosystem Mapping of the Coastal Douglas-Fir Biogeoclimatic Zone	Urban/industrial area within a 1km buffer	URB_1KM
	Rural/agriculture area within a 1km buffer	RUR_1KM
	Forest cover area within a 1km buffer structural stages open and shruby	FOR0_1KM
	Forest cover area within a 1km buffer including structural stages closed and young forest	FOR1_1KM
	Forest cover area within a 1km buffer including structural stages mature and old forest	FOR2_1KM

Source	Covariate decription	Abbreviation
	Savannah area within a 1km buffer	SAV_1KM
	Shrub area within a 1km buffer	SHR_1KM
Terrestrial Ecosystem Mapping of	Urban/industrial area within a 100m buffer	URB_100
the Coastal Douglas-Fir	Rural area within a 100m buffer	RUR_100
Biogeoclimatic Zone	Forest cover area within a 100 m buffer structural stages open and shrubby	FOR0_100
	Forest cover area within a 100 m buffer including structural stages closed and young forest	FOR1_100
	Forest cover area within a 100 m buffer including structural stages mature and old forest	FOR2_100
	Savannah area within a 100m buffer	SAV_100
	Shrub area within a 100m buffer	SHR_100
	Distance to nearest Urban area	Near_Urb
	Coniferous crown closure (0-25%) area within a 100m buffer	CR_CL0
Earth Observation for Sustainable	Coniferous crown closure (26-60%) area within a 100m buffer	CR_CL_2
Development(EOSD) Landcover	Coniferous crown closure (>60%) area within a 100m buffer	CR_CL_3
	Broadleaf area within a 100m buffer	BRDLF
Sensitive Ecosystems Inventory	Second growth and older forest class area within a 100m buffer	OF_100
(SEI)	Second growth and older forest class area within a 1km buffer	OF_1KM

Occupancy and detection models

The large size of my study area required me to compile data from related surveys conducted during a single 9 week period (Jewell et al. 2007, Martin et al. 2011) but differing in sampling intensity between years and precluding reliable estimates of colonization and extinction (MacKenzie et al. 2003). I also assumed no variation in site occupancy across years to minimize model complexity, thus assumed a closed population for all species (Mackenzie et al. 2002). The R package unmarked v. 0.8.7 (Fiske and Chandler 2011) provided the framework for all species models, which necessarily include two parts: occupancy and detection (Mackenzie et al. 2002). To estimate detectability, I used 5 site specific covariates as potentially influencing detectability, including: 1) crown closure >60% within 100m, 2) island size (ha), 3+4) forest cover within a 1km buffer of each site classified by structural stage ('early', closed and young forest; 'late', mature or old forest). My fifth covariate was observer identity. For each of 12 focal species I fitted all 32 detectability models (without parameterizing occupancy) and then ranked each by AIC (Akaike 1974) to select top-ranked models for further analysis. To accommodate my reduced but still extensive set of 29 predictor covariates for occupancy modelling, I first used a 'stepwise' covariate selection procedure that was combined with an Information Theoretic (IT) approach, where all covariate combinations are tested. Due to the large number of potential models 2^{29}, I created a hybrid forward-backward stepwise-IT approach that tested subsets of models, before moving on to the next iteration in the modeling approach. In each iteration the model(s) with the lowest AIC values (or a range of AIC values) were retained and used as the new base model for the next model routine iteration. I linked this function to the unmarked package to create a candidate set of models based on the statistical significance of individual

covariates and AIC (Appendix 3). I then ranked all candidate models by AIC and averaged those with ΔAIC ≤ 7 from the top ranked one (Burnham and Anderson 2002).

To test for and account for residual spatial auto correlation in model predictions I added spatial polynomial terms to all averaged models above (Easting, Northing, Easting^2, Northing^2, Easting*Northing; simplifying slightly the procedure of Borcard et al. (1992). The inclusion of spatial covariates resulted in an additional 32 models for each species that were again ranked by AIC and assessed for covariate significance, and averaged (all models with ΔAIC ≤ 7). Goodness-of-fit (GOF) for all models was assessed following MacKenzie and Bailey (2004) and the parametric bootstrapping function in the unmarked package, where H_0 is no difference between χ_{obs}^2 (test statistic for observed data) and χ_b^2 (test statistic for bootstrapped data), using 1000 bootstrap permutations.

Predictive maps

I created landscape-level predictive occupancy maps over my 1,560 km^2 study area using 156,000 100x100m polygons. For each polygon centroid I generated a covariate set identical to that used for survey points, and then estimated probability of occurrence based on my averaged models for each of the focal species. To consolidate focal species maps into an index of forest-associated bird species richness, I created a score for each polygon resembling a single-species habitat suitability index (e.g., Guisan and Thuiller 2005, Beaudry et al. 2010). Specifically, I calculated the polygon scores by summing the weighted probability of occurrence of each species linked to old forest structure via expert questionnaires (Martin et al. 2005a). This process yielded a weighted, forest-association community score that ranged from 0 (no forest-associated species present) to 1 (all forest-associated species present) for each of the map polygons.

I compared individual occupancy maps for each species to the community score map visually and using fuzzy-set map comparison to achieve objectivity, evaluation and repeatability (Hagen 2003), to ask if the community map was driven by a single species and to support my interpretation of the expert-weighted community map. To reduce fuzziness (location-specific spatial variability and noise) I followed Jewell et al. (2007) by using the Map Comparison Toolkit (Visser and de Nijs 2006) to compare each single species map to the weighted forest-associated bird community map, and recorded the fuzzy modification of the Kappa statistic (K_{fuzzy}), the fuzzy version of fraction correct (P_A), and the absolute value of the difference between two maps based on a cell-by-cell comparison ($Diff_{abs}$) (Jewell et al. 2007). K_{fuzzy} is similar to the traditional Kappa statistic because the expected percentage of agreement between two maps is corrected for the fraction of agreement expected by randomly relocating all cells in each map (Visser and de Nijs 2006).

Lastly, to confirm that my bird community predictions did emphasise older forest patches, I tested if the distribution of forest-associated bird communities I identified corresponded with an independently calculated mature and older forest (TEM-MF; >80 years) layer from the TEM data. I did so by calculating the area-weighted mean of my predicted old forest bird community scores for each TEM-MF polygon, averaged them, and then compared this to the average community score over the whole study area, to investigate if the average TEM-MF community scores were higher than the landscape average. I also investigated the change in the community score with change in the TEM-MF area by plotting area weighted mean community scores against patch size as well as an independent agricultural area dataset to reveal potential drawbacks of relying solely on coarse-grained forest metrics to identify areas for high value sites for conservation and/or restoration.

Results

Expert elicitation allowed us to estimate degree of old-forest association for each of 18 candidate bird species, with 12 species obtaining positive scores and thus being adopted as members of an old forest bird community (Table 2.1). Forest-association scores for individual species ranged from 0.06 for Wilson's warbler to 0.81 for the brown creeper (Table 2.1).

Species occupancy and detection

Detectability varied greatly among species, resulting in models with 2-5 covariates per species and individual covariates being selected in 6 to 12 bird species models (Table 2.3). Occupancy models for my 12 forest-associated species included 6-13 statistically significant covariates (α=0.15), with individual covariates being selected in 1 to 7 bird species models (Table 2.4). For example, mature and older forest within a 100m buffer (OF_100) had a positive influence on 6 species (brown creeper, golden-crowned kinglet, purple finch, red-breasted nuthatch, Wilson's warbler, yellow-rumped warbler; for covariates see Table 2.4; averaged estimates in Table A.2.3). Conversely, distance to or amount of urban area within 100m or 1km were negative predictors in 9 of 10 cases where these covariates were selected (Table 2.4). These covariate selection frequencies suggest that 6 of 12 species that were associated with old forest habitat also avoid human-dominated landscapes (i.e. urban, rural and road; e.g., Tables 2.4 and A.2.2). In three species (Pacific wren, purple finch, Townsend's warbler), spatial polynomials improved model fit, with all five spatial terms reaching statistical significance. In no other species did spatial terms improve predictive models. Model goodness-of-fit, as $P(\chi_b^2 > \chi_{obs}^2)$, ranged from 0.36 to 0.70, indicating appropriate fits in all the averaged models.

Table 2.3: Detectability covariate selection frequency (covariate description see Table 2.2). + and – indicate positive or negative influences on the model, respectively.

	CR_CL_3	Is_size	FOR1_1KM	FOR2_1KM	Observer
+	5	1	7	5	
-	3	7	0	1	
total	8	8	7	6	12

Predictive maps

I used averaged, single species models to create predictive maps for 12 forest-associated birds (Figures 2.2 and A.2.1); of which three with relatively high (>0.76) or low (<0.56) association scores are shown to illustrate similarities and differences in predicted occurrence (Figure 2.2). Using all 12 maps as inputs, I next generated a composite map of the forest-associated bird community by weighting each of 12 single-species maps by their forest association scores (Table 2.1) to create a single, weighted composite map (Figure 2.3). Similarity of the weighted forest-associated community map to 12 individual species maps ranged from K_{fuzzy} = -0.24 to 0.17 (mean ± se = -0.08 ± 0.12), where values of 0 indicate randomness, and negative and positive values indicate more disagreement or agreement than expected by chance, respectively. Fraction agreement (P_A) ranged from 32% for olive-sided flycatcher (2nd lowest association score) to 67% for golden-crowned kinglet (3rd highest association score; mean P_A ± se = 49 ± 11). $Diff_{abs}$ ranged from 0.14 to 0.43 (mean ± se = 0.25 ± 0.09; Table 2.5), which confirms my assumption that each indicator species contributed independent information to my composite map.

Table 2.4: Occupancy covariate selection frequency (covariate description see Table 2.2). + and – indicate positive or negative influences on the model, respectively.

	rd_up_1k	rd_up_100	rd_p_1k	rd_p_100	near_road	near_frshw	near_saltw	Is_size	URB_1KM	RUR_1KM
+	4	2	0	2	1	3	1	1	0	1
-	3	2	2	1	3	2	1	1	1	3
total	7	4	2	3	4	5	2	2	1	4

	FOR0_1KM	FOR1_1KM	FOR2_1KM	SAV_1KM	SHR_1KM	URB_100	RUR_100	FOR0_100	FOR1_100	FOR2_100
+	1	2	1	2	1	1	1	2	2	3
-	0	5	3	0	0	3	1	1	2	2
total	1	7	4	2	1	4	2	3	4	5

	SAV_100	SHR_100	Near_Urb	CR_CL0	CR_CL_2	CR_CL_3	BRDLF	OF_100	OF_1KM
+	3	0	0	1	4	0	1	6	4
-	1	2	5	3	0	3	1	1	1
total	4	2	5	4	4	3	2	7	5

Figure 2.2: Individual species maps representing the three species with the highest weight scores: brown creeper (a), red-breasted nuthatch (b) and golden-crowned kinglet (c); and the three with the lowest ones: yellow-rumped warbler (d), olive-sided flycatcher (e) and Wilson's warbler (f).
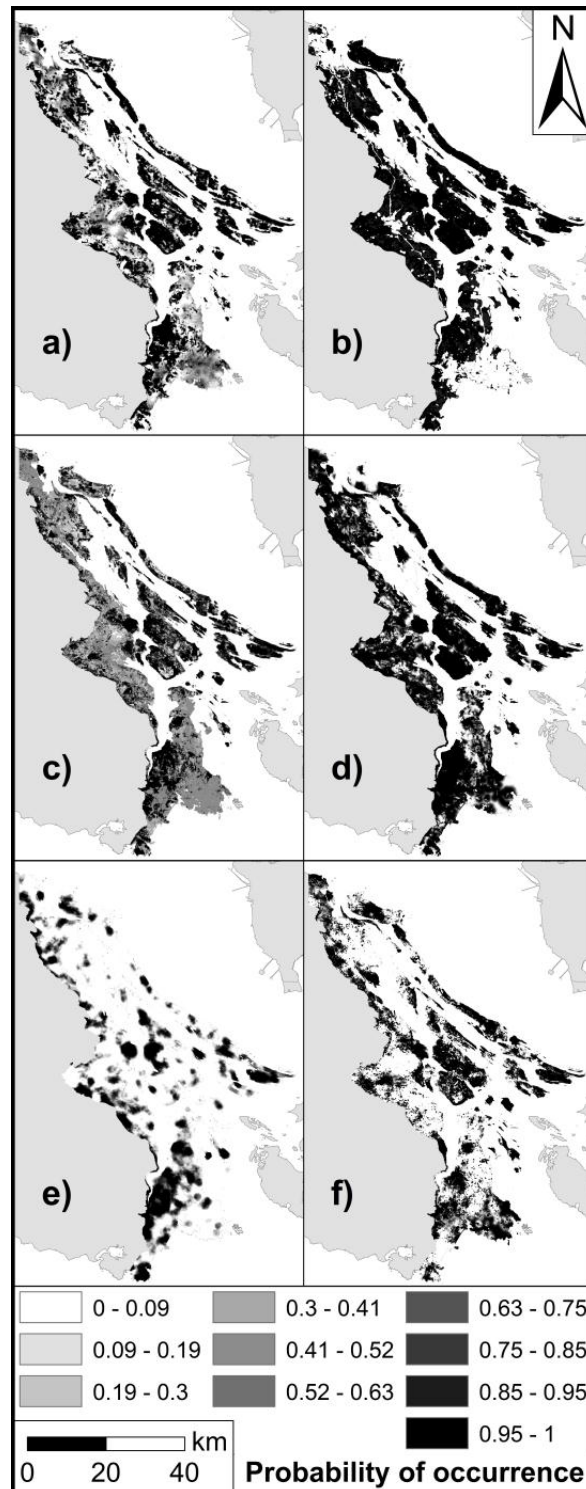
Figure 2.3: Weighted community score map using individual results from the 12 selected species and their corresponding species community weights from Table 2.1.
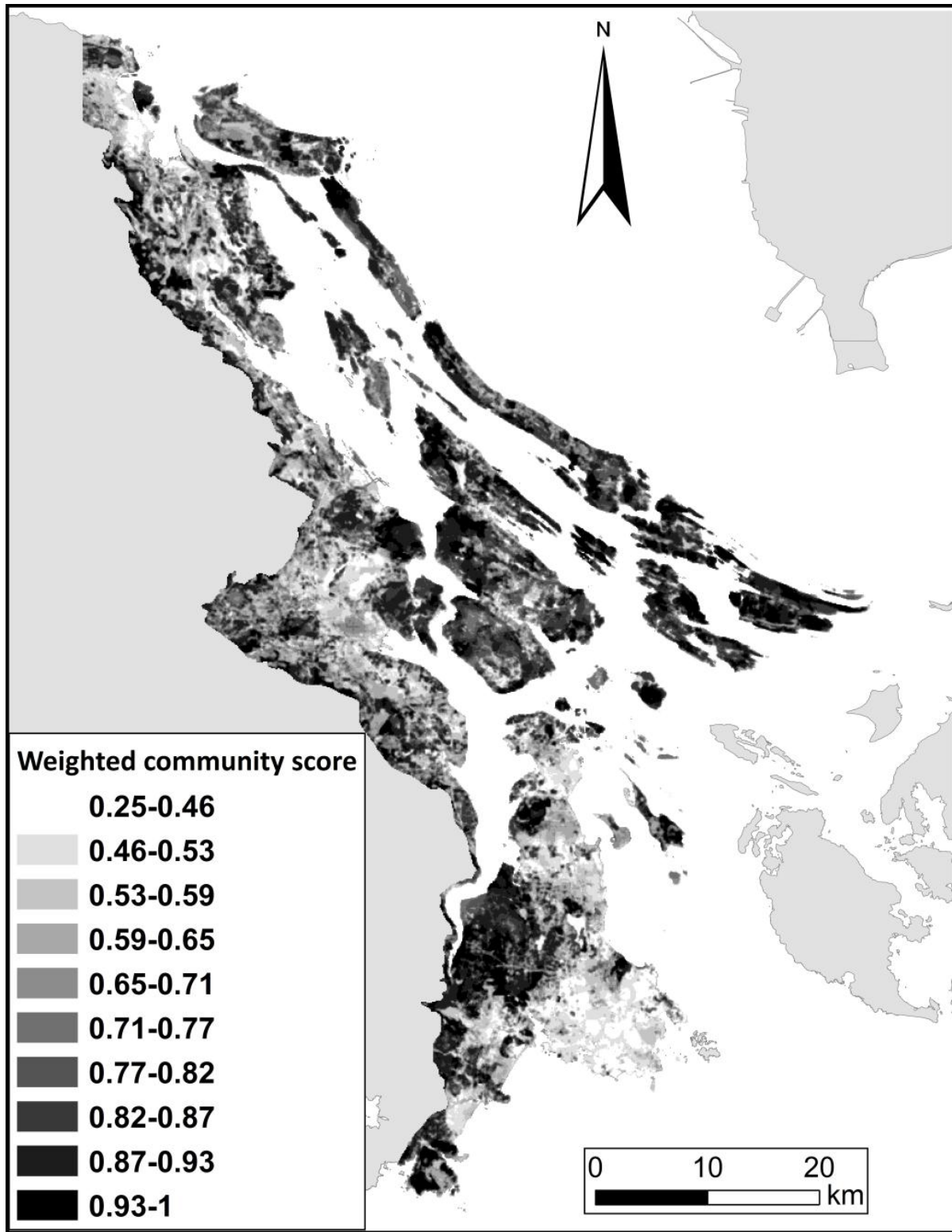
Table 2.5: Map comparison values between the weighted bird community score and individual species.

Species	K_{fuzzy}	P_A	$Diff_{abs}$
brown creeper	0.0357	0.5510	0.2130
chestnut-backed chickadee	-0.1415	0.4769	0.2166
golden-crowned kinglet	0.1661	0.6657	0.1388
olive-sided flycatcher	-0.1989	0.3151	0.4332
pacific wren	-0.0334	0.5141	0.2182
pacific-slope flycatcher	-0.2426	0.3404	0.3424
pine siskin	-0.0862	0.5142	0.1999
purple finch	-0.1043	0.6197	0.1446
red-breasted nuthatch	-0.0154	0.4522	0.2557
townsend's warbler	0.0396	0.6253	0.1612
wilson's warbler	-0.1712	0.3805	0.3732
yellow-rumped warbler	-0.1869	0.4713	0.2470
Average	-0.0783	0.4939	0.2453
Lowest	-0.2426	0.3151	0.1388
Highest	0.1661	0.6657	0.4332

I next compared my predictions on the distribution of forest-associated bird communities to an inventory of mature and old forest habitat (>80 years, 'TEM-MF' mapped throughout my study area using aerial photographs; MES 2008) to test if my composite map correctly identified old forest stands. I found that the predicted mean, area-weighted occurrence of the old forest bird community in polygons identified in aerial photographs as mature forest was 12% larger than the landscape average (0.82 versus 0.73). However, plotting area-weighted bird community scores against the size of TEM-MF polygons, under the assumption that larger polygons should support richer forest bird communities, revealed no relationship (R^2=0.01, Figure 2.4a). In contrast, 11% of the variance in community scores in mature forest polygons was explained by the area of

32

agriculture within a five kilometer buffer around the TEM-MF polygon centroids, as expected if

agriculture reduced the value of adjacent forest as habitat for forest-associated birds (Figure

2.4b).

Figure 2.4: a) Area weighted mean community score of each TEM-MF (>80 years) polygon
plotted against TEM-MF area. The logarithmic trend line equation is: 0.74 + 0.0074 * log(TEM-
MF area); $R^2 = 0.012$; p<0.0001. b) Area weighted mean community score of each TEM-MF
(>80 years) polygon plotted against agricultural area within 5 kilometers. The trend line equation
is: 0.85 – 5.37e-05 * agriculture; $R^2 = 0.11$; p<0.0001.
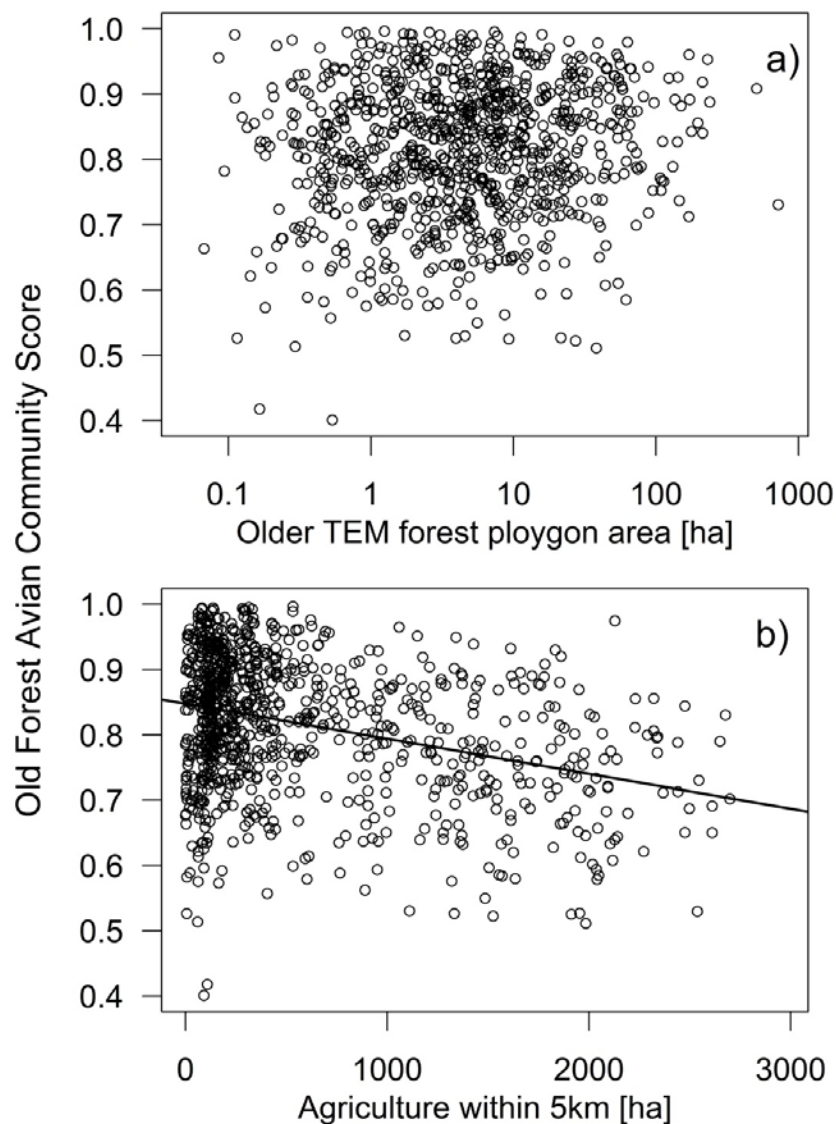
Discussion

I used expert opinion to identify 12 species likely to occur in old-growth Coastal Douglas-fir forests of the Georgia Basin, which now comprise <0.3% of this highly threatened region (Austin et al. 2008). My composite occurrence map, weighted by forest association, predicts the occurrence of old forest bird communities directly (Figure 2.3) and should improve estimates of the conservation value of forest stands based solely on coarse scale indicators such as fragment size, location and opportunistic rare species occurrence records. My methods contrast with DeWan et al. (2009), who pooled detections of 9 'forest interior' species as contributing equally to detectability and occurrence models, despite marked variation in abundance and association with old forest structures (Hayes et al. 2003, Sallabanks et al. 2006). In contrast, the modest overlap of my single-species occupancy maps (Figure 2.2), but overall similarity to my composite, old forest community map (Table 2.5), suggests that the 12 species I used cumulatively indicate habitat conditions typical of historically complex, old growth Douglas-fir forests in the Georgia Basin. Those forests are characterized by high crown and understory complexity (e.g., golden-crowned kinglet, pine siskin, purple finch, Townsend's, yellow-rumped and Wilson's warbler), large trees and canopy gaps (olive-sided and Pacific-slope flycatcher) and large, standing and downed coarse woody debris (brown creeper, red-breasted nuthatch, chestnut-backed chickadee, Pacific wren; Table 2.5; Austin et al. 2008, Vellend et al. 2008). My results therefore support recent suggestions that using multiple species to map high conservation value habitats represents a significant advance over single-species approaches (e.g., Rubinoff 2001, Roberge and Angelstam 2004), particularly given that birds are highly effective indicators of habitat condition (e.g., Elith et al. 2006, Branton and Richardson 2011).

Several studies have shown that failing to account for imperfect detection probability biases estimates of occupancy (Gu and Swihart 2004, Martin et al. 2005b), particularly in species such as forest birds, where detection probabilities are often << 1. To date, however, most studies attempting to incorporate detection probability have used relatively few covariates due to practical limits on software and model selection (Burnham and Anderson 2002, Hein et al. 2009, DeWan et al. 2009). In contrast, I included many potential predictors due to the number of species involved and the number of habitat descriptors available and documented in the literature as influencing the occurrence of old forest-associated birds. Including large sets of candidate predictors requires new ways to select among large sets of candidate models, as demonstrated by Weir et al. (2005) for Anurans. I adopted a similar approach of sequential model development and variable selection to include two covariate scales and avoid the shortcomings of restricting my analyses to a single spatial scale (e.g., Lawler and Edwards 2006). My results confirm that covariates at coarse (1km) and fine scales (100m) were often selected; indicating that including both scales increased the flexibility and fit of my models and, potentially, more accurately predicted species' responses to habitat distribution, type and context (Table 2.4).

My results show that 6 of 12 species identified as being positively associated with old forest habitat also avoided human dominated landscapes, consistent with the idea that species respond differently to habitat type, condition and context (Sallabanks et al. 2006), that suites of species with complementary habitat preferences can be used to estimate community membership and complexity, and that composite scores based on complementarity are likely to provide the most reliable indexes of site value in multi-species conservation plans (Lindenmayer et al. 2008). My method of creating composite old forest bird community maps weighted by expert opinion (Table 2.1) represents a straightforward way to develop indexes representing whole communities

and could be augmented further by adding reliable occurrence maps for other ecologically or culturally significant taxa. In particular, my composite map should identify old forest stands with a better than average potential for population growth in forest-associated birds, given that the most highly ranked stands were on average more isolated from human land uses known to facilitate parasites, predators and competitors of many forest bird species (Andrén 1994, Martin et al. 2005b, 2011, Jewell et al. 2007, Vellend et al. 2008). The relative stability of expert scores, model predictions and composite scores indicates that my maps are relatively stable and probably represent a reliable way to identify high quality bird community areas. I therefore predict that future surveys of focal species inhabiting highly ranked sites will reveal higher average nest success and juvenile to adult ratios, and lower rates of brood parasitism, than sites ranked lower.

My composite map of old forest-associated bird communities should also facilitate forest restoration and conservation plans (Chazdon et al. 2009, Fuller et al. 2010) by helping to discriminate among fragments of similar size and age but different value to birds based on local or landscape context (Figure 2.4). For example, the value of forest as habitat for forest-interior bird species is typically assumed to be positively related to fragment size. In contrast, I found no clear relationship between the size of older forest fragments and occurrence of forest-associated bird communities, but did find that proximity to agriculture markedly reduced the occurrence of forest-associated bird communities regardless of patch size (Figure 2.4). Many other studies also suggest that human land use practices influence bird species abundance and distribution by affecting habitat quality and demographic performance (Andrén 1994, Allombert et al. 2005, Martin et al. 2011). My results complement this work by demonstrating how planners can use

species occurrence data and expert opinion to map high value habitats in ways that incorporate bird community responses to human land use practices explicitly.

Nevertheless, two caveats also suggest that further study is required. First, despite my attempts to account for spatial dependence in data using spatial polynomials in models, model residual spatial dependence after accounting for landscape covariates (e.g., Borcard et al. 1992, Dormann et al. 2007) and following related suggestions (Magoun et al. 2007), there is currently no consensus on the most appropriate method to account for spatial dependence in data on my model predictions. Thus, it is possible that some of the models I created may include species responses to unmeasured habitat or environmental gradients. Second, no unbiased methods yet exist to assess prediction error in my models because estimates of variance explained (*cf* Nagelkerke's R^2; Nagelkerke 1991) suffer uncertainties related to effective sample size and degrees of freedom (reviewed in Fielding and Bell 1997). However, parametric bootstrapping (MacKenzie and Bailey 2004) provided us with helpful measures of model fit, and comparing my predictions to the area and distribution of high-value forest stands identified in aerial photographs suggests that my maps should enhance the outcome of conservation planning by prioritizing forest habitats most likely to support viable populations of old forest-associated birds, and thus as core areas for old growth restoration.

Conclusions

Using individual species maps ranked by habitat affinity, I produced a composite occurrence map for forest-associated birds of the Coastal Douglas-fir zone. A fuzzy-set map comparison showed that my composite map represented the multi-dimensional response of my 12 indicator species to a wide range of coarse and fine scale habitat metrics in predictive models. This composite map should advance the identification and restoration of high conservation value

forests by helping planners to quantify native species responses to human land use change and

forest habitat condition, rather than relying on classical habitat metrics such as patch size, which

may or may not be more influential on species occurrence than adjacent habitat use, proximity to

urban area, or presence of particular enemies. By focusing on common species for this purpose, I

was able to take advantage of already existing occurrence data, minimize survey costs, and avoid

biases associated with mapping rare or cryptic species. The general flexibility of my approach

and ease of including additional survey data and taxa suggest that it represents a useful method

for advancing conservation area design.

Chapter 3: Bird community conservation and carbon payments in Western North America[2]

Introduction

There is a pressing need to develop mechanisms to promote biodiversity conservation in the face of climate and land use change and the competing needs of humans (Ehrlich and Pringle 2008, Butchart et al. 2010, Bayon and Jenkins 2010, Estes et al. 2011). This challenge is particularly severe in human-dominated landscapes, where private ownership prevails and the cost of purchasing properties or compensating land holders for lost opportunity incurred as a result of conservation can be substantial (Naidoo et al. 2006, Wunder 2007). One way of making conservation more affordable is by offsetting those costs via payments for ecosystem services. The use of carbon markets to pay for carbon sequestration is an ecosystem service gaining global attention (Venter et al. 2009a, Agrawal et al. 2011, Phelps et al. 2012), in part because public concerns about the consequences of climate change have motivated 35 nations and 13 sub-national jurisdictions to put a price on carbon (Climate Commission 2013). To the degree that carbon and biodiversity values overlap, carbon offsets could therefore be used to protect forests that would otherwise be logged (Chan et al. 2011, Douglass et al. 2011) or to restore those still supporting valued old forest communities (Schuster and Arcese 2013).

Several outstanding issues arise when considering the role of carbon markets in forest restoration. One issue is that biodiversity values may be lower in stands with the highest returns from carbon sequestration sales because sequestration rates typically peak in stands of

intermediate age (Pregitzer and Euskirchen 2004), although older forests act as carbon sinks and continue to accumulate carbon over time, but at lower rates on average (Luyssaert et al. 2008), resulting in high initial returns on the sale of carbon storage credits, where one carbon credit represents the offset of greenhouse gas emissions by one tonne of carbon dioxide equivalent (CO_2-e). A second issue is whether to develop conservation plans that maximize species richness (α-diversity) within habitats or maximize dissimilarities in community composition (β-diversity) to accommodate landscape complexity and species that utilize multiple habitats (Marsh et al. 2010, Mokany et al. 2011). Under climate change, it has been suggested that an emphasis on community dissimilarity (β-diversity) may deliver more robust conservation plans than those based on species richness (Arponen et al. 2008, Mokany et al. 2011). Here I examine the potential value of carbon credit sales to offset land acquisition costs by developing conservation area designs that maximize β or α-diversity in native old forest and savannah bird communities in relation to forest structure and human land use. Specifically, I ask how protecting forests with high carbon storage versus high carbon uptake is likely to affect conservation outcomes.

