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Hydrogeomorphology, floristics, classification and conservation values of the little-known montane mires of the upper Cudgegong River catchment, Central Tablelands, New South Wales

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Abstract: Mires or peat swamps have a restricted distribution in Australia and are limited to areas where hydrological inputs exceed evapotranspiration. In NSW, mires are restricted to the coast, adjacent ranges or tablelands, and along the Great Dividing Range; most are listed as threatened ecological communities under State or Commonwealth legislation. Due primarily to the relatively high rainfall and suitable geology, the Blue Mountains region includes a number of such threatened mire ecological communities. Most of these mire types are largely included within the Greater Blue Mountains World Heritage Area, although there are notable exceptions, such as the endangered Newnes Plateau Shrub Swamps.

This paper reports on a little-known group of diverse, relatively isolated and largely unprotected mires, in a relatively low rainfall area in the upper Cudgegong River catchment, east of Rylstone in the NSW Central Tablelands, and of their floristic, hydrogeomorphic and typological relationship with other mires of the Blue Mountains. They can be broadly divided into montane bogs, montane fens and hanging swamps. Particular attention is focussed on the largest and most diverse one, Rollen Creek swamp, which contains all three types. It is hoped that highlighting this hitherto unrecognised group of high conservation-value mires will contribute to their improved conservation and encourage further research into mires of eastern NSW.

Key words: peat swamp, wetlands, groundwater-dependent ecosystems, *Eucalyptus camphora*, *Petalura gigantea*, National Heritage, Greater Blue Mountains World Heritage Area.

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Introduction

Wetlands encompass a range of vegetated ecosystems, including those referred to as mires, bogs, fens, swamps and marshes (see Gore, 1983; Mitsch & Gosselink, 2007; Mitsch et al., 2009). Wetlands are characterised by a diversity of hydrological regimes, and may be permanently, seasonally or intermittently inundated or saturated (DECCW, 2010). Even in permanent wetlands, water table depth can vary considerably within a particular wetland and between different wetland types. Such spatial heterogeneity, even within a permanent wetland complex, may result in considerable heterogeneity in substrates, vegetation associations and habitat, often across small spatial scales (e.g., Brown et al., 1982; Keith & Myerscough, 1993; Keith et al., 2006). For example, wetland complexes are often characterised by a complex intergrading of fens, bogs, swamps or marshes (Hájek et al., 2006; Kirkpatrick & Bridle, 1998; Rydin & Jeglum, 2013; Wheeler & Proctor, 2000; Yabe & Onimaru, 1997).

Hydrology and water balance (evapotranspiration compared to precipitation and other hydrological inputs) are the critical factors in determining the development of peaty or organic-rich wetland sediments. A basic requirement for peat formation is that plant biomass production (carbon production) exceeds decomposition (ecosystem respiration or carbon output). Consistently high water tables and a relatively anoxic environment generally provide the necessary conditions for peat accumulation in wetlands. Conditions of seasonal drying or widely fluctuating water tables, and/or negative water balance, result in oxidisation and bacterial decomposition of organic matter, and are not conducive to accumulation of peat (Gore, 1983; Rydin & Jeglum, 2013).

In the international context the term *mire* refers to peat-forming wetlands, and includes bogs and fens (Gore, 1983; Joosten et al., 2017; Rydin & Jeglum, 2013), although the terms may be applied somewhat differently in Australia (see Whinam & Hope, 2005). In the Australian context, bogs are typically low nutrient, acidic, dominated by sclerophyllous sedges and shrubs, and mosses, and may have large parts of the surface raised above the water table, while fens are usually more nutrient-rich, less acidic or alkaline, dominated by softer herbaceous and graminoid vegetation, and usually with a near surface water table (Keith, 2004; Whinam & Hope, 2005). Additionally, in Australia, the term *swamp*, as used in this paper, is a generic term which may refer to a wide range of wetland types, including peat-forming wetlands (such as bogs and fens), and wetlands with primarily mineral substrates with environmental conditions unsuitable for peat development, in contrast for example, to its more specific application in the USA (Cowardin et al., 1979) or Europe (Rydin & Jeglum, 2013). While they receive hydrological inputs from precipitation, surface flow and groundwater in varying proportions, most mires (peat swamps) in Australia, including the diverse mire communities across the Blue Mountains of NSW, are considered groundwater-dependent ecosystems (see NSW Government, 2002; Serov et al., 2012; Whinam & Hope, 2005).

Mire ecosystems in the Blue Mountains are captured within the Montane Bogs and Fens and Coastal Heath Swamps vegetation classes of NSW and the ACT of Keith (2004). These are represented by a number of mire vegetation types, including Blue Mountains Sedge Swamps (BMSS), Newnes Plateau Shrub Swamps (NPSS), Coxs River Swamps and Boyd Plateau Bogs (Benson & Keith, 1990; Keith & Benson, 1988). These are characterised by considerable spatial heterogeneity across a number of environmental gradients, within and between individual swamps and swamp types (e.g., Holland et al., 1992a; Holland et al., 1992b). Variation in vegetation across the hydrological gradient (from ephemeral to permanent saturation) is particularly evident within the upland mires developed on sandstone geology (e.g., Benson & Baird, 2012; Holland et al., 1992a). The term *montane*, as used in this paper, follows its usage in various vegetation publications in NSW (e.g., Bell et al., 2008; Hunter & Bell, 2007; Keith, 2004; NSWSC, 2004). In describing the Montane Bogs and Fens vegetation class, Keith (2004) indicated an elevation range of 600–1500 m, which includes the elevation range of the study area. Following Keith (2004), bogs and fens identified in this study are thus referred to as montane bogs and montane fens. In their classification of the native vegetation of southeast NSW, Tozer et al. (2010) attributed higher elevation mire vegetation in the southern half of the Blue Mountains (the northern extent of their study area does not extend beyond Lithgow) to the Tableland Bogs, Tableland Swamp Meadows (in the Montane Bogs and Fens vegetation class) and Blue Mountains-Shoalhaven Hanging Swamps (in the Coastal Heath Swamps vegetation class) types.

The upper Cudgegong River catchment (east of Rylstone) on the NSW Central Tablelands drains an area surrounded by the Great Dividing Range (at elevations over 1000 m) and forms part of the inland flowing Macquarie River catchment (Fig. 1). Part of the upper Cudgegong River catchment (in the Dunns Swamp area) is included in Wollemi National Park in the Greater Blue Mountains World Heritage Area (GBMWA) (Fig. 2). No comprehensive vegetation or wetland mapping has been undertaken across this catchment, although previous reports have identified and briefly described a range of wetland vegetation types in the Dunns Swamp area (Bell, 1998a, b; Tame, 1997). During doctoral thesis fieldwork on the biology of the endangered mire-dwelling Giant Dragonfly, *Petalura gigantea*, Baird (2012) identified a number of previously undocumented and relatively isolated mires, with disjunct flora and fauna occurrences, in and adjoining the upper Cudgegong River catchment. Baird and Benson (2017) subsequently highlighted the value of one of these mire systems, Rollen Creek swamp, which occurs largely in Coricudgy State Forest, in support of a proposal to add Coricudgy State Forest to the National Heritage list; ultimately a candidate for potential addition to the GBMWA (GBMWAAC, 2015). The added inclusion of specific areas on the edges of the GBMWA, including Coricudgy State Forest, would significantly enhance the currently established biodiversity

values of the GBMWA, and has been recommended (Benson & Smith, 2015).

The aim of this paper is to draw attention to the overlooked mires of the upper Cudgegong River catchment, with a particular focus on the Rollen Creek mire system, the largest and most diverse, to contextualise them in relation to other mires of the Blue Mountains, to contribute to improving their conservation, and to encourage further research on mires in southeastern Australia.

Upper Cudgegong River study area

Location and physiography

The study area is the upper Cudgegong River catchment between the Great Dividing Range and Rylstone, centred approximately on Dunns Swamp, Wollemi National Park (32° 50' 04" S, 150° 12' 21" E). The area is located in the NSW Central Tablelands, in the north-western part of the Blue Mountains and of the Sydney Basin bioregion (Thackway & Cresswell, 1995). The Great Dividing Range forms the watershed of the upper Cudgegong River, which drains inland to the Macquarie River (Fig. 1 & 2). Drainages to the north, east and south of the upper Cudgegong River watershed drain to the coast. Elevation decreases from east to west; from 1256 m on the basalt peak of Mt Coricudgy on the Great Dividing Range, westward to 571 m at Rylstone. Additional high elevation areas (above 1000 m) along the catchment watershed, and associated with the Great Dividing Range, include the remnant basalt-capped plateau of Nullo Mountain, and the isolated basalt-capped peaks of Mt Darcy and Mt Coorongooba.

The geology of the study area consists of underlying Permian sandstone, conglomerate, shale and siltstone; Triassic Narrabeen Sandstone, shale and conglomerate, forming characteristic 'pagoda country'; and Tertiary basalt-capped peaks and remnant plateaus. The Capertee, Collingwood, Inglewood, Lees Pinch, Munghorn Plateau, Nullo Mountain and Coricudgy soil landscapes dominate the study area (Kovac & Lawrie, 1991). The Inglewood (Yellow Earths), Lees Pinch (Shallow Soils) and Munghorn Plateau (Siliceous Sands) soil landscapes have developed primarily in association with the Triassic Narrabeen sandstone geology, while Nullo Mountain and Coricudgy soil landscapes (Krasnozems) have developed in association with the basalt-capped peaks and remnant plateaus, where *in situ* weathered Tertiary basalt and basalt colluviums overlie the Narrabeen sandstone. Capertee (Yellow Podzolics) and Collingwood (Red Podzolics) soil landscapes have developed in association with the Permian geology (Kovac & Lawrie, 1991). The peaty sediments of the mires in the area were not differentiated in the coarse-scale soil landscape mapping of Kovac and Lawrie (1991).

The study area includes the Dunns Swamp area of Wollemi National Park, a popular tourist site, and parts of Nullo Mountain and Coricudgy State Forests. With the exception of logging of better quality timber from the slopes of Mt Coricudgy, there is no evidence of recent logging activity in the immediate Coricudgy State Forest area which is mainly essentially undisturbed shrubby woodland with little timber value. Variably cleared and farmed freehold lands are mainly concentrated along the Cudgegong River valley and its tributaries, such as Coxs Creek, and in the Nullo Mountain area. The lower Cudgegong valley around Rylstone was explored by the botanist Allan Cunningham in the 1820s, and the area taken up for pastoral settlement soon after. Occupational licences in the upper Cudgegong in the Parish of Kelgoola were offered for sale in 1843 (*Sydney Morning Herald* 18/1/1843).

Climate

Rainfall across the study area is characterised by high inter-annual variability, with average annual rainfall decreasing with elevation and from east to west. Highest mean monthly rainfalls occur between November and March (late spring to early autumn), with lowest mean monthly rainfalls from July to September (winter to early spring). Mean annual rainfall at the Nullo Mountain AWS (1130 m elev.) was 955 mm/annum (1994–2017) (www.bom.gov.au). Mean maximum monthly temperature at Nullo Mountain was 16.7° C (9.2–24.0° C) and mean minimum monthly temperature was 8.0° C (2.5–13.6° C). Mean Annual rainfall at the property "Kelgoola" (747 m elev.) in the upper Cudgegong River valley near Rollen Creek swamp and Mt Coricudgy, was 809 mm (1963–2006), also with high inter-annual variability (486–1235 mm/annum). The higher elevation areas along the Divide, including Mt Coricudgy, Mt Coorongooba and Mt Darcy, can also be expected to have higher rainfall than that of the nearby upper Cudgegong valley at "Kelgoola" due to an orographic effect, and similar to that of Nullo Mountain. Mean annual rainfall at Rylstone (Ilford Rd AWS, 605 m elev.), at the western edge of the study area, is lower, with 655 mm/annum. Mean maximum monthly temperature at Rylstone was 22.6° C (14.4–30.7° C) and mean minimum monthly temperature was 8.2° C (1.2–15.9° C) (www.bom.gov.au).

Previous vegetation studies in the upper Cudgegong

Previous vegetation studies in the upper Cudgegong are mainly confined to the Dunns Swamp area (Bell, 1998a; Gellie & McRae, 1985; Tame, 1997). Gellie and McRae (1985) referred more broadly to the Cudgegong Swamps. Building on a brief report by Tame (1997), and with limited additional fieldwork, Bell (1998a) described Cudgegong River Swamp Grassland, Upper Cudgegong Alluvial Sedgeland, Upper Cudgegong Alluvial Reedland, Upper Cudgegong Alluvial Shrub-swamp and Upper Cudgegong Sphagnum Bog in the Dunns Swamp area of Wollemi National Park.

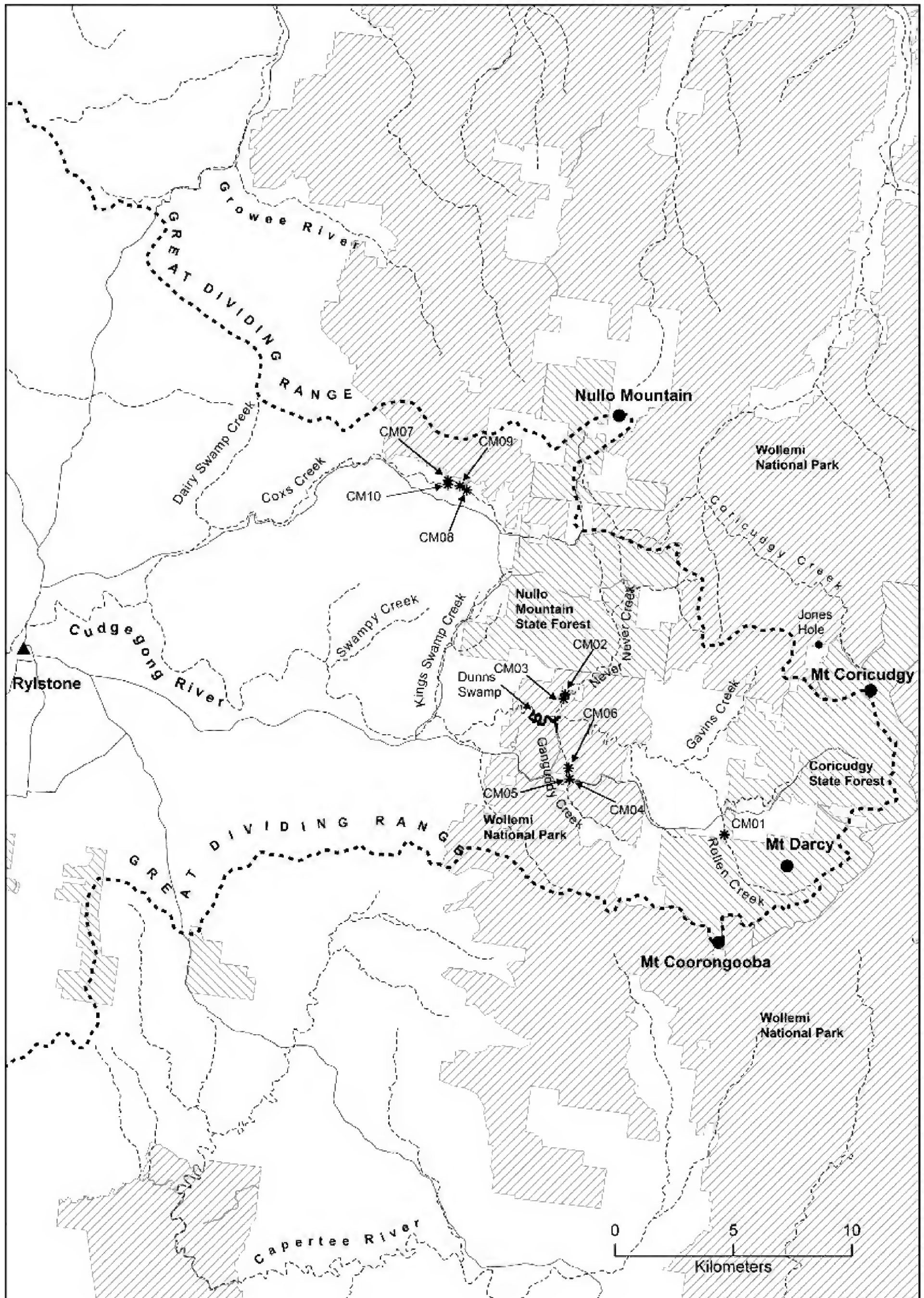


Fig. 2. Map of the Upper Cudgegong River catchment study area and locations of surveyed mires. The site codes in this diagram are explained in Table 1.

Although Bell (1998a, b) did not identify the *Carex* fen vegetation along Never Never Creek and other similar sites, he noted the need for additional survey of wetland vegetation communities across the upper Cudgegong area to clarify their floristic relationships and distributions. Bell (1998a) noted that the Cudgegong River Swamp Grassland vegetation occurred only in waterlogged peaty alluviums and the Upper Cudgegong Alluvial Sedgeland occupied “grassy bogs”. Tame (1997) suggested the substrate of this vegetation type was probably a shallow sandy peat overlying impermeable clays. Bell (1998b) noted that Cudgegong Sphagnum Bogs only occurred in limited areas on heavy soils with poor drainage, along creek-lines or in small localised patches, but also referred to it occurring on peaty sands.

Bell (1998a) also noted the mention by Ford (1989) of an area of swamp near Mt Darcy (“along Rollen Creek”) with apparent similarities to his Cudgegong River Swamp Grassland. This is the first known reference to a swamp system along Rollen Creek, later described by Baird (2012) and Baird and Benson (2017).

Methods

Reconnaissance fieldwork (by IRCB) was carried out across the upper Cudgegong River catchment and adjoining areas in 2007-2009, searching for evidence of the endangered *Petalura gigantea* and to identify mires with potential breeding habitat for this obligate, groundwater-dependent, mire-dwelling dragonfly (Baird, 2012, 2014). Identification of upper Cudgegong mire sites has been based on a range of sources; aerial photograph interpretation (API, using Google Earth); previous vegetation reports from the Dunns Swamp area (Bell, 1998a, b; Tame, 1997); additional information provided by former NPWS ranger Chris Pavich (pers. comm.); data gathered by Baird (2012) (which included recording mires along 4 creek systems), and the results of a vegetation survey of the swamp system along Rollen Creek in February 2017 (Baird & Benson, 2017); and additional vegetation surveys across the study area in October 2017. Swamps were surveyed across Coxs, Ganguddy, Kings Swamp, Never Never and Rollen creeks (Fig. 2). Swamps on the remaining creek systems are on private property. Vegetation surveys consisted of walking transects targeted to cover as much local variation as possible and to record all readily identifiable plant species (and any evidence of threatened faunal species) across the range of identified floristic and hydrogeomorphic variability within the swamps surveyed. The presence of localised seepage areas or of an emergent water table provided further confirmation of the presence of suitable hydrological conditions for the development of organic-rich substrates. A sediment probe (1.8 m long × 8 mm diameter steel rod) was used to measure depth, and qualitatively assess characteristics, of organic-rich sediments in lower Rollen Creek swamp.

Results

Mires of the upper Cudgegong River catchment

Swamp vegetation in the upper Cudgegong River catchment was identified along the Cudgegong River and its tributary creeks - Coxs, Dairy Swamp, Ganguddy, Gavins, Kings Swamp, Mill, Never Never, Rollen, Sugarloaf and Swampy creeks (Fig. 2). Mire vegetation communities (with persistent high water tables and organic-rich or peaty sediments) were confirmed by survey along Coxs, Ganguddy, Never Never and Rollen creeks. Mire vegetation was identified along the main Cudgegong River valley immediately upstream and downstream of the confluence with Rollen Creek, but was not surveyed due to its location on private property. The presence of mire was not confirmed along the highly degraded Kings Swamp drainage. Details of mires surveyed are shown in Table 1. Occasional small isolated patches of seepage-fed hanging swamp on valley sides are also scattered across the upper study area on Narrabeen Sandstone, particularly in the Rollen Creek catchment. Additional small mire patches can be expected to occur elsewhere across the study area in locations that are difficult to access or identify on Google Earth imagery.

The mires of the upper Cudgegong River and its tributaries above Dunns Swamp, and along upper Coxs Creek, occur primarily in the Munghorn Plateau soil landscape on Triassic geology (see Kovac & Lawrie, 1991). Mire sediments included organic sands, sandy peats, sapric peats and fibrous peats, depending on the hydrological regime and topographic context. Some patches of swamp with sedgeland-heath vegetation, with floristic similarities to the wetter sedgeland-heath bogs, occur on heavier clayey soils (e.g., along lower Ganguddy Creek above Dunns Swamp) and cannot be considered mires. Depths of soft, saturated, organic-rich/peaty sediments throughout the main valley floor along lower Rollen Creek swamp (CM01) were generally >1.8 m, frequently with no major differences in density or texture observed across that depth. Dense sandy-gravel material was encountered, however, below 1.6 m in several locations. In one area near the confluence with a tributary near the Coricudgy State Forest boundary, some narrow bands of sandy-gravel material at different depths indicated historical deposition events, probably associated with post-fire erosion, upstream and/or upslope. Predictably, depth of organic-rich swamp sediment progressively decreased, and sand content increased, towards swamp margins on lower valley side slopes.

Considerable hydrogeomorphic and floristic diversity was observed in the mires across the study area, including montane bogs and montane fens, broadly referable to the Tableland Bogs and Tableland Swamp Meadows typologies, respectively, of Tozer et al. (2010), within the Montane Bogs and Fens vegetation class of Keith (2004), and hanging swamps on valley sides, with similarities to the Blue Mountains-Shoalhaven Hanging Swamps typology of Tozer et al. (2010), within the Coastal Heath Swamps vegetation class of Keith (2004). The rare, swamp-dwelling small tree,

Eucalyptus camphora subsp. *camphora*, is widely distributed across the upper Cudgegong catchment within each of these mire variants, along associated upstream drainage lines, and in some other swampy areas on heavier soils, including along lower Ganguddy Creek.

The main mire systems

Rollen Creek swamp, most of which occurs in Coricudgy State Forest, is one of only two extensive and relatively undisturbed (based on API, using Google Earth) mire systems in the upper Cudgegong catchment. The Rollen Creek swamp system appears to include the full range of hydrogeomorphic and floristic variation observed within the identified mires in the study area, including montane bogs (Tableland Bogs) and montane fens (Tableland Swamp Meadows) on valley floors, and hanging swamps on valley sides (Figures 3-11). Due to its outstanding diversity, size, condition and its representativeness of the range of mires in the study area, the Rollen Creek swamp system is described in detail below.

In addition to the extensive area of valley-floor sedgeland-heath bog (Tableland Bog) along Rollen Creek, several isolated and very small patches of variably degraded, seepage-fed sedgeland-heath bog (CM08-10) were also identified adjacent to, and intergrading with, the alluvial *Carex*-dominated fen (Tableland Swamp Meadow) (CM07) along upper Coss Creek (Fig. 13). A narrow band of hanging swamp (CM04) of sedgeland-heath and *Sphagnum*, with conspicuous springs, also occurs just above Ganguddy Creek, near to and upstream of the Coricudgy Road bridge.

The other large mire system in relatively good condition is along Never Never Creek and its lower tributaries above Dunns Swamp (Kandos Weir impoundment) in Wollemi National Park. The Never Never Creek mire system is dominated by *Carex* fen (Tableland Swamp Meadow) along the main valley floor (CM03) with scattered *Leptospermum* shrubs (Fig. 12), although floristic variants occur in some marginal areas and in slightly higher gradient tributary swamp (CM02), in association with *Eucalyptus camphora*. The Rollen Creek and Never Never Creek mire systems are surrounded by high quality, dry sclerophyll eucalypt-dominated woodland on adjoining slopes.

The swamps along the main valley floors of the Cudgegong River (above and below Dunns Swamp) and its tributaries, including Coss Creek (Fig. 13) are generally low gradient, alluvial swamp meadows. They are generally located on cleared freehold land, with a history of grazing, nutrient enrichment and weed invasion, and mostly range in condition from moderately- to highly-degraded. They range from *Carex*-dominated fens with organic-rich alluvium in wetter areas (Tableland Swamp Meadows) which may be variably inundated with shallow water, to seasonally wet tussock grasslands on largely mineral soils at the drier end of the hydrological gradient. Some of the swamp meadows include small pools along the main drainage line (possible chain-of-ponds system) with reedland (marsh) vegetation

of *Typha* sp., *Phragmites australis* and/or *Eleocharis sphacelata*. Along the main Cudgegong River valley above Dunns Swamp, around the confluence with Rollen Creek, areas of highly degraded Tableland Swamp Meadow sometimes also appear to have small remnant patches of sedgeland-heath bog (Tableland Bog) on their margins. Based on API (using Google Earth), additional areas of Tableland Swamp Meadow with possible Tableland Bog were identified along Gavins and Sugarloaf creeks, upper tributaries of the Cudgegong River; these swamps on private properties are mostly surrounded by cleared grazing land and were not visited.

A localised part of the upper Kings Swamp drainage includes some plants which occur in mires elsewhere in the area (e.g., *Callistemon citrinus*, *Eucalyptus camphora*, *Leptospermum obovatum*), with some groundwater seepage evident, but it is highly degraded and occurs on cleared grazing land. Regardless of its potential pre-disturbance state, it is not treated as mire here. A degraded and grazed narrow swamp with *Callistemon citrinus* and *Eucalyptus camphora* subsp. *camphora* also occurs along upper Mill Creek. This small swamp patch on otherwise cleared, private property was observed but not surveyed and it was not possible to determine its substrate or hydrological characteristics. It is also not treated as mire here.

Another large swamp system is in Jones Hole, on Jones Hole Creek (a headwater tributary of Coricudgy Creek), to the northwest of Mt Coricudgy (Fig. 2), in the coastward flowing Hunter River catchment, not the Cudgegong River. It is mostly in Wollemi National Park though partly in Coricudgy State Forest. The vegetation of this difficult-to-access valley-floor swamp system (32° 48' 16" S, 150° 20' 11" E, 720–850 m elev., ~3250 m length) is unknown, but it is reasonable to assume, due to its elevation and proximity to the higher rainfall area of adjacent Mt Coricudgy, that it is characterised by a high water table and variably peaty sediments. API (using Google Earth) suggests that it is predominantly a *Carex*-dominated fen, with areas of intergrading sedgeland-heath bog. The swamps in Jones Hole and along Never Never Creek have some history of cattle grazing prior to National Park gazettal (C. Pavich, pers. comm.). Conspicuous, abundant and structural plant species recorded for each of the identified and surveyed mires in the study area are included in Table 2. This is an incomplete list as some grasses and monocots were not identified, and some seasonal species may not have been evident at the time. Only two conspicuous bryophytes are listed, but the total bryophyte richness is likely to be high.

Other swamps

The study area includes a diversity of swamp vegetation types distributed across the hydrological gradient; much of the swamp area is characterised by only seasonally or intermittently wet sandy alluviums or heavier clayey soils along valley floors and cannot be considered mires. This includes seasonally wet tussock grasslands on largely mineral

soils at the drier end of the hydrological gradient, which generally occur in the lower elevation and lower rainfall parts of the catchment. Most of these seasonally wet tussock grasslands are surrounded by cleared grazing areas and occur along the Cudgegong River and its tributaries below Dunns Swamp, and along lower Coxs Creek; they have a history of grazing, nutrient enrichment and weed invasion; and are moderately- to highly-degraded.

Some shrubby swamps on sandy alluviums, including in the Dunns Swamp area and upper Ganguddy Creek, have a scrub of *Leptospermum polygalifolium* (often colonising disturbed sites), and are probably referable to the Upper Cudgegong Alluvial Shrub-swamps of Bell (1998a). Upper Cudgegong Alluvial Reedland (fringing vegetation around the Kandos Weir impoundment) and Upper Cudgegong Alluvial Shrub-swamp of Bell (1998a) in the Dunns Swamp area are not mires and were not considered potential habitat for

P. gigantea by Baird (2012). Other reedland (marsh) patches, however, occur in association with the *Carex* fens and can be considered part of the mire systems. Swamps in the lower parts of the Cudgegong River catchment (east of Rylstone) below Dunns Swamp and along lower Coxs Creek, occur in the Capertee soil landscape on Permian geology (Kovac & Lawrie, 1991).

Additional seasonally wet swamps occur to the north of the catchment watershed in headwaters of the northward flowing Growee River catchment (e.g., Spring Log Swamp on Spring Log Creek), but these are on primarily mineral sediments and are not mires. Growee Swamp (500 m elevation), which occurs further downstream on the Growee River to the north, was not visited, but it occurs on cleared grazing land in a lower rainfall area, and is also assumed to be on seasonally drying alluvial sediments or heavier clayey soils.

Table 1: Identified mires of the upper Cudgegong River catchment (arranged by decreasing elevation) showing - site code, location, coordinates, approximate elevation range (upper to lower), approximate length or area, mire type and condition. An unsurveyed Cudgegong River mire above Dunns Swamp has been included. Mire types - Tableland Bog (TB), Tableland Swamp Meadow (TSM), and Hanging Swamp (HS). The locations of these mires are indicated in Fig. 2.

Mire site	Location	Coordinates (Google Earth)	Elevation (m) (Google Earth)	Length or area	Mire type	Condition
CM01	Rollen Creek	32° 52' 47" S, 150° 17' 01" E	745-700	3 km	TB, HS, TSM	Good
CM07	Coxs Creek	32° 44' 27" S, 150° 09' 56" E	740-660	10 km	TSM	Poor-Good
CM08	Coxs Creek margins adjoining CM07	32° 44' 43" S, 150° 10' 18" E	724	20 × 20 m	TB	Poor
CM09	Coxs Creek margins adjoining CM07	32° 44' 37" S, 150° 10' 07" E	722	40 × 40 m	TB	Poor
CM10	Coxs Creek margins adjoining CM07	32° 44' 35" S, 150° 09' 48" E	718	150 × 40 m	TB	Poor-Moderate
CM04	Above Ganguddy Creek	32° 51' 22" S, 150° 12' 52" E	680-678	100 × 5 m	HS	Good
CM05	Ganguddy Creek below and adjoining CM04	32° 51' 22" S, 150° 12' 52" E	678	12 × 10 m	TB	Moderate -Good
CM02	Never Never Creek tributary	32° 49' 26" S, 150° 12' 49" E	677-665	200 m	TB/TSM	Good
CM06	Ganguddy Creek tributary at junction with Ganguddy Creek	32° 51' 07" S, 150° 12' 51" E	674	15 × 10 m	TB	Moderate
CM03	Never Never Creek	32° 49' 32" S, 150° 12' 46" E	672-656	1.7 km	TSM	Good
Not surveyed	Cudgegong River above Dunns Swamp, upstream and downstream of the confluence with Rollen Creek	32° 05' 55" S, 150° 17' 04" E (indicative location)	741-719		TSM, TB	Very poor

Table 2: Native plant species (including mosses) recorded for each of the mires surveyed in the upper Cudgegong River catchment. Refer to Table 1 for details.

SPECIES	Family	CM01	CM02	CM03	CM04	CM05	CM06	CM07	CM08-10
<i>Asperula gunnii</i>	Rubiaceae	1							
<i>Baeckea utilis</i>	Myrtaceae	1							
<i>Baloskion fimbriatus</i>	Restionaceae		1						1
<i>Baloskion australe</i>	Restionaceae	1							
<i>Baumea</i> sp.	Cyperaceae	1	1		1		1	1	1
<i>Blechnum nudum</i>	Blechnaceae	1			1	1			
<i>Bulbine bulbosa</i>	Liliaceae	1							
<i>Callistemon citrinus</i>	Myrtaceae	1	1	1	1	1	1		
<i>Carex appressa</i>	Cyperaceae							1	
<i>Carex gaudichaudiana</i>	Cyperaceae	1	1	1	1	1	1	1	1
<i>Centella asiatica</i> (or <i>cordifolia</i>)	Apiaceae	1							1
<i>Centrolepis?</i>	Centrolepidaceae	1							
<i>Comesperma retusum</i>	Polygalaceae	1							
<i>Drosera binata</i>	Droseraceae	1			1				1
<i>Drosera peltata/auriculata</i>	Droseraceae				1				
<i>Drosera spathulata</i>	Droseraceae	1							
<i>Eleocharis sphacelata</i>	Cyperaceae			1				1	
<i>Empodisma minus</i>	Restionaceae	1			1				
<i>Epacris microphylla</i>	Ericaceae	1			1				
<i>Epacris paludosa</i>	Ericaceae	1				1	1		
<i>Eriocaulon scariosum</i>	Eriocaulaceae	1							
<i>Eucalyptus camphora</i> subsp. <i>camphora</i>	Myrtaceae	1	1	1	1	1	1		1
<i>Eucalyptus pauciflora</i>	Myrtaceae	1							
<i>Gahnia sieberiana</i>	Cyperaceae				1				
<i>Geranium neglectum</i>	Geraniaceae	1	1						1
<i>Gleichenia dicarpa</i>	Gleicheniaceae	1			1	1			
<i>Gonocarpus micrantha</i>	Haloragaceae	1							
<i>Goodenia</i> sp.	Goodeniaceae	1			1				
<i>Gratiola peruviana</i>	Scrophulariaceae						1		
<i>Gymnoschoenus sphaerocephalus</i>	Cyperaceae	1							
<i>Hakea microcarpa</i>	Proteaceae	1							1
<i>Hybanthus?</i>	Violaceae	1							
<i>Hydrocotyle</i> sp. (<i>sibthorpioides</i> ?)	Apiaceae	1					1		
<i>Hypericum</i> sp. (<i>gramineum</i> ?)	Hypericaceae	1							
<i>Isachne globosa</i>	Poaceae	1							
<i>Isotoma fluviatilis</i> ?= <i>Pratia surrepens</i>	Campanulaceae	1							
<i>Juncus</i> spp.	Juncaceae	1		1				1	1
<i>Lepidosperma limicola</i>	Cyperaceae	1							
<i>Leptospermum continentale</i>	Myrtaceae	1	1		1				1
<i>Leptospermum grandifolium</i>	Myrtaceae				1	1	1		
<i>Leptospermum obovatum</i>	Myrtaceae	1	1	1		1	1		1
<i>Lepyrodia</i> sp.	Restionaceae	1			1				
<i>Lythrum salicaria</i>	Lythraceae							1	
<i>Patersonia fragilis</i>	Iridaceae	1							
<i>Phragmites australis</i>	Poaceae		1	1				1	
<i>Polytrichum</i> sp. (<i>Dawsonia</i> sp.?)	Polytrichaceae					1			
<i>Pultenea divaricata</i>	Fabaceae	1							
<i>Pultenea</i> sp.	Fabaceae	1							
<i>Ranunculus</i> sp.	Ranunculaceae	1							
<i>Senecio</i> sp.	Asteraceae	1							
<i>Sphagnum cristatum</i>	Sphagnaceae	1			1	1	1		1
<i>Spiranthes australis</i>	Orchidaceae	1							
<i>Stylidium graminifolium</i>	Stylidiaceae	1							1
<i>Tetrarrhena juncea</i>	Poaceae	1							
<i>Typha</i> sp. (<i>orientalis</i> ?)	Typhaceae							1	
<i>Utricularia dichotoma</i>	Lentibulariaceae	1							1
<i>Viola caleyana</i>	Violaceae	1				1			
<i>Xyris ustulata</i>	Xyridaceae	1							
<i>Xyris gracilis</i>	Xyridaceae	1							

Rollen Creek mire system: location and physiography

Because of its size, range of mire variation and good condition, the mire system along Rollen Creek deserves particular attention. Apparently once known as Rotten Creek (G. Summers pers. comm.), Rollen Creek is an upper Cudgegong River tributary about 13 km east of Olinda. The upper swamp is located in Coricudgy State Forest, with the contiguous downstream section of good quality swamp on freehold land (“Inglewood”) extending to where the Coricudgy Road crosses ($32^{\circ} 52' 18''$ S, $150^{\circ} 16' 59''$ E). Though cattle have historically had some access to at least parts of the swamp upstream of the Coricudgy Road crossing, it is generally unsuitable for cattle, largely due to its high water table and soft deep peaty soils (G. Summers pers. comm.). A somewhat degraded area of sedgeland-heath bog remnant also occurs downstream of the Coricudgy Road crossing (between the road and cleared grazing land) towards the Cudgegong River junction, but has been impacted by grazing, fire and a previous timber mill with associated sawdust dump which was located below the crossing (G. Summers pers. comm.).

The Rollen Creek mire system (CM01) is narrow and elongate, oriented roughly NNW-SSE along the creek valley, about 3 km long (about 2.5 km in the State Forest) and varies in width from about 10 to 100 m, but is mostly <40 m wide. The catchment is Narrabeen Sandstone with the exception of the isolated peaks of Mt Coorongooba (~1060 m) and Mt Darcy (1079 m), with their residual basalt, Coricudgy soil landscapes. Approximately 20% of the slopes of Mt Coorongooba, and almost 50% of the slopes of Mt Darcy are in the Rollen Creek catchment. The area of the catchment is approximately 18 km², but with the complexity of the hydrogeology of these Narrabeen sandstones, aquifers may be collecting water from a larger area. There is a conspicuous seepage/spring at the head of the main creek and several small drainage lines enter the swamp system along its length; some also have areas of groundwater-fed peaty swamps with sedgeland-heath vegetation along their lower sections where they join Rollen Creek.

The head of Rollen Creek mire is about 2 km from the Divide (to the south); Mt Darcy and Mt Coorongooba are both within 3 km of the head of the mire. Elevation ranges from about 745 m at the source of the mire to 700 m at the road crossing, giving a low overall gradient of 1.5% (15 m per km) and within the range of Newnes Plateau Shrub Swamps (NPSS) (Benson & Baird, 2012). The downstream end lacks a nick-point waterfall, characteristic of NPSS, but grades into a *Carex gaudichaudiana*-dominated fen above the Coricudgy Road crossing, and transitions to cleared grazing land downstream of the Coricudgy Road crossing, with the exception of the patch of remnant sedgeland-heath bog below the Coricudgy Road. The extent of the fen area is likely to have been increased (with conversion from sedgeland-heath bog) by damming associated with the road

crossing. The mire system is surrounded by dry eucalypt woodland on poor sandy soils on adjoining slopes.

Rollen Creek mire system: vegetation structure and composition

The Rollen Creek mire system is dominated by extensive, valley-floor, sedgeland-heath bog, referable to the Tableland Bogs typology. In addition to the seepage/spring at the head of the main creek, there are small seepage/spring-fed mire patches adjacent to the main valley-floor mire in at least two other locations further downstream. There is also an extensive valley-side hanging swamp dominated by *Gymnoschoenus sphaerocephalus*.

The Tableland Bog along the main Rollen Creek valley floor includes areas of closed and open sedgeland, with a variable proportion of shrubs forming either sedgeland-heath or with a mallee or tall shrub canopy (Figures 3–10). The swamp is essentially treeless, though much of it is dominated by areas of mallee or multi-stemmed, shrubby *Eucalyptus camphora* subsp. *camphora* plants. Towards the head of the mire system, where the valley narrows, older *Eucalyptus camphora* occasionally reach 6–8 m high. In this area, this mallee/tall shrub canopy (sometimes low open woodland) generally forms dense scrub associated with *Callistemon citrinus* and *Leptospermum* shrubs up to 5 m high. Along the lower parts of the swamp system, however, *Eucalyptus camphora* plants are much smaller and often less than 2 m high. Plants with a range of sizes/ages indicate continual recruitment of these species. Parts of the upper swamp system with a canopy of *Eucalyptus camphora* and other large shrubs also have a wet meadow/forb-land ground-layer, which includes various herbs, *Carex gaudichaudiana*, *Sphagnum cristatum* and unidentified grasses (Fig. 3). A similar vegetation variant was also recorded along a small tributary (CM02) adjoining the *Carex* fen along Never Never Creek (CM03), along lower Ganguddy Creek near the Coricudgy Road bridge (CM05, CM06), and in association with small patches of sedgeland-heath bog (CM08–10) along margins of the *Carex* fen along upper Coss Creek (CM07).

Throughout the sedgeland-heath dominated valley-floor bog there may be an open or sparse, small-medium height shrub layer, including *Baeckea utilis*, *Comesperma retusum*, *Epacris microphylla*, *Epacris paludosa*, *Hakea microcarpa*, *Leptospermum obovatum* and *Pultenea divaricata*. The groundcover is generally very dense, mostly >90% cover, predominantly of *Sphagnum* moss, sedges and other graminoids, smaller shrubs and herbs, including carnivorous species such as *Drosera binata* and *Utricularia dichotoma*. Herbs and ferns include *Centella asiatica*, *Geranium* sp., *Gleichenia dicarpa*, *Goodenia* sp., *Isachne globosa*, *Spiranthes australis* and *Viola caleyana*. Sedgeland species within the sedgeland-heath include *Baumea* sp., *Baloskion australe*, *Empodisma minus*, *Gymnoschoenus sphaerocephalus*, *Juncus* sp., *Lepyrodia*

sp., *Tetrarrhena juncea* and *Xyris ustulata*. There are patches with *Sphagnum* hummocks up to 1 m high and the soil surface is spongy on flat and steeper side slopes (Figures 3, 10). There may also be localised areas of *Sphagnum* bog in shadier, wetter and more fire-protected sites along tributary drainage lines.

Carex gaudichaudiana is scattered throughout the valley floor mire in localised patches of fen amongst sedgeland-heath bog in wetter low gradient areas along the drainage lines (Fig. 6), and as a discrete patch of *Carex*-dominated fen referable to Tableland Swamp Meadow at the downstream end near the Coricudgy Road crossing (Fig. 11).

In hanging swamps on side slopes open patches of sedgeland with few or no shrubs may have *Gymnoschoenus sphaerocephalus* and *Gleichenia dicarpa* dominant, with *Empodisma minus*, *Lepidosperma limicola* and *Xyris ustulata* less abundant (Figures 8–10). *Baeckea utilis* generally occurs as scattered individuals across these *Gymnoschoenus*-dominated hanging swamp areas, with occasional other shrubs present. Upper hanging swamp margins are relatively distinct with a band of *Gleichenia* along the edge of the dry eucalypt woodland; *Eucalyptus camphora* also occurs on swamp margins.

The main woodland dominants adjoining the swamps are *Eucalyptus dives*, *Eucalyptus radiata*, *Eucalyptus rossii* and *Eucalyptus dalrympleana*. In one place (32° 53' 24" S, 150° 17' 24" E), the geographically restricted *Eucalyptus corticosa* was recorded in woodland adjacent to the upper margin of the hanging swamp on the eastern side of the Rollen Creek mire. *Eucalyptus pauciflora* may occur on woodland margins sometimes edging into the swamp, as it does in the higher elevation mires of the Newnes and Boyd plateaus further south.

