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Biodiversity and functional diversity of Australian stormwater biofilter plant communities

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1 Title: Biodiversity and functional diversity in Australian stormwater biofilter plant communities.

2
3 **Abstract**

4 Stormwater biofilters are an important part of the urban landscape in many Australian cities.

5 Until recently, plants, the most visible aspect of these systems and most green infrastructure,
6 have been selected primarily for their survivability and aesthetics. However, recent research has

7 identified specific species that enhance biofilter functions such as pollutant removal and flood
8 prevention (via infiltration). Prior to these findings, little attention was paid to developing

9 planting plans that included plant species with specific functional traits, such as specific root

10 length (SRL), percent fine roots (PFR), and relative growth rate (RGR); rather, biofilter planting

11 plans often suggest planting a relatively large number of plant species with the expectation that

12 some species will not survive due to competition and environmental factors. As these unsuited

13 species are lost, species diversity might be expected to be lower in older biofilters. However, it is

14 unknown whether biodiversity or functional diversity (i.e., the diversity of functional traits)

15 actually decreases as biofilters age. To investigate this question, we surveyed plant communities

16 in 32 biofilters along a chronosequence in Melbourne, Perth, and Sydney. From these data, we

17 calculated biodiversity and functional diversity indices from trait (i.e., SRL, PFR, and RGR) data

18 available in the literature for dominant plant species. We found that, although plant species

19 diversity is lower in older biofilters, functional diversity is unaffected by age. These trends

20 suggest biofilter plant communities maintain functional diversity despite losing biodiversity over

21 time. A better understanding of how plant functional traits relate to ecosystem functions would

22 let us design biofilters with better performance and value.

23

Keywords: biofilter function, urban stormwater systems, plant functional traits, functional diversity, urban ecology, water sensitive urban design

1. Introduction

Plants, the most visible aspect of most green infrastructure, play a significant role in the function of stormwater biofiltration systems (Read, Wevill, Fletcher, & Deletic, 2008; Bratieres, Fletcher, Deletic, & Zinger, 2008; Read, Fletcher, Wevill, & Deletic, 2009; Zhang, Rengel, Liaghati, Antoniette, & Meney, 2011; Payne et al., 2014). Stormwater biofilters are a type of green infrastructure (a.k.a, water sensitive urban design, sustainable urban drainage systems) comprised of planted soil-based filter media that is designed to treat urban runoff before either releasing to the receiving environment, typically 24–72 hours following the runoff event, or stored in a cistern for later use, typically for irrigation. Managing urban stormwater runoff using biofiltration can provide multiple ecosystem services (e.g., carbon sequestration, water quality improvement, urban heat mitigation, provision of biodiversity) (Hatt, Fletcher, & Deletic, 2009; Wong & Brown, 2009; Lundy & Wade, 2011; Grant et al., 2012), but we know very little about plant communities in these systems and how they change over time.

Optimizing biofilter design based on ecological theories has not been widely addressed (Levin & Mehring, 2015). The typically positive relationship between biodiversity and ecosystem function (Tilman et al., 1997; Balvanera et al., 2006; Mace, Norris, & Fitter, 2012) may be an important factor in designing ecosystems for specific purposes. Plants can act as ecosystem engineers in biofilters (Levin & Mehring, 2015), and specific plant traits are associated with particular ecosystem functions (Read et al., 2009). Plant species with varying morphologies, physiologies, and growth strategies can form complementary niches with varying effects on the surrounding environment (Levin & Mehring, 2015). In a biofilter, plants such as

Melaleuca ericifolia, a relatively deep-rooted shrub native to Australia, maintain infiltration rates over long periods of time (Le Coustumer, Fletcher, Deletic, Barraud, & Poelsma, 2012) while species like *Carex appressa*, an Australian sedge with a relatively high growth rate and long, fine roots, improve nitrogen removal more efficiently than other species (Read et al., 2009). These plants, with complementary functional traits, can be found growing in the same biofilter in Australia. Planting plans sometimes feature consideration of specific species known to enhance nutrient removal and infiltration, but rarely consider selecting species with specific functional traits explicitly (Payne et al., 2015). This is likely because these traits are not well documented and ecologists are rarely involved in designing green infrastructure (Cameron & Blanuša, 2016).

Plant communities composed of functionally divergent species or traits contain combinations of species that enhance productivity through complementary resource use (Díaz & Cabido, 2001; Petchey & Gaston, 2006). Other ecosystem functions, like pollutant removal and infiltration in biofilters, can be enhanced by specific traits (Payne et al., 2014), such as relative growth rate (Read et al., 2009) and root length (Le Coustumer et al., 2012), respectively. Plant biodiversity and functional diversity measures can be useful tools that managers can use to evaluate biofilters over time. One of these measures, the number of functional groups, is positively correlated to ecosystem function in green roof plant communities (Lundholm, MacIvor, MacDougall, & Ranalli, 2010). However, functional diversity indices that account for the diversity of species traits may be more informative than functional group richness for determining whether a system provides functions that result in ecosystem services (Lundholm, Tran, & Gebert, 2015). Different functional groups may share similar traits that are functionally relevant, so the number of different functional groups does not necessarily account for these similarities. “Functionally redundant” species, those species sharing functional group

classifications, may have important roles in community dynamics (Cadotte, Cavender-Bares, Tilman, & Oakley, 2009), but their presence is not accounted for by counting the total number of functional groups. Additionally, the number of functional groups as a measure of functional diversity can only provide an arbitrary scale for assessing diversity (Petchey & Gaston, 2002). For example, in some classification of functional groups, the difference between trees and grasses is equal to the difference between sedges and grasses. Consequently, functional diversity indices that are based on relative differences between functional traits have been developed (Walker, Kinzig, & Langridge, 1999; Petchy & Gaston, 2002). While functional diversity along an urbanization gradient in green infrastructure has been documented using multiple taxa and functional groups selected based on sensitivity to urbanization (Pinho et al., 2016), the authors found no other studies addressing plant functional diversity in green infrastructure along a chronosequence and no studies investigating plant communities in biofilters, other than in relation to habitat provided for invertebrates (Kazemi, Beecham, & Gibbs, 2011).

In this study, we characterized plant communities in various aged biofilters located in three Australian cities with unique climates. To achieve this, we conducted plant surveys and collected root samples in 32 biofilters in three Australian cities: Melbourne, VIC; Perth, WA; and Sydney, NSW. We calculated biodiversity and functional diversity indices and analyzed these data for correlation to biofilter age. Functional diversity indices were calculated using functional traits (i.e., specific root length, percent fine roots, and relative growth rate) previously found to affect pollutant removal in biofilters. Because species that are not well suited for specific biofilter conditions were likely lost over time, biodiversity could be expected to decrease over time. On the other hand, some species may be gained through natural recruitment, so biodiversity could increase over time. Similarly, functional diversity could increase or decrease

over time because it is often highly correlated to biodiversity. However, because the responsibility of maintenance of biofilter plant communities fall on multiple parties and is often unclear (Ambrose & Winfrey, 2015), we were not able to predict how biofilter plant communities in older systems may be managed compared to newer systems. Consequently, we were not confident that we would find any biodiversity or functional diversity patterns along our chronosequence. Through this work, we aim to describe biodiversity patterns in biofilters to provide a detailed account of how these plant communities change over time and provide insight that could lead to improved management strategies and design that enhances pollutant removal and ecological value.

2. Methods

2.1. Site Selection

In each city, biofilter sites were chosen from a list of biofilters compiled from published accounts and personal communications with municipal officials. Biofilters were selected to represent a range of ages, but maintain consistent design specifications regarding planting density, catchment ratio, and hydraulic loading. Twelve biofilter sites were sampled in Melbourne, 11 in Perth, and 9 in Sydney (Figure 1).

Mean annual rainfall (MAR) for each site was determined using the average annual precipitation measured at the closest rain gauge operated by the Australian Government's Bureau of Meteorology for the period of time between the year of construction of the biofilter to the sampling date. When data were not available for that time period, we used rainfall data from the next closest rain gauge. All rain gauges were located within 10 km of the closest biofilter.

2.1.1. Site Location Information

On average, the 12 biofilter sites sampled in Melbourne, VIC received MAR of 767 mm (Bureau of Meteorology, 2015) during the time between biofilter construction and sampling (Table 1). Seasonally, rainfall was greater in winter months and lower in summer months; average monthly rainfall ranged from about 47 mm in January to 65 mm in October (Bureau of Meteorology, 2015). The selected study sites ranged in age (period of time between construction and date of sampling in October, 2014) from 1.5 to 12 years. Median biofilter age and area were 3.4 years and 24 m², respectively. Representative photographs of sites samples in Melbourne are provided in Appendix A1.

The 11 biofilter sites sampled in Perth, WA received an average MAR of 738 mm (Bureau of Meteorology, 2015) during the time between biofilter construction and sampling. Typical of Mediterranean climates, rainfall was very low in summer months, with most rainfall occurring during winter months; average monthly rainfall ranged from about 10 mm in January to 160 mm in June (Bureau of Meteorology, 2015). The selected sites ranged in age (period of time between construction and date of sampling in November, 2014) from 1.5 to 9 years. Median biofilter age and area were 5.5 years and 200 m², respectively. Representative photographs of sites samples in Perth are provided in Appendix A2.

Nine biofilter sites were sampled in Sydney, NSW. These sites received MAR of 1316 mm (Bureau of Meteorology, 2015) during the time between biofilter construction and sampling. Although more rainfall occurred in winter months than in summer, rainfall was relatively abundant throughout the year, with average monthly rainfall ranging from 70-80 mm in September-December to 130 mm in June (Bureau of Meteorology, 2015). These sites ranged in age (period of time between construction and date of sampling in November, 2014) from 1.8 to

14 years. Median biofilter age and area were 5.3 years and 42 m², respectively. Representative photographs of sites samples in Sydney are provided in Appendix A3.

2.2. Root Characteristics

The following methods were used to quantify the areal densities of root mass (*RM*, g/m²) and root length (*RL*, m/m²) for each biofilter site. Plant root samples were analyzed from filter media cores collected at each site at a sampling frequency of one sample per, 200 m² of biofilter area or five samples, whichever was greater. Within biofilters, at each randomly selected sampling location, we collected cores by driving a 2.5-cm diameter chromium-molybdenum steel soil probe to rooting depth (10–30 cm below soil surface) and emptied the probe's contents into a plastic zip-lock bag. Holes made from the steel probe were backfilled with fine sand and existing surrounding material to minimize disturbance to biofilter infiltration. Bagged samples were stored at 4°C for less than 24 hours before filter media was hand-washed from roots through a 600-μm sieve.

Prior to drying, roots were stored at 4°C until analyzed for root length (all samples stored fewer than 72 hours). The modified grid-line intersect method (Tennant, 1975) was used to determine root length. Briefly, we arranged fresh root segments randomly on a 9-cm diameter petri dish marked with 1.3-cm (0.5-in.) vertical and horizontal gridlines. Next, we examined the roots under a dissecting microscope and counted all intersections of roots with each orientation of gridline. The total number of intersections using this grid size provides an estimate of the root length in cm (Marsh, 1971). Root samples were then dried at 60°C for 48 hours and weighed to get root mass (g). *RM* and *RL* were calculated by dividing root mass and root length, respectively, by the cross-sectional area of the soil probe (4.9 cm²).

2.3. Plant Species Survey and Plant Diversity Indices

Plant surveys were completed at each biofilter site by visually estimating the cover of each plant species for the entire site. One sampler performed all cover estimates to ensure consistency across sites. Plants were identified to species where possible, otherwise to genus. For plants identified to species, we determined whether species were native to Australia using online databases. For sites larger than 250 m², one 0.25-m² quadrat was randomly placed for every ~125 m² of biofilter, with the mean of the visually estimated cover within quadrats used to estimate plant cover of each species for the entire site.

Species richness (S) was determined by counting the number of species for each site. The Shannon Diversity index (H') was calculated using Equation 1.

$$H' = - \sum_{i=1}^S P_i \ln[P_i] \quad (1)$$

Where P_i is equal to the ratio of the cover of individual plant species (i) to the total cover of all plant species.

2.4. Plant Functional Diversity Indices

To evaluate plant functional diversity, we calculated three indices: number of functional groups (FG), Functional Diversity (FD), and Functional Attribute Diversity (FAD). To calculate FG , we categorized surveyed plant species into the following growth form types to represent functional groups: grasses, sedges, rushes, forbs, shrubs, and trees. We selected these plant growth forms to represent functional groups due to their intrinsic differences in morphology and growth strategies. Read et al. (2008) similarly categorized plant species they selected to represent vegetation in Australian biofilters in a column study.

