Monitoring Forest Restoration Effectiveness on Galiano Island, British Columbia: Conventional and New Methods

by

Quirin Vasco Hohendorf
B.Eng., Hochschule Weihenstephan-Triesdorf, 2015

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Abstract

I compared forest structural parameters of treated and untreated plots on a forest restoration site on Galiano Island, British Columbia. The site was replanted with Douglas-fir (Pseudotsuga menziesii (mirb.) Franco) after being intensively logged in the 1970s and then thinned in the early 2000s. I used existing baseline data from 8 permanent plots (5 treated, 3 control) and compared it with forest assessment data collected in the field in the summer of 2017. Additionally, I used 16 temporary plots (8 treated, 8 control). I assessed vegetation percentage cover by plot, coarse woody debris by plot, tree diameter, species and status (n = 846), height (n = 48) and diameter growth (n = 271). I found that treated plots showed improved measures of structural diversity like diameter growth, crown ratios and plant diversity, but I was unable to relate the increased diameter growth to the restoration treatments. My findings suggest that to create a lasting impact, restoration thinning will have to be more frequent or create larger gaps.

I then reviewed the current studies with unmanned aerial vehicles (UAV) in ecological restoration. I evaluated potential use of hobbyist UAVs for small organizations and not-for-profits and found that if applied correctly, UAVs can increase the amount of available data before, during and after restoration. Reproducible and reliable results require trained personnel and calibrated sensors. UAVs can increase access to remote areas and decrease disturbance of sensitive ecosystems. Regulations, limited flight time and processing time remain important restrictions on UAV use and hobbyist UAVs have a limit availability of sensors and flight performance.

Finally, I used images taken from a hobbyist UAV to assess forest structure of the restoration site on Galiano Island and compared my results with the ground measurements. I found a canopy height model (CHM) from UAV images underestimated mean tree height values for the study site on average by 10.2 metres, while also severely underestimating mean stem densities. Using a 2 metre threshold, I delineated canopy gaps which accounted for 6 % of the canopy. UAV images and the resulting CHM represent a new visualization of the study site's structure and can be a helpful tool in the communication of restoration outcomes to a wider audience. They are not, however, sufficient for monitoring or scientific applications.

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List of Abbreviations

BVLOS	beyond visual line-of-sight
CHM	canopy height model
CWD	coarse woody debris
DBH	diameter at breast height
DEM	digital elevation model
DTM	digital terrain model
EVLOS	extended visual line-of-sight
GCP	ground control point
GIS	geographic information system
GPS	global positioning system
IQR	inner quartile range
RBG	Red-green-blue. Primary colours representing visual light
SfM	Structure-from-motion technology
UAV	unmanned aerial vehicle
VLOS	visual line-of-sight

Acknowledgements

I would like to acknowledge the Lkwungen-speaking peoples on whose traditional territory the University of Victoria stands and the Songhees, Esquimalt and W̱SÁNEĆ peoples whose historic relationships with the land continue to this day.

My research was focused on what is now known as District Lot 63, Galiano Island. I would like to acknowledge that my work was conducted in the shared, asserted and unceded territory of the Penelakut, the Lamalcha, and the Hwlitsum Nations, other Hul'qumi'num speaking peoples, SENĆOŦEN and W̱SÁNEĆ speaking peoples, and any others with rights and responsibilities in and around what is now known as Galiano Island. I would like to acknowledge that my work was conducted on the ceded territory of the Tsawassen First Nation. I am very grateful for the privilege of having been able to conduct my work within these shared traditional territories.

I would like to express my gratitude to everyone who supported me on this journey: To my graduate supervisor Dr. Eric Higgs and committee member Cecil C. Konijnendijk for supporting me and allowing me the freedom to turn my ideas into this project. Thank you to everyone at the Galiano Conservancy Association and especially Keith Erickson. Keith, along with Herb Hammond were participants in the original treatments and helped me understand the thinking behind it. Thank you to my lab group, my cohort and the School of Environmental Studies for making my two years in Victoria such an unforgettable experience. Thank you to the University of Victoria for financially supporting my graduate studies and to the Lorene Kennedy Graduate Student Research Award committee for supporting my fieldwork on Galiano Island. Last but not least, I would like to thank my partner, my friends and my family who kept me motivated along the way.

"Damn good coffee!" - Dale Cooper, Twin Peaks

Dedication

In memory of Ken Millard who was the heart of the restoration treatments on DL63 and inspired us all to work hard for conservation and restoration.

Chapter 1: Introduction

1.1 Ecological restoration

The standard definition of ecological restoration, "is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER, 2004, p. 3). In the light of decreasing biodiversity and land loss, it is more important than ever to restore degraded systems and ecological restoration becomes increasingly recognized as an important tool in protecting the environment (Aronson and Alexander, 2013). Ecological restoration is no replacement for conservation but an additional measure that needs to be taken globally to counteract degradation and destruction of natural systems (Aronson and Alexander, 2013; Keenleyside et al., 2012; Suding, 2011).

Ecological restoration first evolved as a discipline in the 1980s, but its roots in North America date back at least to the 1930s, when Aldo Leopold conducted the first documented restoration project at the University of Wisconsin-Madison (Greenwood, 2017). Many new ideas and concepts in ecology also influenced restoration ecology and the field evolved from a simple "bring back what was before" to a complex discipline, dealing with a changing climate (Falk and Millar, 2016), heavily altered and novel ecosystems (Hobbs et al., 2013), and alien invasive species (Head et al., 2015).

To be successful, restoration projects need to be effective, efficient and engaging (Keenleyside et al., 2012). Ecological restoration is effective when interventions re-establish ecosystem structure, function and composition in the short and long-term by increasing the resilience against future disturbance and encouraging ecological, social and cultural sustainability of the project. Efficient restoration considers different scales, enhances the ecosystem services provided by the restored ecosystem and ensures long term maintenance and monitoring. Available resources are used so that they have the most possible impact. Ecological restoration is engaging when project planners collaborate with local communities, scientists and other stakeholders throughout the whole project and when monitoring results are communicated effectively to all stakeholders. This increases the support for restoration projects, improves monitoring and builds capacity and understanding for ecological processes (Keenleyside et al., 2012).

Restoration must not only meet ecological needs, but also consider social and cultural needs to be successful (Perring et al., 2015; Wiens and Hobbs, 2015). Services provided by restored ecosystems often include social and cultural benefits like recreation, food resources or clean water (Keenleyside et al., 2012). These should be incorporated in the goal setting, planning and monitoring regime in a quantifiable way.

In early restoration, monitoring was often neglected which complicated the assessment of restoration success (Wortley et al., 2013). This resulted in many projects with low success and declining support from funders and local communities. A review of scientific papers on restoration success in 2013 showed that monitoring of restoration success is becoming increasingly important. The authors found 301 publications that evaluate restoration outcomes in the 28 years covered by the study, with most studies published between 2008 and 2012 (Wortley et al., 2013). The authors relate this development to increasing maturity of restoration projects. Monitoring can improve restoration success by contributing to adaptive management (AM). AM uses an iterative process of management decisions as a means of dealing with uncertainty in the process. An important part of AM is learning about the system while managing it and so further improve future management. It follows six steps to manage a project. Assessment, design, implementation, monitoring, evaluation, adjustment and repeated assessment (Murray & Marmorek, 2003). The “Ecological Restoration for Protected Areas” IUCN guidelines recommend a seven-phase process to ecological restoration which includes AM as its main element (Keenleyside et al., 2012). AM has been recognized as an excellent strategy for successful restoration (Dellasala et al., 2013; Gaylor et al., 2002), and is being implemented many projects around the globe, for example in federal forests in the USA (Dellasala et al., 2013; Franklin and Johnson, 2012) and the restoration of Springbrook world heritage rainforest in Australia (Keenleyside et al., 2012).

1.2 Ecological Restoration of Forests

Deforestation and forest degradation are the second largest source of anthropogenic carbon emissions (IPCC, 2007). The effects of elevated amounts of carbon in the earth’s atmosphere on biodiversity and human livelihoods, have led to an increased recognition for countermeasures like re-forestation and forest restoration (Ciccarese et al., 2012). Additionally, intact and functioning forest ecosystems are critical for important ecosystem services, such as clean water, air, firewood and timber supply (Ciccarese et al., 2012). Ecosystems with long-lived species are especially hard to restore, due to long planning periods and high uncertainties about future environmental conditions (Golladay et al., 2016; Hamann and Wang, 2006). This is especially true for forests, due to the slow growth and long lifetimes of trees. We cannot predict precisely how the climate will have changed in 50 or even in 200 years, when a now young stand will have reached a mature state and forests therefore forest management has to deal with a degree of uncertainty (IPCC, 2007). While most young forests will eventually undergo succession towards old-growth stands, the goal of forest restoration is to help the succession and accelerate the process (Parks Canada Agencies, 2008).

Long term planning under these conditions is challenging, but there is significant consensus that especially in forests adaptive management strategies are a good way of responding to the challenge (Golladay et al., 2016; Hiers et al., 2016), and among others, Parks Canada (2008) and Keenleyside et al. (2012), suggest using adaptive management in their guidelines for ecological restoration. Since the publication of the guidelines, adaptive management has become even more popular (Hobbs, 2016).

1.3 The Coastal Douglas-fir zone

My study site on is located on Galiano Island, one of the southern Gulf Islands, between the Lower Mainland and Vancouver Island in British Columbia, Canada. The study site is in the heart of the moist-maritime Coastal Douglas-fir biogeoclimatic zone (CDF) (Nuszdorfer et al., 1991). The CDF zone covers less than one percent of British Columbia and appears only at elevations up to 260 m (figure 1-1) (Nuszdorfer et al., 1991). The climate is cool mesothermal, with mild wet winters (800 mm precipitation) and warm and dry summers (200 mm of precipitation) (Nuszdorfer et al., 1991). Mean temperatures range from 3°C to 17°C with an annual mean of 10°C (Nuszdorfer et al., 1991). Douglas-fir (*Pseudozuga menzesii* (Mirb.) Franco) is the most common tree species throughout the zone (Nuszdorfer et al., 1991). Arbututs (*Arbutus menziesii)* Pursh and Garry oak (*Quercus garryana)* Douglas ex Hook. are less common but almost exclusively occur in the CDF zone in Canada (Nuszdorfer et al., 1991). Only 3% of the CDF zone is protected, with mostly small, isolated, and patches and few large protected areas (> 250 ha)

(Nuszdorfer et al., 1991). Almost one third of the CDF has been transformed from forest to some other form of land use (Nuszdorfer et al., 1991). Only about 10% of the forest is more than 120 years old and less than 1% is old-growth (Nuszdorfer et al., 1991). Land transformation, invasive species introduction and the change of ecological processes have led to the listing of many species as endangered (Nuszdorfer et al., 1991). The CDF zone has a very limited extent, but has significant species richness and distinctive ecological communities that make well-connected and better protected management necessary (Nuszdorfer et al., 1991).

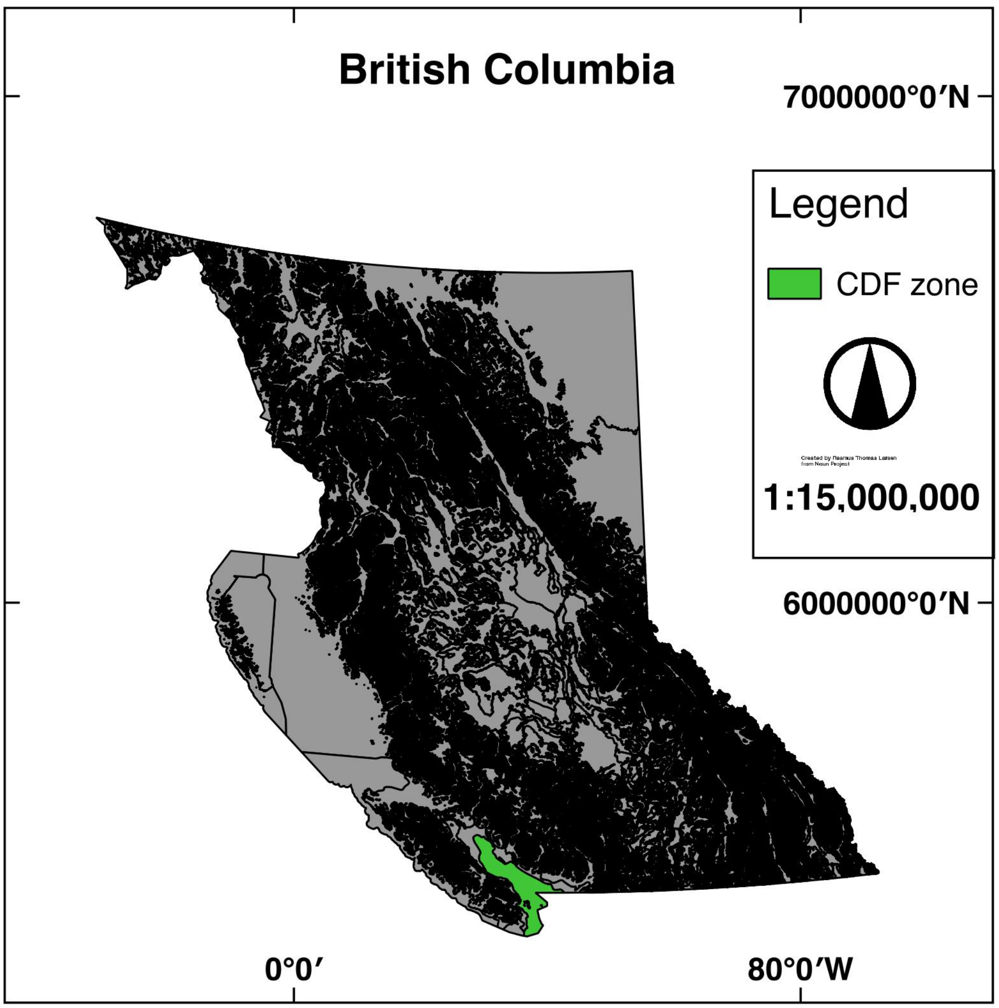


Figure 1-1: Overview of British Columbia with the Coastal Douglas-fir zone (green)

1.4 The Galiano Conservancy Association and Restoration of a Douglas-fir plantation

The Galiano Conservancy Association (GCA) is a local land trust that was formed in 1989. Formed out of a desire to stop unsustainable logging practices on Galiano Island in the 1970's, Forest conservation and restoration has always been a core concern of the GCA. With clear-cut logging happening all over the island in the 1970's the community started to stand up against logging companies to protect their island´s ecosystems, which consequently led to the formation of the GCA as a land trust.

In 1998, early in its history, the GCA acquired a highly-degraded forest lot (District Lot 63, or DL63) that would become part of the Mid Galiano Island Protected Area Network. The Mid Galiano Island Protected Area Network covers 616 hectares and spans from west to east roughly in the middle of the long and narrow island. The site was partially clear-cut in 1967 and again in 1978 and only about 4% of the 61.5 ha were left intact (Gaylor et al., 2002). The first cut removed all trees from 20% of the land area and all remaining woody biomass was piled and burned to create an easier environment for planting (Gaylor et al., 2002). After the second cut, slash and topsoil were piled in windrows and burned. This was done partly to fight laminated root rot, a fungal disease caused by *Phellinus weirii-1* (Murrill) R. L. Gilbertson, but the large windrows did not fully combust (Gaylor et al., 2002). This left coarse woody debris in various sizes and degrees of combustion. After both cuts the open areas were re-planted with Douglas-fir seedlings from off-island provenance (Gaylor et al., 2002).

The restoration of the Douglas-fir plantation started in 2003 by the GCA with the help of many volunteers (Scholz et al., 2004). All the restoration work was done without the use of power tools or combustion engines as a nod to low impact techniques. For the erection of snags, moving of big logs, and the pulling of trees, the GCA used chain hoists and skylines, techniques specifically designed for the project (Scholz et al., 2004). The treatments included dispersal of the coarse woody debris (CWD) formerly piled in windrows, erection of large snags to mimic wildlife trees, control of invasive species, of loosening compacted soil on roads and timber landings, pulling, topping, and girdling of trees, and planting of native plant species (Scholz et al., 2004). The restoration of DL63 is a unique restoration project because of its low-impact approach. The project is of special importance to the GCA: many of its early members were directly involved in the restoration efforts and the low-impact approach directly reflects values held by many members.

Before starting the restoration of District Lot 63, the GCA collected extensive baseline data. The GCA divided the forest into 47 polygons of varying sizes according to ecosystem types, by assessing aerial photographs and later confirming and correcting the extend of the polygons by ground sampling. The creek at the east side of the property, and a buffer of 20 m on both sides, were excluded from the sampling and treatments. Depending on their relative size, each polygon was sampled with one to eight temporary 20 x 20 m sampling plots. The plots were randomly distributed, but locations were manually corrected to avoid edge effects, roads and openings.

The GCA then established eight permanent plots on the study site – five in areas where restoration treatments took place, and three control plots outside the treatment areas. Additionally, the GCA established two permanent plots in a neighbouring mature Douglas-fir forest. Those plots are part of a 1-hectare SI/MAB plot. The SI/MAB plot is an internationally used monitoring plot for biodiversity recommended by the Smithsonian Institute (SI) and the UNESCO

Program on Man and the Biosphere (MAB) (Roberts-Pichette and Gillespie, 1999). The GCA laid out all permanent plots using the guidelines described by Roberts-Pichette and Gillespie in *Terrestrial Vegetation Biodiversity Monitoring Protocols* (Roberts-Pichette and Gillespie, 1999). The plots were 20 x 20 m as suggested for young, even-aged stands. The plots were laid out square to the general slope, and all corners A-D were marked with metal pins (Figure 2). I was not able to find some of these metal pins and had to reestablish several corners using a compass and measuring tapes. Each quadrat bears an individual ID and all four corners were marked with GPS points and are available as a shapefile for GIS use. For plots on a slope, the GCA used slope correction to set up an exact 20 x 20 m square in the plane.

Monitoring strategies were included in the original "Restoration Plan" (Gaylor et al., 2002) and the "Monitoring Baseline" (Scholz et al., 2005). The GCA designed an adaptive array of monitoring strategies to assure that monitoring will persist in the future, even with the uncertainties that beset a small non-profit charitable organization (Scholz et al., 2005). However, monitoring was not executed as planned. Two students, one graduate and one undergraduate, did subsequently collect data about stand structure, soil nutrients, and species composition as part of their thesis work (Harrop-Archibald, 2010; Meidl, 2013).