<u>Carbon and biodiversity in the Georgia Basin</u>

The Coastal Douglas-fir (CDF) ecozone of the Georgia Basin (British Columbia, Canada, Meidinger and Pojar 1991) is a classic example of an endangered but extraordinarily diverse region that has been rapidly converted to exclusive human use ($\geq 60\%$) (Austin et al. 2008). The CDF retains $\leq 0.3\%$ of historic old forests (>250 years) (MES 2008) and $\leq 10\%$ of oak woodland and savannah (Lea 2006), which provide habitat for 117 species at risk of extirpation, which represents the highest density of species of global and provincial concern to conservation of any ecozone in BC (Austin et al. 2008). Because regional, provincial and federal authorities own

<20% of the region and only ~9% is already conserved, cost-efficient routes to conservation are urgently needed to help reduce the risk of extirpation for those species and related ecosystems.

Prior to European colonization the CDF occurred as uneven-aged forest (often >300 years) dissected by shallow and deep-soil meadow and woodland communities (Meidinger and Pojar 1991, Mosseler et al. 2003) maintained in part by aboriginal land management practices to enhance hunting opportunities and root and fruit harvests (MacDougall et al. 2004, Dunwiddie and Bakker 2011, McCune et al. 2013, Turner 2014). In addition to recent human-caused disturbances, oak woodland and savannah community distributions are predicted to shift under future climate conditions, and only a small fraction of the current protected areas have the potential to accommodate this shift (Pellatt et al. 2012). The resulting land use heterogeneity within the region and potential for humans to directly or indirectly affect native species richness (Gonzales and Arcese 2008, Martin et al. 2011, Bennett et al. 2012, Schuster and Arcese 2013) make this system ideal for studying trade-offs involved when attempting to maximize α- versus β-diversity in conservation plans, while simultaneously maximizing ecosystem service values represented as total carbon stored or sequestration potential. To do so, I compared systematic conservation scenarios that maximized old forest and savannah bird biodiversity (α-diversity) or their dissimilarity (β-diversity), and then quantified their relative costs given alternate carbon markets, and in relation to increasing targets for the total area conserved (Noss et al. 2012) (Table 3.1).

Table 3.1: Summary of diversity features, land cost metrics, conservation targets and carbon prices used in 144 Marxan scenarios. PCT= Pacific Carbon Trust; StC = Carbon Storage * CC; SeqC = Carbon Sequestration potential * CC; TotC = StC + SeqC.

Diversity features (n=2)	Property cost metrics (n=4)	Conservation Targets [%] (n=9)	Carbon credit value (CC) (n=3)
α-diversity (maximize Old Forest + Savannah individually)	Total Land value (TLV)	10 to 50 (in 5% steps)	9 $/T (lowest price PCT has paid for credits so far)
β-diversity (maximize β-score)	TLV – StC		12.5 $/T (half the cost PCT charges, as well as roughly the average price PCT is paying for credits)
	TLV – SeqC		25 $/T (the price that PCT is charging for credits)
	TLV – TotC		

Methods

Ethics statement

Permits or permission for the use of bird point count locations were obtained from Parks Canada (locations in National Park Reserves), private land owners (locations on private land), or did not require specific permission as they occurred on public right of ways (e.g., roadsides, regional parks). As private land owners did not want their information posted publically please contact the authors for contact details. The field studies did not involve endangered or protected species. This study did not require approval from an Animal Care and Use Committee because it was a non-invasive observational field study, and did not involve the capture and handling of wild animals.

Biodiversity data

I used trained observers to conduct 1,770 point counts on mainland BC and 53 islands from 30 Apr – 11 Jul, 2005 - 2011 (Figure 3.1, 48.7° N, 123.5° W) to record all birds detected in 10 min, 50m radius counts between 5 AM – 12 PM at 713 sample locations (>100m apart). Locations were re-visited 1-12 times and geo-referenced via a GPS (GPS60, Garmin Ltd, Kansas, USA). I extended the approach of Schuster & Arcese (2013) geographically (from 1560km^2 to 2520 km^2) by adding 601 counts to create predictive distribution models for 47 bird species and 25 covariates based on remote-sensed data and models incorporating imperfect detectability (Mackenzie et al. 2002). To estimate detectability I used one site specific (crown closure) and three observation specific (time of date, Julian date and observer identity) covariates. I associated bird species indicators with the habitats they were expected to occupy by using 11 experts to rank the likelihood of observing 47 species in 10 focal habitat types using photographic and text descriptions of herbaceous, shrub, woodland, wetland, four forest types (pole, young, mature and old), and 2 human-dominated habitats (rural, urban), to create two community metrics indicating Old Forest (OF, Schuster and Arcese 2013) and Savannah (SAV) habitats standardized between 0 and 1 by dividing through the maximum value possible (details in Appendix 4), where:

$$OF = \frac{-2*Herb - 1*Shrub - 0.5*Pole + 0.5*YFor + 1*MFor + 2*OFor}{7}$$

$$SAV = \frac{2*Wood + 2*Herb + 1*Shrub}{5}$$

These metrics match my goals given the region's history and focus on Old Forest and Savannah community conservation (see Introduction). Specifically, each species contributed to

the cumulative Old Forest or Savannah community score, weighted by its expert opinion score

for the given sub-type, summed across species to create community specific association scores

from 0 to 1, and corresponding to none versus all members of the community expected to be

present. The metrics were then projected spatially as predictive maps of community occurrence

over the entire study area (2520km^2, Figure 3.1) as 1ha hexagonal polygons (Figure A.4.1-2,see

also (Schuster and Arcese 2013)).

Figure 3.1: Georgia Basin study area including bird point count locations. Dark grey area indicates the extent of the CDF study region and black dots represent bird point count locations.
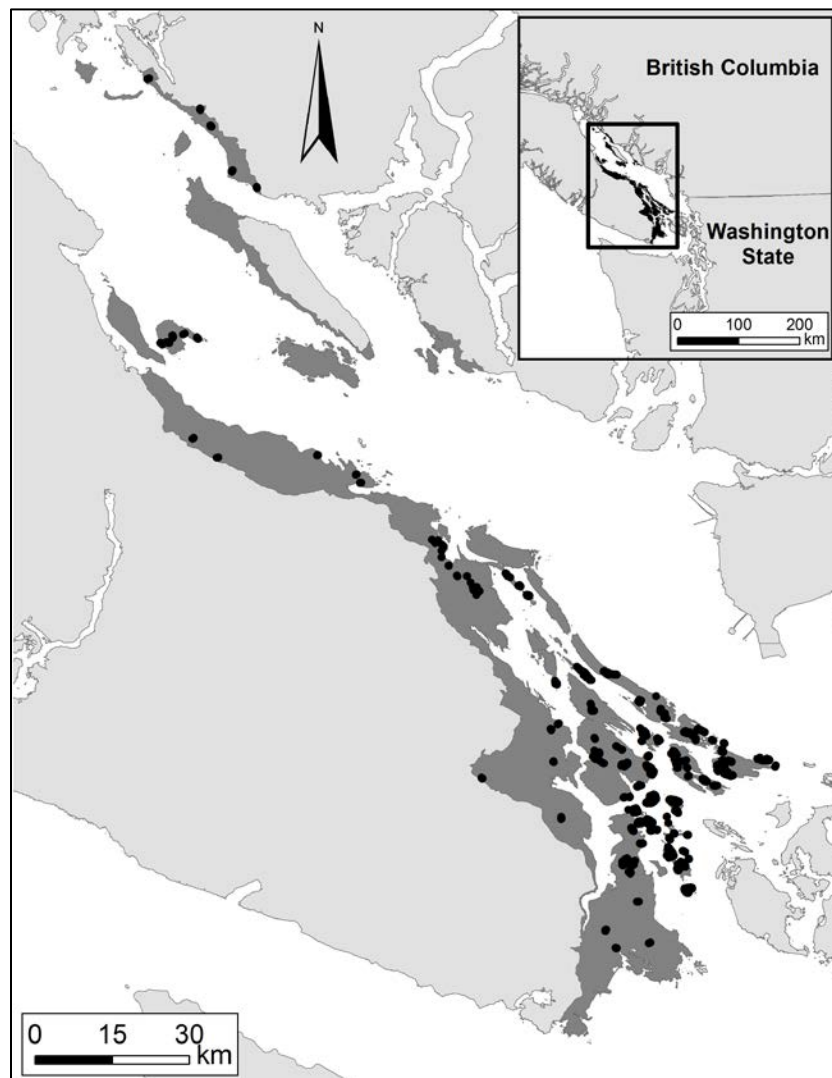
Carbon estimates

Forest carbon storage and sequestration rates were estimated for all forested land in the study area using terrestrial ecosystem mapping (TEM; MES 2008) and FORECAST (Kimmins et al. 1999). FORECAST is a stand-level forest ecosystem simulator that is one of two models approved by the BC Ministry of Forests for carbon budget assessments (Ministry of Environment 2011), and the only model calibrated for use in the CDF (Blanco et al. 2007) and linked to TEM (Seely et al. 2004). To facilitate carbon analysis TEM polygons were stratified into homogenous analysis units based on site series. Site series are representative ecosystems in each Biogeoclimatic subzone of British Columbia e.g., Douglasfir-Salal or Western redcedar-Grand fir-Foamflower (MES 2008). Carbon contents were simulated for all ecosystem pools including: soil organic matter, above-ground litter, above and below-ground tree biomass, plant biomass, deadwood, and dead below-ground biomass. Net ecosystem carbon storage was calculated for each analysis unit considering the fact that only certain ecosystem pools are generally eligible for forest carbon offsets. Net ecosystem carbon storage was limited to: above and below-ground tree biomass, deadwood biomass, and dead below-ground biomass. Each AU was simulated for a period of 300 years with results reported for annual time steps to create carbon storage curves. FORECAST results were subsequently assigned to individual TEM polygons by estimating the age of each polygon subsection based upon the current assigned structural stage and estimated productivity class and assigning them values from the corresponding analysis unit (Seely et al. 2004). The age estimates were derived from ranges provided by Meidinger *et al.* (1998) for regional forest ecosystems. Ages of old stands (structural stage 7) were set at 200 to be conservative. Age estimates were verified against a subset of TEM polygons (Southern Gulf Islands of Southwestern BC) for which direct age estimates were available (n=254 out of 25885

polygons). For conservation prioritization analysis I used predicted net ecosystem carbon storage and net ecosystem carbon sequestration estimates for 20 years from now, even though carbon offset contracts are usually done on a 100 year timeframe. I restricted carbon sequestration estimates to a 20 year timeframe as FORECAST estimates were only provided for this period, due to uncertainty about regional fire frequency in the future. Further details on this analysis are provided in Appendix 5.

Cadastral layer and property costs

I incorporated spatial heterogeneity in land values (Ando et al. 1998, Polasky et al. 2001, Ferraro 2003, Naidoo et al. 2006) in my plan by using cadastral data and 2012 land value assessments (Integrated Cadastral Information Society of BC, ICIS). However, because there is no centralized entity curating cadastral data for British Columbia, I combined data from ICIS, the BC Assessment agency and the Integrated Cadastral Fabric. Doing so required processing to remove stacked and overlapping polygons and slivers. The combined cadastral layer included 193,623 polygons. Current assessments were available for 187,139 polygons, but missing for 3,281 polygons or reduced relative to market value (3,203 polygons) due to taxation or administrative reasons unrelated to my work (e.g farm or managed forest land). For these 6,484 polygons I applied an inverse distance weighted interpolation to estimate land values by splitting cadastral polygons into 10 groups based on polygon size to accommodate high size related heterogeneity in assessed cost using R v.2.15.2 (R Development Core Team 2012) and packages gstat v.1.0-14 (Pebesma 2004) and sp v.1.0-1 (Bivand et al. 2013).

Marxan inputs

I used Marxan (Ball et al. 2009) to prioritize cadastral polygons for inclusion in conservation area designs by using them as planning units (n = 193,623). I calculated biodiversity and carbon estimates for each planning unit using ArcGIS v.10.1 (ESRI 2012) and area weighted sums in Geospatial Modelling Environment v.0.7.2.1 (Beyer 2012).

To determine whether maximizing β- versus α- diversity affected conservation outcomes I created two sets of diversity features as inputs to Marxan. First I included the diversity features individually in the analysis and set conservation targets for Old Forest and Savannah scores as the percentage of total old forest or savannah habitat existing within the study region. The second approach I used was to pre-specify a β-diversity metric to combine biodiversity features with the goal to specifically maximize highly diverse habitat patches. For this purpose I created the following metric:

$$\beta - score = \frac{2 * OF * SAV}{OF + SAV}$$

This represents the Old Forest and Savannah community dissimilarity, using a scaling factor of 2 to create β-scores between 0 and 1 (Figure A.4.3). In Marxan analyses I set targets for the β-score, while still including Old Forest and Savannah metrics (without setting a target) to keep track of individual community representation. I used a total of four property cost metrics per diversity scenario: i) Total land value (TLV) for each property, which is the sum of the assessed property value and any improvement on that parcel; ii) TLV minus the current carbon storage (T) times the carbon credit value ($/T); iii) TLV minus the amount of potential carbon sequestration over 20 years times the carbon credit value; and iv) TLV minus ii and iii combined (Table 3.1). Here I used $12.5 Canadian per carbon credit, which is half the amount that Pacific

Carbon Trust (PCT), a crown corporation established in 2008 to deliver greenhouse gas offsets in the province of British Columbia (http://pacificcarbontrust.com, date accessed: 2013-12-10), sells credits for and about the average amount they pay for credits. Prices currently paid on the Voluntary Carbon Market would represent an alternative to PCT value, but because PCT prices are specific to the study region and represent a range of carbon contracts established in the province of BC I used PCT values for my analysis. I further focus on the gross cost for both land purchase and carbon contracts and do not include transaction, survey or implementation costs for either approach here.

Marxan scenarios

I used the two diversity scenarios α (Old Forest + Savannah) and β (β-score) in combination with the four cost scenarios (Table 3.1). An important consideration for this study was what level to set the required conservation target to, in order to ensure the study system will maintain viable populations of native species and is resilient to predicted environmental change in the future. As there is debate about what constitutes appropriate conservation goals (Noss et al. 2012) I used a range of conservation targets (10-50%) to investigate the potential trade-offs of different targets. I calibrated each diversity scenario to ensure robust analysis by initially setting the diversity target to 50% (the most costly to reach) and the number of restarts to 100, as I was not so much interested in the spatial representation of the reserve design but rather its cost effectiveness (Ardron et al. 2010). For the same reason I also refrained from setting boundary length modifiers. For each diversity scenario I created Marxan solutions for combinations of the following species penalty factors (SPF's): 1-10,15,20 and number of iterations: 10k, 50k, 100k, 500k, 1M, 5M, 10M, 25M, 50M, 100M, for a total of 65 Marxan calibration analyses per diversity scenario. I created cumulative distribution functions using number of solutions on the

y-axis, solution cost on the x-axis for SPF and Marxan score for number of iterations (Ardron et al. 2010). Based on the results I used the following values for SPF and number of iterations respectively: Old Forest + Svannah (3/10M); β-score (3/10M). I investigated each Marxan calibrations analysis' summed solution file (Ardron et al. 2010) to make sure every restart met its targets, excluding ones that missed the target by > 5%. For ease of computation I created an R function to batch run Marxan (Appendix 6).

I held the calibrated values constant in subsequent analyses and ran Marxan scenarios for the two diversity metrics in combination with the four cost metrics, using the baseline carbon credit value of $12.5. For each combination I further varied the conservation target from 10 – 50%. From each run I recorded the cost of the total reserve system averaged over the number of restart (100), while ensuring conservation targets were met. To examine the amount of remaining Old Forest and Savannah communities protected by maximizing β-diversity I compiled community scores as Marxan features in these scenarios without setting targets, allowing us to keep track of Old Forest and Savannah representation without affecting the analysis. I used the results from these analyses to compare the reserve prices within each diversity scenario as well as across scenarios. In addition I calculated the potential cost savings between fee simple acquisition scenarios (TLV only) and ones that utilize the sale of carbon credits. As market prices of carbon credits are highly variable I extended my approach to include variation in carbon credit value, by repeating the entire analysis for two additional carbon credit values: i) $9 per credit (the lowest rate PCT has ever paid for credits), and ii) $25 per credit (the price PCT sells credits for). In total 144 Marxan scenarios were investigated (Table 3.1). All results presented here relate to the baseline carbon credit value of $12.5 unless otherwise stated.

Results

Land acquisition cost, diversity and planning goals

Acquisition costs of conservation networks increased from \$180M to \$2.45B as targets increased from 10 to 50% of remaining Old Forest and Savannah bird communities when maximising α-diversity (Figure 3.2), but reduced slightly when maximizing β-diversity (\$172M to \$2.16B, 10-50% target; Figure 3.2), representing savings of 4-15% as compared to equivalent α-diversity scenarios depending on the conservation target (Figure 3.3a). Savings were due in part to a reduction in total area needed to reach a given target when maximising β versus α-diversity (mean = 7%, range = 5 - 11%; Figure 3.3b). The amount of standing and sequestration potential carbon in conserved landscapes also declined slightly when maximizing β-diversity (mean= 2%, range= 0.7- 5%; Figure 3.3c). In contrast, representation of Old Forest communities slightly increased (1 – 2.5%) and representation of Savannah declined (-2.0 - -5.7%) when maximizing β versus α-diversity (Table 3.2).

Figure 3.2: Reserve costs sing alpha and beta diversity and a carbon credit value of $12.5 across a range of conservation targets (term definitions in Table 3.1).
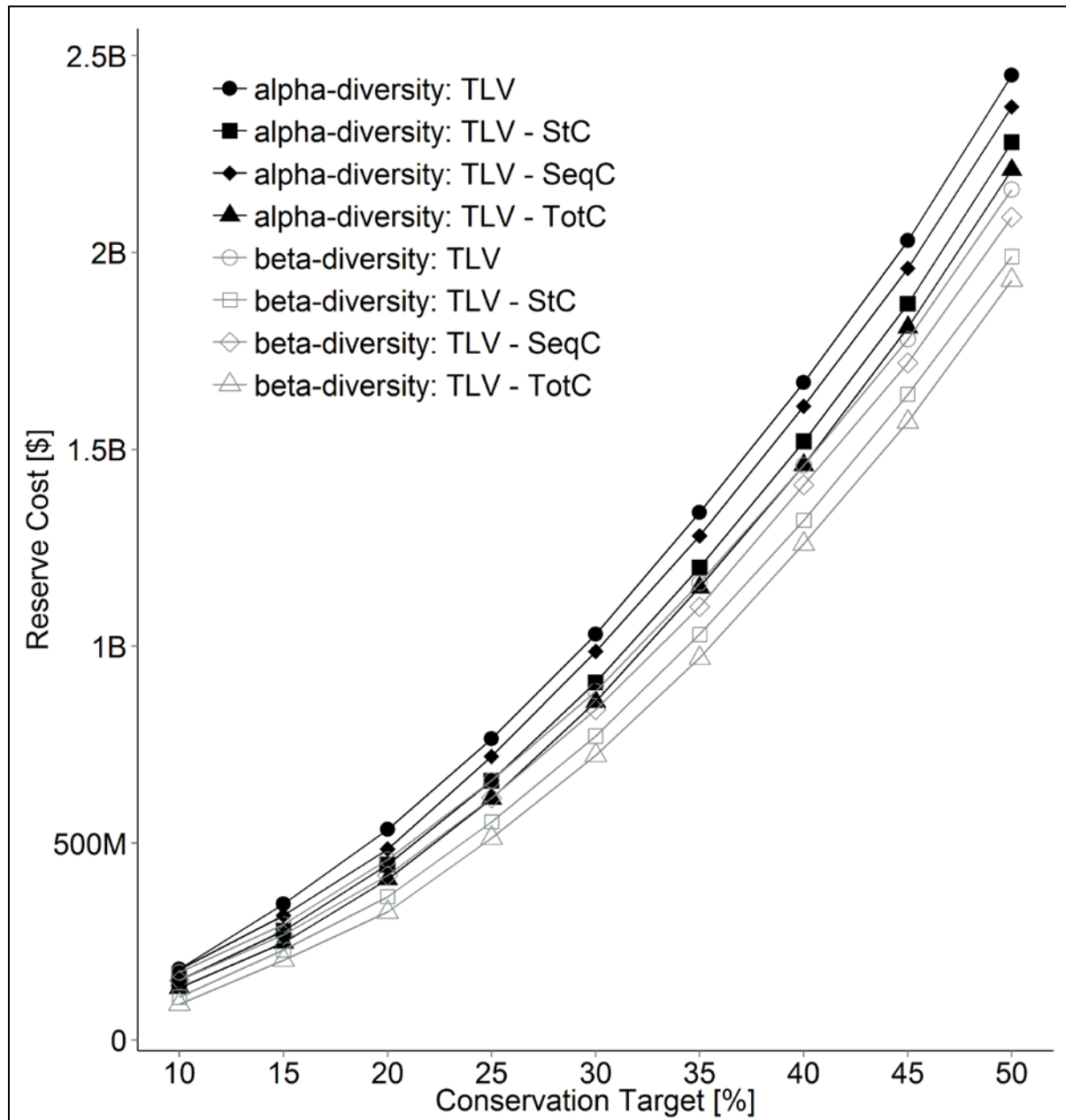
Figure 3.3: Comparison of α and β-diversity scenario results. Presented are the % reductions when using a β-diversity approach for: a) reserve network cost, b) reserve network area, c) total carbon included in the reserve networks. Circles represent TLV, squares StC, diamonds SeqC and triangles TotC (term definitions in Table 3.1) (c).
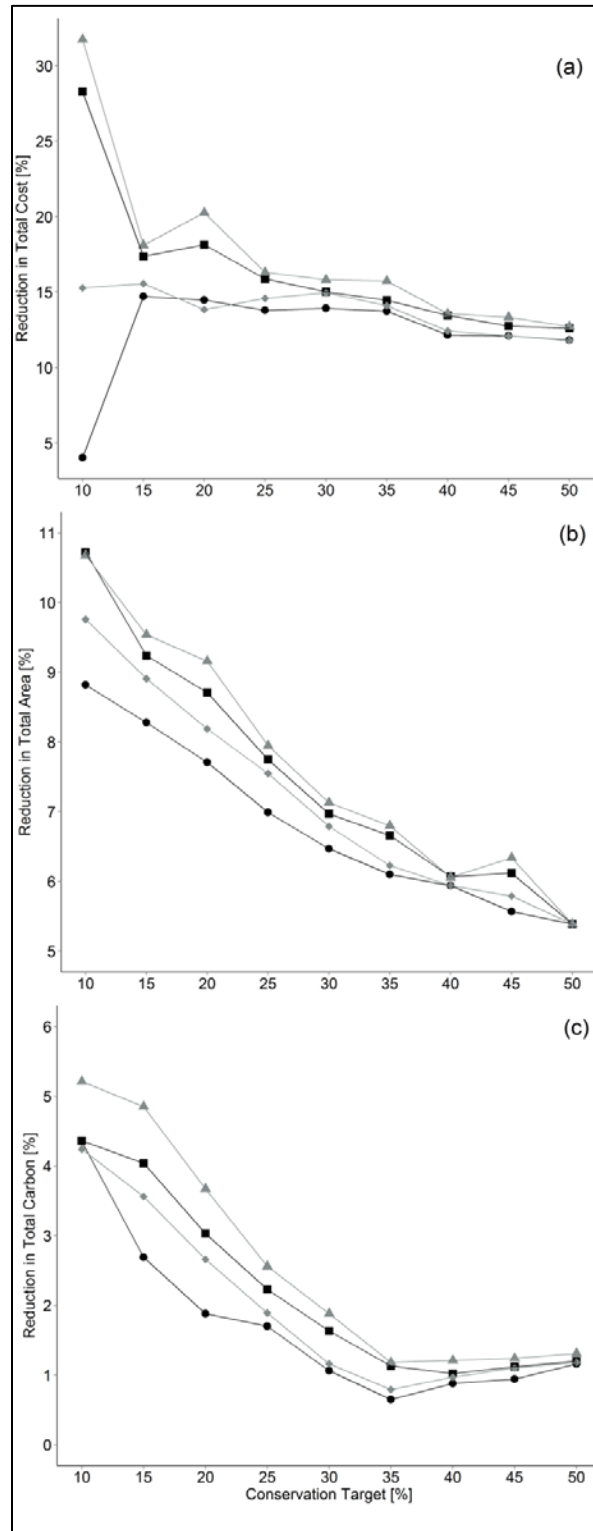
Table 3.2: Relative contribution of Old Forest (OF) and Savannah (SAV) to meeting β-diversity targets under different carbon offset assumptions. Conservation targets (bold) and % actually included in β-diversity scenarios (BETA) using carbon offset value of $12.5/t. β-targets were met in each case and OF was generally overrepresented and SAV underrepresented. StC = standing carbon credits; SeqC = sequestration potential credits; TotC = StC + SeqC.

Scenario		Percent of Target Bird Community Protected [%]								
		10	**15**	**20**	**25**	**30**	**35**	**40**	**45**	**50**
Total Land Value (TLV)	BETA	10.00	15.00	20.00	25.00	30.00	35.00	40.00	45.00	50.00
	OF	11.12	16.31	21.60	26.81	32.07	37.01	42.18	47.29	52.41
	SAV	8.80	13.57	18.09	22.87	27.59	32.49	37.22	42.07	46.85
TLV - StC	BETA	10.00	15.00	20.00	25.00	30.00	35.00	40.00	45.00	50.00
	OF	11.18	16.40	21.73	26.89	32.07	37.13	42.23	47.51	52.47
	SAV	8.72	13.38	17.93	22.68	27.45	32.34	37.14	41.78	46.78
TLV - SeqC	BETA	10.00	15.00	20.00	25.00	30.00	35.00	40.00	45.00	50.00
	OF	10.98	16.31	21.67	26.83	32.04	37.10	42.14	47.35	52.43
	SAV	8.96	13.56	18.07	22.85	27.57	32.42	37.30	42.02	46.84
TLV - TotC	BETA	10.00	15.00	20.00	25.00	30.00	35.00	40.00	45.00	50.00
	OF	11.17	16.43	21.72	26.95	31.95	37.18	42.27	47.50	52.48
	SAV	8.72	13.36	17.94	22.70	27.63	32.26	37.10	41.79	46.73

Cost savings given carbon credit sales

Maximizing total (standing + sequestered) carbon resulted in the largest cost savings in both α and β-diversity scenarios aimed at protecting Old Forest and Savannah habitats. Acquisition costs increased from $133M to $2.21B as target increased from 10 to 50% when maximizing α-diversity, which represent potentials offset of $47M – 235M, equivalent to a 10 – 28% cost reduction via carbon credit sales (Figure 3.2). In comparison, acquisition costs were lower for scenarios that maximised β-diversity ($90M to $1.93B), in part because implied carbon credit sales ($82-227M) contributed slightly more to cost reduction (e.g., 11 – 48%; Figure 3.2). Maximising carbon storage and carbon sequestration potential individually reduced acquisition costs to a smaller extent, but carbon storage offered superior savings (Figure 3.2). Overall,

maximizing total carbon returned networks that were 17.5% cheaper on average when maximizing β versus α-diversity compared to 12.3% without using carbon storage and sequestration values (Figure 3.3a).

Conservation targets and carbon price

The cost of conservation networks increased exponentially with increasing targets for all scenarios (Figure 3.2). In β-diversity scenarios the total area that needed to be acquired to reach a conservation target was 11 – 5% lower and acquisition costs 32 – 13% less than scenarios that maximized α -diversity (Figure 3.3). The percent reduction in total acquisition costs due to carbon value also declined as conservation targets increased in α and β-diversity scenarios (Figure 3.4). The magnitude by which acquisition costs were reduced by carbon value was similar across prices considered but maximized at $25/T in most scenarios (Figure 3.4). Relative reduction in cost due to carbon was maximized at the 10% target in all β-diversity scenarios (Figure 3.4b).

Figure 3.4: Cost savings when using carbon credit sales in relation to outright acquisition (TLV scenarios). Carbon credit values used are shown in parenthesis for a) α-diversity and b) β-diversity. Rectangles represent StC, diamonds SeqC and triangles TotC. Carbon credit values are as follows: dotted ($9), black ($12) and grey ($25) (term definitions in Table 3.1).
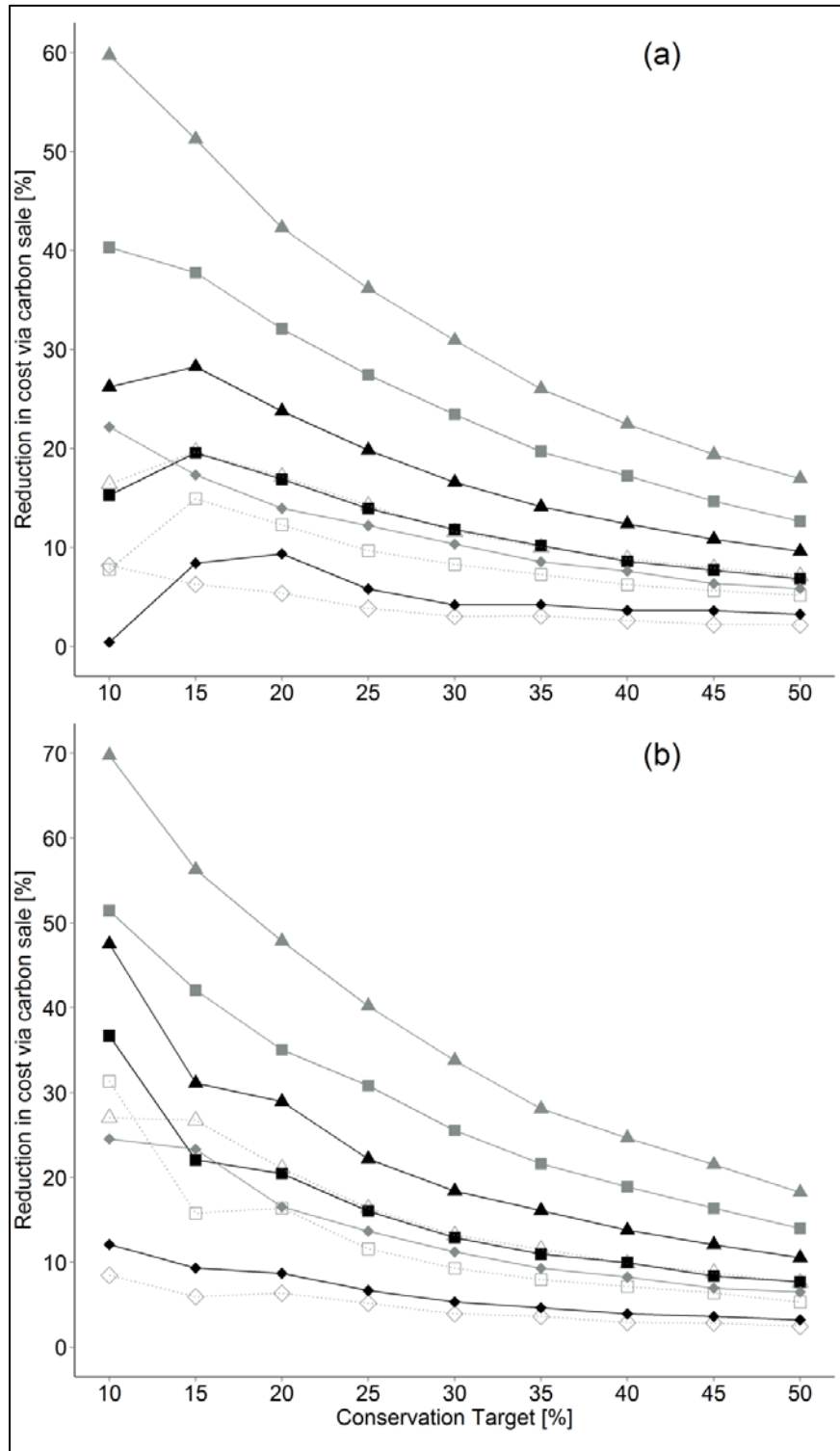
Discussion

Carbon credit sales have the potential to reduce land acquisition costs by up to 48% in Coastal Douglas-fir forest and woodland communities of western North America given carbon values paid in the region ($12.5/T: (PCT 2013); Figure 3.4b). The largest benefits were realized in scenarios that maximized total carbon and bird β-diversity in native woodland and old forest bird communities of the region, because those scenarios achieved their targets by selecting cheaper and slightly smaller networks than scenarios maximizing α-diversity. I now develop these points in light of literature on ecosystem services, land acquisition and conservation applied to threatened plant and animal communities in human-dominated landscapes of the Georgia Basin of western North America.