Functional Diversity (*FD*) and Functional Attribute Diversity (*FAD*) are two indices that quantify the extent of diversity of species' functionalities present in ecological communities (Walker et al., 1999; Petchy & Gaston, 2002). These two indices can be sensitive to which traits are included (Cadotte et al., 2009). By selecting specific traits related to specific ecosystem functions, *FD* and *FAD* can be used to evaluate a plant community's potential role in affecting a specific ecosystem function. Functional Diversity and Functional Attribute Diversity were determined for each site using plant survey results and plant functional trait data from studies that described traits of plants grown in conditions similar to those present in stormwater biofilters (Appendix B, Read et al., 2009; Payne, 2013; Pham, 2016). We calculated *FD* and *FAD* using functional group type and three specific traits that have been positively correlated with pollutant removal and infiltration, two primary functions of stormwater biofilters that can diminish with age (Hatt, Fletcher, & Deletic, 2009): relative growth rate, specific root length, and percent fine roots. These functional diversity indices were calculated for the biofilters described in Section 2.1. These indices were calculated assuming only dominant plant species were functionally relevant, based on the 'mass ratio' theory by Grime (1998). Consequently, we only included plants with greater than 10% cover, which excluded ruderal species that were unlikely to have significant effects on biofilter function and include intended and established species. *FD* and *FAD* have been used to improve planting plans for green roofs (e.g., Van Mechelen, Van Meerbeek, Dutoit, & Hermy, 2015) and evaluate functional diversity in a number of natural and managed ecosystems (e.g., Mokany, Ash, & Roxburgh, 2008; Cadotte et al., 2009; Sabatini, Burton, Scheller, Amatangelo, & Mladenoff, 2014; van der Walt, Cilliers, Toit, & Kellner, 2015). Trait data for these species were scaled to a mean of zero and variance of one in order to determine *FD* (Petchy & Gaston, 2002) and *FAD* (Walker et al., 1999). We constructed

functional dendrograms for each site using a dissimilarity matrix of hierarchically clustered Euclidian distances between species' scaled functional traits (Petchy & Gaston, 2002). We used the unweighted pair-group method with arithmetic means (UPGMA) algorithm to hierarchically cluster species on functional dendrograms (Petchy & Gaston, 2002). The sum of the branch lengths of each dendrogram was used to calculate *FD* at each site (Petchy & Gaston, 2002). *FAD* was calculated by summing the Euclidian distances in the dissimilarity matrix constructed from functional traits of species present at each site (Walker et al., 1999). We calculated these indices using R (R Core Team, 2016) code provided by Cadotte et al. (2009), which requires the 'ape' package (Paradis, Claude, & Strimmer, 2004).

2.5. Data Analysis

We tested normality of observed data distributions using the Shapiro-Wilk normality test ($\alpha=0.05$). After confirming normality on observed or log-transforming data (all data were normally or log-normally distributed), we removed data points that failed the Grubbs' Test for Outliers (two-sided test for opposite outliers, $\alpha=0.05$). One data point was removed from analysis of root mass (Parkfield Lake biofilter, Perth,, 2077 g/m²) based on the Grubbs' Test for Outliers. We did not find any other outliers in our dataset. On the resulting dataset, we performed Pearson's correlation tests ($\alpha=0.05$) to evaluate the relationship between biofilter characteristics (i.e., age, area, and mean annual rainfall) and ecosystem structure measures. We used one-way ANOVAs to test whether the measured variables were affected by biofilter location (city). Statistical analyses were performed using R Statistical Software (R Core Team, 2016).

3. Results

3.1. Root Characteristics

Perth ($430 \pm 360 \text{ m/m}^2$) biofilters contained more than twice the average root length per unit area as Melbourne ($190 \pm 140 \text{ m/m}^2$) and Sydney ($170 \pm 160 \text{ m/m}^2$) biofilters (ANOVA, $p = 0.036$). There were no significant differences in average root mass areal densities between biofilters located in Melbourne ($420 \pm 240 \text{ g/m}^2$), Perth ($750 \pm 590 \text{ g/m}^2$) and Sydney ($520 \pm 413 \text{ g/m}^2$). One species, *Ficinia nodosa*, was present in multiple biofilters in all cities and had greater root length in Perth biofilters ($435 \pm 279 \text{ m/m}^2$) than in Melbourne ($156 \pm 70 \text{ m/m}^2$) and Sydney ($206 \pm 166 \text{ m/m}^2$) biofilters, but not significantly (ANOVA, $p = 0.062$). There were no correlations between root characteristics and biofilter age or area.

3.2. Biodiversity and Functional Diversity

Most biofilter plant species belonged to four families: Cyperaceae, Juncaceae, Poaceae, and Myrtaceae (Table 2). There were a total of 56 species in 19 families across the surveyed biofilters, with 12 species and 11 families present in biofilters in more than one city (Table 2). There were 30, 25, and 19 plant species in Melbourne, Perth, and Sydney biofilters, respectively. Of the plants identified to species, 73, 88, and 82% of species were native to Australia in Melbourne, Perth, and Sydney biofilters, respectively (Table 2). Eighty percent of all plants identified to species in our surveys were Australian natives.

Growth form types used to represent functional groups (FG) included grasses, sedges, rushes, forbs, shrubs, and trees (Table 3). In Melbourne, Perth, and Sydney, there were a total of 27, 29, and 23 dominant species representing each of these functional groups, respectively. There was no significant difference between the number of functional groups at each site between cities, with Melbourne, Perth, and Sydney biofilters containing on average 1.8, 2.5, and 2.0 functional groups, respectively (Table 4). Rushes, sedges, and forbs were the most common functional groups, with 84% of sites containing at least one of these groups. Only one biofilter,

Clifton Hill in Melbourne, contained a grass. Sydney biofilters had fewer rushes than Melbourne and Perth biofilters. Perth biofilters had fewer forbs and more shrubs and trees than Melbourne and Sydney biofilters. Shrubs and trees comprised about 15% of functional groups found in all biofilters. Sedges and forbs were the most redundantly represented functional groups, meaning there were more sites that contained two or more sedge and/or forb species than other groups. Overall, about 1 out of every 3 sites contained redundant species within functional groups (Table 3).

The hierarchical clustering of all biofilter species according to functional traits resulted in a dendrogram with total branch length of 11.99 (Figure 2). That is, if all species were present in one biofilter, *FD* would equal 11.99. The clustering algorithm generally arranged plant species of the same functional group within clades, but not in all cases or at all sites (Figure 2, Appendix C). We calculated the *FD* for each biofilter site using dendrograms that were constructed using only species present in that biofilter. Functional Diversity ranged from 0.09 (Wolseley Grove, Sydney, and Cremorne St., Melbourne, Table 3) to 6.62 (Edinburgh Gardens, Melbourne, Table 4). The lowest *FD* (0.09) calculated (for biofilter sites at Wolseley Grove and Cremorne St.) was equal to the branch length of a single species present, *Lomandra longifolia*. The highest *FD* (6.62) was equal to the sum of the branch lengths of the dendrogram for the site Edinburgh Gardens (Appendix C) that contained *Carex appressa*, *Dianella revoluta*, *Ficinia nodosa*, *Juncus flavidus*, and *Lomandra longifolia*. Attribute Diversity, the sum of Euclidian distances between species in functional trait space, ranged from 3.22 (Alcock St., Perth, Table 4) to 24.2 (Edinburgh Gardens, Table 4). There were no significant differences between biofilters grouped by location (city) for any variable other than root length (Table 4).

Older biofilters tended to have fewer species ($r = -0.38$, $p = 0.03$; Figure 3) and lower species diversity ($r = -0.52$, $p = 0.003$). The correlations between biofilter age and functional trait diversity indices were negative but not statistically significant (FG , $p = 0.12$; FD , $p = 0.09$; FAD , $p = 0.21$). Species richness was positively correlated to Shannon diversity (H' , $r = 0.83$, $p < 0.001$) and two of the three functional diversity indices (Figure 3; FG , $r = 0.67$, $p < 0.001$; FD , $r = 0.46$, $p = 0.009$; FAD , $p = 0.23$). Shannon diversity was positively correlated to all functional diversity indices (Figure 3; FG , $r = 0.73$, $p < 0.001$; FD , $r = 0.62$, $p < 0.001$; FAD , $r = 0.44$, $p = 0.01$). Biofilters with higher Functional Diversity (FD) had more plant cover ($r = 0.39$, $p = 0.03$). Older biofilters also tended to be smaller ($r = -0.37$, $p = 0.04$).

4. Discussion

Overall, we found that biofilter plant communities in three Australian cities were similar in ecological structure. Root length did vary by city, but biodiversity did not. Biofilter plant communities did lose species diversity along the chronosequence, but all measures of functional diversity were unaffected by biofilter age.

Root length density is strongly linked to drought tolerance in plants (Gowda, Henry, Yamauchi, Shashidhar, & Serraj, 2011; Vadez, 2014). Greater root length areal density was found in biofilter plants in Perth systems, where much longer periods between rain events occur compared to Melbourne and Sydney (Winfrey, Hatt, & Ambrose, 2017). Root length did not differ between Melbourne and Sydney, despite Sydney receiving roughly twice as much annual rainfall. Both Melbourne and Sydney receive rainfall relatively evenly throughout the year and stormwater biofilters in these areas often receive about 25–100 times more water than the rainfall depth by design (Ambrose & Winfrey, 2015). Consequently, it appears plants in stormwater biofilters located in Melbourne and Sydney do not need to develop longer plant roots in

comparison to those located in Perth, which had similar rainfall depth to Melbourne, but much longer periods between rain events. Although the comparison was not statistically significant, *F. nodosa* present in Perth biofilters had more than twice the root length as *F. nodosa* in Melbourne and Sydney biofilters.

Root mass and root length were not correlated to plant cover or plant community diversity indices. Because plant cover is not a good predictor of root mass in these systems, plant cover may not be a good indicator of diminished hydraulic conductivity (clogging). Clogging can be a problem in older biofilters (Hatt, Fletcher, & Deletic, 2009). In theory, an accumulation of roots could increase biofilter clogging by forming a mat of fine roots that trap sediment (Archer, Quinton, & Hess, 2002). Despite this, there were no significant correlations between root characteristics and biofilter age in any city. Because clogging does occur in older biofilters (Hatt, Fletcher, & Deletic, 2008), these results may support the argument that clogging in older biofilters is due to the accumulation of fine particulate matter from sediment loading in the upper filter bed layer (Mousavi & Rezai 1999; Hatt, Fletcher, & Deletic, 2009; Le Coustumer et al. 2012) rather than mats of fine roots near to surface. Accordingly, Hatt, Fletcher, & Deletic (2008) suggest to scrape the top 2–5 cm of surface layer to remove accumulated sediment and increase infiltration.

For most sites, functional dendrograms clustered similar functional groups together in clades (e.g., Edinburgh Gardens, Spring St., etc., Appendix C), but a few plant communities contained a tree (*Melaleuca ericifolia*) and a sedge (*Ficinia nodosa*) within the same clade (e.g., Calder Nook, Kirkland Way, etc., Appendix C). These functional groups would likely cluster into clades better by increasing the number of plant traits used to construct the dissimilarity matrix. For instance, if plant height were included, trees and sedges would be less likely to

appear in the same clade. However, we were only interested in evaluating functional diversity of plant traits relevant for pollutant removal in this study. Given the differences between trees and herbaceous species in growth and uptake phenologies, we may have selected different or additional traits to analyze for functional diversity that better represent the roles of biofilter trees in pollutant removal. For instance, trees with higher overall leaf area may intercept more rainfall intrinsically, but not necessarily in practice if these trees are pruned to maintain aesthetics, do not retain their foliage during the rainy season, or if they remain relatively small throughout their lifetime (Xiao & McPherson 2002).

At the ecosystem scale, increased biodiversity can enhance the provision of ecosystem services such as productivity, erosion control, nutrient cycling, and regulation of invasive species in natural ecosystems (Balvanera et al., 2006; Mace, Norris, & Fitter, 2012). Ecosystem services in built ecosystems can also be enhanced by increased biodiversity (Lundholm et al., 2010; Lundholm & Williams, 2015; Lundholm, 2015), but not in every case (Means, Ahn, Korol, & Williams, 2016). Functional trait diversity appears to better predict ecosystem service provision (Lundholm et al., 2015). Although average plant species richness and diversity were similar at sites in all three cities in this study (Table 4), the total number of species in biofilter plant communities in each city varied, with Melbourne having considerably more total species than Sydney (30 versus 19 species). Additionally, despite having a lower percentage of native species, Melbourne biofilters had nearly twice as many native species in total than in Sydney (Table 2).

At the landscape scale, biofilters in Melbourne may be contributing more than those in Sydney to enhancing native biodiversity in urban areas. However, it is difficult to connect increased biodiversity to ecosystem service provision at a spatial scale relevant to society due to the heterogeneity of the environment at these scales (Kremen, 2005).

Plant community structure is inextricably linked to management and, at least in newer biofilter plant communities, initial plantings. Consequently, we were unlikely to detect any significant relationship between plant community characteristics and independent variables like age and area. These factors may explain the lack of significant correlation between total plant cover and most diversity indices (Figure 3). Nevertheless, total plant cover was positively correlated to *FD*, which suggests functionally diverse systems may be more productive, even in these heavily managed systems.

