Survival and growth of planted seedlings of three native tree species in urban forest restoration in Wellington, New Zealand

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Abstract: Planting container-grown native seedlings is a restoration technique widely used to enhance biodiversity, including in urban areas in New Zealand. We measured survival and growth of seedlings of three native tree species (*Aristotelia serrata*, wineberry ($n = 743$); *Cordyline australis*, cabbage tree ($n = 666$); and *Pittosporum eugenioides*, lemonwood ($n = 701$)) planted in 11 forest restoration sites in Wellington, New Zealand, between 2008 and 2012. We used mixed-effect models to identify the relative importance of a range of variables affecting mortality in the early years after planting. Seedling mortality during the first year was 38%, 20% and 11% for *A. serrata*, *C. australis* and *P. eugenioides*, respectively. Although variables explaining mortality were species-specific, initial seedling height and the height of grass surrounding each seedling were present in the best ranked models explaining mortality in all three species. Contrary to expectations, trees were still at substantial risk of mortality ≥2 years after planting. Plants in forest restoration projects managed by community groups experienced similar mortality as those managed by local government. Our results suggest that reduction of cover of surrounding grasses is likely to enhance seedling survival and indicate that initial plant height, rather than time since planting, is the best predictor of the likelihood of mortality of planted seedlings.

Keywords: afforestation; cabbage tree; ecological restoration; lemonwood; saplings; tree mortality; wineberry

Introduction

Over the last decades, afforestation projects have been widely used in attempts to restore degraded habitats (e.g. wasteland, old landfills, or roadsides) in urban areas (Sullivan et al. 2009; Pincetl et al. 2013). Restoring forest habitats in urban areas provides numerous benefits, including the mitigation of soil erosion, increases in property values, and new resources and improved connectivity for wildlife (Pavao-Zuckerman 2008; Oldfield et al. 2013; Alvarado et al. 2014). Most restoration practitioners, including those in New Zealand, use native species, mostly planting container-grown seedlings, to facilitate restoration of forest habitats (Smaill et al. 2011). However, survival rates and the factors influencing mortality are unaccounted for in many restoration projects (Sullivan et al. 2009).

Compared with direct seeding or assisting natural regeneration, planting container-grown seedlings (e.g. one-year-old plants) may reduce the time required to achieve canopy closure (Porteous 1993; Bergin & Gea 2007). Achieving canopy closure is desired in most restoration projects because it suppresses the biomass of some weeds (Sullivan et al. 2009). Furthermore, planting seedlings offers opportunities for community involvement, which has been shown to increase environmental awareness, reinforce a sense of national identity, and improve the health and well-being of the volunteers (Meurk & Swaffield 2000; Dunn et al. 2006; Husk et al. 2013).

There are, however, drawbacks associated with using container-grown seedlings for afforestation. For example, planting potted seedlings raised in nurseries is more expensive and labour intensive than direct seeding or assisting natural regeneration (Porteous 1993; Douglas et al. 2007). Another inevitable drawback is the time required by seedlings to acclimatise to their new environment (e.g. transplanting shock; Burdett 1990). During acclimatisation to site-specific conditions, seedlings experience stress and are vulnerable to unfavourable environmental conditions (e.g. disease, weather extremes), potentially resulting in high mortality.

Understanding plant mortality is a complex process, and is highly context-dependent and species-specific (Holzwarth et al. 2013). Mortality of seedlings, however, generally decreases over time and can become negligible in as little as 2 years after planting (Graham & Filmer 2007; Ledgard & Henley 2009). Opportunities thus exist to implement short-term intensive management aimed at improving initial seedling survival without increasing the already substantial direct and indirect costs associated with urban forest restoration (Douglas et al. 2007).

Aristotelia serrata (Elaeocarpaceae) or makomako/wineberry, *Cordyline australis* (Asparagaceae) or tī kouka/cabbage tree, and *Pittosporum eugenioides* (Pittosporaceae) or tarata/lemonwood are among the fastest growing native trees in New Zealand (Busell 1968; Marden et al. 2005) and are therefore commonly used in ecological restoration projects (Mitcalfe 2002). We monitored survival and growth of 2110 individuals of these three species planted between 2008 and 2012. For one year, we monitored individually identifiable plants, including seedlings planted in 2012 and young trees planted up to 4 years beforehand. Our specific objectives were: (1) to identify the relative importance of variables affecting plant mortality; (2) to identify the most vulnerable period for seedlings after planting; (3) to understand possible differences in plant survival between forest restoration carried out by local governments or community groups; and (4) to measure growth rates of seedlings over time. Such information is necessary to help agencies managing urban forest restoration projects improve their planting and management strategies and thus reduce costs.

