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Invasive Woody Plant Research in St. Edward State Park: Control of Invasive Species for Science and Native Biodiversity

CHERRY LAUREL (*Prunus laurocerasus*) and PORTUGUESE LAUREL (*Prunus lusitanica*) INVASION IN SAINT EDWARD STATE PARK

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Note: *This document reports the results of our 2015 & 2016 research seasons investigating non-native cherry laurel (*Prunus laurocerasus*) and Portuguese laurel (*P. lusitanica*) in St. Edward State Park. It also references the combined* results *from three previous seasons (2011, 2012, and 2013) of research on English holly (*Ilex aquifolium*) at the same site, reported in Stokes (2014), Stokes et al. (2014a & b), and Lopez and Stokes (2016).*

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SUMMARY

We located and removed all cherry laurel *Prunus laurocerasus* and Portuguese laurel *P. lusitanica* from a 9.2 hectare (22.8 acre) area of St. Edward State Park in January – March, 2015. This was the same study area where English holly *Ilex aquifolium* was located and removed in 2011 - 2013 (Stokes et al. 2014b). A total of 231 *P. laurocerasus* (25.1/ha) and 22 *P. lusitanica* (2.4/ha) were located and removed.

Age of trees in our sample, determined by annual ring counts of basal cross-sections, ranged from 1 to 35 years for *P. laurocerasus*, and 1 to 21 years for *P. lusitanica*. Age-abundance patterns suggest rapid population growth for both species, with a doubling time of roughly 3 years for *P. laurocerasus*. *P. laurocerasus* trees were in a rapid growth and biomass accumulation stage of their life history, with steepening size (height, stem diameter, canopy area)-age curves. All measures of size also increased with age in *P. lusitanica*, but a larger sample is needed to more precisely characterize its growth pattern. Native shrub and ground cover vegetation was dramatically reduced under *Prunus* canopy relative to adjacent areas.

Mapping of the known-age *Prunus* sample indicates that both species are spreading rapidly at two spatial scales: contiguous outward expansion of tree clumps, primarily vegetatively, and long distance dispersal, probably via bird-dispersed seeds. *Prunus* was concentrated along forest edges of our study area, likely because edge conditions are more favorable for *Prunus*, and/or because propagule pressure from horticultural plantings in St. Edward Park and in the neighborhood surrounding the park is higher in these locations.

Our results suggest that *Prunus* is a substantial component of a rapidly accelerating invasion of St. Edward Park by non-native, shade tolerant trees. The temporal and spatial patterns revealed in this study suggest that *P. laurocerasus* has the potential to become, in less than two decades, a major component of the forest flora, both in number of individuals and area occupied. *P. lusitanica* may be on a similar trajectory, although it appears to be a more recent and less numerous invader at present. Though less abundant than holly, *Prunus* is increasing at a more rapid rate, and already makes up a substantial proportion of the ongoing invasion of St. Edward Park by non-native trees. Like holly, both *Prunus* species appear to establish well under forest canopy, and thus are potential invaders of natural areas in most of western Washington, where forest is the dominant ecological community type.

The negative effects of *Prunus* and holly on native vegetation that we observed suggests that the proliferation of these non-native trees would likely come at the expense of native plant diversity, and could have large effects on the native forest ecosystem. This is of particular concern for St. Edward Park, which is one of the few exemplars of relatively intact native forest in the Seattle metropolitan region. We present management recommendations and a research agenda that could produce information necessary to inform the management of these species. Maps and figures of the course of *Prunus* and holly spread included in this report and elsewhere (Stokes et al. 2014b) may be useful in conveying to the public the seriousness of the threat posed by these non-native trees and by invasive species generally.

Figure 1. Non-native *Prunus*-dominated understory in mixed mid-successional forest fragment < 100 m N of St. Edward State Park and < 250 m NE of our study area. Cherry laurel (*P. laurocerasus),* along with lesser amounts of Portuguese laurel (*P. lusitanica*) and English holly *(Ilex aquifolium),* forms a dense, nearly continuous, non-native vegetation layer from ground level to a height of 4 – 8 m under a canopy of red alder, bigleaf maple, Douglas-fir, western hemlock, and western redcedar.

INTRODUCTION

The spread of invasive, non-native species is one of the greatest threats to biodiversity and native ecosystems (Richardson et al. 2000, MEA 2005, Primack 2010, Lockwood et al. 2013). Invasive plants pose a particularly serious threat to natural areas (Cronk and Fuller 1995, Reichard and White 2001, Zhao et al. 2013). In the Pacific Northwest, as elsewhere, non-native plants are increasingly invading natural areas, displacing native plant species and reducing habitat quality for native animal species (examples in Boersma et al. 2006, Reichard 2007). A critical need in developing effective management of invasives is a better understanding of the pattern and process of their spread (Lockwood et al. 2013, Zhao et al. 2013).

An especially impactful class of invasive plants is shade tolerant trees (Walther 1999, Walther and Grundman 2001, Webster and Wangen 2009). Unlike many invasive plants that require disturbed or open environments to spread, shade tolerant trees can potentially invade closed-canopy forests, and thus may pose a threat to mid- and late-successional forest ecosystems. Relative to other plant types, invasive trees have slower rates of growth and reproduction, which, at least in forested environments, are difficult for humans to perceive, often obscuring their invasive nature until late in the invasion process. This is problematic, because once established, they tend to be long-lived, system-transforming (Richardson et al. 2000, Richardson and Rejmanek 2011), and difficult to eradicate (Mack and Foster 2009), giving them the potential to radically alter the invaded system more or less permanently.

Three non-native tree species that have recently become naturalized in Pacific Northwest forests are English holly (*Ilex aquifolium*), cherry laurel (*Prunus laurocerasus*), and Portuguese laurel (*P. lusitanica*). All are relatively small, shade tolerant, broadleaf evergreen species that are native to Eurasia and are commonly used in landscaping and horticultural plantings in western Washington. All three species reproduce both vegetatively and by seeds contained in berries, presumably dispersed by birds. The best studied of the three, *Ilex*, has been increasingly identified as invasive in the Northwest in recent decades (Olmstead 2006, Jones and Reichard 2009, Zika 2010), and its invasive character in western Washington forests has recently been demonstrated (Stokes et al. 2014a).

The two *Prunus* species are less widely recognized as invasive in the Northwest, and their occurrence in natural Northwest ecoystems has not been systematically studied. *P. laurocerasus* is a native of the Black Sea region of southwestern Asia (Walther 1999, Hättenschwiler and Körner 2003), and has been widely reported to be invasive in western Europe (Walther 1999, Keil and Loos 2005, Hackney 2006-8, Booy et al. 2015). Its earliest known record of naturalized occurrence in Washington is 1952 (Bennett et al. 2011). *P. lusitanica* is a native of the Iberian Peninsula. It is classified by the IUCN as a “vulnerable” species in its native range (Duarte et al. 2011), but is reported to be invasive but less aggressive than *P. laurocerasus* in the United Kingdom (Booy et al. 2015).

Anecdotal evidence is accumulating that these two *Prunus* species are invasive in Northwest plant communities. Both species have been reported to be invasive in western

Washington (Swearingen and Bargeron 2016), and both are designated as "Emergent Species" in Stanley Park (British Columbia) Natural and Sensitive Areas (SPES nd). *P. laurocerasus,* is classified as a "Weed of Concern" in King County, Washington, and has been identified as the second most abundant non-native tree species in Seattle's city parks (*Ilex* is the first; King County 2016). Like *Ilex*, *P. laurocerasus* is on the Washington State Noxious Weed Control Board's list of plants to be monitored for possible inclusion on the State Noxious Weed list (NWCB 2010).

Little information exists about the effects of *Prunus* on native Pacific Northwest species and ecosystems, however some inferences may be drawn from preliminary findings regarding effects of the ecologically similar *Ilex* (Stokes 2014a, Church and Stokes in prep.). With its dense, fast-growing, evergreen foliage, *Prunus* may suppress or displace native species through shading or other mechansisms. Like *Ilex* (Nickelson 2014), *Prunus* appears to have the capacity to become a dominant forest plant that could profoundly alter native forest structure in novel ways by forming a persistent thicket-like evergreen sub-canopy tree layer, a structural element with no analogue in the region's native forest (Fig. 1, Stokes, pers. obs.). As some of the few invasive plants apparently able to colonize closed-canopy Northwest forest, these shade tolerant trees may have the potential to transform the region's native forests on a large scale.

Despite the seriousness of possible effects of invasive *Prunus* on native ecosystems, little is known about the status, pattern, or process of *Prunus* invasion in the Pacific Northwest. This study addresses this information gap by enumerating and aging all *Prunus* occurring in a large (> 20 acre) area of mainly native, maturing, low elevation western Washington forest, and using this known-age *Prunus* sample to quantify parameters of population growth and spread in this forest system. Our research questions include: What is the current state of *Prunus* in the forest—its population density, age structure, and morphological characteristics? What effect does *Prunus* have on native understory vegetation? What have been the past patterns and rates of numeric increase and spatial spread since the beginning of the invasion? And finally, what are the implications of past spread for the future of *Prunus* in the forest? As our research was conducted in a widespread Northwest forest type (mid successional, western hemlock zone *sensu* Franklin and Dyrness 1988), results of this study may help inform management of *Prunus* in wildland forests throughout the region.

Study Area: Saint Edward State Park

See also study area description in Stokes et al. (2014a).

We conducted this study in St. Edward Park, site of a substantial area (~120 ha) of largely native forest situated in a suburban matrix near Seattle WA (Fig. 2). The study area consisted of a 9.2 ha (22.8 acres) area of forest in the northern section of the park (Fig. 3, Stokes 2014b), the same study area that was surveyed for English holly (*Ilex aquifolium)* in 2011-2013 (Stokes et al. 2014a&b). Vegetation in the study area is characterized as an *Alnus rubra/Polystichum munitum* community (Chappell 2004, Smith 2006), and is mostly dominated by large red alder (*A. rubra*) and bigleaf maple (*Acer macrophyllum*), with substantial but variable amounts of small-to-large Douglas-fir (*Pseudotsuga menziesii*), western red cedar (*Thuja plicata*), and western hemlock (*Tsuga heterophylla*). The shrub layer is substantial, primarily consisting of deciduous species such as salmonberry (*Rubus spectabilus*) and Indian plum (*Oemleria cerasiformis*), as well as lower growing evergreen species such as salal (*Gaultheria shallon*), Oregon grape (*Mahonia nervosa*), sword fern (*P. munitum*), and native blackberry (*R. ursinus*).

Near the center of the park is a seminary building around which several large *P. laurocerasus* hedges were installed at the time of construction in the 1930s (NPS 2006). The hedges, located approximately 200 m from the nearest point in our study area, still exist at the site. Available evidence suggests that the hedges were historically maintained at small size (3 - 5 ft height; NPS 2006). Since 2006 and perhaps earlier, the hedges have been taller (~ 8 ft) but are closely trimmed and do not generally produce flowers or berries (Stokes, pers. obs.).

Figure 2. St. Edward State Park and environs. Dominated by primarily native semi-mature forest, the park (border indicated by yellow line) is largely surrounded by residential development. Aerial photo from Smith (2006).

METHODS

We surveyed the study area for all *Prunus laurocerasus* and *P. lusitanica* from January – March 2015 (Fig. 3).To locate all *Prunus* plants, we systematically traversed the entire study area, walking in lines 3 – 10 meters apart, depending on vegetation conditions and visibility. We are confident that we located all *Prunus* plants > 50 cm tall—and nearly all *Prunus* of any size—because most deciduous vegetation had not leafed out at the time of our searches, and therefore the evergreen *Prunus* was highly conspicuous in the understory. Furthermore, on March 10, 2015, following the survey, Stokes and an assistant systematically searched 40% of the study area and found only one small *P. lauroceras*, a 1-year old plant < 0.3 m in height, that had been missed in the survey.

We determined the location of each *Prunus* using a handheld GPS unit (Trimble Juno SB; estimated error after differential correction < 3 m) or, if the plant was within 25 m of a previously located plant, either *Prunus* or an *Ilex* location that still could be identified (label from 2011-2013 still readable), we recorded distance and bearing from the already located plant using a meter tape and hand-held compass (est. error < 1 m).

Once we located a *Prunus* plant, we recorded the following characteristics of the plant: height (or linear extent of central leader if the tree was bent over), canopy diameter, foliage density (visually assessed as light, medium or dense), trunk diameter (at ground level, 20 cm above ground, and at breast height), and presence/absence of berries. Trunk diameter was measured with dial calipers to the nearest mm. Height and canopy diameter were measured with a meter stick or meter tape to the nearest 0.1 m. We also determined whether the plant originated from seed or vegetative spread, as evidenced by the root structure we observed when we uprooted it (see below), or by distance from nearest conspecific.

Small *Prunus* sprouts were present under or near some *Prunus* trees. We included in our sample all of these sprouts that had a basal stem diameter ≥ 1cm. Any remaining sprouts (< 1 cm basal diameter) farther than one meter from a sampled plant were also sampled, allowing us to map the extent of the sprout-covered area. Thus our sample population includes all *Prunus* plants in the study area that were ≥ 1 cm basal diameter and any smaller *Prunus* plants that were > 1 m from the nearest sampled conspecific. We recorded the numbers of unsampled sprouts at each site and removed them by uprooting.

At the site of each large *Prunus* or group of *Prunus* with continuous canopy, we visually estimated % cover (to nearest 5%) of native evergreen and woody plant species and % bare ground (i.e., % not covered with native vegetation), both under the *Prunus* canopy and in the adjacent area within 5 m of *Prunus* cover.

We removed all *Prunus* we encountered by uprooting when possible, pulling by hand or using a weed wrench. The ground was moist during the field season, allowing us to successfully extract all large roots from the ground in most cases. We inspected the uprooted trees for evidence of root connections with other trees. For trees too large to uproot (n = 29), we cut the trunk at ground level using a bow saw. In 28 of these cases we immediately treated the cut surface with an over-the-counter herbicide (Roundup Concentrate Plus; 18% glyphosate; Monsanto) to kill the plant and suppress sprouting. Glyphosate is a widely used herbicide that binds tightly to soil and is often used for wildland invasive control (Tu *et al*. 2001). Direct application of the herbicide to only the cut surface of the stump limited the quantity of herbicide we used to less than 8 ounces over the entire study area.

We collected a ground-level cross section of the stem for each *Prunus* in the sample. After the cross sections had dried for at least 14 months, we sanded them with 150 grit sandpaper followed by 220 grit. We then moistened the cross sections with water and examined them under a dissecting scope, counting annual growth rings to determine tree age for all individuals. Two of us (Stokes and Thiel) independently counted rings for each cross section. For samples in which our counts differed by more than 1, we re-sanded and re-counted. The final ring counts for all samples were all within 1, and we estimate the accuracy of the sample ages to be ± 1 yr.

The locations of *Prunus* plants, their ages, canopy size, and other variables were entered into an Excel table and then mapped using ArcMap 10.5 (ESRI 2016). The spatial data were projected into State Plane coordinate system (Washington North FIPS 4601 in meters), using a Lambert Conformal Conic Projection, based on the North American Datum of 1983.