Regarding the trend of lower species diversity in older biofilters, one may consider assembly rules for plant community succession in stormwater biofilters (Levin & Mehring, 2015). Plant species from surrounding areas may colonize biofilters while plants that previously colonized or were initially planted may not persist due to competition and/or exclusion via abiotic conditions (e.g., drought, flooding, undeveloped soil matrix, etc.) and maintenance (e.g., pruning and manual removal). Because of these factors, the assembly rules for stormwater biofilters could predict a decrease in plant species over time (Keddy, 1992). The species pool for a biofilter largely depends on the species initially planted, species in the surrounding landscape, and those present in the seed bank of the imported, engineered filter media. Plant species diversity in other types of green infrastructure has been shown to decrease with age in some studies (e.g., green roofs: Rowe, Getter, & Durhman, 2012; and Lundholm, Heim, Tran, & Smith, 2014; and constructed wetlands: Noon, 1996), while others found no effect of age on species diversity (Köhler, 2006; Madre, Vergnes, Machon, & Clergeau, 2014). Noon (1996) suggested that two phases of community succession occur in constructed wetlands. Establishment of plant species through seed dispersal, propagation from the seed bank, and human intervention dominate the first phase. Species-related processes dominate the second

phase of succession (Noon, 1996). However, instead of species-related processes dominating the second phase of succession in biofilters, human intervention likely dominates more than other factors. Indeed, stormwater biofilter plants are often managed throughout their lifespan via pruning, weeding, and replanting multiple times per year (Payne et al., 2015).

Recent and current guidelines suggest planting biofilters with a large number of species (FAWB, 2008; Payne et al., 2015), but there is little information about initial plantings in older biofilter design. If older biofilters were initially planted with fewer species than newer ones, this could help explain the trend we found that fewer species were present in older biofilters. This would be an artifact of our chronosequence approach. Still, whether species were filtered out environmentally or initial plantings in older biofilters included fewer species, this pattern persists despite management efforts, which were unlikely to remove species from biofilters without replacing them.

Contrary to species diversity, functional diversity indices were not correlated to biofilter age. This could suggest that, while species diversity decreases in older biofilters, functional diversity is maintained. Common management practices in Australian biofilters (e.g., pruning, weeding, replanting, and mulching) appear not to decrease functional diversity of biofilter plant communities over time. Interestingly, newer biofilters did not have higher functional diversity than older biofilters despite recent changes to planting plan guidelines that suggest using species that specifically promote better biofilter functioning (Payne et al., 2015).

Maintaining these biofilter plant communities appears to be promoting the retention of functional diversity. Vegetation maintenance protocols for Australian biofilters require frequent monitoring, assessment, and maintenance of plant health and cover over the life of the biofilter (Payne et al., 2015). When unhealthy or dead plants are observed, they are replaced only after the

cause has been addressed, which is typically related to system hydraulics (Payne et al., 2015). Consequently, the presence of intentionally planted species is linked to the biofilter function of stormwater capture through a common maintenance practice. To preserve aesthetics, weed species are removed from biofilters routinely (Payne et al., 2015). In Australian stormwater wetlands, intentionally planted vegetation that is well maintained confers a healthy ecosystem according to public perception (Dobbie, 2013). Consequently, aesthetic value can be preserved in biofilters by maintaining the appearance of intentional care (Gobster, Nassauer, Daniel, & Fry, 2007). Overall, management of biofilter vegetation appears to maintain functional diversity through facilitating the health and survival of intentionally planted species and removing those that may degrade biofilter function and aesthetics.

To better understand the relationship between plant traits and biofilter functions, these should be measured *in situ*. To identify which plant traits or combinations of traits are correlated to specific biofilter functions, plant communities should be surveyed for plant functional traits that were not included in this study, such as root and leaf characteristics, above- and belowground biomass, nutrient content, and canopy density. Biofilter functions, such as infiltration, carbon storage, pollutant removal, and microclimate regulation, should also be measured. Information on the relationships between plant traits and ecosystem function in these systems can improve design guidelines for planting plans and maintenance across climates and regions by providing guidance on developing planting plans that enhance ecosystem services through enhanced biofilter function. Biofilters are a common landscape feature in Australian cities that provide multiple benefits linked to ecological structure. Understanding plant functional traits as they relate to ecosystem functions would let us design biofilters with better performance and value. However, as stormwater managers and landscape planners increasingly work together

413 to design and manage urban ecosystems, goals of enhancing biodiversity to enhance landscape-
414 scale benefits will need to be balanced with goals of increasing functional diversity to enhance
415 ecosystem-scale benefits.

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Table 3. Functional groups present at each site. Numbers indicate number of species present at site belonging to each functional group.

Table 4. Diversity indices and biofilter characteristics. Abbreviations: *RM* = root mass areal density, *RL* = root length areal density, *S* = species richness, *H'* = Shannon Diversity index, *FG* = number of functional groups, *FD* = Functional Diversity, *FAD* = Functional Attribute Diversity.

Table 1. Biofilter site characteristics. Mean annual rainfall was calculated for the period of time between the year of construction of the biofilter to the sampling date.

Biofilter Site ID	Mean Annual Rainfall (mm)	Age (yrs.)	Area (m ²)
Melbourne (average $\pm$ SD)	767 $\pm$ 257	5.4 $\pm$ 3.9	133 $\pm$ 224
Alleyne Ave 1	633	9	76
Alleyne Ave 2	613	3	12
Avoca Cr	511	10	13
Clifton Hill	558	7	100
Cremorne St	619	12	15
Edinburgh Gardens	516	3	700
Fernhill Rd	1,041	2	10
Hereford Rd	1,196	4	100
Morrison Reserve	1,042	2	500
Parker St	511	10	22
Spring St	983	2	25
Stringybark Blvd N	983	2	25
Perth (average $\pm$ SD)	738 $\pm$ 86	4.8 $\pm$ 2.2	428 $\pm$ 581
Alcock St	804	4	135
Barlee St	644	5	75
Calder Nook	672	6	240
Channel View	674	9	200
Kirkland Way	737	3	350
Mead St	935	1	1,000
Parkfield Lake	803	2	2,000
Sotheby Dr	735	6	150
Splendid Gardens	735	6	300
Strelly St	644	5	105
Welcome Meander	735	6	150
Sydney (average $\pm$ SD)	1316 $\pm$ 110	5.6 $\pm$ 3.7	73 $\pm$ 69
Bay St	1,207	5	120
Birubi Ave	1,411	5	18
Dawson Ave	1,123	2	200
Karuah Rd	1,430	7	17
Kooloona Cr	1,450	6	24
Marriott St	1,311	3	130
Normurra Ave	1,364	8	1.5
Wolseley Grove	1,236	14	107
Young St	1,311	3	42

Table 2. Plant species list for all sampled biofilters. Presence of plant species in city is designated by “x”. Plants native to Australia and references for native status are shown in the last two columns. Functional groups indicated by letters: F = forbs; G = grasses; R = rushes; S = sedges; Sh = shrubs; T = trees. This table has been adapted from Winfrey, Hatt, & Ambrose (2017).

Family	Species Name	Melbourne	Perth	Sydney	Native to Australia?	Reference	Functional Group
Acanthaceae	<i>Hygrophyla costata</i>	x			No	a	F
Amaryllidaceae	<i>Agapanthus</i> sp.			x	—*		F
Asteraceae	<i>Heterotheca grandiflora</i>	x			No	b	F
	<i>Leucophyta brownii</i>		x		Yes	c	Sh
	<i>Sonchus oleraceus</i>	x			No	b	F
	<i>Taraxacum</i> sp.	x	x	x	—		F
Cyperaceae	<i>Baumea articulata</i>		x		Yes	c	S
	<i>Baumea juncea</i>		x		Yes	d	S
	<i>Carex appressa</i>	x		x	Yes	d	S
	<i>Carex fascicularis</i>	x			Yes	d	S
	<i>Carex tereticaulis</i>	x			Yes	d	S
	<i>Cyperus</i> sp. 1	x			—		S
	<i>Cyperus</i> sp. 2		x		—		S
	<i>Eleocharis acuta</i>		x		Yes	d	S
	<i>Ficinia nodosa</i>	x	x	x	Yes	d	S
	<i>Gahnia trifida</i>		x		Yes	d	S
	<i>Lepidosperma longitudinale</i>		x		Yes	d	S
Euphorbiaceae	<i>Euphorbia</i> sp.	x	x	x	—		F
Fabaceae	<i>Trifolium</i> sp.	x	x	x	—		F
Goodeniaceae	<i>Goodenia ovata</i>	x			Yes	e	Sh
Iridaceae	<i>Iris</i> sp.	x			—		F
Juncaceae	<i>Juncus amabilis</i>	x			Yes	d	R
	<i>Juncus australis</i>	x			Yes	e	R
	<i>Juncus flavidus</i>	x		x	Yes	d	R
	<i>Juncus kraussii</i>		x		Yes	c	R
	<i>Juncus pauciflorus</i>		x		Yes	d	R
	<i>Juncus subsecundus</i>		x		Yes	d	R
Lamiaceae	<i>Lycopus australis</i>	x			Yes	e	F
	<i>Westringia fruticosa</i>			x	Yes	c	Sh
Myrtaceae	<i>Callistemon phoeniceus</i>			x	Yes	f	Sh
	<i>Eucalyptus</i> sp. 1		x		—		T
	<i>Eucalyptus</i> sp. 2			x	—		T
	<i>Leptospermum</i> sp.	x		x	—		T
	<i>Lophostemon confertus</i>	x			Yes	c	T
	<i>Melaleuca ericifolia</i>	x		x	Yes	e	T
	<i>Melaleuca nesophila</i>		x		Yes	f	T
	<i>Melaleuca thymifolia</i>	x			Yes	f	Sh

Onagraceae	<i>Gaura lindheimeri</i>	x			No	a	F
Plantaginaceae	<i>Plantago</i> sp.	x		x	–		F
Poaceae	<i>Alopecurus pratensis</i>		x		No	b	G
	<i>Austrodanthonia</i> sp.		x		–		G
	<i>Cynodon dactylon</i>	x	x	x	No	b	G
	<i>Microlaena stipoides</i>	x			Yes	b	G
	<i>Pennisetum setaceum</i>			x	No	b	G
	<i>Poa labillardierei</i>	x			Yes	c	G
	<i>Sporobolus virginicus</i>	x			No	b	G
Polygonaceae	<i>Polygonum</i> sp.	x			–		F
Proteaceae	<i>Banksia</i> sp.			x	–		T
	<i>Grevillea</i> sp.		x		–		Sh
	<i>Hakea francisiana</i>		x		Yes	f	T
	<i>Isopogon formosus</i>		x		Yes	c	Sh
Rutaceae	<i>Correa</i> sp.		x		–		Sh
Scrophulariaceae	<i>Myoporum parvifolium</i>		x		Yes	e	Sh
Xanthorrhoeaceae	<i>Dianella revoluta</i>	x	x	x	Yes	c	F
	<i>Lomandra hystrix</i>			x	Yes	c	F
	<i>Lomandra longifolia</i>	x		x	Yes	c	F
Total number of species:		30	25	19			
Percent native:		73%	88%	82%	(All species: 80%)		

*We did not determine native statuses of plants that were identified to genus only.

a. USDA (2017)

b. CABI (2017)

c. ANBG (2017)

d. Western Australian Herbarium (2017)

e. Royal Botanic Gardens and Domain Trust (2017)

f. ANPSA (2017)

Table 3. Functional groups present at each site. Numbers indicate number of species present at site belonging to each functional group.

Biofilter Site ID	Functional Groups						# FG within sites
	Grass	Rush	Sedge	Forb	Shrub	Tree	
Melbourne							
Alleyne Ave 1				1			1
Alleyne Ave 2			1				1
Avoca Cr			1				1
Clifton Hill	1	1			1		3
Cremorne St				1			1
Edinburgh Gardens		1	2	2			3
Fernhill Rd		1					1
Hereford Rd		1	1			1	3
Morrison Reserve		1	1	2			3
Parker St			1				1
Spring St		2	2	1			3
Stringybark Blvd N		1					1
# of species within FG	1	8	9	7	1	1	
Perth							
Alcock St			1			1	2
Barlee St		1	1				2
Calder Nook		1	1		1		3
Channel View		1	1		2		3
Kirkland Way		1	1			1	3
Mead St		1	1		1		3
Parkfield Lake		1					1
Sotheby Dr		1	1				2
Splendid Gardens		1	1			1	3
Strelly St		1	1	1			3
Welcome Meander		1	1			1	3
# of species within FG	0	10	10	1	4	4	
Sydney							
Bay St			2	1			2
Birubi Ave			1				1
Dawson Ave		1	2			1	3
Karuah Rd		1		1			2
Kooloona Cr			2	1			2
Marriott St			1	2			2
Normurra Ave		1	1	1			3
Wolseley Grove				1			1
Young St			1	2			2

<i># of species within FG</i>	<i>0</i>	<i>3</i>	<i>10</i>	<i>9</i>	<i>0</i>	<i>1</i>
# of species within FG for all cities	1	21	29	17	5	6

Table 4. Diversity indices and biofilter characteristics. Abbreviations: *RM* = root mass areal density, *RL* = root length areal density, *S* = species richness, *H'* = Shannon Diversity index, *FG* = number of functional groups, *FD* = Functional Diversity, *FAD* = Functional Attribute Diversity.