Methods

Study sites

Over the course of one year, we monitored plant survival and growth in 11 forest restoration sites within Wellington, New Zealand (Fig. 1 and Appendix S1). Elevation at these sites ranged between 50 and 200 m above sea level. The areas of the sites were between 1 and 10 ha and the sites differed in topography and previous land use (e.g. former landfills or pasture). Landfills were closed by adding layers of soil (including fine-grained, coarse-grained and highly organic soils), followed by sowing of vegetation (mostly turf grass but occasionally some native shrubs). Restoration projects were carried out by the Wellington City Council (WCC, n = 6) or local community groups (n = 5). During the monitoring period (May 2012 to September 2013), annual temperatures averaged 12.9°C, monthly rainfall averaged 98.2 mm, and the strongest recorded wind gust was 140.8 km h^{-1} (NIWA 2013), values representative for the period from 2008 to 2012. However, of the 224 mm total rainfall during January–March 2013, 93% occurred as three independent events, and this period was also characterised by high sunshine hours (760 h), which led to unusual, short-term drought conditions (Turner 2013).

We monitored the growth and survival of the cohorts of seedlings of *A. serrata*, *C. australis*, and *P. eugenioides* planted between 2008 and 2012. In any given year, planting of all cohorts occurred between May and September and never exceeded densities of 2 plants m^{-2}. Planting techniques were similar across all sites and planting years. Seedlings were planted into sites where grass was the main vegetation cover (mostly non-native species; composition at sites and around planted seedlings not quantified). Sites were mown 2 weeks before planting, but not around seedlings thereafter, such that grass cover around some seedlings was up to 1.5 m tall by the end of the study. Wellington City Council's Berhampore Nursery propagated and grew all seedlings in PB3 containers containing 1.7 litres of potting soil. The height of the seedlings when planted out at the restoration sites ranged between 40 and 80 cm.

Growth and survival of planted seedlings

We monitored newly planted seedlings for a year after planting; planting dates differed among sites (22 May 2012 to 9 September 2013; Appendix S1). On the initial monitoring day, plants were randomly selected, individually marked with aluminium tags, and their location recorded on topographical maps. We also recorded whether or not seedlings were appropriately planted (e.g. firmly planted into the ground). Using detailed maps provided by WCC, we located trees planted between 1 and 4 years earlier (there were no 4-year-
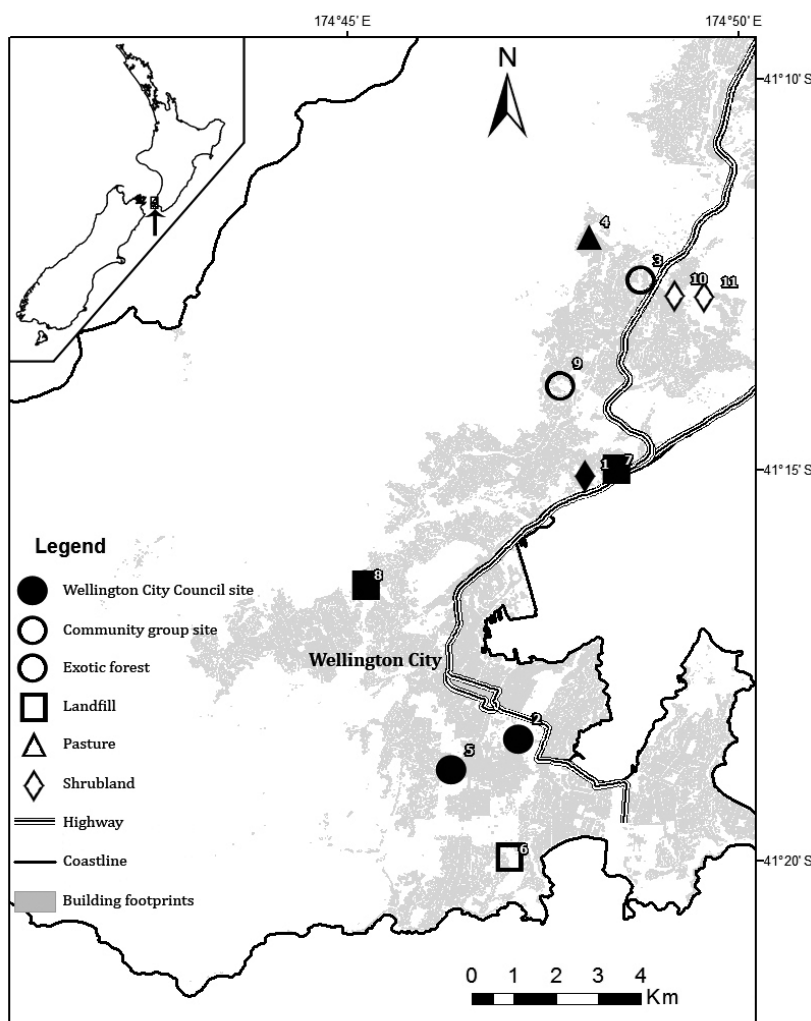

