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Kiri Joy Wallace & Bruce D. Clarkson

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REVIEW ARTICLE

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Urban forest restoration ecology: a review from Hamilton, New Zealand

Kiri Joy Wallace and Bruce D. Clarkson

Environmental Research Institute, University of Waikato, Hamilton, New Zealand

ABSTRACT

Restoration of urban forest improves quality of life for city residents and is important for boosting native biodiversity. However, the scientific knowledge required to inform successful restoration is largely lacking for urban forest ecosystems, which differ from rural forests. Here we review two decades of urban forest restoration research in Hamilton, New Zealand and summarise key findings across seven major ecological topics: i. species traits, filters and thresholds, ii. species richness and target ecosystems, iii. tree regeneration, iv. seed banks and seed rain, v. seed predation, vi. enrichment planting and vii. restored forest function. We then discuss general urban restoration principles that increase the efficacy of urban restoration efforts including restore to a minimum of 10% indigenous cover in cities, develop a step-wise restoration plan and prioritise partner engagement. The purpose of this review is to aid urban forest restoration across New Zealand and globally.

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Introduction

It is paramount we understand the dynamics of planted native forests to restore them effectively (Oldfield et al. 2015; Miller et al. 2016; Wallace et al. 2017). This is critical for urban forests in particular because of benefits they provide, such as ecosystem services (Dobbs et al. 2011; Endreny et al. 2017), enhanced human health and well-being (Alberti 2005; Brown et al. 2014), and havens for native biodiversity (Aronson et al. 2014b; Threlfall et al. 2016). About 86% of New Zealand citizens live in cities (The World Bank 2014), where their opportunity to regularly connect with nature is often a local forested park. Despite the manifold benefits, most New Zealand urban centres have little native forest cover remaining (Clarkson et al. 2007c). Forest fragments are instead relegated to upland rural areas, and even there, restoration work is often needed (Norton et al. 2018).

Restoration ecology is a novel, expanding discipline both globally (Perring et al. 2015; Crouzeilles et al. 2016) and in New Zealand (Laughlin and Clarkson 2018; Norton et al. 2018). Restoration actions vary widely from alterations of the abiotic such as alteration of soil composition (Janzen 2000) to manipulation of biotic elements such as the attraction of birds to promote seed dispersal (Robinson and Handel 1993; Reid and Holl 2012). New

CONTACT Kiri Joy Wallace kwallace@waikato.ac.nz

Zealand practices typically include removal of non-native, invasive species (eg trapping/poisoning of introduced predatory mammals), followed by re-introduction of native species (eg planting of native tree seedlings). These practices have recently been extended to the urban context (Overdyck et al. 2013; Clarkson and Kirby 2016) in an effort to maximise the many benefits of native biodiversity where people live (McPherson et al. 1997; Matsuoka and Kaplan 2008; Johnson and Handel 2016; Acosta et al. 2018).

Urban forests are distinctly different from rural forests both ecologically (eg greater fragmentation and non-native species pressure) and environmentally (eg urban heat island, higher pollution levels), and are more dynamic than rural forests (Groffman et al. 2016; Doroski et al. 2017). Unique challenges to urban forest restoration worldwide include the urban heat island effect (Samuel et al. 2016), fragmented city landscapes (Drinnan 2005; Clarkson et al. 2007c) and non-native species invasion (Trammell et al. 2012; La Sorte et al. 2014). Planted urban forests are therefore faced with additional pressures and require intensive management to return to a functional native state (Ruiz-Jaén and Aide 2006).

Urban restoration ecology is a young, growing scientific field in New Zealand (Clarkson and Kirby 2016), and research in Hamilton has been occurring for about two decades since the mid 1990s (Clarkson and Bylsma 2016). We have found that the underpinnings of urban forest restoration must be scientific to ensure success because little is known about either reconstruction of the forest from scratch (ie plantings in former pasture or parkland, Clarkson and Kirby 2016) or restoration of degraded, extant forest patches. Urban forest restoration by trial and error is costly, and resulting failures are both discouraging to practitioners and condemning of future funding approval. Instead, we propose an evidence-based approach developed with partners and practitioners, informed by, ecologists and applied through practice oriented principles. This completes the full cycle of discovery and implementation, allowing restoration efforts to be successful.

The applied practice of urban ecological restoration has only emerged in Hamilton recently (Clarkson and McQueen 2004; Clarkson and Kirby 2016). Prior to 2000 most efforts focused on either care of existing forest remnants or constituted new plantings characterising revegetation more than ecological restoration. After 2000 fuller recognition of the potential of Hamilton gullies as places to reconstruct indigenous forest and the decision to set aside 60 ha of public land for the establishment of Waiwhakareke Natural Heritage Park shifted the approach to one of ecological restoration. A wider vision for the city as a driver of a regional, landscape scale restoration was raised as early as 2004 (Clarkson and McQueen 2004). Together, the Gully Reserves Management Plan (Hamilton City Council 2007), the Waiwhakareke Natural Heritage Park Operative plan (Hamilton City Council 2011) and the Hamilton Operative District Plan (Hamilton City Council 2017) are the main elements of the planning and policy framework. It is within this context that further ecological research was initiated to assist government agencies and community groups with their restoration endeavours.

The purpose of this review is to synthesise research done in Hamilton in order to aid urban forest restoration efforts across New Zealand and globally. We focus on the restoration of native forest flora but acknowledge that actions to re-instate native fauna should follow closely to achieve fully functioning urban forest ecosystems. There are some aspects of restoration (ie below-ground interactions) that are in early stages of investigation in Hamilton and while not discussed here, are recognised as important in restoration.

The review has two parts: first, we present restoration research from our Hamilton studies grouped by seven ecological topics (Figure 1): i. species traits, filters and thresholds, ii. species richness and target ecosystems, iii. tree regeneration, iv. seed banks and seed rain, v. seed predation, vi. enrichment planting and vii. restored forest function. These studies contribute theoretical and applied advancements to the field of urban restoration ecology by describing how planted urban forests develop and how to best manage them. The second section, 'General Urban Restoration Principles', summarises broader topics including socio-political dimensions to consider for successful urban restoration projects.