Maximizing β versus α diversity

Prioritizing β over α-diversity in Old Forest and Savannah bird communities reduced acquisition costs by up to 15% land purchase only, and up to 32% including carbon values (Figure 3.3a). One reason for these savings is that the fraction of Old Forest bird habitat included in conservation networks was larger in β as compared to α-diversity scenarios (Table 3.2), resulting in more carbon stored per unit area conserved (Figure 3.3b,c). However, over-representation of Old Forest relative to Savannah communities also reduced acquisition costs in scenarios not including carbon, indicating that Old Forest habitat was on average less valued than Savannah habitat in the region, perhaps due to high human amenity values (Table 3.2; e.g., Vellend et al. 2008). Scenarios maximizing β-diversity also met conservation targets by protecting less total area and providing nearly 11% cost savings compared to α-diversity scenarios (Figure 3.3b). The benefits of β-diversity scenarios were largely independent of carbon value or conservation target levels (Figure 3.3b). Prior studies of the consequence of

emphasizing α- versus β-diversity in conservation planning have concluded that a focus on β-diversity is likely to enhance long term persistence in diverse species assemblages and reserve networks (Fairbanks 2001, Wiersma and Urban 2005, Marsh et al. 2010). My results broaden these conclusions by showing that scenarios that maximize β-diversity may also reduce the cost of conservation by reducing the area required to meet realistic targets for land acquisition.

Conservation cost and carbon

My results support the utility of carbon credit sales in regions where land cost is high and where old or re-growing forests offer habitat for valued focal communities. My results therefore compliment suggestions that carbon credit sales have the potential to advance conservation and mitigate the impacts of climate change (Venter et al. 2009a, 2009b, Bradshaw et al. 2013) but extend those suggestions by providing a spatially explicit, empirical example applied to a landscape with high conservation and cultural values (Austin et al. 2008, Vellend et al. 2008, Arcese et al. 2014). The largest cost reductions with carbon credit sales were obtained by including both carbon storage and sequestration potential (Figure 3.4), indicating that flexibility in carbon credit sales with respect to forest age can also increase economic efficiency. Although my results are based on a 20 year time-frame due to uncertainty about fire frequency, however more typical projects use a 100 year time-frame (Ministry of Environment 2011, Bradshaw et al. 2013). New data on long term fire frequencies could support this approach.

My finding that carbon storage reduced costs more than sequestration (Figure 3.2 and Figure 3.4) is partly a consequence of logging history. Nearly 30% of non-anthropogenic landcover includes mature forests (≥ 80 years-old, Figure A.5.1). The predominance of young forest (< 80 years old) has the potential to reduce adjacency between older, high-value forest and savannah habitat with rich and diverse native bird communities. Young forest patches may

provide lower-cost opportunities to link high-value OF patches if acquisition costs can be offset by higher sequestration. Nevertheless, most scenarios preferentially included older stands (>80 years) with more carbon storage and lower sequestration rates (Figure A.5.1). Several other studies have suggested that carbon credits could be used to advance conservation, particularly on private land, to compensate land owners for forgone opportunity costs (Crossman et al. 2011, Douglass et al. 2011, Evans et al. 2014). I extended these suggestions by providing a particularly detailed example to demonstrate how land use planners might offset the costs of conservation area design by acquiring habitats that simultaneously maximize biodiversity and realize the economic potential of carbon credit sales.

Although my results were robust over a range of carbon values, carbon markets remain unpredictable. Nevertheless, carbon markets are of substantial size, the European Union Emissions Trading System for example included 2.1 billion metric tons in 2011 (Newell et al. 2013). In 2013 China, the largest national source of greenhouse gases (19.1% of total emissions), introduced pilot emission trading schemes (Lo 2013, Wang 2013), joining a growing number of countries with national emission trading schemes (Climate Commission 2013). Voluntary carbon markets that are currently the biggest market place for forest carbon offset projects in countries like Canada had a market volume of $572M in 2011 (Newell et al. 2013). Figure 3.4 illustrates the changes to reduction in purchase costs using a range of carbon credit values. If carbon markets were to diminish or be abolished for e.g., political reasons as recently happened in Australia this would have devastating consequences for this approach, negating the benefits of carbon credit sales. If on the other hand carbon markets take off and probably policy related changes increase the price of carbon substantially the cost savings possible via carbon credit

sales would be even more substantial than shown here and greatly benefit conservation efforts in forest ecosystems.

Conservation targets

A key goal of my work was to demonstrate novel routes to protecting high-value, Old Forest and Savannah bird communities at landscape scales in western North America. However, the amount of habitat needed to achieve those goals remains uncertain. Policy-driven targets for biodiversity conservation place goals for terrestrial habitat conservation at 17% by 2020 (Convention on Biological Diversity 2010), but recent reviews suggest much higher targets be set (25-75%; Noss et al. 2012). I used a range of conservation targets and explored their influence on reserve design, carbon value and the conservation of Old Forest and Savannah ecosystems. I found that as targets increased, carbon contributed proportionally less to reducing acquisition costs for all scenarios (Figure 3.4) as achieving higher targets required the inclusion of more expensive parcels. Although total carbon generally increased linearly with higher conservation targets, acquisition costs increased exponentially, causing a decline in relative carbon benefit (Figure 3.4). Even for the largest targets (50%) in α and β-diversity scenarios, carbon values reduced acquisition cost by 9.6 and 10.5%, respectively ($235, 227M; Figure 3.4) at $12.5 per Ton.

A number of uncertainties in my study also have the potential to limit its interpretation. First, actual purchase costs may differ from assessed or predicted values (Carwardine et al. 2010, Knight et al. 2011). Second, it may not be feasible to protect the areas offering the highest conservation value and least cost regionally, particularly if local representation or the augmentation of existing conservation areas is emphasized (Pressey et al. 2007). Third, carbon markets remain unpredictable (see above) and will have considerable influence on the cost-

effectiveness of carbon credit sales for conservation. Assuming that carbon markets develop further, my results demonstrate that carbon value has the potential to substantially reduce land acquisition costs in human-dominated landscapes, particularly in the Georgia Basin of western North America, where diverse Old Forest and Savannah bird (Schuster and Arcese 2013) and plant (Bennett and Arcese 2013) communities still persist in relatively isolated, mature forest and woodland habitats.

Chapter 4: Efficient routes to land conservation given risk of covenant failure[3]

Introduction

There is a pressing need to develop mechanisms for promoting biodiversity conservation in the face of climate change and accelerating demands for land use (Ehrlich and Pringle 2008, Butchart et al. 2010, Bayon and Jenkins 2010, Estes et al. 2011). Developing such mechanisms has proved particularly challenging in human-dominated landscapes with mostly private ownership where the cost of land purchase and/or opportunity costs of conservation can be substantial (Naidoo et al. 2006, Wunder 2007). In these cases, one cost-effective route to conservation may include private land conservation covenants, which are registered on the land title to prohibit future land use changes that are likely to reduce conservation values in exchange for monetary or other compensation (Merenlender et al. 2004, Knight et al. 2011).

Potential advantages of conservation covenants include a much lower initial cost than direct land purchase (Pence et al. 2003) and the ability to enable voluntary conservation agreements in cases where landowners wish to retain ownership (Langholz and Lassoie 2001, Winter et al. 2005, Knight et al. 2010). As a consequence, conservation covenants and other private land management agreements have gained global attention as an important policy tool for meeting conservation objectives (Fishburn et al. 2009, Gordon et al. 2011). To date, few studies have addressed the potential complexities of such investments to ask if covenants or land purchase are most likely to maximize the conservation of high-biodiversity landscapes in a region (Armsworth and Sanchirico 2008, Fishburn et al. 2009). In particular, serious questions

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[3] A version of this chapter is currently under review as Schuster, R., and P. Arcese. Efficient routes to land conservation given risk of covenant failure.

exist regarding the long-term costs of monitoring and enforcing covenants given the risk that property owners might violate the covenant or challenge it legally (Knight et al. 2010). Because covenant violations and related legal disputes appear to be increasing and may become more frequent after repeat sales of properties to new owners (Rissman & Butsic 2011), the long-term costs of covenants due to monitoring, legal or remediation costs to covenant holders could be substantial (Byers et al. 2005). Violations may also reduce conservation values via land conversion, resource extraction or removal of habitat. If these costs prove substantial the value of covenants as conservation tool may be over-estimated.

In landscapes dominated by private ownership, conservation non-governmental organizations can find it challenging to compete in retail markets (Pasquini et al. 2011) and conservation covenants may represent a viable alternative to purchasing high-biodiversity lands given willing owners (Parker 2004, Knight et al. 2010). However, little is known about the cost-effectiveness of covenants versus land purchase given the risk of dispute and potential losses to biodiversity over time. Although land purchase implies more certain control of land management activity, covenants vary in what they protect, ranging from simply protecting forest canopy to prohibiting all activities that could potentially reduce biodiversity (Parker 2004, Rissman 2013).

Given these uncertainties, I conducted a series of simulations based on land and biodiversity values in the Georgia Basin of northwestern North America to estimate the cost and effectiveness of land purchase versus conservation covenants to restore critically endangered Old Forest and Savannah habitats and associated bird communities to pre-logging levels. As regional, provincial and federal authorities own <20% of the region and only ~9% is preserved for conservation, cost-efficient routes to meeting conservation goals are urgently needed. Here I

asked the following questions about the long term (100 years) costs and potential conservation value of conservation covenants compared to land purchase: 1) To what extent does dispute rate reduce the cost-effectiveness of conservation covenants as compared to purchasing conservation land? 2) Assuming that covenant violations reduce the area of high-biodiversity landscapes protected, what is the area of the high-biodiversity landscape that is protected after 100 years, given the alternate tactics of financing and managing conservation covenants versus purchasing conservation land?

To answer these questions I created detailed distribution maps for 47 bird species and combined them to create a composite, beta-diversity score where high values represent high-biodiversity landscapes in the region. I contrasted a land purchase scenario developed in Chapter 3 with scenarios involving conservation covenants and a range of assumptions about covenant dispute rates and costs in the Georgia Basin of western Canada.

Methods

Study region

I studied a 2520 km^2 portion of the Coastal Douglas-fir (CDF) ecological zone of the Georgia Basin of British Columbia (BC), Canada (49°N 123.5°W, Figure 4.1). The CDF includes a critically endangered but diverse suite of old forest and savannah plant and animal communities endemic to the region, but much of the historic range of the CDF has been converted to nearly exclusive human use ($\geq 60\%$) (Austin et al. 2008). The current CDF area retains $\leq 0.3\%$ of historic forests (>250 years) (MES 2008) and $\leq 10\%$ of oak woodlands extant prior to European contact (Lea 2006).

Figure 4.1: Coastal Douglas-fir region showing the study area in red.
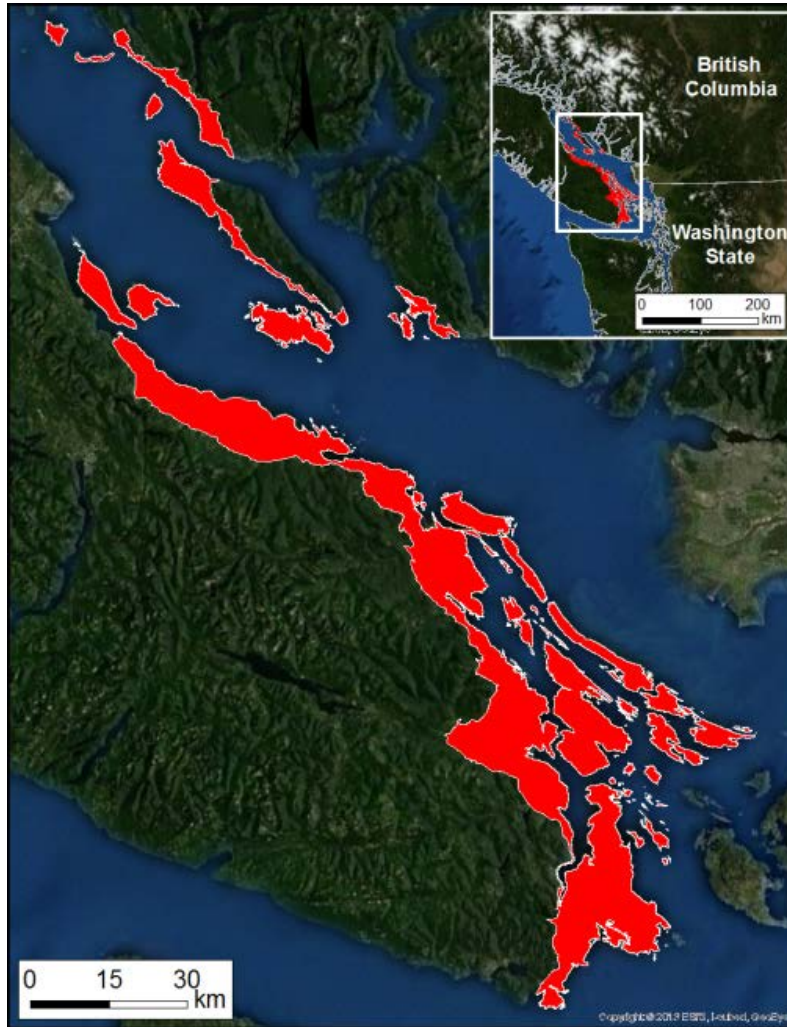
Land purchase cost scenario

To identify ideal conservation networks given fee-simple land purchases, I built on the results of Chapter 3, where I used distribution models for 47 bird species based on 25 remote-sensed predictor variables and models incorporating imperfect detectability (e.g., Mackenzie et al. 2002) to create composite scores of bird communities to represent native species biodiversity in Old Forest and Savannah habitats (Chapter 3). I combined Old Forest and Savannah composite scores into a beta-diversity metric to represent high-biodiversity landscapes. I used

cadastral data from the Integrated Cadastral Information Society of BC, the BC Assessment agency and the Integrated Cadastral Fabric, to identify planning units (actual properties) and land value assessments (as of 2012) to represent property cost. I then used Marxan (Ball et al. 2009) to prioritize cadastral polygons according to their beta-diversity metric for inclusion in conservation networks by using them as planning units (PU's; n = 193,623; Chapter 3). I created conservation networks to protect 20% of the total beta-diversity in the study region and retained 100 Marxan solutions to allow for variability in spatial network configuration and cost.

Covenant cost metrics and assumptions

To compare land purchase versus conservation covenant scenarios, I identified the properties selected in land purchase Marxan solutions also as covenant locations, assuming in both cases that owners were equally willing to sell or place a covenant on their property. I did not estimate opportunity costs of other land uses here since opportunity costs are not typically paid by conservation organizations (the focus of my study), but are often at least partly compensated via tax credits (Parker 2004, e.g., in BC Natural Area Protection Tax Exemption Program). I did not estimate devaluation of land given covenants as there is as yet no consensus on its magnitude or direction (Anderson and Weinhold 2008).

Covenant costs reflect land management experience at The Nature Trust of British Columbia (http://www.naturetrust.bc.ca/) and Islands Trust Fund (http://www.islandstrustfund.bc.ca/) following similar examples in literature (Main et al. 1999, Parker 2004). I compiled land managers estimates of fixed covenant costs including: legal, financial advice, registration and endowment fees (Table 4.1). Property size affected several variable costs related to ecological baseline surveys, appraisals and land boundary survey costs and I used land manager estimates to parameterize equations and produce scalable predictions

(Table 4.1). A third cost identified by land managers involves reoccurring costs related to annual

monitoring and staff time in replies to land owner questions or requests (Table 4.1). I calculated

all costs as present day Canadian dollars (August 2013) to allow for comparability and

consistency of results, instead of using discount rates equal to the inflation rate for costs that are

incurred over time and report future dollar values (Wilson et al. 2009).

Table 4.1: Covenant costs estimates from The Nature Trust of BC and Islands Trust Fund. All variable costs follow a saturating curve in the form of: cost = Intercept + Slope * ln(covenant size [acres]), with the constraint that the cost can't be below 'minimum cost'.

	Cost [$]
Fixed costs	
Land owner	
legal cost	300
financial advice	300
Covenant registration	200
Endowment	10000
Covenant holder	
legal cost	4000
Variable costs	
Ecological baseline	
minimum cost	1000
Intercept	2185
Slope	1957
Appraisal	
minimum cost	1500
Intercept	0
Slope	1957
Land survey	
minimum cost	1000
Intercept	300
Slope	1957
Reoccurring costs [yearly]	
Covenant monitoring	760
Staff cost to reply to Land owner request	152

Conservation covenant scenarios

As the goal of efficient land conservation here is to maximize the fraction of the high-biodiversity landscape that is protected (i.e., not developed further) given an investment in conservation, I calculated the cost-effectiveness of alterative scenarios as the fraction of the high-biodiversity landscape protected, divided by the total reserve network cost for each scenario (cf Wilson et al. 2007). I standardized this value by the cost effectiveness of land purchase to directly compare it to the land purchase scenario.

I used values reported by Rissman & Butsic (2011) to estimate the distribution of dispute costs and created a cost profile bound between $1000 and $400,000 following a power function (cost[$]=4845.78 * disputes$^{-0.701}$). I also tested cost profiles that allowed for dispute costs over $400k, by adding a truncated normal distribution with mean of $400k and SE of $1M, and a 1% probability that costs are from that distribution but found no marked effects on results and thus restricted my analysis to a range reported in literature. To find the covenant dispute rate that results in the cost effectiveness of land purchase to exceed that of a conservation covenant scenario, I used dispute rates of 2.8 out of 10000, 1000 and 100 covenants per year, which is equivalent to 0.028, 0.28 and 2.8%. The 2.8 value reflects the average number of yearly disputes derived from results published by Rissman & Butsic (2011). The 2.8 value does not reflect a dispute rate as Rissman & Butsic (2011) had no data on the total number of covenants this was out of (i.e. the denominator in the rate), which is why I chose a range of dispute rates. For each year (n=100) in my analysis, each covenant in a reserve system was subject to a potential dispute at the assumed rate. Given a dispute, a randomly drawn dispute cost was added to that properties covenant cost. To quantify the potential negative effect of disputes on biodiversity I estimated biodiversity loss by letting biodiversity loss follow the same distribution as dispute cost, but

bounded between 0 and 100%. Due to uncertainty in actual strength of association between

biodiversity loss and dispute cost, I relaxed the association slightly by allowing for variation in

the biodiversity loss value following a normal distribution around the estimate (SD=5% of total

biodiversity loss possible). All analyses were conducted using R v.3.0.2 (R Core Team 2013),

custom distributions were parameterized using package distr v.2.5.2 (Ruckdeschel et al. 2006)

and the R script for my analysis can be found in Appendix 7.

Results

Given a goal of protecting 20% of the high-biodiversity landscape in conservation

networks, land purchase scenarios protected a mean of 370 km^2 (range = 365-374 km^2) at a mean

cost of $457M (range = $441 – 470M) (Figure 4.2a). In comparison, the cost of an equivalent

area under conservation covenants averaged $43.9M in year 1 (range = $42.6 – 45.0 M) and

$162M cumulative to year 100 (range = $157 – 166M) including monitoring and staff costs

(Figure 4.2a), representing a 65% reduction in cost compared to land purchase. Including dispute

rates of 0.028 and 0.28% increased the long term costs of covenant scenarios by 2 and 23%,

respectively (Figure 4.2a). However, at the highest dispute rate (2.8% of covenants/ year),

network cost increased 3.4 fold (mean = $546M, range = $524 – 570 M), well above scenarios

without disputes, and it exceeded the cost of land purchase (Figure 4.2a).

My baseline, high-biodiversity scenario aimed to protect 20% of high-biodiversity

landscapes via land purchase or covenant, but in the absence of disputes. Given a low dispute

rate (0.028%), the high-biodiversity landscape protected declined by a mean rate of 0.75% (range

= 0.11 – 1.49%) after 100 years compared to the baseline protection (Figure 4.2b). An

intermediate dispute rate (0.28%) returned a mean decline of 7.31% (range = 5.25 – 9.25%), and

the highest dispute rate (2.8%) returned a mean decline of 53.62% (range = 49.33 – 57.7%) after

100 years (Figure 4.2b).

I defined the long-term cost effectiveness of conservation covenants as the fraction of

high-biodiversity landscapes protected, divided by the total cost of the network. I standardized

this value by the cost effectiveness of land purchase to directly compare covenant and land

purchase scenario. At low and intermediate dispute rates, covenants outperformed land purchase

(2.1 - 2.8 times more cost-effective after 100 years, Figure 4.3). However, a high dispute rate

drove the cost-effectiveness of covenants below that of land purchase within 50 years and was

only 39% as cost-efficient as land purchase after 100 years (Figure 4.3).

Figure 4.2: a) Conservation network cost comparison between land acquisition and conservation covenants of varying dispute rates. b) Biodiversity loss of varying covenant dispute rates in conservation networks and an initial 20% protection level of current biodiversity in the CDF ecological zone. Solid lines represent mean values for each approach, and the corresponding ribbons show minimum and maximum values for the 100 Marxan solutions.
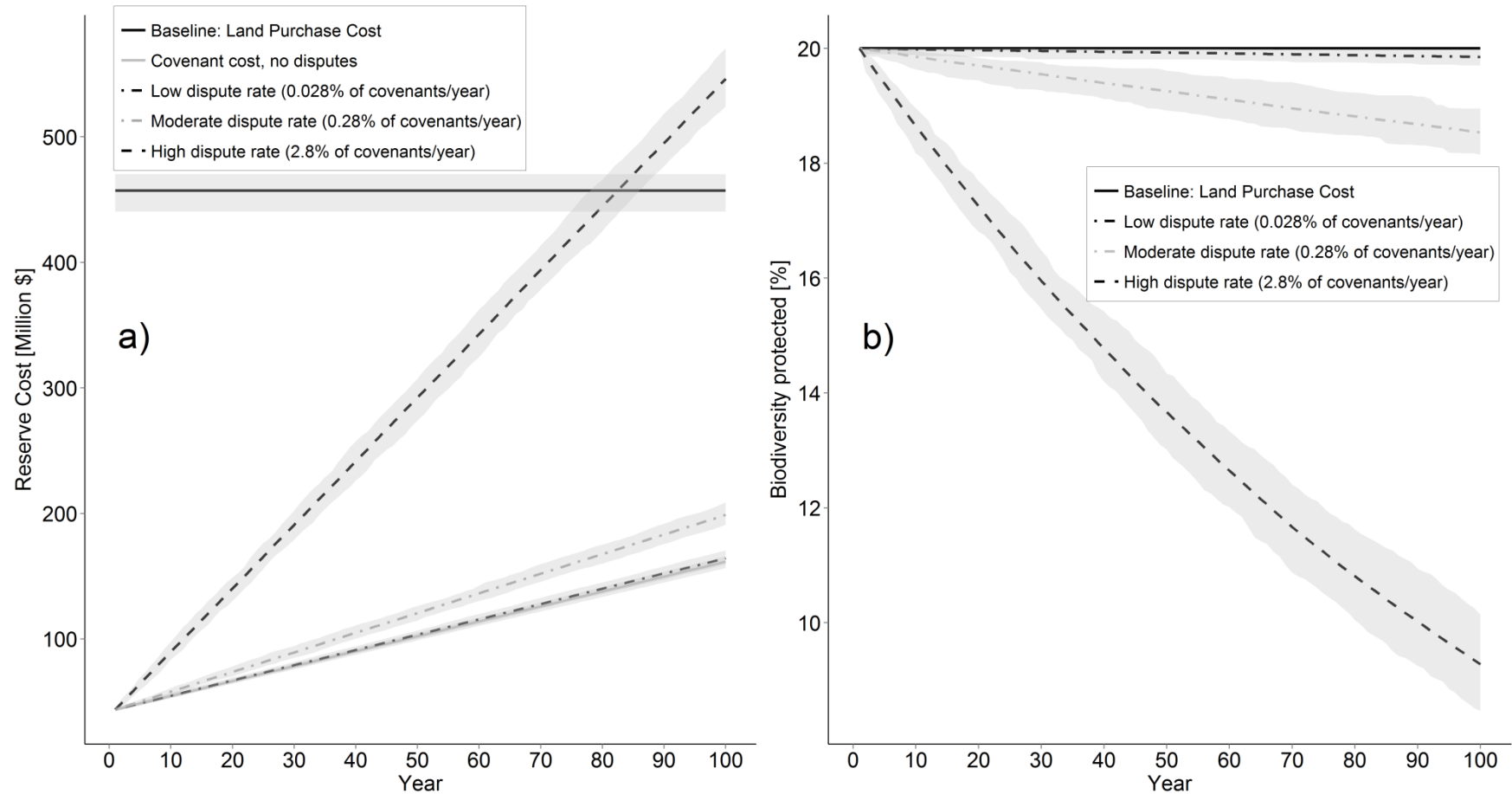
Figure 4.3: Long term cost effectiveness defined as rate of biodiversity protected divided by the reserve network cost. Values are relative to the baseline land purchase scenario. Solid lines represent mean values for each scenario, and the corresponding ribbons show minimum and maximum values for the 100 Marxan solutions.
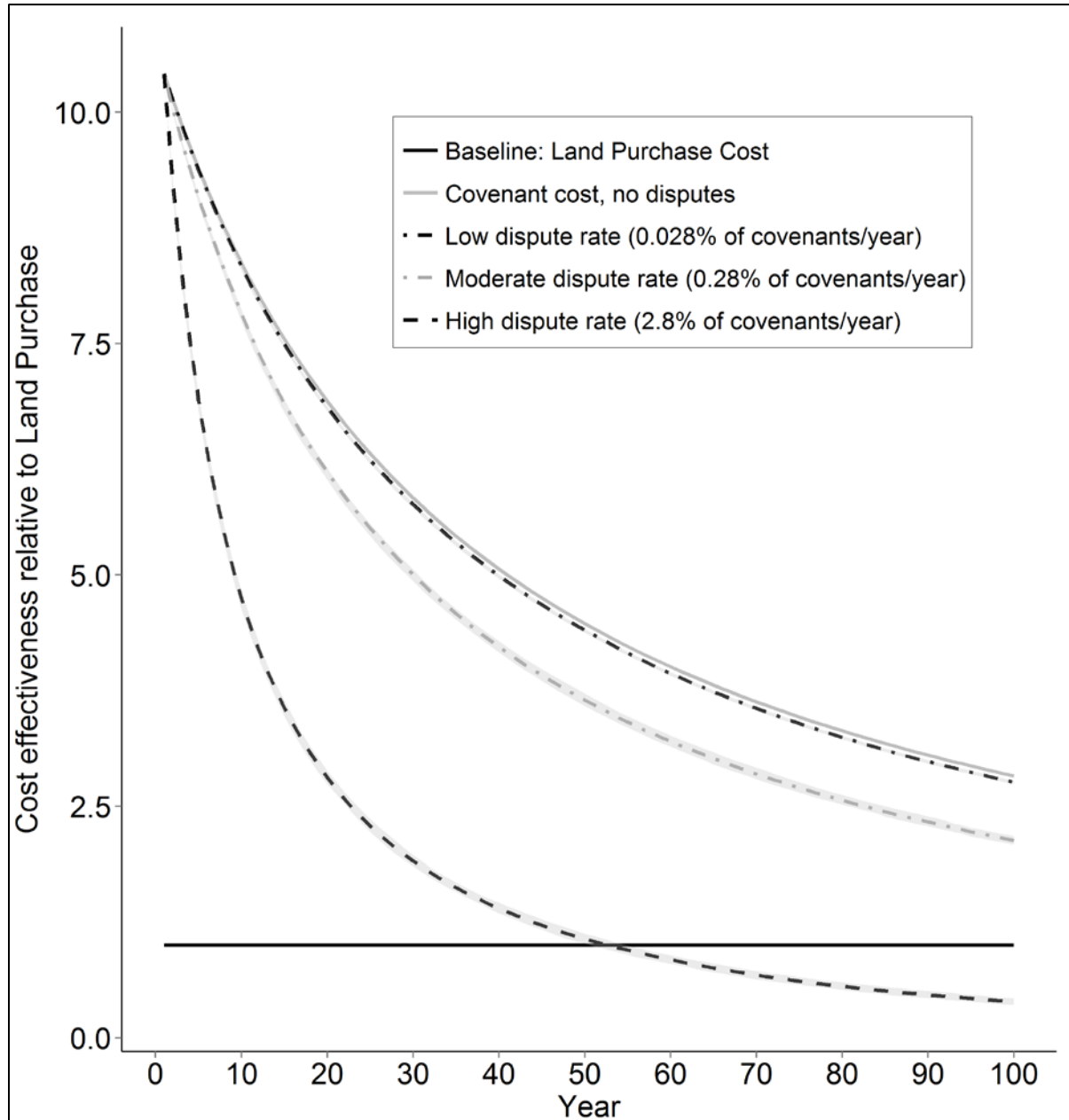
Discussion

I found that covenant violations and disputes can substantially affect the long-term cost-effectiveness of conservation strategies that employ covenants and land purchase to protect high-biodiversity landscapes. In particular, land purchase outperformed covenants as a cost-effective approach to protection when dispute rates were high, in part because disputes are also likely to involve reduced biodiversity protection (Figure 4.2b). These results raise questions about the cost-effectiveness of conservation covenants and potential liabilities to covenant holders. In contrast, the low initial cost of covenants vs land purchase suggests that as long as disputes are rare, conservation covenants are likely to outperform land purchase in terms of their cost-effectiveness of biodiversity conservation (Figure 4.3). I now develop these points in light of literature on land acquisition and conservation covenants and point out several remaining uncertainties.

Covenant dispute rate

I showed that the cost-effectiveness of covenant versus land purchase approaches to conservation is likely to depend on covenant dispute rate (Figure 4.3). This suggests that minimizing dispute rate should be a key goal for conservation organizations when covenants are created, because disputes raise costs and reduce the cost-effectiveness of biodiversity conservation. However, given the current paucity of data on the frequency and cost of disputes (Byers et al. 2005, Rissman and Butsic 2011), formalizing the experience of conservation organizations with historically drafted covenants is urgently needed to identify potential pitfalls and thereby reduce dispute rates in future. Anecdotal data suggest that disputes and legal challenges rise with the number of consecutive new owners of covenanted properties. If true,

these observations imply that any existing language in covenants that engenders risk should be remedied before ownership is transferred.