Biofilter Site ID	<i>RM</i> (g/m ²)	<i>RL</i> (m/m ²)	<i>S</i>	<i>H'</i>	<i>FG</i>	<i>FD</i>	<i>FAD</i>
Melbourne (average ± SD)	420± 240	240± 192	4.42± 3.9	0.74± 0.8	1.8 ± 1.0	2.24± 2.3	9.29± 5.7
Alleyne Ave 1	419	152	1	0 ^a	1	— ^b	— ^c
Alleyne Ave 2	345	455	2	0.64	1	0.60	7.75
Avoca Crescent	365	58	1	0	1	0.60	7.75
Clifton Hill	287	437	10	1.45	3	4.33	7.25
Cremorne St	489	16	1	0	1	0.09	3.93
Edinburgh Gardens	1,080	60	5	1.52	3	6.62	24.17
Fernhill Rd	587	157	1	0	1	0.29	5.63
Hereford Rd	289	206	5	1.24	3	4.01	6.04
Morrison Reserve	372	310	8	1.48	3	4.61	13.30
Parker St	405	112	1	0	1	0.60	7.75
Spring St	72	202	12	2.03	3	2.61	12.94
Stringybark Blvd N	370	142	6	0.51	1	0.29	5.63
Perth (average ± SD)	750± 590	430± 363	4.91± 1.9	0.98± 0.5	2.5± 0.7	2.91± 1.6	6.37± 3.2
Alcock St	35	740	2	0.11	2	3.60	3.22
Barlee St	946	452	7	1.53	2	0.60	7.75
Calder Nook	591	1,232	4	0.58	3	0.60	7.75
Channel View	446	759	5	1.00	3	5.19	10.25
Kirkland Way	654	273	4	0.64	3	3.60	3.72
Mead St	436	213	8	1.47	3	2.30	3.65
Parkfield Lake	2,077	101	3	1.04	1	1.80	11.19
Sotheby Dr	318	552	3	0.92	2	3.60	3.72
Splendid Gardens	301	132	7	1.29	3	5.19	10.87
Strelly St	1,429	32	6	1.57	3	2.30	4.07
Welcome Meander	967	252	5	0.61	3	3.18	3.93
Sydney (average ± SD)	520± 413	168± 161	5.00± 1.3	0.87± 0.5	2.0± 0.7	3.08± 2.2	6.53± 4.1
Bay St	407	109	6	1.14	2	4.43	9.47
Birubi Ave	163	505	4	0.24	1	0.70	4.11
Dawson Ave	728	64	6	1.44	3	6.02	13.17
Karuah Rd	229	88	4	0.66	2	0.29	5.63
Kooloona Cr	1,384	10	7	0.67	2	3.60	1.31
Marriott St	260	190	5	1.30	2	5.19	9.64
Normurra Ave	810	220	4	0.82	3	4.20	9.70
Wolseley Grove	629	14	3	0.06	1	0.09	3.93
Young St	82	314	6	1.49	2	3.18	1.80

^aDiversity index is equal to zero for sites with species richness less than 2 (i.e., monocultures have no diversity).

^bFunctional Diversity could not be calculated for this site because plant trait data were not available for the dominant species present in this plant community (*Iris* sp.)

^cFunctional Attribute Diversity could not be calculated for this site because plant trait data were not available for the dominant species present in this plant community (*Iris* sp.)

List of Figures:

Figure 1. Location map of biofilters in Melbourne, Perth, and Sydney, Australia. Created using Google Maps.

Figure 2. Functional trait dendrogram of all dominant (>10% cover) plant species in all sampled biofilters, except Alleyne Avenue 1, where *Iris* sp. was dominant but trait data were unavailable. Groupings are based on functional similarities in trait matrix. Letters in parentheses correspond to the following functional group classifications: forb (f), grass (g), rush (r), sedge (s), shrub (sh), and tree (t).

Figure 3. Correlogram of biofilter characteristics (age, area, mean annual rainfall) and biofilter traits (i.e., total plant cover (percent cover), root mass areal density, root length areal density, species richness (S), Shannon diversity (H'), number of functional groups (FG), Functional Diversity (FD), and Functional Attribute Diversity (FAD)). Numbers are correlation coefficients (r). Significant positive correlations are designated by light gray circles. Significant negative correlations are designated by dark gray circles. The circle diameter corresponds to the absolute value of the correlation coefficient, r .

Figures:

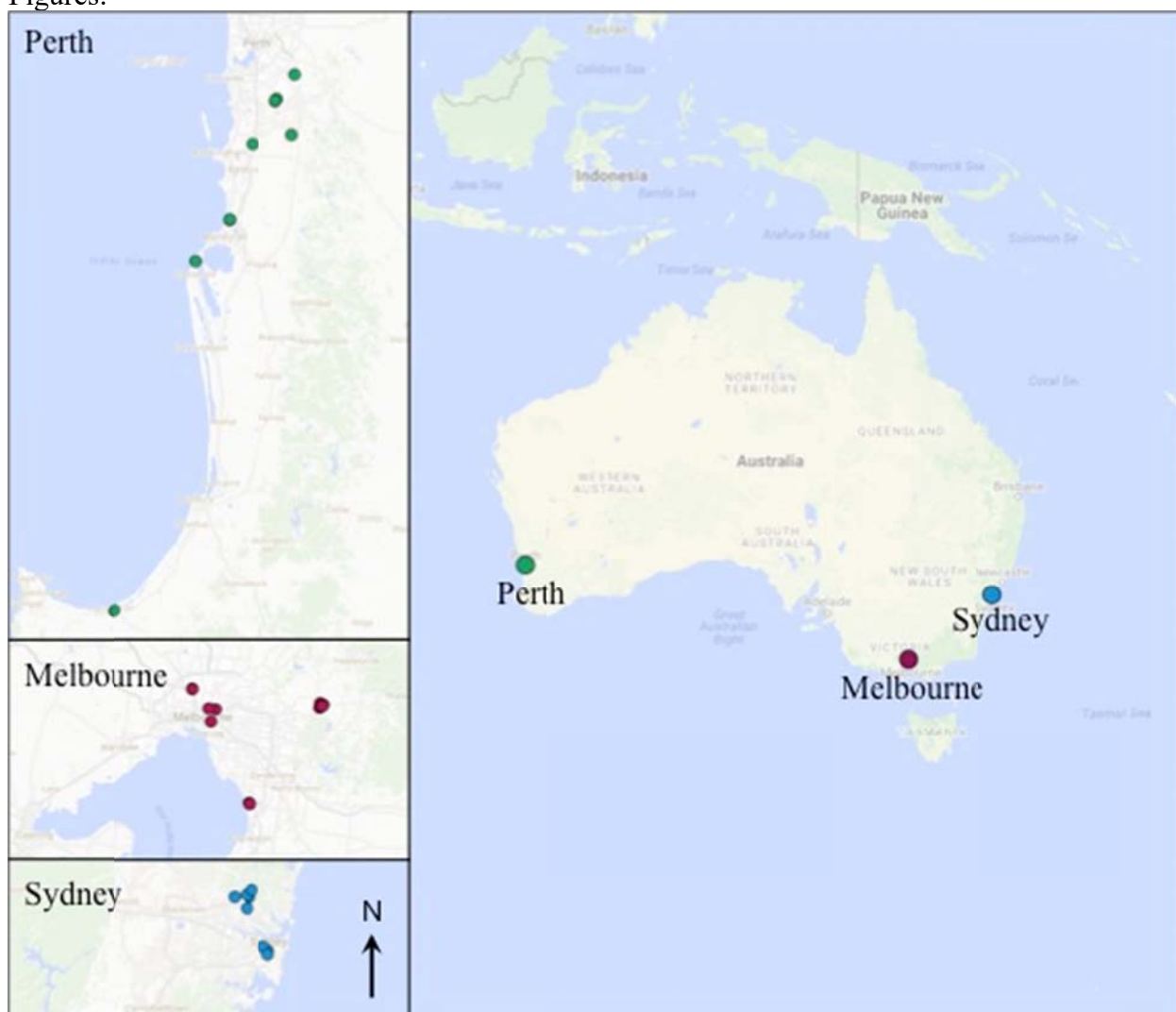


Figure 1. Location map of biofilters in Melbourne, Perth, and Sydney, Australia. Created using Google Maps.

Functional Trait Dendrogram for Biofilter Plant Species

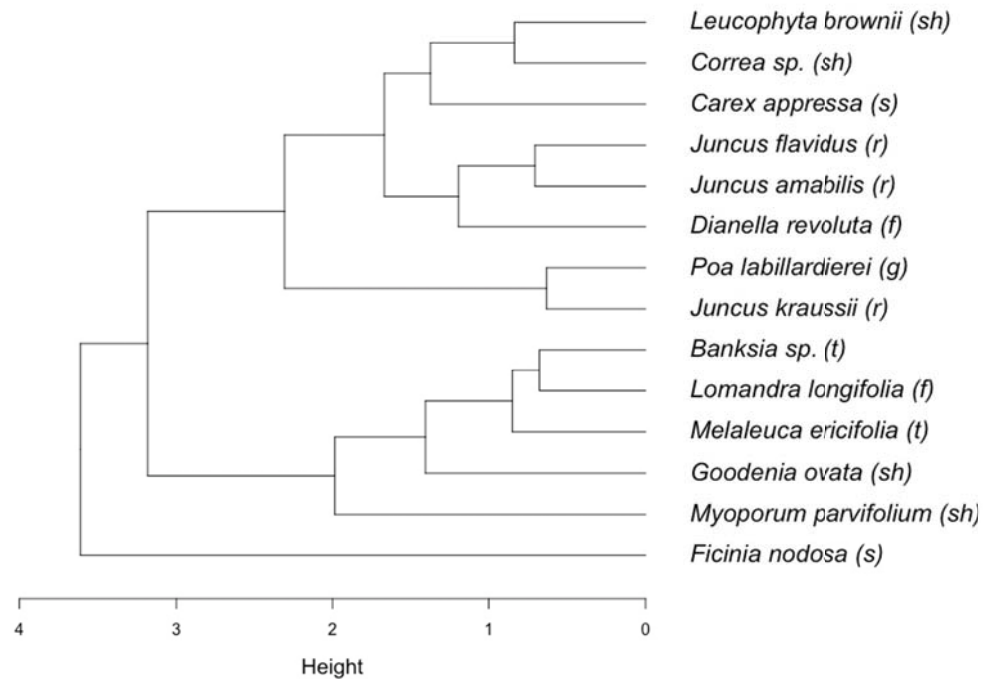


Figure 2. Functional trait dendrogram of all dominant (>10% cover) plant species in all sampled biofilters, except Alleyne Avenue 1, where *Iris* sp. was dominant but trait data were unavailable. Groupings are based on functional similarities in trait matrix. Letters in parentheses correspond to the following functional group classifications: forb (f), grass (g), rush (r), sedge (s), shrub (sh), and tree (t).

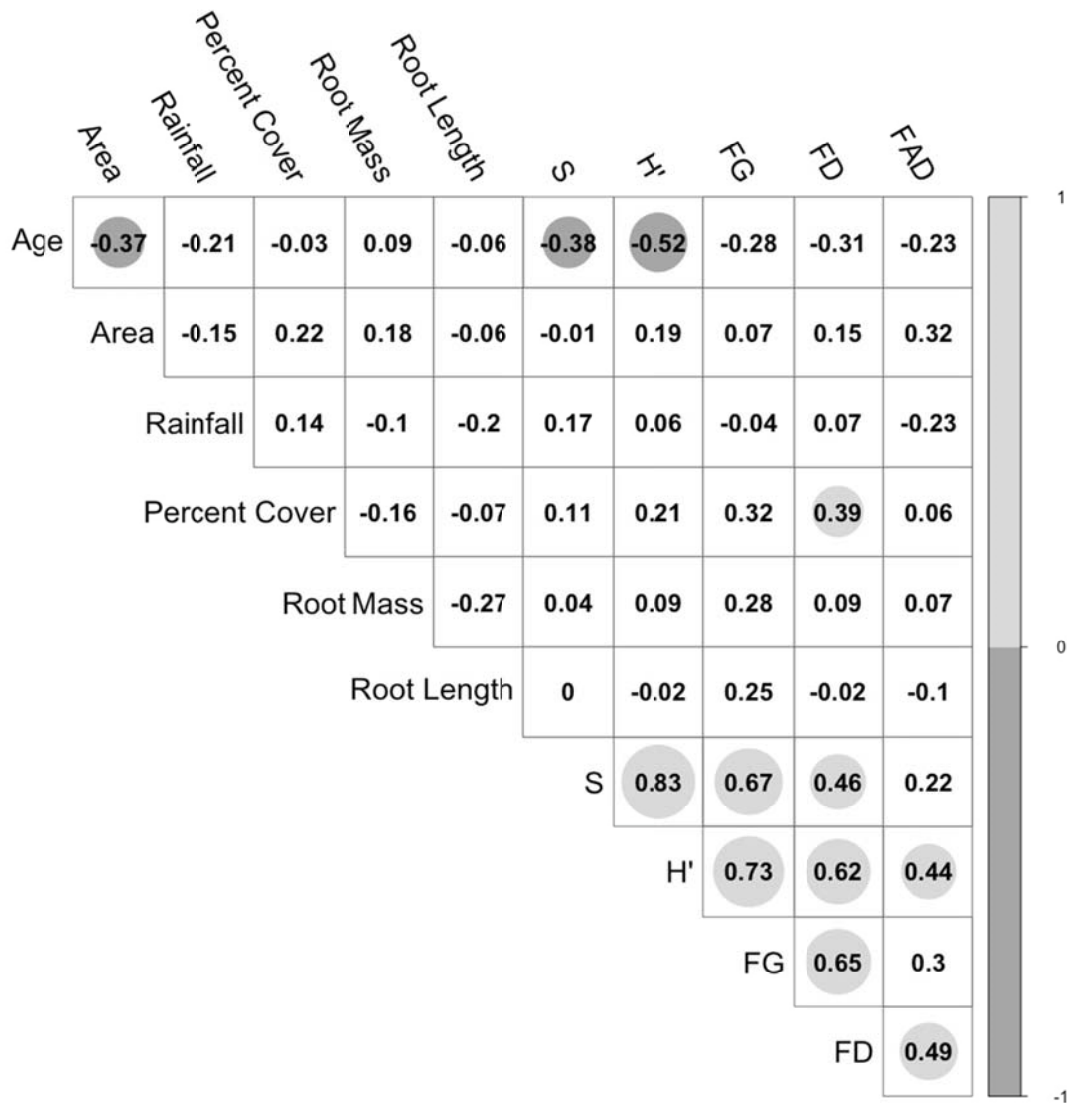


Figure 3. Correlogram of biofilter characteristics (age, area, mean annual rainfall) and biofilter traits (i.e., total plant cover (percent cover), root mass areal density, root length areal density, species richness (S), Shannon diversity (H'), number of functional groups (FG), Functional Diversity (FD), and Functional Attribute Diversity (FAD)). Numbers are correlation coefficients (r). Significant positive correlations are designated by light gray circles. Significant negative correlations are designated by dark gray circles. The circle diameter corresponds to the absolute value of the correlation coefficient, r .