Hamilton research

Study sites

Hamilton is located on New Zealand's North Island, which was historically 75% covered in the temperate rainforest but 66% of which is now cleared for agriculture and

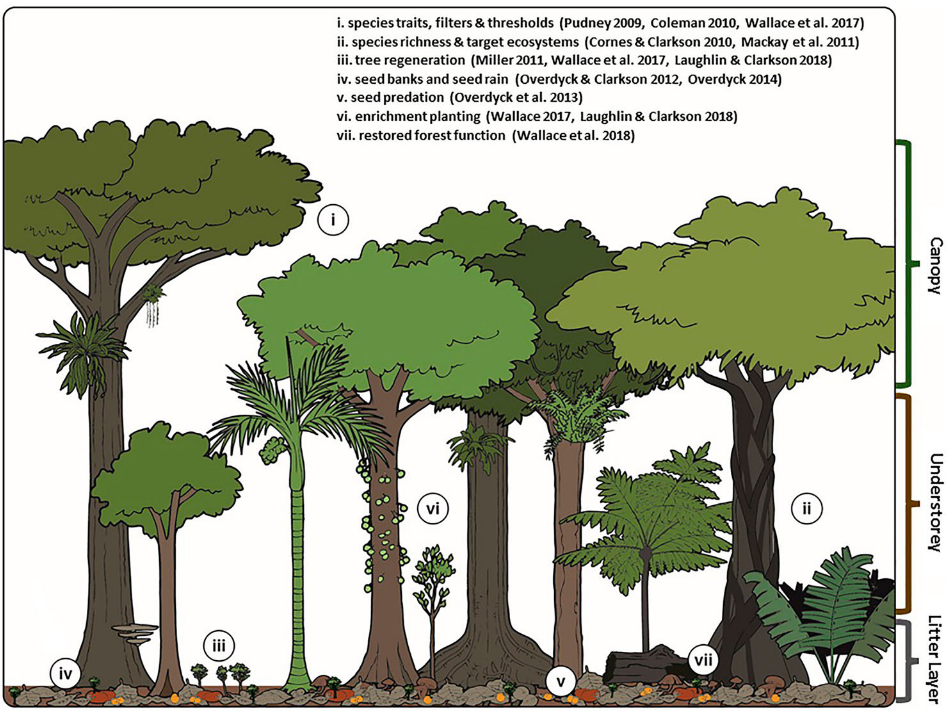

Figure 1. Ecological topics to consider in an urban forest undergoing restoration, listed with related research references from Hamilton, New Zealand. Topics are defined as: i. species traits, filters and thresholds largely determine community composition and successional progress, ii. species richness and target ecosystems are measures for setting restoration planting goals, iii. tree regeneration is an index of restoration success and necessary for a self-perpetuating forest, iv. seed banks and seed rain determine non-native weed control and enrichment planting requirements, v. seed predation indicates pest animal presence and when planting must occur by over sowing seed, vi. enrichment planting of late-successional species is often required in urban forest but must occur only when appropriate microclimate conditions develop, vii. restored forest function should be a primary goal of restoration work.

silviculture (Nicholls 1980). Data were collected from restored urban forest patches in Hamilton (37.7870°S, 175.2793°E), population 160,000. Hamilton has an annual mean precipitation of 1110 mm with mean minimum and maximum temperatures of 8.7°C and 18.9°C, respectively (NIWA), and 2.1% indigenous forest cover (Clarkson et al. 2007b). Data from research outputs discussed in this paper were collected from forests throughout and closely surrounding the Hamilton ecological district (Figure 2) from the mid-1990s to 2017. These forests fell broadly into four categories based on location and restoration status: (1) remnant or secondary forest in the Hamilton surrounds, (2) remnant or secondary forest within Hamilton (3) unrestored forest within Hamilton (de-forested and never actively restored) and (4) restored forest within Hamilton (de-forested then reconstructed, usually by planting from scratch into retired pasture). We summarise research from these studies by ecological topic below.

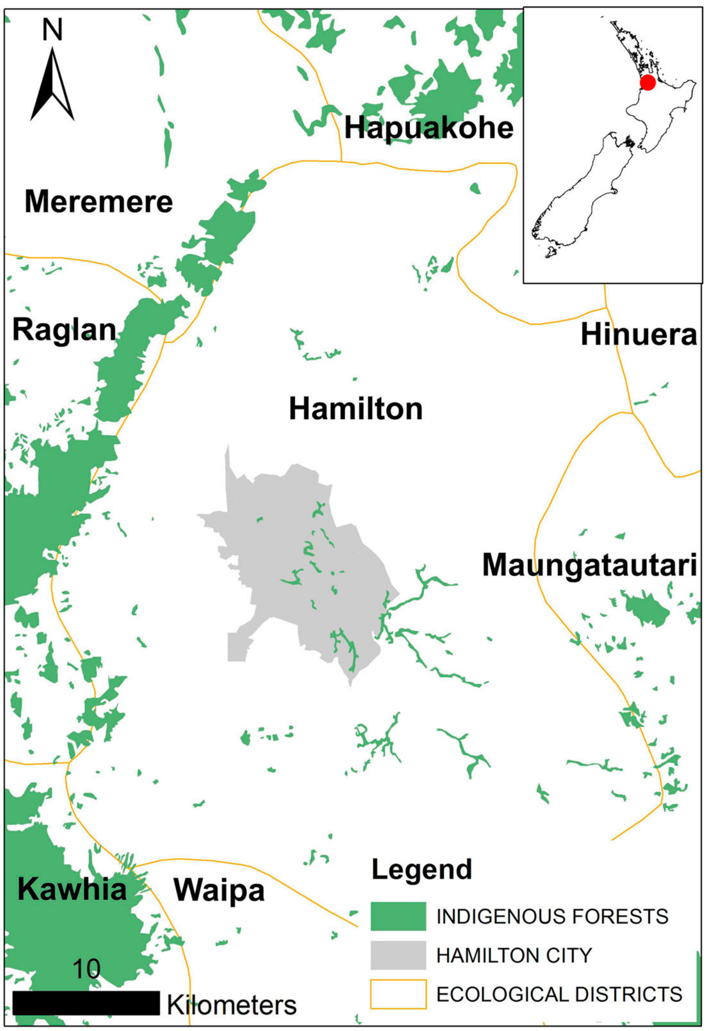

Figure 2. Placement of Hamilton city boundaries, Hamilton Ecological District and surrounding forest.

i. Species traits, filters and thresholds

Plant species traits often pose a substantial filter determining what species survive in the urban context. The Hamilton ecological district hosts at least 343 vascular plant species, but only 195 are found in the entirety of Hamilton's city boundaries (Overdyck 2014). This limitation is in part due to a lack of niches, as reconstructed forests inventoried in Overdyck (2014) had not matured yet to be able to facilitate more specialist late-successional species. Although this barrier disappears with time, the limited ranged of species typically used in restoration plantings also restricts the traits found in restored urban plant communities. This is often due mainly to logistical difficulties in the acquisition of seeds and successful propagation. Hamilton's planted forest species complement is only a subset of what is present rurally. For example, urban native canopy species represent only 46% that of rural native canopies, and urban understory species 26% that of rural canopies (Overdyck and Clarkson 2012). Although the inhospitable urban environment may filter as to prevent a full representation of rural forest species, these low percentages should be increased to represent a larger suite of species traits. A variation of traits and range of natural genetic variability can create forest community resilience in the face of disturbance (Sandel et al. 2011), global climate change (Laughlin 2014) and invasion (Funk et al. 2008).