About 48 native species, including most of the conspicuous, abundant and structural species, were recorded in Rollen Creek mire (Table 2). This is an incomplete list as some grasses and monocots were not identified, and some seasonal species may not have been evident at the time. Appendix 1 includes the list of species recorded in Rollen Creek swamp with reference to species also recorded in NPSS (Benson & Baird, 2012; Benson & Keith, 1990) and Boyd Plateau Bogs (Keith & Benson, 1988; Kodela et al., 1996).

Rollen Creek mire system: hydrology

Given the relatively low average annual rainfall compared to the nearby basalt-capped mounts and the higher elevation parts of the Blue Mountains further south, and its high inter-annual variability at the property “Kelgoola”, the presence of the well-developed mire system along Rollen Creek indicates a strong groundwater influence, in addition to rainfall input. This is exemplified by the presence of springs and hanging swamps with groundwater seepage adjacent to the valley floor mire. The Narrabeen Sandstone of the upper Blue Mountains is also characterised by complex

groundwater hydrology and the presence of aquifers which support a diversity of groundwater-dependent swamp communities (e.g., Baird, 2012; DLWC, 1999a, b; Marshall, 2005). The expected higher rainfall in the higher elevation areas above 1000 m, around the catchment watershed in the headwaters of the upper Cudgegong River catchment, is likely to contribute significantly to the presence of the groundwater-dependent mires through both direct runoff, and infiltration into the groundwater system in the relatively permeable geology.

The characteristic swamp vegetation and extensive occurrence of peaty/organic-rich mire sediments indicates a permanently high water table in Rollen Creek mire. Peaty sediments include fibrous peats and more highly decomposed sapric peats, which require such hydrological conditions to develop. Depth and characteristics of the peaty/organic-rich sediments can be expected to vary considerably across the swamp system, as occurs in NPSS for example, but the depths recorded in the downstream part of Rollen Creek swamp (>1.8 m) typically exceeded depths recorded in NPSS by Benson and Baird (2012). They are, however, similar to depths recorded in the somewhat similar, low gradient montane bog and fen complex at the head of Long Swamp (part of the Coxs River Swamps of Benson & Keith, 1990) in Ben Bullen State Forest (Baird, 2014; IRCB, unpubl. obs.; Martin, 2017).

The hanging swamps on the valley side, dominated by *Gymnoschoenus sphaerocephalus*, and contiguous with the main valley floor mire (Figures 8–10), are similar to those which occur elsewhere in the Blue Mountains on sandstone geology. These hanging swamps are typically associated with the presence of a low-permeability aquitard (e.g., claystone layer). They develop where groundwater is redirected downslope following the slope of the aquitard to emerge on the valley side as seepage. These hanging swamps along Rollen Creek include groundwater seeps and springs with associated groundwater-dependent species such as burrowing crayfish (*Euastacus* sp.) (Fig. 10).

The predominance of *Sphagnum* in the Rollen Creek mire system is indicative of persistent or permanent wetness. The hummocks are slow growing and the large hummocks were associated with large shrubs with a cluster of basal stems, or large tussocks of sedgeland vegetation, which appear to provide support for the hummock growth and provide moderate levels of shade. Larger *Sphagnum* hummocks in Boyd Plateau Bogs are similarly associated with multi-stemmed shrubs and sedgeland tussocks (IRCB unpubl. obs.). The *Sphagnum* is vulnerable to fire during drought, when the hummocks may dry out (see Hope et al., 2009; Whinam, 1995; Whinam & Chilcott, 2002; Whinam et al., 1989).



Fig. 3: Dense *Sphagnum* cover and hummocks in *Carex* and grass-dominated area in the upper section of Rollen Creek swamp, with *Eucalyptus camphora* subsp. *camphora*. Photo by Ian Baird



Fig. 4: Lignotuberous *Eucalyptus camphora* subsp. *camphora* resprouting after fire along a drainage line through a *Gymnoschoenus*-dominated patch of swamp in the upper Rollen Creek. Photo by Ian Baird



Fig. 5: Dense montane sedgeland-heath bog along the Rollen Creek valley floor. Photo by Ian Baird



Fig. 6: A patch of *Carex gaudichaudiana* fen amongst sedgeland-heath in a low gradient section of Rollen Creek swamp. Note the emergent water table (bottom centre) and *Eucalyptus camphora* subsp. *camphora* in front of adjoining woodland at rear. Photo by Ian Baird



Fig. 7: Valley floor bog with a patch of *Empodisma minus* and *Tetrarrhena juncea*-dominated ground-layer, with low *Eucalyptus camphora* subsp. *camphora* (rear and right). Photo by Ian Baird



Fig. 8: *Gymnoschoenus*-dominated hanging swamp in the upper section of Rollen Creek. The fringing low *Eucalyptus camphora* subsp. *camphora* is in front of adjoining upslope woodland at rear. Photo by Ian Baird



Fig. 10: Seepage in a *Gymnoschoenus*-dominated section of a hanging swamp in the upper Rollen Creek with *Sphagnum*, *Gleichenia dicarpa* and scattered shrubs of *Baeckea utilis* evident. A *Euastacus* sp. crayfish burrow is visible in the pool. Photo by Ian Baird



Fig. 9: Resprouting *Gymnoschoenus sphaerocephalus* tussocks post-fire in a hanging swamp in the upper Rollen Creek swamp. Note the large size (and age) of the bases of tussocks, fire-killed shrubs and fringing resprouting *Eucalyptus camphora* subsp. *camphora* (rear). Photo by Ian Baird



Fig. 11: *Carex gaudichaudiana* –dominated montane fen (centre left) in a low gradient area of the downstream end of the Rollen Creek swamp, with fringing sedgeland-heath bog. Photo by Ian Baird



Fig. 12: Valley-floor *Carex* fen along Never Never Creek above Dunns Swamp, with scattered *Eucalyptus camphora* subsp. *camphora* and *Leptospermum* shrubs. Photo by Ian Baird



Fig. 13: Valley-floor *Carex* fen with flowering *Lythrum salicaria* along Coxs Creek (looking upstream) showing cleared grazing land on private property on right-hand (southern) side and good quality bushland on the left-hand side. Small patches of fringing sedgeland-heath bog, with some *Eucalyptus camphora*, occur in embayments along fen margins at right. Photo by Ian Baird

Discussion

Comparison of montane mires of the upper Cudgegong River catchment with other montane mires of the Blue Mountains

We found a range of mires with considerable floristic and hydrogeomorphic diversity across the upper Cudgegong River catchment. These mires can be broadly divided into montane bogs, montane fens and hanging swamps; all groundwater dependant ecosystems typically found in high rainfall areas. Keith (2004) indicated a rainfall range of 800–1500 mm/year for the Montane Bogs and Fens vegetation class and 1000–1500 mm/year for the Coastal Heath Swamps vegetation class. The Cudgegong mires are at the extreme lower limits of these ranges. The upper

Cudgegong montane bogs (Tableland Bogs) and montane fens (Tableland Swamp Meadows) occur in a lower rainfall area, and at relatively lower elevation, compared to similar Blue Mountains mires further south. Mires in Ben Bullen State Forest and on the Newnes Plateau are above 900 m (but most above 1000 m), compared to those at 660–745 m in the upper Cudgegong (Table 1). Mean annual rainfall for these central Blue Mountain mires generally exceeds 1000 mm, and on the Boyd Plateau and nearby areas (above 1100 m elevation) exceeds 1100 mm.

The hanging swamps of the upper Cudgegong River catchment, particularly along upper Rollen Creek, also occur in a lower rainfall area than similar hanging swamps on sandstone in the central Blue Mountains (BMSS), particularly those on the eastern side of the Blue Mountains at similar elevation, where mean annual rainfall also exceeds 1100 mm, compared to mean annual rainfall of 809 mm at “Kelgoola”, near Rollen Creek mire. In upper parts of the Blue Mountains, where the best developed hanging swamps on sandstone occur, mean annual rainfall may exceed 1300 mm. In lower elevation areas with considerably lower rainfall and higher evapotranspiration in the lower reaches of the upper Cudgegong River catchment east of Rylstone (mean annual rainfall <750 mm), to the north of the upper Cudgegong River catchment in the Growee River catchment, and to the south in the Capertee River catchment, suitable conditions for mire development are evidentially absent.

Floristically, the Rollen Creek mire is similar to NPSS with about 65% of the 48 recorded native species in common, particularly shrub species (see Appendix 1), though the predominance of *Eucalyptus camphora*, restricted mainly to these swamps, is a conspicuous point of difference. *Eucalyptus camphora* subsp. *camphora* is geographically restricted to Nullo Mountain and the upper Cudgegong River catchment, with a localised disjunct population in swamps in the Megalong Creek valley near Katoomba. It is also of note in being restricted mainly to swamp habitat; indeed there are very few *Eucalyptus* species that are found in wet or poorly drained sites and *Eucalyptus camphora* appears to be confined to this habitat. The only similar swamp-inhabiting eucalypt in the Sydney Basin bioregion (there are almost 100 eucalypt species in the the GBMWA; Benson & Smith, 2015) is the closely related *Eucalyptus aquatica*, which is found in analogous montane mires in the Southern Highlands (Penrose State Forest) (Shepherd & Keyzer, 2014).

The abundance of *Callistemon citrinus* along Rollen Creek is another conspicuous difference. This species does not occur in NPSS or Boyd Plateau Bogs, but is the only *Callistemon* found in BMSS (and some transitional to NPSS). Interestingly, *Callistemon pityoides*, the typical montane bog *Callistemon* (found in Boyd Plateau Bogs, some NPSS, and montane bogs elsewhere), was not recorded anywhere in the upper Cudgegong swamps, nor does it occur in BMSS. Other conspicuous NPSS species not so far recorded are *Celmisia* sp. (aff. *longifolia*), *Boronia deanei* subsp. *deanei* and *Grevillea acanthifolia* subsp. *acanthifolia*. *Celmisia* and *Boronia deanei* also occur in Boyd Plateau Bogs. These species may be more drought-sensitive and given the much less extensive

swamp areas on the upper Cudgegong, in comparison with the main Blue Mountains area further south, could have been extirpated by past periods of extensive drought, assuming they have been present in the past.

Although *Leptospermum grandifolium* was recorded in the small patch of hanging swamp above Ganguddy Creek (CM04) and a nearby small tributary bog (near and downstream of the bridge on the Coricudgy Road) (CM06), the apparent lack of the species in Rollen Creek swamp and other surveyed sites is noteworthy, considering that it is a common shrub in NPSS, BMSS and Boyd Plateau Bogs (interestingly, single plants of *Epacris paludosa*, which was otherwise only recorded from Rollen Creek swamp, were also recorded in CM05 and the nearby CM06). There is also a general lack of dominance by *Leptospermum* species, compared to their abundance in many NPSS. The only species recorded in Rollen Creek swamp was *Leptospermum obovatum*, which is common in Rollen Creek swamp, occurs in some NPSS, and is common in Boyd Plateau Bogs. *Leptospermum continentalis* also occurs on heavier soils on drier margins of swamps. *Baeckea utilis*, the only *Baeckea* species present, and widely distributed in Rollen Creek swamp, is a typical montane bog and wet heath species which occurs in and around Boyd Plateau Bogs and some higher elevation Newnes Plateau swamps, often on heavier soils, but does not occur in BMSS. *Baeckea utilis* is typically replaced by *Baeckea linifolia* in most NPSS and in the higher elevation Coastal Heath Swamps of the region such as BMSS.

Most other abundant species are similar to those occurring in NPSS, including *Lepidosperma limicola*, *Empodisma minus*, *Epacris paludosa* and *Leptospermum obovatum*. Although *Gymnoschoenus sphaerocephalus* occurs in small patches within the main valley floor bog and in upstream areas where the valley narrows, it is mostly restricted to the hanging swamps and seepage/spring-fed mire patches adjacent to the main valley floor mire. Rollen Creek was the only mire in the upper Cudgegong catchment where *Gymnoschoenus* was recorded, although it has been recorded just east of Mt Coricudgy, in a previously undocumented small hanging swamp (990 m elev.) where *Petalura gigantea* was recorded (Baird, 2012: Appendix 1, site PMC01). *Gymnoschoenus* is widely distributed in mires on sandstone elsewhere in the Blue Mountains (in NPSS and BMSS) and the Southern Highlands, and montane bogs of the Northern and Southern Tablelands (mostly on granite), but it appears to be absent from the granite-based Boyd Plateau Bogs. The only occurrence in Tableland Bog on granite in the Blue Mountains region which is known to one of the authors (IRCB) is in a small montane bog south of the Kowmung River near Trailers Mountain, although it may also occur in several small unsurveyed bog patches nearby. Floristically about 58% of the 48 recorded native species in Rollen Creek swamp are shared with the Boyd Plateau Bogs, including herbaceous species such as *Centella asiatica*, *Geranium* sp., *Hypericum* sp., and *Viola caleyana* (Appendix 1).

Areas of hanging swamp dominated by *Gymnoschoenus sphaerocephalus* (Figures 8–10) are not referable to any

of the swamp vegetation types in the upper Cudgegong area identified by Bell (1998a). With the exception of the occurrence of *Eucalyptus camphora* and *Baeckea utilis*, these hanging swamps are more similar floristically and hydrogeomorphically to those occurring elsewhere on Narrabeen Sandstone in the Blue Mountains, such as the BMSS of Keith and Benson (1988).

The presence of extensive areas of *Sphagnum cristatum* along Rollen Creek, often forming large hummocks within the sedges and shrubs, was noteworthy. In comparison with Rollen Creek swamp, *Sphagnum* is either very restricted or absent from NPSS and BMSS (but note the extensive and unusual *Sphagnum* cover in the fen/marsh in Goochs Crater in the upper Wollangambe River catchment and in a BMSS in McCrae's Paddock in Katoomba). The much more extensive *Sphagnum* cover in Rollen Creek swamp (and elsewhere in mires and other parts of the upper Cudgegong catchment) is surprising considering the lower rainfall recorded at nearby "Kelgoola", compared to NPSS and other central Blue Mountains mires. Its persistence is likely to be the result, at least in part, of a sustained groundwater influence and the nearby presence of higher elevation peaks upstream around the catchment watershed contributing additional rainfall inputs as a result of an orographic effect. According to Chris Pavich (pers. comm.), *Sphagnum* was more abundant in the upper Cudgegong before the drought of the 1930-40s, with a subsequent increase in fire and grazing probably contributing to its further disappearance (C. Pavich pers. comm.). Rollen Creek swamp, however, has not been subject to heavy grazing or frequent anthropogenic fire (G. Summers pers. comm.), which may have helped prevent loss of *Sphagnum*. *Sphagnum* is generally much more abundant in montane bogs on granite, such as those of the Boyd Plateau, and nearby areas on metasedimentary geology (e.g., near Mt Werong and in Jenolan State Forest), than in the swamps developed on sandstone in the Blue Mountains, such as the NPSS and BMSS (also see Downing et al., 2007; Whinam & Chilcott, 2002). Differences in fire history and climatic factors between these areas are likely to be contributing factors.

While there are small patches of *Sphagnum* bog in the upper Cudgegong which are broadly consistent with Bell (1998a)'s Cudgegong Sphagnum Bogs, these are treated here as part of a variable upper Cudgegong montane bog type. *Sphagnum* also occurs along various drainage lines (often on heavy soils) and in localised patches within the other swamp types (including areas transitional between *Carex* fen and sedgeland-heath bog) where suitable conditions occur. The Cudgegong River Swamp Grassland and Upper Cudgegong Alluvial Sedgeland of Bell (1998a) are also treated as part of the montane fens identified in this study. As a result of more extensive survey across the study area, and based on their broadly similar floristics and potential organic-rich substrates, as described by Bell (1998a) in the Dunns Swamp area, Baird (2012) noted that there is considerable gradation between these swamp vegetation types.

In summary, the Rollen Creek swamp includes extensive valley floor mire of intergrading bog and fen, with some

very small valley-side seepage/spring areas, and extensive valley-side hanging swamp dominated by *Gymnoschoenus*. The valley-side seeps, springs and hanging swamps are most similar to other such groundwater-dependent mire expressions developed on Narrabeen Sandstone in the Blue Mountains, such as the NPSS and BMSS. While these seeps and hanging swamps are hydrogeomorphically similar to similar mires developed on sandstone geology elsewhere in the Blue Mountains, they are characterised by a somewhat distinctive floristic assemblage. BMSS form part of the Coastal Heath Swamp vegetation class (NSWSC, 2007), although most are above the nominal 600 m upper elevation range indicated by Keith (2004) for the Coastal Heath Swamps. NPSS are considered transitional between the Coastal Heath Swamps and Montane Bogs and Fens vegetation classes (NSWSC, 2005b), with that transition occurring between Bell and Clarence. The valley floor mire along Rollen Creek, however, with areas of bog and fen, shows greater affinity, respectively, to the Tableland Bogs and Tableland Swamp Meadows of Tozer et al. (2010), within the Montane Bogs and Fens vegetation class. The valley-floor bogs along Rollen Creek and elsewhere in the upper Cudgegong have floristic and hydrogeomorphic similarities with montane bogs of the Southern, Central and Northern Tablelands (e.g., Hunter & Bell, 2007; Tozer et al., 2010). The *Carex* fens along Coxs Creek and Never Never Creek, at the downstream end of Rollen Creek swamp, and similar areas across the upper Cudgegong River catchment, have affinities with the *Carex* fen vegetation of Northern NSW (Hunter, 2013; Hunter & Bell, 2009), the *Carex-Poa* fen vegetation of Long Swamp in Ben Bullen State Forest and nearby areas (part of the Coxs River Swamps of Benson & Keith, 1990), the swampy meadows of the Central Tablelands (see Mactaggart et al., 2008; Mactaggart, 2008) and the Mountain Hollow Grassy Fens (DEC, 2006). This complexity highlights the value of this and other geographically isolated mires of the upper Cudgegong River catchment.

Conservation value of Rollen Creek swamp and other remaining mires of the upper Cudgegong River catchment

Mires are geographically restricted ecosystems in Australia and their extent and health have been considerably reduced and degraded since European settlement through urban and transport infrastructure development, agriculture, drainage, grazing, and more recently through mining impacts, particularly the impacts of subsidence from longwall coal mining, a Key Threatening Process in NSW (CoA, 2005, 2010; NSWSC, 2005a). Climate change and fire are also recognised as significant threats to these ecosystems (e.g., Baird & Burgin, 2016; CoA, 2010; Keith et al., 2014; Pemberton, 2005).

In the Sydney Basin bioregion, Temperate Highland Peat Swamps on Sandstone (THPSS) is listed as an Endangered Ecological Community (EEC) under the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (CoA, 2005). THPSS also includes NPSS and BMSS. Under the NSW *Biodiversity Conservation Act 2016* (which replaced the NSW *Threatened Species Conservation Act*

1995) NPSS is also listed as an EEC (NSWSC, 2005b), and Blue Mountains Swamps (including the Blue Mountains Sedge Swamps of Keith and Benson (1988)) is listed as a Vulnerable Ecological Community (NSWSC, 2007). Although not specifically included in the description of THPSS, the authors consider that some of the mires of the upper Cudgegong, particularly the Rollen Creek valley floor mire and hanging swamps, are clearly referable to the THPSS EEC, and have been simply overlooked because of lack of documentation. For a similar reason, none of the valley-floor bogs and fens in the upper Cudgegong are specifically identified in the Montane Peatlands and Swamps of the New England Tableland, NSW North Coast, Sydney Basin, South East Corner, South Eastern Highlands and Australian Alps bioregions EEC determination (NSWSC, 2004). In circumscribing the Montane Peatlands and Swamps EEC, the NSWSC (2004) referred to swamps above 400–600 m elevation. The montane bogs and fens of the upper Cudgegong are also referable to this EEC and are clearly an important part of the complex of endangered montane mire communities distributed across the tablelands and adjacent ranges of NSW.

Habitat for rare species-flora

The mires of the upper Cudgegong River catchment are isolated from other montane mires. The nearest montane mires to the north occur at high elevation at Barrington Tops (Mort, 1983), although somewhat drier and more floristically impoverished wet heath vegetation, with some floristic affinity to the wet heath in the upper Cudgegong mires, occurs on heavier soils on Coolah Tops (Binns, 1997). The nearest to the south occur on the Newnes Plateau and the adjacent Ben Bullen State Forest.

This isolation contributes to the significance of some rare and restricted plant species in the mires and adjacent woodlands of the upper Cudgegong. For example, the distribution of the mallee *Eucalyptus camphora* subsp. *camphora*, illustrates the disjunct biogeography of many montane species restricted to specialised habitats. This taxon is geographically restricted to higher elevation mires and swampy drainage lines along this part of the Divide, with populations at Nullo Mountain and in the upper Cudgegong River catchment (Rollen Creek swamp in particular has a large population), and a highly localised and disjunct population 90 km further south in the Megalong Creek Valley near Katoomba, where it occurs in swamp patches with different floristics on heavier clayey soil and colluviums on Permian geology (600 m elev.). This locality in the Megalong Creek valley is also noteworthy for the occurrence of the rare and locally endemic *Callistemon megalongensis* (Craven, 2009; Udovicic & Spencer, 2012) and *Callistemon purpurascens* (Douglas & Wilson, 2015).

Another rare eucalypt, *Eucalyptus corticosa*, a locally endemic tree species restricted to the upper Cudgegong valley east of Rylstone, occurs in eucalypt woodland on shallow infertile soils on sandstone ridges and was recorded on the outer edge of the *Gymnoschoenus*-dominated hanging swamp along Rollen Creek. *Veronica blakelyi*, a small shrub restricted to the western Blue Mountains, near Clarence,

near Mt Horrible, on Nullo Mountain and in the Coricudgy Range, is listed as Vulnerable under the NSW *Biodiversity Conservation Act 2016* and was recorded in woodland adjoining Rollen Creek swamp. Additional rare plant species are likely to occur in the woodlands adjoining these swamps, including in Coricudgy State Forest.

Habitat for rare species- fauna

The mires of the upper Cudgegong, particularly Rollen Creek, may also provide habitat for specialist mire fauna. Baird (2012) considered that the upper Cudgegong mires, particularly the valley-floor bogs and *Gymnoschoenus*-dominated hanging swamps, provide potential habitat for the endangered dragonfly, *Petalura gigantea*. The record of a single *Petalura* in a small hanging swamp (one of several such previously undocumented swamps in close proximity) east of Mt Coricudgy in Wollemi National Park, a considerable distance from the nearest known populations on the Newnes Plateau (Baird, 2012), strongly suggests that a population occurs somewhere within the complex of mires in the upper Cudgegong. Rollen Creek swamp is the most likely habitat, although surveys on three occasions have not recorded the species.

Rollen Creek swamp also appears to provide suitable habitat for the endangered Blue Mountains Swamp Skink, *Eulamprus leuraensis*, which is only known from mid-upper elevation mires in the central Blue Mountains (BMSS) and the Newnes Plateau (NPSS). This groundwater-dependent species has not been recorded further north than the Newnes Plateau (Gorissen, 2016; LeBreton, 1996). Evidence of foraging and tunnelling activity of Swamp Rats, *Rattus lutreolus*, a common species of BMSS and NPSS, was also observed during fieldwork. Additional obligate mire-dwelling fauna are likely to occur here (e.g., skinks and invertebrate stygofauna), but more detailed surveys will be required to identify them.

A burrowing spiny crayfish, *Euastacus australasiensis*, is widely distributed in Blue Mountains mires (including hanging swamps), and *Euastacus* burrows and partial remains were observed by the authors in the Rollen Creek mire system (including hanging swamps) and (by IRCB) in the small hanging swamp east of Mt Coricudgy where *Petalura gigantea* was recorded. While these are likely to be *Euastacus australasiensis* (see McCormack, 2012), the possibility exists that an unidentified swamp-dwelling taxon is involved. In the forested headwaters of the Cudgegong River in Coricudgy State Forest, below Mt Coricudgy and upstream of Rollen Creek, a large, stream-dwelling, burrowing spiny crayfish species, *Euastacus vesper*, closely related to *Euastacus spinifer* of eastern drainages, has recently been described (McCormack & Ahyong, 2017). The discovery of this apparently highly-localised species provides further evidence of the high conservation value of Coricudgy State Forest and of the opportunities for further research in this area.

Value of mires for palaeo-ecological, evolutionary and climate studies

Previous palaeoecological studies of Blue Mountains mires, based upon pollen and charcoal analysis of radiometrically-dated sediment cores, have greatly increased our understanding of the developmental history of these mires and their past climates and vegetation, particularly since the Last Glacial Maximum (~21 000 years BP). While the oldest sediment core ages suggest some mires may have commenced development around 13 000 years BP, sediment cores from other swamps have maximum ages within the Holocene (e.g., Black et al., 2008; Chalson & Martin, 2009; Fryirs et al., 2014; Martin, 2017). The mires of the upper Cudgegong River catchment can be expected to be a similarly rich source of knowledge related to species distributions, the developmental history of these mires and of climatic change since the Last Glacial Maximum, and expand our existing understanding of vegetation change across southeastern Australia and the Sydney Basin during this period.

The biogeography of the mallee *Eucalyptus camphora* subsp. *camphora*, geographically restricted to localised high elevation patches of suitable habitat along the Divide, is similar to that of many montane species with disjunct populations restricted to specialised habitats. Such disjunct distributions are likely to reflect, or be the result, of past climatic fluctuations to some extent, either during the Last Glacial Maximum or as a result of previous glacial/interglacial cycles. For *Eucalyptus camphora*, two other subspecies have also been recognised; subsp. *relicta*, found further north at Guyra and in Queensland, and subsp. *humeana*, occurring from Wee Jasper south into Victoria. The distribution of these taxa, with their greater morphological and geographic variation, presumably indicates older geographic separation than that within subsp. *camphora*, and provides evidence of past climate-related divergence. Other montane species restricted to specialised habitats, such as mires, have similar biogeographic patterns; understanding these patterns is of considerable scientific interest, particularly in the context of a rapidly changing climate.

Conclusion

The upper Cudgegong River catchment includes a complex of endangered peat swamp or mire types which are geographically disjunct from their nearest neighbours and characterised by some distinctive floristic assemblages. These include areas of montane bog, montane fen and hanging swamp. The presence of rare species such as the mallee *Eucalyptus camphora* subsp. *camphora*, and the potential for endangered fauna such as *Petalura gigantea* to be present, further highlights their value. In the context of the relatively low rainfall where these groundwater-dependent mires occur (and the upper Cudgegong is at their climatic limits), there is a surprising hydrogeomorphic and floristic diversity across these different mire types. With their relatively low rainfall, these mires may be particularly vulnerable to climate change.

Historically there has been considerable loss and degradation of the mires in the upper Cudgegong River catchment through land clearing and agriculture, beginning in the 1840s, and evidence of this damage provides a strong imperative to protect those examples that have survived. Conservation of these mires and their associated flora and fauna will benefit from further survey, mapping and biodiversity census. In addition to improved management of identified mires on private lands, including the possible use of biodiversity conservation covenants and incentives, improved recognition of the inherent values of these mires will be fundamental to their long-term conservation. Rollen Creek swamp is unique in the area, in terms of its size, floristic and hydrogeomorphic diversity, and good condition, and its conservation must be a priority. National Heritage listing of Coricudgy State Forest would provide a substantial foundation to highlight the values of this mire system and its surrounding woodland landscape, including the biodiversity associated with the significant, higher rainfall, basalt-capped island peaks adjoining the GBMWA. Such listing is a prerequisite for nomination of this and other high biodiversity areas for addition to the GBMWA and is recommended.

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Appendix 1: Native plant species (including mosses) recorded in Rollen Creek mire (TB=Tableland Bog, TSM=Tableland Swamp Meadow, HS= Hanging Swamp), and whether those species have also been recorded in Newnes Plateau Shrub Swamps (NPSS) (Benson & Baird, 2012) or Boyd Plateau Bogs (Kodala et al., 1996).

PLANT SPECIES	Family	Rollen Creek mire	NPSS	Boyd Plateau Bogs
<i>Asperula gunnii</i>	Rubiaceae	TB	NPSS	BPB
<i>Baeckea utilis</i>	Myrtaceae	TB, HS	NPSS	BPB
<i>Baloskion australe</i>	Restionaceae	TB, HS	NPSS	BPB
<i>Baumea</i> sp.	Cyperaceae	TB, HS	NPSS	BPB
<i>Blechnum nudum</i>	Blechnaceae	TB	NPSS	BPB
<i>Bulbine bulbosa</i>	Liliaceae	TB		
<i>Callistemon citrinus</i>	Myrtaceae	TB, HS		
<i>Carex gaudichaudiana</i>	Cyperaceae	TB, TSM		BPB
<i>Centella asiatica</i> (or <i>cordifolia</i>)	Apiaceae	TB		BPB
<i>Centrolepis?</i>	Centrolepiaceae	TB		
<i>Comesperma retusum</i>	Polygalaceae	TB		BPB
<i>Drosera binata</i>	Droseraceae	TB, HS	NPSS	BPB
<i>Drosera spathulata</i>	Droseraceae	TB	NPSS	
<i>Empodisma minus</i>	Restionaceae	TB, HS	NPSS	BPB
<i>Epacris microphylla</i>	Ericaceae	TB	NPSS	BPB
<i>Epacris paludosa</i>	Ericaceae	TB, HS	NPSS	BPB
<i>Epilobium gunnianum</i>	Onagraceae			BPB
<i>Eriocaulon scariosum</i>	Eriocaulaceae	TB		
<i>Eucalyptus camphora</i> subsp. <i>camphora</i>	Myrtaceae	TB, HS		
<i>Eucalyptus pauciflora</i>	Myrtaceae	TB	NPSS	BPB
<i>Gahnia sieberiana</i>	Cyperaceae		NPSS	
<i>Geranium neglectum</i>	Geraniaceae	TB		BPB
<i>Gleichenia dicarpa</i>	Gleicheniaceae	TB, HS	NPSS	BPB
<i>Gonocarpus micrantha</i>	Haloragaceae	TB	NPSS	BPB
<i>Goodenia</i> sp. (<i>bellidifolia</i>)	Goodeniaceae	TB	NPSS	
<i>Gymnoschoenus sphaerocephalus</i>	Cyperaceae	TB, HS	NPSS	
<i>Hakea microcarpa</i>	Proteaceae	TB	NPSS	BPB
<i>Hybanthus?</i>	Violaceae	TB		
<i>Hydrocotyle</i> sp. (<i>sibthorpioides?</i>)	Apiaceae	TB	NPSS	BPB
<i>Hypericum</i> sp. (<i>gramineum?</i>)	Hypericaceae	TB	NPSS	
<i>Isachne globosa</i>	Poaceae	TB		
<i>Isotoma fluviatilis?</i> = <i>Pratia surrepens</i>	Campanulaceae	TB		BPB
<i>Juncus</i> spp.	Juncaceae	TB, TSM	NPSS	BPB
<i>Lepidosperma limicola</i>	Cyperaceae	TB, HS	NPSS	
<i>Leptospermum continentale</i>	Myrtaceae	TB	NPSS	BPB
<i>Leptospermum myrtifolium</i>	Myrtaceae		NPSS	BPB
<i>Leptospermum obovatum</i>	Myrtaceae	TB, HS	NPSS	BPB
<i>Lepyrodia</i> spp.	Restionaceae	TB	NPSS	
<i>Patersonia fragilis</i>	Iridaceae	TB	NPSS	BPB
<i>Pultenea divaricata</i>	Fabaceae	TB, HS	NPSS	
<i>Pultenea</i> sp.	Fabaceae	TB		
<i>Ranunculus</i> sp.	Ranunculaceae	TB		
<i>Scirpus polystachyus</i>	Cyperaceae			BPB
<i>Senecio</i> sp.	Asteraceae	TB		
<i>Sphagnum cristatum</i>	Sphagnaceae	TB, HS		BPB
<i>Spiranthes australis</i>	Orchidaceae	TB		BPB
<i>Stylidium graminifolium</i>	Stylidiaceae	TB	NPSS	BPB
<i>Tetrarrhena juncea</i>	Poaceae	TB	NPSS	
<i>Utricularia dichotoma</i>	Lentibulariaceae	TB	NPSS	BPB
<i>Viola caleyana</i>	Violaceae	TB	NPSS	
<i>Xyris ustulata</i>	Xyridaceae	TB, HS	NPSS	BPB
<i>Xyris gracilis</i>	Xyridaceae	TB	NPSS	
TOTAL		48	33	31
Number RC shared with NPSS and/or BPB			31	28
Percentage RC shared with NPSS and/or BPB			65%	58%

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Insects associated with flowering of *Rhodomyrtus psidioides* (Myrtaceae): Is this a Myrtle Rust (*Austropuccinia psidii*)-induced Plant-pollinator interaction Extinction Event?

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Abstract: The threatened Australian endemic rainforest tree *Rhodomyrtus psidioides* (Myrtaceae) is visited and pollinated by a taxonomically diverse assemblage of mainly small, ecologically unspecialised, insects. Flower structure suggests that it may also be adapted for wind-pollination. However, the recent (2010) invasion by the aggressive fungal pathogen Myrtle rust (*Austropuccinia psidii*) has resulted in the local extinction of both the floral resource and associated plant-insect relationships. Here I table observed insect visitors to the flowers of *Rhodomyrtus psidioides* made before the impact of Myrtle rust - no other records appear to have been published.

Key Words: *Rhodomyrtus psidioides*, Myrtaceae, threatened plants, extinction, subtropical rainforest, Lorien Wildlife Refuge, anthophilous insects, wind-pollination.

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Introduction

In the 1990s, as part of my PhD investigation of the pollination ecology of lowland subtropical rainforests in northern New South Wales (Williams 1995), I undertook the observation and hand collection of insects that visited the flowers of the small myrtaceous tree *Rhodomyrtus psidioides* (G. Don) Benth.). This was one of numerous mass-flowering tree and shrub species I investigated over a three-season period (1990-1993); owing to time constraints observations of *Rhodomyrtus psidioides* were random rather than following a rigorous experimental protocol. Nevertheless, the proximity of several plants at one site (Lorien Wildlife Refuge) allowed opportunity for frequent casual observation, almost on a daily basis during seasonal flowering events in that period.

Rhodomyrtus psidioides is currently proposed for listing in New South Wales as *critically endangered* (Preliminary Determination, NSW Scientific Committee 2017) as it is severely threatened from infection from *Austropuccinia psidii* (Myrtle rust) an introduced pathogen first noted in NSW in 2010. Plants are extremely susceptible with all parts of the plant being affected and populations are threatened with extinction. Quantitative findings of recent very large declines in *Rhodomyrtus psidioides* populations due to *Austropuccinia psidii* infection reported in Carnegie *et al.* (2016) are supported by field botanists who have encountered the species during routine botanical surveys and seed collecting over multiple years (B. Makinson *in litt.* April 2016).

Rhodomyrtus psidioides is a large shrub or small tree endemic to Australia, distributed from Gosford on the central coast of New South Wales to Gympie in southeastern Queensland (Harden, 1991). Populations flower synchronously but flowers on individual trees open sequentially, however, not all regional populations flower each year (Williams 1995). The flowers are large (1.4cm) and usually clustered, individually last for 3-7 days, are bisexual but self-incompatible (Adam & Williams, 2001), fragrant, creamish-white in colour, and with yellow, slightly sticky pollen that is readily expelled from dehiscent anthers (Williams, 1995). Little nectar was evident in the flowers that were microscopically examined (Williams, 1995). The anthers are brush-like, with the stigma extending slightly above, the stigmatic surface being broad and laterally flattened apically (Fig. 1) (Williams, 1995). Flower morphology partly agrees with the wind-pollinated (anemophilous) syndrome (Faegri & van der Pijl, 1979) such that in addition to being adapted for pollination by biotic vectors the floral structure suggests flowers are also facultatively wind-pollinated (Williams & Adam, 2010). The exine sculpture is indistinct – smooth (Williams & Adam, 1999), indicating no special modification for biotic dispersal. Thus the species is considered cryptically ambophilous, a previously poorly recognized biotic-abiotic pollination strategy now considered to be expressed by rainforest angiosperms more widely (Bullock, 1994, Williams & Adam, 2010).

Floral visitor observations were undertaken principally of plants growing on the margin of a subtropical lowland rainforest at Lorien Wildlife Refuge, approximately

3km north of Lansdowne (31°45'00"S, 152°32'30"E). Observations were carried out there during November 1990 and November and December 1992 (occasional observations were continued in subsequent years). At the nearby Lansdowne Nature Reserve (31°47'30"S, 152°32'30"E), a small floodplain rainforest remnant, a single day (19 Nov. 1990) of observations was additionally undertaken. At both sites a small number of mass-netted insect samples were also collected (Williams 1995). These gave indications of the nature of visitor assemblages at particular moments, but are insufficient to allow any statistical analysis.

Rainforest at Lorien Wildlife Refuge and Lansdowne Nature Reserve represent vegetation communities listed as *endangered ecological communities* (respectively ‘Lowland Rainforest in the NSW North Coast and Sydney Basin Bioregions’ and ‘Lowland Rainforest on Floodplain in the New South Wales North Coast Bioregion’) originally under the NSW *Threatened Species Conservation Act 1995*; this now supplanted by the *Biodiversity Conservation Act 2016*. Both formations are also listed as *critically endangered* under Federal legislation (see ‘Lowland Rainforest of Subtropical Australia’, *Environmental Protection and Conservation Act 1999*) because of the extent of past agricultural clearing and their now limited extent. Conservation has been a major consideration in their recent management. However, the unforeseen and widespread entry of the South American fungal pathogen ‘Myrtle rust’ (*Austropuccinia psidii*) (Invasive Species Council, 2011, Makinson, 2018) into the region around 2010 has resulted in the death of all mature *Rhodomyrtus psidioides* trees.



Figure 1: *Rhodomyrtus psidioides* flower showing extended stigma and expanded stigma surface.

Table 1. Insect taxa recorded visiting the flowers of *Rhodomyrtus psidioides* (1990-1992)

(Insects determined to family- Lorient Wildlife Refuge records cited first, Lansdowne Nature Reserve indicated with an asterisk '*'; multiple species given in parentheses; 'sp./spp.' = number of species uncertain).

COLEOPTERA-beetles	
Aderidae	<i>Aderus</i> sp.
Boganiidae	<i>Athertonium</i> sp., * <i>Athertonium</i> sp.
Cerambycidae	<i>Syllitus</i> sp.
Chrysomelidae	<i>Crepidodera</i> sp., <i>Ditropoda</i> spp. (2), <i>Monolepta australia</i> , <i>M. ?minuscula</i> , * <i>Crepidodera</i> sp., * <i>Monolepta</i> sp.
Cleridae	<i>Scrobiger splendidus</i>
Coccinellidae	<i>Harmonia testudinaria</i> , <i>Rhizobius</i> sp., <i>Scymnus</i> sp.
Corylophidae	<i>Sericoderus</i> spp. (2), * <i>Sericoderus</i> sp.
Curculionidae	<i>Cytallia sydneyensis</i> , undetermined spp. (4), * <i>Cytallia sydneyensis</i> , * <i>Orthorhinus</i> sp., *undetermined spp. (2)
Dermestidae	<i>Anthrenus</i> sp.
Elateridae	<i>Megapenthes futilis</i> , <i>Microdesmes collaris</i> , * <i>Drymelater</i> sp., * <i>Megapenthes futilis</i>
Latridiidae	<i>Corticicara</i> sp.
Melyridae	<i>Helcogaster</i> spp. (2), <i>Neocarphurus ?angustibasis</i>
Mordellidae	<i>Mordella inusitata</i> , <i>Mordella</i> sp., <i>Mordellistena</i> sp., * <i>Mordellistena</i> sp.
Nitidulidae	<i>Notobrachypterus</i> sp.
Oedemeridae	? <i>Ischnomera</i> spp. (2), <i>Pseudolychus</i> spp. (2)
Phalacridae	? <i>Olibroporus</i> sp.
Ptilidae	<i>Acrotrichis</i> sp.
Scarabaeidae	<i>Diphucephala ?pygmaea</i> , <i>Phyllotocus scutellaris</i> , * <i>Diphucephala ?pygmaea</i>
DIPTERA-flies	
Bombyliidae	<i>Geron</i> spp. (2)
Calliphoridae	? <i>Calliphora</i> sp., <i>Stomorhina</i> sp.
Dolichopodidae	<i>Amblypsilopus ?broulensis</i> , <i>Diaphorus</i> sp.
Drosophilidae	<i>Drosophila</i> spp. (2)
Empididae	undetermined sp.
Lauxaniidae	<i>Melanina</i> sp., <i>Steganopsis melanogaster</i>
Scatopsidae	undetermined sp.
Tachinidae	undetermined spp.
HEMIPTERA-bugs	
?Jassidae	undetermined spp.
Miridae	undetermined sp.
Psyllidae	undetermined sp.
HYMENOPTERA-wasps and ants	
Braconidae	undetermined sp./spp.
Encyrtidae	undetermined sp./spp.
Eulophidae	undetermined sp./spp.
Formicidae	? <i>Camponotus</i> sp., * <i>Crematogaster</i> sp.
Pergidae	? <i>Neoeurys</i> sp.
Pteromalidae	undetermined sp./spp.
Vespidae	<i>Polistes humilis</i>
HYMENOPTERA/Apiformes-bees	
Apidae	<i>Amegilla ?pulchra</i> , <i>Apis mellifera</i> , <i>Trigonula carbonaria</i>
Colletidae	<i>Hylaeus ?ofarrelli</i> , <i>Hylaeus</i> sp., <i>Leioproctus</i> sp., * <i>Heterapoides</i> sp.
PSOCOPTERA-book lice, bark lice	
*Caeciliidae	* <i>Caecilius ?lineatus</i>
Ectopsocidae	<i>Ectopsocus</i> sp. near <i>meridionalis</i>
THYSANOPTERA-thrips	
Phlaeothripidae	<i>Haplothrips</i> sp., * <i>Haplothrips</i> sp.
Thripidae	<i>Heliethrips haemorroidalis</i> , <i>Thrips setipennis</i> , <i>Thrips</i> sp., *? <i>Thrips</i> sp.