Figure 1. Location of the 11 forest restoration sites in Wellington City, New Zealand. Symbols represent the previous land use of the sites. Filled symbols indicate sites managed by Wellington City Council; unfilled symbols indicate sites managed by community groups.

old *P. eugenioides*), and tagged them as described above. We tagged up to 30 trees per species, per planting year, and per site (Appendix S1). In total, we monitored 2110 plants.

We assessed plant mortality 6 and 12 months later at each site and attempted to identify the cause of mortality. Plants with no leaves during investigations were classified as 'standing dead', following the convention of Holzwarth et al. (2013). Plants ripped out of the ground, probably by high wind gusts, were considered 'uprooted'. Plants found cut after mowing were recorded as 'mown'. Plants that could not be re-located were recorded as 'missing' and considered dead. The remaining plants were considered alive.

Variables affecting mortality

We recorded a range of variables at each site, or about each planted seedling, that could affect seedling mortality (Table 1). Plant age was determined from the difference between planting date and the initial site-specific monitoring date. Plant height was measured as the distance from the ground to the highest green leaf. Slope and aspect were measured with a clinometer and compass, respectively. We transformed aspect values to values ranging from 0 to 2 using the transformation described in Beers et al. (1966). North-eastern slopes, commonly exposed to the sun, had a value of 2, north-western and south-eastern slopes had a value of 1, and south-western slopes had a value of 0.

The amount of canopy cover was measured using a convex densiometer with 24 squares by counting the number of squares shaded by overhead vegetation. Holding the device directly above the plant, we estimated canopy cover facing all four compass directions and reported the overall mean. We assessed soil conditions according to three variables. First, we used a proving ring penetrometer to assess soil compaction from the force (penetration resistance) required to penetrate the device to a soil depth of 15 cm (kg cm^{-2}). We then excavated to 15 cm depth and collected 5 g of soil from each of nine randomly selected points per site and from the bulked sample per site estimated phosphorus (P) concentration using the bicarbonate extraction method (Olsen et al. 1954) and total nitrogen (N) concentration using the Dumas dry combustion principle (Leco Corp., St. Joseph, MI; Landcare Research Laboratory in Palmerston North). During each site visit (n = 5), we categorically estimated grass cover in a 1.5 × 1.5

m quadrat surrounding each plant as 0, 0–20, 20–40, 40–60, 60–80 or >80 percent-cover. We calculated the mean of grass cover for each plant over time by averaging the midpoints of each recorded interval. We also recorded if grass height in the quadrat was ever higher or lower than the maximum height of the planted seedling. We estimated relative rodent abundance at each site based on three tracking tunnels set at least 50 m apart (Porteous 1993). Tracking tunnels baited with peanut butter were set for one night in September 2012 and March 2013 coinciding with expected rodent annual population minima and maxima (Ogden & Gilbert 2009). We measured rabbit abundance according to the Modified McLean Scale (National Pest Control Agencies 2012). We recorded whether or not mulch had been applied to the planted seedlings at each measurement. Mulch was defined as a 3-cm layer of wood chips, mainly of *Pinus* spp., covering the ground around each seedling. Finally, we also recorded whether the restoration at that site was carried out by the local government agency or a community group.

