Restoration of Degraded Trail Crossings in Wet and Riparian Areas Using Balsam Poplar Cuttings, Blue Rapids Provincial Recreation Area, Alberta

by

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Abstract

Protected areas are established and maintained in Alberta, Canada for the conservation and preservation of natural features and to facilitate their use and enjoyment for outdoor recreation and education. The use of off-highway vehicles (OHV) is a popular recreational activity in Alberta's parks. OHV use can cause immediate negative impacts on natural areas. These impacts include: soil compaction, altered water infiltration dynamics, inhibition of root growth, and direct damage to native plants, which in turn facilitates the establishment of non-native species. Consequently, the plant community structure on the directly impacted trail and in adjacent forest, can be affected. The structure and function of such degraded ecosystems can be restored through active restoration, including stopping continued damage, recontouring of surface area, and revegetation. One way to quickly re-establish trees or shrubs is through the use of hardwood cuttings, which are stems cut from donor plants. Balsam poplar (*Populus balsamifera* L.) is a common native tree species in Alberta that is well-suited for such an approach.

Blue Rapids Provincial Recreation Area (BRPRA), located outside of Drayton Valley, Alberta, Canada, has supported OHV recreational use for several decades on trails that were originally developed for oil and gas activities (e.g., seismic lines), but is also positioned to conserve the area's natural features. BRPRA has several kilometers of unmanaged OHV trails, which traverse wet and riparian areas of the North Saskatchewan River. Some of these trails have been decommissioned and targeted for active restoration.

This study focused on the revegetation of those decommissioned OHV trails with the use of balsam poplar cuttings. To better understand the factors influencing plant establishment on restored trails I investigated: i) the efficacy of three pre-planting treatments for restoration with balsam poplar cuttings; and ii) if the plant community composition differed on enhanced trails, adjacent edges and adjacent forested areas, as well as if there were differences at each position within two years after trail enhancement.

We established a revegetation experiment by planting balsam poplar cuttings prepared with three different treatments (rooted, unrooted, and direct plant) on 15 decommissioned OHV trails within BRPRA. For the rooted and unrooted treatments cuttings were collected from live trees at BRPRA during dormancy, whereas for the direct plant treatment cuttings were collected in early summer. Rooted cuttings were initially rooted in the greenhouse and transferred to the field at the end of the first growing season, whereas for the unrooted and direct plant treatments cuttings were planted in the field in early summer. Survival and growth of the planted cuttings was monitored during the year of establishment through the end of the subsequent growing season. We also investigated the plant community and abiotic variables on restored trails, on their adjacent edges and in the adjacent 10 m forested area both during the year the re-vegetation experiment was established and one year later.

The rooted treatment showed overall better survival and taller plants, but did not perform better than the other two treatments in height growth difference during the second growing season; the unrooted treatment had the second-best survival and direct plant had the poorest survival. All three treatments were recommended, since each one might be practical to specific needs of different restoration programs. Initial diameter of the cutting had a nearly significant positive influence on survival during the first growing season and height growth in both growing seasons, optimal initial diameters ranged from 4 to 8 mm. Thus, the use of cuttings with initial diameters from 4 to 8 mm and collected during dormancy is recommended for maximum survival and growth. Environmental conditions on the trail also influenced survival and height growth.

Enhanced OHV trails and their associated edges had different plant communities when compared to adjacent forested areas. The increased abundance of non-native species and increase in graminoids on trails and edges is of concern. Trails seem to be favouring the establishment of annual vegetation. Therefore, the planting of native woody species might help plant succession on these trails, although, this cannot yet be predicted. Longer-term monitoring of the enhanced OHV trails can help inform their successional trajectories, including evaluating survival and growth patterns of the three restoration planting treatments.

Dedication

Aos meus heróis, e pais, Cristina e Eraldo.

To my heroes, and parents, Cristina and Eraldo.

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List of Abbreviations

Abbreviation/Acronym	Description
AEP	Alberta Environment & Parks
AIC	Akaike Information Criterion
AICc	Akaike Information Criterion corrected for small sample sizes
ANOVA	Analysis of variance
BRPRA	Blue Rapids Provincial Recreation Area
EPBR	Eagle Point-Blue Rapids
EPBRPC	Eagle Point-Blue Rapids Parks Council
EPBRPS	Eagle Point-Blue Rapids Park System
EPPP	Eagle Point Provincial Park
FGRMS	Alberta Forest Genetic Resource Management and Conservation Standards
NMS	Non-metric multidimensional scaling
MANOVA	Multivariate analysis of variance
MRPP	Multi-response permutation procedure
OHV	Off-highway vehicle
perMANOVA	permutation-based MANOVA
PPA	Provincial Parks Act
PR	Penetration Resistance
PR in	Penetration Resistance in planting hole area
PR out	Penetration Resistance outside planting hole area
SE	Standard error
USDA	United States Department of Agriculture
VasCan	Database of Vascular Plants of Canada

Chapter 1. General introduction

1.1. Recreation Areas: Uses and Impacts

Provincial Parks and Provincial Recreation Areas are protected areas in Alberta administered by Alberta Environment and Parks (AEP 2017). Together these two types of protected areas are key components of the overall Alberta Government protected areas strategy for conservation and recreation. The Provincial Parks Act regulates activities and restrictions in Provincial Parks and Provincial Recreation Areas (AEP 2017). Provincial Parks are established and maintained "for the preservation of Alberta's natural heritage, for the conservation and management of flora and fauna, for the preservation of specified areas, landscapes and natural features and objects in them that are of geological, cultural, historical, archeological, anthropological, paleontological, ethnological, ecological or other scientific interest or importance, to facilitate their use and enjoyment for outdoor recreation, education and the appreciation and experience of Alberta's natural heritage and to ensure their lasting protection for the benefit of present and future generations" (PPA 2000). Provincial Recreation Areas differ from parks in that, as per the Provincial Park Act (2000), they are instead established and maintained primarily "to facilitate their use and enjoyment for outdoor recreation by present and future generations".

Various activities occurring within protected areas can be considered as recreational use, such as hiking, jogging, bird watching, biking, photography, horseback riding and off-highway vehicle (OHV) use (Charman and Pollard 1995; Jordan 2000). These recreational activities can cause stress on, or damage to, ecological communities, through trampling, habitat disturbance and modification, increased competition caused by addition of non-native species, nutrient loading, pollution, fragmentation, and edge effects (Charman and Pollard 1995; Jordan 2000; Monz et al. 2013).

The effects of recreational activities on soil and vegetation properties can be immediate and may remain for decades (Ozcan et al. 2013; Korkanç 2014). The extent and intensity of recreational user impacts can be influenced by many factors: amount of use, timing of use, type and behavior of use, and type and condition of the environment (Monz et al. 2013). Trampling, which can result from hiking, horseback riding and OHV use, is the most commonly researched recreational impact (Cole and Bayfield 1993; Jordan 2000; Monz et al. 2013; Korkanç 2014). Although trampling is commonly researched, the ecological effects caused specifically by OHV use have had limited study worldwide (Brooks and Lair 2005; Trip and Wiersma 2015), and little experimental research has been done to assess the effects of OHV use in wet habitats (Hannaford and Resh 1999).

1.2. Effects of Off-highway Vehicle Use

Off-highway vehicles (OHVs) are motorized vehicles that are usually small enough to access areas where standard vehicles cannot go; additionally, they are able to traverse rough terrain including gravel, mud, sand, and water (Taylor no date). OHV trails are typically 2 to 4 m wide and have exposed soil on the surface (Brooks and Lair 2005). The use of OHVs and presence of their associated vehicular trails in natural areas can result in changes in the physical and chemical properties of ecosystems and alterations in plant populations and their community structure (Hannaford and Resh 1999; Jones 1999; Brooks and Lair 2005). Vehicular trails and their use cause direct impacts (i.e. on trail), such as loss of vegetation cover and erosion, indirect impacts (i.e. off-trail), such as increased sediment and nutrient loading in surrounding vegetation, and landscape impacts, such as habitat fragmentation and spread of non-native species (Brooks and Lair 2005; Ouren et al. 2007; Trip and Wiersma 2015).

OHV activity alters soil structure and density, increases soil temperatures, breaks soil crusts, and disrupts fine gravel surfaces that stabilize soils and control soil erosion (Wilshire 1977; Ouren et al. 2007; Trip and Wiersma 2015). Application of surface forces to soils, including from recurrent OHV use, leads to soil compaction; compacted soils require a greater force before they can be penetrated, as compared to uncompacted soils (Wilshire et al. 1978; Webb 1983). Through surface subsidence compacted areas (i.e. trails) can become lower than the adjacent ground surface area; this can lead to a redirection of water flow from the surface towards the trail (Meyer 2002). Compaction also reduces soil infiltration rates substantially. In initially dry soils, the infiltration capacity drops as a logarithmic function of the number of OHV passes (Webb 1983). This is even more problematic for wet soils and can result in faster and more frequent ponding of water. Moreover, compaction contributes to accelerated soil erosion (Webb 1983). Vehicular trails and crossings can result in redirection of water runoff in a concentrated stream, accelerating soil erosion rates and resulting in changes in speed, direction and quality of water moving along the trail (Wilshire 1977; Webb 1983; Brooks and Lair 2005; Taylor no date). Furthermore, when soil particles moved by erosional processes are deposited they can bury fertile soils, resulting in reduction of productivity and soil nutrient availability (Wilshire 1977).

In addition to erosion, Meyer (2002) identified surface failure as an important impact of OHV use on wet soils. Surface failure occurs when water pools in low-lying areas on trails and saturates the soil; then the downward pressure of an external force (e.g. vehicle’s wheels) damages the soil structure, resulting in muddy sections and deep holes on trails. In wet conditions some trail segments become nearly inaccessible to OHVs (Arp and Simmons 2012). When this happens, OHV users may avoid the degraded surfaces; this results in widening of the trail (Meyer 2002). When avoidance is no longer possible OHV users may deviate from maintained trails and create new trails adjacent to the maintained ones (Meyer 2002; Arp and

Simmons 2012). This process is called trail braiding. Trail braiding expands the impacts of OHV use through the landscape (Meyer 2002; Arp and Simmons 2012).

The changes incurred to soil characteristics by OHV use affect the composition and structure of vegetation that can grow on the trail; the ability to support vegetation is compromised after recurrent OHV use. Changes in soil strength and structure inhibit the growth of root systems and reduce water infiltration. Vegetation is directly affected through decreased size and abundance of native species (Hannaford and Resh 1999), reduction of root biomass through erosional exposure and of above-ground portions of plants through breakage or crushing (Wilshire 1977). This then leads to declines in photosynthetic capacity and reproduction (Ouren et al. 2007). Additionally, dust generated by vehicles can suppress plant growth (Ouren et al. 2007). Changes in soil temperature caused by the lack of vegetation cover alter germination times of seeds and general growth rates of plants, and the reduction of soil nutrient availability inhibits plant growth (Wilshire et al. 1978).

Reduced native vegetation cover on OHV trails can, in turn, facilitate the establishment of non-native species, which outcompete native species, leading to a decrease in native plant biodiversity (Charman and Pollard 1995; Ouren et al. 2007). Seeds of non-native species can get attached to the tires or other parts of OHVs, and in this way disperse to distant locations. Tracks left by OHVs can create seedbeds for these non-native species, facilitating their establishment (Jones 1999). The lack of native vegetation allows non-native species to easily occupy sites with disturbed soil in areas where they were not previously found (Webb et al. 1983; Brooks and Lair 2005). As a result, the vegetation community structure on OHV trails can be affected and plant community composition on and adjacent to the trail can differ from pre-disturbance conditions and from the surrounding area (Meyer 2002).

Vegetation and soil in wet and riparian areas are more sensitive to OHV damage than in dry areas. According to Trip and Wiersma (2015) the impacts caused by recreational vehicles are greater when the soil moisture is greater. Soil moisture level dictates how stable the soil is and how resistant it can be to mechanical stresses. In general, the higher the soil moisture the less stable the soil is. To clarify, at saturation, finely textured soils have reduced load bearing capacity: soil voids are filled with water, not leaving space for air and particle movement; this results in more strongly altered soil structure when the area is subjected to surface forces (Meyer 2002).

1.3. Conservation in Recreational Areas

The frequency of recreation and tourism activities, including OHV use, is increasing in protected areas (Monz et al. 2013) and on crown land. OHV use in North America has increased substantially since the 1960's (Slaughter et al. 1990; Taylor no date). Protected status of an area does not influence the amount of OHV traffic, which has been found to be similar outside and within protected areas (Trip and Wiersma 2015). We thus need studies that quantify the effects of such traffic and provide insights into management actions that could mitigate the undesirable effects of OHV use.

Some management actions have been described to help mitigate recreational use impacts on protected areas. Reducing and controlling the use of recreational motorized vehicles in sensitive areas (e.g. providing protection for the terrain and directing traffic to designated protected routes) helps to avoid direct impacts. Also, controlling the opening of new unmanaged trails avoids the spreading of the impacts along the landscape (Wilshire et al. 1978; Slaughter et al. 1990, Hannaford and Resh 1999; Trip and Wiersma 2015).

1.4. Restoration of Degraded Wet and Riparian Areas

Restoration is "the process of repairing human caused damage to the diversity and dynamics of indigenous ecosystems"; the process begins with the judgment that an ecosystem is damaged by humans to a point that it will not recover naturally in the near term (i.e. 50 years) and that continued degradation may occur (Jackson et al. 1995). Restoration has also been defined as the return of disturbed ecosystems to conditions similar to the pre-disturbance state (Wissmar and Beschta 1998).

Riparian areas are the interface between aquatic and terrestrial systems, spanning from the zone of submerged aquatic vegetation to upland areas (Boudell et al. 2015). From an ecological standpoint, riparian systems are essential, as these areas act as an ecological corridor, are often biodiversity hotspots, and are critical in the life histories of many species (Naiman and Décamps 1997). Riparian systems are intimately tied to the wetland systems they surround. Both systems function synergistically with each other, providing vital services that enhance the functioning of the ecosystem including water filtration, carbon storage, and nutrient cycling (Euliss et al. 2006; Mitsch and Gosselink 2007). However, riparian areas are in a state of substantial stress, ranking as one of the most degraded systems across the globe (Wevill and Florentine 2014). Given the continued pressure of resource development, urbanization and agriculture, coupled with climate change-induced stressors, riparian areas are not able to fulfill their ecological roles; hence, there is a critical need for their restoration (Naiman and Déchamps 1997).

Active restoration typically includes activities designed to restore the structure and function of the ecosystem. For example, restoration can first ensure that continued damage is stopped through management actions that will reduce or eliminate human activities related to the degradation. Subsequently, recontouring of the surface area to the previous condition can be done; finally, the area can be revegetated with native plant species (Kay and Graves 1983; Slaughter et al. 1990; Charman and Pollard 1995; Jones et al. 2018).

The re-establishment of native vegetation on degraded areas can be achieved with the use of various techniques that range from hands off approaches, like just stopping the disturbance and leaving the area isolated for natural regeneration (i.e., passive recovery), to more involved processes such as planting seeds, bare root stock seedlings or hardwood cuttings (i.e., active restoration) (Jones et al. 2018; Landhäusser et al. 2019).

The effects of recreational OHV use differ depending on soil, terrain, vegetation and traffic characteristics (Slaughter et al. 1990; Brooks and Lair 2005; Arp and Simmons 2012; Trip and Wiersma 2015). Wet and riparian areas are particularly sensitive to the effects caused by OHV use showing low resilience and long recovery times following recreational vehicular disturbance (Charman and Pollard 1995; Arp and Simmons 2012). Full recovery in these site types may take a substantially longer period as compared to drier sites or may not occur naturally (Charman and Pollard 1995; Arp and Simmons 2012). Therefore, they might be particularly in need of active restoration.

1.5. Active Restoration with Hardwood Cuttings

In more sensitive areas, such as wet and riparian areas in temperate climates, active restoration of disturbed areas using native tree species is often necessary (Driver 2017). Aspen (*Populus tremuloides* Michx.) and balsam poplar (*Populus balsamifera* L.) forests are abundant in the forested regions of Alberta, Canada. Despite their abundance, the planting of aspen seedlings as a restoration strategy for vegetation recovery has proved to be more challenging than the planting of balsam poplars (Irwin 1985; as cited in DesRochers et al. 2004).

Propagation through hardwood cuttings is a restoration strategy that uses segments of live stems cut from donor plants (i.e., cuttings), generally hardwood tree and shrub species that root easily. Cuttings are generally taken during a period of dormancy, which means during the winter in Alberta, and planted in late spring/early summer (Schreiber 2015). The plant parts that propagate vegetatively (naturally or through hardwood cuttings) originate from dormant buds (Dickmann et al. 2001). After planting in late spring/early summer, as temperatures rise and days get longer adventitious roots are formed on the cutting, vegetative buds break dormancy, and new shoots emerge from them (Hartmann et al. 1990; Dickmann et al. 2001).

The primary species utilized for hardwood cuttings propagation in Alberta are willows (*Salix* spp.) and balsam poplar (DesRochers et al. 2004; Cows and Fish 2007; Polster 2013). These plants have the capacity to develop adventitious roots from cut stem sections (Schroeder and Walker 1991) and these rooted cuttings can then be easily established in the field (Dickmann et al. 2001). They also have fast growth rates (Douglas et al. 2016), which is an important characteristic for plants to be used successfully in restoration.

Balsam poplar is an early and intermediate successional hardy and fast-growing deciduous broadleaf tree species (Zasada and Phipps 1990). It grows on upland and floodplain sites but is mostly concentrated on flood plain (Zasada and Phipps 1990). This species reaches flowering age at around 8 to 10 years-old; but it can live up to 200 years (Zasada and Phipps 1990). Balsam poplar can reproduce sexually, via seeds, and asexually, through vegetative propagules (Zasada and Phipps 1990; Dickmann 2001). This species has the ability to sprout from existing or adventitious buds at the stump and/or root collar (Zasada and Phipps 1990; Dickmann 2001). Balsam poplar can also form roots on buried branch fragments (Zasada and Phipps 1990; Braatne et al. 1996; Dickmann 2001), due to pre-formed root primordia, which are initiated on stems during the first growing season (Krasny 1988). For the same reason, balsam poplar can be vegetatively propagated through hardwood cuttings (Dickmann 2001). The natural rooting of fallen branch segments in balsam poplar is common on wet and riparian habitats (Braatne et al. 1996; Dickmann 2001). This is possible because balsam poplars are flood tolerant (Krasny 1988; Braatne et al. 1996), since their stems are able to form new roots even when covered by silt deposited during flooding. Other tree species, such as aspen, do not have this capability (Dickmann 2001).

The establishment of balsam poplar trees on a site influence soil development. The dominance of balsam poplar trees promotes an increase in: forest floor nitrogen content; soil carbon and cation exchange capacity; and depth of the forest floor (Zasada and Phipps 1990). Balsam poplar stem cuttings have been used for rapid re-vegetation of degraded areas with the aim of returning wildlife habitat, stabilizing riparian soils and protecting soil structure (Stanturf et al. 2001; DesRochers et al. 2004). Because they are native to BRPRA, and in reason of their ability to root from stem cuttings, their fast growth rates, resistance to flooding, and associated benefits to soil structure; I decided to use balsam poplars on this restoration project.

Factors influencing cutting performance

Establishment, survival, and height growth of balsam poplar cuttings may be affected by factors including: season of collection, initial planting environment, diameter of planted cutting, and edaphic factors (i.e., soil moisture and compaction).

The season in which the cutting is collected may be important; cuttings collected during dormancy have been commonly used in both greenhouse and field experiments (Schroeder and Walker 1991; Stanturf et al. 2001; DesRochers et al. 2004). However, cuttings collected in late spring/early summer (‘green’ cuttings) can also be used for propagation, as balsam poplar cuttings can form adventitious roots also from stem cuttings from trees that have leafed out. The planting of green cuttings is not commonly done (Dickmann et al. 2001; Stanturf et al. 2001), since balsam poplar have more carbohydrate reserves during dormancy, and by the time spring arrives, those reserves are depleted (Dickmann et al. 2001). As a result of more extensive root system development in response to near optimum growing conditions, survival and height growth are usually higher for cuttings initially planted in a greenhouse when compared to those planted directly in the field; yet few studies have compared performance of cuttings initially grown in the greenhouse and then transferred to the field (i.e., rooted cuttings) with the performance of cuttings initially established in the field (Schroeder and Walker 1991; DesRochers et al. 2004). The initial diameter of the cutting can be important because it reflects available carbohydrate reserves (Dickmann et al. 2001; Zalesny Jr. et al. 2005; Tilley and Hoag 2008; Douglas et al. 2016); thus, larger diameter cuttings are expected to perform better than smaller diameter cuttings.

Edaphic factors may be important because after cuttings are planted they get established and produce new adventitious roots. These roots form the initial fine-root network, which provides a large absorbing surface through lateral expansion and elongation into the soil (Dickmann et al. 2001; Douglas et al. 2016). Root penetration can be hampered by high soil strength (Zalesny Jr. et al. 2005). Increasing soil moisture generally increases root and shoot growth up until it reaches the point of saturation (Zalesny Jr. et al. 2005). The fine absorbing "feeder" roots of most plants are concentrated near the soil surface, in the top 10 cm of the soil profile (Dickmann et al. 2001). Thus, the successful initial rooting of a cutting requires correct moisture conditions near the soil surface: too much water causes anoxia and drought conditions result in desiccation and death (Dickmann et al. 2001; Zalesny Jr. et al. 2005).

1.6. Eagle Point – Blue Rapids Enhancement Project: Restoration of Decommissioned Off-highway Vehicle Trails in Wet and Riparian Area Crossings

Blue Rapids Provincial Recreation Area (BRPRA) and Eagle Point Provincial Park (EPPP), located east of the town of Drayton Valley, Alberta together form the Eagle Point-Blue Rapids (EPBR) Park System. The Park System was established in 2007, following community encouragement, to help balance recreational, industrial and conservation uses in this area of crown land, and to prevent further severe environmental degradation (EPBRPS 2011). These two protected areas are cooperatively managed by Alberta Parks and the Eagle Point-Blue Rapids Parks Council (EPBRPC 2017). Eagle Point Provincial Park was created to provide non-motorized recreational activities, such as hiking, biking and skiing for the community and Blue Rapids Provincial Recreation Area was created to provide motorized recreational options (i.e., OHV use), as well as provisions for hunting (EPBRPS 2011) within a setting that also allows for industrial activities (primarily oil, natural gas, sand, and gravel extraction). OHV activity is the most common cause of continued disturbance on trails, although OHV use was not the original cause of disturbance. Most of the trails within BRPRA were created for activities related to the oil and gas industry, such as opening of paths for installation of pipeline, wells and seismic lines. Those paths are now used for motorized OHV recreation.

Blue Rapids Provincial Recreation Area has several kilometers of unmanaged OHV trails. Some of these trails traverse wet areas and watercourse features in the riparian ecosystem of the North Saskatchewan River. The current trail layout, condition and use of these trails have resulted in severe vegetation loss, erosion, compaction and sedimentation of these areas (EPBRPS 2012a). Furthermore, decades of recurrent OHV use in areas with vulnerable soils have led to very large swales that seasonally fill with water. These are of concern from an environmental perspective because: i) when they become unnavigable by most OHVs, the required circumvention of the swales causes widening of trails, and further loss of adjacent habitat; ii) when precipitation causes them to overflow it results in sedimentation into adjacent watercourses, water bodies and surrounding vegetated areas; and iii) they create new sub-optimal habitat that attracts aquatic and semi-aquatic fauna and flora, which may then be disturbed by future OHV use.

The Eagle Point-Blue Rapids Park System reclamation strategy encourages, guides and facilitates the conversion of disturbed sites into restored areas. The Parks Council is responsible for reclamation of disturbed areas impacted by recreational activities, such as OHV use, that took place both prior to the establishment of the Park System, and the ones that are still ongoing (EPBRPS 2012a). For disturbance originated from oil and gas activities, the Alberta Government (2013) requires reclamation of wellsites and associated facilities in forested lands and wetlands. Alberta Environment (2002) defines reclamation as “the process of reconverting disturbed land to its former or other productive uses”, whereas ecological restoration is considered “the process of assisting the recovery and management of ecological integrity”. Disturbed areas within the EPBR System that are under the Parks Council’s responsibility for reclamation are to be ecologically restored, rather than reclaimed. According to the Eagle Point-

Blue Rapids Parks System reclamation strategy (2012a), ecological restoration should aim to restore the system to one that is similar to the native vegetation community. Initially, woody pioneer species should be planted to act as a nurse crop, create a cool microclimate, loosen soils, reduce compaction and add organic matter (through litterfall) to the topsoil. Additional phased successional plantings of shade-tolerant tree species can be applied after four to six years and seven to fifteen years; however, that may not be necessary, as the first planting of woody species may be enough to initiate succession.

The Blue Rapids Wetland Enhancement Project was initiated in 2016 to apply ecological restoration activities on decommissioned OHV trails in Blue Rapids Provincial Recreation Area. Funding for the enhancement resulted from an initiative partnering the community, provincial and federal governments, and industry. The enhancement project activities included: i) preventing further vehicular access in the wet and riparian areas; ii) constructing crossing structures to avoid direct access of vehicles to sensitive wet and riparian areas; iii) eliminating swales through recontouring of the decommissioned wet and riparian trail crossing areas; and iv) re-establishing vegetation at wet and riparian trail crossing areas through a propagation and planting program. Trails that went through the enhancement activities are hereafter called 'enhanced trails'. Information on the specific enhancements applied to each sample crossing can be found on Appendix 1. The project also included an outreach component to engage and inform the community.

