Using Unmanned Aerial Vehicles and Wildlife Camera Trapping for the Early Assessment of Road Reclamation Effects in Northern Ontario.

By

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ABSTRACT

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The forest industry has been performing forest road reclamation to regenerate forests after harvest, to maintain wildlife habitat and to limit forest access by the public, all to help maintain large, contiguous wild forests. International forest certification bodies like the Forest Stewardship Council (FSC) and Sustainable Forestry Initiative (SFI) use third-party accreditation to ensure that forest management best practices are followed, wood products are harvested sustainably, and sensitive habitat and wildlife species are protected. To access new resources, previously disturbed areas must be returned to similar, natural forest conditions, which includes the reclamation of access roads. Deactivating access roads inherently creates challenges for monitoring site rejuvenation by removing the very pathways needed to monitor those areas. This study attempted to develop a new method for inexpensive, repeatable, large-scale monitoring and measurement of regenerating forest areas considered inaccessible from the ground.

This study was done using off-the-shelf unmanned aerial vehicles, or drones. Recent technological advancements has allowed for the application of random photo sampling methods to orthorectified mosaic tiles derived from drone imagery. The goal was to use automated software, and accurately classify percent vegetation cover (VC) on deactivated logging roads and correlate reclamation ‘success’ to wildlife species presence/absence. Two conventional image classification software packages (ERDAS Imagine and eCognition Developer) were tested to assess VC values on road reclamation treatments with overall orthomosaic classification accuracies reaching 97.86%. The largest influence on road regeneration VC was found to be road ecotype. Lack of significant difference in the photo sampling results suggests the previously applied reclamation efforts in this study area were unsuccessful. The wildlife monitoring efforts found a significant difference in the use of different road treatments and ecosite by species, with active/non-reclaimed roads having the lowest species presence and activity.

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INTRODUCTION

Forest activities in Canada and Ontario are regulated by third-party organizations that assess forestry practices and set rigorous laws and regulations grounded on science-based indicators (NRCAN 2017). Forest Stewardship Council Canada (FSC) and Sustainable Forestry Initiative (SFI) are examples of certification systems in Canada that perform audits of forest operations, require annual reporting, and offer chain-of-custody from the forest through to point-of-sale. Certification processes also help to ensure conservation of biodiversity, protection of sensitive habitat, protection of wildlife and species at risk, protect water quality, respect of Aboriginal and treaty rights, and prevent illegal logging or the importing of illegal wood products (FSC 2004; NRCAN 2017; SFI 2015). Since forest certification was adopted in the 1990s, Canada now has the largest amount of the worlds independently certified forests, 37%, at over 168 million hectares (NRCAN 2017).

FSC and SFI national standards require the maintaining of large, contiguous areas of representative forest habitat, conserving biodiversity and protecting wildlife and species at risk with connectivity between key habitats (FSC 2004; NRCAN 2017; SFI 2015). This is done through standard operating procedures in forest management plans (Lawson 2009a; Lawson 2009b) where wildlife habitat, in addition to forested land, are maintained through silviculture and road removal/reclamation. Road building, maintenance and removals are scheduled by the licensee in coordination with stakeholders such as local citizen committees and the provincial governments to account for as much of the stakeholder road use as possible. Annual reports of these efforts are given to the FSC and SFI certification boards monitoring the efforts of forest companies and the success of forest management strategies. The challenge then, is monitoring the vast boreal forests that cover Ontario and Canada regularly, effectively and in a timely manner.

Landsat imagery and traditional aerial photography techniques have provided some solutions to monitoring (Lambin 1999; Zhang et al. 2016) but lack the fine detail needed to assess changes in vegetation cover under 5 meters down to the individual plant level (Environment Canada 2011). Therefore, improved methods and accurate technological capabilities are required to assess forest disturbances and road reclamation efforts to meet the minimal habitat disturbance goals of third-party certification, government and industry stakeholders. Two points need improvement: developing better monitoring programs and employing site adapted forestry (Jeglum 2003). Both of which are challenges with limited staff and resources. Site adapted forestry practices consider the surrounding ecosystems, locally adjusting generalized practices applied over larger areas to better suit different forest conditions (i.e. *Picea mariana* lowland versus *Pinus banksiana* upland). Before the advent of low-cost unmanned aerial vehicle (UAV) platforms, low-level imagery came from manned aircraft (Anderson & Gaston 2013; Duniway *et al*. 2012), flying close to the tree canopy coming with inherently higher risks.

Unmanned aerial vehicles have been used for many applications in natural resource fields like automated object identification for wildlife monitoring (Christiansen *et al*. 2014; Dulava *et al*. 2015) with various sensors in remote or potentially dangerous areas (van Gemert *et al.* 2014; Watts *et al*. 2010) or mapping illegal forest harvesting and tropical deforestation (Koh and Wich 2012). Random point sampling can be applied effectively to aerial surveys with UAVs by using ground truth data measurements to calibrate the image classifications and perform accuracy assessments. As UAV technology is used more often in scientific research, rigorous methodologies and consistent methodological reporting are required to enable meaningful comparisons of results from different studies.

The objective of this study was to assess the success of the road reclamation efforts on a site in the English River Forest (ERF). There were two main objectives. First, to measure the amount of vegetation growing across treatment types. Second, to measure the reclamation-vegetation effects on wildlife in the area. The factors used in assessing included, levels of vegetation growth, vegetation types, road deterioration, use by people, use by wildlife and which mammal species were present. The study area was designated as an area of interest by the industrial partner, Resolute Forest Products, as it had already undergone reclamation and decommissioning efforts and with the results of this study, aim to improve reclamation best practices in the future. The roads being studied were already decommissioned at different intervals (between 0 to 10 years ago) and road treatment methods varied throughout the study area. Ideally, the roads would have been treated in a manner to allow for stronger statistical testing, but that was not in the means of control of this study. Differences in the timing of treatments across the area were accounted for by adding a time-since-disturbance factor to the statistical model.

There were two hypotheses tested for road vegetation cover. First, that percent vegetation cover would be the lowest on *Active* roads and that *Abandoned* roads should have the highest level of vegetation cover. The assumption was that A*bandoned* roads have had the most time to recover post-disturbance compared to *Site Prepared* road segments. The second hypothesis was that the "road eco-type" will affect the level of cover; road eco-type was the cumulative effects of deactivation status, deactivation treatment, aggregate type, road design, surrounding forest type, topography, hydrology and the underlying structure. With all these features considered, the road network in the ERF study area was divided into four, general "road eco-types"; *Sand*, *Lowland-Decommissioned*, *Upland-Decommissioned* and *Upland-Abandoned*. A third road treatment hypothesis was that road segments with higher intensity reclamation efforts would see an increase in the number of plant species found on site. This hypothesis was derived from the mid-level disturbance theory (Walker 2012), providing a higher potential of species establishment over the active and abandoned road segments.

The objective of the wildlife monitoring program was to understand which animal species were present in the area, how they might be using the road network and potentially establish population estimates from the trail cameras. The industrial partner had past anecdotal evidence from managers, contractors and the public but was lacking hard evidence of animal populations in this area. The effects of road reclamation on wildlife in Ontario is also in its infancy, lacking a baseline of activity. From literature (D'hondt *et al.* 2011; Switalski and Nelson 2011; Tigner *et al.* 2014; Whittington *et al.* 2011), it was expected that the roads would see heavy use by predators and less so by prey species. The hypothesis developed for this portion of the study was that there would be different levels of animal use across the different road treatment types. It was expected that deactivated road segments would be used more often by wildlife and less so on active road segments. The second wildlife objective was to test the line-of-sight visibility across the different treatments. The hypothesis was that abandoned roads would have higher levels of vegetation and the lowest line of sight measurements, followed by treated roads, and active road segments would have the highest values for line-of-sight. Like the vegetation measurement assumptions, the abandoned roads would have had the most opportunity to regenerate as compared to the treated roads and the active roads, leading to lower visibility.

LITERATURE REVIEW

ROADS

Road Density

The total land area of this study was measured using Google Earth and QGIS shapefiles, with linear measurements being 6 km north/south (0/180°) by 14.5 km east/west (90/270°) for a total of 87 km^2 or 8,700 hectares. Decommissioned and active road length within the study area had 385 different segments for a total of 121,160 meters or 121.16 km, which gives an average of 1.39 km of road/ km^2 of area. If the most current roads network shapefiles are used, 210 segments are listed as decommissioned, assuming they were no longer a part of the landscape, the remaining total road length within the study area was 51,611 m, or 51.61 km, putting the road density at 0.59 km of road/ km^2. Natural resource management of English River Forest is directed by the policy and the legislation put in place by the Crown to manage the public interests found therein (Lawson 2009a). Forest management in Ontario is also directed by many other documents and guides provided by the Ministry of Natural Resources and Forestry (MNRF) OMNR (2014).

The only animal species at risk in the English River Forest that is actively managed for is the woodland caribou, *Rangifer tarandus caribou* (Lawson 2009b). Projected woodland caribou habitat (FMP-8) for the year 2009 was 475,295 ha of summer habitat and 112,013 ha winter. The current FMP projects for the year 2029 that caribou habitat will be 537,618 and 94,440 ha for summer and winter respectively. The FMP start levels (Lawson 2009b) indicate that 29% of the caribou zone was designated as suitable for caribou and minimum plan target values are to maintain at least the plan start values to the end of the plan, year 2019, to allow permanent occupancy of the area by woodland caribou (FMP-13). The area of this study area outlined below falls on the southern area of known potential caribou occupancy in the Brightsands/Moberly Lake conservation area shown on the right side of Figure 1. Access was restricted beyond the two deactivation points into the study area to ATVs and foot traffic only, on deactivated treatment areas.

As part of the 2009-2019 English River Forest FMP (Lawson 2009a), one of the management objectives is to reduce the road density in the license area below the 2009 plan start level of 0.856 km of road per km^2 of productive land, and below 0.849 km^2 in the identified caribou zone.

The caribou zone in the FMP ensures any harvested stands in that area are successfully regenerated, hardwood tree species are minimized, conifer species are either maintained or increased through silviculture, and the areas are assessed on years 3, 7 and 10 of the FMP.

ECOLOGICAL EFFECTS.

Wildlife

The English River Forest is on the southern limits of the woodland caribou Ontario range. Caribou, *Rangifer tarandus,* is a large ungulate species found throughout the northern hemisphere on the tundra and into the boreal forests of Canada, Russia, Scandinavia, and Alaska. Currently, the woodland sub-species of caribou, *Rangifer tarandus caribou*, has been listed as vulnerable worldwide (Gunn 2016), threatened under the Species at Risk Act (SARA), and Schedule 1 (COSEWIC 2011; Gunn 2016; NCASI 2011). The government of Canada (GOC 2017) lists 16 separate populations of caribou within Canada, one of which is extinct, and four are of concern. The Committee on the Status of Endangered Wildlife in Canada listed the woodland caribou as threatened in the year 2000 and said that the sub-species would be extinct by 2100 if the rate of habitat loss continued (COSEWIC 2011; ST 2000). Ecosystem degradation of the boreal old growth forests, key woodland caribou habitat, caused by industrial expansion and motorized recreation results in further fragmentation of caribou habitat, and increased predator presence (Environment and Climate Change Canada 2017; MNR 2009; Parks Canada 2017; Vors *et al.* 2007).

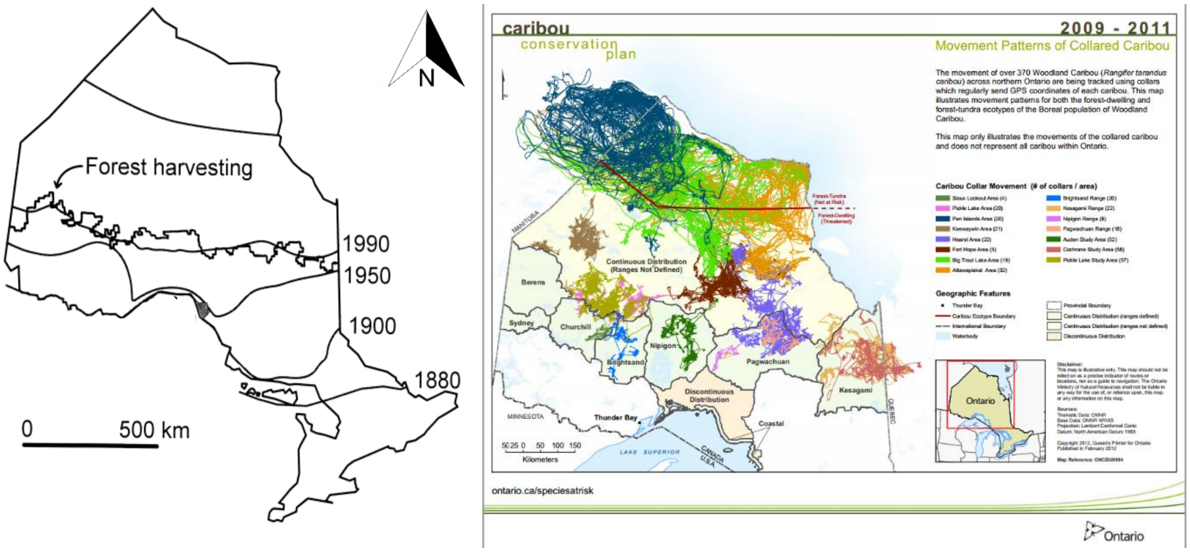

Figure 1. Range recession of woodland caribou in Ontario (Schaefer 2003). Right: recorded GPS tracks of different caribou populations within Ontario between 2009 and 2011.

Caribou diets consist of mostly lichens, although there has been recent evidence that they also feed on fungi and other vegetation during summer months (Thompson *et al.* 2014). Although several previous studies suggest that boreal caribou across Canada require ground lichens (Bloomfield 1980; Cringan 1956; Stardom 1975), they may only prefer them (Ahti and Hepburn 1967; Cumming and Beange 1987). Thompson *et al.* (2014) studied food selection behaviour through collared caribou and recorded what the animals were eating, when and where. In winter, where food choice was limited, lichens were the central portion of the animals' diets. However, in summer months when there are higher quality food choices, caribou were found to be opportunistic with food selection. They were also found to have sub-optimal diets considering the other diet options available to them. While the range of plants consumed did expand in the summer months, lichens remained the main component of the caribou diets. This is important because it can have implications on forest operations in areas that are known as caribou habitat or feeding areas with sensitive lichen beds (Vors *et al.* 2007).

Predator and Prey Relationships

Islands play an essential role for caribou in their attempt to isolate themselves from predators (Carr *et al*. 2012; Cringan 1956; Cumming and Beange 1987). Smaller islands while possibly providing less sustenance for caribou, may be more stable than larger islands (Lomolino *et al.* 2010; Walker 2012) and be somehow less attractive to moose who like to feed on the buds of young deciduous trees (Gagne *et al.* 2016) and to bear and wolves which prey on caribou and moose (Patterson *et al.* 2013). However, this may also be a consequence of islands being isolated from disturbances such as fires or industrial activities such as forestry or mining and the building of infrastructure to support them. If caribou associated disturbance with danger, smaller islands tend to receive less disturbance and may be more attractive to them. Gagne *et al.* (2016) suggested that the percentage of deciduous vegetation can explain the pattern of avoidance in forest stands and cut blocks. Blocks with higher proportions of hardwoods were selected more often by moose and wolves, while caribou selected stands with low basal area, and mature conifer forests with heath and lichen beds. This results in caribou spending less time in fully closed mature conifer forests.

Use of roads

Woodland caribou move away from suitable habitats that have roads or similar linear features nearby (Dussault *et al.* 2012; Latham *et al.* 2011; Pinard *et al.* 2012). This avoidance is an adaptive response to the potential for predator interactions that could prey on calves or weaker individuals within the caribou population. The primary source of calf mortality has been shown to be from black bear, followed by wolves (Dussault *et al.* 2012; Faille *et al.* 2010; Pinard *et al.* 2012). The higher the road density, the higher the risk, leading to increased pressure for dispersion from previously inhabited areas. Research has also shown (Dussault *et al.* 2012; Faille *et al.* 2010) that calving females have high fidelity to calving areas and will return areas where they have had previous success rearing their you. If a new road is put into this habitat, calves are at higher risk of predation because those new roads allow easier access that was not there previously. Potential increase in predation can result in lowered recruitment rates to the local population and a decrease in numbers. This may be because females having had previous success rearing calves in those areas, perceive higher risks in going to a new location with uncertain success than staying with increased predation.

Black bear, *Ursus americanus,* behaviour on or near roads was suggested as possibly being learned (Brody and Pelton 1989), where experience has shown the animals that the cost/benefits were in their favour. The animals learn that people may be a source of food and that there is little risk of harm. Where there is a higher risk of mortality, such as highway collision or risk from hunting, there is still a high presence of bear activity (Brody and Pelton 1989; Clark *et al.* 1993). Therefore, one can assume that on deactivated roads, in remote areas with little human activity, black bear learn that these linear features have a low risk of human/bear interaction and that the probability of mortality is quite low. Brody and Pelton (1989) showed that road density had no effect on road crossing by black bear, but roads with lower amounts of vehicle traffic had higher active crossing by the individuals within the study. Even during bear hunting season where hunters were using those roadways, they saw that there was little to no change in behaviour either in or out of protected areas.

Like people, animals will use the path of least resistance to travel to save on energy expenditure. Wolves will use roads if it suits them to move through and maintain their territory, but also in search of prey. Having greater mobility increases their encounter rate with prey and the probability of success in the capture of prey (Courbin *et al.* 2014; Whittington *et al.* 2011). Wolf presence was found to have a direct, negative influence on woodland caribou presence (Boan *et al.* 2014) at broader landscape scales with greater than a 2000-meter radius. Under 2000 meters, moose presence accounted for half of the variance within the model explaining non-presence of caribou. Johnson *et al.* (2004) showed that with the northward advancement of forest harvesting there was reduced caribou forage and an increase in the abundance of predators, correlating with the recession of caribou within the southern portions of Ontario's boreal forest.

In Jasper National Park (Alberta, Canada), winter tracking of 2 separate wolf packs showed that they only travelled within 25 meters of roads or linear features 21% of the time and travelling through covered areas such as forest, river channels or other landscape features the other 79% of the time (Whittington *et al.* 2005). They were also seen to travel on average up to 5 times farther on low-use roads and had higher affinity to the edges of low use trails compared to high traffic roads in the park. This is interesting given that the study area is mountainous terrain which could make travel more difficult off-trail. Latham *et al.* (2011) showed that out of all possible linear features in northern Alberta landscape, roads were the least favoured by wolves in both snow and snow-free conditions; it was 40 times less likely for a wolf to cross a road than seismic lines regardless of snow conditions. Alternatively, Dickie *et al.* (2017) found that wolves on average would select linear features over forest stands, except selected seismic lines in the summer and trails in the winter, opposite to what Latham *et al.* (2011) had found.

One study in Poland (Jędrzejewski *et al.* 2004) found four main variables affecting wolf sightings; distance from the country's eastern border, amount of forest cover, lengths of roads and forest fragmentation. These four factors explained 60% of the variance within their models. While in Ontario, Newton *et al.* (2017) found that among collared wolves, anthropogenic features were selected more than river or lake shorelines and other natural linear features, especially during summer months. Newton *et al.* (2017) also showed no functional difference in selection between old cut areas, deciduous or mixed forests, sparse or barren land, and conifer, with some avoidance of lowland and recently cut areas. Moreover, as road density increases, so does use by wolves (Newton *et al.* 2017) because they do not have to deviate as far from their main direction of travel to use them. This enables the predators to travel farther, faster and expend less energy to do so.

Previous studies have shown that moose tend to select habitat that provides higher quality forage and presence increasing with distance from roadways (Eldegard *et al.* 2012; Laurian *et al.* 2008). Moose were also more likely to use or be found near, smaller or less-used roads than major roads and more so at night compared to daytime hours (Eldegard *et al.* 2012). Eldegard *et al.* (2012) also showed that males were found more often closer to roads than females. The behavioural differences between males and females was suggested as a predator-avoidance perceived strategy by the females, regardless of whether there were predators present or not. On roads less used by people, such as operational roads like those in this study, the perceived risk of predation is lower and may result in more activity on or near them. Also, Laurian *et al.* (2008) found that home ranges with higher proportions of road segments were 11% larger than moose home ranges that had little to none, with forest roads and highways crossed at similar rates.

An exception to this was seen particularly along major traffic routes, where roadside vegetation sodium concentrations were higher in the vegetation as a result of roadway salting (Laurian *et al.* 2008). Areas where salt run-off pools along the sides of roads were found to be selected by moose just as often as the adjacent forests further away from the road segments. This would not necessarily be a behaviour found in remote or active logging areas where there would be no road salt application, but minerals that are associated with aggregates and vehicle dust may still act as attractants.

Management challenges

The report on caribou research programs in Canada (NCASI 2011) states that Ontario contains what is thought to be 15% of Canada's woodland caribou, yet there are no exact population estimates. There is some understanding of the ecologic influences affecting the species such as predators, habitat, or food availability, but we lack the physical count of individuals (past and present) living in Ontario's boreal forests, making it more difficult to manage for this species effectively.

A growing concern with black bear is that their steadily increasing population in the boreal forest may be putting increasing pressure on other species, such as woodland caribou (Pinard *et al.* 2012). Caribou will adapt their behaviours to avoid interactions with the increase in wolf populations that follow the northward expansion of moose into the known caribou range. However, avoidance of black bear may not yet be adopted by caribou with the rotation of forest harvest planning putting new harvest areas in caribou habitat and changing the habitat suitability. Younger, early successional stands tend to favour deer, moose, and black bear, while at the same time are unfavourable for caribou (Dussault *et al.* 2012), which have cumulative adverse effects on caribou numbers with loss of preferred habitat and increased predators. Pinard *et al.* (2012) showed that 93% of predation events on caribou calves were done by black bear, whereas wolves were responsible for just over 6% of calf fatalities. Dussault *et al.* (2012) recorded that 95% of caribou mortalities were black bear caused. For that reason, black bear likely have the most significant potential impact on the boreal woodland caribou recruitment in the English River Forest study area.

The primary concern for moose management in the English River Forest is that the forest license falls under the caribou forest management area, and as a result, operation and long-term planning are adapted towards managing for caribou habitat. Moose prefer much different forest conditions than caribou. Moose tend to prefer young forest stands and lowland or wetlands which provide aquatic feeding areas (Bjørneraas *et al.* 2011; Lawson 2009a; Lawson 2009b). Cederlund and Okarma (1988) found that females on average preferred young to medium-aged forest stands and clear cuts as foraging areas and avoided mature stands and bogs.

Eldegard *et al.* (2012) also speculate that with road avoidance behaviours, forage availability is lowered, especially during winter months, possibly acting as a limiting factor to population sizes. Neumann *et al.* (2009) concluded from their efforts that in the heavily managed forests of Sweden that human activities did not necessarily have an impact on movement rates/behaviours of monitored moose. Males were found to be almost indifferent to anthropogenic disturbance during the fall as they were engaged in rutting activities (Neumann *et al.* 2009).

It was suggested that a higher than normal wolf to moose ratio was one of the main factors in limiting population growth of moose in Pukaskwa National Park with human hunting adding more pressure to the species (Bergerud *et al.* 2007; Bergerud *et al.* 1983). This could be the case for other parts of the boreal forests of Ontario, including the English River Forest. The challenge then is which species takes priority. Generally, modern forestry practices, using natural disturbance pattern emulation strategies, compliments the foraging and movement behaviours of moose (Bjørneraas *et al.* 2011).

Linear Feature Avoidance

It is well documented that caribou are sensitive to and may avoid active roads (Cumming and Beange 1987; Dyer *et al.* 2001; Gunn 2016; Macdonald *et al.* 2012) as they are exposed to higher risks of predation and mortality in these areas (Whittington *et al.* 2011). Dyer *et al.* (2001) showed that in northern Alberta, caribou were six times less likely to cross a road than any other randomly generated line within their home range, yet seismic lines seemed to have little to no impact on caribou movement. This is an interesting and important point to understand; it is not necessarily the linear feature through a forest that affects animal behaviour, but the function of a road and its continued use and persistence through time by people that affects these avoidance behaviours.

EDGE EFFECT

Through use and inherent design of roads, vegetation can struggle to colonize the surface of roads. Higher temperatures and arid conditions near the center of the road can create desert-like growing conditions making it difficult for boreal species to establish, while the lower ditch portions of the road structure can be subjected to heavy water runoff and flood-like conditions (Klimeš 1987; Özkan and Gökbulak 2017). Nevertheless, road edges can also provide preferable habitat to some wildlife species while providing forage and shelter for animals travelling along roadways (Forman and Alexander 1998; Seiler 2001). Deer and moose, for example, can often be seen foraging on the grass and shrubs commonly found along the rights-of-way and ditches, providing food and easy routes. Trees growing on forest edges were also found to have smaller basal areas compared to their contiguous counterparts. Edges also reduced canopy cover, bryophyte cover, increased Coarse Woody Debris (CWD) (Kumar *et al.* 2017), and increased broadleaf regeneration (Harper *et al.* 2015).

COMPACTION

Due to the nature of their construction and use, road soils are densely compacted and low in organic matter. To increase the success in restoring vegetation, Lloyd *et al.* (2013) found abandoned road succession to shrubs and trees was much slower than if the roads were treated. Their study showed that recontouring the roadbed broke up the hardpan surface and lower soil horizons. This, in turn, was shown to permit water to percolate down instead of running over the surface, bringing organic content and nutrients with it. The broken road surface also provided purchase for wind-blown seeds to establish, their root systems penetrating the lower horizons, accelerating the stabilization or recovery of the roadbed to a more natural, pre-disturbed condition. Froehlich (1979) measured the effects on growth of young pine trees growing on trails previously compacted by logging equipment. After 16 years, measured trees had up to a 12% reduced growth rate on compacted trails.

Comparison of abandoned roads and roads that have been deactivated or decommissioned (Lloyd *et al.* 2013) suggests that abandoned roads may not return to their original state but turn into a different ecological, stable state (Walker 2012) with long-term ecological and economic consequences (Brecka *et al.* 2018). Increased aggregate bulk density from compaction by vehicles can significantly reduce root length and aboveground biomass (Luce 1997; Nadian *et al.* 1997) reducing the plant's ability to seek nutrients from the soil, limiting growth and carbon uptake. That could mean that the road may stay permanently non-treed. They remain vegetated only by low standing woody shrubs and herbaceous plants, leaving it as a distinct linear feature on the landscape and possibly failing the FMP requirements of the silvicultural regeneration. The road could also act as a catalyst to begin altering the surrounding site into a different forest stand type. For example, in a low-lying area where the surrounding forest was lowland black spruce (*Picea mariana)* and larch (*Larix laricina)*, the higher, more dry soils could allow for a species like jack pine to grow, possibly altering the long-term forest composition. It is not necessarily a negative effect having different tree species growing on a roadbed where none were expected to grow, but it may have long-term, cumulative effects on the area that managers should consider when planning into the future. The altered ecological state of the site may change its functional potential and have consequences for system resilience and recovery from future disturbances (Lloyd *et al*. 2013).

Another study (Ozkan and Gokbulak 2017), looked at what happens to soil conditions such as moisture and soil temperature when the vegetation cover type changes from a forested condition to strictly herbaceous vegetation found on roads and ditches. Removal of woody vegetation significantly increased the soil temperature and soil moisture. Luce (1997) showed an increase in soil moisture, hydraulic conductivity and establishment on sites that were mechanically treated by ripping versus non-treated roads. Lloyd *et al.* (2013) also looked at the effects of carbon and nitrogen cycling and storage potential. After treatment of the road, total soil carbon was estimated to be six times the amount of the control to a depth of 25 centimetres. They found that on a decommissioned road, nitrogen content within the soils increased by one order of magnitude. This is one of the stronger arguments for intensive active restoration of forest roads. The increase in organic matter and nitrogen in treated roads would provide more suitable conditions for plant/tree growth, acting as a potential carbon sink and restoring ecological processes to those areas.

EFFECTS OF LINEAR FEATURES

Deyr *et al.* (2001; 2002) found that road avoidance by caribou increased proportionally with the increase in the level of traffic on the road segment. Laurian *et al.* (2008) also found similar results with moose in Quebec, where the animals were more likely to avoid roads with higher levels of traffic than those with less, even with favourable resources considered attractants for moose such as vegetation with high salt content and refuge from biting insects (Thompson and Stewart 1998). On heavy trafficked roads, noise (Forman and Alexander 1998) and mortality become a serious factor to animal life crossing the road (Seiler 2001). The animal often misjudges the speed of vehicles or the sudden appearance of the vehicles spooks them and attempt to flee these would-be-predators, which often happens with birds or small mammals. Many animals have now learned to associate roadways as a danger and avoid them. This restricts their movement abilities, cutting off potential habitat, food resources and reducing the probabilities of finding mates — all of which acts as compounding, indirect factors in a sensitive species' like caribou, survival. The larger the road, the more significant the potential cumulative impacts.

Seed dispersals of key hardwood species were studied in northern USA (Hughes and Fahey 1988), showing that maximum dispersal distances were 15 meters (*Acer saccharum*) up to 25 meters (*Betula alleghaniensis*) in 'good' seed years. Greene *et al.* (1999) calculated that aeolian seeds such as *Populus spp.* with their light, fluffy seeds could reach 250 meters from source trees if there were no obstructions. Heavier seeds from species like *P. glauca* were mostly limited to 75 meters from sources (Greene *et al.* 1999). Greene and Johnson (1995) listed the maximum recorded dispersal distance for *B. papyrifera, A. rubrum, P. resinosa, P. strobus, and P. glauca* as 475 meters. Seedling recruitment was limited to these distances in disturbed areas, limiting the speed of vegetation recovery for that site. The dispersal and establishment were found to decline exponentially from the forest edge, meaning that without a seed source of sexually mature trees nearby there would be little to no aeolian-based seedling growth. It would be difficult to have natural seeding onto the roads due to harvesting layout designs having the roads at the center of blocks, and most often the farthest away from seed source trees.

HABITAT FRAGMENTATION

Several studies (Forman and Alexander 1998; Holbrook and Vaughan 1985; Mech *et al.* 1988; Mladenoff *et al.* 1999) have shown that as road densities increase, critical thresholds are crossed and have negative impacts on viable habitats and populations (Seiler 2001). Non-paved, dirt roads in the United States that cross and divide wildlife habitat total more than 630,000 km in length (Forman and Alexander 1998) and more than 20% (1,600,000 km^2) of the landscape is ecologically affected by roads.

There are five strategies outlined in the ERF FMP (Lawson 2009b) 1) prioritizing zones for road regeneration based on providing caribou habitat, 2) increasing public awareness and signage of the intent to restore the roads to natural conditions, 3) using appropriate silviculture methods to restore the roadbeds to a forested area that matches the surrounding forests, 4) applying treatments as soon as possible and monitoring of site regeneration, and 5) restricting vehicle access to protect and promote forest regeneration on the road surface. It is noted that this is to block road vehicles only and not ATVs as experience has proven that completely blocking access is not possible and people will do almost anything to get around or through obstructions put in place. Any operational road with no access provisions or restrictions is still required to have a prior discussion with the local trappers before roads in an area can be progressively decommissioned.

Monitoring roughly 50 million hectares of forest in Ontario on a regular and consistent basis is a monumental challenge. Two key points highlighted in Best Forestry Practices (Jeglum 2003) that need development were: developing better monitoring programs and employing site adapted forestry. Both present challenges with limited staff and resources. Remote sensing has been used previously to assess habitat fragmentation by roads using Landsat image data (Meddens *et al.* 2008). By classifying landscapes with remote sensing techniques, it is possible to measure the amount of land cover remaining as intact forest or the expansion/contraction of access road networks.

Figure 4 is an example of the level of detail that is now available from satellite data. Data sources like *Planet* (Figure 2) and other high definition satellites, provide highly detailed 4-band image data for processing and classification. On a landscape level, large disturbances are visible and measurable, but small details are missed. Looking closer at a site-sized scale detail is gained, but the regional context and the sense of impact on the landscape can be lost. This is an example of why it is important to define 'habitat fragmentation' along with the area and temporal context that it applies to. The boreal forest is a complex system that is regenerated through short-term disturbances. The problem of fragmentation logically arises from semi-permanent or permanent disturbances, such as roads, that either alter that land base or prevent it from returning to a forested state.

