Throwing shade: performance of native Pacific Northwest shrubs in shading out invasive reed canarygrass (*Phalaris arundinacea*)

by

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Abstract

Reed canarygrass (*Phalaris arundinacea*) is an invasive grass common in wetlands and riparian areas throughout the Pacific Northwest. It is highly adaptable and resistant to many control methods, but is vulnerable to shading. We sought to control reed canarygrass by establishing desirable native shrubs to overtop and shade it. Plots were rototilled, mulched, live-staked, and monitored for 2-6 growing seasons. We tested 1) effective planting densities by live-staking hardhack (*Spiraea douglasii*) at 50, 30, and 15 cm spacing, 2) relative species performance by planting hardhack, red-osier dogwood (*Cornus sericea*), and thimbleberry (*Rubus parviflorus*), all at 30 cm densities, and 3) alternative site preparation methods by using cardboard mulch or excavating the top 20 cm of topsoil. Higher planting density significantly reduced reed canarygrass cover and biomass. Both hardhack and red-osier dogwood successfully suppressed reed canarygrass, though thimbleberry did not. No significant differences between site preparation methods were observed.

Keywords: reed canarygrass; *Phalaris arundinacea*; *Spiraea douglasii*; invasive species management; live staking; planting density

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List of Abbreviations

BBRP	Boundary Bay Regional Park
GSPE	Growing seasons post-establishment
RCG	Reed canarygrass (*Phalaris arundinacea*)

Chapter 1. Introduction

1.1. Reed Canarygrass Invasion and Genetics

Reed canarygrass (*Phalaris arundinacea*; hereafter referred to as RCG) is a clonal grass which has become invasive throughout much of North America (Galatowitsch et al. 1999). It grows 0.5-2.0 m tall and has a hollow stem (Baltensperger & Kalton 1958). Leaves are flat and broad, typically 0.5-2.5 cm wide (Vincent et al. 1986). It has three-flowered spikelets on 7-40 cm long panicles (Carlson et al. 1996). Seeds are 4 mm long on average and weigh approximately 1 mg (Grime et al. 1981). The plant is perennial and grows from seed, rhizome, or tiller buds (Apfelbaum & Sams 1987; Sheaffer & Marten 1995). The species is highly variable in colour and physical traits, making morphological identification of different varieties impossible (Baltensperger & Kalton 1958; Merigliano & Lesica 1998).

Figure 1.1: RCG panicle during flowering (source: Matt Lavin).

RCG is native to Europe, Asia, and parts of North America, and it has also been introduced in many of the world's temperate areas (Galatowitsch et al. 1999; Maurer & Zedler 2002; Lavergne & Molofsky 2004; Jakubowski et al. 2013). With rare exception, most RCG growing wild in North America today are invasive, hybrid genotypes (Lavergne & Molofsky 2004; Lavoie et al. 2005; Jakubowski et al. 2013). Native North American varieties were believed to have been uncommon prior to invasion (Townsend & Hebda 2013), and have since declined further, now relegated to parts of Alaska and northern Canada (Lavoie et al. 2005; Jakubowski et al. 2013). It is believed RCG was first introduced to North America by European settlers in the mid to late 1800s (Lavergne & Molofsky 2004, 2007). It was primarily used to stabilize soil and provide forage for livestock (Galatowitsch et al. 1999; Lindig-Cisneros & Zedler 2002; Kim et al. 2006). Numerous introductions of different Eurasian varieties have followed since then, prompting further hybridization and increasing invasion success (Lindig-Cisneros & Zedler 2002; Lavergne & Molofsky 2004, 2007). Townsend and Hebda (2013) conducted pollen analysis in sediment cores extracted from Swan Lake, British Columbia, and found that RCG quickly went from being absent to abundant in the early 1900s, displacing many native species.

1.2. Reed Canarygrass Growth and Impacts

Invasive RCG is now common and especially problematic in the Pacific Northwest, the central Prairies, the American Midwest, and the Great Lakes region, where it displaces native species (Galatowitsch et al. 1999; Perry & Galatowitsch 2004; Kinney 2011). It thrives in a range of wet and moist areas, and can be found in marshes, wet meadows, swamps, prairie potholes, fields, stream banks, floodplains, lake shores, bogs, fens, ditches, and roadsides (Tu 2004; Lavergne & Molofsky 2007). It is especially common in disturbed, eutrophic, and human-modified sites such as degraded wetlands, abandoned fields, and ditches (Kercher & Zedler 2004; Jakubowski et al. 2013). Plants attain the highest biomass in saturated soils and are capable of enduring overland flooding (Gomm 1978; Ashworth 1997).

In invaded areas, RCG commonly establishes dense, monotypic stands (Galatowitsch et al. 1999; Kercher & Zedler 2004; Lavergne & Molofsky 2004). When

dominant, the grass forms a dense canopy which precludes establishment of other species (Maurer & Zedler 2002; Fierke & Kaufman 2005, 2006; Kim et al. 2006; Miller et al. 2008). Stems often lodge following seed dispersal, which further densifies the canopy (Mueller 1941; Apfelbaum & Sams 1987). Once established, RCG spreads rapidly, and can quickly alter the structure, composition, and function of previous vegetative communities (Lavergne & Molofsky 2004). In riparian areas, RCG can become so dense that it reduces surface water movement, impedes salmonid movement, and increases sedimentation by replacing streambank vegetation (Sheaffer & Marten 1995, Werner & Zedler 2002; Lavergne & Molofsky 2004; Miller et al. 2008). Werner and Zedler (2002) describe how an invasion of RCG and cattail reduced surface water movement leading to increased sedimentation, degradation of soil microstructure, and the loss of *Carex* tussocks. RCG's impacts to soil include reduced soil organic matter and bulk density, which are important for the growth of native species (Werner & Zedler 2002).

RCG also impacts animal communities and diversity. The plant is rarely used for nesting by birds (Littlefield 1999; Niemuth 2000). It has poor food value, as high alkaloid concentrations make the plant unpalatable to most animals (Coulman 1995). Spyreas et al. (2010) found RCG reduced arthropod, small mammal, and plant diversity in wetlands. Weilhoefer et al. (2017) found distinctly different arthropod communities in invaded and uninvaded wetlands, with significantly lower biodiversity in the former. The decrease in overall plant and animal biomass and diversity disrupts food webs and nutrient cycling, further degrading wetland ecosystems (Miller et al. 2008; Weilhoefer et al. 2017).

Continued introduction of novel varieties throughout North America in recent decades may be contributing to increased plasticity and invasive traits (Lavoie et al. 2005; Lavergne & Molofsky 2007). Much of RCG's success as an invader is attributable to its rapid and vigorous growth (Lavergne & Molofsky 2004). Shoots emerge early in the spring and sometimes in late winter in the Pacific Northwest, giving the plant an early competitive advantage (Hutchison 1992; Lavergne & Molofsky 2004; Waggy 2010). Growth is rapid throughout spring, and rhizomes also exhibit high growth rates when they begin development in the summer (Apfelbaum & Sams 1987; Hutchison 1992). Growth is especially vigorous in areas high in nitrogen and phosphorus (Kercher & Zedler 2004). Plants can tolerate variable water levels, soil moisture regimes, nutrient regimes, sedimentation, and photoperiods (Brandle 1983; Townsend & Hebda 2013). They can also tolerate moderate levels of salinity (McWilliams et al. 2007).

Seed production is high compared to most grasses (Kellogg et al. 2003). Tu (2004) found RCG in the Pacific Northwest produces an average of 600 seeds per inflorescence. Seeds are quick to germinate. Leck (1996) observed significant germination within 2 weeks, and Grime et al. (1981) observed some germination within 9 days. Germination rates are high (Leck 1996; Vecrin et al. 2007). Seeds are capable of tolerating a range of environmental conditions and stressors (Lindig-Cisneros & Zedler 2001). They can tolerate variable temperatures (Apfelbaum & Sams 1987), lighting (Lindig-Cisneros & Zedler 2001), soil depths (Comes et al. 1981; Leck 1996), soil moisture (Leck 1996; Ashworth 1997; Kellogg et al. 2003), flooding regimes (Leck 1996), and relative humidity (Leck 1996). General seed viability remains unclear. Grime et al. (1981) found germination rates declined from 87% at 3 months to 65% at 1 year. Toole and Brown (1946) observed 1% of seeds buried 22 inches below soil to germinate after 31 years, though viability is generally accepted to be much shorter (Leck 1996). Generally, germination rates are highest in saturated (but not flooded) soils (Ashworth 1997; Maurer & Zedler 2002, Lavergne & Molofsky 2004), high-light conditions (Lindig-Cisneros & Zedler 2001), and shallow soil depths (Leck 1996). Seeds are gravity-dispersed, though they are buoyant and may also be dispersed via waterways (Comes et al. 1978; Coops & van der Velde 1995).

Rhizome growth is also consistently high, and rhizomes and roots form a dense mat in the soil (Kellogg et al. 2003; Lavergne & Molofsky 2004). Klimešová and Šrůtek (1995) found over half the root system in monotypic stands was contained within the upper 20 cm of soil. Mueller (1941) found most rhizomes contained within the top 3-13 cm of soil, and a total of 117 m of rhizomes were observed within one square meter of soil. Rhizomes may develop within 26 days after germination (Apfelbaum & Sams 1987). They are also resistant to competition, still sprouting at high rates under 30% plant cover (Bonilla-Warford & Zedler (2002). RCG can also root from nodes, often following the lodging of stems (Mueller 1941; Apfelbaum & Sams 1987), and most plant fragments are capable of rooting (Sheaffer & Marten 1995; Maurer & Zedler 2002).

RCG is particularly problematic in restored wetlands. Restoration prescriptions frequently leave exposed soil which is sensitive to invasion (Adams & Galatowitsch 2005). A study of restored prairie potholes in Iowa, Minnesota, and South Dakota found that 41 of 62 wetlands were dominated by RCG within 10 years of restoration (Mulhouse & Galatowitsch 2003). Projects like these often occur on sites with a long history of

cultivation, excavation, or drainage, making these legacies of disturbance difficult to overcome (Lindig-Cisneros & Zedler 2002).

1.3. Control Methods

A number of practices have been used to control RCG infestations. Chemical control is common, with glyphosate frequently sprayed (Hovick & Reinartz 2007; Miller et al. 2008). This approach is generally considered the most time- and cost-efficient, and effective for short term results (Apfelbaum & Sams 1987; Miller et al. 2008). In one study, yearly application of glyphosate effectively controlled 94-96% of RCG five months after the second application (Miller et al. 2008). Hovick and Reinartz (2007) found that herbicide alone performed poorly, though outcomes improved considerably when combined with plowing or mowing. Others have cautioned against using herbicide as a long-term solution, as plants are quick to re-establish (Apfelbaum & Sams 1987). Herbicide requires frequent and ongoing applications to remain effective and it may not be feasible in certain areas, such as near water bodies (Stockhouse et al. 2000).

Mechanical control is also common, utilizing physical means such as cutting, mowing, tilling, or seedhead trimming. Miller et al. (2008) found yearly mowing achieved 72-73% RCG control. Control by mowing and tilling can be challenging as seeds, rhizomes, or plant fragments left in the soil are likely to re-establish (Apfelbaum & Sams 1987). Hovin et al. (1973) found that seedhead trimming was successful when compared to a control which cut seedheads after seeding, with 4 of 12 clonal stands completely failing to establish after this treatment. Others have failed to control RCG using this method (Apfelbaum & Rouffa 1983), possibly because it does target rhizomes.

Some have experimented with burying plants under enough soil to prevent re-establishment. Seed banks for RCG are generally well developed, and have been found exceeding 5,000 seeds/m^2 in the upper 2 cm of soil (Leck & Simpson 1994). Ensuring the area of concern is sufficiently buried is crucial. Leck (1996) found burying seeds with 5 cm of soil reduced germination rates by only 20% compared to surface rates. Sites should also not be disturbed for several years after treatment, as buried seeds have been observed germinating after 30 years once excavated (Toole & Brown 1946).

Another method of control is covering or solarization. This involves smothering live plants and/or seedbanks under black or transparent plastic or rubber tarps. Tarps use solar energy to increase soil temperatures and kill weeds or their seeds. This practice has achieved mixed results. In one case, clipping RCG at ground level and applying a plastic tarp for two growing seasons reduced RCG but failed to eliminate it (Apfelbaum & Rouffa 1983).

Flooding has also been tried as a form of cultural control, which aims to achieve cost-effective, long-term control by modifying environmental conditions (Pedigo & Rice 2014). In one experiment, flooding decreased RCG biomass by 73% and lowered survival (Maurer & Zedler 2002). However, RCG's adaptability for variable soil moisture regimes and hydroperiods makes flooding a challenging approach. Flooding must be either permanent or of sufficient duration. Kercher and Zedler (2004) found constant flooding reduced RCG in prairie mesocosms, though RCG became more dominant when subjected to early-season or intermittent flooding. RCG attains higher biomass when soil is saturated and when water levels fluctuate (Gomm 1978; Kercher & Zedler 2004). As for seeds, Comes et al. (1978) found 24 months of inundation was sufficient to reduce seed germination to 0%. However, seeds are buoyant and may disperse to new, uninfested areas. Less is known about the effects of flooding on rhizomes, and the duration of flooding required to render all plant parts incapable of regenerating is unclear.