This thesis focuses on activity (iv) of the enhancement project: re-establishing vegetation at wet and riparian trail crossing areas through a propagation and planting program in Blue Rapids Provincial Recreation Area. The main goal of this study was to examine approaches to propagate and establish planted cuttings of a woody pioneer tree species characteristic of the vegetation community in the Park (i.e., balsam poplar) on previously disturbed wet and riparian areas; this was intended to initiate the process of ecological succession on a trajectory towards sites developing similar structure and function to the surrounding areas.

In Chapter 2, I evaluate the efficacy of three planting treatments for restoration with balsam poplar cuttings (i.e., direct plant, unrooted, and rooted) following recovery from OHV disturbance to determine how these treatments affected plant survival and height growth after controlling for initial diameter, soil penetration resistance, depth to water table, disturbance and damage. I also determined if the plant community composition differed on enhanced trails, adjacent edges and adjacent forested areas (10 m from edges), as well as if there were differences at each position within two years after trail enhancement.

In Chapter 3, I summarize the results of this study and present management implications and practices along with potential avenues for further research to help inform wet and riparian areas enhancement and OHV trail management.

Chapter 2. Restoration of trails degraded by off-highway vehicle use: performance of hardwood cutting treatments and effects of disturbance and restoration on plant community composition

2.1. Introduction

Protected areas are established and maintained in Alberta for the conservation, preservation, and management of flora, fauna, and natural features of scientific and cultural importance, while also facilitating their use and enjoyment for outdoor recreation and education (PPA 2000). Therefore, protected areas need to meet competing demands for recreation and conservation of ecosystems and biodiversity. Recreational activities occurring within these areas include hiking, jogging, biking, and off-highway vehicle (OHV) use (Charman and Pollard 1995; Jordan 2000). The use of OHVs is becoming an increasingly popular recreational activity in Alberta's recreational areas and on crown land (Monz et al. 2013).

OHVs use and presence of their associated vehicular trails in natural areas can result in changes in the physical and chemical properties of ecosystems and alterations in plant populations and community structure (Hannaford and Resh 1999; Jones 1999; Brooks and Lair 2005). They cause direct impacts (i.e., on trail), indirect impacts (i.e., off-trail, causing edge effects), and landscape impacts (e.g., fragmentation; Brooks and Lair 2005; Ouren et al. 2007; Trip and Wiersma 2015). OHV activity alters soil properties; in particular the repeated application of forces from OHV passes on the soil can cause soil compaction (Wilshire et al. 1978; Webb 1983), which reduces water infiltration and inhibits root growth (Hannaford and Resh 1999). Native species are damaged and thus reduced in abundance on trails (Hannaford and Resh 1999), this in turn facilitates the establishment of more competitive non-native species (Charman and Pollard 1995; Ouren et al. 2007). Non-native species can often easily occupy sites with disturbed soil in areas where they were not previously found (Webb et al. 1983; Brooks and Lair 2005). As a result, the vegetation community structure can be affected and plant community composition on and adjacent to the trail can differ from pre-disturbance conditions and from the surrounding area (Meyer 2002). Effects may also extend beyond the trail and edge into the adjacent forest, due to edge effects resultant from the flow of energy, nutrients and species across the edge towards the forest (Murcia 1995). These ecosystem changes are particularly problematic on wet and riparian areas, as the impacts caused by recreational vehicles are greater when the soil moisture is greater (Trip and Wiersma 2015). Wet and riparian areas are particularly sensitive to the effects caused by OHV use showing low resilience and long recovery times following recreational vehicular disturbance (Charman and Pollard 1995; Arp and Simmons 2012).

Blue Rapids Provincial Recreation Area (BRPRA), located along the North Saskatchewan River outside of Drayton Valley, Alberta, Canada, has supported OHV recreation for several decades; this, at times, poses a conflict with its objective to conserve its unique and essential natural heritage. BRPRA has several kilometers of unmanaged OHV trails, which were originally established as pipelines rights-of-way, seismic lines and access roads by the local oil and gas industry. Some of these trails traverse wet areas and watercourse features in the riparian ecosystem of the North Saskatchewan River. The current trail layout, condition and use of these trails have resulted in severe vegetation loss, erosion, compaction and sedimentation of the wet and riparian watercourse crossing areas (EPBRPS 2012a). For these reasons, the structure and function of these crossing areas are in the process of being restored. This can be achieved through active restoration activities, such as first ensuring that continued damage is stopped by decommissioning the impacted trail crossings, then recontouring the surface areas to previous conditions (e.g., removing swales), and finally revegetating the decommissioned trail crossings with native plant species (Kay and Graves 1983; Slaughter et al. 1990; Charman and Pollard 1995; Jones et al. 2018).

Ecological restoration activities are being applied in BRPRA through the Blue Rapids Wetland Enhancement Project, which was initiated in 2016. The project activities include: i) preventing further vehicular access in the wet and riparian areas; ii) constructing crossing structures to avoid direct access of vehicles to sensitive wet and riparian areas; iii) eliminating swales through recontouring of the decommissioned wet and riparian trail crossing areas; and iv) re-establishing vegetation at wet and riparian trail crossing areas through a propagation and planting program. Trails that went through the enhancement activities are hereafter called 'enhanced trails'.

This study focused on revegetation of the enhanced OHV trail crossings within Blue Rapids Provincial Recreation Area. One way to rapidly re-establish native vegetation on degraded areas is through planting hardwood cuttings (Jones et al. 2018). Cuttings are segments of live stems cut from donor plants, generally hardwood tree and shrub species that root easily. Balsam poplar (*Populus balsamifera* L.) is one of the primary tree species utilized for hardwood cuttings propagation in Alberta (DesRochers et al. 2004; Cows and Fish 2007; Polster 2013).

Balsam poplar is an early and intermediate successional hardy and fast growing deciduous broadleaf tree species that grows mainly on floodplain sites (Zasada and Phipps 1990). It reaches flowering age around 8 to 10 years-old, and trees live up to 200 years (Zasada and Phipps 1990). Balsam poplars can propagate via seeds and vegetatively (Zasada and Phipps 1990; Dickmann 2001). Vegetative propagation occurs through the natural establishment of buried branch fragments, or through the planting of hardwood cuttings (Zasada and Phipps 1990; Dickmann 2001). This is possible due to pre-formed root primordia, which are initiated on stems during the first growing season (Krasny 1988). This tree species is tolerant to flooding, and thus commonly found in wet and riparian areas (Krasny 1988; Braatne et al. 1996). Balsam poplars can also influence soil development when dominant at a site, causing an increase in forest floor nitrogen content and soil carbon (Zasada and Phipps 1990). Their stem cuttings have been used for rapid re-vegetation of degraded areas with the aim of returning wildlife habitat, stabilizing riparian soils and protecting soil structure (Stanturf et al. 2001; DesRochers et al. 2004). Because they are native to BRPRA, and in reason of their ability to root from stem cuttings, their fast growth rates, resistance to flooding, and associated benefits to soil structure; I decided to plant balsam poplars on this restoration project.

Cuttings can be collected during dormancy or during late spring/early summer (‘green’); they can be initially planted in the field, or can be first rooted in a greenhouse; variability in initial diameter is related to variability in their available carbohydrate reserves, which can also affect survival and growth (Dickmann et al. 2001; Zalesny Jr. et al. 2005; Tilley and Hoag 2008; Douglas et al. 2016). It is of interest to submit cuttings to different planting treatments, or propagation methods, in an attempt to see which treatments maximize their survival and growth, and also which type of cutting is best suited to the enhanced trails in BRPRA. As a result of near optimum growing conditions, mostly related to water availability and lack of stresses, survival and height growth are usually greater for cuttings initially planted in a greenhouse, in response to the development of root systems, when compared to those planted directly in the field (Schroeder and Walker 1991). However, few studies have compared performance of cuttings initially grown in the greenhouse and then transferred to the field with the performance of cuttings established as unrooted cuttings in the field (Schroeder and Walker 1991; DesRochers et al. 2004).

Edaphic factors can affect cutting establishment in the field. The development of the initial fine-root network can be hampered by high soil strength (Zalesny Jr. et al. 2005). As fine absorbing “feeder” roots are mostly concentrated in the top 10 cm of the soil profile (Dickmann et al. 2001), drought or water saturation on the soil surface might cause plant death (Dickmann et al. 2001; Zalesny Jr. et al. 2005).

This study focused on the restoration of decommissioned off-highway vehicle (OHV) trails in wet and riparian areas of Blue Rapids Provincial Recreation Area (BRPRA). The main goal of this study was to examine the establishment of planted cuttings of a woody pioneer species characteristic of the vegetation community in the Park (i.e., balsam poplar) on previously disturbed wet and riparian areas; this was intended to initiate the process of ecological succession on a trajectory towards wet and riparian crossing areas developing similar structure and function to the surrounding areas.

I first evaluated the efficacy of three pre-planting treatments for restoration with balsam poplar cuttings (i.e., direct plant from green cuttings, unrooted from dormant cuttings, and rooted from dormant cuttings) following recovery from OHV to determine how the treatments affected height growth and survival after controlling for initial diameter of the cutting and the abiotic variables soil penetration resistance and depth to water table. Since cuttings planted in a greenhouse are exposed to better conditions for root development, I hypothesized that the rooted treatment would exhibit the best survival and height growth as a result of established and more extensive root systems in the plants when they were transferred to the field. The unrooted treatment would present the second best results, since cuttings collected during dormancy would have higher carbohydrate reserves, whereas the direct plant treatments would be the least successful because carbohydrate reserves were used by the donor tree to leaf out during spring, before cuttings were collected. Cuttings with larger initial diameter were expected to have higher height growth than those with smaller initial diameter because of their larger amount of carbohydrate reserves. Considering that soil compaction inhibits root system growth, I hypothesized that soil compaction would have a negative effect on plant height growth and survival. And since the fine root system is established mostly in the top 10 cm of soil, I expected that either saturation or drought near the soil surface would lead to increased mortality and poorer growth of planted cuttings.

Second, I also determined if the trail, adjacent edge and adjacent undisturbed forest had different plant community compositions on enhanced OHV trails, as well as if there were differences at each position within two years after trail enhancement. Considering that wet soils are sensitive to the impacts caused by OHV use and that it may take a long time period for these areas to be fully recovered, I hypothesized that within locations there would not be a difference between plant community composition and diversity indices between the year planting treatments were applied (2017) and one year after treatment (2018). Considering that impacts caused by recurrent OHV use on trails can inhibit native plant growth and development instead promoting establishment of competitive non-native species, I hypothesized that there would be higher cover of non-native species on trails and in the edge than in the adjacent forest plant community. Since the plant community on trails has been disturbed by OHV use and thus differs from pre-disturbance conditions and from the surrounding area, as well as edges are directly influenced by the trail environment, I hypothesized that there would be a higher cover of early-successional graminoids and exposed soil on trails and edges, and that forested areas would have higher cover of native species, shrubs, forbs and litter.

2.2. Methods

2.2.1. Study Area

Blue Rapids Provincial Recreation Area (BRPRA) is located 13 km east of Drayton Valley, Alberta and 142 km southwest from the city of Edmonton, within the Dry Mixedwood and Central Mixedwood Natural Subregions of Alberta, Canada (Figure 2-1; AP 2015). BRPRA covers an area of 36.4 km². The area is ecologically significant, as it includes an area that is transitional between the Lower Foothills Subregion to the West and Central Parkland Subregion to the East. The park also functions as a protective buffer for a section of the North Saskatchewan watershed (EPBRPC 2017). The mean maximum temperatures in the growing season are 17.2°C in May, 20.6°C in June, 23.0°C in July and 22.0°C in August. Mean monthly precipitation during the growing season is 61.0 mm in May, 101.9 mm in June, 103.4 mm in July and 76.3 mm in August, with a mean annual precipitation of 550.6 mm (30 year climate normal

1981-2010, Entwistle station; closest station from BRPRA, 44 km away from the Park, with similar elevation, 780 m). Elevation ranges from 716 m to 876 m in BRPRA (EPBRPS 2012b). BRPRA soils are formed mainly by grey wooded forest soils of the Luvisolic order on the upland areas above the valley and soils of the Regosolic order on the active floodplain (EPBRPS 2011).

Six distinct terrestrial vegetation communities have been identified amongst the native vegetation: riparian forbs and shrubs; balsam forest; aspen forest; spruce forest; pine forest; and aspen – high bush cranberry – ostrich fern (assemblage of each classification is described in the document EPBRPS 2012b). The canopy includes *Picea glauca* (Moench) Voss., *Populus balsamifera* L., and *Populus tremuloides* Michx. Common forbs include *Cornus canadensis* L., *Equisetum hyemale* L., *Fragaria virginiana* Miller and *Solidago canadensis* L.. Common shrubs include *Rosa acicularis* Lindl., *Shepherdia canadensis* (L.) Nutt. and *Cornus sericea* L.; common graminoids include *Poa pratensis* L., *Calamagrostis canadensis* (Michaux) Pal. de Beau., and *Bromus pumpellianus* Scribner. The most common mosses in BRPRA are *Hylocomium splendens* (Hedw.) Schimp., *Pleurozium schreberi* (Brid.) Mitt. and *Ptilium crista-castrensis* (Hedw.) De Not. (EPBRPS 2012b). Introduced species observed include *Bromus inermis* Leysser, *Melilotus officinalis* (L.) Lam., *Tanacetum vulgare* L. and *Taraxacum officinale* F.H. Wigg.

BRPRA's land base is comprised of 15% water, 77% native vegetation and 8% disturbed areas. The most common disturbance types occurring in the park include industrial oil and gas facilities, access roads, and managed and unmanaged recreational facilities. BRPRA was established as a protected area in 2007. However, prior to the protected status, it held an extensive unmanaged off-highway vehicle (OHV) trail system that operated (and is still operating) for several decades. At present, the area remains a popular destination for recreational activities and OHV use for the residents of Central Alberta. This has resulted in unsanctioned OHV use and opening of new trails (EPBRPS 2011, EPBRPS 2012a). Blue

Rapids Provincial Recreation Area has decommissioned the use of some of these trails and is applying enhancement activities to restore their ecological function. The enhancement project activities have included: i) preventing further vehicular access in the wet and riparian areas through blocking their access; ii) constructing crossing structures to avoid direct access of vehicles to sensitive wet and riparian areas; iii) eliminating swales through recontouring of the decommissioned wet and riparian trail crossing areas; and iv) re-establishing vegetation at wet and riparian trail crossing areas through a propagation and planting program.

2.2.2. Experimental Design and Data Collection

We targeted a total of 15 enhanced crossings within Blue Rapids Provincial Recreation Area (BRPRA) for restoration (Figure 2-2, Appendix 2). We conducted experimental planting and sampled the vegetation during the summers of 2017 and 2018. The crossings targeted for restoration had access to OHVs blocked; eight of them had structures constructed to redirect OHV routes, such as bridges, and all of them were recontoured and blocked with the use of physical barriers (e.g. rocks, logs and nets). This way, it was possible to differentiate maintained trails from decommissioned enhanced trail crossings to be targeted for restoration. The enhancement activities took place from October 2016 until March 2017. All of the sampled crossings were previously disturbed by OHV activities, did not have significant natural regeneration, were located in wet or riparian areas, and were at least 11 m long and 2 m wide, in order to meet experimental requirements. Trails were mapped with the use of a handheld Geographic Positioning System (GPS) GARMIN GPSmap 62st.

Stem collection

On March 13, 2017, 1110 balsam poplar dormant stem cuttings, ranging from 40 cm to 100 cm in length and 2 mm to 9 mm in diameter at the terminal bud, were collected in Blue Rapids Provincial Recreation Area.

Stem collection occurred at four different sites within Blue Rapids Provincial Recreation Area (Figure 2-3) in order to exceed a minimum level of genetic diversity of the ecosystems at future restored sites, following the Alberta Forest Genetic Resource Management and Conservation Standards (FGRMS 2016). Each collection site was located at least 600 m apart from the other sites: site 1 and site 2 were 600 m apart from each other; site 2 and site 3, 700 m; site 3 and site 4, 900 m; site 1 and site 4, 2 km (Coordinates in Appendix 3). The minimum number of stems we collected at a site was 40 but the numbers mostly ranged between 100 and 250 stems collected per site, from a minimum of 10 different trees on each patch. This follows the collection requirements from the Alberta Forest Genetic Resource Management and Conservation Standards, which allows the collection of at least 10 plants per patch for the propagation of balsam poplar and shrubs in Alberta (FGRMS 2016).

Right after collection, stems were put into buckets with snow, to keep them cool during transportation. Collected stems were stored in freezers at the University of Alberta, North and Augustana campuses, at temperatures around -20°C inside plastic bags from March until June, when they were prepared and planted. Those were the procedures used for collection of dormant cuttings.

Green cuttings were collected in early summer, in the week of June 12, 2017, prior to planting, in the same collection sites as dormant cuttings. According to standard 19 of the FGRMS (2016) deployments without nursery and within nine months of the collection date do not need registration, and propagules (up to 5000) can be from one single genotype. When collecting greenwood, we followed the same minimum requirements followed for collection of dormant cuttings, but that is not necessary under the FGRMS. 'Green cuttings', were collected from green wood, which means the stems had leafed out in the beginning of spring, prior to collection. We collected about 50 stems at each site. Stems ranged from 40 to 100 cm in length and 2 to 6 mm in diameter. Some of the cuttings prepared from these stems had to be stored in the fridge at +5 °C after collection (before soaking) for a maximum of three days due to timing between collection, soaking and planting and the logistics in the field, but in general, stems were not stored for a long time after collection.

Treatments

Our experimental revegetation treatments involved planting of cuttings of balsam poplar (*Populus balsamifera* L.) and willows (*Salix* spp.). Cuttings are segments of live stems cut from donor plants.

For the willow treatment, *Salix* spp. stems were collected during dormancy on collection sites 1, 3 and 4; and stored in freezers at -20°C; after storage, on the week of June 12, 2017, they were cut into 25 cm cuttings and then soaked in water for 24 hours; these cuttings were planted directly into the sample trail crossings. Most of the willows did not survive, after eight weeks only seven plants were alive, while all the others (n= 143) were dead. We suspect the poor survival was due to the size and quality of the willows we collected, their stems were thin (ranging from 1.4 mm to 4 mm in diameter) and branched. The identification of dormant willows to species level at collection day was also challenging, which resulted in eight different unknown willow species being collected. Because of these challenges I decided not to include any further consideration of willows in the thesis.

The balsam poplar stem cuttings were used to examine three different planting treatments: unrooted, rooted, and direct plant. Stems submitted to the unrooted treatment were collected during dormancy (i.e. in the winter – March 2017) at collection sites 1, 2 and 3 and stored in freezers at -20°C. After storage, during the week of June 12, 2017, they were cut into 25 cm lengths and then soaked for 24 hours; these cuttings were then planted directly into the trail crossings targeted for restoration in Blue Rapids Provincial Recreation Area (i.e., in the field). Stems submitted to the rooted treatment were also collected during dormancy (March 2017) at collection sites 1, 2 and 3 and stored in freezers at -20°C. On June 12, 2017, they were cut into 10 cm cuttings and then soaked for 24 hours; these cuttings were then planted in a greenhouse at Bonnyville Forest Nursery where they were grown for approximately two months.

During the week of August 14, 2017, after development of roots and foliage, they were planted into the sample trail crossings in Blue Rapids Provincial Recreation Area. For the direct plant treatment, I used stems that had been collected at all four collection sites during the week of June 12, 2017. The side shoots and leaves were removed, buds were left on the stems, the stems were cut into 25 cm sections and then soaked for 24 hours. After soaking, these cuttings were planted directly into the sample crossings in Blue Rapids Provincial Recreation Area.

Cutting preparation and planting followed previously developed protocols. Cuttings were manually prepared with hand pruners, to ensure they had a vegetative bud located within 1 cm of the tip (DesRochers et al. 2004). Some of the longer stems (e.g. 100 cm long) were divided into multiple cuttings (Stanturf et al. 2001). Stems were cut into 25 cm or 10 cm cuttings depending on the treatment. Cuttings that were planted directly in the field were cut longer than cuttings initially planted in the greenhouse because DesRochers and Thomas (2003) found that 10 cm cuttings had good rooting success when planted in a greenhouse, whereas longer cuttings contain larger carbohydrate reserves which would benefitgrowth in the field. Schroeder and Walker (1991) used 25 cm and Zalesny Jr. et al. (2005) used 20 cm cuttings in their field trials. Longer cuttings are less likely to dry in non-irrigated conditions because they can hold more water and their aboveground area exposed to air is proportionally smaller (DesRochers and Thomas 2003). For the cuttings that were going to be first planted directly in the field, the base of each cutting was cut at a 45° angle to create a greater surface area in order to facilitate nutrient and water absorption (Woods 2011). All cuttings were soaked for 24 hours at room temperature, placed vertically inside laboratory bottles with cold tap water covering three-fourths of the cutting length, until the moment they were planted. This was done because soaking hardwood cuttings in water prior to planting (post-storage) increases short-term survival rates by effectively hydrating vascular tissues, which may translate to increased biomass production (Phipps et al. 1983, Stanturf et al. 2001, Schaff et al. 2002, DesRochers et al. 2004, Tilley and Hoag 2008).

The experiment was set out in a block design. Each of the 15 wet or riparian crossings was treated as an experimental unit (block) with cuttings of different treatments planted within each block. Cuttings from the three balsam poplar treatments were planted in separate 1.5 x 1.5 m plots in the crossings; at least two plots of each treatment were put in almost all crossings. Individual plots had five cuttings (all from the same treatment). All plots had cuttings that originated from different collection sites to ensure genetic diversity of balsam poplar cuttings within a plot. Cuttings were separated by 1.06-1.5 m distance within a plot (Figures 2-4 and 2-5). Plots were positioned 1 m apart from each other and 1 m away from the edges of the trail. A random number generator was used to decide the position of treatments on a specific crossing. Figure 2-6 represents the planting layout.

Sampled crossings did not have uniform conditions and characteristics. They differed in size, location, and presence or absence of water features. Appendix 1 contains details of crossings, treatments applied, and total number of cuttings planted on each crossing.

Deployment

Rooted treatment cuttings were first planted in a greenhouse at Bonnyville Forest Nursery. The cuttings were planted vertically into seven Styroblock 512A 60/220 trays. Each tray housed a maximum of 60 cuttings. The trays were spray painted and labeled to indicate the site where cuttings were collected from. These cuttings were grown in the greenhouse for two months (from June 12 to August 19, 2017), until the roots were developed. They were grown with natural light and photoperiod for the summer of 2017. Day and night temperatures were not controlled but followed environmental conditions. These cuttings were planted at the sampled crossings in BRPRA at the end of the growing season, in the week of August 28, 2017.

Deployment of the two unrooted cutting treatments (unrooted and direct plant) in the field occurred in the sample crossings within Blue Rapids Provincial Recreation Area (BRPRA) during the week of June 12, 2017. A total of 690 cuttings from all treatments were planted in the experimental units in BRPRA. Holes of approximately 25 cm depth and 20 cm width were dug with the use of shovels; we did this because digging holes with extra space around the plant can facilitate plant growth and root development (Marlow 2013). After digging the holes and prior to planting, the soil surrounding each cutting was prepared to make sure the soil was loose enough and also to remove excessive and large rocks that could hamper root development; during planting, soil was worked in order to avoid air pockets (Marlow 2013, Douglas et al. 2016, McCarthy et al. 2017). The cuttings were placed vertically in the ground with approximately 80% of their total length below the soil surface. We did this based on Polster (2013) who recommended at least 75% of the total length of cuttings should be below soil, but in drier conditions, or when water regime cannot be controlled, this should be increased to approximately 90%. Approximately 5 cm of each cutting was sticking out of the ground and one bud was left above the soil surface for most balsam poplar cuttings; in some cases where the soil could not handle a 20 cm depth hole (e.g., presence of rocks or high compaction) more than one bud was left above ground.

Monitoring

Following planting, a monitoring system was established. Trail crossings and plots were marked with metal tree tags and identified by number and letter codes. Monitoring also entailed site documentation through the use of photographs and field notes to document restoration success and unsanctioned OHV use on the sampled trail crossings.

Survival of planted cuttings (alive/dead status) was recorded at the end of the first growing season (last week of August 2017) and at the beginning (first week of June 2018) and the end of the second growing season (last week of August 2018). However, given the vegetative growth of balsam poplar, a plant that looked dead previously could re-sprout from the roots or from another bud in the stem. When this occurred, the plant was considered as alive for all time periods.

Cuttings had height zero at planting day, as the buds where shoots would emerge from were the baseline from which future height growth was measured. Initial diameter of each cutting was measured at the base of the terminal bud on the planting day using calipers. Height growth of the shoots emerging from the cuttings was measured at the end of the first growing season (last week of August 2017) and at the beginning (first week of June 2018) and the end of the second growing season (last week of August 2018).

Disturbances or stresses that could be influencing a cutting were visually observed in the planting area (i.e., a 20 cm radius area around each cutting) and recorded in categories as follows: off-highway vehicle (OHV) use (visual signs of wheels); flooding; competition (when other plants were covering more than 50% of the 20 cm radius area around each cutting); animal trampling; herbivory; fallen tree; and none observed (when no obvious disturbance was visually observed around the cutting). When more than one disturbance type or stress was observed around a cutting, the one with the highest influence observed was selected as a factor. I also recorded damage level for each cutting using a scale based on proportion of the cutting damaged: undamaged (0-10% damage); lightly damaged (11-50%); severely damaged (51-100%); and dead. For data analyses I reduced damage to two categories: undamaged (0-10% damage) and damaged (11-100% damage). Damage included disturbance to the stem and leaves, including absence of leaves.