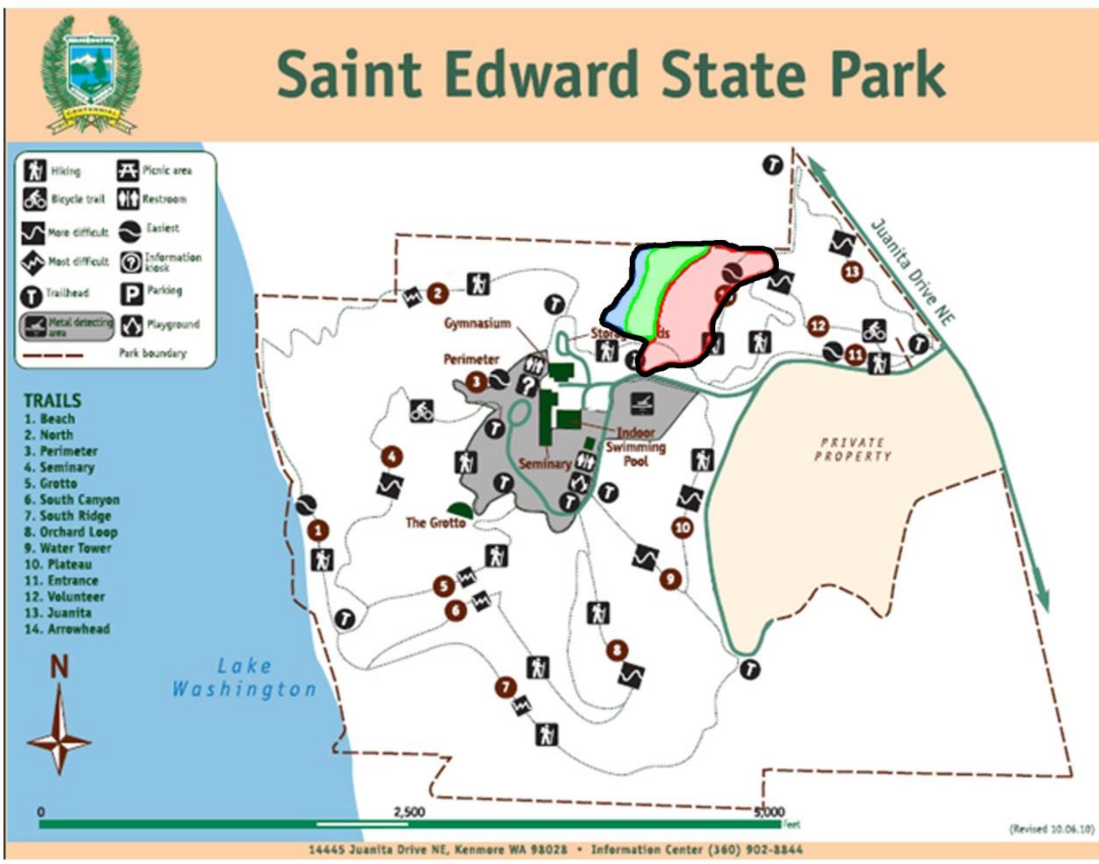

Figure 3. Location of study area (in dark outline) at St. Edward State Park, sampled in winter 2015. The same study area was previously surveyed for *Ilex aquifolium* over three years: 2011 (red), 2012 (green), and 2013 (blue). Bordering the north edge of Park and study area, is a residential neighborhood, with most homes constructed in the 1950's to the 1970's, and an elementary school and grounds established in 1957. Both *Prunus* species are planted in the neighborhood. Several large *P. laurocerasus* plantings were installed near the seminary building in the 1930's (NPS 2006). 2010 Washington State Parks map.

RESULTS AND DISCUSSION

Prunus *in St. Edward State Park*

We located, sampled, and removed 231 *P. laurcerasus* and 22 *P. lusitanica* individuals in the 9.2 ha study area, or approximately 25.1 and 2.4 trees/ha respectively (Table 1). This compares to 58.7 sampled *Ilex* per hectare in the same study area in 2012 (Table 1). We also removed a total of 212 and 2 unsampled small sprouts of the two species respectively. Thus the total stem density (sampled trees plus unsampled sprouts) of the two species was 48.2 and 2.6 stems/ha respectively (compared to 140 stems/ha for *Ilex* in 2012; Stokes et al. 2014b). Of the total sampled trees, 83 *P. laurcerasus* (9.0/ha) and 12 *P. lusitanica* (1.3/ha) were $\geq$ 10 years of age. Relative to $\geq$ 10 yr-old *Prunus*, nearly twice as many $\geq$10 yr old *Ilex* were found in the same area in 2012 (n = 161, 17.5/ha).

To explicitly compare *Prunus* and *Ilex* abundance at a point in time, we compared numbers of trees that were at least 10 years old in 2012 (i.e., all *Prunus* in our study that were at least 13 years old when sampled in 2015, and all *Ilex* that were at least 10 years old in 2012). Using this metric, at the time when there were 161 10+ year old *Ilex* (17.5/ha), there were 56 (6.1/ha) *P. laurocerasus* and 7 *P. lusitanica* (0.8/ha). Thus, assuming negligible mortality of 10+ year old *Prunus* in the last 3 years, we infer that in 2012, among established trees, *Ilex* was nearly three times as abundant as *P. laurocerasus*, and 20 times as abundant as *P. lusitanica* in the study area. Possible reasons for the greater abundance of *Ilex* include earlier initial invasion, greater propensity for sprouting, greater rates of seed production or vegetative establishment, and more rapid dispersal.

Unsampled sprouts (basal diameter < 1 cm and within 1 meter of a sampled tree) were likely very young; average age of sampled *P. laurocersus* in this size range (basal diameter < 1 cm) was 2.23 years (sd = 0.88, range = 1 – 4, n = 30). We conclude that by excluding these small sprouts from our sample, we did not miss any 10+ year old trees. The ratio of unsampled sprouts to sampled trees was 0.92 sprouts per sampled tree for *P. laurcerasus*, and 0.09 for *P. lusitanica*. These are both lower than the value for *Ilex* (1.43), which may suggest that the *Prunus* species are less prone to sprouting, although other factors, such greater mean age of the *Ilex* sample may also be important.

Of the 253 *Prunus* trees sampled, we were able to remove 224 (89%) by uprooting by hand or using a weed wrench. All unsampled *Prunus* sprouts (n = 214) were removed by uprooting. The 29 sampled *Prunus* that could not be uprooted (28 *P. laurcerasus* and 1 *P. lusitanica*), were cut at the base, with glyphosate herbicide applied to the cut stumps of 28 of the 29. The maximum basal diameter of trees removed by uprooting was 8.5 cm; the minimum basal diameter of trees that could not be removed by uprooting was 3.9 cm. Approximately half (49%, n = 17) of the 35 trees with basal diameter $\geq$ 3.9 cm and $\leq$ 8.5 cm could be uprooted. These are very similar to the results for *Ilex* (Stokes et al. 2014b), suggesting a similar relationship between tree size and difficulty of removal for these three species.

Prunus growth form ranged from tree-like to low spreading shrub-like. Both *Prunus* species had a relatively high rate of damage from falling canopy trees and limbs. Of the sample of 231 *P. laurocerasus*, 10 (4%) were broken off (but still alive), 13 (6%) were

Table 1. Number, density, age, and size of sampled Cherry laurel (a) and Portuguese laurel (b) in St. Edward State Park study area (9.2 ha. [22.8 acres]), sampled Jan. – Mar., 2015. English holly sample (same study area, sampled 2011-2013) shown for comparison (c).

a. Cherry laurel *Prunus laurocerasus*

Sample	N	Density (ha^{-1})	Age (yrs)		Basal diam. (cm)		Height (m)*		Crown diam. (m)*	
			avg (SD)	range	avg (SD)	range	avg (SD)	range	avg (SD)	range
All	231	25.1	9.1 (5.9)	1 – 35	2.7 (2.8)	0.2 – 24.3	2.8 (2.2)	0.1 – 13.7	1.0 (1.0)	0.0 – 5.5
≥ 10 yrs	83	9.0	15.5 (4.8)	10 – 35	5.1 (3.5)	1.0 – 24.3	4.9 (2.3)	0.4 – 13.7	2.0 (1.0)	0.1 – 5.5
< 10 yrs	148	16.1	5.4 (2.2)	1 – 9	1.4 (0.6)	0.2 – 3.3	1.6 (0.9)	0.1 – 4.2	0.5 (0.3)	0.0 – 1.5
≥ 13 yrs**	56	6.1	17.8 (4.4)	13 – 35						
< 13 yrs**	175	19.0	6.3 (2.9)	1 – 12						

* *Trees that were previously broken off (n = 10) are excluded from averages for height and canopy diameter.*
** *Trees that were ≥ 10 years old in 2012, i.e., comparable to ≥ 10 year-old tree sample in the Ilex survey (panel c below).*

b. Portuguese laurel *Prunus lusitanica*

Sample	N	Density (ha^{-1})	Age (yrs)		Basal diam. (cm)		Height (m)*		Crown diam. (m)*	
			avg (SD)	range	avg (SD)	range	avg (SD)	range	avg (SD)	range
All	22	2.4	10.1 (6.4)	1 – 21	3.2 (2.7)	0.2 – 11.8	3.2 (2.3)	0.1 – 7.2	1.5 (1.0)	0.1 – 3.4
≥ 10 yrs	12	1.3	15.1 (4.1)	10 – 21	4.8 (2.6)	1.9 – 11.8	4.4 (1.7)	2.1 – 7.2	2.0 (0.7)	1.1 – 3.4
< 10 yrs	10	1.1	4.2 (2.3)	1 – 8	1.2 (0.8)	0.2 – 2.7	0.8 (0.8)	0.1 – 2.4	0.4 (0.4)	0.1 – 1.1
≥ 13 yrs**	7	0.8	18.3 (2.1)	15 – 21						
< 13 yrs**	15	1.6	6.3 (3.6)	1 – 11						

* *Trees that were previously broken off (n = 7) are excluded from averages for height and canopy diameter.*
** *Trees that were ≥ 10 years old in 2012, i.e., comparable to ≥ 10 year-old tree sample in the Ilex survey (panel c below).*

c. English holly *Ilex aquifolium*

Sample*	N*	Density (ha^{-1})*	Age (yrs)**		Basal diam. (cm)		Height (m)***		Crown diam. (m)***	
			avg (SD)	range	avg (SD)	range	avg (SD)	range	avg (SD)	range
All	540	58.7	8.5 (8.0)	1 – 46	1.8 (3.2)	0.1 – 35.0	1.5 (1.9)	0.1 – 18.0	0.7 (1.0)	0.1 – 10.5
≥ 10 yrs	161	17.5	18.7 (7.8)	10 – 46	4.5 (4.8)	0.6 – 35.0	3.3 (2.7)	0.5 – 18.0	1.7 (1.4)	0.2 – 10.5
< 10 yrs	379	41.2	4.3 (2.3)	1 – 9	0.7 (0.4)	0.1 – 3.3	0.7 (0.5)	0.1 – 3.0	0.2 (0.2)	0.1 – 1.3

* *Sample numbers and density updated with 8 additional individuals >3yrs old discovered in 2015 (i.e., missed in 2011-13 surveys).One (cut ca 2009) individual was 27 yrs old in 2012; all others were < 10 yrs old in 2012. All except 2 had spread vegetatively from the 27yr old tree.*
** *Age could not be determined for 3 small individuals.*
*** *Trees for which height (n = 15) or canopy diameter (n = 10) was not determined are excluded from averages for those measures.*

prostrate, and another 4 (2%) had a steeply angled trunk. Among *P. lusitanica*, 6 (27%) were broken off and 3 (14%) were prostrate. Fourteen (6%) *P. laurocerasus* had a substantial amount (>20%) of dead or yellowing foliage, and 10 (4%) had been browsed. Five (23%) *P. lusitanica* had yellowing foliage. One (5%) had been previously cut near the base. While small numbers make comparisons tentative, it appears that at St. Edward Park, *Prunus* is similar to *Ilex* in level of damage caused by falling trees and limbs. The range of growth form was also similar, however *Prunus* more commonly had a shrub growth form, and also more frequently had unhealthy foliage than *Ilex. P. laurocerasus* may also be more subject to herbivory.

We found no dead *Prunus*, and we presume that it, like *Ilex*, has a low mortality rate in St. Edward Park, at least after becoming well established (e.g., ≥ 10 years old). However, because dead *Prunus* trees are probably less distinctive and identifiable than *Ilex*, and given the greater frequency of unhealthy foliage in *Prunus*, we cannot be certain that mortality rates are as extremely low in *Prunus* as they appear to be in *Ilex* (Stokes et al. 2014a).

P. laurcerasus in the study area ranged from 1 to 35 years of age, with many young individuals and declining numbers of older ages (Fig. 4a). Sixty-four percent (n = 148) of sampled trees were less than 10 years old. Within this general pattern of declining numbers with age, there was variability in representation of ages and years of establishment (Fig. 5a), with anomalously low levels of establishment in some years (e.g., 1989-1991, 1995-1997, 2004), and high levels in others (e.g., 1992-1994, 2007). These anomalies may reflect unusually unfavorable and favorable germination or establishment conditions in some years. *P. lusitanica* in the study area ranged from 1 to 21 years of age. Forty-five percent (n = 10) of sampled trees were less than 10 years old (Fig. 4b). There was no clear trend in establishment rate over the species' 21-year period of presence (Figs. 4b & 5b.), perhaps because of small sample size.

The maximum ages of the two *Prunus* species (35 and 21 years) suggests that the invasions of those species began more recently than the *Ilex* invasion (oldest sampled *Ilex* was 46 years old in 2012), at least within the study area. Opportunistic observations indicate that there are forested areas of the park outside our study area (e.g., the SE corner of the park) where invading *P. lusitanica* are larger and presumably older than those sampled in our study (D. Stokes, pers. obs.).

Of the plants for which reproductive mode of origin could be conclusively determined, 31 of 225 (14%) of *P. laurcerasus* versus 13 of 21 (62%) of *P. lusitanica* originated from seed, with the rest originating vegetatively from roots or branches of established plants. By comparison, 36% of *Ilex* in the same study area originated from seed (Stokes et al. 2014b). The greater proportion of seed-originated *P. lusitanica* may reflect the fact that this species is a more recent invader, that it has a lower tendency to sprout from roots, or it may be an artifact of small sample size.

While all of these species spread vegetatively, the greater capacity for long-distance spread via seed makes seed-origination a particularly important component of the invasion, accounting for all spread into unoccupied habitat. Most (75%) *P. laurocerasus* trees originating from seed established in the early 1990's – early 2000's, with less seed establishment occurring earlier (6%) and later (19%; Fig. 6a). Establishment of *P. lusitanica* from seed did not show a trend over time (Fig. 7a). The cumulative numbers of seed-originated trees at the time of our study suggests a rapid increase in the numbers for both species over the course of the invasion (Figs. 6b & 7b). However, because our methods likely lead to an undercount of the youngest age classes (unsampled small individuals [< 1 cm basal diameter] near sampled trees), and because there may be some (unknown) level of mortality of established trees, this pattern must be interpreted cautiously (see also discussion p. 23).

The height of *P. laurocerasus* trees in the study area ranged from 0.1 m to 13.7 m, with an average of 2.8 m (sd = 2.2, n = 221). The second oldest *P. laurocerasus* in the study area (31 years) was the tallest (13.7 m) and had the largest stem diameter (24.3 cm basal diameter) in the sample population. The oldest tree (35 years) had a broken top. Its measured height (10.4 m) and basal diameter (16.5 cm) were the second greatest in the sample. Tree height (Fig. 8a) and diameter at base (Fig. 9a) were strongly positively correlated with age. Both height-age and basal diameter-age curves became progressively steeper with age, indicating that in the environment at St. Edward Park the age range of trees in our sample is a stage of accelerating biomass accumulation.

