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### **Disturbance and climate change risks to forest carbon sinks and potential management responses**

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# Disturbance and climate change risks to forest carbon sinks and potential management responses

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June 2018

A dissertation submitted to Bangor University  
in candidature for the degree Philosophiae Doctor

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## ***Summary***

Climate change is a global crisis facing forest management. There are risks to many ecological goods and services from forests due to changes in productivity, mortality, pathogen, and wildfire dynamics. Likewise, there are opportunities such as increases in productivity or targeted funding for climate adaptation and emission reductions. To manage those risks and opportunities, we need the fundamental knowledge of forest carbon (C) cycles. Overall, my dissertation aims to improve our understanding of forest carbon dynamics and how they may respond to natural disturbances, climate change and management activities. This purpose falls within the context of the need to adapt to and mitigate climate change for the ongoing provision of ecological goods and services from forest ecosystems such as timber and biodiversity.

The thesis starts with a critical analysis of six papers I have previously published. That chapter includes a synthesis of findings, critique of methods used, and identifies some areas for future research. Each subsequent chapter represents a contributing article.

The overall findings of this dissertation are *(i)* that although forests are often GHG sinks, moderate or high intensity natural disturbances can reverse that flux. *(ii)* That climate change effects on productivity may increase or decrease the natural sinks or even create emission sources in forests that may otherwise have been sinks. *(iii)* That management strategies to increase species diversity and resilience may be effective at reducing risks of emissions, but they must be assessed for individual ecosystems and may be insufficient to fully offset disturbance or climate change effects.

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## **1. Critical Analysis**

Caren C. Dymond

### *1.1. Summary*

Scientific knowledge of forest carbon cycles is fundamental to understanding and modelling the changing climate and for developing management responses. In this critical analysis I summarize six articles that have contributed to this body of knowledge and evaluate them within the context of the broader literature. In these articles we provide evidence that natural disturbances and tree productivity declines can contribute to increasing atmospheric greenhouse gas (GHG) concentrations by reducing biological CO<sub>2</sub> sinks and increasing GHG emissions from forests (Kurz et al. 2008a, b, Dymond et al. 2010, 2016). Furthermore, in Dymond et al. (2014) and Hof et al. (2017) we go on to argue that reducing those emissions risks may be possible through management efforts to increase resilience through tree species diversity.

The evaluation suggests that the earlier papers have tended to over-estimate the impact of natural disturbances on the GHG balance of forests. The more recent publications address some of these inaccuracies. The short time span in the early papers and omission of climate change impacts on productivity are also addressed in the two most recent papers. Future papers that assess forest management strategies may be strengthened if they considered other theoretical frameworks (e.g. vulnerability assessments) rather than just resilience theory. The evaluation noted that all six papers use a book-keeping or mechanistic approach to modelling forest carbon dynamics rather than an inversion modelling or other methodology. However, there does not appear to be a strong consensus in the literature for a particular protocol. The conclusion is that, despite their weaknesses, the articles submitted for this dissertation provide a meaningful contribution to the knowledge of forest GHG sinks and emissions, climate change impacts in forests, and response options for forest managers.

### *1.2. Introduction*

Climate change is the most important crisis facing forest management globally. Changes in productivity, mortality, pathogen, and wildfire dynamics threaten timber supplies,

biodiversity, and community stability. Furthermore, there are potential opportunities such as increases in productivity and targeted funding for climate adaptation and emission reductions. To reduce those risks, we need the fundamental knowledge of forest carbon (C) cycles. In this critical analysis, I summarize six articles that have contributed to this body of knowledge and evaluate them within the context of the broader literature (Kurz et al. 2008a, b, Dymond et al. 2010, 2014, 2016, Hof et al. 2017). Furthermore, I also assess them within the context of my research strategy which is: to provide scientific knowledge to forest managers and policy makers that will enable them to reduce C emissions or increase C sinks as part of the suite of ecological goods and services provided by forests.

I have structured this analysis into topics to show a progression of thought from the earlier papers that examined the C cycle, to later work that assessed management responses under climate change. The first topic focuses on natural disturbance impacts on forest C dynamics because these papers provided important contributions to the literature on the global C cycle. The second topic focuses on a paper which integrates forest C dynamics, climate change impacts on productivity, and C storage in wood products. This article demonstrates substantial methodological improvements over the older papers and the importance of uncertainty in future tree productivity to the GHG balance of temperate coniferous forests. Given the relatively recent publication year (2016) the evaluation centers on modelling limitations. The third topic focuses on management strategies because these papers ultimately achieve my research strategy stated above and add significantly to the literature. Resilience theory provides the context for these papers, which I evaluated as case studies. The fourth topic focuses on evaluating the modelling methods underlying all six papers reviewed and discusses why there was a shift in C modelling between the older three papers and the newest two articles. The fifth topic focuses on some potential future research needs to address my research strategy and improve the modelling. The sixth topic offers some conclusions.

### *1.3. Natural Disturbance Impacts on C*

Three of the papers submitted for this dissertation quantified the impact and importance of natural disturbances on the greenhouse gas (GHG) balance of boreal and temperate forests in Canada (Kurz et al. 2008a, b, Dymond et al. 2010). The contribution of a forest to the

atmospheric concentration of GHG is the difference between: the uptake of CO<sub>2</sub> by photosynthesis, and the release of CO<sub>2</sub> by respiration, natural and anthropogenic disturbances, and forest products whether through combustion or decomposition (McGuire et al. 2001, Kurz et al. 2013). Significantly reduced rates of CO<sub>2</sub> sequestration can occur as a result of natural disturbances because they generally have immediate, negative impacts by reducing tree growth rates or causing mortality (Conard and Ivanova 1997, Nepstad et al. 1999). In addition, disturbances will cause increases in emissions in the short-term due to increased decay or combustion of dead organic matter.

Research on natural disturbance impacts on carbon show a wide distribution of results depending on drivers in the study area. These drivers typically include wildfires and storms which cause stand-replacing mortality but have recently begun to include insect outbreaks. Comparing against subsequent publications, the evaluation of the first three papers in this dissertation includes lessons learned regarding: how to set up simulations of the future, limitations of the modelling, and the assumptions of disturbance intensities.

#### 1.3.1. Relative importance of natural disturbances

In Kurz et al. (2008a) we assessed the relative importance of natural disturbances to the future risks of net GHG emissions from the managed forests of Canada (2.4 million km<sup>2</sup>). We integrated prior knowledge from Kurz and Apps (1992), Kurz and Apps (1999) and Amiro et al (2001) on management, wildfire and insect impacts on forest C stocks and GHG fluxes with new, probabilistic models of disturbance size and frequency, and improved estimates of pest disturbance impacts. Consequently, our results showed a 100% probability of the forests of Canada being a net source of GHG emissions due to insects and fires, over the period 2006-2022 (the end of the analysis). This national-scale analysis included the regional insect-outbreaks reported in Kurz et al. (2008b) and Dymond et al. (2010) described below.