Canada has committed under the United Nations Framework Convention on Climate Change (UNFCCC) to take actions to limit climate change (Government of Canada, 2010). These actions include the promotion of "…sustainable development approaches (e.g. promote the conservation and enhancement of sinks and reservoirs of all GHGs, and take into account climate change in economic and environmental decision making)" (Government of Canada, 2010, p. 2) and regular updates on the progress in fulfilling these commitments (Government of Canada, 2010). One of these measures of promotion is the EcoAction Community Funding Program, which helped finance community based climate action on conserved forest land. In 2010 Canada reported about successful projects and included the restoration of the provincially and globally endangered Coastal Douglas-Fir forest on District Lot 63, undertaken by the GCA on Galiano Island, BC (Government of Canada, 2010). “Restoration efforts undertaken will increase carbon sequestration on the site. This will help reduce the impacts of climate change. Restoration will also increase biodiversity, improve ecosystem health and enhance the site’s ability to adapt to the impacts of a changing climate.” (Government of Canada, 2010, p. 134). The project is also explicitly mentioned as a success of Canadas restoration efforts on the IUCN hosted website www.infoflr.org. Until now, the success of the DL63 restoration project has not been evaluated. This thesis is the first comprehensive evaluation of the effects of the forest restoration on Galiano Island, and will contribute to the continuing adaptive management of the site.

1.5 Remote sensing and Unmanned Aerial Vehicles

Environmental remote sensing, the practice of recording electromagnetic waves from a distance to gather information about objects on the earth’s surface, started with the invention of airplanes and cameras, but did only gain a global importance after the launch of the first satellites in the 1950s and 1960s when it was first coined “remote sensing” by the United States Office of Naval Research (Cracknell, 2007, Khorram et al., 2012) . Remote sensing can be used to detect any kind electromagnetic energy, from gamma to radio waves. However, most commonly used is visible and infrared light (Khorram et al., 2012). The technology was quickly adapted for military reconnaissance during World War One and remote sensing data soon became popular for civilian applications because of its ability to provide data for large areas with relative high spatial and temporal resolution (Rees, 2013).

Unmanned aerial vehicles (UAVs), commonly known as drones, are the newest development in remote sensing (Adão et al., 2017). UAVs are small, remotely controlled systems, capable of autonomously following a pre-programmed flight path and usually carry one or more sensors, most commonly digital cameras. Both UAV's and their sensors are affordable compared with many other remote sensing technologies, and have gained popularity for recreational, commercial, and military applications and research. Many classifications of UAVs exist, but for UAVs in ecology Anderson and Gaston (2013) describe four categories: *Large*, *Medium*, *Small and Mini*, and *Micro and Nano*. Large UAVs weigh about 200 kg, are as large as small airplanes, require a runway for takeoff and full aviation clearing. However, they allow for an operating range of about 500 km and flight times of up to two days. Medium UAVs weight about 50 kg, have similar start and landing requirements to large UAVs, but are cheaper and easier to handle due to their reduced size. Their operating range is similar to large UAVs, but flight times are only about 10 hours (Anderson and Gaston, 2013). Small and mini UAVs weigh less than 30 kg (small) and less than 5 kg (mini), can only be flown within line-of-sight, require small open areas and minimal equipment for takeoff and landing, and can be controlled by flight planning software or directly by radio control. With an operating range of less than 10 km and a flight time of less than two hours, their application is limited to smaller areas (Anderson and Gaston, 2013). Micro and nano UAVs weigh less than 5 kg, require barely any space for takeoff and landing and are flown within line of sight, controlled by flight planning software or direct radio control. Operating range is similar to small UAVs, but flight times are even shorter (< 1 hour). In this thesis, I focused on micro UAVs. They are currently the most common because of their affordability and easy handling (Anderson and Gaston, 2013).

Regulations for UAV use vary from country to country. Technical developments are occurring rapidly, cost/performance is lowering. Most countries require permissions when UAV are used for commercial or scientific applications, and often require registration of the UAV and insurance for damage caused by the vehicle (Stöcker et al., 2017). In addition, the maximum flight height, the weight of the UAV including any attachments and distance to sensitive airspace like airports or hospitals are restricted in most countries (Stöcker et al., 2017). Usually, operation of UAV has to be within visual line of sight (VLOS). In the US, UK, Italy, Spain and South Africa the use of an extended visual line of sight (EVLOS), where an additional observer helps keeping visual contact to the UAV, is possible (Stöcker et al., 2017). Flying beyond visual line of sight (BVLOS) are almost always subject to higher level regulations and require exceptional approval or special flight conditions (Stöcker et al., 2017).

1.6 Conceptual Foundation and Organization of the Thesis

My research focused on assessing the effectiveness of a forest restoration project on Galiano Island, which I explore in depth in chapter 2. My project is part of an ongoing monitoring effort that had been largely held back by insufficient resources since the inception of the restoration in 2003. I explored alternative ways of monitoring restoration effects because of the uncertainty of available funding. Initial experimentation with a UAV for canopy gap mapping led me to focus on UAV applications in ecological restoration and their future potential in a review of current literature in chapter 3. I conceived and executed a trial of UAV derived images for the monitoring of restoration effectiveness on my study site on Galiano Island (chapter 4).

I have written up the results as three manuscripts for potential publication. (chapter 2 to 4). Working alongside my committee in coming months, I propose to submit chapter 2 to the journal *Ecological Restoration*, chapter 3 to *Restoration Ecology*, and chapter 4 to *Forests*. Formatting is according to journal standards and therefore differs slightly between chapters.

Chapter 2: Restoration effectiveness in a Young Douglas-fir Forest

0. Abstract

We assessed the outcomes of the restoration of a 40-year-old Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco plantation in British Columbia, Canada. The main restoration processes undertaken between 2003 and 2006 were thinning by pulling, topping, and girdling trees. We used existing baseline data from 8 permanent plots (5 treated, 3 control) and compared it with forest assessment data collected in the field in the summer of 2017. Additionally, we used 16 temporary plots (8 treated, 8 control) to cover restoration effects in areas of the forest that were not covered by the permanent plots. We assessed tree diameter, species and status (n = 846), height (n = 48) and diameter growth (n = 271). We also assessed understory percentage cover of vascular plants by species and all pieces of coarse woody debris with diameters larger than 7.5 cm in the 8 permanent plots. Analysis with generalized mixed effect linear models showed that treated areas displayed increased diameters, higher diameter growth, increased plant diversity, increased crown ratio, and more snags, but lower basal area, tree heights, and density. Control plots showed a stronger increase in volumes of coarse woody debris but volumes were still lower than treated plots. We were unable to relate the increased diameter growth to the restoration treatments. Our findings suggest that to create a lasting impact, restoration thinning will have to be more frequent or create larger gaps.

1. Introduction

Calls for re-forestation and forest restoration have become more urgent, with two billion ha of degraded forest globally (Minnemayer et al., 2011), continuing global deforestation, a worldwide loss of biodiversity, and directional climate change (Ciccarese et al., 2012; Mansourian et al., 2005). Moreover, threats to forests are increasing. A rise in global temperatures poses a significant threat to future forests as tree species with small populations or fragmented ranges may not be able to migrate fast enough to keep up with the changing conditions (Aitken et al 2008). Invasive insects and mammals pose an additional threat to trees, especially in combination with weather extremes weakening the trees (Dumroese, 2014).

Intact and functioning forest ecosystems are critical to the provision of ecosystem services such as clean water, air, opportunities for recreation, and perhaps most importantly in the context of climate change, carbon sequestration (Ciccarese et al., 2012). Once degraded, forests are especially challenging to restore, due to long planning periods, slow tree growth, and uncertainties about future environmental conditions (Golladay et al., 2016; Hamann and Wang, 2006).

With increasing threats, it is no longer enough to conserve forests. There is also a need to actively restore forests to re-create habitat for species that rely on old-growth structures (Halme et al., 2013). Internationally, several commitments to sustainable forest management and forest restoration have been agreed. These include the New York Declaration on Forests (UN Climate Summit, 2014), the Bonn Challenge ((IUCN) International Union for Conservation of Nature, 2018), the Aichi Biodiversity Targets (specifically Target 15) (UN Environment, 2018), the United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (REDD+) (UN-REDD Programme, 2016), and the United Nations Framework Convention on Climate Change (UNFCCC) (Protocol, 1997). Canada has committed under the UNFCCC to take actions to limit climate change (Kingsberry et al., 2010). Those actions include ecological restoration, such as for example the federally funded restoration of a provincially and globally endangered coastal Douglas-fir ecosystem on Galiano Island, BC (Kingsberry et al., 2010). Global commitments have increased awareness of, and attention for forest restoration, but resources for treatments remain limited since there is no immediate financial benefit.

Forest restoration increasingly focuses on landscape level approaches that may be more appropriate than traditional approaches to address the large scale of the problem (Stanturf et al 2014a). The probably most prominent approach is Forest Landscape Restoration (FLR) as defined by the IUCN (IUCN and WRI, 2014), a concept that focuses on restoring forested landscapes rather than individual sites. Landscape-level thinking requires the balancing of different land uses and stakeholders. The FLR approach focuses on the restoration of ecological function and strategies are not limited to traditional restoration to a “natural” state but can include any other combination of species and land. Restored landscapes increase ecosystem goods and services for local communities and but have global implications with increased carbon storage capacities. Restoration strategies are based on local conditions, knowledge and traditional land use. FLR actively engages and involves stakeholders and goals and practices are aligned with their values to improve livelihoods. Restored landscapes explicitly include many land uses such as agroforestry, managed forests and protected land (IUCN and WRI, 2014).

Thinning is commonly used in forest restoration to increase spatial heterogeneity and improve ecological function (Fajardo et al., 2007; Versluijs et al., 2017). Another restoration strategy with growing importance is the re-establishment of fire regimes in forests that historically had frequent low intensity fires, but where fires have been suppressed in the past decades. This often includes removal of fuel and mechanical thinning to reduce fuel loads before prescribed burning, which may otherwise lead to unwanted high intensity fires.

Measures to prepare forests for future conditions or transforming degraded forest ecosystems to functioning systems can include assisted migration of tree species and even introduction of non-native species that can fulfill similar functions to historic species that may not be able to persist into the future due to climate change. In Canada, assisted migration is being tested and considered for *Pinus albicaulis* (Whitebark pine) (Mclane and Aitken, 2017).

Focusing on ecological function can help avoid unsustainable goals and objectives in the light of climate change (Stanturf 2014). Just as in ecological restoration more generally, ecological forest restoration is moving away from the idea of a historical baseline, and it is becoming increasingly common to work towards a functioning ecosystem that fulfills a specific set of functions. This may include planting non-native genotypes or species and can include silvicultural management strategies (e.g., restoration forestry) since there can be large overlap between silviculture and forest restoration. Methods for forest restoration are mainly based on planting, but increasing focus is placed on soil, hydrology, and fire regimes. Especially in developing countries that are part of the REDD+ there is an increasing focus on social aspects of restoration on ecological functions like food production and firewood.

Uncertainty remains about whether common forest management methods like thinning are effective in improving structural diversity, especially if model systems are lacking. Here we focus on a restoration project in a provincially and globally endangered coastal Douglas-fir ecosystem on Galiano Island, British Columbia (Kingsberry et al., 2010). The restoration aimed to

"…increase carbon sequestration on the site […] increase biodiversity, improve ecosystem health and enhance the site's ability to adapt to the impacts of a changing climate." (Kingsberry et al., 2010, p. 134). In 2002, the local land trust, the Galiano Conservancy Association (GCA) created a restoration plan for a 61.5-hectare property it owned, and restoration treatments happened in 2003 and 2006. Management included pre- and post-assessments of the site and the establishment of permanent plots for continued monitoring (Gaylor et al., 2002). The site provides an opportunity to assess the effects of small-scale restoration on forest stand dynamics. In the absence of monitoring, it remained unknown how effective this restoration project was in increasing biodiversity, improving ecosystem health, and enhancing the site's ability to adapt to the impacts of a changing climate.

We investigated the performance of the restoration treatments in providing improved structural diversity by assessing the present plant composition and forest canopy structure of the restoration forest and adjacent control areas. We hypothesized that: 1) the treated areas will show elevated stand height, increased diameter growth, lower stem density, higher diversity in understory plant species, higher volume and diameters of coarse woody debris (CWD), and higher percentage cover of understory vegetation than the un-treated areas; we generally expected a higher spatial variability in the treated areas; and 2) both un-treated and treated areas will show lower diversity, volume and diameters of CWD, and percentage cover of understory vegetation than the reference stand.

2. Methods

2.1. Study Site

The study area is located along the Strait of Georgia, a major inlet of the Pacific Ocean between Vancouver and Vancouver Island on Canada's West Coast (figure 2-1). The study area is situated in the heart of the moist-maritime Coastal Douglas-fir bio-geoclimatic zone (CDFmm) (Krakowski et al., 2009). Relatively steep slopes and elevations from sea level up to about 140 m characterize the topography of the area.

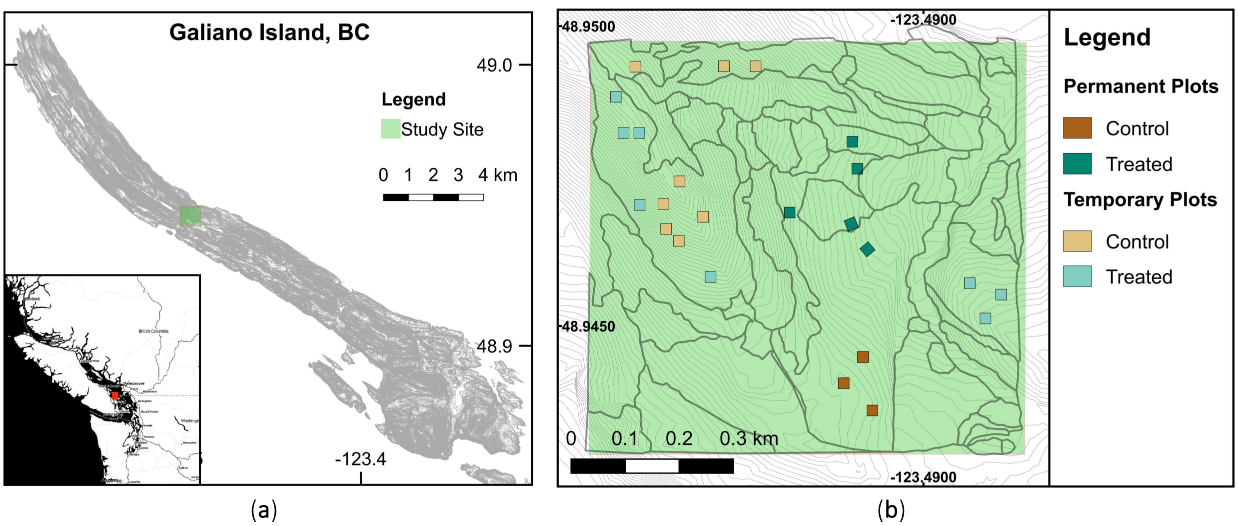


Figure 2-1: (a) Location of Galiano Island in Western Canada and study site on Galiano Island, British Columbia, Canada. (b) Overview of the study site with permanent and temporary plots

Old forests in the area are characterized by a moderately open to closed canopy of *Pseudotsuga menziesii* (Mirb.) Franco (Douglas fir), with some *Abies grandis* (Douglas ex D. Don) Lindl. (grand fir) and *Thuja plicata* (Donn ex D.) Don (Western red cedar). The understory is dominated by *Mahonia nervosa* (Pursh) Nutt. (dull Oregon-grape), *Gaultheria shallon* Pursh (Salal), *Holodiscus discolor* (Pursh) Maxim. (oceanspray), *Rubus ursinus* Cham. & Schltdl. (Pacific trailing blackberry), *Trientalis borealis* Hook. (broad-leaved starflower), *Polystichum munitum* (Kaulf.) C. Presl (sword fern), and *Pteridium aquilinum* (L.) Kuhn (bracken fern). The moss layer is dominated by *Eurhynchium oreganum* (Sull.) A. Jaeger (Oregon beaked-moss), *Rhytidiadelphus triquetrus* (Hedw.) Warnst. (electrified cat's-tail moss) and *Hylocomium splendens* (Hedw.) B.S.G. (step moss) (Green and Klinka 1994). Sites are relatively dry and soils with very poor to medium nutrient regimes (Pojar et al., 2004).

The study site was partially clear-cut logged in 1967 and then again in 1978. Only about 4 % of the 61.5 ha were left intact after the two forestry passes (Gaylor et al., 2002b). Remaining coarse woody debris was bulldozed into piles (windrows), set on fire. but did not combust fully. These windrows were not replanted and some remain visible on the site.

After both cuts the open areas were re-planted with *P. menziesii* seedlings from off-island (Gaylor et al., 2002b). The canopy now consists of *P. menziesii* with some *Alnus rubra* Bong. (red alder), *Acer macrophyllum* Pursh (bigleaf maple), *A. grandis*, and *T. plicata*. The restoration treatments were planned carefully with the help of a forest manager and carried out entirely by hand. Treatments included pulling of trees to mimic natural soil disturbance and gap creation, topping trees to create gaps and establish snags. Girdling trees caused a slower death of some trees and created food trees for wildlife as well as delayed gaps which were intended to extend the effects of the treatments longer into the future. About half the study site was restored between 2003 and early 2006. In treatment areas about 50% of the trees were culled (min 40%, max 60%) by girdling, pulling, or topping.

2.2. Permanent plots

We used eight permanent plots established by the GCA. Five plots were in areas where restoration treatments took place (TR1 – TR5), and three control plots outside the treatment areas (CO1 – CO3). As a reference, we used two permanent plots in a neighbouring mature Douglas-fir forest (MA1 and MA23) that are part of a 1-hectare biodiversity monitoring plot that was laid out by the GCA following the *Terrestrial Vegetation Monitoring Protocol* by the Ecological Monitoring and Assessment Network (Roberts-Pichette and Gillespie, 1999). All plots in the study were 20 x 20 m as suggested for young, even-aged stands (Roberts-Pichette and Gillespie, 1999). Quadrat side A-B was placed square to the general slope (parallel to the overall contour lines), and all corners A-D were marked with metal pins (figure 2-2). The coordinates of the permanent plots were recorded by the GCA with a TRIMBLE handheld GPS device. Photographs of the sites helped with re-identification of the sites. All trees were tagged with a unique ID for identification during the installation of the original plots. For plots where we were not able to find all four metal pins, we re-installed the missing marker using two measuring tapes and a compass. Additional to the tree mapping according to Roberts-Pichette and Gillespie (1999) the GCA collected data on soil type, vegetation percentage cover by species, slope, and coarse woody debris (CWD).

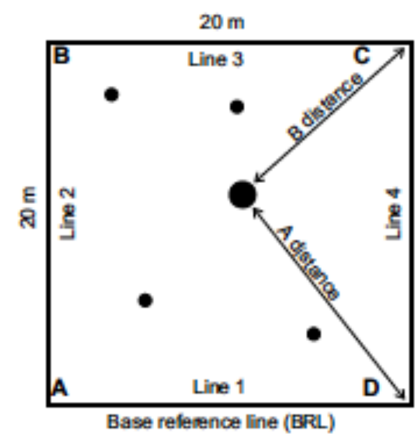


Figure 2-2: Layout of permanent plots and assessment of tree location, according to the protocol suggested by Roberts-Pichette and Gillespie (1999)

2.3. Field Methods

We repeated a full assessment of all ten permanent plots. We measured the diameter at breast height (DBH) of all trees, estimated vegetation percentage cover by layer, assessed length and diameter of all pieces of coarse woody debris (CWD) with a diameter larger than 7.5 cm, and retrieved six depth measurements for L, F, and H layer (B.C. Ministry of Forests and Range and B.C. Ministry of Environment, 2010).