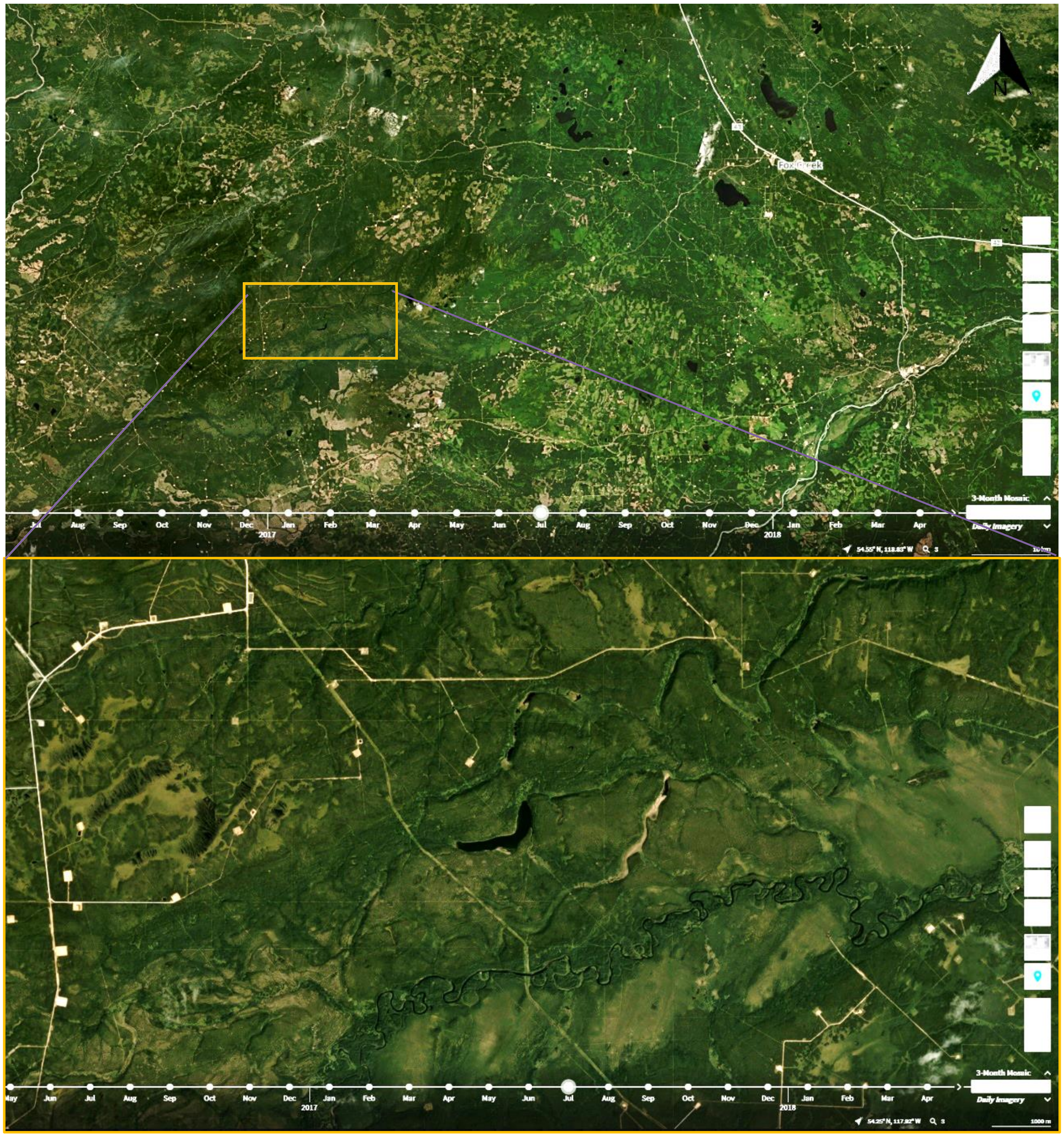

Figure 2. Screenshots from Planet.com/explorer web browser showing the level of habitat fragmentation in western Alberta. The second image is a magnified section of the top image. The white squares in the images are oil and gas rig and pump stations. The lighter green patches in the upper image are mostly forest cut blocks.

MANAGEMENT CHALLENGES

Roads are one of the costliest factors of any resource-based operation. As outlined in *8.5.2 Primary Roads* (Lawson 2009b), the estimated costs for each kilometer were $40,000 for construction of the road, $10,000 for maintenance and $8,000 for each water crossing. Roads are expensive to build, expensive to maintain, and expensive to remove. However, they are required to gain access and move sought-after resources such as timber. Once a road is established there are strong pressures from the public to keep those roads in place (Hunt and Hupf 2014; Mihell and Hunt 2011). If the forest license holder wishes to deactivate a road, they must confer beforehand with the province and the public before commencing, and deconstructive work with all three parties sometimes has conflicting interests (Bliss 2000; Hunt and Davis 2017; Lawson 2009a; Tindall 2003).

Another growing concern of forestry activity in Canada is a loss of biodiversity. Euskirchen *et al.* (2000) found that there was no difference in species richness, Simpson's dominance or evenness between boreal forest systems that were clear-cut, forest edges or the forest interiors. This finding goes against how the public see typical boreal forest harvest practices (Bliss 2000). While forests of Ontario are managed with public interests in mind (ex. forest certification), the 5 points outlined in the MNRF crown land guide in of themselves, have conflicting interests (OMNR 1993). Crown land is the term used to describe land that is owned and controlled by the federal or provincial governments (OMNR 1993; Neimanis 2011). People understand that biodiversity is important and that roads affect habitat, migration and several other biodiversity-related factors, yet public surveys in northern Ontario showed that most people were against the closure of roads or restrictions to their use in any form (Hunt and Davis 2017; Hunt and Hupf 2014; Mihell and Hunt 2011). This creates a dichotomy in decision making for crown lands.

WILDLIFE MONITORING AND TRAIL CAMERAS

Radio collars and telemetry have been used heavily to track and monitor animal movements in the wild with a range of results in many types of studies (Amstrup and Beecham 1976; Brody and Pelton 1989; Clark *et al.* 1993; Courbin *et al.* 2014). One challenge in using this technique is that it is labor-intensive and has the potential to put increased stress on the animal subjects. Researchers must actively locate and capture the animals to attach the GPS collars.

Collaring and telemetry fall under the Capture and Recapture (CR) methodology for population studies where there is a clear identification of individuals (Foster and Harmsen 2012; Rowcliffe *et al.* 2008). This allows researchers to know where the individual is, its movements and how often that individual returns to a location. CR methods also assume that there are high populations in relatively small areas so that the recapture rates are significant. This inherently makes population studies difficult for more rare species or species that do not have unique identifiers like stripe or spot patterns, or other colour variations that can be seen using camera traps. To get around this, Noss *et al.* (2003) assumed that when unable to make clear distinctions between individuals in different photographs from the same site, it might be assumed that it is the same individual. This likelihood is increased if it is known that the species maintains distinct territories and it is unlikely that other members of the species will transit through. However, this model requires a high level of understanding of the individuals, population, or species being studied to make those assumptions.

Another method employed by Rowcliffe *et al.* (2008), was to use an adapted chemistry formula for measuring movement speed and the detection areas of the cameras to predict how often a sighting would occur. Researchers were then able to estimate population densities without having to mark and recapture individuals or the need for unique, identifying marks. The second advantage to the Rowcliffe *et al.* (2008) method is that it is not dependant on the spacing of camera traps to the sizes of home ranges of the animals being studied. Rovero and Marshall (2009) report that they set a minimum 1-hour interval between trigger events if what looked like the same animal had remained in the area. Therefore, for events where an animal may be browsing in the target area and triggers multiple recordings, they would all be counted as one event up until that one-hour mark. However, if there were a distinctly different individual or another species that triggered an event within the 1-hour mark, then that 1-hour time would be reset to zero and a new trigger 'event' is reported.

Camera traps (trail cameras, motion cameras, remote cameras) are being used more frequently in wildlife monitoring (Burton *et al.* 2015; Cusack *et al.* 2015; Kays *et al.* 2010; Meek *et al.* 2014; Kolowski and Forrester 2017: Rovero and Marshall 2009). Between the years 2008 and 2013, Burton *et al.* (2015) found 266 camera trap studies published with mammalian carnivores being the focus (64.7%). This study also found that 40% of those relied upon "opportunistic or targeted" camera locations. Benefits to using camera traps were that they are non-invasive, low labour and maintenance, produce quality, time and date stamped data points tied to specific geographic locations and can provide extra insight by recording animal behaviors (Kays *et al.* 2010). Limitations of using camera traps are that they can lack a robust, well-defined methodology in their deployment due to the challenges in accessing remote areas and the lack of standardized reporting in publications (Burton *et al.* 2015). Another limitation to camera trapping is the costs of purchasing the camera equipment, with budget restricting the number of cameras, possibly restricting the sample sizes and area, as was the case for this study, and other camera trap studies (Burton *et al.* 2015; Cusack *et al.* 2015; Meek *et al.* 2014).

Kolowski and Forrester (2017) studied the feature-based placement of camera traps and had capture rates up to 9.7 times higher than that of nearby cameras that were randomly placed through the study area acting as the control. The features used were trails and logs or snag-like features on the landscape. Kolowski and Forrester (2017) showed in Figure 3, that after an accumulation of 385 \`camera nights\` (nights x camera sets) the number of species captured on trails versus randomly sampling became similar, and by 1000 \`camera nights\` confidence intervals almost completely overlap. This resulted in no significant difference between trail-feature placements or randomly placed cameras. This can be used along with Cusac *et al.* (2015) results to argue that given enough time, having biased camera trap placement will even out compared to randomly placed cameras within the study area.

The challenge in using camera traps is making inferences from the data collected. Because camera deployment changes from study to study, with few studies accounting for placement bias, they lack robust population models. Burton *et al.* (2015) found that out of over 260 articles, only 3 produced estimates that addressed imperfect detection and model assumptions using the random encounter models. Where camera traps were used for relative abundance studies, the model assumes that there must be a constant level of detectability across all sites, which is rarely addressed (Burton *et al.* 2015). It is also challenging to study population metrics using unmarked individuals. Accurate measurement of animal movements through their environment is difficult and a complex problem when considering all factors involved such as terrain, shelter, predator-prey avoidance, vegetation density, and vegetation types.

Studies have shown that predators use these road networks to move through their environment (D’hondt *et al.* 2011; Switalski and Nelson 2011; Tigner *et al.* 2014; Whittington *et al.* 2011) while others have said that prey species, specifically ungulates, generally circumvent roads to avoid predators (Cumming and Beange 1987; Dyer *et al.* 2001; Dyer *et al.* 2002; Forman and Alexander 1998; Laurian *et al.* 2008; Whittington *et al.* 2011). By using camera trapping techniques, the human influence on animal behaviours was removed from the site to elicit a more natural behaviour which was captured by the cameras. Since the goal was to see which animals were using the different road types, the camera locations were therefore biased to only measure activity and presence on the road network and not in the surrounding forested land area.

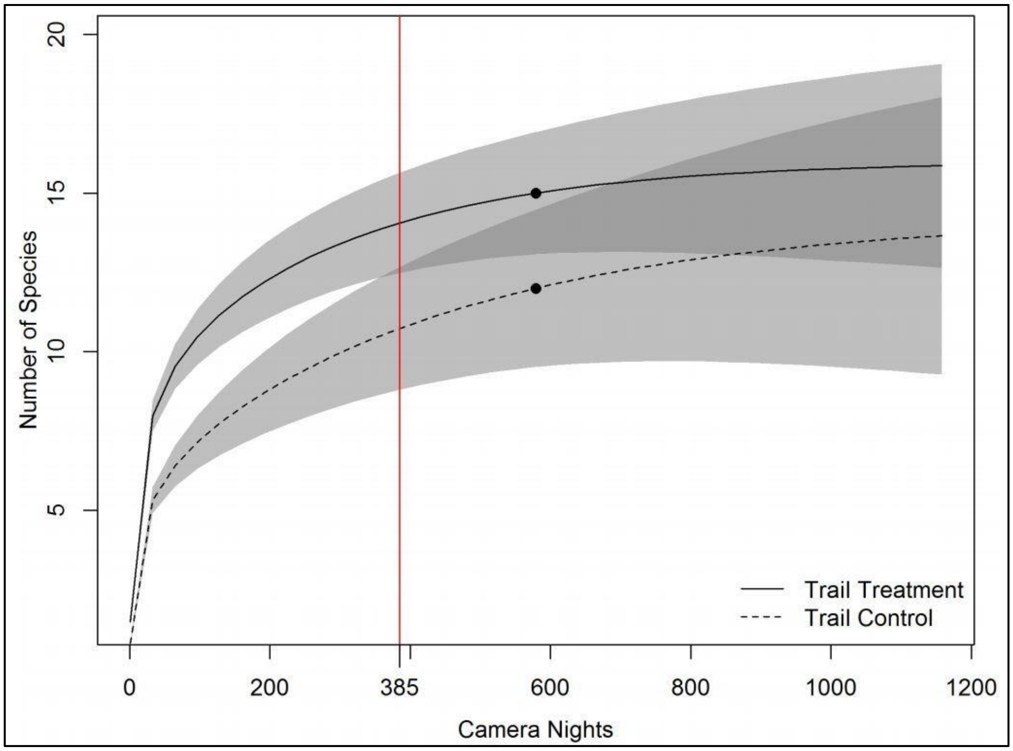

Figure 3. Results from Kolowski and Forrester (2017) comparing feature biased camera placement versus random placement. The vertical line shows where to the two location times become statistically similar in capture rates.

VEGETATION MAPPING AND UNMANNED AERIAL VEHICLES

Comparing Image Data Sources

Previous studies have used remotely sensed data to accurately classify and calculate land cover values using various levels of imagery. From coarse Landsat data at 30-meter resolution (Block 2009; Congalton 1991; Lambin 1999; Meddens *et al.* 2008) to Very High Resolution (VHR) from 5 to 0.5-meter satellite imagery (Uddin *et al.* 2015) to ground level photos directly over top of the subject (Luscier *et al.* 2006). Each level has their advantages and disadvantages concerning scale in land cover classification. 30-meter pixel resolution works well to monitor large area disturbances like harvest cut blocks and for assessing land cover changes through time. VHR data allows us to see finer details (Figure 4), that gives us the ability to see at a stand to stand level compared to the regional scale of Landsat but is expensive. Imagery taken at low altitude gains the ability to see individual plants on the ground, but we lose the context of the regional influence on those individuals. The moderate resolution imaging spectroradiometer (MODIS) provides data for the entire world but is limited to 250-meter, 500-meter and 1-kilometer spatial resolution (Zhang *et al.* 2006).

There are many techniques to classify and qualitatively assess Vegetation Cover from remotely sensed data (Lu and Weng 2007). Supervised classification was used to map an area of *Kalmia spp.* cover over large areas in Newfoundland (Franklin *et al.* 1994). It used the Compact Airborne Spectrographic Imager (CASI) with 2.3-meter resolution and six bands, including multiple NIR bands, acquiring overall 96% percent accuracy of classified cover types on aerial imagery, and 86% on satellite imagery. Even with multispectral data, Franklin *et al*. (1994) only accurately classified road surfaces 93% of the time using the Compact Airborne Spectrographic Imager (CASI) and bare soil 72% of the time from Landsat. WorldView-2 remotely sensed imagery with a 1.8 m (multispectral) ground sampling distance (GSD), reported an R^2 value of 0.79, p=0.05 for their Normalized Difference Vegetation Index (NDVI) and Modified Soil Adjusted Vegetation Index (MSAVI) classification of grasslands (Wiesmair *et al*. 2016). The larger challenge of accurate vegetation classification using only RGB imagery remains. Reported remote sensing results vary in standards of comparison (ARSET 2016; Olofsson *et al.* 2014), as seen in the examples above, making it difficult to evaluate method efficacy between studies.

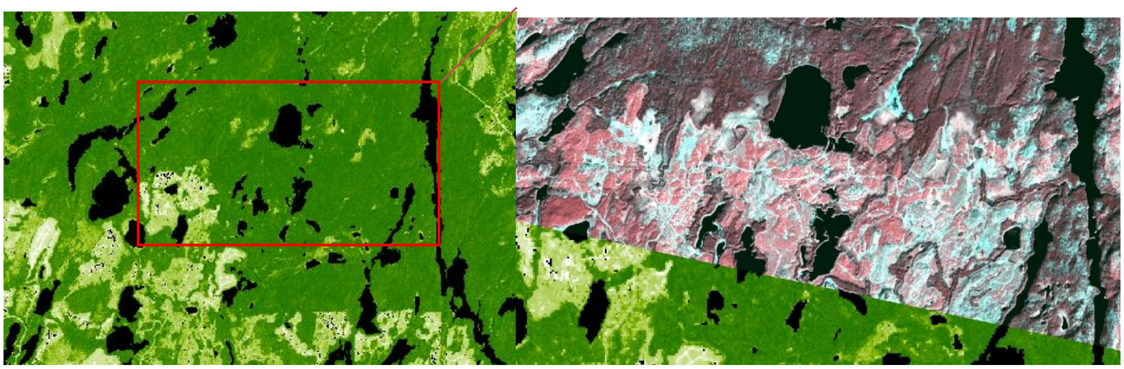

Figure 4. Left: An example of a 30-meter Landsat Forest Cover image from 2000. (Earth Explorer GFCC30TC_p026r026_TC_2000). Right: An overlaid high-resolution satellite image with 3-meter pixel resolution with NIR false colour showing vegetation from summer 2017 (planet.com 20171025_161911_0f31).

Unmanned Aerial Vehicles

As unmanned aerial vehicle technology develops, rigorous methodologies need to be better established to apply them to scientific research. A structured methodology applies to using automated object identification for wildlife monitoring (Christiansen *et al.* 2014; Dulava *et al.* 2015) with various sensors in remote or potentially dangerous areas (van Gemert *et al.* 2014; Watts *et al.* 2010) or mapping illegal forest harvesting and tropical deforestation (Koh and Wich 2012). A recent study in Labrador tested whether UAVs could perform long-range surveys searching for caribou targets. The study assessed factors that lead to successful target detection and was the first study to legally fly beyond line of sight with a UAV in Canada (Patterson *et al.* 2015). Chianucci *et al*. (2016) showed that vegetation cover could be measured using only a stock, off-the-shelf consumer grade, 3-band camera (Red, Green, Blue) mounted to a fixed-wing UAV where images were collected at 170 meters above ground level (AGL). This provided 7.5 cm pixels for classification of leaf-area in dense beech (*Fagus* spp.) forests and reached an $R^2 = 0.7$, $p=0.05$. While this was not strictly land cover values, it proved that RGB imagery could be used to assess vegetation indices with relatively high levels of accuracy. However, it is necessary to continue studying its uses in forestry.

In the past 5 to 10 years of literature, there are more studies using UAVs as a platform for data collection. As developers add new programming to the control functions, the opportunities for field applications and research grow with it. The stigma that UAVs are hard to fly and that the regulations are too restrictive is not true. Although some training and understanding of the rules are required, that can be said for many tools in natural resources fields. Researchers are learning first hand the potential that UAVs provide and can take that experience back to their workplace for assessment.

RECLAMATION AND DECOMMISSION

Figure 5. Left, the extent of excavation of the water crossing and road berm preventing road vehicle access. Right, evidence of access beyond the water crossing removal and berm. Both photos were taken as part of the study area assessment, May 15th, 2017.

Defining Reclamation

Switalski *et al.* (2004) provided an updated cost analysis of different road reclamation methods in mountainous terrain and broke down of the benefits for each. The authors compare four methods (gating \$100-\$2,800, permanent barriers \$800-\$1000, ripping \$400-\$1,200/km and full recontour \$3,000-200,000/km) and if each of those fixes fill stability problems, erosion concerns, wildlife security and their costs. The cost-benefit ratios may be partially intrinsic value, but it is up to the forest manager to what extent they wish to reclaim their roads and if there is a need to proceed with a full recontour. In Ontario, operational costs should be less due to relatively easier operating conditions (non-mountainous terrain) and would, therefore, have a lower cost to benefit ratio.

The term 'reclamation' has been used in several cases with slightly varying connotations or definitions. Bradshaw (1984) defined reclamation as efforts to improve the quality of the land by restoring some of the pre-disturbance ecosystem functions. Bradshaw's perspective seemed to be more utilitarian compared to the more ecological mindset of today but was the end-goal is the same; restoration of previously 'derelict' or degraded land. Jackson *et al*. (1995) define ecological restoration as repairing the anthropogenic damage to the diversity and dynamics of the original system. This definition is more inclusive of which species were present pre-disturbance, species interactions and ecosystem functions/services more so than just presence and structure. The Society for Ecological Restoration defines restoration as the process of renewing and then maintaining ecosystem health (Higgs 1997), which generalizes the ideas of renewal and ecosystem, making the term restoration more ambiguous. It has been argued that the term restoration is restrictive in that it relies on a historical benchmark that requires a high level of experience and expertise to interpret what those historical conditions were (Higgs 1997) and didn't consider future conditions. Consequently, when considering road reclamation, it is essential to define the terms first, so the stakeholders understand the end goal of the project.

After disturbance, regardless if it is anthropogenic or natural, ecological processes will move that site through successional stages along defined, seral pathways to a final established state and, Bradshaw (1984) goes to say that restoration is doing this by artificial means. If the disturbance is relatively minor, then the trajectory of restoration, natural or artificial, should return to its previous stable state given the required time. However, if the disturbance is substantial, then the trajectory may be altered and may end at an altered stable state (Mallik 2003; Walker 2012), changing species assemblages and possibly the ecosystem services of that site. It is generally accepted that the loss of forested land to non-forested land by roads, is minimal, less than 470 hectares per year (OMNR 2016), but could have long-term, cumulative effects on plant assemblages and ecosystems if left unmanaged.

Five road reclamation strategies outlined in the ERF FMP (Lawson 2009b) are: 1) prioritizing zones for road regeneration based on providing caribou habitat, 2) public awareness and signage of the intent to restore the roads to natural conditions, 3) using appropriate silviculture methods to restore the roadbeds to a forested area that matches the surrounding forests, 4) application of treatments as soon as possible and monitoring of site regeneration, and 5) the restriction of vehicle access to protect and promote forest regeneration on the road surface. With any operational road without access provisions or restrictions, the forestry company is still required to have a prior discussion with the local trapline holders before roads in an area can be decommissioned. It is noted that this is to block road vehicles only and not ATV's as experience has proven that completely blocking access is not possible and people will do almost anything to get around or through obstructions put in place.

Benchmarks for Restoration

Boreal forest resilience to disturbance may be challenged soon and may struggle to adapt to the rapid climatic changes that are projected to occur (Brecka *et al.* 2018; Graham *et al.* 1990; Meyer 2006). This may mean that human disturbance may begin to have more significant impacts, which may cause alternate forest states and compositions, affecting forest management decisions and boreal forest wood supply. To better understand the temporal changes that occur on site-specific or regional scales, frequent and accurate monitoring of these areas is required (Lambin 1999). The challenge therein is that the Canadian boreal forest (nrcan.gc.ca/forests/boreal/17394, March 2018) is over 550 million hectares and 50 million hectares in Ontario alone (ontario.ca/page/forest-regions, March 2018). The question is then, how do we monitor that much area on a regular basis?

Two key points that were highlighted in Best Forestry Practices (Jeglum 2003) were to a) develop a monitoring program and b) employ site adapted forestry, although both are easier said than done. This same guideline listed in its mitigation and reclamation strategy to establish and maintain vegetation cover with the establishment of graminoids, grasses, or shrubs. This ensures the minimal amount of nutrient loss, minimizes erosion and attempts to restore the natural processes. While this solves the short-term goal of providing vegetative cover, Holl (2002) argued that planting aggressive ground cover species like these on reclamation areas may inadvertently inhibit the long-term recovery trajectory. Establishing dense vegetation like alder and willow or allowing it to establish can delay the forest succession or create a localized alternate state and change the route of succession, creating an ecosite different to the one that previously occupied that area.

Boan *et al.* (2014) suggested that overstory tree species composition, structure and age were not enough to predict where and when there would be caribou use, suggesting that vegetation alone was not the only characteristic that should be managed. If management strategies remain a-spatial and only account for things such as percent of available habitat for species of concern and not the inter-relationships with other species present in the area, then the populations may continue to decline despite best intentions. This would suggest that more complex ecosystem models such as Structural Equation Models, may be needed to account for the highly varied and multi-level factors that influence the successful regeneration of an eco-site and the return of species like caribou to those areas.

Access to Crown Land

In Ontario and across Canada, public opinion on road management is highly varied, and at times conflicting. Since forest managers must consider the publics' interest for their strategic and operational planning (CFSA 1994), public opinion on how roads should be handled is taken in by the planning committee, and appropriate actions are decided. If there is a consensus on what should be done with the access roads (leave open, return to crown authority, decommission) then it is easy for the planning team to decide; but that is rarely the case.

Often, individuals believe public or crown land and crown resources belong to the public; therefore they are entitled to have unmitigated access to those areas, regardless if there was no previous access (Mihell and Hunt 2011). These individuals have been found to show the least support for road/access management in forest management planning, further supporting a '*cognitive hierarchy*' model to decision making. Those who showed more support for road management, closures or deactivations (of which only 10% of study participants supported complete deactivation) had more biocentric, environmental values (Mihell and Hunt 2011; Hunt and Hupf 2014). Recreational use was also considered in this same study, looking to see if there was a trend related to peoples' support for road closure and the proportion of those individuals who owned or used off-road vehicles like ATVs or snowmobiles.

The lowest support for road closures were people found to have high usage of forest roads and off-road vehicles for the pursuit of consumptive and recreational activities. As a result, managers cannot simply decide based on the desire for access roads based recreational activities alone, because those public values may only represent a portion of the overall public vote. Even if the majority agreed with closure and a road was physically deactivated with public notice and signage stating that the road was closed to public access, there are those few individuals who ignore those facts and will continue to use what remains of the roadbed.

Hunt and Hupf (2014) showed that treated roads, deactivated/decommissioned, had less motor vehicle traffic than those that remained open or untreated during the first five weeks of the moose hunting season. They also showed that there was no significant difference in traffic levels between road quality and treatment methods. This means that although deactivation dissuades people from continuing to use closed roads, it does not entirely remove traffic. Enforcement during hunting seasons may aid in reducing the number of people using roads that are closed to the public (Hunt and Hupf 2014). However, given the amount of area and the low number of Conservation Officers in the area, the probability that a person being charged for unlawful entry is low; where individuals see the personal benefits as outweighing the costs.

Benefits to Road Reclamation

There has been more effort to restore forested habitat connectivity (Switalski *et al.* 2004) by removing roads which can act as travel corridors or as physical barriers to sensitive wildlife species (Walker 2012). Removing the linear feature can blend the land back into the surrounding forests, reducing the amount of forest edge. The increased contiguous forest area then has increased timber potential from the lowered risk associated with forest edge-based disturbances such as overheating and windthrow. Moreover, depending on the silviculture methods used, if the roads are seeded/planted with commercial tree species instead of quick growing shrubs, those trees then can add to the fibre volume in that area while reducing the amount of productive forest land to non-forest land.

POLICY

Environmental conservation has mainly focused on 1) *preserving* ‘natural’ areas by placing them under a protected status within the boundaries of human activity, 2) preserve, improve or restore the levels of biodiversity outside of natural areas and 3) attempt to restore and maintain environmental and ecological integrity in areas where traditional boundary style methods of conservation are not practical (Cumming 2016). These ideals have led to a movement of conservation efforts to expand and acquire more protected areas with a growing variety of types and levels of protection around the world. However, by its definition, the act of preservation hampers new growth or change, always trying to return a system to an anthropocentric, previously defined ‘natural’ level (Monbiot 2014).

The *Forest Management Guideline for the Conserving Biodiversity at the Stand and Site Scales* (OMNR 2010a; OMNR 2010b) states access roads should be designed as temporary in significant habitat areas. They should be removed as soon as possible after harvest operations and the area returned to productive forest. OMNR (2010) gives a list of standards, guidelines and best practices for road decommissioning to meet the forest management planning and SFI/FSC requirements. Roads planned to be returned to the Crown remain available to the public while accessible and if the road is no longer needed it is to be decommissioned, allowing it to regenerate ‘naturally’. Roads are assessed for environmental and safety concerns, and appropriate deactivation methods are determined based upon biological, water quality, engineering and safety factors. Depending on conditions, almost all water crossings will be removed preventing road vehicle access (OMNR 2010a; OMNR 2014a). According to the supplementary materials (Lawson 2009b), road berms and ditching will occur in areas to protect any sylvicultural investments or to deactivate roads. Other decommissioning techniques available to forest managers include scarifying, slash piling to aide in planting, seeding silviculture or natural regeneration of the site. The current forest management plan also states that natural ingress of brush such as alders and dogwoods will be encouraged as this acts as an obstruction to predator movements and line-of-sight.

Within the MNR’s Crown Land Management guide (OMNR 1993), the mission of the MNR was stated as:

> *“further the public interest by preserving clear title (legal ownership) of Crown Land which enables us to: safeguard the environment; promote conservation; ensure sustainable use of land through an integrated (terrestrial/aquatic) ecosystem approach to management; and*

> *support social and economic development of land where such development is compatible with the environment and other interests".*

It is important to note the order of the MNR's goals outlined above, where the basic, guiding principles are that environmental and ecosystem integrity are to be regarded as of utmost importance while supporting social and economic development. The Ministry of Natural Resources and Forestry (MNRF) have in place the "Forest Management Planning Manual for Ontario's Crown Forests" (FMPM) to help guide forest license holders in the management of crown forests they are utilizing for timber or wood fibre extraction (OMNR 2009). Road networks allow for resources to be extracted, yet they also have tremendous impacts on forests and the wildlife that live therein.

The definition of the actions to be taken when looking to decommission a road in a Forest Management Plan is: "*For roads or road networks identified for transfer to MNR where MNR's management intent is to not maintain the road for public use, the physical work that will be undertaken to render the road impassable to vehicular traffic, enhance public safety and reduce potential environmental damage (e.g., removal of a water crossing(s)). The roadway will degenerate over time.*" (PG 431, OMNR 2009). For roads held by the licensee or being returned to the MNR, tables list all *"road construction, maintenance, monitoring, access controls and decommissioning"* and will identify:

> *(a) the corridors for primary roads (20 years);*
> *(b) the corridors for primary and branch roads planned for construction (10 years);*
> *(c) the corridors for primary and branch roads planned for construction (5 years);*
> *(d) the operational road boundaries (5 years);*
> *(e) the areas of concern within the corridors and operational road boundaries;*
> *(f) the 100-metre wide crossing of each area of concern;*
> *(g) the acceptable alternative locations or location restrictions for each crossing of an area of concern;*
> *(h) the roads that will be maintained;*
> *(i) the roads and associated water crossings that will be monitored;*
> *(j) the segments of roads that will have access controls implemented, and the type of access control activities; and*
> *(k) the segments of roads that will be decommissioned, and the type of decommissioning activities.*

In the Roads and Water Crossings monitoring section, any recently decommissioned roads are not mentioned. The only mention through the document was a list of roads to be decommissioned in the future and decommission activities. Roads with water crossings that have been decommissioned are checked for compliance under the *Fisheries Act (*Environment and Climate Change Canada 1999*)* protecting essential fish habitat, but there was no description in the FMPM of any review or compliance checks for roads that have no water crossings within their boundaries.

In the *Environmental Guidelines For Access Roads and Water Crossings* (OMNR 1990), which is supposed to outline the best practices in building access roads to ensure 'environmental' protection only covers water crossings and the protection of fish habitat. The only outlines for when a road is no longer in use are *physical* or *natural abandonment*. Both suggestions have little to no site reclamation, and physical abandonment is only designed to prevent vehicle access (e.g. berms, trenching, water crossing removals). This guide's (OMNR 1990) solution to unused roads is abandonment; again, assuming the road will degenerate or disintegrate over time with no real solution to repairing or removing the linear features and restoring them to a forested condition.