These results differ from a study completed by Pilli et al. (2016) where the natural disturbance impacts on the GHG balance of the EU forests were minor (up to approximately 12 MtCO<sub>2</sub> yr<sup>-1</sup>) compared with the overall large net sink (approximately -450 to -325 MtCO<sub>2</sub> yr<sup>-1</sup>) and the uncertainty between different models. The main effect of natural disturbances was to substantially lower biomass stocks regionally when storms caused direct mortality.

The main drivers of the GHG balance in the EU are tree growth rate, the fuelwood harvest rate ( $\text{m}^3$  per year), and changes in land-use.

Similarly, both natural and anthropogenic disturbances were included in a sensitivity assessment of the forest GHG balance in the Yucatan peninsula of Mexico (Mascorro et al. 2015). Their analysis used the same modelling framework as [Kurz et al. \(2008a\)](#) and Pili et al. (2016). Predictably, the scenario with the lowest disturbance rate had the largest  $\text{CO}_2$  sinks compared to the one with the highest disturbance rate, however the large range was unexpected, a difference of over 112  $\text{MtCO}_2\text{e}$  cumulative from 2002 to 2010 for an area of 3.2 million ha (Mascorro et al. 2015). Although the sensitivity of the forest GHG balance to disturbances is much higher in the Yucatan than in the EU, the main drivers are similar: tree harvesting and changes in land-use.

### 1.3.2. Importance of insects

In [Kurz et al. \(2008b\)](#) we provide the first assessment of the importance of an insect outbreak on forest C dynamics and the resulting positive feedback process to climate change. Our results indicated that the resulting tree mortality from a bark beetle outbreak was substantial enough to convert the forests from a natural  $\text{CO}_2$  sink to a net GHG emitter over a large forested area ( $374,000 \text{ km}^2$ ). Within months of attacking its host, the mountain pine beetle (*Dendroctonus ponderosae*) causes tree mortality. The widespread and high intensity of the outbreak caused a switch from C sink to source. Specifically, in the scenario that included the beetle population outbreak and the forest management response of increasing the short-term harvest rate ( $\text{m}^3$  per year), the forests were a net source of emissions annually from 2001 to 2020 (the end of the simulation). (These emissions included the assumption of instantaneous oxidation of harvested wood). Without the beetle or increased harvest rate, the forests would have only been a source from 2003 to 2006 due to wildfires.

Although the relatively high importance of the mountain pine beetle to the C cycle was unknown, insects and diseases have a long-documented impact on ecosystem function, above-ground biomass and specifically timber volume. Moreover, recent analyses have specifically focused on GHG fluxes (Hicke et al. 2012). Their review found that net ecosystem productivity decreases immediately following insect attacks and diseases outbreaks, with

some studies reporting a switch from being a net C sink to a net C source to the atmosphere. This GHG balance may persist for a long time afterwards while emissions from dead organic matter decomposition exceed CO<sub>2</sub> sinks in tree growth. The degree of impact varies widely and recovery times to C sink range from a year or two (Brown et al. 2012) to decades (Peters et al 2017). The mountain pine beetle outbreak plus increased harvest intensity on the standing forest were severe enough to appreciably reduce the net ecosystem exchange of the entire managed forest of Canada (2.4 million km<sup>2</sup>) (Stinson et al. 2011). However, the knowledge gaps on growth reductions and effectiveness of pest and pathogen management efforts remain substantial (Hicke et al. 2012).

Unlike [Kurz et al. \(2008b\)](#), who modelled an insect (*D. ponderosae*) that directly causes tree mortality, [Dymond et al. \(2010\)](#) modelled a defoliator, spruce budworm (*Choristoneura fumiferana* Clem.). [Dymond et al. \(2010\)](#) established the model parameters based on intensive empirical data of the effects of spruce budworm on growth and mortality rates in four different host types (a combination of species and age class). The *C. fumiferana* outbreak was projected to shift the study area in southern boreal forest of Quebec, Canada from a net C sink to a net C source from 2011–2024 (the end of the simulation). This result was surprising because of the lower intensity of the *C. fumiferana* compared to the mountain pine beetle or wildfire. In terms of volume losses, the impacts of the *C. fumiferana* in our projection were lower than values from the literature of previous outbreaks. We speculated that changes in tree species composition and age distribution may have occurred in the study area between the time of the historical outbreak and the forest inventory we used to project the future. It is unclear if the change in host distribution would have been natural, resulting from historical outbreaks preferentially damaging one type of host (which is supported by empirical data), or natural combined with deliberate management action.

### 1.3.3. Evaluating the projections

We can evaluate the projections in [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#) now that some of the period has been observed. We can assess the projections from 2006 to 2014 against new results produced using the same modelling framework but with empirical data on harvest (m<sup>3</sup> per year), planting (ha per year) and natural disturbances (ha per year). [Kurz et al. \(2008a\)](#) projected that all years from 2006 to 2014 would be net GHG sources for the



managed forest of Canada. However, Metsaranta et al. (2017) reported that only six of those nine years were estimated as net GHG sources for the same study area. The scale of the GHG sources reported in Metsaranta et al. (2017) was in the order of  $-70 \text{ MtCO}_2\text{e yr}^{-1}$  which is consistent with the 30-60% percentile band in Fig. 2.1. The three years of net sinks reported in Metsaranta et al. (2017) that were not anticipated in [Kurz et al. \(2008a\)](#) are likely to have occurred because the mountain pine beetle outbreak collapsed quicker than anticipated in our projections. The peak of beetle-infested area actually occurred in 2006 rather than in 2008 as we assumed in the [Kurz et al. \(2008a, b\)](#) projections (BC Gov. 2017a). Moreover, by 2009 the outbreak area was surveyed at  $44,000 \text{ km}^2$  in contrast to the area of  $70,000 \text{ km}^2$  in the earlier projections. Further contributing to the unexpected sink that occurred in three of the years was the lack of a *C. fumiferana* outbreak in central Canada. [Kurz et al. \(2008a\)](#) and [Dymond et al. \(2010\)](#) assumed that the *C. fumiferana* population dynamics would follow its historical return interval and develop during 2012-2016, but no new outbreak has occurred as of 2016 (NFDP 2017). In retrospect, we should have conducted a sensitivity analysis of the GHG fluxes without natural disturbances and included more conservative assumptions regarding the insect outbreaks.

#### 1.3.4. Evaluating modelling limitations

Limitations of [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#) methods include the short time-span of the projections, little incorporation of climate change impacts, and the assumption that harvested wood would instantaneously oxidize.