Table 2. Numbers of individual visiting insects and taxa recorded from selected single *Rhodomyrtus psidioides* tree sampling events (1990-1992)

Samples collected only during the morning at each site (from Williams, 1995)

	No. of individuals	No. of taxa	No. of individuals	No. of taxa
Lorien Wildlife Refuge Nov. 1990	tree 1		tree 2	
total Coleoptera	47	24	14	11
total Diptera	36	14	51	19
total all Hymenoptera	4	3	6	6
s/total bees	0	0	0	0
total visitors	107	47	103	45
Total <6mm in size	101		103	
Lorien Wildlife Refuge, Nov. 1992	tree 1			
total Coleoptera	38	13		
total Diptera	23	17		
total all Hymenoptera	9	9		
s/total bees	0	0		
total visitors	87			
total <6mm	86			
Lansdowne Nature Reserve, Nov. 1990	tree 1			
total Coleoptera	467	12		
total Diptera	13	7		
total all Hymenoptera	2	2		
s/total bees	1	1		
total visitors	534	32		
total <6mm in size	532			

Results and Discussion

Sampling and observation results show insect taxa recorded visiting the flowers of *Rhodomyrtus psidioides* in the 1990-1992 period (Table 1) and numbers of individual visiting insects and taxa recorded from selected single *Rhodomyrtus psidioides* tree sampling events (1990-1992) (Table 2). No vertebrates were seen visiting flowers. Table 1 underestimates the numbers of insect species owing to difficulties in identifying to family groups such as small Diptera and microhymenoptera. Insect visitors were predominantly (99%) in the <6mm size class (Table 2, Williams, 1995). Only *Amegilla ?pulchra* and the introduced ‘honey bee’ *Apis mellifera* constituted notable size exceptions. Although individual temporally-discrete sampling events can result in seemingly large numbers of individuals and taxa (Table 2), in general over the period of the study, insects were often few in number at any one time of observation; no taxon exhibited mass attraction responses to open blossoms, most individual blossoms were devoid of insects when observations were made, and even the otherwise ubiquitous *Apis mellifera* was usually absent. This seemed counter-intuitive given that blossoms were massed, conspicuous and fragrant, but might be explained by the small quantity of nectar that individual flowers seemed to offer. No specific visitation patterns were observed. Rather, insects appeared to recruit randomly to flowers throughout each day.

The single day of observations, and the single netted sample collected at Lansdowne Nature Reserve (Table 2), is too small to establish an understanding of the possible full suite

of visitors there, however, observations indicated that the assemblage likely mirrors that recorded at Lorien Wildlife Refuge; this being a mixture of ‘incidental’ visitors (e.g., Dolichopodidae, Psyllidae, microhymenoptera) and potential pollinators dominated by small species, ecologically unspecialised for pollination, that are commonly encountered on a range of mass-flowering rainforest trees and large shrubs elsewhere in the region (Williams, 1995, G. Williams unpubl. records). The few species with specialised morphological adaptations to a floricolous habit were represented by Mordellidae, the scarab *Phyllotocus scutellaris*, apid bees, and the bombyliid fly genus *Geron*. All visitors, regardless of their degree of adaptation to feeding upon floral resources, have the potential to transport pollen loads. In the case of thrips and other minute insects, only single or small numbers of grains are anticipated to be transported, and movements are largely restricted to adjacent flowers and plants; their contribution to out-crossing thus being individually small, though cumulatively over time potentially significant. Only *Amegilla ?pulchra* and *Apis mellifera* undertake relatively frequent or long distance foraging movements between dispersed individual plants, with the former known to regularly exhibit ‘trap-lining’ foraging strategies (see Williams & Adam, 2010, Willmer, 2011).

Myrtle rust has spread globally and was first detected in New South Wales in April 2010. Its infection results in crown dieback, branch death and mortality of sensitive members of the Myrtaceae (Invasive Species Council, 2011, Carnegie *et al.*, 2016, Pegg, 2017, Makinson, 2018). Myrtle rust has the potential to fundamentally alter the ecology

of Australia's vegetation communities. At Lorien Wildlife Refuge *Rhodomyrtus psidioides* and the related and highly sensitive *Rhodamnia rubescens* (Benth.) Miq. have been severely attacked but species of the related Myrtaceae genera *Archirhodomyrtus*, *Corymbia*, *Eucalyptus*, *Lophostemon*, *Syncarpia*, *Syzygium* and *Tristaniopsis* have been seemingly unaffected. All these genera recruit a taxonomically wide assemblage of putative pollinators, most of which are small in size, that frequent a diversity of mass-flowering shrubs and trees with open insect-adapted floral structures (G. Williams pers. obs.).

Although *Rhodomyrtus psidioides* is able to resprout from root stock, vegetative regrowth over the ensuing years is constantly re-infected, causing gross leaf deformity and tip mortality (Fig. 2). Consequently, sucker growth, persistent as it has been, has not been able to successfully progress to reproductive maturity. This scenario is exhibited by numerous other populations in the region, for example in littoral rainforest at Harrington (G. Williams pers. obs.) and ornamental plantings at Diamond Beach (T. Wright pers. comm.). Although none of the insect visitors recorded constitute species with known obligate plant dependencies, *Rhodomyrtus psidioides* flower – putative pollinator interactions are extinct, at least locally. But should the species be able later to re-establish viable populations, a resident assemblage of pollinators would be present to reconstitute its pollination suite.



Figure 2: Root sucker tip mortality on *Rhodomyrtus psidioides* resulting from Myrtle rust attack.

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Paul Adam (University of New South Wales) offered numerous insights over the years into the nature and conservation of Australia's rainforests, Dan Bickel, David McAlpine, Courtenay Smithers (Australian Museum), Laurence Mound and Ian Naumann (CSIRO) kindly assisted with the identification of Diptera, Psocoptera, Thysanoptera and microhymenoptera. Kim McKay (Director and CEO), Rebecca Johnson (Director, AMRI) and Derek Smith (Collection Manager) are thanked for facilitating access to the resources of the Australian Museum. Dan Bickel is also thanked for comments on an original draft of the manuscript and Terry Wright (Diamond Beach) kindly provided observations on a population he had planted near Taree.

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A new classification of savanna plant communities on the igneous rock lowlands and Tertiary sandy plain landscapes of Cape York Peninsula bioregion

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Abstract: Classifying and mapping landscapes are tools to simplify complex systems into the discreet subsets widely used in landscape management. In 1999, the Queensland Government adopted a Regional Ecosystems approach as a state-wide landscape classification scheme. For the Cape York Peninsula bioregion in north-eastern Australia, Regional Ecosystems (RE) were initially recognised based on a pre-existing vegetation map and classification for the bioregion. The classification had been developed using expert-techniques based on extensive field plot data. Here, we use numerical analyses to classify the field plot data and identify savanna plant communities associated with two widespread landform groups in the bioregion (the old loamy and sandy plains (land zone 5) and the hills and lowlands on igneous rocks (land zone 12). Communities were identified at the plant association level, using species importance values calculated from foliage cover and vegetation height at each plot. We developed a descriptive-framework for each community using statistically based characterising species and biophysical attributes. We recognise 57 communities compared with 110 that had been previously identified using expert-techniques. This classification is used to recommend refined Regional Ecosystems under the government's regulations. The descriptive-framework supported consistent descriptions of communities and assignment of new sites to the classification. We conclude that incorporating quantitative methods in classifying and describing plant communities will improve the robustness and defensibility of Regional Ecosystems and their use in landscape management across Queensland.

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Introduction

Vegetation classification is a globally used tool for land management and for investigating ecological diversity at multiple scales. Consistent vegetation classification schemes which cross geographical and administrative boundaries are therefore highly desirable (ESCAVI 2003; Rodwell 2006; Jennings *et al.* 2009; De Cáceres *et al.* 2015). Recognising this, the Queensland government adopted the Regional Ecosystem (RE) approach as a state-wide classification scheme in 1999. This is a triple-tiered hierarchy with the first division being based on the Interim Biogeographical Regions of Australia (Thackway & Cresswell 1995). The second division of the hierarchy is ‘land zone’; a concept that involves broad geological divisions with consideration of geomorphological processes and soils (Wilson & Taylor 2012). Examples of land zones include ‘alluvial river and

creek flats’, ‘coastal dunes’ or ‘hills and lowlands on granitic rocks’. The third level of the classification scheme is termed ‘vegetation community’ and is a plant community, recognised at the plant association level (Figure 1). A Regional Ecosystem is therefore defined as “a vegetation community, or communities, in a bioregion that are consistently associated with a particular combination of geology, landform and soil” (Sattler & Williams 1999). REs can therefore contain one or more vegetation communities. REs are mappable, with a distinctive signature recognisable from remote sensing imagery at the landscape scale of 1:100,000. REs are revised and updated when new data is supplied. To this end, each bioregion has a technical committee to review and implement proposed changes based on appropriate data. This technical review committee performs the same function as similar panels in other Australian and international jurisdictions (EVSWG 2017; OEH 2018; USNVC 2018).

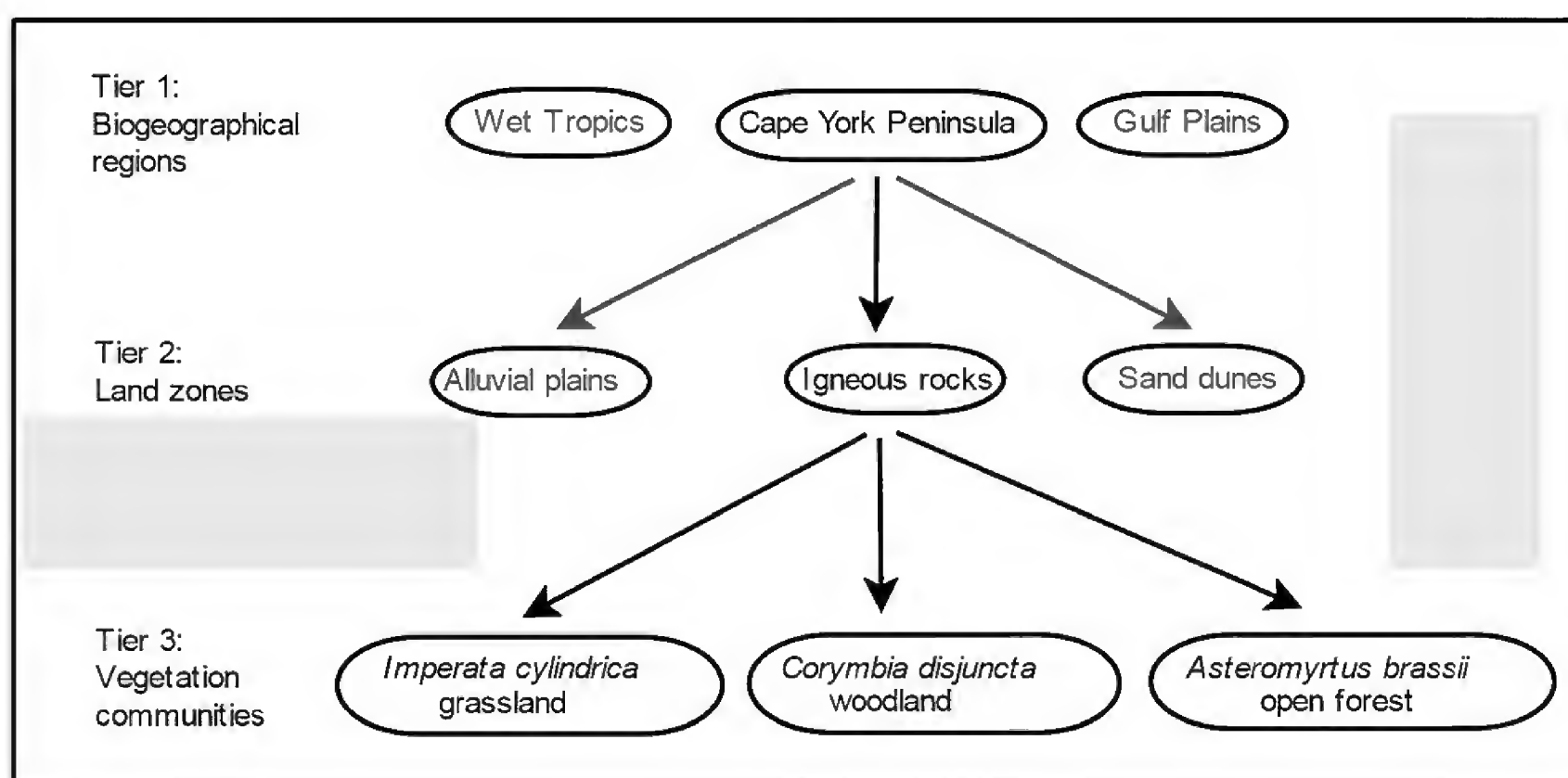


Figure 1: Regional Ecosystem classification scheme. Regional Ecosystems are a triple-tiered hierarchy. The first tier is biogeographical regions based on the Interim Biogeographical Regions of Australia. The second tier is broad geological / geomorphological groups (labelled land zones). The third tier are plant communities recognised at the association level (labelled vegetation communities)

For Cape York Peninsula (the Peninsula) a vegetation map and qualitatively-based classification at the plant association level was developed as part of the Cape York Peninsula Land Use Study (CYPLUS) carried out in the early 1990s (Neldner & Clarkson 1995). With the adoption of the RE framework, the CYPLUS vegetation classification was converted to a Regional Ecosystem classification using qualitative methods. The vegetation map was also revised in the context of a state-wide RE mapping program at a scale of 1:100,000, an exercise that ultimately necessitated a revision of the RE classification of the Peninsula.

A best-practice framework for vegetation classification is centred around standardised methods of data collection and classification techniques (De Cáceres *et al.* 2015). Following this best-practice, the RE classification framework has accompanying documentation describing a standardised survey and mapping methodology for Queensland (the methodology) (Neldner *et al.* 2017c). It outlines a consistent

set of classification protocols for defining vegetation communities which align with both the Beadle (1981) definition of a plant association and the necessary emphasis on canopy species used in classifications for vegetation mapping. These protocols identify the pre-dominant layer within a vegetation’s structure as that contributing most to the above-ground biomass (Neldner *et al.* 2017c). Communities are then defined using the height, cover and dominant species in the pre-dominant layer, with subordinate consideration given to associated species in other layers (Neldner *et al.* 2017c). Plant associations are thus defined as a community where the pre-dominant layer has a uniform floristic composition and exhibits a uniform structure. This forms the basis for mapping and survey projects at all scales across the state and is embedded in legislation. Currently however, implementation of these classification protocols relies on qualitative techniques and subjective sorting of plot data into similar groups. The use of

qualitative techniques is widespread and common in remote areas with limited researchers such as in Queensland, but they have acknowledged problems based primarily around their lack of transparency, repeatability and consistency between researchers (Mucina 1997; Kent 2012; Oliver *et al.* 2012). A good outcome from such processes is heavily dependent on a researcher's knowledge of the vegetation of the area and the biases introduced by their assumptions of the ecological and biophysical processes important to landscape function and biodiversity. Consequently, qualitative methods do not produce communities which are statistically comparable (Harris & Kitchener 2005; Kent 2012; Oliver *et al.* 2012). Using quantitative techniques in the classification process can help to overcome some of these problems allowing consistent, statistical information to be produced about community composition and structure.

A classification scheme has widest applicability if it can perform two major tasks: firstly, determine communities with transparent and repeatable techniques, and secondly provide consistent and reliable assignment of new sites to the classification scheme (De Cáceres & Wisser 2012). The aim of this study is to address these requirements by incorporating quantitative analyses into the classification of vegetation communities within the RE framework. Specifically, we aim to classify the savanna communities of two land zones on the Peninsula at the association level, assess the adequacy of the preferential sampling design used and develop a descriptive-framework which incorporates statistically derived characterising species for assigning new site data into these communities. We use this framework to describe REs suitable as distinct vegetation mapping units.

Methods

The Cape York Study area

Cape York Peninsula bioregion covers 120,000 km² in the monsoon tropics of north-eastern Australia and lies between 10 and 16 degrees south (Figure 2). Elevations range from sea level to approximately 800 m. The annual average rainfall varies between 1000–2000 mm with 80% falling in the wet season between December and March (Horn 1995). Temperatures range from an average annual monthly minimum of 14 °C in winter (July) to an average monthly maximum of 35 °C in summer (December) (BoM. 2016). Our study encompasses the savanna communities on two of the ten land zones on the Peninsula (Neldner 1999); the old loamy and sandy plains (land zone 5) and the hills and lowlands on igneous rocks (land zone 12). These communities on land zone 5 cover 45,000 km² (40% of the bioregion) and on land zone 12, 6,500 km² (5% of the bioregion). Land zone 5 is distributed across the full extent of the bioregion while land zone 12 occurs primarily along a north-south spine associated with the Great Dividing Range (Figure 2).

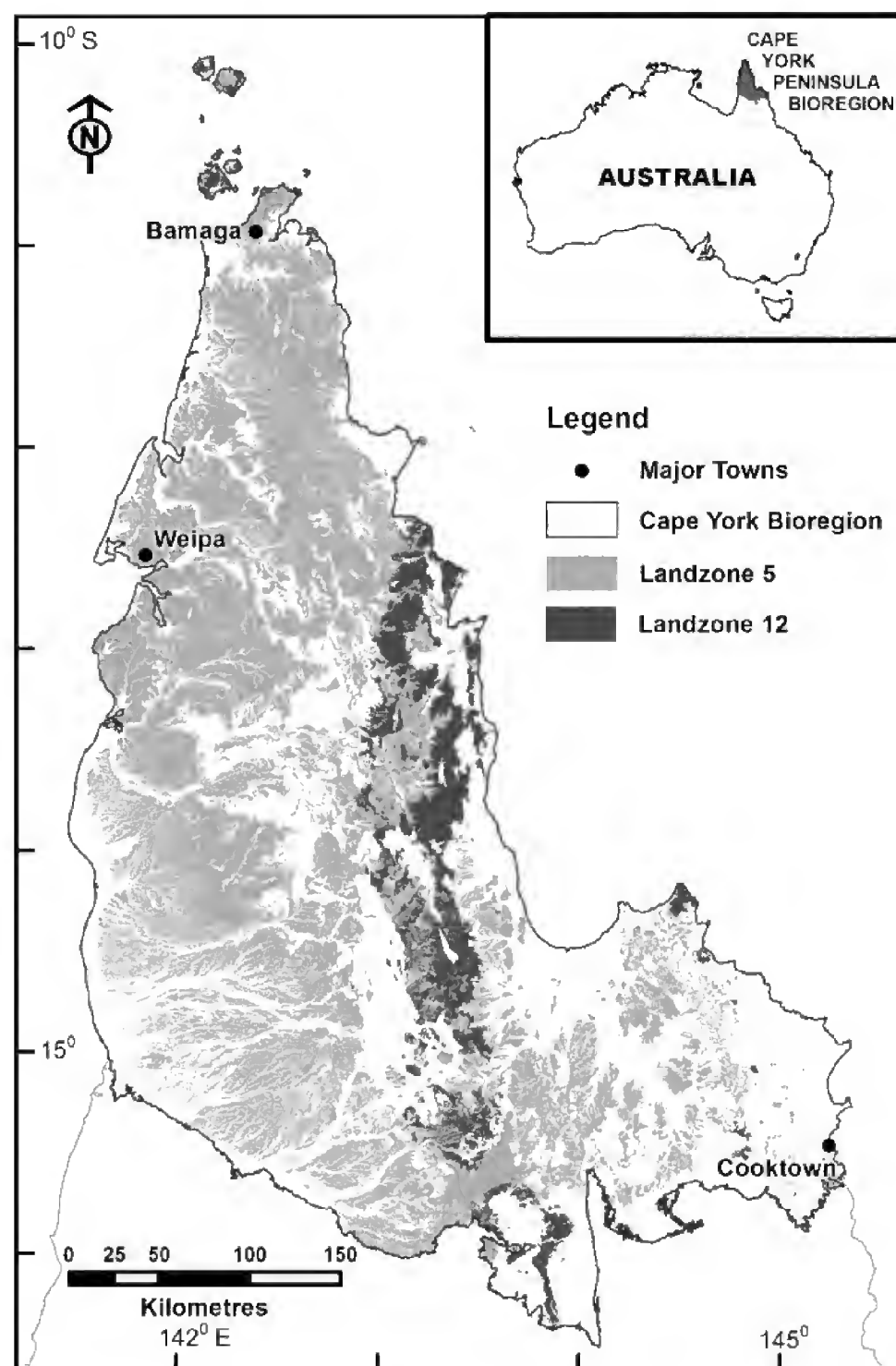


Figure 2: Distribution of the two land zones on Cape York Peninsula classified in this study.

Data Collation

During the mapping process two major types of vegetation data were collected; observational sites and vegetation plot data. These were sampled from 1990 to 2015, with the majority between 1992 and 1996 as part of the original mapping project (Neldner & Clarkson 1995). The observational sites were collected in large numbers as rapid records made during field traverses of the mapping area. They include records of geolocation, dominant species in the pre-dominant layer and vegetation structure. The survey design for locating vegetation plot data was preferential, with locations chosen based on either interpreted photo-patterns from air photos and ease of access, or on plant assemblages identified during the collection of observational sites. Observational site data were extracted from GIS coverages associated with the mapping project and vegetation plot data from the Queensland government 'CORVEG' database. The latter were categorised as either 'detailed' plots, containing data appropriate for use in determining the vegetation classification, or 'non-detailed' plots, containing incomplete data or data collected using different methods. Detailed plots contained data on percent foliage projected cover (%FPC) for each species in each woody vegetation layer recorded

along a 50m transect using the line intercept method (Neldner *et al.* 2017c). The average height of each layer was also recorded. The ground layer had species abundance recorded as an estimate of %FPC in 1 m² quadrats at 10 m intervals along the 50 m transect (five quadrats in total) and averaged. There were 192 detailed, 38 non-detailed plots and 4,670 observational sites on land zone 5 and 96 detailed, 45 non-detailed sites and 1,424 observational sites on land zone 12. Vegetation communities in which the pre-dominant canopy was the ground layer we refer to collectively as grasslands, but this group includes sedgeland and rock pavements with scattered herbs and forbs as well as true grasslands (Neldner *et al.* 2017c).

It was necessary for our quantitative analysis to accord with the classification protocols and principles outlined in the methodology as they are embedded in all current regional ecosystem mapping relied upon for regulatory purposes. To this end, previous research identified transformations to detailed plot data suited to incorporating vegetation structure into the classification of plant associations (Addicott *et al.* 2018). As a result, %FPC for each species was multiplied by the height of the layer in which it occurred prior to data analysis. This generated a species importance value for every species in each plot. The protocols also specify using dominant species and removing species which have low occurrence or abundance (here termed 'sparse') from a dataset is recommended in general scientific practice when exploring ecological patterns in data. Sparse species may mask relationships of interest at the relevant scale, their occurrence and identification may be dependent on survey design or their distribution may be spatially and temporally inconsistent (Grime 1998; McCune & Grace 2002; Kent 2012). To standardise the removal of sparse species across plots we excluded those whose contribution to total foliage cover was never >1%. For example, if a species contributed <1% in one plot and >1% in others it was retained in **all** plots in which it occurred. A species whose contribution to total foliage cover was <1% in every plot was excluded. This resulted in removing 175 taxa from the land zone 12 and 254 taxa from the land zone 5 analyses. Standardising the removal of sparse species in this way provides a consistent method across the dataset which does not delete infrequent but dominant taxa (Field *et al.* 1982), eliminates most of the unidentified taxa in a site without affecting the identification of communities at the association level (Pos *et al.* 2014) and improves the recognition of communities identifiable at landscape mapping scales (Addicott *et al.* 2018). We acknowledge that sparse species and unidentified taxa may be new, rare and/or endangered species and hence of high conservation significance. However, for this study they are not critical to vegetation classification at landscape scales (Addicott *et al.* 2018). Non-native species were also excluded, and any remaining taxa not reliably identified to species were amalgamated to genus level. The ground layer at wooded plots was excluded to identify communities suitable as mapping units (Neldner & Howitt 1991; Archibald & Scholes 2007; Mucina & Daniel 2013). In the final analyses there were 351 species with 241 occurring on land zone 5, 258 on land zone 12, and 148 shared between the two land zones. Plant nomenclature follows Bostock & Holland (2015).

Identifying plant communities

Plots were allocated to either land zone 5 or 12 based on field observations as well as geology, regolith and / or soils mapping available at plot each location. We analysed the detailed plots in each land zone to look for groups of co-occurring plant species using agglomerative hierarchical clustering and the software package PRIMER-E v6 (Clarke & Gorley 2006). We produced a similarity matrix (square-root transformation, Bray-Curtis coefficient) and ran the CLUSTER routine, using unweighted pair group mean averaging, to form clusters. To choose the level of cluster division for identifying plant communities we used a combination of three evaluation methods: 1) the SIMPROF routine which determines clusters significantly different to each other (Field *et al.* 1982), 2) Indicator Species Analysis (Dufrêne & Legendre 1997) (in the 'labdsv' R package (Roberts 2013)) which determines clusters maximising species occurrence and 3) generalised linear models (GLM) in a multivariate framework (Lyons *et al.* 2016) (available in the 'optimus' R package (Lyons 2018) to estimate the relative performance of differing cluster divisions in predicting species foliage cover. This last method uses GLMs and Akaike's Information Criteria (AIC). AIC is summed across individual species, and the final sum-of-AIC score is used as a measure of how well the cluster division predicts species cover. A lower sum-of-AIC score indicates a better prediction. In situations where the three evaluators produced differing results, we formed a subset of plots and tested cluster divisions within the subset.

Assigning plant communities to the Regional Ecosystem framework

Our final plant communities were evaluated by the *technical review committee for regional ecosystems of the Cape York Peninsula bioregion* whose role was to evaluate and give effect to proposals to modify Regional Ecosystems classifications. During this process the committee assigned plant communities to regional ecosystems based on expert-judgement of non-floristic variables as outlined by the methodology (Neldner *et al.* 2017c), potentially producing REs containing communities with different dominant species and low floristic similarity to each other. For example, communities which did not have predictable or mappable occurrences or were <100 ha in total area of distribution were grouped with those on closely associated landforms and similar ecological niches. Communities recognised as successional temporal variants, or condition states, of a climax association were also grouped into one RE. Where the committee requested more evidence to support proposed changes, we used the classification protocols as a guide for conducting further analyses. Consequently, we tested for floristic differences between sites on different geomorphological areas and soil types (using the ANOSIM routine), for differences in canopy height (using an unpaired t-test) and investigated whether differences in the ground layers of sites were coincident with geomorphological areas or soil divisions (using *n*MDS ordination and GIS overlay). One additional role of the committee was to

identify communities not represented in the analyses but recognisable from aerial photo interpretation, non-detailed plot data and observational sites. There were therefore two types of communities in the final classification scheme; those identified through quantitative analysis and those identified by expert-techniques. The latter communities will be reviewed when further detailed sampling data and quantitative analyses are available.

Creating community descriptions and assigning new sites

An important aspect of a vegetation classification scheme is to allow description and identification of its plant communities (De Cáceres & Wiser 2012). To this end we compiled a descriptive-framework based on characterising species, vegetation structure and landform, including geographical distribution when it aided identification. Characterising species were those used to describe the floristic and structural composition of a community (De Cáceres *et al.* 2015) and were identified for the quantitatively defined communities using each species' frequency, average cover and strength of association with a community. To determine the strength of each species' association with a community, we calculated a phi-coefficient of association (Chytrý *et al.* 2002) based on cover, using the JUICE software package (Tichý 2002). Each group was standardised to equal number of plots. A phi-coefficient of 100 means a species occurs only in that community, while values approaching zero indicate the species is equally abundant in several communities. The phi-coefficient values were also used to identify species with a significant association to a community using Fisher's exact test ($p < 0.05$) (Chytrý & Tichý 2003). We listed species frequency and average cover using the technical-description routine within the CORVEG database, which also allowed identification of vegetation structure. We defined characterising species as those with a phi-coefficient of association > 6 or occurring in $> 70\%$ of sites. A phi-coefficient of > 6 was chosen to ensure a minimum of one statistically associated species with each community. Landform and additional vegetation structure information was taken from plot sheets and observational data where available. Geographical distribution came from the final mapping. Where communities were represented by fewer than three sites in analyses we used non-detailed or observational sites for additional information. To describe qualitatively determined communities we used species, structure and landform information from non-detailed plots and observational sites, and, where it was diagnostic, mapped distribution. These community descriptions are necessarily less robust but allow indicative recognition in the field.

The ease and certainty with which new sites can be reliably allocated into a classification scheme outside of an analysis process is important (De Cáceres & Wiser 2012) and we expected our descriptive-framework to enable this. To test this, we used the 'non-detailed' plots previously excluded from analysis as 'new' sites. We matched the information available from each plot to that in the descriptive-framework, subjectively assigning it to a vegetation community and rating its level of fit-to-community as high or low. These

non-detailed sites had a variety of vegetation information available ranging from a community label with or without a limited species list (and sometimes growth form) to complete species lists with alternative abundance measure such as classes, stem density or basal area and an indication of which layer species occurred in. In sites which had only a label (or label and a species list) we took the label as an indicator of dominance and structure. We also used landform information where it was provided on the site pro-forma.

Along with defining a classification via consistent analytical techniques, labelling communities using consistent naming conventions is important (De Cáceres & Wiser 2012). Neldner *et al.* (2017c) outlines these for the RE framework. In this, a limited number of characterising species are listed in order of dominance, with punctuation to indicate relative abundance and frequency, followed by the structural formation. Associated habitat characteristics, such as landform or soil descriptors are included in labels where they are diagnostic. We followed these conventions to develop community labels.

Assessing sampling adequacy

Knowledge of bias in a sampling design allows an understanding of the strength and weaknesses of results. We reduced bias by using plots with standardised plot size, collection methods, data attributes, data quality and season of survey. The standard plot size of 500 m² has been shown to adequately capture the species diversity at the plot level in savanna and woodland communities (Neldner & Butler 2008). Data collection methods follow the standard survey methods outlined in Neldner *et al.* (2017c). Seasonality is an issue in the ground layer as many species occur only in the wet and early dry season. In sites dominated by woody vegetation, excluding the ground layer removed this potential bias. Plots dominated by the ground layer were surveyed between May and August (the early dry season). Despite standardising these aspects of survey design however, we expected some bias due to preferential rather than random selection of plot locations. Therefore, we assessed how well the field sampling captures firstly the environmental variability across the landscape, and secondly the community and species richness.

To test how well the environmental variability was sampled, we followed the convention of testing those variables expected to limit plant species growth, dividing them into climate and soil themes. We used four climate variables, two temperature variables (average annual temperature, and the coefficient of variation of temperature seasonality) and two rainfall variables (annual average rainfall, and the mean moisture index of the lowest quarter), available as ANUCLIM datasets (Xu & Hutchinson 2013). The soil variables were grouped in to soil nutrients (organic carbon content, and phosphorus) and soil structure (available plant water capacity, permeability, drainage, and slope) (Lyons *et al.* 2017; Neldner *et al.* 2017a). All soil datasets came from Australian Soil and Resource Information System (McKenzie *et al.* 2012; ASRIS 2014), with the slope derived from the digital elevation model for the Peninsula

(GeoScience Australia *et al.* 2009). In addition to these climate and soil variables, we assessed how well the survey sampled variation in vegetation structure by using a maximum persistent greenness GIS coverage (JRSRP 2017). This coverage is derived from LANDSAT imagery classification and, on the Peninsula, equates to density of woody vegetation layers, with a higher greenness index indicating denser woody vegetation. While density of woody vegetation is significantly correlated with the climate variables ($r = 0.6$, $p < 0.0001$), the R^2 value of the 4-way multiple regression is 34% indicating the predictability of density of woody vegetation using these variables is relatively low (Appendix 6, Figures A6.1-A6.4). We are therefore confident that assessing woody vegetation density will provide useful additional information on bias in sampling of vegetation structure. All these datasets were accessed as raster coverages. Using the DOMAIN software program (Carpenter *et al.* 1993) we calculated the similarity of the environmental envelope at any grid point to that at any plot or observational site. DOMAIN uses the input variables to create an environmental envelope for each grid cell and then calculates the similarity between each grid cell and any site in a Euclidean p-dimensional space using the Gower metric. The similarity is bounded in one direction, with values close to 100% for maximum congruence, and can be displayed spatially. Because observational sites assist in identifying the assemblages recorded in the detailed plots we investigated the amount of environmental variability captured by both types of data.

To assess the community and species richness surveyed by the detailed plots we estimated total population richness, and calculated the proportion captured by our sampling. To assess the species richness surveyed we used the full species dataset (with weeds removed), as our classification analyses used only a subset of species surveyed. To estimate population richness from our samples we derived 1,000 model-populations using bootstrap techniques. We then calculated an unbiased population estimate of richness by 1) estimating the bias, by subtracting the sample richness from the mean richness of the model-populations, and 2) subtracting this bias from the sample richness. Using the bootstrap model-populations we also defined 90% confidence intervals (using the 0.05 and 0.95 quantiles around the mean of the 1,000 model-populations). All calculations were done in the R environment (R Development Core Team 2014) using the 'bootstrap' package (Efron & Tibshirani 1993).

Results

Assessing sampling adequacy-Environmental variability

The survey design comprehensively sampled the full environmental variability in each land zone. Between 99 and 100% of the total area of each land zone was >90%-similar to any observational site for all variables. Results were similar for detailed plots for climate, vegetation structure and soil nutrient variables. Between 99 and 100% of the total area of

land zone 5 and 98% of land zone 12 was >90%-similar to any detailed plot (in the respective land zone). These results were slightly lower for soil structure, with 98.6% of land zone 5 and 95% of land zone 12 >90%-similar to any site. Appendices 3a and 3b have detailed tables and indicative maps of areas of lower similarity to sites. The detailed GIS coverages of these areas are available from the senior author if more detail is required.

Community Richness

We found the survey design reliably sampled the community richness of land zone 5 but not that of land zone 12. On land zone 5 it captured 95% of the estimated total community richness. Nineteen of an estimated 20 communities were sampled in detailed sites, within the 90% CI (19 - 21). On land zone 12 the survey captured 89% of the community richness (24 of an estimated 27 communities), outside the 90% CI of 25 - 29 (Table 1).

Species Richness

The survey did not reliably capture the full species richness on either land zone, with the number of species sampled lying outside of the 90% CIs (Table 1). There were 775 species sampled on land zone 5 and 673 on land zone 12, representing 86% of the estimated species richness on either land zone (Table 1).

Table 1: Sampled and expected community and species richness. The expected number of communities and species and the 90% confidence intervals (CI) are calculated from bias corrected estimates of 1,000 bootstrap model-populations.

	Number sampled	Number expected	90% CI
Community richness			
Land zone 5	19	20	19 - 21
Land zone 12	24	27	25 - 29
Species richness			
Land zone 5	775	904	889 - 920
Land zone 12	673	785	771 - 798

Plant Communities

There were 57 communities in our study's final classification, 27 on land zone 5 and 30 on land zone 12. Seventy-five percent of these were identified by quantitative methods and 25% by qualitative techniques and less detailed plot data (Table 2). Two communities were recognised after additional analyses requested by the technical review committee (Appendix 4). Incorporating quantitative analysis resulted in fewer communities on both land zones than the expert-based classification with an overall reduction of 49%. Individually, the reduction was higher on land zone 5 (54%) than land zone 12 (42%), driven by the larger decrease in the number of woodlands and shrublands identified (Table 2). Whilst most of the final REs consisted of one plant community,

in 11 instances, the review committee assigned several communities to individual REs. The 27 communities on land zone 5 were assigned to 21 REs, and the 30 on land zone 12 to 23 forming some REs with more than one community (appendix 2). Because the detailed descriptions, conservation status and ecological notes for individual REs and their communities are available on-line we have not included it in this manuscript (<http://www.qld.gov.au/environment/plants-animals/plants/ecosystems>). However, to portray the communities and REs recognised, we have included the short label descriptions, mapped areas and notes for the REs in Appendix 2. To illustrate the floristic relationships between the communities and REs on each land zone we formed community dendrograms and ordination plots from the detailed plot data (Appendix 5).

Table 2: The number of communities in each formation on each land zone. The quantitative analysis resulted in a reduction in

the number of vegetation communities. ‘a priori’ classification = vegetation communities in the pre-existing, qualitatively derived, classification.

	Grasslands	Shrublands	Woodlands
Land zone 5 (45,000 km ²)	806 ha	1,904 km ²	46,089 km ²
Quantitatively derived	1	1	17
Qualitatively derived	1	1	6
Total after review (no. of REs)	2 (1)	2 (2)	23 (18)
<i>a priori</i> classification	4	7	48
Land zone 12 (5,500 km ²)	154 km ²	110 km ²	5,236 km ²
Quantitatively derived	5	3	16
Qualitatively derived	1	1	4
Total after review (no. of REs)	6 (5)	4 (4)	20 (14)
<i>a priori</i> classification	7	6	38

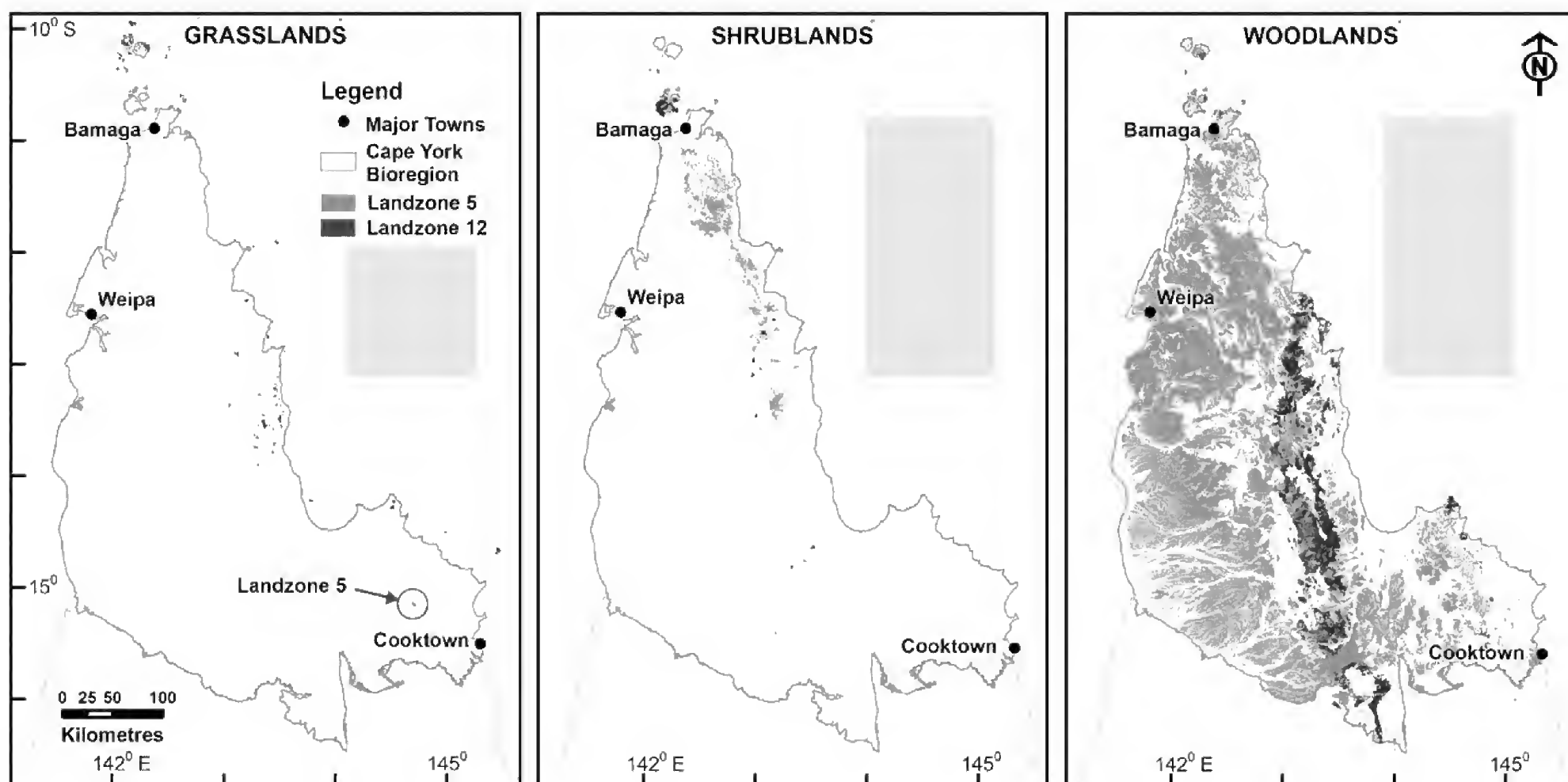


Figure 3: Distribution of the vegetation formations across Cape York Peninsula bioregion included in this study.

Summary of plant communities and formations of land zone 5 (old loamy and sandy plains)

Grasslands are of limited extent on land zone 5 (0.01% of the land zone) and contain two communities. One occurs only on islands in the Torres Strait and the other in southern Cape York Peninsula (Figure 3). Shrubland communities cover 4% of the land zone (Figure 3), the most extensive of which (1,900 km²) occur on the deep sand plains in the north-east and east of the bioregion. The second occurs only on the Torres Strait islands. Woodlands dominate land zone 5 (95% of the area) (Figure 3) and can be broadly categorised into four groups; 1) *Eucalyptus tetrodonta* dominated woodlands, 2) other Eucalypt and *Corymbia* dominated woodlands, 3) *Melaleuca* dominated woodlands, and 4) *Asteromyrtus*

dominated woodlands. The *Eucalyptus tetrodonta* woodlands dominate the landscape, covering 42,870 km². *Melaleuca* dominated woodlands cover the next largest area of 2,825 km², the *Asteromyrtus* dominated woodlands 1,044 km² and Eucalypts and *Corymbia* species other than *Eucalyptus tetrodonta* cover the smallest area (528 km²).

Summary of plant communities and formations of land zone 12 (hills and lowlands on granitic rocks)

Grasslands are again of limited extent on land zone 12, covering 2% (Figure 3). The most widespread of these was the rock pavements with scattered herbs and forbs associated with the tops of the major mountain chains on the mainland and the Torres Strait islands (66 km²). The remaining five are

all dominated by Poaceae species. Shrublands cover 12% of the land zone (Figure 3), with three of the four communities dominated by *Melaleuca* species. The fourth, covering the largest area (57 km²), is dominated by an endemic species, *Leptospermum purpurascens*. Despite having the largest area, its range is restricted to the hills and mountains associated with Iron Range in the centre of the bioregion. Woodlands are again the most widespread formation (75% of land zone) (Figure 3). These are dominated by *Eucalyptus tetradonta* woodlands (41% of woodland area) and ironbark woodlands (*Eucalyptus cullenii* and *Eucalyptus crebra*) (28%). Other Eucalypt and *Corymbia* dominated woodlands cover 21%. *Melaleuca* woodlands cover 3% of the land zone, a much smaller area than on land zone 5. The remaining area is covered by one mixed species low woodland and two *Acacia* communities (both of which occur only in the Torres Strait islands).

Assigning new sites into the classification scheme

Using the descriptive-framework (Appendix 1) we were able to incorporate all 83 non-detailed sites into the classification scheme. The characterising species provide the most useful information; strength of association allowed us to rank characterising species in importance for a community. The species information in the non-detailed sites could then be matched to this, even when not all characterising species were recorded at a site. While the characterising species was the most useful individual piece of information, the most powerful tool for assigning sites in to the classification was the combination of characterising species plus vegetation structure information. Landform became diagnostic where the characterising species overlapped (particularly the *Eucalyptus tetradonta* woodlands). We could assign 66% of sites (55) with a high level-of-fit to community. These were the sites that contained quantitative abundance and structure data collected using different methods. The sites assigned with a low level-of-fit to community were those with only a community label to indicate abundance and structure.