The height of *P. lusitanica* trees in the study area ranged from 0.1 m to 7.2 m, with an average of 3.2 m (sd = 2.3, n = 15). The tallest tree (7.2 m) and the tree with the largest basal diameter (11.8 cm) was one of the two oldest trees (Two trees were 21 years old). Tree height (Fig. 8b) and diameter at base (Fig. 9b) were also positively correlated with age. The basal diameter-age curve became progressively steeper with age, however the height-age curve did not. A larger sample of *P. lusitanica* is required to determine if the age range of our sample is a stage of accelerating biomass accumulation in this species.

Nearly all (95 – 99%) of the sampled *P. laurocerasus* were below the typical maximum height of the species in Europe (6 – 12 m [Walther 1999]; 14 m [Booy et al. 2015]) and all had stem diameters far below the maximum (60 cm; Rushforth 1999). Similarly, all *P. lusitanica* in our sample were below the maximum height observed in Europe (10 m [Arbolapp 2017]; 12 m [Booy et al. 2015]). Information on lifespan of these species is scarce, but there is some indication that 50 – 150 years is expected for both species in urban settings (SelecTree nda&b). Thus, there is potential for substantial future size increase and persistence among the *Prunus* presently existing in St. Edward Park.

As with height and stem diameter, *P. laurocerasus* canopy diameter (Fig. 10a) was highly correlated with age, and increased at an increasing rate in the age range of our sample trees. Area covered by P. *laurocerasus* canopy ranged from < 0.01 m^2 to a maximum of 23.8 m^2 for the second oldest tree (31 yrs) in the sample (the oldest tree had a broken top). Canopy diameter and age were positively correlated in the smaller *P. lusitanica* sample as well (Fig. 10b), however the rate of increase did not increase over time.

We saw no flowers or berries on any of the *Prunus* in our sample area, as we conducted our study at a time of year when flowers and berries are not present in these species.

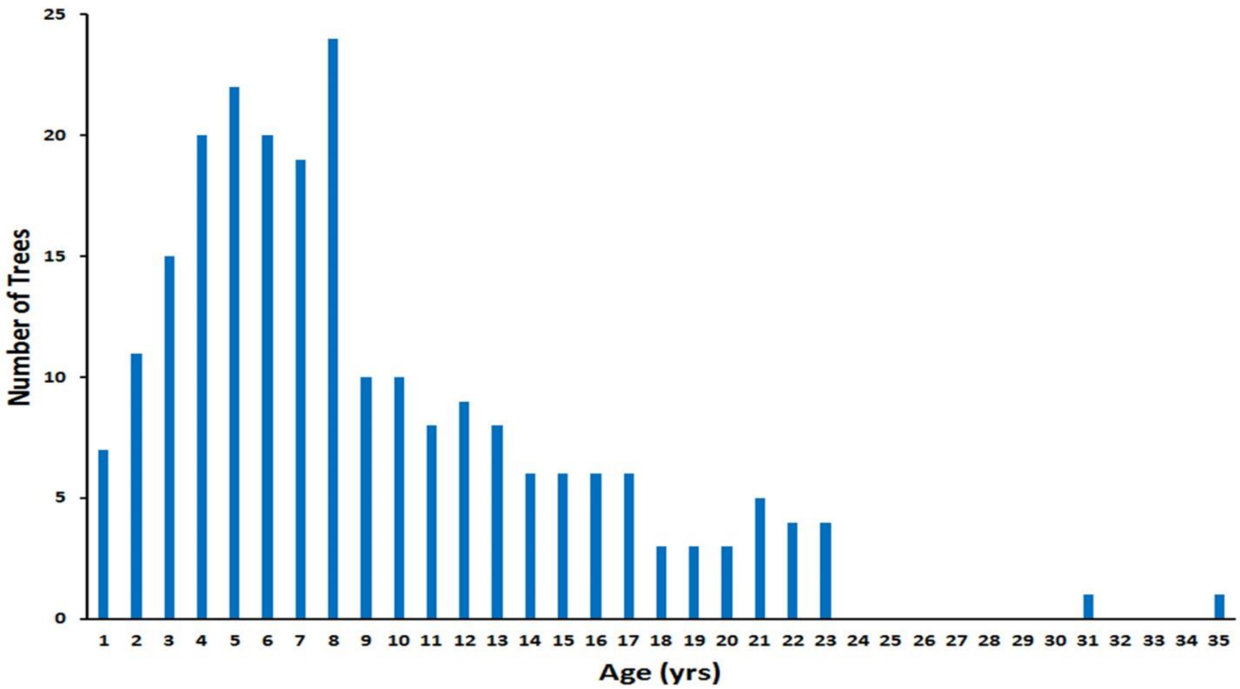

Figure 4a. Ages of sampled *P. laurocerasus* (n = 231) in St. Edward State Park study area. All individuals $\geq$ 1 cm in basal diameter or > 1 m from nearest sampled tree were sampled. Young trees (< *ca* 5 yrs) are underrepresented because small individuals (< 1cm basal diameter) within 1 m of sampled trees were not sampled (see text).

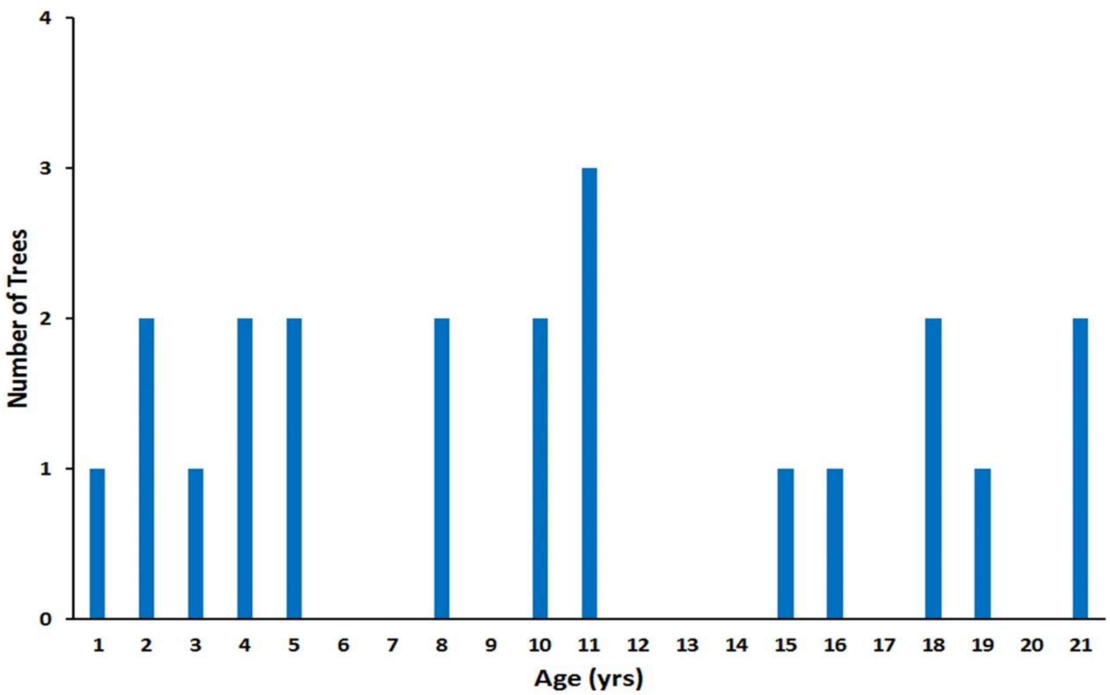

Figure 4b. Ages of sampled *P. lusitanica* (n = 22) in St. Edward State Park study area. All individuals $\geq$ 1 cm in basal diameter or > 1 m from nearest sampled tree were sampled. Young trees (< *ca* 5 yrs) are underrepresented because small individuals (< 1cm basal diameter) within 1 m of sampled trees were not sampled (see text).

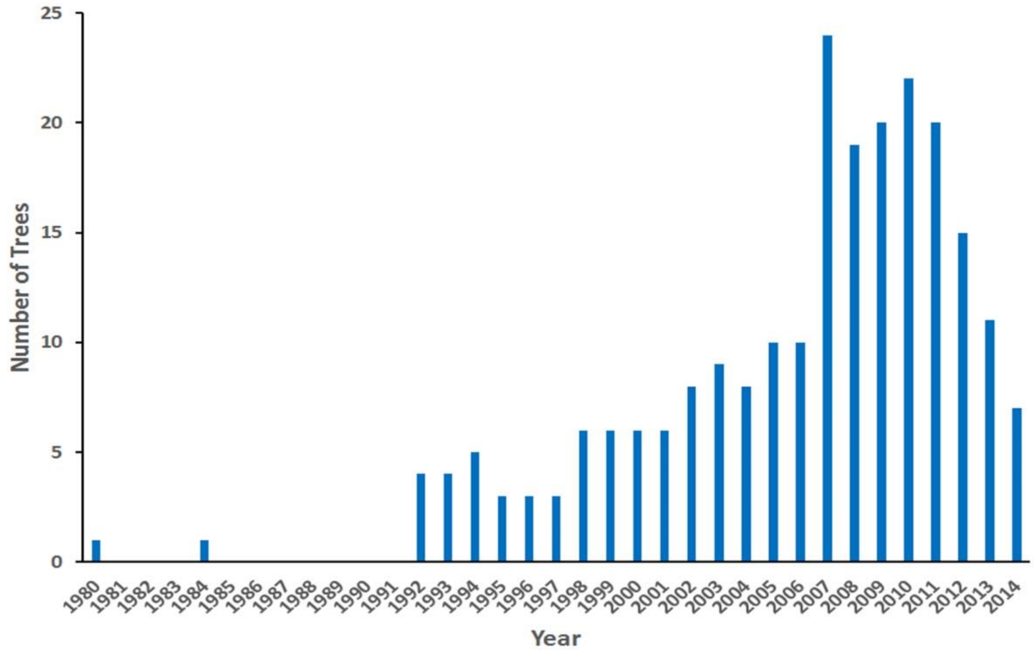

Figure 5a. Year of establishment of *P. laurocerasus* (n = 231) in St. Edward State Park sample area. Data collected in 2015. Young trees (< *ca* 5 yrs; 2011 and later) are under-represented due to incomplete sampling of small individuals (< 1cm basal diameter) within 1 m of sample trees.

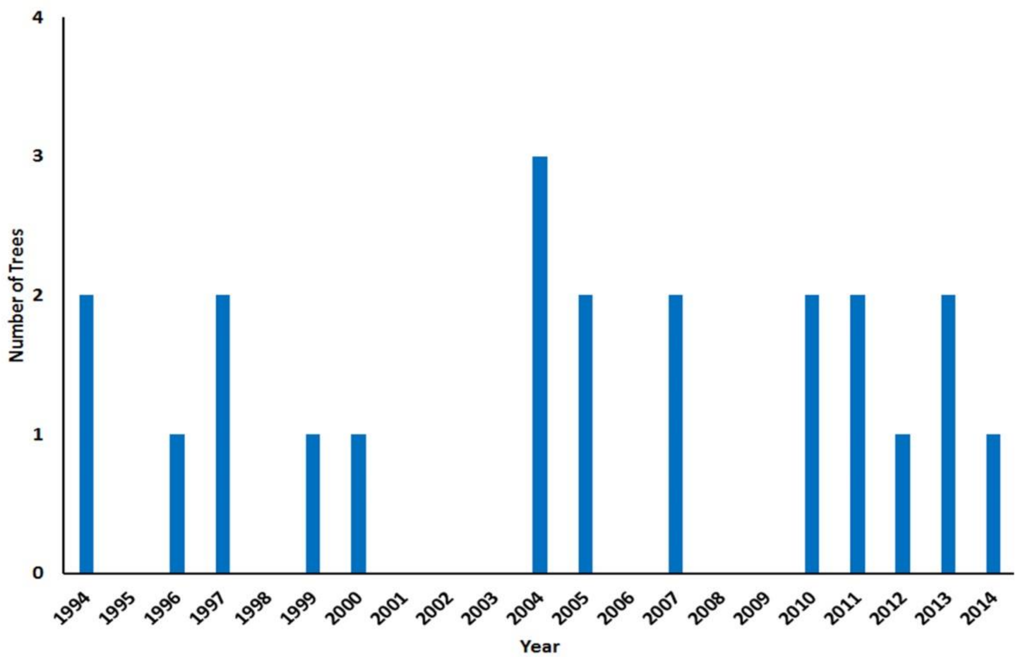

Figure 5b. Year of establishment of *P. lusitanica* (n = 22) in St. Edward State Park sample area. Data collected in 2015. Young trees (< *ca* 5 yrs; 2011 and later) are under-represented due to incomplete sampling of small individuals (< 1cm basal diameter) within 1 m of sample trees.

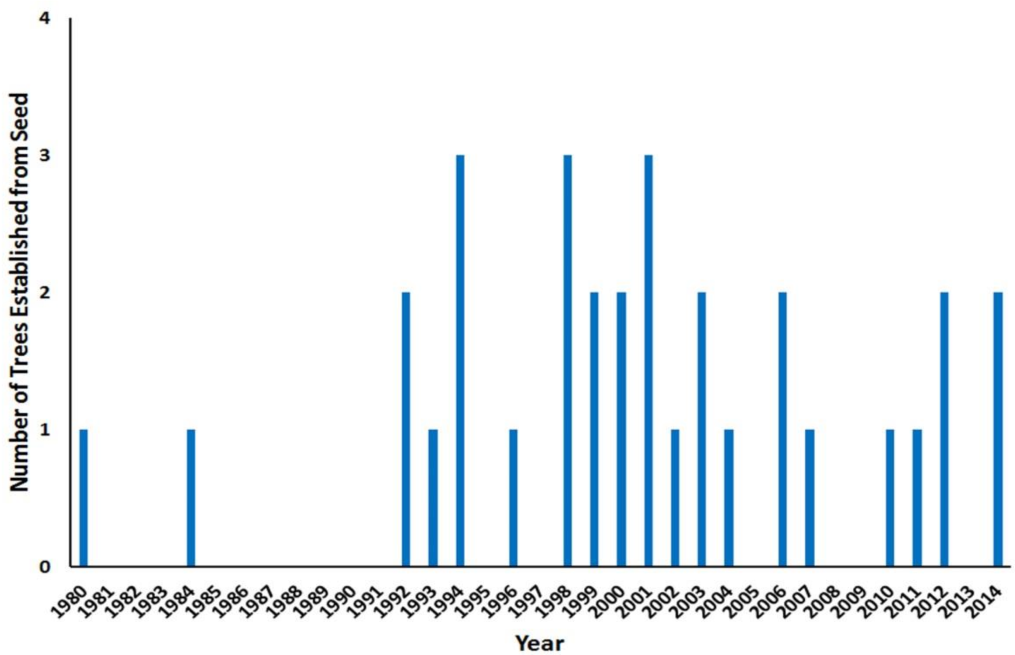

Figure 6a. Establishment of *P. laurocerasus* resulting from seed (n = 32) in the 2015 study area in St. Edward State Park. Establishments in recent years (2011 and later) may be undercounted due to incomplete sampling of small individuals (< 1cm basal diameter) within 1 m of sample trees. While most of these were of vegetative origin, a small number may have originated from seed.