Forest Certification

Forest activities in Canada and Ontario are regulated by third-party organizations that assess forestry practices and set laws and regulations based on scientific indicators (NRCAN 2017). Forest Stewardship Council Canada (FSC) and Sustainable Forestry Initiative (SFI) are two examples of certification systems in Canada that perform the audits of forest operations, require annual reporting, and offer chain-of-custody from the forest through to the point-of-sale. They are ensuring conservation of biodiversity, protection of sensitive habitat, protection of wildlife and species at risk, protect water quality, respect of Aboriginal and treaty rights, and prevent illegal logging or the importing of illegal wood products (FSC 2004; NRCAN 2017; SFI 2015). Since forest certification was adopted in the 1990s, Canada now has the largest areas, over 168 million hectares, or 37% of the world's independently certified forests (NRCAN 2017).

FSC and SFI national standards require the maintaining of large, contiguous areas of representative forest habitat, conserving biodiversity and protecting wildlife and species at risk, such as woodland caribou, with connectivity between key habitats (FSC 2004; NRCAN 2017; SFI 2015). This is done through standard operating procedures in forest management plans (Lawson 2009a; 2009b) where wildlife habitat in addition to forested land is maintained through silviculture and road removal/reclamation. Annual reports of these efforts are given to the FSC and SFI certification boards monitoring the efforts of forest companies and the success of forest management strategies. As an example, the FSC Canada National Boreal Standard, which covers the entire boreal forest within the country, requires reporting of current forest conditions, including roads and linear disturbance density to serve as a reference during landscape impact assessments. Collection of that information is delegated to the individual forest license holder and is an expensive and extensive effort.

Best Management Practices

The document with the most information, the *Forest Management Guide for Conserving Biodiversity at the Stand and Site Scales* (OMNR 2010a), s. 5.1.1.3 states that roads planned for decommissioning should be executed on the same schedule as water crossing removal. The only environmental concern listed in the entirety of the road decommissioning guidelines are erosion and sedimentation of any watercourses that transect the road, which can affect water quality and fish habitat. Out of all the documents reviewed that pertain to road reclamation in Ontario, only the Stand and Site guide (OMNR 2010a) suggests the rolling back of grubbed materials such as earth, stumps or other organics that were pushed aside in the construction of the road. The Landscape Guide was also referenced within this section yet was inaccessible at this time.

UNMANNED AERIAL VEHICLES

The advancement of unmanned aerial vehicles (UAVs) in recent years has mainly been a result of the increased demand in the hobbyist market. The public demand wanted more than just a radio-controlled helicopter as a toy and the industry moved towards a platform for professional quality photographs and videos. Combined with the mass development of accelerometers for cell phones, the technology has become highly miniaturized and affordable (Anderson and Gaston 2013). Systems that used to cost thousands of dollars twenty years ago, now cost hundreds. Also, these highly accurate, lightweight GPS systems, toys that used to just take simple pictures have now become a serious tool that can be used to collect scientific data, repeatedly, accurately and affordably.

UNMANNED AERIAL VEHICLES IN NATURAL RESOURCES MANAGEMENT

Before the advent of low-cost UAV platforms, imagery came from flying airplanes and helicopters flying at low levels, close to the tree canopy and had higher inherent risks. More and more research is being completed as UAV technology develops along with new methodologies to apply them. Whether it is using automated object identification for wildlife monitoring (Christiansen *et al.* 2014; Dulava *et al.* 2015), using various sensors in potentially dangerous circumstances (van Gemert *et al.* 2014; Watts *et al.* 2010) or mapping illegal forest harvesting and tropical deforestation (Koh and Wich 2012). A recent study in Labrador tested the utility of UAV long-range surveys for caribou targets and what factors lead to the successful detection of a target (Patterson *et al.* 2015). It the first study to legally fly beyond line of sight with a UAV in Canada. In British Columbia, Goodbody *et al.* (2017) reviewed some capabilities of UAVs specific to forestry. The platforms were highlighted as an effective means of collecting digital aerial photogrammetry and creating point clouds. These point clouds were then used for tree species identification, building canopy models and deriving wood volumetrics.

Other studies have shown that analyzing pixel information within large-scale imagery is possible. Percent vegetation cover was previously determined using a subset sample to detect a minimum of 4% difference in cover (Moffet 2009) and measure the above-ground biomass of invasive plant species (Blumenthal *et al.* 2007). Images of the study areas were broken into different age groups or time-since-disturbance, fire, with three images being selected at random from each cohort. Cover was then measured as the proportion of each category within each image to be confirmed with the ground measurements later. Although this study was completed using a small fixed-wing plane, this method can be applied to the imagery collected by a UAV. In post-analysis, statistic validity was calculated using linear regressions and compared the ground truth measurements and image measurements (Moffet 2009). This method is similar to the approach taken in this study; ground truth measurements were held against the remotely sensed imagery measurements done in the lab. Then a linear regression was used to compare the two sets of measurements with the assumption that there should be no significant difference between the two sets of measurements.

It has been shown that vegetation cover could be measured using only an RGB camera mounted to a fixed-wing UAV (Chianucci *et al.* 2016). Images collected at 170 meters AGL, providing 7.5 cm pixels for classification of leaf-area in dense beech forests (*Fagus spp.*) forests reached an $R^2 = 0.7$, $p=0.05$. While this is not strictly land cover values, it proved that RGB imagery could be used to assess vegetation indices with relatively high levels of accuracy. The rule set from Chianucci *et al.* (2016) was tested against the orthomosaic tiles from this study to see if it was broad enough in definition to apply to a boreal setting. While it was able to generally classify the vegetation, there were lots unclassified areas and relatively high levels of misclassified objects caused by differences in lighting and plant species. It was found that while a good starting point for classifying boreal imagery, it needed significant adjustments and prevented a ‘drag and drop’ workflow.

If the target of the imagery was to measure the leaf area index of small herbaceous plants, a higher image resolution with a small GSD is required, leading to a smaller extent and covering less area (Kamada and Okabe 1998). Therefore, it is crucial for the researcher to understand this trade-off and to plan for data collection accordingly, ensuring images will provide appropriate information for the task at hand. It is also important to note that overcompensating with higher than necessary image resolution also poses challenges in post-analysis. If one is attempting to answer landscape-level questions, having a smaller extent may lead to missing points of interest, holes in the data or must be compensated by taking more photographs. This means more data storage and more hours in post analysis and computational time.

ACCURACY ASSESSMENTS

For any remote sensing project, accuracy assessments are necessary to be able to validate any results. In the comparison by Meinel *et al.* (2001) between ERDAS and eCognition, both programs returned overall accuracies of 89.6%, but the parameters were not equal. The image resolution was 1 meter for eCognition compared to the 4-meter resolution that was tested on ERDAS, and eCognition was able to separate two more classes than ERDAS with the same level of accuracy. Huang and Ni (2008) also showed the performance difference with a pixel-based accuracy of 75.67%, with a Kappa of 0.7025 and their object-based accuracies resulting in 83.27% and a 0.7792 Kappa. Luscier *et al.* (2006) used eCognition to measure the amount of area held by four classes of vegetation and reported K values ranging from 80-100% on various levels of different vegetation covers. Luscier *et al.*’s (2016) success of classifying vegetation allowed them to accurately and quickly image the study sites while reducing field time, costs and reducing the impact of in situ measurements.

Having broader classification levels should allow for the use of automated, unsupervised classification of the aerial imagery, further increasing the efficiency. This would give basic results such as percent cover of the road surface or percent tree to non-woody plant ratios, giving a basic representation to forest managers. Once the images are classified, an observer can then review them and visually identify different species or even assess the health of the vegetation. On the other hand, gathering intensive field data like Leaf Area Index (LAI), recording plant ID, soil/aggregate types, and the surrounding forest ecotype that could influence the species found on the site can be used later to make more detailed estimates with the image data. If the post analysis is found to be inaccurate because the of the intricacy of the ground detail, the user could degrade or run a resample algorithm on the ground data reducing the data resolution and making it easier to produce more accurate predictions from the photos. Increasing the number of field sample plots will increase the accuracy of the regression (Duniway *et al.* 2012). It is also important that there are enough field ground truth plots to represent all the different classification levels and maintain statistic validity across all expected image classes.

A disadvantage of traditional aerial photo sampling is that the imagery only captures the top layer of vegetation and cannot penetrate to the ground layer if there is a dense cover (Duniway *et al.* 2012; Moffet 2009). Shorter plants such as lichens, horsetail or moss that tend to grow in the shade could be covered from aerial imagery and may not be measured if there is only a vertical image of the plot. This can be alleviated with using drones by taking a series of vertical photos on either side of the ground plots, and using the angle of view, to see underneath taller shrubs or trees (Kamada and Okabe 1998).

METHODS

STUDY AREA

The study area (Figure 6) of this project was based in the lower-middle area of the English River Forest (ERF) forest (Appendix 1) in northwestern Ontario. The ERF forest license is held by Resolute Forest Products Canada Inc., (previously known as AbitibiBowater Inc. at the start of the 2009-2019 FMP). The total land area for the ERF is 1,032,763 hectares (ha) with 777,167 ha (75.25% of total area) of production area, 5% non-productive, 2% of the land base being protected and the remaining area is listed as non-forested (Lawson 2009a).

It was reported in the 2007 Forest Management Plan that the road density for the ERF was 0.856 km/km^2 of ‘productive land’. The study area falls between the Moberly Lake/ Brightsand River conservation area to the east, the Tri Lake area to the west and Baltic Lake on the north side with the Wagner private forest bordering the south. The Moberly lake area is a conservation area highly valued as a tourism destination, and the Tri Lake area is popular among fishing enthusiasts. Both of which increase vehicle presence in those areas. The overall area was found to be an essential source of moose aquatic feeding habitat (Lawson 2009a). Total land area of the study area was calculated to be 87 km^2 of the roughly 10,000 km^2 total license area. As of May 24th, 2017, calculated road density of the entire ERF license using all listed roads was 0.606 km of road per square kilometre (6,256.34 km of road/ 10,327.63km^2). If the roads marked as decommissioned were removed from the list of all listed SFL roads, and no longer considered part of the road network, road density is 0.588 km/km^2 for the English River Forest SFL. For future comparison, road density of 0.59 km/km^2 will be used as it was assumed, for the time being, that roads listed as decommissioned were no longer considered present on the landscape.

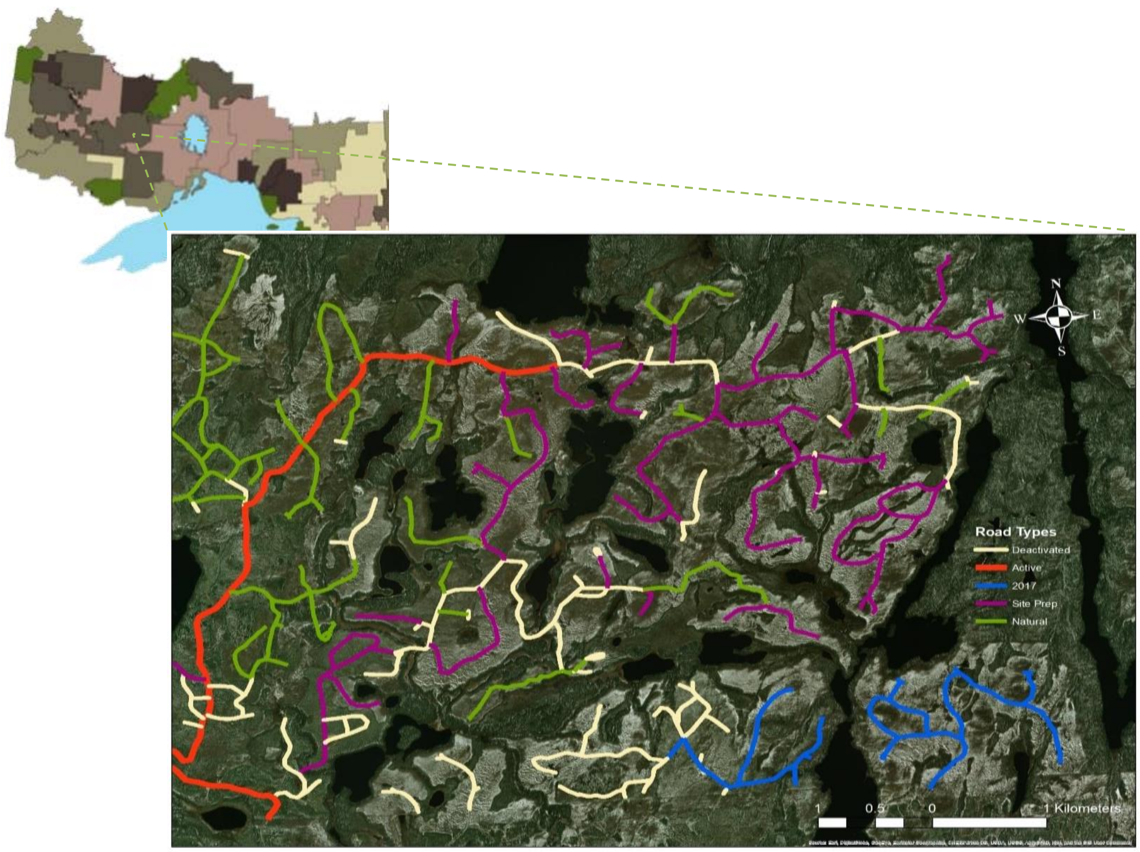

Figure 6. Study area in the English River Forest. Entry is gained via the southwest side on the remaining active (red lines) road, in the bottom left of the map. The various road segments are colour coded according to their status and treatment method. The northwestern Ontario forest management licenses are shown in the top left inset.

ROAD DEACTIVATION

Decommissioning efforts were applied to different road segments (one to ten years ago) that best complemented silvicultural efforts in adjacent cut blocks, Figure 6. All deactivation efforts were completed before commencement of this study, with separate sections deactivated on different years due to logistic constraints (See “TSD”, Appendix 12). This included road deactivation or trenching, at the point of entry, at any water crossings that were removed and at the midway points along main strategic roads connecting different sections. Trenches were created at the time using an excavator by digging a channel perpendicular to the road direction, down through the roadbed. The excavated material was then placed on either side of the channel increasing the height of the banks on either side attempting to prevent vehicle traffic from passing. Road surface scalloping was completed along the lengths of different sections (Figure 7). Scalloping was done using an excavator moving along the length of the road scooping soil from the roadbed or the road edge and piling the removed sediment onto the road surface in a criss-cross pattern creating a mogul-like texture. Trenching and scalloping were done to try and prevent motorized vehicle traffic travel along the roadway, breaking up the drivable surface and speed up the ecological recovery of the site. If there were large rocks or boulders, logs and other organic debris nearby the operator placed these items on the road as another source of obstruction.

Other than the central access road which remains active, the in-block road segments and skid trails were either site prepared the same as the surrounding forest harvest area or left abandoned, free of mechanical disturbance. Since the area was going to be aerially seeded to replant the forest blocks, it was assumed that the site prepared roads segments would also be seeded.

Figure 7. Examples of reclamation techniques. Left, scalloping of the roadbed with aggregates and coarse woody debris. Right, a water-crossing removal at a road entrance from an active road segment creating a channel or trench. The culvert removal also acts to potentially bar vehicle access.

CAMERA TRAPPING

18 Cabela's *Outfitter 14MP Black Infrared HD* trail cameras were used for wildlife monitoring. This model is powered by eight AA batteries, stores 14MP (day) or 5MP (night) images and 720p HD video with audio and uses 40 'black' infrared LEDs to capture photos and video in low light conditions. The infrared LEDs allowed for night capture events without a traditional, visible flash, preventing altered behaviours. These trail cameras use Passive Infrared motion sensor (PIR) with a maximum detection area of 30° each side of center for the first 25 feet, then narrowing going out to a 100-foot limit of the daylight detection range (85 feet nighttime range). Camera settings used were: 14MP resolution for images, trigger speed was set at 1 second with a 3-image burst mode and a 10-second video with a one-minute delay (Rovero and Marshall 2009) between triggers. PIR sensitivity was initially tested at the high setting but field testing showed that a low sensitivity setting was the most efficient.

Cameras were set up at each site to capture the most area of the reclaimed road as possible. Units were placed on the outside edges of the deactivation area, or at road center facing parallel to the road direction, so that more of the road surface remained in the field of view. The cameras were attached to suitable, stable trees with either the adjustable strap or base plates provided and a python cable lock. All cameras were placed at least 30 centimetres above the tops of any ground vegetation to ensure a clear view of the sites; generally, between 0.5 meters to 1.5 meters above ground level. They were also adjusted to ensure the PIR sensor and the cameras were aimed at the targeted path of travel.

Locations were plotted using geo-referenced digital maps, on the GIS road shapefile and Google Earth roads layer for post analysis (Figure 8). To standardize the use of camera trapping techniques, (Meek *et al.* 2014) distances between camera locations were then measured from the established sites and tabulated, (Appendix 17). The average distance between cameras was 3,237 meters with a minimum distance of 61 meters (cameras C6 toC20b) and a maximum distance of 7,112 meters (cameras C17 to C19b).

Figure 8. Camera Locations within the English River Forest study area.

Since one goal of the camera trapping was to monitor natural behaviours along the road networks, no lures or baits were used to attract animals in front of the cameras. Attractants can result in elevated abundance counts or completely different behaviours of photographed species (Meek *et al.* 2014). Baiting could lead to seeing species not typically found in that specific area, or species that would not normally interact, changing inter-species interactions such as predator/prey interactions.

For a standardized approach, a triggering *event* was referred to when a subject (animal or person) passed in front of the passive infrared sensor triggering the camera to fire and capture the programmed number of photos or video (Meek *et al.* 2012). It must be noted here that when the batteries were changed part way through the study, some of the cameras' settings were reset to default which went unnoticed by the operators, resulting in different capture programs such as the 3-photo burst. These cameras still triggered normally but did not capture the file types as planned.

Photographs were sorted afterwards by location and date, and with all false triggers separated. Captures were tagged with the camera number, location, site type, file number and location, subject name and species, sex if possible, date and time of the event, an identifier for unique individuals if possible and the activity of the subject.

GROUND VEGETATION PLOTS

Vegetation data survey methods (Figure 9) were completed using a amalgamation of the Ontario Permanent Sample Plots (Sharma *et al*. 2008) and Permanent Growth Plot survey methods (OMNR 2015), the quadrat method for vegetation sampling (Peet *et al*. 1998; Stohlgren *et al*. 1995) and Canada's National Forest Inventory Ground Sampling Guidelines (Natural Resources Canada 2008). Sites were established at random points along road sections that were accessible and stratified by road type; active, natural regeneration/abandonment and site prepared. Transects were 50 meters long centred mid-road, with five, 5m^2 circular plots (radius = 1.26 meters) and the center plot (plot 3) placed in the center of the road. Two regeneration survey plots were placed at each end of the transect (plots 1 and 5) at ± 25 meters, and survey plots (2 and 4) fell on the road edge/transition area centered with half of the area road surface and half forest floor (or ditching if present).

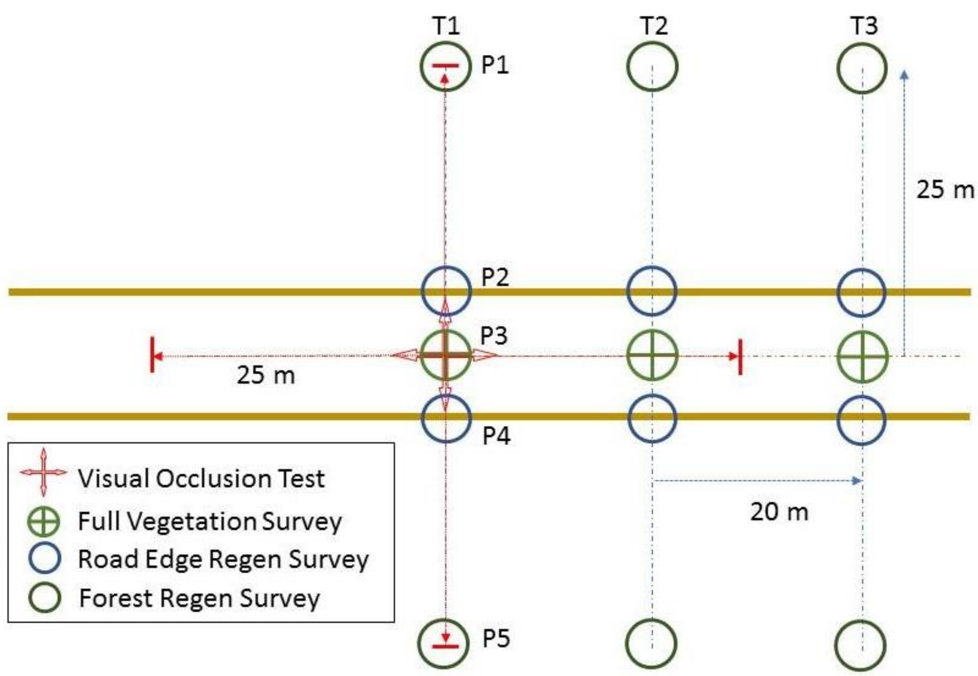

Figure 9. Ground truth plot and visual occlusion test layout. Transects (T1, T2, T3) were established with T1 being the closest to the main access point, with 20 meters between each transect. Each circle plot was 5 m^2, with plots 1 and 5 being representative forest measurements. Plots 2 and 4 were road edge, and plot 3 was placed at road center.

The transect distance of ± 25m, was assumed to be far enough away from the road that there should be no 'edge effect' on vegetation within plots 1 and 5 (Harper *et al.* 2015). The center plot, consisted of a full vegetation survey, assessing all plant species (trees, shrubs, sedges and other vascular species) growing on the road surface. The assumption being tested was that sites with more intensive reclamation effort should see an increase in plant species richness and abundance (Walker 2012). Coarse woody debris (CWD) was measured along each transect, with intercept, small and large end diameters, along with species (if possible), origin, length, distance from road center, state of decay and if there were any signs of wildlife use. Duff materials were also recorded, and an average duff depth was measured within each plot to the nearest centimeter.

Vegetation cover was measured as the amount of ground area within each plot covered by the vertical projected column of plant species, where small gaps and openings are included, for each species found within the plot boundary giving a total cumulative 'Canopy Cover' (Bedell 1998; Karl *et al.* 2016). Cover percentage for one species cannot exceed 100%, but the total sum (vegetation cover) of all species present within the plot may total to more than 100% coverage (Peet *et al.* 1998). This is to account for multiple layers of vegetation that could potentially be growing on each site; moss and lichens, grasses and sedges, small shrubs, to trees. Hypothetically, a plot could be completely covered in lichen (100%), could also support multiple blueberry plants (50% coverage) and have several pine branches overhanging the plot (25% cover). This plot would then have a hypothetical cover value of 175%, accounting for the structure of multiple layers of vegetation. Using this method allows for more of a vertical comparison of vegetation between plots along the transects and between sites. Due to time limitations, only the tallest member of each species was measured for heights, age, diameter at the standard diameter at breast height (DBH) of 1.3 meters or 30 centimetres if the plants were shorter than 1.3 meters.

Sites 1, 2 and 3 were located on active road segments, sites 4 and 5 were in the south access tile which was previously deactivated over ten years ago. Sites 6 and 7 were in the northeast, deactivated section, passed the culvert pull trench put across the road. Sites 8 and 9 were located on the northwest section on a road segment that was abandoned and left to regenerate naturally. Sites 10 and 11 were also on abandoned roads but further south in the middle west tile near site 3, which all had mature forest stands surrounding them. Finally, sites 12 and 13 were in the southeast corner of the study area, on roads recently deactivated (within the year), having been abandoned since the area was harvested.

Treatment significance was tested with a factorial ANOVA. The linear model tested was: Cover ~ Road type * Ecosite * Plot + Site * Plot. Sites, while randomly distributed through the study area, were fixed factors along with plot. Road type and ecosites were random, as these were uncontrolled factors that were measured. Significance was then tested using the Tukey's Honest Significant Difference (HSD) test to identify which treatments led to different levels of cover.

Visual Occlusion

A visual occlusion test (Bowkett *et al.* 2008) was done using a one-meter square, white placard. This was done to estimate the visual line of sight for predators and prey, correlating to the amount of vegetation growing on each site. A white $1m^2$ placard was held vertically, 1m off the ground at 25 meters away and facing site center. The percent of the card remaining visible from center was then measured for each 90° in rotation around the site: two records parallel and two records perpendicular to the road direction (Figure 10). Visual estimations were recorded, and photographs were taken of the card from the center, also at 1-meter AGL, simulating predator line of sight for comparison. Higher percentages of placard visibility left uncovered (top left and bottom right in Figure 10) indicates a longer line of sight for a predator and potentially higher risk for prey. This visual test was completed at each site, testing the hypothesis that there should be no difference in line of sight visibility between either direction if road reclamation was successful. Significance was test again using a factorial analysis. The model used was Percent obstruction ~ Ecosite + Site + Time since disturbance + Road vs Sight line + Road forest. Line of sight visibility, or percent obstruction was the dependant variable. Site and sightline were fixed factors, while ecosite, time-since-disturbance, road treatment and road forest were random. Significance was then tested using the Tukey's Honest Significant Difference (HSD) test to identify which treatments had significant changes in line of sight.

Figure 10. Sample photos of the visual occlusion test at site 5. Top left: 100% at 14° from 25 meters. Top right: 2% at 280°. Bottom left: 2% visibility at 98°. Bottom right: 90% at 189° from 25 meters.

IMAGE SAMPLING

Aerial photos were collected using the DJI Inspire 1 and an iPad to control flight parameters. All flights were completed under the Special Flight Operations Certificate Number 17-18-00045262. The ground-measurement sites were flown with the UAV, taking images of the vegetation at each site at 120 meters above ground level (AGL). Aerial image sampling of the road network was limited to areas that were safely accessible by ATV, preventing damage to vegetation or create more disturbance to the site. This limitation reduced the number of launch points of the UAV and the range which could be flown while remaining legally within line of sight (500 m restriction) of the machine. For more detailed information on the specifications of the UAV system and the breakdown of changes in image resolution, see Appendix 13 and Appendix 14.

After vegetation sites were photographed, the UAV was also flown at 50 and 120 meters over lengths of the road network capturing images that were later used for the random sampling of road vegetation for the different road treatment methods. Images were then filtered to remove those with high levels of motion blur, were not downward facing or had unsuitable lighting conditions to reduce the 3D model errors, streamline the computational processes and the building of the orthomosaics for the large-area sampling.

Image Classification

The images collected from the aerial mapping flights were first brought into a photogrammetry, structure-from-motion computer program called Agisoft Photoscan. This program allowed the automated colour matching and assembly of all images (roughly 4 000) by overlapping identical pixels from adjacent and overlapping images and rendering all the pixels in a 3D, virtual space. A dense point cloud of image pixels was created, then blended to form an orthorectified, mosaic image or orthomosaic. Using the standard DJI Inspire 1 camera, all imagery was GPS-located with internal IMU measurements and standard Red/Green/Blue (3-band) colour. The large orthomosaic covering the study area was then broken into smaller sections for easier handling and faster automated classification.

The next step was to begin classifying the orthomosaic tiles. Two software packages were used; ERDAS Imagine and eCognition Developer 64. ERDAS analyzes images pixel by pixel and classifies by individual pixels (Meinel *et al.* 2001) whereas eCognition uses object-based algorithms to classify images (Meinel and Neubert 2004). The first step was to run an *Unsupervised Classification* in ERDAS, labelling the classes accordingly, then subset the sample areas that match the ground truth plots of the same sites. An *unsupervised classification* separates the pixel information and sorts them into different bins or classes. The number of classes was dictated by the user for all imagery analyzed in ERDAS Imagine and were all run with 100 different classes. These were then reduced by the user with the *recode* tool to the required number for the poly-class and the two-class images that were later tested.

Concurrently, those same orthomosaics were also imported into eCognition and classified based on a user-defined rule set. A random tile was used to create a 'training' rule set, classifying the image into feature classes like vegetation, soil, water, lichen or other features found in the image. Once the parameters were found acceptable, that rule set was then applied to the remaining orthomosaics, automatically classifying the images. That rule set was:

- Rule Set:
    - Segmentation:
        - Multiresolution Segmentation: Scale of 50 [shape:0.1, compactness:0.5]
    - Classification:
        - Unclassified w/ Mean Layer 1 (red band) >200 at New Level: ***road***

- With Arithmetic Feature 1 > 0.0655 at New Level: ***vegetation***
- Unclassified with Mean Layer 2 (green band) <= 120 at New Level: ***shadow***
- Unclassified with Mean Layer 3 (blue band) < 139 at New Level: ***lichen***
- Unclassified at New Level: ***nonVeg***

For the second classification line, the arithmetic equation was (2*[Mean Layer 2]-[Mean Layer 1]-[Mean Layer 3])/(2*[Mean Layer 2]+[Mean Layer 1]+[Mean Layer 3]). The resultant image was then exported from eCognition and brought into ERDAS to perform the accuracy assessments.

Accuracy Assessment

The next step was to assess the classification accuracy. To validate the imagery for random sampling of other areas, ground truth plots used to train the parameters of the aerial photos must be kept separate from the samples used to test for accuracy and precision (Congalton 1991; Luscier *et al.* 2006). Validation data were collected in the same manner as the samples used for training (Karl *et al.* 2012). The eCognition files had to be brought into ERDAS Imagine in the ERDAS viewer, as this software was the only one of the two used that had an accuracy assessment tool.

For this study, two different Accuracy Assessment (AA) methods were used. First, an AA was completed using the ERDAS *Accuracy Assessment Tool* found within ERDAS, using a pixel-based error matrix (Congalton 1991). Then again, using an area-based error matrix (Olofsson *et al.* 2014). AAs were also completed comparing poly-class (Conifer, Hardwood, Lichen, Road, Soil, Water) classified images and then only assigning two classes (Vegetation and Non-Vegetation) for the same images. The AAs were completed on the same 3 test ortho tiles, '*south access', 'middle',* and the 'no*rth'* tiles for all iterations of the AA process. Assessments were done in ERDAS by generating a selection matrix of 50 random pixels per class (ARSET 2016; Olofsson *et al.* 2014) within the image using a stratified random sampling method (Congalton 1991) to ensure that each class received a minimum number of samples but still be randomly placed within the class areas. If simple random sampling were used to do the AA, there was a chance that smaller but likely important classes would be under-sampled and not adequately tested.

For those images that had two classes, at least 100 sample points were generated; for the images with five classes, over 250 sample points were generated. The original unclassified images were used to aid in assessing what the sampled classified pixels were and assigned a '*true*' class number that in the AA matrix under '*reference class*'. This process was repeated for all three image tiles, for the two different programs and with the poly-class and 2-class images.

The *User's* (correctly classified points divided by the total classified points for a class) and *Producer's Accuracy* (correctly classified points divided by the total points of the same class) are also listed for each class and the *kappa-statistics* (*K*) showing the class/sample agreement while accounting for chance (Olofsson *et al*. 2014). The two accuracies along with the overall accuracy and the error matrix itself can tell us how accurately the automated classification was. Overall accuracy gives a relative understanding of the accuracy but lacks detail in which classes were confusing the classification. An image with an overall accuracy of 80% (Luscier *et al.* 2006) to 85% is generally considered accurate depending on the intended use of the mapped features (ARSET 2016; Olofsson *et al.* 2014). Generally, the higher the associated risk involved if features are misclassified, then a higher level of required overall accuracy (95-99%).

The second accuracy report method, the area-based error matrix (Olofsson *et al*. 2014), was used because it uses a population error matrix. This adjusts accuracy levels based on the proportion of area and reduces area bias in reporting (ARSET 2016) while providing standard error and confidence intervals. For this set of AAs, the ARSET (2016) method was used to derive the classification accuracies for the same images as the pixel-based approach. The difference comes in the addition of calculating the proportional area of each class within each image, then using that value to calculate a weighted *Standard Error of Area* and *95% Confidence Intervals* (CI) in hectares for each class. An unbiased *User's Accuracy* is then calculated by dividing the correctly classified area values by the total proportion of areas classified of that class. Using this area-based method, we can report the accuracies for each class within each image but also the CI and the amount of area that a particular class represents.

Aerial sample plots

After accuracy assessments, the next step was to sample the mosaic to correlate the ground data with the aerial imagery. A circle plot was drawn using the georeferenced image with a radius of 1.26 m (5 m^2) matching the ground truth plots that were done in the field. This circle was saved as an Area of Interest (AOI) file to be used in the next steps. A subset image was created, with the saved classified image as the input file (ex NW_Naturaltile), and the output file named for the plot number and location (ex. Site 1, Transect 1, Plot1). This resulted in 15 circle-plots aerial samples/site that match the exact location of the ground truth plots at each site, which have been classified and ready to be assessed for vegetation cover, Figure 11. This same process was then applied to the *eCognition* classified images using the same AOI files. The two different aerial sample data sets (ERDAS and eCognition) were then used to contrast between the two programs the calculation of percent vegetation (Figure 24). By using the classified ortho-tile as the sampled layer, it ensures that all plots within the site have uniform classifications across all plots.