Data analysis

We used generalised mixed-effect models with a logit-link function to analyse mortality of *A. serrata*, *C. australis* and *P. eugenioides* on a per individual basis. Due to potentially unrecorded differences among sites, we treated the variable 'site' as a random effect. We standardised all independent variables to a mean of zero and standard deviation of 0.5 because variables were measured in units with different scales (Schielzeth 2010). In order to determine how much variance of the recorded mortality is explained by the models, we generated the conditional (R^2_c) and marginal (R^2_m) coefficients of determination (Nakagawa & Schielzeth 2013).

We checked that variables were not highly correlated (e.g. $r \geq 0.7$) using Spearman correlation coefficients (Booth et al. 1994). We excluded from the models those plants that were accidentally mown or inadequately planted, due to the small percentage (<1%) of plants of each species affected.

We used a model inference approach based on Akaike Information Criterion (AIC) to identify the relative importance of variables affecting plant mortality. We compared all possible models, also known as an all-subset approach, because of

Table 1. Potential variables involved in the mortality of seedlings of three species (*Aristotelia serrata, Cordyline australis, Pittosporum eugenioides*) transplanted in Wellington City, measurement units, measurement scale and references to support each candidate variable.

Variable	Units	Scale	Reference
Seedling age	Years	Individual	Graham & Filmer (2007)
Seedling height	cm	Individual	Holzwarth et al. (2013)
Slope	Degrees	Individual	Larcher (1975)
Aspect	Degrees	Individual	Larcher (1975)
Canopy cover	Number of shaded squares	Individual	Bergin & Gea (2007)
Soil compaction	kg cm^{-2}	Site	Sullivan et al. (2009)
Total N	%	Site	Larcher (1975)
P content	mg P kg^{-1} soil	Site	Larcher (1975)
Grass cover	%	Individual	Sullivan et al. (2009)
Grass height	Higher or lower than the top of the seedling	Individual	Smaill et al. (2011)
Rabbits	Modified McLean Scale	Site	Camara (2011)
Rodents	Proportion of tracking tunnels	Site	Campbell (1978)
Mulch	Yes or no	Individual	Bergin & Gea (2007)

the exploratory nature of our study (Grueber et al. 2011) and the complexity of processes involved in studies of plant mortality (Holzwarth et al. 2013). For each possible model, we generated the AIC, the difference in AIC relative to the model best supported by available data (ΔAIC), and Akaike weights (AICw). All models with a ΔAIC < 4 were considered supported by the data (Burnham & Anderson 2002). To assess the relative importance of each explanatory variable (RVI), we summed AICw from all models supported by the data containing variables of interest (Bartoń 2009).

We fitted separate linear regression models for each age cohort to test whether relative seedling growth rates differed among species. Relative growth rate (R_h: cm cm^{-1} yr^{-1}) of seedlings was calculated following the convention of Alvarez-Aquino et al. (2004) ($R_h = (\log_e H_f - \log_e H_i)/t_f - t_i$, where H_1 and H_2 are the heights of the seedling at initial planting (t_i) and upon remeasurement one year later (t_f), respectively. In this study, time was measured in years hence ($t_f - t_i$) = 1). We used post hoc multiple comparison analysis (Tukey's Honest Significant Differences test) to assess differences in the R_h among species.

All statistical analyses were conducted in Program R (R Development Core Team 2012), using the lme4 (Bates et al. 2010) and MuMIn (Bartoń 2009) packages. Smoothing splines were used to assist with graphical interpretation of the relationship between height and mortality (Bates et al. 2010).

Results

Mortality of newly planted seedlings 12 months after planting was 38%, 20%, and 11% for *A. serrata*, *C. australis*, and *P. eugenioides*, respectively (see Fig. 2 for mortality across the different ages of plants). For all three species, mortality was greater during the second monitoring period (months 7–12) (Table 2). Most of the recorded plant mortality was determined from either a lack of 'live' leaves or the plant disappearing completely (56.9% and 32.6% of total mortality, respectively). Accidentally mown and inadequately planted trees resulted in 4% of the total mortality (Table 2).

Relative importance of the variables in plant mortality

Before we generated models to explain plant mortality, we omitted rabbit abundance as a variable since there was no evidence of rabbits at any of the sites.

The relative importance of variables in plant mortality was species-specific (Table 3). However, for all three species, the

Table 2. Specific causes and plant mortality (*n*) for each of the three tree species studied in urban restoration projects, accumulated 6 and 12 months after the initial monitoring date. Trees varied from newly planted to 4 years post-planting at time of first monitoring.