As the number of permanent plots was relatively small, we set up another sixteen temporary sampling plots in other parts of the property with comparable ecological site conditions; eight plots that were treated in the same way and at the same time as the treated permanent plots (NTR1 – NTR8) and eight control plots in untreated areas of the study site (NCO1 – NCO8). These samples had a simplified sampling design (no CWD data and DBH categories, instead of exact diameter). We randomly distributed the temporary plots in pre-mapped treatment and control areas, using QGIS' "random points" tool (QGIS Development team, 2018).

We measured length and the center diameter of all pieces of CWD with diameters larger than 7.5 cm (B.C. Ministry of Forests and Range and B.C. Ministry of Environment, 2010). The sampling of understory vegetation followed the guidelines described in (B.C. Ministry of Forests and Range and B.C. Ministry of Environment, 2010). We assessed species by layer and percent area cover in the plot.

The DBH of all trees was obtained in the sample plots. We measured diameters of snags, but did not include these measurements in the basal area calculations. We re-sampled about five trees per plot for height, crown width, and depth, to estimate the live crown percentage, with the exact number depending on the previous assessments. In plots where many of the previously measured trees had died, we replaced the trees with trees of similar size. DBH were measured with a standard circumference tape, tree height with a Nikon Forestry Pro laser rangefinder. In addition, we recorded tree status according to (B.C. Ministry of Forests and Range and B.C. Ministry of Environment, 2010).

2.4. Analysis

Four datasets were used in the analysis. A "temporary" dataset included all data points of the permanent plots in 2017 and data from all 16 temporary plots (n_{PlotTR} = 13, n_{PlotCO} = 11), a "permanent" dataset included eight permanent plots (n_{PlotTR} = 5, n_{PlotCO} = 3) on the study site and data points from 2007 (shortly after restoration treatments) and 2017. A "height" dataset with 42 heights (n_{Fd} = 35, n_{Dr} = 7) was used for the analysis of tree heights and finally a "vegetation dataset with percentage cover by species for all vascular plants in the permanent plots (n_{PlotTR} = 5, n_{PlotCO} = 3). The permanent dataset was used for calculation of DBH growth and CWD calculations. The permanent dataset therefore is a subset of the temporary dataset. The temporary dataset only includes diameter, height, status, and species of trees, and vegetation percentage cover by layer. The temporary dataset allowed assessment of diameter distribution, vegetation analysis and tree heights.

All statistical analysis was done using R statistical software (R Core Team, 2017). For CWD, we compared CWD volume and number of CWD pieces per plot using an ANOVA. A Shapiro-Wilk test for normality of volumes and count of CWD pieces did not lead us to reject the hypothesis that the samples come from a normal distribution (p_{Vol} = 0.7193, p_{No} = 0.3642), and a visual inspection of the distribution confirmed this assumption. We therefore used simple linear regression models with volume (count) as our response variable and treatment, plot ID and year of assessment as explanatory variables. We did not adjust for the unequal sampling size (5 treated, 3 control).

Vegetation data were examined with R's mvabund package using the ManyGLM function (ManyGLM; R-package, (Wang et al., 2012)). Mvabund addresses the mean-variance relationship of multivariate data by fitting a generalized linear model (GLM) to every plant species individually. Assumptions of the model are also easier to interpret in a model-based framework. A negative-binomial distribution was used to account for the high number of zeros in the vegetation data. The residuals showed an even spread. We calculated the Shannon Index for each plot individually and averaged the value by treatment. This did not address the uneven sample size.

To test for effects of treatments on canopy structure, we compared tree height, density (number of living trees per plot), diameter, and basal area between treatments with mixed effect linear regression models, after using the Shapiro-Wilk test for normality and visual inspection of the variables. To avoid pseudo replication and to account for the unbalanced sampling design, the Plot ID was included as a random effect in the models. DBH was modelled only for the two most common species *P. menziesii* (n_{Fd} = 725) and *A. rubra* (n_{Dr} = 40) individually and height was modelled with the smaller subsample of about five trees per plot (n_{Fd} = 35, n_{Dr} = 7). Sampling sizes varied strongly between tree species (see Figure 2-4 (b) below) and would have affected the model outcomes. All tree species other than *P. menziesii* and *A. rubra* had sample sizes that were too small for statistical analysis and did not appear in all plots. Plot based data (basal area, number of snags and density) was modelled including all tree species.

To make predictions about the effects of treatments on growth we calculated the change in diameter between 2007 and 2017 ("DBH growth") for *P. menziesii* (n_{Fd} = 178) and *A. rubra* (n_{Dr} = 40). Since there are only historical data for permanent plots, diameter growth analysis was limited to trees in the eight permanent plots on the study site. All dead trees were excluded from the analysis because of uncertainty of mortality year. Effects of treatments on DBH growth, were modelled using a generalized linear mixed effect model. To account for unbalanced samples (n_{PlotCO} = 3, n_{PlotTR} = 5) and avoid pseudo-replication, we included the plot ID as a random effect in our model. Calculations were done with the 'nmle' package in the statistical software R (Pinheiro et al., 2017). All other individual tree based analysis was done using only the two most common tree species *P. menziesii* and *A. rubra* with two individual models.

3. Results

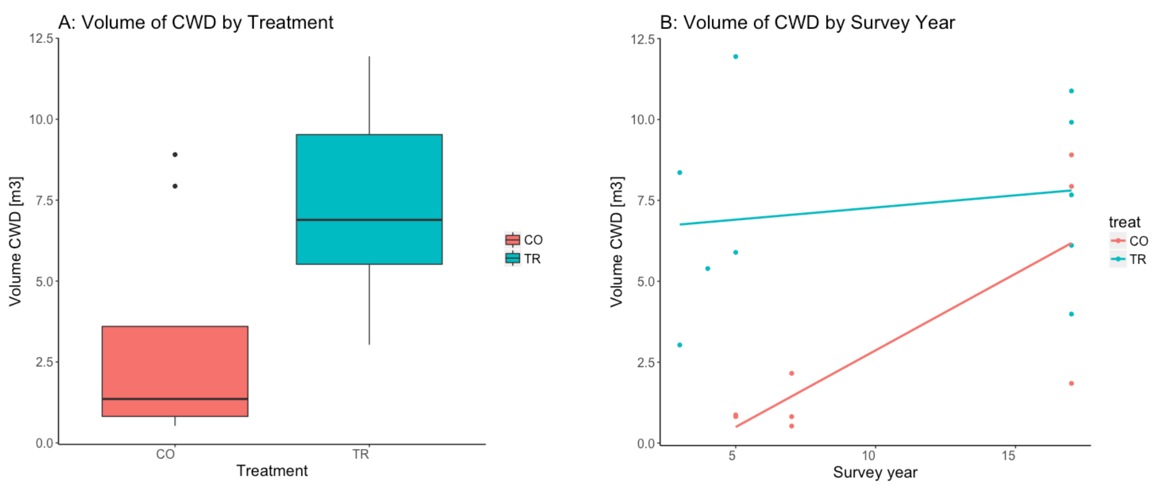
Treated areas showed a higher diversity and higher cover of understory plants, were more structurally diverse, and had higher volumes of CWD. We were however not able to connect all of these differences to restoration treatments. Tree heights and basal area in treated areas were lower than expected. Table 2-1 summarizes all results.

Table 2-1: Summary of all measures of stand structure and diversity by treatments. Fd = Douglas-fir, Dr = Red Alder. Values are group means.

	CWD Vol [m^3 ha^-1]	CWD Dia [cm]	Cover Herb [%]	Cover Shrubs [%]	Height Fd [m]	DBH Fd [cm]	DBH Dr [cm]	Basal Area [m^2 ha^-1]	Density [ha^-1]	Snags [ha^-1]
Treated	192.81	13.79	6.62	4.53	25.24	24.62	13.78	35.58	800.44	568.10
Control	155.69	19.41	5.27	4.18	26.62	23.30	16.23	42.78	1073.88	407.14
Mature Reference	NaN	NaN	8.50	1.00	51.81	86.60	71.50	88.32	311.56	133.11
Estimate	4.1412	-11.2123			-2.5155	4.22	-2.83		8.3493	13.0036
Std. Error	2.2207				2.0787	0.96	1.98		1.2793	2.1252
t-value	1.8650				-1.2100	4.39	-1.43		6.5270	6.1190
p-value	0.10	0.00			0.23	0.00	0.15	0.10	0.00	0.00

3.1. Coarse Woody Debris

Volume of CWD has increased for CO and TR in the last ten years (figure 2-3(b)). CO plots showed a strong increase in CWD, but volumes were still lower than in TR plots (figure 2-3(a)). Results were similar for the number of pieces of CWD. Both CO and TR showed a steady increase in number of pieces and they have very similar numbers. The ANOVA showed a significant difference in volume of CWD by treatment (mean Sq = 83.539, F = 14.1640, p = 0.004461) and a significant difference on the number of pieces (Mean Sq = 2030.6, F = 6.9740, p = 0.0268767).



(a) (b)

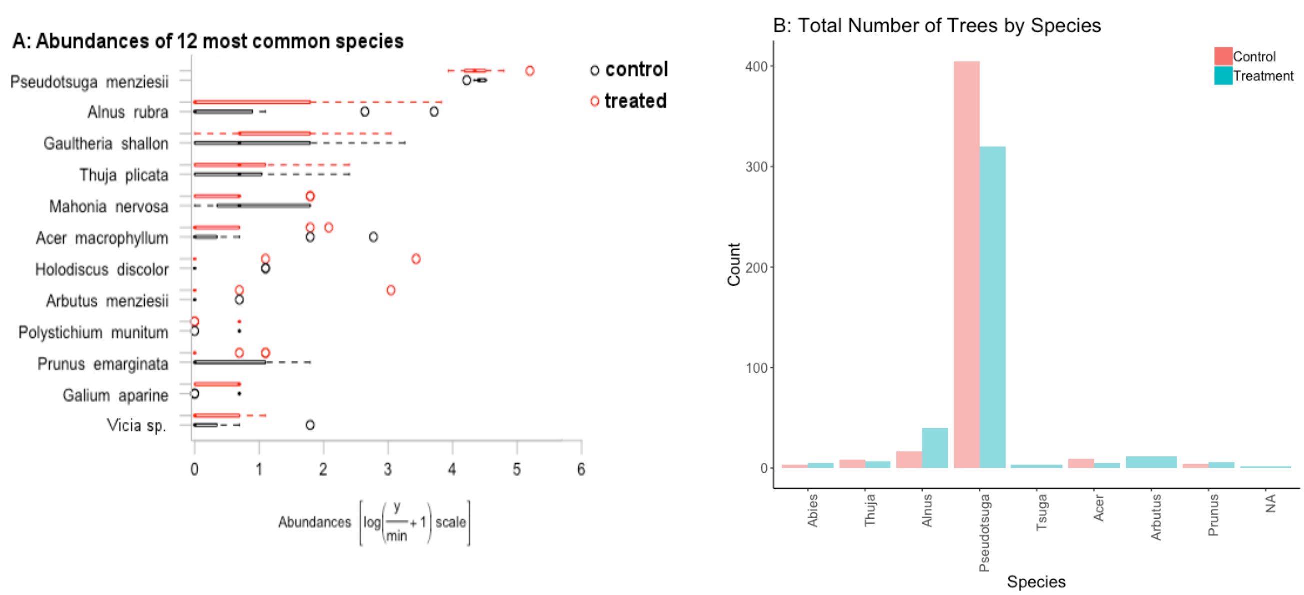
Figure 2-3: Comparison of volumes of coarse woody debris (CWD). CO = untreated control, TR= treated. **(a)** Boxplot of CWD by treatments. The lower and upper hinges correspond to the first and third quartiles (the 25th and 75th percentiles). Whiskers extend 1.5*IQR from hinge. **(b)** Volume of CWD by survey year. Each dot represents one plot.

Most pieces of CWD had small diameters, and differences in diameter distribution between treatments were negligible. The proportion of CWD with small diameter (10 - 30cm) showed an increase for both CO and TR.

3.2. Understory Vegetation

All species found in the study are species common to the area. *Cytisus scoparius* (L.) Link (scotch broom), a common invasive species in the area was present, but only in very small numbers. The orchid species *Epipactis helleborine* (L.) Crantz (broadleaf helleborine), a common exotic species, was present as well. *Cirsium arvense* (L.) Scop. (Canada thistle) and *Cirsium vulgare* (Savi) Ten. (bull thistle), both exotic thistles, were present. Single individuals of *Ilex aquifolium* L. (English holly) another exotic species, were present in two plots.

Most species appeared in both CO and TR plots, with similar abundances. *M. nervosa* showed a similar mean but higher abundances in CO plots, *Prunus emarginata (Douglas ex Hook.) D. Dietr.* (bitter cherry) was more abundant in CO and *Galium aparine* L. (cleavers) was less abundant in CO (Fig. 2-4(a)). All twelve most abundant tree and shrub species were common species. Of the six tree species, *P. menziesii* was the most abundant in all plots (figure 2-4(b)).



(a) (b)

Figure 2-4: **(a)** Abundance of 12 most common plant species in the study plots. **(b)** Species count by treatment.

The mean Shannon Index was higher for TR plots (0.92) than it was for CO (0.78) and highest for MA plots (1.49).

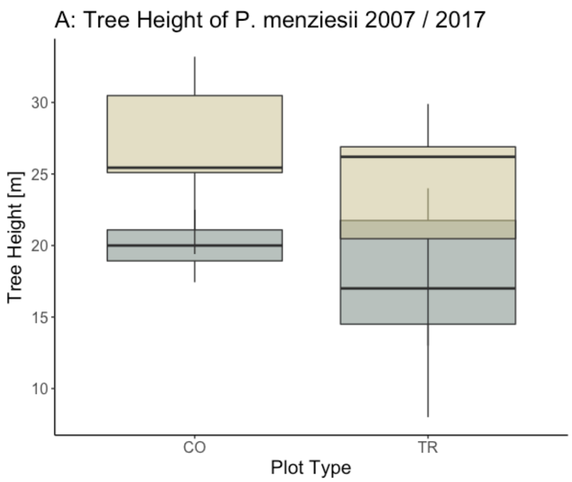
3.3. Diameter, Height, Density, Basal Area and Growth

The most common canopy tree species was *P. menziesii*, with some *A. rubra* and few *Arbutus menziesii* (arbutus), *P. emarginata*, *A. grandis*, *A. macrophyllum,* and *T. plicata* (Fig. 2-4(b)).

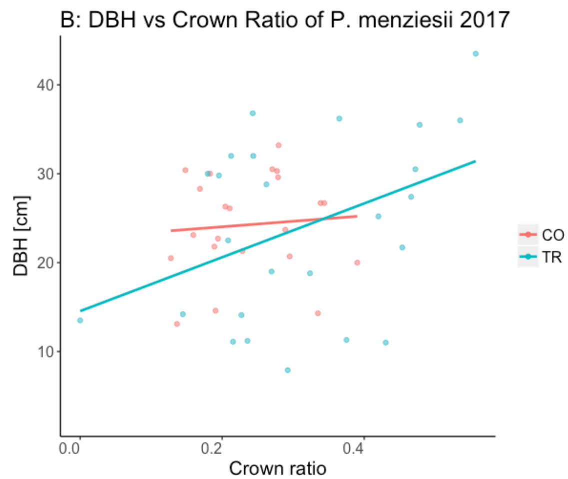
3.3.1. Tree Height

Tree height for *P. menziesii* increased for all plots between 2007 and 2017. TR plots showed a wider range of tree heights and a lower mean tree height (figure 2-5(a)). The results of a linear mixed effects model suggest a strong negative effect of treatments on tree height (Estimate = -5.14695, p = 0.005294). DBH was another strong predictor of height (Estimate = 0.43165, p = 6.462e-13).

Crown ratio ($CrRt = \frac{\textbf{Tree Height}-\textbf{Branch Height}}{\textbf{Tree Height}}$) was on average smaller in the CO plots, and TR supported lower live branches (figure 5(b)). The analysis with a linear mixed effects model showed a small but insignificant negative effect of treatments on crown ratio, however (estimate = -0.0538480, p = 0.557812). The only significant predictor of crown ratio was DBH (estimate = 0.0091957, p = 0.000487). Effects of DBH were minimal. The correlation between DBH and crown ration was stronger for trees in TR plots, than for trees in CO plots.



(a)



(b)

Figure 2-5: Comparison of tree heights by treatment and survey year.**(a)** Tree height by treatment in 2007 (grey) and 2017 (beige). The lower and upper hinges correspond to the first and third quartiles (the 25th and 75th percentiles). Whiskers extend 1.5*IQR from hinge. **(b)** Tree height by crown ratio of *Pseudotsuga menziesii* trees.

3.3.2. Density, Basal Area and Snags

Mean density for TR was 800.44 trees/ha and 1073.88 trees/ha for CO plots. Density decreased for both treatments, it was lower for TR than CO plots in 2007 and remained lower in 2017 (figure 2-6(a)). Densities by treatments were more similar in 2017 than they were in 2007. Basal area differed strongly in 2007 (shortly after the treatments) since many trees were culled in TR plots (figure 2-6(b)). Basal area increased for both treatments, but the increase was stronger for TR (from 21.91m^2 ha^{-1} to 39.97 m^2 ha^{-1} for TR and from 31.74 m^2 ha^{-1} to 44.09 m^2 ha^{-1} for CO).

The mean number of snags per plot decreased from 2007 to 2017 for both treatments and spread decreased as well (figure 2-6(c)). Diameter of snags increased for both treatments (from 10.67cm to 11.93cm for TR and from 7.97cm to 11.88cm for CO).