Time is a crucial factor when modelling forest C dynamics because tree life spans and slow decay rates result in the C dynamics in any year being temporally autocorrelated to past years, decades or even centuries. The three papers describing natural disturbance impacts on GHG emissions ([Kurz et al. 2008a, b](#), [Dymond et al. 2010](#)) only reported on short-time frames (20 to 25 years) relative to tree life spans (typically 200 – 400 years in British Columbia). In these papers we relied on historical empirical data for tree productivity, decay rates, and natural disturbance frequency and intensity with the exception of the mountain pine beetle. Before we began our projects, the beetle outbreak was already many-times the scale of historical outbreaks. This was due in part to warmer temperatures associated with a changing climate (Taylor et al. 2006). Our projection of the beetle outbreak was based

largely on the availability and spatial contiguity of host trees and was not constrained by historical data ([Kurz et al. 2008b](#)). The uncertainty caused by not including climate change effects on productivity, decay or other natural disturbances meant our confidence in longer simulations was low. Longer simulations may be justified in a journal article with accompanying caveats, but in our experience, too many caveats render science irrelevant to policy makers or forest managers. The shorter time spans in [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#) were highly relevant to climate change mitigation policy, in particular for Canada to decide whether to include forests under its Kyoto Protocol commitments.

One of the implications of the short time-frames is that the forests remained a net GHG source throughout the simulation periods in [Kurz et al. \(2008b\)](#) and [Dymond et al. \(2010\)](#). However, it is logical that the forests could potentially recover their C sink status after the insect outbreaks ended. Our subsequent modelling study of British Columbia forest C dynamics and GHG fluxes incorporated plausible climate change effects on tree productivity, decay rates and wildfire over a longer simulation period ([Metsaranta et al. 2011](#)). We projected a return to a net GHG sink as soon as 2020 under the most optimistic climate change scenario, or not at all before 2080 for the most pessimistic scenario. (This study assumed 50% storage of harvested C in wood products).

The lack of comprehensive climate change effects is a major limitation to [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#) because future frequency and intensity of natural disturbance events are expected to increase globally ([Dale et al. 2001](#), [Allen et al. 2010](#)). However, the net effect on forest GHG fluxes will vary locally and regionally with differences in the disturbance events themselves ([Boulanger et al. 2017](#)), potential increases in productivity ([Gonsamo et al. 2017](#)), disturbances alleviating constraints on productivity ([Reyer et al. 2017](#)), as well as the effect of other ecosystem and human responses ([Martin and Watson 2016](#), [Seidl et al. 2016](#)). Therefore, our more recent studies ([Dymond et al. 2016](#), and [Hof et al. 2017](#)) incorporate climate change effects and make longer projections (until 2114 in the latter paper). This is still short relative to tree life spans (typically 200 – 400 years) and stand-replacing fire return intervals (from 100 to 200 years) in British Columbia, but uncertainties increase over time and longer-term projections would go well beyond those of the current Global Circulation Model projections.

In addition to most climate change effects, storage of forest C in products was also not included in [Kurz et al. \(2008a, b\)](#) or [Dymond et al. \(2010\)](#). Storage and emissions of forest C in the manufacturing of wood products, during their use, and in disposal can greatly affect the net C balance of commercially managed forests. For example, one study estimated that assuming instantaneous emission of logged C over-estimated British Columbia's forest GHG balance by as much as 180% (Dymond 2012). Our studies, [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#), assessed only the GHG or C balance in the forest itself, essentially assuming instantaneous oxidation of all the C in harvested wood. Because these studies focus on quantifying the impact of natural disturbances, this simplifying assumption had less of an impact than it would for studies specifically evaluating mitigation or adaptation strategies. However, since this assumption limits the value of the studies as a source of evidence for forestry policy-making, I developed and parameterized a model of the forest product manufacturing sector in British Columbia including updating model parameters for wood markets and disposal in the US and Canada (Dymond 2012).

#### 1.3.5. Evaluating disturbance impacts

The conclusion of [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#) that natural disturbances play key roles in GHG balances of some forests hinge on the parameterization of the intensity of wildfires and the models ability to simulate the recovery dynamics following an insect outbreak.

A major driver of inter-annual variability of forest C fluxes in Canada and globally are wildfires (Kasischke et al. 2005, Stinson et al. 2011). The wildfire modelling in the three early papers submitted as part of this dissertation ([Kurz et al. 2008a, b](#), [Dymond et al. 2010](#)) relied on default parameters from the Carbon Budget Model of the Canadian Forest Sector v. 3 (CBM-CFS3) (Kurz et al. 2009). However, my subsequent analysis (unpublished) has shown that these parameter values led to emissions that are too high for British Columbia. In the British Columbia Provincial Greenhouse Gas Inventory report, which relies on the same model, the average C emission due to fire is 54 tC ha<sup>-1</sup> (1990-2008 data from BC Gov. 2017b). In contrast, a detailed empirical study of similar, coniferous temperate forests after a fire found that its fuel consumption rate ranged from 19 to 30 tC ha<sup>-1</sup> (Campbell et al. 2007). Kasischke et al. (2005) reviewed boreal wildfire emission estimates and found the

range was from 11 to 28 tC ha<sup>-1</sup> (annual average by severity class for the boreal region 1992-1995-2003, excluding deep peatland fires). A more recent study of fuel consumption in Alaskan fires estimated on an annual basis, 30 tC ha<sup>-1</sup> under drier conditions and 17 tC ha<sup>-1</sup> under cooler and moister conditions from 2006-08 (Kasischke and Hoy 2012). As a result of the availability of data to parameterize a range of fire severities, when setting up the LANDIS-II model for our more recent papers we used the fire consumption data from Campbell et al. (2007) for parameterization. When additional data become available, or when different forest types are included in the analysis (Ghimire et al. 2012), we will reassess this parameterization. The implication of over-estimating wildfire impacts is that those emissions can outweigh the effect of management activities intended to increase CO<sub>2</sub> sinks. This could create a disincentive to the adoption of mitigation activities. From a purely scientific perspective, over-estimating the emissions due to wildfire would lead to over-estimation of tree growth rates in a modelling framework that is calibrated against forest C stocks.

In addition to overestimating the disturbance impacts of wildfire, the models of Kurz et al. (2008a, b) and Dymond et al. (2014) (described below) are also likely to have overestimated the impact of mountain pine beetle outbreaks because the models only simulated single-age cohorts - the current forest canopy trees. In contrast, subsequent studies have shown that non-canopy tree cohorts and understory vegetation can have increased growth rates due to less competition from the beetle-killed individuals (Brown et al. 2012, Alfaro et al. 2015, Campbell and Antos 2015). Our more recent papers used the Forest Carbon Succession Extension (ForCSv2) to LANDIS-II, which includes multi-species and multi-age cohorts, to simulate that release mechanism (Hof et al. 2017). As with fire, model outputs that over-estimate the impact of insect outbreaks may act as a disincentive for mitigation activities. On the other hand, if management activities are effective at reducing insect impacts, but those potential impacts were over-estimated to begin with, resources could end up being wasted on unnecessary mitigation activities. A range of intensities could be incorporated as part of an uncertainty analysis.