Burning is another form of cultural control which has proven suitable in some cases. Apfelbaum and Rouffa (1983) found limited success in controlling RCG in an infested park with a 2-3 year burn cycle. However, the grass persisted in some parts, especially disturbed margins within the park. Hutchison (1992) also cautions against prescribed burns in areas where a native seedbank is depleted or absent, which are prone to reinvasion. The timing of burns is also important. Howe (1995) tested 3-year burn cycles, with one treatment conducted in the spring and another in the summer. He found that while RCG biomass decreased following spring burns, it increased following summer burns.

A more promising approach to effective long-term cultural control is shading, which has been described as the "Achilles heel" of RCG (Hovick & Reinartz 2007), and has been shown to reduce tiller numbers (Smith et al. 1990). A number of authors have

advocated the use of woody plants for the purpose of shading RCG growth and spread (e.g., Kim et al. 2006; Hovick & Reinartz 2007; Miller et al. 2008). Light availability during summer months is a strong predictor of RCG biomass (Kercher & Zedler 2004). Forman (1998) found that although aboveground biomass did not respond to shading, belowground biomass was significantly reduced, up to 81%. Shading also helps reduce germination and establishment, and may contribute to depleting RCG seed banks. When light was decreased by 96% in a Minnesota wetland, RCG seedling establishment was reduced by 87% (Iannone & Galatowitsch 2008). Lindig-Cisneros and Zedler (2001) tested germination rates in darkness compared to different light intensities and photoperiods. They observed germination rates between 67 and 91% under light, but only 1.2% in darkness. Another study found no germination in darkness (Leck 1996). A few studies have attempted to control RCG by establishing desirable woody species, yielding promising results. For instance, a study which planted willow stakes at 0.6 m spacing saw RCG biomass decrease by 45 and 68% following the first and second growing seasons, respectively (Kim et al. 2006). Hovick and Reinartz (2007) argued that most sites could eliminate RCG within 10-20 years following these methods.

One limitation of this approach is that trees and larger shrubs may not be suitable for establishment or use in certain sites and may be incompatible with certain ecotypes or restoration goals. In urban areas, zoning requirements, land-use plans, and infrastructure such as hydroelectric right-of-ways may preclude tree establishment. In these cases, only shrubs or graminoids may be suitable. Literature on using graminoids to shade out RCG has shown promise in limited contexts. Some success has been observed in species with denser growth and more early-season canopy development (Lindig-Cisneros & Zedler 2002; Maurer & Zedler 2002), but research is considered lacking. Hovick and Reinartz (2007) found that shrubs generally had higher survival rates than trees when established amongst treated RCG, and argued they were a more promising approach.

1.4. Project Background

We sought to determine whether locally-abundant native shrubs were suitable for shading and outcompeting RCG and to test the effects of planting density and site

preparation methods on competitive outcomes. Our first site is a hydroelectric right-of-way in Bear Creek Park, Surrey, BC, where no tall vegetation is permitted. Here, Experiment 1 was established in 2015 and designed to test the effects of planting density, a research question similar to Kim et al. (2006). Using willows, Kim et al. (2006) tested planting densities of 1.21 m, 0.91 m, and 0.60 m, and 0 (control, no willows), and found a clear trend: higher densities were more effective in suppressing RCG. We expanded on this by using a densely-growing shrub and by including higher-density treatments, up to 15 cm. We used live stakes of hardhack (*Spiraea douglasii*), a native shrub common to wetlands throughout the Pacific Northwest. Hardhack roots readily from live stakes when harvested and planted during dormancy. Carr and Merchant (1981) observed 95% rooting success when using live stakes with rooting hormone, and rates exceeding 90% are common for stakes even without hormone (Darris 2002). Easily propagated wetland plants like hardhack are often readily found on site, making them convenient for restoration projects (Tu 2004). This also has the advantage of ensuring regionally-appropriate genetic stock adapted to local site conditions.

Building on this idea, Experiment 2 (established 2016 at the same site) aimed to test the performance of other shrubs relative to hardhack. For this experiment, density was standardized and hardhack was compared to red-osier dogwood (*Cornus sericea*) and thimbleberry (*Rubus parviflorus*). Dogwood was chosen because, like hardhack, it is known to root readily from hardwood cuttings (Darris 2002). Thimbleberry was chosen primarily because it grows densely as it spreads by rhizomes and has large leaves up to 20 cm across (Maxwell 1990) which overlap each other, growth characteristics conducive to shade. Thimbleberry grows up to 2 m tall, though generally shorter (Oleskevich et al. 1996), and its similarity in height with RCG made it useful for investigating whether there is a critical height at which competitive outcomes with RCG shift. We used plugs for thimbleberry so we could also test the performance of nursery stock instead of live stakes.

Experiment 3 (established 2018) was designed as an identical replicate for Experiment 1, testing hardhack planting density in a different site, and Experiment 4 (established 2019) was designed to test alternative methods of site preparation and whether they affect competitive outcomes. Both are located at Boundary Bay Regional Park, Delta, BC, where the study area is being managed as an old-field site and thus limited to shrubs and herbaceous vegetation. Hovick and Reinartz (2007) demonstrated

the importance of site preparation to give desirable woody plants an early competitive advantage when establishing amongst RCG. They treated experimental plots to different combinations of herbicide, plowing, burning, and/or mowing. Across the 23 species they sought to establish, mean survival in experimental treatments averaged between 37.6 and 50.5%, and only 7.1% in the untreated control. We prepared most treatments by rototilling and mulching with woodchips prior to planting. Experiment 4 sought to test how this method compared to treatments for topsoil excavation and cardboard mulch. Topsoil excavation was chosen because it has the advantage of removing most live RCG material while also depleting most of the seedbank, giving desired plants time to establish before adjacent RCG encroaches or seedbanks replenish. Cardboard was chosen as an inexpensive, easily applied mulch alternative.

1.5. Goals & Objectives

This research will evaluate the effects of planting density, shrub species, and site preparation methods on the ability of native shrubs to outcompete RCG in the short- and long-term. Comparable studies (e.g., Kim et al. 2006; Hovick & Reinartz 2007; Miller et al. 2008) have showed promising development of these ideas, though monitoring was no more than 2 years and long-term success remains unclear. This is a common criticism of studies on RCG control (Adams & Galatowitsch 2005), though this study will use data collected over a longer time frame. I also hope to build on the successes of these studies by determining if the use of shading is the most effective mechanism for lasting RCG control. As with trees and larger shrub species, it is hoped the native shrub species chosen will effectively suppress RCG. These findings will have implications for site managers facing RCG infestations, and will allow them to better predict the outcome of different management strategies. Specifically, the objectives of the study are as follows:

1. Identify the most effective planting density of hardhack stakes
2. Measure the performance of hardhack relative to red-osier dogwood and thimbleberry
3. Measure the performance of cardboard mulch and topsoil excavation as site preparation methods

Chapter 2. Site Descriptions

2.1. Bear Creek Park

2.1.1. Surficial Materials and Soils

Bear Creek Park in Surrey, BC, along with much of the surrounding area, is composed of Capilano sediments, which include raised marine and fluvial deposits, marine and non-marine silts, and riverine sand (Armstrong & Hicock 1980). The experimental plots are located in an area of raised beach composed of medium to coarse sand deposits up to 5 m thick and high in fossilized marine shell casts (Armstrong & Hicock 1980; Turner et al. 1997). Surficial materials are primarily contributed by fluvial and colluvial activity (Sandquist 2000).

A soil description at this site is provided by Hennigar et al. (2019). Two test pits were dug and analyzed, one adjacent to blocks A-F, another in blocks G-L. Both pits had an LFH horizon extending from 0-10 cm depth, an A horizon extending from 0 to between 20 and 30 cm, and a B horizon extending down to 60 cm. The LFH horizon was composed of a rhizo-mull, with a gradual transition to an Ah layer. The A horizon was a dark brown colour and contained many fine RCG roots. The B horizon ranged from light brown to light grey and from silty loam to silty clay loam. It contained few roots and less than 5% course fragments. Gleying and mottling were evident, though the water table was not encountered up to 60 cm at the time of digging in autumn. A previous group of students observed the water table at 20 cm depth during winter (Browne et al. 2015).

2.1.2. Climate, Topography, and Hydrology

Bear Creek Park is located within the coastal western hemlock very dry maritime subzone (CWHxm) biogeoclimatic subzone. The area experiences warm, dry summers and mild, wet winters, typical of CWHxm subzones (Table 2.1).

Figure 2.1: Climatic characteristics of a typical Coastal Western Hemlock dry maritime subzone biogeoclimatic zone compared to climate characteristics of Surrey, BC.

	Mean precipitation (mm)			Mean temperature (°C)		
	Annual	Driest month	Wettest month	Annual	Warmest month	Coldest month
CWHxm[1]	1505	39	251	9.3	17.0	1.8
Surrey, BC[2]	1405	39	214	9.6	17.3	2.0

[1] Pojar et al. 1991
[2] Surrey climate data from https://en.climate-data.org/north-america/canada/british-columbia/surrey-4345/

The eponymous Bear Creek runs south through the park before joining the Serpentine River approximately 5 km southeast. A number of smaller creeks and streams feed into Bear Creek on the northwest side of the park. The park area is relatively flat and the study site is approximately 28 m elevation, gently sloping southwest towards Bear Creek.

2.1.3. Vegetation

The immediate study area is predominantly a RCG monoculture, though some other plants are present in low abundance. Bentgrass (*Agrostis* sp.), vetch (*Vicia* sp.), field horsetail (*Equisetum arvense*), creeping buttercup (*Ranunculus repens*), Canada thistle (*Cirsium arvense*), and common rush (*Juncus effusus*) were found amongst the RCG and in study plots. Within study plots, field horsetail is the only non-experimental plant which exceeds trace amounts (i.e., < 1%, though only in a few plots). A few open patches dominated by common rush are present amongst the RCG, though it is unclear why they remain uninvaded. Patches of hardhack, red-osier dogwood, and black twinberry (*Lonicera involucrata*) are present near the study plots.

2.1.4. Disturbance History and Stressors

The earliest known surveys in this region were undertaken in 1858, associated with clearing for settlement, agriculture, and commercial logging (Sandquist 2000). The last of Surrey's large trees had been logged by the 1930s, and Bear Creek Park was established shortly after. Although some trees have regrown and intact forest cover exists in much of the park, the hydro right-of-way which spans the study area is subject to regular clearing. Any large trees or shrubs are removed in a width of approximately 100 m.

Invasive species and the ongoing loss and degradation of wetland habitat threaten the park. Between 1827 and 1990, the proportion of land area with wetlands dropped from 10% to 1% in the Lower Mainland (Boyle et al. 1997). The park remains under threat from urbanization, which has impacted hydrology and nutrient cycling in remaining wetland areas (Faulkner 2004). Urbanization and soil compaction have decreased soil permeability, and an estimated 80% of rain becomes overland flow (Parsons 2015). As a result, stormwater flows have caused flooding, scour, and erosion in the Bear Creek watershed (Parsons 2015). Invasive species are abundant in the park, and a number of aggressive invaders such as Himalayan blackberry (*Rubus armeniacus*) and knotweed (*Fallopia* spp.) are abundant near the study area. Like other urban parks, heavy use of trails creates edge effects and visitors may act as dispersal agents for invasive seed.

2.2. Boundary Bay Regional Park

2.2.1. Surficial Materials and Soils

The site at Boundary Bay Regional Park (BBRP) is located on modern lowland sediments deposited since the last glaciation. Armstrong and Hicock (1979) mapped much of BBRP as flood-deposited sandy to silt loam overbank sediments, typically less than 2 m thick and overlying deeper riverine sediments. A layer of porous, glaciomarine-derived sediments, likely containing groundwater reservoirs, are present underneath (Turner et al. 1997). Due to the study site's close proximity to the tidal flats, the plots are

also influenced by marine shore sediments. These deposits are sand to sandy loam and up to 2 m thick (Armstrong & Hicock 1979).

The study plot area at Boundary Bay is mapped as Seaview soil (Canadian Soil Information Service 2013), a poorly-drained Rego Humic Gleysol with a high water table (ponding occurs at the study site during part of the year). The study area is textured as silt loam. Deeper layers of Seaview are increasingly saline and often textured as sand or gravelly sand (Luttmerding 1981). Luttmerding (1981) describes the vegetation typical of Seaview soils as mainly salt tolerant grasses, forbs, and shrubs.

2.2.2. Climate, Topography, and Hydrology

BBRP is located within the coastal Douglas-fir moist maritime subzone (CDFmm). BBRP experiences warm, dry summers and mild, wet winters (Table 2.2).