Invasion by non-native species is a major limitation to the restoration of comprehensive forest assemblages, because non-native species have traits which confer competitive advantage (Pudney 2009; Coleman 2010; Trammell et al. 2012). Pudney (2009) found that in open canopy conditions in Hamilton forests, significantly greater numbers of the non-native *Lonicera japonica* (Japanese honeysuckle) occurred. This non-native liane's climbing habit allows it to smother native vegetation of small stature, essentially limiting establishment and therefore halting natural succession. This may be avoided through an integrated control approach including spot spraying, cutting and pasting, and encouragement of a closed canopy (dense restoration plantings) to block light (Wall and Clarkson 2006; Pudney 2009). In Hamilton forests where the native canopy tree *Dacrycarpus dacrydioides* (Kahikatea) and introduced *Salix cinerea* (Grey Willow) co-exist, the *S. cinerea* is able to gradually take over the canopy and suppress *D. dacrydioides* regeneration (Coleman 2010). Where *S. cinerea* densities exceeded 2 per 10 square m, *D. dacrydioides* seedlings were no longer present. These non-native plants act as filters constraining the growth of native plants in urban forests while also preventing forest development across vital ecological thresholds such as canopy closure.

Canopy closure in newly-planted forests is the first, most important threshold in forest development (Doroski et al. 2017). Wallace et al. (2017) studied the dynamics of a chronosequence of New Zealand planted urban forests aged 3–70 years, looking for thresholds of ecosystem properties most important for native tree regeneration. Using breakpoint analyses, they found that distinct thresholds existed in urban forests about twenty years after initial plantings (Figure 3). At twenty years canopy openness dropped to less than 5%, causing senescence of competitive non-native herbaceous ground weeds, and stabilisation of the microclimate (humidity and soil temperatures in particular). These conditions were significant drivers causing native tree seedling regeneration only after the canopy closure threshold was crossed. The forest will then develop naturally through other successional stages, including canopy gap formation through eventual dieback of the planted early-successional tree species. These gaps provide light levels required by some important

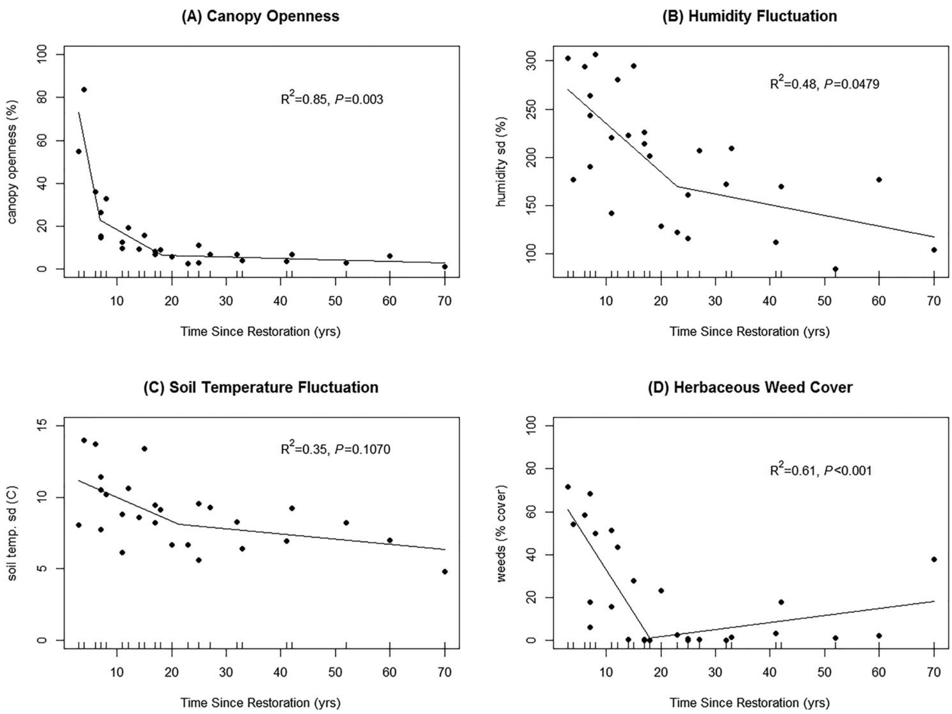

Figure 3. Breakpoints, or thresholds, of ecosystem properties significantly affecting native tree regeneration in a restored urban forest chronosequence ($n = 27$). Thresholds all occurred approximately 20 years after restoration planting. (A) Canopy openness had two thresholds, occurring at 7.0 and 18.1 years, (B) humidity fluctuation had a threshold point at 23.1 years (C) soil temperature fluctuation at 21.3 years and (D) herbaceous weed cover at 17.9 years. Reproduced with permission from Wallace et al. (2017).

late-successional saplings to fully recruit into the canopy (Knowles and Beveridge 1982; Lusk and Ogden 1992). This later stage is appropriate for enrichment plantings if low seed dispersal into the site is limiting spontaneous recruitment, and may require 'managing the gap' actions if light-demanding weeds take advantage of the newly available light resource (Whaley et al. 1997).

In conclusion, a larger array of early and mid-successional native species with a wide variety of traits should be planted to form urban forests. This pioneering cohort should be managed to cross the threshold of canopy closure through the management of invasive non-native weeds. Once forests are able to cross the canopy closure threshold, environmental filters are altered, and the microclimate will be appropriate for spontaneous native tree regeneration and late-successional enrichment plantings.

ii. Species richness and target ecosystems

With suitable management, native woody species richness increases during the first few decades following initial forest restoration plantings (Mackay 2006; Shoo et al. 2015; Wallace et al. 2017). MacKay et al. (2011) found on average a planted forest canopy species count doubled from 8 species in the first decade to 17 species by the third decade after planting. The main drivers during this dynamic window of forest succession include canopy closure, subsequent weed suppression and development of a suitable microclimate (Wallace et al. 2017). But these events cannot occur without the planting of pioneer species and follow-up management such as non-native weed control and enrichment plantings.

To achieve species richness goals it is sometimes helpful to adopt an extant or historical reference ecosystem as a restoration target (Bakker et al. 2000). Definition of the target, reference ecosystem has been the topic of some debate (Pickett and Parker 1994; Aronson et al. 1995; White and Walker 1997). Purists advocate complete elimination of introduced species and re-creation of historical ecosystems. This is difficult without excellent records, in landscapes lacking examples of extant indigenous cover, and when attempting to shape ecological communities resilient to global climate change (Laughlin 2014; Perring et al. 2015). Other members of the restoration ecology community have advocated novel ecosystems, which have also been hotly debated (Hobbs et al. 2009, 2014; Murcia et al. 2014; Aronson et al. 2014a). The novel ecosystem approach suggests a mix of native and non-native species are acceptable in a restored ecosystem and is driven by elements of restoration theory such as a narrow focus on achieving specific ecological functions, accommodating human social values and sheer practical feasibility. However, a novel ecosystem is not substantiated by evidence to be as ecologically valuable as a natives-only ecosystem (Pauchard et al. 2018)The novel ecosystem approach may also have a disproportionally negative impact in easily-invaded island ecosystems such as New Zealand, due to vacant niches and a high proportion of specialised, endemic species that evolved in isolation (Alpert et al. 2000; O'Dowd et al. 2003).