#### 1.4. Climate Change Effects on Productivity

In the 1980s and 1990s, climate change increased forest productivity globally (Boisvenue and Running 2006). However, regionally productivity only increased in areas where water was not a limiting factor. Projecting future changes of productivity is complicated by the uncertainty in climate predictions, including in emissions scenarios, extreme weather events, precipitation events, Wieder et al. 2015), the variability and uncertainty in species and provenance responses to, or forcing of, changing climatic factors (e.g. Bonan 2008, Martínez-Sancho et al 2017), the non-climatic limitations on productivity such as nutrient availability (e.g. Norby et al. 2010), and a lack of data to parameterize and run models (Carpenter 2015). Nevertheless, these projections are needed to inform management strategies for both mitigation and adaptation (Lindner et al. 2010, Keenan 2015).

In Dymond et al. (2016) we estimated the landscape-scale C impacts of climate-induced productivity changes. The results indicate that by 2050 forests that are currently cold and wet will have increased productivity, become larger C sinks and store more C with than without climate change. However, forests where productivity is already water-limited are at risk of becoming net C sources by 2050. This paper synthesized method improvements over the four earlier papers submitted as part of this dissertation including: changing productivity and decay rates over time due to climate change, spatially explicit modelling of multi-aged and multi-species stands, and including C stored in products in estimating the C emissions (more detail is provided in the section below on evaluating the overall modelling methods). To my knowledge, this was the first paper to include C stored in forest products within an assessment of climate change effects on forests. In addition, in Dymond et al. (2016) we defined a new C metric: Net Sector Productivity (NSP):

$$NSP = NPP - (Rh + E_d + E_p) \quad (Eq. 1)$$

where NPP is net primary productivity, Rh is heterotrophic respiration, E<sub>d</sub> is direct disturbance emissions and E<sub>p</sub> is emissions from wood products.

##### 1.4.1. Evaluating modelling limitations

Due to time limitations, we did not represent the mountain pine beetle in Dymond et al. (2016), despite a recent outbreak in the study area in north-western British Columbia. I

don't think this was a major problem with the study given the wide variability in productivity rates that we tested, which likely encompassed the effects of this outbreak. We chose the use of an ensemble approach to climate scenarios, rather than selecting individual climate models with specific emissions scenarios. The strength of an ensemble approach is that it represents the average and standard deviation of different climate models' output. When separate climate futures are analyzed, the obvious question is, which one is more likely? In this study area, in a location transitional between coastal and continental climates, we have no credible answer to that question of probabilities. Other modelling limitations are discussed below in the section on evaluating overall methods for modelling C.

### *1.5. Management Strategies to Mitigate through Adaptation*

My primary concern with the analyses of [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#) was that we did not assess management options to reduce the impact of natural disturbances on C stocks or emissions. This was not unusual in the literature at that time, in which the question of management response to the risks from climate change was only examined theoretically. For decades, authors have argued for various management strategies to mitigate or adapt to climate change but provided little quantification of potential benefits, estimates of costs, or potential trade-offs (Layser 1980, Englin and Callaway 1993, Noss 2003). The main exception is assisted migration, which has had some modelling of effectiveness (O'Neill et al. 2008a). As a result of my concerns, [Dymond et al. \(2014\)](#) and [Hof et al. \(2017\)](#) explicitly evaluated management strategies.

Forests and many other ecosystems may be net sources or net sinks of CO<sub>2</sub> in exchange with the atmosphere. This means that there is the potential to reduce emissions, increase uptake, or better maintain C storage (Smyth et al. 2014). However, as shown by [Kurz et al. \(2008b\)](#), an increased intensity and area of natural disturbances due to climate change could sabotage such management efforts. Furthermore, C storage is not the only, or even the most important, ecological good or service provided by our forests, managers are likely to be more interested in timber production, wildlife conservation, regulation of water quality and fisheries, etc. Therefore, my research program has focused on assessing strategies that can help adapt forests to climate change, and in doing so also mitigate

climate change by reducing emissions, increasing sinks, or maintaining C storage at the same time.

#### 1.5.1. Resilience and the insurance hypothesis

In the studies cited above on natural disturbances and climate change effects, a recurring theme was uncertainty. One field of thought for managing ecological goods and services in the context of an uncertain future is to increase the resilience of forests through innovative management (Campbell et al. 2009). Unfortunately, the term resilience has a spectrum of scientific and common-language definitions, leading to confusion and ambiguity (Fisichelli et al. 2015). Resilience is defined in my work as *“the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks”* (Walker et al. 2004). For example, a conifer forest attacked by an insect will recover to be a conifer forest again if resilient, rather than, perhaps, transforming to a savanna-type ecosystem if not resilient. This is the definition often used for ecosystem studies and it incorporates the self-organizing qualities of complex systems and the potential for multiple stable states (Folke 2006). An earlier definition of resilience, now often called “engineering resilience” focuses on recovery time to a single stable state and resistance to the disturbance. The latter definition was rejected by Campbell et al. (2009) for managing BC’s forest ecosystems in a changing climate. This decision was based on the recognition of the profound uncertainties that global change has brought to our ecosystems and the reduced predictability of management outcomes. Ecosystem functions of forests may persist after a disturbance, but not necessarily in the same stable-state as before. Similar reasoning was used by Reyer et al. (2015) in their review of forest resilience and tipping points. A third definition is social-ecological resilience: *“(1) the amount of disturbance a system can absorb and still remain within the same state or domain of attraction, (2) the degree to which the system is capable of self-organization (versus lack of organization, or organization forced by external factors), and (3) the degree to which the system can build and increase the capacity for learning and adaptation”* (Folke 2006). The inclusion of adaptability and socio-economic components, in addition to the potential for multiple ecological steady states, makes this concept especially appealing for managed forest systems.

One way to increase resilience that is supported by ecological theory is by increasing species diversity. The insurance hypothesis poses that greater species diversity reduces the impacts of disturbances on the ecosystem as a whole while increasing and buffering productivity (Yachi and Loreau 1999). The proposed mechanism is that having a greater variety of species will result in a smaller proportion of the stand or landscape being susceptible to the impact of disturbance, such as a host species for a pathogen. This hypothesis assumes that different species are functionally different pieces of the ecosystem, rather than occupying the same niche, although this is not always true (Paquette and Messier 2011). For example, Poorter et al. (2015) document a clear positive effect of functional diversity on neotropical forest C stocks. The introduction to [Dymond et al. \(2014\)](#) provides more context from the literature. Colloquially, this is known as bet-hedging, and diversification of investments is a common strategy for reducing risks to peoples' savings (as modelled by economists through portfolio theory).

Whether considering biological or financial risks, logic indicates that there must be limitations to the diversification that can be achieved through management efforts and the resilience that diversification can provide. Foremost, before any management strategies to increase resilience are likely to be implemented, there will need to be an economic analysis of trade-offs between costs and benefits over the short and long term. For example, the economics of harvesting may depend on the dominance of a single species, either because it is the least costly to log or because it has the highest value. Furthermore, if a forest is already highly diverse a saturation-effect can occur (Poorter et al. 2015), implying that management actions that increase diversity will likely have limited ability to create a meaningful change to ecosystem function. It is also possible that management efforts that have a low impact on the landscape may have limited ability to affect diversity or resilience. For example, in [Dymond et al. \(2014\)](#) we were able to show that management actions on 0.3% to 1% of the forest annually was sufficient to increase or decrease the Berger Parker Dominance index over 80 years, so affecting 56%–62% of the forest cumulatively within the model. A lower harvest rate ( $\text{m}^3$  per year) or a reliance on more natural regeneration may not be as effective. Even if diversity and resilience can be altered in simulation modelling, those strategies may not be implementable given the potential tradeoffs between different forest values such as economics, recreation, wildlife habitat etc. For example, conservation



areas are unlikely to accept logging activities just so a more resilient species mix can be planted.