Figure 2.2: Climatic characteristics of Tsawwassen, BC.

	Mean precipitation (mm)			Mean temperature (°C)		
	Annual	Driest month	Wettest month	Annual	Warmest month	Coldest month
Tsawwassen, BC[1]	1035	30	153	10.0	16.9	3.5

[1] Tsawwassen climate data from https://en.climate-data.org/north-america/canada/british-columbia/tsawwassen-12274/

BBRP is flat to gently undulating, with slopes of less than 5%, and mostly 3 m or less in elevation (Luttmerding 1981). A small salt marsh estuary is present on the north end of the park, fed by two small channels: one entering from the other end of the park towards the southeast, and another entering from Tsawwassen to the southwest. Otherwise, there are no waterways in the park, though depressional and ephemeral wetlands are present. Drainage in the park ranges from poorly drained to moderately

well drained, depending on soils, though it is always slow and water is mostly removed through subsurface or groundwater flow (Canadian Soil Information Service 2013).

2.2.3. Vegetation

The inland area of BBRP, where the study site is located, contains old field, wetland, and woodland communities. Weber (2007) provides a detailed list of plants surveyed in BBRP. Most of the site is unforested, though some common trees include Douglas-fir (*Pseudotsuga menziesii*), paper birch (*Betula papyrifera*), black hawthorn (*Crataegus douglasii*), and black cottonwood (*Populus trichocarpa*). Common shrubs include hardhack, Himalayan blackberry, Scotch broom (*Cytisus scoparius*), Nootka rose (*Rosa nutkana*), common snowberry (*Symphoricapos albus*), and salmonberry (*Rubus spectabilis*). The site is mostly herbaceous plants, of which RCG is the most dominant, abundant throughout the park.

The experimental plots are situated within a RCG monoculture stand approximately 9 ha in size. Himalayan blackberry, Canada thistle, and bull thistle are abundant near the experimental plots. Purple loosestrife (*Lythrum salicaria*), Scotch broom, and two unidentified grass species were also observed during field visits, though they were not abundant.

2.2.4. Disturbance History and Stressors

Most of the arable land in Delta was deforested and converted to agricultural production by the late 1800s, though much of this land has now been urbanized in recent decades (Murray 2006). Although the study site and BBRP have a legacy of agriculture, soils on site have not been significantly altered, and are considered to be in their native state (Canadian Soil Information System 2013).

The park area faces many of the same stressors from urbanization as Bear Creek (section 2.1.4), though the park is less developed. There are fewer trails, structures, and paved areas within the park, and fewer areas accessible to the public. Invasive plants are arguably the greatest stressor in Boundary Bay. Park staff and volunteers regularly conduct invasive species removals, mapping, and other

maintenance. Managers at BBRP are looking to preserve much of the park, including the study area, as old-field habitat, which is threatened by invasive species, noxious weeds, and woody encroachment. Of these, RCG is a top priority and managers are currently using mowing, tillage, and herbicide to combat it in an attempt to restore old-field habitat species complexes.

Chapter 3. Methods

3.1. Experimental Design

We established four experiments, two at Bear Creek Park (Experiments 1 and 2) and two at Boundary Bay Regional Park (Experiments 3 and 4). The location of the study plots within each site was chosen as a suitable spot by previous students, who determined favourable experimental conditions in the uniform RCG cover and microtopography (Browne et al. 2015; Driediger et al. 2016; Smith & McGuffin 2018; Hennigar et al. 2019; Froc et al. 2020). The students also noted a uniform water table after digging 2 test pits and assessing the site's soil profile. They also designed the experimental treatments and collected data from 2015 to 2019.

Table 3.1: Summary of experiments, locations, date of plot setup, treatment, site preparation used, and shrub establishment.

#	Experiment	Location	Date Installed	Treatments	Site Preparation	Percent Shrub Establishment[1]
1	Hardhack Density Trials	BC	Feb 2015	H15	Rototill, woodchips	38
				H30		47
				H50		48
2	Native Shrub Trials	BC	Mar 2016	HH30	Rototill, woodchips	84
				ROD30		94
				TH30		92
3	Density Replicate	BB	Feb 2018	H15	Rototill, woodchips	16[2]
				H30		14[2]
				H50		22[2]
4	Site Preparation Trials	BB	Feb 2019	CB+ROD30	Rototill, cardboard	17
				CB+HH30	Rototill, cardboard	21
				EX+HH30	Excavation	29

Note: BC = Bear Creek Park, BB = Boundary Bay Regional Park. Treatment codes represent shrub species, planting density in centimeters, and experimental site preparation methods. H/HH = hardhack, ROD = red-osier dogwood, TH = thimbleberry. CB = cardboard, EX = excavation.

[1] Establishment is recorded as the percentage of surviving shrubs after one growing season.

[2] Due to poor establishment of hardhack in the first growing season of Experiment 3, these plots were restocked in February 2019.

Each experiment had three treatments. A summary of experimental treatments can be found in Table 3.1. Experiment 1, hardhack density trials, contained treatments for hardhack staked at 15, 30, and 50 cm spacing (H15, H30, and H50, respectively). Experiment 2, native shrub trials, contained treatments for hardhack, red-osier dogwood, and thimbleberry, all staked at 30 cm spacing (HH30, ROD30, and TH30, respectively). Experiment 3, hardhack density replicate, contained identical treatments and site preparation as Experiment 1. Experiment 4, site preparation trials, contained treatments for red-osier dogwood with cardboard, hardhack with cardboard, and hardhack in excavated plot, all at 30 cm spacing (CB+ROD30, CB+HH30, and EX+HH30, respectively).

A randomized complete block design was used for all experiments, with each block containing one plot from each treatment. At Bear Creek, blocks A-F correspond to Experiment 1 (Browne et al. 2015) and blocks G-L to Experiment 2 (Driediger et al. 2016). At Boundary Bay, blocks M-R contain Experiments 3 and 4 combined, and thus contain six plots per block (Smith & McGuffin 2018; Hennigar et al. 2019). The layout of treatments within each block was randomized. There are a total of 72 plots, 18 for each experiment. There is a buffer of approximately 2 m between each block. Each plot is 2.5 m by 2.5 m for a total of 7.5 m by 2.5 m per block (or 7.5 m by 5.0 m at Boundary Bay). Following plot establishment, all sampling and monitoring was conducted within the inner 1.5 m by 1.5 m of each plot (Fig. 3.1).

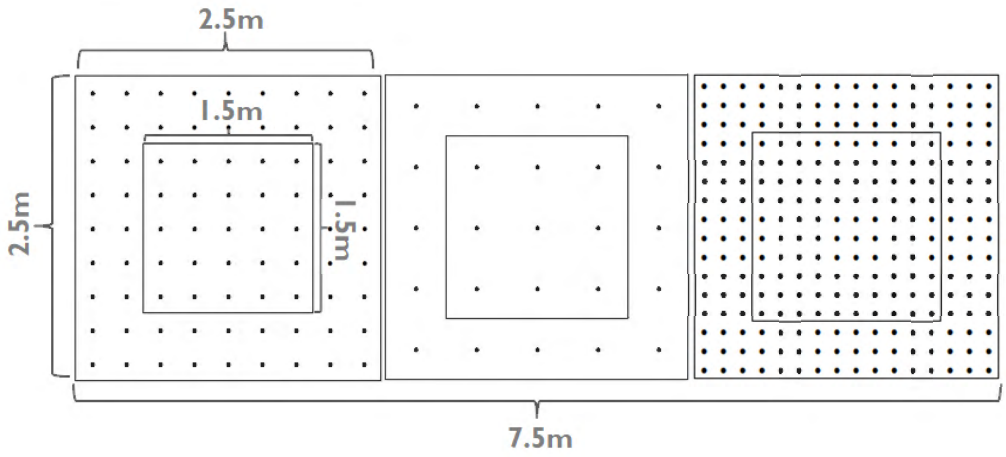

Figure 3.1: Diagram of block layout in density trials. Dots represent planted stakes in 30 cm, 50 cm, and 15 cm spacings (Adapted from Marcoux et al. 2017).

A set of four control plots was also established in Bear Creek in September, 2020 to serve as a reference point for conditions representative of the untreated site. A 10 m by 10 m area adjacent to blocks A-F, and another adjacent to blocks G-L was delineated. Within each of these two areas, two control plots, each 50 cm by 50 cm, were established in a randomized location. Control plots were marked using pin flags.

3.2. Plot Establishment

Site preparation for each experiment was conducted in either February or March each year. Blocks were rototilled by tractor. A layer of woodchip mulch approximately 10 cm thick was then applied (plots for Experiment 3 received an additional “top up” approximately 5 cm thick after 1 year). For Experiment 4, cardboard or excavation were used in place of woodchip mulch. Cardboard was single-face corrugated cardboard and was applied one layer thick, with stakes planted through it. Excavation entailed removal of the top 20 cm of soil prior to staking. In October 2017 at BBRP, prior to establishment of the plots for Experiments 3 and 4, a 5 ha portion of field dominated by RCG was mowed and sprayed with glyphosate by site managers. A full map of blocks and plot layout for both sites can be found in Appendix A (Fig. A.1; A.2).

Stake harvesting took place during late winter to ensure dormancy. Hardhack and dogwood stakes for all treatments were harvested from patches growing at their respective study sites. Some additional stakes gathered from Deer Lake Park, Burnaby, were used for blocks G-L. Thimbleberry plugs were supplied from Nat’s Nursey Ltd. in Langley, British Columbia. All harvested hardhack and dogwood stakes were 0.4 cm to 1.0 cm in diameter, and cut to approximately 50 cm long, on average. Stake harvest took place up to a month before planting. Prior to planting, stakes were tied into bundles and stored under a tarp in a shaded area on site. Some stakes were stored in a campus refrigerator. Planting was done by hand at corresponding planting densities (square spacing). Stakes were planted by hand to a depth of roughly 20 cm and the soil around them lightly compacted. After planting, 20, 15, or 9 randomly selected stakes (for treatments with 15, 30, or 50 cm spacing, respectively) from each sampling plot received individual tags for monitoring.

3.3. Data Collection

3.3.1. Percent Cover

We assessed percent cover using both ocular estimates (yearly from 2015 – 2018) when the plants were shorter (e.g., less than 1.3 m in height) and digital photographs (2020) when the plants were taller. For ocular estimates, the sampling plot was delineated and divided into quadrants to capture the variation across each plot. A Daubenmire quadrat (20 cm by 50 cm frame) was then placed within each quadrant, for a total of four sub-samples averaged per plot. Cover was estimated for RCG, planted shrubs, and other species present in noticeable amounts.

Plants had grown too large for the use of Daubenmire quadrats by 2020. For percent cover estimates for Bear Creek in September 2020, I used a 6-foot stepladder and a mounted camera to photograph the plot center from approximately 3.5 m in height by climbing the stepladder and holding the tripod with the camera over the plot center. For each photograph, a measuring tape was stretched diagonally across the plot from one corner to another and marked with flagging tape at a fixed point along the tape (1.77 m) to indicate plot center. Each image was then edited to be overlain with a square grid that was centered upon the flagging tape and rotated in line with the measuring tape. This systematic placement and orientation eliminated bias in sampling area. The area within the square was then used to estimate cover of both RCG and planted shrubs. To further reduce bias and variation, a student volunteer and I separately recorded estimates, and the average was used for analysis. Percent cover for Boundary Bay was estimated visually in September, 2020, as plants had grown too large for Daubenmire quadrats, but did not yet require a ladder. Estimates were conducted by two trained volunteers, who recorded an agreed upon figure. Percent cover for the control plots at Bear Creek was estimated visually in September, 2020.

3.3.2. Reed Canarygrass Biomass

RCG biomass was harvested from the inner sampling plots at Bear Creek in 2017 and 2020 and from the plots at Boundary Bay in 2020. Sample size varied by time

period. The inner sampling plot was divided into quadrants, and each quadrant was assigned a sampling area by placing a Daubenmire quadrat in a fixed location in each quadrant. In 2020, a single 50 cm by 50 cm sampling quadrat was sampled within the center of the plot. This location ensured there was no overlap with areas previously sampled in 2017 in Bear Creek, without introducing sampling bias. Aboveground biomass within the sampling area was harvested by clipping from the soil surface. Only the sampling year's growth was harvested. Biomass was stored in paper bags.

Each batch of samples was oven-dried at 150 °F for 48 hours. Each batch had samples evenly spaced within the same oven for consistency in drying. Further reductions in mass were found to be negligible past 48 hours. Each sample was then weighed, and the combined weight (minus the mean bag weight times the number of bags) was recorded for each plot. As the total sampled area was variable, biomass values from all sampling years and plots were standardized to g/m^2.

Table 3.2: List of data collected across different experiments and years. Experiment establishment year in brackets (e.g. "est. YEAR"). Tickmarks under a given number represent the collection of a given parameter following that number of growing seasons. All data were collected between August and September for a given sampling year.