Regardless of theoretical stance, many restoration projects land somewhere between these two extremes and are usually defined at the end more by funding, project design and project longevity.

Here, as recommended by the Society for Ecological Restoration International (McDonald et al. 2016), and New Zealand experts (Norton et al. 2016) we advocate the use of an extant reference target ecosystem to use as a general restoration guide. This is part of a broader practical approach to restoring ecological integrity (Lee et al. 2005) to protect unique, endemic flora and fauna, and feasible in New Zealand due to its relatively recent settlement and land use records.

Reference ecosystems should be geographically close to the restoration site and share ecosystem properties such as landform, soil type and previous plant community type. For urban restoration sites, it is best to pair with existing urban forest rather than rural forest because city conditions are unique, and this will set an achievable benchmark for species richness. The high edge to interior ratio inherent to small urban forest patches affects an array of properties, such as altered vegetation structure, increased sunlight availability, greater fluctuations in humidity, greater soil compactness and more exposure to pollution (Matlack 1993; Young and Mitchell 1994; Murcia 1995; Harper et al. 2005; Malmivaara-Lämsä et al. 2008). Urban forests are therefore particularly disrupted by edge effects because of their small patch size and isolation, compounded by the altered environment of the surrounding urban matrix (eg asphalt causing the urban heat island, Oke et al. 1989).

Selection of target ecosystems must include subsequent surveys to determine species richness and measure other ecosystem properties. For example, Cornes and Clarkson (2010) assessed the largest urban indigenous forest remnant in Hamilton, Claudelands Bush. The resulting plant species list (134 species) and water table data provide targets for forest restoration elsewhere in Hamilton and also improve adaptive management of

Claudelands Bush, especially when considered together with three earlier ecological assessments of this forest fragment (Gudex 1955; Boase 1985; Whaley et al. 1997).

Regular inventories of valuable ecological sites like Claudelands Bush are important for awareness of how species richness or distribution may be changing across the urban landscape over time. Managers can then consider landscape scale richness, which impacts restoration efforts by providing local seed-sourcing options and spontaneous seed dispersal. Repeated monitoring also deepens understanding of population changes and lag effects (eg management practices or climate change), providing insight for how to manage other restored or reconstructed forests in a changing environment.

Two large-scale inventories have occurred city-wide in Hamilton: the Key Ecological Sites of Hamilton City: Volumes I, II and III (Downs et al. 2000) and Key Ecological Sites of Hamilton City (Cornes et al. 2012). Both were commissioned by the Hamilton City Council and conducted by the University of Waikato Environmental Research Institute, who established a permanent vegetation plot network for the surveying. If urban forest remnants are scarce and inventories like this cannot be conducted, it is then best to expand beyond the urban zone tosurvey the surrounding landscape by accessing knowledge on indigenous forest species across the whole ecological district (Clarkson 1981; Clarkson et al. 2007a) and region (Leathwick et al. 1995).

iii. Tree regeneration

Achieving late-successional tree establishment in restored forests signifies the crossing of an important threshold in forest dynamics (Oliver and Larson 1990). Late-successional canopy tree species are long-lived, vital elements of the forest community, and are therefore necessary recruits for long-term urban forest restoration (Labatore et al. 2017). Without regeneration of late-successional trees, planted early-successional trees may thrive for several decades but then senesce without replacement, causing native canopy collapse and re-invasion by non-native invasive species (Doroski et al. 2017).

Late-successional tree species regeneration is dependent on forest age, and requires specific regeneration conditions, such as a stable forest floor microclimate (Wallace et al. 2017). Young forest patches with open canopies often possess ground layers invaded by competitive herbaceous non-native weeds that hinder tree regeneration. Both these barriers are overcome upon crossing the threshold of canopy closure (Doroski et al. 2017), sometimes as soon as five years post-planting when herbaceous weeds are shaded out (Laughlin and Clarkson 2018) and more so twenty years post-planting, when humidity and soil temperature levels stabilise (Figure 3; Wallace et al. 2017).

Wallace et al. (2017) found that total and late-successional native plant regeneration (trees and epiphytes) in Hamilton and New Plymouth increased significantly over the first 70 years after initial restoration plantings (Figure 4). This increase was of particular note for late-successional trees, which rarely occurred in planted forests under ten years old (~1000 stems/ha), but were more common (~10,000 stems/ha) in forests older than that (Figure 4B). Obligate epiphytes (which must grow on trees) did not colonise until forests were at least 20 years old and remained at relatively constant densities after that (growing on ~800 host trees/ha, Figure 4E).

Urban forests typically require intensive management to cross the threshold into a closed canopy system. Initial tree plantings may succumb to intense competitive pressure from non-native weed species in cities (Miller 2011). Initial plantings of densely spaced (eg 1 plant per m^2), tall plants (eg 1 m), use of weed matting and other means of weed control

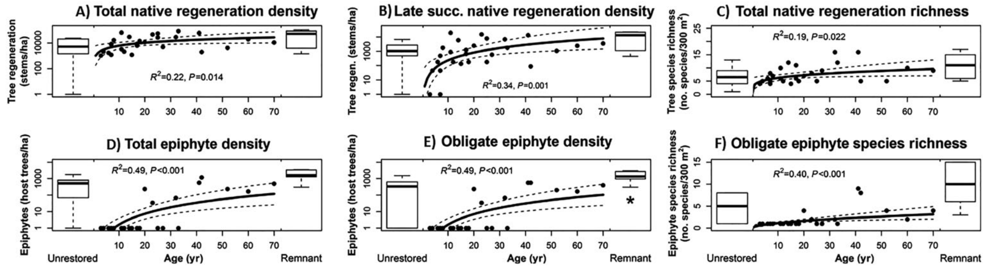

Figure 4. Native tree and epiphyte regeneration in restored forests over time (central section of each bivariate plot, $n = 27$), compared with unrestored ($n = 6$) and remnant forests ($n = 6$) (boxplots to left and right, respectively). (A) Total native tree regeneration density, (B) late-successional native tree regeneration density, (D) total epiphyte density and (E) obligate epiphyte density have *y*-axes shown in log-scale. Boxplots represent 25th, 50th and 75th percentiles and whiskers extend to data extremes. Scatterplot points represent restored sites; the solid lines represent the fitted values from a linear regression model on log-transformed data, and the dashed lines represent 95% confidence intervals. Asterisks indicate a significant difference between unrestored and remnant forests ($\alpha = 0.05$). Adapted with permission from Wallace et al. (2017).

are helpful in countering competition in the first decades and reducing the need for expensive herbicide use. If herbicide is used, it should be minimised and tailored to the site to avoid killing regenerating native plants and causing pollution.