Inclusion of results in mapping

Vegetation mapping and classification are two separate processes often accompanying each other (Franklin 2013). In this survey the process was iterative, with the mapping (and accessibility) driving the choice of transects, and the outcomes feeding back to change the qualitative classification depicted in the mapping. Continuing this process, the results of our classification analyses were used to revise the Regional Ecosystem mapping to reflect the updated vegetation communities and REs. As part of the mapping, individual mapped areas (i.e. polygons) are also assigned levels of reliability for attributes and locational accuracy. Polygons which contained detailed plots were given a high reliability in the mapping, as were areas containing non-detailed plots assigned in to the classification with a high fit-to-community. Polygons containing non-detailed plots with low fit-to-community were mapped with a low reliability and identified as requiring further survey.



Figure 4: RE 3.5.19 *Asteromyrtus lysicephala* and *Neofabricia myrtifolia* open heath to shrubland on sand sheets.



Figure 5: RE3.5.36a *Eucalyptus tetradonta* and *Corymbia nesophila* woodland on undulating plains.



Figure 6: RE 3.5.41b *Melaleuca viridiflora* low open woodland +/- *Petalostigma banksii* on plains.



Figure 7: RE 3.12.10a *Eucalyptus cullenii* +/- *Corymbia clarksoniana* woodland on granite hills and footslopes.



Figure 8: RE 3.12.28 *Leptospermum purpurascens* tall shrubland on igneous hills.



Figure 9: RE 3.12.48a *Heteropogon triticeus* dominated grasslands on igneous headlands and offshore islands.

Discussion

We present, for the first time, a bioregional scale classification of vegetation communities, within the Regional Ecosystem framework, incorporating quantitative analyses. After initial assignment of sites to land zones, we allocated sites to communities using 1) numerical classification based on floristic attributes, and 2) statistical analysis of vegetation structure and environmental factors. These communities were incorporated into the RE framework by an expert panel peer-review process. We developed a descriptive-framework to characterise the vegetation communities (using statistically derived floristic attributes and non-statistically derived abiotic variables), and used this to assign new sites to the classification. In so doing we addressed the two main tasks of a classification scheme (as outlined by De Cáceres & Wisser -2012) - to determine vegetation communities using transparent and repeatable techniques, and to provide a framework for consistent and reliable assignment of new sites into the classification scheme.

While our classification incorporates as much quantitative analysis as available data allows, 25% of communities were still identified using expert-based techniques. This was done using plots with different data collection methods, or observational data from helicopter flights over inaccessible areas of the bioregion, meaning the data could not be used in the analyses. Communities identified by expert-based techniques therefore represent ‘known unknowns’ and provide a targeted direction for future data collection.

A notable outcome of the quantitative analysis was the 49% reduction in the number of communities recognised, compared to the expert-driven process. Quantitative analysis allows experts to test their interpretation of the factors influencing landscape function; in this case, unquantified floristic and biophysical attributes. One question our analysis asks is, ‘Does the floristic composition of the landscape reflect the divisions chosen by experts, based on their assumptions about the importance of these attributes?’ The 49% reduction suggests that, in this case, it does not. Quantifying the differences between the expert and quantitatively derived communities is beyond the scope of this paper, but is the focus of ongoing work. However, one function of quantitative analysis is to help gain consensus among experts about the species driving vegetation community differences.

Preferential-sampling designs are biased in several ways compared to stratified random-sampling designs (Diekmann *et al.* 2007; Hédl 2007; Michalcová *et al.* 2011). It is well recognised that the statistical power of preferential-sampling designs is lower (Lájer 2007), but much of the aim of vegetation survey and mapping is to distinguish patterns using descriptive procedures rather than to produce inferential results from null hypothesis significance testing (De Cáceres *et al.* 2015). Roleček *et al.* (2007) found that preferential sampling designs cover a greater range of environmental extremes than random sampling designs for the same level of survey effort; our results appear to agree with this. Despite an initial perception that 51,500 km² would not be adequately sampled with 288 detailed plots, this survey covered the environmental

variability and community richness comprehensively on land zone 5, and adequately on land zone 12. We suggest this is due to the two-tiered system of data collection, with large numbers of rapid observational sites augmented by detailed vegetation plots in representative locations. The small difference in the sampling adequacy between land zones is likely due to accessibility. Whilst the landscapes of the old loamy and sandy plains of land zone 5 are relatively well traversed by roads, allowing access for detailed plot collections, the landscapes of the igneous rocks of land zone 12 have mountainous terrain with few roads providing limited access.

Although our sample design adequately surveyed landscape variability and community richness, our analyses show this is not so for species richness. This differs from other studies that found preferential-sampling had a higher likelihood of sampling the full species richness than stratified-random sampling, as researchers tend to choose sample locations with higher species richness (Michalcová *et al.* 2011). In this survey, however, locations were chosen on a perception of representativeness of distinctive communities, rather than species richness, potentially explaining the difference to other studies. Our survey's design of detailed plot locations evidently captures the communities present, but not the full floristic variability within those communities. This result agrees with the intuitive assessment that sampling such a large area with so few sites would not provide comprehensive coverage; and with Lawson *et al.* (2010) who found high levels of floristic heterogeneity within regional ecosystems in south-east Queensland.

A major function of a classification scheme is to allow new site data to be assigned to it (De Cáceres & Wiser 2012). In the authors' experience, an important issue when using a qualitatively-derived classification for this task, is ambiguity in allocating new sites into the scheme. A descriptive-framework based on quantitative data helped overcome this by allowing us to allocate sites with different data collection methods to the classification scheme with a high level-of-fit to community, enhancing the repeatability of allocating new sites. This, in turn, increases the classification's applicability by allowing 1) easier recognition of community types, 2) greater confidence in identifying sites from communities new to the classification, and 3) the classification to become a dynamic scheme responsive to new information. Our descriptive-framework does not fit the definition of membership rules outlined by De Cáceres & Wiser (2012), (in that the same rules used to define communities are not used to allocate new sites into it) but it performs a similar function.

A potential benefit of incorporating quantitative analyses in the Regional Ecosystem framework is to allow a display of relationships between communities not obvious in a qualitative classification. An area with many similar REs, may have less diversity than an area with fewer dissimilar REs. For instance, a result of the committee process of allocating communities to REs, based on non-floristic variables, is that REs can contain communities dominated by different species with low similarity to each other. Dendrograms, scatter plots and similarity matrices produced by quantitative analyses provide a visualisation and measure

of the similarities between REs and their vegetation communities (Appendix 5). For example RE 3.12.18 has two communities 'a' and 'b' (Appendix 5, figs 5.2 and 5.4). RE 3.12.18b is found in small patches scattered through larger areas of 3.12.18a, on the same landform, and not predictable enough to be reliably mapped at 1:100,000 scale. Displaying these relationships between communities may be useful in conservation planning, for example.

Incorporating quantitative analyses in the Regional Ecosystem framework will enhance its already wide use. As well as the current comparisons of spatial and temporal change of REs (Accad *et al.* 2017), statistical comparisons between vegetation communities at a cross-bioregion scale will become possible (Goodall 1973). We anticipate quantitatively-based vegetation communities will aid investigations into questions such as the assumptions behind their use as surrogates for biodiversity (Sattler & Williams 1999), the environmental drivers of the patterns of community distribution, and the phylogenetic diversity of communities. Importantly it will provide statistically-backed base-line data against which to measure the effects of future changes, such as climate and land use. REs are used by a wide cross-section of the public and form part of legislation at multiple tiers of government. With vegetation communities (the base-line level of the RE hierarchy) based on quantitative analyses, REs are more robust and readily defensible, providing legislators and users with greater confidence in the classification scheme.

Conclusion

To standardise classification procedures across large geographic areas and multiple administrative boundaries is one of the globally-recognised goals of vegetation science (Jennings *et al.* 2009; Walker *et al.* 2013; De Cáceres *et al.* 2015). These procedures are generally described as standardised data collection methods, classification schemes and quantitative classification techniques. In Australia, most state governments have adopted approaches which work towards achieving these goals (Sun *et al.* 1997; Gellie *et al.* 2017). In Queensland this is well advanced. As well as having state-wide Regional Ecosystem mapping at 1:100,000 scale, there is a standardised classification scheme, data collection methods and qualitative classification techniques. Extending our quantitative classification approach to the Regional Ecosystem framework across the remainder of Cape York Peninsula and other bioregions in Queensland, will further the achievement of these globally recognised goals.

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Appendix 1: Descriptive-framework for quantitatively derived vegetation communities on land zone 5 and 12 in Cape York Peninsula bioregion.

'Frequently occurring spp.' are produced using the technical-description routine in the QLD government database 'CORVEG'. Frequency = % of total sites in which a species occurs; average %cover = mean of all cover values >0 for each species. Formatting is as outputted from this routine, with % after the frequency and no percent symbol associated with average %cover. For example, *Corymbia stockeri* (93%, 3) means this species occurs in 93% of sites and when it occurs has an average cover of 3%. Statistically **highly** associated species are those with a phi-coefficient of association >6 with a community. Statistically **significantly** associated are those significantly associated with a community ($p < 0.05$) using Fisher's exact test. A species may be highly associated with a community but not significantly associated if it also has a strong association with another community. For example, *Dodonaea polyandra* is highly associated with RE 3.5.5, but not significantly as it is also highly associated with RE 3.5.42. Communities recognised using qualitative techniques are not included.

Version 10 RE (number of sites used in NA)	Short label description	Frequently occurring spp. (% frequency, average %cover) (Species not occurring in the canopy layer are labelled with their layer, E = emergent, T2 = Second tree layer, S1 = shrub layer)	Statistically significantly associated spp. ($p < 0.05$) (phi association coefficient)	Statistically highly associated spp. (phi association coefficient)	Occasional spp with high average %cover (% frequency, average %cover) (Species not occurring in the canopy layer are labelled with their layer, E = emergent, T2 = Second tree layer, S1 = shrub layer)	Additional information
Land Zone 5						
3.5.5 (6 sites)	<i>Corymbia novoguineensis</i> +/- <i>C. tessellaris</i> woodland on sand plains on northern Cape York Peninsula	<i>Corymbia novoguineensis</i> (100%, 16)	<i>Corymbia novoguineensis</i> (40.1) <i>Corymbia tessellaris</i> (22.8) <i>Gardenia</i> sp. (19.5) <i>Welchiodendron longivalve</i> (12.2) <i>Acacia polystachya</i> (12.1) <i>Acacia crassicarpa</i> (10.3) <i>Bridelia tomentosa</i> (8) <i>Tabernaemontana orientalis</i> (6.9) <i>Mallotus nesophilus</i> (6.9) <i>Terminalia muelleri</i> (6.2) <i>Syzygium suborbiculare</i> (6.1) <i>Livistona muelleri</i> (6)	<i>Corymbia novoguineensis</i> (40.1) <i>Corymbia tessellaris</i> (22.8) <i>Gardenia</i> sp. (19.5) <i>Dodonaea polyandra</i> (13.1) <i>Welchiodendron longivalve</i> (12.2) <i>Acacia polystachya</i> (12.1) <i>Acacia crassicarpa</i> (10.3) <i>Bridelia tomentosa</i> (8) <i>Tabernaemontana orientalis</i> (6.9) <i>Mallotus nesophilus</i> (6.9) <i>Terminalia muelleri</i> (6.2) <i>Syzygium suborbiculare</i> (6.1) <i>Livistona muelleri</i> (6)	<i>Welchiodendron longivalve</i> (17%, 12) <i>Corymbia tessellaris</i> (50%, 11) <i>Corymbia nesophila</i> (13%, 10)	Occurs in the northern Peninsula area and Torres Strait Islands.
3.5.6 (4 sites)	<i>Eucalyptus phoenicea</i> woodland on sandy outwash plains	<i>Eucalyptus phoenicea</i> (100%, 17)	<i>Eucalyptus phoenicea</i> (41.8) <i>Eucalyptus portuensis</i> (15.4) <i>Melaleuca nervosa</i> (7.3)	<i>Eucalyptus portuensis</i> (41.8) <i>Eucalyptus portuensis</i> (15.4) <i>Xanthorrhoea johnsonii</i> (14.7) <i>Melaleuca nervosa</i> (7.3) <i>Acacia torulosa</i> (6.8)	<i>Eucalyptus portuensis</i> (25%, 10) <i>Grevillea glauca</i> (100%, 2) T2 <i>Xanthorrhoea johnsonii</i> (25%, 15) S1	On the Battlecamp sandstone plateau. <i>Eucalyptus portuensis</i> can be co-dominant

3.5.9 (11 sites)	<i>Eucalyptus tetradonta</i> , <i>Corymbia stockeri</i> +/- <i>C. setosa</i> woodland on sand plains	<i>Erythrophleum chlorostachys</i> (100%, 6) <i>Eucalyptus tetradonta</i> (100%, 4) <i>Corymbia stockeri</i> (93%, 3)	<i>Corymbia setosa</i> (17.8) <i>Neofabricia sericisepala</i> (12.8) <i>Erythrophleum chlorostachys</i> (10.2) <i>Melaleuca nervosa</i> (6.1)	<i>Corymbia setosa</i> (64%, 4) <i>Corymbia novoguineensis</i> (9%, 9)	Widespread in the southern half of the bioregion and occurs predominantly on the Holroyd Plain in close association with 3.5.37a. At the northern extent of this community <i>Corymbia novoguineensis</i> can dominate the canopy and <i>C. setosa</i> is absent. <i>Corymbia setosa</i> is not always present but there is often a combination of mixed species, including heaths, in the community. <i>C. stockeri</i> is likely to be subsp. <i>peninsularis</i> .
3.5.15b (1 site)	<i>Dapsilanthus spathaceus</i> open sedgeland with emergent shrubs.	<i>Dapsilanthus spathaceus</i> (100%, 19) <i>Germainia capitata</i> (100%, 10) <i>Schoenus sparteus</i> (100%, 9) <i>Scleria</i> sp. (100%, 6) <i>Pandanus</i> sp. (100%, 6)	<i>Dapsilanthus spathaceus</i> (42.7) <i>Germainia capitata</i> (30.9) <i>Schoenus sparteus</i> (29.3) <i>Scleria</i> sp. (23.9) <i>Pandanus</i> (22.1) <i>Poaceae</i> (19.5) <i>Tricostularia undulata</i> (13.8) <i>Eriocaulaceae</i> (9.5) <i>Xyris</i> sp. (9.5)		Occurs in mosaic patches with 3.5.15a on Torres Strait Islands
3.5.19 (14 sites)	<i>Asteromyrtus lysicephala</i> and/or <i>Neofabricia myrtifolia</i> and/or <i>Jacksonia thesioides</i> open heath to shrubland on sand sheets	<i>Asteromyrtus lysicephala</i> (86%, 8) <i>Jacksonia thesioides</i> (86%, 3) <i>Neofabricia myrtifolia</i> (86%, 5) <i>Choriceras tricornae</i> (71%, 3) <i>Hibbertia banksii</i> (71%, 1) <i>Allocasuarina littoralis</i> (79%, 3) E.	<i>Asteromyrtus lysicephala</i> (24.3) <i>Allocasuarina littoralis</i> (12.8) <i>Thryptomene oligandra</i> (11.5) <i>Neofabricia myrtifolia</i> (11.3) <i>Hibbertia banksii</i> (8.9) <i>Leucopogon ruscifolius</i> (8.8) <i>Choriceras tricornae</i> (8.4) <i>Gompholobium nitidum</i> (8.2) <i>Grevillea pteridifolia</i> (6.8) <i>Eucalyptus chlorophylla</i> (6.3)	<i>Leucopogon ruscifolius</i> (29%, 4) <i>Thryptomene oligandra</i> (21%, 7)	Heath formation differentiates this from RE 3.5.42 which is a low woodland to open forest.
3.5.25 (1 site)	<i>Eucalyptus leptophleba</i> woodland on plains	<i>Eucalyptus leptophleba</i> (100%, 21) <i>Eucalyptus chlorophylla</i> (100%, 2) <i>Corymbia clarksoniana</i> (100%, 2)	<i>Eucalyptus leptophleba</i> (42.4) <i>Eucalyptus chlorophylla</i> (6.3)		
3.5.34 (4 sites)	<i>Corymbia nesophila</i> open forest on sand rises in the Torres Strait	<i>Corymbia nesophila</i> (100%, 20)	<i>Corymbia nesophila</i> (22.5) <i>Acacia simsii</i> (9.7)	<i>Corymbia clarksoniana</i> (50%, 13) <i>Corymbia stockeri</i> (50%, 6) <i>Jacksonia thesioides</i> (50%, 15) S1	Occurs on Torres Strait Islands. <i>Corymbia stockeri</i> is likely to be <i>Corymbia stockeri</i> subsp. <i>peninsularis</i> .

3.5.35 (13 sites)	<i>Eucalyptus tetradonta</i> , <i>Corymbia nesophila</i> woodland with a heathy understory	<i>Eucalyptus tetradonta</i> (100%, 8) <i>Corymbia nesophila</i> (92%, 5) <i>Asteromyrtus brassii</i> (77%, 4) T2. <i>Neofabricia myrtifolia</i> (69%, 4) T2 <i>Neoroepa banksii</i> (69%, 6) S1 <i>Acacia calyculata</i> (77%, 2) S1	<i>Neoroepa banksii</i> (14.1) S1 <i>Lophostemon suaveolens</i> (10.3) <i>Asteromyrtus brassii</i> (8) T2 <i>Acacia calyculata</i> (6.7) S1 <i>Neofabricia myrtifolia</i> (6.3)	<i>Lophostemon suaveolens</i> (15%, 13)	Characterised by a dominance of heath species in the understory. The understory species mix of this community may change depending on fire history.
3.5.36	<i>Eucalyptus tetradonta</i> and <i>Corymbia nesophila</i> woodland to open forest on undulating plains and remnant plateaus				
3.5.36a (40 sites)	<i>Eucalyptus tetradonta</i> and <i>Corymbia nesophila</i> woodland on undulating plains	<i>Eucalyptus tetradonta</i> (100%, 11) <i>Corymbia nesophila</i> (100, 10) <i>Erythrophleum chlorostachys</i> (45%, 2)	<i>Corymbia nesophila</i> (13) <i>Eucalyptus tetradonta</i> (7.6)		Occurs on sandy and loamy plains off remnant bauxite plateaus.
3.5.36b (10 sites)	<i>Eucalyptus tetradonta</i> and <i>Corymbia nesophila</i> open forest on remnant plateaus	<i>Eucalyptus tetradonta</i> (100%, 15) <i>Corymbia nesophila</i> (100%, 15) <i>Erythrophleum chlorostachys</i> (50%, 3)	<i>Corymbia nesophila</i> (19.7) <i>Eucalyptus tetradonta</i> (12.3) <i>Parinari nonda</i> (6.8) <i>Planchonia careya</i> (6.5) T3, S1		Occurs on bauxite plateaus. May occur as a woodland.
3.5.37	<i>Eucalyptus tetradonta</i> +/- <i>Corymbia stockeri</i> woodland to tall open forest on erosional plains and remnant plateaus				
3.5.37a (14 sites)	<i>Eucalyptus tetradonta</i> +/- <i>Corymbia stockeri</i> woodland on erosional plains	<i>Eucalyptus tetradonta</i> (100%, 12) <i>Corymbia stockeri</i> (93%, 7) <i>Erythrophleum chlorostachys</i> (43%, 2)	<i>Corymbia stockeri</i> (9.7) <i>Eucalyptus brassiana</i> (7) <i>Eucalyptus tetradonta</i> (6.9)	<i>Xanthorrhoea johnsonii</i> (21%, 5) S1	Occurs predominantly on erosional plains and rises but can occur on edges of bauxite plateaus. <i>Corymbia stockeri</i> is likely to be subsp. <i>peninsularis</i> .
3.5.37b (16 sites)	<i>Eucalyptus tetradonta</i> +/- <i>Corymbia stockeri</i> tall woodland on tertiary remnant plateaus	<i>Eucalyptus tetradonta</i> (100%, 18) <i>Corymbia stockeri</i> (81%, 9) <i>Erythrophleum chlorostachys</i> (69%, 7)	<i>Eucalyptus tetradonta</i> (14.2) <i>Corymbia stockeri</i> (12.6) <i>Erythrophleum chlorostachys</i> (11.6)		Occurs predominantly on the Kimba plateau and northern extensions. <i>Corymbia stockeri</i> is likely to be subsp. <i>peninsularis</i> .

3.5.38	<i>Eucalyptus tetradonta</i> +/- <i>E. cullenii</i> or <i>E. tetradonta</i> +/- <i>C. stockeri</i> and <i>Melaleuca</i> spp. woodland on remnant surfaces	<i>Eucalyptus tetradonta</i> (100%, 9) <i>Corymbia stockeri</i> (100%, 4) <i>Eucalyptus cullenii</i> (86%, 6)	<i>Eucalyptus cullenii</i> (25) <i>Corymbia disjuncta</i> (7.5)	Both subspecies of <i>Corymbia stockeri</i> may occur. Occurs on weathered remnant surfaces often with ironstone nodules to surface. <i>Melaleuca stenostachya</i> may occur but will have lower biomass than the <i>C. stockeri</i> and <i>Eucalyptus cullenii</i> .
3.5.38a (7 sites)	<i>Eucalyptus tetradonta</i> , <i>Corymbia stockeri</i> +/- <i>E. cullenii</i> woodland on remnant surfaces	<i>Eucalyptus tetradonta</i> (100%, 9) <i>Corymbia stockeri</i> (100%, 4) <i>Eucalyptus cullenii</i> (86%, 6)	<i>Eucalyptus cullenii</i> (25) <i>Corymbia disjuncta</i> (7.5)	Both subspecies of <i>Corymbia stockeri</i> may occur. Occurs on weathered remnant surfaces often with ironstone nodules to surface. <i>Melaleuca stenostachya</i> may occur but will have lower biomass than the <i>C. stockeri</i> and <i>Eucalyptus cullenii</i> .
3.5.38b (7 sites)	<i>E. tetradonta</i> +/- <i>C. stockeri</i> woodland with a <i>Melaleuca</i> spp shrub layer on remnant surfaces	<i>Eucalyptus tetradonta</i> (100%, 8) <i>Corymbia stockeri</i> (86%, 7) <i>Melaleuca viridiflora</i> (100%, 3) T2 <i>Melaleuca stenostachya</i> (43%, 4) <i>Corymbia clarksoniana</i> (43%, 2) <i>Melaleuca stenostachya</i> (57%, 4) T2	<i>Corymbia stockeri</i> (7.9) <i>Melaleuca stenostachya</i> (7.9)	Occurs on weathered remnant surfaces often with ironstone nodules to surface.
3.5.39 (26 sites)	<i>Eucalyptus tetradonta</i> +/- <i>Corymbia clarksoniana</i> woodland on sand plains	<i>Eucalyptus tetradonta</i> (100%, 13) <i>Corymbia clarksoniana</i> (73%, 4) <i>Erythrophleum chlorostachys</i> (54%, 4)	<i>Eucalyptus tetradonta</i> (7.1)	<i>Corymbia stockeri</i> (15%, 7) <i>Corymbia tessellaris</i> (12%, 7) <i>Eucalyptus leptophleba</i> (4%, 8)
3.5.40 (5 sites)	<i>Melaleuca stenostachya</i> +/- <i>Eucalyptus chlorophylla</i> woodland +/- <i>M. viridiflora</i> shrub layer on outwash plains	<i>Melaleuca stenostachya</i> (100%, 9) <i>Eucalyptus chlorophylla</i> (60%, 8) <i>Melaleuca viridiflora</i> (60%, 3)	<i>Melaleuca stenostachya</i> (35.7) <i>Eucalyptus chlorophylla</i> (17.3) <i>Corymbia hylandii</i> (8.7) <i>Acacia brassii</i> (7.2)	Occurs on weathered remnant surfaces often with ironstone nodules to surface.
3.5.41	<i>Melaleuca viridiflora</i> +/- <i>Corymbia clarksoniana</i> woodland to low open woodland on plains	<i>Melaleuca viridiflora</i> (60%, 8) <i>Corymbia clarksoniana</i> (43%, 2)	<i>Melaleuca viridiflora</i> (8.7)	Occurs on weathered remnant surfaces often with ironstone nodules to surface.

3.5.41a (14 sites)	<i>Melaleuca viridiflora</i> +/- <i>Corymbia</i> <i>clarksoniana</i> woodland on plains	<i>Corymbia clarksoniana</i> (86%, 12) <i>Melaleuca viridiflora</i> (50%, 9) <i>Melaleuca viridiflora</i> (79%, 2) T2	<i>Corymbia clarksoniana</i> (17.7) <i>Melaleuca viridiflora</i> (8.7)	<i>Melaleuca viridiflora</i> (64%, 2) S1	
3.5.41b (1 site)	<i>Melaleuca viridiflora</i> +/- <i>Petalostigma</i> <i>banksii</i> low open woodland on plains	<i>Melaleuca viridiflora</i> (100%, 7) <i>Petalostigma banksii</i> (100%, 5) T2	<i>Petalostigma banksii</i> (14.6) <i>Melaleuca viridiflora</i> (10.3) <i>Wrightia saligna</i> (6.1)		
3.5.42 (14 sites)	<i>Asteromyrtus brassii</i> and/or <i>Neofabricia</i> <i>myrtifolia</i> low open forest to woodland on sand plains	<i>Neofabricia myrtifolia</i> (93%, 9) <i>Asteromyrtus brassii</i> (93%, 6) <i>Allocasuarina littoralis</i> (71%, 3) T2	<i>Leucopogon yorkensis</i> (24.1) <i>Neofabricia myrtifolia</i> (20.3) <i>Asteromyrtus brassii</i> (19.4) <i>Callitris intratropica</i> (16.9) <i>Choriceras tricornis</i> (11.4) <i>Allocasuarina littoralis</i> (11.2) <i>Dodonaea polyandra</i> (10.6) <i>Welchiodendron longivalve</i> (10) <i>Alyxia spicata</i> (9.2) <i>Otax pendula</i> (6.8)	<i>Welchiodendron longivalve</i> (36%, 5) <i>Leucopogon yorkensis</i> (29%, 21) <i>Choriceras tricornis</i> (43%, 4) <i>Dodonaea polyandra</i> (36%, 4) <i>Callitris intratropica</i> (21%, 5) <i>Alyxia spicata</i> (71%, 2) S1 <i>Neoroepora banksii</i> (21%, 5) S1 <i>Melaleuca viridiflora</i> (50%, 5)	Woodland to open forest structure differentiates this from RE 3.5.19.
Land Zone 12					
3.12.7 (2 sites)	<i>Corymbia</i> <i>clarksoniana</i> , <i>Eucalyptus brassiana</i> open forest on granite ranges	<i>Corymbia clarksoniana</i> (100%, 20) <i>Eucalyptus brassiana</i> (100%, 12)	<i>Corymbia clarksoniana</i> (35.5) <i>Chionanthus ramiflorus</i> (29.4) <i>Eucalyptus brassiana</i> (28.9) <i>Melaleuca nervosa</i> (21.8) <i>Calophyllum sil</i> (15.5) <i>Wikstroemia indica</i> (13.8) <i>Celtis paniculata</i> (12) <i>Polyscias elegans</i> (12) <i>Ganophyllum falcatum</i> (11.9) <i>Litsea glutinosa</i> (11.9) <i>Cupaniopsis anacardioides</i> (6.9)	<i>Corymbia stockeri</i> (50%, 5) <i>Chionanthus ramiflorus</i> (50%, 20) T2 <i>Melaleuca nervosa</i> (50%, 12) T2	
3.12.9 (1 site)	<i>Corymbia tessellaris</i> +/- <i>Welchiodendron</i> <i>longivalve</i> +/- <i>Eucalyptus cullenii</i> open forest on foothills of granite hills.	<i>Corymbia tessellaris</i> (100%, 30)	<i>Corymbia tessellaris</i> (51.1) <i>Tabernaemontana pandacaqui</i> (6.9)		

3.12.10	<i>Eucalyptus cullenii</i> +/- <i>Corymbia clarksoniana</i> woodland or <i>E. chlorophylla</i> woodland on granitic ranges	<i>Eucalyptus cullenii</i> (100%, 12) <i>Corymbia clarksoniana</i> (69%, 5)	<i>Eucalyptus cullenii</i> (28.8) <i>Dendrolobium umbellatum</i> (14.8) <i>Dolichandrone heterophylla</i> (8.1) <i>Acacia oraria</i> (6.1)	<i>Dendrolobium umbellatum</i> (38%, 8) T2 <i>Dendrolobium umbellatum</i> (38%, 7) S1	This vegetation community is unmappable at the RE mapping scale and interspersed with vegetation community 3.12.10b. These communities are mapped collectively as 3.12.10
3.12.10a (13 sites)	<i>Eucalyptus cullenii</i> +/- <i>Corymbia clarksoniana</i> woodland on granitic ranges	<i>Eucalyptus cullenii</i> (28.8) <i>Dendrolobium umbellatum</i> (14.8)			
3.12.10b (2 sites)	<i>Eucalyptus chlorophylla</i> woodlands on granitic ranges	<i>Eucalyptus chlorophylla</i> (19.6) <i>Corymbia confertiflora</i> (9.6)	<i>Eucalyptus chlorophylla</i> (19.6) <i>Corymbia confertiflora</i> (9.5)	<i>Corymbia confertiflora</i> (50%, 3)	This vegetation community is unmappable at the RE mapping scale and interspersed with vegetation community 3.12.10a. These communities are mapped collectively as 3.12.10.
3.12.11 (6 sites)	<i>Corymbia stockeri</i> +/- <i>Welchiodendron longivalve</i> woodland on steep to rolling granite hills	<i>Corymbia stockeri</i> (100%, 23) <i>Dalbergia densa</i> (10.9) <i>Welchiodendron longivalve</i> (9.4)	<i>Corymbia stockeri</i> (45.9) <i>Dalbergia densa</i> (10.9) <i>Cryptocarya exfoliata</i> (9.8) <i>Welchiodendron longivalve</i> (9.2)	<i>Eucalyptus cullenii</i> (17%, 10) <i>Corymbia tessellaris</i> (17%, 8) <i>Welchiodendron longivalve</i> (17%, 8) <i>Dalbergia densa</i> (83%, 2) S1	
3.12.18	<i>Eucalyptus leptophleba</i> and <i>Corymbia clarksoniana</i> woodland or <i>Corymbia disjuncta</i> woodland on steep to low igneous hills	<i>Eucalyptus leptophleba</i> (100%, 13) <i>Corymbia clarksoniana</i> (86%, 3)	<i>Eucalyptus leptophleba</i> (35.1) <i>Dendrolobium umbellatum</i> (9.9) <i>Eucalyptus platyphylla</i> (8.4) <i>Flueggea virosa</i> (6.7)	<i>Eucalyptus platyphylla</i> (14%, 8) <i>Dendrolobium umbellatum</i> (43%, 7) T2 <i>Dendrolobium umbellatum</i> (43%, 3) S1	
3.12.18a (7 sites)	<i>Eucalyptus leptophleba</i> +/- <i>Corymbia clarksoniana</i> woodland to open woodland on steep to low igneous hills	<i>Eucalyptus leptophleba</i> (35.2) <i>Dendrolobium umbellatum</i> (10) <i>Flueggea virosa</i> (6.7)			
3.12.18b (2 sites)	<i>Corymbia disjuncta</i> +/- <i>C. clarksoniana</i> woodland on steep igneous hillslopes	<i>Corymbia disjuncta</i> (100%, 8) <i>Corymbia clarksoniana</i> (100%, 1)	<i>Corymbia disjuncta</i> (29.4) <i>Ptilostigma malabaricum</i> (6.9) <i>Alstonia</i> sp. (6.3)		Restricted to Orchid Creek area.

3.12.28 (5 sites)	<i>Leptospermum purpurascens</i> tall shrubland on igneous hills	<i>Leptospermum purpurascens</i> (100%, 19)	<i>Leptospermum purpurascens</i> (44.3) <i>Shonia tristigma</i> (10.7) <i>Hovea</i> sp. (7.7)	<i>Leptospermum purpurascens</i> (44.4) <i>Shonia tristigma</i> (10.7) <i>Acacia brassii</i> (10.2) <i>Hovea</i> sp.(7.7) <i>Lomandra banksii</i> (6.2)	<i>Shonia tristigma</i> (20%, 6) S2
3.12.30 (1 site)	<i>Imperata cylindrica</i> +/- <i>Heteropogon contortus</i> +/- <i>Mnesithea rotboellioides</i> tussock grassland on steep slopes of igneous hills	<i>Imperata cylindrica</i> (100%, 52) <i>Heteropogon contortus</i> (100%, 18) <i>Mnesithea rotboellioides</i> (100%, 11)	<i>Imperata cylindrica</i> (71.3) <i>Heteropogon contortus</i> (41.6) <i>Mnesithea rotboellioides</i> (32.5) <i>Cymbopogon refractus</i> (19.6)	<i>Imperata cylindrica</i> (71.4) <i>Heteropogon contortus</i> (41.6) <i>Mnesithea rotboellioides</i> (32.5) <i>Cymbopogon refractus</i> (19.6)	
3.12.32 (2 sites)	<i>Schizachyrium</i> spp. +/- <i>Rhynchosia</i> spp. grasslands on granite hills and rck pavmet	<i>Schizachyrium</i> sp. (100%, 9) <i>Ectrosia</i> sp. (100%, 8) <i>Rhynchosia minima</i> (100%, 5) <i>Eriachne</i> sp. (100%, 4)	<i>Ectrosia leporina</i> (26.8) <i>Rhynchosia</i> sp. (19.8) <i>Eriachne</i> sp. (18.1) <i>Schizachyrium</i> sp. (13.8) <i>Polycarpaea spirostylis</i> (12.2) <i>Scleria</i> sp. (10.5) <i>Fimbristylis</i> sp. (9.4)	<i>Ectrosia leporina</i> (26.9) <i>Rhynchosia</i> sp. (19.7) <i>Eriachne</i> sp. (18.1) <i>Schizachyrium</i> sp. (13.6) <i>Polycarpaea spirostylis</i> (12.2) <i>Scleria</i> sp. (10.5) <i>Fimbristylis</i> sp.(9.4) <i>Pseudopogonatherum contortum</i> (6.9) <i>Ptilotus</i> sp. (6.9) <i>Indigofera</i> sp. (6.7)	Restricted to granite hills south of Coen.
3.12.34 (2 sites)	Igneous rock pavements associated with mountains and some offshore islands	<i>Rhynchosia</i> sp. (100%)	Rock pavement (99.5)	Rock pavement (99.5)	Occurs as a complex of rock pavements (rather than boulder piles) with shrubland that can include <i>Acacia umbellata</i> , <i>Canarium australianum</i> , <i>Psidium</i> spp., <i>Terminalia</i> spp. and <i>Carissa ovata</i> growing in areas where soil accumulates
3.12.39	<i>Eucalyptus crebra</i> +/- <i>Corymbia hylandii</i> or <i>Lophostemon suaveolens</i> low woodland to low open forest on granite hills				

3.12.39a (2 sites)	<i>Eucalyptus crebra</i> +/- <i>Corymbia hylandii</i> low woodland to low open forest on skeletal soils on gullies and foothills of granite hills	<i>Eucalyptus crebra</i> (100%, 10)	<i>Eucalyptus crebra</i> (22.2) <i>Diospyros</i> sp. (19.1) <i>Acacia disparrima</i> (17.7) <i>Manilkara kauki</i> (16.9) <i>Terminalia subacrotora</i> (14.7) <i>Micromelum minutum</i> (10) <i>Alphitonia excelsa</i> (9.8) <i>Argophyllum verae</i> (9.8) <i>Bombax ceiba</i> (9.8) <i>Cupaniopsis fleckeri</i> (9.8) <i>Psychotria</i> sp. (7.2)	<i>Corymbia hylandii</i> (29.8) <i>Cochlospermum gillivraei</i> (27.4) <i>Eucalyptus crebra</i> (21.9) <i>Diospyros</i> sp. (19.1) <i>Acacia disparrima</i> (17.7) <i>Manilkara kauki</i> (16.9) <i>Terminalia subacrotora</i> (14.7) <i>Gardenia</i> sp. (12.8) <i>Erythrophleum chlorostachys</i> (11.2) <i>Micromelum minutum</i> (10) <i>Alphitonia excelsa</i> (9.8) <i>Argophyllum verae</i> (9.8) <i>Bombax ceiba</i> (9.8) <i>Cupaniopsis fleckeri</i> (9.8) <i>Psyrax</i> sp (8.8) <i>Sterculia quadrijida</i> (8) <i>Psychotria</i> sp. (7.2) <i>Euroschinus falcatus</i> (6.7)	<i>Corymbia hylandii</i> (50%, 19) <i>Cochlospermum gillivraei</i> (50%, 18)
3.12.39b (1 site)	<i>Lophostemon</i> <i>suaveolens</i> low open forest on upper slopes of granite ranges.	<i>Lophostemon suaveolens</i> (100%, 30) <i>Eucalyptus crebra</i> (100%, 8)	<i>Endiandra glauca</i> (53.2) <i>Myrsine variabilis</i> (21.9) <i>Litsea breviumbellata</i> (16.9) <i>Cyclophyllum coprosmoides</i> (13.8) <i>Rhodammia</i> sp. (13.8) <i>Cryptocarya vulgaris</i> (9.8) <i>Guioa acutifolia</i> (9.8) <i>Mischocarpus</i> sp. (9.8)	<i>Endiandra glauca</i> (100%, 30) S1	
3.12.40 (7 sites)	<i>Corymbia nesophila</i> +/- <i>Eucalyptus</i> <i>tetradonta</i> woodlands on igneous hills and rises	<i>Corymbia nesophila</i> (100%, 27)	<i>Corymbia nesophila</i> (52) <i>Eucalyptus tetradonta</i> (16.9) <i>Coelospermum reticulatum</i> (9.8)	<i>Corymbia nesophila</i> (51.7) <i>Eucalyptus tetradonta</i> (16.6) <i>Neoroepora banksii</i> (10.6) <i>Coelospermum reticulatum</i> (9.8) <i>Antidesma ghaesembilla</i> (8.7) <i>Erythrophleum chlorostachys</i> (7.5) <i>Hibbertia</i> sp. (7)	<i>Eucalyptus tetradonta</i> (57%, 12)
3.12.41 (6 sites)	<i>Eucalyptus tetradonta</i> woodland +/- heath species on granite hills and rises	<i>Eucalyptus tetradonta</i> (100%, 6) <i>Neofabricia myrtifolia</i> (83%, 3) T2 <i>Asteromyrtus brassii</i> (67%, 3) T2 <i>Acacia rothii</i> (67%, 5) T2 <i>Xanthorrhoea johnsonii</i> (83%, 3) S1	<i>Asteromyrtus brassii</i> (19.1) <i>Acacia rothii</i> (17.8) <i>Neofabricia myrtifolia</i> (12.4) <i>Eucalyptus tetradonta</i> (10.2) <i>Grevillea glauca</i> (10) <i>Xanthorrhoea johnsonii</i> (9.8) <i>Hibbertia banksii</i> (8.9) <i>Allocasuarina littoralis</i> (8.4) <i>Corymbia nesophila</i> (7.6) <i>Banksia dentata</i> (6.3)	<i>Corymbia nesophila</i> (50%, 8) <i>Asteromyrtus brassii</i> (50%, 7) <i>Corymbia stockeri</i> (17%, 8)	

3.12.42 (12 sites)	<i>Eucalyptus tetradonta</i> woodland on low to undulating granite hills.	<i>Eucalyptus tetradonta</i> (100%, 11) <i>Corymbia clarksoniana</i> (67%, 3)	<i>Eucalyptus tetradonta</i> (20.7)	<i>Eucalyptus tetradonta</i> (20.4) <i>Erythrophleum chlorostachys</i> (7.3)	<i>Eucalyptus brassiana</i> (8%, 17) <i>Corymbia stockeri</i> (8%, 15)
3.12.43 (5 sites)	<i>Welchiodendron longivalve</i> , <i>Acacia brassii</i> low woodland on igneous hills	<i>Welchiodendron longivalve</i> (100%, 7) <i>Acacia brassii</i> (75%, 6)	<i>Welchiodendron longivalve</i> (24) <i>Acacia brassii</i> (17.4)	<i>Welchiodendron longivalve</i> (24) <i>Acacia brassii</i> (17.4) <i>Eucalyptus platyphylla</i> (13.5) <i>Bursaria incana</i> (8.3) <i>Petalostigma pubescens</i> (7.6) <i>Livistona muelleri</i> (6.1)	<i>Eucalyptus cullenii</i> (25%, 13) <i>Eucalyptus platyphylla</i> (25%, 10) <i>Neofabricia myrtifolia</i> (25%, 6)
3.12.43a (4 sites)	<i>Welchiodendron longivalve</i> and <i>Acacia brassii</i> low woodland on undulating igneous rises to steep hills.	<i>Welchiodendron longivalve</i> (100%, 10) <i>Acacia brassii</i> (100%, 6)	<i>Welchiodendron longivalve</i> (24) <i>Acacia brassii</i> (17.4)	<i>Welchiodendron longivalve</i> (24) <i>Acacia brassii</i> (17.4) <i>Eucalyptus platyphylla</i> (13.5) <i>Bursaria incana</i> (8.3) <i>Petalostigma pubescens</i> (7.6) <i>Livistona muelleri</i> (6.1)	<i>Eucalyptus cullenii</i> (25%, 13) <i>Eucalyptus platyphylla</i> (25%, 10) <i>Neofabricia myrtifolia</i> (25%, 6)
3.12.43b (1 site)	<i>Melaleuca viridiflora</i> and <i>Welchiodendron longivalve</i> shrubland on rocky igneous headlands	<i>Corymbia tessellaris</i> (100%, 10) <i>Jacksonia thesioides</i> (100%, 6)	<i>Corymbia tessellaris</i> (20.4) <i>Jacksonia thesioides</i> (19.6)	<i>Corymbia tessellaris</i> (20.4) <i>Jacksonia thesioides</i> (19.6) <i>Acacia crassicaarpa</i> (13.5) <i>Croton armhemicus</i> (8.2) <i>Syzygium suborbiculare</i> (8.1)	<i>Corymbia tessellaris</i> (100%, 6) E <i>Acacia crassicaarpa</i> (100%, 4) E Restricted to continental Torres Strait islands.
3.12.44 (3 sites)	<i>Melaleuca citrolens</i> low open woodland on low granite hills and rolling rises	<i>Melaleuca citrolens</i> (100%, 3)	<i>Melaleuca citrolens</i> (24.7) <i>Melaleuca foliolosa</i> (11.3) <i>Petalostigma banksii</i> (8.9)	<i>Melaleuca citrolens</i> (24.7) <i>Melaleuca foliolosa</i> (11.3) <i>Petalostigma banksii</i> (8.8)	<i>Melaleuca foliolosa</i> (33%, 3) <i>Petalostigma banksii</i> (33%, 3)
3.12.45 (5 sites)	<i>Melaleuca viridiflora</i> low woodland to low open woodland occurs on steep igneous hills and footslopes	<i>Melaleuca viridiflora</i> (100%, 13)	<i>Melaleuca viridiflora</i> (22.9)	<i>Melaleuca viridiflora</i> (22.9) <i>Choriceras tricornis</i> (8.1)	<i>Eucalyptus tetradonta</i> (20%, 4) E <i>Choriceras tricornis</i> (20%, 13) S1
3.12.47	Mixed heath species low woodland to wetter dwarf shrubland on igneous hills				
3.12.47a (4 sites)	Mixed heath species tall shrubland on igneous hills	<i>Asteromyrtus hysicephala</i> (100%, 3) S1 <i>Melaleuca viridiflora</i> (40%, 6) <i>Allocasuarina littoralis</i> (40%, 4) <i>Neofabricia myrtifolia</i> (40%, 4) <i>Melaleuca arcana</i> (40%, 4) <i>Asteromyrtus brassii</i> (40%, 3)	<i>Allocasuarina littoralis</i> (17) <i>Choriceras tricornis</i> (11.1) <i>Asteromyrtus hysicephala</i> (10.6) <i>Xanthorrhoea johnsonii</i> (9.9) <i>Melaleuca arcana</i> (9.8) <i>Jacksonia thesioides</i> (8.9) <i>Neofabricia myrtifolia</i> (7.3)	<i>Allocasuarina littoralis</i> (17) <i>Choriceras tricornis</i> (11.1) <i>Asteromyrtus hysicephala</i> (10.6) <i>Xanthorrhoea johnsonii</i> (9.9) <i>Melaleuca arcana</i> (9.8) <i>Jacksonia thesioides</i> (8.9) <i>Neofabricia myrtifolia</i> (7.3)	
3.12.47b (1 site)	<i>Asteromyrtus hysicephala</i> , <i>Choriceras tricornis</i> , <i>Jacksonia thesioides</i> dwarf shrubland on igneous slopes with impeded drainage	<i>Asteromyrtus hysicephala</i> (100%, 6) <i>Choriceras tricornis</i> (100%, 5) <i>Jacksonia thesioides</i> (100%, 4) <i>Melaleuca viridiflora</i> (100%, 3)	None	<i>Asteromyrtus hysicephala</i> (10.6) <i>Melaleuca viridiflora</i> (13.3) <i>Grevillea pteridifolia</i> (7)	

3.12.48	<i>Heteropogon triticeus</i> or <i>Themeda triandra</i> or <i>Schizachyrium</i> <i>fragile</i> tussock grassland on rocky igneous coastal headlands and islands	<i>Heteropogon triticeus</i> (100%, 42) <i>Sarga plumosum</i> (80%, 14)	<i>Heteropogon triticeus</i> (63.4) <i>Sarga plumosum</i> (32.2) <i>Ipomoea</i> sp. (9.8) <i>Eragrostis</i> sp. (8.3)	<i>Heteropogon triticeus</i> (63.5) <i>Sarga plumosum</i> (32.2) <i>Lomandra</i> sp. (15.8) <i>Lepturus repens</i> (13.9) <i>Ipomoea</i> sp. (9.8) <i>Salsola australis</i> (9.8) <i>Sesuvium portulacastrum</i> (9.8) <i>Eulalia mackinlayi</i> (9.6) <i>Lithomyrtus obtusa</i> (9.5) <i>Bulbostylis barbata</i> (8.8) <i>Eragrostis</i> sp. (8.3)	<i>Lomandra</i> sp. (20%, 13) <i>Lepturus repens</i> (20%, 10) <i>Lithomyrtus obtusa</i> (20%, 6) E
3.12.48a (5 sites)	<i>Heteropogon triticeus</i> +/- <i>Sarga plumosum</i> grasslands on igneous headlands and offshore islands	<i>Heteropogon triticeus</i> (100%, 42) <i>Sarga plumosum</i> (80%, 14)	<i>Heteropogon triticeus</i> (63.4) <i>Sarga plumosum</i> (32.2) <i>Ipomoea</i> sp. (9.8) <i>Eragrostis</i> sp. (8.3)	<i>Heteropogon triticeus</i> (63.5) <i>Sarga plumosum</i> (32.2) <i>Lomandra</i> sp. (15.8) <i>Lepturus repens</i> (13.9) <i>Ipomoea</i> sp. (9.8) <i>Salsola australis</i> (9.8) <i>Sesuvium portulacastrum</i> (9.8) <i>Eulalia mackinlayi</i> (9.6) <i>Lithomyrtus obtusa</i> (9.5) <i>Bulbostylis barbata</i> (8.8) <i>Eragrostis</i> sp. (8.3)	<i>Lomandra</i> sp. (20%, 13) <i>Lepturus repens</i> (20%, 10) <i>Lithomyrtus obtusa</i> (20%, 6) E
3.12.48c (1 site)	<i>Schizachyrium</i> spp., <i>Aristida</i> spp. grasslands on igneous headlands	<i>Aristida</i> sp. (100, 20) <i>Schizachyrium</i> sp. (100%, 20)	None	<i>Aristida</i> sp. (43.3) <i>Schizachyrium</i> sp. (35.7)	

Appendix 2. Regional Ecosystems and vegetation communities on land zone 5 and 12 in Cape York Peninsula bioregion

Short label descriptions only are included. For more detailed descriptions please visit the website <https://environment.ehp.qld.gov.au/regional-ecosystems/>. Communities are grouped by vegetation formation.