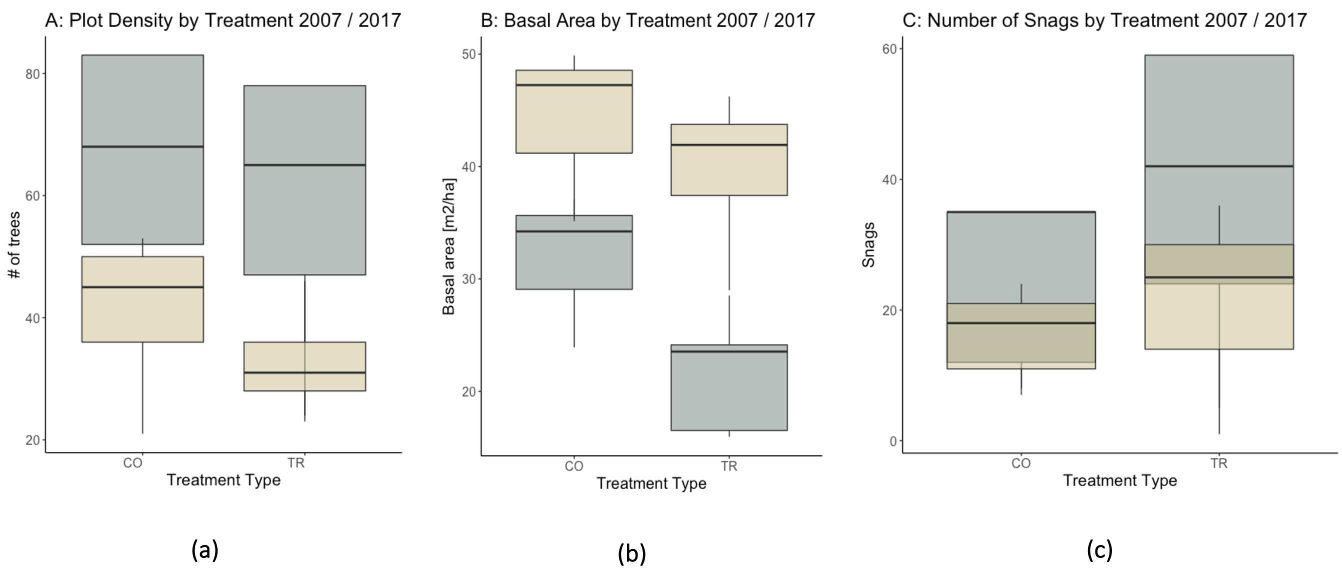


Figure 2-6: Density, basal area and snags of all species by treatment in 2007 (grey) and 2017 (beige). The lower and upper hinges correspond to the first and third quartiles (the 25th and 75th percentiles). Whiskers extend 1.5*IQR from hinge. **(a)** Density by treatment. **(b)** basal area by treatment. **(c)** Number of snags by treatment

3.3.3. Diameter Distribution and Growth

Mean DBH increased for both treatments. Mean DBH was higher for TR plots than for CO in 2017, but was lower in 2007 (figure 2-7(a)). This increase in mean DBH explains the increase of basal area in TR plots even with a decrease in density.

Mean diameter growth differed between TR and CO plots ($Growth_{COmean}$ = 0.35 cm a^{-1}, $Growth_{TRmean}$ 0.54 cm a^{-1}). The mean for both treatments was very similar but there were some trees with very high growth rates in TR plots (figure 2-7(b)). Overall, diameter growth was higher for trees with larger diameter.

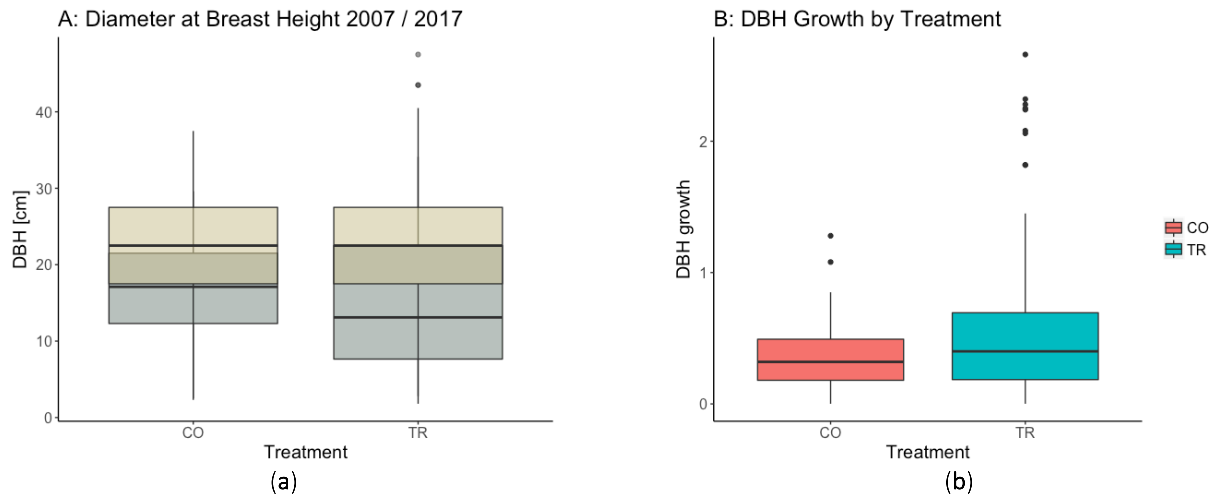


Figure 2-7: The lower and upper hinges correspond to the first and third quartiles (the 25th and 75th percentiles). Whiskers extend 1.5*IQR from hinge. **(a)** Boxplot of diameter at breast height in 2007 (grey) and 2017 (beige) by treatment; **(b)** Diameter growth per year by treatment. max CO = 1.28 cm a-1, mean CO = 0.347975 cm a-1, max TR = 2.66 cm a-1, mean TR = 0.54 cm a-1

We were unable to fit a model that properly explained the variation in diameter growth. In the generalized linear mixed effects models, treatment only had a very small and statistically insignificant effect on *P. menziesii* (Estimate = 0.1770389, p = 0.408) and a small but significant effect on *A. rubra* (Estimate = -1.90892, p = 0.010706). For *P. menziesii*, the previous diameter in 2007 had the only significant effect on diameter growth (Estimate = 0.0870893, p = 3.97e-06). The diameter growth of *A. rubra* was mainly influenced positively by percentage cover of substrate water (Estimate = 2.97826, p = 0.000158) and negatively by the slope gradient (Estimate = -0.21722, p= 0.001688).

4. Discussion

We found that treated areas showed a higher diversity and cover of understory plants, were more structurally diverse, and had higher volumes of CWD. We were however not able to connect all of these differences to restoration treatments. Moreover, tree heights in treated areas were lower than expected.

Even though we found a lower density for TR plots, lower basal area, a larger crown ratio (longer crowns), higher diameter growth, higher volumes of CWD a higher percentage cover of understory plants, and a higher diversity of plant species, these differences were relatively small and in most cases not statistically significant. The parameters were closer to values in our reference stand (MA) in TR plots than they were in CO and diameter and diameter growth had a wider range for TR plots than for CO plots, which is a sign of increased structural diversity, which may hint at positive effects of the treatments, but could not be confirmed by statistical models. Other structural parameters were not showing the expected results. Mean tree heights were lower in treated plots than in the control.

Even though volumes of CWD were still higher in TR plots, control plots gained large amounts of CWD volume in the last 10 years whereas volumes in TR only showed a small increase. This is a sign that the stand underwent its stem exclusion phase, where dominant trees out-shade sub-dominant trees and ultimately results in a higher tree mortality (Spies and Cline, 1988). Restoration treatments may have slowed down this development, decreasing the rate of dying trees and consequently CWD on the ground for TR plots. The higher volumes in TR are most likely due to remaining debris from the windrows that were re-distributed throughout the TR plots as part of the original restoration efforts. Generally, CWD volume increases with the age of the forest and the productivity, and CWD volumes in the neighbouring mature forest were indeed higher. We considered the higher volume of CWD in TR plots therefore as a success. According to Feller (Feller, 2003) there are no studies on CWD volume in CDF old-growth forests, and therefore we were not able to compare the measured amounts with “ideal” values. The number of snags decreased for both treatments, most likely caused by decay of small diameter snags which were now part of the CWD on the ground.

TR plots showed significantly more trees of *A. rubra. A. rubra* is a nitrogen fixer and its leaf litter helps improve soil quality by increasing nitrogen content (Tarrant and Miller, 1963). Mixed leaf litter of *P. menziesii* and *A. rubra* decomposes faster that litter alone (Fyles and Fyles, 1993). The higher number of *A. rubra* trees is not a result of the restoration treatments: the trees were already present before the treatments.

The basal area of both TR and CO plots increased, but treatments increased the basal area of TR plots more than in the CO plots. Density of trees decreased for both TR and CO, which supported lower life branches and therefore longer crowns. Our results are in line with other studies in a variety of forest ecosystems that have found that thinning decreases tree density and basal area (Battaglia et al., 2010; Fajardo et al., 2007; Harrod et al., 2009; Stephens and Moghaddas, 2005; Vaillant et al., 2009). Bailey and Tappeiner (Bailey and Tappeiner, 1998) found that live crown ratio was significantly higher in thinned Douglas-fir stands than in un-thinned stands, which corresponds with our findings of longer crowns in TR plots. Other studies on thinning treatments in Douglas-fir forests found that thinning had no effect on basal area of *P. menziesii* (Wilson et al., 2009). We saw similar results than Wilson and Puettmann (Wilson and Puettmann, 2007) who showed that thinning in young *P. menziesii* stands in western Oregon and

Washington, United States increased spatial variability, supported lower live branches and had greater growth.

Unexpectedly, diameters of the dominant tree species *P. menziesii* were only slightly higher, and mean tree height of all species was lower for TR plots than it was for CO. Other studies have found that thinning increased diameter (Harrod et al., 2009; Vaillant et al., 2009) and height (Battaglia et al., 2010; Harrod et al., 2009; Stephens and Moghaddas, 2005; Vaillant et al., 2009). Thinning increases the amount of resources available to remaining trees which is expected to increase their growth. This effect appears to not have been strong enough to be reflected in our results.

We identified a higher diversity of vascular plants in TR plots but did not find any old-growth associated understory plants in TR or CO plots. A study by Lindh and Muir (2004) found that thinning of young Douglas-fir forests increased the cover of old-growth associated understory plants, but did have no effect on basal area of *P. menziesii* (Wilson et al., 2009), an effect we were not able to confirm.

In the light of our hypotheses, we were surprised not to see stronger signals across most indices for the treated plots. This may have several reasons. First, with five permanent treatment plots and three permanent control plots, the study design was unbalanced. The outcomes may have been affected, even though we tried to account for the unbalanced design by choosing appropriate models. We did not reanalyze the data using a weighted approach to the unbalanced design, but this will be undertaken prior to any further publication of these results. A preliminary re-examination of the data suggests the restoration response many in fact be higher than accounted for in the present analysis.

Second, the treatments did not show a significant effect on the diameter growth even though the diameter growth mean was significantly higher in TR plots. This may have been caused by a poor model fit. None of our included variables were able to explain the variation in DBH growth well. Higher diameter growth may be caused by better soil or moisture conditions in the TR plots, instead of the thinning treatments. TR and CO plots differed in their structural diversity before the restoration treatments. Particularly mean diameter, density and species distribution differed significantly between CO and TR before the treatments and made it harder to fit appropriate models.

Third, even though the data spanned ten years, the time difference may not have been enough to show significant differences. Forests are very long lived ecosystems that react slowly to changes. Consequently, we may see stronger effects over time (Wilson and Puettmann, 2007).

On the other hand, young forest stands are dynamic systems, that react quickly to disturbances. Young, dense stands undergo “self-thinning”, a process that significantly reduces stem density in the years after canopy closure. On our study site, natural death of trees significantly reduced stem density between 2007 and 2017 on untreated control sites (figure 2-6 (a)). Many of the canopy gaps the GCA created were relatively small and were closed by surrounding trees relatively quickly. Gap sizes of the restoration treatments may therefore not have been large enough. This is supported by an overall similarity between treatments.

5. Conclusion

Past studies have suggested that restoration cannot always return ecosystems to a previous "natural" state (Benayas et al., 2009; Jones et al., 2018). If possible, efforts should be focused on the most important areas and most effective treatments, but when resources for restoration treatments are limited, it may be prudent to simply remove the disturbance and let natural succession do its work.

Given the right conditions, natural regeneration or passive restoration, can provide ecological and social benefits at significantly lower costs than active restoration (Chazdon et al., 2016). This is however limited by political, social and economic barriers and depends on the severity of the disturbance (Chazdon et al., 2016). Additionally, passive restoration allows for less engagement of local stakeholders in in the restoration process, and therefore removes the possibility of creating jobs and a deeper understanding of the ecological processes involved in the restoration treatments.

Based on our findings we conclude that even moderate pre-commercial thinning with intensities of approximately 50% of trees in young Douglas-fir forests can improve structural diversity and biodiversity, but single treatments at a young age are not enough. Young forest stands show fast growth and high flexibility towards disturbances. Especially when resources for restoration treatments are limited it may therefore be beneficial to focus on the creation of larger gaps and leave the remaining stand untreated. This creates a heterogeneous matrix and gap creation have been shown to improve biodiversity (Muscolo et al., 2014). Our study can help focus often limited resources in ecological restoration to where they can have the most impact. Given that the last restoration treatments happened more than ten years ago and that the forest is still relatively young, a continuation of treatments could further improve the structural diversity of the study site.

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Chapter 3: The Potential for Hobbyist Unmanned Aerial Vehicles in Ecological Restoration

0. Abstract

We explore the potential of relatively inexpensive hobbyist unmanned aerial vehicles (UAV) as a tool in in ecological restoration for small and not-for-profit organizations. First, we summarize existing UAV technology, current commercial and scientific applications and future developments. Then UAVs are evaluated for their application in improving restoration outcomes. Sensors available for the smallest class of UAVs include digital cameras, infrared cameras, multi- and hyperspectral cameras and LiDAR sensors. If applied correctly, UAVs can increase the amount of available data before, during and after restoration and therefore help improve scientific understanding of ecological processes involved in restoration. This can help in setting more effective, efficient and engaging restoration goals and better monitor if these goals have been met. UAVs can increase access to remote areas and decrease disturbance of sensitive ecosystems. Regulations, limited flight time and processing time remain important restrictions on UAV use. The loss of field expertise and hands-on experience can be a serious concern for volunteer education. Resulting data and available sensors for hobbyist UAVs presently limit their application for monitoring and scientific research.

1. Introduction

Remote sensing and aerial photography provide access to larger spatial coverage and detailed analyses in ecology (Aplin, 2005). Since the 1970, satellite-based data have provided improved resolution, wider temporal and spatial coverage, multiple data types, and relative affordability.

Consequently, remote sensing has become an integral part of ecological research and informed restoration planning (Lovitt et al., 2018). Unmanned aerial vehicles (UAVs), commonly known as drones, are the newest development in remote sensing (Adão et al., 2017). UAVs are small, remotely controlled systems, capable of autonomously following a pre-programmed flight path and usually carry one or more sensors, most commonly digital cameras. Both, UAV's and their sensors, are affordable compared with many remote sensing technologies. Unmanned aerial systems (UAS) usually consist of one or more UAVs, equipped with sensors and a ground control station (Pádua et al., 2017). The resolution of images obtained with UAV's is comparable or better than that obtained with traditional remote sensing instruments, which makes up for the lack of vast landscape coverage (Anderson and Gaston, 2013). Many remote sensing data analysis software can be used to analyze UAV data, while special software is available to extract the full potential of UAV images.

Since UAV's are easy to use and offer improved spatial and temporal resolution at a very low cost (Anderson & Gaston, 2016), they are employed for commercial applications such as surveying, agriculture, construction, photo- and videography, replacing or enhancing other remote sensing methods (Drone Deploy, 2018). UAVs have been used for research in fields as varied as hydrology and geology, measuring stream flow (Tauro et al., 2016), water levels (Bandini et al., 2017), and volcanic activities (Amici et al., 2013). Some studies have used UAVs in ecological research (Reif and Theel, 2017). Even though ecology represents a much smaller market for UAV products than forestry or agriculture and, hardware and software applications can and have been adapted for ecological research (Anderson and Gaston, 2013; Crutsinger et al., 2016). Relatively inexpensive UAVs for hobbyists have risen in popularity in the last years and are now widely available. This has created interest in using a UAV with smaller and not-for-profit organizations.

Ecological restoration is not limited to a specific ecosystem and can take place in any kind of system, from coral reefs (e.g. Rinkevich, 2014), to grasslands (Barr et al 2017), wetlands (Kelly et al 2011), rivers (Palmer et al. 2005), tropical, temperate and boreal forests (Zahawi et al, 2013, Dumroese, 2015, Hekkala 2014). Goals of ecological restoration are not just based on current conditions, but are informed by historical and future biotic and abiotic conditions (Suding et al. 2015). Planning restoration projects therefore requires a range of information about biotic, abiotic, social and cultural factors affecting the ecosystem that is to be restored.

Keenleyside *et al.* (2012) describe three principles of successful ecological restoration: effectiveness, efficiency and engagement (Keenleyside et al., 2012). Before starting a restoration of a disturbed site, it is important to set realistic and achievable goals, which are then further refined by measurable objectives (Keenleyside et al., 2012). The goals will then inform planning, implementation and monitoring of the project and allow for quantitative assessment of the project success. Goals can be effective when focusing on project specific values, efficient by considering specific constraints on the project, and engaging when considering that understanding and support from local stakeholders are crucial for the long-term success of the project. Successful projects often require adaptive management, where monitoring allows for detection of potential problems and revision of restoration strategies. The required monitoring can take many forms and depends on restoration goals and objectives.

After defining goals, restoration practitioners and researchers encounter many challenges achieving them, and often there is no one definite way of achieving a restoration goal. Based on eight recent studies published in Restoration Ecology, Matzek *et al.* (2017) suggested five overall directives that can help restoration projects in achieving their goals more effectively. The authors suggest **1) to follow ecological theory, 2) harness technological advances, 3) reject dogma, 4) encourage self-critique and 5) respect stakeholders' limitations** to improve future performance of restoration projects.

In this article, we review the characteristics of current hobbyist UAV technology and highlight the role of UAV's in recent ecological restoration studies. We will also examine how relatively inexpensive UAV's can support the application of five directives for successful restoration projects as proposed by Matzek *et al.* (2017) as mentioned above. Finally, we will discuss the reliability of hobbyist UAV data and future developments in the field. In our review, we will focus on micro UAV's. Micro UAV's are defined by weights of less than 5kg, whereas mini UAVs weight up to 30 kg and large, usually tactical, UAVs weigh up to 150 kg (Ballari et al., 2016). Micro UAVs (hereafter simply 'UAVs') are ideal for ecological research since they are affordable and accessible platforms that are easy to handle, transport, and set-up.

2. Current UAV technology and use

Benefits of UAVs are their high spatial and temporal resolution, flexibility, accessibility, and low operational cost. They can fill the gap between satellite or airplane remote sensing that covers large areas with coarse resolution and traditional ground measurements, which are useful for very small areas. UAV's can survey areas of a few km^2 with relative ease, while larger areas are better suited for other remote sensing technologies (Cordell et al., 2017; Cruzan et al., 2016). In fact, UAV remote sensing is likely to add to or replace traditional methods in many fields and offer new opportunities for ecological assessments (Linchant et al., 2015; Pádua et al., 2017).

UAVs can be used in most climatic zones and weather conditions although rain and strong winds prevent flights. Baena et al. (2017) describe their successful use of UAVs for plant conservation in different regions of the world and ecosystems, "…ranging from Peru's hyper-arid vegetation to the dry forests of the Caribbean and finally to the humid forest of South Africa and the Brazilian Amazon." (Baena et al., 2017). UAVs have been used to study the micro-topography of Antarctic moss beds (Lucieer et al., 2012), for search and rescue operations in mountain environments (Silvagni et al., 2017), and archeological mapping in the Amazonian rainforest (Khan et al., 2017).