Urban areas do not typically host high vertebrate herbivore densities, and therefore focus on protection against hare, goat and pukeko browse sometimes does not require as much attention (Cornes et al. 2008) as rural areas. Instead, clear communication between urban land managers, practitioners and contractors is most important to orchestrate appropriate planting timing, location and follow-up care to cross the threshold into native tree regeneration conditions.

Even when conditions are conducive to tree regeneration there may not be adequate propagules available either in the seed bank or dispersed from nearby forest to provide for late-successional tree establishment. This is often the case in urban forests, which have a long history of disturbance or are isolated, and it is important to recognise when enrichment planting is necessary.

iv. Seed banks and seed rain

Successful establishment of late-successional trees in restored forests is paramount to ensuring diverse, resilient and sustained forests. Forest seed bank and seed rain composition largely determine future forest composition (Labatore et al. 2017). Determining what seeds are available from these sources can inform management decisions well in advance, such as weed control measures and required enrichment species planting. Early, well-informed decision making can change the trajectory of forest ecosystem development and save long-term costs and effort.

Native New Zealand forest seeds persist for only a few years in the seed bank (Rowarth et al. 2007), hence restored urban forest seed banks are typically dominated by non-native wind-dispersed herbaceous weeds and native ferns (Overdyck and Clarkson 2012). This implies that while ferns may recolonise spontaneously, and sometimes facilitate a desirable ecological trajectory (Brock et al. 2018), a developing urban forest will require a management plan for weed control and native woody enrichment planting. Overdyck and

Clarkson (2012) surveyed canopy vegetation and seed banks of restored Hamilton forest patches and found that despite native canopy compositions, seed banks were predominantly comprised of non-native species (46 non-native species out of 69 total species). When then compared with seedbanks of rural remnant forests surrounding Hamilton, only 64% of the native species were found in both forest types, indicating that the natives that do persist in the seedbank are not fully representative.

If most woody native seeds do not persist in the seed bank, seed rain is then of utmost importance. Overdyck (2014) discovered that seed rain in Hamilton forest patches primarily consists of native ferns and early-successional woody species dispersed by wind and water (Figure 5). Arrival of the typically large, fleshy-fruited native woody species is primarily via avian mutualisms, suggesting that urban areas without dispersal agents such as the native wood pigeon, the Kereru (*Hemiphaga novaeseelandiae*), will not reach the full forest species complement.

Efficient management of urban forests undergoing restoration requires knowledge regarding species in the seed bank and seed rain. It can generally be assumed that late-

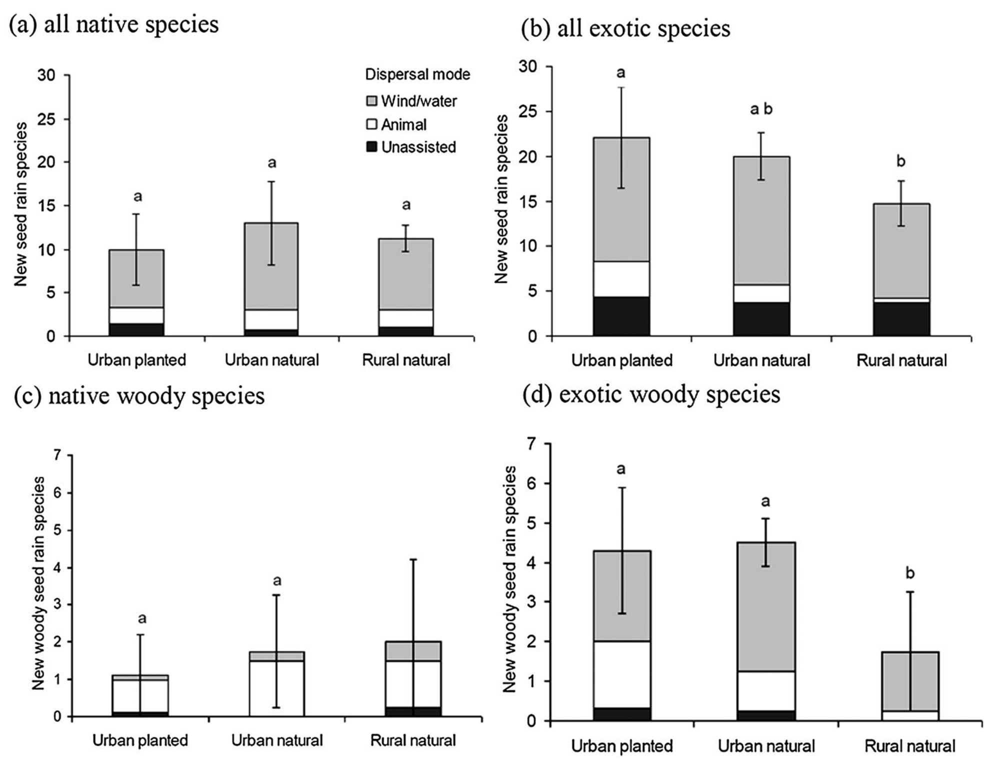

Figure 5. Mean (±SD) of species that are newly arriving in a forest patch (a) all native species, (b) all exotic species, (c) native woody species and (d) exotic woody species in the annual seed rain (and not present in vegetation at sites) in urban planted ($n = 9$), urban natural ($n = 4$) and rural natural ($n = 4$) forests. Different letters denote significant differences between overall treatment means for Tukey's pairwise comparisons, $p < .05$. Here 'Urban planted' denotes restored urban forest patches, 'Urban natural' are urban forest remnants, and 'Rural natural' are rural forest remnants. Reproduced with permission from Overdyck (2014).

successional native species will be lacking, but ferns will re-colonise spontaneously, and non-native weed species will be prevalent.

v. Seed predation

Native, late-successional tree species' seeds may come to rest on a restored urban forest floor, either passively through seed rain or intentionally through sowing by land managers. Direct sowing can be a cost and labour efficient option for introducing late-successional plant species to forests undergoing restoration (Cole et al. 2011). However, seed predation poses an obstacle to the establishment (Labatore et al. 2017), which is exacerbated for New Zealand tree species that did not co-evolve with introduced seed-predators like rats (Daniel 1973).

A study in Hamilton forest patches investigated broadcast seeding of three late-successional, large, fleshy-seeded tree species (*Beilschmiedia tawa*, *Elaeocarpus dentatus* and *Litsea calicaris*) to determine best practice for discouraging seed predation and improving seedling establishment (Overdyck et al. 2013). The factorial design included a control and three factors: caging, removal of fleshy fruit pericarp and incorporation into fertilizer-enriched clay balls (Figure 6). Caging and clay balls significantly increased survival and establishment. Uncaged seeds had 58% loss compared with caged seeds, which only suffered 4% loss. Uncaged seeds with pericarp removal that were also in clay balls had an intermediate loss of 35%. Use of the clay ball doubled the seedling establishment rates after germination in *B. tawa* (6% vs. 12%).