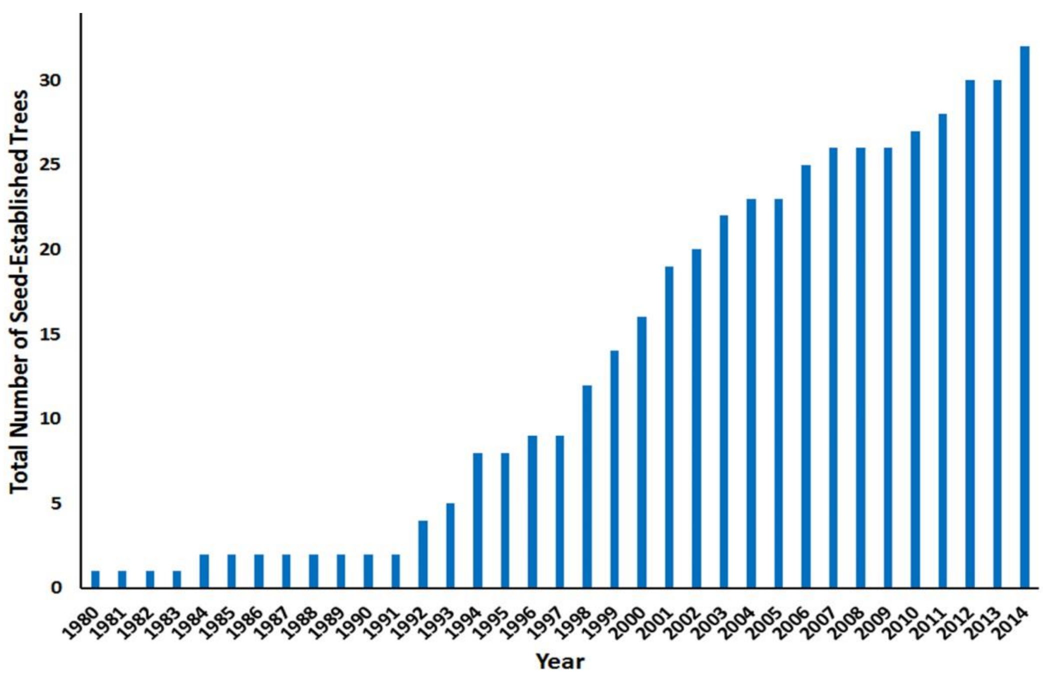

Figure 6b. Cumulative number of *P. laurocerasus* in the study area resulting from seed (n=32). This figure may not accurately represent the pattern of population growth due to a) likely under-representation of young (2011 and later) plants, and b) it does not include possible mortality over this timespan (see text, pp. 12 & 23).

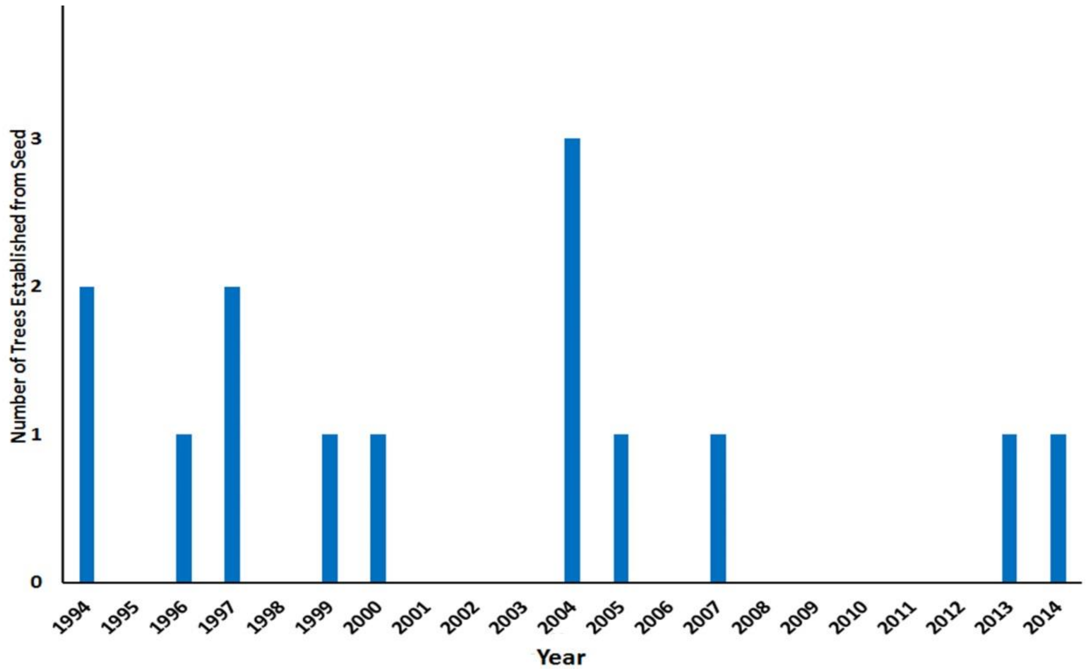

Figure 7a. Establishment of *P. lusitanica* resulting from seed (n = 14) in the 2015 study area in St. Edward State Park. Establishments in recent years (2011 and later) may be undercounted due to incomplete sampling of small individuals (< 1cm basal diameter) within 1 m of sample trees. While most of these were of vegetative origin, a small number may have originated from seed.

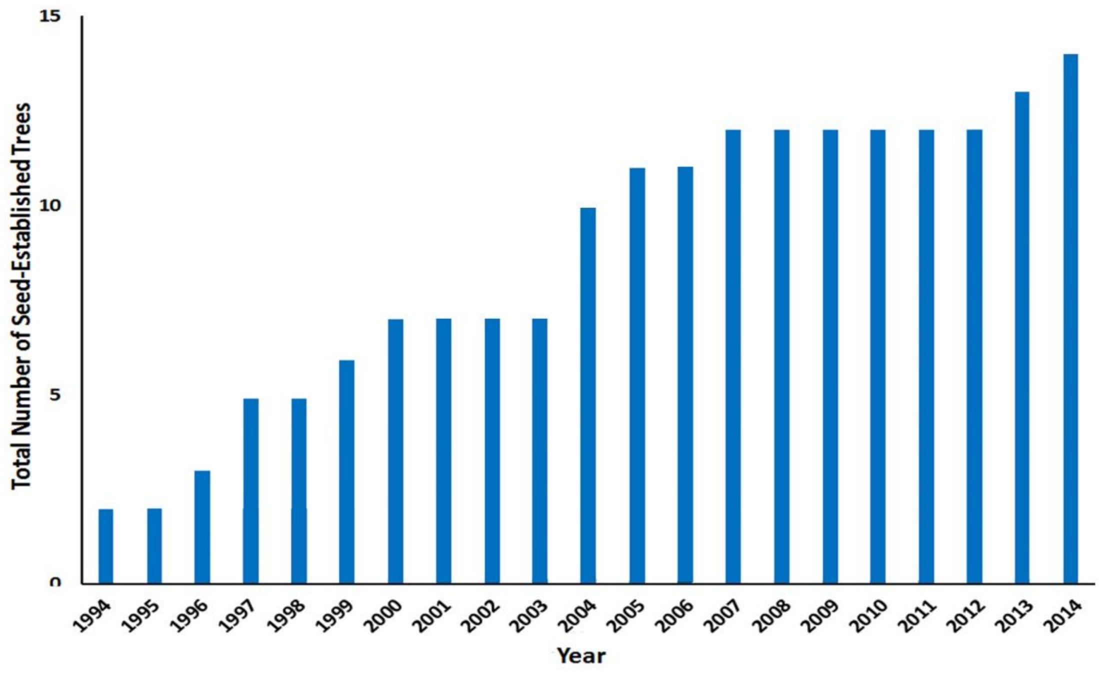

Figure 7b. Cumulative number of *P. lusitanica* in the study area resulting from seed (n=14). This figure may not accurately represent the pattern of population growth due to a) likely under-representation of young (2011 and later) plants, and b) it does not include possible mortality over this timespan (see text, pp. 12 & 23).

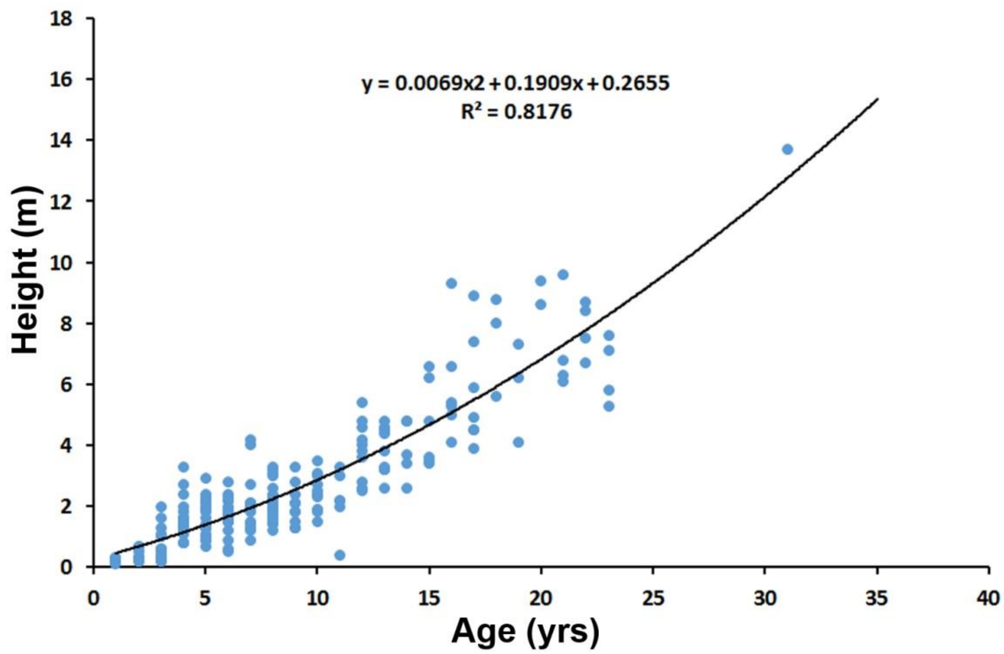

Figure 8a. *P. laurocerasus* height by age (n = 221) in St. Edward State Park. Trees that had been previously broken off (n = 10) are excluded.

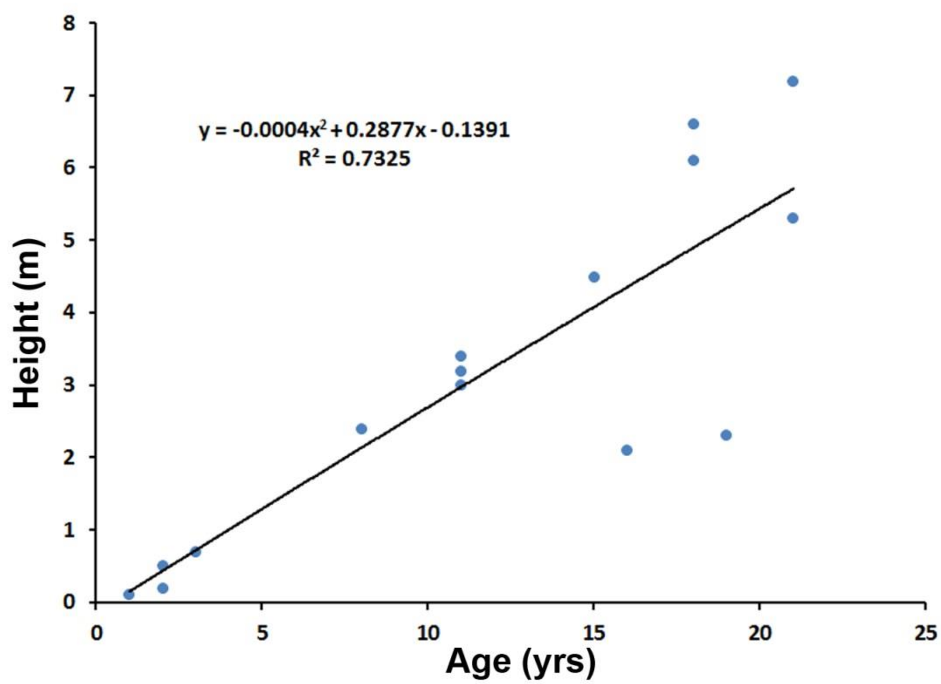

Figure 8b. *P. lusitanica* height by age (n = 15) in St. Edward State Park. Trees that had been previously broken off (n = 6) or cut down (n = 1) are excluded.

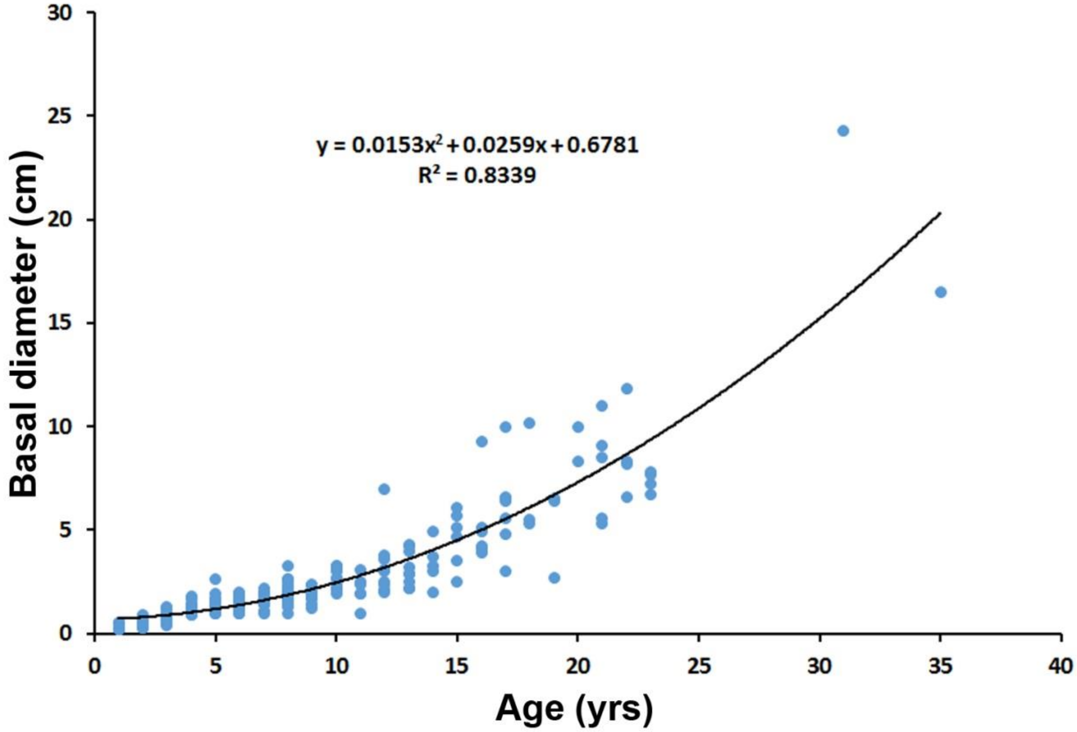

Figure 9a. Stem diameter by age for all *P. laurocerasus* (n = 231) in St. Edward State Park study area, 2015.