Dispute cost profile

I used a dispute cost profile based on limited data (Rissman and Butsic 2011) but note that costs may exceed $400k. Moreover, I used an inverse dispute cost profile, whereas the frequency of large costs may be a key concern in some organizations. Data are clearly required to clarify dispute cost profiles and facilitate economical risk analyses (Boyd et al. 2000, Game et al. 2013). However, more complex cost profiles are highly speculative and modest variation around the upper end of dispute costs did not significantly change my results.

Biodiversity loss connected to covenant dispute

To my knowledge, the potential for biodiversity loss via covenant violation was not explored in detail prior to my research. I found that, even at intermediate covenant dispute rates, the area of the high-biodiversity landscape conserved was predicted to decline by >7% after 100 years (Figure 4.3). Higher dispute rates resulted in the loss of over half of the originally protected high-biodiversity landscape, because disputes often involve land conversion, loss of protected elements or other compromises to site integrity (Smith 2009). Although my assumption that biodiversity loss and dispute cost vary directly is simplistic and as of yet not supported by data, it seemed reasonable to us to speculate that covenant violations and resulting alteration to a property (e.g., clearing the land or building a structure) would lead to a loss of high-biodiversity habitats on a covenanted property. My results emphasize that covenant disputes add management costs, but may also reduce high-biodiversity habitats, and that both of these costs should therefore be considered when comparing biodiversity conservation via covenant versus fee simple land purchase.

Conclusion

My results suggest that over the long-term, the cost-effectiveness of conservation covenants can outperform land purchase as a strategy to protect biodiversity, as long as the rate of disputes and legal challenges to covenants are low. To achieve low dispute rates in the long term I suggest two main avenues i) formalize the experience of conservation organizations and historically-drafted covenants to identify potential pitfalls and help avoid them and ii) remedy existing terms of covenants that engender risk before ownership is transferred. Both these actions should help to ensure that conservation covenants are a preferred option in biodiversity protection. Activities that could improve our ability to compare the cost-effectiveness of land purchase and covenants as approaches to biodiversity protection include obtaining better quantitative data on: i) covenant dispute rates and cost profiles over time, and ii) biodiversity loss given a dispute. My findings should apply to areas with similar patterns of private ownership and human impact as occurs in the Georgia Basin of western North America now experiencing similar development pressures and high land purchase and/or opportunity costs of conservation. In regions with less human impact, human demands for land purchase or use may be lower, potentially shifting the cost profiles I present in favour of a land purchase approach. If dispute rates are lower in these regions, conservation covenants might still represent the most cost-effective approach. Conservation covenants, if used with fixed term contracts, could further provide a more flexible approach than land purchase, to adapt to rapid environmental changes related to climate change.

Chapter 5: Conclusions

My thesis helped reach the overall goal of evaluating alternatives to successfully funding conservation investments via four novel advances to land use planning for conservation. These were i) providing new methods to identify areas of high-value biodiversity for conservation and/or restoration (Chapter 2); ii) contrasting α and β-diversity approaches in systematic conservation planning (Chapter 3); iii) investigating the potential conservation network cost reductions using carbon credit sales (Chapter 3); and iv) analysing the long term cost-effectiveness of conservation covenants compared to land purchase for conservation (Chapter 4). In the remainder of this chapter I present some of the limitations of this study and summarize the management implications of this work. I will introduce my plans for further research on the topics of this thesis, incorporating changes to species distributions due to climate and land use change, downscaling conservation prioritization scenarios from region wide to municipal levels, and ways to improve the conservation covenant cost-effectiveness analysis. Finally, I present general conclusions of my work by putting the previous chapters into perspective of the broader conservation literature.

Limitations of this study

Through providing robust distribution models for 47 bird species (Chapter 2), integrating carbon offsets as well as α- and β-diversity maximization into conservation planning (Chapter 3), and analyzing the long term cost-effectiveness of conservation covenants (Chapter 4), my thesis advances systematic conservation planning in several ways, and contributes important new information to help guide conservation and restoration in the Coastal Douglas-fir Biogeoclimatic zone. However, my work also has several potential limitations, some of which I propose to

address in the future, while others may be difficult to overcome due to logistical constraints and uncertainties in ecological and social systems.

Biodiversity data

Due to the expanse of the study area it would be logistically and economically very challenging to collect enough data in one field season to allow for single-season model development. The bird point count data used were collected over multiple years (seasons) by multiple observers for varying purposes. As a result the closure assumption of the occupancy models might have been violated, leading to biased estimates (Mackenzie et al. 2002, 2006, Mackenzie and Royle 2005). As the data were not sufficient to include colonization and extinction effects (MacKenzie et al. 2003), I had to combine datasets in order to be able to use this approach, although less uncertainty about colonization and extinction events would be possible with the former approach (Mackenzie and Royle 2005). Allowing for colonization and extinction would further allow to specifically link to demographic models and identify source and sink locations, especially if abundance data were used instead of detection/non-detection data (Royle 2004). In practice multi-season abundance data are hard and costly to collect and in many cases "presence only" data is the only available type of data that can be used for species distribution models, e.g., from natural history collections (Graham et al. 2004, Elith and Leathwick 2009). The detection/non-detection data I used represents a huge improvement on these types of data in terms of quality, and the inferences that can be drawn from them are generally less biased than those from 'standard' presence only data (Brotons et al. 2004, Hastie and Fithian 2013).

Cadastral polygon and land value data

As there is currently no centralized entity that curates cadastral data for British Columbia, I had to use three sources for the cadastral polygon data (Integrated Cadastral Information Society of BC, the BC Assessment agency and the Integrated Cadastral Fabric). As a result, there might be issues with the correct parcel boundaries in some places and when combining the data I noticed in several places that polygon overlaps were quite substantial (several $100m^2$) and in many cases individual polygons appeared twice in the data set. The total initial area of overlap from the cadastral layer was 1123 km^2 I tried to correct these errors using several GIS tools and manual inspection, but because there were over 190k polygons in my data set some errors might still be present. These overlaps could potentially lead to biased results in Marxan analysis if the same biodiversity values were included twice (once in each overlapping part), and caution should be taken when applying my analysis in a practical setting. Further, assessed land values were hugely discounted for some properties, or missing entirely and there is no analytical way to estimate true land value for those properties (R. Paxton, BC Assessment, personal communication). I chose to interpolate missing or discounted land values using inverse distance weighting of adjacent land values, but this is only an approximation of true land value and might introduce some bias in the conservation prioritization results. This potential issue affected a total of 3.3% of cadastral polygons, and given that only 0.7% of polygons were selected in Marxan solutions for Chapter 4 and parts of Chapter 3 this has the potential to introduce errors in the results. By using inverse distance weighting on 10 polygon size groups I am confident to have been able to minimize this risk though and my method clearly represents an improvement on using discounted and therefore erroneous land value assessments without correcting for this bias.

Current protected areas and connectivity in conservation prioritization

I did not include currently protected areas as the focus of my work was not to build on existing protected areas, but to investigate general cost-effectiveness of conservation approaches. I further did not take connectivity between protected parcels in my conservation prioritization scenarios into account, because estimating connectivity via the boundary length modifier in Marxan is highly subjective, project specific, and that has the potential to bias results substantially. Doing so allowed me to answer my research questions, but in order to be applicable for prioritization of land purchases or conservation covenants in practice, careful thought has to be used to overcome both these shortcomings. Further, I have used a region wide approach without refining the prioritization by municipal or regional district boundaries, although a substantial part of conservation efforts and related prioritization decisions lies with local governments in the study region. Local government level conservation prioritizations will have to be completed to allow for actionable plans focused on each regional district in the study region. As with all prioritization software tools the results are to be understood as general guidelines, and for on-the-ground prioritization these analyses need to be informed further by decision makers in the area to ensure prioritization outcomes are realistic and meet the approval and needs of stakeholder groups involved (Knight et al. 2009).

Conservation covenant data

Finally, caution should be taken when interpreting and generalizing from the results regarding Chapter 4 (conservation covenants versus land purchase) and generalizing its conclusions. Caution is warranted due to the lack of data and the level of uncertainty related to long term predictions (in this case 100 years). The main goal of this chapter was to provide a conceptual framework for comparing the long term cost effectiveness of conservation covenants

to land purchase, while still parameterizing the analysis with as much data as is available at the moment. At the same time I wanted to identify current shortcomings in data available on this issue, to point out potential areas of future research. The medium to long term success rate of conservation covenants is one of the biggest gaps in data related to this issue, but these data could help improve our understanding of the cost effectiveness of conservation covenants substantially.

The potential for biodiversity loss via covenant violation has not been explored previously. As a result of this lack of data on biodiversity loss I have linked biodiversity loss directly to conservation covenant violation cost. I do realize that this might be biased at both extremes (the low and high dispute cost rate). Initial biodiversity loss could be higher than relative dispute cost, e.g., if an integral part of the biodiversity on a covenant is easily disturbed, even at very low dispute costs. At the same time the highest dispute cost does not necessarily mean that all biodiversity is lost. As with dispute rates, I chose this approach to raise awareness of the potential issues, which is why I refrained from using a more complex distribution for biodiversity loss, which would have been very speculative and potentially distracted from the goal of this study.

Management implications of results

The CDF is considered one of British Columbia's most endangered ecosystems (Austin et al. 2008). Recently a number of local government agencies, Land Trusts and conservation organizations have joined conversation efforts for the CDF and created the Coastal Douglas-fir and Associated Ecosystems Conservation Partnership (CDFCP). "*The CDFCP arose from the recognition of a need for a more strategic and collaborative approach among those involved and interested in conservation efforts in Coastal Douglas-fir ecosystems, and was developed through*

a series of discussions and workshops including different levels of governments, non-government conservation organizations, and community residents who believe that by working together, we can more effectively achieve our shared conservation goals. The CDFCP promotes shared stewardship and will identify conservation priorities, reduce duplication of effort, share resources and information, and provide support to its participants."[4]

In addition, Parks Canada and The Nature Conservancy Canada have engaged local scientists to identify remaining areas of older coastal Douglas-fir forests and oak meadows for potential acquisition using remote-sensed information on land cover, but without the benefit of detailed data on the distribution of plant or animal species potentially representing old forest ecological values (De Wan et al. 2009). Enhancing the planning database to include distribution maps of species of interest to conservation, particularly those indicative of old-growth Coastal Douglas-fir forests and oak meadows, has the potential to dramatically increase the effectiveness of resulting conservation network designs for the region, and is considered a high priority by all stakeholders in the area.

I have developed methods to maximize the efficiency of conservation actions for the protection and restoration of critical wildlife habitat in a region where land conversion has decreased native species richness and eliminated old-growth forest and woodland habitats. My research provides novel solutions to conservation network design, enhances the persistence of valued native species, and provides immediate guidance to land use planners trying to maximize the value of their investments in conservation covenants and acquisition (Wiersma and Nudds 2009). My results will also advance the application of conservation planning tools by

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[4] http://www.cdfcp.ca/index.php/about-the-cdfcp/about-the-partnership

determining how carbon offsets can be used to produce the most economically and biologically efficient solutions for land acquisition and protection using systematic planning software.

Ongoing Applications

I am currently applying results from this thesis at three different scales from local to regional to Provincial. For Islands Trust (IT, local scale) I created a report showing the contribution of the IT area to conservation in the Coastal Douglas-fir (CDF) Biogeoclimatic zone. In a second project I extended my results to the wider CDFCP region in order to identify the extent of a range of biodiversity features in several land tenure categories in the CDFCP region, and to investigate the roles the different categories could have in habitat protection and what to focus future conservation efforts on. On a provincial level I am working with The Nature Trust of British Columbia (TNTBC) to develop a product that allows TNTBC to more thoroughly and efficiently evaluate potential securement opportunities for conservation using scientific rules of thumb, spatial arrangement, and extent of compatible or incompatible land uses. To extend my results to a cross boundary scale I started to work with the North Pacific Landscape Conservation Cooperative (NPLCC) on a project striving to synthesize existing regional models of invasive/native species distribution, forest and savannah ecosystems and climate change to prioritize land acquisition and conservation investment throughout Georgia Basin and integrate those results with US partners to facilitate cross-boundary planning for endangered forest/savannah habitat in the Pacific Northwest.

Future work

My analyses have contributed to the overall goal of evaluating alternatives to successfully funding conservation investments by identifying high biodiversity areas (Chapter 2) and assessing alternative mechanisms for funding conservation goals (Chapter 3,4). In this section I

am proposing six ways to extend these findings, and make the results from this thesis more applicable for local managers and land use planners as well as address some of the limitations of this thesis presented earlier.

Extending biodiversity predictions to incorporate future climate and land use change scenarios

I will project the predictive species distribution models created for Chapters 2 and 3 into the future. I will achieve this by adjusting the current modeling approach to include climate predictor variables for the region (Wang et al. 2006, 2012) and re-build the models using these predictors. I will then use predicted future climate values to extrapolate model results into the future to investigate potential distribution changes of individual bird species in relation to climate change, similar to (Matthews et al. 2011). In addition, I will create predictive scenarios of land use change in the future (Sohl et al. 2013) and investigate a suite of land development scenarios to adjust current predictions of species distributions. I will individually investigate climate change and land use change effects on bird diversity in the region, as well as use a combined approach to identify potential compounding effects that both have on the future extent of Old Forest and Savannah ecosystem related bird species diversity. Lastly, I will investigate the long term sustainability of proposed conservation networks in light of climate and land use change effects.

Broadening biodiversity goals to increase the representation of distinct species communities

The focus on forest and savannah ecosystems using bird species as indicators in the CDF made sense for my study with a focus on identifying high quality biodiversity landscapes in the CDF for conservation and restoration. Prior to European colonization the CDF occurred as

uneven-aged forest (often >300 years old) dissected by shallow and deep-soil meadow and woodland communities (Meidinger and Pojar 1991, Mosseler et al. 2003) maintained in part by aboriginal land management practices to enhance hunting opportunities and root and fruit harvests (MacDougall et al. 2004, Dunwiddie and Bakker 2011, McCune et al. 2013, Turner 2014). As birds only represent one taxonomic group in the CDF, I propose to expand the range of biodiversity features to be used in conservation prioritization by predictive models on plant species distribution for the 'Garry oak ecosystem'(GOE) (Bennett 2014). The Coastal Douglas-fir Conservation Partnership has further identified a number of biodiversity features of interest to them, which I have recently included in technical report to them. These are element occurrences of Douglas-fir/Oregon Grape communities and Sensitive Ecosystem Inventory data (http://www.env.gov.bc.ca/sei/, date accessed: 2011-10-10) such as 'Coastal Bluff', 'Terrestrial Herbaceous', 'Wetland' and 'Riparian'. The latter two would additionally be useful in conservation efforts to create reserve networks that buffer streams with carbon as a way to connect parts of the landscape, using recent approaches developed for connecting freshwater systems or land and sea ecosystems (Moilanen et al. 2008, Linke et al. 2011, Makino et al. 2013).

Incorporating existing protected areas and network connectivity to increase reserve network resilience

I will extend the approach of spatial prioritization for conservation in the area, by taking into account exiting protected areas. I will use these areas as the conservation network baseline for further analysis and employ Marxan to add to the currently existing reserve network to ensure compactness of future reserve systems. This approach will be taken in collaboration with members of the Coastal Douglas-fir and Associated Ecosystems Conservation Partnership

(CDFCP, http://www.cdfcp.ca/) and other conservation groups in the area to ensure my results are realistic and applicable to regional goals. To buffer streams with carbon, as mentioned above, I will include connectivity in the reserve system to ensure proposed reserve networks will allow for species to move as habitats get less suitable due to climate change or are under threat due to land use change or development. How to best include connectivity in a reserve network is a question of ongoing research in the spatial conservation prioritization literature and there is much debate on the appropriate metrics to use and how to include them in a prioritization approach. I propose to test several metrics and ways to accommodate this, including graph theoretic approaches to Marxan (Bode et al. 2008a, Kininmonth et al. 2010) and recent developments from Landscape Ecology (Pascual-Hortal and Saura 2006, McRae et al. 2008, Urban et al. 2009).

Extending conservation covenant cost-effectiveness analysis to help reduce covenant dispute rates

To help improve the reliability of cost-effectiveness calculations for conservation covenants I plan to survey regional Land Trusts to collect data on covenant dispute rates and costs. This additional data should help make the cost-effectiveness analysis more realistic and identify the current rate of conservation covenant disputes, and allow for better estimates of the long term costs of conservation covenants compared to land purchase. Data on biodiversity loss related to covenant dispute will be harder to collect, but I am hoping that the largest land trust in North America, The Nature Conservancy (TNC), will have at least some data on this. My plan is to seek collaboration with TNC in Canada and the US in order to collect data on biodiversity loss related to conservation covenant violations. In case these data are not available directly, I hope to at least approximate this loss by collecting biodiversity data from conservation covenant properties that have experienced a covenant violation. As conservation covenants usually have

an ecological baseline assessment done when they are put in place, a repeat ecological baseline assessment after a violation could allow for the collection of before and after disturbance data and help parameterize the biodiversity loss function of the conservation covenant cost-effectiveness analysis.

General Conclusions of this thesis

Identifying high biodiversity areas

The cost and effort required to reliably map the distribution of rare species and communities targeted for conservation often results in the adoption of coarse-scale targets and 'ad hoc' criteria in site prioritization plans, frequently at the cost of ecosystem representation and species diversity (e.g., Arponen et al. 2008, Fuller et al. 2010). An alternative approach, used here, is to develop predictive distribution maps for representative taxa that, based on known habitat affinities, ease of detection and abundance, can effectively estimate native species richness and can be used to prioritize areas of conservation interest (e.g., Branton and Richardson 2011). Predictive distribution maps of common indicators could help planners identify habitat patches likely to support valued communities of native species (Müller et al. 2003, Hijmans and Graham 2006) if imperfect detection (Mackenzie et al. 2002), biased sampling and validation are addressed.

In practice "presence only" data are often the only available type of data that can be used for species distribution models, e.g., from natural history collections, despite their limitations such as sampling bias (Graham et al. 2004, Elith and Leathwick 2009). The systematically collected, long term detection/non-detection data I used represents a huge improvement on these types of data in terms of quality, and the inferences that can be drawn from them are generally

less biased than those from 'standard' presence only data (Brotons et al. 2004, Hastie and Fithian 2013). I further provided a method that helps to discriminate among habitat fragments of similar size and age but different value to birds based on local or landscape context, which should help facilitate forest restoration and conservation plans (Chazdon et al. 2009, Fuller et al. 2010).

By drawing on the cumulative experience of expert birders in the Costal Douglas-fir zone, I was further able to quantify the local scale habitat affinity of a suite of bird species to inform the creation of my composite maps. This approach is a fairly inexpensive method to gather information on localized data of bird species habitat affinity and I believe that with the information provided in Chapter 2, as well as Appendices 1 and 8, land managers will be able to apply my methods in other study regions, especially with the rising availability of detection/non-detection data through open data repositories (e.g., GEBIF http://www.gbif.org/) or citizen science project (e.g., eBird http://ebird.org) . To make my model approach widely applicable, I only used freely available data as variables in my models to allow easy access to practitioners interested in applying my species distribution modelling methods in other regions (Jewell et al. 2007). Variables were derived for the most part from either Landsat (Wulder et al. 2008) or aerial photographs (MES 2008) (Table 2.2 and A.4.1). I further made the code to generate my model sets freely available (Appendix 3) and the methodology is detailed in Chapter 2 and Appendix 4. The high biodiversity Old Forest and Savannah habitat areas identified in Chapter 2 formed an integral part of Chapters 3 and 4, as these results represented the biodiversity inputs of all conservation prioritization and biodiversity loss analyses conducted in Chapters 3 and 4.

Carbon credit sales and maximizing β-diversity as mechanisms for reaching conservation

goals

In Chapter 3 I provided an example of a rigorous and cost-effective approach to use for conserving lands in human-dominated landscapes, by showing that carbon credit sales have the potential to enhance conservation outcomes in human-dominated landscapes through reducing the net acquisition costs of conservation land. These results compliment suggestions that carbon payments have the potential to advance conservation and mitigate the impacts of climate change (Venter et al. 2009a, 2009b, Bradshaw et al. 2013), but extend those suggestions by providing a spatially explicit, empirical example in a landscape with high conservation and cultural values (Austin et al. 2008, Vellend et al. 2008, Arcese et al. 2014). I showed that maximizing β versus α-diversity may further help reduce costs and enhance landscape heterogeneity. This is promising news as prior studies of the consequence of emphasizing α- versus β-diversity in conservation planning have concluded that a focus on β-diversity is likely to enhance long term persistence in diverse species assemblages and reserve networks (Fairbanks 2001, Wiersma and Urban 2005, Marsh et al. 2010). Under climate change, an emphasis on community dissimilarity (β-diversity) may deliver more robust conservation plans than those based on species richness (Arponen et al. 2008, Mokany et al. 2011). My results broaden these conclusions by showing that scenarios that maximize β-diversity may also reduce the cost of conservation by reducing the area required to meet realistic targets for land acquisition.

Conservation area design does not exist in isolation from government needs and budgets; hence a number of adjustments are still needed before this theoretical analysis can facilitate land management in practice. First, current protected areas were not selected by default in this analysis as I focused on the academic problem that any parcel in the study region could be

purchased with the same likelihood, which is clearly not the case in practice as there might be unwilling sellers, unrealistic asking prices or other issues that hinder land purchase (Knight et al. 2011). A further complication with using currently protected areas in the analysis would be that land managers might want to build on existing protected areas to reduce overall management costs, as the long term management cost per ha should decrease with property size (Armsworth et al. 2011, Adams et al. 2012). This could lead to suboptimal, more expensive conservation networks. Other issues with using currently protected areas are that they might not be located in the highest biodiversity value spots, given a history of protecting lands in locations of low productivity or natural resource value (Pressey 1994). This would lower the biodiversity values protected and require additional land purchases to reach conservation goals. It has also been suggested that under future climate conditions only a small fraction of current protected areas in the CDF have the potential to accommodate shifts in oak woodland and savannah communities (Pellatt et al. 2012), illustrating that current protected areas might not be the best places for long term protection of high value biodiversity. This further emphasises the importance of focussing on protecting areas of high β-diversity as I have done here.

Second, I did not consider the spatial distribution of proposed conservation networks, nor did I take into account or set the spatial configuration goal for the reserve networks created, such as ensuring the connectivity between habitat patches. This omission might negatively affect the long term performance of proposed conservation networks due to a lack source – sink population connections. In Marxan connectivity is usually achieved by using the Boundary length modifier and visual inspection of the reserve network (Ardron et al. 2010), ideally in collaboration with the land manager and other stakeholders that would finance and/or manage the reserve network (Knight et al. 2009). The boundary length modifier (BLM) is used to consolidate areas for

management efficiency and/or general connectivity, to allow for the long term species persistence as well as source-sink dynamics over time (Rayfield et al. 2011). Recent studies have extended this BLM approach to more realistic metrics of connectivity, especially for freshwater systems (Moilanen et al. 2008, Linke et al. 2011). Marxan now also allows for directionality in connectivity, which is important for such things as stream conservation efforts, because downstream effects will most likely differ from upstream effects (Makino et al. 2013). In the CDF these approaches could be used to create reserve networks that buffer streams with carbon as a way to connect the landscape. Setting connectivity metrics can be subjective and usually involves several iterations specific to a given project (Knight et al. 2009). As such, I did not attempt to speculate how to best incorporate connectivity. In practice, this is an area requiring further investigation such that the results can be applied in a meaningful way. This would ideally be done in collaboration with the land manager and other stakeholders that would finance and/or manage the reserve network (Knight et al. 2009).

Contrasting the long term cost effectiveness of conservation covenants and land purchase

In Chapter 4 I provided an assessment of alternative mechanisms for funding conservation goals and showed that the long term cost-effectiveness of covenant versus land purchase is impacted by covenant dispute rate. This suggests that minimizing dispute rate should be a key goal for conservation organizations when covenants are created, because disputes raise costs and reduce the cost-effectiveness of biodiversity conservation. Further, I raise the point that the potential for biodiversity loss via covenant violation has not been explored previously. My results suggest that over the long-term, the cost-effectiveness of conservation covenants can outperform land acquisition as a strategy to protect biodiversity, as long as the rate of disputes and legal challenges to covenants and the related biodiversity loss are low. One way to help

achieve low dispute rates would be to ensure that the legal wording of covenant contracts leaves little room for challenges to the covenant.

Previous work has shown that conservation covenants can cost much less than direct land purchase (Pence et al. 2003) and further enable voluntary conservation agreements where landowners wish to retain ownership (Langholz and Lassoie 2001, Winter et al. 2005, Knight et al. 2010). This is a main reason why conservation covenants and other private land management agreements have gained global attention as an important policy tool for meeting conservation objectives (Fishburn et al. 2009, Gordon et al. 2011). Prior to my work, few studies had examined the potential complexities of such investments to ask whether covenants or land purchase maximize conservation of high-biodiversity landscapes (Armsworth and Sanchirico 2008, Fishburn et al. 2009) and none have compared the long-term cost-effectiveness of both approaches.

One of the biggest challenges to placing conservation covenants on high biodiversity value landscapes is willing landowners (Knight et al. 2011). In Canada, and British Columbia in particular, there are incentives such as tax credits (Parker 2004, e.g., Natural Area Protection Tax Exemption Program), but none compensate land owners for the lost opportunity costs of developing their land or using it in other revenue-generating manners like agriculture or forestry, as proposed elsewhere (Drechsler et al. 2007, Klimek et al. 2008, Bunn et al. 2013). Currently, a conservation covenant approach is hugely dependent on individuals wanting to create a legacy and see their property protected into the future (Moon and Cocklin 2011). As conservation covenants represent a viable way to protect high value biodiversity in the medium-to-long term, we should improve incentives such as tax exemption programs (e.g., the Natural Areas Protection Tax Exemption Program in the study region) that also attract participation from land

owners interested in economic gain rather than conservation in its own right. Compensating land owners for lost opportunity costs would certainly shift the cost profile of conservation covenants closer to a land purchase approach, but might still be cheaper than the costs of purchasing the land out-right. There is a potential range of options that might facilitate covenant payments. Additional approaches include those that could be extended (the tax advantages mentioned above), as well as some that are partially realized, like the carbon offsets investigated in Chapter 3 and others not yet widely used, such as biodiversity offsets (Bull et al. 2013).

In addition to the issue of cost effectiveness, there is another question as to the effectiveness of covenants versus land purchase because the latter implies total management control. For example, in the study region with its history of low intensity fires for stand regeneration (MacDougall et al. 2004, Dunwiddie and Bakker 2011), sole ownership of islands allows for management actions that include low intensity fires. However, ownership does not allow for this management action in parts of the landscape where fire cannot be used, indicating that overall effectiveness of conservation will vary for many reasons other than covenant status. Covenant effectiveness will also depend on the covenant type. While covenants sometimes protect against all likely threats (by including protection against all kinds of development rights), other times they just reserve some rights (e.g., protection of the forest canopy) (Parker 2004, Rissman 2013). For example, a miniature golf course underneath a covenanted old growth canopy is unlikely to maintain biodiversity protection.

Given that the vulnerabilities of both land purchase and conservation covenant approaches are manifold, they require careful thought. Activities which could immediately improve our ability to compare the cost-effectiveness of these approaches include obtaining better quantitative data on: i) covenant dispute rates and cost profiles over time, and ii)

biodiversity loss due to disputed covenants. Gathering dispute rate and cost profile data might be

a challenge as this is fairly sensitive data that conservation organizations might not be willing to

share, but this is the only way to empirically test the cost-effectiveness of conservation covenants

compared to land purchase.

References

Adams, V. M., R. L. Pressey, and N. Stoeckl. 2012. Estimating land and conservation management costs: the first step in designing a stewardship program for the Northern Territory. Biological Conservation 148:44–53.

Agrawal, A., D. Nepstad, and A. Chhatre. 2011. Reducing emissions from deforestation and forest degradation. Annual Review of Environment and Resources 36:373–396.

Akaike, H. 1974. A new look at the statistical model identification. IEEE Transactions on Automatic Control 19:716–723.

Allombert, S., A. Gaston, and J. Martin. 2005. A natural experiment on the impact of overabundant deer on songbird populations. Biological Conservation 126:1–13.

Anderson, K., and D. Weinhold. 2008. Valuing future development rights: the costs of conservation easements. Ecological Economics 68:437–446.

Ando, A., J. Camm, S. Polasky, and A. Solow. 1998. Species Distributions, Land Values, and Efficient Conservation. Science 279:2126–2128.

Andrén, H. 1994. Effects of Habitat Fragmentation on Birds and Mammals in Landscapes with Different Proportions of Suitable Habitat: A Review. Oikos 71:355–366.

Araújo, M. B., P. H. Williams, and R. J. Fuller. 2002. Dynamics of extinction and the selection of nature reserves. Proceedings of the Royal Society of London. Series B: Biological Sciences 269:1971–1980.

Arcese, P., R. Schuster, L. Campbell, A. Barber, and T. G. Martin. 2014. Deer density and plant palatability predict shrub cover, richness and aboriginal food value in a North American island archipelago. Diversity and Distributions in press.

Ardron, J. A., H. P. Possingham, and C. J. Klein (Eds.). 2010. Marxan Good Practices Handbook, Version 2. Page 165. Pacific Marine Analysis and Research Association, Victoria, BC, Canada.

Armsworth, P. R., L. Cantú-Salazar, M. Parnell, Z. G. Davies, and R. Stoneman. 2011. Management costs for small protected areas and economies of scale in habitat conservation. Biodiversity & Conservation 17:3589–3606.

Armsworth, P. R., and J. N. Sanchirico. 2008. The effectiveness of buying easements as a conservation strategy. Conservation Letters 1:182–189.

Arponen, A., A. Moilanen, and S. Ferrier. 2008. A successful community-level strategy for conservation prioritization. Journal of Applied Ecology 45:1436–1445.

Arroja, L., A. C. Dias, and I. Capela. 2006. The Role of Eucalyptus Globulus Forest and Products in Carbon Sequestration. Climatic Change 74:123–140.

Austin, M. A., D. A. Buffett, D. J. Nicolson, G. G. E. Scudder, and V. Stevens. 2008. Taking Nature's Pulse: The Status of Biodiversity in British Columbia. Pages 1–268. Biodiversity BC, Victoria, BC.

Ball, I. R., H. P. Possingham, and M. E. Watts. 2009. Marxan and relatives: Software for spatial conservation prioritisation. Pages 185–195 *in* A. Moilanen, K. Wilson, and H. P. Possingham, editors. Spatial conservation prioritisation quantitative methods and computational tools. Oxford University Press, Oxford.

Bayon, R., and M. Jenkins. 2010. The business of biodiversity. Nature 466:184–5.

Beaudry, F., A. M. Pidgeon, V. C. Radeloff, R. W. Howe, D. J. Mladenoff, and G. A. Bartelt. 2010. Modeling regional-scale habitat of forest birds when land management guidelines are needed but information is limited. Biological Conservation 143:1759–1769.

Bennett, J. R. 2014. Comparison of native and exotic distribution and richness models across scales reveals essential conservation lessons. Ecography 37:120–129.

Bennett, J. R., and P. Arcese. 2013. Human Influence and Classical Biogeographic Predictors of Rare Species Occurrence. Conservation Biology 27:417–421.