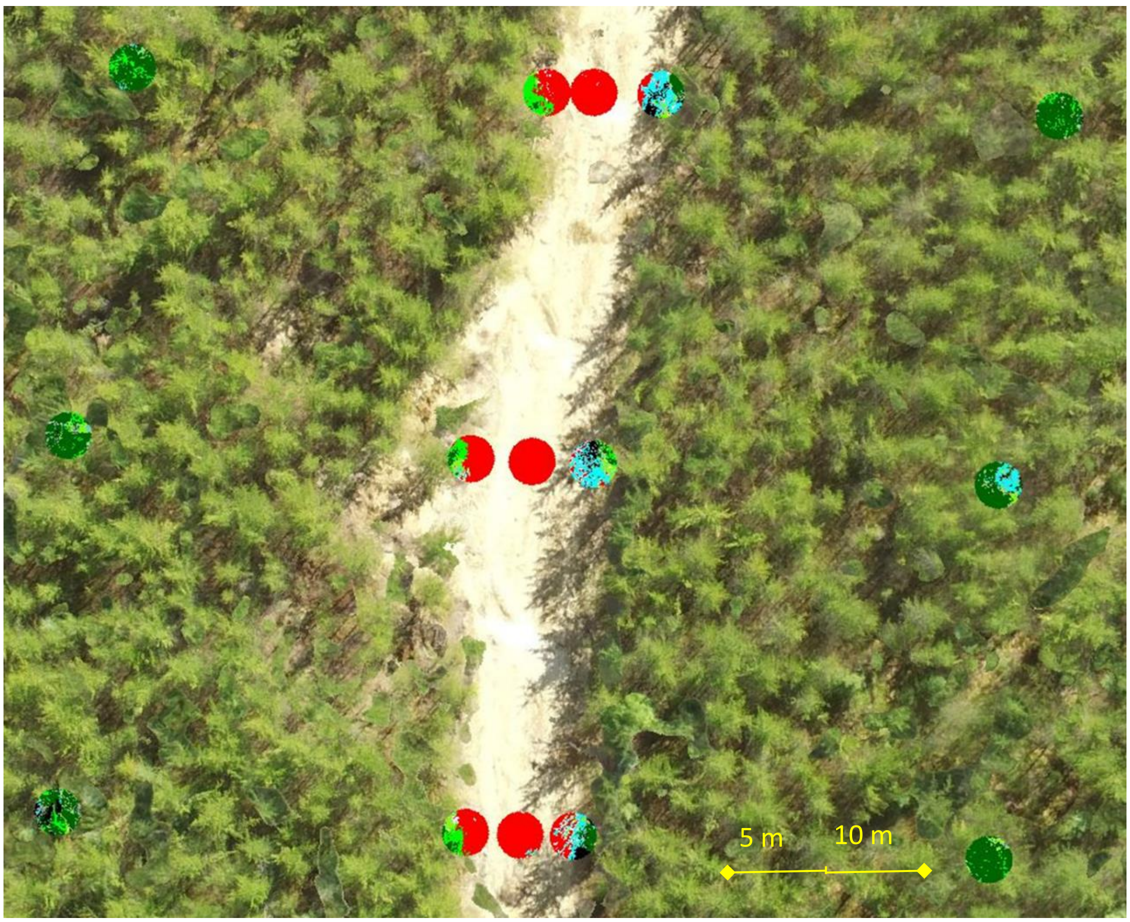

Figure 11. Example of the completed aerial samples that have been clipped from the classified image tile. The 15 plot locations match the measured ground truth plots, at ± 25 meters, road edge and road center for the three transects.

Pixel attributes

Once the subset sample plots were collected for each site, their pixel characteristics were retrieved from their attribute tables. These attributes were then sorted into their classes and colour coded for easy visual representation, like soil/road surface (white or red), conifer and other vegetation (shades of green), and CWD (browns and tan) which were later compared to the percent cover values from the ground-truth plots. Although this process is longer than using a recode step to condense the 100 classes down to the few classes like road, soil, vegetation, or water, it was done this way to have as much detailed information throughout the process, for as long as possible, as it might have been needed later.

The attribute table was then copied over into a spreadsheet with the number of pixels and classification. Pixels were summed according to their designated class and tallied for comparison. To assess the comparison of measurements for each method, the total vegetation cover from both the ground and aerial samples were matched, correlated, and tested. For this study, it was assumed that the ground plot cover measurements are "true" or the correct values, and the aerial cover values were tested against the ground measurements. To compare aerial measurements of vegetation cover to the ground cover, for this part of the study, the ground cover values were normalized to a maximum of 100% cover. This was done because the cameras can only see the top-most vegetation layer, regardless of what may be growing underneath.

Random Point Generation

This step aimed to prove that inaccessible or decommissioned road segments can continue to be monitored and assessed for regeneration by using off-the-shelf unmanned aerial vehicles technology, effectively and accurately. After running the validation of the aerial sampling plots over the ground-truth plots, it is possible to apply that same methodology to the rest of the Orthomosaic to measure vegetation regrowth on the road surfaces. For this part of the study, a random sampling method was chosen. ArcGIS software was used to create the points, with the "*Create Random Points*" tool. Since the objective was to measure vegetation on the road surfaces the road layer vector files were used as the constraining feature class, meaning any points created would fall along that vector. As this was a proof of concept, a minimal point intensity was used. Subsequently, if the number or distribution of points does not satisfy the statistical burden, the tool can be re-run quickly with new input values until they meet the requirements of the user. A new vector point shapefile was created as the center points for the orthomosaic sampling and overlaid onto the map (Figure 12).

The next step was to import these newly created points into the image classification software (ERDAS Imagine) to begin the random sample point, vegetation cover assessment. The road vector file was also brought in to illustrate how well the road surfaces in the classified images line up with the vector file (bottom of Figure 13). To expedite the cover measurements for the study area, sample points derived in ArcGIS were snapped to the closest position in the classified image that represents the equivalent road segment. Samples were created using the same methods as the ground truth comparisons. For each randomly generated point, a 5m^2 circle plot was drawn at approximate road center, where an AOI file is named, saved and a subset of the classified image is created (Figure 14).

The subset plot attribute tables were opened, and the pixel counts for both vegetation and non-vegetation were condensed into a table (Appendix 12), and other pertinent information was collated with the random sample points. The data file was then imported into R-Studio for statistical analysis and tested using a factorial ANOVA and a Tukey HSD post-hoc. The linear model was 'RandomSamplePoint2' = Percent vegetation ~ Ecosite + Road type, where percent vegetation was the dependent variable representing the measured ground cover, while ecosite and road type where independent and random factors. The post-hoc tested significance in cover between treatments.

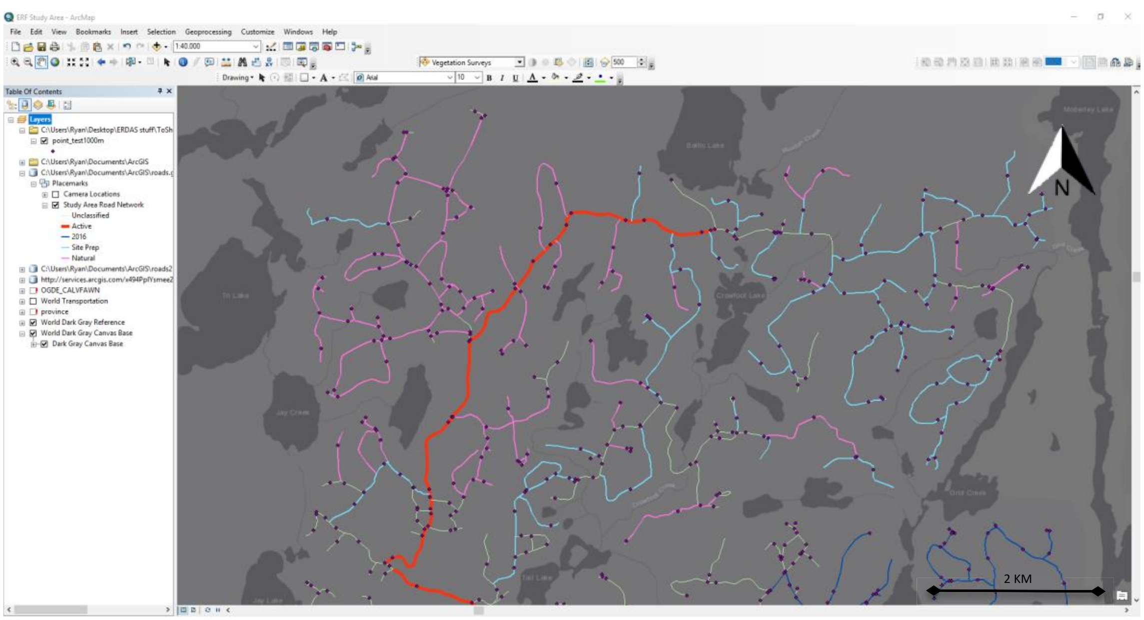

Figure 12. Random sampling points along the English River Forest study area road network, created using the 'Create Random Points' tool in ArcGIS.

Figure 13. Randomly generated points overlaid onto the mosaic raster files used to classify vegetation and road surfaces within the study area. An example of how the varying levels of geospatial data entry and inaccuracies can lead to shifts in alignment between vector files (white lines) and the roads (red) in orthorectified images.

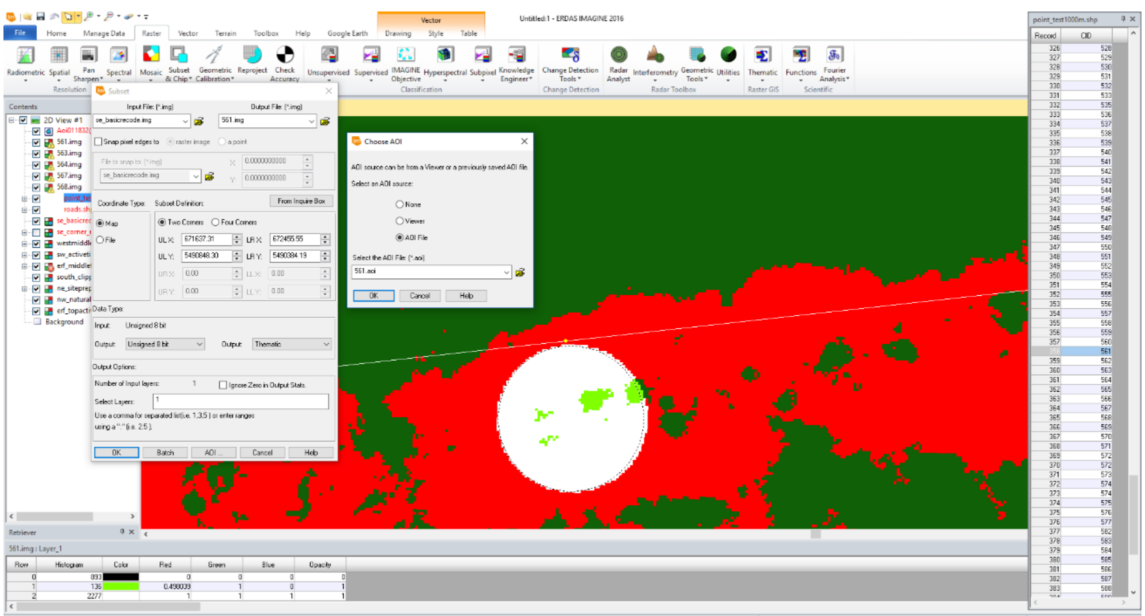

Figure 14. An example of the random point road cover sampling. The random point vector layer is brought into the contents window (left) with the attribute table opened (right) for point ID. The AOI is drawn onto the road center representing the generated random points. Subset image function (middle left) is used to create the sample plot, deriving percent cover values for that 5m^2 plot and recorded from the attribute table (bottom).

RESULTS

TRAIL CAMERA RESULTS

The first set of cameras were installed May 15th, 2017 and the last data collection was completed on October 3rd, 2017 for a total of 142 days. There was a total of 14,856 events, creating 30,838 photos and videos with 580 significant photo and video captures. This equals an average of 104.6 of events per day for the whole study area and an average of 217 files per day. Meaning there was an average of 4 significant captures between all cameras on site each day. 3.9 percent of the events (1.88 % of files) had significant captures meaning they were triggered by either an animal or a person.

Review of the photos showed that people (35 file captures) were only in the area in the late summer/early fall during hunting season and were clearly, actively hunting along the roadway. Since the files are date and time stamped, these show that the people were both legally and illegally hunting in the area, which will be discussed in the next section.

All other captures (545) were wildlife. The nine, identifiable species captured were 130 captures of *Alces alces (*moose*), Ursus americanus (*black bear*)* had 289 captures*,* 6 *Vulpes vulpes (*red fox*),* 67 *Lynx canadensis (*Canadian lynx*),* 36 captures of *Canis lupus (*Wolf*),* 3 captures of *Bonasa umbellus* (ruffed grouse*),* 1 *Sylvilagus floridanus (*eastern cottontail*),* 9 *Antigone canadensis (*sandhill crane*),* and 1 *Sciurus carolinensis (*red squirrel*).* 9 captures were unidentifiable. All of which is summarized and shown in Figure 15.

Table 1. Camera units associated with the number of events and the associated number files created from them. Captures are files where there were animals or people found within the photo or video excluding study personnel. Total events were calculated from May 15th to October 3rd, due to the time restrictions of this study. 14,856 events produced 30,838 files with 580 significant captures.

Camera	C5a	C5b	C6	C8	C11	C12	C15	C16	C17	C18	C19	C19b	C20a	C20b	Totals
Events	210	1943	840	2052	326	59	255	1223	176	2688	1019	458	1204	2403	**14856**
Files	420	4879	1679	4103	651	118	691	2446	332	5356	2037	915	2408	4803	**30838**
Captures	0	71	8	80	76	22	43	36	32	98	27	9	2	76	**580**

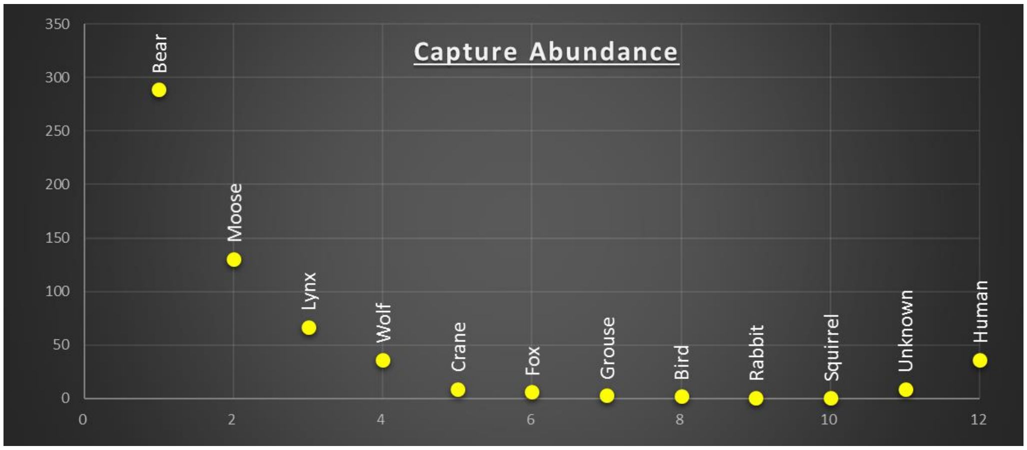

Figure 15. Species capture abundance for the English River Forest study area from May 15th to October 3rd, 2017. Bear sightings occurred in 289 captures, moose had 130, lynx 67, wollves had 36, sandhill cranes had 9, fox was 6, grouse had three captures, and both rabbit and squirrel were captured only once each. There were nine captures where the animal in question was unidentifiable, two where the subject was too close to the camera to be identified, but with the body design could be narrowed down to a bird species and lastly, there were 35 captures of people (not part of the study team).

Events were mapped according to their time of capture for each species to determine if there was a more active versus non-active time of day for that species. This can then inform the average speed of movement for each species. Figure 16 shows the time of day activity for each animal species with more than three separate capture events through the study period. Records for moose show a trend where they were most active between midnight and noon. Wolves were the only species that seemed to be active throughout the 24-hour period. The other three species, lynx, fox and bear had significant trends for that species. Lynx were only seen between midnight and noon, while fox were seen in the morning before 10 AM and bear were most active between 12 AM and 1 PM with only four out of 280 events between 2 and 7 PM over the study period. This was found to be different from the findings from Amstrup and Beecham (1976) which interestingly showed peak black bear activity between 8 AM and 9 PM. These results suggest that when applying a species speed to the density equation (Rowcliffe *et al.* 2008) other than wolf and moose, the speeds only apply to the active times, and not a full 24 hours (Rovero and Marshall 2009).

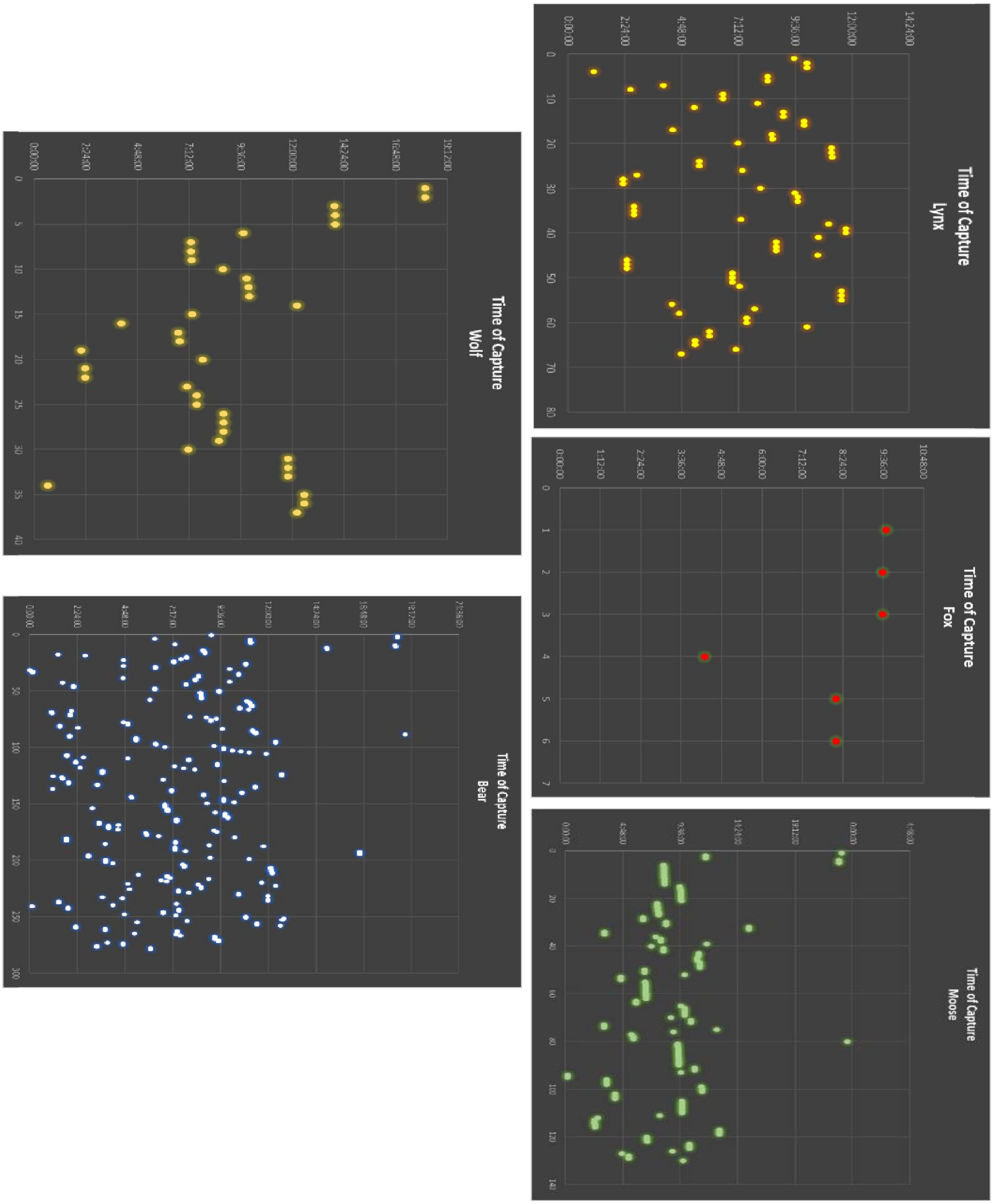

Figure 16. Time of day capture for the main species within the ERF study with multiple captures from top left to bottom right; Lynx, Fox, Moose, Wolf and Bear. This shows the active periods for each species. Lynx was seen to be most active from 12 AM to 12 PM, fox were seen between 3 AM to 10 AM, while moose were seen throughout the day but with most activity occurring between 12 AM and 3 PM. Wolf sightings occurred through the day, and black bear were found to be most active from 12 AM to 12 PM but with some activity was seen throughout the 24-hour period.

GROUND VEGETATION SAMPLING RESULTS

There was a total of 55 different plant species identified (Appendix 2). The most abundant species was *Pinus banksiana,* found in 108 plots. The next most abundant species were *Alnus viridis*, green alder, in 72 plots*, Betula papyrifera*, white birch, in 65 plots*,* and *Salix spp.* in more than 50*.* Rare species only found in one or two plots included cow vetch (*Vicia cracca*), starflower (*Trientalis borealis*), creeping snowberry (*Gaultheria hispidula*) and serviceberry (*Amelanchier spp.).* Most species found were considered typical boreal species (Legasy *et al.* 1995) with several species that are typically found in disturbed sites with exposed or shallow soils. While there was a relatively high species richness for the study area, the ERF study was lacking evenness or relative abundance; having high proportions of a select few species and minimal sample numbers of most species, Figure 17.

For sites 1 through 13, there were respectively, 11, 12, 18, 16, 16, 25, 18, 21, 18, 16, 22, 25 and finally 20 species found, Figure 18. This shows that the active road segments had the lowest number of species present while the highest diversity was found on naturally regenerating sites. Sites 5, 10, 3, 7 and 11 were the treated sites with some form of reclamation and had a wider range of plant species richness than the other road types, with no significant difference between either the active or natural treatments.

Figure 19 shows the break down of species present at each site, highlighting the trend associated with plant communities found on the different road treatments with the more distinct communities being sites 5, 6, and 10. The remaining ten sites were generally found to have the same plant communities.

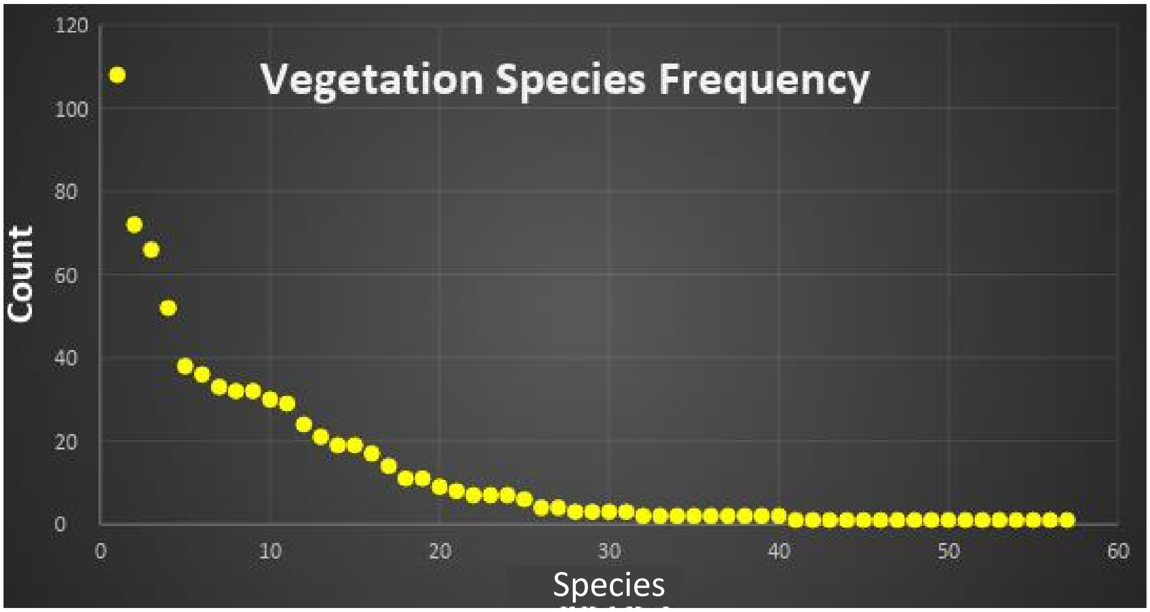

Figure 17. Vegetation counts for each species found within the study area. This shows common species like Jack pine (Pj) with 108 counts and green alder (Ag) at 72, compared to more rare species like vetch (Vtc) with only one incidence within the ERF study. See Appendix (Appendix 2) for species codes.

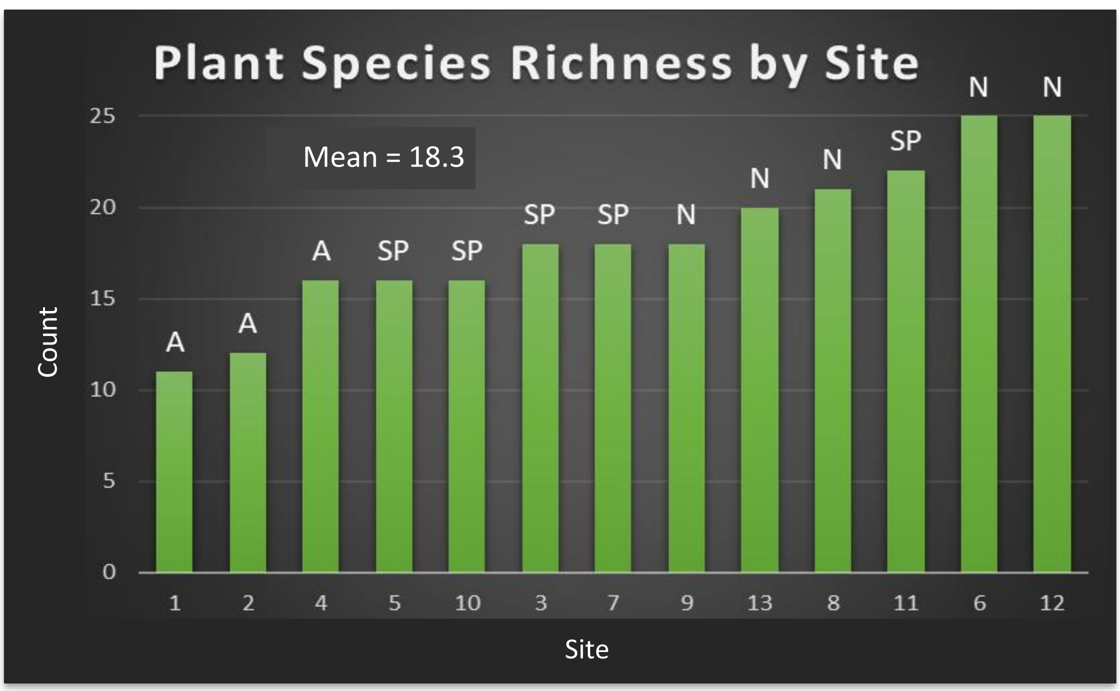

Figure 18. Ranked site (X-axis) specific vegetation species richness for the ERF study area. Road types are labelled: (A) Active, (SP) Site Prepped and (N) is Natural/Abandoned. The average number of species was calculated to be 18.3 across sites 1-13 with a standard deviation of 4.3 across all sites.

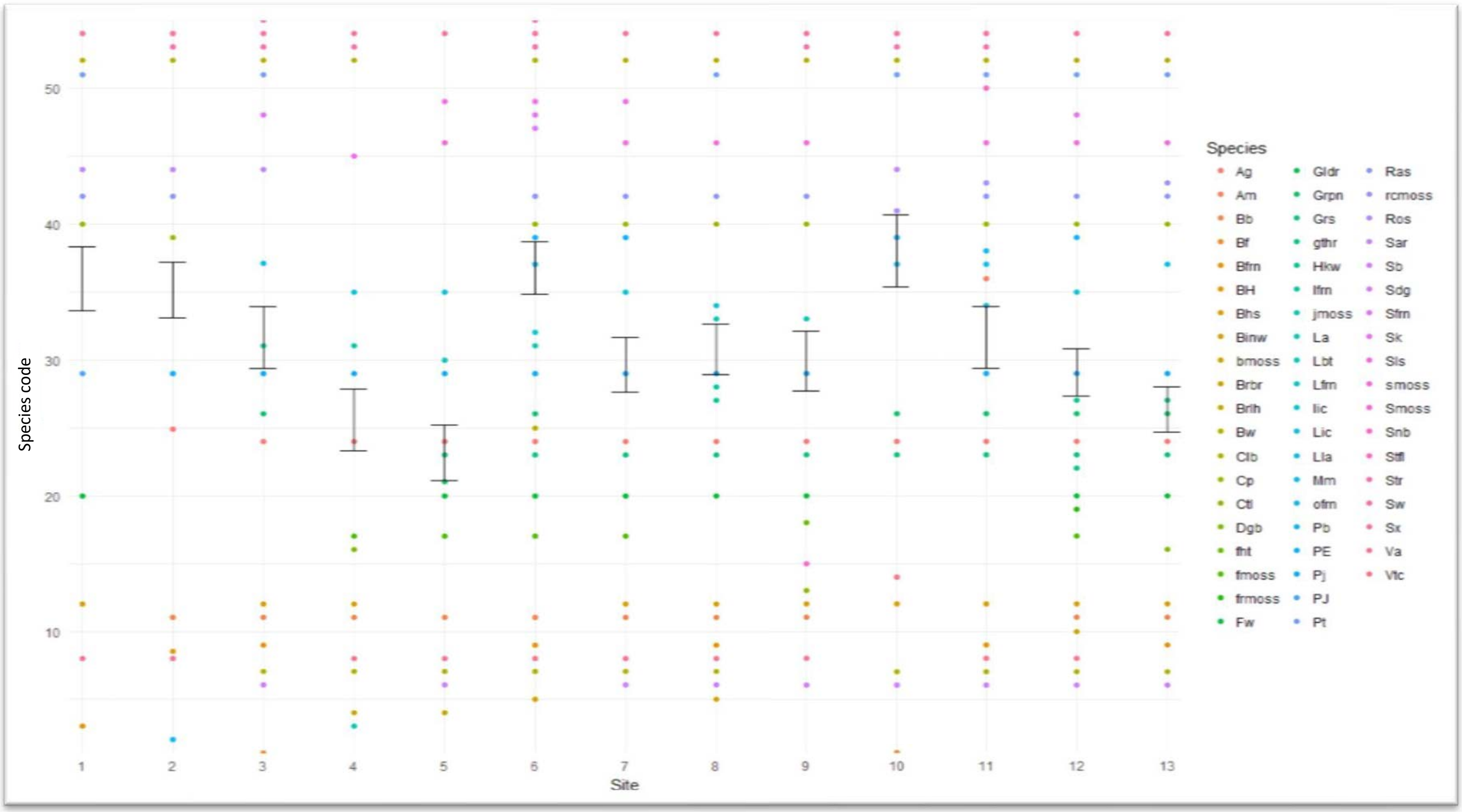

Figure 19. Site-specific vegetation species presence map with standard deviations between sites. The error bars show the relative difference in plant communities found between the 13 different sites. Species codes are listed in Appendix 2. A full report of species at each site can be found in Appendix 4 and Appendix 5.

SITE COVER

Ground cover varied widely throughout the ERF study area, from zero percent cover at *Site 3* up to 391%, multiple layer, cumulative ground cover at *Site 12*. Sites *One*, *Two* and *Three* were active sites. *Four*, *Five*, and *Twelve* were deactivated treatments. Sites *Six*, *Seven* and *Thirteen* were decommissioned. Finally, sites *Eight*, *Nine*, *Ten* and *Eleven* were abandoned road segments. Figure 20 shows the change in cover values across all sites and which sites are significantly different in cover percent. *Site 1,* an active road segment, had the lowest average percent cover at 0.68% and *Sites 11* (2.01%) *and 12* (1.87%) statistically had the highest average cover values of all 13 sites.