RE Label	Short Description	Area km ² (areas <1 km ² in Ha)	No. of Vegetation communities	Derivation	No. of Sites	Comments & Distribution
Land zone 5						
<i>Grasslands</i> (0.01% of land zone)						
3.5.15b	<i>Dapsilanthus spathaceus</i> open sedgeland with emergent shrubs.	120 Ha		Quantitative	1	Occurs in mosaic patches with 3.5.15a only on Torres Strait Islands.
3.5.29	<i>Themeda triandra</i> and <i>Heteropogon contortus</i> closed tussock grasslands on erosional plains.	686 Ha	1	Qualitative	0	The grassland is a fire climax community, with scattered vine forest remnants clumped on low granite mounds which protrude above the undulating sand plain, offering some protection from fire. Moa Island in the Torres Strait and also on coastal areas and islands in the north-east of the bioregion.
<i>Shrublands</i> (4% of land zone)						
3.5.19	<i>Asteromyrtus lysicephala</i> and <i>Neofabricia myrifolia</i> open heath to shrubland on sand sheets.	1,902	1	Quantitative	14	Heath formation differentiates this from RE 3.5.42 which is a low woodland to open forest. From Coen to the McHenry Uplands. Also occurs just north of Hopevale.
3.5.43	<i>Asteromyrtus brassii</i> + <i>Melaleuca saligna</i> tall shrubland on residual sand plains.	230 Ha	1	Qualitative	0	Restricted to Moa Island in the Torres Strait.
<i>Woodlands</i> (95% of land zone)						
3.5.5	<i>Corymbia novoguineensis</i> +/- <i>C. tessellaris</i> woodland on sand plains on northern Cape York Peninsula.	117	1	Quantitative	6	Occurs on northern Cape York Peninsula and Torres Strait islands.
3.5.6	<i>Eucalyptus phoenicea</i> +/- <i>E. tetradonta</i> woodland on sandy outwash plains.	401	1	Quantitative	4	On the Battlecamp sandstone plateau.
3.5.9	<i>Eucalyptus tetradonta</i> and <i>Corymbia stockeri</i> and/or <i>C. setosa</i> on sand plains.	5,346	1	Quantitative	11	Western side of Great Dividing Range and Kalpowar Plains. Widespread in the southern half of the bioregion and occurs predominantly on the Holroyd Plain in close association with 3.5.37a.
3.5.15	<i>Melaleuca viridiflora</i> and <i>Asteromyrtus symphyocarpa</i> low woodland on colluvial plains.	33	2			
Vegetation communities:						
3.5.15a	<i>Melaleuca viridiflora</i> +/- <i>M. saligna</i> +/- <i>Corymbia</i> spp. low woodland or tall shrubland on residual sands.	32		Qualitative	0	Occurs on depositional and erosional plains. Widespread throughout the bioregion.
3.5.15b (Included in grasslands)						
3.5.21	<i>Corymbia clarksoniana</i> +/- <i>C. tessellaris</i> open forest on coastal ranges and lowlands.	54	1	Qualitative	0	On east coast from Cooktown to Lockhart River.
3.5.24	<i>Eucalyptus chlorophylla</i> +/- <i>Corymbia clarksoniana</i> woodland on erosional plains.	290	1	Qualitative	0	Occurs in the south-east of the bioregion on the Laura Basin and south of Cooktown.

RE Label	Short Description	Area km ² (areas <1 km ² in Ha)	No. of Vegetation communities	Derivation	No. of Sites	Comments & Distribution
3.5.25	<i>Eucalyptus leptophleba</i> and <i>E. platyphylloides</i> woodland on outwash plains.	45	1	Quantitative	1	South-east of bioregion.
3.5.26	<i>Eucalyptus platyphylloides</i> +/- <i>Corymbia clarksoniana</i> woodland to open forest on flat wet plains.	22	1	Qualitative	0	Occurs in the south-east of the bioregion close to the Wet Tropics Bioregion boundary.
3.5.27	<i>Melaleuca citrolens</i> +/- <i>M. foliolosa</i> +/- <i>M. viridiflora</i> low open woodland on plains.	711 Ha	1	Qualitative	0	Restricted to Rinyirru (Lakefield) National Park (CYPAL) within the Laura Basin.
3.5.32	<i>Asteromyrtus brassii</i> +/- <i>Syzygium angophorooides</i> +/- <i>Acmena hemilampra</i> open forest on residual sand rises and sheets.	228 Ha	1	Qualitative	0	Restricted to Moa Island, Torres Strait.
3.5.34	<i>Corymbia nesophila</i> and <i>C. stockeri</i> open forest on sand rises in the Torres Strait	116 Ha	1	Quantitative	4	Restricted to Moa Island, Torres Strait.
3.5.35	<i>Eucalyptus tetradonta</i> and <i>Corymbia nesophila</i> woodland with heathy understorey on sand plains.	2,241	1	Quantitative	13	Areas dominated by <i>Corymbia nesophila</i> sometimes occur. The understorey species mix of this community may change depending on fire history. Extensive on sandplains from the upper reaches of the Archer River and Temple Bay through the McHenry Uplands to Bamaga.
3.5.36	<i>Eucalyptus tetradonta</i> and <i>Corymbia nesophila</i> woodland on undulating plains and remnant plateaus. Vegetation communities:	17,751	2			
	3.5.36a <i>Eucalyptus tetradonta</i> and <i>Corymbia nesophila</i> woodland on undulating plains.	9,880		Quantitative	40	Occurs from Cape Melville, across to the Kimba plateau and throughout the north of the bioregion.
	3.5.36b <i>Eucalyptus tetradonta</i> and <i>Corymbia nesophila</i> woodland to open forest on plateaus.	7,870		Quantitative	10	Small unmapable areas of 3.5.36a occur sporadically throughout the range of this community. Occurs on the Weipa Plateau and other remnant bauxite plateaus. 3.5.36b is generally taller than 3.5.36a. Occurs more often as an open forest in the north.
3.5.37	<i>Eucalyptus tetradonta</i> and <i>Corymbia stockeri</i> +/- <i>Erythrophloeum chlorostachys</i> woodland on erosional plains and deep massive sands. Vegetation communities:	5,587	2			
	3.5.37a <i>Eucalyptus tetradonta</i> and <i>Corymbia stockeri</i> woodland.	4,585		Quantitative	15	Occurs on depositional and erosional plains. Widespread throughout the bioregion.
	3.5.37b <i>Eucalyptus tetradonta</i> ± <i>Erythrophloeum chlorostachys</i> ± <i>Corymbia stockeri</i> tall woodland.	998		Quantitative	13	Predominantly on the Kimba Plateau, but may occur on other tertiary remnant plateaus in small patches below the scale of mapping. Typical vegetation is on the Kimba Plateau in an area known as the Desert. Occurs on tertiary remnant plateaus whilst 3.5.37a does not. 3.5.37b is generally taller than 3.5.37a (above 22m).
3.5.38	<i>Eucalyptus tetradonta</i> and <i>E. cullenii</i> woodland or <i>E. tetradonta</i> and <i>C. stockeri</i> +/- <i>Melaleuca</i> spp. woodland on remnant surfaces.	275	2			

RE Label	Short Description	Area km ² (areas <1 km ² in Ha)	No. of Vegetation communities	Derivation	No. of Sites	Comments & Distribution
Vegetation communities:						
3.5.38a	<i>Eucalyptus tetradonta</i> and <i>E. cullenii</i> +/- <i>Corymbia stockeri</i> woodland on remnant weathered surfaces.	269	1	Quantitative	6	Central plains on the Peninsula from Musgrave to the Torres Strait Islands.
3.5.38b	<i>Eucalyptus tetradonta</i> and <i>Corymbia stockeri</i> often with <i>Melaleuca stenostachya</i> .	640 Ha	1	Quantitative	7	Occurs in the central Peninsula from Coen to Bramwell Junction.
3.5.39	<i>Eucalyptus tetradonta</i> and <i>Corymbia clarksoniana</i> woodland on sand plains.	9,848	1	Quantitative	25	Mainly occurs on plains in the Laura basin, but also in the central west Peninsula and on some Torres Strait Islands.
3.5.40	<i>Melaleuca stenostachya</i> +/- <i>Eucalyptus chlorophylla</i> +/- <i>M. viridiflora</i> woodland on outwash plains.		1	Quantitative	5	Occurs throughout Cape York including the Torres Strait Islands.
3.5.41	<i>Melaleuca viridiflora</i> +/- <i>Corymbia clarksoniana</i> woodland on plains.	2,386	2			
Vegetation communities:						
3.5.41a	<i>Melaleuca viridiflora</i> +/- <i>Corymbia clarksoniana</i> woodland on plains	318 Ha	1	Quantitative	14	Central plains on the Peninsula from Musgrave to the Torres Strait Islands.
3.5.41b	<i>Melaleuca viridiflora</i> low open woodland +/- <i>Petalostigma banksii</i> on plains.	336 Ha	1	Quantitative	1	Occurs in the central Peninsula from Coen to Bramwell Junction.
3.5.42	<i>Asteromyrtus brassii</i> and/or <i>Neofabricia myrtifolia</i> low open forest to woodland on sand plains.	426	1	Quantitative	14	Occurs throughout Cape York including the Torres Strait Islands.
Land zone 12						
<i>Grasslands</i> (2% of land zone)						
3.12.30	<i>Imperata cylindrica</i> +/- <i>Mnesithea rotboellioides</i> closed tussock grassland on steep slopes of igneous hills.	37	1	Quantitative	1	Occurs mainly in the northern Mellwraith Range, but also near Temple Bay and on some Torres Strait Islands.
3.12.32	<i>Schizachyrium</i> spp. +/- <i>Rhynchosia</i> spp. grasslands on shallow soils on undulating granite hills.	10	1	Quantitative	2	Restricted to granite hills south of Coen.
3.12.34	Igneous rock pavements associated with mountains and some offshore islands.	66	1	Quantitative	2	Mellwraith Range, Iron Range and Altamouli Range. Torres Strait Islands.
3.12.48	<i>Heteropogon triticeus</i> or <i>Themeda triandra</i> or <i>Schizachyrium fragile</i> tussock grassland on rocky igneous coastal headlands and islands.	20	3			
Vegetation communities:						
3.12.48a	<i>Heteropogon triticeus</i> dominated grasslands on igneous headlands and offshore islands.	11	1	Quantitative	5	Occurs on Torres Strait Islands
3.12.48b	<i>Themeda triandra</i> tussock grassland on igneous headlands and islands.	533 Ha	1	Qualitative	0	Occurs on Torres Strait Islands and other offshore islands and headlands along the east coast.
3.12.48c	<i>Schizachyrium</i> spp. +/- <i>Aristida</i> spp. grasslands on igneous headlands.	356 Ha	1	Quantitative	1	Restricted to headlands near the northern most extent of Cape York Peninsula and some east coast islands.
<i>Shrublands</i> (12% of land zone)						
3.12.28	<i>Leptospermum purpurascens</i> tall shrubland on igneous hills.	58	1	Quantitative	5	Occurs on the western edge of Iron Range.

RE Label	Short Description	Area km ² (areas <1 km ² in Ha)	No. of Vegetation communities	Derivation	No. of Sites	Comments & Distribution
3.12.43b	<i>Melaleuca viridiflora</i> and <i>Welchiendendron longivalve</i> shrubland on rocky igneous headlands.	9a Ha		Quantitative	1	Occurs on most continental Torres Strait Islands.
3.12.46	<i>Melaleuca stenostachya</i> shrubland on exposed igneous headlands and hills.	52	1	Qualitative	0	Occurs on granite hills between Musgrave Roadhouse and Archer River crossing as well as on some Torres Strait Islands.
3.12.47b	<i>Asteromyrtus lysicephala</i> , <i>Choriceras tricornis</i> , <i>Jacksonia thesioides</i> dwarf shrubland on igneous slopes with impeded drainage			Quantitative	1	Occurs west of the Iron Range.
<i>Woodlands</i> (75% of land zone)						
3.12.7	<i>Eucalyptus brassiana</i> and <i>Corymbia clarksoniana</i> open forest on granite ranges.	103	1	Quantitative	2	Occurs on western McIlwraith Range and the Melville Range.
3.12.8	<i>Corymbia clarksoniana</i> +/- <i>C. tessellaris</i> open forest on coastal granite ranges and lowlands.	185	1	Qualitative	0	McIlwraith Range and Iron Range, with extensive patches on some Torres Strait islands.
3.12.9	<i>Corymbia tessellaris</i> +/- <i>Welchiendendron longivalve</i> +/- <i>E. cullenii</i> open forest on footslopes of granite hills.	49	1	Quantitative	1	East of McIlwraith Range and Torres Strait islands close to the Australian mainland.
3.12.10 [#]	<i>Eucalyptus cullenii</i> +/- <i>Corymbia clarksoniana</i> woodland or <i>E. chlorophylla</i> woodland on granitic ranges.	1,677	2			
Vegetation communities:						
3.12.10a	<i>Eucalyptus cullenii</i> +/- <i>Corymbia clarksoniana</i> woodland on granite hills and footslopes.	25		Quantitative	13	[#] This RE is mapped as 3.12.10 as the vegetation communities are unmapable at the RE mapping scale. Occurs along the length of the Great Dividing Range. Also in small patches on ranges from Cooktown to the Altamou Range.
3.12.10b	<i>Eucalyptus chlorophylla</i> woodlands on granite hills.	273 Ha		Quantitative	2	Occurs along the length of the Great Dividing Range. Also in small patches on ranges from Cooktown to the Altamou Range.
3.12.11	<i>Corymbia stockeri</i> +/- <i>Welchiendendron longivalve</i> woodland on steep to rolling granite hills.	137	1	Quantitative	6	Southern Torres Strait islands and near Lockhart River.
3.12.18	<i>Eucalyptus leptophleba</i> and <i>Corymbia clarksoniana</i> woodland or <i>C. disjuncta</i> woodland on steep to low igneous hills.	483	2			
Vegetation communities:						
3.12.18a	<i>Eucalyptus leptophleba</i> and <i>Corymbia clarksoniana</i> woodland to open woodland on steep to low igneous hills.	481		Quantitative	7	Central Peninsula along the Great Dividing Range.
3.12.18b	<i>Corymbia disjuncta</i> woodland on steep igneous hillslopes.	153 Ha		Quantitative	2	Restricted to hills northeast of Archer River Roadhouse.
3.12.23	<i>Acacia brassii</i> low open forest on igneous hills.	18	1	Qualitative	0	Torres Strait islands.
3.12.37	<i>Eucalyptus platyphylla</i> +/- <i>Corymbia</i> spp. woodland to open forest on coastal igneous headlands and footslopes.	547 Ha	1	Qualitative	0	Restricted to Horn and Muralag Islands in the Torres Strait.
3.12.38	<i>Corymbia clarksoniana</i> and/or <i>C. nesophila</i> and/or <i>C. stockeri</i> low woodland on acid volcanic hills.	17	1	Qualitative	0	Torres Strait Islands.
3.12.39	<i>Eucalyptus crebra</i> +/- <i>Corymbia hylandii</i> or <i>Lophostemon suaveolens</i> low open forest to low woodland on skeletal soils on gullies and foothills of granite hills.	63	2			

RE Label	Short Description	Area km ² (areas <1 km ² in Ha)	No. of Vegetation communities	Derivation	No. of Sites	Comments & Distribution
Vegetation communities:						
3.12.39a	<i>Eucalyptus crebra</i> +/- <i>Corymbia hylandii</i> low open forest on skeletal soils on gullies and foothills of granite hills.	52		Quantitative	2	Occurs on Cape Melville
3.12.39b	<i>Lophostemon suaveolens</i> , <i>Eucalyptus crebra</i> low open forest on upper slopes of granite ranges.	11		Quantitative	1	Occurs on the Altanmoui Range north of Cooktown.
3.12.40	<i>Corymbia nesophila</i> +/- <i>Eucalyptus tetradonta</i> woodlands on igneous hills and rises.	368	1	Quantitative	7	Occurs from Musgrave to Temple Bay on the Coen-Yambo Inlier. Small areas west of Cooktown and on some Torres Strait Islands.
3.12.41	<i>Eucalyptus tetradonta</i> +/- <i>Corymbia nesophila</i> woodland on igneous hills and rises.	120	1	Quantitative	6	Occurs mainly around Iron Range but also found south of Coen and north of Bamaga.
3.12.42	<i>Eucalyptus tetradonta</i> +/- <i>Corymbia clarksoniana</i> woodland on low to undulating granite hills.	2,445	1	Quantitative	12	Occurs along the length of the Coen -Yambo Inlier from Musgrave to Portlands Rds. Also on ranges from north-west of Cooktown.
3.12.43	<i>Welchiodendron longivalve</i> and <i>Acacia brassii</i> low woodland on undulating igneous rises to steep hills.	14	2			
Vegetation communities:						
3.12.43a	<i>Welchiodendron longivalve</i> and <i>Acacia brassii</i> low woodland on undulating igneous rises to steep hills.	13		Quantitative	4	Occurs near Wolverton and Temple Bay as well as on some Torres Strait Islands.
3.12.43b	(Included in shrublands)					
3.12.44	<i>Melaleuca citrolens</i> low open woodland on low granite hills and rolling rises.	61	1	Quantitative	3	Occurs on granite ranges from Coen to Palmerville.
3.12.45	<i>Melaleuca viridiflora</i> +/- <i>Eucalyptus</i> spp. low woodland to low open woodland on steep hills and footslopes.	141	1	Quantitative	5	Occurs along the length of the Great Dividing Range from the Hann River up to Portlands Rds.
3.12.47 [#]	Mixed heath species low woodland to wetter dwarf shrubland on igneous hills	352	2	Quantitative		
Vegetation communities:						
3.12.47a	Mixed heath species low woodland on igneous hills					#This RE is mapped as 3.12.47 as the vegetation communities are unmappable at the RE mapping scale.
3.12.47b	(Included in shrublands)			Quantitative	4	

Appendix 3a. Assessment of the sampling adequacy of survey design on land zone 5 and 12, Cape York Peninsula bioregion

Sampling adequacy of landscape variability

Table 3.1: Total area of land zone 5 and 12 at different similarity levels to any site for each environmental variable. For example, 818 ha of land zone 5 is between 75 – 89% similar in climate to any observational site. This represents 0.01% of the total area of land zone 5. The minimum similarity in climate of any grid cell to any observational site is 81%. Figures are rounded to the nearest hectare or km².

LZ 5	Observational sites				Analysis Sites					
	% Similarity Class	ha	km ²	% total area	Minimum %similarity	% Similarity Class	ha	km ²	% total area	Minimum %similarity
Climate	<75%	0	0	0%	81	<75%	205	2	0.003%	70
	75 - 89%	818	8	0.01%		75 - 89%	26076	261	0.41%	
	90-95%	3283	33	0.1%		90-95%	400620	4006	6.32%	
	>95%	6333855	63339	99.9%		>95%	5911054	59111	93.26%	
Vegetation Density	<75%	241	2	0.004%	63	<75%	3657	37	0.06%	12
	75 - 89%	659	7	0.01%		75 - 89%	6360	64	0.10%	
	90-95%	673	7	0.01%		90-95%	40744	407	0.64%	
	>95%	6336315	63363	99.98%		>95%	6287275	62873	99.2%	
Soil Nutrient	<75%	76	1	0.001%	66	<75%	2229	22	0.04%	0
	75 - 89%	53	1	0.001%		75 - 89%	261	3	0.004%	
	90-95%	197	2	0.003%		90-95%	20946	209	0.33%	
	>95%	6331552	63316	99.99%		>95%	6278650	62787	99.16%	
Soil Structure	<75%	0	0	0	84	<75%	21888	219	0.35%	0
	75 - 89%	426	4	0.01%		75 - 89%	65998	660	1.04%	
	90-95%	6601	66	0.10%		90-95%	493978	4940	7.80%	
	>95%	6324721	63247	99.89%		>95%	5749884	57499	90.81%	
LZ 12	Observational sites				Analysis sites					
	% Similarity Class	ha	km ²	% total area	Minimum %similarity	% Similarity Class	ha	km ²	% total area	Minimum %similarity
Climate	<75%	0	0	0.0%	84	<75%	149	1	0.02%	60
	75 - 89%	2903	29	0.3%		75 - 89%	17770	178	1.9%	
	90-95%	19622	196	2.1%		90-95%	178764	1788	19.5%	
	>95%	894385	8944	97.5%		>95%	720226	7202	78.5%	
Vegetation density	<75%	64	1	0.01%	59	<75%	524	5	0.06%	5
	75 - 89%	167	2	0.02%		75 - 89%	12768	128	1.4%	
	90-95%	1147	11	0.1%		90-95%	14875	149	1.6%	
	>95%	915365	9154	99.9%		>95%	888575	8886	96.9%	
Soil nutrient	<75%	651	7	0.1%	35	<75%	6879	69	0.8%	27
	75 - 89%	2615	26	0.3%		75 - 89%	12865	129	1.4%	
	90-95%	14165	142	1.6%		90-95%	21919	219	2.4%	
	>95%	884381	8844	98.1%		>95%	860150	8602	95.4%	
Soil structure	<75%	20	0.2	0.002%	68	<75%	1773	18	0.2%	48
	75 - 89%	5135	51	0.6%		75 - 89%	42218	422	4.7%	
	90-95%	28428	284	3.2%		90-95%	116632	1166	12.9%	
	>95%	868186	8682	96.3%		>95%	741159	7412	82.2%	

Appendix 3b: Areas of low sampling adequacy by survey design on land zone 5 and 12, Cape York Peninsula bioregion

Figure 3.1: Distribution of areas of land zone 5 and land zone 12 which are <90%-similar to any site for each environmental variable. Because such large areas of both land zones were >90%-similar to any site, for display purposes we show only areas with <90%-similarity. Areas on land zone 12 correspond largely with areas of rainforest which are not included in this study. These maps are indicative only. GIS layers are available from the first author if more detail is required

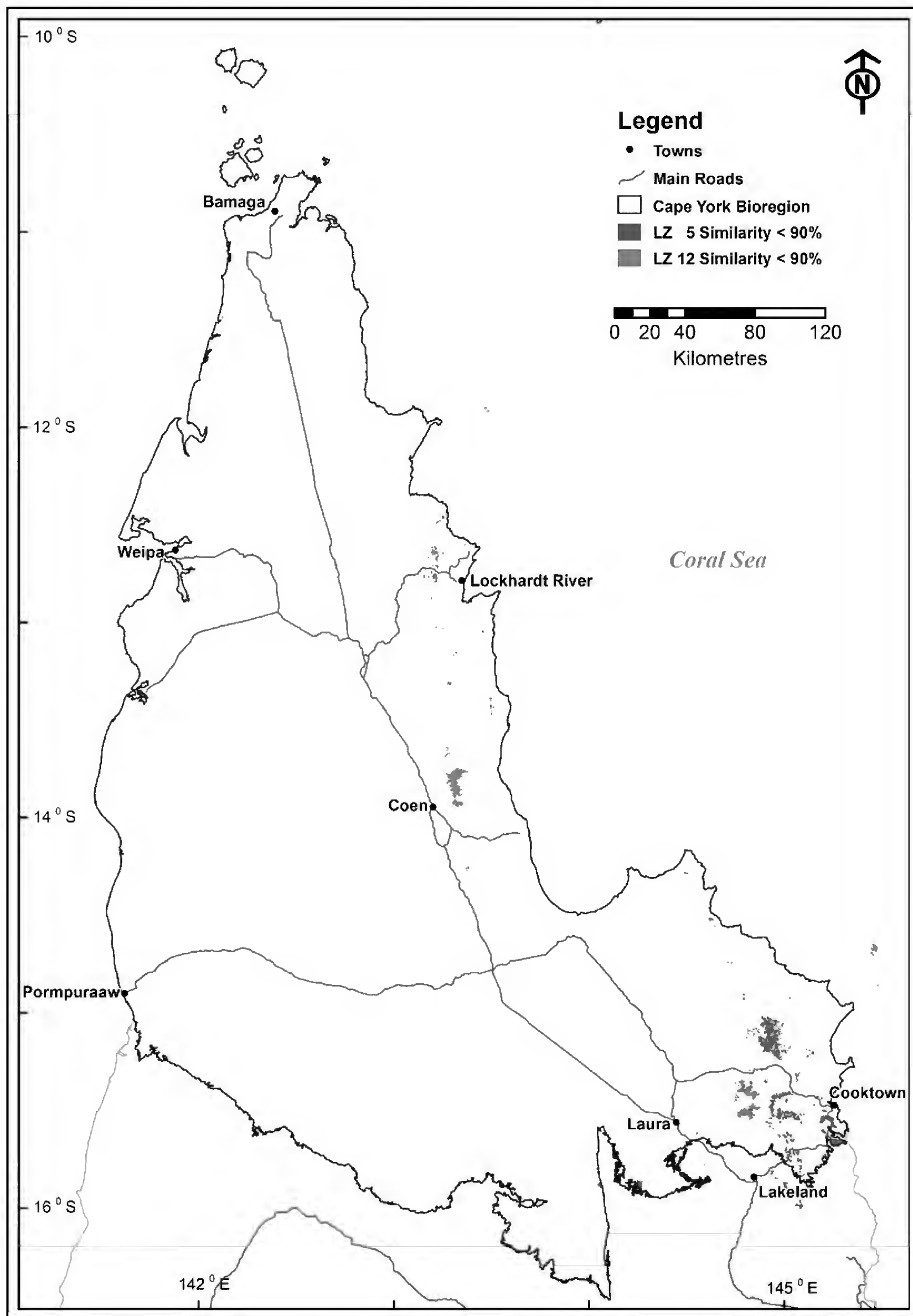


Figure 3.1a: Climate.

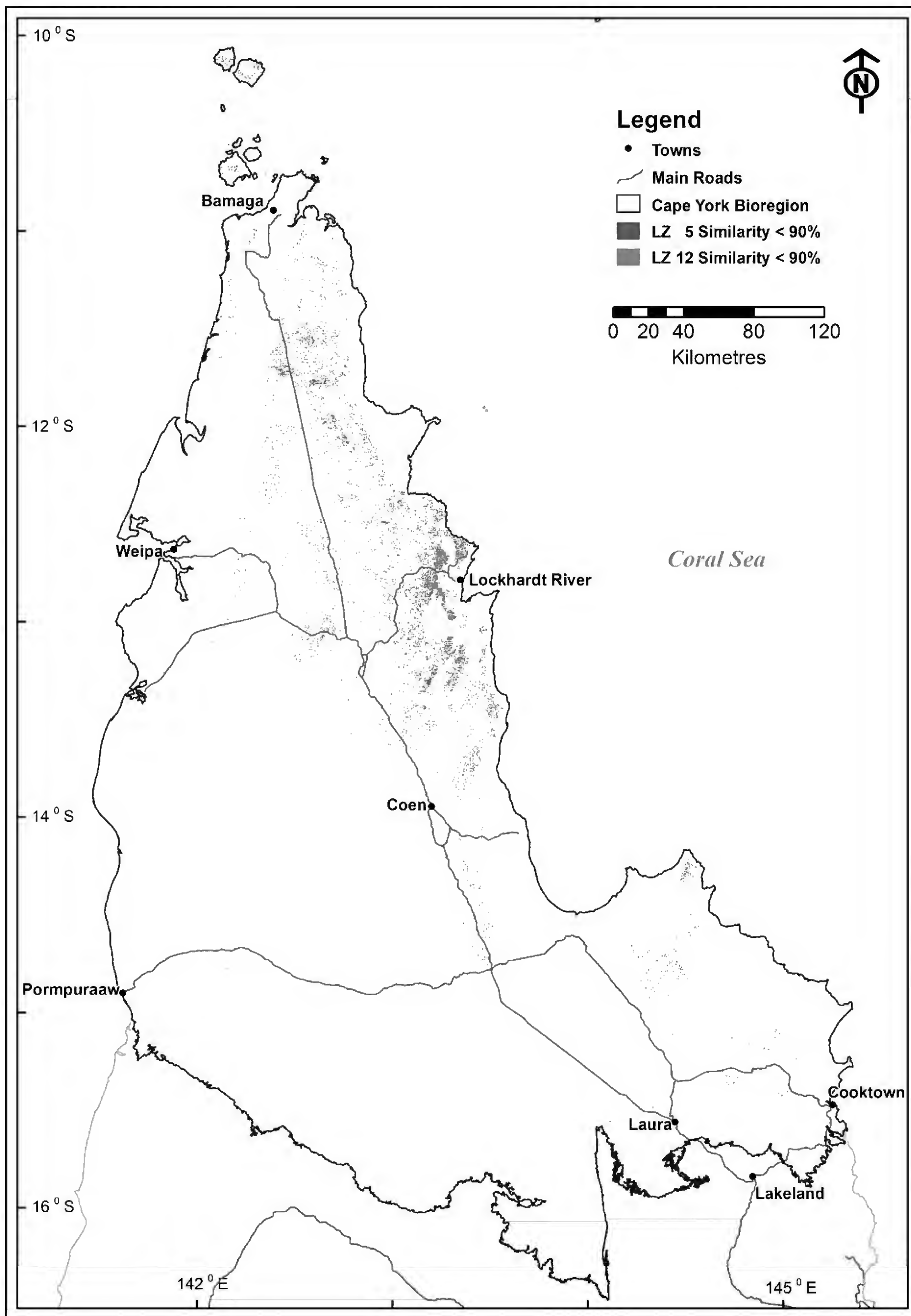


Figure 3.1b: Woody vegetation density (represented by maximum persistent greenness)

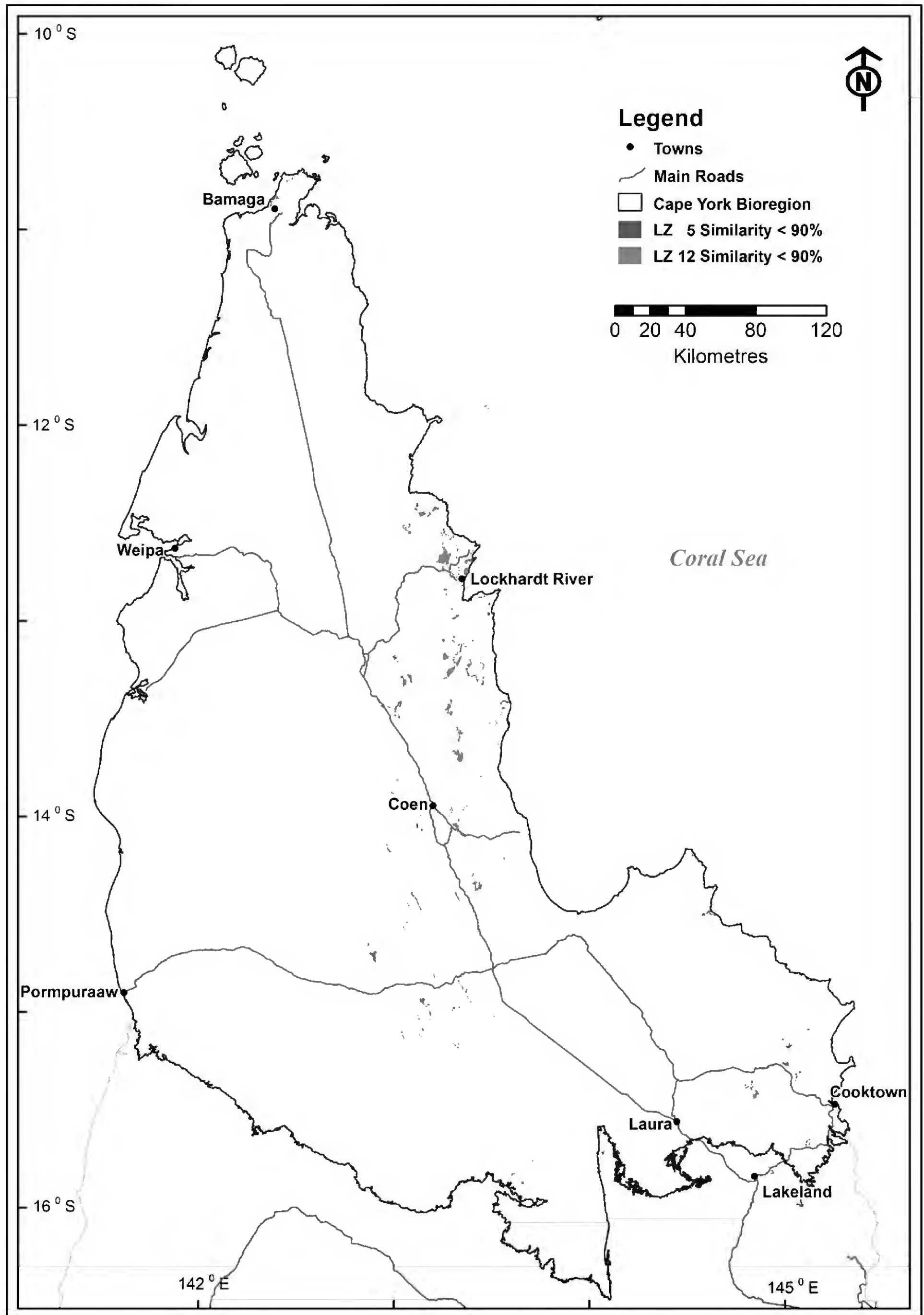


Figure 3.1c: Soil nutrient

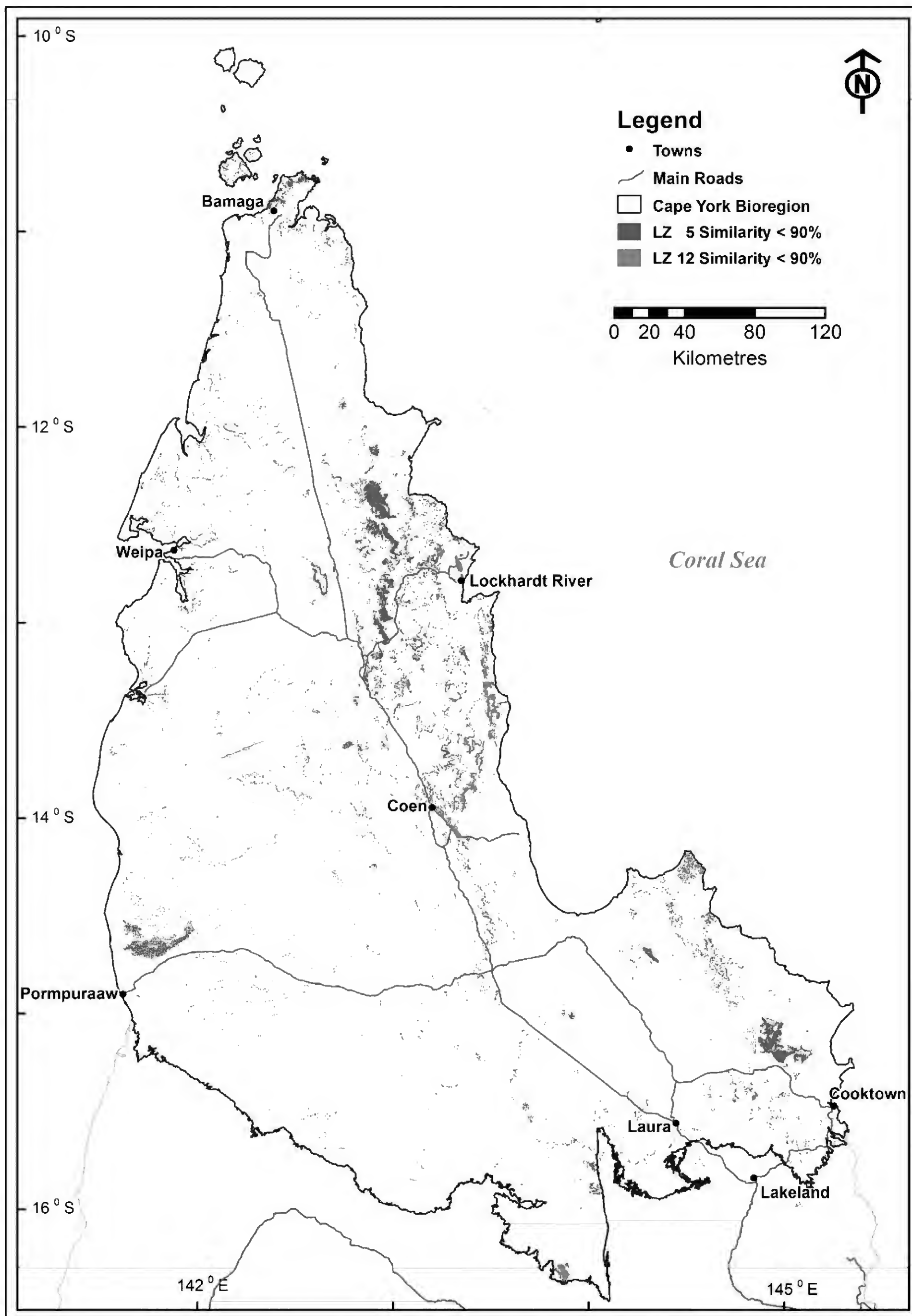


Figure 3.1d: Soil structure

Appendix 4: Additional analysis requested by the technical review committee and recommendations.