Bennett, J. R., M. Vellend, P. L. Lilley, W. K. Cornwell, and P. Arcese. 2012. Abundance, rarity and invasion debt among exotic species in a patchy ecosystem. Biological Invasions 15:707–716.

Beyer, H. L. 2004. Hawth's analysis tools for ArcGIS. Spatial Ecology LLC, Marshfield, WI. http://www.spatialecology.com/htools/index.php.

Beyer, H. L. 2010. Geospatial Modelling Environment 0.5.1 Beta. Spatial Ecology LLC, Marshfield, WI. http://www.spatialecology.com/gme/.

Beyer, H. L. 2012. Geospatial Modelling Environment (Version 0.7.2.1). (software). URL: http://www.spatialecology.com/gme.

Bivand, R. S., E. J. Pebesma, and V. Gómez-Rubio. 2013. Applied Spatial Data Analysis with R. Page 405. Springer, New York, NY.

Blair, R. B. 1996. Land Use and Avian Species Diversity Along an Urban Gradient. Ecological Applications 6:506–519.

Blanco, J. A., B. Seely, C. Welham, J. P. (Hamish) Kimmins, and T. M. Seebacher. 2007. Testing the performance of a forest ecosystem model (FORECAST) against 29 years of

field data in a Pseudotsuga menziesii plantation. Canadian Journal of Forest Research 37:1808–1820.

Bode, M., K. Burrage, and H. P. Possingham. 2008a. Using complex network metrics to predict the persistence of metapopulations with asymmetric connectivity patterns. ecological modelling 214:201–209.

Bode, M., K. A. Wilson, T. M. Brooks, W. R. Turner, R. A. Mittermeier, M. F. McBride, E. C. Underwood, and H. P. Possingham. 2008b. Cost-effective global conservation spending is robust to taxonomic group Proceedings of the National Academy of Sciences 105 :6498–6501.

Borcard, D., P. Legendre, and P. Drapeau. 1992. Partialling out the spatial component of ecological variation. Ecology 73:1045–1055.

Boulinier, T., J. D. Nichols, J. E. Hines, and J. R. Sauer. 2001. Forest fragmentation and bird community dynamics: inference at regional scales. Ecology 82:1159–1169.

Bowen, M. E., C. A. McAlpine, A. P. N. House, and G. C. Smith. 2007. Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna. Biological Conservation 140:273–296.

Bowen, M. E., C. A. McAlpine, L. M. Seabrook, A. P. N. House, and G. C. Smith. 2009. The age and amount of regrowth forest in fragmented brigalow landscapes are both important for woodland dependent birds. Biological Conservation 142:3051–3059.

Boyd, J., K. Caballero, and R. D. Simpson. 2000. Law and economics of habitat conservation: Lessons from an analysis of easement acquisitions, the. Stan. Envtl. LJ 19:209.

Bradshaw, C. J. A., D. M. J. S. Bowman, N. R. Bond, B. P. Murphy, A. D. Moore, D. A. Fordham, R. Thackway, M. J. Lawes, H. McCallum, S. D. Gregory, and others. 2013. Brave new green world--Consequences of a carbon economy for the conservation of Australian biodiversity. Biological Conservation 161:71–90.

Branton, M., and J. S. Richardson. 2011. Assessing the Value of the Umbrella-Species Concept for Conservation Planning with Meta-Analysis. Conservation Biology 25:9–20.

Brotons, L., W. Thuiller, M. B. Araújo, and A. H. Hirzel. 2004. Presence-absence versus presence-only modelling methods for predicting bird habitat suitability. Ecography 27:437–448.

Bull, J. W., K. B. Suttle, A. Gordon, N. J. Singh, and E. J. Milner-Gulland. 2013. Biodiversity offsets in theory and practice. Oryx 47:369–380.

Bunn, D., M. Lubell, and C. K. Johnson. 2013. Reforms could boost conservation banking by landowners. California Agriculture 67.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer Verlag, New York, NY.

Butchart, S. H. M., M. Walpole, B. Collen, A. van Strien, J. P. W. Scharlemann, R. E. A. Almond, J. E. M. Baillie, B. Bomhard, C. Brown, J. Bruno, K. E. Carpenter, G. M. Carr, J. Chanson, A. M. Chenery, J. Csirke, N. C. Davidson, F. Dentener, M. Foster, A. Galli, J. N. Galloway, P. Genovesi, R. D. Gregory, M. Hockings, V. Kapos, J.-F. Lamarque, F. Leverington, J. Loh, M. A. McGeoch, L. McRae, A. Minasyan, M. Hernández Morcillo, T. E. E. Oldfield, D. Pauly, S. Quader, C. Revenga, J. R. Sauer, B. Skolnik, D. Spear, D. Stanwell-Smith, S. N. Stuart, A. Symes, M. Tierney, T. D. Tyrrell, J.-C. Vié, and R. Watson. 2010. Global biodiversity: indicators of recent declines. Science 328:1164–8.

Byers, E., K. M. Ponte, and L. T. Alliance. 2005. The conservation easement handbook. Land Trust Alliance Washington, DC.

Campbell, R. W., N. K. Dawe, I. McTaggart-Cowan, J. M. Cooper, G. W. Kaiser, M. C. E. McNall, and G. E. J. Smith. 1997. The Birds of British Columbia: Passerines (Flycatchers Through Vireos). UBC Press.

Carwardine, J., K. A. Wilson, S. A. Hajkowicz, R. J. Smith, C. J. Klein, M. Watts, and H. P. Possingham. 2010. Conservation Planning when Costs Are Uncertain. Conservation Biology 24:1529–1537.

Carwardine, J., K. A. Wilson, M. Watts, A. Etter, C. J. Klein, and H. P. Possingham. 2008. Avoiding costly conservation mistakes: the importance of defining actions and costs in spatial priority setting. PLoS One 3:2586.

Chan, K. M. A., L. Hoshizaki, and B. Klinkenberg. 2011. Ecosystem Services in Conservation Planning: Targeted Benefits vs. Co-Benefits or Costs? PLoS ONE 6:e24378.

Chan, K. M. A., R. M. Pringle, J. Ranganathan, C. L. Boggs, Y. L. Chan, P. R. Ehrlich, P. K. Haff, N. E. Heller, K. Al-Khafaji, and D. P. Macmynowski. 2007. When agendas collide: human welfare and biological conservation. Conservation Biology 21:59–68.

Chan, K. M. A., M. R. Shaw, D. R. Cameron, E. C. Underwood, and G. C. Daily. 2006. Conservation Planning for Ecosystem Services. PLoS Biol 4:e379.

Chazdon, R. L. 2008. Beyond Deforestation: Restoring Forests and Ecosystem Services on Degraded Lands Science 320 :1458–1460.

Chazdon, R. L., C. A. Peres, D. Dent, D. Seil, A. E. Lugo, D. Lamb, N. E. Stork, and S. E. Miller. 2009. The Potential for Species Conservation in Tropical Secondary Forests. Conservation Biology 23:1406–1417.

Climate Commission. 2013. The Critical Decade: Global Action Building On Climate Change (full report as pdf). Page 40.

Condit, R., W. D. Robinson, R. Ibáñez, S. Aguilar, A. Sanjur, R. Martínez, R. F. Stallard, T. García, G. R. Angehr, L. Petit, and others. 2001. The Status of the Panama Canal Watershed and Its Biodiversity at the Beginning of the 21st Century Long-term ecological studies reveal a diverse flora and fauna near the Panama Canal, harbored within a corridor of forest stretching from the Caribbean to t. BioScience 51:389–398.

Convention on Biological Diversity. 2010. Strategic plan for biodiversity 2011–2020 and the Aichi targets. Secretariat of the Convention on Biological Diversity, Montreal.

Crossman, N. D., B. A. Bryan, and D. M. Summers. 2011. Carbon Payments and Low-Cost Conservation. Conservation Biology 25:835–845.

Daily, G. C., T. Söderqvist, S. Aniyar, K. Arrow, P. Dasgupta, P. R. Ehrlich, C. Folke, A. M. Jansson, B. O. Jansson, N. Kautsky, and Et Al. 2000. The value of nature and the nature of value. Science 289:395–396.

Devine, W. D., C. A. Harrington, and D. H. Peter. 2007. Oak woodland restoration: understory response to removal of encroaching conifers. Ecological Restoration 25:247–255.

DeWan, A. A., P. J. Sullivan, A. J. Lembo, C. R. Smith, J. C. Maerz, J. P. Lassoie, and M. E. Richmond. 2009. Using occupancy models of forest breeding birds to prioritize conservation planning. Biological Conservation 142:982–991.

Dobson, A. P., J. P. Rodriguez, W. M. Roberts, and D. S. Wilcove. 1997. Geographic distribution of endangered species in the United States. Science 275:550–553.

Dormann, C. F., J. M. McPherson, M. B. Araújo, R. Bivand, J. Bolliger, G. Carl, R. G. Davies, A. Hirzel, W. Jetz, W. Daniel Kissling, I. Kühn, R. Ohlemüller, P. R. Peres-Neto, B. Reineking, B. Schröder, F. M. Schurr, and R. Wilson. 2007. Methods to account for spatial autocorrelation in the analysis of species distributional data: a review. Ecography 30:609–628.

Douglass, L. L., H. P. Possingham, J. Carwardine, C. J. Klein, S. H. Roxburgh, J. Russell-Smith, and K. A. Wilson. 2011. The Effect of Carbon Credits on Savanna Land Management and Priorities for Biodiversity Conservation. PLoS ONE 6:e23843.

Drechsler, M., F. Wätzold, K. Johst, H. Bergmann, and J. Settele. 2007. A model-based approach for designing cost-effective compensation payments for conservation of endangered species in real landscapes. Biological conservation 140:174–186.

Dunwiddie, P. W., and J. D. Bakker. 2011. The future of restoration and management of prairie-oak ecosystems in the Pacific Northwest. Northwest Science 85:83–92.

Egoh, B., B. Reyers, M. Rouget, M. Bode, and D. Richardson. 2009. Spatial congruence between biodiversity and ecosystem services in South Africa. Biological Conservation 142:553–562.

Egoh, B., M. Rouget, B. Reyers, A. Knight, R. Cowling, A. Vanjaarsveld, and A. Welz. 2007. Integrating ecosystem services into conservation assessments: A review. Ecological Economics 63:714–721.

Ehrlich, P. R., and R. M. Pringle. 2008. Where does biodiversity go from here? A grim business-as-usual forecast and a hopeful portfolio of partial solutions. Proceedings of the National Academy of Sciences 105:11579–11586.

Elith, J., C. H. Graham, R. P. Anderson, M. Dud'ik, S. Ferrier, A. Guisan, R. J. Hijmans, F. Huettmann, J. R. Leathwick, A. Lehmann, J. Li, L. G. Lohmann, B. A. Loiselle, G. Manion, C. Moritz, M. Nakamura, Y. Nakazawa, J. McC. M. Overton, A. T. Peterson, and S. J. Phillips. 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29:129–151.

Elith, J., and J. R. Leathwick. 2009. Species distribution models: ecological explanation and prediction across space and time. Annual Review of Ecology, Evolution, and Systematics 40:677–697.

Ellis, E. C. 2011. Anthropogenic transformation of the terrestrial biosphere. Philosophical transactions. Series A, Mathematical, physical, and engineering sciences 369:1010–35.

ESRI. 2009. ArcGIS 9.3. Economic and Social Reserach Institute Inc., Redlands, CA. http://www.esri.com/.

ESRI. 2010. ArcGIS 10. Economic and Social Reserach Institute Inc., Redlands, CA. http://www.esri.com/.

ESRI. 2012. ArcGIS 10.1 Economic and Social Reserach Institute Inc., Redlands, CA. http://www.esri.com/.

Estes, J. A., J. Terborgh, J. S. Brashares, M. E. Power, J. Berger, W. J. Bond, S. R. Carpenter, T. E. Essington, R. D. Holt, J. B. C. Jackson, R. J. Marquis, L. Oksanen, T. Oksanen, R. T. Paine, E. K. Pikitch, W. J. Ripple, S. A. Sandin, M. Scheffer, T. W. Schoener, J. B. Shurin, A. R. E. Sinclair, M. E. Soulé, R. Virtanen, and D. A. Wardle. 2011. Trophic downgrading of planet Earth. Science 333:301–6.

Evans, M. C., J. Carwardine, R. J. Fensham, D. W. Butler, K. A. Wilson, H. P. Possingham, and T. G. Martin. 2014. The economics of biodiverse carbon farming: regrowth management as a viable mechanism for restoring deforested agricultural landscapes. Global Environmental Change in press.

Fairbanks, D. 2001. Species and environment representation: selecting reserves for the retention of avian diversity in KwaZulu-Natal, South Africa. Biological Conservation 98:365–379.

Ferraro, P. J. 2003. Assigning priority to environmental policy interventions in a heterogeneous world. Journal of Policy Analysis and Management 22:27–43.

Ferrier, S., G. Manion, J. Elith, and K. Richardson. 2007. Using generalized dissimilarity modelling to analyse and predict patterns of beta diversity in regional biodiversity assessment. Diversity and Distributions 13:252–264.

Fielding, A. H., and J. F. Bell. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24:38–49.

Fishburn, I. S., P. Kareiva, K. J. Gaston, and P. R. Armsworth. 2009. The Growth of Easements as a Conservation Tool. PLoS ONE 4:6.

Fiske, I. J., and R. B. Chandler. 2011. unmarked : An R Package for Fitting Hierarchical Models of Wildlife Occurrence and Abundance. Journal Of Statistical Software 43:128–129.

Fuller, R. A., E. McDonald-Madden, K. A. Wilson, J. Carwardine, H. S. Grantham, J. E. M. Watson, C. J. Klein, D. C. Green, and H. P. Possingham. 2010. Replacing underperforming protected areas achieves better conservation outcomes. Nature 466:365–367.

Game, E. T., J. A. Fitzsimons, G. Lipsett-Moore, and E. McDonald-Madden. 2013. Subjective risk assessment for planning conservation projects. Environmental Research Letters 8:45027–45038.

Gardner, T. A., J. Barlow, R. Chazdon, R. M. Ewers, C. A. Harvey, C. A. Peres, and N. S. Sodhi. 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecology Letters 12:561–582.

Goldman, R. L., H. Tallis, P. Kareiva, and G. C. Daily. 2008. Field evidence that ecosystem service projects support biodiversity and diversify options. Proceedings of the National Academy of Sciences of the United States of America 105:9445–9448.

Gonzales, E. K., and P. Arcese. 2008. Herbivory more limiting than competition on early and established native plants in an invaded meadow. Ecology 89:3282–3289.

Gonzalez, A., J. H. Lawton, F. S. Gilbert, T. M. Blackburn, and I. Evans-Freke. 1998. Metapopulation dynamics, abundance, and distribution in a microecosystem. Science 281:2045–2047.

Gordon, A., W. T. Langford, M. D. White, J. A. Todd, and L. Bastin. 2011. Modelling trade offs between public and private conservation policies. Biological Conservation 144:558–566.

Graham, C. H., S. Ferrier, F. Huettman, C. Moritz, and A. T. Peterson. 2004. New developments in museum-based informatics and applications in biodiversity analysis. Trends in Ecology & Evolution 19:497–503.

Groom, J. D., and T. C. Grubb. 2002. Bird species associated with riparian woodland in fragmented, temperate-deciduous forest. Conservation Biology 16:832–836.

Gu, W., and R. K. Swihart. 2004. Absent or undetected? Effects of non-detection of species occurrence on wildlife-habitat models. Biological Conservation 116:195–203.

Guisan, A., and W. Thuiller. 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters 8:993–1009.

Hagen, A. 2003. Fuzzy set approach to assessing similarity of categorical maps. International Journal of Geographical Information Science 17:235–249.

Hastie, T., and W. Fithian. 2013. Inference from presence-only data; the ongoing controversy. Ecography 36:864–867.

Hayes, J. P., J. M. Weikel, and M. M. P. Huso. 2003. Response of birds to thinning youg douglas-fir forests. Ecological Applications 13:1222–1232.

Hazlitt, S. L., T. G. Martin, L. Sampson, and P. Arcese. 2010. The effects of including marine ecological values in terrestrial reserve planning for a forest-nesting seabird. Biological Conservation 143:1299–1303.

Heikkinen, R. K., M. Luoto, R. Virkkala, and K. Rainio. 2004. Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in an agricultural–forest mosaic. Journal of Applied Ecology 41:824–835.

Hein, C. D., S. B. Castleberry, and K. V Miller. 2009. Site-occupancy of bats in relation to forested corridors. Forest Ecology and Management 257:1200–1207.

Hijmans, R. J., and C. H. Graham. 2006. The ability of climate envelope models to predict the effect of climate change on species distributions. Global Change Biology 12:2272–2281.

Jewell, K. J., and P. Arcese. 2008. Consequences of parasite invasion and land use on the spatial dynamics of host populations. Journal of Applied Ecology 45:1180–1188.

Jewell, K. J., P. Arcese, and S. Gergel. 2007. Robust predictions of species distribution: Spatial habitat models for a brood parasite. Biological Conservation 140:259–272.

Jönsson, M. T., S. Fraver, and B. G. Jonsson. 2009. Forest history and the development of old-growth characteristics in fragmented boreal forests. Journal of Vegetation Science 20:91–106.

Kavanagh, R. P., and M. A. Stanton. 2005. Vertebrate species assemblages and species sensitivity to logging in the forests of north-eastern New South Wales. Forest Ecology and Management 209:309–341.

Kimmins, J. P., D. Mailly, and B. Seely. 1999. Modelling forest ecosystem net primary production: the hybrid simulation approach used in FORECAST. Ecological Modelling 122:195–224.

Kininmonth, S. J., G. De'ath, and H. P. Possingham. 2010. Graph theoretic topology of the Great but small Barrier Reef world. Theoretical Ecology 3:75–88.

Klimek, S., A. gen Kemmermann, H.-H. Steinmann, J. Freese, and J. Isselstein. 2008. Rewarding farmers for delivering vascular plant diversity in managed grasslands: A transdisciplinary case-study approach. Biological Conservation 141:2888–2897.

Knight, A. ., R. M. Cowling, M. Difford, and B. M. Campbell. 2010. Mapping Human and Social Dimensions of Conservation Opportunity for the Scheduling of Conservation Action on Private Land. Conservation Biology 24:1348–1358.

Knight, A. T., R. M. Cowling, H. P. Possingham, and K. A. Wilson. 2009. From theory to practice: Designing and situating spatial prioritization approaches to better implement conservation action. Pages 249–259 *in* A. Moilanen, K. A. Wilson, and H. P. Possingham, editors. Spatial conservation prioritization. Quantitative methods & computational tools. Oxford University Press.

Knight, A. T., H. S. Grantham, R. J. Smith, G. K. McGregor, H. P. Possingham, and R. M. Cowling. 2011. Land managers' willingness-to-sell defines conservation opportunity for protected area expansion. Biological Conservation 144:2623–2630.

Lamb, D., P. D. Erskine, and J. A. Parrotta. 2005. Restoration of degraded tropical forest landscapes. Science 310:1628–1632.

Langholz, J. A., and J. P. Lassoie. 2001. Perils and Promise of Privately Owned Protected Areas. BioScience 51:1079–1085.

Lawler, J. J., and T. C. Edwards. 2006. A Variance-Decomposition Approach to Investigating Multiscale Habitat Associations. The Condor 108:47–58.

Lea, T. 2006. Historical Garry oak ecosystems of Vancouver Island, British Columbia, pre-European contact to the present. Davidsonia 17:34–50.

Lees, A. C., and C. A. Peres. 2009. Gap-crossing movements predict species occupancy in Amazonian forest fragments. Oikos 118:280–290.

Lindenmayer, D., R. J. Hobbs, R. Montague-Drake, J. Alexandra, A. Bennett, M. Burgman, P. Cale, A. Calhoun, V. Cramer, P. Cullen, D. Driscoll, L. Fahrig, J. Fischer, J. Franklin, Y. Haila, M. Hunter, P. Gibbons, S. Lake, G. Luck, C. MacGregor, S. McIntyre, R. Mac Nally, A. Manning, J. Miller, H. Mooney, R. Noss, H. Possingham, D. Saunders, F. Schmiegelow, M. Scott, D. Simberloff, T. Sisk, G. Tabor, B. Walker, J. Wiens, J. Woinarski, and E. Zavaleta. 2008. A checklist for ecological management of landscapes for conservation. Ecology Letters 11:78–91.

Linke, S., E. Turak, and J. Nel. 2011. Freshwater conservation planning: the case for systematic approaches. Freshwater Biology 56:6–20.

Lo, A. Y. 2013. Carbon trading in a socialist market economy: Can China make a difference? Ecological Economics 87:72–74.

Luck, G. W. 2007. A review of the relationships between human population density and biodiversity. Biological Reviews 82:607–645.

Luyssaert, S., E.-D. Schulze, A. Borner, A. Knohl, D. Hessenmoller, B. E. Law, P. Ciais, and J. Grace. 2008. Old-growth forests as global carbon sinks. Nature 455:213–215.

MacDougall, A. S., B. R. Beckwith, and C. Y. Maslovat. 2004. Defining conservation strategies with historical perspectives: a case study from a degraded oak grassland ecosystem. Conservation Biology 18:455–465.

MacKenzie, D. I., and L. L. Bailey. 2004. Assessing the fit of site-occupancy models. Journal of Agricultural, Biological, and Environmental Statistics 9:300–318.

MacKenzie, D. I., J. D. Nichols, J. E. Hines, M. G. Knutson, and A. B. Franklin. 2003. Estimating Site Occupancy, Colonization, and Local Extinction When a Species Is Detected Imperfectly. Ecology 84:2200–2207.

Mackenzie, D. I., J. D. Nichols, G. B. Lachman, S. J. Droege, J. A. Royle, and C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248–2255.

Mackenzie, D. I., J. D. Nichols, J. A. Royle, K. H. Pollock, L. L. Bailey, and J. E. Hines. 2006. Occupancy estimation and modeling: inferring patterns and dynamics of species. Elsevier, Burlington, MA.

Mackenzie, D. I., and J. A. Royle. 2005. Designing occupancy studies: general advice and allocating survey effort. Journal of Applied Ecology 42:1105–1114.

Magoun, A. J., J. C. Ray, D. S. Johnson, P. Valkenburg, F. N. Dawson, and J. Bowman. 2007. Modeling Wolverine Occurrence Using Aerial Surveys of Tracks in Snow. The Journal of Wildlife Management 71:2221–2229.

Main, M. B., F. M. Roka, and R. F. Noss. 1999. Evaluating Costs of Conservation. Conservation Biology 13:1262–1272.

Makino, A., M. Beger, C. J. Klein, S. D. Jupiter, and H. P. Possingham. 2013. Integrated planning for land--sea ecosystem connectivity to protect coral reefs. Biological Conservation 165:35–42.

Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. Nature 405:243–53.

Marsh, C. J., O. T. Lewis, I. Said, and R. M. Ewers. 2010. Community-level diversity modelling of birds and butterflies on Anjouan, Comoro Islands. Biological Conservation 143:1364–1374.

Martin, T. G., P. Arcese, and N. Scheerder. 2011. Browsing down our natural heritage: Deer impacts on vegetation structure and songbird populations across an island archipelago. Biological Conservation 144:459–469.

Martin, T. G., P. M. Kuhnert, K. Mengersen, and H. P. Possingham. 2005a. The power of expert opinion in ecological models using Bayesian methods: impact of grazing on birds. Ecological Applications 15:266–280.

Martin, T. G., B. A. Wintle, J. R. Rhodes, P. M. Kuhnert, S. A. Field, S. J. Low-Choy, A. J. Tyre, and H. P. Possingham. 2005b. Zero tolerance ecology: improving ecological inference by modelling the source of zero observations. Ecology Letters 8:1235–1246.

Matthews, S. N., L. R. Iverson, A. M. Prasad, and M. P. Peters. 2011. Changes in potential habitat of 147 North American breeding bird species in response to redistribution of trees and climate following predicted climate change. Ecography 34:933–945.

McCune, J. L., M. G. Pellatt, and M. Vellend. 2013. Multidisciplinary synthesis of long-term human--ecosystem interactions: A perspective from the Garry oak ecosystem of British Columbia. Biological Conservation 166:293–300.

McRae, B. H., B. G. Dickson, T. H. Keitt, and V. B. Shah. 2008. Using circuit theory to model connectivity in ecology, evolution, and conservation. Ecology 89:2712–2724.

Meidinger, D., and J. Pojar. 1991. Ecosystems of British Columbia. British Columbia Ministry of Forests, Victoria, BC.

Meidinger, D., R. Trowbridge, A. Macadam, and C. Tolkamp. 1998. Field Manual for Describing Terrestrial Ecosystems. Crown Publications Inc., Victoria, Canada.

Merenlender, A. M., L. Huntsinger, G. Guthey, and S. K. Fairfax. 2004. Land trusts and conservation easements: Who is conserving what for whom? Conservation Biology 18:65–76.

MES. 2008. Terrestrial Ecosystem Mapping of the Coastal Douglas-Fir Biogeoclimatic Zone. Madrone Environmental Services LTD., Duncan, BC. Mandrone Environmental Services LTD., Duncan, BC.

Millenium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Synthesis. Page 137pp. World Resources Institute Washington DC. Island Press, Washington, DC.

Ministry of Environment, B. C. 2011. Protocol for the Creation of Forest Carbon Offsets in British Columbia. British Columbia Ministry of Environment, Victoria.

Moilanen, A., and A. Arponen. 2011. Setting conservation targets under budgetary constraints. Biological Conservation 144:650–653.

Moilanen, A., J. Leathwick, and J. Elith. 2008. A method for spatial freshwater conservation prioritization. Freshwater Biology 53:577–592.

Mokany, K., T. D. Harwood, J. M. Overton, G. M. Barker, and S. Ferrier. 2011. Combining α - and β -diversity models to fill gaps in our knowledge of biodiversity. Ecology Letters 14:1043–1051.

Moon, K., and C. Cocklin. 2011. Participation in biodiversity conservation: motivations and barriers of Australian landholders. Journal of Rural Studies 27:331–342.

Moore, J., A. Balmford, T. Allnutt, and N. Burgess. 2004. Integrating costs into conservation planning across Africa. Biological Conservation 117:343–350.

Mosseler, A., I. Thompson, and B. A. Pendrel. 2003. Overview of old-growth forests in Canada from a science perspective. Environmental Reviews 11:S1–S7.

Müller, R., C. Nowicki, W. Barthlott, and P. L. Ibisch. 2003. Biodiversity and endemism mapping as a tool for regional conservation planning – case study of the Pleurothallidinae (Orchidaceae) of the Andean rain forests in Bolivia. Biodiversity and Conservation 12:2005–2024.

Murcia, C. 1995. Edge effects in fragmented forests: implications for conservation. Trends in Ecology & Evolution 10:58–62.

Nagelkerke, N. J. D. 1991. A note on a general definition of the coefficient of determination. Biometrika 78:691–692.

Naidoo, R., A. Balmford, R. Costanza, B. Fisher, R. E. Green, B. Lehner, T. R. Malcolm, and T. H. Ricketts. 2008. Global mapping of ecosystem services and conservation priorities. Proceedings of the National Academy of Sciences of the United States of America 105:9495–9500.

Naidoo, R., A. Balmford, P. J. Ferraro, S. Polasky, T. H. Ricketts, and M. Rouget. 2006. Integrating economic costs into conservation planning. Trends in ecology & evolution 21:681–7.

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, Dr. Cameron, K. M. A. Chan, G. C. Daily, J. Goldstein, P. M. Kareiva, E. Lonsdorf, R. Naidoo, T. H. Ricketts, and Mr. Shaw. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7:4–11.

Newell, R. G., W. A. Pizer, and D. Raimi. 2013. Carbon markets 15 years after kyoto: Lessons learned, new challenges. The Journal of Economic Perspectives 27:123–146.

Nicholson, E., M. I. Westphal, K. Frank, W. A. Rochester, R. L. Pressey, D. B. Lindenmayer, and H. P. Possingham. 2006. A new method for conservation planning for the persistence of multiple species. Ecology Letters 9:1049–1060.

Noss, R. F., and A. Y. Cooperrider. 1994. Saving nature's legacy: protecting and restoring biodiversity. (R. F. Noss and A. Y. Cooperrider, Eds.). Island Press, Washington D.C., USA.

Noss, R. F., A. P. Dobson, R. Baldwin, P. Beier, C. R. Davis, D. A. Dellasala, J. Francis, H. Locke, K. Nowack, R. Lopez, C. Reining, S. C. Trombulak, and G. Tabor. 2012. Bolder Thinking for Conservation. Conservation Biology 26:1–4.

Parker, D. P. 2004. Land trusts and the choice to conserve land with full ownership or conservation easements. Natural Resources Journal 44:483–518.

Pascual-Hortal, L., and S. Saura. 2006. Comparison and development of new graph-based landscape connectivity indices: towards the priorization of habitat patches and corridors for conservation. Landscape Ecology 21:959–967.

Pasquini, L., J. A. Fitzsimons, S. Cowell, K. Brandon, and G. Wescott. 2011. The establishment of large private nature reserves by conservation NGOs: key factors for successful implementation. Oryx 45:373–380.

PCT. 2013. Pacific Carbon Trust Carbon Offset Portfolio. http://pacificcarbontrust.com/assets/Uploads/Pricing/PCTpricing2009-2012.pdf, date accessed: 2013-08-10.

Pebesma, E. J. 2004. Multivariable geostatistics in S: the gstat package. Computers & Geosciences 30:683–691.

Pellatt, M. G., S. J. Goring, K. M. Bodtker, and A. J. Cannon. 2012. Using a down-scaled bioclimate envelope model to determine long-term temporal connectivity of Garry oak (Quercus garryana) habitat in western North America: implications for protected area planning. Environmental management 49:802–815.

Pence, G. Q. K., M. A. Botha, and J. K. Turpie. 2003. Evaluating combinations of on-and off-reserve conservation strategies for the Agulhas Plain, South Africa: a financial perspective. Biological Conservation 112:253–273.

Perhans, K., C. Kindstrand, M. Boman, L. B. Djupström, L. Gustafsson, L. Mattsson, L. M. Schroeder, J. A. N. Weslien, and S. Wikberg. 2008. Conservation Goals and the Relative Importance of Costs and Benefits in Reserve Selection. Conservation Biology 22:1331–1339.

Phelps, J., E. L. Webb, and W. M. Adams. 2012. Biodiversity co-benefits of policies to reduce forest-carbon emissions. Nature Climate Change 2:497–503.

Polasky, S., J. D. Camm, and B. Garber-Yonts. 2001. Selecting Biological Reserves Cost-Effectively: An Application to Terrestrial Vertebrate Conservation in Oregon. Land Economics 77:68–78.

Possingham, H., I. Ball, and S. Andelman. 2000. Mathematical Methods for Identifying Representative Reserve Networks. Pages 291–305 *in* S. Ferson and M. Burgman, editors. Quantitative methods for conservation biology. Springer-Verlag, New York, NY.

Pregitzer, K. S., and E. S. Euskirchen. 2004. Carbon cycling and storage in world forests: biome patterns related to forest age. Global Change Biology 10:2052–2077.

Pressey, R. L. 1994. Ad hoc reservations: forward or backward steps in developing representative reserve systems? Conservation biology 8:662–668.

Pressey, R. L., M. Cabeza, M. E. Watts, R. M. Cowling, and K. A. Wilson. 2007. Conservation planning in a changing world. Trends in Ecology & Evolution 22:583–592.