In [Dymond et al. \(2014\)](#) and [Hof et al. \(2017\)](#) we have assessed different management objectives to learn more about applying resilience theory to forest management. These objectives were: to reduce the susceptibility of landscapes to the mountain pine beetle and assisted migration for climate change adaptation.

#### 1.5.2. Reducing landscape susceptibility

In [Dymond et al. \(2014\)](#) we reported on a historical retrospective approach to model various management regimes and assessed if they could increase ecological and economic resilience to a pest outbreak. This was one of the first studies to conduct this kind of assessment at a landscape scale (cf. Temperli et al. 2012). We applied a regime to increase the proportion of harvest volume captured from the most dominant tree species (*Pinus contorta*) early in the simulation period. Furthermore, we modelled the planting of a greater diversity of species (in both richness and evenness) and allocated a greater proportion of the area to natural regeneration (EMR strategy). This regime resulted in greater ecological and economic resilience compared to business-as-usual as indicated by the higher Shannon Diversity Index, higher post-beetle outbreak growing stock ( $m^3$ ), higher post-beetle outbreak harvest rates ( $m^3$  per year), and higher, more consistent net revenue over time. Assuming that the growing stock represents the above-ground C stocks, the EMR regime was also the best for maintaining C stocks in the landscape. Therefore, the EMR regime was more effective for both mitigation and adaptation to climate change compared with business-as-usual. In [Dymond et al. \(2014\)](#) we also included a cost-benefit analysis. We found that accelerating the logging of the most abundant, but lower value trees (*Pinus contorta*) created lower revenue in the first decade, and that reforesting mixed species increased costs, but those costs were offset due to the availability of more valuable species and higher harvest rates after the beetle outbreak. A sensitivity analysis indicated a high level of robustness in the results. Our study showed that, despite the common assumption that adaptation will increase costs, economic viability may benefit from reducing forest health risks.

In [Dymond et al. \(2014\)](#) we identified some limitations on management's ability to affect diversity and resilience. The mixed planting regime, which had the same harvest variables as

the business-as-usual strategy but higher diversity in planted tree species, did not increase the landscape's resilience to the beetle outbreak as indicated by lower post-beetle growing stock ( $\text{m}^3$ ) and harvest rates ( $\text{m}^3$  per year), and increased costs compared to both the EMR regime and business-as-usual. In contrast, a companion study with similar management strategies but a different landscape assessed that mixed planting did increase ecological and economic resilience to the beetle outbreak compared with business-as-usual (Dymond et al. 2015).

### 1.5.3. Assisted migration

Assisted migration is a controversial strategy for adapting to climate change in a commercial forestry context (Aubin et al. 2011). The general concept in forestry is to move species or provenances (locally-adapted populations) to locations that will be likely to suit them better in the future, under the assumption that natural migration or evolution will be too slow to maintain productivity, timber supply, biodiversity or other ecological goods and services in forests (Pedlar et al. 2012). The designation of provenances and evidence from 'common garden experiments' form the basis of seed-use practices across Canada and parts of the USA (Michigan Forest Division 1957) and Europe (Bärring et al. 2017). These practices have traditionally required the availability of geographically-local source populations for seeds used in plantations to maximize productivity. It appears that the rate of climate change will outpace the rate of evolution in trees, resulting in potentially significant shifts in forest productivity (Wang et al. 2006, Aitken et al. 2008, O'Neill et al. 2008a). Despite any potential controversy, assisted migration has been embraced by the Canadian provinces that regulate seed selection for the majority of commercial forestry in the country (Pedlar et al. 2011, O'Neill et al. 2017) and research has started elsewhere, including Europe (e.g. Bärring et al. 2017). For the most part, the Canadian regulatory changes only allow seed to be moved fairly short distances, with a focus on matching provenances to future climates and maintaining timber supply.

Hof et al. (2017) assessed potential management regimes to increase resilience in C dynamics, species diversity and harvest rates ( $\text{m}^3$  per year). The novelty of this work is that it brought together modelling advances made by Dymond et al. (2016) for estimating future climate change effects with adaptive planting regimes (either increased use of locally-native

species or range expansion of species from elsewhere in British Columbia). We found that future precipitation rates, rather than temperatures drove the forest C dynamics. This conclusion resulted from the precipitation increasing or decreasing depending on the climate scenario, creating quite different consequences for productivity. Decreasing precipitation while temperature increases would likely reduce productivity in ecosystems that are already water-limited. Examining management regimes, we found that the current planting standards for species led to relatively low levels of resilience. In contrast, planting a greater diversity of tree species (compare to current standards) was generally beneficial for increasing the resilience indicators of C stocks and fluxes, NPP and tree species diversity, but had lower harvest rates (m<sup>3</sup> per year). We think that this is an example of the insurance hypothesis, where no matter which climate scenario is simulated, one or more of the tree species in the mixture will thrive. The second-best planting regime added *Pinus contorta* to the stocking standard in three ecoregions. However, that species is not currently planted in those habitats because of high snowfall damage of seedlings, which was not modelled. In addition, *P. contorta* is susceptible to a high number of insect and pathogen species in British Columbia. We conclude that, in the north-west of British Columbia, increased GHG emissions from forests may be counteracted to some extent by increasing the tree species diversity. An economic assessment of the different planting regime options is underway. Because this is a recent publication, I do not add to the comparisons with other studies presented in its Discussion.

#### 1.5.4. Evaluating relevance to resilience theory

Together, the three case studies ([Dymond et al. 2014](#) 2015, [Hof et al. 2017](#)) contribute to identifying the ecosystems and situations where evidence supports the insurance hypothesis and resilience theory (Table 1.1). More importantly, they also identify some cases where the theory is not supported by the results of the simulations. [Dymond et al. \(2014\)](#) provided a case study of a landscape dominated by *P. contorta* (75% of standing volume in 1980) where the business-as-usual management regime increased that dominance. Our results showed that under these conditions, management regime that aim to increase resilience can increase the diversity (Shannon Index), however not necessarily the resilience to a pest outbreak in the short-term. The regime to only plant mixed species but not change the harvest regime resulted in lower levels of ecological and economic

indicators than business-as-usual. In part this is due to the pest outbreak occurring only two decades into the simulation.