Urban forest studies further beyond Hamilton have confirmed that seed predation is a substantial factor limiting late-successional tree establishment. In North America, Labatore et al. (2017) introduced canopy tree seeds to urban forests and discovered that seedling recruitment increased significantly where seed predators were reduced. These results, paired with their work showing a lack of native seed rain, suggests that urban forests without canopy species' seed introduction and subsequent protection were destined to experience canopy collapse and revert to non-native species dominated urban shrublands.

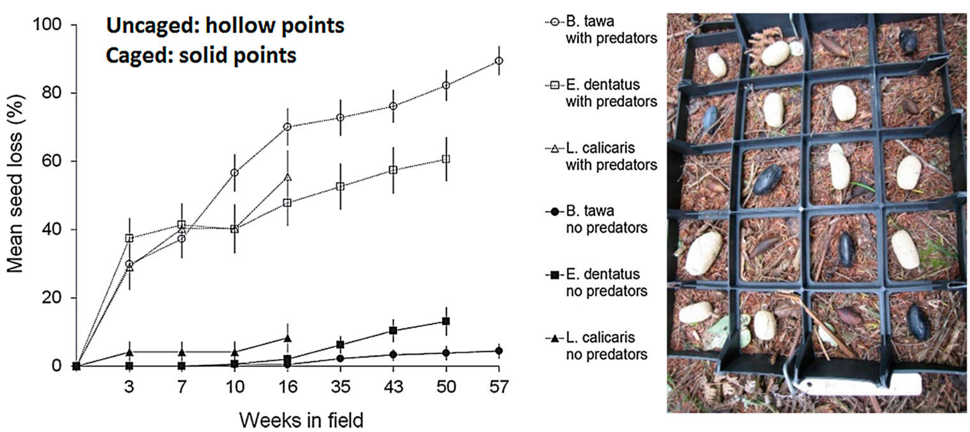

Figure 6. Cumulative seed loss (mean ± SE, $n = 36$) for *B. tawa*, *E. dentatus* and *L. calicaris*, with seed predator access (uncaged treatment) and no seed predator access (caged treatment) at one rural and two urban forest sites. Species start dates have been standardised to time zero although start dates in the field were staggered due to the timing of seed collection, so experiment length was shortened accordingly for the species collected later. Adapted with permission from Overdyck et al. (2013).

While restored urban forests can be enriched with late-successional plant species through direct seeding, resulting seedling recruitment is only likely if seed predators are excluded or controlled. This may be done by caging seeds, encasing them in clay balls (Overdyck et al. 2013), or intensifying non-native omnivorous mammal control to levels adequate for native bird protection (Saunders and Norton 2001). Further, in cases where germination and establishment are successful, several more years of care must be invested to ensure plant survival to a hardier size and age.

vi. Enrichment planting

Due to lack of late-successional species in seed banks and seed rain, and limitations posed by seed predation, forests undergoing restoration often require intervention by enrichment planting of late-successional tree seedlings or saplings after formation of an early-successional tree canopy (Overdyck et al. 2013, Suganuma and Durigan 2015, Bertacchi et al. 2016). This is frequently the case in urban areas, where forest undergoing restoration are typically geographically separated from native seed sources and pollinators, dispersal agents are limited, and control of seed predators is difficult.

Experimental work in herbaceous weed-infested Hamilton restored forests demonstrated that enrichment was possible by planting tall (>1 m) *B. tawa* seedlings (Wallace 2017). *B. tawa* exhibits the classic characteristics of a late-successional tree species seedling, including a slow growth rate, extreme shade-tolerance (Knowles and Beveridge 1982, Carswell et al. 2012), need for a stable understorey microclimate (Clarkson and McQueen 2004) and provision of dense shading once mature. Despite the necessity of an established canopy to protect *B. tawa* from frosts and desiccation while very young, it later grows faster under indirect light provided through small canopy gaps (Knowles and Beveridge 1982), which warm the air and soil slightly. An initial planting height of >1 m under a restored canopy limited suppression by the aggressive monoculture groundcover weed *Tradescantia fluminensis*, when concurrent mulching and weeding did not significantly help over 4 years of establishment and growth (Wallace 2017).

Enrichment planting is occurring on a wide scale at a large forest restoration project in Hamilton named Waiwhakareke Natural Heritage Park. This forest ecosystem has been reconstructed from scratch by planting early-successional native forest species followed by enrichment plantings after approximately seven years of growth. Laughlin and Clarkson (2018) found enrichment plant survival is ultimately determined most by canopy closure (ie planted forest age) and canopy composition. Under young, open canopies, planted enrichment seedlings survived best if overshadowed by mainly species in Myrtaceae (tea tree spp., genera *Leptospermum* and *Kunzea*), but under older, closed canopies, planted enrichment seedlings survived best with a more diverse canopy composition including broad-leaved species (Figure 7A). Growth rates improved with canopy age (Figure 7B).

Many epiphytes, which are plants that grow on trees, are also considered late-successional plant species due to their microclimate sensitivity (León-Vargas et al. 2006; Bryan et al. 2011; Clarkson 2011). Just as with other enrichment species, they may need to be actively introduced into urban forests undergoing restoration, but only once conditions are suitable. The shrub epiphyte *Griselinia lucida* is native to the Hamilton Ecological District and was once noted as a conspicuous epiphyte in the New Zealand lowland rainforest (Dawson 1966). However, increased forest fragmentation and urbanisation have lowered humidity levels in forest canopies, which is detrimental to *G. lucida*

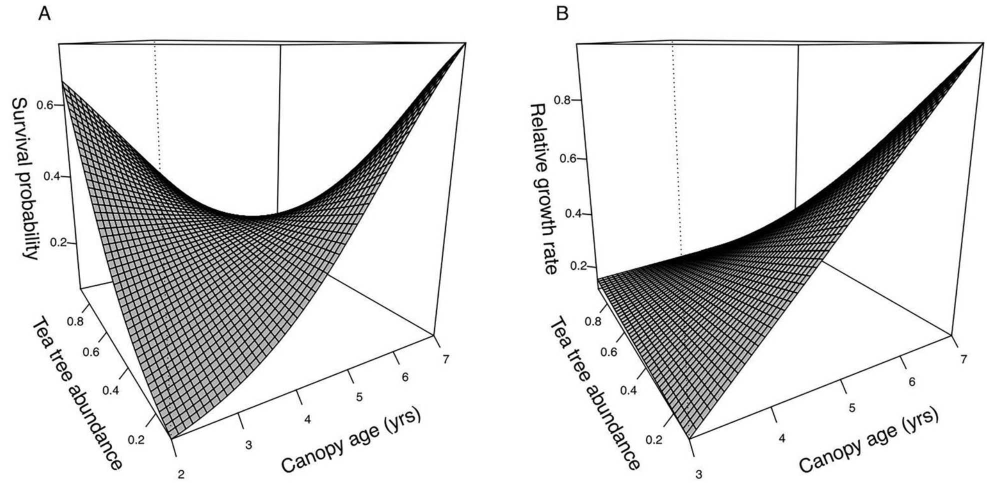

Figure 7. Interaction plots illustrating the curved surfaces of generalised linear model-fitted predictions of seedling (A) survival and (B) relative growth rate across all planted enrichment species as influenced by the interacting effects of canopy age (years) and tea tree (*Leptospermum* and *Kunzea* spp.) relative abundance. The interaction between tea tree abundance and canopy age was significant for both response variables. Reproduced with permission from Laughlin and Clarkson (2018).