R Core Team. 2013. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

R Development Core Team. 2010. R: A language and environment for statistical computing, http://www.r-project.org. R Foundation for Statistical Computing, Vienna, Austria.

R Development Core Team. 2012. R: A language and environment for statistical computing 2.15.2, http://www.r-project.org. R Foundation for Statistical Computing, Vienna, Austria.

Rayfield, B., M.-J. Fortin, and A. Fall. 2011. Connectivity for conservation: a framework to classify network measures. Ecology 92:847–858.

Rissman, A. R. 2013. Rethinking property rights: comparative analysis of conservation easements for wildlife conservation. Environmental Conservation 40:222–230.

Rissman, A. R., and V. Butsic. 2011. Land trust defense and enforcement of conserved areas. Conservation Letters 4:31–37.

Roberge, J.-M., and P. E. R. Angelstam. 2004. Usefulness of the Umbrella Species Concept as a Conservation Tool. Conservation Biology 18:76–85.

Royle, J. A. 2004. N-Mixture Models for Estimating Population Size from Spatially Replicated Counts. Biometrics 60:108–115.

Rubinoff, D. 2001. Evaluating the California Gnatcatcher as an Umbrella Species for Conservation of Southern California Coastal Sage Scrub. Conservation Biology 15:1374–1383.

Ruckdeschel, P., M. Kohl, T. Stabla, and F. Camphausen. 2006. S4 Classes for Distributions. R News 6:2–6.

Sallabanks, R., R. A. Riggs, L. E. Cobb, and S. W. Dodson. 2006. Bird-Habitat Relationships in Grand Fir Forests of the Blue Mountains, Oregon. Forest Science 52:489–502.

Schuster, R., and P. Arcese. 2013. Using bird species community occurrence to prioritize forests for old growth restoration. Ecography 36:499–507.

Scott, J. M., F. Davis, B. Csuti, R. Noss, B. Butterfield, C. Groves, H. Anderson, S. Caicco, F. D'Erchia, T. C. Edwards Jr., J. Ulliman, and R. G. Wright. 1993. Gap Analysis: A Geographic Approach to Protection of Biological Diversity. Wildlife Monographs 123:3–41.

Seely, B. 2012. Evaluation of Carbon Storage within Forests in the Coastal Douglas Fir Zone (unpiblished technical report). Canadian Wildlife Service.

Seely, B., J. Nelson, R. Wells, B. Peter, M. Meitner, A. Anderson, H. Harshaw, S. Sheppard, F. L. Bunnell, H. Kimmins, and D. Harrison. 2004. The application of a hierarchical, decision-support system to evaluate multi-objective forest management strategies: a case study in northeastern British Columbia, Canada. Forest Ecology and Management 199:283–305.

Sinclair, A. R. E., D. S. Hik, O. J. Schmitz, G. G. E. Scudder, D. H. Turpin, and N. C. Larter. 1995. Biodiversity and the need for habitat renewal. Ecological Applications 5:579–587.

Smith, A. H. 2009. Conservation Easement Violated: What Next-A Discussion of Remedies. Fordham Envtl. L. Rev. 20:597.

Sohl, T. L., K. L. Sayler, M. A. Bouchard, R. R. Reker, A. M. Friesz, S. L. Bennett, B. M. Sleeter, R. R. Sleeter, T. S. Wilson, M. Knuppe, and T. Van Hofwegen. 2013. Spatially explicit modeling of 1992 to 2100 land cover and forest stand age for the conterminous United States. Ecological Applications.

Thomas, C. D., B. J. Anderson, A. Moilanen, F. Eigenbrod, A. Heinemeyer, T. Quaife, D. B. Roy, S. Gillings, P. R. Armsworth, and K. J. Gaston. 2012. Reconciling biodiversity and carbon conservation. Ecology letters.

Turner, N. J. 2014. Ancient Pathways, Ancestral Knowledge: Ethnobotany and Ecological Wisdom of Indigenous Peoples of Northwestern North America. 2 vol. McGill Queen's University Press, Montreal.

Urban, D. L., E. S. Minor, E. A. Treml, and R. S. Schick. 2009. Graph models of habitat mosaics. Ecology letters 12:260–73.

Vellend, M., A. D. Bjorkman, and A. McConchie. 2008. Environmentally biased fragmentation of oak savanna habitat on southeastern Vancouver Island, Canada. Biological Conservation 141:2576–2584.

Venter, O., W. F. Laurance, T. Iwamura, K. A. Wilson, R. A. Fuller, and H. P. Possingham. 2009a. Harnessing carbon payments to protect biodiversity. Science 326:1368.

Venter, O., E. Meijaard, H. Possingham, R. Dennis, D. Sheil, S. Wich, L. Hovani, and K. Wilson. 2009b. Carbon payments as a safeguard for threatened tropical mammals. Conservation Letters 2:123–129.

Visser, H., and T. de Nijs. 2006. The Map Comparison Kit. Environmental Modelling & Software 21:346–358.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human Domination of Earth's Ecosystems Science 277 :494–499.

De Wan, A., P. J. Sullivan, A. J. Lembo, C. R. Smith, J. C. Maerz, J. P. Lassoie, and M. E. Richmond. 2009. Using occupancy models of forest breeding birds to prioritize conservation planning. Biological Conservation 142:982–991.

Wang, Q. 2013. China has the capacity to lead in carbon trading. Nature 493:273.

Wang, T., A. Hamann, D. L. Spittlehouse, and S. N. Aitken. 2006. Development of scale-free climate data for Western Canada for use in resource management. International Journal of Climatology 26:383–397.

Wang, T., A. Hamann, D. L. Spittlehouse, and T. Q. Murdock. 2012. ClimateWNA-High-Resolution Spatial Climate Data for Western North America. Journal of Applied Meteorology & Climatology 51:16–29.

Weir, L. A., J. A. Royle, and P. Nanjappa. 2005. Modeling anuran detection and site occupancy on North American Amphibian Monitoring Program (NAAMP) routes in Maryland. Journal of Herpetology 39:627–639.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46:120–139.

Wiersma, Y. F., and T. D. Nudds. 2009. Efficiency and effectiveness in representative reserve design in Canada: The contribution of existing protected areas. Biological Conservation 142:1639–1646.

Wiersma, Y. F., and D. L. Urban. 2005. Beta Diversity and Nature Reserve System Design in the Yukon, Canada. Conservation Biology 19:1262–1272.

Wilcox, B. A., and D. D. Murphy. 1985. Conservation Strategy: the Effects of Fragmentation on Extinction. American Naturalist 125:879–887.

Wilson, K. A., M. Lulow, J. Burger, Y.-C. Fang, C. Andersen, D. Olson, M. O'Connell, and M. F. McBride. 2011. Optimal restoration: accounting for space, time and uncertainty. Journal of Applied Ecology 48:715–725.

Wilson, K., J. Carwardine, and H. P. Possingham. 2009. Setting conservation priorities. Annals of the New York Academy of Sciences 1162:237–64.

Wilson, K., E. C. Underwood, S. a Morrison, K. R. Klausmeyer, W. W. Murdoch, B. Reyers, G. Wardell-Johnson, P. a Marquet, P. W. Rundel, M. F. McBride, R. L. Pressey, M. Bode, J. M. Hoekstra, S. Andelman, M. Looker, C. Rondinini, P. Kareiva, M. R. Shaw, and H. P. Possingham. 2007. Conserving biodiversity efficiently: what to do, where, and when. PLoS biology 5:e223.

Winter, S. J., K. J. Esler, and M. Kidd. 2005. An index to measure the conservation attitudes of landowners towards Overberg Coastal Renosterveld, a critically endangered vegetation type in the Cape Floral Kingdom, South Africa. Biological Conservation 126:383–394.

Wulder, M. A., J. C. White, M. Cranny, R. J. Hall, J. E. Luther, A. Beaudoin, D. G. Goodenough, and J. A. Dechka. 2008. Monitoring Canada's forests. Part 1: Completion of the EOSD land cover project. Canadian Journal of Remote Sensing 34:549–562.

Wunder, S. 2007. The efficiency of payments for environmental services in tropical conservation. Conservation biology 21:48–58.

Zipkin, E. F., E. H. C. Grant, and W. F. Fagan. 2012. Evaluating the predictive abilities of community occupancy models using AUC while accounting for imperfect detection. Ecological Applications 22:1962–72.

Appendices

Appendix 1:   **Chapter 2 expert elicitation descriptions.**

Introduction for experts:

Thank you for taking the time to assist us with our survey.

We are creating predictive occurrence maps for up to 18 bird species to aid in identifying areas within the Coastal Douglas-fir (CDF) zone of south-eastern Vancouver Island and the Southern Gulf Islands (see Figure) that are likely to support bird communities representing mature and older forest and woodland habitats. Once assembled, these and other maps of native and exotic plants, habitat types and human land use, will be used to identify existing forest and woodland stands likely to contribute positively to the restoration of 'old growth' CDF communities in future. We are especially interested in identifying bird species that, based on expert opinion, are likely to be present in mature or older CDF forest and woodland stands (e.g., stands with large diameter trees, well-vegetated light gaps, and multi-layer canopy structure).

We are requesting your help because detailed information on the dependence of individual species on old forest and woodland habitats are generally unavailable for the coastal Douglas-fir zone. We are therefore reaching out to experienced birders to help us by filling out an online survey requiring perhaps 15-20 minutes of your time.

Habitat dependence questions:

Please read the following descriptions of habitats found in the CDF Biogeoclimatic Zone, then rank the listed bird species for each habitat type using the embedded drop-down menus.

Herbaceous: Early successional or herbaceous communities maintained by environment or disturbance (e.g., flooding, grazing, fire, agriculture); dominated by forbs, graminoids, ferns. Invading or residual shrubs and trees may be present (tree cover < 10%, shrubs < 20%), time since disturbance < 20 yrs via forest succession or non-forested communities maintained in this stage.

Shrub/Herb: Early successional or shrub communities maintained by environment or disturbance; dominated by shrubby vegetation that is <10m tall. Seedlings and advance regeneration may be abundant (tree cover < 10%, shrub cover > 20%), time since disturbance < 40 years via forest succession.

Pole/Sapling: Trees > 10 m tall, typically densely stocked, have overtopped shrub and herbaceous layers; younger stands are vigorous (usually > 10-15 years old); older stagnated stands (up to 100 years old) are also included. Self-thinning and vertical structure not yet evident in the canopy, time since disturbance < 40 years via forest succession and up to 100+ years for dense stagnant stands.

Young Forest: Self-thinning has become evident and the forest canopy has begun to differentiate into distinct layers (dominant, main canopy, and overtopped); vigorous growth and a more open stand than in the Pole/Sapling stage, time since disturbance generally 40-80 years, depending on tree species and ecological conditions.

Mature Forest: Trees established after the last disturbance have matured; a second cycle of shade-tolerant trees may have become established; understories become well developed as the canopy opens up; time since disturbance generally 80-250 years.

Old Forest: Older, structurally complex stands comprised shade-tolerant and regenerating tree species, but often including long-lived, older seral stage trees sometimes dominating the upper canopy. Snags and coarse woody debris in all stages of decomposition, understory of deciduous and regenerating confers typical. Shrub layer well-developed and dense, especially in light gaps. Time since last stand-replacing disturbance generally > 250 years.

Please rank the following bird species in terms of their dependence (low/medium/high) on the before mentioned habitat types in the Coastal Douglas-fir Zone.

(followed by drop-down menus for selecting habitat associations)

Appendix 2:   **Bird model details.**

Table A.2.1: Reclassification scheme for Terrestrial Ecosystem Mapping (TEM) structural stages.

Structural stage	description	forest class
1,2	open	0
3	shrub/herb	0
4	pole/sapling	1
5	young forest	1
6	mature forest	2
7	old forest	2

Table A.2.2: Reclassification scheme for Terrestrial Ecosystem Mapping (TEM) classes.

TEM code	Description	Reclass	TEM code	Description	Reclass	TEM code	Description	Reclass
CS	Cw-Slough sedge	Forest	OR	Oceanspray - Rose	Savannah	Wf52	Sweet gale - Sitka sedge	Shrub
DC	FdPl-Cladina	Forest	QB	Garry Oak - Brome/mixed grasses	Savannah	Wf53	Slender sedge - White beak-rush	Shrub
DF	Fd-Sword fern	Forest	RO	Rock Outcrop	Savannah	Wm04		Shrub
DG	FdBg - Oregon grape	Forest	SC	Cladina - Wallace's selaginella	Savannah	Wm05	Cattail	Shrub
DO	Fd - Oniongrass	Forest	TA	Talus	Savannah	Wm06	Great bulrush	Shrub
DS	FdHw-Salal	Forest	AS	Trembling aspen - Slough sedge	Shrub	Wm50	Sitka sedge - Hemlock -parsley	Shrub
HD	HwCw-Deer fern	Forest	BE	Beach	Shrub	Wm51	Three-way sedge	Shrub
HK	HwFd-Kindbergia	Forest	CD	Act - Red-osier dogwood	Shrub	Ws50	Hardhack (pink spirea) - Sitka sedge	Shrub
RB	Cw-Salmonberry	Forest	CW	Act-Willow	Shrub	Ws51	Sitka willow-Pacific willow-Skunk cabbage	Shrub
RC	CwSs-Skunk cabbage	Forest	Ed01	Tufted hairgrass - Meadow barley	Shrub	Ws52	Red alder - Skunk cabbage	Shrub
RF	Cw-Foamflower	Forest	Ed03	Arctic rush - Alaska plantain	Shrub	BK	Break water	Rural
RK	CwFd - Kindbergia	Forest	Em01	Widgeon-grass	Shrub	CF	Cultivated Field	Rural
RP	Cw - Indian-plum	Forest	Em02	Glasswort - Sea milkwort	Shrub	CO	Cultivated Orchard	Rural
RS	Cw-Sword fern	Forest	Em03	Seashore saltgrass	Shrub	CV	Cultivated Vineyard	Rural
RT	Cw-Black twinberry	Forest	Em05	Lyngbye's sedge	Shrub	GC	Golf Course	Rural
RV	Cw - Vanilla-leaf	Forest	GB	Gravel Bar	Shrub	RW	Rural	Rural
SS	Ss-Salmonberry	Forest	LA	Lake	Shrub	DM	Dam	Urban
AM	Arbutus-Hairy manzanita	Savannah	LS	Pl - Sphagnum	Shrub	ES	Exposed Soil	Urban
BA	Rock Outcrop	Savannah	MU	Mudflat Sediment	Shrub	GP	Gravel Pit	Urban
CB	Sand Clif	Savannah	OW	Shallow Open Water	Shrub	IN	Industrial	Urban
CL	Cliff	Savannah	PD	Pond	Shrub	MI	Mine	Urban
DA	FdPl - Arbutus	Savannah	RA	Nootka Rose - Pacific Crab Apple	Shrub	RN	Railway Surface	Urban
FC	Fescue-Common camas	Savannah	RE	Reservoir	Shrub	RZ	Road Surface	Urban
GO	Garry Oak - Ocean Spray	Savannah	RI	River	Shrub	TZ	Mine Tailings	Urban
LM	Dunegrass - Beach pea	Savannah	Wb50	Labrador tea - Bog-laurel -Peat-moss	Shrub	UR	Urban/ Suburban	Urban
OM	Garry Oak - Moss	Savannah	Wf51	Sitka sedge - Peat-moss	Shrub			

Table A.2.3: Individual species model covariate averages.

Species	brown creeper	chestnut backed chickadee	golden-crowned kinglet	olive-sided flycatcher	pacific wren	pacific-slope flycatcher	pine siskin	purple finch	red-breasted nuthatch	townsend's warbler	Wilson's warbler	yellow-rumped warbler
Intercept	2.16	9.20	1.57	3.42	0.46	11.02	14.34	2.84	1.85	1.81	1.35	4.49
near_road	-	-	-0.05	-	0.06	-8.25	-	-	-	-2.04	-	-
near_frshw	-0.76	-	-	-	-	13.65	-	-	-	1.46	2.45	-2.84
near_saltw	-	1.53	-	-1.49	-	-	-0.08	-	-	-	-	-
URB_1KM	-	-	-	-4.59	-	-	-	-	-	-	-	-0.19
RUR_1KM	-	-1.99	-	-	-	-	-0.09	-	-	-	-3.63	-
FOR0_1KM	-	-	-	0.64	-	-	-	-	-	-	-	-
FOR1_1KM	-	-1.91	-	0.03	0.14	-4.79	-	-0.02	-	-0.44	1.25	-
FOR2_1KM	-1.25	-	-	-	-0.05	-	-	-	-	-	-	-2.41
SAV_1KM	-	0.13	-	-	-	-	-	-	-	0.28	-	-
SHR_1KM	-	-	-	-	-	8.17	-	-	-	-	-	-
URB_100	-	-1.23	0.01	-	-	-	-	-	-9.59	-	-	-0.20
RUR_100	-	-	-	-	-	-	-	-	-	-	0.11	-1.78
FOR0_100	-	0.61	-	-0.13	-	-	-	-	0.26	-	-0.52	-
FOR1_100	-	-	-0.67	0.63	-	-	-	0.01	1.72	-	-	-
FOR2_100	-	3.09	-	-2.48	-0.10	-	-	-	2.95	-	-	-0.03
SAV_100	-	0.40	-	-	0.07	-	-	-	-	0.07	-0.41	-
SHR_100	-	-	-	-	-	-	-1.42	-	-0.63	-	-	-
Near_Urb	-0.04	-1.17	-0.49	0.02	-	-	-	-	-	-	-2.56	-
CR_CL0	-	-4.20	-1.91	-	-	-	-	-0.21	-	-	0.13	-
CR_CL_2	3.51	-	-	1.55	1.69	-	11.15	-	-	1.73	-	-
CR_CL_3	-	-	-0.02	-	-	-0.06	-	-	-	-	-0.35	-
BRDLF	-	-	-	-	-	-0.89	-	-	-	-	6.34	-
OF_100	1.05	-	1.54	2.34	-	-	-	0.13	2.15	-	1.68	2.80
OF_1K	-	-	-	0.59	-	-3.58	-	2.06	-	-	0.07	2.62
Is_sie	-	-	-	1.21	-	3.38	-1.91	-	-	-	-	-
rd_up_100	-0.25	-	-	-	-0.01	-	-	0.21	-	-	1.58	-
rd_p_100	-	0.34	-	-	-	-	-	-0.02	-	-	-	-
rd_up_1k	-1.22	3.79	-	-	-0.54	2.91	1.04	-0.13	-	-	-	-
rd_p_1k	-	-	-	2.82	-	-6.88	-	-0.06	-	-	-	-
UTM_E	-	-	-	-	-0.07	-	-	0.62	-	-1.08	-	-
UTM_N	-	-	-	-	0.07	-	-	-0.25	-	0.10	-	-
UTM_E_UTM_N	-	-	-	-	-	-	-	-1.84	-	1.73	-	-
UTM_E2	-	-	-	-	-0.07	-	-	0.91	-	-0.53	-	-
UTM_N2	-	-	-	-	0.07	-	-	-0.25	-	0.25	-	-

Figure A. 2.1: Additional occupancy maps for 6 species: a) chestnut backed chickadee; b) pine siskin; c) townsend's warbler; d) pacific-slope flycatcher; e) pacific wren and f) purple finch.
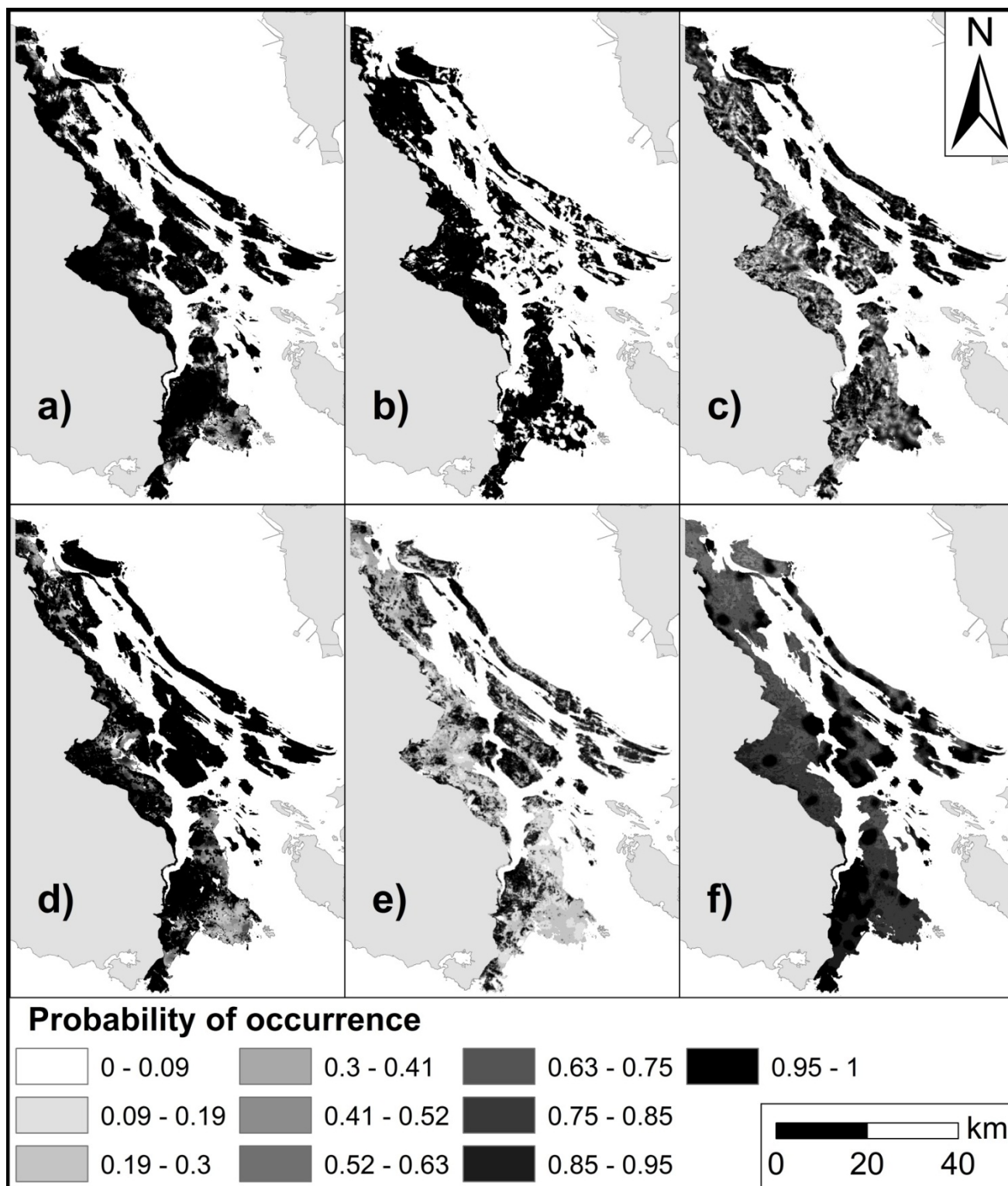
Appendix 3: **Stepwise model selection code.**

```
# Example use of f.AICc.occu.sig function
covar <- read.csv("sitecovar.csv")
surcov <- read.csv("survcovar.csv")
var<-as.vector(c("X1","X2","X3","X4"))

y<- read.csv("y.csv")
AMGOUMF<- unmarkedFrameOccu(y,siteCovs=covar, obsCov=surcov)
AMGO<- occu(~ + XS1 + XS2 + XS3 ~ 1, AMGOUMF)
AMGOr<- f.AICc.occu.sig(start.model=AMGO, blocks=var,max.iter=30, AICcut=1)



f.AICc.occu.sig <- function(start.model, blocks, max.iter=NULL, detocc = 1, AICcut = 1,
print.log=TRUE){
# f.AICc.occu.sig: a function for "stepwise" regression using occupancy models of package unmarked
# start.model: initial model e.g. occu(~1~1, UMF)
# detocc: if set to 1 (default) runs the function on the occupancy side; 2 does the detectability
# Parts based on Forward.lmer by Rense Nieuwenhuis (http://www.rensenieuwenhuis.nl/r-sessions-
32/)
# with some additions by for Nick Isaac
# Author: Richard Schuster (mail@richard-schuster.com)
# 08 October 2012

  modlst <- c(start.model)
  x <- 2
  if (detocc == 1) coeff <- length(start.model@estimates@estimates$state@estimates)
  else coeff <- length(start.model@estimates@estimates$det@estimates)

  best <- FALSE
  model.basis <- start.model
  keep <- list(start.model)
  AICmin <- AIClst <- AICc(start.model)
  #cutoff for when to exclude values
  cutoff <- 20

# Maximum number of iterations cannot exceed number of blocks, but this is also the default
if (is.null(max.iter) | max.iter > length(blocks)) max.iter <- length(blocks)

# Setting up the outer loop
for(ii in 1:max.iter){
    models <- list()
    coeff <- coeff + 1
    cnt <- 1
    for (jj in 1:length(keep)) {
     # Iteratively updating the model with addition of one block of variable(s)
     for(kk in 1:length(blocks)) {
```

```r
    if (detocc == 1) form <- as.formula(paste("~. ~. + ", blocks[kk]))
    else form <- as.formula(paste("~. + ", blocks[kk], "~. "))

    if (class(dummy <- try(update(keep[[jj]], form))) == "unmarkedFitOccu") {
      flag <- 0

      #check if there is any NAN's in the SE's
      if (detocc == 1) {
        for(dd in 1:length(sqrt(diag (vcov(dummy@estimates@estimates$state))))) {
          if(diag (vcov(dummy@estimates@estimates$state))[[dd]] < 0
             || sqrt(diag (vcov(dummy@estimates@estimates$state))[[dd]]) > cutoff)
            flag <- 1
            break
        }
      }
      else {
        for(dd in 1:length(sqrt(diag (vcov(dummy@estimates@estimates$det))))) {
          if(diag (vcov(dummy@estimates@estimates$det))[[dd]] < 0
             || sqrt(diag (vcov(dummy@estimates@estimates$det))[[dd]]) > cutoff)
            flag <- 1
            break
        }
      }
      if (flag == 0) {
        for (bb in 1:length(AIClst)) {
          if (round(AICc(dummy),digits=6) == round(AIClst[bb], digits=6)) {
            flag <- 1
            break
          }
        }
      }

    }
    else {flag <- 1}

    if (flag == 0) {
        models[[cnt]] <- dummy
        modlst[[x]] <- models[[cnt]]
        AIClst <- c(AIClst, AICc(models[[cnt]]))
        x <- x + 1
        cnt <- cnt + 1
    }
  }
}

if(length(LL <- unlist(lapply(models, function(x) {AICc(x)}))) == 0) break
keep <- list()
```

```
  k = 1
  cont <- 0
  #check for improvement in AIC, if none stop loop
  for (mm in order(LL, decreasing=FALSE)) {

   if (LL[mm]< AICmin + AICcut) {
    if (detocc == 1) {
     if (length(models[[mm]]@estimates@estimates$state@estimates) == coeff) {
      keep[[k]]<- models[[mm]]
      k <- k + 1
      if (LL[mm]<AICmin) {
       AICmin <- LL[mm]
       cont <- 1
      }
     }
    }
    else {
     if (length(models[[mm]]@estimates@estimates$det@estimates) == coeff) {
      keep[[k]]<- models[[mm]]
      k <- k + 1
      if (LL[mm]<AICmin) {
       AICmin <- LL[mm]
       cont <- 1
      }
     }
    }
   }
   else break
  }

  if (length(keep) == 0) break
 }
## Create Model List
fitlst <- fitList(fits = modlst)
modsel <- modSel(fitlst,nullmod=NULL)

## Return the gathered output
return(list(model=model.basis, modlst=modlst, fitlst=fitlst, modsel=modsel))
}
```

Appendix 4: **Extended bird models including Bayesian spatial autocorrelation.**

Following are the details of my biodiversity models as they have been presented in Schuster and Arcese (2013), plus some modifications to reflect the changes I have made to my methodology since that paper was published.

Study area and sampling methods

I focused on a 2,520 km^2 portion of the Coastal Douglas-fir zone of BC that includes >2000 islands from 0.0003 - 32,000 km^2 (e.g., Vancouver Island). Roughly 40% of the CDF region still occurs as uneven-aged forest, interspersed with shallow soil balds, deep soil meadows and woodland/savannah habitat. Mature CDF forests support long-lived conifers and a mainly deciduous tree and shrub sub-canopy subject to disturbance and, as a result, are structurally complex (Meidinger and Pojar 1991, Mosseler et al. 2003). As part of a larger program on avian conservation (Jewell et al. 2007, Martin et al. 2011), trained observers conducted point counts on 53 islands from 30 Apr – 11 Jul, 2005 - 2011 (except 2006). Trained observers recorded all birds detected in a 50m radius in a 10 min period between 5 AM – 12 PM, at 713 sample locations (mean distance between all locations = 19 km). Total visits to each location ranged from 1-12 (mean = 1.86), with each location recorded by handheld GPS (GPS60, Garmin Ltd, Kansas, USA).

Expert rankings

I asked 11 professional ornithologists with >5 years of experience with local birds to estimate the degree of association of 47 species (Table A.4.1) to old forest (OF) and savannah/woodland (SAV) CDF habitats. Specifically, experts ranked species' according to their expected association (low = -1, medium = 0 or highly associated = 1) with each of 10 focal habitat types in present-day CDF habitats using photographic and text descriptions of herbaceous, shrub, woodland, wetland, four

forest types (pole, young, mature and old), and 2 human-dominated habitats (rural, urban). (For detailed descriptions of my approach see Appendix 8).

Expert ranks were then averaged for each species and habitat type. Old forest (OF) association scores for each species were then calculated by summing a species' rank in each habitat, multiplied by following weights: herbaceous (-2), shrub/herb (-1), pole/sapling (-0.5), young forest (+0.5), mature forest (+1) and old forest (+2). Doing so resulted in a score for each species that ranged from a minimum of -7, indicating no association to old forest structure, to a maximum of 7, indicating a strong association to old forest structure. This score was then standardized to fall between -1 and 1 by dividing by the maximum value possible (7). All birds with positive forest association scores were therefore considered to be members of the CDF old forest (OF) bird community, with those species having higher forest association scores contributing most to composite maps. In this paper I extended this approach to also include Savannah habitats (SAV), which is detailed in the main text.

Landscape covariates

Because birds respond to many fine and coarse scale habitat features (Lawler and Edwards 2006) I developed covariate descriptors of landscape condition and context using coarse (1km) and fine (100m) scale features to advance early work conducted only at coarse scales (DeWan et al. 2009). For modelling species detection and occurrence, I chose candidate predictors based on their proven ability to predict species occurrence at site and landscape levels in similar exercises or regions (Guisan and Thuiller 2005, Jewell et al. 2007). All covariate names appear in Table A.4.2 and were derived from the following sources: (i) Terrain Resource Information Management (TRIM, http://archive.ilmb.gov.bc.ca/crgb/pba/trim/specs/specs20.pdf, date accessed: 2011-10-10); (ii) Sensitive Ecosystems Inventory (SEI): East Vancouver Island and the Gulf Islands (http://www.env.gov.bc.ca/sei/, date accessed: 2011-10-10); (iii) Earth Observation for Sustainable

Development Landcover (EOSD LC 2000,(Wulder et al. 2008)); (iv) aerial photographs to calculate the islands sizes; and (v) Terrestrial Ecosystem Mapping (TEM) of the CDF Zone (MES 2008).