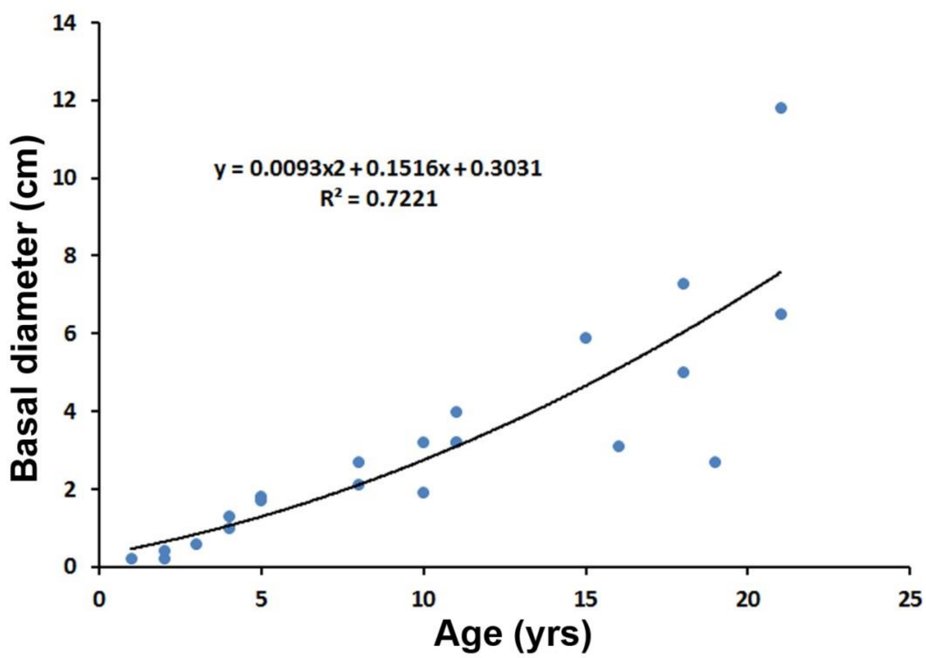

Figure 9b. Stem diameter by age for all *P. lusitanica* (n = 22) in St. Edward State Park study area, 2015.

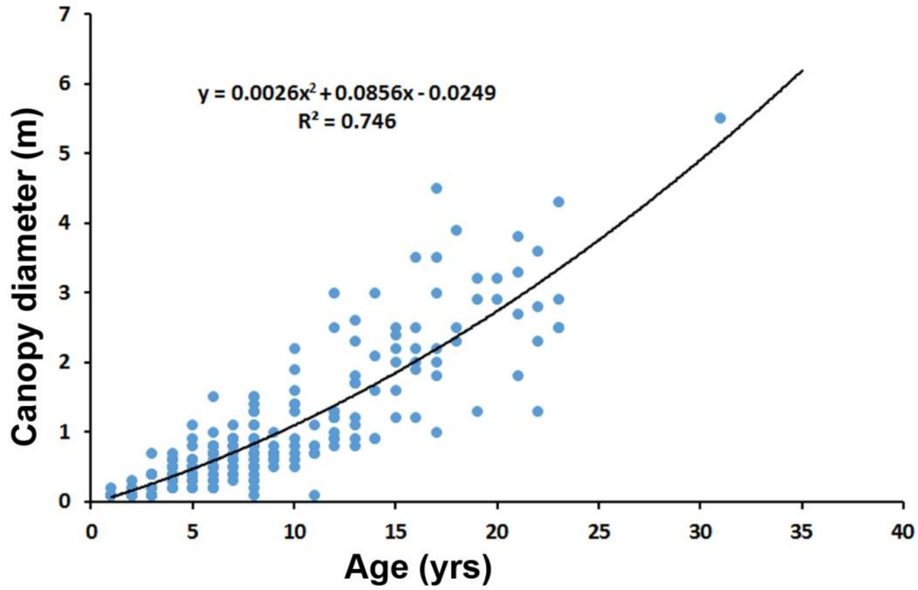

Figure 10a. *P. laurocerasus* canopy diameter by age (n = 221) in St. Edward State Park. Trees that had been previously broken off (n = 10) are excluded.

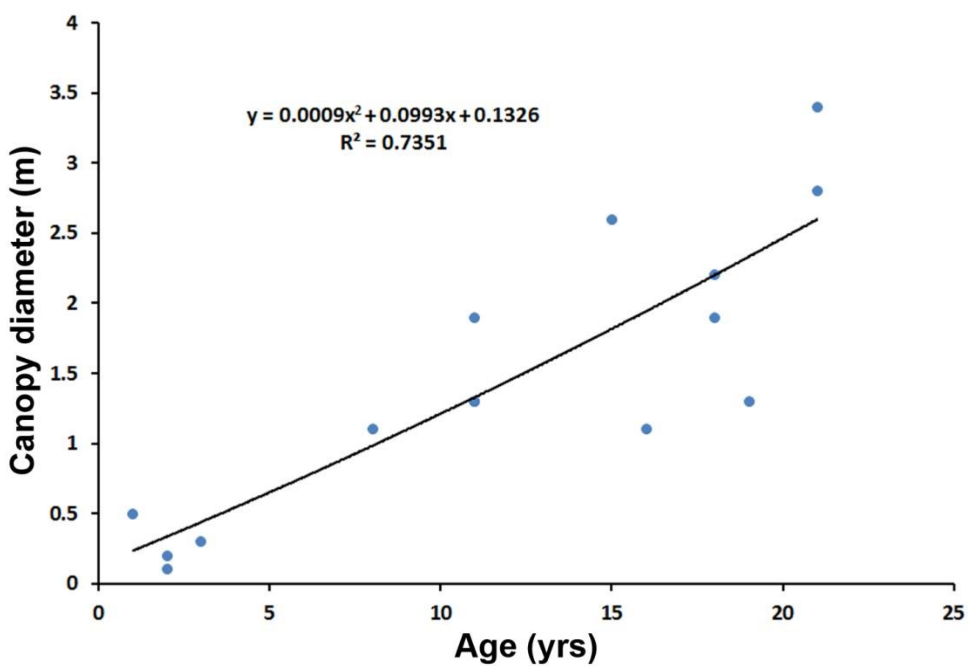

Figure 10b. *P. lusitanica* canopy diameter by age (n = 15) in St. Edward State Park. Trees that had been previously broken off (n = 6) or cut down (n = 1) are excluded.

Effects of Prunus *on native species*

Native shrub and ground cover vegetation was dramatically reduced under *Prunus* canopy. We compared native woody and evergreen vascular plant cover under the canopy of 23 large study-area *Prunus* trees or groups of trees ("clumps") with the vegetation in the adjacent area within a 5 m radius of the *Prunus*. Under *P. laurocerasus* canopy, native vegetation cover averaged 21%, versus 63% in the adjacent area (Fig 11a). Nearly all native vegetation cover under *Prunus* canopy was located near the canopy dripline, with very little cover more than 0.5 m inside the drip line. Substantial areas under large tree clumps were virtually devoid of vegetation. Reduction of native cover was also observed under *P. lusitanica*, however the difference was marginally insignificant ($p = 0.06$), perhaps because the sample size was very small ($n= 5$; Fig 11b).

We observed a total of 14 species of native woody and evergreen vascular ground cover plant species under and around *Prunus* trees at our 23 comparison sites (Table 2). All native species with at least 5% coverage either under or adjacent to *Prunus* were less abundant under *Prunus* canopy, although small sample size precluded tests for statistically significant differences for all but the most common species (Fig. 12).

While apparently indicating that *Prunus* has strong negative effects on native plant species, these results are based on small samples and must be considered preliminary. Furthermore, how the negative effects of *Prunus* compare to those of native tree species is unknown, and merits investigation.

Table 2. Native plant species found in ground cover or shrub layer within a 5 m radius and under the canopy of *Prunus* trees and tree clumps (n = 23 sites), listed in order of frequency of occurrence.

Species	**Common Name**	**No. of sites Present**	**Adjacent Present (≥5%)**	**Under Present (≥5%)**
Polystichum munitum	Sword fern	23	23 (21)	19 (16)
Rubus ursinus	Native blackberry	20	20 (15)	19 (8)
R. spectabilis	Salmonberry	17	16 (15)	8 (4)
Oemleria cerasiformis	Indian plum	15	15 (13)	9 (4)
Mahonia nervosa	Dwarf Oregon grape	10	9 (3)	7 (2)
Thuja plicata	Western red cedar	8	8 (2)	1 (0)
Gaultheria shallon	Salal	7	7 (4)	4 (2)
Vaccinium parvifolium	Red huckleberry	6	6 (2)	2 (0)
Sambucus racemosa	Red elderberry	5	5 (0)	0 (0)
Oplopanax horridus	Devil's club	2	2 (1)	1 (1)
Athyrium filix-femina	Lady fern	2	2 (0)	0 (0)
Tolmiea menziesii	Youth on age	1	1 (0)	1 (0)
Corylus cornuta	California hazelnut	1	1 (0)	0 (0)
R. parviflorus	Thimbleberry	1	0 (0)	1 (0)
**Hedera helix*	English ivy	1	1 (1)	1 (0)

**non-native*

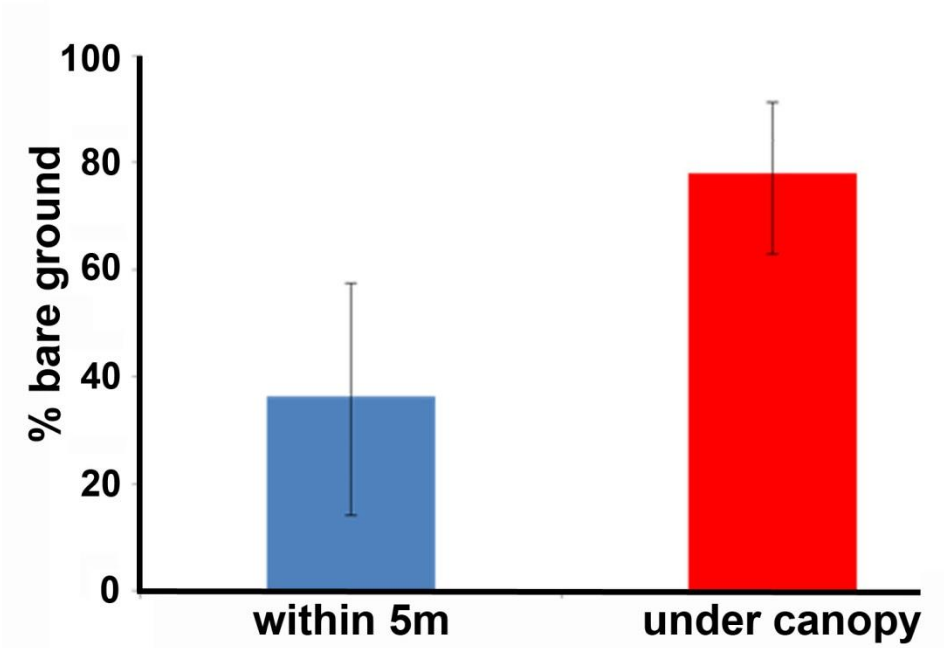

Figure 11a. Mean percent (± 1 SD) of area not covered by native woody or evergreen vascular plant species (bare ground) under and adjacent to *P. laurocerasus* at 18 paired comparison sites. Paired t-test on arcsin-transformed values ($t = 8.49, df = 17, p < 0.0001$).

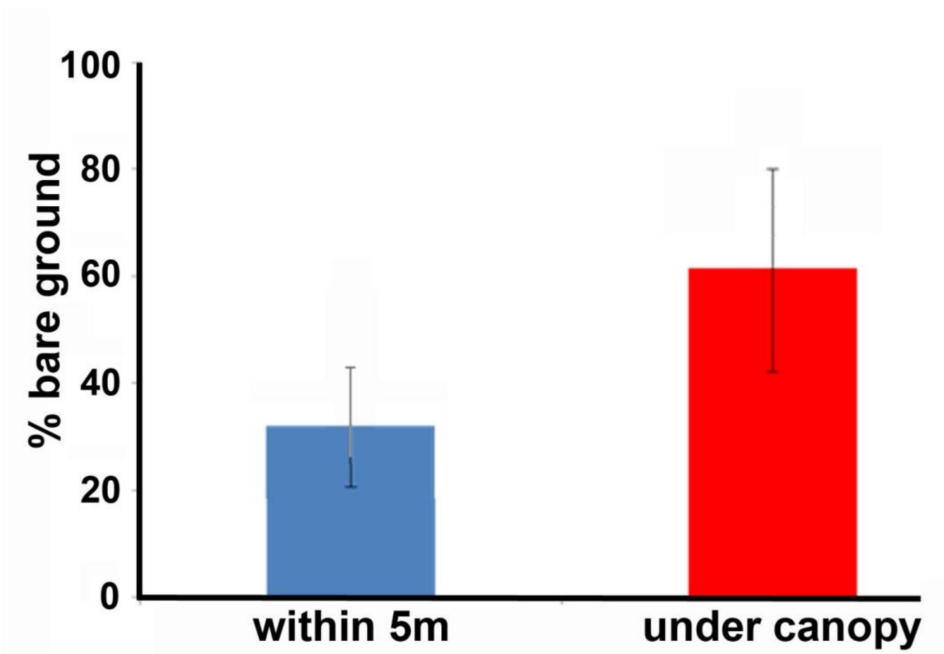

Figure 11b. Mean percent (± 1 SD) of area not covered by native woody or evergreen vascular plant species (bare ground) under and adjacent to *P. lusitanica* at 5 paired comparison sites. Paired t-test on arcsin-transformed values ($t = 2.69, df = 4, p = 0.055$).

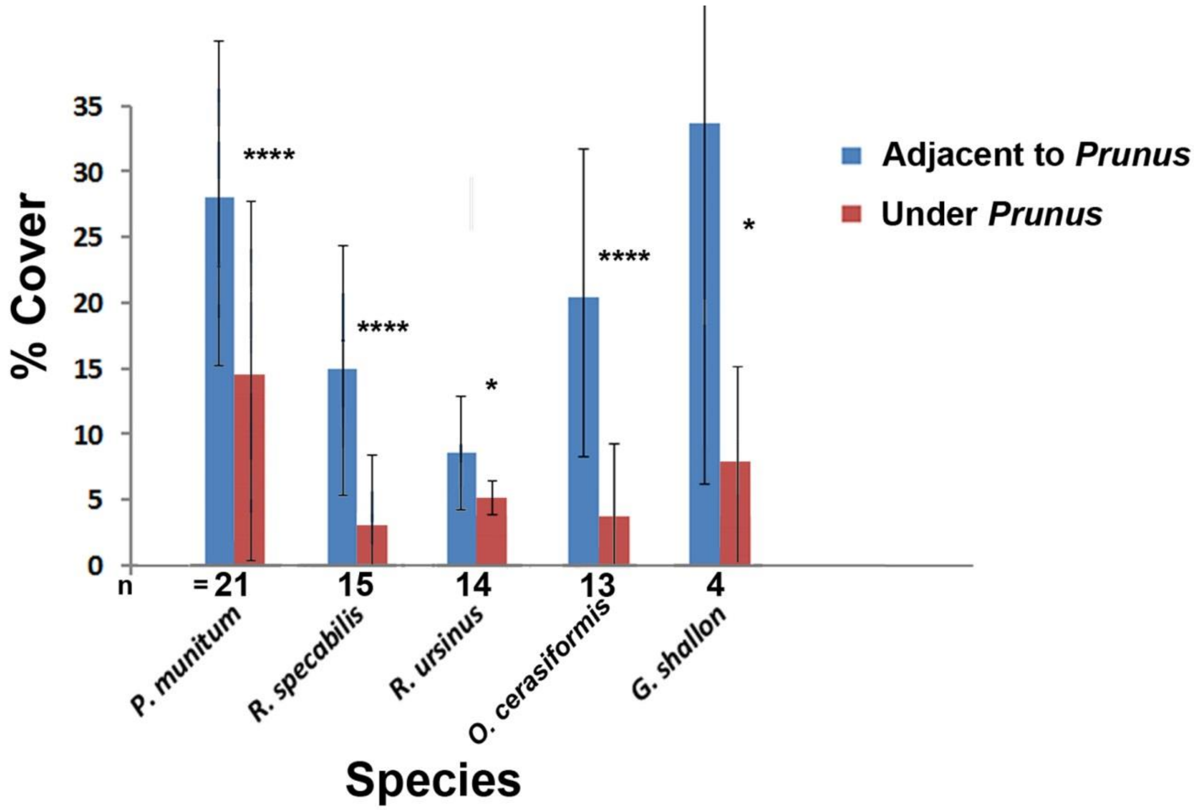

Figure 12. Mean percent (+ SD) cover of native ground cover and shrub species under and adjacent to (≤ 5 m) *Prunus* canopy at 23 comparison sites in St. Edward study area. For each species, the sample includes only sites where that species accounted for at least 5% cover adjacent to or under the *Prunus.* Only species with at least 4 comparison sites shown. Asterisks indicate statistical significance *(**** = $P < 0.001$, * = $P < 0.1$;* paired t-tests).