Causes of plant death	*Aristotelia serrata*		*Cordyline australis*		*Pittosporum eugenioides*	
	6	12	6	12	6	12
Standing dead	40	127	21	38	8	29
Uprooted	2	4	3	3	2	2
Missing	12	39	9	58	3	14
Mown	1	1	1	3	0	3
Planted inappropriately	1	1	5	5	1	1
Mortality[1] (%)	7.5	23.1	5.9	16.1	2.0	6.9
Total marked	743		666		701	

[1]Mortality calculated per 6-month period and per 12-month period, respectively (i.e. the 12-month total mortality includes the 6-month total).
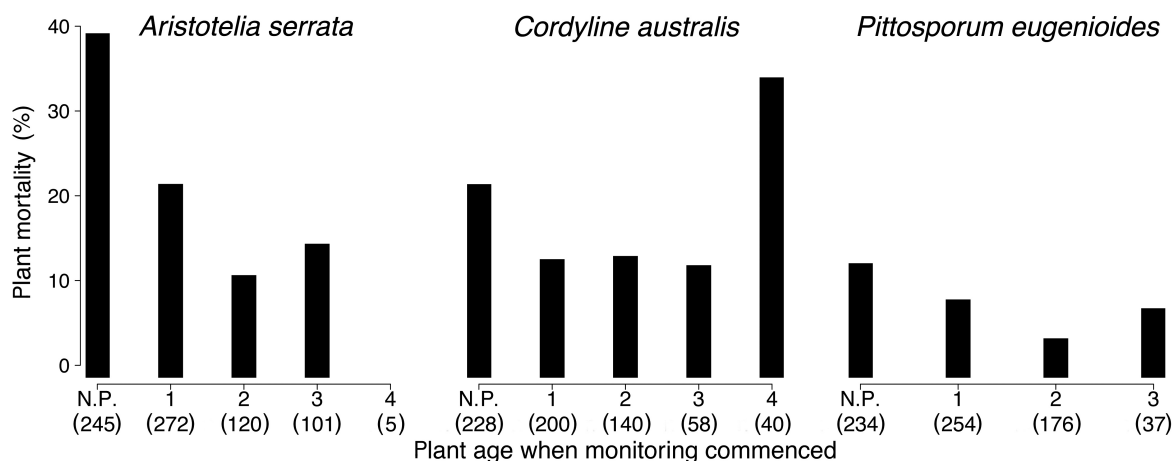

Figure 2. Plant mortality per species 12 months after the initial monitoring date of different aged plants (newly planted (N.P.) to 4 years old). Sample size (*n*) is indicated under each age cohort.

height of the planted seedling and grass cover were common variables in the best models explaining mortality. The initial height of the plant was overall the most important variable for the three species (>0.96 Relative Variable Importance). The probability of mortality decreased as initial plant height increased (Fig. 3).

Plants that were shorter than the surrounding grass experienced greater mortality than taller ones. Information related to height of the grass was included in the optimum model explaining mortality of all species. While grass cover increased plant mortality in all species, spreading a layer of mulch reduced the mortality. However, these two variables were more relevant in the mortality of *A. serrata* and *C. australis* than of *P. eugenioides* (see Table 3).

The relationship between mortality and age was only important for predicting mortality of *C. australis* (Table 3). Mortality generally decreased with age; however, there was a mortality peak in 4-year-old *C. australis* (Fig. 2) indicating that this relationship was not linear.

The effect of other variables varied among species. Soil compaction was included in the best models explaining mortality of *A. serrata*. Mortality of *A. serrata* increased when they were planted in highly compacted soils. Slope and canopy cover were included in the best models explaining mortality of *C. australis*. Mortality of this species increased when planted on less shaded and steeper slopes.

Aspect, rodent abundance, and P concentrations in soil were not included in the best models explaining mortality of any of the three species and N in soil was only included for *A. serrata*. While trees planted and managed by community groups trended lower in mortality than those managed by WCC (12% vs 18% per annum), the agency managing a restoration site was also not included in the best models explaining mortality of the species (Table 3).