Experiment	1 **Hardhack Density** (est. Feb 2015)	2 **Native Shrubs** (est. Mar 2016)	3 **Hardhack Density Replicate** (est. Feb 2018)	4 **Site Prep** (est. Feb 2019)
Growing seasons post-establishment	1 2 3 4 5 6	1 2 3 4 5	1 2 3	1 2
Percent cover (RCG)	☒☑☑☑☒☑	☑☑☑☒☑	☒☑☑	☑☑
Percent cover (shrubs)	☒☒☑☑☒☑	☒☑☑☒☑	☒☒☑	☒☑
RCG biomass	☒☒☑☒☒☑	☒☑☒☒☑	☒☒☑	☒☑
Shrub density	☒☒☒☒☒☑	☒☒☒☒☑	☑☑☑	☑☑

3.3.3. Stake Density and Survivorship

The density of surviving plantings per plot was measured for both sites in September, 2020. At Bear Creek Park, pin flags were used to mark out a 1.0 m by 1.0 m square centered within each plot, and all shrubs of the planted species occurring therein were counted. Because some plants had multiple stems, a rule of 10 cm distance or more was used to delineate stems as belonging to a separate plant. Counts were then averaged to calculate the number of surviving plants per unit area for each treatment.

At Boundary Bay, a square PVC quadrat of 1 m^2 was placed approximately centered within each plot in order to place pin flags delineating the same area. Once flagged, the density of surviving plantings was calculated using the same methods as those described for Bear Creek Park.

3.4. Data Analysis

All statistical analysis was conducted on RStudio version 1.2.5033, running R version 3.6.2 (R Core Team 2020). In addition to the base software, the packages ggpubr, lsmeans, multcomp, multcompView, emmeans, car, and dplyr were used in analysis and/or making graphs. Data were analyzed for normality using a Q-Q plot with fitted residuals. Mean, standard deviation, minimum, and maximum values were recorded for each parameter under analysis. One-way ANOVA were used to test for significant effects from treatment or block on collected data. Significant results were analyzed using a Tukey honest significant differences (HSD) post-hoc test. Welch's two-sample t-tests were used to compare significant changes in a treatment between growing seasons post-establishment (GSPE). Some response variables were also compared using Pearson's product-moment correlation. All significance testing used an α of 0.05, though marginally significant ($0.05 < p < 0.1$) results were also noted.

Chapter 4. Results

4.1. Experiment 1: Hardhack Planting Density Trials

RCG percent cover in the 4 control plots was consistently high, ranging from 80 to 90%, averaging 86 ± 5% (mean ± standard deviation). RCG cover gradually and consistently increased across all treatments from year to year. From 2 to 6 GSPE, RCG mean percent cover per treatment group increased from 12 to 17% for H15, from 14 to 43% for H30, and from 33 to 48% for H50. Treatment had a significant effect on RCG cover during every year of sampling (ANOVA; $p < 0.05$ for 2 and 3 GSPE, $p < 0.01$ for 4 and 6 GSPE; Fig. 4.1a), with a significant difference between the H15 and H50 treatments (Tukey HSD; $p = 0.038$, 0.011, 0.0030, and 0.0038 for 2, 3, 4, and 6 GSPE, respectively; Fig. 4.1a). p values for response variables in all experiments in 2020 are summarized in Table 4.1.

Table 4.1: *p* values (from ANOVA) for all measured parameters in Experiments 1 through 4. All data is from 2020 only.

	Experiment 1: Density Trials	**Experiment 2:** Species Trials	**Experiment 3:** Density Replicate	**Experiment 4:** Site Prep Trials
Parameter	***p*‡**	***p*‡**	***p*‡**	***p*‡**
RCG cover (%)	**	***	*	ns
RCG biomass (g/m^2)	**	**	•	ns
Shrub cover (%)	***	***	***	*
Shrub density (plants/m^2)	***	**	***	•

‡ *p* values represented with significance codes: *** < 0.001, ** < 0.01, * < 0.05, • < 0.1 > ns

RCG biomass in control plots was also consistently higher than that of the density treatments. Biomass in control plots ranged from 1240 to 1500 g/m^2, averaging 1379 ± 125 g/m^2. Biomass was lowest when hardhack was planted at higher densities. Unlike cover, mean RCG biomass declined from 3 to 6 GSPE in all treatments, though

this decline was not significant. There was no significant difference between treatments at 2 GSPE (ANOVA; $p = 0.101$; Fig. 4.2a), but at 5 GSPE there was a significant difference between the H15 and H50 treatments (Tukey HSD; $p = 0.0075$) and between H15 and H30 (Tukey HSD; $p = 0.025$).

Hardhack cover was greatest in higher planting densities. This was significant for 3, 4, and 6 GSPE (ANOVA; $p = 0.013$, 0.011, and 0.00036, respectively; Fig. 4.3a). At 6 GSPE, data showed a significant difference between the H15 and H50 treatments (Tukey HSD; $p = 0.00055$; Fig 4.3a) and between H15 and H30 (Tukey HSD; $p = 0.0019$). RCG cover was inversely proportional to hardhack cover. This was significant across all sampling years (Pearson's correlation; $p = 0.00028$, 0.013, and <0.0001 for 3, 4, and 6 GSPE, respectively; Fig. 4.5).

Live shrub density at 6 GSPE was lower than planting density at the time of establishment in all treatments, though the relationship remained roughly proportional. Mean shrub density at 6 GSPE was 13.8, 7.5, and 5.2 plants/m^2 in H15, H30, and H50 treatments, respectively. Both H50 (Tukey HSD; $p = 0.00041$; Fig. 4.4a) and H30 (Tukey HSD; $p = 0.0059$) were significantly different from H15, though not from each other. Assuming 100% establishment and no self-thinning, densities of 44.5, 11.0, and 4.0 plants/m^2 would be expected for H15, H30, and H50, respectively.

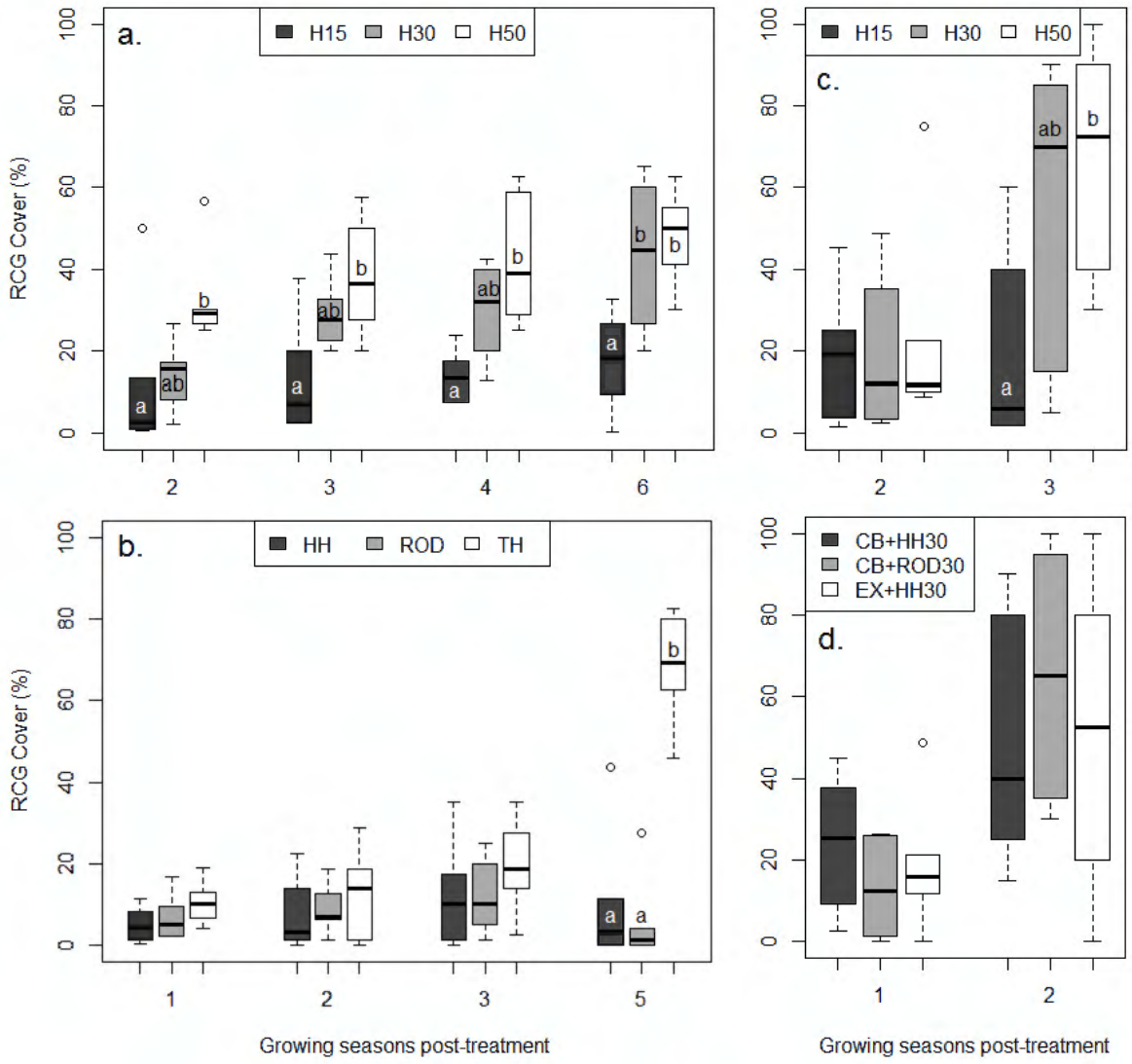

Figure 4.1: **Changes in RCG percent cover across growing seasons since establishment in (a) hardhack density trials, (b) shrub species trials, (c) hardhack density replicates, and (d) site preparation trials. Treatments H15, H30, and H50 represent hardhack planting densities of 15, 30, and 50 cm spacings, respectively. Treatments HH, ROD, and TH represent hardhack, red-osier dogwood, and thimbleberry, respectively. Treatments CB+HH30, CB+ROD30, and EX+HH30 represent hardhack at 30 cm planting density with cardboard mulch, red-osier dogwood at 30 cm planting density with cardboard, and hardhack at 30 cm planting density with topsoil excavation, respectively. Boxplots represent lower and upper quartiles, with whiskers representing minimum and maximum values (excluding outliers). Inner lines represent the median. Letters represent least significant mean groupings at $\alpha = 0.05$ (where applicable).**