My dataset comprised 25 predictor covariates of site and landscape condition (Table A.4.2), derived at each of 713 avian point count locations. The data source satellite and aerial photography imagery was collected between 2002 and 2004, which was in part supplemented by ground work until 2008. All covariates were created using Geospatial Modeling Environment (Beyer 2012) in conjunction with ArcGIS 10.1 (ESRI 2012) and R v. 2.15.2 (R Development Core Team 2012). Due to their widely varying scales, all covariates were standardized about their mean value, to ensure that importance was not driven by measurement scale (White and Burnham 1999).

Occupancy and detection models

The large size of my study area required us to compile data from related surveys conducted during a single 9 week period (Jewell et al. 2007, Martin et al. 2011) but differing in sampling intensity between years and precluding reliable estimates of colonization and extinction (MacKenzie et al. 2003). I also assumed no variation in site occupancy across years to minimize model complexity, thus assumed a closed population for all species (Mackenzie et al. 2002). The R package unmarked v. 0.9-9 (Fiske and Chandler 2011) provided the framework for all species models, which necessarily include two parts: occupancy and detection (Mackenzie et al. 2002). To estimate detectability I used one site specific (crown closure) and three observation specific (time of date, Julian date and observer identity) covariates. For each of 47 focal species I fitted all 16 detectability models (without parameterizing occupancy) and then ranked each by AIC (Akaike 1974) to select top-ranked models for further analysis. To accommodate my reduced but still extensive set of 25 predictor covariates for occupancy modelling, I first used a modified 'stepwise' covariate selection procedure linked to the unmarked package to create a candidate set of models based on the statistical

significance of individual covariates and AIC. I then ranked all candidate models by AIC and averaged those with $\Delta AIC \leq 7$ from the top ranked one (Burnham and Anderson 2002).

Predictive maps

I created landscape level predictive occupancy maps over my 2,520 km^2 study area using 252,000 1ha hexagons. For each hexagon centroid I generated a covariate set identical to that used for survey points, and then estimated probability of occurrence based on my averaged models for each of the focal species. To consolidate focal species maps into an index of forest-associated bird species richness, I created a score for each polygon resembling a single-species habitat suitability index (Guisan and Thuiller 2005, Beaudry et al. 2010). Specifically, I calculated the polygon scores by summing the weighted probability of occurrence of each species linked to old forest structure via expert questionnaires (Martin et al. 2005a). This process yielded a weighted, forest-association community score that ranged from 0 (no forest-associated species present) to 1 (all forest-associated species present) for each of the map polygons.

Major additions to Schuster and Arcese (2013)

In addition to extending the extend of Schuster and Arcese (2013) I also addressed two uncertainties they identified to improve and assess the fit of my models by testing for spatial autocorrelation using Bayesian approaches to estimate Moran's I in model residuals and goodness of fit using 'area under the curve' (AUC). Where spatial autocorrelation resulted in values of Moran's I > 0.2, I added a spatial autocovariate to models. To assess model goodness of fit I used Markov Change – Mote Carlo (MCMC) iteration to calculate AUC by comparing model results to Bayesian inference Using Gibbs Sampling (BUGS) estimates of latent occupancy state, using a modification of Zipkin et al. (2012).

The tests for spatial autocorrelation in model residuals resulted in Moran's $I > 0.2$ in six of the 47 bird species. For five species (dark eyed junco, Eurpean starling, fox sparrow, northern rough-winged swallow, song sparrow) the inclusion of a first order autocovariate term resulted in a reduction of Moran's I below 0.2. For one species (varied thrush) I did not find a first order autocovariate distance that removed autocorrelation entirely (Table A.4.). Bayesian model goodness of fit (GOF) estimates for AUC ranged from 0.659 – 0.991 (mean = 0.881) (Table A.4.). Based on the expert elicitation results I included 16 bird species in the OF score (Table A.4.1, Figure A.4.1) and 13 in the SAV score (Table A.4.1, Figure A.4.2). Five additional species had positive OF scores, but were not included in the OF community score, because their 95% credible intervals of model GOF did not span AUC values of 0.8, which I used as my cut-off to indicate models of good fit (bald eagle, common raven, warbling vireo, Western tanager) or model predictions were unrealistic (Swainson's thrush), predicting high probability of occurrence in urban areas, despite the fact that this species is mostly associated with undisturbed forests in BC (Campbell et al. 1997). The CDF score I created by combining OF (Figure A.4.1) and SAV (Figure A.4.2) predictions varied from 0 to 0.79, illustrating the fact that there is no area of complete overlap between the two biodiversity metrics, which was to be expected given the fact that I used fine scale (1ha) polygons as my predictive units (Figure A.4.3).

Table A.4.1: Bird species and elicitation results. Yellow fields show species that have positive OF scores, but have not been included in the combined metric, due to low GOF or unrealistic predictions (reasons in text). OF score = old forest association score. SAV score = savannah association score. OF/SAV weight = the weight positive OF/SAV scores got in producing the old forest/savannah community association metrics.

Bird Species	OF score	OF weight	SAV score	SAV weight
American Goldfinch	-0.396	-	0.236	0.092
American Robin	-0.143	-	0.255	0.099
Bald Eagle	0.617	-	-0.782	-
Barn Swallow	-0.429	-	-0.182	-
Bewick's Wren	-0.221	-	0.055	0.021
Brown-headed Cowbird	-0.532	-	0.400	0.156
Brown Creeper	0.831	0.077	-0.600	-

Bird Species	OF score	OF weight	SAV score	SAV weight
Chestnut-backed Chickadee	0.636	0.059	-0.255	-
Chipping Sparrow	-0.487	-	0.255	0.099
Common Raven	0.682	-	-0.673	-
Dark-eyed Junco	-0.144	-	0.327	0.128
European Starling	-0.331	-	-0.218	-
Fox Sparrow	-0.249	-	-0.271	-
Golden-crowned Kinglet	0.734	0.068	-0.636	-
Hairy Woodpecker	0.818	0.075	-0.673	-
Hammond's Flycatcher	0.685	0.063	-0.727	-
House Finch	-0.275	-	-0.076	-
House Sparrow	-0.169	-	-0.691	-
House Wren	-0.092	-	0.051	0.020
MacGillivray's Warbler	-0.123	-	-0.109	-
Northwestern Crow	-0.212	-	-0.109	-
Northern Flicker	0.481	0.044	-0.073	-
Northern Rough-winged Swallow	-0.325	-	-0.323	-
Orange-crowned Warbler	-0.126	-	0.036	0.014
Olive-sided Flycatcher	0.539	0.050	-0.455	-
Pine Siskin	0.578	0.053	-0.382	-
Pileated Woodpecker	0.877	0.081	-0.673	-
Pacific-slope Flycatcher	0.787	0.072	-0.491	-
Purple Finch	0.235	0.022	-0.182	-
Red-breasted Nuthatch	0.831	0.077	-0.400	-
Rufous Hummingbird	-0.175	-	0.364	0.142
Red-winged Blackbird	-0.130	-	-0.709	-
Savannah Sparrow	-0.532	-	0.036	0.014
Song Sparrow	-0.214	-	0.073	0.028
Spotted Towhee	-0.184	-	0.164	0.064
Swainson's Thrush	0.429	-	-0.345	-
Townsend's Warbler	0.760	0.070	-0.655	-
Tree Swallow	-0.186	-	-0.364	-
Varied Thrush	0.770	0.071	-0.764	-
Violet-green Swallow	-0.200	-	-0.200	-
Warbling Vireo	0.104	-	-0.255	-
White-crowned Sparrow	-0.552	-	0.309	0.121
Western Tanager	0.675	-	-0.473	-
Wilson's Warbler	-0.013	-	-0.273	-
Winter/Pacific Wren	0.779	0.072	-0.673	-
Yellow-rumped Warbler	0.412	0.038	-0.436	-
Yellow Warbler	-0.221	-	-0.364	-

Table A.4.2: Occupancy model covariate description including data source and covariate abbreviation used here. The data source satellite and aerial photography imagery was collected between 2002 and 2004, which was in part supplemented by ground work until 2008.

Source	Covariate description	Abbreviation
TRIM	Total amount of unpaved road length within a 1km buffer	rdl_up_1k
	Total amount of unpaved road length within a 100 buffer	rdl_up_100
	Total amount of paved road length within a 1km buffer	rdl_p_1k
	Total amount of paved road length within a 100 buffer	rdl_p_100
	Nearest road	near_road
	Nearest freshwater source	near_frshw
	Nearest shoreline	near_saltw
Aerial photographs used to draw island polygons	Island size	Is_size
Terrestrial Ecosystem Mapping of the Coastal Douglas-Fir Biogeoclimatic Zone	Rural/agriculture area within a 1km buffer	RUR_1KM
	Forest cover area within a 1km buffer including structural stages closed and young forest	FOR1_1KM
	Forest cover area within a 1km buffer including structural stages mature and old forest	FOR2_1KM
	Herbaceous area within a 1km buffer	HRB_1KM
	Savannah area within a 1km buffer	SAV_1KM
	Shrub area within a 1km buffer	SHR_1KM
	Wetland area within a 1km buffer	WET_1KM
	Urban/industrial area within a 100m buffer	URB_100
	Rural area within a 100m buffer	RUR_100
	Forest cover area within a 100 m buffer including structural stages closed and young forest	FOR1_100
	Forest cover area within a 100 m buffer including structural stages mature and old forest	FOR2_100
	Herbaceous area within a 100m buffer	HRB_100
	Savannah area within a 100m buffer	SAV_100
	Shrub area within a 100m buffer	SHR_100
	Wetland area within a 100m buffer	WET_100
	Distance to nearest Urban area	Near_Urb
Earth Observation for Sustainable Development(EOSD) Landcover	Crown closure within a 100m buffer, combining EOSD crow closure categories (dense, open and sparse) into one measure	CR_CL

Table A.4.3: Results of tests for residual spatial autocorrelation and area under the receiver operation curve (AUC). Columns on the left show initial results with maximum Moran's I values from the MCMC approach. AUC mean and 95% credible interval values are further shown. For bird species with max. Moran's I values > 0.2 autocovariates were added to averaged models to try to account for residual autocorrelation. The distance column shows which autocovariate distance was chosen per bird and columns to the right of this show the Moran's I improvements as well as changes to AUC values.

Bird species	max Moran's I	AUC mean	AUC 2.5	AUC 97.5	distance	max Moran's I	AUC mean	AUC 2.5	AUC 97.5
AMGO	0.17	0.92	0.80	0.96					
AMRO	0.05	0.98	0.97	0.99					
BAEA	0.15	0.73	0.69	0.76					
BARS	0.10	0.99	0.99	0.99					
BEWR	0.11	0.90	0.86	0.94					
BHCO	0.10	0.97	0.96	0.98					
BRCR	-0.10	0.96	0.94	0.98					
CBCH	-0.17	0.96	0.93	0.97					
CHSP	0.09	0.88	0.84	0.91					
CORA	0.09	0.66	0.58	0.75					
DEJU	0.20	0.91	0.88	0.93	250m	0.09	0.92	0.90	0.94
EUST	0.26	0.86	0.83	0.88	250m	0.06	0.89	0.86	0.91
FOSP	0.25	0.90	0.88	0.92	200m	0.12	0.90	0.82	0.94
GCKI	0.11	0.86	0.80	0.90					
HAFL	0.16	0.84	0.61	0.98	1km	0.11	0.76	0.62	0.84
HAWO	-0.08	0.84	0.77	0.90					
HOFI	-0.09	0.92	0.88	0.95					
HOSP	0.09	0.98	0.94	0.99					
HOWR	0.10	0.87	0.84	0.89					
MGWA	0.19	0.97	0.95	0.99					
NOCR	0.12	0.88	0.86	0.90					
NOFL	-0.09	0.78	0.72	0.84					
NRWS	0.21	0.83	0.76	0.89	200m	0.18	0.78	0.58	0.91
OCWA	0.07	0.99	0.96	0.99					
OSFL	0.08	0.90	0.88	0.92					
PISI	0.06	0.96	0.69	0.99					
PIWO	0.06	0.82	0.73	0.90					
PSFL	0.08	0.99	0.98	0.99					
PUFI	0.14	0.99	0.95	0.99					
RBNU	-0.09	0.96	0.87	0.99					
RUHU	0.13	0.86	0.82	0.88					
RWBL	0.17	0.99	0.98	0.99					
SAVS	0.09	0.95	0.86	0.98					
SOSP	0.22	0.80	0.81	0.85	250m	0.09	0.85	0.82	0.87

Bird species	max Moran's I	AUC mean	AUC 2.5	AUC 97.5	distance	max Moran's I	AUC mean	AUC 2.5	AUC 97.5
SPTO	0.19	0.92	0.91	0.94					
SWTH	0.14	0.79	0.74	0.84					
TOWA	0.06	0.97	0.95	0.98					
TRES	0.12	0.84	0.76	0.91					
VATH	0.26	0.79	0.68	0.88	---				
VGSW	0.04	0.89	0.85	0.93					
WAVI	0.16	0.74	0.69	0.79					
WCSP	0.13	0.88	0.85	0.90					
WETA	0.07	0.66	0.56	0.77					
WIWA	0.18	0.88	0.84	0.92					
WIWR	0.08	0.81	0.77	0.84					
YRWA	0.09	0.79	0.74	0.84					
YWAR	0.17	0.85	0.81	0.88					

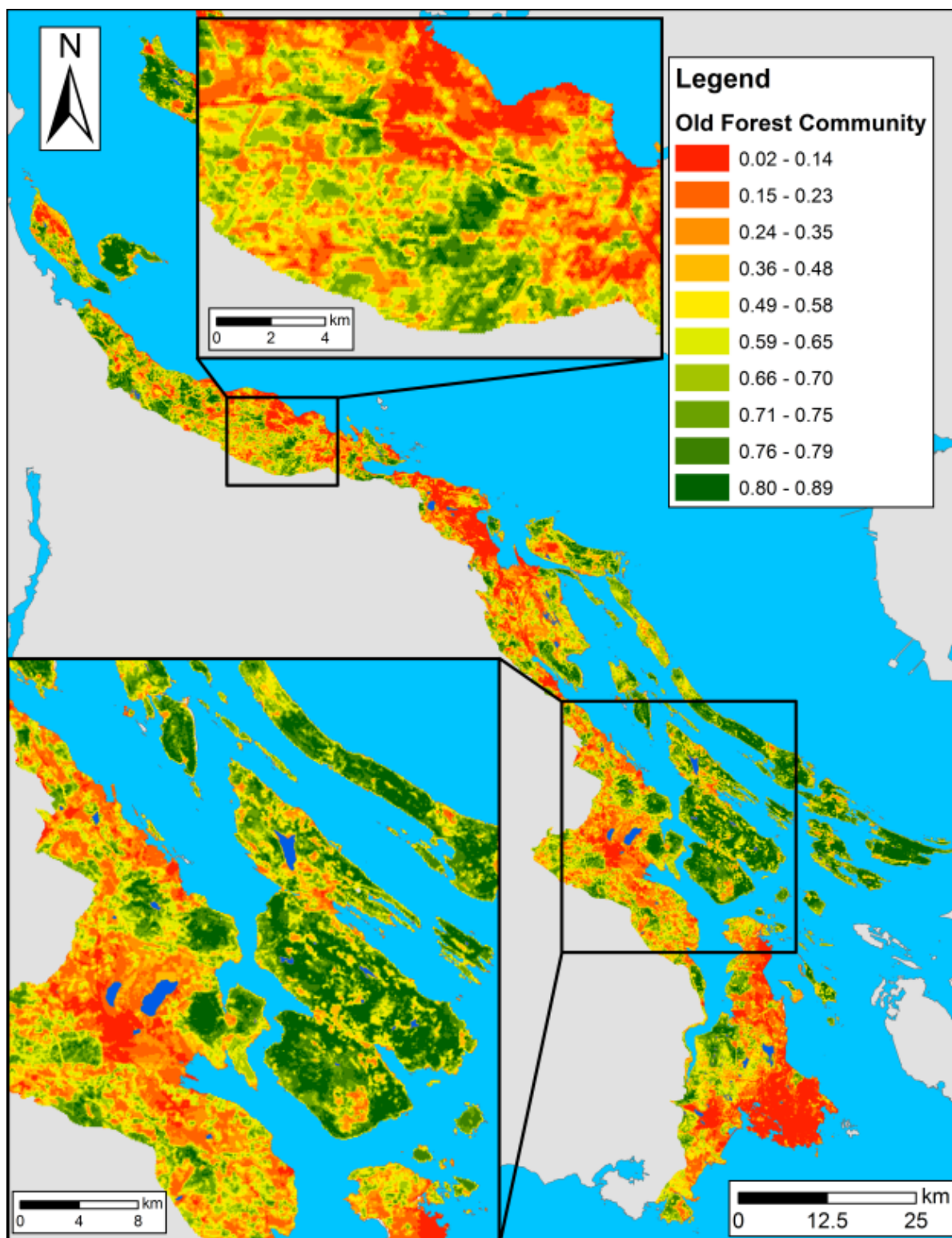**Figure A.4.1: Old Forest score representation.**

Figure A.4.2: Savannah score representation.
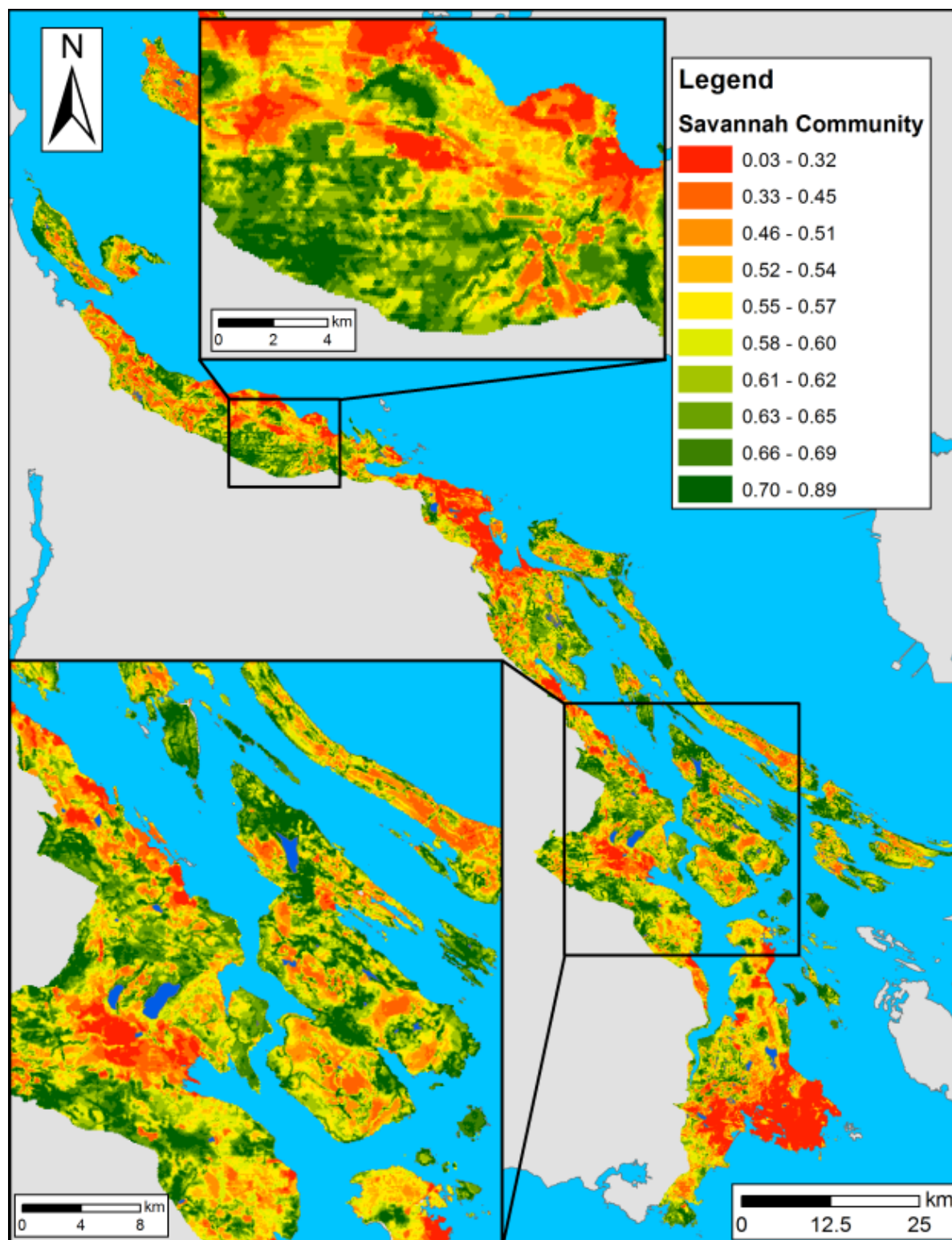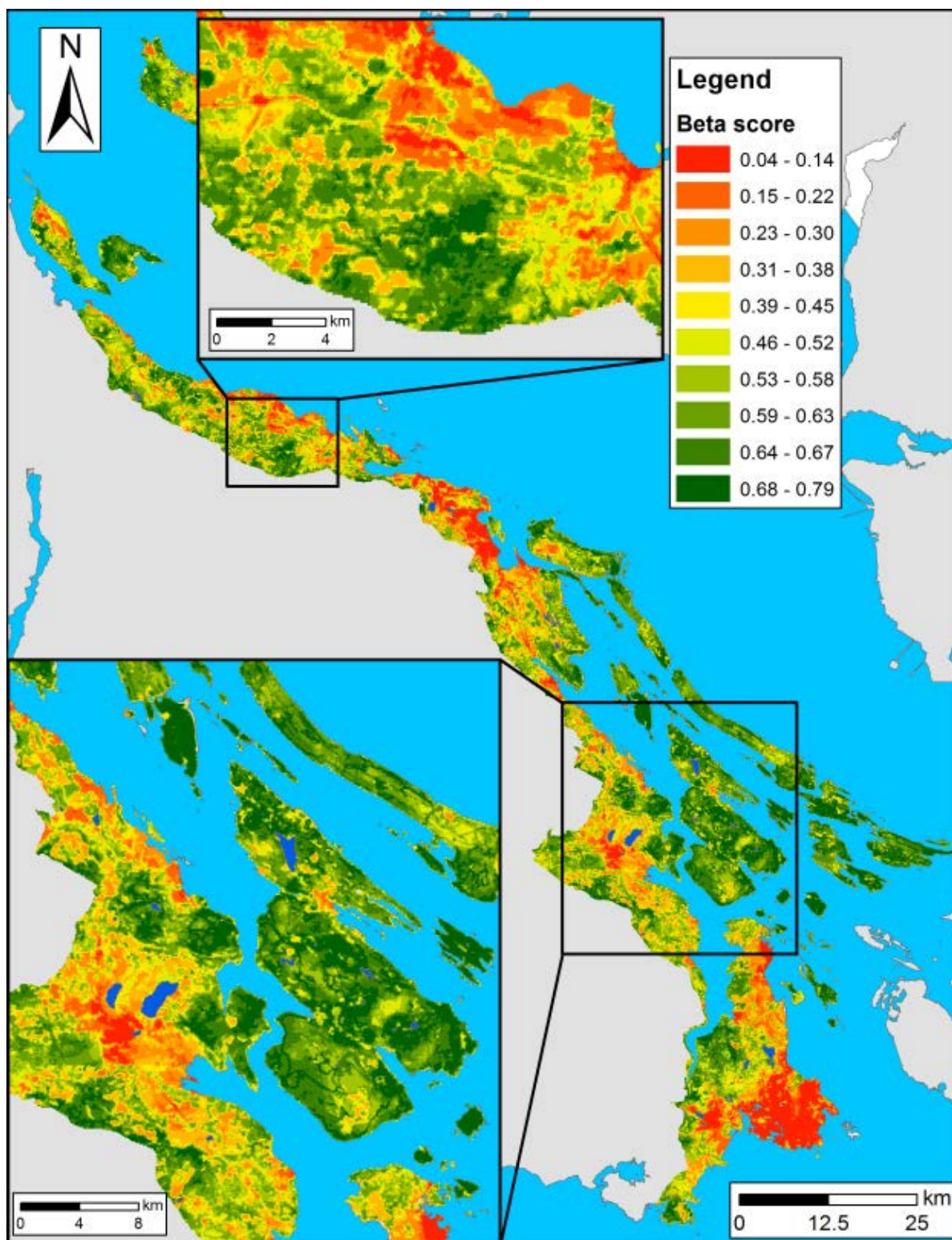
Figure A.4.3: Beta Score representation.

Appendix 5:   **Standing carbon and carbon sequestration potential estimates.**

Standing carbon estimates from model FORECAST in the CDF ranged from 0 to 615 t/ha (mean ± SD = 96.1 ± 92.95) and the total amount of carbon in the study area was estimated at 20.84 Mt (Figure A.5.1). Estimates for sequestration potential over the next 20 years ranged from -9 to 130.6 t/ha (mean ± SD = 37.4 ± 31.9) with a total sequestration potential for the area of 7.98 Mt (Figure A.5.2).

The methodology for standing carbon and carbon sequestration potential estimates is described in a technical report (Seely 2012). Terrestrial Ecosystem Mapping (TEM) data from Parks Canada CDF areas were combined with TEM data from non-Parks areas (MES 2008). The merged TEM data were structured such that each polygon was divided into up to three discrete subsections. Each subsection (defined in the data as a proportion of the whole polygon area) had a site series and structural stage defined. Site series are representative ecosystems in each Biogeoclimatic subzone of British Columbia (MES 2008). For the carbon modeling Seely (2012) used FORECAST (Kimmins et al. 1999), a stand-level forest ecosystem simulator that is one of two models approved by the BC Ministry of Forests for carbon budget assessments (Ministry of Environment 2011), and the only model calibrated for use in the CDF (Blanco et al. 2007) and linked to TEM (Seely et al. 2004).

To facilitate the carbon analysis, the TEM polygons were stratified into a series of relatively homogenous analysis units (AUs) based upon site series. The starting conditions for the FORECAST simulations were created by running the model in set-up mode up to 2000 years, to allow for the accumulation of dead organic matter (including soil organic matter) that would be representative of the disturbance history of the site. The final set-up run was conducted with nutrient feedback turned on to allow the site quality representation in the model to stabilize.

To represent the impact of endemic disturbance agents, including pests and pathogens, the simulations included periodic mortality events that occurred throughout the simulation at a regular interval and intensity for each species. Carbon storage curves (annual time step) were created for each AU and stored in a database. Carbon contents were simulated for all ecosystem pools including: soil organic matter, above-ground litter, above and below-ground tree biomass, plant biomass, deadwood, and dead below-ground biomass. Net ecosystem carbon storage was calculated for each AU considering the fact that only certain ecosystem pools are generally eligible for forest carbon offsets. Net ecosystem carbon storage was limited to: above and below-ground tree biomass, deadwood biomass, and dead below-ground biomass. Each AU was simulated for a period of 300 years with results reported for annual time steps.

FORECAST results were subsequently assigned to individual TEM polygons by estimating the age of each polygon subsection based upon the current assigned structural stage and estimated productivity class (Seely et al. 2004). These age estimates are derived from ranges provided by Meidinger (1998) for regional forest ecosystems. Ages of old stands (structural stage 7) were set at 200 to be conservative. Age estimates were verified against a subset of TEM polygons (Southern Gulf Islands) for which direct age estimates were available (n=254). For conservation prioritization analysis we used predicted net ecosystem carbon storage and net ecosystem carbon sequestration estimates for 20 years from now (the analysis timeframe predictions could be made with confidence).

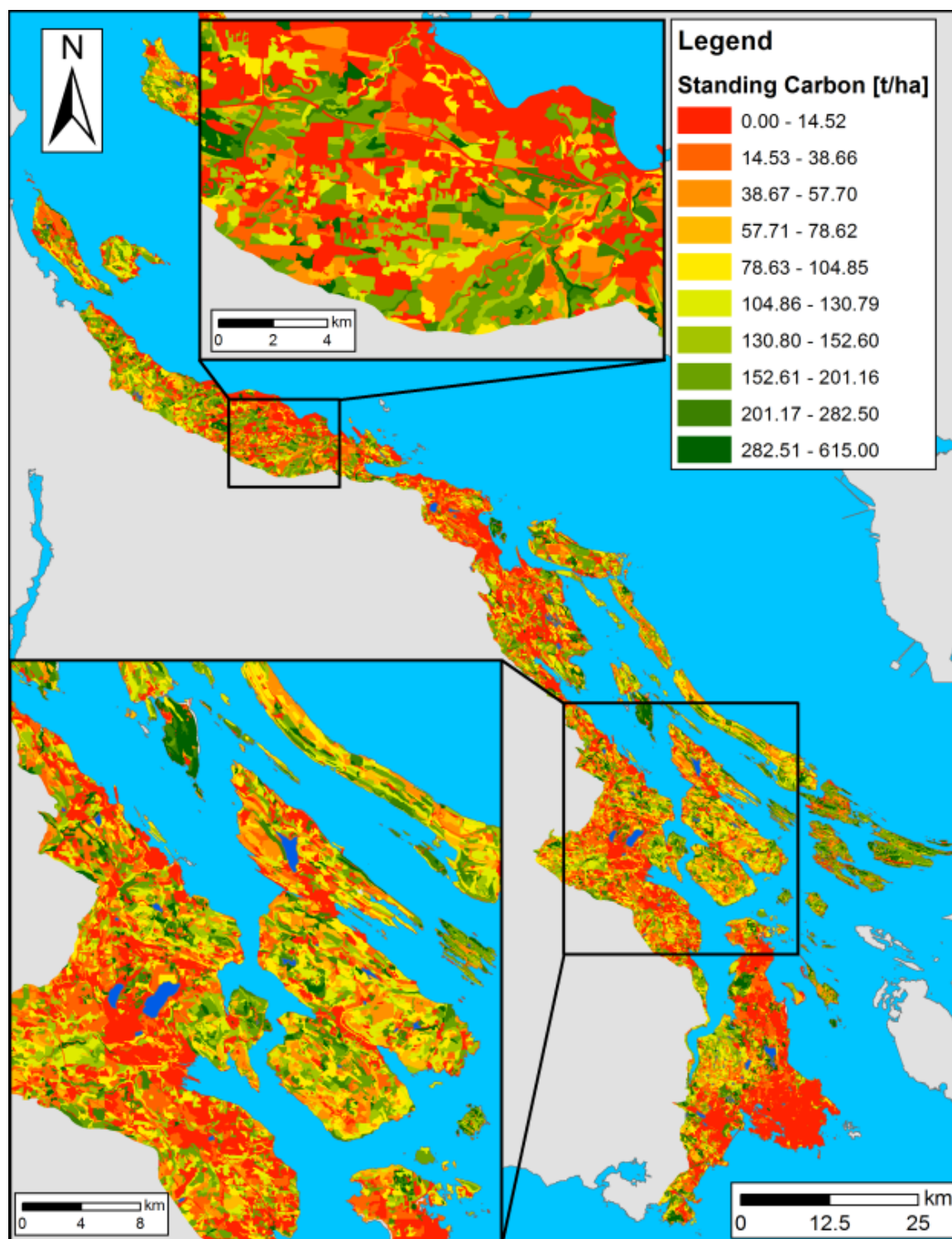**Figure A.5.1:** Net ecosystem carbon storage within forests of the CDF zone in British Columbia.