However, UAVs are still a very young technology and they come with inherent limitations. Citizens tend to be concerned about privacy infringements by UAV use (Winter et al. 2016, Finn et al 2014), which requires open communication of UAV applications to the local communities when working in populated areas. Due to their overhead or birds-eye perspective, UAVs are limited to surveys of parameters that are visible from above and not blocked by tree canopy or other covers. Newer sensors can penetrate canopy, but limitations of the birds-eye perspective remain. UAV sensors are also limited to data based on electromagnetic waves reflected from a surface. This excludes acoustic or chemical analysis of the study site. Direct impacts of the UAV also need to be considered, especially when flying close to the ground and when studying wildlife, which may show stress reactions to the vehicle. UAVs are becoming increasingly more affordable, but especially sensors other than standard digital cameras are still expensive in acquisition. Current quick development of technology makes technology obsolete quickly. UAV technology is diverse and we summarize current technology in respect to their application to ecological restoration:

2.1 Several types of UAVs for different purposes

UAVs can be classified in two general categories: Fixed-wing and multi-rotor powered UAVs. Fixed-wing systems can cover larger areas due to their longer flight length and faster speeds. They are generally susceptible to vibrations (Wallace et al., 2011), but are especially useful when larger areas need to be captured and the required flight times are longer (Toth and Jóźków, 2016). This makes them especially useful in agriculture and forestry applications.

Multi-rotor UAVs are currently only able to fly for 15-30 min but are more stable in flight, more flexible when flight space is limited, and can deliver higher resolution images (Cruzan et al., 2016; Pádua et al., 2017). They are especially useful in areas with limited start and landing area since they can take of vertically, and when stable images of smaller areas are required. Multi-rotor UAVs are most useful for inspection, surveying, construction, emergency response, law enforcement and cinematography, and still images (Pádua et al., 2017).

(a) (b)

Figure 3-1: Two examples of common UAVs. (a) DJI Inspire 2 multi-rotor UAV. (b) SenseFly eBee Classic fixed-wing UAV. Images were obtained from the manufacturers' websites (https://www.dji.com/; https://www.sensefly.com/)

2.2 Temporal and spatial flexibility

UAVs can provide images at a higher spatial and temporal resolution than other remote sensing technologies. The parameters for spatial and temporal resolution are almost completely set by the user and are not constrained by satellite revisiting periods or pre-determined spatial resolution (Anderson and Gaston, 2013). UAVs have been used in ecological studies with sub-centimeter resolution of intertidal reefs in Australia (Murfitt et al., 2017), and can theoretically be used for constant monitoring when several UAVs are used (Fetisov et al., 2012; Merino et al., 2012). Most research studies currently use a spatial image resolution of 1-10 cm per pixel (Ballari et al., 2016; Cordell et al., 2017; Lovitt et al., 2018), as compared to the freely available satellite data which usually has a resolution of 10-60 m per pixel for multispectral data and < 2 m per pixel for ortho-photographs (Díaz-delgado, 2017). There are still few studies of frequently repeated assessments although Vega et al. (2015) flew UAVs at four different dates throughout the cropping season. Similarly, Dempewolf (2017) determined growth of tree terminal shoots in Germany with UAV's flying repeatedly at four times throughout the growing season.

Image processing becomes increasingly faster, and near-real-time creation of 3-D models is already available for commercial applications (Stefanik et al., 2011, Lockheed Martin, 2018). This will allow for processing of the data while it is being collected and insufficient data quality due to bad image quality could be corrected while researchers are still in the field instead of having to wait to process images in the office. Near-real-time object detection has been tested in avalanche response (Bejiga et al., 2017), but could also be used in wildlife monitoring as a way of detecting nearby animals with thermal image sensor.

2.3. Affordability and Accessibility

UAVs are very affordable compared to other remote sensing methods and even consumer grade models can be platforms for scientific studies (Cruzan et al., 2016; Dempewolf et al., 2017; Marteau et al., 2017; Surový et al., 2018). Moreover, a UAV is relatively easy to use and does not require extensive training (Crutsinger et al., 2016). In fact, Smith et al. (2015) acknowledge that UAV's have driven the diffusion of remote sensing due to their affordability and easy usage.

Regulations can limit the use of UAVs, with many countries now requiring permits or limit the areas where UAVs can be used. Regulations are necessary for airspace safety. However, regulations are lagging behind rapid technological development (Stöcker et al., 2017b). Regulations vary by country and are in many cases still in development. A current summary of regulations can be found in Stöcker *et al.* (2017b), but it remains necessary to stay informed about local regulations before applying UAVs for ecological research.

2.4. Availability of open source software and platforms

Several open source kits are available in addition to commercially available UAVs. Open source software makes processing of UAV derived data widely accessible and can improve the reproducibility of analysis. Open source flight control software like ArduCopter (Robotics Inc.; http://ardupilot.org/copter/) allow for specialized set-ups and DIY solutions. Zahawi et al. (2015) used low budget UAVs with Ecosynth (http://ecosynth.org/) open-source software and an arducopter-based platform to monitor tropical forest recovery in Costa Rica. A relatively inexpensive (< $1500 US) UAV was used to quantify forest structure metrics. Zahawi et al. (2015) found that modeled tree height from UAV data was a strong predictor of tree height measured in the field. Accuracy was comparable to similar studies using LiDAR data. The authors estimated above-ground biomass and predicted frugivorous bird abundance using their canopy height data.

Lehmann et al. (2017) used hobbyist grade UAVs to map invasive species in a savannah type ecosystem in Bahia State, Brazil and used freely available software (ArduPilot Mega 2.6 (APM2.6; http://ardupilot. com); VisualSfM software (Wu, 2013); CloudCompare (http://www.danielgm.net/cc/); Quantum GIS (https://www.qgis.org/)) to manage 3-D data for point cloud creation. They wanted to encourage invasive species mapping by showing the possibilities of a UAV worth less than $2000. Similarly, Dandois & Ellis (2013) used open source software Ecosynth (http://ecosynth.org/) and Bundler (http://www.cs.cornell.edu/~snavely/ bundler/) to map vegetation spectral dynamics. Hopefully software for UAV image processing will continue to develop and represent a true alternative to commercial software as it has already happened in geographic information systems (Quantum GIS (https://www.qgis.org/)) and statistical software (R statistical software (R Core Team, 2017)).

2.5. Wide range of sensors

UAVs can be equipped with many types of sensors. While weight of sensors used to be a limitation, recent developments resulting in miniaturization makes it possible for UAV’s to carry several sensors and take images with different bandwidths and channels simultaneously (Pádua et al., 2017). Digital cameras for visible (RGB) and near-infrared (NIR) light were used most commonly in the studies citied in this article. RGB images cover the spectrum visible to the human eye (400 – 700nm) while NIR sensors capture light with longer wavelengths from 800nm to 2500nm. Most conventional digital cameras can detect infrared light after removing the infrared filter. This can be used to expand the bandwidth of RGB cameras to include near infrared light (e.g. Honkavaara et al., 2013).

Several studies have shown that multispectral data from UAVs can be used in restoration monitoring. Multispectral sensors commonly include the visible spectrum and a portion of infrared light, categorized in 5- 12 bands. The inclusion of infrared light allows for the calculation of vegetation indices like the Normalized Difference Vegetation Index scores (NDVI) or the Enhanced Vegetation Index (EVI) because plants reflect the infrared spectrum differently than most other surfaces. Michez *et al*. (2016) describe the use of visible and near-infrared orthopohotos and a supervised classification algorithm in assessments of invasive plant species abundance in two riparian forests in Belgium. Lishawa *et al*. (2017) field observations and UAV data in a study of Typha removal in the Great Lakes was assessed using NDVI, blue band reflectance and vegetation height that were well correlated to field observations (Lishawa et al. 2017). Lehmann *et al*. (2017) detected oak splendour beetle (*Agrilus biguttatus* (Fabricius)) infections by comparing NDVI data from a compact digital camera modified to detect NIR reflection. The authors used a multi-resolution segmentation and subsequent object-based classification to distinguish between healthy and invested branches and found that the classification matched a previous field survey well. Romero-Trigueros *et al.* (2017) measured citrus trees health in agricultural plantations with a multispectral camera used several flights per day.

Hyperspectral data can be used for inspection of forestry operations, wildfire detection, health monitoring, and forest preservation (Colomina & Molina 2014). Hyperspectral sensors cover hundreds or thousands of bands in narrow bandwidths (5-20nm) compared to only 5-12 bands in multispectral data. Multispectral and visible light data therefore lack spectral precision and bandwidth and are therefore not suited for the analysis of chemical and physical properties (Adão et al. 2017). However, high data volumes complicate analysis and storage of hyperspectral data (Adão et al. 2017).

Figure 3-2: RBG canopy photo of a Douglas-fir forest that was taken to assess restoration effectiveness.

Light Detection and Ranging (LiDAR) laser scanners are used in mapping of terrain and plant cover because they can penetrate plant cover. LiDAR sensors on UAVs are a recent development and are still relatively expensive and uncommon. LiDAR has been commonly used as a remote sensing tool from airplanes, but the acquisition is expensive and can take time. Wallace, Musk & Lucieer (2014) tested the use of UAV laser scanners for forest inventory. After merging point clouds from up to 19 flights for six plots the authors compared plot level metrics for tree height, and individual tree height and stem position. Their results showed that UAV laser scanning delivers results comparable to ground measurements, while being faster and being able to cover a higher number of trees than realistically possible from the ground.

Thermal images can be used for water stress assessment when combined with multispectral data (Anderson & Gaston 2013). Santesteban *et al*. (2017) determined water stress in grape vines by using an open source UAV platform equipped with a thermal camera with a pixel resolution of 13 x 13 cm. Berni *et al.* (2009) found that UAV thermal data can determine water stress in olive trees in the south of Spain by comparing it with field measurements of temperature and leaf conductance and remotely sensed canopy temperature data from airplane. UAV images were better in distinguishing tree crowns because of a higher resolution than images from airplanes. Similar methods could be used in monitoring of plant recovery after restoration treatments.

2.6. Multiple UAV image analysis software

UAV imagery often requires post-processing to be meaningful for the assessment of ecological metrics like water status, plant vigour, biomass, or disease monitoring of plants. Different sensor types allow for different applications and require different pre-processing. UAV data can be used to create 3-D point clouds, raster images, false colour images with different spectral footprints, stitched Orthophotos or thermal maps.

Orthophotos are aerial images that have been orthorectified to represent a geographical location. In UAVs, orthophotos often consist of many photos that have been merged into one image, using image stitching software. Orthophotos can be used in monitoring of rewilding projects like the Knepp wilderness in the UK (Knepp Wilderness, Thomson Ecology 2016) or in wildlife studies (Rey et al. 2017).

3-D ground or canopy models can be created from 2-D images using Structure-from-Motion technology (SfM) (Dandois & Ellis 2013). This technology, originally intended for ground-based photography is now used to calculate biomass and elevation mapping from UAV data (Nex & Remondino 2014). In SfM an image feature detection algorithm detects features across several images using image feature descriptors (Dandois et al. 2017). Those features are then represented as a point with x, y and z coordinates and the final result of the SfM algorithm is a '3-D point cloud'. A 3-dimensional point cloud (hereafter 'point cloud') is a set of points with 3-dimensinal spatial information (x, y, and z coordinates) that represent a physical surface (Weinmann 2016). SfM is highly dependent on the quality of the images, and the quality of results can vary widely. 3-D models are useful for volume estimates or elevation models, for analysis of canopy structure or in restoration planning (Dandois & Ellis 2013; Lovitt et al. 2018; Zahawi et al. 2015). Elevation models can be converted to raster images to be used for tree crown detection using a watershed analysis (Mongus & Žalik 2015). Dufour et al. (2013) compared 3-D models derived from LiDAR, radar and UAV images for riparian vegetation monitoring in the northwest of France. They found that UAVs allowed for assessments before and after restoration treatments and can deliver 3-D surface models with a very high resolution. UAV imagery was cheaper, faster and easier to process compared to LiDAR and radar, but spatial coverage was limited. UAVs can be used to determine past conditions with methods used in archeology (Çabuk et al. 2007; Lambers et al. 2007; Oczipka et al. 2009; Verhoeven 2009; Chiabrando etal. 2011; Rinaudo et al. 2012). Wallace et al (2016) used a UAV to map canopy structure with SfM in Australia. Their results showed that UAV derived data are comparable with LiDAR 3-D point clouds. These data can be used indirectly for assessments of hydrology, microclimate, and biodiversity. Lovitt et al (2018) found that seismic lines generally show lower elevation and more moisture than the surrounding forest in a study of the effects on seismic lines on boreal peatlands microtopography with UAV derived 3-D terrain models. It is therefore unlikely that the seismic lines will recover without active restoration.

Specific objects like tree crowns or breeding birds can be detected from 2-D images with visual light or multispectral properties. Object based image segmentation algorithms can be used for automatic or semi-automatic object detection (Carle et al. 2014). Michez et al (2016) describe the use of UAS in assessment of invasive plant species abundance using visible and near-infrared orthopohotos and a supervised classification algorithm in two study sites in Belgium. They found that invasive species detection is highly species dependent. Results for *Heracleum mantegazzianum* reached the best accuracies with a 97% detection rate, whereas the other two species (*Fallopia sachalinensis/Fallopia japonica* and *Impatiens glandulifera*) in the study only reached 68% and 72%. The applicability of the method therefore depends on the target species.

3. Reliability and concerns with UAV use

More research is needed comparing field based methods and remote sensing, especially when using hobbyist UAVs. Dufour *et al*. (2013) pointed out that few studies compared field based approaches and remotely sensed data. The authors concluded that remotely sensed data can not completely replace field based assessments, especially for understory assessments in areas with dense canopy cover, tree age, or soil properties.

UAVs can affect the behaviour of target species. Barnas *et al*. (2018) researched the effects of fixed-wing UAV flights on nesting behaviour of lesser snow geese (*Anser caerulescens*) and found that survey flights significantly affected the behaviour of the geese. The birds were more active and spent less time resting compared to a control group. Borelle & Fletcher (2017) found that UAV flights always have an effect on nesting birds after examining eleven studies on shorebirds conducted with UAVs and their recorded effects on behaviour of nesting birds. This will have to be considered when monitoring the effects of restoration on wildlife with UAVs. It is also necessary, as with every sampling method, to be aware of possible effects the sampling has on the subject. On the other hand, UAV surveys can reduce interference and disturbance compared to direct surveys done on the ground (Jones et al. 2006; Sarda-Palomera, Francesc et al. 2012).

3-D point clouds derived from SfM vary in quality and may need to be combined with ground proofing or data fusion with other remote sensing data if high precision is required. Tomastik *et al.* (2017) assessed the accuracy of SfM derived point clouds by comparing coordinates of the derived point cloud and coordinates of ground control point measured in the field. Their models received a sub-decimetre accuracy. Dandois *et al.* (2017) went a step further and assessed the accuracy of individual points of the point cloud. They reported that the feature detection algorithm has a significant effect on the sampling quality and more attention should be paid to the development of these. Mlambo *et al*. (2017) assessed the application of SfM for measuring greenhouse gas emissions in the context or REDD+ forest restoration efforts. The authors assessed the accuracy of tree heights measured from SfM derived point clouds and compared them to LiDAR derived models and ground measured tree heights. The UAV derived models were strongly correlated with LiDAR data in an open canopy forest but performed poorly in closed canopy forests. The authors conclude that SfM point clouds are well suited for the assessment forest with sparse canopies, but are not yet able to perform well in closed canopy forest since the SfM technique is not able to accurately map the ground.

Sensor calibration and data processing are important steps in avoiding error in the results from UAV derived data. Spectral data values differ under different lighting conditions, and it is therefore necessary to either control environmental conditions or correct noise resulting from environmental conditions in the pre-processing phase (Adão et al. 2017). Pre-flight calibration of hardware including satellite navigation system and spectral sensors can increase data quality significantly. Conventional navigation grade GPS is not precise enough for geo-referencing with an error low enough for research applications. To improve the precision of geo-referencing ground control points (GCPs) are necessary. GCPs are highly visible markers that are placed around the edges of the study site and which location is measured on the ground with a high-precision GPS. Those know GPS locations can them help to correctly geo-reference UAV images. Newer, better, direct geo-referencing (Global Navigation Satellite System (GNSS) and Inertial Navigation System (INS)) can make the use of GCPs less important (Adão et al. 2017). Pre-processing after data collection helps improve data quality and corrects for uncalibrated sensors and varying environmental conditions. Spectral calibration (Lucieer et al. 2012) and geometric corrections (Hruska et al. 2012) use targets of know reflectance in the field. As opposed to remotely sensed data from satellite or airplane, there is no need for atmospheric correction (Adão et al. 2017).

4. Future developments

UAVs consisting of several UAVs and a control station will be increasingly used in monitoring, with first applications in near real time forest fire reporting (Merino et al. 2012). This may include 'drone swarms', a group of identical UAVs that can replace each other once the batteries need to be recharged. Therefore at least one UAV can continuously be in flight, delivering a constant monitoring.

Sensors will become increasingly small and light, which will allow for a more common use of LiDAR, thermal, multispectral and hyperspectral sensor on small UAVs. Flight times will increase, safety mechanisms on board will be improved and UAVs will become increasingly dust and weather proof (Adão et al. 2017; Crutsinger et al. 2016). Increasing possibilities for software development could drive the use of "crowd-sourced" UAV imagery for monitoring or sampling of larger areas (Crutsinger et al. 2016).

New classes of UAVs like 'ornithocopters', which mimic the flight mechanics of birds are still experimental but may become useful in monitoring of areas where disturbance through bigger UAVs is unwanted (Anderson & Gaston 2013).

UAV aerial sampling (e.g. Random transects) will get increasingly standardized and to be transferable and comparable between studies. This will require standard methods and sampling protocols as well as standardized sensor calibration. Data quality is a problematic issue with UAV data since many applications in ecology are still in an early or experimental stage (Reif & Theel 2017). Camera calibration and data normalization are important steps to avoid unreliable data.

5. UAVs in Ecological Restoration

Respecting the five directives (1) to follow ecological theory, 2) harness technological advances, 3) reject dogma, 4) encourage self-critique and 5) respect stakeholders' limitations) for successful restoration by Matzek *et al*. (2017), UAVs with their versatile nature, quick and uncomplicated use, but also their limitations, will contribute to successful restoration in several ways.

Adapting UAV applications will harness technological advances. UAVs themselves are a relatively new technology in ecological restoration, and they can provide scientific research with more frequent and finer scale assessments as well as carry sensor that are already available from other remote sensing sources but have been too expensive. UAV retrieved data in combination with new statistical modelling processes and analysis tools can help in the planning of ecological restoration. Freely available open-source software and affordable UAV platforms increase the availability of such data and allow for highly individualized monitoring regimes with relatively little effort. Images derived from a birds-eye perspective have certain limitations as mentioned above, but do allow for a new and unusual perspective on restoration projects. This can help in communicating restoration goals and monitoring results to stakeholders by providing an intuitive way understanding spatial data.

UAVs are able to provide more data and higher spatial and temporal resolution than it was possible with other forms of remotely sensed imagery. This can help in providing scientific evidence about the effectiveness of restoration treatments. While evidence can be helpful in challenging conventional beliefs, it is unlikely that UAVs will be helpful in rejecting dogma, as defined by Matzek *et al.* (2017, 111) as "…restoration principles that are generally regarded as true, but that should not be slavishly obeyed". Research about climate change denial has found that providing more accurate facts does not result in a change of opinion as people selectively search for evidence that supports their own opinions and reject opposing evidence, even if it is more convincing (Van der Linden, 2015).