Table 3. Results of multi-model inference. Relative importance of each independent variable on the mortality probability of *A. serrata, C. australis* and *P. eugenioides*. R^2_m and R^2_c = marginal and conditional variance explained by the best model; #1 = indicating whether or not the variable is included in the best model; β ± SE = regression coefficients and unconditional standard errors averaged across all models of the set of <4ΔAIC models; RVI = relative importance of each variable for the set of <4ΔAIC models (* β ± 2*SE does not intersect 0).

|	*Aristotelia serrata* $R^2_m = 0.38$, $R^2_c = 0.39$			*Cordyline australis* $R^2_m = 0.20$, $R^2_c = 0.21$			*Pittosporum eugenioides* $R^2_m = 0.42$, $R^2_c = 0.43$		
	#1	β ± SE	RVI	#1	β ± SE	RVI	#1	β ± SE	RVI
Plant age		−0.02 ± 0.16	0.03	✓	−0.63 ± 0.20*	0.90		−0.71 ± 0.36	0.04
Height	✓	−0.97 ± 0.21*	1.00	✓	−0.48 ± 0.22*	0.96	✓	−1.38 ± 0.43*	1.00
Slope		0.04 ± 0.12	0.17	✓	0.32 ± 0.14*	0.95		0.10 ± 0.16	0.14
Aspect		−0.06 ± 0.10	0.19		−0.17 ± 0.13	0.48		−0.15 ± 0.17	0.19
Canopy cover		−0.06 ± 0.13	0.18	✓	−0.38 ± 0.16*	0.89		−0.11 ± 0.17	0.14
Grass cover	✓	0.41 ± 0.12*	1.00	✓	0.54 ± 0.16*	1.00		0.25 ± 0.26	0.21
Mulch	✓	−2.08 ± 0.54*	1.00	✓	−1.31 ± 0.65*	0.97		−0.95 ± 0.86	0.36
WCC/CGs		0.35 ± 0.59	0.22		−0.99 ± 0.16*	0.63		0.07 ± 0.70	0.12
Rodents		0.12 ± 0.20	0.20		0.33 ± 0.18	0.70		0.40 ± 0.33	0.31
Grass height	✓	0.73 ± 0.22*	1.00	✓	0.50 ± 0.31	1.00	✓	0.58 ± 0.40	0.55
Soil compaction	✓	0.71 ± 0.24*	0.96		0.48 ± 0.29	0.58		0.41 ± 0.33	0.32
Total N	✓	0.59 ± 0.27*	0.88		0.25 ± 0.27	0.34		−0.05 ± 0.32	0.11
P content		0.05 ± 0.17	0.19		0.07 ± 0.15	0.25		0.11 ± 0.30	0.11
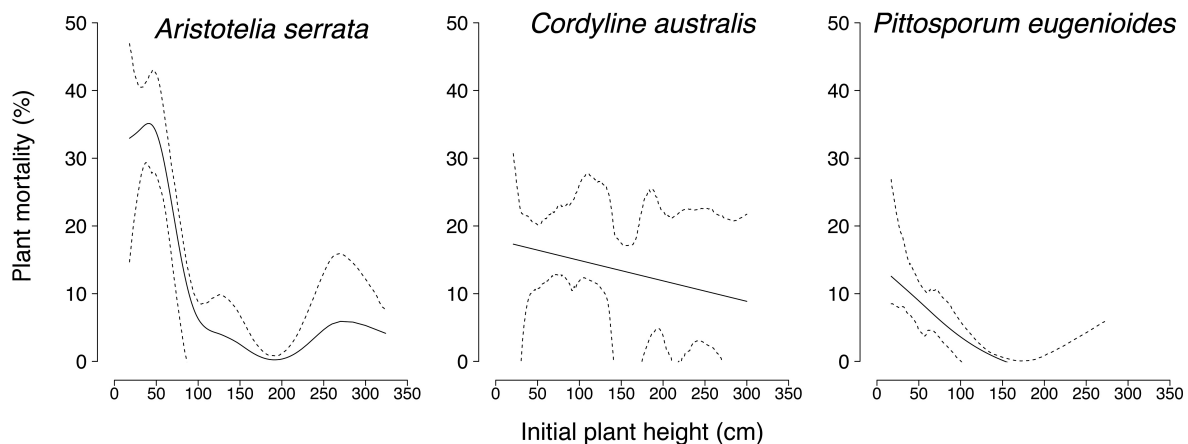
Figure 3. Cubic smooth spline of annual plant mortality versus the initial height of seedlings and trees, up to 4 years old, per species. Dashed lines show the 95% confidence interval.