List of Appendices:

Appendix A. Photographs of sampled sites that were representative of biofilters in each city.

Appendix B. Trait matrix of dominant plant species' traits present in surveyed biofilters.

Appendix C. Functional dendrograms of each site.

Appendix A. Photographs of sampled sites that were representative of biofilters in each city.

Appendix A1. Melbourne biofilters.



Figure A1-1. Biofilter on Alleyne Avenue. Species richness was 1 at time of sampling. Age was 9 years at time of sampling.



Figure A1-2. Biofilter on Fernhill Rd. Species richness was 1 at time of sampling. Age was 2 years at time of sampling.



Figure A1-3. Biofilter on Stringybark Creek Blvd N. Species richness was 6 at time of sampling. Age was 2 years at time of sampling.



Figure A1-4. Biofilter on Stringybark Creek Blvd N. Species richness was 5 at time of sampling. Age was 3 years at time of sampling.

Appendix A2. Perth biofilters.



Figure A2-1. Biofilter on Kirkland Way. Species richness was 4 at time of sampling. Age was 3 years at time of sampling.



Figure A2-2. Biofilter on Welcome Meander. Species richness was 5 at time of sampling. Age was 6 years at time of sampling.



Figure A2-3. Biofilter on Strelly St. Species richness was 6 at time of sampling. Age was 5 years at time of sampling.



Figure A2-4. Biofilter on Alcock St. Species richness was 2 at time of sampling. Age was 4 years at time of sampling.

Appendix A3. Sydney biofilters.



Figure A3-1. Biofilter on Bay St. Species richness was 6 at time of sampling. Age was 5 years at time of sampling.



Figure A3-2. Biofilter on Young St. Species richness was 6 at time of sampling. Age was 2 years at time of sampling.



Figure A3-3. Biofilter on Wolseley Grove. Species richness was 3 at time of sampling. Age was 14 years at time of sampling.



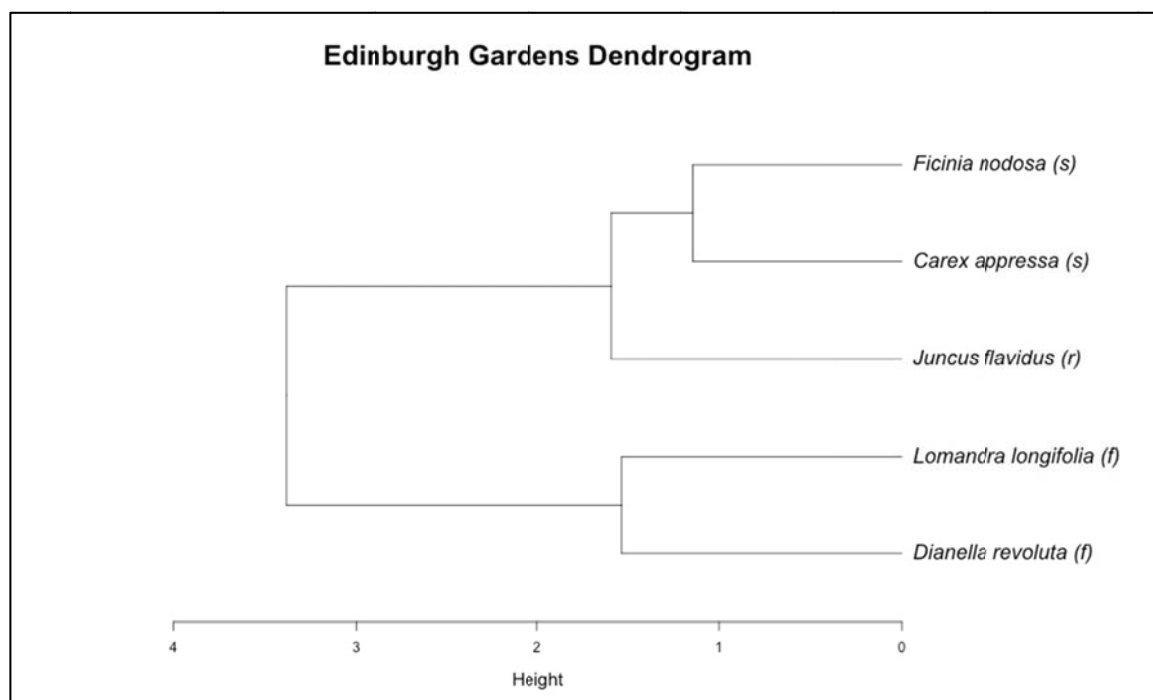
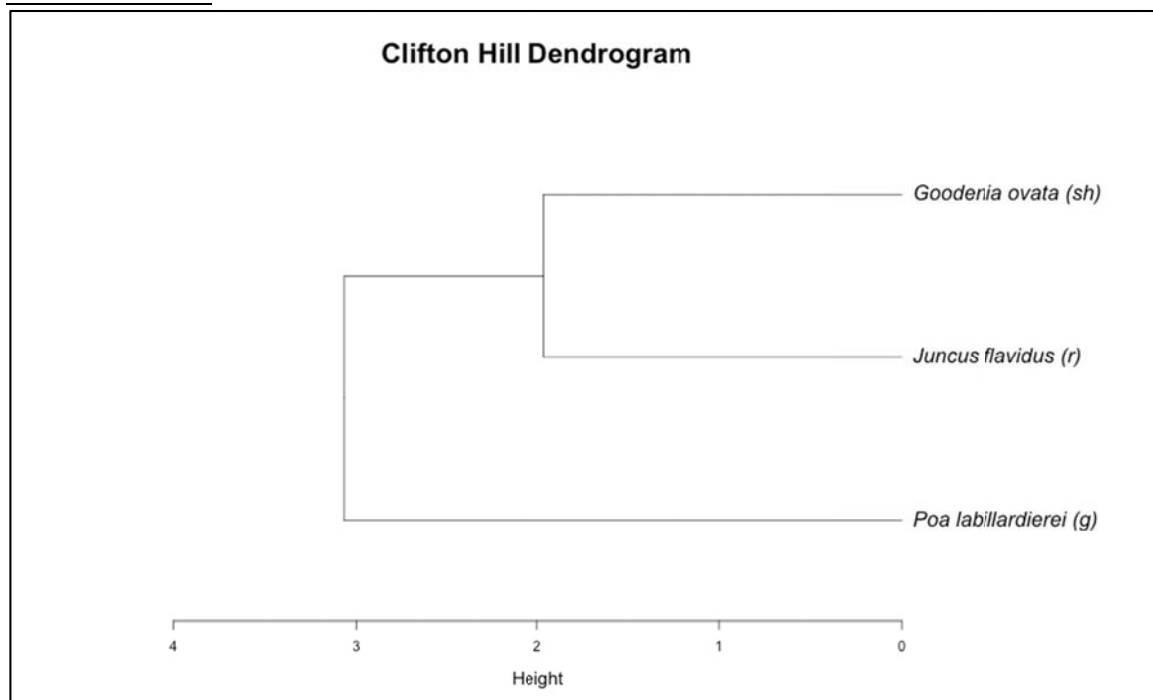
Figure A3-4. Biofilter on Normurra Ave. Species richness was 4 at time of sampling. Age was 8 years at time of sampling.

Appendix B. Trait matrix. Values for plant traits were gathered from Read et al. (2009), Payne (2013), and Pham (2016).

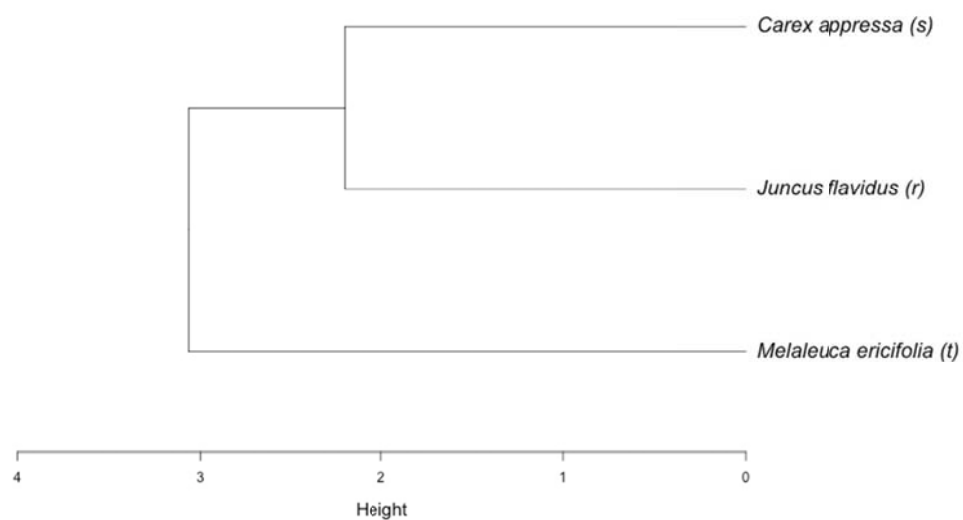
Species	Growth Form	Relative Growth Rate (mg/g/day)	Specific Root Length (m/g)	Percent Fine Roots (%)
<i>Banksia</i> sp.	Tree	8.5	40	47.5
<i>Carex appressa</i>	Sedge	13	70	45
<i>Correa</i> sp.	Shrub	8	55	32.5
<i>Dianella revoluta</i>	Forb	5	20	20
<i>Ficinia nodosa</i>	Sedge	17	85	50
<i>Goodenia ovata</i>	Shrub	12.5	55	35
<i>Juncus amabilis</i>	Rush	15	65	45
<i>Juncus flavidus</i>	Rush	12	70	40
<i>Juncus kraussii</i>	Rush	7.5	42.5	80
<i>Leucophyta brownii</i>	Shrub	8	40	27.5
<i>Lomandra longifolia</i>	Forb	11	35	25
<i>Melaleuca ericifolia</i>	Tree	12	40	37.5
<i>Myoporum parvifolium</i>	Shrub	4.5	20	15
<i>Poa labillardierei</i>	Grass	10.5	105	45

Appendix C. Functional dendrograms constructed for each site. These dendrograms were used to calculate functional diversity indices based on Cadotte et al. (2009). We did not construct dendrograms for sites with only one species. Letters in parentheses correspond to the following functional group classifications: forb (f), grass (g), rush (r), sedge (s), shrub (sh), and tree (t).