survival (Bryan et al. 2011). Therefore it is suggested that *G. lucida* be restored in the urban landscape, but only once restored forest canopy closure is achieved and provides high, stable humidity levels (Bryan and Clarkson 2013). The same approach is relevant for other members of the shrub epiphyte guild, epiphytic ferns and vines.

Enrichment planting of all late-successional plant guilds is typically necessary in urban forest patches undergoing reconstruction from scratch, but may also be needed in urban forest remnants that have merely been degraded or severely isolated. As with forest species richness generally, it is vital to be aware of what enrichment species may be missing. Miller (2011) compared late-successional understorey vegetation of the forest floor between remnant forests in and outside the Hamilton city boundaries. Urban forest remnants comprised only 61.5% of the native understorey species found in rural forest remnants. In a similarly designed study, Bryan (2011) found that urban forest remnants hosted only 55.2% of the native epiphyte species found in rural forest remnants. Both studies also found that the late-successional species that were found in urban remnants occurred in lower densities than in rural forests. Despite altered conditions (ie microclimates) in the urban context, it is likely that urban forests are suitable for a much larger proportion of the plant species found in rural remnants and should be enriched accordingly with representation from all guilds, including trees, shrubs, herbs, epiphytes and perhaps hemiparasites like mistletoes and parasitic plants such as *Dactylanthus taylorii*.

vii. Restored forest function

Forest restoration is fully achieved when both forest structure and function are reinstated. In this context, function encompasses any ecological processes or ecosystem services. Restoration work often focuses exclusively on plant community structure or richness, without consideration of restoring functional processes (Montoya et al. 2012; Wortley et al. 2013). This is unfortunate, as forest restoration is a valuable tool for recovering degraded ecological functions. When successful in an urban setting, the benefits are twofold: support of native biodiversity and ecosystem service provision for urban citizens (Alberti 2005).

Creating a species-rich, complex forest structure can be a means of kick starting ecological function (Derhé et al. 2016) that may be expected for the forest's successional stage (Guariguata and Ostertag 2001). Increased ecological function due to re-introduced link between even a few species can form a positive feedback cycle, enabling more complex ecosystem connectivity. For example, the single species of tree, *B. tawa* hosts endomycorrhizal fungi in its root system, feeds many herbivorous insects, grows large fleshy fruits for bird sustenance, and provides establishment sites for epiphytes (Knowles and Beveridge 1982). Practitioners should include plants with varying functional traits when planning species composition (Laughlin 2014) and shape plans to intentionally replace missing functional links between native species.

Research on restored forests in Hamilton has demonstrated that not all ecological functions are connected to restorated urban forest structure. Wallace et al. (2018) studied connections between the restored forest canopy and nutrient cycling in the forms of decomposition and denitrification. They found that while restoration of a native, evergreen, closed canopy affected decomposition, denitrification by soil microbes was driven by other abiotic factors completely, such as drainage (Figure 8). This research highlights the importance of knowing what drivers are behind desired ecosystem functions and specifically managing those for functional restoration success.

Functioning urban green spacesare important because they provide numerous ecosystem services of human benefit (Gaston et al. 2013). Urban forests, in particular, provide economically valuable services such as flooding and climate mitigation (Dobbs et al.

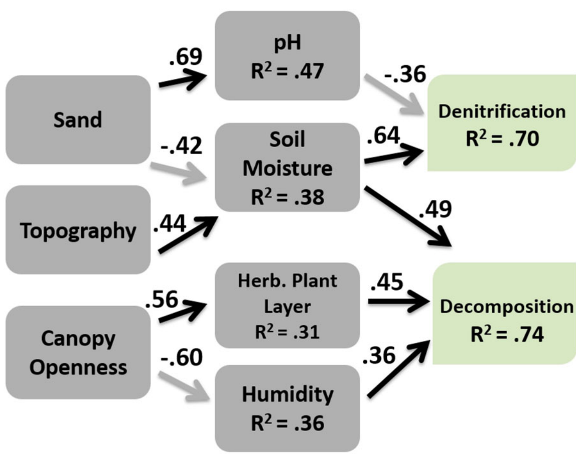

Figure 8. A structural equation model illustrating drivers of decomposition ($n = 17$) and denitrification ($n = 27$). The ecosystem functions of decomposition and denitrification are shown in light green, and their drivers in dark grey. Values by arrows are standardised path coefficients. R2 values are shown in the box of each response variable. For clarity positive pathways are black and negative pathways are grey. Model fit was assessed using Fisher's C statistic, where good-fitting models yield small C statistics and p-values > .05. This model fit the data well (Fisher's $C = 59.96$, df = 46, $p = .081$). Reproduced with permission from Wallace et al. (2018).

2011; Pickett et al. 2011) and pollutant filtering (Pickett et al. 2011). They are also a means of human re-connection to nature, recreational activity and social cohesion (Groffman et al. 2016). Therefore, restoration plans should include the re-instatement of both ecological structure and function.

General urban restoration principles

Successful ecological restoration in an urban context requires attention to several overarching socio-political principles. Landscape-level coordinated strategies for restoring to a minimum threshold of indigenous cover across a city is vital for safeguarding restored biodiversity. This landscape approach requires cooperation between private landowners and multiple public entities (eg city and regional councils, DOC). Careful step-wise plan development and implementation should be orchestrated between these entities, and these partnerships periodically strengthened. We expand on these principles below.

Restore to a minimum of 10% indigenous cover

An important determinant of biodiversity persistence is the total fraction of the landscape remaining in indigenous ecosystems. If a threshold of less than 10% indigenous cover is crossed, a major decline in native species richness is likely; inmany cases less than half the total species may remain due to local extinctions (Diamond 1975; Hanski 2015; Desmet 2018).