An ANOVA was run first on Cover versus Plot and Site and the interaction between Site and Plot in R-Studio. The resulting P-values were less than an alpha of 0.05, therefore *Road Type, Ecosite, Plot*, and *Road Type:Plot*, *Ecosite:Plot* interactions were found to have significant effects on measured ground vegetation cover values, with *Plot* explaining most of the variation within the model. Vegetation found at the ground center plot was less than the road edge and forest plots (Figure 22) and will be discussed more in depth in the next section. A Tukey HSD calculated the differences in means for the different sites with a 95% Confidence Interval to define which sites were significantly different from each other, shown as differing letters in Figure 20.

Figure 21 shows the measured cover values and the average cover values for each site along with the species richness. As seen below, plant species richness is not necessarily related to cover values, but more so to treatment type and site. This is highlighted by sites 6 and 7, where the average cover values are much lower than other treated sites, yet species richness is equal to or higher than their corresponding values. Similarly, site 12 has the same richness as site 6 (Figure 18), yet site 12 average cover was significantly higher.

Table 2. ANOVA testing significant factors of the measured percent cover values.

Cov1<-aov (Cover~Road type*Ecosite*Plot+Site*Plot, data=prj)

	Df	Sum Sq	Mean Sq	F value	Pr(>F)	
Road type	3	12.95	4.315	16.220	8.48e-09	***
Ecosite	3	10.29	3.431	12.896	2.79e-07	***
Plot	4	37.24	9.309	34.992	< 2e-16	***
Site	6	2.62	0.436	1.641	0.14265	
Road type:Plot	12	14.87	1.239	4.658	4.37e-06	***
Ecosite:Plot	12	9.11	0.760	2.855	0.00188	**
Plot:Site	23	9.54	0.415	1.558	0.06710	.
Residuals	110	29.26	0.266			

Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

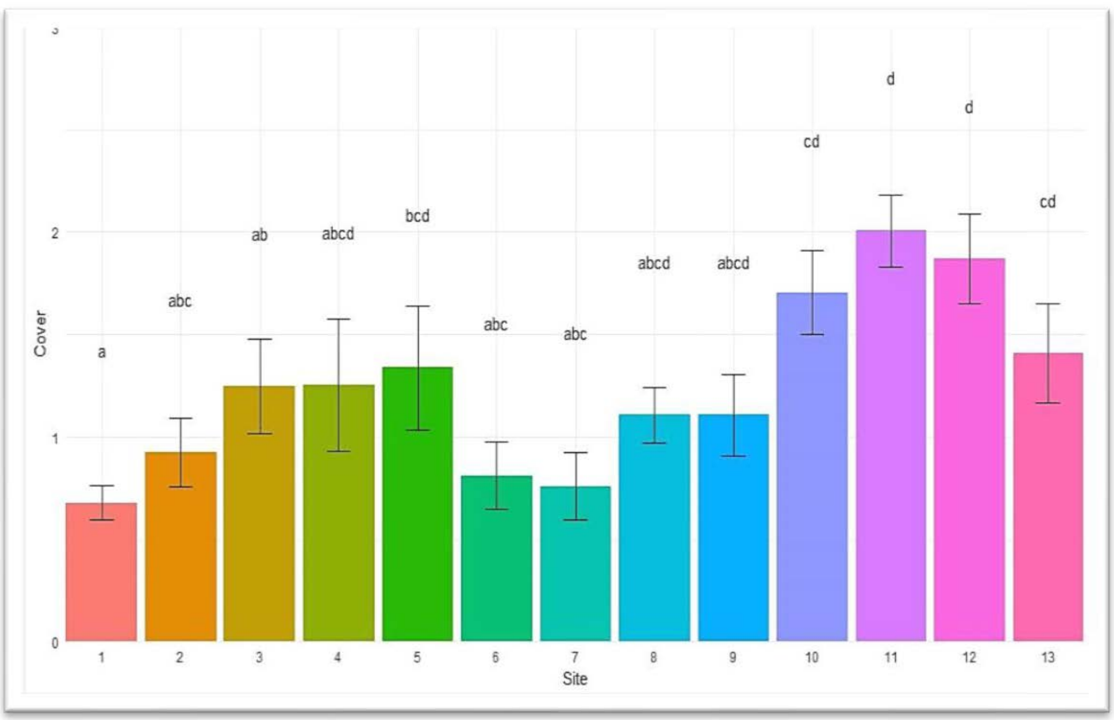

Figure 20. Vegetation cover values averaged over all ground truth plots per site, for sites 1-13. Each unique assigned letter value represents significantly different levels of cover. Produced in R-Studio using ground cover metrics.

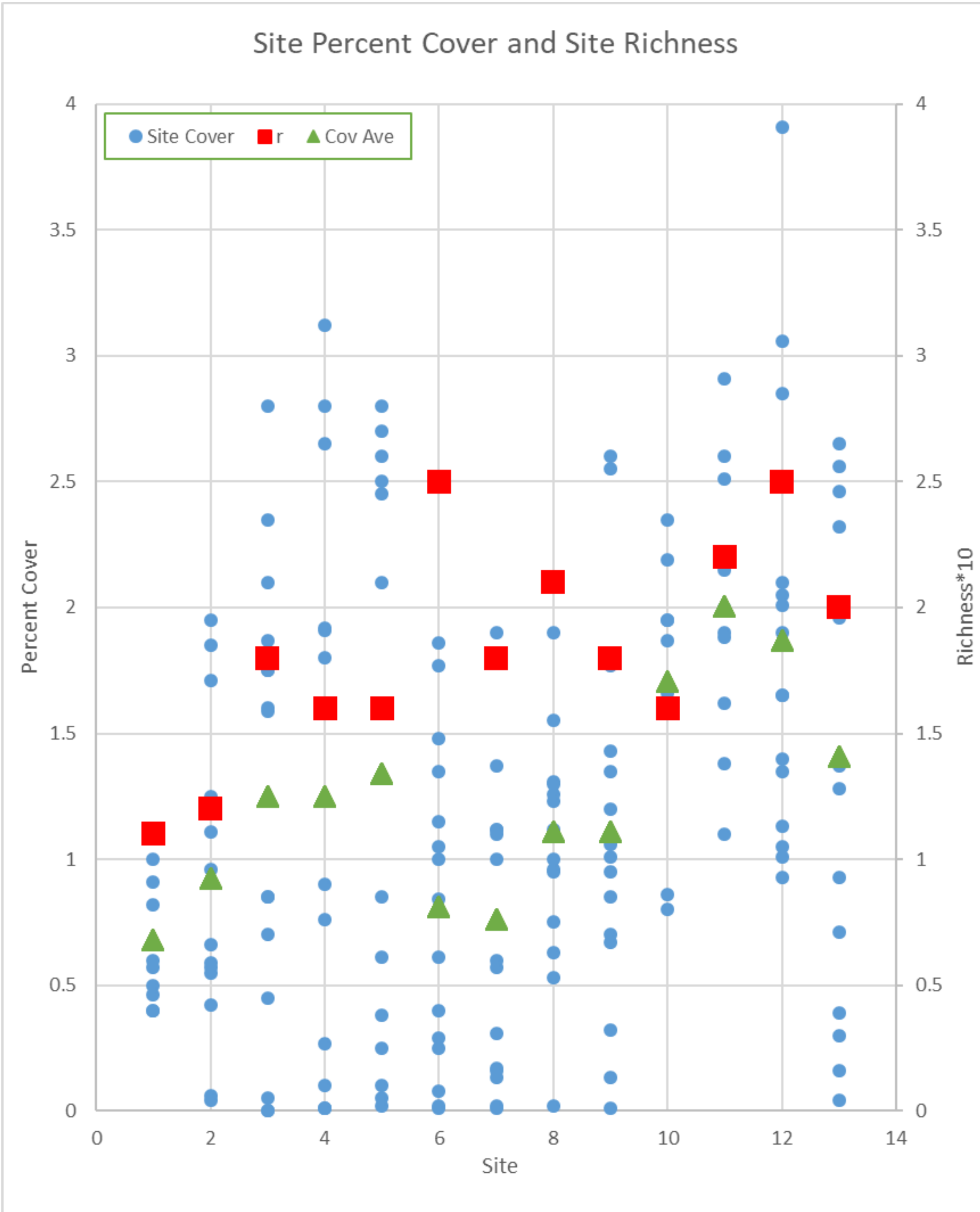

Figure 21. A comparison of site cover and species richness within each site. Site cover values are marked by the blue circles, with site cover average in the larger green circle. The red square represents the site species richness curve (Appendix 4) with cover values (divided by 10).

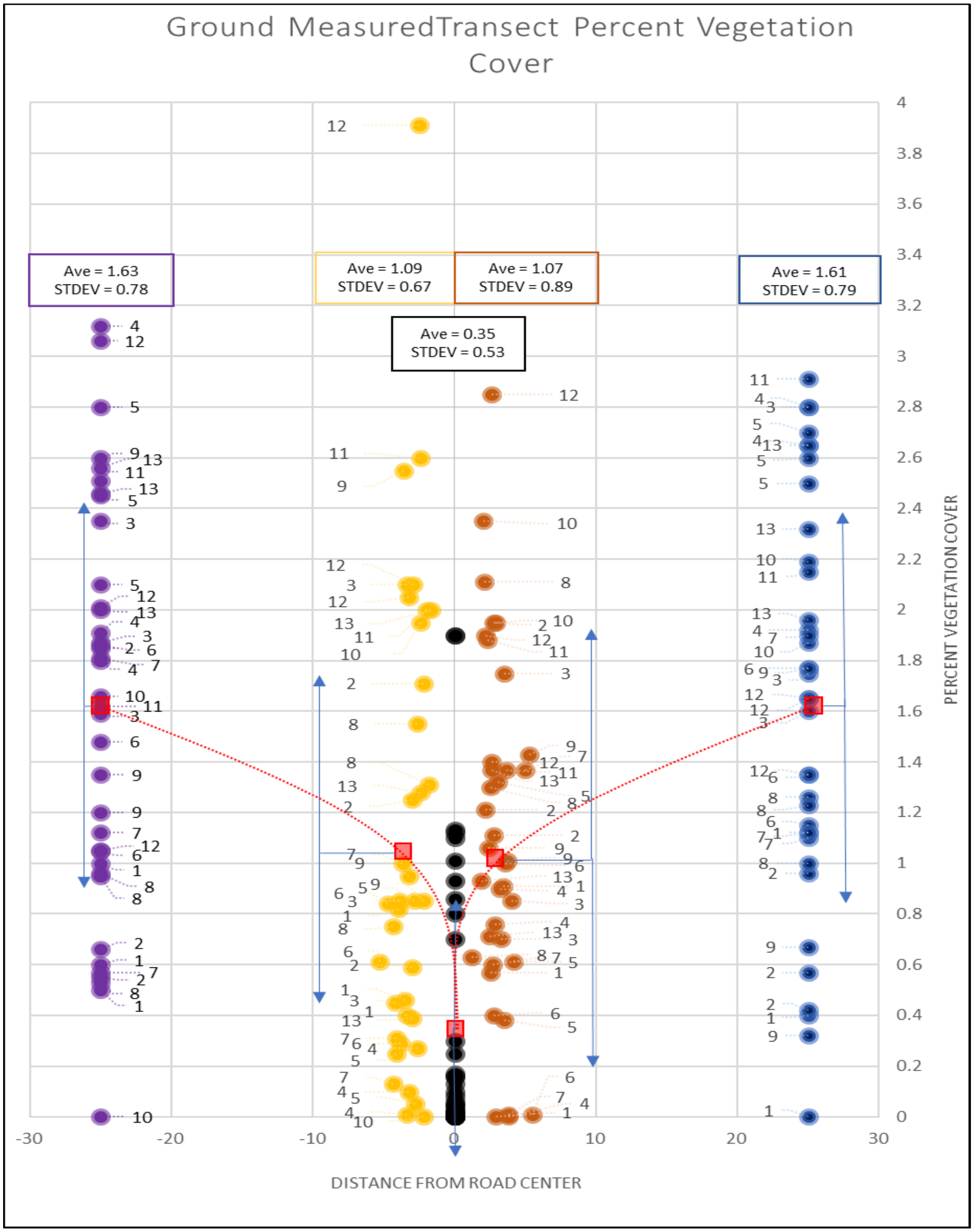

Figure 22. Summary of the ground truth measurements of Vegetation Cover along the length of the 50-meter transects (X-axis). Road center plots were done at 0 meters (black), road edge (light and dark orange) between ± 2 and 5 meters and forest plots at ± 25 meters (blue and purple). Plot points are labelled with site number with the vertical axis representing cumulative percent cover (0 to 400 %). The red squares show the average values for each plot distance; the red dashed line showing the increase of cover from road center outwards. The blue arrows show the standard deviation of cover for each distance.

VISUAL OCCLUSION TEST

The ANOVA done for the line of sight tests showed that the direction of orientation compared to the road had a significant effect on the amount of visibility, and Time Since Disturbance (TSD) had no effect. The ANOVA test was done with having two variables for *RoadvsSightline*, dependant on whether the line-of-sight ran parallel or perpendicular to the road and was statistically significant (close to zero P-value). In other words, measurements were taken looking along the road or looking perpendicular to the direction of the road into the surrounding forest. Because there were only two variables given, the significant P-value in the output below means that one is different from the other and looking at the data, it is evident that the roads having less vegetation maintain greater LOS than the surrounding forest cover. While *Site* had no significant effect on line-of-site with a P-value of 0. 45605, *Ecosite* again was a significant factor (0.00477) on the amount of vegetation and visual obstruction. The strongest predictor of obstruction was sighting direction, at close to zero. A Tukey HSD shows that *Ecosite 2* was significantly different from sites *1 and 3*, while 1, 3, and 4 remain relatively the same between sites for visual line of site (Appendix 11).

Table 3. ANOVA testing significant factors of visual occlusion or line-of-sight values from plot center.

VisAOV<-aov (Percent obstruction~Ecosite+Site+TSD+Road vs Sightline+Road forest)

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Ecosite	3	1.630	0.5433	5.110	0.00477 **
Site	9	0.959	0.1066	1.002	0.45605
RoadvsSightline	3	4.931	1.6438	15.460	1.26e-06 ***
Residuals	36	3.828	0.1063		

Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

AERIAL IMAGE SURVEY RESULTS

The total number of photos of the road network was 4,302 images resulting in an orthomosaic image with a ground sampling distance or pixel resolution of 5 cm x 5 cm. 104 training plots were used to compare the ground truth calibration plots where the values assessed in situ were compared against the aerial imagery collected with the UAV.

Figure 23 shows the correlation between the ground vegetation cover (green line) and the two different aerial training sets from ERDAS (orange dotted line) and eCognition (blue hashed line). All three measurements were of plot vegetation cover across sites 1, 3, 4, 5, 6, 7, 9 and 10. Ground measurements were normalized to a maximum 100% cover to run comparisons of aerial measurements. The Pearson correlation coefficient between *Ground* measurements and aerial measurements from *eCognition* was 0.573 and *ERDAS* had a stronger relation at 0.632 while the relationship between the two aerial measurements was a lot closer with a coefficient of 0.882. The somewhat weak correlation between the ground and aerial cover values will be discussed further in the next section.

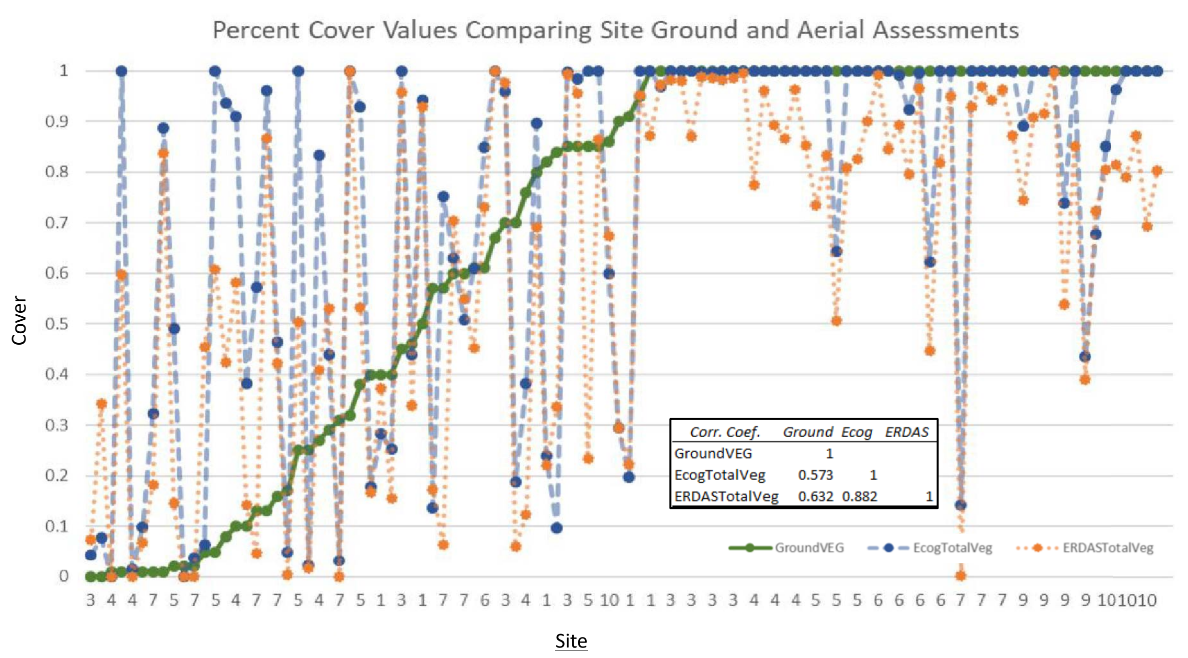

Figure 23. A linear comparison of the three different assessments of vegetation cover (ground, ERDAS Aerial, and eCogAerial). The correlation coefficient (CC) values are shown in the graph, with the correlation between ground assessed values between eCog Aerial cover was 0.573, Ground and ERDAS was 0.632, while the comparison between the two aerial measurements resulted in a CC of 0.882.

ACCURACY ASSESSMENTS

Table 4 shows the summary of the accuracy assessments that were done on the test images; North, Middle and South. The full assessment (Appendix 7 through Appendix 10) show the differences levels of accuracy between the two programs, the two methods and the changes in *Producer's Accuracy* between the classes. The highest AA attained was using the 'Area-based' error matrix with an overall accuracy of 97.86% on the North orthomosaic image, with two classes, and using the ERDAS software. The Standard Error of Area (SEA) was just over half a hectare (0.533 ha) for both classes (vegetation and non-vegetation) and a 95% Confidence Interval (CI) of ± 1.044 ha (Appendix 9). The highest scoring eCognition overall accuracy was 95.72%, the third highest score, with the 'Area-based' error matrix approach on the 2-Class South orthomosaic image. The SEA reported at 1.095 ha while the 95% CI was larger than the top overall scoring image at ± 2.147 ha (Appendix 10). The lowest overall AA was the South ortho image classified using ERDAS with five classes and using the 'Pixel-based' error matrix, scoring a 52.55 overall accuracy. When this same dataset is run through the 'Area-based' approached, the overall accuracy increased to 69.24% overall accuracy. Interestingly, when the classified images were tested using the 'Area-based' AA approach, all but one image were improved or scored the same overall accuracy level compared to the traditional 'Pixel-based' approach. The only AA to decrease was the North Poly-Class image and went from an overall score of 64.45% using pixels, down to 64.18% with the area approach (Table 4). This latter decrease gives evidence of the overestimation in class accuracies from the 'pixel' matrix accuracy assessment, and the 'area' matrix has corrected for this.

After comparing the AA reports for the Poly Class images, top of Table 4, the eCognition software performed much better using the object classification algorithms compared to the pixel-based image classification that ERDAS uses. For each image, both the 'Pixel' and 'Area' based matrices had significantly higher overall scores for the eCognition classification. Moreover, comparing the two programs with the same AA method for poly-class orthomosaic images derived from unmanned aerial vehicles, there was between 17.80 – 34.40% increase in overall accuracy from ERDAS to eCognition.

Two out of the three classified images using only the two classes had higher overall accuracies using ERDAS's pixel-based classification over eCognition's object classification using the 'Area-based' error matrix (Bottom of Table 4) but only by a relatively small margin. Overall AA scores ranged from 86.67% to 97.86%, which was a significant improvement over the Poly class scores. The ERDAS 2-Class scores for *Middle* (95.97%) and *North* (97.86) were 2.52% and 5.86% higher than the same images done eCognition. For the *South* tile using the 'Area' matrix, eCognition had an overall score of 95.72%, a 3.41% increase over the ERDAS iteration. However, if considering only the 'Pixel' error matrices for the 2 Class images, eCognition scored higher on the South (+6.67%), Middle (+2.67%) and North (+0.80%) images. In summary for 2-Class orthomosaics from UAV images, this study found that ERDAS was marginally more accurate by an average of 1.66% over eCognition when using the ‘Area-based’ error matrix for accuracy assessments.

Table 4. Summary of Accuracy Assessments on 3 UAV image ortho-tiles (N, NW and S) using two classification programs. The table compares ERDAS Imagine 2016 (orange) versus eCognition Developer 64 9.0 (blue) and the difference in accuracy in the righthand columns. Multiple classes include Conifer, Deciduous, Road surface, Lichen, and Shadow versus the two Class method of classifying Vegetation and Non-Vegetation. There is also a comparison of Accuracy Assessments by Pixel-based (light blue) versus Area-based (light green) approaches.

ERDAS South Poly Class			ERDAS Middle Poly Class			ERDAS North Poly Class		
Pixel-based Error Matrix -		52.55	Pixel-based Error Matrix -		57.20	Pixel-based Error Matrix -		64.45
Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy
Conifer	83.64	45.54	Conifer	57.26	73.96	Conifer	64.86	96.00
Vegetation	19.61	25.00	Vegetation	66.67	40.38	Vegetation	58.54	37.50
Lichen	54.00	49.09	Road	45.83	91.67	Lichen	62.50	34.88
Road	62.00	88.57	Soil	68.00	51.52	Road	80.00	100.00
Soil	40.82	83.33	Water	14.29	40.00	Soil	60.87	42.42
Area-based Error Matrix		69.24	Area-based Error Matrix		59.69	Area-based Error Matrix		64.18
Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	83.64	83.55	Conifer	57.26	79.93	Conifer	64.86	96.85
Vegetation	19.61	14.27	Vegetation	66.67	39.01	Vegetation	58.54	32.46
Lichen	54.00	41.74	Road	45.83	91.64	Lichen	62.50	27.81
Road	62.00	86.03	Soil	68.00	67.09	Road	80.00	100.00
Soil	40.82	86.13	Water	14.29	3.24	Soil	60.87	34.92
eCog South Poly			eCog Middle Poly Class			eCog North Poly Class		
Pixel-based Error Matrix -		85.14	Pixel-based Error Matrix -		75.00	Pixel-based Error Matrix -		78.40
Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy
Conifer	94.44	81.93	Conifer	97.14	81.60	Conifer	95.92	92.16
Road	84.00	93.33	Road	98.04	78.13	Vegetation	80.77	85.71
Lichen	73.58	82.98	Lichen	48.00	61.54	Lichen	0.00	0.00
			Soil	32.00	57.14	Road	66.00	67.35
						Soil	54.00	55.10
Area-based Error Matrix		90.92	Area-based Error Matrix		94.09	Area-based Error Matrix		86.36
Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	94.44	95.74	Conifer	97.14	99.32	Conifer	95.92	99.28
Road	84.00	77.63	Road	98.04	87.44	Road	80.77	91.36
Lichen	73.58	69.85	Lichen	48.00	18.04	Shadow	0.00	0.00
			Soil	32.00	54.63	Lichen	66.00	47.28
						Soil	54.00	32.77

ERDAS South 2 Classes			ERDAS Middle 2 Classes			ERDAS North 2 Classes		
Pixel-based Error Matrix -		86.67	Pixel-based Error Matrix -		88.00	Pixel-based Error Matrix -		91.20
Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy
Conifer	97.00	85.09	Conifer	98.00	85.96	Conifer	100.00	87.21
Road	66.00	91.67	Road	68.00	94.44	Road	78.00	100.00
Area-based Error Matrix		92.31	Area-based Error Matrix		95.97	Area-based Error Matrix		97.86
Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	97.00	94.12	Conifer	98.00	97.69	Conifer	100.00	97.68
Road	66.00	79.68	Road	68.00	71.16	Road	78.00	100.00
eCog South 2 Classes			eCog Middle 2 Classes			eCog North 2 Classes		
Pixel-based Error Matrix -		93.33	Pixel-based Error Matrix -		90.67	Pixel-based Error Matrix -		92.00
Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy	Accuracy	User's Accuracy	Producer's Accuracy
Conifer	96.00	94.12	Conifer	94.00	92.16	Conifer	92.00	95.83
Road	88.00	91.67	Road	84.00	87.50	Road	92.00	85.19
Area-based Error Matrix		95.72	Area-based Error Matrix		93.45	Area-based Error Matrix		92.00
Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy	Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	96.00	99.54	Conifer	94.00	99.01	Conifer	92.00	98.44
Road	88.00	44.64	Road	84.00	45.10	Road	92.00	67.63

RANDOM POINT SAMPLING

Results of the random point survey ended with 74 samples across different levels of road reclamation classes (*Abandoned* [12], *Active* [15], *Deactivated* [20], *Decommissioned* [20], *Site Prepped* [1], and *Skidder Trail* [6]). As there was only 1 *Site Prepped* segment sample and there was little photo coverage of this segment type. Therefore, for the statistical tests it was removed due to the lack of replicates. The road types versus the amount of cover are summarized in Figure 24 with the proportion of visible road surface, and the total vegetation cover being inverses of each other. The orange dotted curve is labelled with the sampled road treatments, and the green curve shows the vegetation percent cover measured from the orthomosaics. Of the random samples, the highest percent cover reached on an active road segment was 23% while the lowest cover value of the non-permanent segments was 96.8%. Much variability was seen in the samples from *decommissioned,* for example, having far-ranging cover values from 0 – 100%.

The random point samples ANOVA tested for significant effects on vegetation, RSPCover. Road type (P-value 1.4e-06) was the most significant predictor for cover, followed by ecosite (P-value 0.00575). In the RSPCover2 ANOVA model, TSD was found not to be a determinant predictor of cover (P-value 0.43379) most likely since TSD only ranged between 0 to 10 years, and there has not been enough time for the vegetation to grow.

Table 5. ANOVA test on the random sampling photos of the different road treatments.

RSPCover2<-aov(Percent vegetation~Ecosite+Road type)

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Ecosite	3	1.333	0.4444	4.572	0.00575 **
Road type	4	4.049	1.0122	10.415	1.4e-06 ***
Residuals	65	6.317	0.0972		

Signif. codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

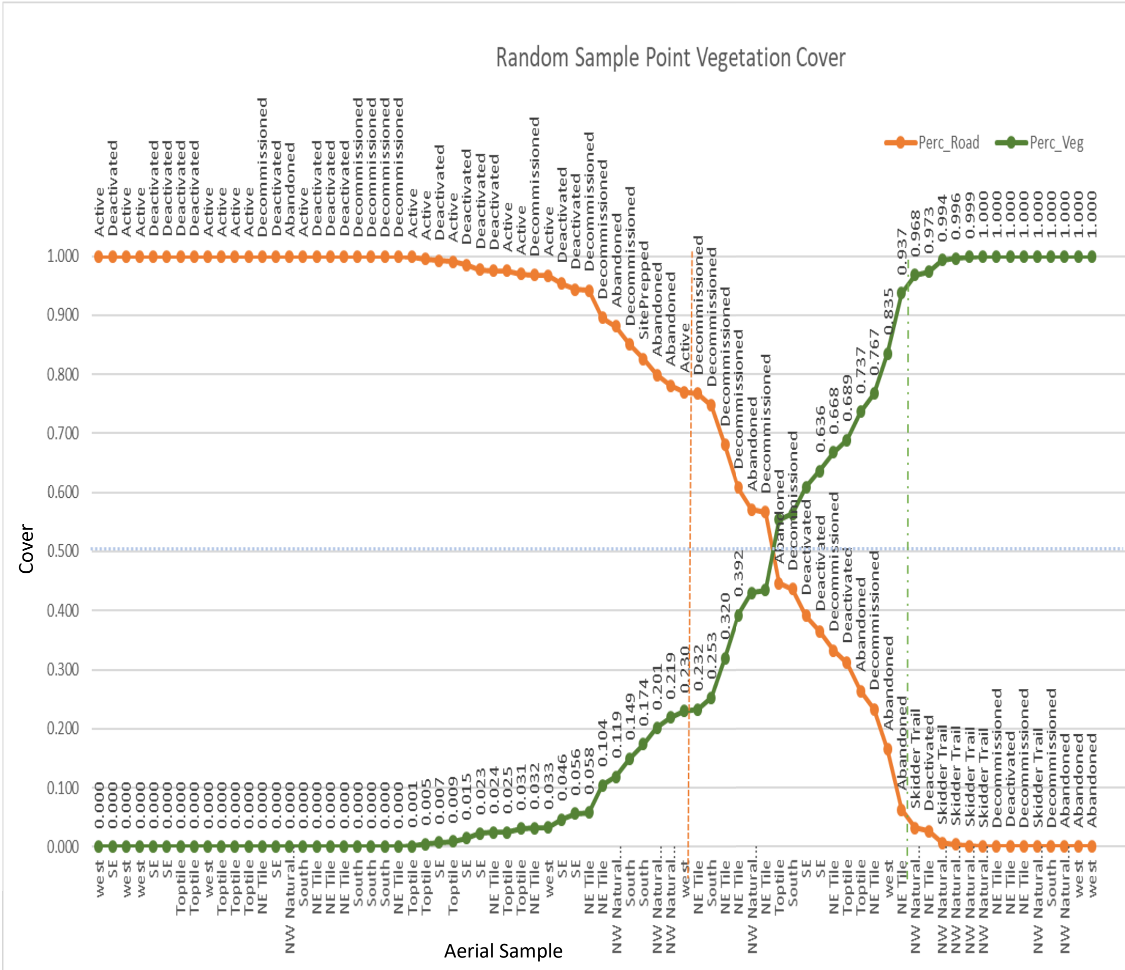

Figure 24. Summary of the random point sampling along linear features found within the Orth mosaic. The two values are the complimentary sum of either the percent of vegetation (green) or non-vegetation (orange) cover. Road cover values are equal to 1 – (the value of vegetation cover, directly opposite). The tile locations are shown along the horizontal axis, and road type sampled is shown at each point along the orange curve. The vertical orange line depicts where the last active road sample falls along the curve, and the green dashed line separates the skid trails or non-permanent roadway samples.

Road type was found to be a significant (P-value of 1.4e-06) influence on cover values. A Tukey HSD revealed which of the five levels had a significant difference. In Table 6, the P-values reveal that active roads were significantly different from all other road segment types (abandoned, deactivated decommissioned and skid trail segments). It also showed that skid trail segments, which were all almost completely grown in were different from deactivated segments, and at an alpha of 0.1, were different from decommissioned roads. Decommissioned and deactivated roads only altered in the relative methodology, resulting in practically the same cover values, and at this point in time, there was no significant difference between abandoned and the deactivated/decommissioned roads.