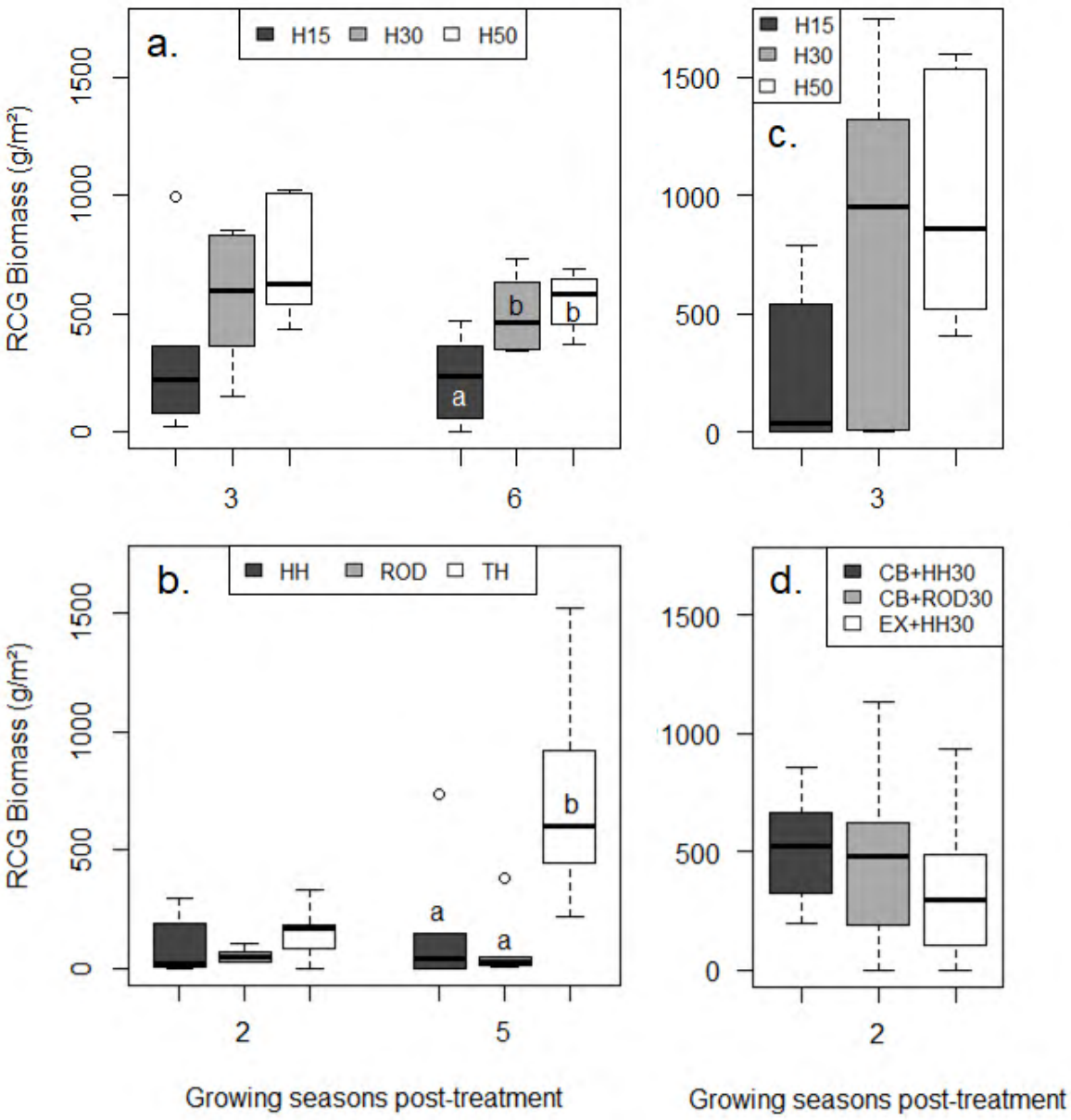

Figure 4.2: **Changes in RCG biomass across growing seasons since establishment in (a) hardhack density trials, (b) shrub species trials, (c) hardhack density replicates, and (d) site preparation trials. Treatments H15, H30, and H50 represent hardhack planting densities of 15, 30, and 50 cm spacings, respectively. Treatments HH, ROD, and TH represent hardhack, red-osier dogwood, and thimbleberry, respectively. Treatments CB+HH30, CB+ROD30, and EX+HH30 represent hardhack at 30 cm planting density with cardboard mulch, red-osier dogwood at 30 cm planting density with cardboard, and hardhack at 30 cm planting density with topsoil excavation, respectively. Boxplots represent lower and upper quartiles, with whiskers representing minimum and maximum values (excluding outliers). Inner lines represent the median. Letters represent least significant mean groupings at $\alpha = 0.05$ (where applicable).**

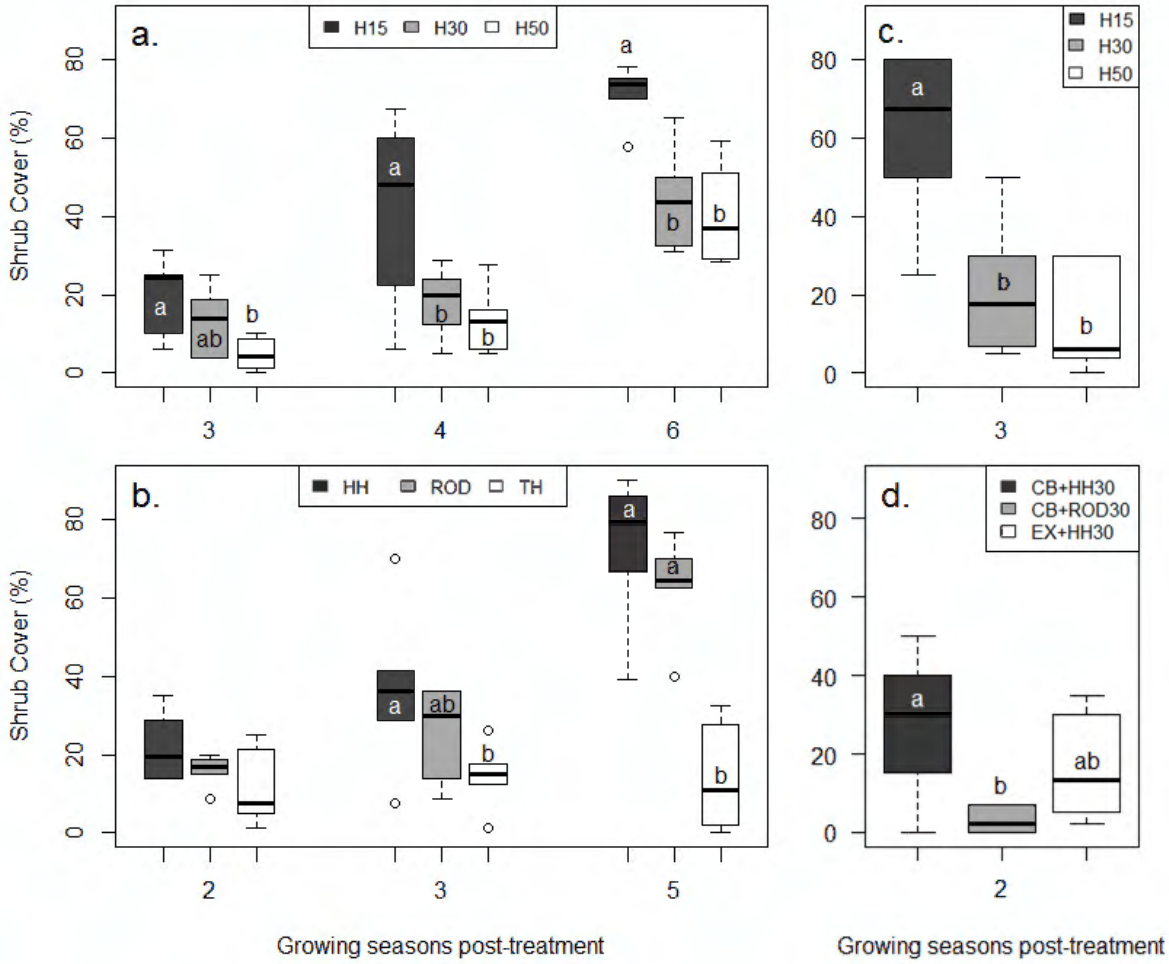

Figure 4.3: **Changes in shrub percent cover across growing seasons since establishment in (a) hardhack density trials, (b) shrub species trials, (c) hardhack density replicates, and (d) site preparation trials. Treatments H15, H30, and H50 represent hardhack planting densities of 15, 30, and 50 cm spacings, respectively. Treatments HH, ROD, and TH represent hardhack, red-osier dogwood, and thimbleberry, respectively. Treatments CB+HH30, CB+ROD30, and EX+HH30 represent hardhack at 30 cm planting density with cardboard mulch, red-osier dogwood at 30 cm planting density with cardboard, and hardhack at 30 cm planting density with topsoil excavation, respectively. Boxplots represent lower and upper quartiles, with whiskers representing minimum and maximum values (excluding outliers). Inner lines represent the median. Letters represent least significant mean groupings at $\alpha = 0.05$ (where applicable).**

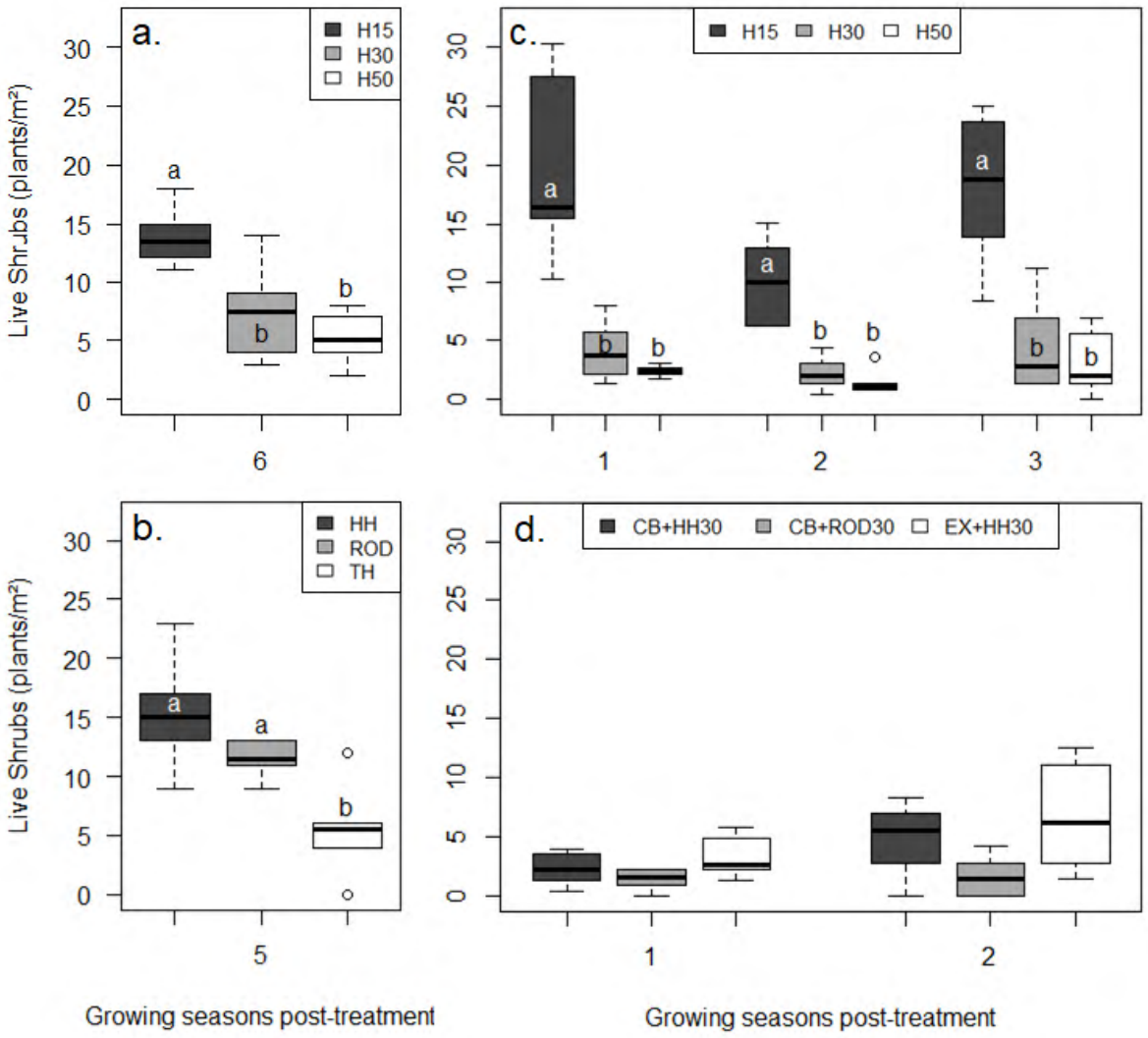

Figure 4.4: **Changes in live shrub density across growing seasons since establishment in (a) hardhack density trials, (b) shrub species trials, (c) hardhack density replicates, and (d) site preparation trials. Treatments H15, H30, and H50 represent hardhack planting densities of 15, 30, and 50 cm spacings, respectively. Treatments HH, ROD, and TH represent hardhack, red-osier dogwood, and thimbleberry, respectively. Treatments CB+HH30, CB+ROD30, and EX+HH30 represent hardhack at 30 cm planting density with cardboard mulch, red-osier dogwood at 30 cm planting density with cardboard, and hardhack at 30 cm planting density with topsoil excavation, respectively. Boxplots represent lower and upper quartiles, with whiskers representing minimum and maximum values (excluding outliers). Inner lines represent the median. Letters represent least significant mean groupings at $\alpha = 0.05$ (where applicable).**

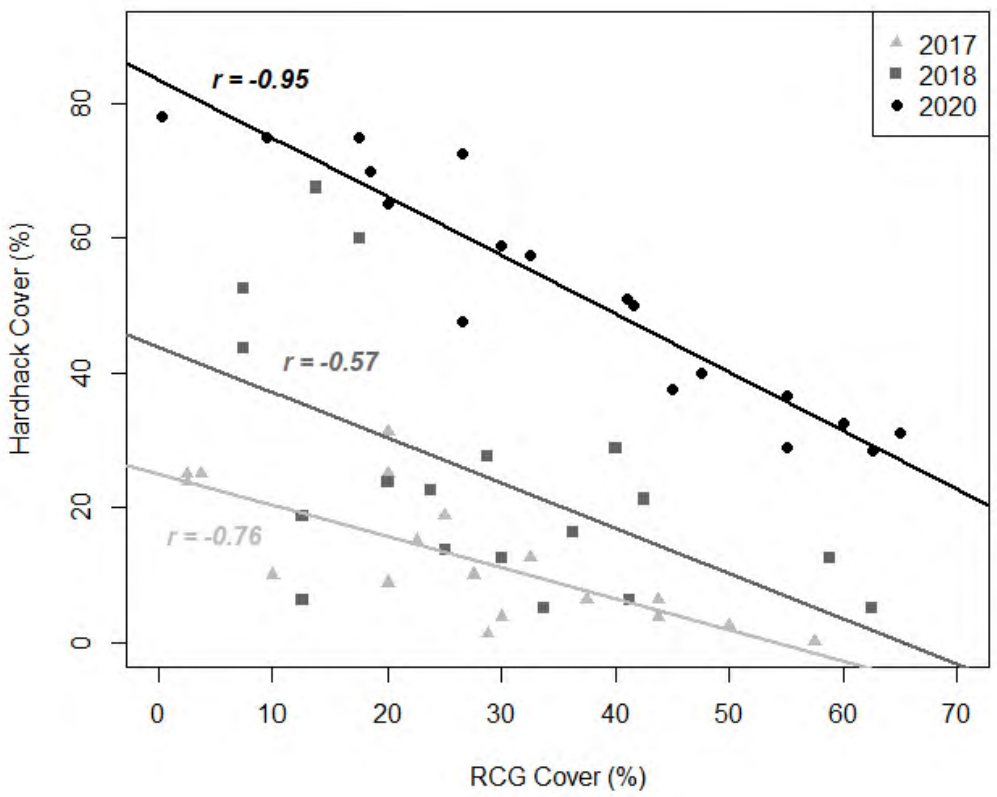

Figure 4.5: Relationship between hardhack percent cover and RCG percent cover in density trials (blocks A-F) for three sampling years. *r* values represent Pearson's correlation coefficient and best-fit lines are provided

4.2. Experiment 2: Native Shrub Trials

RCG percent cover gradually increased within most treatments from 1 to 3 GSPE. From 3 to 5 GSPE, both hardhack and dogwood treatments saw similar declines in RCG cover, though this was not significant; in the same period, RCG cover significantly increased in the thimbleberry treatment (Welch's two-sample t-test; $p < 0.0001$; Fig 4.1b), with mean cover increasing from 19% to 68%. Hardhack and dogwood treatments averaged 10% and 6% RCG cover, respectively, at 5 GSPE. Differences in RCG cover per treatment grouped by year were only significant at 5 GSPE (ANOVA; $p < 0.0001$; Fig 4.1b). The thimbleberry treatment had significantly higher RCG cover than hardhack and dogwood treatments (Tukey HSD; both $p < 0.0001$; Fig 4.1b).