Unsanctioned OHV use occurred in seven of the 15 sampled crossings (Appendix 3). These crossings were excluded from data analyses, since most of the cuttings were dead. From the eight remaining crossings, two were adjacent, or had part of their area adjacent, to active recreational trails, which resulted in influencing impacts from OHV use, such as soil movement and wheels passing near cuttings. Two crossings suffered isolated unsanctioned OHV use. Because these two crossings were protected by barriers, the access was not easy for the riders, that is why it seems that those were isolated events in which most cuttings were not affected, but a few cuttings in the crossings incurred some level of damage.

I measured soil penetration resistance (i.e. soil compaction) using an AMS pocket penetrometer (range from 0 to 4.5 kg/cm², divided into 0.25 kg/cm² increments). Measurements were taken around the planting area of each cutting (outside and inside the “digging hole” area) in 2018, to analyse the effects of local compaction on each cutting. For the vegetation survey (described below), measurements were taken in each 1 m x 1 m quadrat at the trail, edge and 10 m locations, as close to plot centre as possible.

In June of each year I installed 60 cm lengths of rebar on each trail crossing to record the groundwater table level. Three rebars were installed on regular sized (that included up to 10 plots) trail crossings and four rebars were installed on longer crossings (with more than 10 plots). Rebars were placed in between the first two plots, the middle plots and the last two plots of each crossing. I removed the rebars at the end of each summer and measured the distance from the soil surface to the rusted area on the rebar as an indicator of the water table depth.

Vegetation community sampling

Vegetation sampling was conducted to examine changes in vegetation communities between disturbed area (i.e., trail), the immediately adjacent area (i.e., edge), and adjacent forest (i.e., 10 m perpendicular from the edge) on the 15 sample crossings targeted for restoration within BRPRA during the summers of 2017 and 2018. Sampling occurred the last two weeks of July and the first week of August, to ensure vegetative growth and structure development (e.g. flowers, fruits or seedheads on graminoids) on plants had peaked. Two transects were placed on each crossing: one at 1 m above the lowest relief (i.e., the wetter area); and one at the highest relief, or less wet area. Each transect was placed on trail and then extended from the trail edge into the forest. A coin was flipped in order to decide which direction to place transects. Quadrats (1 m x 1 m) were placed on each transect at three sample points: trail; edge; and 10 m from the edge (Figure 2-7).

The vascular plant community (i.e., shrubs, forbs and graminoids – see Appendix 4 for detailed list) was sampled using 1 x 1 m quadrats located at each of the sample points. Percent cover (0-100) of each species/taxon was estimated, by a single observer, to the nearest 1/10th percent for species for less than 1% cover and to the nearest 1% for species with more than 1% cover.

Species identification was done in the field when possible, with the use of the field guide Johnson et al. (1995). I collected and pressed a sample of each plant species encountered. For species that could not be identified in the field, voucher specimens were collected for identification in the lab, with the use of the keys Moss and Parker (1983), Royer and Dickinson (2007), and through comparison with University of Alberta herbarium samples. Species were also double checked by plant specialists at the University of Alberta in order to confirm identification. Species scientific names were confirmed using the USDA plants (website: http://plants.usda.gov/) and the VasCan (website: http://data.canadensys.net/vascan/) databases.

A few samples could not be identified at the species level, either because they were too small or because they were not fully developed (e.g. lack of flowers, fruits or seedheads on graminoids). Although these samples could not be identified they could be differentiated with confidence from all other species in the plot; thus they were considered as different species and included in the analyses at the genus level or as “unknown species”.

Cover values for non-vascular plant species (forest floor mosses), forest floor lichens, and abiotic categories (mineral soil, rock, litter and wood) were also recorded in the 1 m x 1 m quadrats. Mosses were considered along with abiotic cover values. Although some forest floor lichens and mosses could be tentatively identified at the species level, because I was not confident in their identification, I excluded them from diversity and richness calculations.

2.2.3. Data Analyses

I used R 3.5.1 (R Core Team 2018) for all statistical analyses except for the plant community composition multivariate statistics where I used PC-ORD (Version 7.04 MjM Software Design, Gleneden Beach, OR).

Planting treatments

I was interested in identifying which factors were important in determining survival and growth of the balsam poplar, so I used a model selection approach (described below). I included in the analysis all cuttings planted in experimental crossings not affected by continuing unsanctioned OHV use (n=8 crossings, with N=62 plots within them), although a few of the included crossings had some minor OHV influence (described earlier in methods).

Effects on survival were analyzed for three separate time periods: survival over the first summer (Spring to Fall 2017; n=195 cuttings), over the first year (Spring 2017 to Spring 2018; n=340 cuttings), and through the end of the second growing season (Spring 2017 to Fall 2018; n=340 cuttings). The number of cuttings is smaller for summer 2017 than 2018 because rooted cuttings were not planted in the field in the first summer.

Effects on height growth were analysed for the first growing season (2017) for unrooted and direct plant treatments (n=133 cuttings) and the second (2018) growing season (Height Fall 2018 – Height Spring 2018) for all treatments (n=137 cuttings). Dead and severely damaged plants (with negative height growth) were excluded from analyses for height growth (n= 62 in 2017; n= 142 in 2018).

The variables treatment, basal diameter at planting day, penetration resistance in and outside planting area, damage (undamaged and damaged or stressed), disturbance type, and depth to water table were potential explanatory variables. Initial diameter was included when significant.

In order to make the decision towards which model to select amongst several candidate models, the one that best describes the data and the outcome should be given preference. The model selection based on Akaike Information Criterion (AIC, AICc for small sample sizes) estimates the information loss when comparing the probability distribution associated with the true generating model to the probability distribution associated with the model to be evaluated (Burnham and Anderson 2002; Wagenmakers and Farrell 2004). According to Akaike (1973) when choosing the model with the lowest AIC value we are also choosing the model with the lowest information loss, and consequently the one that has higher probability to explain the observed data. Although, when two models have a Δ AIC (*model*AIC - *bestmodel*AIC) equal or lower than 2, they are considered equivalent, so Akaike weights can be used in the decision. Because the AIC is an unbiased estimator of minus twice the expected likelihood of the model, we can obtain an estimate of the relative likelihood of a model by transforming ΔAIC into Akaike weights, the formula is: exp(-0.5 ΔAIC) of a model divided by the sum of exp(-0.5 ΔAIC) across all models. Akaike weights are the probability that a model is the best model (i.e., ranges from zero to one) (Wagenmakers and Farrell 2004). To examine which explanatory variables influenced survival and height growth I used the exhaustive screening approach, without interactions, and AIC value corrected for small sample sizes (AICc) as the Information Criterion (Burnham and Anderson 2002). I chose the five best models for each response variable, based on the smallest AIC value and their relative Akaike weights, with the use of the glmulti package (Calcagno 2013). I reran all top five models selected by glmulti in order to compare their AICc values and Akaike weights and confirm which was the best overall model. I visually assessed the residuals and distributions of the final best model for each response variable to ensure assumptions of normality, homogeneity of variance and linearity were met. To examine which explanatory variables best explained survival I used generalized linear mixed-effects models (glmer function, lme4 package, Bates et al. 2018). To examine which variables best explained height growth I used linear mixed-effects models using the lmer function in the lme4 package (Bates et al. 2018). I square-root transformed the response variable height growth 2018 to meet the assumptions of normality.

For the survival and height growth analyses crossing and plot within crossing were considered random effects. Penetration resistance outside of planting hole in 2017 and 2018 were too correlated to be added to the same model, so I used the average of the 2017 and 2018 values.

For the top AICc model, I estimated population marginal means (Searle et al. 1980) with the emmeans package (Lenth et al. 2018; default settings: p-value adjustment using the Tukey method, and Kenward-Roger method for degrees-of-freedom; $\alpha = 0.05$) to compute contrasts between the treatment groups in pairwise post-hoc tests (Lenth 2018).

Plant community

I initially used multi-response permutation procedures (MRPP) to test for statistically significant differences in community composition (McCune and Grace 2002) between groups. I compared the crossings with unsanctioned OHV use along with the ones with no OHV use, they were not significantly different (p= 0.054) and were not indicated as an appropriate grouping variable (A= 0.004), so I combined enhanced crossings both with and without unsanctioned

OHV use for the vegetation community analyses. I compared transects in wetter and drier locations, they were also not significantly different (p= 0.07) and were not a good grouping variable (A= 0.001). “Wetter” and “drier” transect locations within a crossing did not present difference in wetness, as all crossings were located in wet and riparian areas. So for the multivariate analyses I averaged the community data for the two plots at a given position from the two transects (wetter and drier) at each crossing.

I used linear mixed effects analysis of variance (ANOVA) models (lme function in the package nlme; Pinheiro et al. 2016) to test for differences among positions (trail, edge, 10 m from edge) and years (2017, 2018) for both abiotic and biotic variables. Abiotic response variables included depth to water table, penetration resistance, cover of mineral soil, rock, litter, wood, and mosses. In a few sample crossings OHV users removed the rebars we used to measure depth to water table, which resulted in missing values, so I used the mean depth to water table value of the year to fill those missing values. Biotic variables included: cover of shrubs, forbs, mosses, native species, non-native species, and total vegetation; richness (per plot) of shrubs, forbs, graminoids, and total species; and Simpson diversity (Magurran 2004) of shrubs, forbs, graminoids and all plants. Position (trail, edge and 10 m) and year (2017 and 2018) were considered fixed variables while crossing was considered a random effect. I visually assessed normality and homogeneity of variances of residuals for all models. Log and square root transformations were applied when necessary, residuals were checked again after transformation. When position or the position*year interaction was significant (α=0.05) I used the emmeans package (Lenth et al. 2018; default settings: p-value adjustment using the Tukey method, and Kenward-roger method for degrees-of-freedom; α = 0.05) to conduct pairwise post-hoc tests as contrasts between trail, edge and 10 m for positions, between 2017 and 2018 for years and among positions within years for the interaction.

I used non-metric multidimensional scaling (NMS) unconstrained ordination to examine the patterns in the plant community composition among the three different sampled trail positions (trail, edge, 10 m into adjacent vegetation) (McCune and Grace 2002). I excluded species that occurred in only one plot; the main matrix contained 120 species. I used PC-ORD for ordination, with Sørenson (Bray-Curtis) as the distance measure. A matrix of composition dissimilarities is used for multivariate ecological abundance data collected at different sampling locations to quantify the difference between samples (Greenacre and Primicerio 2013; Faith et al. 1987). The nature and strength of the relationship between values of the dissimilarity measure and its corresponding ecological distances will indicate patterns in the data (Faith et al. 1987). I used the Bray-Curtis dissimilarity measure for distance as it is robust in terms of rank and linear correlation, and a good option for ordinations (Faith et al. 1987). I completed 250 runs with real data and 250 Monte Carlo randomized runs, ranging from a six-dimensional to a one-dimensional solution. I evaluated the scree plot and the stress to determine the number of dimensions for the final solution (McCune and Grace 2002) and ran a final NMS with the number of dimensions determined from the preliminary runs (n=3). I then calculated Pearson correlation coefficients for species and environmental variables (cut-off $R^2 > 0.3$). Environmental variables included water table, penetration resistance, mineral soil, rock, litter, wood, total bare ground, in addition to cover of native and non-native species, fungi and lichen, cover, richness and diversity values for shrubs, forbs, graminoids and total plants.

I used permutation-based MANOVAs (perMANOVA; PC-ORD) to test for statistically significant differences in community composition according to position, year, and their interaction. 4999 randomizations were used and significance was based on the proportion of randomized trials with a response value greater or equal to the observed response value. When the perMANOVA had a significant result (it did for position), I ran follow up post-hoc pairwise comparisons among predictors (trail, edge and 10 m) following the same procedures as for the

ANOVAs using Bonferroni-adjusted α values (family-wise α = 0.05): comparisons among positions α = 0.016 (i.e. 0.05/3).

I used Indicator Species Analyses (Dufrêne and Legendre 1997; PC-ORD) to determine which species were associated with each position for both years together (as the community composition did not differ among years; p= 0.067), using 4999 permutations in the Monte Carlo test of significance; species that had an indicator value >20 and were significant at α = 0.05 were included.

2.3. Results

<u>*Planting treatments*</u>

Survival

The top models (Δ AICc < 2) for survival to the end of the first growing season included two to four explanatory variables (Table 2-1); the best overall model to explain survival to the end of the first summer included treatment, initial diameter and penetration resistance outside the planted cutting. In this time period the unrooted treatment was more likely to survive than the direct plant (rooted treatment not included). Despite being included in the top model, treatment, initial diameter and penetration resistance did not have a significant effect on survival, although treatment and initial diameter had a nearly significant effect (Table 2-2). Initial diameter of the cutting had a positive influence on survival. Penetration resistance had a possible positive influence. At the end of the first growing season (2017), mean survival of the field planted cuttings was not significantly different for the unrooted treatment compared with the direct plant. While not tested statistically, the cuttings planted in the greenhouse had similar survival numbers to the unrooted treatment, and consequently were also similar to direct plant (Figure 2-8a).

The top models (Δ AICc < 2) for survival over the first year included three to six explanatory variables (Table 2-1); the best overall model included treatment, disturbance and penetration resistance outside the planting hole. In this period, the rooted and unrooted treatments were more likely to survive when compared to direct plant. Plants influenced by competition, trampling, herbivory or no observed disturbance were more likely to survive, respectively, when compared to those influenced by OHV use. Penetration resistance had a nearly significant positive effect on survival (Table 2-2). Survival over the first year, when all plants were in the field, was lowest for the direct plant treatment, with intermediate survival rates in the unrooted treatments, and rooted having the highest survival, all three treatments were significantly different (Figure 2-8b; Table 2-3).

The top models (Δ AICc < 2) for survival over the entire study period (Spring 2017 through Fall 2018) included three to four explanatory variables (Table 2-1); the best overall model included treatment, disturbance and penetration resistance outside the planting hole area. The rooted and unrooted treatments were more likely to survive when compared to the direct plant treatment (Table 2-2). Plants influenced by competition, herbivory and without evidence of disturbance were more likely to survive when compared to the ones influenced by OHV use. Penetration resistance did not have a significant effect on survival during the second growing season, but it had a possible positive influence (Table 2-2). By the end of the second growing season (2018), mean survival of the three treatments all remained significantly different from one another (Figure 2-8c; Table 2-3).

Height growth

The top models (Δ AICc < 2) to explain height growth in the first growing season included three to four explanatory variables (Table 2-1); the best overall model included treatment, initial diameter and penetration resistance in the planting holes area. During the first summer, unrooted cuttings had higher height growth in the field, as compared to the direct plant treatment. Cuttings with higher initial diameter had greater height growth. Penetration resistance in the planting area was negatively associated with height growth (Table 2-4). When height growth was compared between the two treatments that were planted in the field during the first growing season (unrooted and direct plant), while controlling for initial diameter and penetration resistance in planting area, they were significantly different (Table 2-5). Mean height growth was significantly higher for the unrooted treatment compared with the direct plant treatment (Table 2-5). While not statistically compared, cuttings planted in the greenhouse had mean height growth of 45.9 cm ($\pm$ 1.1 cm, n= 81) in the same time period (Figure 2-9a).

The model that best explained height growth in the second growing season (2018) (Δ AICc < 2) included damage and initial diameter, but not treatment. Damaged plants had lower height growth as compared to undamaged plants. Plants with higher initial diameter had greater height growth (Table 2-4). When height growth in the second growing season was compared among all three treatments, controlling for damage level and initial diameter, the treatments were not significantly different (Table 2-5). The average height growth (absolute, not correcting for initial diameter) in the second growing season appeared to be higher for the unrooted treatment compared with the rooted and direct plant treatments, but treatments were not significantly different from each other (Figure 2-9b; Table 2-5).

At the end of the first growing season (2017), more than 50% of the alive plants had initial diameter on planting day between 4 and 8 mm; 75% of the dead plants had initial diameter between 2 and 4 mm (Figure 2-10a). Height growth was positively correlated to initial diameter at the end of the first and second growing seasons (Figures 2-11 a and b).

Looking at the number and percentage of plants in different damage categories in the second growing season, the direct plant and unrooted treatments had a higher number of dead plants in comparison to other damage levels. In contrast, the rooted treatment had more lightly and severely damaged plants, when compared to dead plants (Figure 2-12). The disturbance type with higher frequency and proportion of dead plants was OHV. Competition and herbivory had a more balanced distribution of damage levels when compared to OHV (Figure 2-13). “None observed” was the most frequent disturbance type among all treatments, disturbances influenced all treatments in similar frequencies and proportions (Figure 2-14).

Plant community

Total vegetation cover was lower on the trail than at the edge or in the forest in both years but there was evidence of recovery one year after the restoration activities in terms of an increase in total vegetation cover on the trail and at the edge (Tables 2-6, 2-8; Figure 2-15a). Trail plots had a much higher percent cover of non-native species (Tables 2-6, 2-8; Figure 2-15b) than edge and 10 m plots while the opposite was true for cover of native species (Tables 2-6, 2-7; Figure 2-15c). The cover of shrubs on trail and edge was similar, but it was higher 10 m from the edge. There was a decrease in shrub cover from 2017 to 2018 at all positions (Tables 2-6, 2-7; Figure 2-16a). Forb cover was higher on the trail when compared to the edge and 10 m locations (Tables 2-6, 2-7; Figures 2-16b). Graminoid cover was higher at the edge than 10 m from the edge while the trail had intermediate cover and did not differ from the other two positions (Tables 2-6, 2-7; Figure 2-16c).

Total species richness was higher at the edge when compared to trail and forest, with evidence of increase one year after the restoration activities (Tables 2-6, 2-7; Figure 2-17a). Shrub richness was similar in the edge and forested areas and lower on trails, but it did not differ among years (Tables 2-6, 2-7; Figure 2-17b). Forb richness was higher at the edge, when compared to trail and forest, and it increased in all three positions from 2017 to 2018 (Tables 2-6, 2-7; Figure 2-17c). Graminoid richness differed among all three positions, edges had the highest richness of graminoids, followed by trails, and forested areas had the lowest. Graminoid richness increased substantially on edge and trail from 2017 to 2018 (Tables 2-6, 2-7; Figure 2-17d), but on forested areas this increase was similar to the pattern observed for increases in total species, shrub and forb richness. Species diversity (all vascular plants) was similar on trails and forested areas and it was higher at the edge. There was an increase in diversity of all vascular plants one year after the restoration activities (Tables 2-6, 2-7; Figure 2-18a). Simpson diversity of shrubs was lower on trail and similar on edge and forest; there was no difference between 2017 and 2018 (Tables 2-6, 2-7; Figure 2-18b). Diversity of forbs was highest at the edge, trail and forest had similar lower diversities, and there was not a difference between years (Tables 2-6, 2-7; Figure 2-18c). Graminoid diversity was lower on forested areas, and similar among edges and trails; it increased in all positions from 2017 to 2018, with substantial increase on edges and trails (Tables 2-6, 2-7; Figure 2-18d).

The perMANOVA of understory vegetation composition showed that there was no significant interaction between position and year ($p = 0.97$) and no significant difference among years ($p = 0.067$) but there were significant differences in community composition among the positions (perMANOVA; $p = 0.0002$). Post-hoc tests showed that all positions differed from each other (10 m vs edge $p = 0.0008$; 10 m vs trail $p = 0.0002$; and edge vs trail $p = 0.0002$). The NMS 3-dimensional solution (final stress = 17.9 after 158 iterations) explained 65.6% of the variation in the dataset for 2017 and 2018 together. The unconstrained ordination illustrated the separation among the three positions (trail, edge and 10 m), and which variables were associated with the different positions (Figure 2-19). Cover of mineral soil was negatively correlated with axis 1 and positively associated with the trail plot locations, while litter, shrubs (cover, diversity and richness), native, non-native, graminoids and total vascular plant cover were positively correlated with axis 1 and associated with edge and 10 m plots. None of the measured variables appears to be correlated with axis 2, whereas non-native cover and graminoids were negatively correlated with axis 3, which did not separate among the three trail positions.

The indicator species analysis revealed significant indicator species for each of the different positions: trail, edge and 10 m (Table 2-9). The trail included five indicator species: one woody (*Populus balsamifera*), three forbs (*Melilotus albus, Plantago major*, and *Trifolium* *hybridum*) and one graminoid (*Juncus bufonius*); three of these (*Melilotus albus, Plantago major*, and *Trifolium hybridum*) were non-native. Edge had one shrub (*Symphoricarpos occidentalis*), seven forbs (*Achillea millefolium, Anemonastrum canadense, Equisetum arvense*, *Symphyotrichum ciliolatum, Solidago canadensis*, *Taraxacum officinale*, and *Vicia americana*) and two graminoids (*Bromus inermis* and *Bromus pumpellianus*) as indicators, two of these (*Bromus inermis* and *Taraxacum officinale*) were non-native. The 10 m position had one tree (*Picea glauca*), three shrubs (*Cornus sericea*, *Rosa acicularis*, and *Rubus idaeus*), and six forbs (*Aralia nudicaulis*, *Eurybia conspicua*, *Maianthemum stellatum*, *Petasites frigidus* var. *palmatus*, *Rubus pubescens* and *Viola renifolia*) included as significant indicator species, all native.

There were important differences in abiotic environment between trail, edge and forest. Ground cover of mineral soil and woody debris had effects of position and year on them. Mineral soil cover was different among the three positions, with trails presenting more exposed mineral soil, followed by edges, and forest presented the lowest exposed mineral soil cover. There was a decrease in exposed mineral soil from 2017 to 2018. Wood cover was higher on forest, followed by edge, and lowest on trails, presenting an increase from 2017 to 2018. Rock, litter and moss ground cover had effects of position only. Rock cover was similar at forest and edges and higher on trails. Litter cover was similar on forest and edge and substantially lower on trails. Forest floor mosses cover was lower on trails than at the edges or in the adjacent forests (Tables 2-6, 2-7; Figures 2-20 a and b). Depth to water table was significantly higher in 2017 than in 2018 (Tables 2-6, 2-7; Figure 2-21a). Across both years soil penetration resistance on the trail was significantly higher when compared to the edge and to 10 m into the forest; the latter two positions did not differ (Tables 2-6, 2-7; Figure 2-21b).

2.4. Discussion

Survival and height growth of balsam poplar cuttings

This study increased our understanding of the factors that influence survival and height growth of balsam poplar cuttings planted in decommissioned OHV trails in wet and riparian areas. The insights here obtained can also be relevant to the oil and gas industry, since originally the trails were pipeline right-of-way, seismic lines, or access roads. The reclamation of sites disturbed by oil and gas activities after activity is stopped is required by the Alberta Government (2013), so companies might also use information on options proposed here to reclaim these areas. While previous studies focused primarily on the use of cuttings collected during winter dormancy and either grown in the greenhouse or planted as unrooted cuttings in the field (Schroeder and Walker 1991; DesRochers et al. 2004; McCarthy et al. 2017), my study extended this to include the use of unrooted cuttings collected during early summer post-dormancy and immediately planted. The results of this study revealed higher survival for the rooted treatment, followed by unrooted, and the direct plant treatment presented the lowest survival, at the end of the second growing season. Height growth was influenced by treatment during the first growing season, with unrooted presenting the best field performance. Surprisingly, treatment did not have an effect on height growth during the second growing season, which might reflect the influences that disturbances and environmental conditions have on plant growth.

While new insights have been gained from this study, we are cautious about the results being generalized, as in this study planting occurred in one year (2017). Thus, these results might reflect the specific conditions of that year. According to Ceulemans and Deraedt (1999) field performance in poplars varies depending on climatic conditions. The times of bud set and bud break, and leaf development and leaf drop, are linked to environmental (i.e., regional) and climatic (i.e., weather) conditions (Ceulemans and Deraedt 1999). The variances caused by environmental adaptation are not of concern here, as the cuttings were planted within the same area where they were collected from. However, because planting occurred in only one year, the effects of weather conditions were not taken into consideration.

The study results supported my hypothesis that the rooted treatment would result in the best survival, followed by the unrooted treatment, with the direct plant treatment having the lowest survival rates. Most of the mortality occurred during the first growing season, in which rooted (presenting greenhouse mortality) and unrooted (presenting field mortality) treatments performed similarly, despite their different growth environments, and direct plant had the highest mortality.

It is important to reinforce that all rooted plants were transferred to the field at the end of the first growing season, which means they were all alive in the field at this time period. In this sense, the initial mortality of these plants occurred during the greenhouse stage, so during the period of highest mortality rates, rooted cuttings were not established in the field yet. Then, survival rates over the first year indicate a drop from 100% to about 85% for the rooted treatment, which is similar to the mortality rates experienced by the other treatments. The rooted treatment, then, has the advantage of skipping one season in the field and starting with 100% survival after the first growing season, which results in higher survival rates at the end of the second growing season. Another advantage of the rooted treatment is the size needed for cuttings; cuttings initially planted in a greenhouse can be 10 cm long, in contrast to 25 cm long cuttings for the ones initially established in the field. In this case, less plant material needs to be collected to produce rooted cuttings. It is also easier to use basal and central sections of the original branches, and to avoid using the tips, when preparing these cuttings (the advantages of that are described further in the paragraph about initial diameter below).