Seedling growth

Seedling height growth rates were species- and age-specific (Table 4; Fig. 4). Species identity had a significant effect on the R_h of all age cohorts. Newly planted *A. serrata* had the greatest R_h (2.94 ± 0.03 cm cm^{-1} yr^{-1}). Across age cohorts, *C. australis* had the lowest R_h of the three species (Table 4).

Discussion

There were three main results from our study. First, while the set of variables that best explained mortality was species-specific, two variables (height of the planted seedling and height of the surrounding grass with respect to the plant) were consistently present in the best models of the three native species we considered. Second, contrary to our expectations, two of the three species continued to suffer ≥10% annual mortality more than 2 years following planting. Third, after accounting for environmental site differences and the use of mulch, plants in forest restoration managed by community groups had similar mortality to those managed by WCC.

Mortality observed in our study was comparable with that reported from other forest restoration projects that used container-grown seedlings in New Zealand (e.g. Sullivan et al. 2009; Smaill et al. 2011; Gillon 2014) and elsewhere (Alvarez-Aquino et al. 2004). Most mortality occurred during summer and autumn (between 6 and 12 months after the initial monitoring date) when Wellington experienced unusual drought

Table 4. Results of the age-specific linear regression models testing for a generic effect of species on relative growth rate (R_h), the mean relative growth rates (cm cm^{-1} yr^{-1}) and Tukey's HSD post hoc comparison tests between *Aristotelia serrata*, *Cordyline australis* and *Pittosporum eugenioides*.

Plant age	R^2	F	d.f.	Species	R_h mean ± SE †
Newly planted	0.17	55.14***	2,521	*A. serrata*	2.94 ± 0.03[a]
				C. australis	2.51 ± 0.04[a]
				P. eugenioides	2.57 ± 0.04[b]
One year old	0.01	4.89**	2,629	*A. serrata*	2.55 ± 0.02[a]
				C. australis	2.49 ± 0.03[b]
				P. eugenioides	2.52 ± 0.03[ab]
Two years old	0.05	10.74***	2,402	*A. serrata*	2.59 ± 0.02[a]
				C. australis	2.49 ± 0.03[b]
				P. eugenioides	2.59 ± 0.01[a]
Three years old	0.21	23.66***	2,173	*A. serrata*	2.67 ± 0.02[a]
				C. australis	2.42 ± 0.04[b]
				P. eugenioides	2.61 ± 0.04[a]
Four years old	0.24	10.97**	1,30	*A. serrata*	2.72 ± 0.07[a]
				C. australis	2.47 ± 0.08[b]

**$P < 0.01$, ***$P < 0.001$

† Within each age cohort, species not sharing the same letter displayed significantly different ($P < 0.05$) R_h mean.
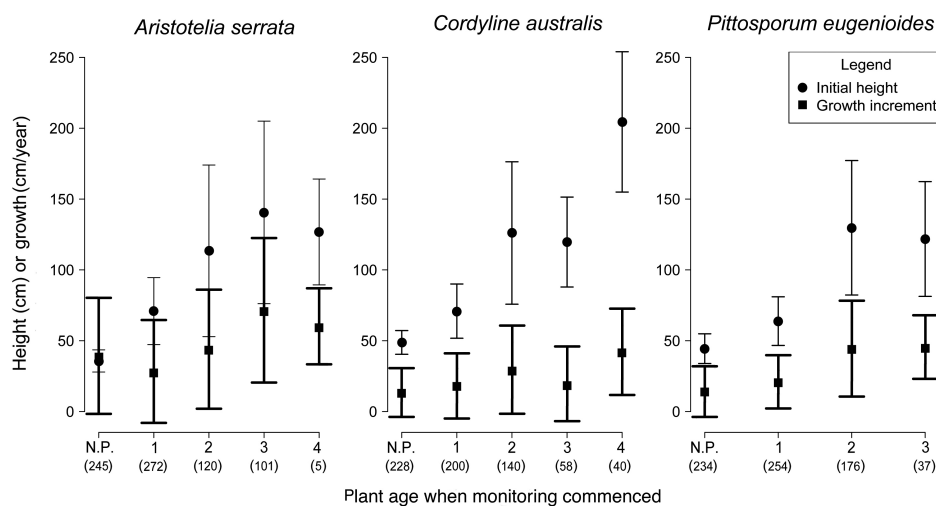