Data for RCG aboveground biomass showed a similar result. Biomass collected at 2 GSPE showed low means (between 90 g/m^2 for hardhack and 157 g/m^2 for thimbleberry) and no significant differences in treatment. From 2 to 5 GSPE, mean RCG biomass did not significantly change for hardhack and dogwood treatments, but increased to 720 g/m^2 for thimbleberry (Welch's two-sample t-test; $p = 0.028$; Fig. 4.2b). At 5 GSPE, the thimbleberry treatment had significantly more RCG biomass than both hardhack and dogwood treatments (Tukey HSD; $p = 0.023$ and 0.010, respectively; Fig 4.2b).

Mean shrub percent cover showed no difference among treatments at 2 GSPE. It increased between 2 and 5 GSPE for hardhack (22% to 73%; Welch's two-sample t-test; $p = 0.00046$; Fig. 4.3b) and dogwood (16% to 63%; Welch's two-sample t-test; $p = 0.00011$). Shrub cover did not significantly change between years for thimbleberry, averaging 14% at 5 GSPE. However, thimbleberry cover was significantly lower than both hardhack and dogwood (Tukey HSD; both $p < 0.001$; Fig 4.3b) at 5 GSPE. The relationship between shrub cover and RCG cover was inversely proportional, though this relationship was only significant at 3 and 5 GSPE (Pearson's correlation; $p = 0.029$, <0.0001, respectively; Fig. 4.6).

Mean live shrub density at 5 GSPE ranged from 5.5 plants/m^2 in the thimbleberry treatment to 15.3 plants/m^2 in the hardhack treatment. Both hardhack and dogwood density (Tukey HSD; $p = 0.00077$ and 0.029, respectively; Fig. 4.4b) were significantly higher than thimbleberry, though no different from each other.

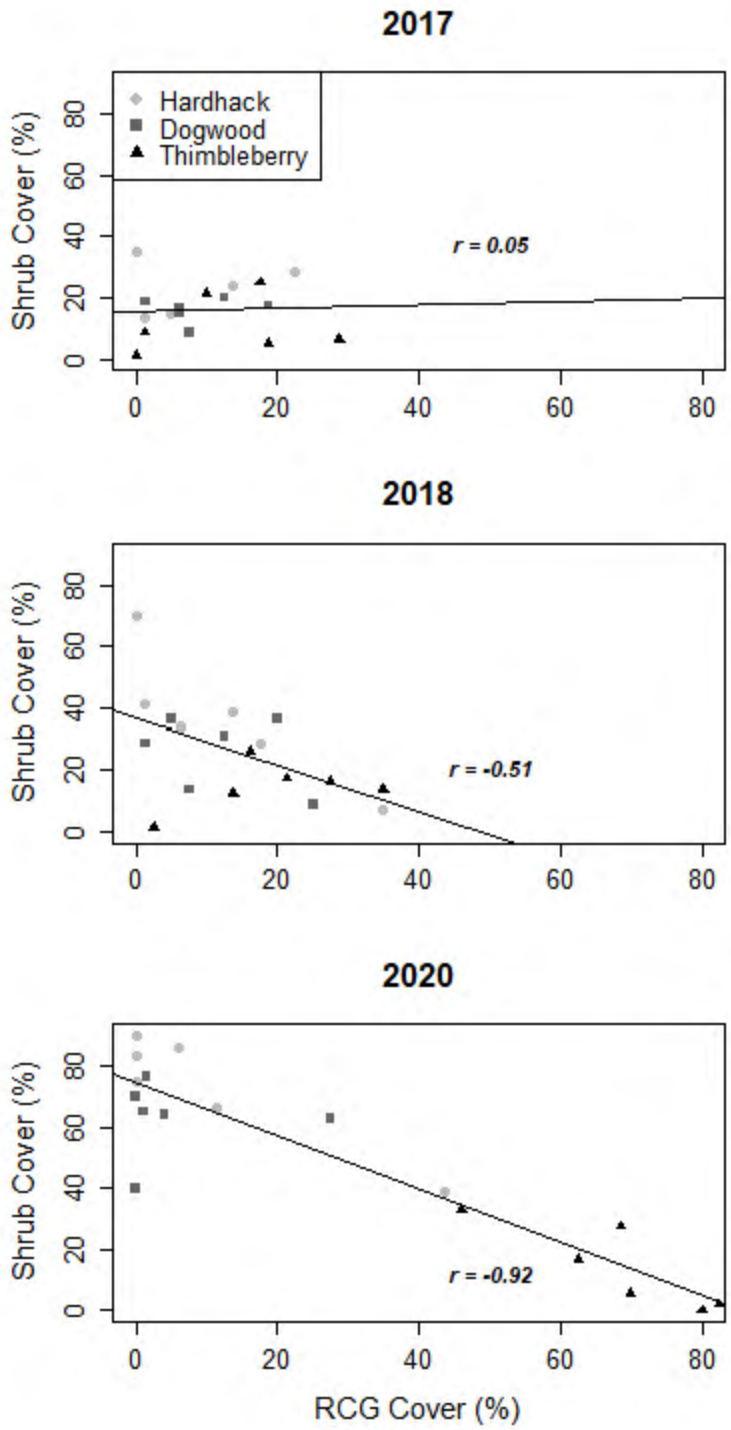

Figure 4.6: **Relationship between shrub percent cover and RCG percent cover in native shrub trials (blocks G-L) for across years since treatment (2016). *r* values represent Pearson's correlation coefficient and best-fit lines are provided.**

4.3. Experiment 3: Planting Density Replicate

Mean RCG percent cover increased in all treatments between 2 and 3 GSPE. This was only significant in the H50 treatment, which increased from 23.3 to 67.5% (Welch's two-sample t-test; $p = 0.021$; Fig. 4.1c). Mean RCG cover at 3 GSPE for H15 and H30 was 19.3 and 55.8%, respectively. Mean RCG cover was significantly higher in H50 than in H15 (Tukey HSD; $p = 0.039$; Fig. 4.1c), though neither were different from H30.

Mean RCG biomass was higher in lower-density treatments at 3 GSPE, though these differences were only marginally significant (ANOVA; $p = 0.079$; Fig. 4.2c). Mean biomass was 233.3, 832.0, and 964.9 g/m^2 in H15, H30, and H50 treatments, respectively. RCG biomass was strongly correlated with RCG cover in all experimental plots at Boundary Bay in 2020 (Pearson's correlation; $p < 0.0001$).

Mean shrub percent cover increased with higher density planting treatments. Shrub cover at 3 GSPE averaged 61.7, 21.2, and 12.7% in H15, H30, and H50 treatments, respectively. Cover in H15 was significantly higher than both H30 and H50 (Tukey HSD; $p = 0.00356$ and 0.00070, respectively; Fig. 4.3c). Shrub cover was strongly and inversely correlated with RCG cover in both experiments at Boundary Bay in 2020 (Pearson's correlation; $p < 0.0001$).

Live shrub density was consistently higher in higher-density planting treatments. Treatment means changed little over time. At 3 GSPE, shrub density was significantly higher in H15 than in H30 and H50 (Tukey HSD; $p = 0.00054$ and 0.00021, respectively; Fig. 4.4c). Shrub densities in H30 and H50 were not significantly different. This pattern held true across all sampling years. Shrub density at 3 GSPE averaged 18.1, 4.4, and 3.0 plants/m^2 in H15, H30, and H50 treatments, respectively.

4.4. Experiment 4: Site Prep Trials

Experiment 4 generally showed little or no difference between treatments. RCG cover showed no significant treatment effect at either 1 or 2 GSPE. RCG cover

increased in all treatments between 1 and 2 GSPE, though this was only significant for the CB+ROD30 treatment (Welch's two-sample t-test; $p = 0.0057$; Fig. 4.1d).

RCG biomass showed no significant differences between treatments (Fig. 4.2d). Biomass at 2 GSPE averaged 516.3, 484.9, and 353.1 g/m^2 in CB+HH30, CB+ROD30, and EX+HH30 treatments, respectively.

Mean shrub cover at 2 GSPE was 27.5, 3.0, and 16.5% in CB+HH30, CB+ROD30, and EX+HH30 treatments, respectively. CB+HH30 was significantly higher than CB+ROD30 (Tukey HSD; $p = 0.014$; Fig. 4.3d), but neither treatment was significantly different from EX+HH30.

Site preparation treatments showed only a marginally significant effect on the density of live shrubs at 2 GSPE (ANOVA; $p = 0.051$; Fig. 4.4d). Shrub density at 2 GSPE averaged 4.9, 1.6, and 6.7 plants/m^2 in CB+HH30, CB+ROD30, and EX+HH30 treatments, respectively. Treatments showed no significant difference at 1 GSPE, or between 1 and 2 GSPE.

Chapter 5. Discussion

5.1. Planting Density

Results demonstrate that shrub planting density had a significant treatment effect, with higher planting densities resulting in lower RCG cover and biomass, and higher shrub cover and live shrub density (Fig. 4.1, 4.2). This was true of density experiments at both sites, though the effect on RCG biomass was only marginally significant at Boundary Bay. For Experiment 1 at Bear Creek, H15 performed significantly better across all metrics compared to H50, with H30 in between and generally not significantly different from either. For data collected at 6 GSPE, the H30 treatment performed no differently than H50 across all metrics, while H15 was significantly different from both. The H15 treatment at Boundary Bay performed much better than H30 and H50 for shrub cover and density, though only slightly reduced RCG cover and biomass. Most of the plots at Boundary Bay in the H30 and H50 treatments appear to have failed to establish. Even at 6 GSPE, both RCG and shrub cover in Experiment 1 appeared to be increasing. However, because RCG biomass at Bear Creek did not change from 3 to 6 GSPE, it appears unlikely that RCG cover will continue to increase. In general, results indicate an inverse relationship between staking density and RCG presence.

The success of the H15 treatments, despite H30 performing only marginally better than H50 in both density experiments, suggests evidence of density-dependent competitive thresholds. Kim et al. (2006) also found that higher densities of staked willows reduced RCG cover, though this relationship appeared linear. However, the densest treatment in that study was 60 cm spacing between plants and plots were only monitored for 2 years. Thresholds may have appeared with denser treatments or a longer monitoring period. The importance of high-density planting has been argued by others. Molofsky et al. (1999) grew RCG in variable soil moisture, neighbour species, and neighbour density, and found that density was the greatest predictor of RCG growth. Competitors which form a dense canopy early in the season are especially advantageous (Maurer & Zedler 2002; Perry & Galatowitsch 2004; Hovick & Reinartz 2007). Adams and Galatowitsch (2008) found evidence of two-way density dependent effects when testing competition between RCG and native forb-graminoid mixes, both

seeded at variable rates, in wetland mesocosms. RCG in that study was capable of suppressing native plants at low seeding densities, while native plants required much higher seeding rates to significantly suppress RCG growth. These studies have all shown density-dependent effects, but no evidence of critical thresholds.