Urban development in New Zealand is typically associated with depletion of lowland indigenous ecosystems, including some of the country's richest, most diverse forests (Molloy 1980). Because of such development, the core of twenty main New Zealand urban centres range between <1%–8.9% remnant indigenous cover remaining (Clarkson et al. 2007b), making cities focal areas for restoration action. Clarkson et al. (2018) restated 10% as a minimum restoration target in severely biodiversity-depleted environments such as urban centres where indigenous cover is less than 10% (Grimm et al. 2008; Gillespie et al. 2012). This target can be achieved through encouragement of restoration work on both private and public land and should consist of large, connected forest patches to maximise support of biodiversity (Hanski 2015). In realising this threshold, key considerations include spatial configuration, ecosystem representation and connectivity (Clarkson et al. 2018). A clear, city-wide strategy for implementing landscape-scale restoration through a metapopulation approach will help reach this 10% target by facilitating important ecological processes such as dispersal (eg Wellington, New Zealand has a restoration strategy for the landscape scale).

Develop a step-wise restoration plan

A logical, achievable step-wise plan is vital in efforts to restore an urban forest ecosystem (Clarkson and Bylsma 2016). The high degree of invasion in cities increases non-native weed control requirements and the harsh abiotic conditions, (eg modified, polluted soils) can quickly suppress native plantings. A carefully tailored plan with clear steps (Table 1) suitable for the scale of the project will help avoid expensive pitfalls and

Table 1. Important steps to follow when forming an urban ecological restoration project.

	Restoration plan steps	Examples
1	Engage with all relevant partners	Local government, iwi, community groups, businesses
2	Define restoration goals	target ecosystem, non-native species removal, increased ecological function
3	Build landscape-scale vision	Understand connectivity with neighbouring ecosystems
4	Form long-term timeline	For a forest, this should be decade to century length
5	Create accurately scaled project budget	Plant costs, labour costs, administration costs
6	Acquire funding	Through granting agencies, local government, donations
7	Form restoration methods with scientific underpinning	Correct density of plantings, large enough plants, timely enrichment plantings
8	Perform restoration actions with partners	Conduct restoration plantings with partners present, follow up care
9	Monitor outcomes to gauge success	Annual monitoring of plant survival (quantitative if possible), photo points
10	Adapt methods moving forward	Change species mix being planted based on survival monitoring results

reduce risk of practitioner burnout and project failure. Plans may range from region-wide biodiversity strategies to local park management guidelines.

This step-wise approach should be applied from conception to the eventual on-the-ground restoration work. Plantings themselves should be dense, cared for intensively and then eventually enriched. Urban restoration work is best accomplished with a patient 'quality' over 'quantity' mindset. A specific planting plan using ecological underpinning can be formed using various online resources specific to a region or city, eg Hamilton Gully Guide (Wall and Clarkson 2006).

Ongoing monitoring is crucial in gauging success or failure, yet it does not occur in most restoration projects. Ruiz-Jaen and Aide (2006) conducted a review of 468 studies that employed seeding or planting techniques for terrestrial restoration and found only 14% evaluated restoration success afterward. If benchmarks of success are defined initially, and monitoring reveals whether they are attained, adaptive management can occur. Adaptive management allows precious resources to be allocated to the most effective, efficient management methods. These may be different from initial methods, or those used elsewhere because restoration approaches are rarely a one-size-fits-all. Waiwhakareke Natural Heritage Park has permanent monitoring plots that are surveyed regularly to measure long-term progress (Grove et al. 2006; Cornes et al. 2008), as well as ground-truthing after new plantings to determine immediate establishment success (Nepia et al. 2015). Findings are then carefully discussed in an advisory group and management approaches adapted where necessary to minimise wasted resources.

Prioritise partner engagement

Partner engagement is crucial for long-term success in the management of public urban green spaces. Waiwhakareke Natural Heritage Park in Hamilton New Zealand is a 60 ha urban forest restoration project that began on abandoned public pastureland in 2004. This project management includes the vital steps in a restoration project (Table 1), and continues successfully after 15 years (Clarkson et al. 2012). Waiwhakareke has been a joint effort between local government and community, standing the test of political inconsistency and charitable group leadership turnover. This is due to continued emphasis on partner engagement, which has made the progression of the restoration plan possible.

Positive, varied, partner engagement lends powerful support to restoration projects, which is critical for meeting long-term goals. An additional aspect of partner involvement is often overlooked, however. This is the benefit partners receive in return through connection to such projects via on-the-ground restoration work. Urban green spaces do not have to be 'complete' for citizens to enjoy them; in fact, participating in early stages of ecological restoration can be a therapeutic experience. Volunteer planting and weeding provides a connection urban residents need with nature and with each other (Matsuoka and Kaplan 2008). Finally, investment by many in the early stages will help ensure continued investment and guardianship long-term.

Clear communication of progress throughout a project is important for continued partner engagement. A diversity of outputs tailored to a wide range of partners and end users is most effective. This may occur through updates in newsletters, websites or social media, or more interactive methods such as working bee gatherings, meetings, academic conferences or workshops (Clarkson and McQueen 2004). Acknowledgement of all who are involved and celebration of milestones will keep support strong and encourage continued communication and cooperation between multiple partners.

Conclusion

Restoration of the urban forest is a new and important initiative in many cities in New Zealand and beyond. Here we have reviewed ecological restoration research from Hamilton, New Zealand and summarised key findings to help inform successful restoration of urban forest ecosystems. By understanding the importance of species traits, filters and thresholds, restoration plantings can be designed using appropriate pioneer species, with the goal of crossing ecological thresholds like canopy closure. We have emphasised why increased native species richness of plantings is important for resilience, and that awareness about the indigenous cover on the surrounding landscape can be helpful. When choosing a reference ecosystem, urban forest remnants are useful targets for restoration as they indicate what species will tolerate urban conditions. In reviewing tree regeneration, seed banks and seed rain, and seed predation, we highlight why it is important to understand barriers to what late-successional species can and will regenerate spontaneously. When desirable late-successional species do not regenerate spontaneously, we encourage enrichment planting under the right environmental conditions. Finally, we encourage practitioners to remember to restore for both forest structure and ecological function.

We also discussed general principles determining urban restoration project success. First, that restoring to a minimum of 10% indigenous ecosystem cover in a city is a necessary target for maintaining a healthy level of native biodiversity. Secondly, forming a stepwise restoration plan with well-timed and comprehensive steps is important for efficient, sustainable project progression. Finally, we emphasised why creating and maintaining partner engagement is more important than ever when working in urban settings.

Despite the complexities of urban forest restoration, the benefits are manifold and worth the substantial effort. Native fauna like birds, lizards and indigenous bats are supported by restored urban forest habitat (Dekrout et al. 2014). Restored forests will have positive ramifications on the quality of our waterways by filtering water running into our rivers (Collier et al. 2008) and lakes (Duggan 2012), and cleaning air in our city streets (Pugh et al. 2012). Finally, urban forests enrich the lives of city citizens by providing green spaces for recreation, community involvement and connection with our precious biological heritage.

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ORCID

Kiri J. Wallace http://orcid.org/0000-0003-1236-6846

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