A better understanding of ecological processes and easier assessment of ecological experiments can inform the move away from long held believes without rejecting scientific evidence. Collecting more data on existing and new restoration projects will help to test beliefs about the best methods and can, when necessary, inform new methods. Matzek et al. (2017) write about the example of including non-native species in restoration treatments to restore ecological function instead of limiting the species selection to native species alone. UAVs could for example be used to closely and regularly monitor the non-native species' spread and therefore draw results about benefits of non-native species. This could help prove or reject the long held believe of seeing non-native species as purely negative.

Frequent and comprehensive monitoring with UAVs will encourage self-critique. Monitoring, which historically has been lacking in ecological restoration (Wortley et al. 2013), is simplified and significantly reduced in cost compared to traditional ground measurements when using UAVs. This monitoring will need to be ground truthed and standard sampling methods will have to be developed and repeatability of assessments will need to be secured to create reliable monitoring results.

Inexpensive, easy and fast UAV assessments respect stakeholder and practitioner limitations by decreasing costs and focusing intensive efforts on areas that are most in need of restoration treatments. Such assessments of current ecological conditions can increase the efficient use of resources and optimize the limited resource, making sure land managers and restoration practitioners get the most value out of their limited budget. Rapid digital mapping in combination with GIS also allows for simple inclusion of the interests of several stakeholders. Monitoring, often lacking in ecological restoration, is simplified and significantly reduced in cost compared to traditional on-the-ground assessments. However, restoration monitoring with traditional ground measurements can be quicker and more efficient than introducing a high-tech solution like UAVs. Most ground measurements have been proven to deliver repeatable results with a good accuracy and are carried out with relatively simple tools. This makes traditional methods more accessible for volunteers without specific training and less prone to technological failure or weather conditions. UAVs are therefore most useful for projects that have a relatively large spatial extent and does not have an established volunteer group. UAV remote sensing can be a very useful tool, but should remain just one of many.

UAVs can make fieldwork safer, especially when used in remote areas and areas that are hard to access. Traditional fieldwork often is in dirty, dull and dangerous conditions or even inaccessible (Watts, Ambrosia, and Hinkley 2012). UAVs are most useful for small to medium sized areas of up to several hectares, areas with high spatial variability, applications that need frequent or fast monitoring and can be used under a cloud cover which is not possible with satellite photography.

If applied well, UAV assessments will help to make restoration projects more effective by increasing the available data for the assessment of restoration outcomes, efficient by saving time and resources and engaging by providing intuitive new perspective on restoration projects and offer more frequent updates of monitoring data.

On the other hand, when relying entirely on remotely sensed data, there is no chance for ground proofing the data. Retrieving field measurements from UAV images removes the hands-on experience of collecting the data and removes the step of critically thinking about data quality. When measuring data in the field by hand, outliers or measuring errors can often be distinguished with common sense. This step can be more challenging when metrics are derived from digital 3-D models that are harder to intuitively understand. Increased use of airspace by micro UAVs has caused conflict with civilian aircrafts. Dystopian visions of total surveillance caused by widespread use of UAVs may be science-fiction, but privacy issues can be problematic when using UAVs. With more monitoring done with remote sensing methods, the risk of accidentally documenting people's activities increases. Normalization of UAVs in public will increase the risk of abusing this technology by hacking the drones of others or using drones as a tool in illegal activities. The recent attack on the Venezuelan president with an amateur drone demonstrates this very serious concern. Maduro was attacked with what appeared to be a makeshift explosive attached to a micro UAV (Herrero & Casey, 2018). As affordable UVAs become increasingly widespread, regulations around their use and data collection become increasingly important.

6. Conclusion

Ecological restoration projects are often unsuccessful in reaching their goals and obtaining the expected results because of unclear or unspecific goals, unrealistic expectations, and no or little monitoring (Keenleyside et al. 2012). One of the biggest challenges for ecological restoration now and in the future, is consistent monitoring after treatments. UAVs can help to establish baseline data before restoration treatments and in combinations with geographic information systems help in the planning process of treatments. After the treatments, UAVs can help in many monitoring applications, and because it can be done regularly and quickly, adaptive management (reacting to changes or unexpected developments) can be improved by managers.

UAVs allow for a plethora of applications in restoration ecology of which some have already been established as common techniques and others have been tried. Fields where UAVs are commonly applied now (e.g. forestry or agriculture) can help contribute to an understanding of ecological processes and improved planning of ecological restoration projects.

With increasing miniaturization and affordability of sensors the use of UAVs in restoration ecology will grow in future years. Due to their limitations mentioned above, it is unlikely that UAVs will replace regular ground measurements completely, but they can make fieldwork easier and faster. UAVs can also allow for restoration planning, execution and monitoring in areas that were previously inaccessible, or where funds (especially for monitoring) are limited.

In rewilding projects where we may want to exclude humans to create wilderness areas, the use of UAVs for monitoring of vegetation recovery and species abundance of animals and plants could be a way of minimizing human impact. Effects of low flying UAVs on animal behaviour will have to be considered.

UAV's, just like any other remote sensing technology can always only be a tool in working towards a restoration goal. Defining clear and measurable goals remains the most important factor in planning and executing a successful restoration goal.

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Chapter 4: Assessing Canopy Structure Using a Hobbyist UAV and 'Structure from Motion' Technology in a Restored Douglas-fir Forest

0. Abstract

We compared forest structural metrics from aerial images derived from a hobbyist unmanned aerial vehicle (UAV) and ground measurements to demonstrate the applicability of UAVs for restoration monitoring. We found a canopy height model (CHM) from UAV images underestimated mean tree heights on average by 10.64 m compared to ground measurements but both data showed a statistically significant correlation. Stem densities for UAV data were underestimated by 375 stems ha^{-1} on average and both data sources showed no correlation. Canopy gaps accounted for 6% of the canopy, with an average gap size of 58 m^{2}. Most gaps were smaller than 20 m^{2}. UAV images and the resulting CHM represent a new visualization of the study site for the communication of restoration outcomes to a wider audience but did not meet requirements for monitoring of results or scientific studies. Changes in the sampling methods such as a better digital elevation model and the use of ground control points would improve the results. However, it is unlikely that hobbyist UAVs are able to produce reliable and reproducible results.

1. Introduction

Regular evaluation of restoration outcomes through monitoring can help improve practices and allow for the wise use of limited resources (Jones et al. 2018). We demonstrated the applicability of a consumer grade unmanned aerial vehicles (UAV) in forest restoration monitoring by testing the accuracy of mean tree height and tree density measures against ground measurement data from permanent plots. The investigation focused on whether a UAV survey is accurate enough to provide useful information for restoration practitioners.

Thinning is a common method in forest restoration to improve ecological diversity and function (Fajardo et al. 2007; Versluijs et al. 2017). The creation of a diverse canopy and gaps plays an important role in recreating old growth structures. Parameters adapted from forest management such as density, canopy height, basal area, canopy closure and biomass are commonly used in monitoring of forest restoration (Ruiz-Jaen & Aide 2005; Zahawi et al. 2015). These parameters are especially useful for planning when forest restoration is incorporated within silvicultural treatments. For example, Getzin et al. (2012) used very high resolution UAV derived ortho-rectified photographs to examine the relationship between floristic biodiversity and canopy gap size in beech dominated mixed forests. They found that fine scale spatial information of gaps was strongly correlated with plant biodiversity. Until recently, such monitoring of canopy structure was time consuming and labour intensive, because it had to rely on transects (Runkle 1992) or on visual assessment of the canopy cover (Seischab et al. 1993). Visual assessments are quick but often subjective and imprecise (Coops et al. 2007). With the increased availability of remote sensing and especially UAV data, gap assessments can be done quick and assisted by algorithms that delineate canopy gaps (Zielewska-Büttner et al. 2016).

The most important advances in monitoring in the last decade are linked to the increasing availability of remotely sensed data. Many satellite-based remote sensing data are now freely available (e.g. Landsat, Sentinel) in resolutions of up to 10m/pixel. This includes visible light, multispectral, hyperspectral, LiDAR and radar data. The increased availability and affordability of UAVs, commonly known as drones, have added very high resolution aerial images to the toolkit of restoration scientists and practitioners. With its easy and relatively inexpensive deployment, UAV-based monitoring is likely to contribute to the implementation of successful adaptive management, a strategy that requires long-term monitoring. Adaptive management has been identified as the best strategy for a successful restoration project; however, it is rarely successfully implemented since requiring considerable resources (Perring et al. 2015).

The low cost of UAVs for monitoring has resulted in various applications for agriculture (e.g. Torres-Sánchez et al., 2015), construction (e.g. Bang et al., 2017), forestry (e.g. Tang and Shao, 2015) and increasingly ecological research (e.g. Dandois and Ellis, 2013). For example, UAVs have been used for the monitoring of riparian vegetation restoration (Dufour et al. 2013), bog restoration (Knoth et al. 2013), invasive species removal (Lishawa et al. 2017), tropical forest recovery (Zahawi et al. 2015) and post-fire forest recovery (Aicardi et al. 2016). UAVs have also been used in the monitoring of small and patchy ecosystems such as oak forests in Germany that are not well suited for traditional remote sensing technologies which require large areas for optimum results (Lehmann et al. 2015). Canopy height models (CHM) derived from airborne stereo photography produce accurate estimates of timber volume and basal area of forest stands (Straub et al. 2013; Wang et al. 2015) and detection of gaps (Betts et al. 2005). CHMs derived from UAV imagery are now being used (Ota et al. 2017). Such methods developed for forest management can be used for the monitoring of forest restoration projects for estimation of canopy structure and biomass.

UAV data can be combined with other remote sensing tools. UAV remotely sensed data are usually limited to relatively small areas. However, high resolution UAV data in combination with low resolution satellite data can work as a promising way of monitoring larger areas of forest (e.g. Puliti et al., 2018).

With 2 billion hectares of forest in need of restoration globally, new ways of thinking about restoration projects are necessary (Stanturf, 2014). More and more projects are planned at a landscape scale, with an increasing focus on social and cultural values of several landowners and stakeholders. UAVs can help by providing appealing data visualization and reducing time and resources needed to monitor remote areas that are difficult to access (e.g. Reif and Theel, 2017).

Keenleyside *et al.* (2012; also McDonald et al. 2016) describe three principles for successful restoration of protected areas. Projects need to be effective, efficient and engaging. Engagement requires collaboration with local communities and communication of restoration treatments and effects to them. Communicating the results of restoration monitoring to stakeholders, local communities, other scientists, practitioners and the general public is an important part of ecological restoration and contributes to the success of a project (McDonald et al., 2016). Communication can happen using image based remote sensing products such as ortho-photographs or canopy height models (CHM), allowing a wide audience to intuitively understand restoration results. The birds-eye view provided from a low flying UAV can spark interest and help stakeholders understand scientific results (David et al. 2016).

We used current unmanned aerial vehicle (UAV) technology to monitor forest structural parameters of a restoration project in the coastal Douglas-fir zone (CDF) in British Columbia. The intent was to assess an off-the-shelf consumer (or “prosumer”) grade micro UAV to demonstrate their application in restoration monitoring by presenting a typical workflow for UAV image processing and comparing results for mean tree heights and density to ground measurements from seventeen plots. Additionally, the UAV images were used to derive canopy gaps as another measure of canopy structure.

2. Materials and Methods

The study area is located on Galiano Island, British Columbia, Canada (48°56'47.4"N, 123°29'36.6"W) along the Salish Sea, a major inlet of the Pacific Ocean between Vancouver and Vancouver Island (figure 4-1). The 61.5 ha site is in the heart of the moist-maritime Coastal Douglas-fir biogeoclimatic zone (CDFmm) (Krakowski et al. 2009). Relatively steep slopes and elevations from sea level up to about 140 m characterize the topography of the area and a small creek runs from south to north across the eastern side of the property. Vegetation, soil and moisture regime differ across the site and ecosystem types were previously delineated with 50 individual polygons (Gaylor et al. 2002) (table 4-1).

Table 4-1: Ecosystem types on the study site

ECOSYSTEM TYPE	Stage	Area (Ha.)	% Total Area
Douglas-fir – Salal	Pole / Sapling	19.1	32.4
Douglas-fir – Salal	Young Forest	1.5	2.5
Douglas-fir, Grand fir – Oregon grape	Shrub / Herb	0.4	0.7
Douglas-fir, Grand fir – Oregon grape	Tall Shrub	0.2	0.3
Douglas-fir, Grand fir – Oregon grape	Pole / Sapling	13.6	23
Douglas-fir, Grand fir – Oregon grape	Young Forest	11.3	19.2
Douglas-fir, Grand fir – Oregon grape	Mature Forest	1.1	1.9
Western Red Cedar, Grand fir – Foamflower	Shrub / Herb	0.6	1
Western Red Cedar, Grand fir – Foamflower	Pole / Sapling	2.4	4
Western Red Cedar, Grand fir – Foamflower	Young Forest	2.8	4.7
Western Red Cedar – Skunk cabbage	Tall Shrub	1.5	2.5
Western Red Cedar – Skunk cabbage	Young Forest	1	1.7
Other		3.6	6.1

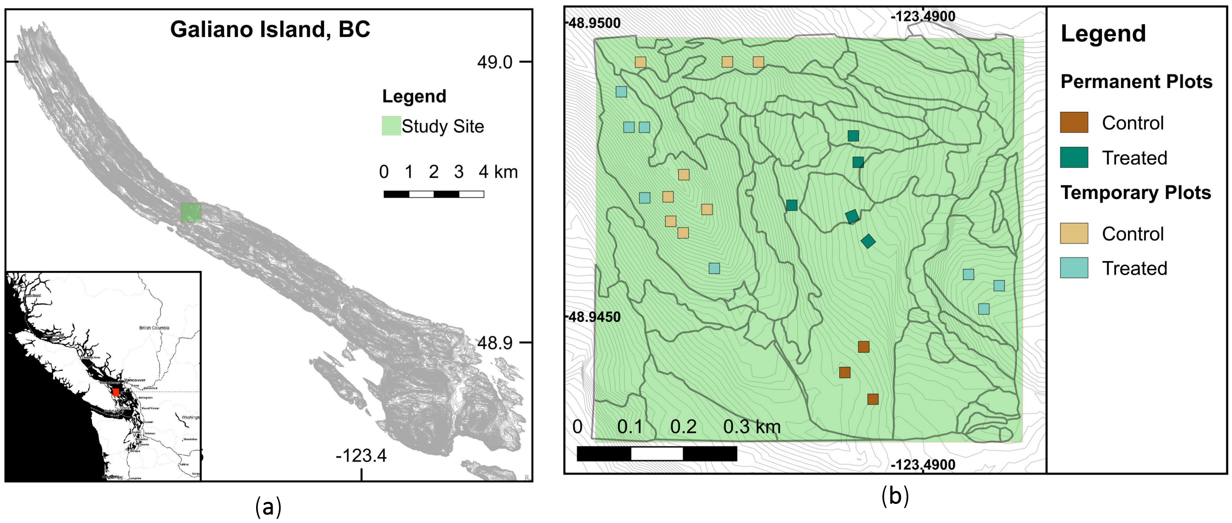


Figure 4-1: Location and contour map of the 61.5 ha study site on Galiano Island, British Columbia.

The local land trust, the Galiano Conservancy Association, conducted restoration thinning on the young, coniferous forest to increase structural diversity and biodiversity starting in 2004. Restoration thinning was deemed necessary after the forest was partially clear-cut logged in 1967 and 1978 with only approximately 4 % of the area left intact in 1978 (Gaylor et al. 2002). Remaining coarse woody debris were bulldozed into piles or windrows and set on fire, but did not combust fully. These windrows were not replanted and are still visible.

In the following season, the open areas were re-planted with *Pseudotsuga menziesii* (Mirb.) Franco (Douglas-fir) seedlings from non-local stock (Gaylor et al. 2002). About half the study site was restored in 2004 and early 2005. In an assessment of the site before the treatments the GCA found several ecosystem types in different stages (Table 1). Restoration consisted of thinning and creation of small gaps where 40-60% of trees were culled by girdling, pulling or topping. The canopy now consists mainly of *P. menziesii,* with minor contributions of *Alnus rubra* Bong. (red alder), *Acer macrophyllum* Pursh (bigleaf maple), *Abies grandis* (Douglas ex D. Don) Lindl. (grand fir), and *Thuja plicata* (Donn ex D.) Don (Western redcedar).

Tree height and density estimates were derived from (1) UAV derived images obtained in late summer 2017, and (2) traditional forestry methods collected with a laser rangefinders and tree counts from the ground in early summer 2017.

Aerial images were taken with a DJI Mavic Pro (https://www.dji.com/mavic; consumer grade UAV with standard camera; table 4-2). The images were originally intended for the creation of a stitched ortho-photo. We used DJI's flight planning software DJI GS Pro (https://www.dji.com/ground-station-pro) to plan the flight. Horizontal overlap was set to 90% and side overlap to 60% at a flight altitude of 85 meters above launch point. The software allows for quick flight planning and flight plans can be changed in the field if necessary. The survey area can be manually selected on an offline map and flight paths are calculated automatically according to the mentioned pre-set parameters (image overlap, flight altitude). The software does not allow for correction of the flight height according to the ground topography. Because of this and because of limited battery time, we flew the property in four separate flights, always starting the highest possible point that was accessible and setting the flight height to 85 m above ground. The actual height above ground varied depending on the topography.

Table 4-2: Characteristics of the DJI Mavic Pro consumer grade Unmanned Aerial Vehicle (https://www.dji.com/mavic/info#specs).

Weight (Battery & Propellers Included)	734 g (exclude gimbal cover)
Max Speed	65 kph in Sport mode without wind
Overall Flight Time	21 minutes (In normal flight, 15% remaining battery level)
Satellite Positioning Systems	GPS / GLONASS
Sensor	1/2.3" (CMOS), Effective pixels:12.35 M (Total pixels:12.71M)
Lens	FOV 78.8° 28 mm (35 mm format equivalent) f/2.2 Distortion < 1.5% Focus from 0.5 m to ∞
ISO Range	photo: 100-1600
Electronic Shutter Speed	8s -1/8000 s
Image Size	4000×3000

Ground measurements were collected by measuring tree heights of randomly selected trees (on average five trees per plot) in seventeen 20 m x 20m plots for a total of 111 trees. Three height measurements per tree with a laser rangefinder were averaged to receive a height value. Andersen et al. (2006) achieved a precision of +-0.27 m with a laser rangefinder by comparing the measurements with height measurements by total stations. Luoma et al. (2017) found a standard deviation of 0.5 m when comparing tree measurements by users with different levels of experience using a clinometer. Sibona et al. (2017) reported similar precision for laser rangefinders in a comparison of LiDAR, rangefinder and direct measurements after felling. We therefore considered values measured on the ground as accurate to at least 0.5m. We counted all trees in those plots and calculated densities by hectare. For 42 trees, we also recorded the exact location by measuring the distance to two plot corners (Roberts-Pichette & Gillespie 1999).