Although the H15 treatment was successful in both sites, it is unclear why there was a large difference between sites in the performance of hardhack in the H30 and H50 treatments. It is possible that this discrepancy is time-dependant, and plants at Boundary Bay may still establish. Indeed, the results for hardhack cover in these treatments are almost identical at both sites when compared at 3 GSPE. After 3 GSPE, the maximum hardhack cover in any plot at Bear Creek was only 31%, whereas at 6 GSPE, the minimum had increased to 29%. However, I suspect this will not be the case for Boundary Bay. Shrub mortality was high at the site and, excepting the H15 treatment, many shrubs appeared stressed.

Despite lower cover and densities, hardhack in the H50 treatment at Bear Creek may ultimately prove successful. Hardhack cover increased considerably (from 14 to 40%) between 4 and 6 GSPE, and may continue to increase as the plants develop more branches and new clones. In favourable conditions, hardhack achieves high densities (Darris & Gonzalves 2009). Eventually, hardhack growth in all density treatments can be expected to level off, though it is unknown when this will occur and if it will be at different levels. Still, even if this is not the case, 50 cm and possibly lower density hardhack staking can be useful depending on goals.

5.2. Shrub Species

In Experiment 2, both hardhack and red-osier dogwood established effectively and suppressed RCG, which showed little or no cover and biomass by 2020. The HH and ROD treatments showed no significant difference in these metrics or in shrub cover and density in any sampling year. Thimbleberry performed poorly and failed to establish in meaningful quantities. Many surviving thimbleberry plants appeared stunted and stressed. These plots were largely overgrown with RCG, which had achieved cover and biomass similar to that of control plots by 2020, or after 5 GSPE.

It is unclear why thimbleberry failed to establish in all of its plots. The finding is somewhat surprising, given that thimbleberry is known for rapidly establishing in and colonizing disturbed areas, spreading aggressively by both rhizome and seed (Maxwell 1990). It is possible some unknown microsite effect proved unfavourable, but thimbleberry is highly adaptable to variable site conditions (Oleskevich et al. 1996). The failure to establish may have been due to poor stock, unfavourable growing conditions for plugs, or possibly unsuitable genetics. However, given the remarkably high RCG cover and biomass in these plots, as well as high initial rates of thimbleberry establishment (92%), it seems likely that RCG simply proved a more effective competitor. Although thimbleberry can grow up to 2 m in height, stressed plants are generally much shorter (Oleskevich et al. 1996). All thimbleberry plants at Bear Creek were less than 1 m in height. Because most work on shading RCG has been done with either small forbs and graminoids or large shrubs and trees, it is unknown whether there is a critical height required for overtopping RCG at which competitive outcomes shift dramatically. This effect has been shown with other plants. Hill et al. (1995) surveyed tree seedlings in a New York right-of-way and found that their growth was initially slow, but increased dramatically when they grew tall enough to overtop the herbaceous layer. If thimbleberry had established and reached heights taller than most grasses but shorter than hardhack, that may have helped answer the question of critical height thresholds.

Microsite effects may have influenced results in the shrub species trials. Although the site at Bear Creek was a relatively homogenous RCG monoculture at the time of plot establishment, some smaller patches of rush, thistle, horsetail, or other grasses exist within it. In reviewing footage of the setup for plots G-L, it was evident that some patches within the treated area were not as heavily dominated by RCG as the area with plots A-F. This may explain the lower rates of RCG cover observed in 2020 for hardhack at 30 cm spacing in blocks G-L (10%) versus the same treatment in blocks A-F (43%). It may also explain the higher rates of all-shrub establishment (84-94% in plots G-L versus 38-48% in plots A-F). However, if this was a significant contributor to outcomes, it would also be expected to decrease RCG dominance in thimbleberry plots, which instead reached pre-treatment baseline levels. RCG surrounding the plots also appeared no less vigorous than normal in 2020.

5.3. Site Preparation

Site preparation treatments did not significantly affect any response variable except shrub cover. The CB+ROD30 had lower shrub cover than the CB+HH30 treatment, though this difference appears due to shrub species, as both treatments used cardboard. Few dogwood plants were found during sampling, and those present were visibly stressed and stunted. The failure of dogwood to establish suggests there is likely a difference in hardhack and dogwood's suitability for the site at Boundary Bay.

The treatments of Experiment 4 were originally meant to be compared with the standard 10 cm of woodchip mulch applied in other treatments (especially those also using 30 cm planting density) and were slated to be established in 2018, alongside Experiment 3. Constraints delayed the treatments by a year, which made statistical comparison using H30 as a control challenging. However, even after comparing standardized GSPE (rather than year), no significant differences between woodchip mulch, cardboard mulch, or excavation were detected.

The failure of the dogwood and cardboard treatment at Boundary Bay may be related to soils. The site is mapped as Seaview soils (Canadian Soil Information Service 2013), which are described as being increasingly saline at deeper layers, resulting in salinity being the greatest limiting factor for vegetation establishment in these soils (Luttmerding 1981). Although RCG is not known to establish in saline environments, it has demonstrated limited salinity tolerance (McWilliams et al. 2007). As most belowground RCG biomass is contained within the upper 20 cm of soil (Comes et al. 1981; Klimešová & Šrůtek 1995), it was likely not negatively impacted by the saline conditions in deeper layers of soil typical of Seaview soils (Luttmerding 1981). The opposite may be true for hardhack, the roots of which can grow up to 80 cm deep (Collison et al. 2005). The rooting depth of red-osier dogwood is unknown, though root structure is described as high-biomass with no taproot (Holle & Simberloff 2004). Despite no taproot, its large size suggests rooting below 20 cm, and it may have also been impacted by saline soil layers. Differences in salt tolerance may also explain why hardhack and dogwood showed the same performance at Bear Creek but different performance at Boundary Bay, with most dogwood failing to establish in the latter. This outcome may have also been due to differences in the water table, which was higher in Boundary Bay. However, this seems unlikely, given that dogwood has been described

as highly resistant to flooding and high water tables (Brink 1954). More detailed soil analysis, particularly at Boundary Bay, may yield explanations.

5.4. Other Findings

Despite high aboveground RCG biomass in some treatments, all-treatment biomass averaged only 375 g/m^2 at Bear Creek in 2020. This is low compared to the control plots, which averaged 1379.2 g/m^2, a value similar to controls in comparable studies. Averages of 598.4 g/m^2 (Howe 1995), 784 g/m^2 (Collins & Allinson 1995), 1352.7 g/m^2 (Klopatek & Stearns 1978), and 1520 g/m^2 (Kätterer et al. 1998) have been recorded in RCG monocultures growing under favourable conditions. Although control plots were not established in Boundary Bay, monoculture growth appeared more consistently vigorous across the site, and all-treatment biomass averaged 564 g/m^2, more than at Bear Creek. Both sites had at least one experimental plot which had higher RCG biomass than the maximum in control plots. Interestingly, while biomass production in excess of 500 g/m^2 within a single growing season in favourable monoculture conditions is common, some plots exhibited similar biomass production even in the presence of established shrubs. For instance, 3 of 10 plots in Bear Creek with biomass exceeding 500 g/m^2 in 2017, including the highest, occurred within shrub cover over 50%. This was similar in 2020. The opposite was true of plots at Boundary Bay. Less surprisingly, of the 18 plots there exceeding 500 g/m^2 in 2020, only 1 occurred amongst shrub cover over 30%. However, plots at Boundary Bay generally had much lower shrub cover, so the effect of shrub cover on RCG biomass between sites is not clear.

Initial establishment appears to have played some role in treatment success, with some exceptions. Although it was statistically unfeasible to compare mean RCG and shrub cover per treatment group across all treatments relative to percent establishment (which included different densities, species, and GSPE), there did appear to be an inverse correlation between percent establishment and future RCG cover. Interestingly, however, there was no correlation between establishment and future shrub cover. The most surprising exceptions were the TH treatment, which eventually failed despite 92% establishment, and the H15 treatment at Boundary Bay, which attained high shrub cover despite only 16% establishment (though Experiment 3 was restocked after the first year).

Although the stark difference in establishment between sites may have been due to non-site-specific constraints (such as unfavourable weather or poor planting techniques), this finding suggests that effective planting densities may require adjustment depending on anticipated site suitability, and lower densities may be sufficient in some areas.

Another noteworthy finding is that total cover of both RCG and planted shrubs closely followed the -3/2 power law of self-thinning, which models the rate at which plants in even-aged stands die as they densify and biomass limits are reached (Lonsdale & Watkinson 1983). Although this law has been contested (Lonsdale 1990), it has been observed in wetland willows (Kim et al. 2007) and in mixed-species grasslands (Nie et al. 1997). The law is generally used to study monotypic rather than interspecific competition. It has never been studied in RCG specifically, but further study of similar trials may be used to predict growth limits following successful shrub establishment.

5.5. Uncertainties & Limitations

Results showed considerable variation. Outliers were common in most response variables, especially pertaining to RCG, whereas shrub cover tended to be more consistent. Though treatments had a significant effect on response variables in most experiments, there was still considerable variation. This is to be expected, to some extent, with in situ studies on vegetation management (Howe 1995). The growth and competitiveness of RCG is known to be influenced by even small differences in numerous factors, such as climatic variables, soil conditions, and moisture (e.g., Gomm 1978, Lindig-Cisneros & Zedler 2001, Kercher & Zedler 2004). The same can also be said of the planted shrubs. Hovick and Reinartz (2007), for instance, observed high variability in shrub establishment amongst RCG not just between different species, but within the same species in different plots. A number of uncertainties and variations in microsite conditions likely contributed to observed variation for both RCG and planted shrubs. Though the extent to which they influenced outcomes is uncertain, a few uncertainties in particular merit discussion.

Differences in soil conditions between sites and even between plots likely influenced results and should be investigated further. A basic soil profile was completed

for Bear Creek. I also conducted preliminary tests for soil moisture, temperature, and electrical conductivity, which are unreported as no significant differences were detected. No soil description or analysis was conducted for Boundary Bay, though this may have yielded insights into the failure of dogwood to establish there, or other site differences. Similar studies have found clues in soil conditions. Kim et al. (2006) found soil moisture did not impact RCG growth, but affected the growth of planted willows. Mueller (1941) found that RCG was more likely to root from nodes when topsoil was wet, and others have found a general preference for wet soils (Gomm 1978; Kercher & Zedler 2004). This may contribute to the dominance of RCG at Boundary Bay, which has a high water table. The finding that both hardhack and RCG grew more rapidly in Boundary Bay (Fig. 4.1, 4.3) may suggest that the site is higher in soil nutrients. Because RCG is a better competitor in high-nutrient conditions (Kercher & Zedler 2004), this may explain why the H15 treatment was the only one to establish in significant quantities, despite lower density plantings establishing successfully at Bear Creek.

Some of the uncertainty in observed results may be explained by limitations in methods. For instance, visual cover estimates are a quick and simple way to estimate vegetation density, but lack precision and are prone to subjective interpretations. Live shrub counts were also variable. Because numbers per plot were generally low, increasing the sampling area would have likely resulted in more precise estimates. As well, hardhack often had multiple stems, and differentiating between different plants was not always obvious. The imprecise timing of data collection, which occurred in either August or September, may have skewed results. Plot setup may have also caused microsite variations. For instance, it's possible that woodchip mulch was not applied evenly, herbicide at Boundary Bay was not sprayed uniformly, or the EX+HH30 treatment was not excavated to a consistent depth.

Inter-plot shading may have influenced findings. Because plots within each block at Bear Creek were oriented north to south and the sun remains low on the horizon for part of the growing season, this may have influenced the growth of shrubs depending on their plot position. I found no statistical significance in response variables depending on plot position, though there may have been a small effect. This would have been especially important in Experiment 2, which used species of different sizes. For instance, dogwoods growing in the southern plot would have potentially shaded smaller hardhack

growing in the center plot, which may have shaded smaller thimbleberry growing in the northern plot.

Sampling design may also explain the finding that although RCG biomass was strongly correlated with cover in both sites and sampling years, this correlation proved far stronger in 2017 than in 2020. During sampling in 2020, I noticed a number of RCG shoots, particularly those growing under the shade of shrubs, were thinner and felt more flimsy compared to the more vigorous shoots in control plots. These plants appeared stressed, most likely from competing with shrubs. The thinness of culms in these stressed shoots may account for this weakening of the cover-biomass correlation. This could explain why Experiment 1 saw a slight increase in RCG cover, despite no change in biomass, from 2017 to 2020 (i.e., more, albeit thinner shoots). Another possibility is that the accumulation of thatch may have contributed to overestimation of cover in 2020 relative to 2017. Sampling in September instead of August was likely a factor as well. Although thatch was excluded from both cover and biomass measurements, discriminating between living and dead plants was easier when collecting biomass samples than when taking visual cover estimates. During the 2017 sampling (2-3 GSPE), plots at Bear Creek would have had little or no thatch, whereas many plots had accumulated a thick layer at the time of sampling in 2020 (5-6 GSPE). This may have influenced cover estimates, though thatch was largely obscured by the canopy.