Dispersion and spread of Prunus

Both species of *Prunus* in our sample occurred at variable densities across the study area (Figs. 13 & 14). Greater numbers of both old and young plants of both species were present in forest edge areas, primarily in the northern portion of the study area near residential development, and along the southern edge where the forest borders a road and grass fields, and an area of horticultural plantings. The overall dispersion pattern of the trees was strongly clumped, a pattern that would be even more pronounced if the small unsampled individuals within 1 m of our sampled trees were included.

As is the case for *Ilex* (Stokes et al. 2014a), clumps of *Prunus* consisted of trees of different ages and appeared to result from one or more founder trees producing additional neighboring individuals. Most of the trees within clumps, including the unsampled sprouts, were the result of vegetative spread and had root systems that were connected to older trees. Clumps appeared to spread by radiation generally outward from the oldest trees (Figure 15). Trees that originated from seed were widely dispersed across most of the study area (Figs. 16 & 17). The presence of young seed-originated trees in previously unoccupied locations indicates that establishment from seed is ongoing.

Thus, like *Ilex*, the spatial distribution of *Prunus* is a product of both long-distance seed dispersal, resulting in widely spaced individuals and clumps, and localized vegetative spread and perhaps seedfall, leading to outward expansion of clumps. Both processes, seed dispersal and vegetative spread, appear to be active. Young individuals resulting from seed—potentially the founders of future clumps—were found far from other *Prunus* trees, and most existing clumps included very young plants, indicating ongoing clump expansion. The number of young seed-originated plants was higher for *P. laurocerasus* than for *P. lusitanica*, perhaps indicating that propagule pressure is higher in this species, a likely consequence of a more advanced invasion with greater numbers of mature plants acting as seed sources.

The concentration of *Prunus* at the north and south forest edges may result from more favorable conditions for establishment in edge environments, heavier propagule pressure in those areas, or both. Large *P. laurocerasus* hedges, planted in the 1930's, still exist around the seminary building, approximately 200 m from the southwest border of our study area. These hedges are currently maintained in closely trimmed form and do not typically produce flowers or berries; however it is possible that they have not always been maintained in this way. In the neighborhood along the north border of the study area, both *Prunus* species are common in residential landscaping (Stokes, pers. obs.), and a forest fragment approximately 250 m NE of our study area is heavily dominated by *P. laurocerasus* (Fig. 1).

In our earlier study of *Ilex* invasion, we determined that *Ilex*, at least by the time it reaches 10 years of age, has an extremely low mortality rate, which allowed us to use our *Ilex* sample to construct models of the past and future course of the invasion (Stokes et al. 2014a). We are unable to be as certain of very low mortality for *Prunus* because, relative to *Ilex*, *Prunus* is less readily identifiable after it has died (thus we would be less likely to detect dead individuals), and because we observed a greater (albeit still very low) proportion of *Prunus* with unhealthy foliage. Nonetheless, we observed few unhealthy *Prunus*, and no dead *Prunus*, and well-established young trees (i.e., in the age range of our sample trees $\geq$ 10 years old) generally tend to have very low mortality rates (Runkle 2000), particularly among shade-tolerant (Lorimer et al. 2001) and fast growing (Wyckoff and Clark 2002) species, such as the *Prunus* in our study area. Therefore, we expect $\geq$ 10 year-old *Prunus* mortality rates to be very low, and we undertake here the same temporal modeling of the *Prunus laurocerasus* (*P. lusitanica* sample size was too low) population and areal extent that we did for *Ilex* (Stokes et al. 2014a), with the caveat that precise trajectories are less certain than for the *Ilex* models.

With the above qualifications noted, it appears that like *Ilex*, *P. laurocerasus* is quickly becoming a much more prominent component of the forest at St. Edward Park (Figs. 18 and 19). In the space of little more than 25 years, it has increased in our study area from one 10 year-old tree in 1990 to the current 83 $\geq$ 10 year-old trees (and 231 trees of all ages), distributed widely over the study area. The pattern of increase closely follows an exponential function (Fig. 18a) with a doubling time of approximately 3 years. In the absence of density dependent constraints on population growth and spread, and without the removal of *Prunus* from the study area, *P. laurocerasus* would have exceeded 250 $\geq$ 10 year-old trees by 2020, and approximately 800 by 2025. Projected total areal extent of canopy follows a similar pattern (Fig. 18b), from 350 m^2 at present to more than 1000 m^2 in 2020 and 3000 m^2 by 2025. Thus, while the *Prunus* invasion in the study area apparently began more than 10 years after the *Ilex* invasion, and the *Prunus* population is less than half of that of *Ilex*, the rate of increase of *P. laurocerasus*, in both population and area covered, appears to be substantially more rapid (*Ilex* doubling time was 7.5 years and 5 years for population and area respectively [Stokes et al. 2014a]).

The small sample (n = 22) of *P. lusitanica* does not permit reliable modeling of its population growth and spread; however its demographic profile (Fig. 7b) and spatial distribution (Fig. 20) suggest a trajectory that is qualitatively similar to *P. laurocerasus*, although it is in an earlier stage of invasion.

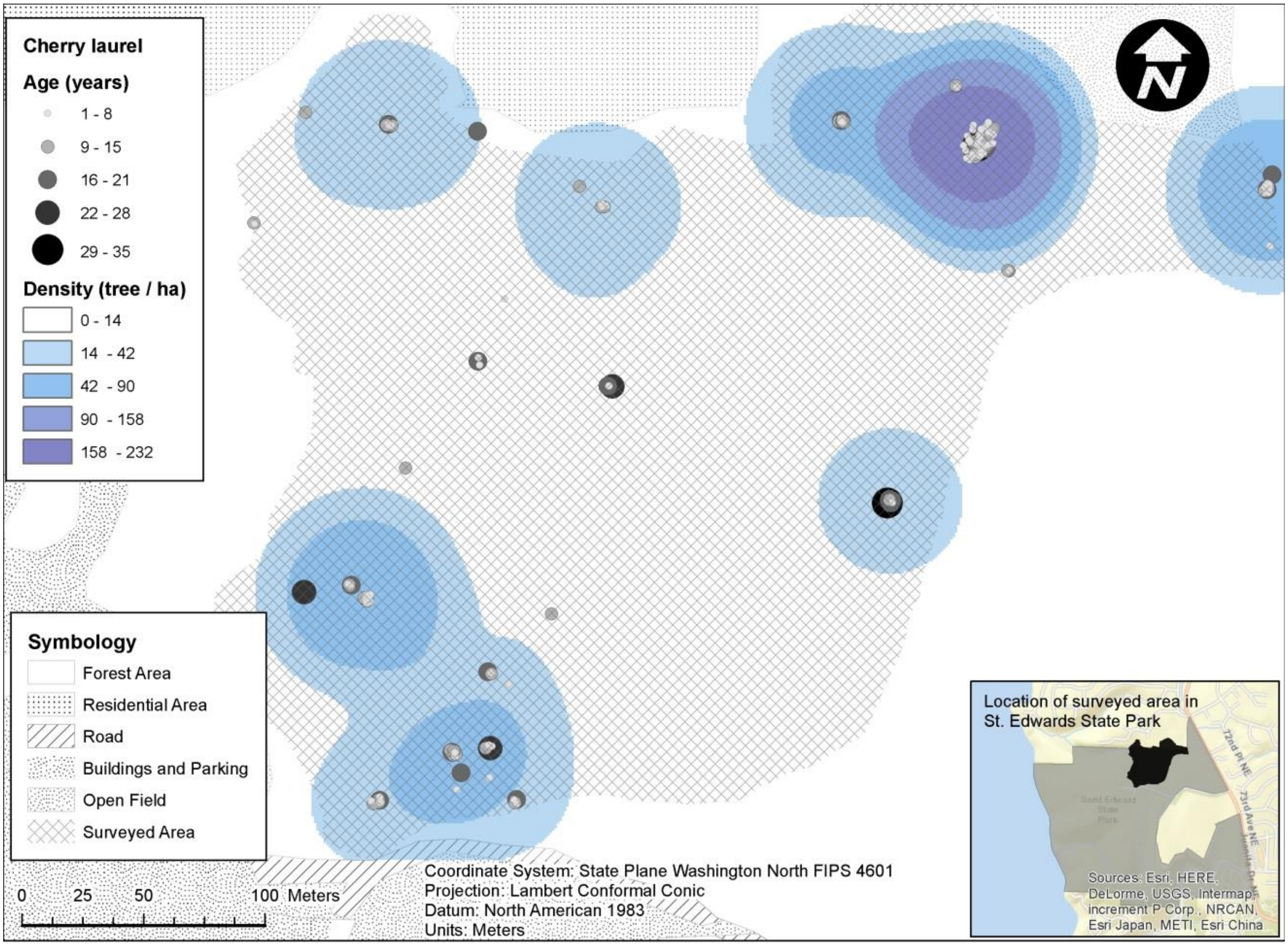

Figure 13. Spatial distribution, density, and age of all (n = 231) *P. laurocerasus* sampled and removed in the St. Edward Park study area in 2015.

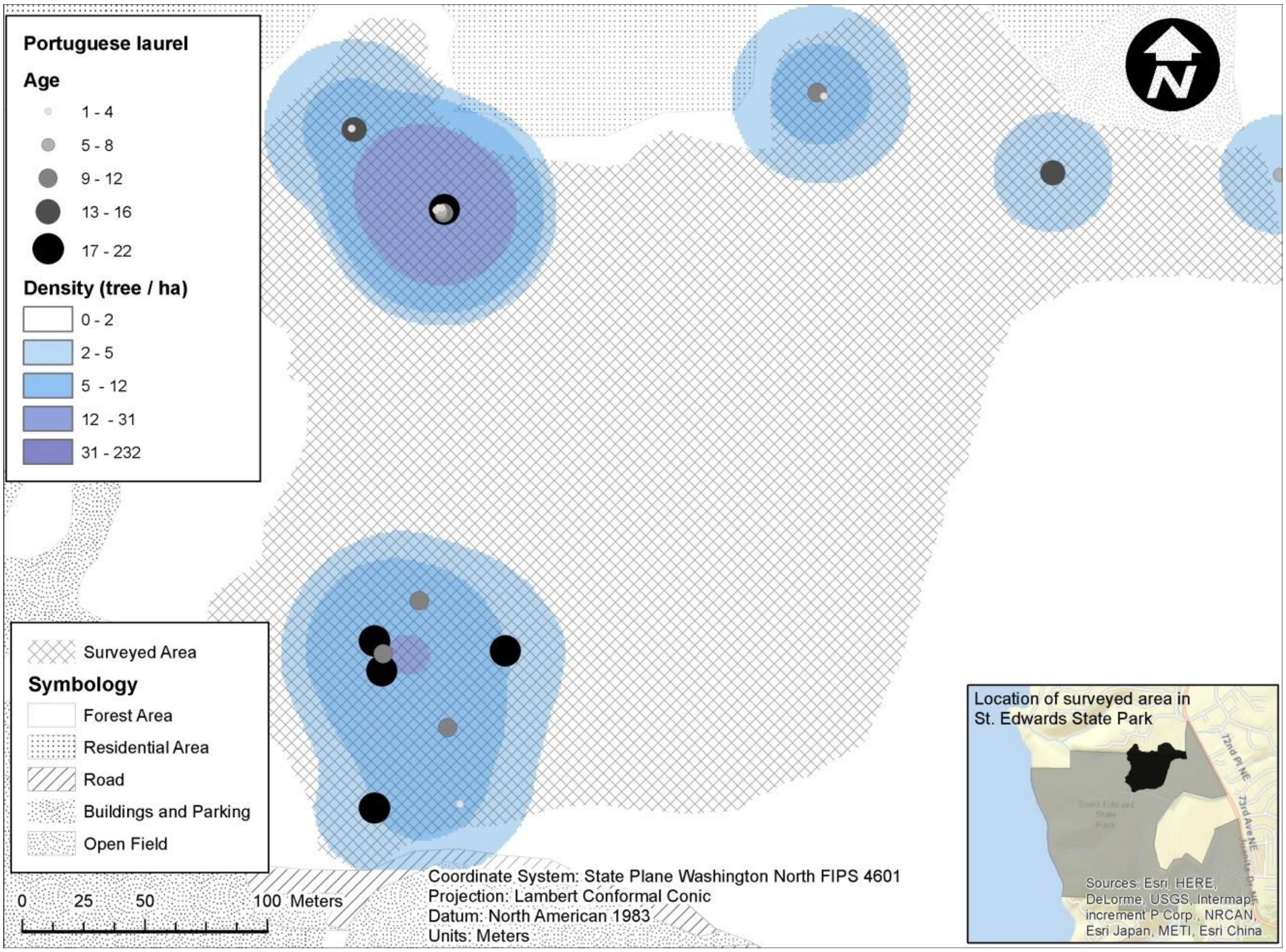

Figure 14. Spatial distribution, density, and age of all (n = 22) *P. lusitanica* sampled and removed in the St. Edward Park study area in 2015.

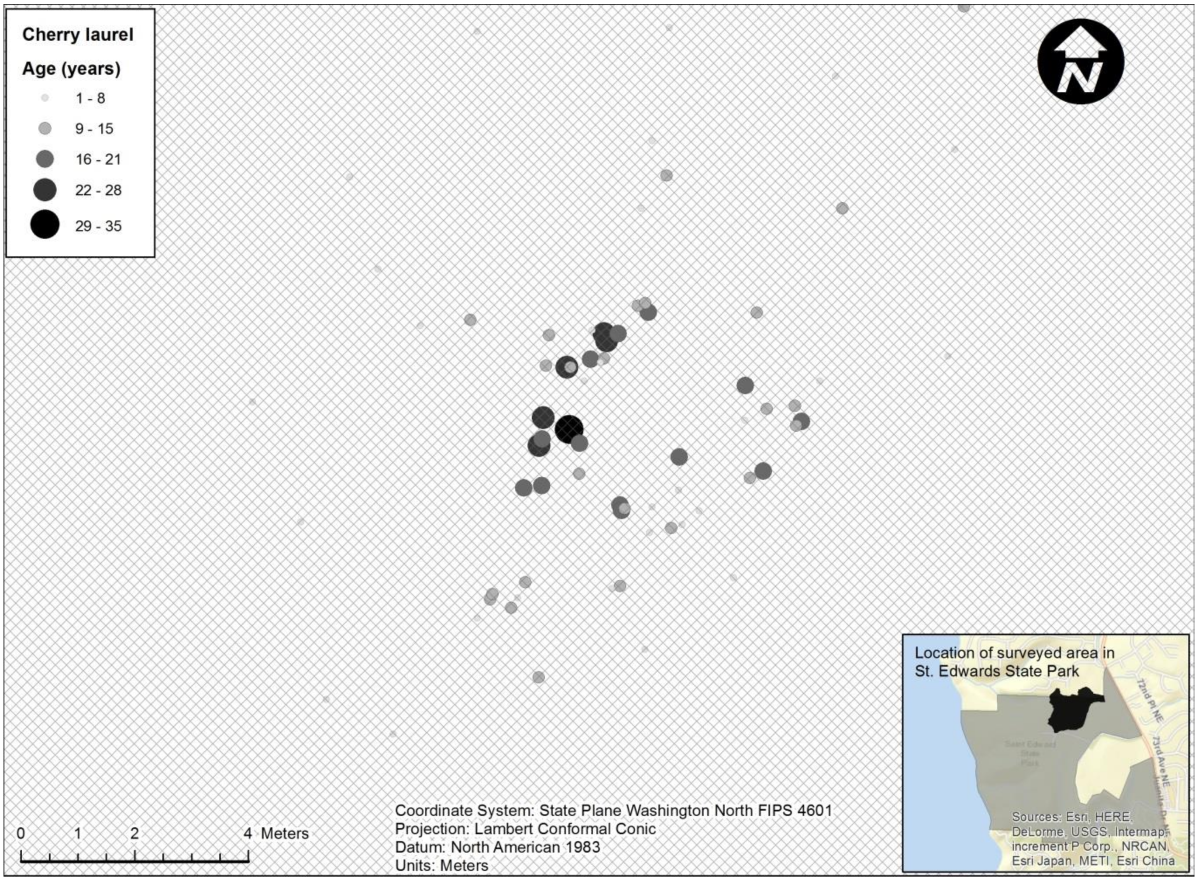

Figure 15. Close-up view of all sampled stems in largest *P. laurocerasus* clump, located in NE corner of study area (see Fig. 13). Note declining tree age with distance from largest tree.