Table 6. Summary of the Post-hoc test on the random photo sampling ANOVA. Significant interactions are underlined below.

```
> TukeyHSD(RSPCover2)
  Tukey multiple comparisons of means
    95% family-wise confidence level
Fit: aov(formula = Percent Vegetation ~ Road type)
```

Road type comparison	difference	lower	upper	p adj
Active-Abandoned	-0.48218193	-0.820955695	-0.1434082	0.0015342
Deactivated-Abandoned	-0.11849692	-0.437895887	0.2009020	0.8353479
Decommissioned-Abandoned	-0.05066986	-0.370068824	0.2687291	0.9916819
Skidder Trail-Abandoned	0.35127693	-0.086078119	0.7886320	0.1735426
Deactivated-Active	0.36368501	0.064914637	0.6624554	0.0094036
Decommissioned-Active	0.43151207	0.132741700	0.7302824	0.0012624
Skidder Trail-Active	0.83345886	0.410933744	1.2559840	0.0000059
Decommissioned-Deactivated	0.06782706	-0.208780556	0.3444347	0.9583943
Skidder Trail-Deactivated	0.46977385	0.062618458	0.8769292	0.0157613
Skidder Trail-Decommissioned	0.40194678	-0.005208604	0.8091022	0.0546706

DISCUSSION

This research attempted to answer several hypotheses. First, road treatment was found to have a significant difference in vegetation cover levels. However, the study area was deactivated at different stages, with some areas being completed ten years ago while the latest was completed only one year before data collection. It was difficult to differentiate long-term effects between reclamation methods due to the high variance in vegetation cover measurements across those treatments. Given more time, it is expected that there would be more significant differences in cover amounts and types found across treatments. Plant species richness was greatest on abandoned roads which have had the most time post-disturbance to recover and establish, with the active road segments having the lowest plant richness. *Road Type, Ecosite, Plot*, and *Road Type:Plot*, *Ecosite:Plot* interactions were found to have significant effects on measured vegetation cover values across the study. Sites 1 – 4, 6 and 7 were significantly different from sites 5, and 10 – 13 with site 1 and sites 11 and 12 being the most distinct (Figure 20).

Second, the hypothesis that unmanned aerial vehicle derived orthomosaic imagery could be used for accurate measurement of vegetation cover values was proven true by the high accuracy assessment scores (Table 4). High accuracies then led into the primary objective; testing if those orthomosaics could be used for accurate, random photo sampling. Several studies have used UAV data in various ways, but none were found that explored the abilities of very-high resolution orthomosaic data. The “RSPCover2” ANOVA model showed *Road Type* was the main predictor of vegetation in the random aerial sampling tests. *Active* roads with at most 23% cover, and in-block, *skid trails* with no less than 96% cover, the difference likely stemmed from the use of aggregates, compaction and disturbance to the soil. No difference between *abandoned* roads and *deactivated*, *decommissioned* or *skid trails* was likely due to the variability in vegetation cover on these segments and because these areas were only recently reclaimed and have not had enough time to develop fully (Lloyd *et al.* 2013). The other primary influence of the random aerial-sample vegetation cover was found again to be ecosite; the environment surrounding the roads logically influences the recovery of the reclaimed areas. If the surrounding landscape is highly productive, with mature forest, then the ecosystem will fill in that gap more quickly than a dry, barren and unproductive site like those of sites 4 and 5.

Unfortunately, not all wildlife monitoring objectives were met during this study. It was not possible to complete populations estimates for the identified species. Difficulties in accessing the target areas and limited field time prevented the development of a mathematical model with adequate degrees of freedom. The hypothesis that camera traps could be used to monitor and observe wildlife species and their natural behaviours on deactivated road segments was found to be possible. The camera traps were able to show which animal species were using the road networks, when they were using the roads and provided data on behavioural patterns for future inferences on species interactions, Figure 16.

This project was ambitious and consisted of a lot of different parts and objectives. Through the extent of this study, it was hoped that the camera traps would capture evidence of a caribou in the area but unfortunately, that did not happen. The intention was to try to understand the effects of road reclamation on vegetation in the area and then tie that to the local wildlife activity. In this section, the results of the study will be reviewed, and discuss explanations of those findings.

WILDLIFE POPULATIONS

If wolf predation on moose and caribou is a concern, previous efforts have shown that although wolves can and do use linear features when it suits them, the majority of travel is done away from the linear features and in forested areas (Dussault *et al.* 2012; Faille *et al.* 2010; Latham *et al.* 2011; Pinard *et al.* 2012). The behaviours observed (easy trotting, relaxed stance, no active scenting or direct staring/tracking) found on the camera traps would suggest that the observed wolves were not actively hunting but only transiting from one part of their territory to another. This study cannot conclusively argue that the wolves found herein were exclusively using the roads, as the camera study design did not allow for off-road placements. The results from this test can only prove that wolves and bear, predators of woodland caribou and moose, were present on the deactivated roads within 20 years of deactivation treatment.

Of the camera traps deployed in this study, only the cameras that were on deactivated roads [6] captured wolves. This may be a result of human avoidance behaviour or a hunting strategy where the deactivated areas tended to have more cover than active roads and less human activity, similar to what Whittington *et al.* (2005) and Dickie *et al.* (2017) found. However, because most of the roadways in the ERF study had young vegetation growth and remained relatively open to travel even with having been treated, they provided forage for species like moose and remained an easy travel corridor. Combined with the LOS visual occlusion test proving that the linear feature remains, and with results similar to Hall *et al.* (2016), roadways provide easy forage, visual advantages, and allow scent trails of prey to be followed more easily by predator species.

If future studies require population estimates, they should consider camera installation either just at snowmelt, or the year prior. That way the time-consuming work is already completed and all that is required is to visit each camera and turn them on at the start of the study. Unfortunately, complications in camera establishment prevented the establishment of a statistically viable model. It took several field outings to the research area to install all the cameras on site, with unintended camera resets occurring and changing settings on some units, theft, equipment failure, and a relatively short study period made any statistical inferences weak at best. There simply were not enough degrees of freedom to create a density or population model. While species densities were not attainable, species presence/absence observation was still possible. Using the image data of each location, assumptions on the use of different roads were made by correlating which species were seen on different road treatments.

Political Nature/Social Demand of Roads

Theoretically, before the roads were built, the lack of access was never a public concern. However, after the roads were built and that access was created, some people feel that because it is public or Crown land, that it is their right to have that road remain open so that they can continue to access their favourite hunting or fishing spot. Mihell and Hunt (2011) have shown that when the question of the value of remoteness, versus access, is asked, the general response is: "Remoteness is good and road deactivation is a good way to maintain it, but do not remove 'my' road; remove a different road." This represents a classic not-in-my-backyard opinion. When then the subject of maintenance costs arises, the general response is defensive with an "it is a forestry/crown road, so they should pay for it" attitude when it is only a select few people using the road for recreation. Should the industrial sector or the province then be forced to pay for the maintenance and remain liable for a road after they are done in that area for the foreseeable future? Evidence given by Mihell and Hunt (2011) and Hunt and Hupf (2014) showed that people would continue to use closed roads, creating their own trails around obstructions if need be. Evidence of which was observed during this study. At several deactivation points in the English River Forest, trails had been cut into the surrounding vegetation by public users when the road had been clearly marked with signage that the road was closed. This makes it difficult to meet restoration goals for those areas that are disturbed by repetitive vehicle access and lowering the chances that sensitive species like caribou would return. This argues in favour of more intensive road removal and reclamation, especially near the beginning of access points to protect the area.

The camera traps also caught illegal hunting activities within the English River Forest. The cameras recorded hunters, at night, driving along the still active road segments in their vehicles while using spotlights to search for animals, on more than one occasion (Figure 25). It is assumed that these individuals were unaware of the camera traps in the area at the time, as later in the study those same cameras were stolen. This puts into question then, what other regulations were or will be broken? The challenge here is that the public wishes for these access roads to remain open so that they can utilize the public resources, yet, when it is thought that no one is looking, there are some that exploit that access. In cases like this, should the road network be entirely removed as part of a much bigger (and much more expensive) land and wildlife management strategy regardless of the public's desire for the opposite? One strategy that foresters could use would be to levy the status of the endangered caribou's natural range in this area against the public desire for the roads to remain open; adapting their forest management plan procedures so that the road removal would be completed in the interest of protecting a threatened species.

The ERF study area was designated as a preferred moose aquatic feeding area (Lawson 2009a; Lawson 2009b; OMNR 2010a), which means that it is of high interest to hunters. The roads created to access the timber now act as main access routes for hunters seeking those moose. This was affirmed by our camera traps capturing hunting activities. The high interest in the English River Forest as a recreational hunting area means increased vehicle activity. This increase in activity reduces the likelihood of successful reclamation and prolonging the time the linear features remain on the land base. If the licensee assumes that because they have made some level of effort to deactivate the road, it will regenerate regardless of the intensity of that treatment, this study proves that it is not the case. However, because of the value to hunters, the road will most likely remain open in one form or another from continued use. Again, if there is a concern for moose populations in Ontario, intensive road removals in crucial areas such as aquatic feeding areas like this could be argued for by the licensee to protect wildlife populations.

Figure 25. Evidence of illegal hunting activities in the ERF. Note the time stamps of 2:21 AM and 23:50 (11:50 PM) on two separate dates and different vehicles. The accompanying video files show the use of spotlights and active searching for wildlife.

EFFECTIVENESS OF ROAD TREATMENT

OMNR's (2014) caribou management guide suggested that cases where roadbeds were constructed on or from sandy soils "may encourage lichen regeneration" and could remain in the effort to promote lichen growth. There was no evidence found during this project to support that theory. The south access section of the ERF study area access roads consisted solely of sand-based aggregate. This section of roads, deactivated roughly ten years prior, had no evidence of lichen bed expansion or encroachment onto the road. In an undergraduate research study linked with this project (Dorland 2018), vegetation was assessed on the deactivation scallops of aggregate/soils from the roadbed and road edge. The excavated soils taken from the road edge that had lichen present showed no signs of outward spread from these 'seed' mounds placed onto the sand-based road. This evidence suggests that either the management recommendations in the Woodland Caribou management guides need improvements or, reclamation strategies in Ontario need further research into their suggested best practices.

Anecdotal evidence found that in areas where roads ran along harvested areas showed a trend in seedling establishment along the road edges and ditches. Figure 26 highlights this observation. The presence of ditching may have acted as a seed trap when the site was aerially seeded or seeds from leave trees within the neighbouring cut blocks. The ditching is lower than the surrounding road and forest soils, acting as a gravity well, collecting seeds and other organic debris. There was also standing water which could have trapped windblown seeds and help to provide moisture for seedling development. In storm events, it is possible that any seeds that previously landed on the road surface would have been washed away, ending up in the ditch alongside the roads. This would further delay re-vegetation of the road surface and add to the edge effect along the corridors. From this observation alone, a suggestion for more parallel site preparation or reclamation strategies (ex. claw ripping along the road) would likely increase vegetation establishment on the road surface, acting like the ditches, collecting seeds and moisture. There is a potential cumulative effect for this as well. If the road edges establish first and silvicultural strategies rely on seeds released from either seed trees left on site or from aerial release programs, the established plants could act as a net/wall, catching any seeds blown in laterally. This could be a meaningful avenue of future research, studying the long-term effects of road edge establishment of vegetation on overall road reclamation success.

Figure 26. Example of vegetation establishment along road edges. Aerial images were collected in late May 2017 at 120 meters, and the scalloping is visible along the road surface as the lighter and darker swirls that make up the aggregate mounds and troughs.

Small, localized disturbances which cause a patch-like disturbance on the landscape increase the species diversity and richness where soils were disturbed (Jonsson and Esseen 1990). This supports the scalloping method when deactivating or decommissioning roads in the boreal forest, but it needs to be done effectively to initiate regeneration. Simon *et al.* (2011) showed that when windthrow creates pit-like features from the root mats lifting, those low-lying depressions see a decrease in tree seedling survival. In the ERF site, scallops were most often taken from just inside of the road edge and deposited back onto the middle of the road. This works as a short-term deterrent to vehicle traffic, but within a few years, these pits and mounds, if not covered in vegetation, eroded away and were no longer an impedance. Luce (1997) comparing road ripping treatment depths, argued that shallow ripping was effective in plant establishment, but only for the first year. The south access road network, which was entirely sand was a prime example of this. Rain, wind, animals and recently all-terrain vehicles have levelled off the road, and it has become relatively ‘drivable’ with little hindrance or vegetation growing on the entire length of the road segment. Signs of erosion were found beginning to erode the obstructions on the northern section of the study area with forest rejuvenation just beginning. However, the roads remain drivable to ATVs, especially in the spring and summer leaving it at risk of damage to the little vegetation that is growing there. Anecdotal evidence (Figure 27) shows an example of subversion around the road deactivation where it was maintained by the public to access the central areas within the ERF study area. The trench was supplemented by large boulders on the rear side of the excavation to help preserve its structure. Large rocks and coarse woody debris were placed on each side of the trench to try and prevent people from going around. Beyond this, there were signs of damage to vegetation from topping and ATV trails that has eroded reclamation scallops. Winter camera data not part of this study also showed on the area's central camera placed behind several trenches and deactivations that there was still recreational use of these old roads.

Figure 27. An example of road closure/deactivation (left) and the cut trail subverting it (center and right). The deactivation trench was put in place attempting to prevent access beyond this point, into the central area.

Sites, where there has been a relatively intense disturbance to the forest soils like in road reclamation, tend to have "young" soils and must go through multiple vegetative stages such as forbs, grass, shrubs and then finally to a treed state which can take hundreds of years (Macdonald *et al*. 2012; Walker 2012). If the goal is to return decommissioned roads to forested land in a reasonable timeframe, decommissioning or reclamation strategies should be designed to enable rapid soil and vegetation development. For woody-shrub regeneration on rangeland, time since last fire was found to be the main factor in shrub cover of the site (Moffet 2009). A longer time since disturbance leads to more established and extensive root systems and increased stem densities. Grant *et al*. (2011) found that both short-term CWD and canopy cover had no significant effects on the growth of low vegetation cover on reclaimed roads. This makes sense because over the initial short growth periods (1-5 years) there tend to be nutrient flushes in the soils (Walker 2012) providing the pioneer plants with enough nutrients to establish themselves and CWD has not yet begun to decay. Merely ensuring that there is CWD placed onto a reclaimed road surface does not guarantee increased vegetation growth either.

CAMERA TRAPS AND VEGETATION

The issue of seasonality and the limited study length was acknowledged and considered. Rain events can affect the efficacy of the camera traps, with moisture condensing on the camera lenses or the PIRs, interfering with animal observations. The wind was also a factor that caused significant interference and false triggers. Because there were set delays between triggers, an animal may have passed in front of the sensor, but it may not have been counted because the camera had just recently triggered by moving vegetation.

Field tests beforehand in similar conditions revealed that placing cameras in areas of high vegetation density proved ineffective. Roads densely covered in alder or willow resulted in visibility being practically zero in any direction, and the cameras did not work. First, there was no solid mounting structure to attach cameras. Second, zero visibility meant that the focal point of the camera was beyond where it could sense an animal passing by, meaning it would not detect anything moving in front of it. The vegetation was just too dense and blocked the view of the camera. Since alder and willow have a high sail effect in the wind, even a small breeze would fire the camera, creating high numbers of false triggers. The third reason for not actively maintaining a camera trap in these areas was that for us to establish a site, we would have to create a path through the vegetation that was not there before. To later retrieve that data, we would have to retrace the same path, further impacting the site and potentially skewing the data. This would defeat the objective of viewing natural behaviours based on the effects of the deactivation methods; disturbing the site by removing some vegetation or creating pathways could potentially alter those behaviours. For those reasons, it was decided not to pursue installing cameras on road sites densely vegetated in shrubs.

After the study was completed, the cameras were left to run over the winter to collect more information about the seasonal use and to test the endurance limits of the equipment for future work. The same reasoning for not creating new paths presented another challenge in how to approach the study during winter months. Winter means snow and deep snow settling onto the roads creates a resistance to the use of roads compared to surrounding forest (Bllomfield 1980; Cumming and Beange 1987; Johnson *et al.* 2004; Stardom 1975). If we were to return to the cameras during the winter, we would be creating artificial pathways making it easier for animals to move through their habitat and again, skew their behaviours.

One challenge that we ran into with the trail cameras was the there was the potential to miss wildlife moving past the camera location. Either because the animal passed outside of the triggering zone, or vegetation caused a false trigger which then 'locked' the camera from taking any more pictures for the designated 1-minute wait time, or having a too narrow field of view. Monitoring success could be improved by having multiple cameras at a single location facing different directions, two cameras placed a set distance apart and facing each other, or by elevating the camera well above the shrub growth found on the site and angled downwards. There are semi-permanent tripods that can be staked on site that cameras can be mounted to or, there are bolt attachments that screw into a tree which then can be used to mount a camera high above ground level.

There is also the option of purchasing and mounting solar panel kits for each camera. This would allow for semi-permanent or permanent installation of the camera traps on site replacing the need for the 8 AA batteries per device and could also power an external storage device such as a raspberry pie computer. The limitation of the 32 GB memory card was the limiting factor in how long the camera can run without having to return to collect the data, as we found throughout this study; the card would fill in about a month while the batteries could have lasted for about four.

Another potential research and development project would be to develop a way of collecting the data from the cameras remotely in an area that has no cellular service. One potential solution would be to use a raspberry pie computer on the camera, and one attached to a drone. Then fly the drone to each camera location, wirelessly connect the two raspberry pies upon arrival, transfer the data from the camera to the drone, then fly the drone back to the access point. This way there would be minimal disturbance to the vegetation and reclamation efforts, little noise disturbance to nearby animals from driving to each camera location, and during the flights to and from each trail camera, the onboard drone camera can photograph the road during its flight.

Although a total of 55 different plant species were found, the evenness of those species is heavily skewed, Figure 17. This same diversity curve has been found in other work (Euskirchen *et al.* 2000), which helps confirm that our results were typical for the boreal setting. This is important to report in reclaimed areas because this shows no significant changes to the ecosystem structure and productivity (Zhang *et al.* 2012) and that no new species were introduced into the area leading to altered equilibrium states (Walker 2012). Contrasting shade tolerance between species, with some species being shade tolerant and others that are not have been shown to also increase site productivity (Zhang *et al.* 2012). This would suggest that planting a mixture of commercial tree species on the roadways and cut blocks could increase the overall site value for the next harvest cycle while improving ecosystem function at the same time. This has direct implications on silvicultural strategies and road reclamation methods. Considering the four most abundant species found in the study were either shade intolerant (Pj, Bw, and Sx) or only slightly tolerant (Ag), we can say that the area was a relatively young site. In this case, species richness alone was found not to be representative of overall site health and production. Considering the top two sites for richness (Figure 18), *sites 12 and 6*, which have the same number of species, their percent cover values (Figure 20) are significantly different with *site 12* having 200% total cumulative ground cover and *site 6* only having roughly 80%. When comparing site richness to average percent cover (Figure 21) a Pearson's correlation results in a coefficient of 0.398, meaning that site richness has a weak relationship with site cover in this case. This supports the conclusion that another factor is mostly responsible for the measured vegetation cover.

Species composition variability may have been dependant on the surrounding forest ecosite; for example, (Figure 19) *site 6* was a low-lying wetland area with a large rock outcropping providing diverse growing conditions in a relatively small area, compared to *site 5* where conditions were 100% sand parent material across the whole site with no variation. *Site 10* was on an abandoned high road with ditching, surrounded by a mature mixed-wood stand with no less than 70% cover across all plots while active s*ite 3,* one kilometre away and part of that same forest stand was statistically no different in species richness but had plots with no cover at all. This again, argues for a more ecosite-based approach to road reclamation.

Initial regeneration of disturbed areas by vegetation is done at different rates by different cohorts of colonizing species (Grant *et al.* 2011; Jonsson and Esseen 1998; Legasy *et al.* 1995; Walker 2012). Hautala *et al.* (2008) showed that with severe disturbances where the whole plant is removed and the belowground tissues are destroyed, those areas see a recovery of bryophytes and graminoids but lack the growth of vascular species. In the English River Forest, similar occurrences were observed on the south access and the northeast access road segments where there was little to no mineral soil, and any hummus and organic layers were sparse and kept away from the road surface. The south access roads where the deactivation treatment excavated some amount of humus from alongside the road and brought it onto the sandy road surface was there any significant vegetation growth (Dorland 2018). By bringing soil and humus onto the roadbed, there is a chance that the excavated material would act as a seed bank and that vascular plants might have active rhizomes that could begin to colonize the surrounding area. It is unclear whether the excavation points were set based on a directive to mitigate 'site disturbance' or if it was a limitation of the excavators reach. By not excavating organic material from further away into the surrounding forest soils and vegetation, recovery may have been hampered and not completed to the maximum potential.

Organic matter build-up on the forest floor may be a possible alternative to alleviate nutrient limitation and moisture retention in these areas. In nutrient-poor sites such as the sandy portion of the study area in this study (sites 4 and 5), plants tend to produce litter with nutrient concentrations lower than that of plants in higher quality sites (Chapin *et al.* 1986) retaining as much as possible and saving them from having to expend more energy to find and replace those lost nutrients. In higher quality sites where nutrients are more available, plants may expend less energy to collect nutrients and may not be as stringent in reallocating nutrients within its structure. However, nutrient-poor sites like sites 4 and 5, have minimal organic matter present, so an introduction of even a low amount of nutrients is still a comparatively high addition to that site. Additively, the low moisture regime of that area would also reduce the rate of decay and the release of those nutrients into the soil, which could reduce nutrient leeching over time. For these reasons, it would be beneficial to retain a supply of organics materials on site during operations specifically for reclamation on roads and nutrient-poor sites.

RECLAMATION ASSESSMENT

The results of the photo sampling of the road network (Figure 24) show that reclamation efforts to fully re-vegetate the site have not succeeded as of yet. The post-Hoc in Table 6 detailed that the only significant differences between the road treatments were between *active* roads and all other treatment types. All but 11 deactivated/reclaimed road samples had less than 50%

vegetation cover, with most falling below the 20% cover mark. If the threshold for success is a vegetation cover greater than 50%, then there was only a 35% success rate. If that threshold was higher to be more in line with what is required for the free-to-grow status of forest cut blocks, then that the success rate drops to about 12% of road segments in the ERF study area.

One logical reason for the lack of success in the reclamation of this study is the lack of a concise definition. The language in the planning manual shows that there is little concern about the effects that access, and operational roads have on the surrounding ecosystem. “The roads will degenerate over time” is a statement designed to minimize the economic costs to the industry, with the language of the definition giving no weight to the long-term consequences. There is no mention of returning the area to a natural benchmark, or of concern for restoring ecosystem processes. The unclear term leaves the goal of reclamation unclear, and with no real consequences of failure, there is little incentive for continued monitoring of the treated roads to ensure successful regeneration of those areas.

The reclaimed roads listed as ‘removed’ were used to make disturbance impact assessments. Since it is not physically possible to audit the entire forest license in a reasonable timeframe, it is logical to use the GIS layers to calculate forest impacts. Therefore, when reporting on current road densities, forest managers would use the available road GIS attributes to calculate densities, buffers and impact assessments. This study showed that the English River forest alone had 210 of 385 road segments listed as decommissioned which put the reported active road density at 0.59km/km^2. The project results show that roads that have been treated and listed as decommissioned have not entirely been removed from the forest, still acting as linear features on the landscape. Therefore, the reported road density (0.59km/km^2) is incorrect, along with the associated disturbance impacts. Understandably, not all decommissioned roads are going to be reclaimed due to cost restrictions, operational strategies, and public inputs, but then they should still be considered a landscape feature or anthropogenic disturbance until they have either grown in or have been entirely removed and confirmed as such.

For cases such as this, where the aim is limiting the long-term disturbance caused by roads, more intensive efforts or changes in strategies will be required. One possible solution would be to implement a program like free-to-grow monitoring for harvested forest areas or incorporate operational roads into the free-to-grow program alongside the harvest areas. This would help check if reclamation efforts were successful, if segments need more attention or have utterly failed and readdressed. Remote sensing through satellite would most likely be preferable, but low-resolution imagery limits monitoring effectiveness, and very-high-resolution satellite data is expensive to obtain for large, extensive areas. Until the costs are reduced, monitoring road reclamation success will have to continue to be done on foot or via manned or unmanned aircraft.

Significant microsite factors included soil temperatures, with pit temperatures much lower than undisturbed plots, potentially lowering growing days and slowing organic matter (OM) decomposition in those areas, contributing to OM build up. Results of Simon *et al.* (2011) mirror this study, suggesting that the scalloping method of deactivation may be inhibiting site regeneration and prolonging the linear effects upon the landscape. By using this deactivation method, they created pits like those Simon et al. (2011) studied to block vehicle traffic and are potentially creating microsites along the road where trees will not or cannot grow.

While the results of this study found similar conditions to Hall *et al.* (2016) with their suggestions to adapt road-shed strategies, a broader consideration and method adaption to surrounding forest conditions adjacent to roadways could further improve reclamation success. In wetter, more nutrient-rich sites with higher levels of organic matter, it may be more appropriate to use a less intensive reclamation strategy like scraping or scalloping with roll-back or pulling organics from the surrounding forest areas. In drier areas, nutrient-poor sites, or roads more heavily constructed, an intensive approach would be required to reach the same levels of success. Ripping of the road surface to below the compacted layer or to the parent material would provide better purchase for seeds, allow root structures to reach nutrients and water. Rolling back organic materials either from beside the road, further inside the cut block or from an outside source should be added as well to improve reclamation best practices (Hall *et al.* 2016; Sanchez *et al.* 2009) to help improve the probability of site regeneration. This would provide nutrients, shelter for seedlings, moisture and can also house vegetative reproductive roots or seed banks adding to the total vegetation cover. The rolling back of organics would most likely have been more useful for those southern, sandy sections of the ERF study area than the scalloping method that was used. So much so, that the mechanical site prep probably could have been foregone in place of moving organic materials on to the roadway.

Anecdotal evidence from this study combined with the significance of road 'ecotype' (Table 2) suggests that a specified forest type deactivation approach may improve the rate of success in road reclamation. Forest managers could refer to the FRI information to modify road reclamation intensity and incorporate the surrounding forest stand type before the actual work would take place.

The trail camera data shows that merely pulling culverts and trenching does not prevent recreational access to closed areas. Moreover, once an ATV gets beyond the obstructions, the damage to vegetation can delay site rejuvenation by years, with the noises from human activity possibly deterring sensitive species from those areas. A more intensive decommissioning effort at strategic entry points would have improved the reclamation success of this study. If greater lengths of the road segments (ex. 100m, 500 m, 1 km, the extent of line of sight) where the roadbeds were entirely removed and debris backfilled, it is probable that there would have been little to no continued use by the public. It would be difficult to traverse with any vehicle and deter almost anyone except the most stubborn to continue past that point. Then beyond those entrance points, there could be less intensive decommissioning efforts as there would be less human disturbance damaging the vegetation growing there. A soil ripper that reached two or three feet into the road surface could be run down the length of the road in the further sections of the road network. The challenge of using a claw ripper in Ontario is that a lot of the roads are built on exposed bedrock outcrops. The troughs would act as aeolian seed traps, the roots would be able to penetrate below into the de-compacted hardpan, and those areas could be completed relatively cheaply (an out and back pass of the machine) to compensate for the costlier entry points.

USE OF UNMANNED AERIAL VEHICLES

The biggest advantage of using unmanned aerial vehicles was that they could reach further than one can on foot or road vehicle, and their use is significantly less expensive than traditional aircraft. With the ability to image road segments that are completely inaccessible by vehicle, we can now see what the conditions are beyond and ensure that the rest of that segment is regenerating. If for example, only a small portion of the road is grown in, preventing access, how do we know what the conditions are like 200 meters, or 500 meters or a kilometer beyond that point? An aerial perspective allows for a broader understanding of what's occurring on the landscape. The second advantage was that the data collected has a very high resolution and accurate (Goodbody *et al.* 2017). For localized, focused studies like this one, the ability to accurately identify individual plants was a significant advantage when studying localized effects.

Third, the positional systems allow for repeat measurements over the same track, as many times as needed and over any timespan, permitting change detection over time. We can return to the same locations in 1 year or 20, re-fly those same tracks and be able to measure the change of vegetation cover compared to today. The advancement in controls for drones has also made applying them in the field accessible as tools for anyone. With minimal training, any staff can use unmanned aerial vehicles to collect accurate and relevant data, making their job easier, safer, and more productive.

Accuracy Assessments

In the comparison by Meinel *et al.* (2001) between ERDAS and eCognition, both programs returned overall accuracies of 89.6%, but the parameters were not equal. To say that these programs were equally effective is unfair. This study used the same imagery for the two programs and directly compared them, using the two different assessments. Huang and Ni (2008) also showed the performance difference with a pixel-based accuracy of 75.67%, with a Kappa of 0.7025 and their object-based accuracies resulting in 83.27% and a 0.7792 Kappa. This mirrors the results of this study, with highly detailed imagery, object-based image classification not only performs better but was more accurate as well. For the multiple class tests, there was a minimum increase in accuracy of 13.95% for automated image classification of orthomosaic images from ERDAS to eCognition. For the images divided into two classes, Vegetation and Non-Vegetation, ERDAS performed better having higher accuracies than the same images with multiple classes.

One caveat to this, images classified in ERDAS were done individually; each ortho tile was run through the same unsupervised classification processes. While class values were the same between runs, it was time-consuming. Where in eCognition, the rule set was developed then applied to the rest of the images without modification. This then leaves room for further refinement in the classification process for eCognition used for this study. If like in ERDAS, each tile was classified on its own in eCognition, with a dedicated process tree, then it is expected that the accuracies would further increase, but also increase the processing hours required to develop the final product. In real-world applications, there is always the balancing of limited resources, time being one of them. If the final product is only marginally improved by expending more time and effort, is it worth it? If a classified image done in one hour gets relatively the same results as one at 5 hours, then economically, the former may be the best option.

Classifying image features into more than two classes, the AA results suggest that eCognition is the better, more accurate option. Whereas with only two classes, ERDAS performed better than eCognition after using the adjusted 'Area-based' AA approach. Moreover, when we review the SEA and 95% CI values from those matrices (Appendix 9 and Appendix 10), we see that the average Standard Error and the 95% Confidence interval is 59% larger in the eCognition classifications (Table 7). The hypothesis that eCognition was the 'better' classification program then was only partially correct. When classifying complex images, with multiple classes eCognition performs at a higher accuracy level than ERDAS, but when using only two classes, ERDAS Imagine seems to be the better program to use.

Table 7. Summary of the 2 Class Standard Error of Area (SEA) and 95% Confidence Interval (CI) in hectares for the South (S), Middle (M) and North (N) images. The SEA and CI values apply to both Vegetation and Non-Vegetation classes within each image. This table is derived from the full Accuracy Reports found in Appendix 9 and Appendix 10.

ERDAS	S	M	N	**Average**
SEA	0.722	0.874	0.533	**0.709**
95%CI	1.415	1.712	1.044	**1.390**
eCog	S	M	N	
SEA	1.095	0.802	1.709	**1.202**
95%CI	2.147	1.572	3.349	**2.356**

Linear Features Mapping

Looking at strictly road presence as a factor of disturbance on the land base, it and a 500-meter buffer have a disturbance factor and counts against the maximum disturbance threshold (OMNR 1993; 2009). Secondly, there is a maximum allowed road density that is designed to limit habitat fragmentation and loss of forested land cover (OMNR 2010a). Once the maximum road density is reached, then if a new road needs to be built, an old road must be removed from the network. The question then is, when does a road no longer count as a road? Does simply pulling the culvert and trenching the entrance and thereby deactivating the road make it no longer a road? Alternatively, by ripping or destroying the road surface and roadbed making it no longer drivable with no vegetation, does that reclassify it into something other than a road? These definitions are still lacking and need to be better outlined by policy and best practices in the future if road reclamation programs are going to be successful. Just because a road has undergone reclamation treatment does not mean it is removed from the landscape. In areas like the English River forest where road density needs to be limited to minimize linear feature disturbance, it is vital to know when a reclaimed road is no longer a road and no longer a factor on the landscape.

There were different factors during this study that may have contributed to variation in the data. Due to the nature of field work, time was a limited resource. Decisions had to be made to either reduce the number of sample plots and increase the intensity/accuracy of data collection or reduce the amount of time spent at each plot/site and maintain a higher sample intensity over the study area. This same challenge also affected the other two field data collection methods with the trail cameras and aerial mapping. Increasing the number of camera trap locations meant an increase in costs and revisit times between those cameras would increase, with the risk of the memory cards becoming full leading to missed camera days. As it was, the cameras were almost at capacity with the current return times. If there were to be increased flight missions, those work hours would then be taken away from either collecting the camera trap memory cards or the vegetation data collection. Ideally, more camera traps should have been used, on more locations, giving improved capture data, increased the number of camera days, reducing the number of missed captures and implications from them could have been stronger. However, limited time and funding prevented this.