In Dymond et al. (2015) (a companion study to [Dymond et al. 2014](#)), we analyzed a more diverse landscape at the start of the simulation and had a business-as-usual approach that maintained more of that diversity. The results of this second paper showed that alternative management regimes did not clearly affect the Shannon Diversity Index until very late in the simulation. Despite the small response of the diversity index, the most aggressive management regime (EMR) was able to increase other resilience indicators. Another difference between the two studies was the economic viability of the regime to plant mixed species: this produced the lowest values in the first study, but similar values to business-as-usual in the second. Dymond et al. (2015) was not submitted as part of this dissertation in the interest of being succinct.

The third case study testing resilience and diversity theories analyzed a different landscape and assessed strategies against reduced productivity due to climate change rather than a pest ([Hof et al. 2017](#)). Nonetheless, the results show that management activities designed to increase tree species diversity were effective in this low diversity landscape, and could provide some resilience to climate change effects, particularly on C fluxes and storage, but there is potentially a trade-off with lower harvest rates.

*Table 1.1. Resilience indicator outcomes after 50-year simulations of management regimes aimed at increasing resilience relative to business-as-usual (BAU) from three landscapes in British Columbia. Outcomes are relative to BAU.*

Management regime	Shannon Diversity Index	Carbon stocks <sup>1</sup>	Carbon fluxes <sup>2</sup>	Harvest rate	Net Present Value <sup>3</sup>	Net Annual Revenue	Context
<u>Dymond et al. (2014)</u> Mixed planting	Higher	Lower	Lower	Lower	Lower	Lower	Low diversity landscape, beetle outbreak after 20 years
<u>Dymond et al. (2014)</u> EMR <sup>4</sup>	Higher	Higher	Larger sinks or smaller sources	Higher	Higher	Higher	Low diversity landscape, beetle outbreak after 20 years
Dymond et al. (2015) Mixed planting	Similar	Similar	Similar	Similar	Higher	Similar	Moderate diversity landscape, beetle outbreak after 20 years
Dymond et al. (2015) EMR <sup>4</sup>	Similar	Higher	Larger sinks or smaller sources	Higher	Higher	Higher	Moderate diversity landscape, beetle outbreak after 20 years
<u>Hof et al. (2017)</u> Diversification	Higher	Higher	Larger sinks or smaller sources	Lower	N/A	N/A	Low diversity landscape, 4 different climate scenarios

<sup>1</sup> Carbon stocks inferred from timber growing stocks for Dymond et al. (2014, 2015).

<sup>2</sup> Carbon fluxes inferred from growing stock changes for Dymond et al. (2014, 2015).

<sup>3</sup> Over 80 years.

<sup>4</sup> EMR (Early pine cut, mixed planting, and increased natural regeneration)

#### 1.5.5. Evaluating management regime approach

Dymond et al. (2014) and Hof et al. (2017) take the approach that we can make pre-determined assumptions about different management regimes and then assess their trade-offs through simulation modelling. This approach is one of many possible strategies.

Harvesting and the associated planting of seedlings are the main forest management activities in British Columbia. However, the six papers did not include dynamic harvest rates that respond to natural disturbances other than the surge in harvest in response to the mountain pine beetle outbreak of 1999-2015. Studies have shown that optimum harvest levels are higher when the modelling of natural disturbances is based on deterministic assumptions of annual average values when compared with stochastic representations of disturbances (Boychuck and Martell 1996, Daniel et al. 2017). Therefore, in our studies the implication is that the modelled harvest rates ( $\text{m}^3$  per year) may be overly optimistic. Because of this over-estimate, the effectiveness of strategies designed to increase resilience through the reduction of natural disturbance impacts may be underestimated with regards to timber supply and economic values. If we used an agent-based model instead, the management decisions would be an emergent property of the criteria and behaviors available to the simulated forest manager and the interactions between those managers and the ecosystem model (Rammer and Seidl 2015). Those results could potentially give a more realistic representation of the socio-ecological system of forest management where variables and parameters are available. Similarly, an optimization model that included climate change effects would allow us to evaluate competing or complimentary forest management strategies in a way consistent with traditional forestry practice (Lagergren and Jönsson 2017). However, both of these approaches do not specifically address the uncertainty that led me to focus on resilience as a management outcome.

Formal risk assessments which incorporate probabilities of different outcomes are specifically designed to accommodate uncertainty. For example, the definition of risk in the Climate Change Risk Assessment for the Forestry Sector of the UK states “*Combines the likelihood an event will occur with the magnitude of its outcome*” (Moffat et al. 2012). This quantitative approach to assessing risk can be very useful where relationships between the climate and the aspects of the forestry sector are well established. For example, Moffat et

al. (2012) report a likely 8% (range 1 to 19%) increase in yield loss due to drought in Wales by the 2080s. However, a careful reading of the text indicates that the indicators are somewhat less certain than the range may show, as they do not include potential increases in productivity due to warmer temperatures or improved water-use efficiency due to increase in atmospheric CO<sub>2</sub> concentration. These kinds of risk assessments are valuable in identifying vulnerabilities to climate change and potential tactics for adaptation (Raymond et al. 2014). However, further analyses are required to assess the effectiveness of different tactics and the potential trade-offs between different socio-ecological values. For example, Crowe and Parker (2008) used portfolio theory from financial asset markets to quantify the potential reduction of risk to growth rates by combinations of seed sources. A fundamental problem with such a detailed, quantitative approach is that, unfortunately, climate projections are more heuristic than probabilistic (Keenan 2015).

### *1.6. Evaluating Overall Approaches to Modelling C*

#### *1.6.1. Strengths and weaknesses of CBM-CFS3*

The oldest three papers (Kurz et al. 2008a, b, Dymond et al. 2010), that form the body of this dissertation relied on a fundamentally different modelling approach than the most recent two papers of this dissertation (Dymond et al. 2016, Hof et al. 2017). The older papers used CBM-CFS3 which has some advantages and some limitations (Kurz et al. 2009). Some of those limitations were overcome by switching to the ForCSv2 extension to LANDIS-II, a free, open-source modelling framework with user support (Scheller et al. 2007, Dymond et al. 2016). A description of CBM-CFS3 along with limitations in the applied studies is described below:

1) CBM-CFS3 uses traditional forestry age and volume tables for growth. Age-volume tables from growth and yield modelling are a convenient source of tree productivity data for representing the past. CBM-CFS3 does allow the application of growth multipliers that can be employed in climate change studies (e.g. Metsaranta et al. 2011) but this does not overcome the fundamental problem that many forestry growth and yield models are not sensitive to a changing climate (cf. Trasobares et al. 2016). In contrast, the productivity inputs to LANDIS-II may come from a variety of different types of models and are structured

in such a way so that they may change over time (Dymond et al. 2015). In [Dymond et al. \(2016\)](#) and [Hof et al. \(2017\)](#) we used a climate-sensitive, mechanistic model to generate the productivity and natural regeneration inputs for ForCSv2 (see the methods section of those papers for details).