Figure 4. Initial plant height and subsequent annual height growth increment for plants of different ages (newly planted (N.P.) to 4 years old). Mean ± standard error. Sample size (*n*) is indicated under each age cohort.

conditions (Turner 2013). In all three species, the probability of mortality decreased as initial height of the plant increased (Fig. 3). However, at one site we also found a number of tall (250–300 cm) *A. serrata* trees uprooted, probably by high winds. Thus, on exposed sites, the susceptibility of tall plants to high winds in Wellington probably contributed to greater than usual mortality of older plants. This was contrary to previous results suggesting that plant mortality in forest restoration projects can become negligible in as little as 2 years following planting (Graham & Filmer 2007; Ledgard & Henley 2009). Our observations indicate that plant height, rather than plant age, is a better predictor for identifying when plant mortality becomes negligible.

Given that mortality was greater in shorter plants, keeping grass below the height of seedlings and trees should reduce competition and increase survival. Proper management during and following planting (e.g. using adequate planting techniques, applying a layer of mulch, and avoiding accidental mowing of plants) also reduced mortality.

The effects of additional site-level and microsite factors on survival varied among species. Consistent with the findings of Sullivan et al. (2009), soil compaction did not influence mortality of *C. australis* or *P. eugenioides*. However, *A. serrata* suffered greater mortality at sites with relatively compacted soils. The preference of *A. serrata* for riparian areas (Davies-Colley et al. 2011) may reflect not just its preference for humid and shaded areas but also its preference for loose soils. *Cordyline australis* experienced greater mortality on steep slopes, and when over-topped by a denser canopy, perhaps reflecting its preference for moist soil and high light conditions (Simpson 2000). From the description provided by Beever et al. (1996), the high observed mortality of 4-year-old *C. australis* plants was likely due to a phytoplasma tree disease known as 'sudden decline'. However, longer-term studies are required to assess the shape of the relationship between mortality and plant age of *C. australis* in urban areas. *Pittosporum eugenioides* appeared the most tolerant of the range of conditions experienced across the 11 sites. These inter-species differences suggest there is potential to maximise survival rates by tailoring the selection of species more specifically to particular site and microsite characteristics (see also Smale et al. 2001; Sullivan et al. 2009; Smaill et al. 2011).

The agency managing the restoration sites was not retained as an important variable in our models explaining observed mortalities of seedlings and trees in Wellington City. We suggest that the more favourable environmental conditions of the sites and the occasional use of mulch by some community groups (variables that were deemed important) contributed to the 6% greater survival rates recorded in sites planted by community groups compared with WCC-planted sites (Appendix S2).

Aristotelia serrata and *P. eugenioides* recorded in this study had higher relative growth rates (R_h) than *C. australis*. This difference in R_h makes *A. serrata* and *P. eugenioides* particularly attractive for restoration projects aimed at achieving rapid canopy closure (Sullivan et al. 2009).

Results from our study offer further guidance for management associated with forest restoration projects in urban areas (see also Sullivan et al. 2009; Oldfield et al. 2013). The results appear robust despite several shortcomings. For example, grass overtopping plants might have impeded our ability to locate shorter individuals during site inspections. These missing trees may have been erroneously categorised as 'dead', ultimately overestimating true mortality. However, mortality could also have been underestimated given that trees

dying from stress may take longer than 12 months to lose all their leaves (Bigler & Bugmann 2003). While our optimum models explained only moderate amounts of variation (R^2_c and $R^2_m < 0.45$), particularly for *C. australis*, this was expected given the large number of potential explanatory variables (Møller & Jennions 2002) and the likely cumulative effects of multiple stressors (Porté & Bartelink 2002). Finally, our study was not based on truly longitudinal data (e.g. following all saplings from planting until 4 years of age) introducing potential error associated with exact age determination of plants. We thus suggest that results presented here are a first step towards understanding age-related seedling and tree mortalities in urban forest restoration projects and should be confirmed by future long-term studies across a range of environmental gradients typically experienced in New Zealand.

Acknowledgements

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Supplementary Material

Additional supporting information may be found in the online version of this article:

Table S1. Study sites, agency managing sites, number of individuals by age group, and initial measuring dates.

Table S2. Summary of environmental covariates and annual plant mortality rates between council-led (WCC) and community-group-led (CG) forest restoration projects (n = 11) in Wellington City, New Zealand.

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