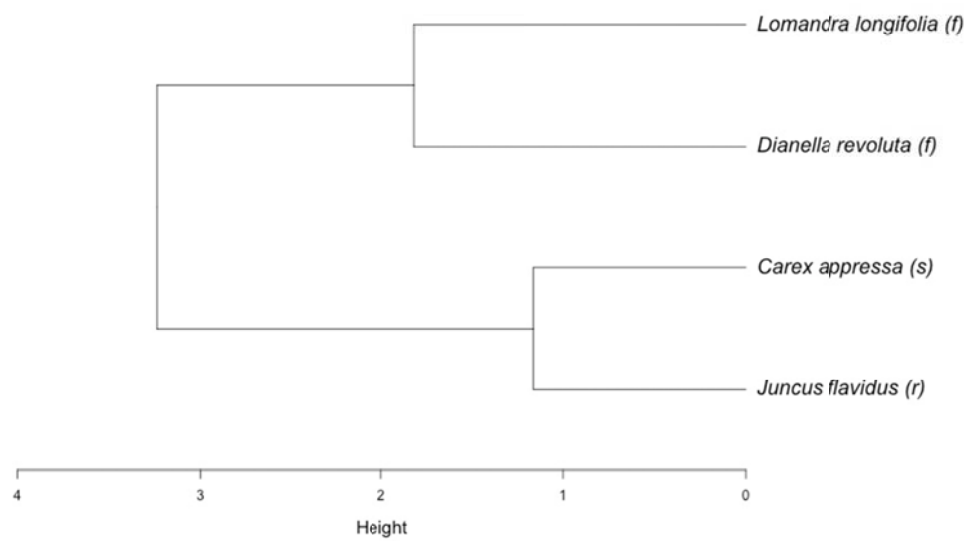
Melbourne sites

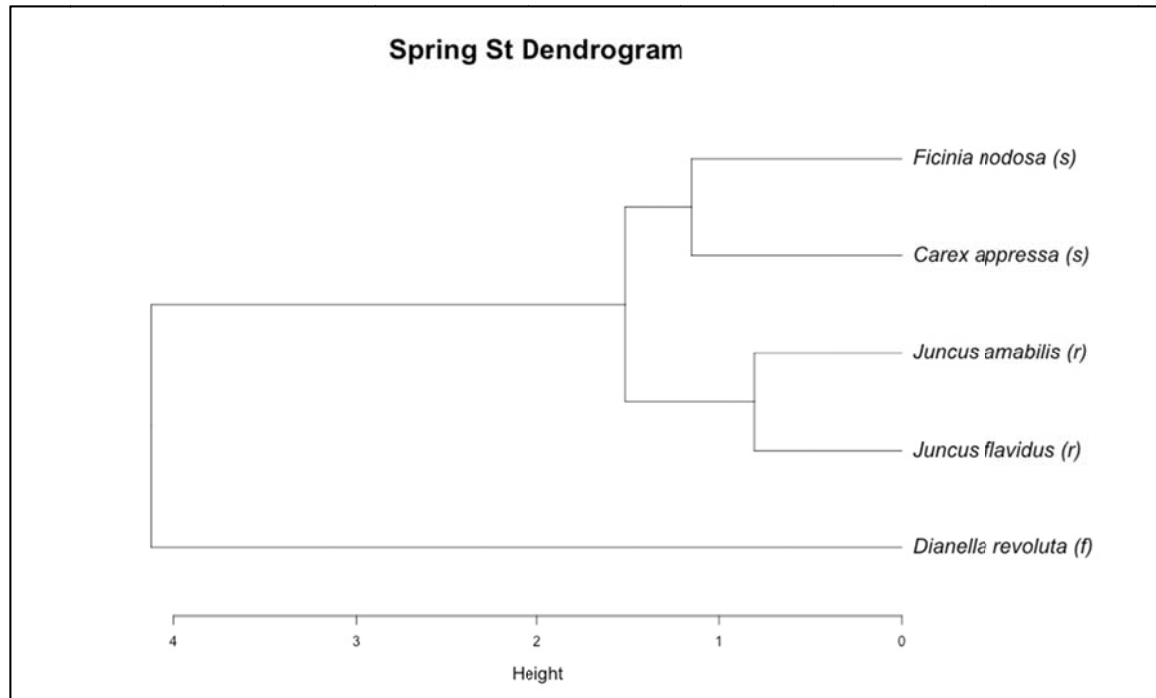


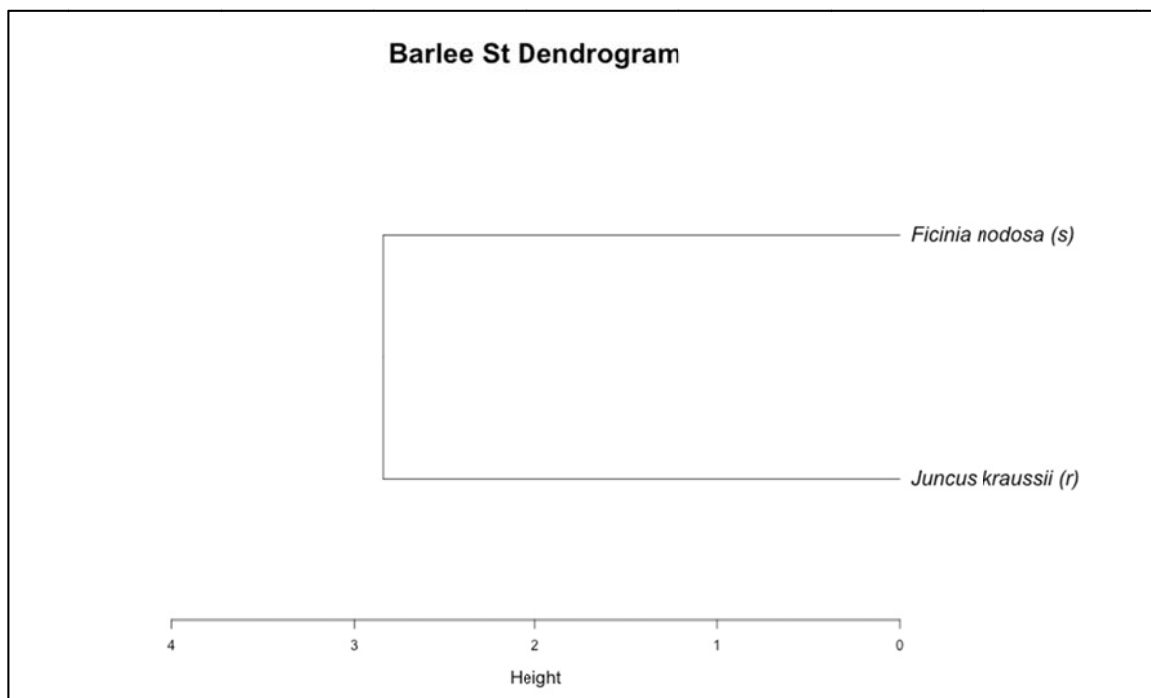
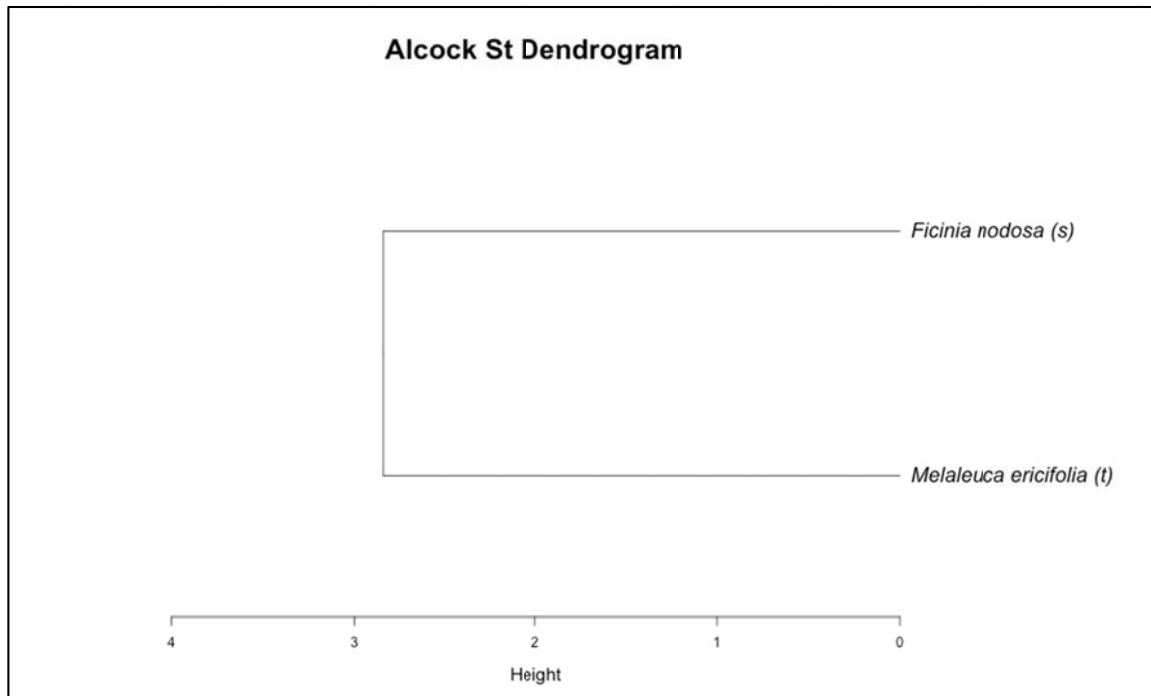
Hereford Rd Dendrogram