The overall survival of unrooted cuttings to the end of the second growing season was just over 50%; while this was much lower than the approximately 75% achieved by the rooted treatment, the unrooted cutting treatment still performed reasonably well, considering 50% as a threshold for acceptable survival rate on restored wet and riparian areas (Sweeney at al. 2002). Survival of the direct plant treatment over the two growing seasons was much lower, at just over 25%, but this difference in survival could be offset by densely planting direct plant cuttings in the field in order to reach target balsam poplar tree densities in the future. The direct plant treatment is more practical and less expensive to apply than the unrooted and rooted treatments. The regulations for deployment of direct plant cuttings are also less restrictive for this type of transplant material under the Alberta Forest Genetic Resource Management and Conservation Standards (FGRMS 2016). Therefore, these benefits can also be considered when making the decision towards using direct plant rather than the other two treatments.

Rooted treatment presented better survival and lower proportion of dead plants in relation to other damage levels, which suggests resilience to environmental stresses. Rooted cuttings experienced optimal conditions during the rooting phase (Schroeder and Walker 1991). These plants produced roots in a greenhouse during the first summer, before being planted in the field. Root extension plays a major role in plant survival, as plants with more extensive root systems will have better access to water and soil nutrients as compared to ones with shorter root systems (Douglas et al. 2016). In contrast, greenhouse-grown plants incur stresses after transfer to the field that were not present in the greenhouse environment (e.g., competition, weather fluctuations, nutrient depression), which could result in longer-term mortality (Gil-loaiza et al. 2016). Considering that plants can physiologically adjust to environmental parameters or stresses (Dickmann et al. 2001), greenhouse rooted plants might be less able to tolerate those stresses than the ones that developed roots in the field. In their study, Schoonover et al. (2011) did not find differences in performance between plants rooted in the greenhouse and transferred to the field and plants initially rooted in the field.

The planting of unrooted balsam poplar cuttings directly in the field has shown to be effective for obtaining high survival rates in previous studies (Wilkinson 1999; DesRochers et al. 2004; McCarthy et al. 2017). The differences in survival between the two initial field treatments, unrooted and direct plant, can potentially be explained by the dynamics of carbohydrate storage in poplars. Cuttings from both treatments can form adventitious roots but their ability to form roots and grow after planting depends on carbohydrate reserves, which build up in late summer, are highest in the fall and are depleted in spring due to leaf-out (Dickmann et al. 2001). That means dormant cuttings have an advantage over the direct plant treatment, in which cuttings were collected and planted in early summer, when their carbohydrate reserves would have been at their lowest. Other than the possible influence of carbohydrate reserves, which was not specifically measured, cuttings from the direct plant treatment were collected from greenwood; in that stage, the plants have started growing and that fresh growth was cut-off when cuttings were collected. These cuttings had to re-start growing, for the second time in the same growing season, after being planted in the field. Unrooted (and rooted) cuttings, on the other hand, were planted, in theory, with full reserves built up during the previous year (Dickmann et al. 2001) and had only 'fresh growth' in the first growing season, which happened after they were planted.

Finally, while we haven't followed longer-term survival yet, we expect that survival rates should stabilize with time; Sweeney at al. (2002) suggested that survivorship gets more stable after the fourth growing season on restored wet and riparian areas. Hence long-term monitoring of the crossings is needed to ascertain whether survival stabilizes with time, and whether this stabilization will happen among all treatments.

As expected, the unrooted treatment performed better for height growth in the first growing season in the field, when compared to the direct plant treatment. This initial difference can be explained by the previously described advantages that dormant cuttings have over the direct plant treatment during the rooting stage. Unrooted balsam poplar cuttings, which originated from dormant cuttings, can establish easily in the field and grow fast (Wilkinson 1999; McCarthy et al. 2017). DesRochers et al. (2004) recommended the use of unrooted balsam poplar cuttings for field planting to recover natural regeneration, due to their efficient performance and low costs. Whereas direct plant cuttings, which are not originated from dormant cuttings, do not have these characteristics. I did not find studies that compared cuttings originating from dormant stems with cuttings originating from stems collected in the summer for further discussion.

Rooted plants, also originated from dormant cuttings, were taller than unrooted and direct plant during the first summer (although greenhouse growth was not statistically tested for the first growing season). This was not surprising given the fact that cuttings in the rooted treatment had the advantage of initially growing in the more stable conditions of a greenhouse (e.g., regular watering schedule, lack of disturbances and/or damage), which in turn facilitated the development of roots. Schroeder and Walker (1991) found that height growth of cuttings in the greenhouse was good regardless of clone or origin of the cutting, which indicates the important influence of environmental conditions and root development in the initial development of cuttings. This is similar to the findings obtained in this study for the time period that cuttings were in the greenhouse, but, unexpectedly, I did not find significant differences in height growth among any of the treatments in the second growing season (after accounting for initial diameter and damage).

Difference in height growth (height in the fall minus height in the spring) in the second growing season was low for all treatments (the average difference in height ranged from just over 1 cm for the direct plant to a maximum of approximately 3.5 cm for the unrooted treatment (the mean total height in 2018 was about 46 cm for rooted, 22 cm for unrooted and 10 cm for direct plant). Low height growth of plants produced by balsam poplar cuttings during the second growing season has been obtained in other studies (DesRochers et al. 2004; Douglas et al. 2016). This could be caused by low proportions of root to shoot ratios (i.e., plants with few roots and long stems) on plants established during the first growing season (DesRochers et al. 2004). This, in turn, could result in more energy being spent on root production during the second growing season, rather than on height growth. Dickmann et al. (2001) explained the differences in root to shoot productions between poplars hardwood cuttings and seeds: cuttings quickly develop leaves from preformed buds, in response to available stored energy that makes simultaneous root and shoot growth possible; seedlings' height growth is slow, since limited initial energy is spent on root development, and because of that, seedlings with only 1 cm^2 of leaf surface already have 17 cm long roots. The fast simultaneous development of above and belowground parts on balsam poplar cuttings might result in unbalanced root to shoot ratios. The lack of treatment effect on height growth at the second growing season could also be caused by the influences of disturbances and suboptimal conditions in the field, as growth can be affected by size of the plant, but also by many other environmental factors (Dickmann et al. 2001; Douglas et al. 2016). Consequently, treatment did not have an effect on height growth during the second growing season, rather, initial diameter of the cutting and damage influenced this response variable.

Initial diameter of the cutting had a positive influence on plant survival in the first summer and on height growth in both summers. Optimal initial diameter for both survival and height growth ranged between 4 and 8 mm in this study, although, we did not have enough samples above 8 mm to infer the benefits of extending this range. Thus, for this specific study, the optimal initial diameter ranged between 4 and 8 mm, but larger initial diameters might also be beneficial. Future research could explore the different effects of initial diameters larger than 8 mm. The benefits of using cuttings instead of seeds are associated with the initial availability of starch, sugar, and protein reserves in the cut stem, which provide energy and carbohydrates to rapid root growth and fast development of leaves from preformed buds (Dickmann et al. 2001). Tilley and Hoag (2008) and Douglas et al. (2016) found cuttings with larger initial diameter had higher height growth and survival when compared to ones of smaller diameters. In contrast, Chater et al. (2017) did not find an association of initial diameter with rooting and growth, albeit in pomegranate. There are a few possible explanations for the influence of initial diameter on survival and height growth. DesRochers and Thomas (2003) indicated that position in the branch where cuttings originated from might be a more important factor affecting plant survival and height growth than basal diameter; in this case, initial diameter can be associated with position in the branch, since cuttings taken from the base are larger than the ones taken from the top. According to Schroeder and Walker (1991), position can affect performance as cuttings taken from the base of the branch have more root primordia and higher rooting rates than the ones coming from the top. In contrast, Smith and Wareing (1972) found that more roots are produced by terminal cuttings than basal ones. Because this study is part of a long-term restoration project, I did not examine root biomass, so I do not have data on rooting capacity. I also did not control for position of the cutting in the branch or quantify carbohydrate reserves. Based on my results, I can associate larger initial diameter with higher survival and height growth, likely as a reflection of carbohydrate reserves and the use of basal sections of original branches, with more root primordia. Future studies could explore the association of survival rates and height growth with initial diameter, carbohydrate reserves, and position in the branch where cuttings were taken from.

The lack of differences among treatments can also be attributed to influences of disturbance and damage on the plants. Environmental conditions in the field and damage incurred became more important during the second growing season, when they were selected in the top models for survival (disturbance) and height growth (damage). Damage affected height directly when plants were trampled or browsed by ungulates, for example. In those cases, the final height (at the end of the second growing season) was lower than the initial height (at the beginning of the second growing season), reaching final negative growth results, but those were not included in the data analyses. Then, the influences of damage analyzed here reflect the differences in height between undamaged and damaged plants that presented positive height growth. The levels of damage included were mostly associated to damaged leaves (e.g., fall off, yellow patches, eaten by herbivores) and stems (e.g., yellow or brown patches, broken side shoots), which were influenced by different environmental conditions (here called disturbances).

Presence of OHV use near the plants, or in the enhanced crossings, which happened in isolated situations, was the disturbance factor that resulted in the highest proportion of dead plants. It is well known that plants cannot withstand the direct impacts caused by OHV use (Hannaford and Resh 1999; Jones 1999; Brooks and Lair 2005; Trip and Wiersma 2015). Even limited vehicular use on a wet area can cause immediate ecological impacts, such as decreased vegetation height and increased plant mortality (Hannaford and Resh 1999). Our results indicated that plants influenced by OHV disturbances are less likely to survive when compared to the ones influenced by competition, trampling, or herbivory. However, in environments where OHV influence is not present, competition and herbivory are considered two major factors affecting balsam poplar growth and survival (Stanturf et al. 2001). Poplars are sensitive to above- and below-ground competition, which can be responsible for plant mortality (Coll et al. 2007; McCarthy et al. 2017). Previous studies have found that growth of hybrid poplars was negatively affected by competing vegetation around the plant, mostly caused by resource competition for light (Coll et al. 2007; Grenke et al. 2016; Henkel-Johnson et al. 2016; Goehing et al. 2019). This influence is even more intensified when the competing vegetation is comprised of graminoids (Grenke et al. 2016; Henkel-Johnson et al. 2016). Goehing et al. (2019) highlighted the importance of controlling aboveground vegetation close to trees in the first two years after planting in order to reduce the effects of competition for resources on plant establishment. In this study, the potential stress caused by competition resulted in similar percentages of undamaged, lightly damaged, severely damaged and dead plants, which means that competition was not a strong factor affecting plant survival or damage. Herbivory, on the other hand, resulted in the highest proportion of severely damaged plants. Poplars are browsed often by ungulates, rodents and other herbivores and this can reduce plant survival and growth (Stanturf et al. 2001; Dickmann et al. 2001). In Sweeney at al. (2002), differences in survival and growth on plants established for restoration of riparian forests were mostly related to intensity of herbivory and competition, indicating the importance of herbivore and weed control.

Therefore, the ideal scenario would be the non-occurrence of unsanctioned OHV use in decommissioned trails. After this is accomplished, future studies could explore the effects of removing opportunities for herbivory and competing vegetation through the comparison of survival and growth based on treatment and type of disturbance control.

Edaphic factors can also influence the performance of balsam poplar hardwood cuttings in the field (Dickmann et al. 2001; Zalesny et al. 2005; Cows and Fish 2007). This study showed a significant negative effect of penetration resistance on height growth and a positive influence on survival. The increased soil density caused by soil compaction can inhibit plant height growth through suppression of root development (Webb et al. 1983; Kozlowski 1999). In Wolken et al. (2010), root growth of balsam poplars was inhibited in compacted soils, due to poor aeration, which resulted in low growth. In the same study, poplars had high survival at conditions of low compaction and high moisture. Our results are similar to the findings of Wolken et al. (2010), since the inclusion of penetration resistance in the top models for survival in all three time periods suggests that this variable had a possible positive influence on survival, while it negatively affected height growth. Kozlowski (1999) explained that mild compaction can benefit initial plant establishment by improving capillary movement of water to the roots.

Depth to water table was not selected as a factor influencing plant survival or height growth in this study. Water availability can influence cutting survival and growth directly: the rooting capacity of cuttings is increased when soil moisture increases up to the point of saturation (Zalesny et al. 2005), and drought inhibits growth and might result in plant death (Dickmann et al. 2001). Thus, the lack of connection between depth to water table and plant survival and growth suggests that water availability was not a limiting (or helping) factor for cuttings establishment in Blue Rapids Provincial Recreation Area. Balsam poplar cuttings can be planted in BRPRA without the extra effort of irrigation, which could be costly and management intensive in a restoration program in drier areas, and also without the negative impacts of too much ground water.

Plant community composition

The results of this study supported my hypotheses that the plant community composition and abiotic cover would differ among the positions in relation to the trail (trail, edge, and forest 10 m from the edge), with a strong presence of non-native species, graminoid and mineral soil cover on trails and edges and native species, shrubs and litter cover in the forest. These differences likely reflect direct effects of OHV disturbance that result in vegetation removal on trails; this, in turn, would have caused exposed soil and reduction in competition allowing the establishment of non-native and early-successional graminoid species. As hypothesized, plant community composition was not significantly different between the years of 2017 and 2018, but there was an increase in total vegetation and non-native species cover, and richness and diversity indices on trails and edges, which was not expected. This suggests that some level of recovery of vegetation seems to be occurring within two years after trail re-contouring. This could be explained by non-severe soil compaction levels on trails, as wet and riparian areas have low resilience when soils are highly compacted, but lower compaction levels might have allowed for the establishment of plants with small root systems, such as annuals.

The elimination of vegetation on vehicular trails is the most obvious direct impact of OHV use on the plant community (Crisfield et al. 2012); the indirect consequences, however, are greater than just local vegetation removal, since effects can extend to the surrounding areas, affecting plant health and native plant distribution (Dale and Weaver 1974; Johnson et al. 1975). The overall trend of lower vascular vegetation cover on trails (i.e. disturbed areas), higher plant cover and species diversity in the edges and a significantly different plant community composition at the forested areas further away from the trail found in this study is consistent with what has been observed in previous studies (Wilshire 1977; Hannaford and Resh 1999; Jones 1999; Brooks and Lair 2005; Harper et al. 2005; Crisfield et al. 2012; Trip and Wiersma 2015).

Total species cover increased on trails within two years after enhancement, indicating a level of recovery of vegetation on trails. However, the plant species that were mostly associated with this increase are forbs and non-natives. The increase in non-native species on trails was expected, considering that these species are adapted to take advantage of disturbed soils and high light availability (Prose et al. 1987; Harper et al. 2005). In addition, OHVs can serve as a means of dispersal of plant propagules, facilitating the establishment of non-native species (Brooks and Lair 2005). Still, it was unexpected that this increase would be significant only within two years after trail enhancement, since wet environments are considered to have low resilience to vehicular disturbances as a consequence of the effects of compaction on soil structure (Webb 1983; Charman and Pollard 1995; Kozlowski 1999; Arp and Simmons 2012; Trip and Wiersma 2015).

In our study, mean penetration resistance (PR) on edges and forests was approximately 0.6 kg/cm^2, but the trails presented mean PR of 1.2 kg/cm^2. Burrows (1982) found that root development did not differ in wet soils with PR of 0.5 kg/cm^2 when compared to control soils with PR of 0 kg/cm^2, but there were negative influences in rooting development after 0.5 kg/cm^2. Wet soils were considered severely compacted after 10 kg/cm^2; dry soils with PR 10 kg/cm^2 showed similar results to wet soils with PR 0.5 kg/cm^2. In Wallace (1987), soils (did not specify if dry or wet) with penetration resistance of 1.6 kg/cm^2 were considered non-compacted; plant growth and biomass were reduced above this threshold but performed well under it. The mean penetration resistance on soils in recreational areas ranges between 4 and 2.6 kg/cm^2 for sites without vehicular use and between 7.5 and 6 kg/cm^2 for sites with vehicular use (Hammitt and Cole 1998; Lei 2004). Considering that the trails focused on in this study are in wet and riparian areas, there might be a level of compaction affecting root development, but the trails are not severely compacted when compared with thresholds found in the literature.

The increase in forbs on trails, and reduced presence of woody species, might be attributed to the effects of soil compaction. According to Kozlowski (1999), plants can expand their roots on compacted soils up to the point where the soil pores are larger than the roots. Small annual plants can become established rapidly and easily on compacted soils when compared to perennial woody species, which need more space for root expansion (Wilshire 1977; Kozlowski 1999). Wilshire (1977) points out that reestablishment of annual plants happens within a few years after disturbance is stopped, but recovery of perennial woody species is a much slower process. According to Webb et al. (1983), disturbed non-compacted soils can become stabilized in a relatively short time period, which results in similar vegetation cover on undisturbed and recovered areas, but complete recovery time would be difficult to predict. Our results might indicate that either the trails under restoration activities in BRPRA were not severely compacted or were structurally recovered during the previous phases of the enhancement project to a point where the present level of compaction allows the growth and establishment of small annual vegetation but inhibits the development of perennial woody species.

Another factor to take into consideration is that many tree and shrub species reproduce sexually and vegetatively (Braatne et al. 1996; Dickmann 2001). Seed dispersal occurs during the summer months, but seed viability is short, since seeds last an average 3 days after becoming wet if a favourable site is not encountered (Braatne et al. 1996). Suitable conditions for seed establishment can occur on intervals of five to ten years on riparian habitats (Braatne et al. 1996). Asexual, or vegetative, propagation, on the other hand, is a more common way of propagation amongst hardwood tree and shrub species. Annual plants, on the other hand, propagate mostly through seeds. These plants go through germination of seeds, flowering and production and dissemination of seeds in only one growing season (Hartmann et al. 1990), whereas many perennial woody plants have a biennial cycle, in which shoots are vegetative in one year and reproductive on the next (Hartmann et al. 1990).Therefore, the differences in life history amongst hardwood trees and shrubs, and annual species might explain the low woody cover on trails, as herbaceous plants rapidly establish via seeds, while woody species take a longer time to get established through vegetative means.

The edges adjacent to trails also experienced an increase in vegetation cover within two years of trail enhancement, which could indicate that edges were also undergoing disturbance from OHV use on trails. Dale and Weaver (1974) showed that vegetation communities on edges can be directly affected by trampling in areas of 1 to 4 m apart from the trail. The vegetation increase in edges was mostly associated with graminoid species. Graminoids can occupy disturbed areas quickly, due to morphological characteristics, such as resistant basal meristems and tissues, that provide high tolerance to trampling disturbances (Charman and Pollard 1995; Trip and Wiersma 2015). Overall, edges presented high species diversity and richness indices in both years. Edge environments can differ in composition, structure and function in comparison to the non-forested and forested ecosystems they are adjacent to (Harper et al. 2005). The higher species richness and diversity on edges result from the lack of competing vegetation on adjacent trails, which increases light availability and energy input in the system resulting in more productivity, as well as from the input of native seeds and propagules, nutrients (through litterfall), and shade from forested ecosystems (Dale and Weaver 1974; Johnson et al. 1975; Brooks and Lair 2005). Higher diversity and richness of species on edge environments have also been observed in previous studies (Dale and Weaver 1974; Johnson et al. 1975; Crisfield et al. 2012).

Besides the fact that vegetation cover on trails and edges increased from 2017 to 2018, those vegetation communities differed from what was found in the forested areas. Trail and edge plant communities were dominated by forbs and graminoids, whereas undisturbed communities had a higher presence of woody indicator species. Taking into account the higher number of forbs and graminoid species on trails and edges, along with the limitations on perennial species establishment, active restoration could be an option to help speed woody species establishment on these sites. Walker et al. (2007) suggested that active restoration can aid primary succession when regeneration of native species does not occur naturally. Bourgeois et al. (2016) revealed that the planting of native trees in degraded wet and riparian ecosystems resulted in the re-establishment of a vegetation structure similar to natural forests two decades after tree planting. Grown trees changed light availability, which in turn influenced herbaceous communities, favoured the establishment of native species, changed soil moisture, nutrients availability and microclimate (Bourgeois et al. 2016). Yet, succession and restoration responses depend on abiotic and biotic conditions specific to each site, including: plant dispersal, germination and growth, and species turnover and ecosystem resilience (Walker et al. 2007). In this context, planting balsam poplar on enhanced trail crossings might help the process of plant succession, although, successional trajectories cannot be predicted, only monitored. This study did not include an examination of similar trails passively restored, as the enhancement project was still too recent; there were not passively restored trails available for comparison. Future research could explore passively restored trails in Blue Rapids Provincial Recreation Area and compare them with the actively restored trails, in order to understand the influence of planting native balsam poplar on the plant community composition in the longer-term.

2.5. Conclusions

Overall, this study provided novel insights into the factors that influence survival and growth of balsam poplar cuttings planted in enhanced OHV trails in wet and riparian areas and how plant communities differ with proximity to the trail in the short-term after trail restoration. Findings can also be relevant to the oil and gas industry, as the actual OHV trails were originally created as pipeline rights-of-way, seismic lines and roads. Considering that other areas in Alberta have similar issues and that reclamation of areas disturbed by oil and gas activities is required by legislation in Alberta, oil and gas companies might look for restoration methods for similar trail systems. The cuttings initially grown in the greenhouse had higher survival compared with those directly planted in the field but did not grow any taller than the other two treatments during the second growing season. Instead, damage seemed to be the most important factor explaining a lack of differences in height growth among the treatments.

Considering that the rooted treatment presented overall higher survival rates and taller plants when compared to the other two treatments, I recommend the use of this treatment in restoration projects that include the possibility of growing cuttings in a greenhouse. For projects where this is not possible, I recommend the use of the unrooted treatment, as it presented acceptable survival rates for restoration of degraded wet and riparian areas. In order to achieve target stem densities of balsam poplar adult trees on decommissioned restored trails in the future, I suggest unrooted cuttings should be densely planted in the field, as mortality of 50% can be expected by the end of the second growing season. The planting layout for this study included a very high planting density. The area of each plot was 4 m² (i.e., each plot was 1.5 m length and wide, plus a 1 m distance from the next plot; 1.5 + 0.5 = 2 m, 2 x 2 = 4 m²), that means we planted 5 cuttings every 4 m², which results in a density of 12500 trees per hectare (trees/ha). This density was chosen in order to account for mortality in the field. Ceulemans and Deraedt (1999) suggest densities of 10,000 stems per hectare for short-rotation poplar plantations. Balsam poplar stand densities vary with stand history; 25-year-old stands include about 8700 trees/ha (Zasada and Phipps 1990). So the unrooted cuttings could be planted in the field in densities of 7 cuttings per 4 m², to account for mortality in the longer term, considering 8700 trees/ha as a target density. The direct plant treatment can be an alternative for restoration projects with time and budgetary limitations, since the application of this treatment does not require advanced collection, storage space, energy consumption, or extra labour; just a group of people to collect the plants in one day, prepare and soak them for 24 hours and plant them the next day. Because of that, and since the plants that survived presented similar height growth rates to the ones from the other two treatments on the second growing season, I also recommend the use of direct plant. Direct plant cuttings should also be planted in high densities, as high as10 cuttings every 4 m² to account for mortality in the longer term, still using 8700 trees/ha as target density. Still, in general, preference should be given to collecting plants during dormancy, and selecting shoots with initial diameters ranging from 4 to 8 mm, to produce cuttings, in order to take advantage of carbohydrate reserves, and consequently, obtain better performance in the field. However, because planting in this study was limited to a single year, care must be taken about generalizing the results; rather I suggest further research could explore planting in different years in order to understand if internal and external conditions (in relation to the cuttings) experienced in 2017, such as plant quality and precipitation regime, influenced the results.

Disturbance/stress and damage came out as important factors influencing survival and height growth of cuttings. Further research can explore the interactions of disturbance and damage with treatment effects. Disturbance and stresses can be managed with the use of herbivore protection (e.g., plant shelters), and weed control (e.g., mowing, and manual control). Controlling for OHV activity is complex, BRPRA has a management plan for OHV use, which includes physical barriers and patrolling of the recreational and decommissioned trails, but unsanctioned use still occurs. An outreach program combined with signage, access control and enforcement, might be a step forward in the interest of informing the population about the consequences of unsanctioned OHV use in decommissioned trails in wet and riparian areas.

Disturbance associated with OHV use on trails at Blue Rapids Provincial Recreation Area has resulted in different plant communities on restored trails and in their adjacent edges as compared to adjacent forested areas. Non-native species and graminoids were positively associated with trail and edge positions, whereas native species and shrubs were associated with forested areas. These changes in plant community composition along the trails will likely continue to shape future plant community composition in the Park if no action is taken. Therefore, active restoration is recommended. The planting of native balsam poplar on trails to be recovered after OHV disturbance might help speed the process of plant succession. The creation of a longer-term plant community monitoring program for these sites could help to achieve a better understanding of their long-term successional trajectories.