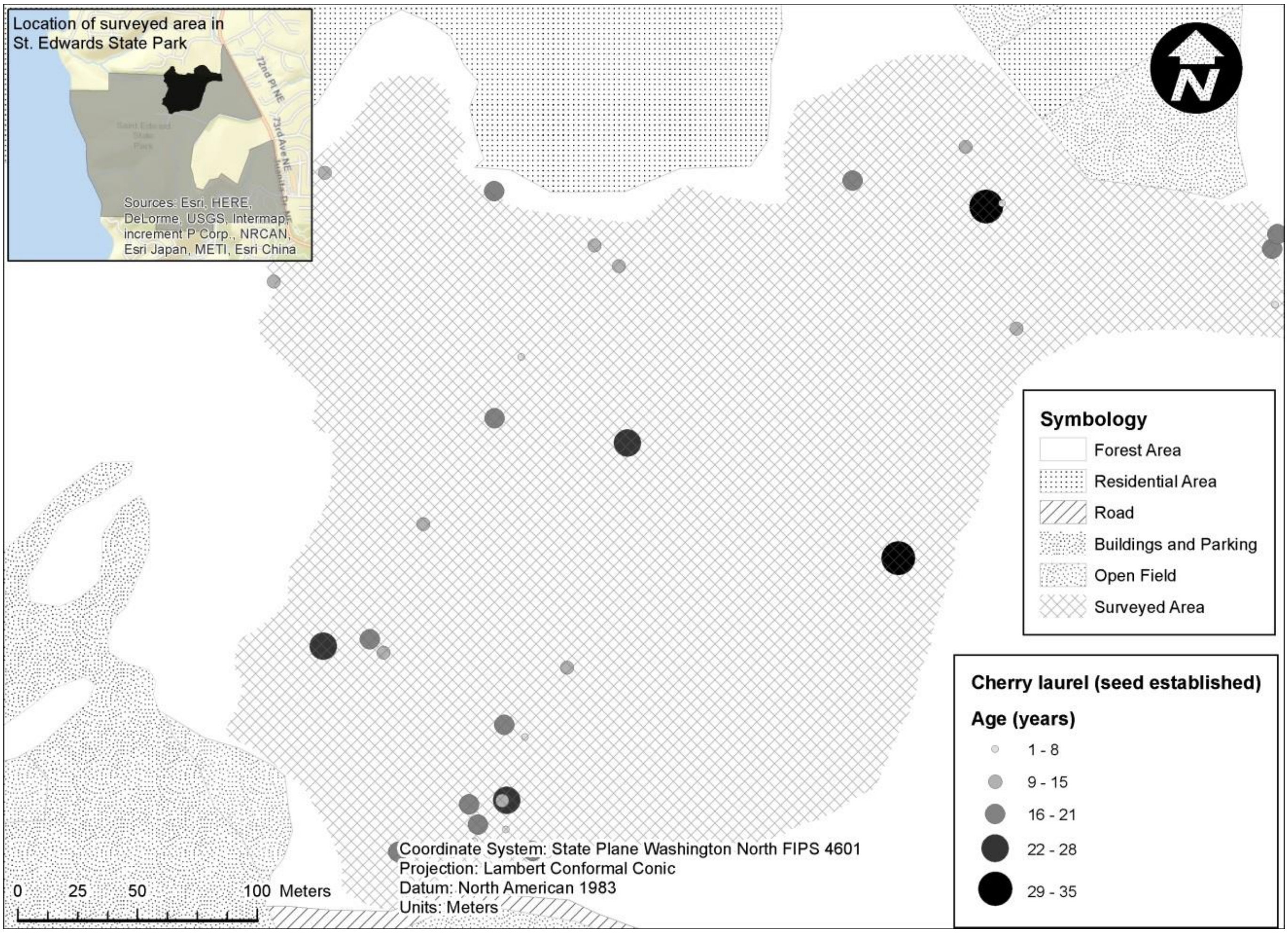

Figure 16. Location and age of seed-originated *P. laurocerasus* (n = 32) in the St. Edward Park study area.

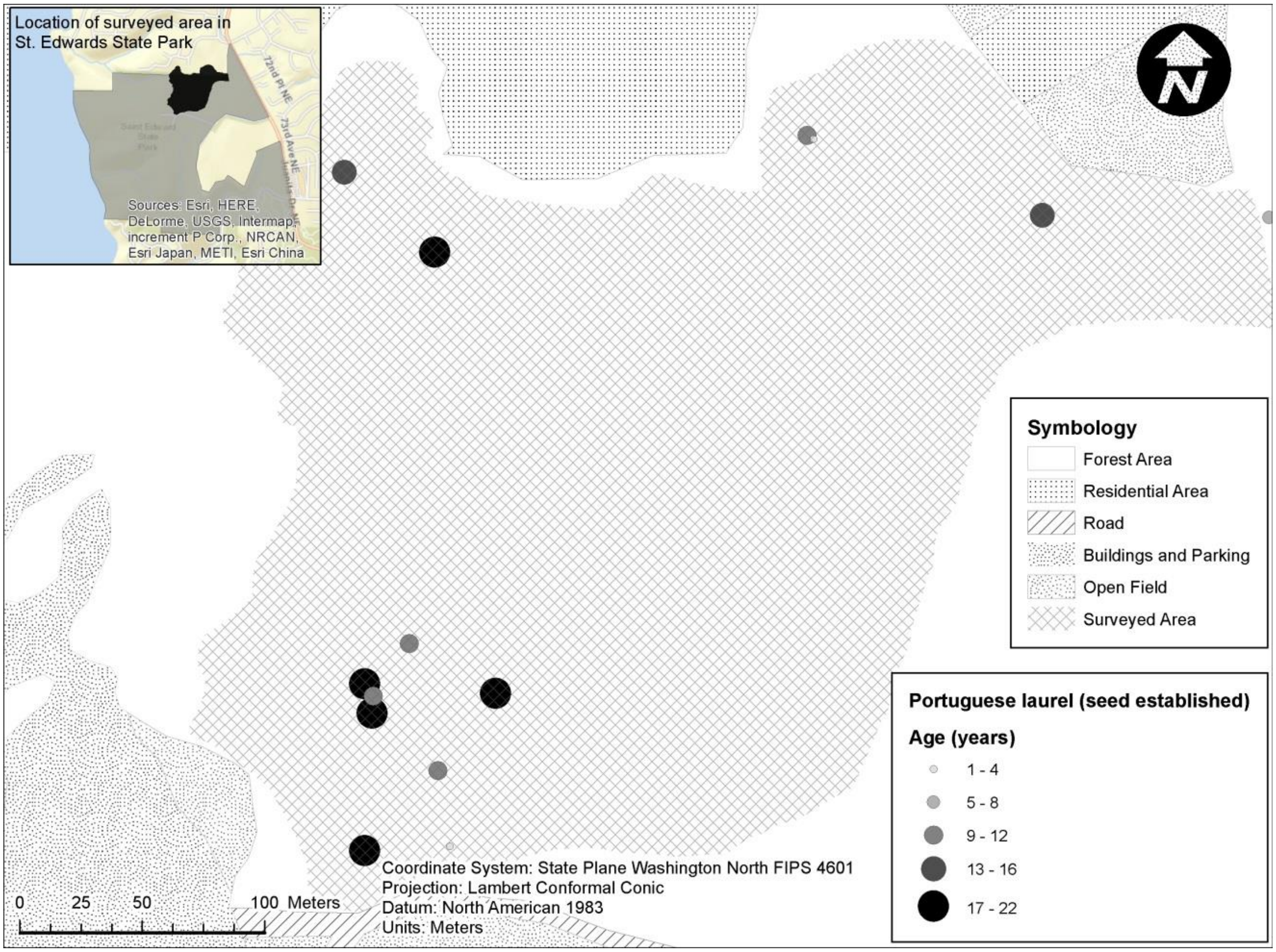

Figure 17. Location and age of seed-originated *P. lusitanica* (n = 14) in the St. Edward Park study area.

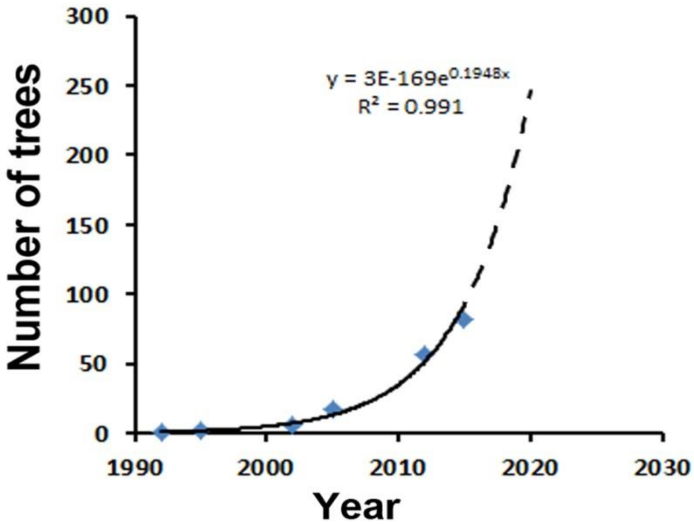

Figure 18a. Number of *P. laurocerasus* $\geq$ 10 years old in St. Edward Park study area from beginning of invasion to 2015, and projected to 2020 (dashed line). Exponential curve fit ($F_{1,4} = 20.8$, $P = 0.01$) from 1990 to 2015 data has a doubling time of approximately 3 years. If current growth rate is sustained (absent removal of the 83 $\geq$ 10 year-old trees in this study), approximately 250 trees $\geq$ 10 years old would exist in the study area by 2020. Note that this assumes negligible mortality of established trees; see text and Stokes et al. 2014a.

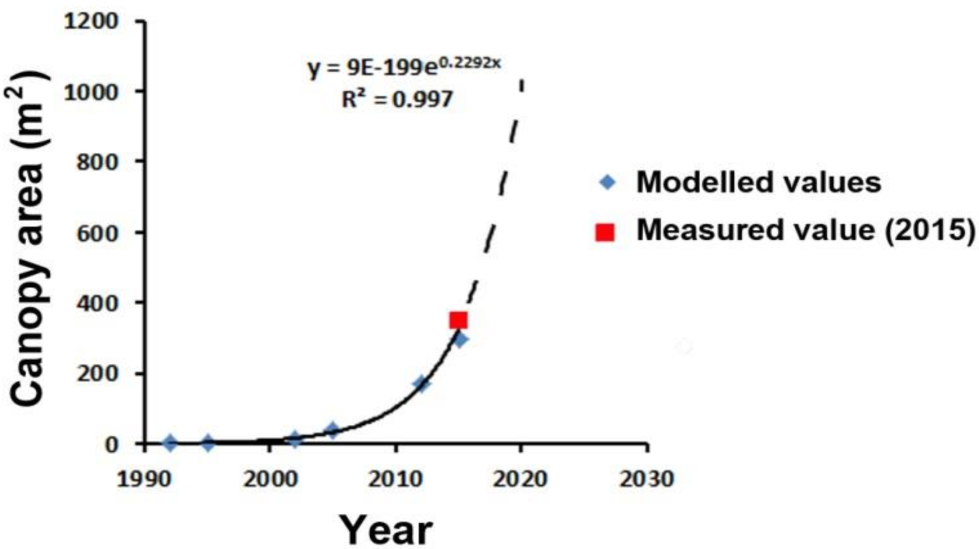

Figure 18b. Total *P. laurocerasus* canopy area (includes only trees $\geq$ 10yrs old) in St. Edward Park study area from beginning of invasion to 2015, and projected to 2020 (dashed line). Modelled values based on tree age and age-canopy area relationship (Fig. 10a). Exponential curve fit (1992 – 2015; $F_{1,4} = 1370.7$, $P = 0.001$) has a doubling time of approximately 3 years. Assumes negligible mortality of established trees; see text and Stokes et al. 2014a.

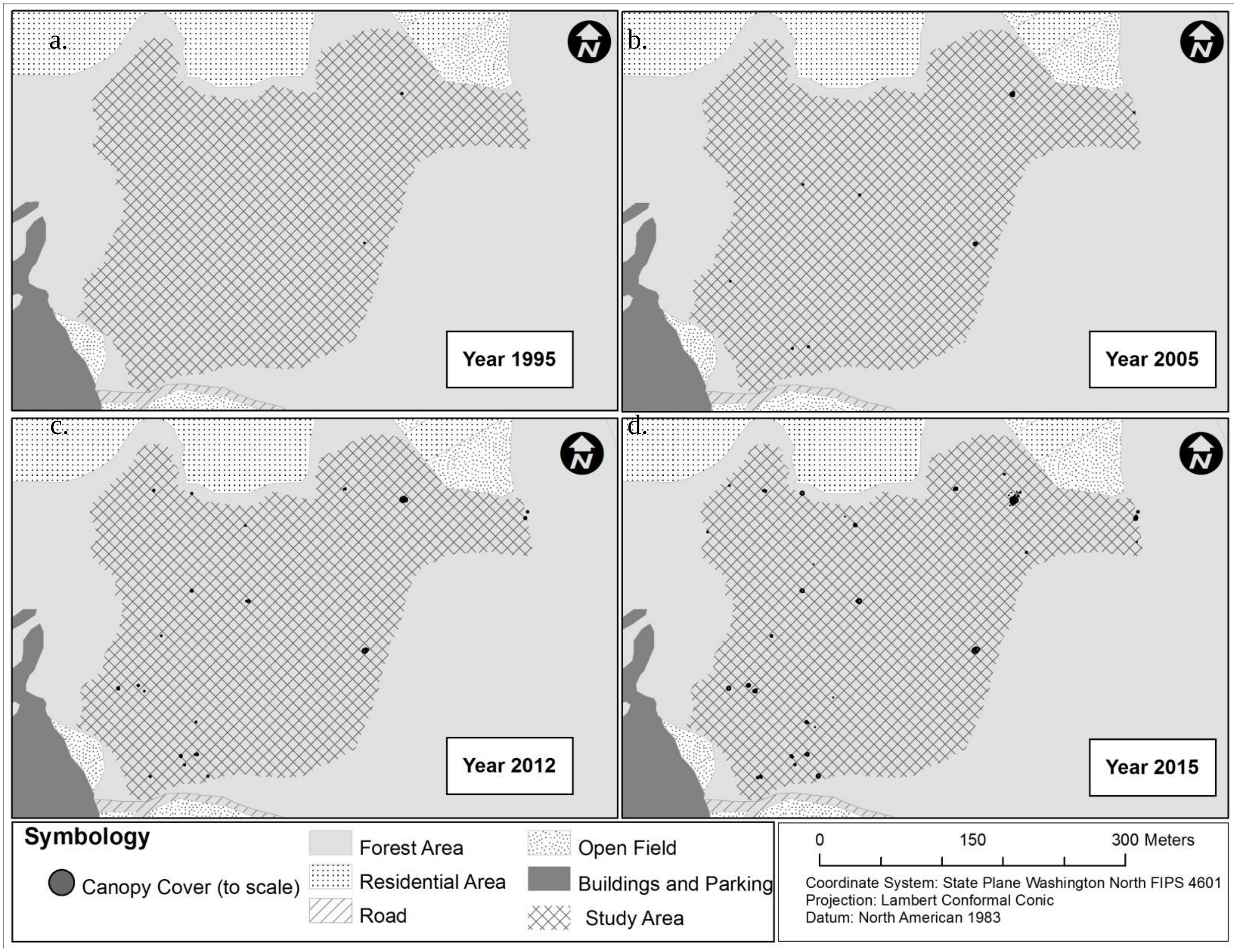

Figure 19a-d. Past and current spatial extent of *P. laurocerasus* in St. Edward study area in four years: 1995, 2005, 2012, and 2015. Canopy depicted at actual size. Past canopy area modeled from canopy radius – age relationship (see Fig. 10a).