The expert panel queried two communities identified by the numerical analysis, requesting further analysis. These were the *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands and the *Eucalyptus tetradonta*, *Corymbia stockeri* woodlands, both distributed across the extent of land zone 5. The final recommendations are discussed below.

Methods

We carried out the initial investigations with the *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands, testing for differences in three attributes; canopy heights of the tallest layer, and floristic differences in the woody and ground layer vegetation (separately). We tested each attribute for differences between landform (Tertiary remnant plateaus and sand plains), soil colour (red, yellow, brown) and soil texture (sand and earth) as recorded on site pro-formas. We used the ANOSIM routine (Clarke and Gorley 2006) which has two outputs; an R statistic and a significance value. The R statistic generally lies between 0 (there is no difference between the groups) and 1 (there is no similarity between the groups) but negative values indicate the within group variation is larger than the between group variation. In the ground layer we firstly looked for distinct species assemblages using *n*MDS and visually assessed whether these were coincident with different landform, soil colours or soil texture using GIS overlay. To test for differences in canopy height we also used an unpaired t-test as well as the ANOSIM routine. Due to the results of these investigations in the *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands, analysis requested by review panel for the *Eucalyptus tetradonta*, *Corymbia stockeri* woodlands was limited to differences in canopy height between landform (again Tertiary remnant plateaus and sand plains) and soil colour (red earths versus all other colours).

Of the 50 sites in the *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands, 32 contained data useful for ground layer analysis and 49 for soil analysis. There were 3 additional sites not included in the original dataset which contained enough information for testing canopy heights. This resulted in 53 sites in the canopy height analysis. There were 31 sites in the *Eucalyptus tetradonta*, *Corymbia stockeri* woodlands.

Results

Eucalyptus tetradonta, *Corymbia nesophila* woodlands

Floristic differences in woody vegetation layers.

There was no floristic difference between soil textures ($R = -0.05$, $p = 0.75$), soil colours ($R = 0.08$, $p = 0.14$) or landform ($R = 0.01$, $p = 0.44$). The negative R value for soil texture indicates that the floristic differences individually on the sandy soils and on the earth soils is greater than the floristic differences between these two soil types.

Floristic differences in ground layer vegetation.

The two-dimensional *n*MDS ordination showed two ground layer species assemblages, one dominated by *Heteropogon triticeus* and the other by *Schizachyrium* species (figure 4.1), but with a lot of variability as evidenced by the high stress level (0.2). However, these assemblages were not significantly associated with either different soil textures ($R = 0.02$, $p = 0.40$), different soil colours ($R = -0.08$, $p = 0.71$) or different landforms ($R = 0.04$, $p = 0.33$). This was also supported by the GIS overlay where there was no clear alignment of these assemblages with different soils or landforms.

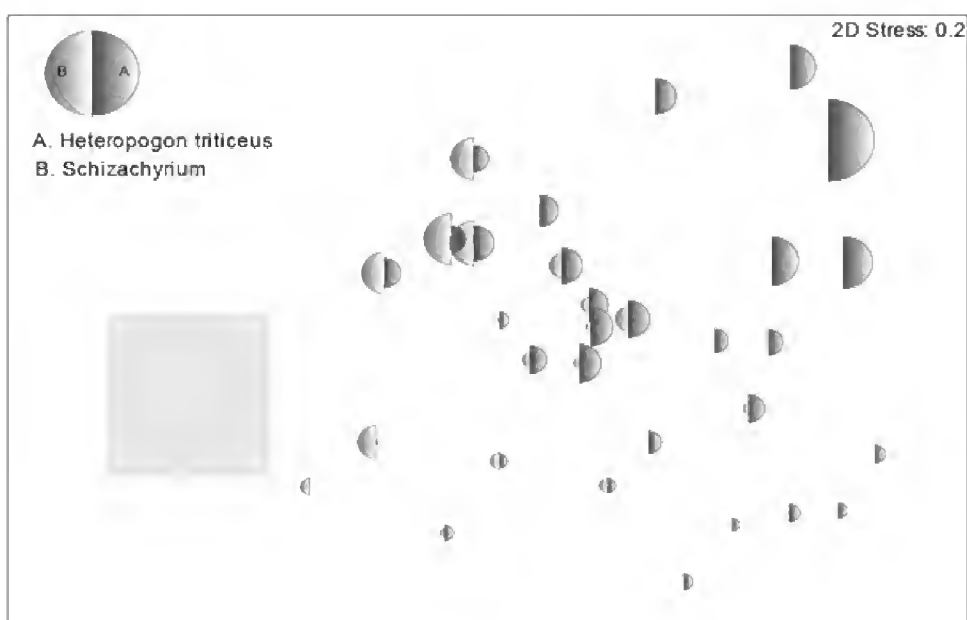


Figure 4.1: Bubble plot showing two species assemblages in the ground layer of the *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands – one dominated by *Schizachyrium* spp, the other by *Heteropogon triticeus*. Abundances are standardised.

Canopy height differences

There was no difference in the canopy heights on different soil textures ($t(47) = 1.1, p = 0.28$) and the ANOSIM results indicated the variability of heights within individual soil textures was greater than between the soil textures ($R = -0.04$). Differences in canopy height on different coloured soils was not straight forward. There was a distinct, but not significant difference between the heights of trees on red earths versus brown earths ($R = 0.86, p = 0.06$), and an indistinct, but significant difference between the heights of trees on red earths versus yellow earths ($R = 0.18, p = 0.03$). The differences in canopy heights between landforms, however, was highly significant ($t(51) = 5.7, p < 0.0001$), with the average height of trees on the Tertiary remnant plateaus being 5.2m taller than those on sand plains. We confirmed these results by running two different ANOSIM analysis. Firstly, we included all sites; 13 on the plateaus and 40 on the plains. These results showed a significant difference ($p = 0.01$), but a large overlap in height ($R = 0.25$). We then ran ANOSIM with an equal number of sites (13) in both landforms (sites from the plains were chosen randomly). The difference in height was again significant ($p = 0.1$) however there was a small overlap in height ($R = 0.84$).

Eucalyptus tetradonta, *Corymbia stockeri* woodlands

There was a significant difference in the canopy heights of trees on both different landforms and different soil colours. The average height difference between trees on Tertiary remnant plateaus and on sand plains was 7.5m ($t(29) = 7.0, p < 0.0001$) and on red earths versus all other coloured soil was 7.2m ($t(29) = 6.4, p < 0.0001$). The ANOSIM results show that there is overlap in tree height on both landform ($R = 0.63$) and soil colour ($R = 0.52$).

Discussion

There are no differences in the floristics of the woody vegetation of the *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands across land zone 5. There is a difference in the floristics of the ground layer, but it is not relatable to differences in soil types or landform and it is possible that the different assemblages are due to disturbance history (Kutt and Woinarski 2007, Miller and Murphy 2017).

There were significant differences in the canopy height of both *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands and *Eucalyptus tetradonta*, *Corymbia stockeri* woodlands on different landforms and soil colour. The red earths, which are most common on the remnant plateaus, grow significantly taller woodlands than other coloured soils, which are most common on the sand plains. From this it is not surprising that the woodlands on the Tertiary remnant plateaus are significantly taller, however, as our ANOSIM results indicate there are areas on sand plains and on yellow earths where woodlands are also tall. This leads us to conclude that the height of woodlands on sand plains is variable, but woodlands on the remnant plateaus are consistently taller.

Recommendation

The classification protocols used in Queensland (Neldner et al. 2017) specify that woodlands with the same dominant species, but with a consistent height difference of 5m, can be split into separate communities. Despite having no consistent floristic differences, the *Eucalyptus tetradonta*, *Corymbia nesophila* woodlands and the *Eucalyptus tetradonta*, *Corymbia stockeri* woodlands on the Tertiary remnant plateaus are consistently ≥ 5 m taller than those on sand plains. However, there is an overlap in height between the plateaus and the sand plains. We therefore recommend the woodlands on the remnant plateaus are recognised as vegetation communities within the appropriate floristically defined regional ecosystem.

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- Kutt, A.S. and J.C.Z. Woinarski. 2007. The effects of grazing and fire on vegetation and the vertebrate assemblage in a tropical savanna woodland in north-eastern Australia. *Journal of Tropical Ecology* 23 (1): 95-106.
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Appendix 5: Floristic similarities between communities on land zone 5 and land zone 12 in Cape York Peninsula bioregion.

Plot data in each community was averaged. The dendrogram was formed using the CLUSTER routine and scatter plots using nMDS ordination in PRIMER v6 (Clarke & Gorley 2006).

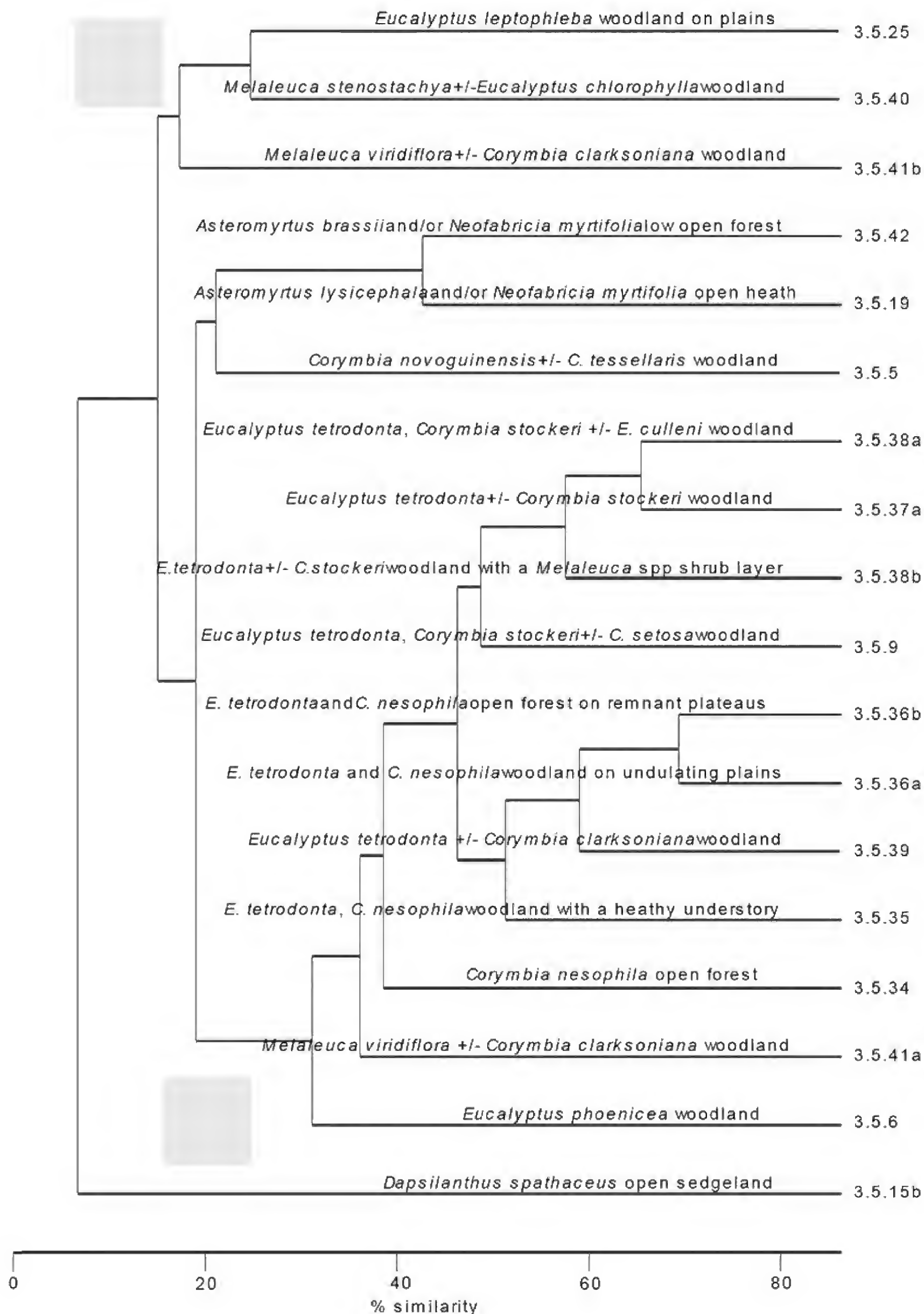


Figure 5.1: Dendrogram showing hierarchical relationships of communities on land zone 5 identified by quantitative analyses.

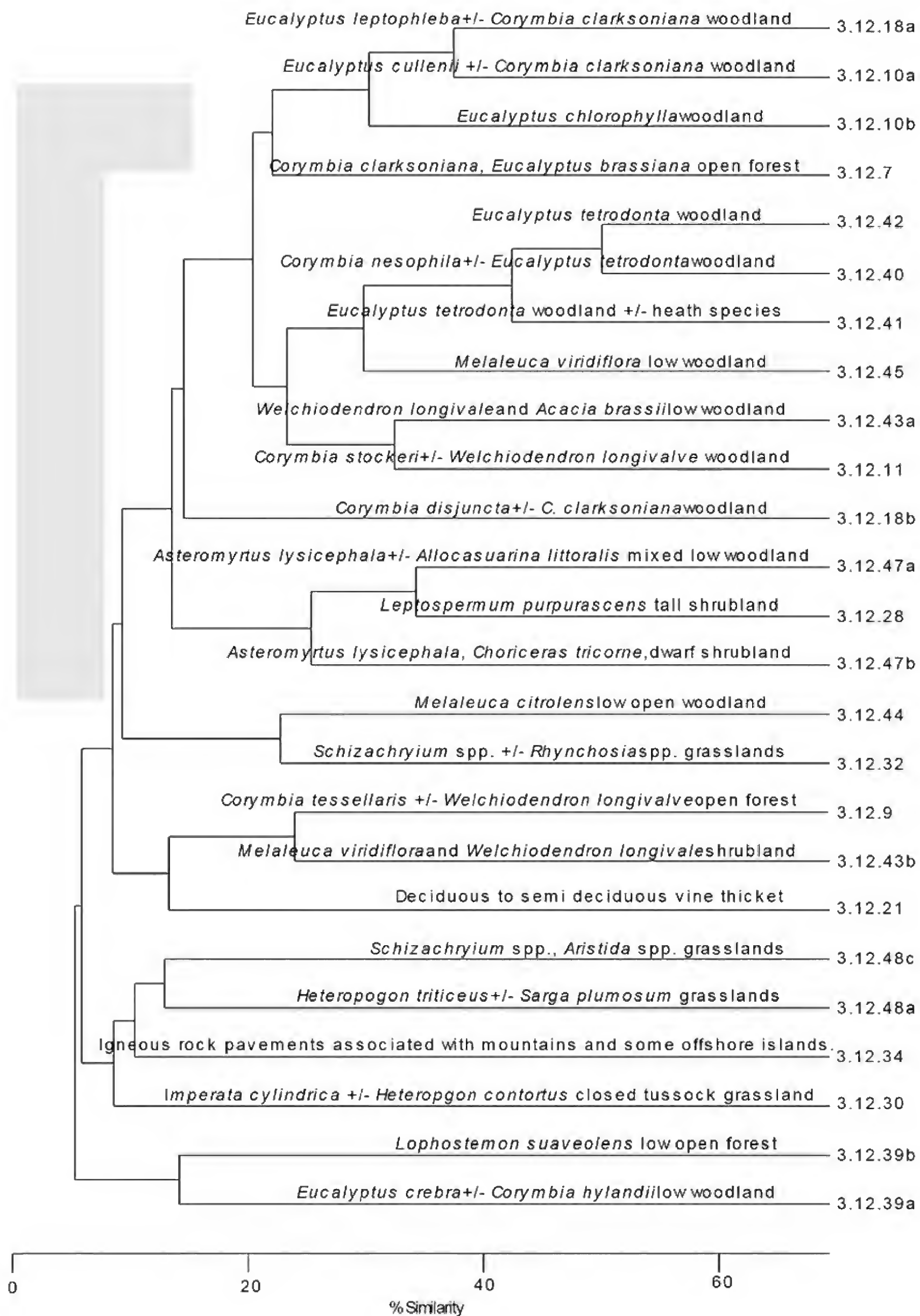


Figure 5.2: Dendrogram showing hierarchical relationships of communities identified by quantitative analyses on land zone 12.

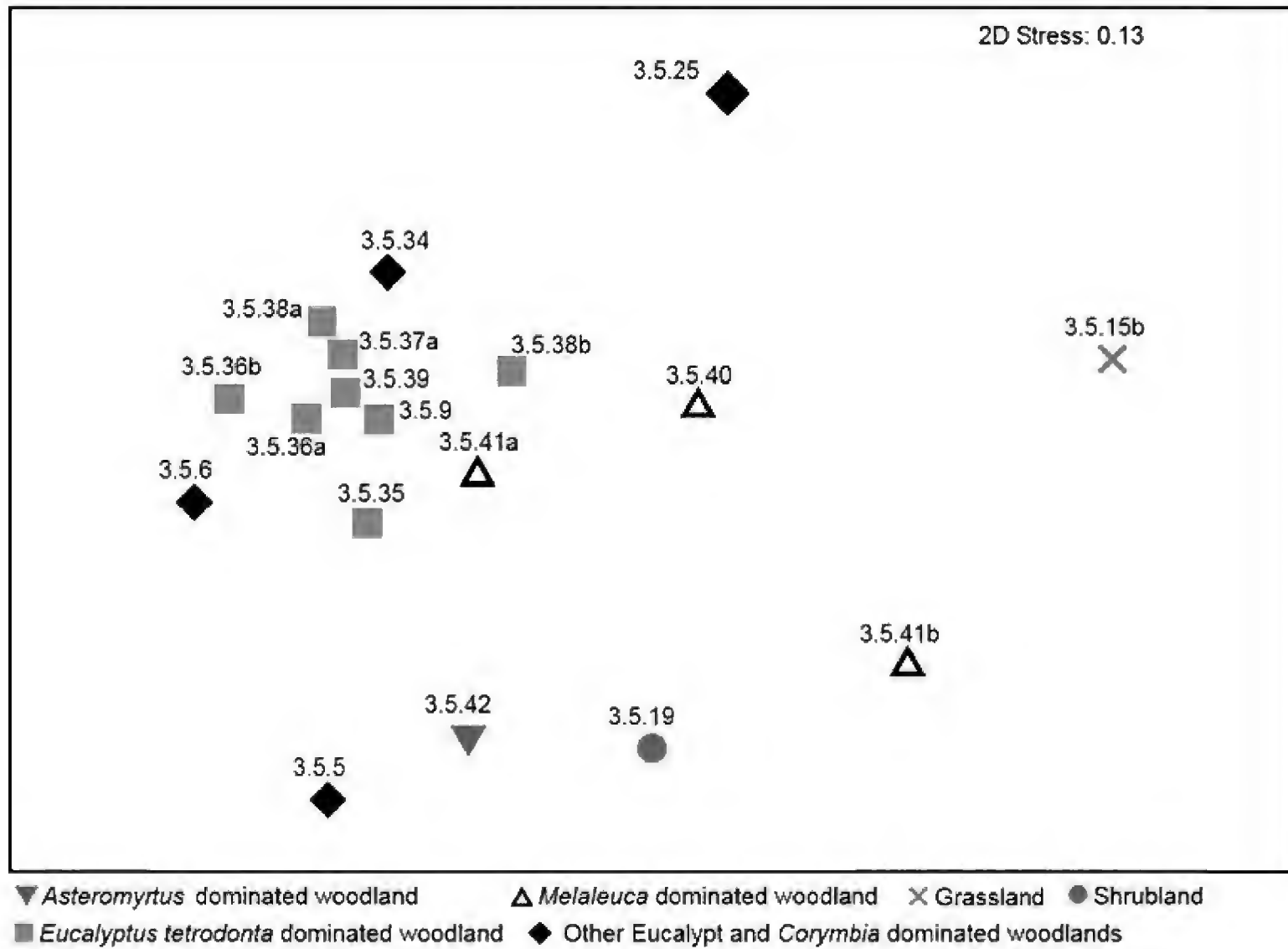


Figure 5.3: Land zone 5 scatter plot showing relative similarity of communities to each other in two-dimensions. Communities close together are more similar to each other. The greater clumping of communities than on land zone 12 scatter plot (Fig.5. 4) indicates a higher level of similarity of communities on land zone 5 than those on land zone 12.

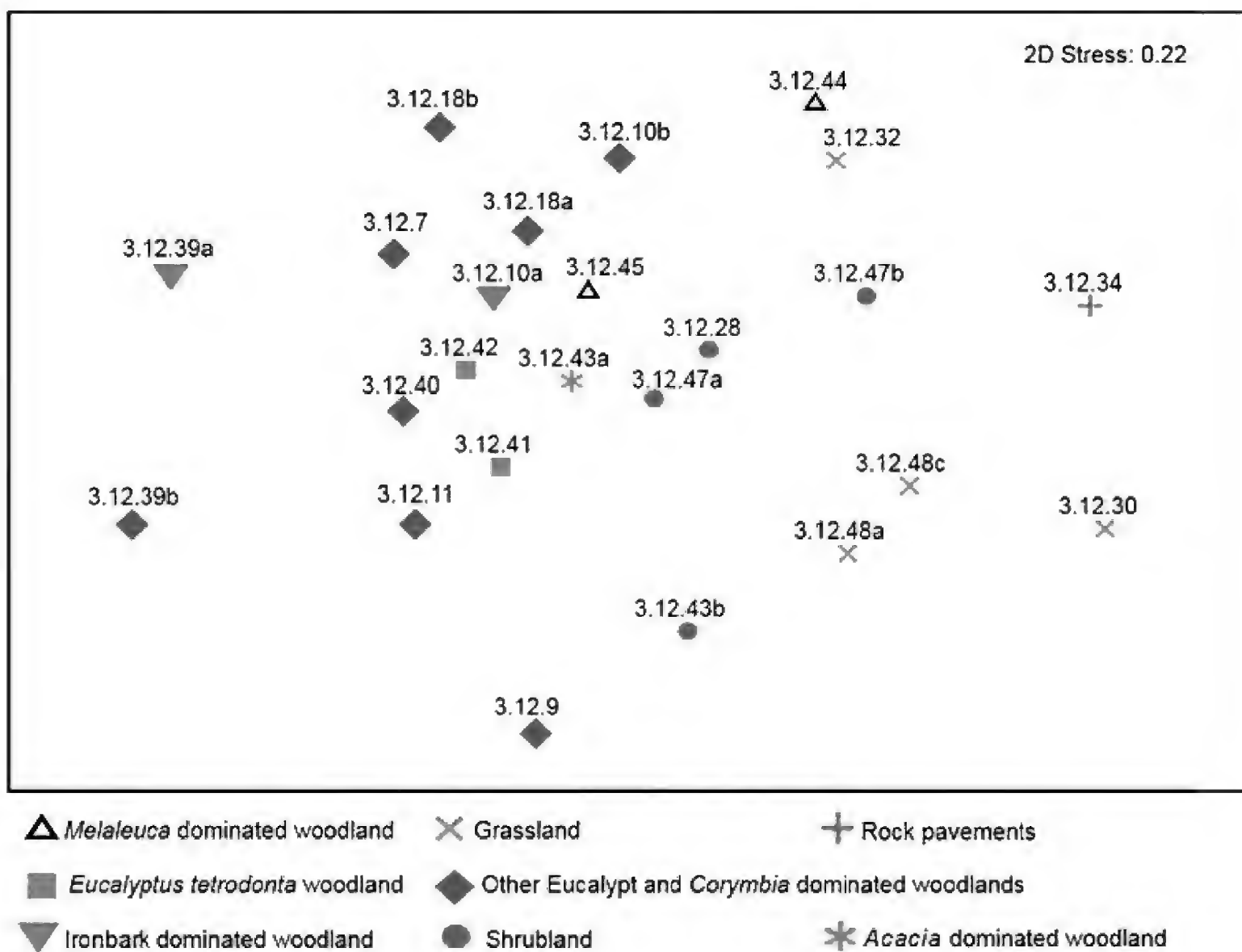


Figure 5.4: Land zone 12 scatter plot showing relative similarity of communities to each other. Communities close together are more similar to each other. The more scattered spread of communities on land zone 12 when compared to the land zone 5 (Fig. 5.3) indicates a lower level of similarity between communities than land zone 5.

Table 5.1: Percent similarity matrix of communities on land zone 5 (calculated with Bray-Curtis coefficient).

	3.5.15b	3.5.19	3.5.25	3.5.34	3.5.35	3.5.36a	3.5.36b	3.5.37a	3.5.38a	3.5.38b	3.5.39	3.5.40	3.5.41a	3.5.41b	3.5.42	3.5.5	3.5.6	3.5.9
3.5.15b																		
3.5.19	9.2																	
3.5.25	8.4	7.6																
3.5.34	6.8	13.6	16.1															
3.5.35	6.6	36.5	9.7	38.8														
3.5.36a	5.9	14.8	9.6	47.8	56.5													
3.5.36b	3.0	9.5	7.5	44.6	50.9	69.4												
3.5.37a	6.0	12.4	15.0	36.2	45.2	61.7	47.8											
3.5.38a	4.4	10.1	17.8	38.7	42.7	52.8	41.3	65.4										
3.5.38b	8.5	16.2	17.3	38.1	37.5	38.7	30.3	53.7	61.4									
3.5.39	4.9	9.5	19.3	33.7	46.5	61.0	57.0	56.0	56.3	40.7								
3.5.40	8.0	20.8	24.7	20.4	18.6	18.1	13.2	22.1	25.1	49.6	17.6							
3.5.41a	7.6	16.6	24.1	32.2	34.7	33.5	28.9	35.0	35.9	35.8	46.9	26.8						
3.5.41b	14.0	19.1	10.6	5.9	6.9	7.3	3.6	7.7	6.5	23.7	10.5	24.1	25.6					
3.5.42	5.5	42.7	4.9	19.4	41.3	24.0	18.3	20.2	15.7	20.2	20.8	18.1	28.5	13.9				
3.5.5	4.5	11.2	5.1	22.0	26.6	25.4	23.9	12.6	9.6	13.3	24.5	10.4	26.6	13.3	31.0			
3.5.6	4.7	13.2	9.1	23.0	33.7	31.5	30.0	35.2	31.2	24.5	35.2	12.3	32.0	4.1	22.3	15.1		
3.5.9	5.9	10.1	13.9	30.9	42.0	46.8	44.2	53.1	49.2	43.8	56.1	20.3	42.1	14.2	17.8	17.1	35.0	

References

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- Neldner, V.J., Niehus, R.E., Wilson, B.A., McDonald, W.J.F., Ford, A.J. & Accad, A. (2017) *Vegetation of Queensland. Descriptions of Broad Vegetation Groups*. Queensland Herbarium, Department of Science, Information Technology and Innovation.

Appendix 6. Investigation into the correlations between persistent greenness index and climate variables.

We tested for correlations between climate variables and woody vegetation density using a 4-way ANOVA in the EXCEL stats package. Woody vegetation is represented by a maximum persistent greenness index (JRSRP 2017). The line-fit plots of woody vegetation density against each climate variable (Fig 6.1 – 6.4) provide a visualisation of the strength of correlation and the low predictability for woody vegetation. While there is a significant correlation between woody density and climate, the spread of actual woody vegetation values compared to expected values portrays the low predictability of woody vegetation density by climate ($R^2 = 0.34$)

Table A6.1: 4-way ANOVA of woody vegetation density against climate variables.

<i>Regression Statistics</i>	
Multiple R	0.58
R Square	0.34
Adjusted R Square	0.34
Standard Error	13.00
Observations	1000

<i>ANOVA</i>					
	<i>df</i>	<i>SS</i>	<i>MS</i>	<i>F</i>	<i>Significance F</i>
Regression	4	86136.3	21534.1	127.3	8.13988E-88
Residual	995	168254.7	169.1		
Total	999	254391			

	<i>Coefficients</i>	<i>Standard Error</i>	<i>t Stat</i>	<i>P-value</i>	<i>Lower 95%</i>	<i>Upper 95%</i>
Intercept	203.77	26.98	7.55	9.67745E-14	150.82	256.71
Temperature seasonality (C of V%)	-0.33	0.09	-3.51	0.0005	-0.51	-0.15
Mean moisture index of lowest quarter	1.75	0.39	4.50	7.63064E-06	0.99	2.52
Annual precipitation (mm)	0.02	0.01	3.23	0.0013	0.01	0.03
Annual mean temperature (°C)	-2.76	0.88	-3.12	0.0018	-4.49	-1.02

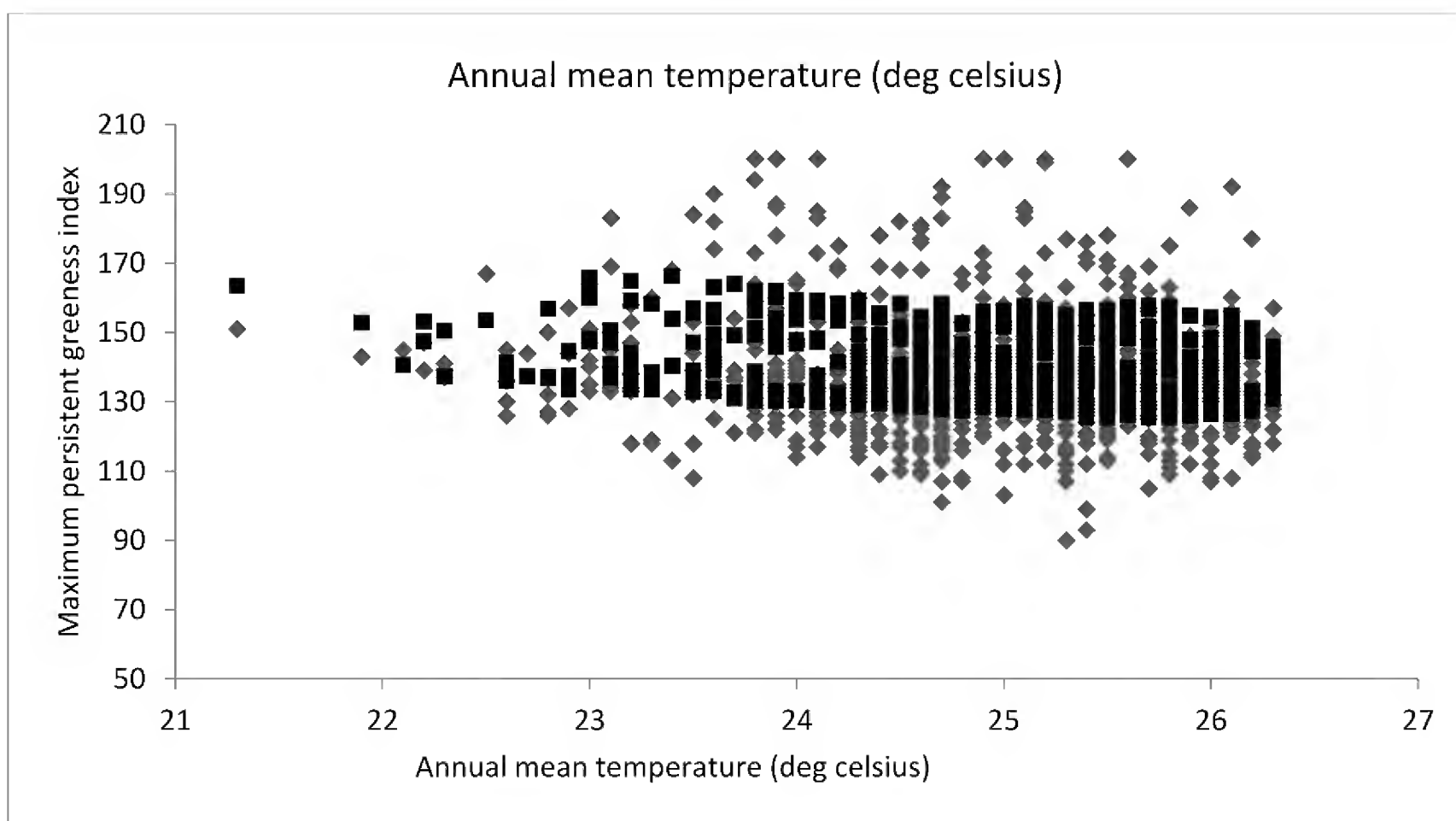


Figure A6.1: Line-fit plots of annual mean temperature against woody vegetation density (represented by maximum persistent greenness index). \blacklozenge = actual maximum persistent greenness index at each observation point, \blacksquare = predicted maximum persistent greenness index at each observation point

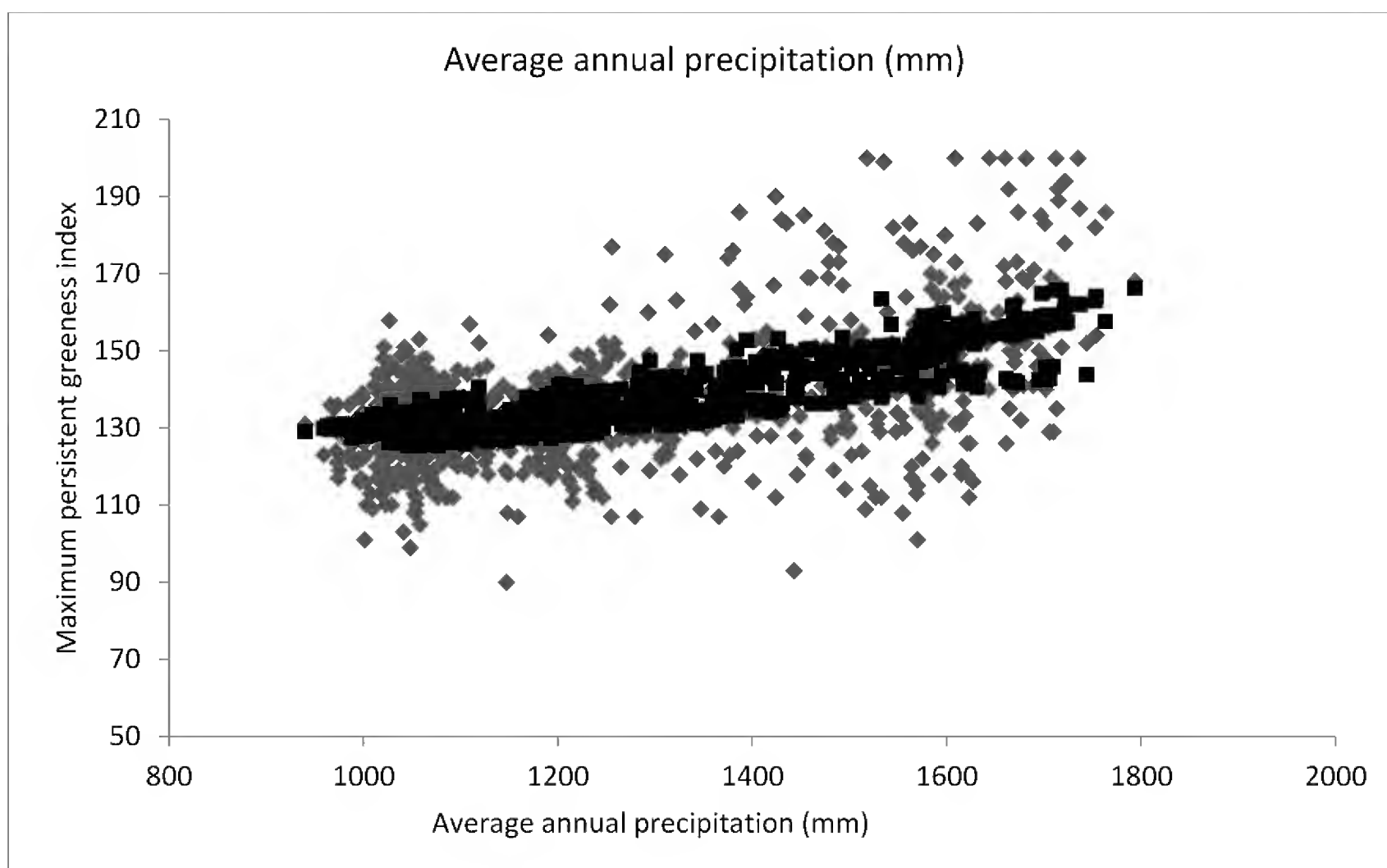


Figure A6.2: Line-fit plots of average annual precipitation against woody vegetation density (represented by maximum persistent greenness index). \blacklozenge = actual maximum persistent greenness index at each observation point, \blacksquare = predicted maximum persistent greenness index at each observation point

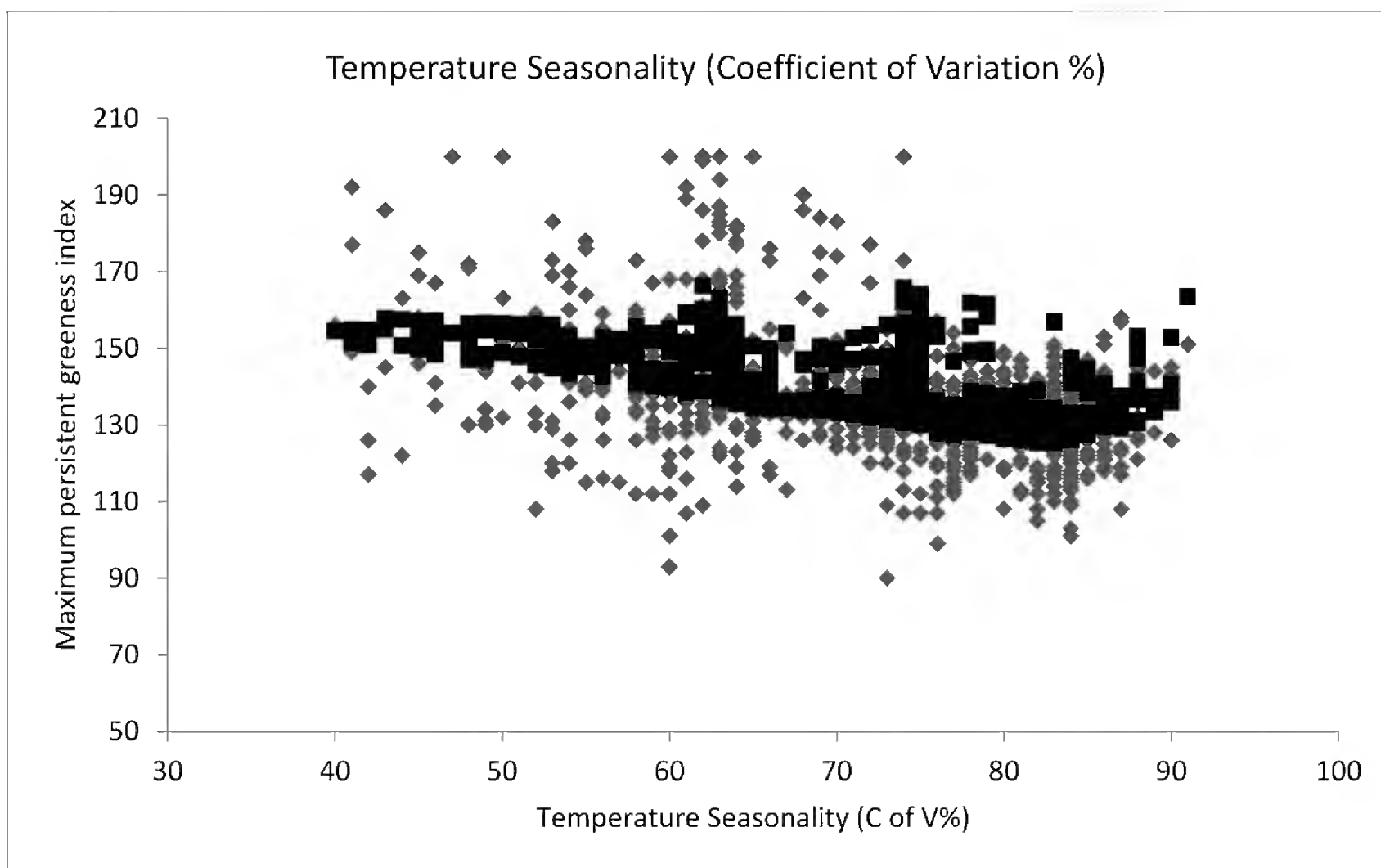


Figure A6.3: Line-fit plots of temperature seasonality against woody vegetation density (represented by maximum persistent greenness index). \blacklozenge = actual maximum persistent greenness index at each observation point, \blacksquare = predicted maximum persistent greenness index at each observation point

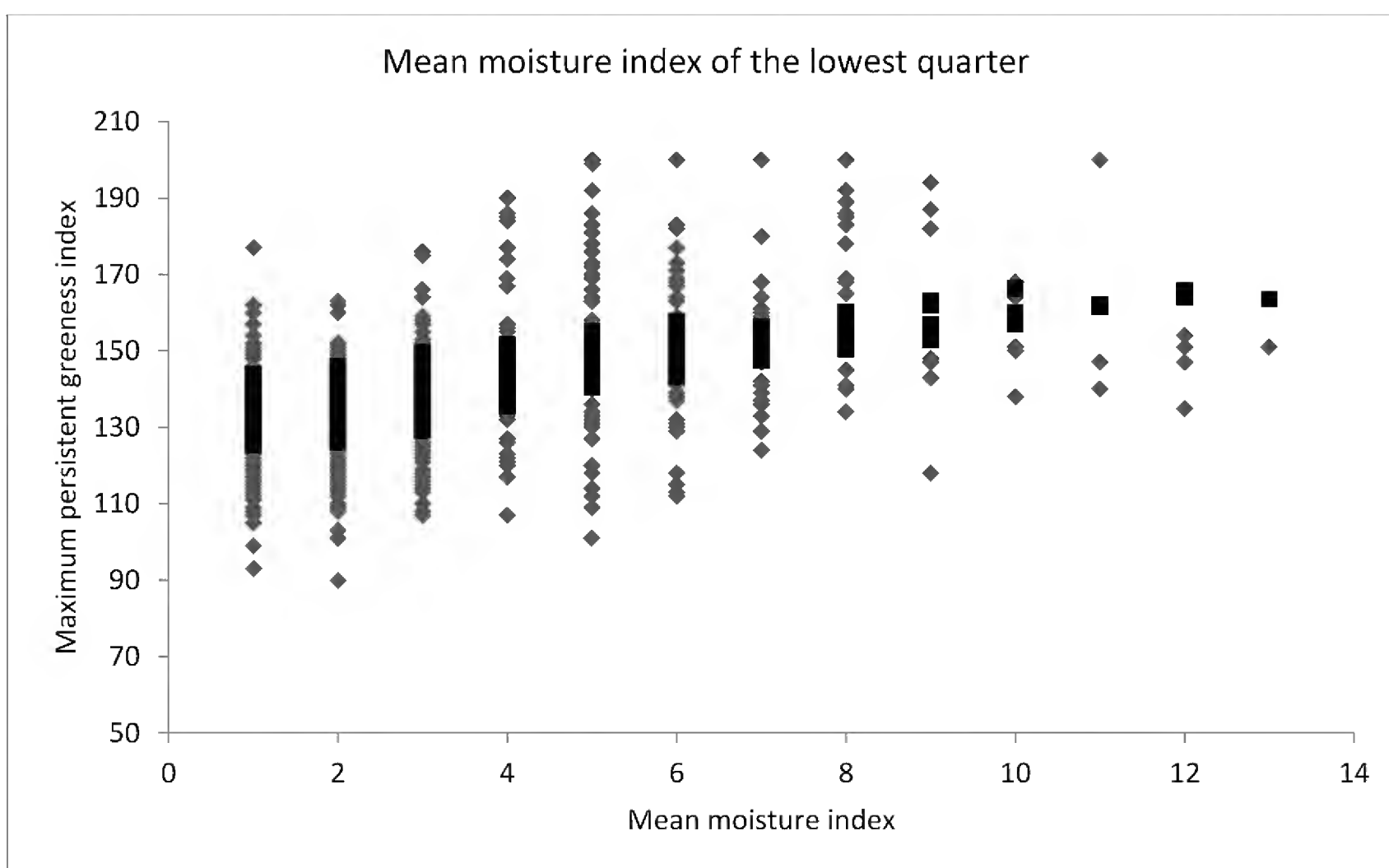


Figure A6.4: Line-fit plots of mean moisture index of the lowest quarter against woody vegetation density (represented by maximum persistent greenness index). \blacklozenge = actual maximum persistent greenness index at each observation point, \blacksquare = predicted maximum persistent greenness index at each observation point

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***Angophora subvelutina* (Myrtaceae) on atypical diatrema habitat at Glenbrook : an addition to the eucalypt list for the Greater Blue Mountains World Heritage Area**

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Abstract: The Greater Blue Mountains World Heritage Area (GBMWA), a natural area of about one million hectares immediately west of Sydney, Australia, is significant for its biodiversity, and particularly for its richness of eucalypt species (species of *Eucalyptus*, *Angophora* and *Corymbia* in the family Myrtaceae), numbered at 96 species in 2010. This paper describes the finding of a previously unlisted *Angophora* species in the GBMWA, and makes a conservation assessment of the population. A population of the Broad-leaved Apple *Angophora subvelutina* F. Muell. occurs at Euroka Clearing south of Glenbrook just within the eastern edge of Blue Mountains National Park, one of the eight conservation reserves that make up the GBMWA. The population numbers over 200 plants and there is evidence that the species has been present at the site since before European settlement. The population includes a mixture of age classes and is considered viable, although substantial intergradation is occurring with the closely related species *Angophora floribunda*. Elsewhere in the Sydney area, the species is relatively uncommon and has been extensively cleared from its relatively fertile habitats. The population in the GBMWA noted here has conservation significance for its size and long history at the site, and for the unusual ecological conditions of the Euroka diatrema, which is an atypical habitat for the species.