Figures

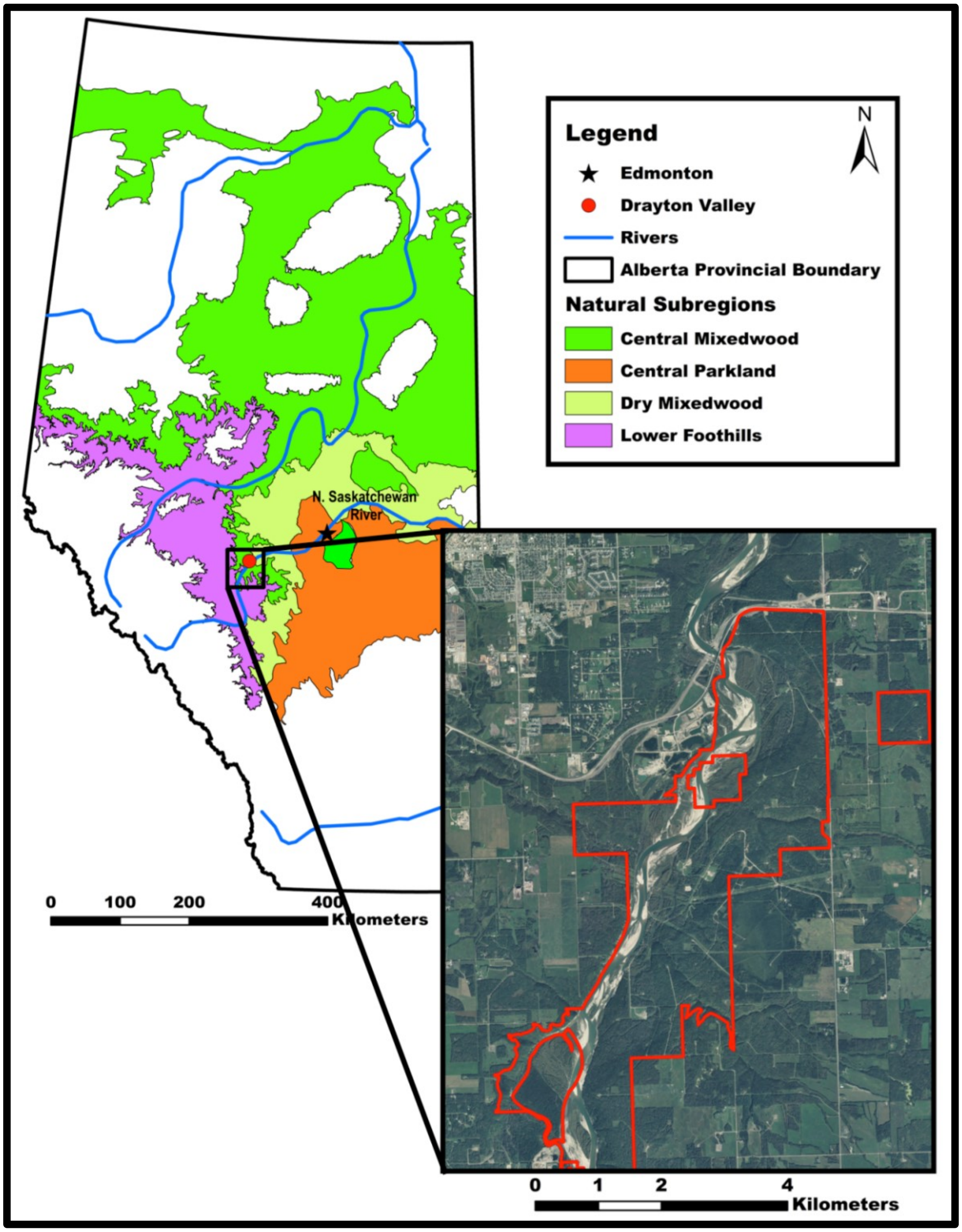

Figure 2-1. Blue Rapids Provincial Recreation Area. Location in Alberta, position relative to Edmonton and Alberta's Natural Sub-Regions. Image courtesy Kerri Widenmaier.

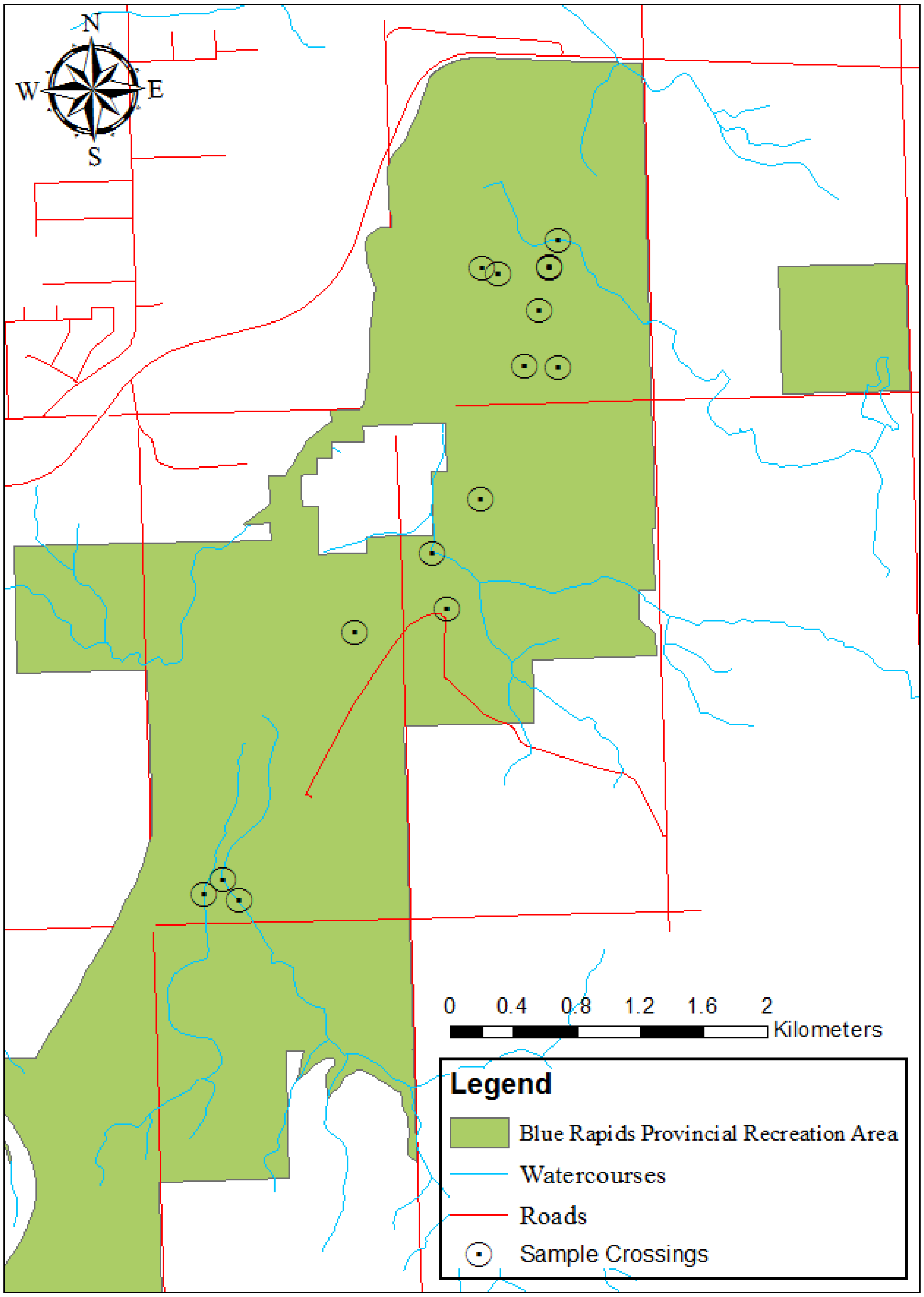

Figure 2-2. The locations of sample wet and riparian area crossings chosen for restoration in Blue Rapids Provincial Recreation Area.

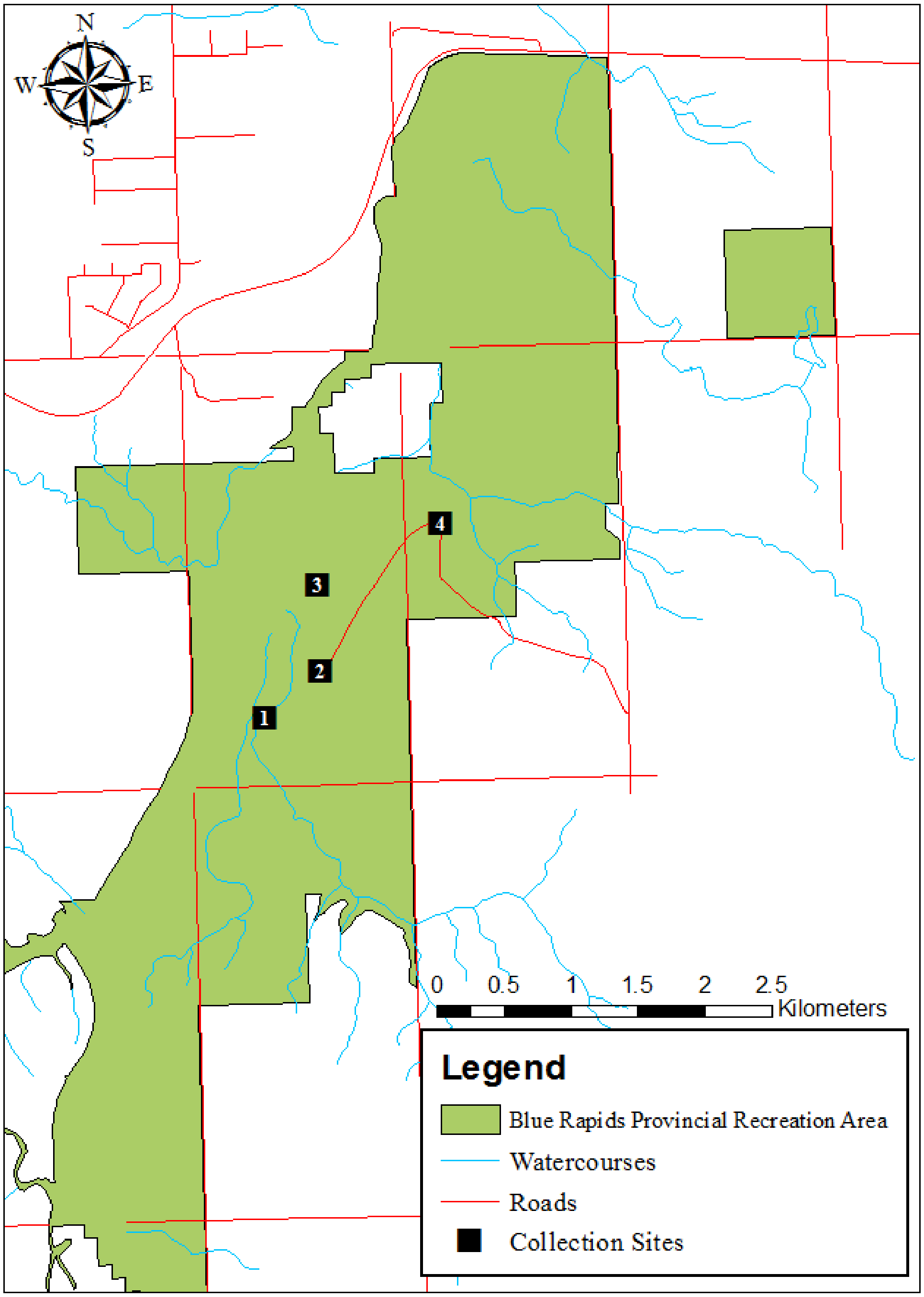

Figure 2-3. Sites where stems were collected within Blue Rapids Provincial Recreation Area.

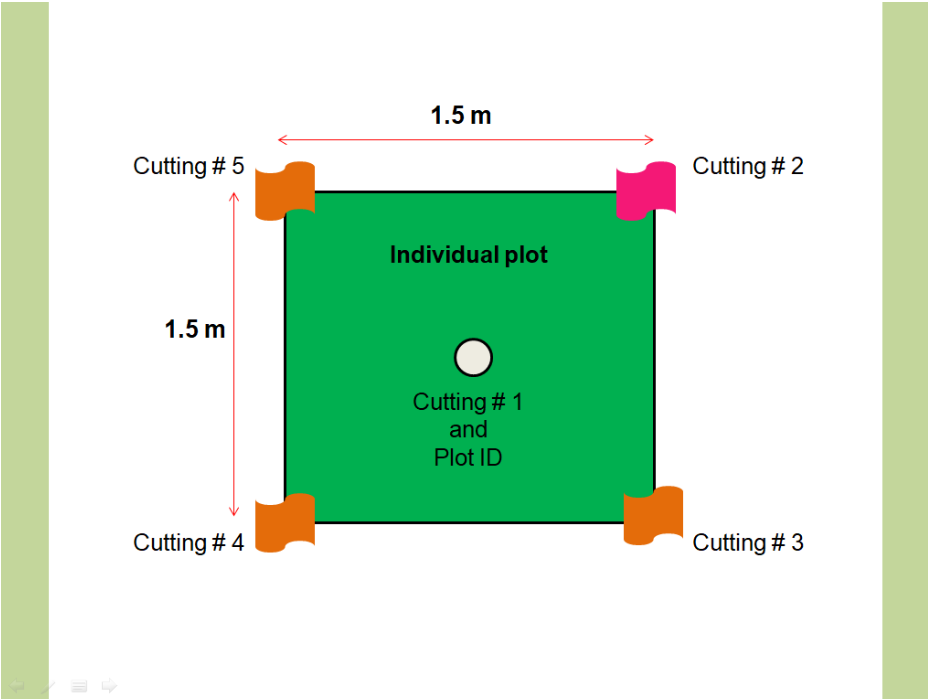

Figure 2-4. Individual plot design. Cuttings were planted at a distance of between 1.06 - 1.5 m from each other. Pigtail and flags identify cuttings 1, 2, 3, 4 and 5, starting from pigtail at #1 and following cutting number 2 in clockwise direction.

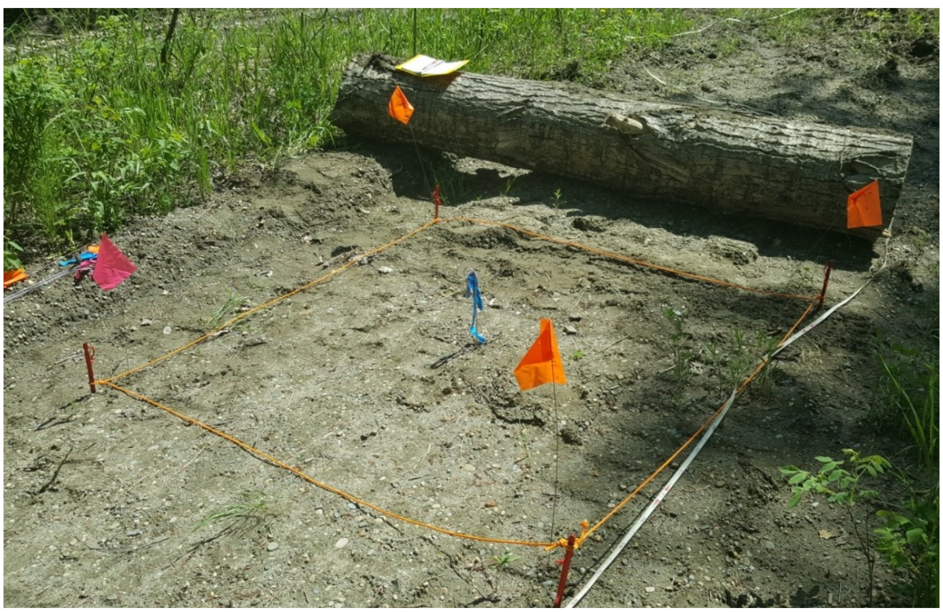

Figure 2-5. Example of the layout of an individual plot within a target crossing. Note the log that is placed across the decommissioned trail to reduce accessibility.

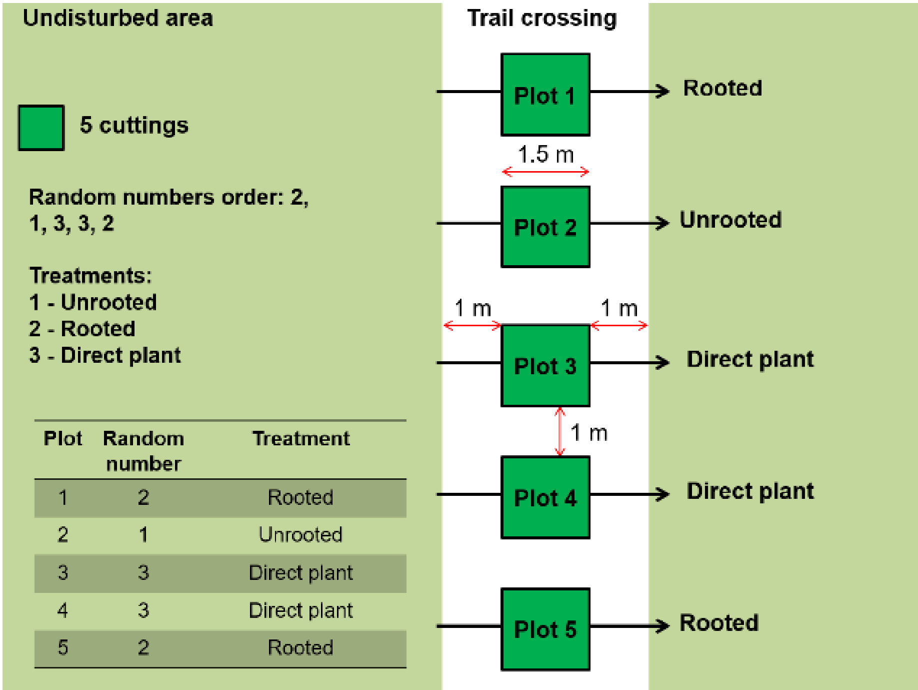

Figure 2-6. Planting layout in hypothetical trail crossing targeted for restoration, showing first five plots on enhanced trail. Each plot received cuttings from a different treatment. Random numbers refer to treatments to be applied, each treatment was given a number. The numbers were assigned by a random number generator and applied to plots in order from plot 1 to plot 5. Plot 1 was positioned in the direction of the main road and each subsequent plot was located further away from plot 1 towards the end of the crossing.

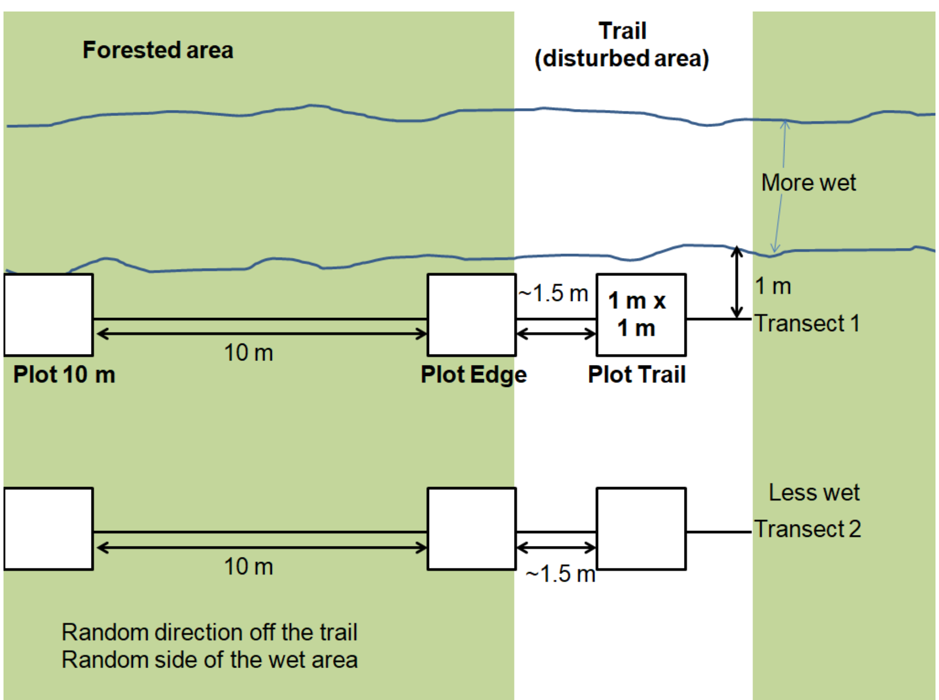

Figure 2-7. Vegetation survey sampling layout in hypothetical target trail crossing.

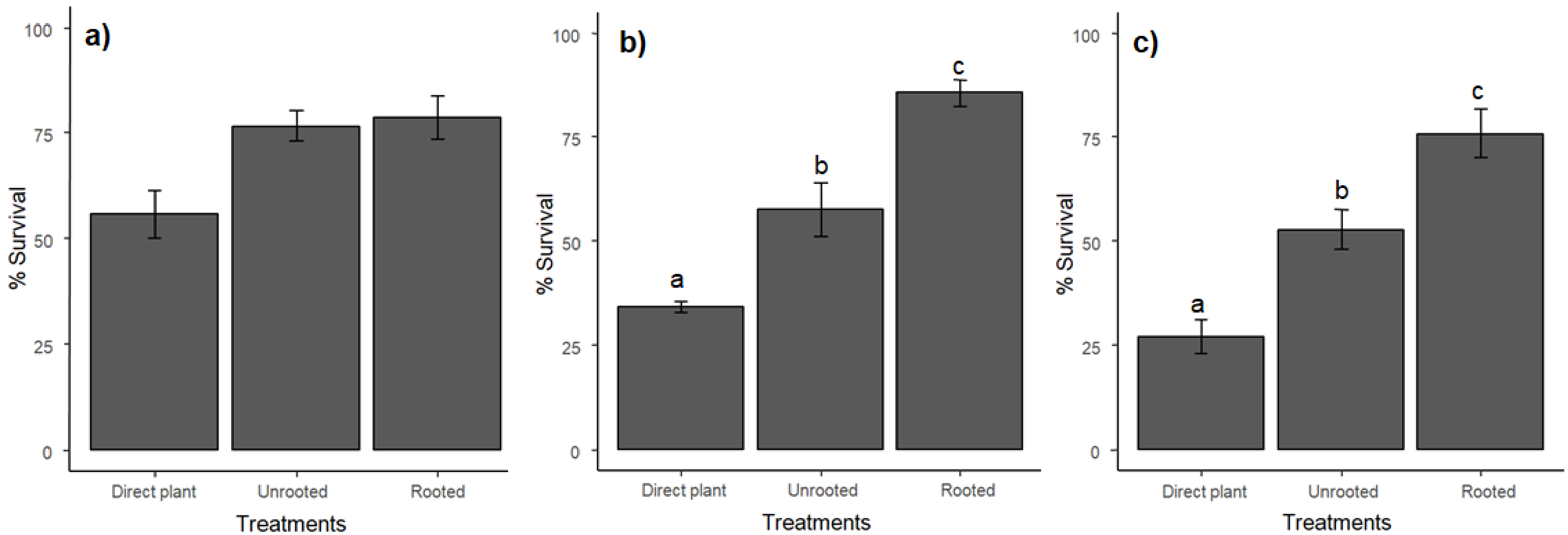

Figure 2-8. Mean survival (%, $\pm$ SE) of the balsam poplar plants among the three planting treatments: a) at the end of the first growing season in 2017 (first summer). The rooted treatment represents survival in the greenhouse (nursing culture measurements), unrooted and direct plant represent survival in the field. Rooted was not statistically compared with field treatments; b) over the first year (spring 2017 through spring 2018 for unrooted and direct plant, and fall 2017 through spring 2018 for rooted); c) through the end of the growing season in 2018 (second summer), unrooted and direct plant represent survival in the field after two growing seasons, and rooted represents survival after one growing season in the field. Bars with different letters (a, b, c) indicate significant differences among treatments after accounting for other explanatory variables in the model (α= 0.05). See also Tables 2-2 and 2-3.

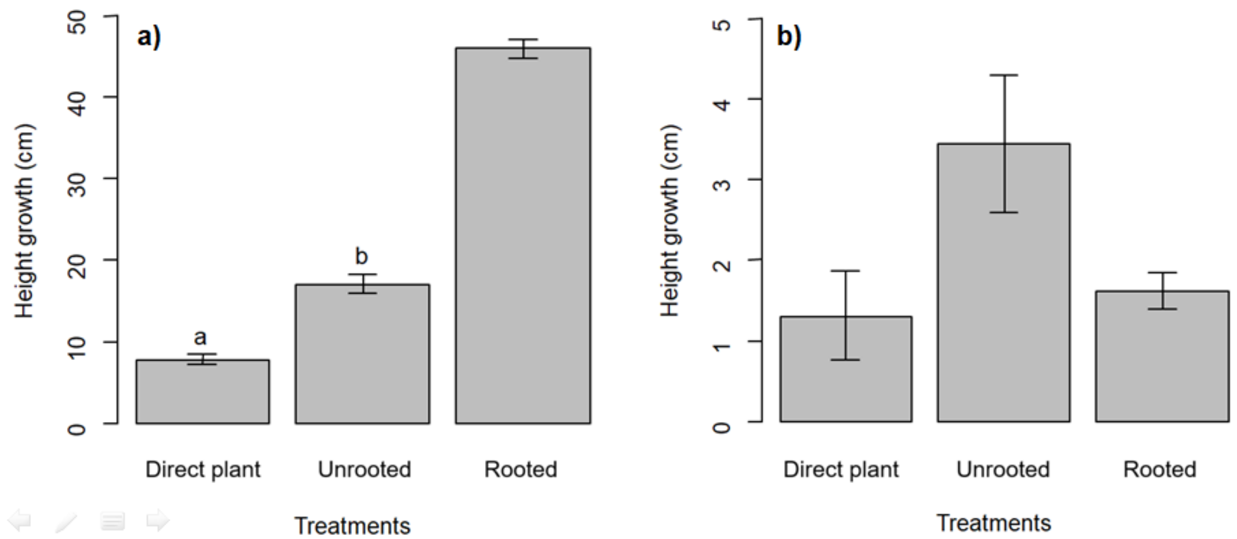

Figure 2-9. Mean height growth (cm ± SE) of the balsam poplar plants produced by the cuttings for the three treatments: a) total height over the first growing season, (Spring to Fall 2017). The rooted treatment represents height growth in the greenhouse, unrooted and direct plant represent height growth in the field. Rooted was not statistically compared with field treatments. Letters indicate significant (α= 0.05) differences among treatments performance in the field; b) growth difference (fall 2018 – spring 2018) over the second growing season, Fall 2018. Direct plant and unrooted presented growth after two growing seasons in the field and rooted presented growth after one growing season in the field. Total mean height for 2018 is the mean height in 2017 plus the difference in height growth in 2018. All treatments represent field performance, treatments were not significantly different. Differences among treatments were obtained after accounting for other explanatory variables in the model for each year (α= 0.05). See also Tables 2-4 and 2-5.