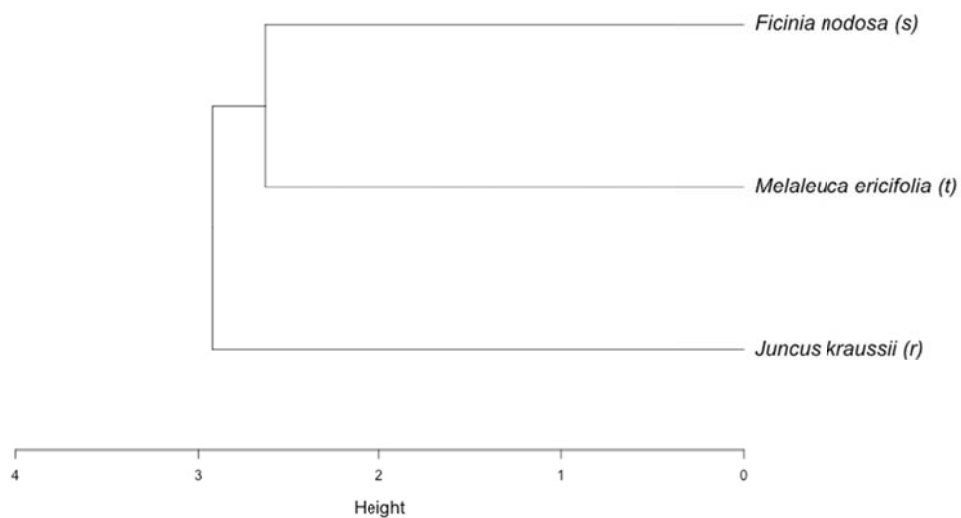
Morrison Reserve Dendrogram



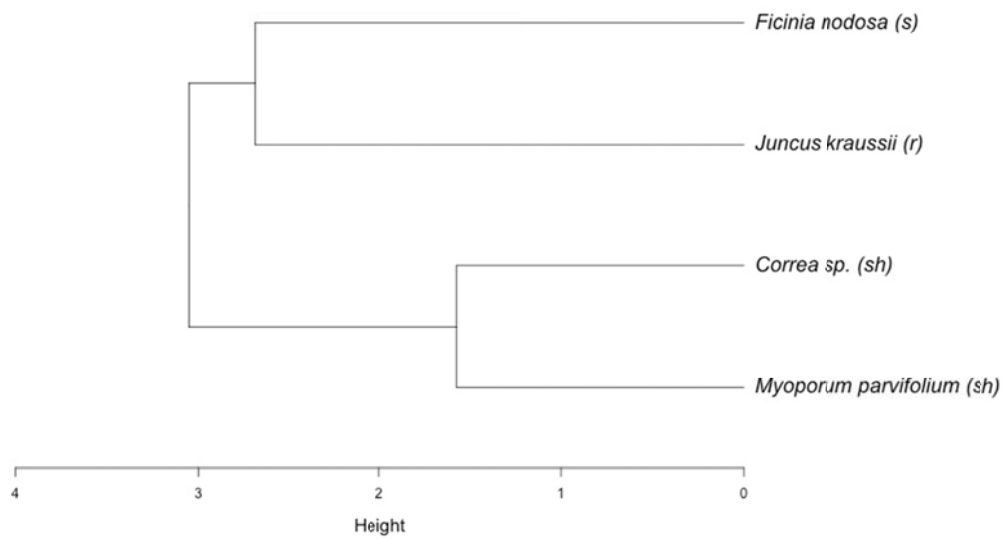


Perth sites

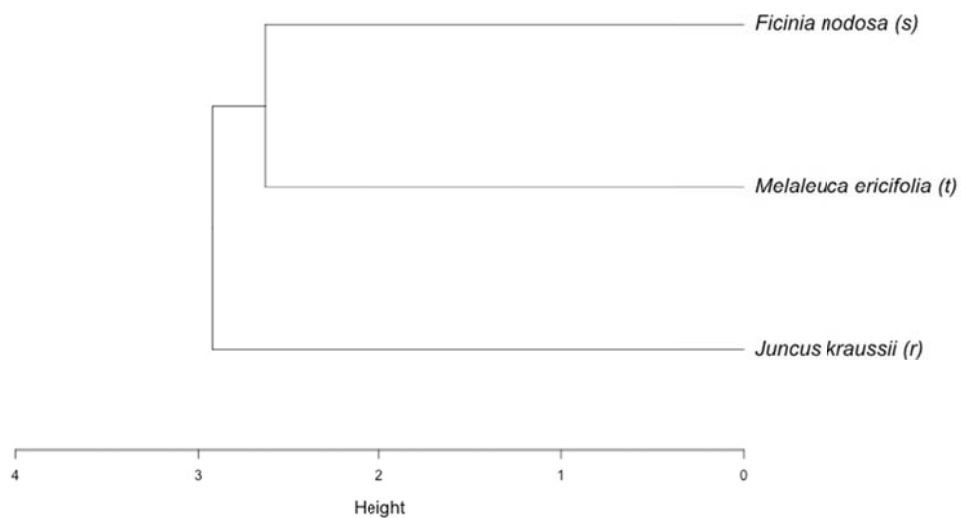
Calder Nook Dendrogram



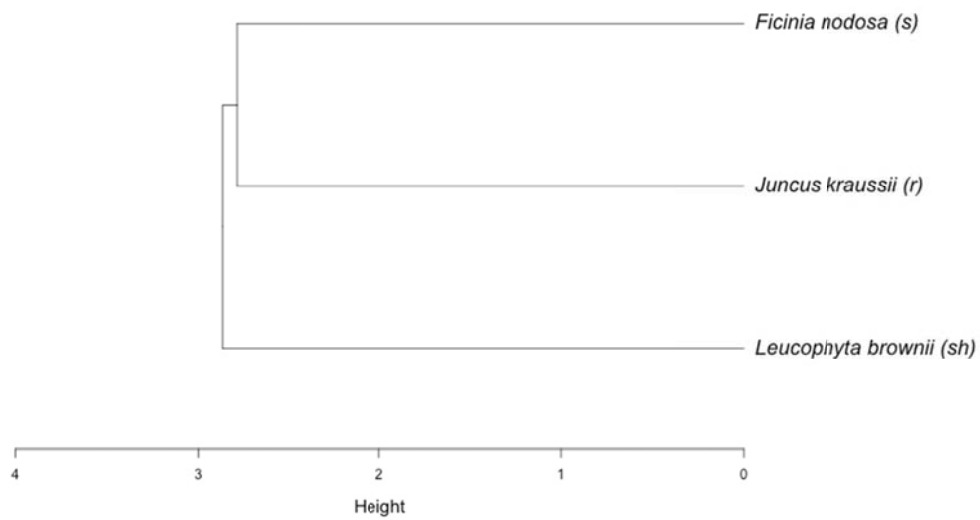
Channel View Dendrogram

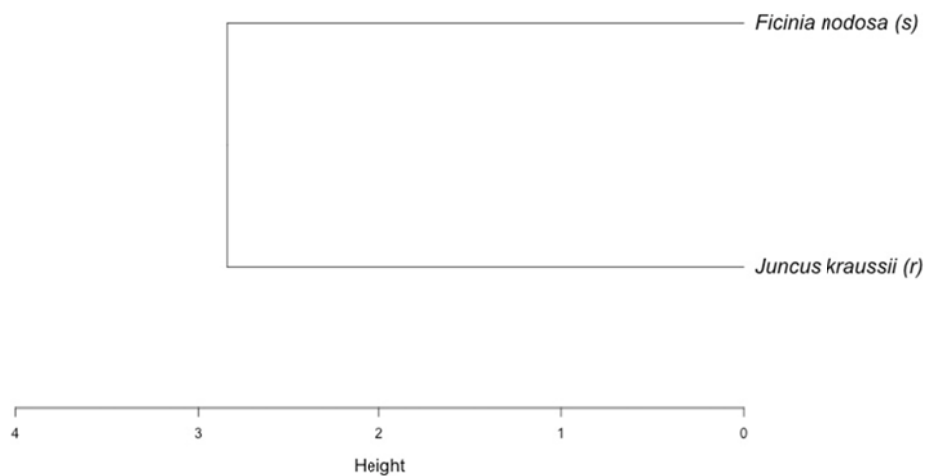
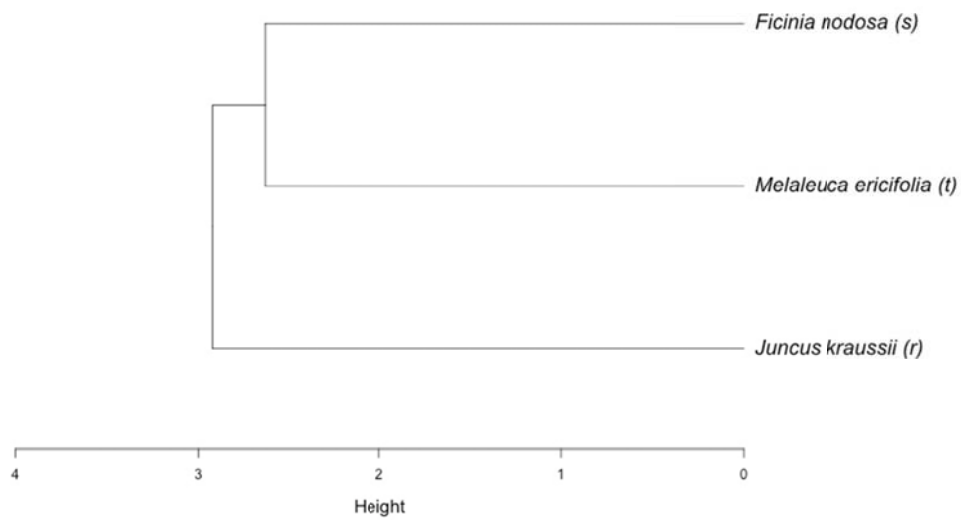


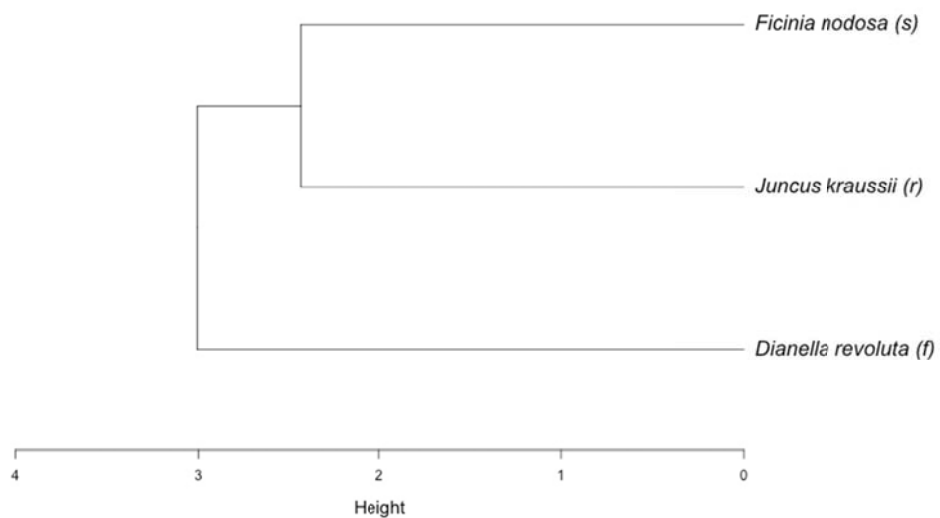
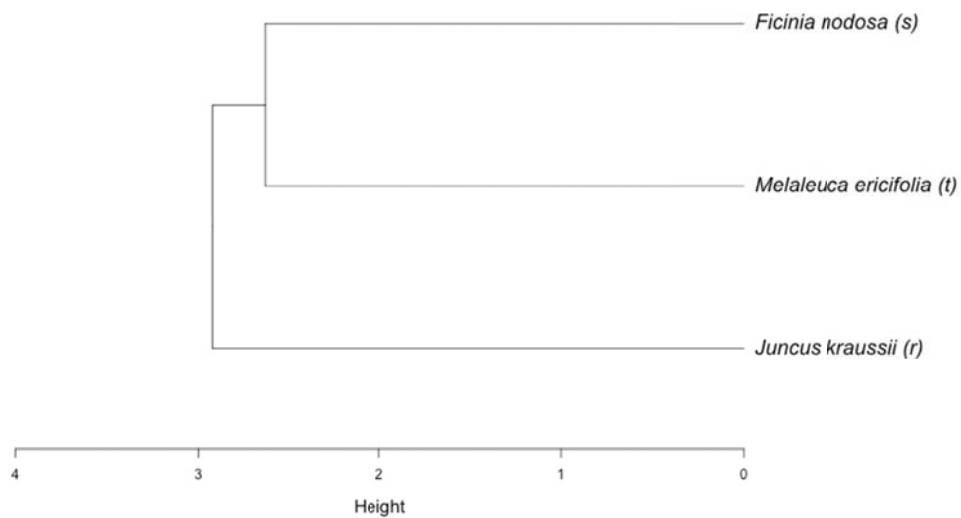
Kirkland Way Dendrogram

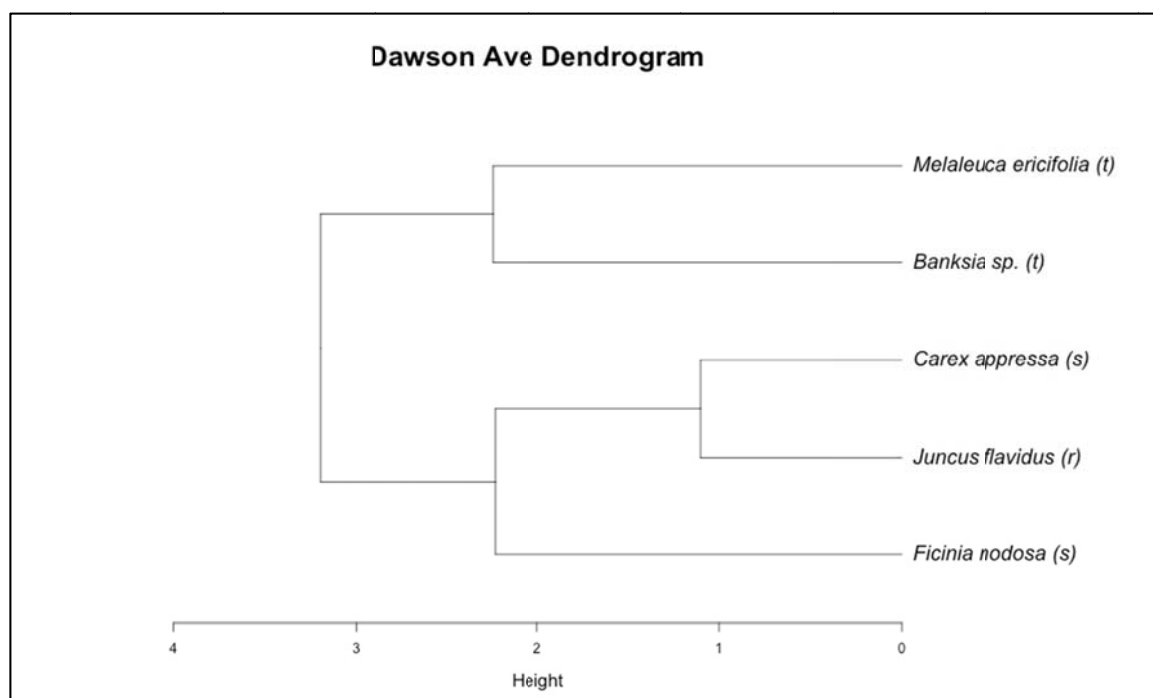
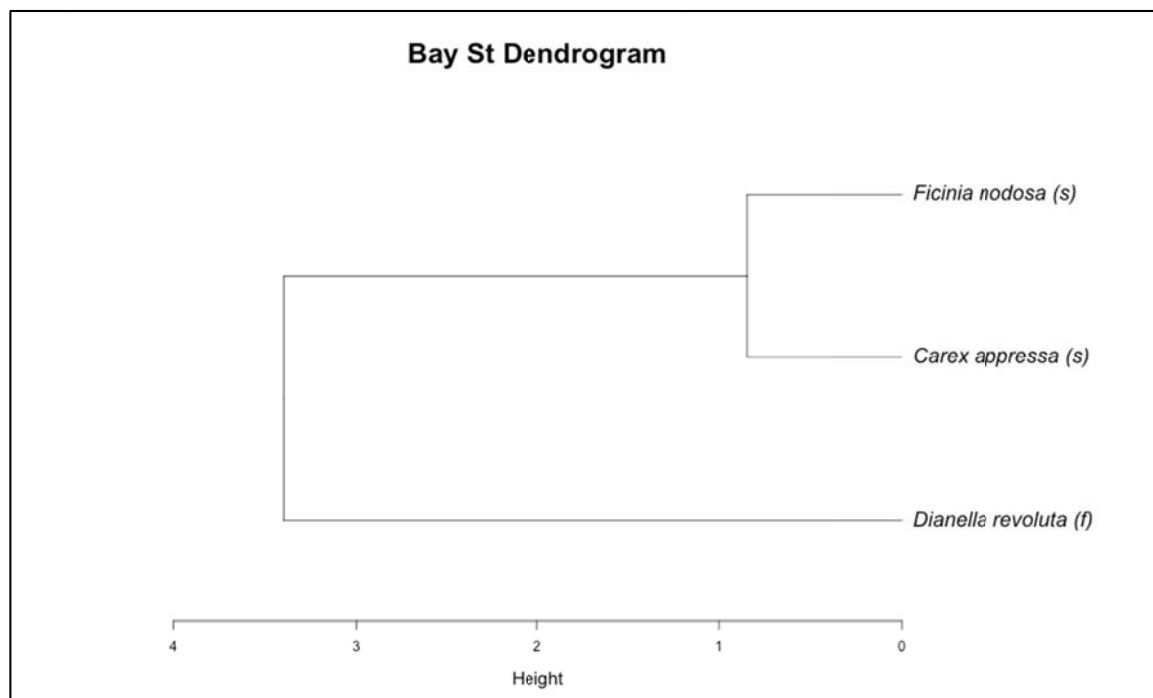