It was assumed that the ground measurements for vegetation cover were the correct or *true* values and that the aerial value for cover was the measurement being tested. This assumption relies heavily on the skill of the ground crew to accurately gauge the percent cover, as there was no way to measure the level of the technician's accuracy (Luscier *et al.* 2006). Field data was collected by crews of two with roughly five years of field experience each, and when assessing vegetation cover both members would give a value, and the average between those was taken. If there were any significantly large differences in the stated cover, both members would re-measure, and another evaluation was given. Most often, the cover values between both members were the same or within (±) 5%. This was deemed an acceptable level of variation for ground measurements. This method was used because of the need to collect as much information as possible in a limited amount of time. Ideally, two separate crews would assess each plot for cover and record all four values in a closed test, then run a statistical analysis to determine if the technicians had any biases; having a consistent over or underestimation of cover percentage and then weight the cover values accordingly. Had there been more time, another step which may have improved actual cover value measurements would have been to break down each plot into smaller sub-sections that could be measured more precisely. This would have increased the confidence of the ground cover values and probably have been more accurate to the true values found in those locations.

A second possible source of error was user bias in the classification accuracy assessments, when deriving the rule sets or assigning classes the user potentially misidentifies the pixels/objects. In this study, all processes were completed by the author, so if there were any user-bias in the classification process, it would be consistent between all images and all steps of classification. During all classification steps, accuracy was a constant concern, and all reasonable precautions were taken to monitor and account for user bias.

Shadows are always a major challenge of any remote sensing project (Movia *et al.* 2016); from urban landscapes with buildings casting shadows, to forests where a tree's shadow can look more like a tree to the viewer than the actual thing. When using individual photos to interpret sites, the operator can classify to each site's specific conditions and lighting, but that is time-consuming and inefficient. The challenge was to classify large contiguous areas simultaneously and have accurate classification across the whole area with changing light conditions and different features. In the southwest tile there was a large number of lichen beds throughout the jack pine stands, and so these were classified as cyan (light blue). By only having the RGB bands available to run classification, it was challenging for the software to distinguish spectral differences between the dark browns of the dry lichen beds and the light brown shadows on the sand (Figure 28). This is one disadvantage of using the DJI Inspire 1 with a standard RGB camera.

Another challenge encountered during this study was in the rendering of the orthomosaic images, as some areas did not receive enough coverage or overlap. Therefore some holes were produced in the orthomosaic, Figure 29. The UAV and the camera flew directly over the road in this area, which means it did not cover the surrounding forests and did not acquire photos of the stands away from the center of the road. When building the orthomosaic, the densely grown black spruce trees on the right side of the road were relatively tall (around 25 meters) compared to the surrounding vegetation and the low flying UAV at around 50 meters created large amounts of variation in the pixel locations between photos. The computer program could not pinpoint where the pixels were in the 3D space, so it omitted those pixels from the final product. This led to the holes that can be seen on the left of the figure. The holes can be cleaned and cropped out,

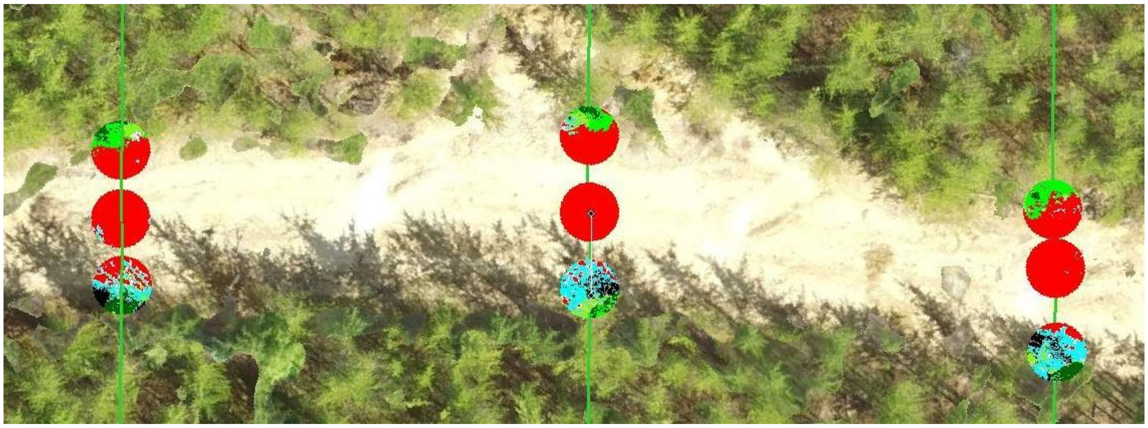

Figure 28. Site 4 showing the classification results from the large-area photo mosaic. Notice the plots along the bottom edge of the road (plots 4) in all three transects have large percentages of 'lichen' (light blue) but is actually the shadow cast by the trees.

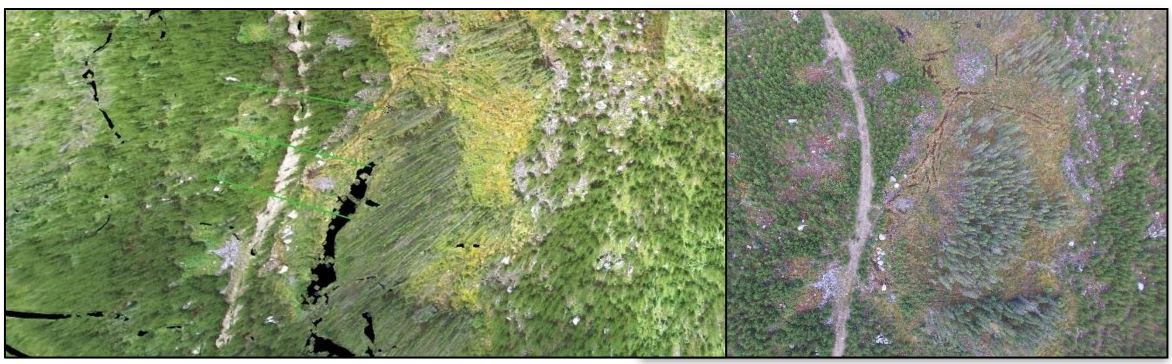

Figure 29. Site 9 showing holes, black areas, in the ortho tile that result from areas with lower amounts of coverage and tall trees that block the camera from penetrating. To the right of the center hole was a stand of mature black spruce which caused the hole to appear in the mosaic.

but that still leaves holes in the data. This may be an issue in the future for operators that are interested in linear feature mapping. The pilot planning a linear mapping mission must be aware of the possibility that they may not get enough overlap in their photos and adapt their flight plan accordingly.

The primary restriction in the use of UAVs for forest and road survey work is the legal restriction to Line-of-Sight when flying (Patterson *et al.* 2015). At the time of writing, Transport Canada Aviation Regulations (http://www.tc.gc.ca/eng/civilaviation/opssvs/flying-drone-safely-legally.html, August 2018) stipulate that all operations must be within visual line of sight up to 500 meters laterally of the pilot, reducing the operational limits from 0.5 nautical miles (926 meters) previously. If we were able to legally fly beyond 500 meters through pre-programmed missions using the road vector GIS layers, then it would have been possible to map most if not all the road segments within the study area and possibly assess the cover levels of the entire road network. As it stands now, the only way to image large areas like this in very high resolution is with traditional manned aircraft or pay for VHR satellite imagery services, both of which are costly.

Expanding on the image classification processing done herein, there are several other classification software packages available that could be tested, including licensed and open-source software packages. Future studies could look to compare the efficiency and accuracy of different programs and workflows, the amounts of time and efforts required to complete the same tasks, and how well the final products (point clouds, orthomosaics, maps) integrate between the different tools and programs from start to finish. Examples include comparing different point cloud handling software such as Agisoft, Trimble Inpho, or Fusion and ERDAS, or compare differences in classification software such as eCognition, ERDAS, QGIS add-ons, and ArcGIS.

Other potential studies should look into the use of ‘learning algorithms’ that are possible now with newer versions of software packages like ERDAS, Agisoft, QGIS. This has the potential to increase classification accuracies and expand the area of coverage. The abilities of these equations grow as the user gives more and more information and training sets that help define image features. The benefit that could be seen in the long run with some extra front-end set up would be a more automated process with larger data sets, which could reduce the overall required working hours needed to produce an accurate final product.

CONCLUSIONS

Overall, using unmanned aerial vehicles for mapping reclaimed roads was found to be successful. Very high-resolution imagery produced highly accurate, classified orthomosaic images that were used to accurately measure percent vegetation cover of road segments that have undergone decommissioning treatments. Their affordability, durability and data collection capabilities make UAVs a valuable tool in the toolbox for any resource manager. In trying to study vegetation cover, if we were to try and access those same areas by foot, we would have disturbed the site, potentially damaging the vegetation and change the cover levels being measured and skewing the results.

The UAV orthomosaic random sampling found no significant effect of reclamation efforts in the study area. While still relatively a 'young' site at roughly 20 years post-harvest, the results suggest that the reclamation strategies implemented at the English River Forest study area were not sufficient to return the site to the desired forested state. Roads that were listed as decommissioned were still present on the landscape and still acting as distinct linear features. The assumed 0.588 km/km^2 active road density for the ERF is most likely inaccurate and underestimating the actual road network disturbance impact of the whole forest license. Within this study, roads listed as removed/decommissioned/reclaimed were still present with some portions even remaining drivable. This has long-reaching implications for linear feature disturbance impacts on the English River forest and in Ontario. A stronger definition of reclamation is needed within Ontario's management guides so that more definitive goals can be set by forest managers. The results of the cover measurements suggest that the generalized reclamation treatments need to be adapted to take the surrounding forest ecology into account if forest companies wish to return unused roads to a forested state.

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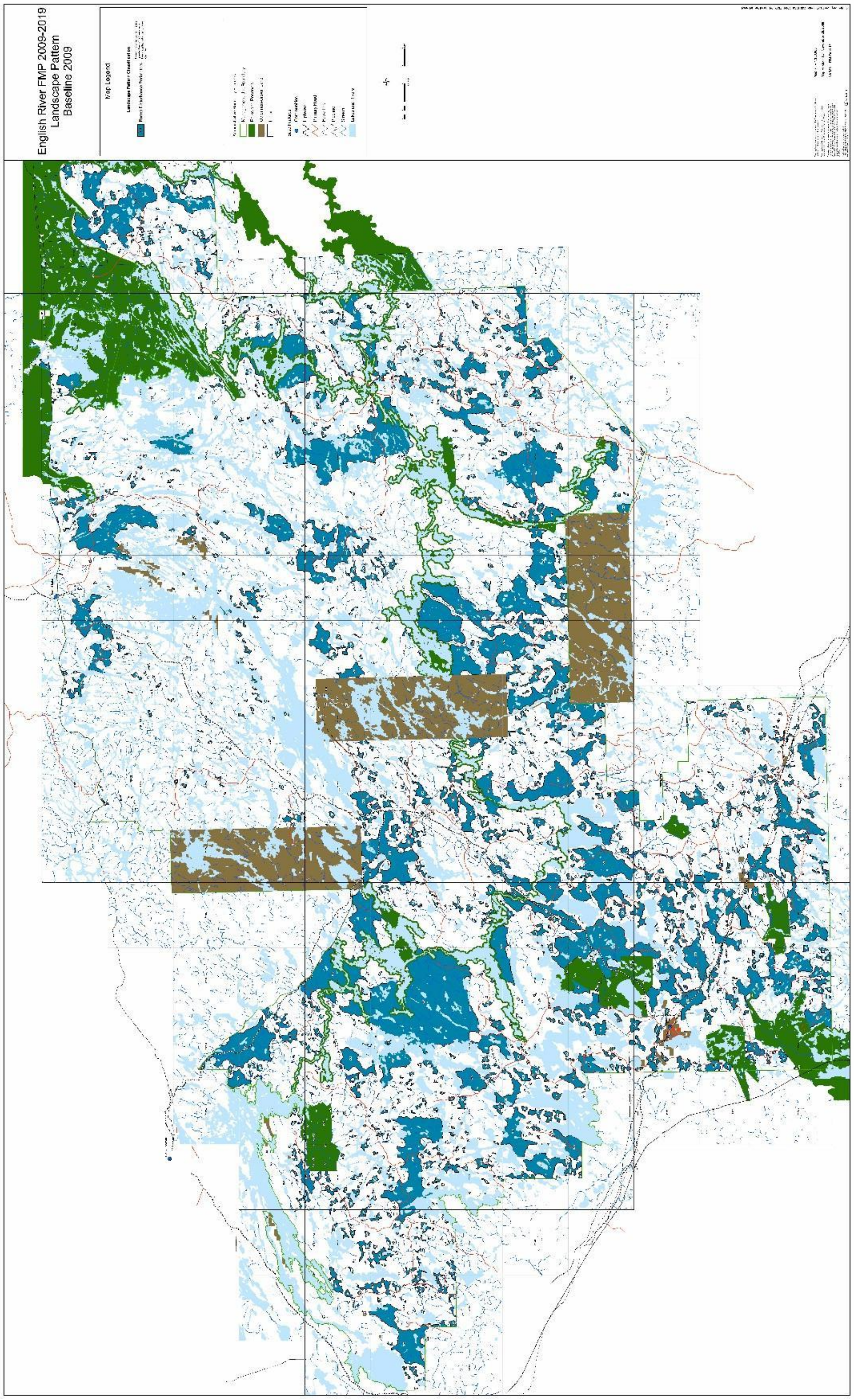
APPENDIX

Appendix 1. English River Forest License



Appendix 2. Vegetation species codes used in the study of percent vegetation cover.

Code	Name	Latin Binomial	Code	Name	Latin Binomial
Bf	Balsam fir	*Abies balsamea*	Pj	Jack pine	*Pinus banksiana*
Pb	Balsam poplar	*Populus balsamifera*	Lbt	Labrador Tea	*Ledum groenlandicum*
BH	Beaked hazel	*Corylus cornuta*	jmoss	juniper moss	*Polytrichum juniperinum*
Brbr	Bearberry	*Arctostaphylos uva-ursi*	Lfrn	lady fern	*Athyrium filix-femina*
Binw	Fringed bindweed	*Polygonum cilinode*	La	Larch	*Larix laricina*
Sb	Black spruce	*Picea mariana*	Lla	Large-leaf aster	*Aster macrophyllus*
Clb	Bluebead lily	*Clintonia borealis*	Lic	lichen spp.	*Cladonia spp.*
Va	Blueberry	*Vaccinium angustifolium*	Am	Mountain Ash	*Sorbus decora*
Bfrn	Bracken fern	*Pteridium aquilinum*	Mm	Mountain Maple	*Acer spicatum*
bmoss	broom moss	*dicranum scoparium*	ofrn	oak fern	*Gymnocarpium dryopteris*
Bb	Bunchberry	*Cornus canadensis*	PE	Pearly everlasting	*Anaphalis margaritacea*
Bhs	Bush honeysuckle	*Diervilla lonicera*	Cp	Pin Cherry	*Prunus pensylvanica*
Ctl	cattail	*typha latifolia*	Ros	Prickly rose	*Rosa acicularis*
Vtc	Cow vetch	*Vicia cracca*	Ras	Raspberry	*Rubus idaeus*
Snb	creeping snowberry	*Gaultheria hispidula*	rcmoss	running club moss	*Lycopodium clavatum*
Dgb	Dogbane	*Apocynum cannabinum*	Sar	Sarsaparilla	*Aralia nudicaulis*
fmoss	feather moss	*Ptilium crista-castrensis*	Sk	Saskatoon	*Amelanchier spp.*
fht	field horsetail	*Equisetum arvense*	smoss	schrebers moss	*Pleurozium schreberi*
frmoss	fire moss	*Ceratodon purpureus*	Sdg	Sedge	*Carex spp.*
Fw	Fire weed	*Epilobium angustifolium*	Sfrn	Sensitive fern	*Onoclea sensibilis*
Gldr	Golden rod	*Solidago spp.*	Sls	Solomon's seal	*Polygonatum spp.*
gthr	goldthread	*Coptis trifolia*	Stfl	Starflower	*Trientalis borealis*
Grs	Grass	*Graminoids*	Pt	Trembling Aspen	*Populus tremuloides*
Ag	Green Alder	*Alnus viridis*	Bw	White birch	*betula papyrifera*
Blrh	Green Bullrush	*Scirpus atrovirens*	Sw	White spruce	*Picea gluaca*
Grpn	Ground pine	*Lycopodium obscurum*	Sx	Willow	*Salix spp.*
Hkw	Hawkweed	*Hieracium cespitosum*	Str	woodland strawberry	*Fragaria vesca*
Ifrn	Interupted fern	*Osmunda claytoniana*			

Appendix 3. Camera number with the number of events and number of files.

Number of events														
Camera	**C5a**	**C5b**	**C6**	**C8**	**C11**	**C12**	**C15**	**C16**	**C17**	**C18**	**C19**	**C19b**	**C20a**	**C20b**
25/05/2017	210	0	35	0	11	0	0	0	0	427	0	0	1204	0
22/06/2017	0	220	152	689	37	0	0	0	0	1250	1019	0	0	1311
26/07/2017	0	482	373	0	116	0	0	1163	0	305	0	0	0	306
02/08/2017	0	390	161	1023	91	13	22	0	34	147	0	458	0	332
26/08/2017	0	603	119	0	56	12	25	0	61	503	0	0	0	395
04/09/2017	0	48	0	340	15	5	0	60	20	18	0	0	0	20
03/10/2017	0	200	0	0	0	29	208	0	61	38	0	0	0	39
	210	1943	840	2052	326	59	255	1223	176	2688	1019	458	1204	2403
	Event	**Total**	**14856**											
Number of files														
Camera	**C5a**	**C5b**	**C6**	**C8**	**C11**	**C12**	**C15**	**C16**	**C17**	**C18**	**C19**	**C19b**	**C20a**	**C20b**
25/05/2017	420	0	69	0	22	0	0	0	0	854	0	0	2408	0
22/06/2017	0	440	304	1377	74	0	0	0	0	2500	2037	0	0	2622
26/07/2017	0	964	746	0	232	0	0	2326	0	609	0	0	0	611
02/08/2017	0	1170	322	2046	181	26	44	0	68	293	0	915	0	663
26/08/2017	0	1809	238	0	112	24	75	0	183	1006	0	0	0	790
04/09/2017	0	96	0	680	30	10	156	120	20	18	0	0	0	39
03/10/2017	0	400	0	0	0	58	416	0	61	76	0	0	0	78
	420	4879	1679	4103	651	118	691	2446	332	5356	2037	915	2408	4803
	File	**Total**	**30838**											

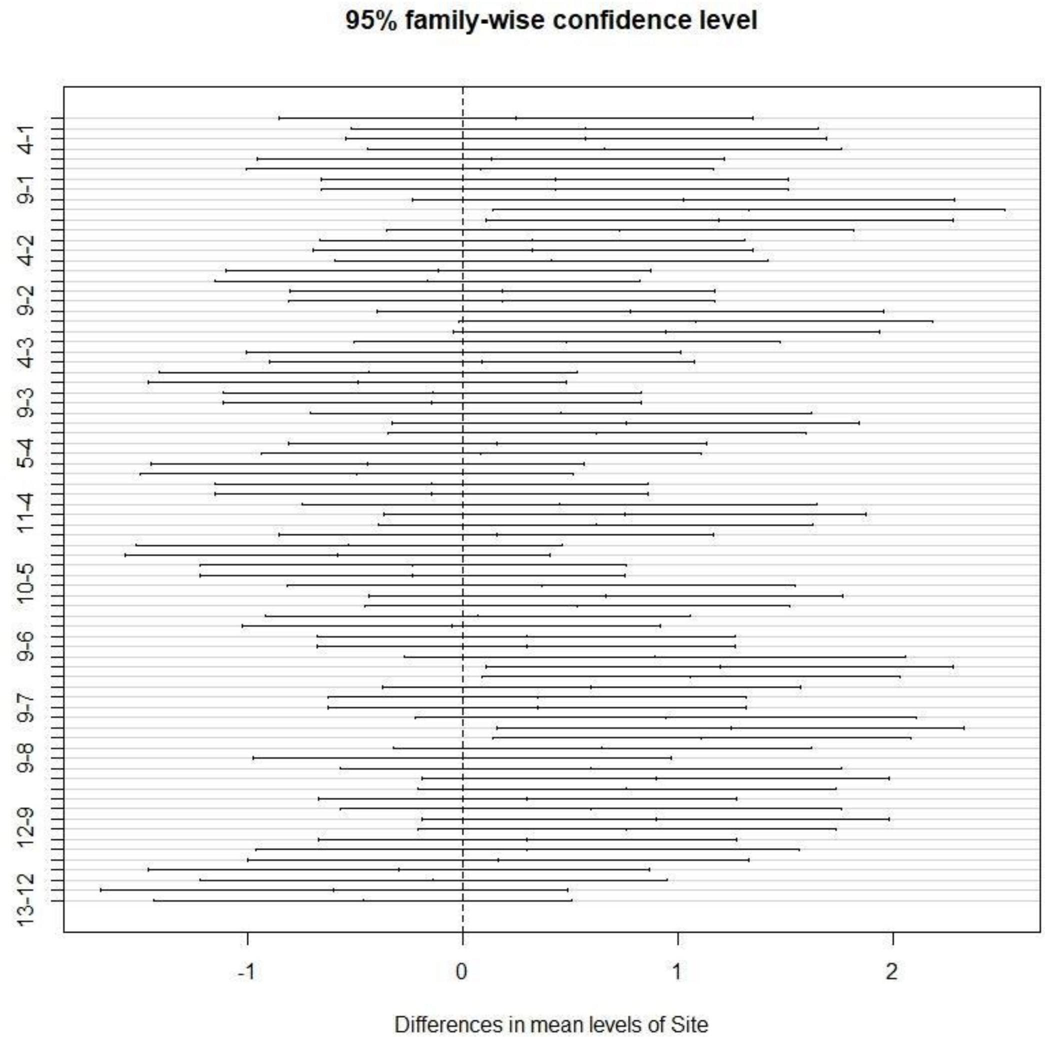
Appendix 4. A summary of species found at each site within the ERF study area.

Site1			Site2			Site3			Site4		
	Species	freq		Species	freq		Species	freq		Species	freq
1	BH	1	1	Ag	13	1	Ag	12	1	Ag	1
2	Bhs	1	2	Bb	2	2	Bb	1	2	Bb	6
3	Bw	3	3	Bhs	2	3	Bf	2	3	Bhs	1
4	Cp	3	4	Bw	9	4	Bfrn	3	4	Brbr	1
5	Fw	2	5	Cp	7	5	Bhs	2	5	Bw	1
6	Pj	10	6	Pb	1	6	Bw	5	6	Clb	1
7	Pt	3	7	Pj	9	7	Clb	3	7	Dgb	1
8	Ras	6	8	Ras	1	8	Grpn	1	8	fmoss	2
9	Sar	1	9	Sar	1	9	jmoss	2	9	jmoss	5
10	Sx	2	10	Sw	6	10	Mm	8	10	La	1
11	Va	1	11	Sx	4	11	Pj	1	11	Lic	2
			12	Va	2	12	Pt	4	12	Pj	11
						13	Sar	3	13	Sk	1
						14	Sb	1	14	Sw	6
						15	Sfrn	1	15	Sx	2
						16	Str	1	16	Va	9
						17	Sw	1			
						18	Sx	3			
Site5			Site6			Site7			Site8		
	Species	freq		Species	freq		Species	freq		Species	freq
1	Ag	3	1	Ag	2	1	Ag	4	1	Ag	5
2	Bb	6	2	Bb	1	2	Bb	1	2	Bb	1
3	Brbr	1	3	Bfrn	1	3	Bhs	2	3	Bfrn	2
4	Clb	2	4	Binw	1	4	Bw	3	4	Bhs	2
5	fmoss	2	5	Brlh	1	5	Clb	3	5	Binw	1
6	Fw	2	6	Bw	8	6	Cp	5	6	Bw	7
7	Gldr	1	7	Clb	2	7	fmoss	1	7	Clb	1
8	Grs	1	8	Cp	2	8	Fw	3	8	Cp	1
9	Lbt	1	9	fmoss	3	9	Grs	2	9	Fw	7
10	Lic	3	10	Fw	3	10	Lic	1	10	Grs	6
11	Pj	10	11	Grpn	1	11	PE	1	11	Hkw	1
12	Sb	1	12	Grs	1	12	Pj	13	12	Ifrn	1
13	Sls	1	13	jmoss	1	13	Ras	5	13	La	2
14	Smoss	4	14	Lfrn	1	14	Sb	4	14	Lla	1
15	Sx	3	15	Mm	1	15	Sls	1	15	Pj	13
16	Va	10	16	PE	2	16	smoss	2	16	Pt	5
			17	Pj	8	17	Sx	4	17	Ras	2
			18	Ras	7	18	Va	2	18	Sb	6
			19	Sdg	2				19	smoss	2
			20	Sfrn	1				20	Sx	9
			21	Sls	1				21	Va	2
			22	Str	1						
			23	Sw	9						
			24	Sx	4						
			25	Va	2						

Appendix 5. Ct'd. Part II. Appendix 4. A summary of species found within each site within the ERF study area.

Site9			Site10			Site11			Site12		
	Species	freq		Species	freq		Species	freq		Species	freq
1	Ag	2	1	Ag	2	1	Ag	9	1	Ag	8
2	Bb	3	2	Bf	1	2	Am	1	2	Bb	8
3	Bhs	1	3	Bhs	2	3	Bfrn	3	3	Bhs	8
4	Bw	2	4	Bw	9	4	Bhs	3	4	bmoss	2
5	Cp	1	5	Clb	1	5	Bw	2	5	Bw	10
6	Ctl	2	6	Grpn	3	6	Clb	1	6	Clb	1
7	fht	1	7	Grs	1	7	Cp	3	7	Cp	1
8	Fw	4	8	Mm	5	8	Grpn	1	8	fmoss	1
9	Grs	1	9	PE	1	9	Grs	3	9	frmoss	1
10	La	4	10	Pt	3	10	Lla	1	10	Fw	2
11	Pj	10	11	Ros	1	11	Mm	4	11	Grpn	2
12	Ras	1	12	Sar	2	12	ofrn	1	12	Grs	2
13	Sb	8	13	Sb	1	13	Pj	1	13	gthr	1
14	smoss	5	14	Sw	1	14	Pt	2	14	Hkw	1
15	Snb	1	15	Sx	4	15	Ras	1	15	lic	2
16	Sw	9	16	Vtc	1	16	rcmoss	1	16	Lic	1
17	Sx	2				17	Sb	1	17	PE	2
18	Va	1				18	smoss	1	18	Pj	13
						19	Stfl	1	19	Pt	2
						20	Sw	1	20	Ras	3
						21	Sx	4	21	Sb	7
						22	Va	1	22	Sfrn	1
									23	smoss	3
									24	Sx	9
									25	Va	2
Site13											
	Species	freq									
1	Ag	11									
2	Bb	9									
3	Bfrn	2									
4	Bhs	8									
5	Bw	7									
6	Clb	2									
7	Cp	6									
8	Dgb	1									
9	Fw	1									
10	Grpn	3									
11	Grs	2									
12	Hkw	2									
13	Mm	1									
14	Pj	9									
15	Pt	2									
16	Ras	4									
17	rcmoss	2									
18	Sb	7									
19	smoss	1									
20	Sx	2									

Appendix 6. Graphed results of the Tukey HSD on vegetation Cover versus Site. This post-hoc tests which sites are different from each other and by what level.



Appendix 7. Full error matrix for ERDAS_Poly images.

ERDASSouth_Poly — Pixel-based Error Matrix

Classified Data	Conifer	Vegetation	Lichen	Road	Soil	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	46	8	1	0	0	55	107763850	26.9409625	0.664168944
Vegitation	40	10	1	0	0	51	17985553	4.49738825	0.11087301
Lichen	6	12	27	2	3	50	11948896	2.987224	0.073643301
Road	5	3	10	31	1	50	10462742	2.6156855	0.064483853
Soil	4	7	16	2	20	49	14088631	3.52215775	0.086830892
Total Reference Points	101	40	55	35	24	255	162253672	40.563418	1
Overall Percent Accuracy	52.55								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	83.64	45.54
Vegitation	19.61	25.00
Lichen	54.00	49.09
Road	62.00	88.57
Soil	40.82	83.33

Area-based Error Matrix

Classified	Conifer	Vegitation	Lichen	Road	Soil	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.555486753	0.096606392	0.012075799	0	0	0.664168944	107763850	26.9409625	0.664168944
Vegitation	0.086959224	0.021739806	0.002173981	0	0	0.11087301	17985553	4.49738825	0.11087301
Lichen	0.008837196	0.017674392	0.039767383	0.002945732	0.004419	0.073643301	11948896	2.987224	0.073643301
Road	0.006448385	0.003869031	0.012896771	0.039979989	0.00129	0.064483853	10462742	2.6156855	0.064483853
Soil	0.007088236	0.012404413	0.028352944	0.003544118	0.035441	0.086830892	14088631	3.52215775	0.086830892
Total Estimated Area Proportions	0.664819794	0.152294034	0.095266877	0.046469839	0.041149	1	162253672	40.563418	1
Class Area Estimates	26.96736321	6.177566571	3.864350152	1.884975511	1.669163				
Standard Error of Area Estimates	0.034506268	0.033141156	0.015038747	0.005512968	0.006771				
Standard Error of Area Estimates (hectares)	1.399692192	1.344318554	0.61002299	0.223624828	0.274664				
95% Confidence Interval in hectares	2.743396696	2.634864366	1.195645061	0.438304662	0.538342				
Overall Accuracy	69.24								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	83.63636364	83.55
Vegitation	19.61	14.27
Lichen	54.00	41.74
Road	62.00	86.03
Soil	40.82	86.13

ERDASMiddle_Poly — Pixel-based Error Matrix

Classified Data	Conifer	Vegetation	Road	Soil	Water	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	71	51	0	0	2	124	166303258	41.5758145	0.660083772
Vegetation	21	42	0	0	0	63	68595967	17.14899175	0.272268175
Road	2	3	11	7	1	24	7990276	1.997569	0.031714661
Soil	1	6	1	17	0	25	8347339	2.08683475	0.033131901
Water	1	2	0	9	2	14	705815	0.17645375	0.002801491
Total Reference Points	96	104	12	33	5	250	251942655	62.98566375	1
Overall Percent Accuracy	57.20								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	57.26	73.96
Vegetation	66.67	40.38
Road	45.83	91.67
Soil	68.00	51.52
Water	14.29	40.00

Area-based Error Matrix

Classified	Conifer	Vegitation	Lichen	Road	Soil	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.377951192	0.271486067	0	0	0.01064651	0.660083772	166303258	41.5758145	0.660083772
Vegetation	0.090756058	0.181512117	0	0	0	0.272268175	68595967	17.14899175	0.272268175
Road	0.002642888	0.003964333	0.014536	0.00925	0.00132144	0.031714661	7990276	1.997569	0.031714661
Soil	0.001325276	0.007951656	0.001325	0.02253	0	0.033131901	8347339	2.08683475	0.033131901
Water	0.000200106	0.000400213	0	0.001801	0.00040021	0.002801491	705815	0.17645375	0.002801491
Total Estimated Area Proportions	0.472875521	0.465314386	0.015861	0.033581	0.01236817	1	251942655	62.98566375	1
Class Area Estimates	29.78437859	29.30813547	0.999026	2.115106	0.77901737				
Standard Error of Area Estimates	0.033730778	0.033713689	0.003552	0.004373	0.00761797				
Standard Error of Area Estimates (hectares)	2.124555452	2.12347909	0.223695	0.275457	0.47982298				
95% Confidence Interval in hectares	4.164128686	4.162019016	0.438441	0.539897	0.94045305				
Overall Accuracy	59.69								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	57.25806452	79.93
Vegetation	66.67	39.01
Road	45.83	91.64
Soil	68.00	67.09
Water	14.29	3.24

ERDASNorth_Poly — Pixel-based Error Matrix

Classified Data	Conifer	Vegetation	Lichen	Road	Soil	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	96	38	10	0	4	148	256339488	64.084872	0.692663172
Vegetation	4	24	7	0	6	41	55394229	13.84855725	0.149682527
Lichen	0	0	15	0	9	24	22346433	5.58660825	0.060383015
Road	0	1	3	16	0	20	15360856	3.840214	0.041507063
Soil	0	1	8	0	14	23	20637119	5.15927975	0.055764223
Total Reference Points	100	64	43	16	33	256	370078125	92.51953125	1
Overall Percent Accuracy	64.45								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	64.86	96.00
Vegetation	58.54	37.50
Lichen	62.50	34.88
Road	80.00	100.00
Soil	60.87	42.42