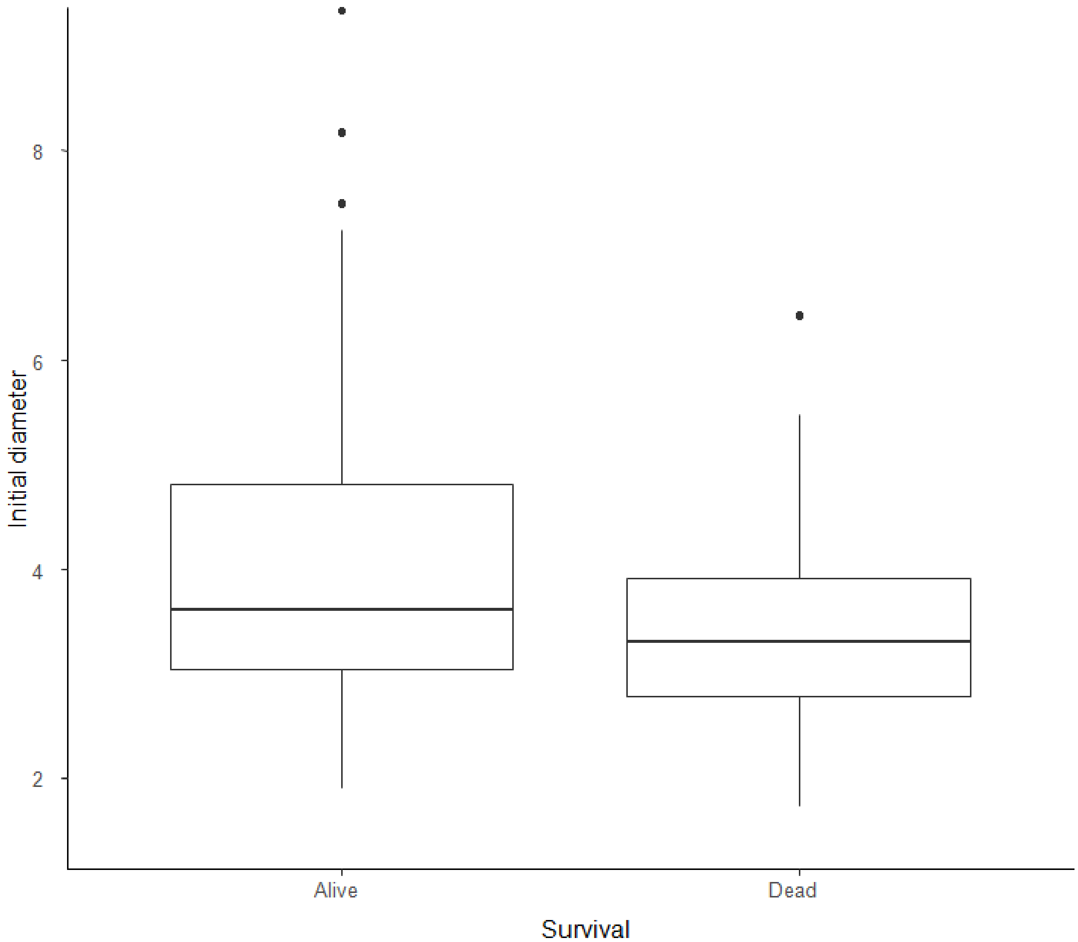

Figure 2-10. Initial diameter of the cutting in relation to survival at the end of the first growing season (2017).

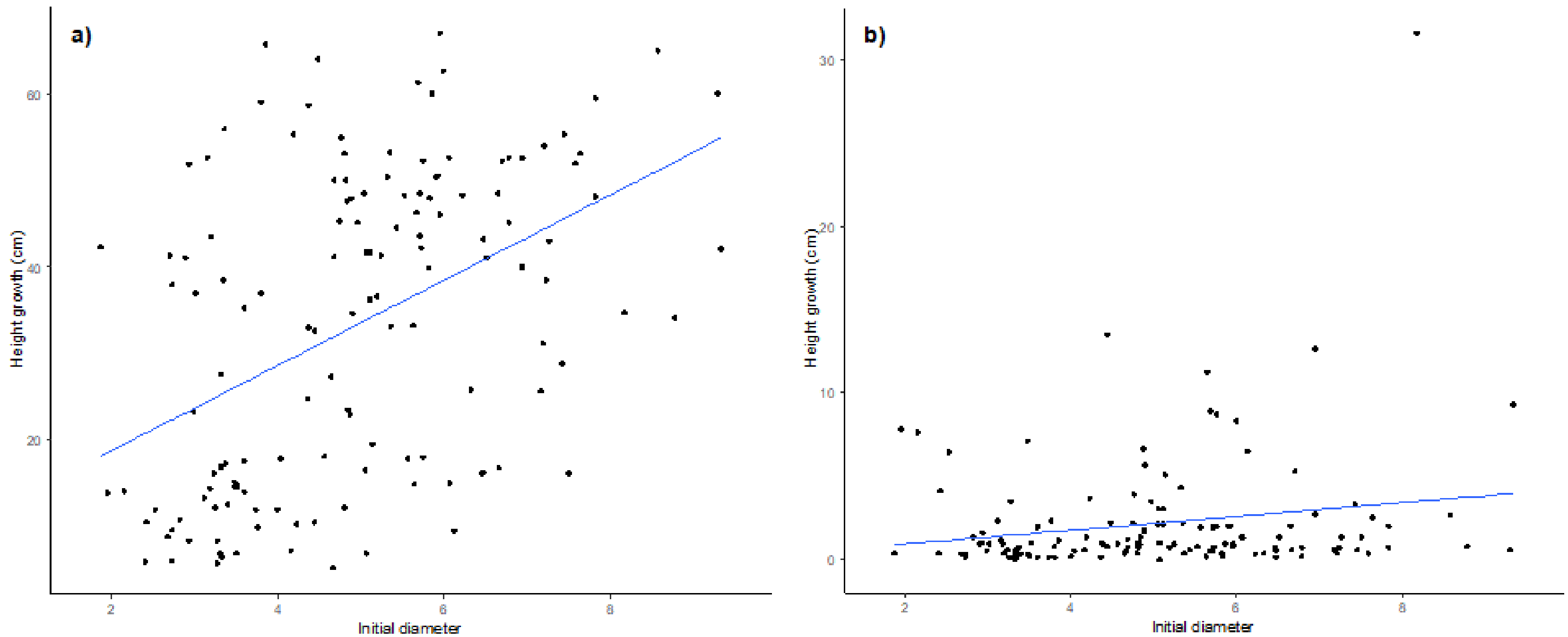

Figure 2-11. Initial diameter of the cutting in relation to height at: a) the end of the first growing season, 2017; and b) the end of the second growing season, 2018 (difference in height: fall – spring).

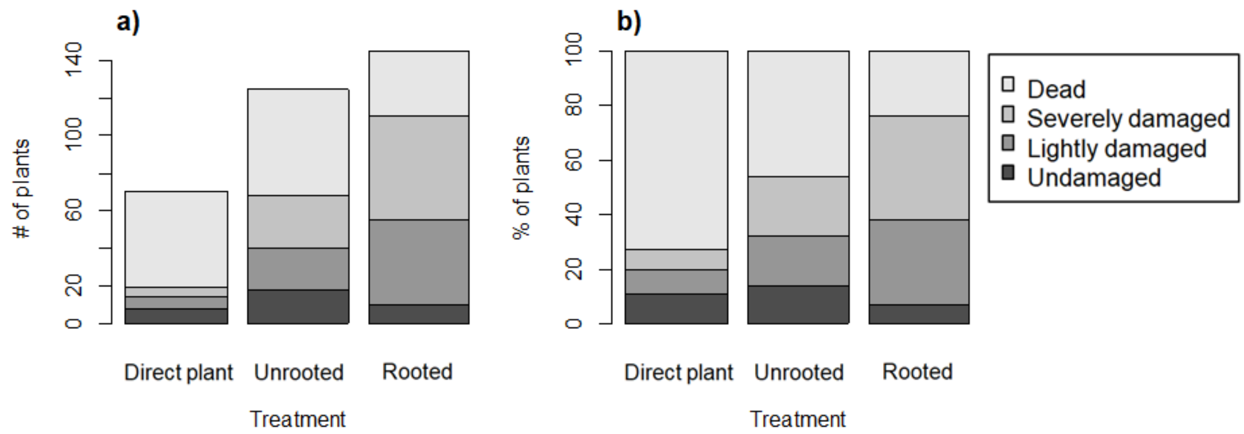

Figure 2-12. Plant damage levels at the end of the second growing season (2018) grouped by each treatment: a) total number of plants; and b) proportion of plants.

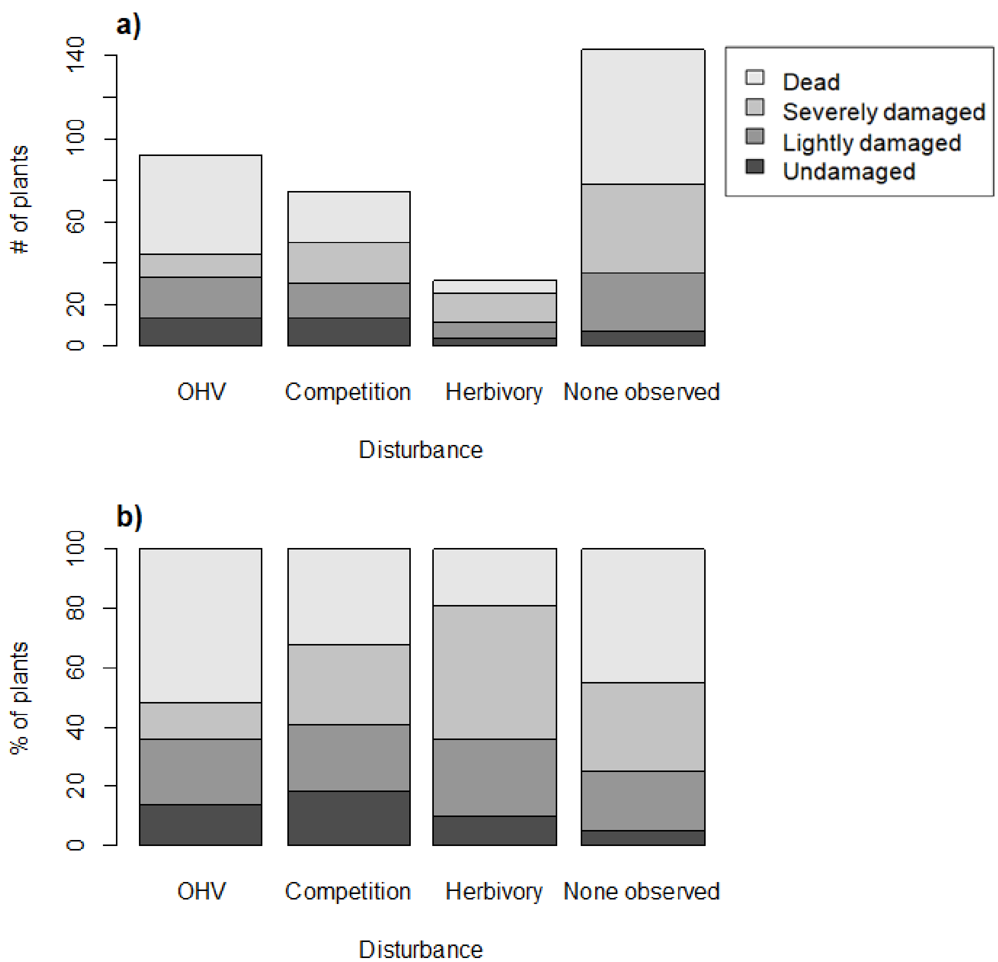

Figure 2-13. Plants influenced by disturbance (or stress in the case of competition) types and their damage levels at the end of the second growing season (2018): a) total number of plants; and b) proportion of plants.

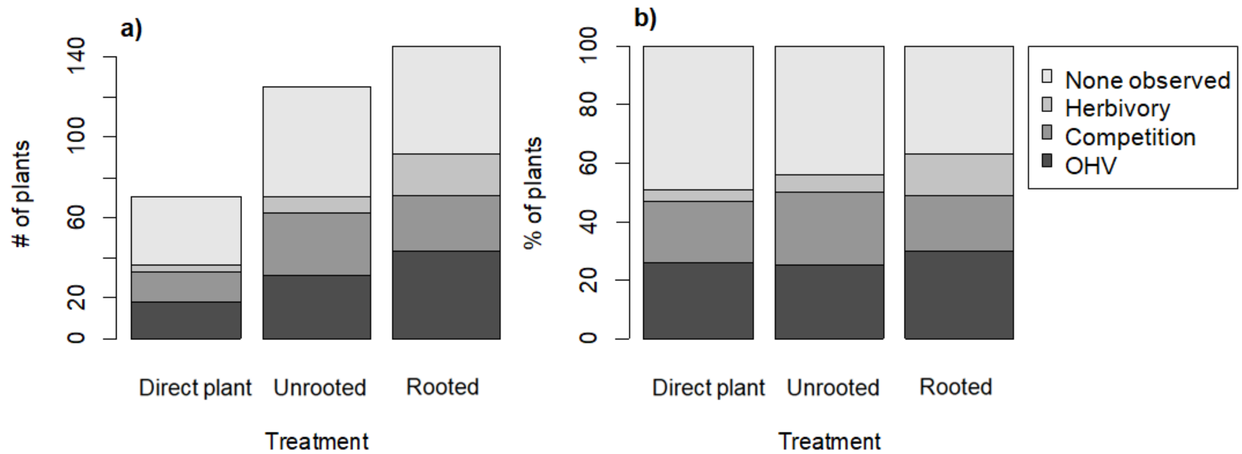

Figure 2-14. Disturbance types at the end of the second growing season (2018) grouped by each treatment: a) total number of plants; and b) proportion of plants.

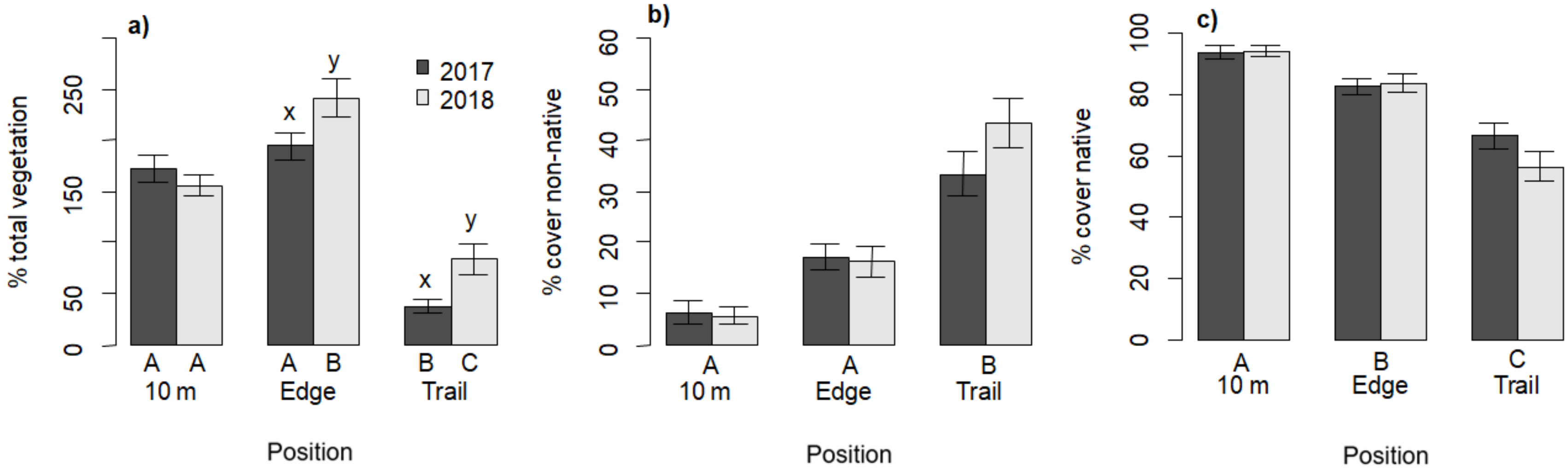

Figure 2-15. Mean percent cover (± SE) for a) total vegetation, b) non-native species, and c) native species, by year (± SE). Years with different letters (x, y) were significantly different. Positions with different letters (A, B, C) were significantly different within a year for vegetation cover and non-native species and in both years for native species. Contrasts can be found in Tables 2-7 and 2-8.

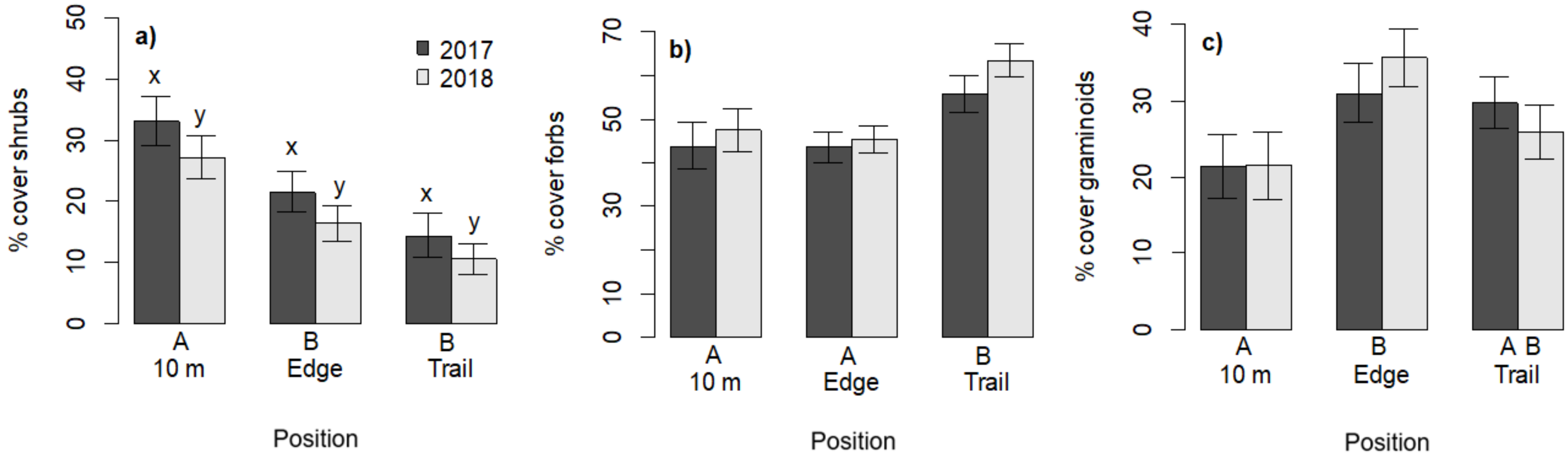

Figure 2-16. Mean percent cover (+ SE) for a) shrubs, b) forbs, c) graminoids, by year. Years with different letters (x, y) were significantly different. Positions with different letters (A, B) were significantly different. Contrasts can be found in Table 2-7.

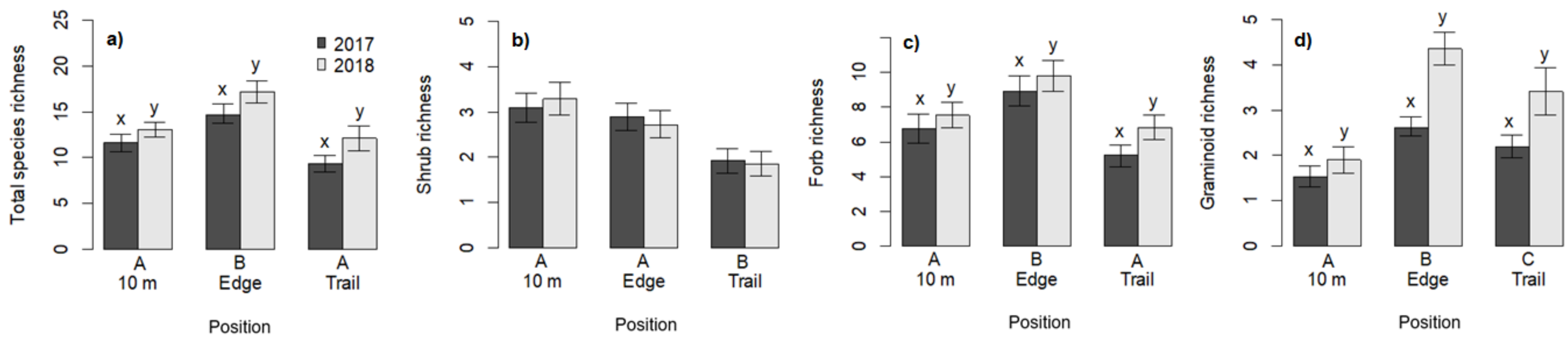

Figure 2-17. Mean richness (+ SE) of a) total species, b) shrubs, c) forbs, d) graminoids, by year. Years with different letters (x, y) were significantly different. Positions with different letters (A, B, C) were significantly different. Contrasts can be found in Table 2-7.

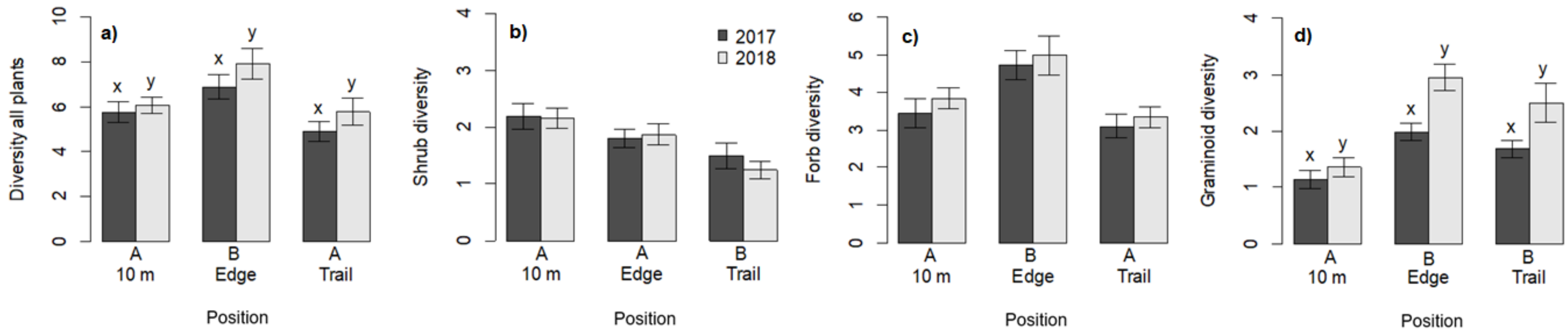

Figure 2-18. Mean (+ SE) Simpson diversity of a) all plants, b) shrubs, c) forbs, d) graminoids, by year. Years with different letters (x, y) were significantly different. Positions with different letters (A, B) were significantly different. Contrasts can be found in Table 2-7.

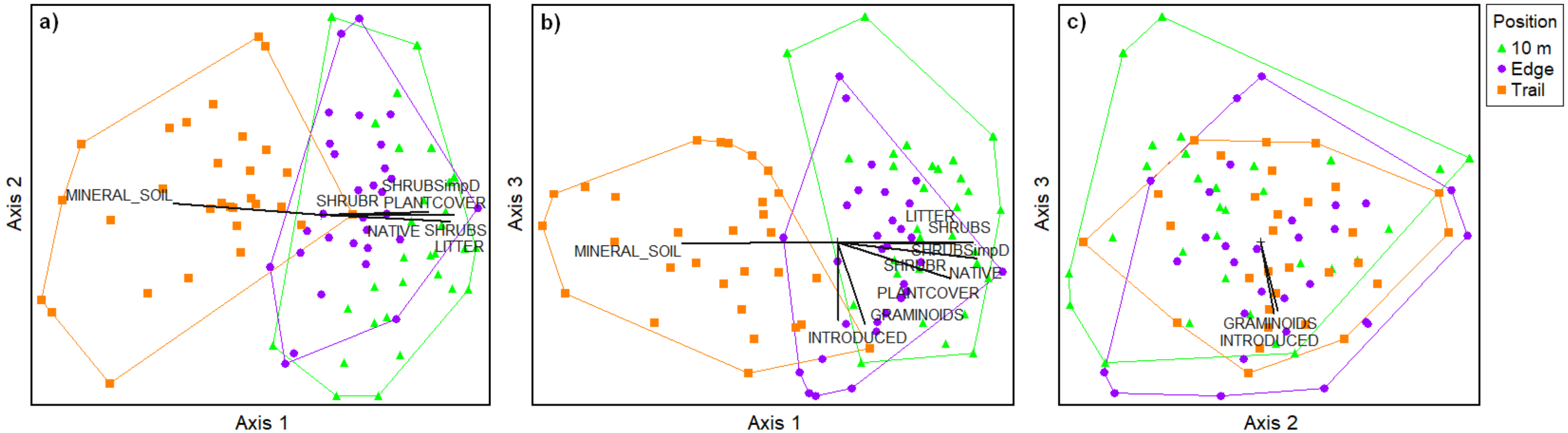

Figure 2-19. Results of non-metric multidimensional scaling ordination of understory plant community composition for 2017 and 2018 altogether. Each symbol is a plot, which is coded by position. The final ordination was a 3-D solution, so three plots are presented: a) the first and second ordination axes; b) the first and third ordination axes; and c) the second and third ordination axes. The angles and lengths of the vectors for the environmental variables overlain on the ordination vectors indicate direction and strength of associations of the variables with the ordination axes (cut-off for displayed variables was $R^2 > 0.3$). Mineral soil, litter, native, non-native, graminoids and shrubs represent cover values; PLANTCOVER – represents total plant cover; INTRODUCED – represents non-native cover; SHRUBR – represents shrub richness; and SHRUBSimpD – Shrub Simpson diversity.