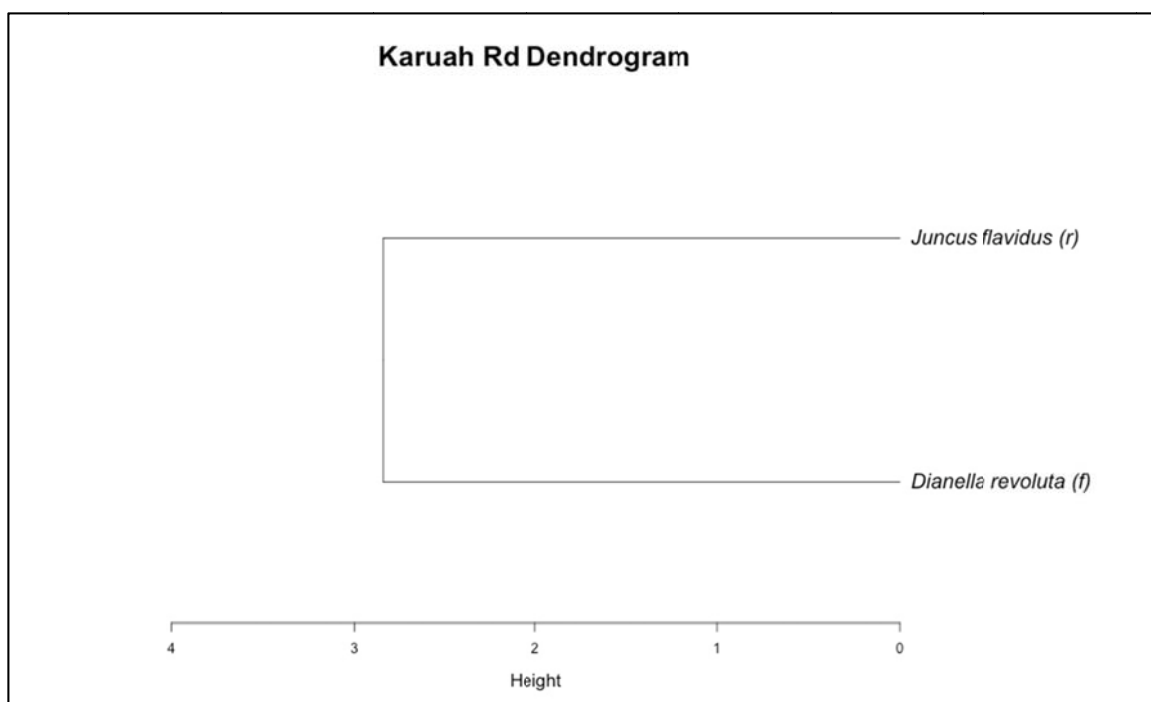
Mead St Dendrogram

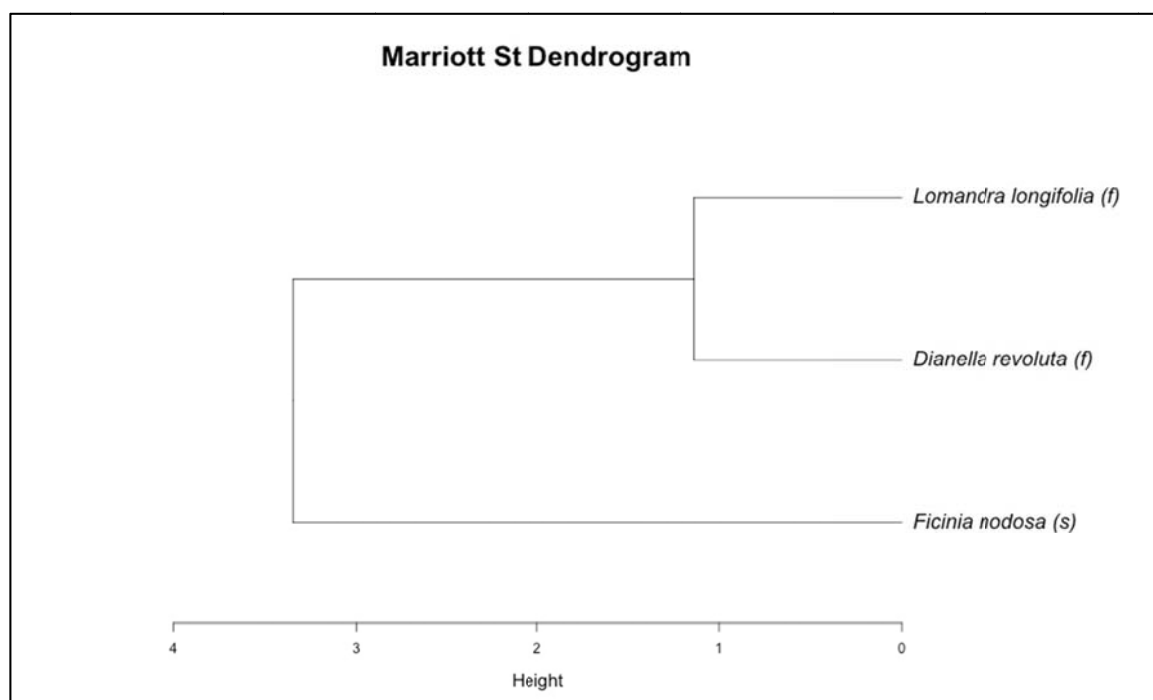
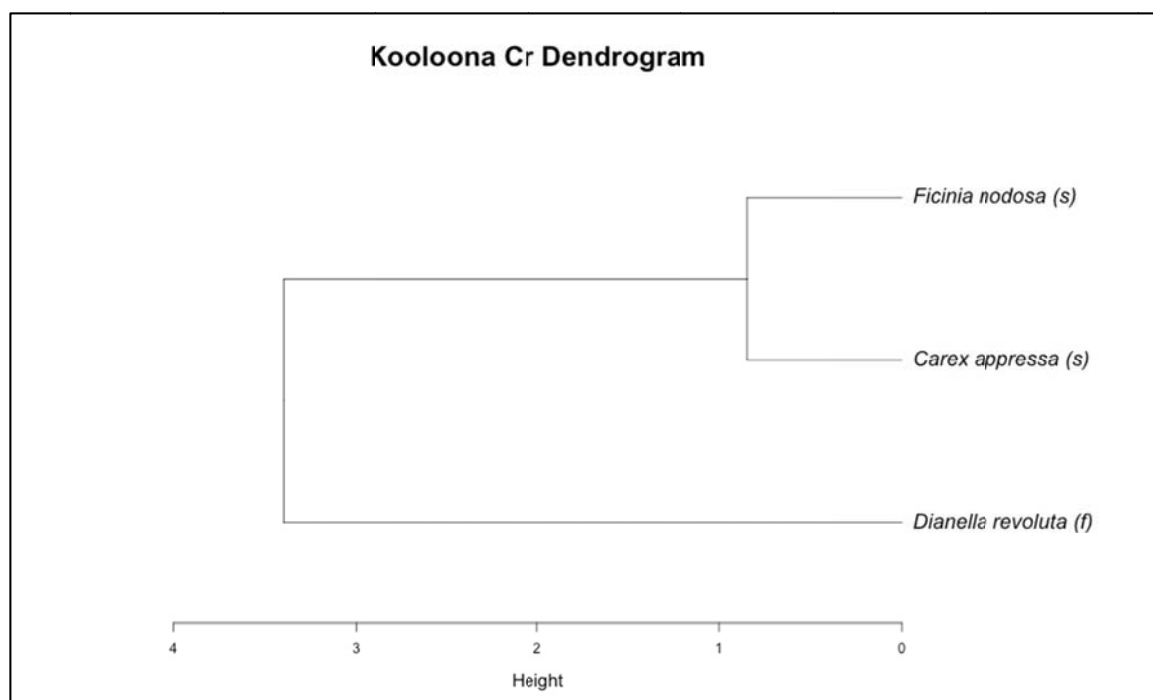


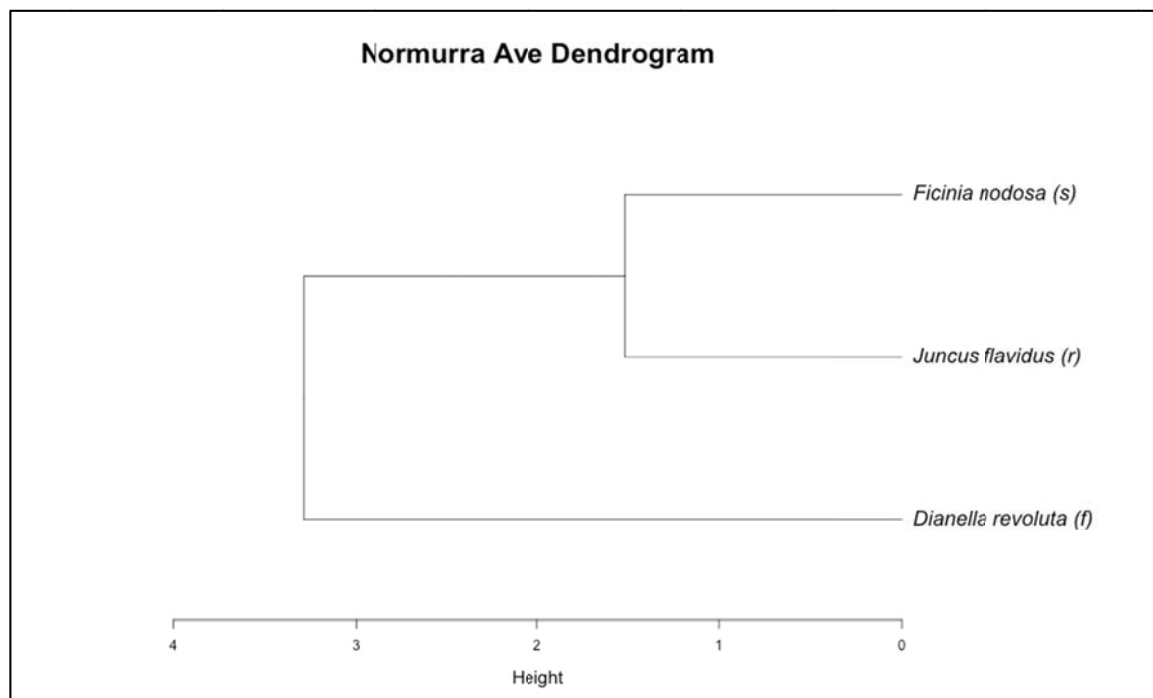
Sotheby Dr Dendrogram**Splendid Gardens Dendrogram**

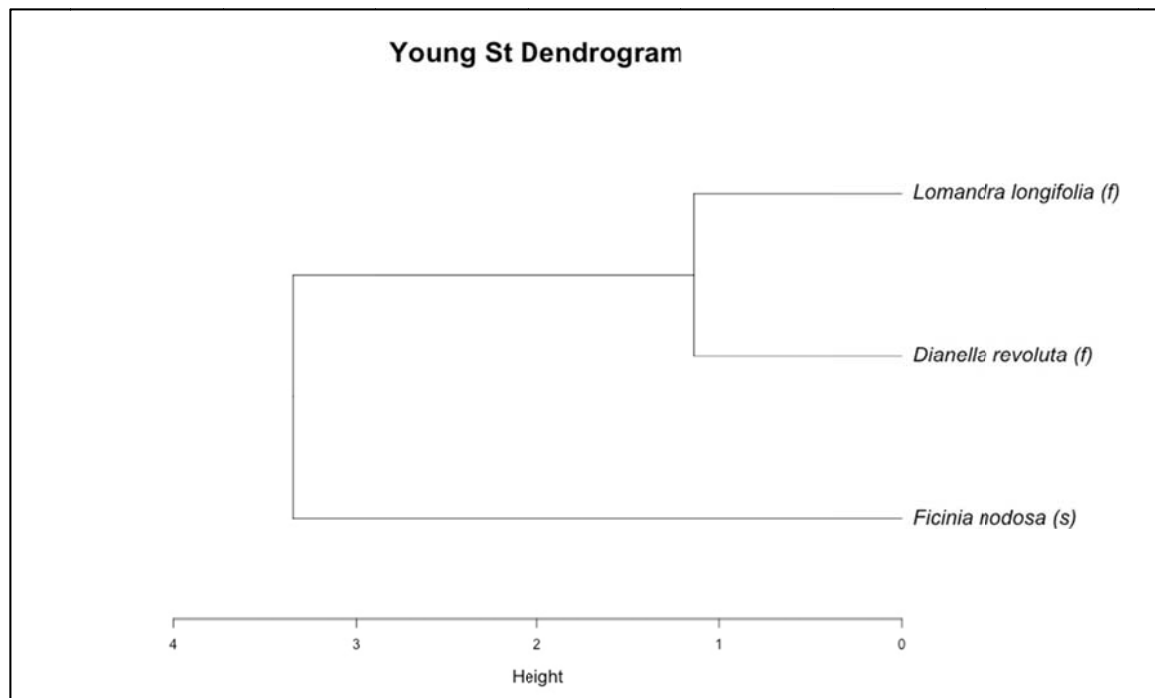
Strelly St Dendrogram**Welcome Meander Dendrogram**

Sydney sites









Appendices References:

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