2) CBM-CFS3 is deterministic, i.e. there is no dynamic feedback from the landscape to the area or intensity of management or disturbances (Kurz et al. 2009). This is an advantage for reporting on what has happened in the past. However, it is a limitation for projecting the future in British Columbia, where climate-envelope modelling has mapped changes from climates that are suitable for closed-canopy forest to climates that are suitable for savannah or grassland ecosystems by 2080 (Wang et al. 2012). This change in tree density is likely to come about by the fire regime switching from lower-frequency, stand replacing fires to more frequent, larger, grassland-type fires. This limitation can be overcome by using the Dynamic Fuels and Fire System available as part of LANDIS-II, where fuels can affect fire behaviour (Sturtevant et al. 2009). Unfortunately, there is no dynamic harvesting or silviculture extension to LANDIS-II.

3) CBM-CFS3 is spatially referenced, not explicit, so a wildfire, for example, may affect simulated-stands that are not adjacent to each other (Kurz et al. 2009). Furthermore, natural regeneration from seed sources outside a stand are not simulated. This design is an advantage at the continental scale but are more limiting for my purpose of producing simulations relevant to local forest managers of British Columbia who think in spatially explicit terms. LANDIS-II is spatially explicit, at the scale chosen by the user (Scheller et al. 2007).

4) CBM-CFS3 can only simulate single-aged stands with up to two species (one hardwood/broadleaved and one softwood/conifer) (Kurz et al. 2009). One of the consequences of the single-age cohort is the overestimate of disturbance impact in stands that are actually more complex, as described above, in the evaluation of disturbances section. Another limitation of this design is that logging is limited to clear-cut and plant, whereas I wanted to evaluate a variety of harvesting and silviculture activities such as partial cutting or natural infill regeneration. Many models, including ForCSv2 can simulate multi-species and multi-age stands (Dymond et al. 2015).



Both CBM-CFS3 (Kurz et al. 2009) and ForCSv2 ([Dymond et al. 2016](#)) are limited by their lack of explicit representation of nutrient availability, which can be a critical factor in forest carbon dynamics (Wieder et al. 2015). Fertilization is, of course also a potential mitigation activity to increase tree growth rates (McDonald et al. 1994, Smyth et al. 2014).

#### 1.6.2. Uncertainty in C modelling

There are a variety of modelling frameworks available for estimating forest C stocks and fluxes with often dramatically different estimates of GHG balance in forests (House et al. 2003). For example, Hayes et al. (2012) compared multiple models, but with methods as consistent as possible to estimate the annual average net ecosystem exchange from 2000 to 2006 in North America. For the forested lands of Canada, estimates ranged from a small sink of  $-31 \text{ MtC yr}^{-1}$  using the same inventory framework as [Kurz et al. \(2008a\)](#), to a large sink of  $-151 \text{ MtC yr}^{-1}$  using inversion modelling. Hayes et al. (2012) state that the large sink estimated by the inversion models is likely to be an overestimate because of biases in vertical mixing and the condition of a west coast boundary effect. The small sink from the forest inventory-based studies, similar to [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#), may be attributable in part to the use of standard forestry yield tables to represent growth rates and those tables may not reflect increased growth rates due to  $\text{CO}_2$  fertilization and warmer conditions in recent decades (House et al. 2003). Another major difference between the modeling approaches is the parameterization of disturbance effects, which may be overestimated in the inventory approach (see above for more detail) and underestimated in inversion models (G. Stinson pers. comm.). The third approach to modelling C dynamics assessed in Hayes et al. (2012) was terrestrial biosphere models. The average net ecosystem exchange for Canada's forests from terrestrial biosphere models fell between the inventory and inversion-based estimates ( $-73 \pm 141 \text{ MtC yr}^{-1}$ ), however the standard deviation within that class of models was even greater than the difference between the average results of the inverse and inventory models. Clearly, there is uncertainty in forest C stock and GHG flux estimates at a continental scale, and given the sensitivity of some ecosystems, natural disturbances contribute greatly to those uncertainties. Based on Hayes et al. (2012), there does not yet appear to be a modelling approach that is clearly and consistently superior to the ones used in this dissertation for estimating recent GHG fluxes from forest ecosystems at the continental scale.

At the stand-scale, there are also uncertainties when models are compared. For example, Wang et al. (2014) used five different models to assess harvesting effects on an aspen forest C stocks and fluxes in the USA. Stemwood C, arguably the pool we should have the most confidence in because of the long history of forestry, ranged from 2,700 to 6,952 g C m<sup>-2</sup>. The GHG balance ranged from 27.5 to 79.3 g C m<sup>-2</sup> yr<sup>-1</sup> (assuming instant oxidation of harvested wood). As a result of this uncertainty, Wang et al. (2014) recommend using multiple models to simulate forest C dynamics.

The global warming impact of natural disturbances in models such as CBM-CFS3 and LANDIS-II can be over-estimated because they do not include the effect of albedo. Especially in coniferous forests, widespread mortality of trees across large areas can increase the reflectance of sunlight and as a result less heat energy is absorbed by the forests, reducing the global warming effect of a disturbance (Bonan et al. 1992). The albedo effect is likely to mean that the global warming implications of the mountain pine beetle, wildfire, and spruce budworm disturbances were overestimated in [Kurz et al. \(2008a, b\)](#) and [Dymond et al. \(2010\)](#). It is still unclear what the relative importance of albedo is, depending on the ecosystem. However, strategies to mitigate climate change may have to take into account if albedo is likely to change, otherwise activities could end up being counter-productive (Bright et al. 2014).

### *1.7. Future Research Needs*

#### *1.7.1. Partial cutting*

Of the various forest management activities practiced in North America, partial cutting<sup>1</sup> (through selective harvesting, high retention forestry, or pre-commercial thinning) has had little investigation in terms of climate change mitigation potential (Ryan et al. 2010, Lempière et al. 2013, Kalies et al. 2016). Where studies on implications for C dynamics have been carried out, the management scenarios are often combinations of changes in harvest

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<sup>1</sup> The Canadian Forest Service defines partial cutting as ‘any cutting in which only part of the stand is harvested’ (CFS 1999)

rate ( $\text{m}^3$  per year) as well as harvesting systems (Bright et al. 2014, Creutzberg et al. 2017, Ling et al. *submitted*) making it difficult to understand the role of partial cutting.

It is unclear under what circumstances partial cutting would be a net benefit to mitigating climate change. A global meta-analysis of 81 papers on partial cutting at the stand-level found that it reduced C stocks (biomass, forest floor and soil), but increased tree growth increments and understory vegetation C stocks compared to uncut stands (Zhou et al. 2013). They also found that the number of years taken to recover the lost above-ground biomass C stocks ranged from 0 to 42 years (the maximum observed) and varies with harvest intensity. Unfortunately, that study was unable to assess whole ecosystem C stocks or fluxes.