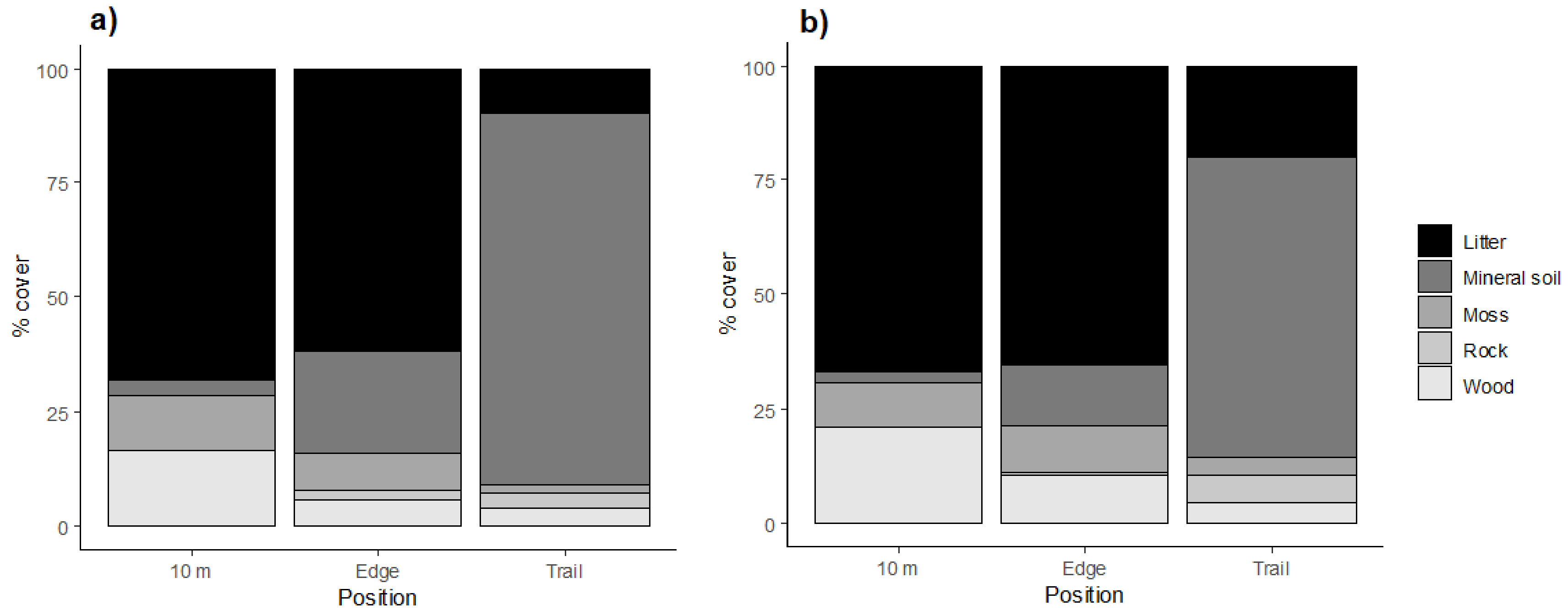

Figure 2-20. Mean cover in proportions of litter, mineral soil, moss, rock, and wood on 10 m, edge, and trail in: a) 2017; and b) 2018.

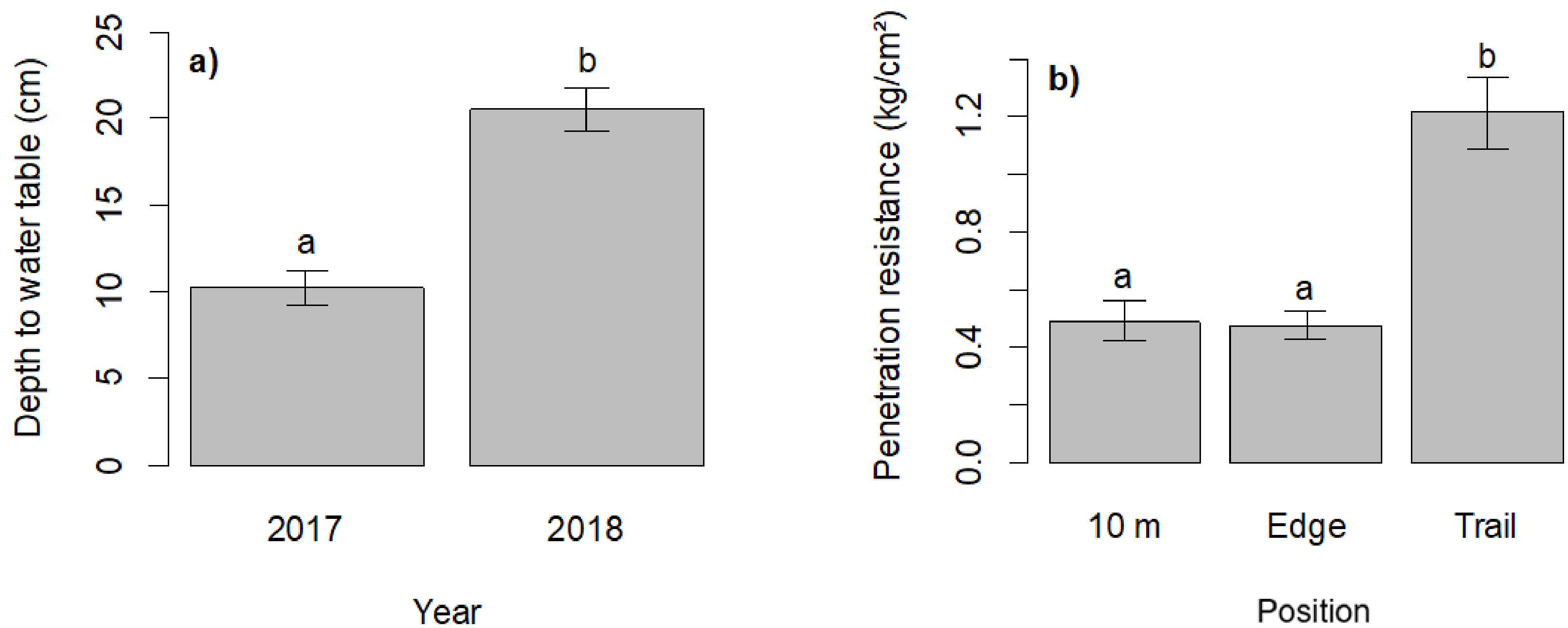

Figure 2-21. Mean abiotic variables: a) depth to water table in 2017 and 2018; and b) Penetration resistance on trail, edge and 10 m (average of 2017 and 2018). Years and positions with different letters (a, b) were significantly different.

Tables

Table 2-1. Results of model selection, including top five best models to describe survival at the end of the first growing season, over the first year (spring 2017 to spring 2018), and through the end of the second growing season (second summer), height growth in 2017, and height growth in 2018. The overall best model (shown in bold) for each response variable had delta Akaike Information Criterion value adjusted for small sample sizes (Δ AICc) zero and the highest Akaike weight.

Variable	Model structure	AICc	Δ AICc	W_i
Survival at the end of 1st growing season	**$\beta_0 + \beta_1$(treatment) + β_2(initial diameter) + β_3(PR out[i])**	**235.24**	**0.00**	**0.33**
	$\beta_0 + \beta_1$(treatment) + β_2(initial diameter) + β_3(PR in[ii])	235.82	0.57	0.25
	$\beta_0 + \beta_1$(treatment) + β_2(PR in)	236.59	1.35	0.17
	$\beta_0 + \beta_1$(treatment) + β_2(initial diameter) + β_3(PR out) + β_4(PR in)	236.89	1.64	0.15
	$\beta_0 + \beta_1$(treatment) + β_2(PR out) + β_3(PR in)	237.61	2.37	0.10
Survival over the 1st year	**$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(PR out)**	**355.59**	**0.00**	**0.28**
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(PR out) + β_4(PR in)	355.70	0.10	0.27
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(initial diameter) + β_4(PR out) + β_5(PR in)	356.51	0.92	0.18
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(PR out) + β_4(PR in) + β_5(water table)	356.74	1.15	0.16
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(initial diameter) + β_4(PR out) + β_5(PR in) + β_6(water table)	357.44	1.85	0.11

Variable	Model structure	AICc	Δ AICc	W_i
Survival through the end of 2nd growing season	**$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(PR out)**	**401.91**	**0.00**	**0.43**
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(initial diameter) + β_4(PR out)	403.73	1.82	0.17
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(PR out) + β_4(water table)	403.81	1.89	0.17
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(PR out) + β_4(PR in)	403.84	1.93	0.16
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(PR out) + β_4(PR in) + β_5(water table)	405.71	3.80	0.06
Height growth 2017	**$\beta_0 + \beta_1$(treatment) + β_2(initial diameter) + β_3(PR in)**	**801.88**	**0.00**	**0.54**
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(initial diameter) + β_4(PR in)	803.18	1.30	0.28
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(initial diameter) + β_4(PR out) + β_5(PR in)	804.66	2.78	0.14
	$\beta_0 + \beta_1$(treatment) + β_2(initial diameter) + β_3(PR in) + β_4(water table)	808.04	6.16	0.03
	$\beta_0 + \beta_1$(treatment) + β_2(disturbance) + β_3(initial diameter) + β_4(PR in) + β_5(water table)	809.08	7.20	0.02
Height growth 2018	**$\beta_0 + \beta_1$(damage) + β_2(initial diameter)**	**325.39**	**0.00**	**0.64**
	$\beta_0 + \beta_1$(damage) + β_2(initial diameter) + β_3(PR in)	327.68	2.29	0.20
	$\beta_0 + \beta_1$(treatment) + β_2(damage) + β_3(initial diameter)	328.71	3.32	0.12
	$\beta_0 + \beta_1$(treatment) + β_2(damage) + β_3(initial diameter) + β_4(PR in)	331.74	6.34	0.03
	$\beta_0 + \beta_1$(treatment) + β_2(damage) + β_3(initial diameter) + β_4(PR out)	333.78	8.39	0.01
	$\beta_0 + \beta_1$(treatment) + β_2(damage) + β_3(initial diameter) + β_4(PR out)	333.78	8.39	0.01

[i] Penetration resistance outside planting hole area.

[ii] Penetration resistance in planting hole area.

Table 2-2. Coefficients of the explanatory variables in generalized linear-mixed effects models predicting survival of balsam poplar cuttings in Blue Rapids Provincial Recreation Area. Includes explanatory variables from the best overall model for each time period. Reference categorical variables contained in the intercept for each model are: direct plant (treatment) for survival at the end of first growing season; direct plant (treatment) and OHV use (disturbance) for survival over the first year; and direct plant (treatment) and OHV (disturbance) for survival through the end of second growing season.

				Confidence intervals of estimate		
Response variable	Explanatory variable	Estimate	Standard error	Lower	Upper	Pr (>(|z|) value
Survival at the end of 1st growing season	Intercept	-1.104	0.615	-2.309	0.100	0.072
	Unrooted treatment	0.770	0.412	-0.037	1.578	0.062
	Initial diameter	0.299	0.156	-0.006	0.604	0.055
	Penetration resistance out	0.284	0.219	-0.145	0.713	0.195
Survival over the 1st year	Intercept	-2.572	0.620	-3.788	-1.357	<0.001
	Rooted treatment	3.072	0.538	2.018	4.127	<0.001
	Unrooted treatment	1.130	0.471	0.208	2.052	0.016
	Competition (disturbance)	1.598	0.479	0.659	2.538	<0.001
	Trampling (disturbance)	2.353	1.650	-0.881	5.588	0.154
	Unknown (disturbance)	1.112	0.451	0.228	1.995	0.014
	Herbivory (disturbance)	3.322	0.779	1.796	4.848	<0.001
	PR out	0.374	0.199	-0.014	0.762	0.058
Survival through the end of 2nd growing season	Intercept	-2.136	0.523	-3.162	-1.111	<0.001
	Rooted treatment	2.493	0.494	1.524	3.461	<0.001
	Unrooted treatment	1.212	0.472	0.288	2.136	0.010
	Competition (disturbance)	1.337	0.447	0.461	2.213	0.003
	Unknown (disturbance)	0.495	0.381	-0.251	1.242	0.193
	Herbivory (disturbance)	1.698	0.635	0.453	2.943	0.007
	PR out	0.293	0.183	-0.066	0.653	0.110

Table 2-3. Estimated marginal means pairwise comparisons of differences between treatment groups for survival at the end of the first growing season, over the first year (spring 2017 to spring 2018), and through the end of the second growing season (second summer).

Response variable	Contrast	Estimate	Standard error	z ratio	P value
Survival at the end of 1st growing season	Direct plant - Unrooted	-0.771	0.412	-1.869	0.062
Survival over the 1st year	Direct plant - Rooted	-3.072	0.538	-5.709	<.001
	Direct plant - Unrooted	-1.130	0.471	-2.402	0.043
	Rooted - Unrooted	1.942	0.436	4.453	<.001
Survival through the end of 2nd growing season	Direct plant - Rooted	-2.493	0.494	-5.046	<.001
	Direct plant - Unrooted	-1.212	0.472	-2.570	0.027
	Rooted - Unrooted	1.281	0.393	3.257	0.003

Table 2-4. Coefficients of the explanatory variables in linear-mixed effects models predicting height growth of balsam poplar cuttings in Blue Rapids Provincial Recreation Area. Includes explanatory variables from the best overall model for height growth 2017 and height growth 2018. Reference categorical variables contained in the intercept for each model are: direct plant (treatment) for height growth 2017; and undamaged (damage) for height growth 2018. Damage was included as a binary variable (damaged and undamaged) in the model.

					Confidence intervals of estimate		
Response variable	Explanatory variable	Estimate	Standard error	t value	Lower	Upper	Pr (>(|t|) value
Height growth 2017	Intercept	-1.774	1.266	-1.402	-4.25	0.70	<0.001
	Unrooted treatment	6.309	1.014	6.219	4.32	8.29	<0.001
	Initial diameter	2.749	0.315	8.724	2.13	3.36	<0.001
	Penetration resistance in	-1.531	0.591	-2.592	-2.69	-0.37	0.011
Height growth 2018	Intercept	1.285	0.226	5.691	0.84	1.73	<0.001
	Damaged	-0.861	0.149	-5.781	-1.15	-0.57	<0.001
	Initial diameter	0.114	0.039	2.949	0.04	0.19	0.004

Table 2-5. Estimated marginal means pairwise comparisons to account for differences between treatment groups for height growth in 2017 and height growth in 2018.

Response variable	Contrast	Estimate	Standard error	Degrees of freedom	t ratio	P value
Height growth 2017	Direct plant - Unrooted	-6.309	1.021	42.79	-6.178	<.001
Height growth 2018	Direct plant - Rooted	-0.309	0.250	86.59	-1.237	0.435
	Direct plant - Unrooted	-0.475	0.245	76.43	-1.935	0.136
	Rooted - Unrooted	-0.166	0.152	56.11	-1.092	0.523

Table 2-6. Results of univariate mixed effect ANOVA models for each abiotic, cover, richness and diversity variable (α= 0.05), testing significance of position, year and the interaction position*year.

Variable	Predictors	Degrees of freedom	F-value	P value
Water table	Position	2	0.000	0.999
	Year	1	54.503	**<.001**
	Position*Year	2	0.010	0.990
Penetration resistance	Position	2	28.896	**<.001**
	Year	1	0.000	1.000
	Position*Year	2	0.000	1.000
Cover				
Mineral soil	Position	2	201.048	**<.001**
	Year	1	5.867	**0.017**
	Position*Year	2	1.523	0.221
Rock[i]	Position	2	17.886	**<.001**
	Year	1	0.001	0.970
	Position*Year	2	0.427	0.653
Litter	Position	2	116.922	**<.001**
	Year	1	2.963	0.087
	Position*Year	2	1.023	0.362
Wood	Position	2	27.193	**<.001**
	Year	1	3.948	**0.049**
	Position*Year	2	0.447	0.640
Moss	Position	2	11.963	**<.001**
	Year	1	0.567	0.452
	Position*Year	2	1.120	0.329
Shrub	Position	2	19.333	**<.001**
	Year	1	4.496	**0.035**
	Position*Year	2	0.062	0.940
Forb	Position	2	11.813	**<.001**
	Year	1	2.224	0.138
	Position*Year	2	0.367	0.693

Variable	Predictors	Degrees of freedom	F-value	P value
	Position	2	6.373	**0.002**
Graminoid	Year	1	0.023	0.878
	Position*Year	2	0.823	0.441
	Position	2	56.826	**<.001**
Native	Year	1	1.223	0.270
	Position*Year	2	2.024	0.135
	Position	2	16.916	**<.001**
Non-native	Year	1	2.711	0.102
	Position*Year	2	3.105	**0.048**
	Position	2	102.018	**<.001**
Total vegetation	Year	1	8.202	**0.005**
	Position*Year	2	5.569	**0.005**
Richness				
	Position	2	20.502	**<.001**
Total species	Year	1	10.434	**0.001**
	Position*Year	2	0.332	0.718
	Position	2	13.586	**<.001**
Shrub	Year	1	0.003	0.957
	Position*Year	2	0.281	0.756
	Position	2	15.302	**<.001**
Forb	Year	1	4.963	**0.027**
	Position*Year	2	0.271	0.763
	Position	2	19.412	**<.001**
Graminoid	Year	1	21.658	**<.001**
	Position*Year	2	2.860	0.060
Diversity				
	Position	2	11.142	**<.001**
All plants	Year	1	4.096	**0.045**
	Position*Year	2	0.348	0.706
	Position	2	11.192	**<.001**
Shrub	Year	1	0.290	0.591
	Position*Year	2	0.448	0.640

Variable	Predictors	Degrees of freedom	F-value	P value
Forb	Position	2	12.570	**<.001**
	Year	1	1.213	0.272
	Position*Year	2	0.030	0.970
Graminoid	Position	2	20.446	**<.001**
	Year	1	17.640	**<.001**
	Position*Year	2	2.098	0.126

[i] Square root-transformed for analysis

Table 2-7. Post hoc pairwise comparisons between positions for abiotic, cover, richness and diversity variables (Bonferroni-adjusted α= 0.05). Includes variables from Table 2-6 that had a significant effect of position.

Variable	Predictors	Estimate	Standard error	Degrees of freedom	t ratio	P value
Penetration resistance	10 m - edge	0.017	0.110	158	0.151	0.987
	10 m - trail	-0.725	0.111	158	-6.528	**<.001**
	Edge - trail	-0.742	0.111	158	-6.678	**<.001**
Cover						
Mineral soil	10 m - edge	-15.283	3.659	158	-4.177	**<.001**
	10 m - trail	-70.565	3.693	158	-19.106	**<.001**
	Edge - trail	-55.282	3.693	158	-14.968	**<.001**
Rock[i]	10 m - edge	-0.401	0.198	158	-2.021	0.110
	10 m - trail	-1.180	0.200	158	-5.891	**<.001**
	Edge - trail	-0.779	0.200	158	-3.890	**<.001**
Litter	10 m - edge	4.133	3.959	158	1.044	0.550
	10 m - trail	55.019	3.996	158	13.769	**<.001**
	Edge - trail	50.885	3.996	158	12.735	**<.001**
Wood	10 m - edge	4.133	3.959	158	1.044	0.550
	10 m - trail	55.019	3.996	158	13.769	**<.001**
	Edge - trail	50.885	3.996	158	12.735	**<.001**
Moss	10 m - edge	2.355	2.174	158	1.083	0.526
	10 m - trail	10.280	2.196	158	4.681	**<.001**
	Edge - trail	7.925	2.196	158	3.608	**0.001**
Shrub	10 m - edge	11.203	2.873	158	3.899	**<.001**
	10 m - trail	17.813	2.902	158	6.139	**<.001**
	Edge - trail	6.610	2.902	158	2.278	0.062
Forb	10 m - edge	1.248	3.541	158	0.353	0.934
	10 m - trail	-14.437	3.576	158	-4.037	**<.001**
	Edge - trail	-15.685	3.576	158	-4.387	**<.001**

Variable	Predictors	Estimate	Standard error	Degrees of freedom	t ratio	P value
Native	10 m - edge	10.776	3.073	158	3.507	**0.002**
	10 m - trail	32.519	3.102	158	10.483	**<.001**
	Edge - trail	21.743	3.102	158	7.009	**<.001**
Graminoid	10 m - edge	-11.887	3.330	158	-3.57	**0.001**
	10 m - trail	-6.089	3.363	158	-1.811	0.165
	Edge - trail	5.798	3.363	158	1.724	0.199
Richness						
Total species	10 m - edge	-3.633	0.826	158	-4.398	**0.001**
	10 m - trail	1.557	0.834	158	1.867	0.152
	Edge - trail	5.191	0.834	158	6.221	**<.001**
Shrub	10 m - edge	0.383	0.253	158	1.513	0.287
	10 m - trail	1.300	0.256	158	5.083	**<.001**
	Edge - trail	0.917	0.256	158	3.585	**0.001**
Forb	10 m - edge	-2.217	0.602	158	-3.682	**<.001**
	10 m - trail	1.073	0.608	158	1.764	0.185
	Edge - trail	3.289	0.608	158	5.410	**<.001**
Graminoid	10 m - edge	-1.783	0.288	158	-6.188	**<.001**
	10 m - trail	-1.075	0.291	158	-3.695	**<.001**
	Edge - trail	0.708	0.291	158	2.434	**0.042**
Diversity						
All plants	10 m - edge	-1.500	0.444	158	-3.381	**0.003**
	10 m - trail	0.533	0.448	158	1.188	0.462
	Edge - trail	2.033	0.448	158	4.536	**<.001**
Shrub	10 m - edge	0.345	0.168	158	2.057	0.102
	10 m - trail	0.799	0.169	158	4.720	**<.001**
	Edge - trail	0.454	0.169	158	2.683	**0.022**
Forb	10 m - edge	-1.200	0.329	158	-3.647	**0.001**
	10 m - trail	0.393	0.332	158	1.184	0.464
	Edge - trail	1.593	0.332	158	4.795	**<.001**

Variable	Predictors	Estimate	Standard error	Degrees of freedom	t ratio	P value
Graminoid	10 m - edge	-1.212	0.194	158	-6.257	**<.001**
	10 m - trail	-0.830	0.196	158	-4.245	**0.001**
	Edge - trail	0.382	0.196	158	1.952	0.128

[i] Square root-transformed for analysis

Table 2-8. Post hoc pairwise comparisons between groups of the interaction position*year for cover of native, non-native and total vegetation. Includes variables from Table 2-6 that had a significant effect of the interaction position*year. Significant P values (α= 0.05) bolded.

Variable	Predictors	Estimate	Standard error	Degrees of freedom	t ratio	P value
	10 m 2017 - Edge 2017	-10.960	4.346	158	-2.522	0.124
	Edge 2017 - Trail 2017	-16.252	4.385	158	-3.706	**0.004**
	10 m 2017 - Trail 2017	-27.212	4.385	158	-6.205	**<.001**
	10 m 2018 - Edge 2018	-10.587	4.346	158	-2.436	0.150
Non-native	Edge 2018 - Trail 2018	-27.251	4.385	158	-6.214	**<.001**
	10 m 2018 - Trail 2018	-37.838	4.385	158	-8.628	**<.001**
	Trail 2017 - Trail 2018	-10.079	4.421	158	-2.280	0.208
	Edge 2017 - Edge 2018	0.920	4.346	158	0.212	0.999
	10 m 2017 - 10 m 2018	0.547	4.346	158	0.126	1.000
	10 m 2017 - Edge 2017	-21.820	15.564	158	-1.402	0.726
	Edge 2017 - Trail 2017	156.286	15.709	158	9.949	**<.001**
	10 m 2017 - Trail 2017	134.466	15.709	158	8.560	**<.001**
	10 m 2018 - Edge 2018	-85.853	15.564	158	-5.516	**<.001**
Total vegetation	Edge 2018 - Trail 2018	156.765	15.709	158	9.980	**<.001**
	10 m 2018 - Trail 2018	70.911	15.709	158	4.514	**<.001**
	Trail 2017 - Trail 2018	-47.141	15.830	158	-2.978	**0.039**
	Edge 2017 - Edge 2018	-47.620	15.564	158	-3.060	**0.031**
	10 m 2017 - 10 m 2018	16.413	15.564	158	1.055	0.898

Table 2-9. Results of indicator species analysis. Species that had an indicator value >20 and were significant (α= 0.05) are listed in order by descending indicator value within each plant community type. Origin and mean cover values for each indicator species are also provided. Abbreviation and Latin binomial of species can be found on Appendix 4.