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Introduction

The Greater Blue Mountains was inscribed on the World Heritage List in 2000 for its outstanding natural values, a major component of which is the high number of eucalypt species and eucalypt-dominated communities (the term ‘eucalypt’ refers to the closely related genera *Eucalyptus*, *Angophora* and *Corymbia* of the family Myrtaceae). In 2000, 91 eucalypt species were known from the Greater Blue Mountains World Heritage Area (GBMWA). A subsequent assessment of the eucalypts in the eight conservation reserves which make up the GBMWA (Blue Mountains, Gardens of Stone, Kanangra-Boyd, Nattai, Thirlmere Lakes, Wollemi and Yengo National Parks and Jenolan Karst Conservation Reserve) listed 96 eucalypt species (Hager & Benson 2010).

The number of eucalypt species recognised in the area is likely to fluctuate given the somewhat equivocal nature of systematic description and changes in the state of scientific knowledge, particularly with the increased application of genetic research. For example, Rutherford et al. (2018), in a genetic study, suggested that at least one of the eight green-leaved ashes (*Eucalyptus cunninghamii*) in the GBMWA (Hager & Benson 2010) showed distinct genetic variation between populations warranting recognition of a new undescribed species, while two other species (*Eucalyptus laophila* and *Eucalyptus stricta*) could not be distinguished from each other.

A different situation applies to the finding of a population of a well-accepted existing species, not previously formally recorded in the GBMWA. Such species may be found in areas that have remained inadequately explored botanically or within lands that are subsequently added to the GBMWA. The species may also be very rare in the GBMWA, with only one or two obscure previous records that have gone unrecognised.

In 2017 Peter Smith noticed that *Angophora subvelutina* F.Muell. (Broad-leaved Apple) was not included in the list of GBMWA eucalypts in the foyer of the World Heritage Exhibition at the Blue Mountains Cultural Centre, Katoomba. The species is also not included in the GBMWA eucalypt list of Hager & Benson (2010). Peter and Judy Smith recalled that *Angophora subvelutina* occurred at Euroka Clearing, Glenbrook, just inside the eastern boundary of Blue Mountains National Park and hence the GBMWA.

Here, we confirm the presence of a previously unlisted (Benson & Hager 2010) eucalypt species (*Angophora subvelutina*) in the GBMWA. We describe and assess the *Angophora subvelutina* population at Euroka Clearing, Glenbrook, and consider the long term viability of this population.

Distribution of *Angophora subvelutina*

Angophora subvelutina was first described by Ferdinand Mueller in 1858. It is a tree that typically grows to about 20 m high with persistent, grey, fibrous-flaky bark and adult leaves which are relatively broad, more or less sessile, and cordate at the base.

Angophora subvelutina usually grows on deep alluvial soils and may be locally abundant. It occurs at scattered locations, mainly east of the Great Dividing Range, from south-eastern Queensland south to the Bega district in southern NSW. In NSW it has been recorded in the North Coast (NC), Northern Tablelands (NT), North Western Slopes (NWS), Central Coast (CC), Central Tablelands (CT), Central Western Slopes (CWS) and South Coast (SC) botanical subdivisions, south to the Araluen district (PlantNET NSW Flora Online 2018). Intergrades with the closely related species *Angophora floribunda* are known from the NWS, CC, CWS and SC, and also occur beyond the known distribution of *Angophora subvelutina* in the Bega district of the far SC, and in the North Western Plains (NWP) (PlantNET NSW Flora Online 2018).

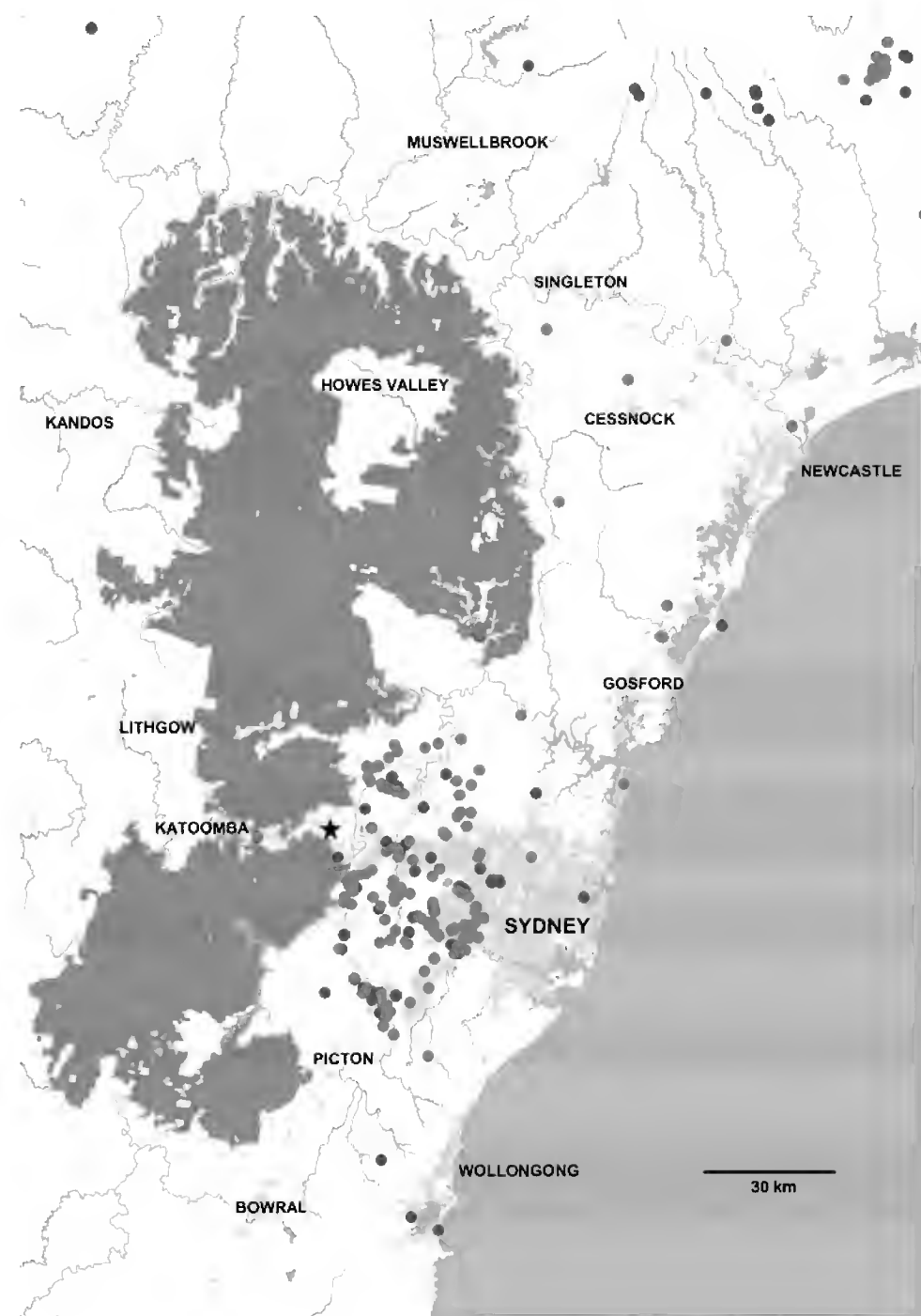


Fig. 1: Map of *Angophora subvelutina* records in the vicinity of the Greater Blue Mountains World Heritage Area. Records from Australasian Virtual Herbarium and NSW BioNet databases, extracted 27 July 2018. Red circles, records from 2000 or later; blue circles, records before 2000; purple star, Euroka Clearing; black star, Sun Valley. Records with inexact locations have not been mapped.

In the Sydney area, specimen records from the Australasian Virtual Herbarium (2018) and sightings records from the NSW BioNet Atlas (2018) indicate that *Angophora subvelutina* is mainly associated with river and creek systems on the Cumberland Plain (Fig. 1). It is found in floodplain forest and on creek banks on deep fertile alluvial soils, but may also

be associated with shale-derived soils with medium to high nutrient levels. In floodplain forest, associated tree species may include *Eucalyptus baueriana*, *Eucalyptus tereticornis*, *Eucalyptus amplifolia* and *Eucalyptus botryoides x saligna* (Benson & McDougall 1998). Most of its original habitat on the floodplains has now been cleared or severely degraded; James et al. (1999) regarded its regional conservation status in western Sydney as Vulnerable. *Angophora subvelutina* is a characteristic species of 'River-Flat Eucalypt Forest on Coastal Floodplains of the NSW North Coast, Sydney Basin and South East Corner Bioregions', which is listed as an Endangered Ecological Community under the NSW *Biodiversity Conservation Act 2016* (NSW Scientific Committee 2011).

Study Area

Our study was undertaken at Euroka Clearing in Blue Mountains National Park, 2.5 km south of the township of Glenbrook. Euroka Clearing is a large semi-cleared area in a circular valley on a volcanic diatreme. Euroka Creek and its tributaries traverse the valley and drain to the Nepean River, some 800 m to the east. It is a popular camping area and is managed as such by NSW National Parks and Wildlife Service. Tree species in the remnant forest at Euroka Clearing include *Allocasuarina torulosa*, *Casuarina cunninghamiana*, *Angophora bakeri*, *Angophora floribunda*, *Angophora subvelutina*, *Corymbia eximia*, *Eucalyptus agglomerata*, *Eucalyptus beyeriana*, *Eucalyptus deanei*, *Eucalyptus eugenioides*, *Eucalyptus fibrosa*, *Eucalyptus punctata*, *Eucalyptus saligna*, *Eucalyptus tereticornis* and *Syncarpia glomulifera*. The vegetation on this diatreme is notable for the diversity of tree species and for its Cumberland Plain influences.

Methods

On 27 June 2017, Peter and Judy Smith collected a specimen (including adult leaves and fruiting capsules) of a tree at Euroka Clearing that they had identified as *Angophora subvelutina* (Broad-leaved Apple). The specimen tree (Fig. 2) was growing on the cleared slope on the eastern side of Euroka Creek, upstream of where the creek makes a right-angle bend. Its height was measured with a clinometer as 15 m on 8 February 2018 and it appeared healthy. The GPS coordinates of the tree were -33.799102, 150.618715 (GDA94 datum). The elevation of the tree was about 85 m asl (determined from a 10 m GIS contour layer). The specimen was given to the National Herbarium of NSW, Royal Botanic Gardens Sydney for identification.

On 8 February and 28 March 2018, Peter and Judy Smith carried out a series of field observations to provide an estimate of the size and age class structure of the *Angophora subvelutina* population at Euroka Clearing. It was soon noted that the population of *Angophora subvelutina* was intermixed with a population of *Angophora floribunda* and that a number of plants were intergrades between the two species. A random

sample of 100 rough-barked *Angophora* plants was selected and each was identified to species (*Angophora subvelutina*, *Angophora floribunda* or intergrade) and classified as a tall tree, low tree (mature tree less than two-thirds the height of the tallest trees) or sapling. Intergrades were identified on leaf characters: leaves that were intermediate between the cordate, virtually sessile leaves of *Angophora subvelutina* and the cuneate, petiolate leaves of *Angophora floribunda*, or else leaves that were a mixture of the two leaf types. A rough estimate was made of the total number of rough-barked *Angophora* plants at Euroka Clearing.

On 27 July 2018, Peter Smith collated and mapped previous *Angophora subvelutina* records in and around the Greater Blue Mountains World Heritage Area. Records were obtained from the specimen database of the Australasian Virtual Herbarium (2018) and the sightings database of the NSW Bionet Atlas (2018).



Fig. 2: *Angophora subvelutina* tree (closest to camera) at Euroka Clearing, Blue Mountains National Park, from which a specimen has been lodged at the National Herbarium of NSW.

Results

The National Herbarium of NSW confirmed that the specimen collected at Euroka Clearing in June 2017 was indeed *Angophora subvelutina*.

The database searches revealed only a single previous record that definitely came from within the GBMWH: a specimen at the National Herbarium of NSW that was collected at Euroka Clearing by T.M. Whaite in 1952. This specimen was from a "tree 20 ft [6 m], bark stiffly fibrous on breccia by creek". There is also a specimen at the N.C.W. Beadle Herbarium, University of New England, collected at "Glenbrook" by T.J. Hawkeswood in 1975, which may have come from Euroka Clearing.

Another record of interest is a 2005 sighting in the Bionet Atlas with coordinates (no description of the location but coordinate accuracy reported as within 30 m) that place it in the Putty Road corridor where it passes through Yengo National Park just north of the Putty Valley Road turnoff, near the head of Snakes Valley Creek. This record warrants further investigation but, if correct, then *Angophora subvelutina* likely occurs within the GBMWhA at this location.

There is also a cluster of four records (one undated specimen and three 2006 BioNet sightings) along the Macdonald River near St Albans, close to the boundary of Yengo National Park. However, the *Angophora subvelutina* population along the river may be restricted to the floodplain outside the GBMWhA.

Another sighting of *Angophora subvelutina* in the BioNet Atlas is from the railway corridor near Bullaburra station in 2011, close to Blue Mountains National Park. This is an unlikely location for the species and, if correct, is probably not a natural occurrence. There are also duplicate *Angophora subvelutina* specimens at the National Herbarium of NSW and the National Herbarium of Victoria that were collected at “Mt Victoria” by J.H. Maiden in 1901. This is another unlikely location for the species but is probably either an error or refers to the general Mt Victoria district rather than the immediate vicinity of the township itself (which is surrounded by Blue Mountains National Park). The specimen was possibly collected in the nearby Kanimbla Valley or Hartley Valley, outside the GBMWhA.

From our fieldwork in February and March 2018, we estimated that the combined population of *Angophora subvelutina*, *Angophora floribunda* and intergrades at Euroka Clearing numbered over 500 individuals, with most individuals found along or near Euroka Creek and its tributaries. The three entities were intermixed and we could see no obvious habitat differences between them.

Based on our sample of 100 plants, the rough-barked *Angophora* composition at Euroka Clearing consisted of 43% *Angophora subvelutina*, 32% *Angophora floribunda* and 25% intergrades (Table 1, Fig. 3). On this basis, we estimate the size of the *Angophora subvelutina* population at Euroka Clearing at over 215 plants. The *Angophora subvelutina* population was of mixed sizes and 19% of the population consisted of saplings of various ages. The population also included some very large old trees, and one notable example had a height of 37 m and a trunk diameter of 1.65 m (Fig. 4). There was pronounced intergradation between the *Angophora subvelutina* and *Angophora floribunda* populations, with about three intergrade plants to every five *Angophora subvelutina* plants.

Table 1: Rough-barked *Angophora* composition at Euroka Clearing in February-March 2018, based on a sample of 100 plants.

Species	Tall trees	Low trees	Saplings	Total
<i>Angophora subvelutina</i>	21	14	8	43
<i>Angophora floribunda</i>	17	13	2	32
Intergrades	13	5	7	25
Total	51	32	17	100

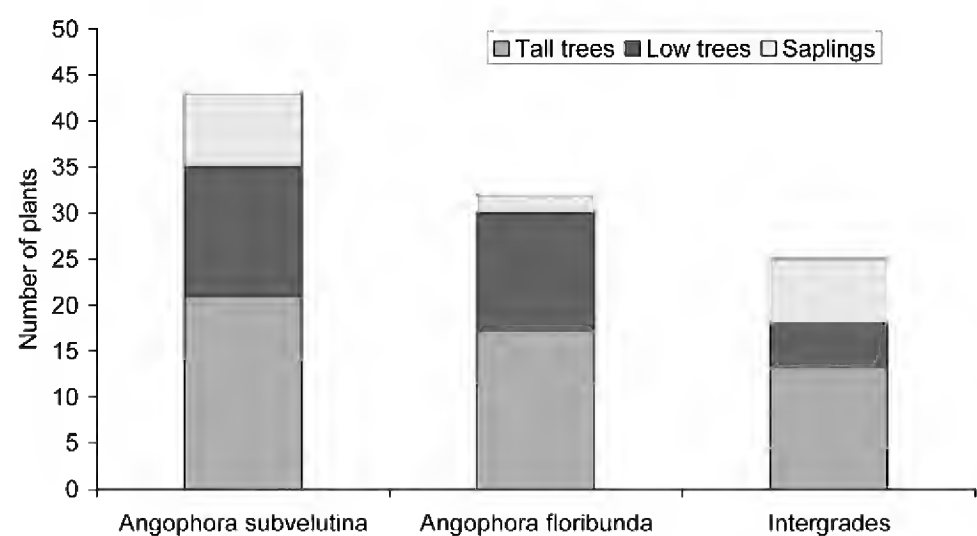


Fig. 3: Rough-barked *Angophora* composition at Euroka Clearing in February-March 2018, based on a sample of 100 plants.



Fig. 4: Large old *Angophora subvelutina* tree at Apple Tree Flat, Euroka Clearing, Blue Mountains National Park. The tree is about 37 m tall, with a diameter at breast height of about 1.65 m. It likely pre-dates European settlement.

Discussion

We have confirmed the presence of a population of *Angophora subvelutina* at Euroka Clearing, Glenbrook in Blue Mountains National Park and thus the Greater Blue Mountains World Heritage Area. The *Angophora subvelutina* population at Euroka Clearing numbers over 200 plants and has clearly been there for a long time. A specimen was collected from the site in 1952 and the population includes a number of trees which are likely to pre-date European settlement, such as the tree in Fig. 4. The long history of the species at the site, the size of the population in February and March 2018, and the range of age classes present, indicate that it is a viable population, despite substantial local hybridisation with another species at the site, *Angophora floribunda*.

This species should now be added to the formal list of eucalypts in the GBMWA. *Angophora subvelutina* is a well known and accepted species. Its late addition to the GBMWA eucalypt list highlights the fact that the GBMWA, an area that is internationally renowned for its biodiversity, remains inadequately explored and documented botanically. Far too few botanists have been afforded the opportunity to work in this area.

The diatrema at Euroka Clearing is an unusual ecological site for *Angophora subvelutina*. It has a creek system but is not a floodplain. The Nepean River is nearby but runs through a gorge and has only a narrow floodplain in this vicinity. The slopes around the diatrema are on sandstone geology and there is a remnant shale cap on the surrounding ridge. The high nutrient soils of the diatrema, together with the influence of the surrounding sandstone and shale geology, appear to provide suitable conditions for both *Angophora subvelutina* and *Angophora floribunda*, as well as for an unusually large number of other eucalypt species that occur at the site. We have also observed *Angophora subvelutina* and *Angophora floribunda* growing together at Campbells Ford beside the Nepean River in Gulguer Nature Reserve east of the GBMWA.

Angophora subvelutina also occurs on another large diatrema at Sun Valley, about 10 km north of Euroka Clearing, outside the GBMWA (Fig. 1). Similarly to Euroka Clearing, both *Angophora subvelutina* and *Angophora floribunda* are present at Sun Valley, with evidence of intergradation between them (Andrew Orme, pers. comm.). The Sun Valley diatrema is a semi-rural area with many houses. The remnant forest on this diatrema, which is dominated by *Eucalyptus amplifolia* (Cabbage Gum), is listed as an Endangered Ecological Community, 'Sun Valley Cabbage Gum Forest in the Sydney Basin Bioregion', under the NSW *Biodiversity Conservation Act 2016* (NSW Scientific Committee 2001).

There is another, smaller diatrema, Machins Crater, about 5 km south-west of Euroka Clearing, within Blue Mountains National Park. Judy and Peter Smith inspected this diatrema on 12 September 2018. We found a single *Angophora floribunda* tree but no *Angophora subvelutina*. The diatrema supports relatively undisturbed forest dominated by *Eucalyptus deanei* (Mountain Blue Gum).

We conclude that the population of *Angophora subvelutina* at Euroka Clearing makes a valuable contribution to the biodiversity of the GBMWA, as well as to conservation of the species in the general Sydney area. This is the only currently known population in the GBMWA. Local populations in the Sydney area outside the GBMWA are considered vulnerable as most floodplain forests have been cleared or severely degraded (Benson & McDougall 1998). We recommend that the Euroka Clearing population be monitored and managed to ensure its long-term viability. Although located within a national park that forms part of a World Heritage Area, the site where the population occurs is managed as a camping ground and day-use area, which may conflict with conservation management of *Angophora subvelutina*. Potential future threats to the population include climatic changes, inappropriate fire regime, lack of adequate regeneration, vegetation clearing, diseases such as the recently introduced Myrtle Rust, and genetic swamping through hybridisation with *Angophora floribunda*.

Acknowledgements

We gratefully acknowledge the assistance that we obtained during this study from Kristina McColl and Andrew Orme of the National Herbarium of NSW, Jacqueline Reid of NSW Office of Environment and Heritage, and local botanist Margaret Baker. Dr Peter Wilson of the National Herbarium of NSW confirmed the identification of the *Angophora subvelutina* specimen. Roger Lembit reviewed the manuscript and gave us helpful feedback. The Greater Blue Mountains World Heritage Advisory Committee was supportive of the study.

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Fate of a rare flowering event in an endangered population of *Acacia pendula* (Weeping Myall) from the Hunter Valley, New South Wales

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Abstract: A rare flowering event in a stand of *Acacia pendula* (Weeping Myall) (family Fabaceae, Mimosoideae) from the Hunter Valley of New South Wales is documented. This species flowers poorly in the region and (with the exception of horticultural specimens) has not been observed to fruit and develop viable seed for over a decade. One stand of this threatened Hunter Valley population of *Acacia pendula* was monitored over a seven month period (January to July 2018) to investigate this poor reproductive output. Despite copious bud production in January and February, the extent and condition of these, and all subsequent flowers rapidly declined, and none progressed to fruit.

Primary reasons for reproductive failure were postulated to be a combination of mass desiccation of capitula following extended dry conditions, infestation by native flower- and phyllode-galling midges and thrips (*Asphondylia* sp., *Dasineura glomerata*, *Kladothrips rugosus*), fungal galls (*Uromycladium* sp.), caterpillars (*Ochrogaster lunifer*), and mistletoe (*Amyema quandang*). Collectively, these stressors appear to be eliminating seed production from the study population; survival is maintained only by the copious root-suckering observed around most plants, particularly after the pressure from stock grazing (cattle, sheep) has been removed. The age of trees studied, based on measures of girth and comparison with growth rates reported for other semi-arid *Acacia*, was inferred to be between 50 and 150 years. The level of *Amyema quandang* (mistletoe) infestation on *Acacia* trees was independent of tree size, and there was no evidence to suggest that mistletoe density alone influenced flowering progress.

Consequences of these observations on future management of *Acacia pendula* in the Hunter Valley are briefly discussed.

Key words: *Acacia pendula*, Hunter Valley, endangered population, flowering fate, health

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Background

The tree *Acacia pendula* Cunn. ex Don (Weeping Myall) (family Fabaceae, Mimosoideae) in the New South Wales Hunter Valley region is a threatened species protected under three elements of legislation. Nationally, the *Environment Protection and Biodiversity Conservation Act 1999* includes *Acacia pendula* as a key species in the Critically Endangered *Hunter Valley Weeping Myall (Acacia pendula) Woodland*. Under the *NSW Biodiversity Conservation Act 2016* (BC Act), an Endangered Population of *Acacia pendula* is listed for the Hunter Valley, and the species also forms a key component of the Critically Endangered *Hunter Valley Weeping Myall Woodland in the Sydney Basin Bioregion*. For the highly modified Hunter region, Weeping Myall and its habitat is one of the most protected plant entities and subsequently presents a range of conservation management challenges wherever it occurs.

Recent debate in the literature over whether or not *Acacia pendula* populations in the Hunter Valley were present prior to European settlement is difficult to fully resolve without detailed cross-population genetic studies (Bell & Driscoll 2014, 2016; Tozer & Chalmers 2015, 2016). As a consequence, the NSW Threatened Species Scientific Committee resolved to retain *Acacia pendula* within its lists of threatened entities under the *Biodiversity Conservation Act 2016* until such clarifying evidence emerges. That being the case, continuing research into the ecology of Hunter Valley populations of *Acacia pendula* is desirable to better understand the taxon and to inform its management, so that government policies and conservation actions can be effectively implemented.

An opportunity arose to study one stand of the Hunter Valley population of *Acacia pendula* in detail following an observation of flower buds on plants in early 2018, a stage that few stands of the species in the region experience (Bell et al 2007). Importantly, successful fruiting and development of seed has never been observed in *Acacia pendula* in the Hunter Valley (critical for conclusive identification), promoting the hypothesis that plants here may be neotonous (retaining juvenile features in the adult phase) or may have lost the capacity for seed production (dispersing instead through vegetative suckers) in response to unfavourable habitat (Bell & Driscoll 2014). This paper documents the fate of a flowering event in *Acacia pendula* over a seven month period in this population, and examines the current age structure and health of individuals within the stand. Conclusions reached on the reproductive output and general health of these plants are considered in the context of future management.

Study Population

Location and habitat

The study population lies near Broke (32° 45' 0.4" S, 151° 6' 7" E) in the Hunter Valley of New South Wales (Fig. 1). This land has been established as part of the Weeping Myall Management Area (WMMA) by Glencore (Bulga Coal), with

the central aim of conserving *Acacia pendula* and its habitat. A monitoring program has been established to inform the management of these plants to ensure that impacts associated with nearby coal mining activities do not denigrate the site. The 3.8 hectare WMMA was fenced and cattle removed in March 2015, and only resident macropods (mostly Grey Kangaroos, *Macropus giganteus*) have grazed the site since. At the time of fencing, twelve individuals of *Acacia pendula* were known from the WMMA.

Currently, ten live individuals of *Acacia pendula* remain within the WMMA (Fig. 2). Seven of the ten individuals are old, well established trees, two are of medium size, and one is represented only by young suckering plants. Most of the older trees have fallen (some comprising two or more trunks) but persist as living plants, their heavy limbs now supported by the ground (Fig. 3). Two of three senescing plants are represented only by vigorous suckering from rootstock following trunk collapse, the third has shown no such suckering and appears dead.



Fig. 1: Location of the Weeping Myall Management Area (WMMA) near Broke in the Hunter Valley, showing local relief (contour interval is 10 m) and extent of landscape clearing.

The WMMA lies in largely cleared, undulating country at a mid-slope position (110 m ASL), on Permian aged geology (Fig. 4). Wollombi Brook, a major feeder stream to the Hunter River, lies c. 2.5 km to the west and is separated by an elevated Jurassic-aged basalt ridgeline housing the historic Milbrodale trig station (c. 170 m ASL). Prior to European settlement, the original vegetation across the study site, as determined by a census of the larger remaining ‘paddock’ trees within a radius of 500 m, likely consisted of a grassy woodland of *Eucalyptus moluccana*, with occasional *Eucalyptus crebra*. There is conjecture as to the origins of the *Acacia pendula* individuals on this site (and elsewhere within the Hunter Valley region), given that their presence in a grassy eucalypt woodland such as this runs contrary to their occupied habitat elsewhere in inland eastern Australia (see Bell & Driscoll 2014).

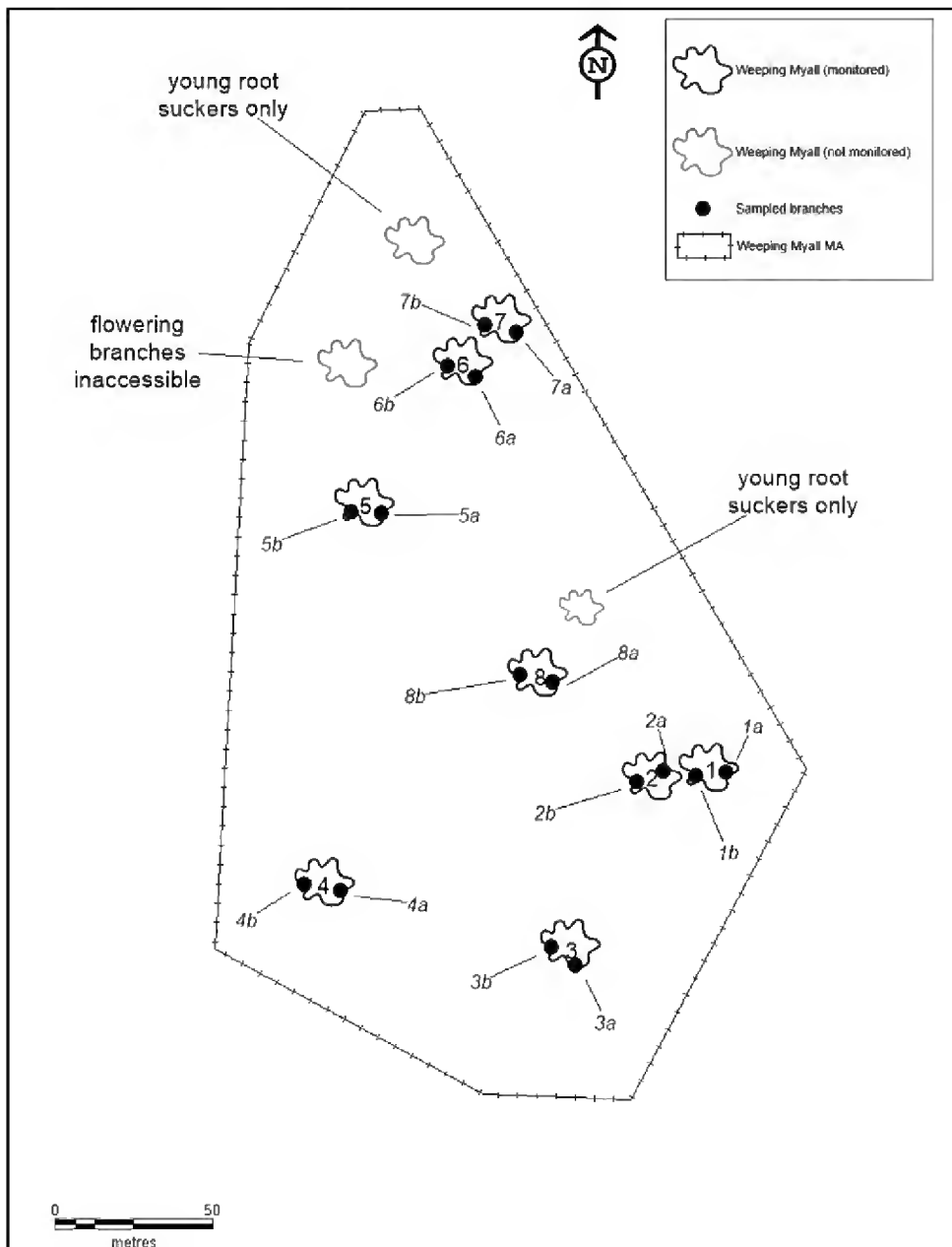


Fig. 2: The study population of *Acacia pendula* in the Weeping Myall Management Area, showing inspection locations on individual trees.



Fig. 3: Two fallen stems of a single individual of *Acacia pendula* (rooted in the centre at the position of observer), showing the canopy of each at extreme left and right.



Fig. 4: Landscape context of the Weeping Myall Management Area (fenced area in middle distance, below and within remnant *Eucalyptus moluccana* woodland).

Acacia pendula within the WMMA conform to morphotype B of Bell and Driscoll (2014), represented by plants with green foliage, slightly pendulous branches on older specimens, flowering irregularly but rarely if ever proceeding to the fruiting stage, and commonly root-suckering. Monitoring of these plants following the exclusion of cattle grazing in March 2015 has shown an eruption of new shoots emanating from roots ('root-suckers', commonly mistaken by some observers as new recruits). Over the course of just two years, the number of stems of *Acacia pendula* rose from 12 in 2013 to 685 in 2015, a 57-fold increase following fencing of the WMMA (visible in Figs 3 and 16). In time, these root-suckers develop into a dense thicket of vegetation shading out a large proportion of native grasses and herbs, and is currently the subject of a separate study.

Land use history

Prior to establishment as a reserve for the protection of *Acacia pendula*, the area formed part of an extensively cleared and modified agricultural landscape. As early as 1821, cattle agistment was granted by land owner Benjamin Singleton for the wider Patrick Plains area, with grazing by cattle and sheep centred on the nearby township of Broke (7 km to the south-east). In the 1850s, subdivision of the land fronting Wollombi Brook began, with partial clearing to accommodate grazing and dairying enterprises (Umwelt 2012). These pursuits remained the sole use of the land for the next 150 years, whereupon properties were purchased for coal mining or biodiversity offsets.

Climatic conditions

Rainfall leading up to the flowering event in January 2018 was well below average for an extended period of time (Fig. 5). Apart from above average falls in the March and October of 2017, little rain fell for the thirteen months prior to flowering (December 2016 to December 2017). Over the course of monitoring (January to July 2018), rainfall remained below the long-term average.

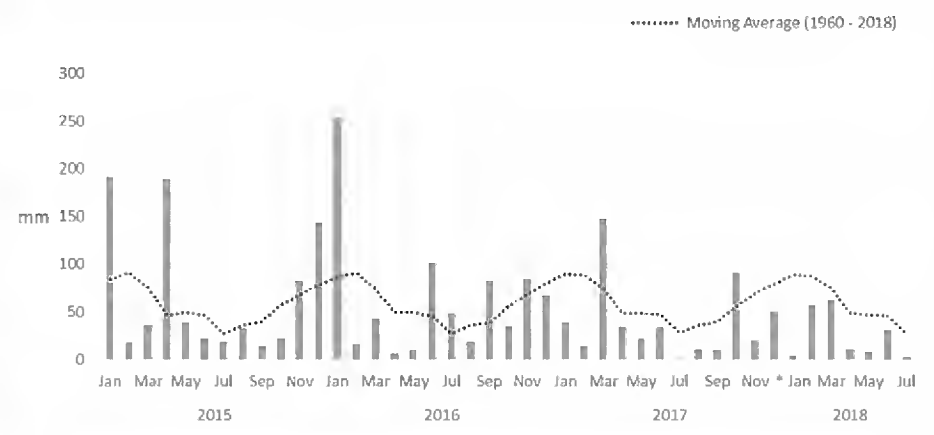


Fig. 5: Rainfall received at Bulga (3.5 km to the west) over the two years prior to flowering. Budding was first observed in January 2018 (* Jan.), following a prolonged dry period. Data source: Bureau of Meteorology (2018).

Methods

Flowering inspections

Eight individuals of *Acacia pendula* were selected for monitoring. For each monitored plant, two observation points were designated that were accessible and where most flower buds were evident at the commencement of the study. As far as possible, inspection points strived to include one receiving high sun exposure and a second receiving low sun exposure, but this was not always possible and was dictated by the extent of flowering on each individual (see Fig. 2). Inspections commenced in late February 2018 and continued monthly until the end of July 2018.

At each monthly inspection, general observations were made pertaining to the proportion and health of buds and flowers, and the presence or otherwise of developing pods. Tagging of specific inflorescences for more regimented monitoring was not undertaken as previous experience had shown high failure rates during flowering in this species, and observations of a more general nature were more likely to gather useful data. Additionally, notes were also made on the extent of flower and leaf galls, and activities of ants and other invertebrates. The presence of buds, flowers and fruits at each inspection point were assigned to one of four numerical categories: 0 (none present), 1 (few present, < 25 visible), 2 (many present, 25–100 visible), 3 (numerous present, >100 visible). Buds and flowers were considered viable and healthy if they were yellow and not dry and ‘crispy’, with no visible signs of galling or flower desiccation. Categorical data on bud and flower presence were averaged across the sixteen inspection points to graphically summarise the progress of flowering over the monitoring period.

Acacia age and health

In the absence of more definitive, non-destructive methods, the assessment of the age of individual *Acacia pendula* trees used stem diameter as a surrogate. The diameter-at-breast height was consequently measured on all *Acacia* plants within the study population (n=12, incorporating both live and dead individuals). In cases where more than one trunk was evident, all were measured but only the largest was used in analyses. For collapsed individuals that lay across the ground surface but remained alive, diameter was measured at approximately 1.7 m above the rooted point of the main trunk. Root suckers were too numerous to measure, and were ignored.

The presence of aerial mistletoe shrubs can impact on the general health and vigour of host species (Reid et al 1994; Watson 2011). In the case of *Acacia pendula*, the number of mistletoe clumps (*Amyema quandang*) was counted on each study plant to allow general observations on whether or not their presence appeared to influence the progress of flowering.

Results

Flowering phenology

Following initial observations of flower buds in early 2018 (Fig. 6), anthesis occurred from March (Fig. 7) but rapidly declined. There was a steady decline in both the number and health of inflorescences over the subsequent six months to July, where no active buds or flowers were evident (Fig. 8). In June, a small number of fresh buds were observed on some trees, suggesting that a second flush of flowering may occur but subsequent observations revealed otherwise. Flowering (open buds) peaked in March but then also underwent a decline to June, and none were present in July. No flowers were observed to progress to the fruiting stage, and no pods were recorded on any monitored tree. Rainfall during this six month period was below the long-term average, with April and May particularly well below average.

All monitored trees displayed at least some bud and flower development over the course of the study. Representative flowers sampled for microscopic examination appeared healthy and properly developed (Fig. 9), but over time these either senesced due to ongoing dry conditions (Figs. 10 & 11), or transitioned into galls. The majority of galls were found to be the result of infestation by the Common Flower Galler (*Dasineura glomerata*) (Fig. 12), and represents the first time *Acacia pendula* has been recorded as a host for this species (P. Kolesik pers. comm.). Previously documented host species include *Acacia deanei*, *Acacia elata*, *Acacia hakeoides*, *Acacia mearnsii*, *Acacia melanoxylon*, *Acacia pycnantha*, *Acacia retinoides* and *Acacia schinoides* (Kolesik et al 2005). Other galls present on inflorescences were attributable to bud galler (*Asphondylia* sp.) (Fig. 13), although these appeared less prevalent than *Dasineura*. On some flowers, woody, bulbous structures attributable to fungal gall (*Uromycladium* sp.) were also observed (Fig. 14). It is unknown if any individual flowers were successfully pollinated during this flowering event, but if so none proceeded to develop pods.



Fig. 6: Budding *Acacia pendula* (photographed 28 February 2018).



Fig. 7: Flowering *Acacia pendula*, approaching anthesis (photographed 27 March 2018).

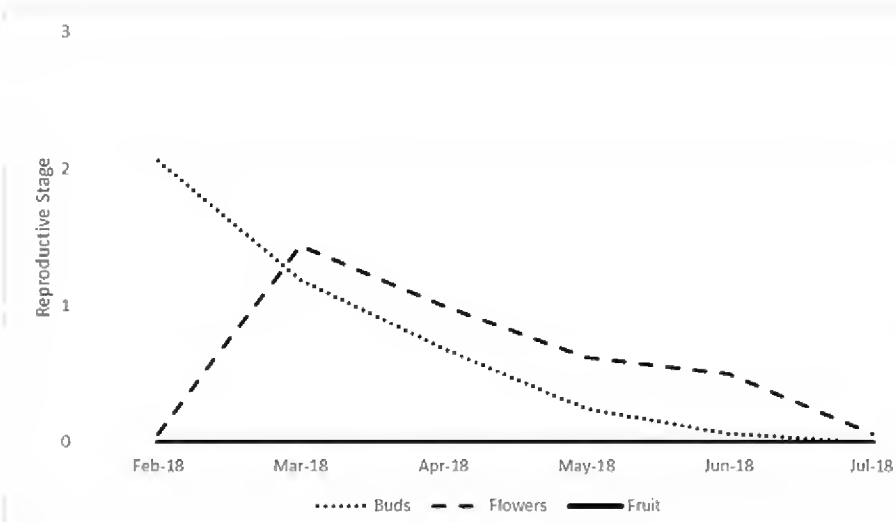


Fig. 8: Schematic summary of flowering fate of eight monitored trees over a six month period in 2018. No seed or fruit was produced. Reproductive Stage: 0 = none present, 1 = few present (< 25 visible), 2 = many present (25–100 visible), 3 = numerous present (>100 visible).



Fig. 9: Capitulum of *Acacia pendula* at anthesis, showing healthy stamens (photographed 28 February 2018).



Fig. 10: *Acacia pendula* inflorescence, showing desiccating capitula and partial dislodgement of stamens (upper capitulum) (photographed 31 May 2018).