A standard photogrammetric and Structure from Motion (SfM) approach similar to Lisein (2013) was used to create a canopy height model (CHM) from UAV data (figure 4-2). Flight paths produced 1313 RGB images of the study site in Agisoft PhotoScanPro software (www.agisoft.com, Agisoft LLC, St. Petersburg, Russia) to align the images using the following settings: medium accuracy, reference preselection, 40,000 key point limit and 10,000 tie point limit. PhotoScanPro automatically uses GPS image positions to align photos. The internal GPS of the UAV was used for image alignment and ortho-rectification which is commonly referred to as direct georeferencing (Uysal et al. 2015).

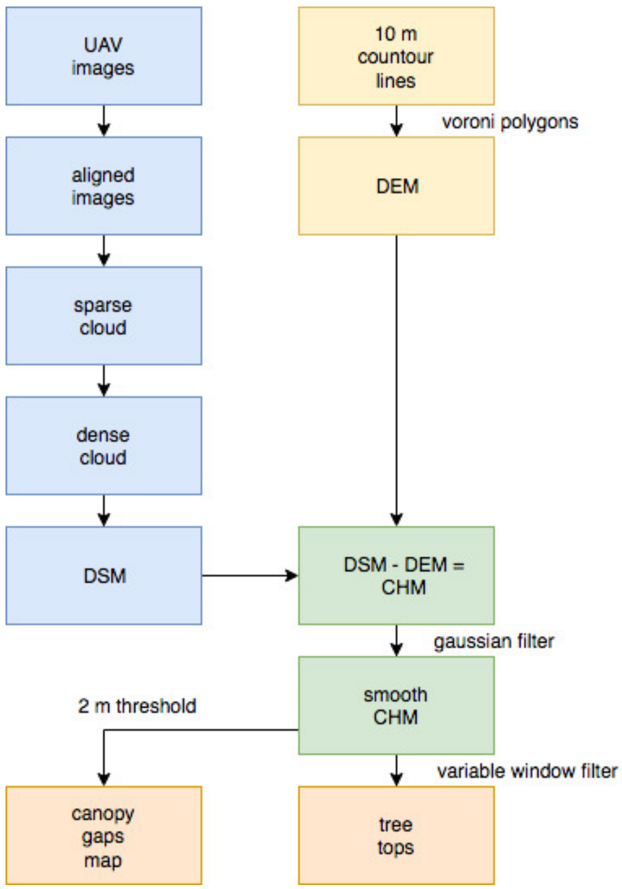


Figure 4-2: Workflow used in tree top and canopy gap detection.

The same software was used to calculate a dense point cloud from overlapping photos using the high quality and medium depth filtering settings, to remove points with extremely different values than their surrounding points. PhotoScanPro uses a SfM approach to create 3-dimensional point clouds from 2-dimensional photos by detecting features across several images and matching them. The software then applies iterative adjustments to estimate the camera orientation and position, and finally the 3-dimensional positions of the features.

Two canopy height models were created after manually deleting artifacts from the point cloud. PhotoScanPro offers a digital elevation model (DEM) function. The first model was created using points classified as ground points to create a DEM with a resolution of 6.02cm/pixel and one using all points classified as high vegetation to calculate a model of the earth's surface including the canopy, commonly known as digital surface model (DSM). Jensen and Mathews (2016) tested the accuracy of DEM from SfM point clouds in open canopy woodland systems. They concluded that SfM products deliver a comparable accuracy to airborne laser scanning with light detection and ranging (LiDAR) products. However, the detection of ground points from standard digital images under closed canopy is challenging (Zahawi et al. 2015). Due to the dense canopy and small gap sizes of our area, the ground model showed large gaps, which necessitated using a DEM derived from 10 m contour lines instead. We then calculated a canopy height model (CHM) by subtracting the DEM from the DSM.

By automatically detecting local maxima in the CHM raster image an algorithm detected tree tops. To avoid errors caused by individual tree branches, the 'CHMsmoothing' function was used in the rLiDAR package (Silva, C.A., Crookston, N.L., Hudak, A.T., and Vierling 2015) with standard settings (Filter = Gaussian, window size = 5 pixel, sigma = 0.67) to smooth the CHM before applying the detection algorithm. Tree tops were detected from the CHM raster file using the 'vwf' function in the ForestTools R-package (Plowright 2018). The 'vwf' function detects tree crowns in the raster data by applying a variable window filter algorithm developed by Popescu and Wynne (2004).

The CHM raster data was used to delineate canopy gaps. All raster cells were considered a gap when the elevation value was lower than 2 m. The threshold of 2 m was used for similar purposes by Brokaw (1982) and Zielewska-Buettner *et al.* (2016) Subsequently, all gaps smaller than 10 m^2 were excluded as demonstrated by Zielewska-Buettner *et al.* (2016). The 10 m^2 threshold was chosen somewhat arbitrarily due to a lack of a generally accepted minimal gap size, but it vaguely represented half the mean tree height.

Similar to Lehmann *et al.* (2017), linear regression models were used to assess the relationship between UAV-derived tree height ("predicted") with field inventory data of tree height ("measured"), and the relationship between predicted and measured stand density.

3. Results

3.1 Tree heights and Density

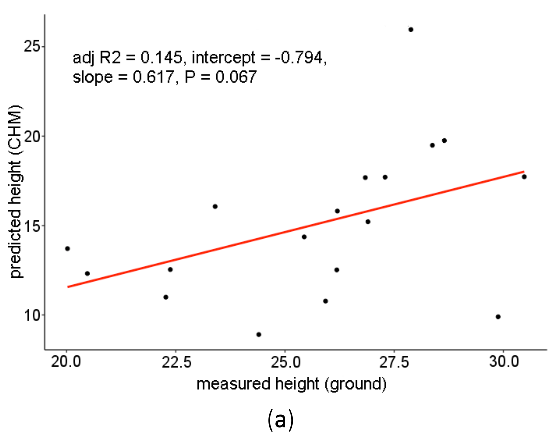
Tree heights derived from the CHM ranged from 7.00 – 46.96 m. with a mean of 16.92 meters (sd = 2.0). Tree density was estimated at 860.25 stems ha^{-1} (sd = 119.9). Mean tree height and density from field measurements were 25.40 m (sd = 3.2) and 904.50 stems ha^{-1} (sd = 269.9) respectively (Table 4-3).

Tree heights measured in the field were on average 10.64 m higher than values derived from UAVs with differences between plot means ranging from 1.93 m to 19.98 m.

Table 4-3: Mean and range of tree height and density from field measurements of 111 trees and predictions from a canopy height model (CHM) using images gathered by an unmanned aerial vehicle.

	Mean Height	*Min Height*	*Max Height*	*SD Height*	*Mean Density*
Model prediction	15.1 (8.9 - 26.0)	10.9 (5.1 - 21.1)	18.6 (11.7 - 28.4)	2.0 (0.6 - 3.3)	508.3
Field measurements	25.7 (20.0 – 30.5)	21.8 (14.4 – 29.7)	29.5 (21.4 – 34.9)	3.2 (1.3 – 7.0)	890.3

There was a significant correlation between tree height measurements and tree height estimations from the CHM (r = 0.67, p = 0.01), but there was no correlation between tree density measurements and tree density estimations by CHM (figure 4-4 (a) and (b)).



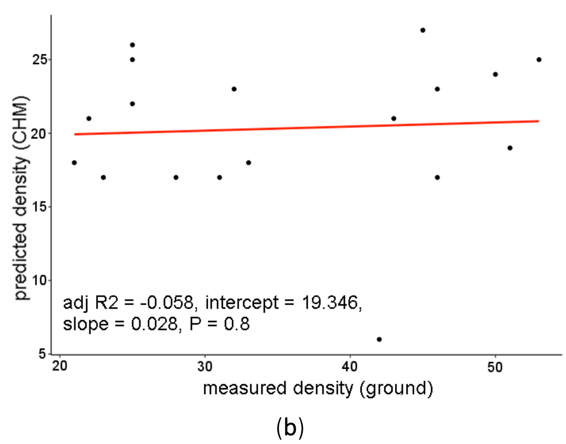


Figure 4-3: (a) Mean plot height measured on the ground vs mean plot height derived from CHM. Each dot represents one 20x20m survey plot; (b) Density measured on the ground vs density derived from CHM.

Values for both tree height and density differed strongly between field measures and SfM derived values (figure 4-4, figure 4-5). The tree density values estimated by the CHM using UAV derived data underestimated tree density in all plots by an average of 15 trees per plot (table 4-3).

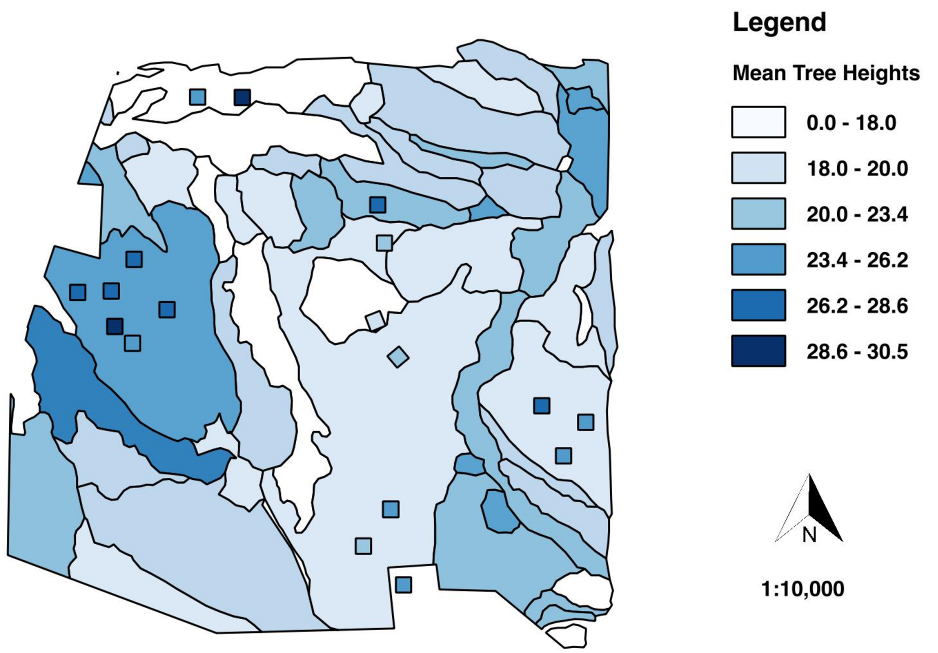


Figure 4-4: Map of tree heights obtained from unmanned aerial vehicle images (polygons) and discrete field measurements of individual trees in 18 square survey plots (squares).

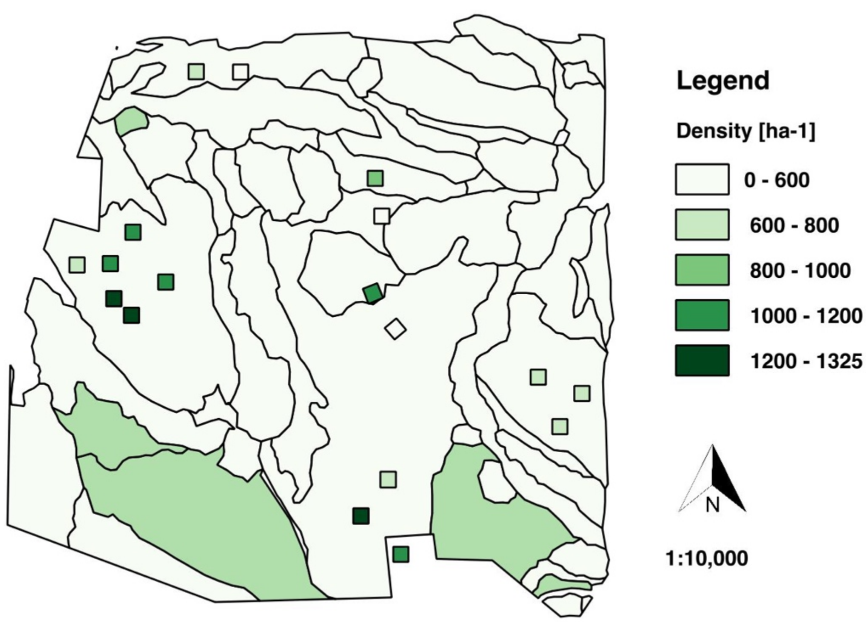


Figure 4-5: Map of tree density obtained from unmanned aerial vehicle images (polygons) and discrete field measurements of individual trees in 18 square survey plots (squares).

3.2. Canopy Gaps

A plot of the CHM raster visualized several important features of the forest structure including areas with low or no tree cover. The windrows, where no trees were re-planted after logging, were visible as long, narrow gaps in the central part or the site (figure 4-7). Old skidder trails were visible as long straight gaps in the canopy, as well as a large landing site in the south west corner of the site. Along the creek on the east side of the property, the canopy was more open, trees were higher and some of the remaining mature trees were clearly visible in figure 4-7. At the far east of the site, the border to the neighboring mature forest was clearly visible with fewer but far taller trees.

The canopy gaps were evenly spread across the study site with most gaps located in the center of the property (figure 4-6). Canopy gaps accounted for 6% of the canopy, with an average gap size of 58 m^2. Most of the gaps were below 20 m^2 with close to 75% of gaps below 50 m^2. There were only three gaps larger than 500 m^2 (Table 4-4).

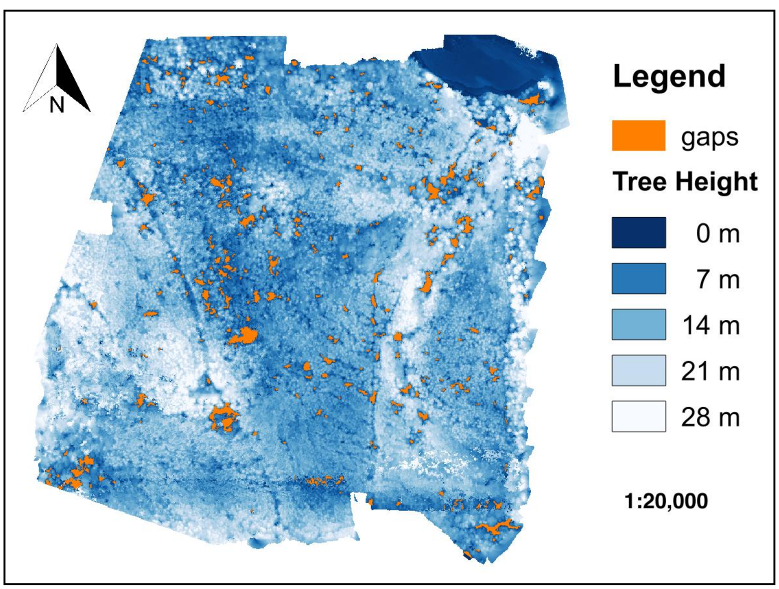


Figure 4-6: Canopy gaps lower than the 2-meter threshold applied to our CHM

Table 4-5: Proportion of canopy gaps of various sizes.

Gap size [m^2]	10-20	20-50	50-100	100-200	200-500	>500
Proportion of total gap size [%]	41.6	30.6	14.1	7.6	5.2	0.9

3.3 Tree Locations

The location of trees subject to field measurements could not be aligned with those represented in the images from the UAV. Figure 4-7 shows predicted and measured tree tops for three plots. We were unable to match up the trees from each dataset. Because of the poor fit, an accuracy assessment was not feasible.

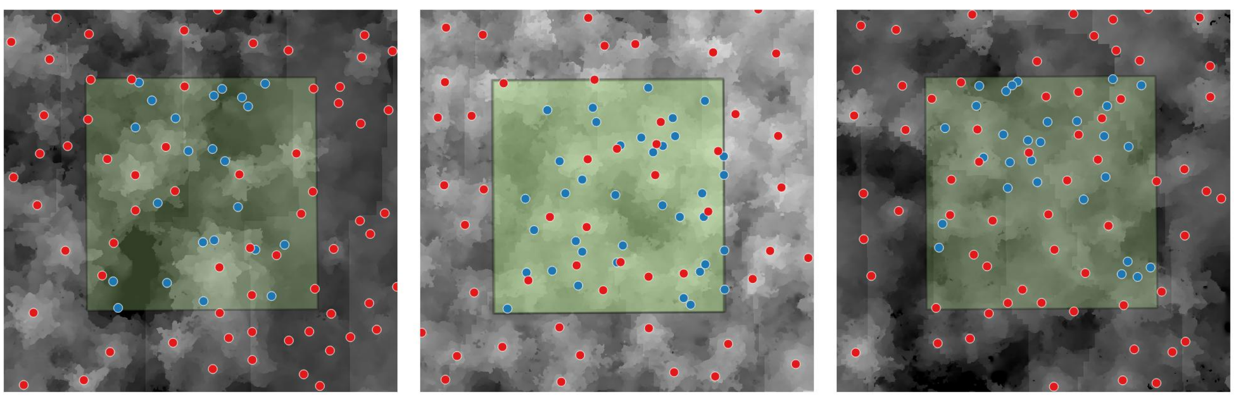


Figure 4-7:Image obtained by an unmanned aerial vehicle showing three plots (green polygon) with tree tops (red dots) and actual location of trees (blue dots). Lighter grey represents higher elevation while dark grey represents low elevation.

4. Discussion

Determining tree heights from UAV images without a DEM that is of similar resolution as the UAV-derived DHM delivered unsatisfying results. The model provided relative height differences between different parts of the study site and therefore an estimate of stand structure. The image can be helpful in detecting areas with better growth and areas with more gaps and therefore be helpful in restoration planning, even if individual tree heights are underestimated. We were lacking a high-quality DEM for the creation of our CHM. DEMs can be created from points classified as ground in the dense point cloud, but in our case, we did not have enough ground points to create a good model, mainly because the canopy was too dense for the UAV to take pictures of the ground. LiDAR data provides better data and is needed for precise surface models, but is expensive. Current miniaturization of LiDAR sensors associated with lower prices could can be carried by UAVs, and will become increasingly affordable.

Density estimates from UAV data were significantly below densities measured on the ground, because the tree top detection algorithm did not detect all trees. Dense canopies make it difficult to detect non-dominant trees as noted by Lisein *et al.* (2013) which coincided with our findings. Densities were underestimated the most in areas with dense, homogenous canopy cover.

Relative heights from our model can be used in detecting areas with better growth and areas with more gaps and therefore be helpful in restoration planning. We could identify many small canopy gaps and very few larger ones. Bradshaw and Spies (1992) used transect sampling for gap detection and found gap distributions similar to our study for mature Douglas-fir forests in Oregon and Washington, with most gaps having smaller sizes. The authors found that old-growth Douglas-fir forests showed generally larger gaps than mature stands in the study. White *et al*. (2018) found that gap detection using point clouds from stereo photography on manned aircrafts images delivered poor results compared to airborne laser scanning. Point clouds derived from UAV SfM deliver better results, but are not as reliable as LiDAR data (Wallace et al. 2016).