Another limitation was that biomass sampling only occurred every few years and only accounted for aboveground biomass. Sampling for belowground biomass is far more labour intensive, as roots and rhizomes require meticulous processing to separate from soil. However, such sampling would allow for a more complete assessment of RCG's condition, and the ratio of aboveground and belowground biomass in different plots may prove useful for study. This may provide clues to stress or vigour of RCG in a plot. Recording both metrics yearly, though time consuming, would provide a more detailed look at how RCG responds to treatments over time. One challenge to this approach is the destructive nature of belowground biomass sampling, which would also destroy any shrubs in the sampled area. A study employing this approach would require plots large enough that biomass samples could be taken from different locations each year without being large enough to impact whole-plot dynamics.

A better understanding of belowground growth and competition is important for shrubs. Both hardhack (Darris & Gonzalves 2009) and thimbleberry (Maxwell 1990) are known to spread primarily by rhizome. During data collection at Bear Creek in 2020, a number of hardhack stems were observed growing just outside of their plots, likely from originally-staked parent plants. Despite having unsampled buffers within each plot, it is possible that roots and rhizomes of different species would have competed or otherwise interacted in a way that influenced the results of Experiment 2. The same may also be true of different hardhack densities in Experiment 1.

Studying belowground interactions is even more important for RCG. Although there is some evidence of success controlling RCG using methods which don't affect belowground parts of the plant, such as seedhead trimming (Hovin et al. 1973), neglecting to also target RCG's belowground vigour is ill-advised. Although high seed production (Tu 2004), high germination rates (Leck 1996), and long seed viability (Toole & Brown 1946) contribute to RCG's success, rhizomes are arguably more important for spread. Apfelbaum and Sams (1987) observed 88% of new shoots emerging from rhizomes or tiller buds, and another study found up to 74% of shoots originated from rhizomes, specifically (Evans & Ely 1941). Although shading has been shown to reduce both aboveground and belowground RCG biomass (Maurer & Zedler 2002), the physiological mechanisms behind this may be subject to edge effects due to the extensive root and rhizome networks which connect plants (Klimešová & Šrůtek 1995; Bonilla-Warford & Zedler 2002). This was addressed, to some extent, with the unsampled buffers in our experimental design, but because the center of any experimental plot in our design was never more than 1.25 m from a plot edge, it is possible (if not likely) that RCG in the center of the plots were still connected to those outside the plot by belowground networks. These belowground networks are capable of distributing nutrients and carbon allocation (Kinmonth-Schultz et al. 2011), so it is possible shade-stressed shoots in experimental plots were aided by plants growing in favourable conditions outside the plots. If this is indeed the case, treatments would thus be more effective when applied at a larger scale which maximizes area:perimeter ratios.

5.6. Future Work & Recommendations

Results underscore the importance of longer-term monitoring in RCG studies, which other authors have called for (Adams & Galatowitsch 2005). Similar studies in establishing native woody plants amongst treated RCG were monitored for only 2 years (Kim et al. 2006; Hovick & Reinartz 2007) or less (Miller et al. 2008). Response variables for RCG and shrubs may show statistical significance soon after treatment, long-term outcomes require lengthier monitoring periods. Our results showed that some variables which were not significant at 2 years post-treatment became significant later on. Even after 6 GSPE in Experiment 1, response variables were dynamic and changed between years. Still, temporal changes appeared less variable than in the first 2 years. Major changes in long-term outcomes for the plots at Bear Creek thus seem unlikely, though this should not be discounted as a possibility. The most notable change occurring after 2 years was the failure of thimbleberry to establish in Experiment 2, and the subsequent prolific increase in RCG cover and biomass between 3 and 5 GSPE. Although experiments at Bear Creek have concluded, ongoing monitoring at Boundary Bay may yield significant differences. This difference in monitoring period, which was 5-6 growing seasons at Bear Creek and only 2-3 at Boundary Bay, is likely a large reason why differences in Experiment 1 were much stronger than in its replicate. Shrubs in most treatments at Boundary Bay appear to have failed to establish. However, hardhack in Experiment 1 also showed high mortality in early years, yet ultimately established successfully. This is also a possibility for Boundary Bay.

Little remains known about belowground interactions in RCG growth and competition, and this should be a priority in future studies. A similar study design which increases plot sizes substantially would be useful in investigating whether RCG plants growing near plot boundaries are connected to and influenced by healthy plants outside the plot, such as by studying the average root network length. A student group in BCIT's FNAM program is currently studying RCG belowground networks, though they are focusing on density rather than length/size. Larger plots would also be useful in studying changes in both aboveground and belowground biomass each year, because they would allow more space for destructive sampling without influencing plot-wide results. This would allow for detailed study of changes in both aboveground and belowground competition, and how this changes temporally and with proximity to unshaded RCG.

Another recommended area for future work is in testing different shrub species using similar methods and a standardized planting density. Hardhack and red-osier dogwood may not be suitable or desirable for a given site. Identifying other effective species would enable the use of this method in a range of different sites and allow for greater flexibility in post-restoration planting designs. Darris (2002) provides a list of shrubs native to the Pacific Northwest which readily root from cuttings. Well-performing species that we did not test include Pacific ninebark (*Physocarpus capitatus*), black twinberry, common snowberry, and salmonberry (Darris 2002). Selecting species which are present in or near a given site is advantageous for reducing costs and increasing the likelihood of establishment by using locally-adapted genetic stock (Tu 2004). For deciduous species, selecting those with early bud break which attain high leaf area indexes earlier in the season is recommended to minimize RCG's sun exposure in a growing season. Selecting species which grow shorter than hardhack would be useful for testing whether there is a critical height threshold required for successfully overtopping RCG. In such a study, closer monitoring of stem etiolation, which occurs when RCG is shade-stressed (Smith et al. 1990; Kim et al. 2006), would be recommended for understanding the predominant mechanism of competition. Shade tolerance should also be considered. Shade tolerant species may be more versatile when grown with other functional groups, but shade intolerant species generally have faster growth, shorter critical periods, and lower declines in growth under competition (Wagner et al. 1999). RCG does not exhibit allelopathy (Chung & Miller 1995), and testing the competitive ability of allelopathic shrub species may also prove useful. Some native species with limited evidence of allelopathy include red elderberry (*Sambucus racemosa*) and common snowberry (Bell et al. 2011).

Beyond testing individual species, future studies should attempt staking groups of shrub species within the same plot to investigate whether creating diverse vegetation communities with multiple species or functional groups are more effective in suppressing RCG than monocultures at the same planting density. Studies have demonstrated the importance of diversity in reducing invasibility. For instance, a 4-year grassland study tested mixes ranging from 3 to 21 native grass and forb species and found that, compared to the 3-species mix, mean invader biomass declined by over 50% with 6 species, and over 90% with 12 species (Zavaleta & Hulvey 2007). This approach should

prioritize functional groups over species richness, as the former increases resilience by re-establishing ecosystem processes (Masters & Sheley 2001).

Controlling RCG by live-staking native shrubs with woodchip mulch is recommended for site managers at Bear Creek Park and BBRP. Staking density should depend on goals and budgets. While the 15 cm spacing was most successful in reducing RCG cover and biomass, the 50 cm spacing still reduced mean RCG biomass compared to the control by more than half at Bear Creek. Even with high-density staking, full eradication of RCG appears unlikely. If goals are not to eradicate RCG completely, but rather to improve habitat structure, diversity, or overall ecological value, then a 50 cm staking density may be best. The obvious advantage of this method is that much larger areas can be treated with the same amount of labour and number of plants. Browne et al. (2015) calculated the person-hours required to plant 1 ha as 2133, 800, and 533 for 15, 30, and 50 cm spacing, respectively. For Bear Creek Park, I recommend staking both hardhack and red-osier dogwood at 50 cm spacing. Hardhack established well, even at 50 cm spacing, and both hardhack and dogwood were effective at reducing RCG. Reduction is a more feasible approach as the site is in a heavily urbanized area, prone to a number of stressors, and abundant in several other invasive species. For BBRP, I recommend a 15 cm staking density if possible, as lower-density treatments failed to establish and RCG at the site appears particularly vigorous. Hardhack should be prioritized over dogwood, though some dogwood should be included as it was never tested at 15 cm spacing and may perform better in higher densities. Because the infested area at BBRP is much larger and high-density staking will take longer, treatment should be done in stages. A long-term approach can better utilize volunteer labour and will have the advantage of adaptive management and allow for testing of different treatments and variations as needed. A 10 cm layer of woodchip mulch is recommended for treatments at both sites; however, applying woodchip mulch accounts for approximately half the person-hours of this method (Browne et al. 2015), so further experimentation with cardboard or other readily-applied mulches is encouraged. If feasible, testing other shrub species, namely black twinberry at Bear Creek Park, Pacific ninebark at BBRP, and common snowberry and salmonberry at both sites, is also recommended.

Chapter 6. Conclusion

RCG's phenotypic plasticity and resistance to conventional control methods make it remarkably challenging to suppress, let alone eradicate. Nonetheless, its pervasive and severe impacts to wetlands and riparian areas throughout the Pacific Northwest require management. This project sought to better understand whether RCG can be outcompeted through overtopping and shading by live-staked native shrubs, and to monitor competitive outcomes over time.

Most treatments were successful in reducing RCG. Higher planting densities of hardhack were more effective in suppressing RCG cover and biomass, and this effect was more pronounced in the 15 cm spacing treatment. RCG cover gradually increased in all planting densities over time, even after 6 growing seasons, but remained low relative to control plots. Red-osier dogwood performed as well as hardhack across all metrics, though thimbleberry plots largely failed to establish and saw RCG return to baseline levels. Site preparation treatments did not yield significant results, though this may have been due to site effects and poor shrub establishment.

The findings are important for site managers facing RCG infestations and will allow them to better evaluate their management options in light of short- and long-term goals and limited budgets. Live-staking with hardhack should be conducted at 15 cm spacing where possible; however, if goals prioritize establishing shrubs across larger areas over RCG eradication, lower density plantings may be sufficient. Both hardhack and dogwood are recommended for use, and experimenting with other native shrubs is encouraged. Although this method is unlikely to eradicate RCG completely in treated areas, it is highly effective in reducing RCG and managing infestations long-term.

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Appendix A.
Images

Figure A.1: Diagram of block layout for blocks A-F (left, density trials) and G-L (right, native shrub trials) at Bear Creek (Hennigar et al. 2019).

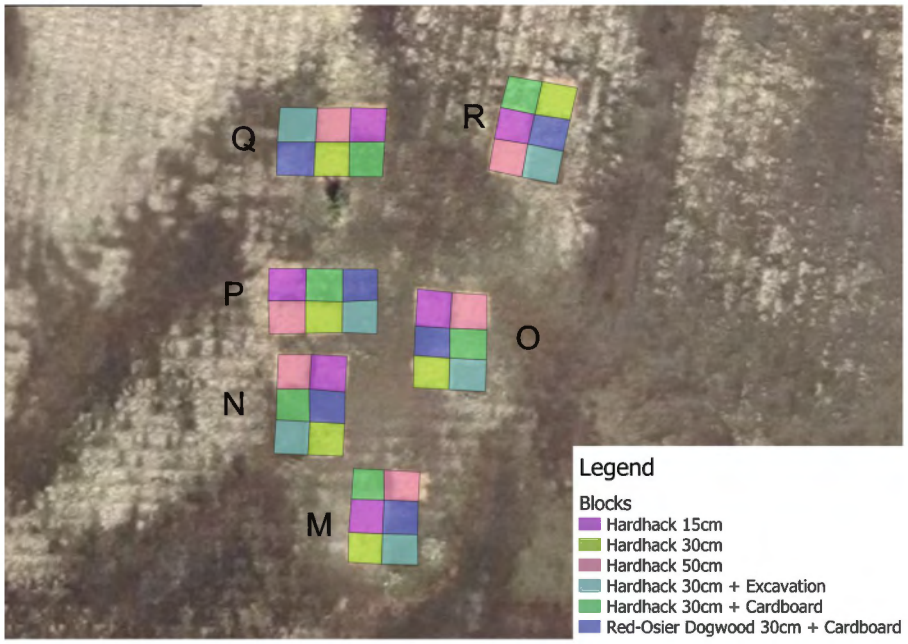

Figure A.2: Diagram of block layout for blocks M-R (density replicate and site preparation trials) at Bear Creek (Hennigar et al. 2019).

Figure A.3: Blocks G-L at Bear Creek after one growing season (April 2017; H. Marcoux).

Figure A.4: Block J at Bear Creek after five growing seasons (September 2020).

Figure A.5: View of the site at Bear Creek. Plots G-L are visible in foreground. Plots A-F are visible in background, center.

Figure A.6: Plots being set up at Boundary Bay (February 2018; H. Marcoux).

Figure A.7: Close up of block M during plot setup at Boundary Bay (February 2018; H. Marcoux).

Figure A.8: Significant RCG regrowth at Boundary Bay since treatment in 2018 (September 2020).

Figure A.9: Biomass samples harvested from plots at Boundary Bay (September 2020).