Predictor	Species	Origin	Observed indicator value (IV)	Mean	Standard deviation	P value
Trail	*Plantago major* L.	Non-native	55.3	25.9	5.3	<. 0.001
Trail	*Trifolium hybridum* L.	Non-native	31.2	19.2	4.2	0.015
Trail	*Juncus bufonius* L.	Native	27.7	8.4	3.3	<. 0.001
Trail	*Populus balsamifera* L.	Native	25.1	14.6	4.4	0.027
Trail	*Melilotus albus* Medikus	Non-native	22.0	13.4	4.2	0.041
Edge	*Equisetum arvense* L.	Native	55.6	34.9	4.0	<. 0.001
Edge	*Taraxacum officinale* F.H. Wiggers	Non-native	45.7	32.7	3.9	0.006
Edge	*Bromus inermis* Leysser	Non-native	40.8	20.9	4.5	0.002
Edge	*Anemonastrum canadense* (L.) Mosyakin	Native	34.2	20.3	4.8	0.012
Edge	*Symphyotrichum ciliolatum* (Lindley) Á. Löve & D. Löve	Native	30.9	18.7	4.6	0.019
Edge	*Solidago canadensis* L.	Native	30.0	21.4	4.2	0.044
Edge	*Bromus pumpellianus* Scribner	Native	29.2	15.7	4.5	0.011
Edge	*Symphoricarpos occidentalis* Hooker	Native	27.6	19.4	4.2	0.048
Edge	*Achillea millefolium* L.	Native	28.7	12.8	3.9	0.003
Edge	*Vicia americana* Muhlenberg ex Willdenow	Native	24.4	13.1	3.8	0.015
10 m	*Maianthemum stellatum* (L.) Link	Native	46.4	24.8	4.4	<. 0.001
10 m	*Rosa acicularis* Lindley	Native	36.9	27.4	4.9	0.047
10 m	*Cornus sericea* L.	Native	31.8	22.6	4.4	0.039

Predictor	Species	Origin	Observed indicator value (IV)	Mean	Standard deviation	P value
10 m	*Rubus pubescens* Rafinesque	Native	31.5	18.3	4.2	0.011
10 m	*Picea glauca* (Moench) Voss	Native	31.2	13.1	4.5	0.003
10 m	*Eurybia conspicua* (Lindley) G.L. Nesom	Native	29.1	15.6	4.2	0.010
10 m	*Rubus idaeus* L.	Native	27.0	14.6	4.2	0.015
10 m	*Aralia nudicaulis* L.	Native	23.5	13.1	3.8	0.017
10 m	*Petasites frigidus* var. *palmatus* (Aiton) Cronquist	Native	22.5	13.0	3.8	0.026
10 m	*Viola renifolia* A. Gray	Native	21.6	12.0	4.3	0.031

Chapter 3. Conclusions

This study provided novel insights into the restoration of decommissioned OHV trails in wet and riparian areas by exploring how survival and height growth of balsam poplar (*Populus balsamifera*) hardwood cuttings were influenced by planting treatment and other factors in the field setting. It also demonstrated how plant communities differ on trails and in their adjacent edges when compared to relatively undisturbed forests, in the short-term after restoration. These insights can be relevant to Parks management decisions and also to the oil and gas industry, as the creation of seismic lines and access roads may result in similar trail systems that are required to be reclaimed after the end of activities. However, because planting in this study was limited to a single year (2017), care must be taken about generalizing the results, since such results could reflect the specific conditions of that year rather than patterns that could consistently occur across years.

I investigated how three planting treatments for restoration with balsam poplar cuttings (i.e., direct plant from greenwood cuttings, unrooted from dormant cuttings, and rooted from dormant cuttings) performed in terms of survival and height growth in the first two field growing seasons, including the influence of initial diameter, soil penetration resistance, depth to water table, disturbance and damage. As hypothesized, the rooted cuttings generally performed better, followed by unrooted cuttings, and the direct plant treatment had the poorest performance in terms of survival and initial summer height growth.

In order to establish balsam poplar plants in decommissioned wet and riparian crossings previously disturbed by OHV use quickly and efficiently, I recommend the use of the rooted treatment, as it had the highest survival rates after plants were transferred to the field; the initial cutting mortality occurred in the greenhouse rather than in the field after the cuttings were planted. Also it provides the advantage of the planted cutting have an existing root system when transferred to the field, which helps buffer it from the short-term edaphic conditions. The rooted cutting also has the benefit that it only requires 10 cm per cutting, so the same amount of material collected can provide more cuttings for restoration use. I recommend the planting of 25% more cuttings in the greenhouse than the number expected to be planted in the field, in order to account for greenhouse mortality. The use of this treatment is recommended for restoration projects that include the possibility (e.g., funding) of growing plants in a greenhouse. For projects in which this is not possible, but where project time allows for winter collection of cuttings, I recommend the use of the unrooted dormant cuttings. Since this treatment presented about 50% survival at the end of the second growing season, I recommend to densely plant unrooted balsam poplar cuttings, in order to achieve the desired future density of trees in the area to be restored. And lastly, for projects with both budget and time limitations (e.g., where a project has to happen immediately without pre-planning in the previous winter), I recommend the use of direct plant, although direct plant cuttings should be planted in very high densities in the field.

My results indicated a positive influence of higher initial diameters of balsam poplar cuttings on survival and height growth. The optimal diameters ranged from 4 to 8 mm, based on the conditions of the cuttings used in this study, so preference should be given to collecting samples with initial diameter in that range, although the use of cuttings larger than 8 mm could also be beneficial. Post-collection, when preparing the original cuttings for planting, it is recommended to use the base of collected stems rather than the tops, in order to take advantage of larger initial diameter, expected higher carbohydrate reserves and more root primordia. In this study, I did not control for the position the cutting was taken from (i.e., base, centre, top) in the original branch or quantify carbohydrate reserves. Future studies could explore the association of survival rates and height growth with initial diameter of the cutting, carbohydrate reserves and position in the original branch.

My findings indicated that disturbance/stress and damage are important factors that influence survival and height growth, particularly in the second growing season. In this sense, future research that includes the combination of cutting treatments with trampling, herbivore and weed control methods could help determine their relative influences on treatment performance. Overall, the results suggest linkages between penetration resistance and establishment of balsam poplar cuttings in the field, as while penetration resistance negatively affected plant height growth, it positively influenced survival. This suggests that the present levels of compaction on restored trails at Blue Rapids Provincial Recreation Area (BRPRA) allow the initial formation of small root systems but inhibit root growth after the point in which roots are larger than soil pores. The influence of different compaction levels could be explored in future studies. I recommend the preparation of soils before planting (e.g., mounding, soil inversion; McCarthy et al. 2017 provides soil preparation methods according to site condition needs). This would reduce top soil compaction on trails to be restored after OHV use. Depth to water table was not associated with performance of balsam poplar cuttings established on the wet and riparian areas focused on in this study. This reinforces the use of poplar cuttings as a good alternative for restoration programs in these areas, since water availability (e.g., too little or in excess) would not act as limiting factor. This also means that costs associated with irrigation can be avoided in these restoration programs.

I also investigated if trail, adjacent edge and its adjacent undisturbed forest had different plant community compositions on enhanced OHV trails, as well as if there were differences at each position within two years after trail enhancement. I found short-term changes in total vegetation cover, diversity and richness indices on trails and edges. Trails presented an increase in forbs and non-native species, whereas edges presented a substantial increase in graminoids. This shift in vegetation communities on trails and edge is of special concern, since these environments differ from the forested areas, which comprised mostly native woody indicator species. These findings indicate that plant community composition differs on trails, adjacent edges and forested areas. Therefore, active restoration could be an option to speed the establishment of native woody species on trails within BRPRA. The planting of native balsam poplar cuttings on these trails, associated with the other recommendations above, might help to initiate (or speed) the process of plant succession on these areas. In addition to that, as trees grow, they might prevent further unsanctioned OHV use on decommissioned trails.

A long-term monitoring program should be created to help understand the successional trajectories of the decommissioned OHV trails that have been actively restored in this study, along with those that have been left to passively restore. Monitoring should occur at the end of each growing season to help determine if treatment has a longer-term effect on survival and height growth of the balsam poplars planted. This could provide insights on whether the advantage of the rooted treatment over the two unrooted cutting treatments remains over the longer term. Likewise, monitoring is needed to better understand if initial diameter and treatment influence performance only during initial establishment of the cutting or if the advantages last for the long-term. Finally, follow-up monitoring activities can also compare actively restored crossings with passively restored crossings within Blue Rapids Provincial Recreation Area. It may be that in both cases the sites will follow a similar trajectory, but in the case of passive restoration it will occur much more slowly. A monitoring program could help us understand if, and how, the planting of balsam poplar cuttings influences plant community composition on actively restored trails compared with those that are left to recover without intervention.

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Appendices

Appendix 1. Characteristics of sampled crossings, including: total number of plots per treatment on each crossing, plot position on trail (plot layout on trail is found in Figure 2-6) size (meters), general characteristics, Blue Rapids Wetland Project enhancement activity initiated in 2016, and OHV use status during the planting and propagation study (summers of 2017 and 2018). Five cuttings of a given treatment were planted per plot. The "Park's code" is the original crossing name recognized by Blue Rapids Provincial Recreation Area.

Target crossing	Park's code	Treatment applied	Plots (n)	Plots position on trail	Size (m)	General characteristics	Enhancement activity	OHV use
1	C2	Direct plant	0	-	Length: 15.1	Riparian area. Watercourse present.	Re-contoured[i]	Isolated event.
		Unrooted	0	-	Width: 2.1			
		Rooted	4	2, 3, 4, 5				
		Willow	6	1 to 6				
2	C5	Direct plant	2	1, 4	Length: 24	Wet area. Seasonal ponding.	Re-contoured, swales filled, access blocked	Absent on trail. OHV use adjacent to 2 last plots.
		Unrooted	5	3, 6, 7, 10, 12	Width: 3			
		Rooted	5	2, 5, 8, 9, 11				
3	C7	Direct plant	2	3, 5	Length: 30	Wet area. Seasonal ponding.	Re-contoured	Absent on trail. OHV use adjacent to 2 first plots.
		Unrooted	5	1, 6, 7, 10, 12	Width: 5			
		Rooted	5	2, 4, 8, 9, 11				
4	C9	Direct plant	2	5, 7	Length: 20.4	Wet area. Seasonal ponding.	Re-contoured	Absent
		Unrooted	2	3, 8	Width: 4			
		Rooted	2	4, 6				
		Willow	2	1, 2				
5	C11	Direct plant	2	1, 7	Length: 21	Wet area. Wooded swamp.	Access blocked	Absent
		Unrooted	2	3, 8	Width: 2			
		Rooted	2	2, 6				
		Willow	2	4, 5				

Target crossing	Park's code	Treatment applied	Plots (n)	Plots position on trail	Size (m)	General characteristics	Enhancement activity	OHV use
6	C13	Direct plant	2	2, 6	Length: 55	Wet and riparian area. Wooded swamp to west and open water to east.	Re-contoured, erosion control (coconut matting)	Isolated event.
		Unrooted	5	3, 4, 8, 11, 14	Width: 2.5			
		Rooted	5	1, 5, 7, 12, 13				
		Willow	2	9, 10				
7	C20	Direct plant	2	3, 5	Length: 26.3	Riparian area. Watercourse present.	Re-contoured, access control	Absent.
		Unrooted	4	2, 6, 7, 9	Width: 3.2			
		Rooted	4	1, 4, 8, 10				
8	C30	Direct plant	2	1, 4	Length: 15.8	Wet area.	Access blocked	Absent.
		Unrooted	2	3, 5	Width: 3			
		Rooted	2	2, 6				
9	C3	Direct plant	2	2, 5	Length: 11	Wet area. Seasonal ponding.	Re-contoured, swales filled	Destructive unsanctioned use.
		Unrooted	3	3, 6, 7	Width: 6.5			
		Rooted	3	1, 4, 8				
10	C4	Direct plant	2	4, 6	Length: 16	Riparian area. Watercourse present.	Re-contoured, access control	Destructive unsanctioned use.
		Unrooted	2	5, 7	Width: 4.5			
		Rooted	2	6, 8				
		Willow	2	1, 2				
11	C14	Direct plant	0	-	Length: 17	Riparian area. Watercourse present.	Re-contoured, access control	Destructive unsanctioned use.
		Unrooted	0	-	Width: 5			
		Rooted	0	-				
		Willow	6	1 to 6				
12	C16	Direct plant	2	1, 4	Length: 35.5	Wet area. Seasonal ponding.	Access control	Destructive unsanctioned use.
		Unrooted	4	3, 6, 12, 14	Width: 3			
		Rooted	4	2, 5, 7, 13				
		Willow	4	8, 9, 10, 11				
13	C17	Direct plant	2	1, 6	Length: 18	Wet area. Seasonal ponding.	Access control	Destructive unsanctioned use.
		Unrooted	2	2, 5	Width: 3			
		Rooted	2	3, 4				

Target crossing	Park's code	Treatment applied	Plots (n)	Plots position on trail	Size (m)	General characteristics	Enhancement activity	OHV use
14	C21	Direct plant	2	3, 10	Length: 30	Wet area. Seasonal ponding.	Re-contoured, filled in low spots	Destructive unsanctioned use.
		Unrooted	2	1, 8	Width: 3			
		Rooted	2	2, 9				
		Willow	4	4, 5, 6, 7				
15	C29	Direct plant	2	1, 4	Length: 28	Wet area. Seasonal ponding.	Access control	Destructive unsanctioned use.
		Unrooted	3	3, 8, 10	Width: 2.5			
		Rooted	3	2, 5, 9				
		Willow	2	6, 7				

[l] Recontouring: filling in low spots with adjacent soil material or lowering of adjacent trail sections to level with low area. Would consider drainage of surface water.

Appendix 2. UTM coordinates and dimensions of crossings targeted for restoration. Crossings 1 - 8 were used for data analyses. Crossings 9 to 15 had unsanctioned OHV use that resulted in damage; thus, they were excluded from analyses. The "Park's code" is the original crossing name recognized by Blue Rapids Provincial Recreation Area. The coordinates format is UTM, Zone 11N map datum NAD 1983.

		UTM Coordinates				
Crossing	Park's code	Easting	Northing	Length (m)	Width (m)	Area (m^3)
1	C2	638967	5896559	15.8	2.1	33.18
2	C5	639398	5896553	24.0	3.0	72.00
3	C7	639327	5896289	30.0	5.0	150.00
4	C9	639234	5895941	20.4	3.7	76.50
5	C11	639447	5895924	21.0	2.0	42.00
6	C13	638959	5895092	55.0	2.5	137.50
7	C20	637331	5892694	26.3	3.2	84.16
8	C30	639396	5896564	15.8	2.8	44.24
9	C3	639070	5896519	11.0	6.5	71.50
10	C4	639454	5896734	16.0	4.5	72.00
11	C14	638660	5894752	17.0	5.2	89.25
12	C16	638752	5894399	35.5	3.0	106.50
13	C17	638166	5894252	18.0	2.7	48.60
14	C21	637214	5892602	30.0	3.0	90.00
15	C29	637431	5892565	28.0	2.5	70.00
					Total Area	1187.43

Appendix 3. UTM coordinates of collection sites in Blue Rapids Provincial Recreation Area. The coordinates format is UTM, Zone 11N map datum NAD 1983.

	UTM Coordinates		
Collection Site	Easting	Northing	Elevation (m)
1	0637427	5892916	750
2	0637839	5893270	751
3	0637819	5893913	751
4	0638737	5894377	792

Appendix 4. Species found in sample plots. Abbreviations represent the first three letters of genus and species names, and the first two letters of subspecies (if appropriate). Nomenclature follows Moss (1983) and USDA plants and Canadensys (VasCan) databases (for most recent accepted name). "Origin" is in relation to the province of Alberta and follows USDA plants and Canadensys (VasCan) database (2018) with confirmation from Alberta Native Plant Council (2018) and ACIMS Plant Species Ranking (2015).

Abbreviation	Species	Origin
Trees		
ALNINCTE	*Alnus incana* (L.) Moench subsp. *tenuifolia* (Nuttall) Breitung	Native
BETPAP	*Betula papyrifera* Marshall	Native
PICGLA	*Picea glauca* (Moench) Voss	Native
POPBAL	*Populus balsamifera* L.	Native
POPTRE	*Populus tremuloides* Michx.	Native
ULMAME	*Ulmus americana* L.	Non-native
Shrubs		
CORCOR	*Corylus cornuta* Marshall	Native
CORSER	*Cornus sericea* L.	Native
ELACOM	*Elaeagnus commutata* Bernhardi ex Rydberg	Native
LINBOR	*Linnaea borealis L.*	Native
LONINV	*Lonicera involucrata* (Rich.) Banks ex Sprengel	Native
PRUVIR	*Prunus virginiana* L.	Native
RIBTRI	*Ribes triste* Pallas	Native
ROSACI	*Rosa acicularis* Lindley	Native
ROSWOO	*Rosa woodsii* Lindley	Native
RUBIDA	*Rubus idaeus* L.	Native
SALIX_SPP	*Salix* spp. L.	Native
SHECAN	*Shepherdia canadensis* (L.) Nuttall	Native
SYMOCC	*Symphoricarpos occidentalis* Hooker	Native
VIBEDU	*Viburnum edule* (Michaux) Rafinesque	Native
VIB_SPP	*Viburnum* spp. L.	Native
Forbs		
ACHMIL	*Achillea millefolium* L.	Native

Abbreviation	Species	Origin
ACHALP	*Achillea alpina* Linnaeus	Native
ACTRUB	*Actaea rubra* (Aiton) Willdenow	Native
ANECAN	*Anemonastrum canadense* (L.) Mosyakin	Native
APOAND	*Apocynum androsaemifolium* L.	Native
ARANUD	*Aralia nudicaulis* L.	Native
ASTCIC	*Astragalus cicer* L.	Non-native
BARORT	*Barbarea orthoceras* Ledebour	Native
CAMROT	*Campanula rotundifolia* L.	Native
CERBEE	*Cerastium beeringianum* Chamisso & Schlechtendal	Native
CHAANG	*Chamerion angustifolium* (L.) Scopoli subsp. *angustifolium*	Native
CICMAC	*Cicuta maculata* var. *angustifolia* Hooker	Native
CIRARV	*Cirsium arvense* (L.) Scopoli	Non-native
COLLIN	*Collomia linearis* Nuttall	Native
COMPAL	*Comarum palustre L.*	Native
CORCAN	*Cornus canadensis* L.	Native
EPICIL	*Epilobium ciliatum* Rafinesque subsp. *ciliatum*	Native
EQUARV	*Equisetum arvense* L.	Native
EQUHYE	*Equisetum hyemale* L.	Native
EQUPAL	*Equisetum palustre* L.	Native
EQUPRA	*Equisetum pratense* Ehrhart	Native
EQUSCI	*Equisetum scirpoides* Michaux	Native
EQUSYL	*Equisetum sylvaticum* L.	Native
EQUVAR	*Equisetum variegatum* Schleicher ex F. Weber & D. Mohr	Native
ERIPHI	*Erigeron philadelphicus* L.	Native
EURCON	*Eurybia conspicua* (Lindley) G.L. Nesom	Native
FRAVES	*Fragaria vesca* L.	Native
FRAVIR	*Fragaria virginiana* Miller	Native
GALBOR	*Galium boreale* L.	Native
GALTET	*Galeopsis tetrahit* L.	Non-native
GALTRIFI	*Galium trifidum* L.	Native

Abbreviation	Species	Origin
GALTRIFL	*Galium triflorum* Michaux	Native
GENAMA	*Gentianella amarella* (L.) Börner	Native
GERBIC	*Geranium bicknellii* Britton	Native
GEUMAC	*Geum macrophyllum* Willdenow	Native
GEURIV	*Geum rivale* L.	Native
HERMAX	*Heracleum maximum* W. Bartram	Native
HIEUMB	*Hieracium umbellatum* L.	Native
LATOCH	*Lathyrus ochroleucus* Hooker	Native
LATVEN	*Lathyrus venosus* Muhlenberg ex Willdenow	Native
LYSTHY	*Lysimachia thyrsiflora L.*	Native
LONVIL	*Lonicera villosa* (Michx.) Schult.	Native
MAISTE	*Maianthemum stellatum* (L.) Link	Native
MELALB	*Melilotus albus* Medikus	Non-native
MELOFF	*Melilotus officinalis* (L.) Lamarck	Non-native
MENARV	*Mentha arvensis* L.	Native
MERPAN	*Mertensia paniculata* (Aiton) G. Don	Native
MITNUD	*Mitella nuda* L.	Native
MOELAT	*Moehringia lateriflora* (L.) Fenzl	Native
PETFRIPA	*Petasites frigidus* var. *palmatus* (Aiton) Cronquist	Native
PETFRIVI	*Petasites frigidus* var. ×*vitifolius* (Greene) Cherniawsky	Native
PLAHYP	*Platanthera hyperborea* (L.) Lindley	Native
PLAMAJ	*Plantago major* L.	Non-native
POTNOR	*Potentilla norvegica* L.	Native
PROTRA	*Prosartes trachycarpa* S. Watson	Native
PRUVUL	*Prunella vulgaris* L.	Native
PYRASA	*Pyrola asarifolia* Michaux	Native
RANMAC	*Ranunculus macounii* Britton	Native
RIBLAC	*Ribes lacustre* (Persoon) Poiret	Native
RIBOXY	*Ribes oxyacanthoides* L.	Native
RORPAL	*Rorippa palustris* (L.) Besser	Native
RORSIN	*Rorippa sinuata* (Nuttall) Hitchcock	Native
RUBPUB	*Rubus pubescens* Rafinesque	Native

Abbreviation	Species	Origin
SCUGAL	*Scutellaria galericulata* L.	Native
SOLCAN	*Solidago canadensis* L.	Native
SONARV	*Sonchus arvensis* L.	Non-native
STAPAL	*Stachys palustris* L.	Native
SYMBOR	*Symphyotrichum boreale* (Torrey & A. Gray) Á. Löve & D. Löve	Native
SYMCIL	*Symphyotrichum ciliolatum* (Lindley) Á. Löve & D. Löve	Native
SYMLAE	*Symphyotrichum laeve* (L.) Á. Löve & D. Löve var. laeve	Native
SYMPUN	*Symphyotrichum puniceum* (L.) Á. Löve & D. Löve var. *puniceum*	Native
TANVUL	*Tanacetum vulgare* L.	Non-native
TAROFF	*Taraxacum officinale* F.H. Wiggers	Non-native
THADAS	*Thalictrum dasycarpum* Fischer & Avé-Lallemant	Native
THAVEN	*Thalictrum venulosum* Trelease	Native
TRIHYB	*Trifolium hybridum* L.	Non-native
TRIPRA	*Trifolium pratense* L.	Non-native
TRIREP	*Trifolium repens* L.	Non-native
URTDIO	*Urtica dioica* L.	Native
VICAME	*Vicia americana* Muhlenberg ex Willdenow	Native
VIOADU	*Viola adunca* Sm.	Native
VIOCAN	*Viola canadensis* L.	Native
VIONEP	*Viola nephrophylla* Greene	Native
VIOREN	*Viola renifolia* A. Gray	Native
Sedges		
CARATH	*Carex atherodes* Sprengel	Native
CARAUR	*Carex aurea* Nuttall	Native
CARCRA	*Carex crawfordii* Fernald	Native
CARDEW	*Carex deweyana* Schweinitz	Native
CARLASAM	*Carex lasiocarpa* subsp. *americana* (Fernald) D. Löve & J.-P. Bernard	Native
CARSIC	*Carex siccata* Dewey	Native

Abbreviation	Species	Origin
ELEACI	*Eleocharis acicularis* (L.) Roemer & Schultes	Native
SCIMIC	*Scirpus microcarpus* J. Presl & C. Presl	Native
Rushes		
JUNALP	*Juncus alpinoarticulatus* Chaix	Native
JUNBAL	*Juncus balticus* Willdenow	Native
JUNBUF	*Juncus bufonius* L.	Native
JUNNOD	*Juncus nodosus* L.	Native
JUNTEN	*Juncus tenuis* Willdenow	Native
Grasses		
AGREXA	*Agrostis exarata* Trinius	Native
AGRSCA	*Agrostis scabra* Willdenow	Native
BECSYZ	*Beckmannia syzigachne* (Steudel) Fernald	Native
BROCIL	*Bromus ciliatus* L.	Native
BROINE	*Bromus inermis* Leysser	Non-native
BROPUM	*Bromus pumpellianus* Scribner	Native
CALCAN	*Calamagrostis canadensis* (Michaux) Palisot de Beauvois	Native
DANINT	*Danthonia intermedia* Vasey	Native
ELYCAN	*Elymus canadensis* L.	Native
ELYREP	*Elymus repens* (L.) Gould	Non-native
ELYTRA	*Elymus trachycaulus* (Link) Gould ex Shinners	Native
FESIDA	*Festuca idahoensis* Elmer	Native
GLYSTR	*Glyceria striata* (Lamarck) Hitchcock	Native
PHAARU	*Phalaris arundinacea* L.	Native
PHLPRA	*Phleum pratense* L.	Non-native
POACOM	*Poa compressa* L.	Non-native
POAINT	*Poa interior* Rydberg	Native
POAPAL	*Poa palustris* L.	Native
POAPRA	*Poa pratensis* L.	Native
POA_SPP	*Poa* spp. L.	Native
SCHPURPU	*Schizachne purpurascens* (Torrey) Swallen subsp. *purpurascens*	Native