One potential area where partial cutting may be an effective mitigation strategy is in areas where clear-cut harvesting occurs despite a high proportion of non-commercial species or defective boles (Marie-Lou Le Francois pers. comm.). The non-economic species are cut, but either burned or left to decay, releasing emissions. Thus, changing management to partial cutting may reduce emissions from harvest residues because the uneconomical trees are un-cut, thus maintaining more C storage and potential GHG sinks. The overall benefit would depend on the proportion of uneconomic trees, if the total area harvested would have to increase to achieve the same harvest rate ( $\text{m}^3$  per year) through partial cutting, and the post-harvest growth rates.

In addition to mitigating climate change, partial cutting may be useful for adaptation.

D'Amato et al. (2011), studied the effect of thinning treatments on above-ground biomass C stocks and increment and identified a trade-off between climate change mitigation and adaptation. Stands with higher stocking densities had higher C storage but lower stand complexity (measured by the: diversity of tree diameters, density of large trees, and species diversity), one indicator of adaptive capacity and resilience (Puettmann et al. 2012).

D'Amato et al. (2011) concluded that a potential strategy for meeting both adaptation and mitigation goals may be to employ partial cutting systems that maintain a proportion of C stores in mature trees, still supply the lumber demanded by society, and enhance structural complexity compared to clear-cut systems.

It appears that partial cutting may be an effective strategy for adaptation in some situations, depending on the ecosystem and climate change effects. In addition to increasing structural complexity, as mentioned above, a second theoretical reason for this being an adaptation strategy is the targeted harvesting of species expected to have reduced productivity in the future due to the changing climate (Steenberg et al. 2011). This selectivity aims to facilitate transition to a composition of species more tolerant of the future climate. A related reason for considering partial cutting for adaptation is to reduce the dominance of species susceptible to pests and pathogens if their threat is increasing ([Dymond et al. 2014](#), 2015). A fifth reason is that partial harvesting systems such as strip cuts or small patch-cuts offer a greater diversity of tree regeneration conditions than in large clear-cuts, for example, partial shading, thus having lower evaporative demand (Ferguson and Carlson 1990). Through this range of microsite environmental conditions, a greater diversity of species may regenerate naturally or through planting. However, growth rates and survivorship of commercially valuable species may be reduced at smaller opening sizes (Lajzerowicz et al. 2006).

Beyond just the tree species, partial cutting allows the logged stand to continue providing adaptive capacity for a variety of ecological goods and services. These include the maintenance of habitat for shade-tolerant plants, wood- or soil-inhabiting insects and fungi, and many other organisms (Gustafsson et al. 2012), increasing the potential for them to maintain their populations despite a changing climate. In addition, they improve the connectivity of habitats within the managed landscape, improving the recolonization and dispersal of organisms that will migrate as the climate changes (Gustafsson et al. 2012).

Given the potential benefits of partial cutting for the adaptation of systems to climate change, and both positive and negative indicators for forest C dynamics, I have identified partial cutting as a research need for forest managers. This future study would likely build on an existing modelling projects in north-western British Columbia ([Dymond et al. 2016](#), [Hof et al. 2017](#)). This approach would leverage the time and effort that has gone into defining modelling parameters and variables. Furthermore, disturbances in that study area tend to be small and the forests dynamics are dominated by gap dynamics, more consistent with partial cutting than large clear-cuts. Other ecosystems, such as those adapted to large, stand-replacing disturbances likely have fewer species that are shade-tolerant and adapted to gap-dynamics therefore partial cutting may be less advantageous.

### 1.7.2. Methods

One of the gaps I have identified in the literature lies between the ecologists and modellers whose fundamental unit is species or vegetation type such as ‘conifer forest’ (e.g. Silva Pedro et al. 2017) and the tree geneticists and forest planners whose fundamental unit is provenances (locally-adapted populations) (Aitken et al. 2008, Crowe and Parker 2008). Models of productivity that incorporated provenance have been limited to a few species for British Columbia (O’Neill et al. 2008b). The recent release of an operational, climate-based system for identifying suitable seed stock for 19 species indicates that modelling limitation will change in the near future (O’Neill et al. 2017) so that an ecological modelling approach can incorporate provenances for British Columbia.

### 1.8. Conclusion

Forest managers and policy makers struggle to make decisions that incorporate climate change knowledge. The six papers reviewed in this critical analysis bring together a suite of information that will help their decisions. First of all, that although forests are often GHG sinks, moderate or high intensity natural disturbances can reverse that flux. Second, that climate change effects on productivity may increase or decrease the natural sinks or even create emission sources. Third, that management strategies to increase resilience may be effective at reducing risks of emissions, but they must be assessed for individual landscapes and may be insufficient to fully offset disturbance or climate change effects.

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Chapter	Pages	Journal article	DOI link
2	49-76	Kurz, W.A., Stinson, G., Rampley, G.J., Dymond, C.C. and Neilson, E.T., 2008. Risk of natural disturbances makes future contribution of Canada's forests to the global carbon cycle highly uncertain. <i>Proceedings of the National Academy of Sciences</i> , 105(5), pp.1551-1555.	<a href="https://doi.org/10.1073/pnas.0708133105">https://doi.org/10.1073/pnas.0708133105</a>
3	77-109	Kurz, W.A., Dymond, C.C., Stinson, G., Rampley, G.J., Neilson, E.T., Carroll, A.L., Ebata, T. and Safranyik, L., 2008. Mountain pine beetle and forest carbon feedback to climate change. <i>Nature</i> , 452(7190), p.987.	<a href="https://doi.org/10.1038/nature06777">https://doi.org/10.1038/nature06777</a>
4	110-143	Dymond, C.C., Neilson, E.T., Stinson, G., Porter, K., MacLean, D.A., Gray, D.R., Campagna, M. and Kurz, W.A., 2010. Future spruce budworm outbreak may create a carbon source in eastern Canadian forests. <i>Ecosystems</i> , 13(6), pp.917-931.	<a href="https://doi.org/10.1007/s10021-010-9364-z">https://doi.org/10.1007/s10021-010-9364-z</a>
5	144-182	Dymond, C.C., Tedder, S., Spittlehouse, D.L., Raymer, B., Hopkins, K., McCallion, K. and Sandland, J., 2014. Diversifying managed forests to increase resilience. <i>Canadian Journal of Forest Research</i> , 44(10), pp.1196-1205.	<a href="https://doi.org/10.1139/cjfr-2014-0146">https://doi.org/10.1139/cjfr-2014-0146</a>
6	183-220	Dymond, C.C., Beukema, S., Nitschke, C.R., Coates, K.D. and Scheller, R.M., 2015. Carbon sequestration in managed temperate coniferous forests under climate change. <i>Biogeosciences Discussions</i> , 12(24).	<a href="https://doi.org/10.5194/bgd-12-20283-2015">https://doi.org/10.5194/bgd-12-20283-2015</a>
7	221-268	Hof, A.R., Dymond, C.C. and Mladenoff, D.J., 2017. Climate change mitigation through adaptation: the effectiveness of forest diversification by novel tree planting regimes. <i>Ecosphere</i> , 8(11), p.e01981.	<a href="https://doi.org/10.1002/ec.11981">https://doi.org/10.1002/ec.11981</a>