Area-based Error Matrix

Classified	Conifer	Vegitation	Lichen	Road	Soil	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.449295031	0.17784595	0.046802	0	0.01872	0.692663172	256339488	64.084872	0.692663172
Vegetation	0.014603173	0.08761904	0.025556	0	0.0219	0.149682527	55394229	13.84855725	0.149682527
Lichen	0	0	0.037739	0	0.02264	0.060383015	22346433	5.58660825	0.060383015
Road	0	0.002075353	0.006226	0.033206	0	0.041507063	15360856	3.840214	0.041507063
Soil	0	0.002424531	0.019396	0	0.03394	0.055764223	20637119	5.15927975	0.055764223
Total Estimated Area Proportions	0.463898204	0.269964874	0.135719	0.033206	0.09721	1	370078125	92.51953125	1
Class Area Estimates	42.91964438	24.97702364	12.55664	3.072171	8.99405				
Standard Error of Area Estimates	0.028162956	0.027730464	0.019124	0.003809	0.01505				
Standard Error of Area Estimates (hectares)	2.605623458	2.56560953	1.769304	0.352402	1.39279				
95% Confidence Interval in hectares	5.107021977	5.028594679	3.467836	0.690708	2.72986				
Overall Accuracy	64.18								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	64.86486486	96.85
Vegetation	58.54	32.46
Lichen	62.50	27.81
Road	80.00	100.00
Soil	60.87	34.92

Appendix 8. Full error matrix for eCog_Poly Images.

eCogSouth_Poly		Pixel-based Error Matrix							
Classified Data	Conifer	Road	Lichen	y	z	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	68	0	4	0	0	72	186879417	46.71985425	0.813457531
Road	4	42	4	0	0	50	8123593	2.03089825	0.035360758
Lichen	11	3	39	0	0	53	34731684	8.682921	0.15118171
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Reference Points	83	45	47	0	0	175	229734694	57.4336735	1
Overall Percent Accuracy	85.14								
Accuracy	User's Accuracy	Producer's Accuracy							
Conifer	94.44	81.93							
Road	84.00	93.33							
Lichen	73.58	82.98							
y	0.00	0.00							
z	0.00	0.00							
Area-based Error Matrix									
Classified	Conifer	Vegitation	Lichen	Road	Soil	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.768265446	0	0.045192	0	0	0.813457531	186879417	46.71985425	0.813457531
Road	0.002828861	0.029703037	0.002829	0	0	0.035360758	8123593	2.03089825	0.035360758
Lichen	0.031377336	0.008557455	0.111247	0	0	0.15118171	34731684	8.682921	0.15118171
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Estimated Area Proportions	0.802471643	0.038260492	0.159268	0	0	1	229734694	57.4336735	1
Class Area Estimates	46.08889433	2.197440624	9.147339	0	0				
Standard Error of Area Estimates	0.023731342	0.005186597	0.024007	0	0				
Standard Error of Area Estimates (hectares)	1.362978167	0.297885326	1.378792	0	0				
95% Confidence Interval in hectares	2.671437207	0.583855238	2.702431	0	0				
Overall Accuracy	90.92								
Area-based Error Matrix									
Unbiased Accuracy	User's Accuracy	Producer's Accuracy							
Conifer	94.44444444	95.74							
Road	84.00	77.63							
Lichen	73.58	69.85							
y	0.00	0.00							
z	0.00	0.00							

eCogMiddle_Poly			Pixel-based Error Matrix						
Classified Data	Conifer	Vegetation	Lichen	Road	Soil	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	102	0	3	0	0	105	131357530	32.8393825	0.930918134
Road	0	50	0	1	0	51	3716615	0.92915375	0.026339292
Lichen	15	0	24	11	0	50	1926233	0.48155825	0.013651027
Soil	8	14	12	16	0	50	1868542	0.4671355	0.013242177
z	0	0	0	0	0	0	2236431	0.55910775	0.015849371
Total Reference Points	125	64	39	28	0	256	141105351	35.27633775	1
Overall Percent Accuracy	75.00								
Accuracy	User's Accuracy	Producer's Accuracy							
Conifer	97.14	81.60							
Road	98.04	78.13							
Lichen	48.00	61.54							
Soil	32.00	57.14							
z	0.00	0.00							
Area-based Error Matrix									
Classified	Conifer	Vegitation	Lichen	Road	Soil	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.904320473	0	0.026598	0	0	0.930918134	131357530	32.8393825	0.930918134
Road	0	0.025822835	0	0.000516	0	0.026339292	3716615	0.92915375	0.026339292
Lichen	0.004095308	0	0.006552	0.003003	0	0.013651027	1926233	0.48155825	0.013651027
Soil	0.002118748	0.003707809	0.003178	0.004237	0	0.013242177	1868542	0.4671355	0.013242177
z	0	0	0	0	0	0	2236431	0.55910775	0.015849371
Total Estimated Area Proportions	0.910534529	0.029530644	0.036328	0.007757	0	0.984150629	141105351	35.27633775	1
Class Area Estimates	32.12032358	1.041732989	1.281529	0.273645	0				
Standard Error of Area Estimates	0.015249808	0.000994077	0.01526	0.001303	0				
Standard Error of Area Estimates (hectares)	0.537957364	0.035067396	0.53833	0.045968	0				
95% Confidence Interval in hectares	1.054396434	0.068732097	1.055127	0.090098	0				
Overall Accuracy	94.09								
Area-based Error Matrix									
Unbiased Accuracy	User's Accuracy	Producer's Accuracy							
Conifer	97.14285714	99.32							
Road	98.04	87.44							
Lichen	48.00	18.04							
Soil	32.00	54.63							
z	0.00	0.00							

eCog North_Poly						Pixel-based Error Matrix			
Classified Data	Conifer	Vegetation	Road	Soil	Water	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	94	0	0	2	2	98	232761125	58.19028125	0.811517768
Road	1	42	0	1	8	52	16301446	4.0753615	0.056834719
Shadow	0	0	0	0	0	0	19089297	4.77232425	0.066554515
Lichen	3	1	1	33	12	50	9959785	2.48994625	0.034724624
Soil	4	6	0	13	27	50	8710317	2.17757925	0.030368375
Total Reference Points	102	49	1	49	49	250	286821970	71.7054925	1
Overall Percent Accuracy	78.40								
Accuracy	User's Accuracy	Producer's Accuracy							
Conifer	95.92	92.16							
Vegitation	80.77	85.71							
Lichen	0.00	0.00							
Road	66.00	67.35							
Soil	54.00	55.10							
Area-based Error Matrix									
Classified	Conifer	Vegitation	Lichen	Road	Soil	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.778394594	0	0	0.016562	0.016561587	0.811517768	232761125	58.19028125	0.811517768
Road	0.001092975	0.045904965	0	0.001093	0.008743803	0.056834719	16301446	4.0753615	0.056834719
Shadow	0	0	0	0	0	0	19089297	4.77232425	0.066554515
Lichen	0.002083477	0.000694492	0.000694	0.022918	0.00833391	0.034724624	9959785	2.48994625	0.034724624
Soil	0.00242947	0.003644205	0	0.007896	0.016398922	0.030368375	8710317	2.17757925	0.030368375
Total Estimated Area Proportions	0.784000517	0.050243663	0.000694	0.048469	0.050038222	0.933445485	286821970	71.7054925	1
Class Area Estimates	56.21714318	3.60274657	0.049799	3.475464	3.588015348				
Standard Error of Area Estimates	0.016424702	0.00350823	0.000694	0.012086	0.012374894				
Standard Error of Area Estimates (hectares)	1.177741322	0.251559378	0.049799	0.86662	0.887347854				
95% Confidence Interval in hectares	2.30837299	0.493056381	0.097606	1.698575	1.739201794				
Overall Accuracy	86.36								
Area-based Error Matrix									
Unbiased Accuracy	User's Accuracy	Producer's Accuracy							
Conifer	95.91836735	99.28							
Road	80.77	91.36							
Shadow	0.00	0.00							
Lichen	66.00	47.28							
Soil	54.00	32.77							

Appendix 9. Full error matrix for ERDAS 2 Class images.

ERDASSouth_2Class — Pixel-based Error Matrix

Classified Data	Conifer	Road	x	y	z	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	97	3	0	0	0	100	137702299	34.42557475	0.848685255
Road	17	33	0	0	0	50	24551373	6.13784325	0.151314745
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Reference Points	114	36	0	0	0	150	162253672	40.563418	1
Overall Percent Accuracy	86.67								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	97.00	85.09
Road	66.00	91.67
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Area-based Error Matrix

Classified	Conifer	Road	x	y	z	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.823224697	0.025460558	0	0	0	0.848685255	137702299	34.42557475	0.848685255
Road	0.051447013	0.099867732	0	0	0	0.151314745	24551373	6.13784325	0.151314745
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Estimated Area Proportions	0.874671711	0.125328289	0	0	0	1	162253672	40.563418	1
Class Area Estimates	35.47967421	5.083743788	0	0	0				
Standard Error of Area Estimates	0.017792408	0.017792408	0	0	0				
Standard Error of Area Estimates (hectares)	0.721720877	0.721720877	0	0	0				
95% Confidence Interval in hectares	1.414572919	1.414572919	0	0	0				
Overall Accuracy	92.31								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	97	94.12
Road	66.00	79.68
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

ERDASMiddle_2Class — Pixel-based Error Matrix

Classified Data	Conifer	Road	x	y	z	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	98	2	0	0	0	100	234899225	58.72480625	0.932351947
Road	16	34	0	0	0	50	17043430	4.2608575	0.067648053
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Reference Points	114	36	0	0	0	150	251942655	62.98566375	1
Overall Percent Accuracy	88.00								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	98.00	85.96
Road	68.00	94.44
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Area-based Error Matrix

Classified	Conifer	Road	x	y	z	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.913704908	0.018647039	0	0	0	0.932351947	234899225	58.72480625	0.932351947
Road	0.021647377	0.046000676	0	0	0	0.067648053	17043430	4.2608575	0.067648053
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Estimated Area Proportions	0.935352285	0.064647715	0	0	0	1	251942655	62.98566375	1
Class Area Estimates	58.91378453	4.071879225	0	0	0				
Standard Error of Area Estimates	0.013871634	0.013871634	0	0	0				
Standard Error of Area Estimates (hectares)	0.873714087	0.873714087	0	0	0				
95% Confidence Interval in hectares	1.71247961	1.71247961	0	0	0				
Overall Accuracy	95.97								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	98	97.69
Road	68.00	71.16
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

ERDASNorth_2Class — Pixel-based Error Matrix

Classified Data	Conifer	Road	x	y	z	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	75	0	0	0	0	75	334080150	83.5200375	0.902728714
Road	11	39	0	0	0	50	35997975	8.99949375	0.097271286
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Reference Points	86	39	0	0	0	125	370078125	92.51953125	1
Overall Percent Accuracy	91.20								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	100.00	87.21
Road	78.00	100.00
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Area-based Error Matrix

Classified	Conifer	Road	x	y	z	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.902728714	0	0	0	0	0.902728714	334080150	83.5200375	0.902728714
Road	0.021399683	0.075871603	0	0	0	0.097271286	35997975	8.99949375	0.097271286
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Estimated Area Proportions	0.924128397	0.075871603	0	0	0	1	370078125	92.51953125	1
Class Area Estimates	85.49992613	7.019605125	0	0	0				
Standard Error of Area Estimates	0.005756324	0.005756324	0	0	0				
Standard Error of Area Estimates (hectares)	0.532572431	0.532572431	0	0	0				
95% Confidence Interval in hectares	1.043841966	1.043841966	0	0	0				
Overall Accuracy	97.86								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	100	97.68
Road	78.00	100.00
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Appendix 10. Full error matrix for eCog_2Class images.

eCogSouth_2Class — Pixel-based Error Matrix

Classified Data	Conifer	Road	x	y	z	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	96	4	0	0	0	100	221611101	55.40277525	0.964639242
Road	6	44	0	0	0	50	8123593	2.03089825	0.035360758
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Reference Points	102	48	0	0	0	150	229734694	57.4336735	1
Overall Percent Accuracy	93.33								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	96.00	94.12
Road	88.00	91.67
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Area-based Error Matrix

Classified	Conifer	Road	x	y	z	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.926053672	0.03858557	0	0	0	0.964639242	221611101	55.40277525	0.964639242
Road	0.004243291	0.031117467	0	0	0	0.035360758	8123593	2.03089825	0.035360758
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Estimated Area Proportions	0.930296963	0.069703037	0	0	0	1	229734694	57.4336735	1
Class Area Estimates	53.43037203	4.00330147	0	0	0				
Standard Error of Area Estimates	0.019069009	0.019069009	0	0	0				
Standard Error of Area Estimates (hectares)	1.095203249	1.095203249	0	0	0				
95% Confidence Interval in hectares	2.146598368	2.146598368	0	0	0				
Overall Accuracy	95.72								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	96	99.54
Road	88.00	44.64
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

eCogMiddle_2Class — Pixel-based Error Matrix

Classified Data	Conifer	Road	x	y	z	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	94	6	0	0	0	100	133283763	33.32094075	0.944569161
Road	8	42	0	0	0	50	7821588	1.955397	0.055430839
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Reference Points	102	48	0	0	0	150	141105351	35.27633775	1
Overall Percent Accuracy	90.67								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	94.00	92.16
Road	84.00	87.50
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Area-based Error Matrix

Classified	Conifer	Road	x	y	z	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.887895011	0.05667415	0	0	0	0.944569161	133283763	33.32094075	0.944569161
Road	0.008868934	0.046561905	0	0	0	0.055430839	7821588	1.955397	0.055430839
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Estimated Area Proportions	0.896763946	0.103236054	0	0	0	1	141105351	35.27633775	1
Class Area Estimates	31.63454783	3.641789925	0	0	0				
Standard Error of Area Estimates	0.022731421	0.022731421	0	0	0				
Standard Error of Area Estimates (hectares)	0.801881275	0.801881275	0	0	0				
95% Confidence Interval in hectares	1.571687299	1.571687299	0	0	0				
Overall Accuracy	93.45								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	94	99.01
Road	84.00	45.10
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

eCogNorth_2Class — Pixel-based Error Matrix

Classified Data	Conifer	Road	x	y	z	Total Classified Points	Total Area (Pixels)	Total Area (hectares)	Stratum Weght (Wi)
Conifer	92	8	0	0	0	100	242720910	60.6802275	0.846242392
Road	4	46	0	0	0	50	44101060	11.025265	0.153757608
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Reference Points	96	54	0	0	0	150	286821970	71.7054925	1
Overall Percent Accuracy	92.00								

Accuracy	User's Accuracy	Producer's Accuracy
Conifer	92.00	95.83
Road	92.00	85.19
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Area-based Error Matrix

Classified	Conifer	Road	x	y	z	Total Area Proportions	Total Area (pixels)	Total Area (hectares)	% of Total
Conifer	0.778543001	0.067699391	0	0	0	0.846242392	242720910	60.6802275	0.846242392
Road	0.012300609	0.141456999	0	0	0	0.153757608	44101060	11.025265	0.153757608
x	0	0	0	0	0	0	0	0	0
y	0	0	0	0	0	0	0	0	0
z	0	0	0	0	0	0	0	0	0
Total Estimated Area Proportions	0.790843609	0.209156391	0	0	0	1	286821970	71.7054925	1
Class Area Estimates	56.7078305	14.997662	0	0	0				
Standard Error of Area Estimates	0.023830719	0.023830719	0	0	0				
Standard Error of Area Estimates (hectares)	1.708793418	1.708793418	0	0	0				
95% Confidence Interval in hectares	3.349235099	3.349235099	0	0	0				
Overall Accuracy	92.00								

Area-based Error Matrix

Unbiased Accuracy	User's Accuracy	Producer's Accuracy
Conifer	92	98.44
Road	92.00	67.63
x	0.00	0.00
y	0.00	0.00
z	0.00	0.00

Appendix 11. TukeyHSD tested on the Visual Occlusion ANOVA.

```
> TukeyHSD(VisAOV)
  Tukey multiple comparisons of means
    95% family-wise confidence level

Fit: aov(formula = PercentObstruction ~ Ecosite + Site + TSD + RoadvsSightline + Road_Forest,
data = VisTest)

$Ecosite
           diff        lwr        upr     p adj
2-1  0.42916667  0.1186807  0.7396526 0.0035928
3-1 -0.01041667 -0.3689350  0.3481016 0.9998248
4-1  0.13333333 -0.2251850  0.4918516 0.7494268
3-2 -0.43958333 -0.8404190 -0.0387477 0.0269806
4-2 -0.29583333 -0.6966690  0.1050023 0.2114594
4-3  0.14375000 -0.2953434  0.5828434 0.8142879
$RoadvsSightline
               diff        lwr        upr     p adj
90-0     0.62134615  0.2934364  0.9492559 0.0000627
180-0    0.07076923 -0.2736840  0.4152225 0.9450394
270-0    0.68884615  0.3194612  1.0582311 0.0000801
180-90  -0.55057692 -0.8784866 -0.2226672 0.0003594
270-90   0.06750000 -0.2865084  0.4215084 0.9553280
270-180  0.61807692  0.2486920  0.9874619 0.0003764
```

Appendix 12. Summary of the randomly sampled points along the road segments found within the ortho-mosaic image. Plot points are sorted by increased %Vegetation Cover found in the 5m² sample areas of road center.

Item	ecosite	CID	AOI	Sample	PhotoTile	TSD	Road_Type	Pixels_Road	Perc_Road	Pixels_Veg	Perc_Veg
6	1	97	97	97	NE Tile	5	Deactivated	2416	1	0	0
26	1	166	166	166	NE Tile	5	Decommissioned	2438	1	0	0
39	1	272	272	272	NE Tile	5	Deactivated	2415	1	0	0
45	1	293	293	293	NE Tile	5	Deactivated	2413	1	0	0
17	2	113	113	113	NW Natural Tile	7	Abandoned	2427	1	0	0
64	4	551	551	551	SE	1	Deactivated	2477	1	0	0
65	4	552	552	552	SE	1	Deactivated	2472	1	0	0
71	4	563	563	563	SE	1	Deactivated	2431	1	0	0
73	4	567	567	567	SE	1	Deactivated	2472	1	0	0
21	3	136	136	136	South	10	Decommissioned	2401	1	0	0
23	3	145	145	145	South	0	Active	2418	1	0	0
27	3	178	178	178	South	10	Decommissioned	2393	1	0	0
56	3	400	400	400	South	10	Decommissioned	2384	1	0	0
2	1	93	93	93	Toptile	0	Active	2440	1	0	0
8	1	99	99	99	Toptile	9	Deactivated	2466	1	0	0
12	1	104	104	104	Toptile	9	Deactivated	2460	1	0	0
13	1	105	105	105	Toptile	0	Active	2443	1	0	0
37	1	269	269	269	Toptile	0	Active	2450	1	0	0
20	2	123	116	166	west	0	Active	2473	1	0	0
29	2	180	180	180	west	0	Active	2459	1	0	0
30	2	186	186	186	west	0	Active	2506	1	0	0
31	2	187	187	187	west	0	Active	2474	1	0	0
25	1	165	165	165	NE Tile	5	Decommissioned	2432	0.999588985	1	0.000411
9	1	100	100	100	Toptile	0	Active	2446	0.99877501	3	0.001225
7	1	98	98	98	Toptile	0	Active	2438	0.995102041	12	0.004898
72	4	564	564	564	SE	1	Deactivated	2450	0.992706645	18	0.007293
3	1	94	94	94	Toptile	0	Active	2420	0.990990991	22	0.009009
74	4	568	568	568	SE	1	Deactivated	2420	0.98534202	36	0.014658
67	4	556	556	556	SE	1	Deactivated	2420	0.977382876	56	0.022617
11	1	103	103	103	NE Tile	5	Deactivated	2436	0.975570685	61	0.024429
10	1	101	101	101	Toptile	0	Active	2375	0.975359343	60	0.024641
15	1	107	107	107	Toptile	0	Active	2350	0.969471947	74	0.030528
5	1	96	96	96	NE Tile	5	Decommissioned	2354	0.967927632	78	0.032072
28	2	179	179	179	west	0	Active	2387	0.967179903	81	0.03282
68	4	558	558	558	SE	1	Deactivated	2357	0.953864832	114	0.046135
70	4	561	561	561	SE	1	Deactivated	2277	0.943638624	136	0.056361
24	1	164	164	164	NE Tile	5	Decommissioned	2278	0.941711451	141	0.058289
58	1	442	442	442	NE Tile	5	Decommissioned	2194	0.895510204	256	0.10449
16	2	111	111	111	NW Natural Tile	7	Abandoned	2142	0.881481481	288	0.118519
22	3	141	141	141	South	10	Decommissioned	2063	0.850721649	362	0.149278
18	2	114	114	144	NW Natural Tile	7	Abandoned	1875	0.798551959	473	0.201448
47	2	306	306	306	NW Natural Tile	7	Abandoned	1872	0.78097622	525	0.219024
32	2	188	188	188	west	0	Active	1927	0.770183853	575	0.229816
44	1	290	290	290	NE Tile	5	Decommissioned	1846	0.767567568	559	0.232432
54	3	394	394	394	South	10	Decommissioned	1782	0.747169811	603	0.25283
36	1	265	265	265	NE Tile	5	Decommissioned	1630	0.680300501	766	0.319699
34	1	256	256	256	NE Tile	5	Decommissioned	1453	0.60820427	936	0.391796
19	2	115	115	115	NW Natural Tile	7	Abandoned	1361	0.570649895	1024	0.42935
48	1	308	308	308	NE Tile	5	Decommissioned	1372	0.566006601	1052	0.433993
40	1	274	274	274	Toptile	0	Abandoned	1071	0.446436015	1328	0.553564
57	3	402	402	402	South	10	Decommissioned	1040	0.4360587	1345	0.563941
66	4	555	555	555	SE	1	Deactivated	970	0.391286809	1509	0.608713
69	4	560	560	560	SE	1	Deactivated	882	0.364312268	1539	0.635688
35	1	258	258	258	NE Tile	5	Decommissioned	795	0.331941545	1600	0.668058
41	1	280	280	280	Toptile	9	Deactivated	685	0.311363636	1515	0.688636
42	1	282	282	282	Toptile	0	Abandoned	644	0.263394683	1801	0.736605
4	1	95	95	95	NE Tile	5	Decommissioned	568	0.232882329	1871	0.767118
61	2	512	512	512	west	7	Abandoned	414	0.165005978	2095	0.834994
43	1	285	285	285	NE Tile	5	Abandoned	152	0.062680412	2273	0.93732
50	2	322	322	322	NW Natural Tile	7	Skidder Trail	80	0.032115616	2411	0.967884
38	1	271	271	271	NE Tile	5	Deactivated	65	0.026705012	2369	0.973295
46	2	307	307	307	NW Natural Tile	7	Skidder Trail	15	0.006270903	2377	0.993729
53	2	333	333	333	NW Natural Tile	7	Skidder Trail	9	0.003795867	2362	0.996204
52	2	330	330	330	NW Natural Tile	7	Skidder Trail	3	0.001228501	2439	0.998771
49	2	318	318	318	NW Natural Tile	7	Skidder Trail	1	0.000425532	2349	0.999574
1	1	92	92	92	NE Tile	5	Decommissioned	0	0	2405	1
14	1	106	106	106	NE Tile	5	Deactivated	0	0	2435	1
33	1	250	250	250	NE Tile	5	Decommissioned	0	0	2385	1
51	2	329	329	329	NW Natural Tile	7	Skidder Trail	0	0	2302	1
60	2	502	502	502	NW Natural Tile	7	Abandoned	0	0	2470	1
55	3	399	399	399	South	10	Decommissioned	0	0	2411	1
62	2	513	513	513	west	7	Abandoned	0	0	2521	1
63	2	514	514	514	west	7	Abandoned	0	0	2510	1

SUPPLEMENTAL INFORMATION

Appendix 13. DJI Inspire 1 Specifications

The DJI Inspire 1 houses the *X3 FC350* high resolution, colour camera, with a 4000 x 3000 resolution, a 20-mm lens, an f-stop of f/2.8 and field of view (FOV) of 94° with a focal length of 3.6 mm. The sensor has 12.4M effective pixels for accurate image analysis but a total of 12.76M pixels over the entire 6.17 x 4.55 mm sensor (https://www.heliguy.com/blog/wp-content/uploads/2015/09/comparison.png, October 2017). The camera is detachable from the UAV platform and interchangeable with other lenses that have the same mount. The camera operates on an electronic gimbal, ZENMUSE X3, with power for the camera drawn from the standard, intelligent, LiPo 6S 4500 mAh battery power-pack that feeds the entire UAV (http://www.dji.com/inspire-1/info).

The X-3 camera does not use a panoramic/fish eye type lense, although there is some, limited amount of distortion. At the height of 20 meters, the camera captures a limited amount of area, forcing the increase in elevation to capture the area of the 40 x 50-meter site. Changes in elevation allow for an increase in the surface area captured, but there is a loss of detailed information due to the static size of the sensor pixels. At 20 meters the GSD is 0.722 cm^2 per pixel, where at 120 meters the GSD is 25.99 cm^2.

Appendix 14. Reference table for Ground Sampling Distance (GSD) of low elevation flights common for UAV missions.

FLIR Vue Pro 640 GSD				Inspire 1 X3 FC350 GSD			
Altitude m	Pixel Area cm^2	Pixel Area m^2	Image Area m^2	**Altitude m**	Pixel Area cm^2	Pixel Area m^2	Image Area m^2
10	0.7943024	0.0000794	26.0277008	**10**	0.1805138	0.0000181	216.6165123
20	3.1772096	0.0003177	104.1108033	**20**	0.7220550	0.0000722	866.4660494
30	7.1487215	0.0007149	234.2493075	**30**	1.6246238	0.0001625	1949.5486111
40	12.7088383	0.0012709	416.4432133	**40**	2.8882202	0.0002888	3465.8641975
50	19.8575598	0.0019858	650.6925208	**50**	4.5128440	0.0004513	5415.4128086
60	28.5948862	0.0028595	936.9972299	**60**	6.4984954	0.0006498	7798.1944444
70	38.9208173	0.0038921	1275.3573407	**70**	8.8451743	0.0008845	10614.2091049
80	50.8353532	0.0050835	1665.7728532	**80**	11.5528807	0.0011553	13863.4567901
90	64.3384939	0.0064338	2108.2437673	**90**	14.6216146	0.0014622	17545.9375000
100	79.4302394	0.0079430	2602.7700831	**100**	18.0513760	0.0018051	21661.6512346
150	178.7180385	0.0178718	5856.2326870	**150**	40.6155961	0.0040616	48738.7152778
200	317.7209574	0.0317721	10411.0803324	**200**	72.2055041	0.0072206	86646.6049383
250	496.4389960	0.0496439	16267.3130194	**250**	112.8211002	0.0112821	135385.3202160
300	714.8721542	0.0714872	23424.9307479	**300**	162.4623843	0.0162462	194954.8611111
400	1270.8838296	0.1270884	41644.3213296	**400**	288.8220165	0.0288822	346586.4197531

Appendix 15. File transfer from eCognition to ERDAS Imagine.

The eCognition files are brought into ERDAS Imagine in the ERDAS viewer, under the *Manage Data* tab, select the *Import Data*. The file format is matched to the exported file from eCognition, *GeoTIFF*, then in the second drop down bar, the file name was selected, and finally, the output file (what it will be called in ERDAS) was placed in the 3rd drop menu and located via the folder options tab. Note that the generated output file is an ERDAS image file (.img). Another way of handling this transfer is to export the classifications as shapefiles. When exporting the resultant shapefiles, select Polygon Raster and select all features except unclassified. Then, for feature attributes, select Object Features>Geometric>Extent>Area (double click 'Area' to add). Then, select Class Related Features>Related to Class>Class Name. You have to double click to create a new class name, and in the new window that is opened, click 'Ok'.

Appendix 16. Impacts of changing forest classification definitions in the forest management planning manual.

In the 2009-2019 FMP it is worth noting that in the land summary section of the management plan, the authors point out the change and addition of two new terms in the classification of production forests. 'Recent disturbance' and 'below regeneration standards' added to in 2004, which attempt to better categorize the current conditions of the forests on the land. That change in terminology changed the management implications for that land area, as described below from section 2.2.3.1 of the Phase 1 2009-2019 English River Forest Management Plan (Lawson 2009a).

> *"From the values above, it is obvious that a significant portion of the area deemed depleted in the 2004 FMP is now deemed below regeneration standards. Also, a portion (about 2,900 ha) of the area classed as barren and scattered in the 2004 FMP is now more accurately classed as 'recent disturbance'. The total of 18 depleted area and barren and scattered in the 2004 FMP is about 126k ha. The total of recent disturbance 19 and below regeneration standards in the 2009 FMP is about 129.6k ha; a difference of about 3,900 ha (just 20 over 0.5 percent of the production forest). Considering the annual cut on this Forest, this amount is considered inconsequential."*

Although just under 4,000 ha out of 1 million may be as inconsequential as the author states, there is a bias inherent in that point of view, a point of view that leads the reader to think small details are not necessarily a priority or may not even matter. That the land area re-classed to below regeneration stands is only half of one percent, it can add to the cumulative total over the yearly and 10-year planning totals and can have direct effects on the forest ecology especially when the 500-meter disturbance buffer is added. They also affirm that the new definition of barren and scattered better reflects the condition of the forest, yet no attributes explaining whether these areas are naturally barren as a normal state or the result of human activity. This change in forest classification, therefore, can have an impact on the disturbance level of a managed forest, and impact forestry operations.

Appendix 17. Linear distances (m) between camera locations. The minimum distance was 61.24 m, the maximum was 7 112.57 m, with an average of 3,237.00 m and a standard deviation of 1 776.44 m. Camera trap locations are shown in Figure 9.

Camera	C5a	C5b	C6	C8	C11	C12	C15	C16	C17	C18	C19	C19b	C20a	C20b
C5a		3697.85	241.81	2137.33	335.86	4041.67	2636.92	3579.57	1238.34	4442.07	5139.57	6025.49	120.31	175.78
C5b	3697.85		3912.37	2285.38	4039.89	824.25	2395.14	4328.39	4529.55	789.63	1841.27	3006.24	3810.88	3875.33
C6	241.81	3912.37		2318.2	169.97	4284.22	2875.72	3553.26	1215.43	4646.83	5321.97	6180.85	121.66	61.24
C8	2137.33	2285.38	2318.20		2456.23	2977.21	2902.49	2320.62	3358.23	2759.50	3188.21	3946.81	2178.03	2223.80
C11	335.86	4039.89	169.97	2456.23		4411.63	2913.59	3699.35	1092.38	4781.91	5468.89	6351.46	262.44	201.75
C12	4041.67	824.25	4284.22	2977.21	4411.63		2169.77	5112.61	4699.81	1281.00	2345.92	3540.99	4151.90	4213.27
C15	2636.92	2395.14	2875.72	2902.49	2913.59	2169.77		5231.92	2810.45	3167.30	4234.13	5394.59	2758.76	2812.48
C16	3579.57	4328.39	3553.26	2320.62	3699.35	5112.61	5231.92		4752.07	4569.02	4441.98	4626.97	3543.99	3551.64
C17	1238.34	4529.55	1215.43	3358.23	1092.38	4699.81	2810.45	4752.07		5342.70	6134.60	7112.57	1248.06	1216.02
C18	4442.07	789.63	4646.83	2759.50	4781.91	1281.00	3167.30	4569.02	5342.70		1093.38	2275.97	4518.84	4594.36
C19	5139.57	1841.27	5321.97	3188.21	5468.89	2345.92	4234.13	4441.98	6134.60	1093.38		1178.37	5193.83	5259.34
C19b	6025.49	3006.24	6180.85	3946.81	6351.46	3540.99	5394.59	4626.97	7112.57	2275.97	1178.37		6078.84	6144.19
C20a	120.31	3810.88	121.66	2178.03	262.44	4151.90	2758.76	3543.99	1248.06	4518.84	5193.83	6078.84		70.53
C20b	175.78	3875.33	61.24	2223.80	201.75	4213.27	2812.48	3551.64	1216.02	4594.36	5259.34	6144.19	70.53	