The quality of tree detection and height estimates from UAV data highly depends on canopy density. Density of the canopy and the instrumentation both affect estimations by models. Birdal *et al.* (2017) were successful at obtaining good estimations of tree heights using a moving window filter algorithm on a digital elevation model in a young, open coniferous forest in Turkey. The authors achieved a root mean square error of 28 cm for tree heights compared to ground measurements. However, in dense canopy conditions, precise tree height estimates are harder to achieve and may require additional data like multispectral images (Dandois et al. 2015a; Panagiotidis et al. 2017). Meng *et al.* (2017) used object-oriented classification ensemble algorithms to improve quality of DTM under dense vegetation. This method uses an additional step to improve the quality of ground points under vegetation by comparing them to surrounding ground points in the open.

The relatively large area of our study site would be better suited for a UAV with extended battery life or a fixed-wing UAV. These vehicles allow for longer flight times and faster flight speeds, and are better suited to cover our whole site in one flight. There are definite drawbacks in covering the site in several flights. For example, a change in lighting conditions can affect the quality of photogrammetric data (Dandois et al. 2015).

We did not have ground control points (GCP) in our images because the images were not originally intended to be georeferenced. GCPs are usually clearly visible rectangular markers of which coordinates are recorded in the field with a high-quality GPS. Additionally, image overlap varied between images and areas of the study site because of the hilly terrain and the constant flight height imposed by the flight planning software. This caused some warps and fragments in parts of the model.

The time required to collect the data was dramatically longer for ground measurements. The field crew spent several days measuring tree heights and counting stems, whereas acquiring all UAV images took just one day. Processing times for UAV images are higher, depend on available computer hardware, but will need at least a full work day. For small restoration sites, ground measurements may therefore remain the most efficient method to acquire structural forest data.

5. Conclusions

It is possible to obtain georeferenced digital images with sufficient quality to create 3-dimensional models of the canopy, but the resulting data quality is not sufficient for monitoring or scientific use. The UAV did not deliver reasonable estimates for structural canopy metrics that can be used as measures for restoration success. Dense canopy and homogenous cover may require better UAVs, trained pilots and more sophisticated pre- and post-processing.

Even with our low accuracy of relative tree height results, restoration practitioners can use these as an indicator of better tree growth and structural diversity, but a confirmation of the results with ground measurements is necessary. Images taken from UAVs and maps produced from these images allow for a unique perspective on the project and a quick overview. Our results can be a helpful visualization for the communication of restoration monitoring results and allow for an almost instant understanding of general canopy structure.

Additionally, all remotely sensed and particularly UAV derived data is geospatial, which means that "…observed areas and objects are referenced according to their geographic location in a geographic coordinate system." (Khorram et al. 2012, 2). Spatially explicit UAV data allows for a spatial and temporal resolution that is not possible to achieve with any other method. This makes UAV data an important spatial planning tool, and can be used for restoration planning in the office to define areas in need of treatments. Areas of interest can be marked and geographical coordinates used directly to input into a GPS device for fieldwork. UAVs can therefore be used as a supporting tool in restoration planning as well as a monitoring tool. While ecological sampling always only delivers an average per plot/polygon/site, UAV mapping can deliver a full mapping of the study site and therefore deliver a more precise assessment. While this remains true for hobbyist UAVs, the data quality only allows for a first assessment of a site and more precise measurements require better technology or the use of conventional ground measurements. The development of sensor systems, UAV technologies, and software is advancing so rapidly that it is reasonably likely that professional quality features suitable to site-level restoration monitoring will be available within a few years. Thus, UAVs may soon be both a powerful and affordable tool for smaller and not-for-profit organizations that conduct restoration monitoring and scientific research.

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Chapter 5: Conclusion

5.1 Summary of findings

I asked if forest restoration efforts at the Galiano Conservancy Association's District Lot 63 restoration site were successful and how UAVs could improve labour and time intensive ground measurements and contribute to successful ecological restoration. I then applied a UAV image analysis workflow to images of the restoration site to demonstrate a potential application in restoration monitoring.

The main findings of chapter 2 were that areas of restoration treatment showed a higher diversity and cover of understory plants, were more structurally diverse, and had higher volumes of CWD. However, I was not able to connect all of these differences to the treatments themselves. Tree heights in treated areas were lower than expected. The results show some positive effects of the restoration treatments on forest structure and plant diversity, but also highlight the importance of appropriate monitoring strategies and a need for appropriate design of monitoring plots.

The main findings of chapter 3 were that UAVs can help to create better restoration goals, help in the planning of treatments and improve monitoring after the treatments. However, positive effects of UAV use are highly dependent on individual projects and stakeholders involved. Negative effects of UAVs on some wildlife species have already been proven and technical, social and legal restrictions of UAVs limit their use in ecological restoration.

Being a relatively new technological development with appropriate standards still under development, UAVs are increasingly used by ecologists to refine the available data on ecosystem recovery and effects of restoration treatments. This may help validate or reject long held hypotheses and theories. Monitoring can be done more often and restoration practitioners can react to problems faster. Restoration outcomes and monitoring results can be communicated faster and better with the help of UAV derived image products. UAVS can increase safety of fieldwork in remote and hard to access environments. Limitations include legal regulations, weather conditions, limited flight time and the need for trained personnel. UAV sensors are limited to electromagnetic radiation that can be sensed from above. Chemical analysis like soil sampling are at least currently not possible and will need to be done by field crews on the ground. Additionally, data quality is currently not always consistent and standards will need to be established. Even though the cost of UAVs has decreased dramatically in recent years, initial investments are still higher than for traditional equipment like tape measures or compasses. Cost of maintenance of UAVs is high and damage to the UAV during use is common. Additionally, increased use of UAVs could lead to a loss of expertise in proven ground based methods and analysis of UAV derived data requires special software, expertise in the use of this software and can consume significant amounts of time.

The main findings of chapter 4 were that UAV images can help in getting an overview of canopy structure, but surveys need to be carried out with care to receive precise results. This includes image overlap and flight height according to the canopy density, time of day and the correct season. Especially in homogenous forests the use of ground control points may be necessary to achieve good results. A pre-existing DEM is necessary under dense canopy to receive good results for tree heights because in contrast to laser scanners, photogrammetry using visible light is not able to penetrate canopy cover. Canopy height models can however deliver a good estimate of relative canopy height and be a useful tool in quickly visually assessing canopy structural measures like tree density, canopy gaps and mean height, both important measures of structural diversity. Technology is changing rapidly, and it is likely that within a few years the quality of data gathered with relatively inexpensive hobbyist UAVs will be sufficient for monitoring and scientific use.

5.2 Greater Context

Treatments for forest restoration can vary greatly, depending on the previous disturbance, the ecosystem, the involved stakeholders and the available resources. Some treatments like wire fencing to prevent grazing or canopy thinning have been found successful over many ecosystems; others such as applying fertilizers or prescribed fire showed mixed effects. Some proved harmful like thinning (Agra et al., 2018). Forest restoration can be as simple as relying on successional processes for the return of a mature forest. However, rapidly changing climate conditions may require us to actively prepare forests for unprecedented climate conditions with methods including assisted migration and supporting new species assemblages. In temperate climates, creating diversity and thereby “spreading the risk” seems to be the best strategy to prepare forest for the future.

A changing climate makes adaptive management more important than ever before in ecological restoration. The necessary monitoring will continue to rely on traditional forestry methods like diameter tapes and laser rangefinders, but an increased use of remote sensing technologies and especially UAVs is likely. These new technologies will increase the amount of available data but data quality standards will have to get established to make gained knowledge transferable.

5.3 Limitations of this Research

I was not able to fully relate restoration treatments to improved ecological conditions in the study site treatment areas. An increased sampling size may have improved the statistical robustness of the analysis and delivered clearer results. Additionally, using adjusted weights in the analysis of tree data would improve the statistical power of the results and could help detecting effects of the treatments. Unfortunately, past data only existed for the eight permanent plots, which limited the possible comparison of before and after data.

The UAV images used to analyze the canopy structure in chapter 4 were of sufficient quality for a relative comparison of structure across the site, but data quality and comparability could have been significantly improved by a higher image overlap, higher image resolution and the use of ground control points. Especially a higher image overlap could have increased the number of ground points, improved my DEM and therefore the canopy heights. Due to time constraints, I was not able to take more images during the 2017 field season.

5.4 Suggestions for Future Research

The results of chapter 2 were inconclusive, which points to reanalysis of the data to weight more effectively the unbalanced data. It also encourages further investigation of effects of thinning treatments on forest structure in the coming years but also the assessment of other indicators of old-growth structures like biomass accumulation and tree regeneration. New thinning treatments on the study site and subsequent monitoring of the effects could give insight in the effectiveness of repeated thinning treatments. A long-term study on different thinning treatments of young Douglas-fir forests in the American Pacific-Northwest found that homogenous thinning over the whole stand does only insignificantly increase the diameter growth of trees with bigger diameters unless remaining densities were extremely low (Puettmann et al., 2016). This is consistent with my results, and it suggests that future treatments should consist of thinning with varying intensities, including gaps and areas with extremely low remaining densities to increase growth of larger trees. According to to Puettmann et al. (2016) extreme thinning does not affect the carbon sequestration of the remaining stand, but an assessment of carbon sequestration on my study site could give valuable insight in these processes. Gaps will also allow for natural regeneration and further the structural diversity. The success of seedling growth will depend on the exclusion of hyper-abundant herbivorous deer.

The size of my study site required me to fly the site in several separate flights to keep visual contact and to account for the short flight times of the UAV. This complicated the creation of a canopy height model, but could be avoided by using UAV sampling, rather than a full assessment of the whole site. Just like conventional ground measurements, images can be taken along an easily accessible and visible transect line or be limited to sampling plots. This reduces the time required for image acquisition and processing. A sampling workflow represents a more feasible option of supporting the restoration monitoring by a charitable organization like the GCA. Due to the nature of UAV images, they are best suited for assessments of canopy gaps and tree heights. If these data are combined with ground measurements of tree diameters and understory vegetation, a comprehensive assessment of restoration success can be achieved. The relatively labour intensive and time consuming assessment of coarse woody debris could be replaced with an estimate based on trees that have fallen and are no more visible in the UAV images.

The applicability of UAVs to monitor forest restoration in temperate forests needs more research. Comparable standards and standardized methods are needed to be able to compare results between studies. In areas where no high-resolution DEM from LiDAR exists, other methods are necessary. Data fusion, the combination of several types of remote sensing data to generate new data, may be a promising approach of overcoming those limitations of UAV data but needs further investigation.

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Appendix A: Design of Permanent Plots

All permanent plots were laid out using the guidelines described by Roberts-Pichette and Gillespie in *Terrestrial Vegetation Biodiversity Monitoring Protocols* (Roberts-Pichette and Gillespie, 1999). The plots have a size of 20 x 20 m as suggested for young, even-aged stands (Roberts-Pichette and Gillespie, 1999). The plots were laid out square to the general slope, and all corners A-D were marked with metal pins (Figure 0-1). I was not able to find some of these metal pins, however and had to reestablish the missing corners with a compass and measuring tape. Each quadrat bears an individual ID and all four corners are marked with GPS points and are available as a shapefile for GIS use. For plots on a slope, The GCA used slope correction to set up an exact 20 x 20 m square in the plane.

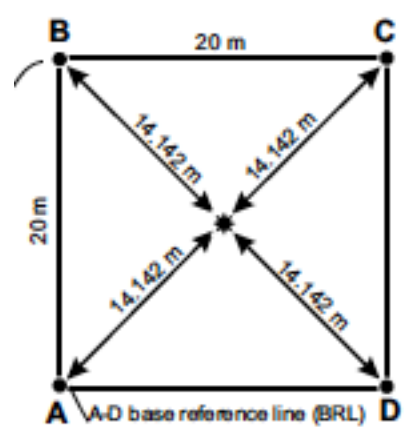


Figure 0-1: Layout of permanent plots (Roberts-Pichette & Gillespie, 1999)

For all plots, the GCA collected the following data Roberts-Pichette & Gillespie (1999):

Essential information

- name of stand and number of plots or stand-alone quadrats
- map of stand showing the plot location(s), their relationship to any prominent feature, and the route to find the plot or plot area
- latitude and longitude of one-hectare plot centre stake
- latitude and longitude and elevation of all corners
- compass bearing of Line A-D - the base reference line (BRL)
- number of each plot or stand-alone quadrat
- plan of hectare plot with all quadrats numbered
- average stand height and canopy depth
- written description of access route to stand and to the plot(s)

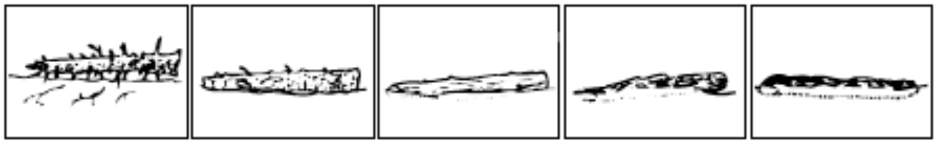
Baseline tree data

- tag number and species of all living and standing trees 10 cm DBH and over
- location of all numbered trees (plotted on a map)
- DBH of all numbered trees
- condition of all numbered trees
- height of about five trees per species and plot
- height to lowest living branch of about five trees per species and plot
- age of stand (determined from off-plot trees)
- photographs from standard positions at standard times and dates
- degree of canopy closure (by quadrat)

Additionally, to the tree mapping the GCA collected data on soil type, vegetation percentage cover by species, slope, and coarse woody debris. The permanent plots are part of the long-term monitoring strategy for the restoration project and allow a detailed description of the change over time.

Coarse woody debris

I measured length and the diameter at the centre of each piece of coarse woody debris (CWD) larger than 7.5 cm in diameter. This differs from the transect sampling suggested by the Ministry of Environment Canada (2010). I recorded the species (if possible), the decay class (figure 0-2), and moss cover per piece of CWD. I then calculated the total volume and average diameter of CWD.



	Class 1	Class 2	Class 3	Class 4	Class 5
Wood Texture	Hard	Sap rot (but still hard, thumbnail penetrates)	Advanced decay (spongy/large peices)	Extensive decay (crumbly-mushy)	Small pieces, soft portions
Portion on Ground	Elevated on support points	Elevated but sagging slightly	Sagging or broken	Fully settled on ground	Partly sunken
Branches	Hard branches with twigs	Soft branches	Branches/stubs absent	Absent	Absent
Bark	Firm	Loose	Trace	Absent	Absent
Wood Appearance	Fresh/recent	Colour fading	Fading colour	Light or brown	Reddish brown
Wood strength	Supports person	May not support person	Breaks easily. Pieces snap	Collapses with weight. Pieces do not snap	Feels firm like ground
Invading Roots	None	None	In sapwood	In heartwood	In heartwood

Figure 0-2: Decay classes as defined by the Ministry of Environment Canada (MOE, 2010)

Vegetation

The sampling followed the guidelines described by the BC Ministry of Forests and Range (2010), except for the tree layer (see below). I assessed species by layer and percent area cover in the plot. I collected any unknown species and verified them with the help of an expert.

A. Tree layer (A1, A2, A3): I repeated the methods used in the baseline assessment, that differ from the standard assessment method for tree mensuration described by the BC Ministry of Forests and Range (2010). I assessed the species and measured the DBH of all trees. I measured snags, but did not include them in the basal area calculations. I re-sampled about five trees per plot for height, crown width and depth, to estimate the live crown percentage, with the exact number depending on the previous assessments. For measurement of the DBH I used a standard circumference tape, for height measurements a laser rangefinder (figure 0-3). In addition, I recorded obvious signs of wildlife use, damage to the trees, and the tree status according to the BC Ministry of Forests and Range (2010) (table 0-1).

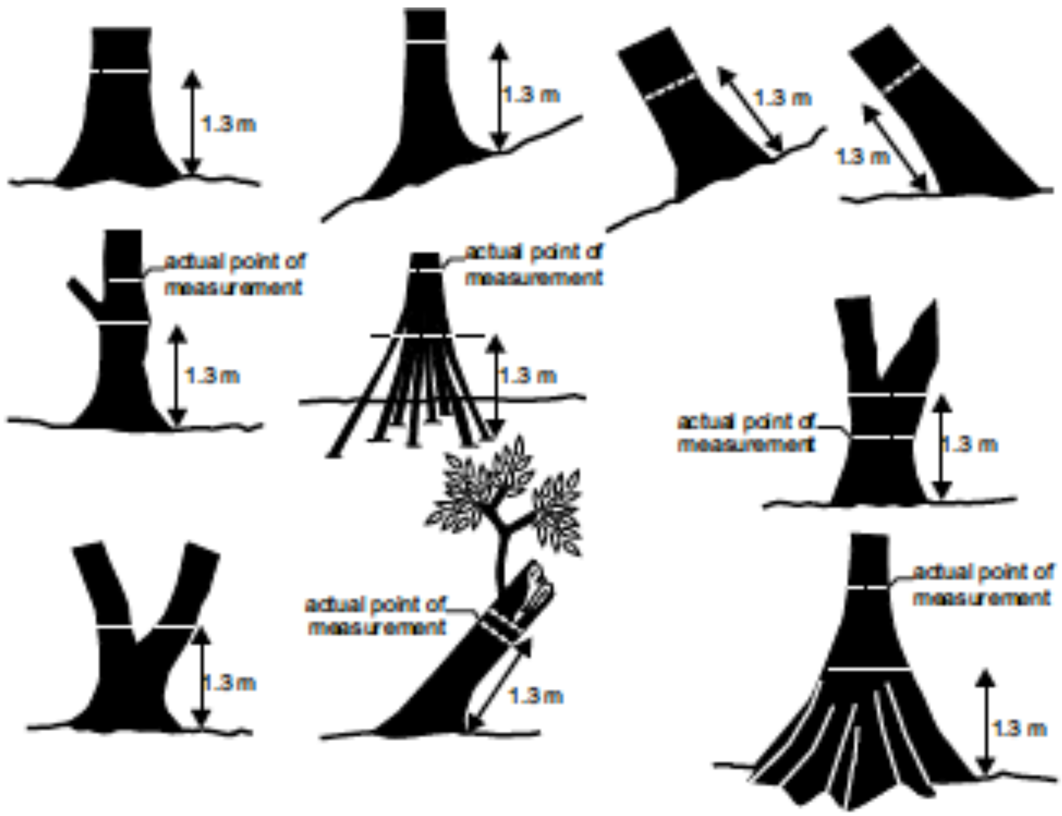


Figure 0-3: How to measure DBH (Roberts-Pichette & Gillespie,1999)

Table 0-1: Tree status (Dallmeier, 1992)

Standing alive	AS
Standing dead	DS
Broken alive	AB
Broken dead	DB
Leaning alive	AL
Leaning dead	DL
Fallen/prone alive	AF
Fallen/prone dead	DF
Standing alive dead top	AD

A. Shrub layer (B1, B2): All tree and shrub species including woody plants between 10 m and 0.15 m are included in this layer. I estimated percentage cover per species.
B. Herbaceous Plants layer (C): All herbaceous species including woody plants less than 15 cm tall are included in this layer. I estimated percentage cover per species.
C. Moss, lichen, liverwort, and seedling layer (D): This layer includes all mosses, terrestrial lichens and liverworts, and tree seedlings (seedlings are trees younger than 2 years, *i.e.* trees that do only show one year of growth). I estimated the total percentage cover of this layer and record all species. Seedlings were of special interest for the assessment of potential for diversification