Figure A.5.2: Net ecosystem carbon sequestration (during the next 20 years) within forests of the CDF zone in British Columbia.
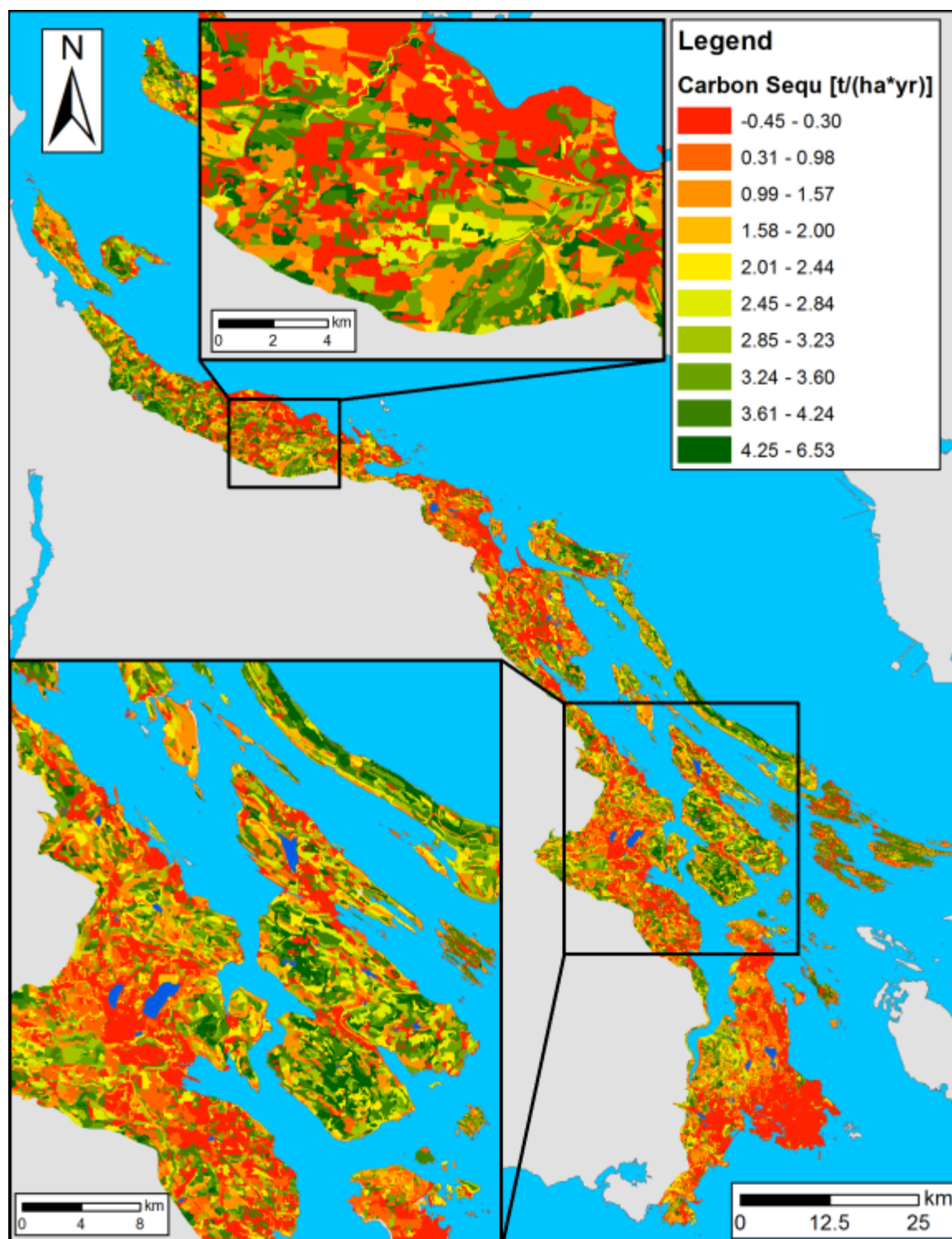
Appendix 6:   **R code for Marxan scenarios and calibration.**

This appendix provides the R code I developed for Marxan calibration that can be used in combination with both the 32 and 64bit versions of Marxan. Function results can be used for Marxan post processing in R.

```
#bare bone run with standard input files
mm <- marxan()



#batch run with SPF, BLM, NREPS, NITNS changing
spf <- c(1,5,10,15,20)
blm <- c(0, 0.1, 0.2, 0.5, 1)
nreps <- c(10, 20, 50, 100)
nitns <- c(10000, 100000, 1000000, 10000000)
mm <- marxan(pu="pu_non_st.csv",
        puvsp="puvsp_non_st.csv",
        spec="spec_non_st.csv",
        bound="bound_non_st.csv",
        spf=spf, blm=blm, nreps=nreps, nitns=nitns, scenname="Full")


########################################################################
########################################################################
## R script for Marxan batch run
## returns data.frames of the best solutions, summed solutions,
## and the summary tables
##
## Parts based on code by Paulo Cardoso
## (http://lists.science.uq.edu.au/pipermail/marxan/2008-May/000319.html)
##
## Author: Richard Schuster
## 11 July 2013
########################################################################
########################################################################
marxan <- function(pu="pu.dat", puvsp="puvsp2.dat", spec="spec.dat", bound="", spf=1,
            blm=0, nitns=100000, nreps=10, scenname="", indir=getwd(), outdir=getwd() ){


 ## Read and obtain Input.dat parameters
 if(file.exists("input.dat"))
   input.file<-dir(pattern="input.dat")
 else
   stop("input.dat file not found")

 input<-readLines(input.file[1],n=-1)

 ## Set input and output directories
 txt <- "INPUTDIR"
```

```
input[grep(txt, input)]<-paste(txt,sprintf("%s/input",indir))
txt <- "OUTPUTDIR"
input[grep(txt, input)]<-paste(txt,sprintf("%s/output",outdir))




## Create data frame for later use with GIS
txt <- "PUNAME"
input[grep(txt, input)]<-paste("PUNAME",pu) # pu file


txt <- "PUVSPRNAME"
input[grep(txt, input)]<-paste(txt,puvsp) # Puvsp file name in input.dat

spffr <- read.csv((file=sprintf("%s/input/%s",indir,spec)))
txt <- "SPECNAME"
input[grep(txt, input)]<-paste(txt,spec) # Spec file name in input.dat

#Boundary file
txt <- "BOUNDNAME"
input[grep(txt, input)]<-paste(txt,bound) # pu file


#gsub(" ","",substr(input[grep(txt, input)],nchar(txt)+1,nchar(input[grep(txt, input)])), fixed=T)

pufr <- read.csv((file=sprintf("%s/input/%s",indir,pu)))
ssolnfr <- bestfr <- data.frame(ID= pufr$id)
summed <- list(TimeStamp=date())


######################### Step 2
#######################################################################

##Loop for sequential sfp
kk <- 2
for(ii in 1:length(spf)) {

  #species penalty factor needs tp be set and saved win spec file for each run
  spffr$spf <- spf[ii]
  write.csv(spffr, sprintf("%s/input/%s",indir,spec), row.names = FALSE)


  ## Loop for sequential Marxan Runs
  for(jj in 1:length(blm)){

    ## Input.dat parameters ##
    txt <- "BLM"
    input[grep(txt, input)]<-paste(txt,blm[jj]) # BLM in input.dat
```

```r
    for (ll in 1:length(nreps)){
     txt <- "NUMREPS"
     input[grep(txt, input)]<-paste(txt,nreps[ll]) # Number of runs in input.dat

     for (mm in 1:length(nitns)){

      txt <- "NUMITNS"
      input[grep(txt, input)]<-paste(txt,sprintf("%i",nitns[mm])) # Number of runs in input.dat

      txt <- "SCENNAME"
      runname <- sprintf("%s_Spf%i_Blm%s_Nrep%i_Iter%i",scenname,round(spf[ii])
                   ,blm[jj],nreps[ll], nitns[mm])
      input[grep(txt, input)]<-paste(txt,runname) # Puvsp file name in input.dat

      write(input,"input.dat")# Re-write input file at each run with the
                      #corresponding parameters changed
      if(file.exists("Marxan_x64.exe"))
        system("Marxan_x64.exe",wait=T,invisible=T) # Call Marxan to execute
      else if(file.exists("Marxan.exe"))
        system("Marxan.exe",wait=T,invisible=T) # Call Marxan to execute
      else
        stop('No Marxan executable found in working directory')

      # saving results for function return
      ssoln <- read.csv((file=sprintf("%s/output/%s_ssoln.txt",outdir,runname)))
      best <- read.csv((file=sprintf("%s/output/%s_best.txt",outdir,runname)))
      summ <- read.csv((file=sprintf("%s/output/%s_sum.txt",outdir,runname)))
      ssoln<-ssoln[order(ssoln$planning_unit),]
      best<-best[order(best$planning_unit),]
      ssolnfr <- data.frame(ssolnfr, ssoln$number)
      bestfr <- data.frame(bestfr, best$solution)
      summed[[kk]] <- summ
      names(ssolnfr)[kk] <- names(bestfr)[kk] <- names(summed)[kk] <- runname
      kk <- kk + 1
     }
    }
   }
 }
 return(list(ssoln=ssolnfr,best=bestfr,sums=summed))
}


###############################################################
#### FUNCTION END
###############################################################
```

Appendix 7: R code for covenant cost-effectiveness analysis.

```
################################################################################
### Setup Covenant analysis (run the Marxan scenario)
### Richard Schuster
### 2013, Sept 25
################################################################################

library(Rmarxan2)
setwd("D:\\R_files\\13_09_25_Ch3_analytical_framework_setup")
cad <- read.csv("Single_Poly_post_cov_top_Land_value_IDW_10_groups_R.csv")

# area adjustment to have it in acres
cov.cst <- data.frame(ID=cad$ID, AREA=cad$AREA, ACR=cad$AREA * 0.000247105,
COST=cad$CALC_TOTAL)

# fixed costs
#Land owner
LO.legal <- 300
LO.finadv <- 300
LO.registr <- 200
LO.endow <- 10000
#Covenant Holder
CH.legal <- 4000

fixed.all <- LO.legal + LO.finadv + LO.registr + LO.endow + CH.legal

# cost vary with property area
# area unit used is acre
LO.bas.b0 <- -2185.30978
LO.bas.b1 <- 1957.45823
LO.bas.min <- 1000
LO.app.b0 <- 0
LO.app.b1 <- 1957.45823
LO.app.min <- 1500
LO.surv.b0 <- 300
LO.surv.b1 <- 1957.45823
LO.surv.min <- 1000


cov.cst$COV.fix <- fixed.all
cov.cst$COV.bas <- ifelse((LO.bas.b0 + LO.bas.b1 * log(cov.cst$ACR)) > LO.bas.min,
              (LO.bas.b0 + LO.bas.b1 * log(cov.cst$ACR)),LO.bas.min)
cov.cst$COV.app <- ifelse((LO.app.b0 + LO.app.b1 * log(cov.cst$ACR)) > LO.app.min,
              (LO.app.b0 + LO.app.b1 * log(cov.cst$ACR)),LO.app.min)
cov.cst$COV.surv <- ifelse((LO.surv.b0 + LO.surv.b1 * log(cov.cst$ACR)) > LO.surv.min,
              (LO.surv.b0 + LO.surv.b1 * log(cov.cst$ACR)),LO.surv.min)
cov.cst$COV.comb <- rowSums(cov.cst[,4:7], na.rm=T)
```

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```
################################################################################
# standard repeat costs
# Monitoring (to see if covenant is intact)
# repeat rate: 1/1 year
# rate used: NCC charge (from Management cost workshop)
mon <- 758
# Staff cost (work done to reply to Land owner request)
# repeat rate: 1/5 years
# rate used: NCC charge (from Management cost workshop)
# presented in yearly portion:
staff <- mon/5


tt <- cov.cst$COV.comb + 50*(mon+staff)

marxan.covenant.pu <- data.frame(id=cov.cst$ID, cost=cov.cst$COV.comb, status=0)
write.csv(marxan.covenant.pu, "input/Cadaster_pu_cost_Covenant.csv", row.names = FALSE)


indir=getwd()
spf <- 3
nitns <- 10000000
# Covenant Marxan costs
puC="Cadaster_pu_cost_Covenant.csv"
# Acquisition Marxan costs
puA="Cadaster_pu_cost_IDW_new.csv"

puvsp="Cadaster_puvsp_no_C_IDW_new_beta_score.csv"
spec="Cadaster_spec_BETA.csv"
#bound="Cadaster_bound_100m_buff.csv"

spffr <- read.csv((file=sprintf("input/%s",spec)))
spffr[spffr$name=="BETA",]$prop <- 0.2 #Set target
write.csv(spffr, sprintf("input/%s",spec), row.names = FALSE)



################################################################################
# Acquisition Marxan run
setwd("./Marxan_Acquisition")
outdir=getwd()

marx.acqu <-  marxan(pu=puA,
             puvsp=puvsp,
             spec=spec,
#              bound=bound,
             spf=spf, nreps=1000, nitns=nitns, scenname="Acqu.T0.2",
```

```
                    indir=indir,outdir=outdir)
setwd("..")
##############################################################################

save.image("Covenants_Acqu_1000_runs.RData")

spd <- function (years, cov.frame, dispute, rD1, pu, mon, staff){

   for (ii in 2:years){
    for (pu in 1:length(cov.frame[,1])){
     if (runif(1) > dispute) {
       #cost of dispute
       dis.cst <- rD1(1) #disp.cost$cost[which.min(disp.cost$prb < runif(1))]
       #dis.cst <- 100000*rexp(1,rate=0.5)

       #metric where bd.loss is losely dependent on dis.cst
       init.bd.loss2 <- dis.cst / 400000
       bd.loss2 <- init.bd.loss2 + rnorm(1,0,0.05)
       bd.loss2 <- ifelse(bd.loss2 < 0, 0, ifelse(bd.loss2 > 1, 1, bd.loss2))
       cov.frame[pu,4+years+ii] <- cov.frame[pu,4+years+ii-1] * (1 - bd.loss2)

       # add dispute cost to covenant cost + set minimum cost to $1000
       min.cst <- 1000
       dis.cst <- ifelse(dis.cst<min.cst,min.cst,dis.cst)
       cov.frame[pu,4+ii] <- cov.frame[pu,4+ii-1] + dis.cst
      }
     else {
       # if there is no dispute carry costs forward but add yearly costs
       cov.frame[pu,4+ii] <- cov.frame[pu,4+ii-1] + (mon + staff)
       # if there is no dispute carry biodiversity values forward
       cov.frame[pu,4+years+ii] <- cov.frame[pu,4+years+ii-1]

     }
    }
   }
  return(as.data.frame(cov.frame))
}

# setup and Marxan runs in folder:
# 13_09_25_Ch3_analytical_framework_setup
library(distr)

setwd("D:\\R_files\\13_12_18_Ch3_analysis_speedup")

load("Covenants_Acqu_1000_runs_red.RData")
rm(list=setdiff(ls(), c("marx.acqu","spd")))

#marx.acqu slots
```

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```
# ssoln
# best
# run
# sums
# mv


# data frame including
# ID
# AREA
# CALC_TOTAL
# Carbon metrics (StC_AWS, SeqC_AWS)
# Biodiv metrics (both AWM and AWS for OF, SAV, BETA)
cad <- read.csv("Polygon_level_Area_Carbon_Biodiv_values.csv")

# area adjustment to have it in acres
cov.cst <- data.frame(ID=cad$ID, AREA=cad$AREA, ACR=cad$AREA * 0.000247105,
COST=cad$CALC_TOTAL)

# fixed costs
#Land owner
LO.legal <- 300
LO.finadv <- 300
LO.registr <- 200
LO.endow <- 10000
#Covenant Holder
CH.legal <- 4000

fixed.all <- LO.legal + LO.finadv + LO.registr + LO.endow + CH.legal

# cost vary with property area
# area unit used is acre
LO.bas.b0 <- -2185.30978
LO.bas.b1 <- 1957.45823
LO.bas.min <- 1000
LO.app.b0 <- 0
LO.app.b1 <- 1957.45823
LO.app.min <- 1500
LO.surv.b0 <- 300
LO.surv.b1 <- 1957.45823
LO.surv.min <- 1000


cov.cst$COV.fix <- fixed.all
cov.cst$COV.bas <- ifelse((LO.bas.b0 + LO.bas.b1 * log(cov.cst$ACR)) > LO.bas.min,
            (LO.bas.b0 + LO.bas.b1 * log(cov.cst$ACR)),LO.bas.min)
cov.cst$COV.app <- ifelse((LO.app.b0 + LO.app.b1 * log(cov.cst$ACR)) > LO.app.min,
            (LO.app.b0 + LO.app.b1 * log(cov.cst$ACR)),LO.app.min)
```

```
cov.cst$COV.surv <- ifelse((LO.surv.b0 + LO.surv.b1 * log(cov.cst$ACR)) > LO.surv.min,
                (LO.surv.b0 + LO.surv.b1 * log(cov.cst$ACR)),LO.surv.min)


# combined initial covenant cost of each parcel in the CDF
# does not include any reoccurring costs (they are calculated below
cov.cst$COV.comb <- rowSums(cov.cst[,5:8], na.rm=T)



################################################################################
# standard repeat costs
# Monitoring (to see if covenant is intact)
# repeat rate: 1/1 year
# rate used: NCC charge (from Management cost workshop)
mon <- 758
# Staff cost (work done to reply to Land owner request)
# repeat rate: 1/5 years
# rate used: NCC charge (from Management cost workshop)
# presented in yearly portion:
staff <- mon/5


################################################################################
################################################################################
##      MARXAN RUNS COMPLETE
##      START COVENANT UNCERTAINTY ANALYSIS
################################################################################
################################################################################


################################################################################
#Setup from Rissman
#Data from Rissman 2010 Fig. 1
inc.rate <- data.frame(year=c(seq(1989,2007,1)),
                issues=c(1,1,NA,1,NA,NA,2,1,NA,3,2,2,5,1,3,3,4,8,5))
fm2 <- glm(issues~year, data=inc.rate, family=quasi(link="log",variance="constant"))
fm2.lin <- lm(issues~year, data=inc.rate)

# for logistic growth
# Formula: N(t) = CC * N0 * exp(rr*t) / (CC + N0 * (exp(rr*t) - 1))
# growth rate rr
rr <- coef(fm2.lin)[[2]]
# Carrying Capacity CC
CC <- 50
# "Population" at year 2013
N0 <- predict(fm2.lin,newdata=data.frame(year=c(2013)))

#Data from Rissman 2010 Fig. 2
costs <- data.frame(value=c(5000,seq(10000,100000,10000),300000,400000),
                incidents=c(12,7,9,4,1,1,2,2,1,1,1,1,1))
```

```
fm1 <- nls(incidents ~ a*value^b, data=costs,start = list(a = 2555, b = -0.655))
sc <- coef(fm1)[[1]]
pw <- coef(fm1)[[2]]

f <- function(x) {
  return(sc*x^pw)
}

#dispute cost range
bins <- seq(1000,400000,500)
pred <- f(bins)

# length of segments
pred.1 <- pred/sum(pred)

#put pred.1 lengths on a vector between 0 and 1
pred.2 <- vector()
pred.2[1]<- pred.1[1]
for (ii in 2:length(pred.1)){
  pred.2[ii] <- pred.2[ii-1] + pred.1[ii]
}

#create lookup data.frame for pred.2 vector values that correspond to bins
disp.cost <- data.frame(prb=pred.2,cost=bins)

D1 <- DiscreteDistribution (supp = bins , prob = pred.1)
dD1 <- d(D1)  ## D1ensity function
pD1 <- p(D1)  ## D1istribution function
qD1 <- q(D1)  ## Quantile function
rD1 <- r(D1)  ## Random number generation

### End setup from Rissman 2010
#############################################################################

# check if all nreps runs met their target
summary(marx.acqu$sums[[2]]$MPM)
summary(marx.acqu$sums[[2]]$Shortfall)

# check acquisition reserve system cost
hist(marx.acqu$sums[[2]]$Cost)
summary(marx.acqu$sums[[2]]$Cost)
mean(marx.acqu$sums[[2]]$Cost)
sd(marx.acqu$sums[[2]]$Cost)

total.beta <- sum(cad$BETA_AWS)

runs <- marx.acqu$run
```

```
mv  <- marx.acqu$mv
sums <- marx.acqu$sums

years <- 100
nruns <- 100
#dispute rate: 2.8/1000 per year
# 2.8 is the average dispute rate from Rissman 2010
dispute <- 1 - (0.28/1000)

run.sum <- data.frame(ID=0,Init.cost=0,Cost.no.disp=0,Init.Beta=0)
run.sum[sprintf("cost.y%02d",seq(1,years,1))] <- 0
run.sum[sprintf("Beta.y%02d",seq(1,years,1))] <- 0

 for (rn in 2: (nruns + 1)){

   #extract polygon ID's that were selected by Marxan run
   #cad$ID[ex==1]
   #extract covenant costs that were selected by Marxan run
   #cov.cst$COV.comb[ex==1]
   #extract BETA values that were selected by Marxan run
   #cad$BETA_AWS[ex==1]

   #extract run to work with
   ex <- runs[rn]
   # cost framework for the covenants per run
   cov.frame <- data.frame(ID=cad$ID[ex==1], Init.cost=cov.cst$COV.comb[ex==1],
         Cost.no.disp=cov.cst$COV.comb[ex==1] + years * (mon + staff),
Init.Beta=cad$BETA_AWS[ex==1])
   cov.frame[sprintf("cost.y%02d",seq(1,years,1))] <- NA
   cov.frame[sprintf("Beta.y%02d",seq(1,years,1))] <- NA

   #setup cost and beta for year 1
   cov.frame[,5] <- cov.frame[,2]
   cov.frame[,5+years] <- cov.frame[,4]

   dum <- as.matrix(cov.frame)
   cov.frame <- spd (years, dum, dispute, rD1, pu, mon, staff)

   #summarize cov.frame rows
   #tempR <- sapply(cov.frame,sum)
   run.sum[rn-1,] <- sapply(cov.frame,sum)
   #run ID into run.sum data frame
   run.sum[rn-1,1] <- sprintf("run_%04d",rn-1)
   #tempR[1] <- sprintf("run_%04d",rn-1)
   #run.sum <- rbind(run.sum, tempR)
 }

#run.sum <- run.sum[-1,]
```

```
save.image("04.Covenant.analysis.all.runs.RM.const.low.1.RData")
################################################################################
################################################################################
#### Plot results (currently only for run 1 as an example
################################################################################
################################################################################
Cst.n.dsp <- seq(run.sum$Init.cost[1],run.sum$Cost.no.disp[1],length.out=years)

tt <- data.frame( Acqu.cst=rep(marx.acqu$sums[[2]]$Cost[1],years),
            Cst.n.dsp=Cst.n.dsp,
            y.cost=as.vector(t(run.sum[1,5:(4+years)])),
            y.biodiv=as.vector(t(run.sum[1,(5+years):(4+2*years)])),
            x= seq(1,years,1))
par(mfrow=c(2,1),cex=1,lwd=1)

# check distributions and how they look like
#use of lookup data.frame in an example
#cost <- vector()
#n <- 10000
#for (jj in 1: n){
#  cost[jj] <- rD1(1) #disp.cost$cost[which.min(disp.cost$prb < runif(1))]
#}
#hist(cost, main="Cost distribution RM.const.low",xlab="dispute cost [$]")
#dd<-summary(cost)
#legend("top", leg = paste(c(names(dd)), c(dd), sep = "="))


plot(tt$Acqu.cst~tt$x, type="l",ylim=c(min(tt$Cst.n.dsp),max(tt)),
    xlab="Years", ylab="Reserve system cost [$]",
    main="Covenant network cost over time RM.const.low (dispute rate: 2.8/10000 per year)",
    lwd=2)
lines(tt$Cst.n.dsp~tt$x, col="red",type="l",lwd=2)
lines(tt$y.cost~tt$x, col="green",type="l",lwd=2)
#lines(tt$y.biodiv~tt$x, col="blue",type="l",lwd=2)

ex.cs1 <- as.expression( c("Baseline: Fee Simple Acquisition Cost", "Covenant cost including
disputes",
                "Covenant cost, no disputes"))
utils::str(legend(-3, .9, ex.cs1, lty = 1, plot = FALSE,
        adj = c(0, 0.6)))  # adj y !
legend(1, max(tt)-max(tt)/10,
       ex.cs1, lty = c(1,1,1), pch= NA,col =c("black","green","red"),
       adj = c(0, 0.6), cex=1.2, lwd=2)

#Biodiv
plot((tt$y.biodiv/total.beta*100)~tt$x, #ylim=c(min(tt$Cst.n.dsp),max(tt)),
    xlab="Years", ylab="Reserve system Biodiversiy value",
    main="Covenant network Biodiversity value over time RM.const.low",
```

```
    col="blue",type="l",lwd=2)
#lines(tt$Cst.n.dsp~tt$x, col="red",type="l",lwd=2)
#lines(tt$y.cost~tt$x, col="green",type="l",lwd=2)
#lines(tt$y.biodiv~tt$x, col="blue",type="l",lwd=2)

#ex.cs1 <- as.expression(  c("Baseline: Fee Simple Acquisition Cost", "Covenant cost including
disputes",
#                    "Covenant cost, no disputes"))
#utils::str(legend(-3, .9, ex.cs1, lty = 1, plot = FALSE,
#         adj = c(0, 0.6)))  # adj y !
#legend(1, max(tt)-max(tt)/10,
#         ex.cs1, lty = c(1,1,1), pch= NA,col =c("black","green","red"),
#         adj = c(0, 0.6), cex=1.2, lwd=2)
par(mfrow=c(1,1))
```

Appendix 8: Expert elicitation introduction to rank 47 bird species.

1. Introduction

Thank you for taking time to complete our survey, which we believe will take 30-50 minutes. We hope the knowledge that you are helping to identify high quality terrestrial habitats for conservation in the Coastal Douglas-fir Zone (CDF, see figure) and our offer of a $100 honorarium makes your effort worthwhile.

We are mapping bird communities of interest to conservation in the CDF using 47 bird species maps based on thousands of `point counts` and expert opinion to associate species with habitat type and condition. Once assembled, these and other maps of native and exotic plant species will be used to identify old forest, woodland and wetland habitats within the CDF that are particularly likely to contribute positively to the persistence of diverse native bird and plant communities in future. Our finished maps will be available to land use planners and facilitate ongoing reserve design exercises aimed at maximizing the persistence of native species and ecosystems of the CDF Zone by identifying and conserving priority sites.

We are requesting your help because detailed information on the reliance of birds on the habitats occurring in the Coastal Douglas-fir zone are lacking for most species, but well-established in the collective experience of many dedicated birders and biologists. We are therefore reaching out to experienced birders to help us summarize this local knowledge by filling out the attached a survey. In return, we promise to gratefully acknowledge your help in all reports and publications we produce, to provide you with copies of those reports, and to send a $100 honorarium to the address you provide.

The accompanying Excel file asks you to rank bird species in terms of their broad 'reliance' (low, medium, high, or unknown) on the 10 habitat types of interest in the CDF zone. Although you will sometimes be uncertain about the correct response, we ask that you select the one that best represents your own experience in the region. At the end of the survey, please provide your contact information in the space provided so we can send you a honorarium.

Thank you very much for your help,

Richard Schuster and Peter Arcese

Centre for Applied Conservation Research
University of British Columbia
#3041 – 2424 Main Mall
Vancouver, BC V6T 1Z4

2. Habitat Reliance in Salish Sea Birds

Please read the following descriptions of habitats found in the CDF Biogeoclimatic Zone, and then indicate the degree to which you expect that each listed bird species 'relies' or 'uses' each habitat type during the spring and summer breeding period (~Apr-Jul) using the embedded, drop-down menus in the accompanying Excel file. To keep acquainted with the habitats described, we have also attached a 1-page 'key' which you may wish to print and have at your side as you complete the spreadsheet.

Herbaceous: Early successional or herbaceous communities maintained by environment or disturbance (e.g., flooding, grazing, fire, agriculture); dominated by forbs, graminoids, ferns. Invading or residual shrubs and trees may be present (tree cover < 10%, shrubs < 20%), time since disturbance < 20 years via forest succession or non-forested communities maintained in this stage.	
Shrub/Herb: Early successional or shrub communities maintained by environment or disturbance; dominated by shrubby vegetation that is <10m tall. Seedlings and advance regeneration may be abundant (tree cover < 10%, shrub cover > 20%), time since disturbance < 40 years via forest succession.	

Pole/Sapling: Trees > 10 m tall, typically densely stocked, have overtopped shrub and herbaceous layers; younger stands are vigorous (usually > 10-15 years old); older stagnated stands (up to 100 years old) are also included. Self-thinning and vertical structure not yet evident in the canopy, time since disturbance < 40 years via forest succession and up to 100+ years for dense stagnant stands.

Young Forest: Self-thinning has become evident and the forest canopy has begun to differentiate into distinct layers (dominant, main canopy, and overtopped); vigorous growth and a more open stand than in the Pole/Sapling stage, time since disturbance generally 40-80 years, depending on tree species and ecological conditions.

Mature Forest: Trees established after the last disturbance have matured; a second cycle of shade-tolerant trees may have become established; understories become well developed as the canopy opens up; time since disturbance generally 80-250 years.

Mature Forest (cont.)	
Old Forest: Older, structurally complex stands comprised shade-tolerant and regenerating tree species, but often including long-lived, older seral stage trees sometimes dominating the upper canopy. Snags and coarse woody debris in all stages of decomposition, understory of deciduous and regenerating confers typical. Shrub layer well-developed and dense, especially in light gaps. Time since last stand-replacing disturbance generally > 250 years.	

Rural/Agriculture: Rural areas are characterize by areas that have residences and other human development scattered and intermingled with forests, range, farm land, cultivated fields or native vegetation. Cultivated fields are non-forested, open areas that are subject to agricultural practices including plowing, fertilization, and non-native crop production which often results in long-term soil and vegetation changes. Although cultivated fields are typically agriculture based our definition includes other green spaces as well, including city parks, baseball fields, residential lawns, and golf courses.

Urban/Industrial: Urban units are characterized by an almost continuous covering over the landscape by residences and human development. Industrial areas dominated by industrial development namely, pulp and paper, lumber mills, oil/gas refineries and so on. These units are characterized by a high degree of ground disturbance; concrete parking lots, large commercial buildings, work yards and other specialized industry infrastructure.
Other urban/industrial areas include exposed soils (predominantly disturbance sites associated with new housing developments), Gravel pits, Mines, Railroad and Road surfaces.

Woodland: coniferous and mixed woodland (≤35% canopy; typically of Douglas fir, arbutus and oak), including and often inter-mixed with herbaceous and maritime meadow or shallow-soil, sparsely vegetated openings (herb-dominated ecosystems of southeastern Vancouver Island and many islands in the Strait of Georgia). In maritime and shallow soil GOE meadows, tree species are mostly or completely absent, but may include sparse cover of e.g., Garry Oak, Douglas-fir and shore pine, and shrubs in sparse to patchy accumulations.

For several reasons, we must exclude from our consideration 'deep soil' GOE meadows that would be expected to succeed relatively rapidly to continuous coniferous forest in the absence of disturbance sufficient to maintain them.

Wetland: Wetlands are characterized with a water table at, near, or above the surface, daily, seasonally or year-round. Soils in wetlands are water-saturated for enough periods that excess water and low soil oxygen concentrations create conditions necessary for water-tolerant plants to dominate. This unit includes floodplains, fens, swamps, marshes, bogs, shallow open water and estuaries.