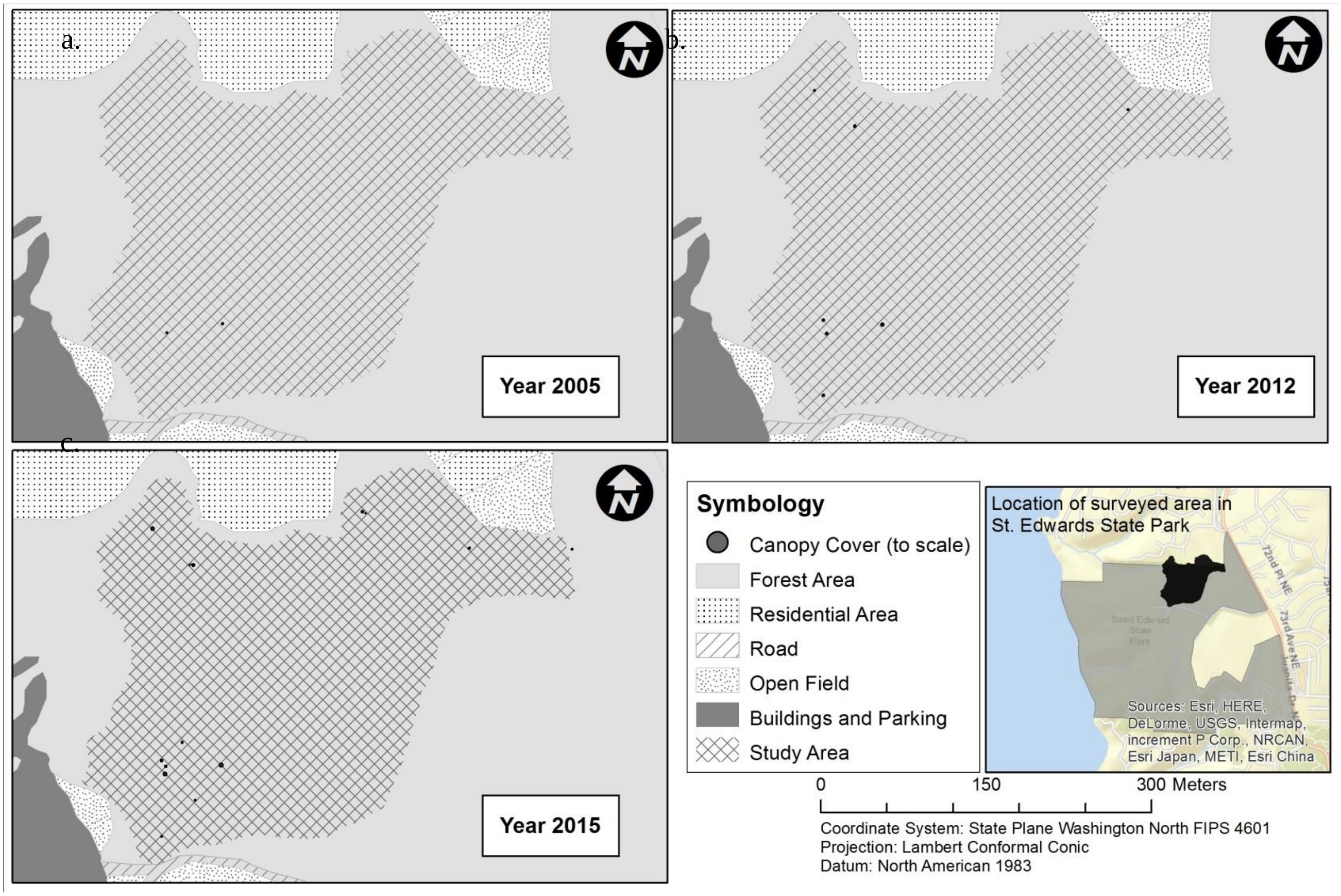

Figure 20a-c. Past and current spatial extent of *P. lusitanica* in St. Edward study area in three years: 2005, 2012, and 2015. Canopy depicted at actual size. Past canopy area modeled from canopy radius – age relationship (see Fig. 10b).

Estimated labor requirements for removal of Prunus *and* Ilex *from the Park*

Our previous study of *Ilex* indicated that approximately 630 person hours (2.3 hrs per acre) plus 49 hours (0.18 hrs/acre) of chainsaw assistance would be required to remove all *Ilex* from the forest at St. Edward State Park (Stokes et al. 2014b). Experience removing *Prunus*, indicates that it is similar to *Ilex* in removal time and effort per tree. As the population of *Prunus* in the study area was approximately half the population of *Ilex* when this estimate was made, we infer that it would take approximately 315 person hours plus approximately 25 hours of chainsaw assistance to remove the all the *Prunus* in the park. Given that the *Prunus* at St. Edward are, on average, smaller than the *Ilex*, it is likely that the actual labor requirement to remove *Prunus* may be somewhat lower than this estimate. Thus, the total resource need for removal of all three invasive tree species from St. Edward is on the order of 850 – 950 person hours, plus approximately 60-75 person-hours of chainsaw assistance.

This is a preliminary approximation that can be more precisely quantified with additional sampling, particularly in forest types and locations not sampled in the current study. Effectiveness and permanence of different removal methods should also be determined to accurately assess resource needs for *Prunus* control over the long term. Because there is a low and quickly reached size threshold (~ 7 cm basal stem diameter) beyond which all of these species usually cannot be uprooted, and because of the apparent rapid growth of these populations, the number of invasives which must be removed by the more labor-intensive methods of chainsawing and herbicide application rapidly increases the longer the invasion proceeds unchecked. Thus, control action sooner rather than later will be advantageous from an economic efficiency perspective.

CONCLUSIONS

P. laurocerasus and *P. lusitanica* have become naturalized and are proliferating in the forest of St. Edward State Park. Our results suggest that since initial establishment in our study area 35 and 21 years ago respectively, both species have quickly increased in numbers and area occupied through both seed and vegetative spread. Our analysis suggests that if current rates of increase continue, *P. laurocerasus* could be a major component of the park's forest in less than 15 years. *P. lusitanica* could also become a prominent species within a few decades, although a larger sample is needed to predict future trends with precision. Given the effects these *Prunus* species appear to have on native vegetation, their projected increase would likely have substantial negative impacts on the park's forest, native plant diversity, and perhaps habitat value.

The earlier a biological invasion is addressed, the greater the likelihood of successful control (Rejmanek 2000). Although it is widespread, *Prunus* in St. Edward State Park appears to still be amenable to control. However, our projections suggest that delaying control for even a few years will result in a substantial increase in number of *Prunus* trees. Moreover, because the *P. laurocerasus,* and likely the *P. lusitanica* are at an age of rapid and accelerating rate of size increase, increasing canopy area and difficulty of removal are expected to compound the negative effects and management challenges presented by the increase in numbers. An immediate management response is recommended, along with a significant research effort that can inform *Prunus* management.

Management recommendations

Designation of P. laurocerasus *and* P. lusitanica *as noxious weeds*

Our results clearly demonstrate the invasive character of *P. laurocerasus* and *P. lusitanica* in mid-successional Pacific Northwest forests, a dominant plant community in low-elevation western Washington. Formal noxious weed designation (e.g., listing as a Class C noxious weed by the Washington State Noxious Weed Control Board) would assist in controlling these species by raising awareness of their invasive character and expanding management options and regulatory control. *Ilex aquifolium* should also be Class C listed (Stokes et al. 2014a).

Immediate development and implementation of a plan for control of Prunus *and* Ilex *in Pacific Northwest west-side wildland forests*

In Northwest forests managed for natural ecosystem processes and native biodiversity, elimination of *P. laurocerasus* and *P. lusitanica* is an appropriate management goal. While widespread, these species are still at a level that is amenable to control, at least in some forests (e.g., St. Edward State Park). Our projections suggest that delaying action for just a few years will result in large numbers of additional *Prunus* trees and added difficulty of removal due to increasing tree size. A management plan should be developed and control measures undertaken while control is still feasible. Control of both *Prunus* species could be combined with *Ilex* control, as all three species occur in the same areas and are controlled by similar methods. At present, with a significant but not unrealistic investment of resources (< 950 person hours), all three major invasive tree species could be eliminated over the entirety of St. Edward Park. In larger protected areas where complete elimination is not practical, control efforts should focus on large seed-bearing trees and the areas most susceptible to invasion. Spatially explicit modeling of invasion risk could help identify invasion-prone areas (Lopez and Stokes 2016). It is likely that a control plan will require modification as more information about the *Prunus* invasion becomes available.

Invasive control on adjacent lands

As the original source of *Prunus* and *Ilex* in wildlands is seed from nearby human-dominated landscapes, and these landscapes are likely to be a continuing seed source, reduction of that seed source would assist in slowing the invasion (Reichard and White 2001). Opportunities for reducing the prevalence of berry-producing *Prunus* and *Ilex* on human-occupied lands near wildlands (e.g., residential neighborhoods), and education of the local public about the invasiveness of *Prunus* and *Ilex* should be pursued. The graphical and map products of this study (e.g., Figs. 18-20; see also figures in Stokes et al. 2014b) can be adapted by land management agencies to communicate the threat posed by these and other invasive species to the public (Mack et al. 2000).

Additional research on the invasion ecology of invasive non-native trees

Additional research is needed to better understand the *Prunus* and *Ilex* invasions and to develop effective management responses. Topics in need of investigation include the vulnerability to invasion of different forest locations and types, particularly mature forest; effects of these invaders on native species; long-term impacts of these invaders on upland, riparian, and in-stream habitats; effectiveness of various control methods; the effects of land management practices, including forest management practices, on spread of these invasives; and likely responses of these invaders to predicted climate change.

Further research needed

Additional data collection to increase the range of plant communities sampled

Additional field data will allow us to investigate questions such as whether some of St. Edward Park's forest types are more invaded than others (e.g., deciduous versus evergreen forest and whether some native plant communities and species are more negatively affected than others. Sampling of conifer-dominated forest (i.e., Douglas-fir-Western Hemlock forest (PSME-TSHE or TSHE-PSME Chappell 2006) in park polygons 4, 5, 7, 11, 16, 21, or 22; Smith 2006) could be compared to our current *Prunus* sample from more deciduous-dominated forest. This field work would also result in elimination of *Prunus* over an additional portion of the park's forest.

Effectiveness of removal techniques

A critical need for management of invasive *Prunus spp.* is to determine the effectiveness of removal methods and the degree of permanence achieved. We will revisit the 2015 study area to determine the effectiveness of pulling and cutting with glyphosate treatment of the stump. Additional questions that should be pursued are: What herbicide is most effective for controlling *Prunus*? Also, does stacking of removed *Prunus* result in new sprouting? And once controlled, how likely is re-invasion?

Improvement of predictive population and cover models with more data

Additional sampling data and analysis of that data can produce a more robust dispersion model of *Prunus* population and canopy increase, both past and future. Specific topics needing more work include determination of actual *Prunus* mortality rates, which can be refined with a larger sample, and investigation of possible spatial and density dependent factors that could limit *Prunus* spread. An improved model will give a more accurate picture of the future threat posed by *Prunus* and the resources necessary for control. We also plan to develop a combined spatial spread model for *Prunus* and *Ilex* to understand combined effects and spatial spread of the invasions.

Effects of woody invasives on other species

Despite a general recognition of negative effects of *Ilex* among land managers, the degree to which it affects and excludes native species is not well known or documented. Even less is known about the effects of invasive *Prunus.* The observations of negative effects presented here, and (regarding *Ilex*) in Stokes et al. (2014a & b) and Church and Stokes (in prep.), should be validated with additional study, with inferences drawn about impacts on forest structure and function.

In addition, better understanding of the mechanisms of the negative effects may be useful in minimizing impacts on native species. In addition to shade and leaf-litter, two additional potential mechanisms of negative impact, water competition and allelopathy, merit investigation. Consumption of soil water by *Ilex* and *Prunus* may have a negative impact on other plants, particularly during dry periods such as those characteristic of summers in the Pacific Northwest. Allelopathy may also contribute to the dramatic decrease in native vegetation present underneath these invasives.

Ecophysiology of Prunus and Ilex

Better understanding of the physiological aspects of *Prunus* and *Ilex* germination, growth, and spread is needed to understand these species' invasiveness and perhaps gain insights into potential means of control. Understanding the physiological function of *Prunus* and *Ilex* clumps may also be important in this regard.

Effects of Prunus and Ilex on riparian and in-stream habitats

Little is known about the effects of these invasives on riparian habitat. In the context of biodiversity conservation and land management, this could be one of the most important dimensions of these invasions, as several federally listed salmonid species occur in areas where these species are invading. By occupying locations that would otherwise be sites of establishment for larger native tree species, these non-native plants may be degrading riparian and in-stream habitat for salmonids. Systematic investigation of the current and potential future impacts of woody invasives on riparian vegetation and in-stream habitat, is needed.

Influence of predicted climate change on proliferation and spread of woody invasives

Evidence from Europe suggests that *Prunus* invasion may be facilitated by environmental changes associated with current and future climate change (Walther 1999, Walther and Grundman 2001, Hättenschwiler and Körner 2003). However no work has been done on this question in North America for *Prunus* or *Ilex*. We can use our records of yearly tree establishments and seed-sourced establishment of these species, as well as our samples of stem cross sections of older trees (161 *Ilex* and 83 *Prunus*) to look for correlations with weather patterns that can shed light on this question. Such research will inform future management of Northwest wildlands as climate change proceeds.

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APPENDIX A

Peer-reviewed publications from our work addressing invasive woody plants in Saint Edward State Park:

Lopez, S., and D.L. Stokes. 2016. Modeling the invasion of Ilex (*Ilex aquifolium*): Spatial relationships and spread trajectories. *Professional Geographer* 68: 399-413.

Stokes, D.L., Church, E.D., Cronkright, D.M., and S. Lopez. 2014. Pictures of an invasion: Ilex (*Ilex aquifolium*) invasion of a Pacific Northwest forest. *Northwest Science,* vol. 88:75-93.