Fig. 11: *Acacia pendula* inflorescence, showing complete dislodgement of stamens from each capitulum, and no development of pods (photographed 31 May 2018).



Fig. 12: *Acacia pendula* capitula freshly infected by galls of *Dasineura glomerata*, showing remnants of anthers and filaments between individual gall chambers (photographed 24 April 2018).



Fig. 13: *Acacia pendula* inflorescence, showing capitulum infected by bud galler (*Asphondylia* sp.) (upper left) and newer, healthy capitula (right) (photographed 31 May 2018).



Fig. 14: Fungal gall (*Uromycladium* sp.) on *Acacia pendula* capitulum, showing its woody texture (photographed 24 April 2018).

Acacia age and health

Across the study population, the average size of *Acacia pendula* trees was 50 cm DBH (diameter-at-breast height), with a standard deviation of 16.2 cm ($n=12$). The smallest tree was 23.2 cm DBH and the largest 82.8 cm DBH (Fig. 15). Three of the twelve individuals (including live and dead plants) possessed two trunks, while a further three had completely fallen trunks (two with copious root suckering) and lay across the ground (Fig. 16). One individual comprised a fallen trunk only with no root suckers and has presumably died, while another showed post-collapse development of roots from its trunk where it lay along the ground (Fig. 17). The large girth of trees within the study area is of some interest, as Boland et al (2006) described *Acacia pendula* with a diameter-at-breast height of “up to 30 cm”, nearly one third of the size of the largest specimen measured here. The large size of *Acacia* stems within the study area may explain why many of them have fallen over but continue to grow while supported on the ground.

Based on reported growth rates of the related *Acacia salicina* elsewhere (Grigg & Mulligan 1999; Jeddi & Chaieb 2012), the estimated age of individual *Acacia pendula* trees is

likely to be between 50 and 150 years. The inferred age of individuals did not appear to influence the extent and success (albeit limited) of flowering, as buds and flowers were observed across all eight study trees ranging between 23 and 67 cm DBH.

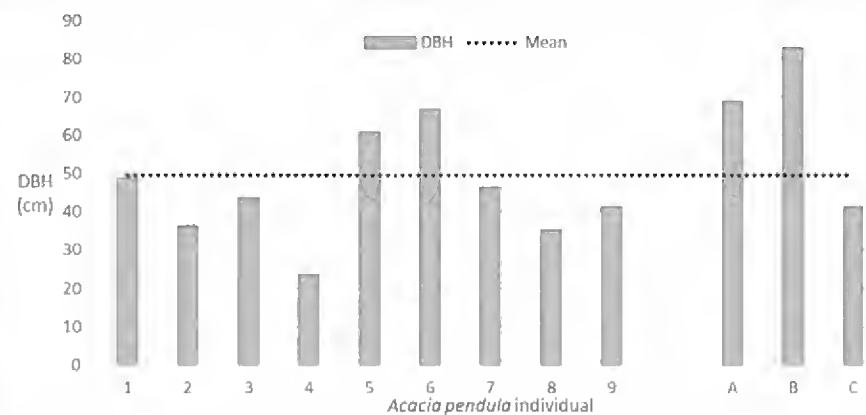


Fig. 15: Diameter-at-breast height (DBH) of *Acacia pendula* individuals within the study area. Only individuals 1–8 were the subject of flower monitoring; individual #9 supported an elevated canopy and was not monitored, while individuals A-C (also not monitored) were collapsed plants with vigorous (#A-B) or no (#C) root suckering.



Fig. 16: Aerial view of *Acacia pendula* (same individual as Fig. 3), showing fallen but still alive trunks (crowns at far left and far right), and copious root suckering in and around the centre.



Fig. 17: New root development mid-way along the collapsed trunk of *Acacia pendula*.

The extent of mistletoe (*Amyema quandang*) growth on *Acacia pendula* plants ranged from 2–38 clumps/tree with a high degree of variance ($n=9$, median of 5, mean of 10.8, SD of 11.7). There was no correlation between the size of *Acacia* trees and the number of mistletoe clumps supported on them. However, general observations suggest that those trees with higher densities of mistletoe appeared in poorer overall health than those with few mistletoes (Fig. 18).

Many trees also displayed evidence of attack by both Bag Shelter Moth (*Ochrogaster lunifer*; family Thaumetopoeidae) and galling-thrips (family Phlaeothripidae). Larvae of *Ochrogaster lunifer* feed on *Acacia* phyllodes and, in some cases, can completely defoliate a tree (Floater 1996). Large silk nests are formed in the canopy (Fig. 19), comprising *Acacia* phyllodes and silk produced by the larvae, and are used for resting during daylight hours. Galling-thrips also attack the phyllodes of *Acacia*, producing galls (Fig. 20) which extensively modify the shape and form of phyllodes (Crespi & Worobey 1998; Morris & Mound 2002). Galls present on *Acacia* within the study population appear attributable to *Kladothrips rugosus*, and although not extensive are present on most trees. All of these invertebrates are native Australian species and form part of the natural ecosystem in which *Acacia pendula* occurs.



Fig. 18: Mistletoe infested *Acacia pendula* showing signs of stress and death of limbs.



Fig. 19: Bag shelter produced by larvae of *Ochrogaster lunifer* in the branches of *Acacia pendula* (photographed 24 April 2018).



Fig. 20: Gall produced by *Kladothrips rugosus* on the phyllodes of *Acacia pendula* (photographed 27 March 2018).

Discussion

A rare flowering event in a population of *Acacia pendula* in the NSW Hunter Valley failed to progress to fruiting, suggesting that at least in the short-term persistence at this location is reliant on asexual reproduction. The 2018 flowering event was the first in that population since at least 2015 (when monitoring began), and such irregularity is reportedly a trait consistent with many other stands of the species in the region and throughout its range (Tame 1992; Boland et al 2006; Bell et al 2007). Fencing and the cessation of cattle grazing at the site in 2015 has been followed by copious emergence of root-suckers from nearly all individuals, yet evidence of successful seed production and subsequent new recruitment remains absent. As with other Hunter Valley stands of this species, long-term survival is likely to be contingent on the appropriate management of stock grazing pressures.

What events lead to the failure of fruit production in *Acacia pendula*? Within the study population during 2018 this appears primarily attributable to infestations of galling insects, accompanied by flower desiccation due to dry conditions. Gall-forming midges of the *Dasineura* and *Asphondylia* genera (family Cecidomyiidae) deposit eggs in the open flowers of *Acacia*, typically within the perianth tube near the ovary. On hatching, larvae then induce the ovary to evaginate and form a number of chambers, so that in some cases entire flower heads can transform into clusters of galls (Kolesik et al 2005; Kolesik 2015). This process results in the loss of flowering material, and hence reproduction in that inflorescence has effectively ceased. In other areas of New South Wales and South Australia, *Dasineura glauca* (Grey Fluted Galler) reportedly often occurs at such high densities that seed production is completely prevented in entire *Acacia pendula* trees (Kolesik et al 2005).

Dasineura glomerata (Common Flower Galler) is prevalent within the study population and appears likely to persist there permanently while ever the host plant remains. *Dasineura glomerata* has not been recorded infecting *Acacia pendula* previously, and represents a new host tree record for the species (P. Kolesik pers. comm.). Other known hosts for *Dasineura glomerata* included several *Acacia* species distributed mainly in coastal and near-coastal locations,

including *Acacia deanei*, *Acacia elata*, *Acacia hakeoides*, *Acacia mearnsii*, *Acacia melanoxylon*, *Acacia pycnantha*, *Acacia retinoides* and *Acacia schinoides*. None of these occur in the immediate locality of the study area, although *Acacia deanei* and *Acacia hakeoides* are present further west in the upper Hunter Valley (c. 60 to 90 km away from the study area), and *Acacia elata*, *Acacia melanoxylon* and *Acacia schinoides* occur in the adjacent mountainous districts. Only *Dasineura glauca* is known to infest *Acacia pendula* (Kolesik et al 2010), with other hosts for this species including the closely related *Acacia omalophylla*. A similar but undescribed gall-midge occurs on other semi-arid *Acacia* species, such as *Acacia aneura* and *Acacia ramulosa* (Kolesik et al 2005).

Individuals of *Acacia pendula* in the study population are also infected (to a lesser degree) by an undetermined species of a second gall-midge, *Asphondylia* sp., and a fungal gall, *Uromycladium* sp. (Pileolariaceae). In some *Acacia* populations, rust disease caused by *Uromycladium* poses a severe threat to the health and survival of infected individuals (e.g. McTaggart et al 2015), although at present this does not appear to be the case within the study population. Phyllodes are similarly attacked by the larvae of *Ochrogaster lunifer* (Thaumetopoeidae) and the galling-thrip *Kladothrips rugosus* (Phlaeothripidae). All of these invertebrates are native Australian species and form part of the natural ecosystem in which *Acacia pendula* occurs. When host plant species are under stress, such as brought about through habitat modification, infestations can severely impact normal growth and reproduction. Where Australian *Acacia* species have become invasive in other parts of the world, deliberate introduction of similar insects has been trialled as a biological control to limit spread (e.g. Impson et al 2008).

Flower desiccation due to dry conditions is a common reason for failure to reproduce in any one season (e.g. Anjum et al 2011). This phenomenon was also suspected to be occurring in the study population of *Acacia pendula* which was regularly under water stress with below average rainfall, despite reasonable falls in February, March and June. These falls did not, however, ensure the retention of flowering material on branches, and for those inflorescences not affected by galls the shedding of stamens to leave ‘bald’ capitulas soon followed. It was not possible to quantify the extent to which flower desiccation affected the overall potential for pollination and seed production, but this is suspected to be high. In a Western Australian study, Gaol and Fox (2002) noted that good winter rainfall was necessary to induce flowering in several *Acacia* species, but that further rain after flowering promoted pod development and seed production. For the *Acacia pendula* plants under study in the Hunter Valley, the abortion of flowering and the lack of pod production occurred despite rainfall in February, March and June.

Although plausible, an absence of pollinators is difficult to advance as a primary cause of flower failure. Most *Acacia* species are self-incompatible, and the transfer of pollen between individuals and populations via pollinating vectors is crucial for outcrossing and seed set (Stone et al 2003). For the bulk of *Acacia* species, this involves unspecialised,

generalist insects (Tybirk 1997). Pollinators of *Acacia pendula* are thought to comprise small native flies, bees and wasps (Bernhardt 1987), all of which are likely to travel over considerable distances visiting multiple stands of flowering plants. Given that the landscape surrounding the study population has been heavily cleared of native vegetation for at least 150 years (now fitting the *fragmented* or *relictual* states of McIntyre & Hobbs 1999), it is possible that the necessary pollinating invertebrates have also declined or disappeared (Kearns et al 1998). Many co-occurring *Acacia* species flower simultaneously, and in such cases such an event serves to attract a number of pollinators which are shared between species. In heavily modified landscapes, co-occurring species are often absent leading to a lack of co-flowering between species, and the threshold needed to attract pollinators may therefore not be reached. Apart from a single individual of *Acacia salicina*, there are no co-occurring *Acacia* within the study population, nor in the immediate vicinity (although good stands of *Acacia flicifolia* do occur 1 km to the north). Stone et al (2003) noted that populations of *Acacia* reduced to relict populations may have already lost their pollinator networks, resulting in lower seed set and dependence on opportunist pollinators. This scenario could also be extended to the study population of *Acacia pendula*, but this requires further investigation.

Recruitment failure as a result of grazing pressure has been documented for other arid-zone *Acacia* species (e.g. Batty & Parsons 1992; Auld 1995), although for the study area *Acacia pendula* impacts from grazing have affected the regeneration of root-suckers. Where recruitment failure is ongoing due to an absence of seed production, there can be important implications for conservation and management. For *Acacia carneorum*, Roberts et al (2017) found this species to be almost entirely reliant on asexual reproduction for persistence in an area, and that relatively few genetically distinct individuals were present across its range despite the often many thousands of stems in a stand. In that case, land managers were encouraged to protect both vegetative root-suckers and true seedlings from threats, as well as to use the few stands that did produce viable seed to augment existing populations through translocations. The lack of seed-producing stands of *Acacia pendula* in the Hunter Valley suggest that a similar recommendation for propagation and translocation of local provenance material cannot be promulgated unless genets originating from outside the region are used. Such an action is not recommended given uncertainty over plant origin in the Hunter Valley (Bell & Driscoll 2014).

Forrest (2016) related flowering events and prolonged recruitment failure from grazing impacts to rainfall patterns for several arid-zone *Acacia*. He found successful reproduction occurred in at least one of the two consecutive years following a La Niña wet period for the arid zone species *Acacia melvillei*, *Acacia homalophylla* and *Acacia loderi*. However, although these wet periods initiated sexual reproduction in these species, other factors appeared to limit success. Gaol and Fox (2002) earlier suggested that a wet winter period was required to induce flowering in some *Acacia*, and that follow up falls were necessary to ensure

seed production. For the study population of *Acacia pendula*, the 2018 flowering event occurred two years after a very wet three-month period from November 2015 to January 2016. South-eastern Australia at this time was in the grip of an El Niño event, and this wet period contrasted strongly with the below average falls received at other times in 2015 and 2016 (refer Fig. 5). Flowering in 2018 was therefore potentially a response to the wet period two years earlier, although without additional data on flowering phenology prior to 2015 this remains conjecture. Apart from this event, examination of rainfall data in the period leading up to flowering shows no clear pattern or spike in rainfall that may have triggered the 2018 flowering event. Winter rainfall was below average in 2017 prior to the documented flowering event, but largely above average in 2016 where no flowering was observed.

High mistletoe density on some *Acacia pendula* within the study area is impacting on the health and vigour of these plants, but desiccation and gall-infestation of flowers was consistent across all study trees, irrespective of the number of mistletoe clumps. However, some trees appear to have suffered branch death as a result of high mistletoe densities. Modification to landscapes associated with agricultural activities are known to increase the density of mistletoes (e.g. Bowen et al 2009; Watson 2011), as the availability of perches for avian vectors becomes greatly reduced. In other studies, mistletoes have been implicated in rapid turnover and increased mortality of host trees (e.g. Reid et al 1994; Reid & Stafford Smith 2000), although susceptibility is not universal (e.g. MacRaild et al 2009).

The general poor health and flowering displayed by *Acacia pendula* within the study population and elsewhere in the Hunter Valley are perhaps symptomatic of wider implications following extensive landscape modification. The study population lies on land that has been largely cleared for grazing purposes for many decades. Henry Dangar's 1828 map of the Hunter River area shows the WMMA to be "open forest country, deep loam soils occasionally intersected by scrubs & ill watered" (Umwelt 2012), but by 1850 subdivision and clearing of the land for grazing purposes had begun. Pastoralism was the first industry established in this part of the Hunter Valley, and in the nineteenth century the Broke area was a centre of pastoral interests based on sheep and cattle grazing (Umwelt 2012). Progressive removal of canopy and shrub species would have occurred during this period to increase the carrying capacity of the land for agriculture and grazing. Such modification to landscapes, with the inherent fragmentation of habitats that ensues, often leads to extinction cascades when the loss of key species in an ecosystem triggers the loss of other species (Fischer & Lindenmayer 2007). It is possible that such removal of key structural and floristic components of the former *Eucalyptus moluccana* woodland over an extended period of time may have led to the poor health and reduced sexual reproduction currently evident in the *Acacia pendula* population at the WMMA.

In any case, *Acacia pendula* trees within the study population are evidently subject to a number of stressors which affect successful and ongoing recruitment. These include but are not limited to infection by various flower- and phyllode-galling

midges and thrips (*Asphondylia* sp., *Dasineura glomerata*, *Kladothrips rugosus*), fungal galls (*Uromycladium* sp.), caterpillars (*Ochrogaster lunifer*), and mistletoe (*Amyema quandang*), together with stress brought about through drought and other climatic extremes. Pollinator absence or decline may also be imposing a different stress on the trees, but as of yet there is no data to confirm this. The absence of any old seed pods beneath all ten of the study trees suggests that these stressors have been operating on and limiting recruitment in them for many years, and that persistence in the area relies solely on asexual reproduction. Arguably, all of these stressors are a result of, or are exacerbated by, a highly modified and cleared landscape, and their collective impacts raise serious questions over how the species can remain viable in such a habitat into the future. Exclusion of stock grazing from *Acacia* populations may be feasible in the short-term at some locations, but management of grazing pressures for the benefit of *Acacia* is uncertain in the long-term, particularly during times of drought when all lands are subject to increased pressure to feed hungry stock.

Such a predicament for *Acacia pendula* has serious implications for conservation management, both here and in the wider Hunter Valley region if the patterns observed in the study population are repeated at other stands. This is particularly so in regard to conservation actions that require the augmentation of existing stands through translocation or supplementary planting. With no seed produced, augmentation planting can only rely on propagation from cutting material which re-distributes the existing poor genetic base. Alternatively, propagation using seed sourced from horticultural specimens (morpho-type A in Bell and Driscoll 2014) will introduce new genetic material into the region, a situation that is unfavourable given conjecture over the origin of existing plants. If *Acacia pendula* is ultimately shown through genetic studies to be a natural component of the contemporary Hunter Valley landscape, it remains unclear why such a disjunct population of the species occurs and persists in seemingly inhospitable habitat well east of its accepted geographical range. Hypotheses around its presence as a relict population from a previous drier climate regime (e.g. DEWHA 2009; OEH 2013), which may help explain the root-suckering habit, require further investigation.

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Managing *Persoonia* (Proteaceae) species in the landscape through a better understanding of their seed biology and ecology

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Abstract: *Persoonia* (family Proteaceae) is a diverse genus of 99 species, mainly of woody shrubs and small trees, that are endemic to Australia. The fleshy fruits that characterise these plants are an important resource in otherwise resource-depauperate ecological communities. However, this genus is highly under-represented in restoration and conservation programs, as its species are notoriously difficult to propagate and transplant in the wild. Understanding the mechanisms that control seed production, viability, dormancy and germination will hasten progress on *Persoonia* propagation. Here we review *Persoonia* studies to reveal the nature of, and variability within, the genus. We identify key factors that need to be addressed; specifically, those affecting fruit set, endocarp degradation and subsequent propagation of *Persoonia*. This synthesis of current knowledge provides important material to inform management of this taxon in the landscape, and outlines several important priorities for future seed biology research on the genus. We outline several important priorities for future seed biology research on the genus.

Keywords: Endocarp, germination, plant ecology, propagation, restoration, seed production

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Introduction

Success in many natural revegetation and restoration projects depends on the establishment of a wide range of species, but is often limited to those that are easy to collect, propagate and establish. Proteaceae are important keystone species in many restoration projects (Roche et al., 1997; Koch, 2007b; Stingemore & Krauss, 2013), be it at the landscape scale, with species of *Banksia*, *Grevillea* and *Hakea* being commonly used, or the translocation of a single threatened species (e.g. *Persoonia pauciflora*). Seed production in woody-fruited Proteaceae can vary significantly, from zero to tens of thousands of seeds on a single plant in any given season (Groom & Lamont, 1998). Seeds of some Proteaceae species are relatively easy to germinate but because many species occur in fire-prone habitats, germination may be cued to fire, and can be difficult to germinate due to specific conditions being required to break the complex dormancy mechanisms (Van Staden & Brown, 1977; Morris, 2000; Morris et al., 2000; Arnolds et al., 2015; Chia et al., 2016).

Persoonia is one genus within the Proteaceae that has very complex dormancy mechanisms. This genus includes many species that are the subject of restoration or conservation projects and include *Persoonia longifolia* (southern Western Australia), *Persoonia pauciflora* (Hunter Valley, NSW), *Persoonia hindii* (Newnes Plateau, NSW) and *Persoonia hirsuta* (Sydney Basin, NSW). However, their inclusion is hampered by consistently poor propagation success (Cambeceades & Balmer, 1995; Ketelhohn et al., 1996; Bauer & Johnston, 1999). For example, *Persoonia longifolia* plants were present in areas of Western Australian jarrah woodland prior to bauxite mining in the 1960s (Koch 2007a; 2007b). The mine site restoration plan for these areas included a target of restoring the ecosystem to a state comparable with pre-mining, but although *Persoonia longifolia* had viable seeds, they could not be germinated and were absent from rehabilitation projects (Koch, 2007b). Recent research on the seed ecology of this species (Chia et al., 2015), including how seasonality and fire affect *in situ* fruit set, dormancy release and germination, identified the seasonal conditions and length of time required for dormancy to break and germination to occur in the soil (Chia et al., 2016).

We see a focus on plant ecology to be of great value for progress in conservation and restoration. It is evident that re-establishing new populations, or augmenting extant populations through translocation, requires a detailed understanding of how the plants interact in the natural environment, if they are to have long-term success. We argue that progress on *Persoonia* seed production and propagation will be facilitated by understanding the mechanisms in nature that control seed production, viability, dormancy and germination. This review aims to identify major research priorities and develop a logical framework to guide future investigations towards a more systematic approach to resolve species persistence in the landscape. Such an approach could also have benefits for other Proteaceae with a similar seed

biology to *Persoonia*, and may be relevant to other families with similarly deep, intractably dormant seeds.

The genus *Persoonia*

Species of *Persoonia* (family Proteaceae) range from low prostrate or spreading shrubs to small trees (Appendix 1); they are characterised by light green foliage with high morphological variability across species, and yellow hermaphroditic flowers (Weston 2003; Fig. 1). A maturing ovule forms a fleshy drupaceous fruit comprising a single hard woody stone containing either one or two seeds (Fig. 2), and an embryo can have up to nine cotyledons (Weston, 2003). All 99 species are endemic to Australia, and together the genus occupies 64 of the 87 national bioregions (IBRA7; Appendix 1), but is largely absent from central arid regions. Including subspecies there are 115 taxa, 72 endemic to eastern Australia, 42 to Western Australia, and one across northern Australia. Nine species are listed as threatened, endangered or critically endangered under the Australian *Environment Protection and Biodiversity Conservation Act, 1999* (eight species occur in NSW; Appendix 1), being impacted by several anthropogenic factors, including land clearing, mining, habitat fragmentation, grazing, slashing and predation. There is a distinct lack of ecological and seed biology data on *Persoonia*; much of this research has been conducted on eastern Australian *Persoonia* species (Table 1).



Fig. 1. Morphological variation within the *Persoonia* genus. **A-** *Persoonia myrtilloides* plant; **B-** *Persoonia levis* plant; **C-** *Persoonia hirsuta* plant, and; **D-** *Persoonia pauciflora* plant (Photos by N. Emery).

Table 1. *Persoonia* species that have been studied for breeding system, fruit set and/or seed germination. Several species have had multiple independent studies that measured one or more component of the reproductive niche (see Appendix 2 for References)

Species	Distribution (state)	Habitat ¹	Rarity ²	Fire response ³	Breeding System ⁴	Fruit set ⁵	Germination ⁶
<i>Persoonia bargoensis</i>	NSW	DSF, DW	En*, Vu†	1	SC		
<i>Persoonia elliptica</i>	WA	DSF, DW	NL	3		39%	
<i>Persoonia glaucescens</i>	NSW	DSF	En*, Vu†	1	NC	18%; 86%	
<i>Persoonia juniperina</i>	NSW, SA, TAS, VIC	DSF, H	NL	2	SC	40%; 30—41.4%	
<i>Persoonia lanceolata</i>	NSW	DSF	NL	1	NC	41%; 88%; 97%	~10.0%
<i>Persoonia levis</i>	NSW, VIC	DSF, DW	NL	3	NC	5—55%; 52%	50.0%
<i>Persoonia longifolia</i>	WA	DSF, DW	NL	3		10%; 98.3%	31.8—94.7%
<i>Persoonia mollis</i> subsp. <i>maxima</i>	NSW	DSF	En*†	1	NC	18%; 89%	
<i>Persoonia mollis</i> subsp. <i>nectens</i>	NSW	DSF	NL	1	NC	35%; 91%	~40.0%
<i>Persoonia myrtilloides</i>	NSW	DSF, DW	NL	1	NC	10—70%	
<i>Persoonia rigida</i>	NSW, VIC	DSF	NL	1	SC	67%	
<i>Persoonia sericea</i>	QLD, NSW	DSF, WSF	NL	2			87.5%
<i>Persoonia virgata</i>	QLD, NSW	DSF	NL	1	NC	36.1—41.6%; 48.9%	50%; 58.8— 87.5%; 100%

¹ DSF = dry sclerophyll forest, DW = dry woodland, H = heath, WSF = wet sclerophyll forest

² En = endangered, NL = not listed under state or national legislation, Vu = vulnerable; * threatened status listed under state/territory legislation; † threatened status listed under the national EPBC Act

³ Ability to resprout following fire: 1 = cannot resprout and reliant on seeds, 2 = can resprout from base only. 3 = can resprout from base and stems; Rymer (2006)

⁴ NC = non-compatible breeding system, SC = self-compatible breeding system; reference list available in Appendix 2

⁵ Fruit set from outcrossed pollination reported in the literature as of 23/04/2018; reference list available in Appendix 2

⁶ Germination results reported in the literature as of 23/04/2018; reference list available in Appendix 2

Field ecology behaviour

Some *Persoonia* species occur across multiple climatic zones, but others are more localised including several of the rarer obligate-seeding species in the Sydney region, and these species often establish alongside roads and tracks. For example, *Persoonia hirsuta* plants occur in drainage lines along track edges with the largest populations along disturbed road easements (N. Emery, pers. obs. 2017). Myerscough et al. (2000) postulated that soil disturbance events might substitute for the effects of fire, particularly in environments with long inter-fire intervals.

Persoonia species lack the proteoid roots (characteristic of most Proteaceae species) that aid inorganic nutrient absorption, yet plants often occur in well-drained, nutrient-poor acidic soils such as the sandstone and shale soils of the Sydney region (Myerscough et al. 2000; Weston 2003). As individual plants can thrive in their environments it is possible that unknown mycorrhizal associations might occur in the roots. *Persoonia pauciflora* plants often occur at the base of Broad-leaved Ironbark (*Eucalyptus fibrosa*) or Spotted Gum (*Corymbia maculata*) trees (N. Emery, pers. obs. 2017), which might indicate a possible relationship between these species.

Flowering times

Peak flowering in most Western Australian *Persoonia* species occurs over winter and spring, and eastern Australian species

predominantly flower during summer and autumn (Bernhardt & Weston 1996; Table 2). Some species such as *Persoonia pinifolia* can produce flowers for most of the year. Eastern Australian *Persoonia* species experience a high frequency of hybridisation (Myerscough et al. 2000) that could be explained by a combination of coinciding distributions, flowering times and/or pollinators, as well as a lack of pre-zygotic barriers for interspecific pollen (Bernhardt & Weston 1996).

Pollination

Many Proteaceous species produce large inflorescences with copious amounts of nectar, making them well-suited for vertebrate pollination (Carolin, 1961; Collins & Rebelo, 1987). *Persoonia* species, in contrast, have small yellow flowers that are most notably pollinated by bees and wasps (Carolin, 1961; Collins & Rebelo, 1987; Bernhardt & Weston, 1996). It was originally postulated that a mutualistic relationship exists between *Persoonia* and small native, hairy *Leioproctus* bees (Hymenoptera: Colletidae) (Bernhardt and Weston 1996). *Leioproctus* have since been observed to pollinate numerous *Persoonia* species including *Persoonia glaucescens*, *Persoonia lanceolata*, *Persoonia mollis* subsp. *maxima*, *Persoonia mollis* subsp. *nectens* and *Persoonia virgata* (Wallace et al., 2002; Rymer et al., 2005). Other bee pollinators known to visit *Persoonia* include *Exoneura* spp. (Bernhardt & Weston, 1996), *Tetragonula carbonaria* (formerly *Trigona*) (Wallace et al., 2002), and the European honeybee *Apis mellifera* (Bernhardt & Weston, 1996; Wallace et al., 2002; Rymer et al., 2005; Chia et al., 2015).

Species	Distribution (state)	Peak flowering time (month)											
		winter			spring			summer			autumn		
		J	J	A	S	O	N	D	J	F	M	A	M
<i>Persoonia subtilis</i>	NSW												
<i>Persoonia subvelutina</i>	ACT, NSW, VIC												
<i>Persoonia juniperina</i>	NSW, SA, TAS, VIC												
<i>Persoonia asperula</i>	NSW, VIC												
<i>Persoonia brevifolia</i>	NSW, VIC												
<i>Persoonia chamaepeuce</i>	NSW, VIC												
<i>Persoonia confertiflora</i>	NSW, VIC												
<i>Persoonia levis</i>	NSW, VIC												
<i>Persoonia linearis</i>	NSW, VIC												
<i>Persoonia rigida</i>	NSW, VIC												
<i>Persoonia silvatica</i>	NSW, VIC												
<i>Persoonia arborea</i>	VIC												
<i>Persoonia gunnii</i>	TAS												
<i>Persoonia moscalii</i>	TAS												
<i>Persoonia muelleri</i> subsp. <i>angustifolia</i>	TAS												
<i>Persoonia muelleri</i> subsp. <i>densifolia</i>	TAS												
<i>Persoonia muelleri</i> subsp. <i>muelleri</i>	TAS												

Manipulative pollination experiments suggest that *Persoonia* species have a breeding system that favours out-crossing (Krauss, 1994; Cadzow & Carthew, 2000; Wallace et al., 2002; Table 1). In *Persoonia mollis*, for example, 20% of out-crossed flowers set fruit compared to just 1% of selfed flowers (Krauss, 1994). Furthermore, pollen tubes were not present in the ovaries of self-pollinated flowers. The experimentally-manipulated result for out-crossed flowers also reflected the natural pollination level, with 17% of unmanipulated flowers setting fruit (Krauss 1994). Similarly, *Persoonia virgata* pollination experiments showed weak self-compatibility, as fruit set was significantly lower in self-pollinated flowers (6.6%) than cross-pollinated flowers (48.9%) (Wallace et al., 2002). Krauss (1994) first noted the possibility of a post-zygotic mechanism within *Persoonia* seeds that caused the majority of selfed fruits to be prematurely terminated between 4 and 30 weeks following pollination. In *Persoonia juniperina* it was reported that 76% of selfed fruits terminated during the maturation period compared to 33% for open-pollinated fruits (Cadzow & Carthew 2000). Alternatively, self-compatibility has been reported in *Persoonia*, as pollen tubes were frequently observed in self-pollinated flowers of *Persoonia rigida*, and final differences in the number of matured fruit from self- and cross-pollinated flowers were not statistically significant (Trueman & Wallace, 1999). Self-compatibility has also been documented for *Persoonia juniperina* and *Persoonia bargoensis* (Cadzow and Carthew 2000; Field et al. 2005).

Fruit set success

Flowers of Australian Proteaceae are hermaphroditic and typically produce a very low rate of fruit set – around 5% (Ayre & Whelan, 1989). Fruit set success varies considerably among *Persoonia* species, and has been documented in

re-sprouting and obligate seeding species (Table 1). Fruit set in *Persoonia longifolia* was reported to be more variable among plants within a population than between populations, and to be positively correlated with plant height and time since last fire (Chia et al., 2015). The availability of carbohydrates that could be transferred from branches to the fruits was reported to be positively correlated with fruit size in *Persoonia rigida*, and fruit set on leaf-bearing branches being 4-6 times higher than defoliated branches (Trueman & Wallace 1999). Minimal vegetative growth during fruit maturation was observed on *Persoonia virgata* plants (Bauer et al. 2001). The slow development of *Persoonia* embryos coupled with the requirement of nutrient uptake for embryo growth, suggests that most of the plant resources are allocated to fruit development during the fruiting season (Strohschen, 1986).

Fruit maturation

Persoonia peak flowering and fruit set precedes a long and highly variable period of fruit maturation reported to require at least 2 months. In some species it can take up to a year for fruits to fully mature and drop from the maternal plant (Trueman & Wallace, 1999; Benson & McDougall, 2000; Wallace et al., 2002; Weston, 2003; Rymer et al., 2005; Chia et al., 2015). It has been reported that *Persoonia pinifolia* embryo maturation is significantly slower than other Proteaceae genera such as *Macadamia*, and the endosperm is almost completely replaced by the embryo at 34 weeks post-anthesis (Strohschen 1986).

Fruit dispersal

Seed dispersal beyond the maternal plant environment may be limited to fruit-drop from the maternal plant (Rice & Westoby, 1981), but the fleshy *Persoonia* fruits are also likely to be consumed and dispersed by birds and

mammals (Weston, 2003; Auld et al., 2007). In one study, 90% of *Persoonia lanceolata* fruits were consumed by Swamp Wallabies (*Wallabia bicolor*) with 98% of these still being viable after being collected from scats (Auld et al., 2007). *Persoonia longifolia* fruits have been reported to be consumed by Brush Tail Wallabies (*Macropus irma*), Western Grey Kangaroos (*Macropus fuliginosus*) and Bobtail Lizards (*Tiliqua rugosa*) (Chia et al. 2015). Many native birds consume *Persoonia* fruits, including the Olive-backed Oriole (*Oriolus sagittatus*), Silver-eye (*Zosterops lateralis*), Pied Currawong (*Strepera graculina*), Regent Bowerbird (*Sericulus chrysocephalus*), Satin Bowerbird (*Ptilonorhynchus violaceus*), Red Wattlebird (*Anthochaera carunculata*) and Lewin's Honeyeater (*Meliphaga lewinii*) (Barker & Vestjans, 1990). However, it is not known whether these vertebrates facilitate dispersal of viable seeds in their scats. *Persoonia longifolia* fruits are commonly found in Emu (*Dromaius novaehollandiae*) scats, but the germinability of these remains very low (Mullins et al., 2002). Cockatoos and other parrots have also been observed to predate on immature *Persoonia* fruit (Weston 2003; K. Chia, pers. comm. 2016). The removal of *Persoonia* seeds may correlate with the rarity and size of plants, as macropods were found to remove significantly more fruits of two common *Persoonia* species (*Persoonia lanceolata* and *Persoonia mollis* subsp. *maxima*) compared with two rare species (*Persoonia glaucescens* and *Persoonia mollis* subsp. *nectens*) (Rymer, 2006). Furthermore, seed removal was significantly positively correlated with plant height in common species only, although plant population size was not reported to be influencing removal (Rymer, 2006).

Seed biology

Within the fleshy *Persoonia* exocarp and mesocarp is the woody endocarp (Fig. 2), which restricts germination as a form of mechanical dormancy. In *Persoonia longifolia* laboratory trials, germination only occurred when all or half of the endocarp was removed (78% and 68% success, respectively - Chia et al. 2016). Norman and Koch (2008) determined that *Persoonia longifolia* endocarps were permeable to water, (increasing in weight by 10-30% following 30 hours of imbibition), but the permeability and hardness of buried endocarps did not significantly differ from the controls after a 2-year soil burial trial, suggesting that endocarp weakening over time is slow. A recent study on *Persoonia longifolia* noted that the removal of the endocarp lid did not increase the rate of imbibition (Chia et al., 2016).

Coupled with the mechanical dormancy mechanism imposed by the endocarp is the physiological dormancy of the embryo, which may require treatment using a chemical stimulant such as gibberellic acid (GA_3), or a combination of warm and cold stratification to improve overall germination success (Mullins et al., 2002; Chia et al., 2016). Mullins et al. (2002) suggested that *Persoonia longifolia* seeds required an unknown period of cold temperatures over winter to maximise overall germination. By contrast, *Persoonia myrtilloides* and *Persoonia levis* seeds showed significantly reduced and no germination, respectively, following a chilling pre-treatment

(Nancarrow 2001). A recent study on *Persoonia longifolia* suggested that the environmental conditions the endocarps are exposed to are more important than the actual burial time. Specifically, germination was highest when endocarps were treated with two simulations of summer rainfall events and a constant summer temperature of 30°C (Chia et al. 2016). Adding more complexity, a heat spike treatment (50°C) improved germination when moisture was limiting, but germination was significantly reduced if long wet cycles were introduced.

It is possible that the proportion of physiologically dormant seeds may be species-specific or vary among years depending on conditions during fruit maturation. Some studies have found that the addition of GA_3 made no difference or had highly variable results, to overall germination (Ketelhohn et al., 1996; Nancarrow, 2001; Chia et al., 2016). For example, GA_3 increased germination of *Persoonia virgata* seeds by at least 50% (Ketelhohn et al., 1996; Bauer et al., 2004). Similarly, it increased germination of *Persoonia levis* seeds, whereas *Persoonia myrtilloides* seeds were unaffected (Nancarrow, 2001). As seeds from *Persoonia myrtilloides* only germinated after 4 months in storage, it is possible that a period of after-ripening or stratification is required to alleviate physiological dormancy. Furthermore, as no indication of viability was given, it is also possible that viable embryos of both species were damaged during the removal of the endocarp.

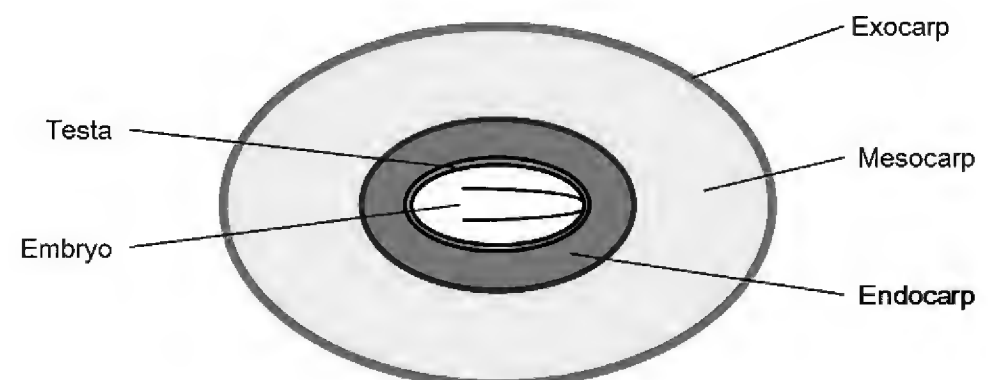


Fig. 2. Diagrammatic cross-section of a typical *Persoonia* fruit (not to scale), comprising mostly a fleshy mesocarp behind a leathery external layer (exocarp). The mesocarp covers the woody stone (endocarp), which protects the seed. *Persoonia* seeds are predominately made up of a testa and non-endospermic embryo, and may contain one or two seeds within the endocarp. The endocarp is the key structure that prevents germination from occurring.

Early studies on *Persoonia pinifolia* and *Persoonia longifolia* noted that germination was negatively affected by microbial contamination (McIntyre 1969; Mullins et al. 2002), but the recommended disinfecting and germinating of seeds under aseptic conditions, has produced mixed results for germination success (Bauer et al., 2004; Chia et al., 2016). Interestingly, contamination of *Persoonia longifolia* seeds was most prevalent in those treated with the smoke stimulant karrikinolide (Chia et al., 2016). *Persoonia pauciflora* seeds treated with GA_3 also suffered from severe contamination despite being surface-sterilised (N. Emery, unpublished data). Microbial growth within the seed could be promoted by both GA_3 and karrikinolide as similar chemical derivatives have a microbial origin (Brian et al., 1954; Light et al., 2009).

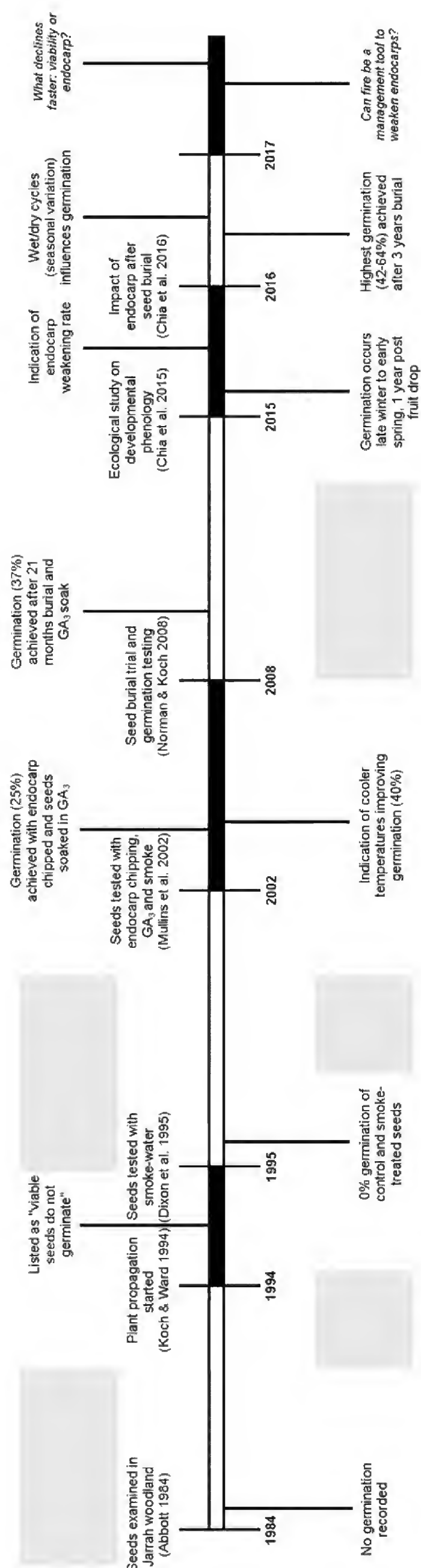


Fig. 3. An example of the framework for *Persoonia longifolia* seed ecology research. Seeds have been tested for germination as part of large mining rehabilitation projects (Abbott 1984; Koch & Ward 1994), conservation studies (Dixon et al. 1995; Norman & Koch

2008), and propagation studies (Mullins et al. 2002; Norman & Koch 2008). Successful germination was not achieved until 2002, when endocarps were physically compromised and seeds were treated with gibberellic acid (GA₃). Germination remained low (25 – 40%); however, there was an indication from direct sowing that germination was higher following cooler temperatures. This link with climate was then rigorously examined in recent studies on the developmental phenology and the endocarp (Chia et al. 2015; Chia et al. 2016). Seed germination was reported to be highest following three years of soil burial, and, moreover, wet and dry cycle length interacted with the rate of endocarp weakening. These results now raise the possibility of fire or heat being used as a management tool for endocarp weakening, and whether endocarps degrade at a faster rate than seed viability *in situ*.

Soil seedbank persistence

In the soil seedbank the endocarp is expected to decompose over time, thereby increasing the rate of oxygen and water reaching the embryo, and allowing the embryo to ‘push out’ of the weakened endocarp. Previous research has reported mixed results of recruitment success in *Persoonia* and, therefore, long-term persistence of seeds in the soil (Auld et al. 2007; Ayre et al. 2009; McKenna, 2007; Chia et al., 2015). *Persoonia pinifolia* fruit, for example, were estimated to have a half-life of one year in the soil seedbank (Auld et al., 2000). The viability of *Persoonia elliptica* seeds in the soil seedbank declined from 39% to 5% after one year (Nield et al., 2015). *Persoonia longifolia* seeds showed a comparatively smaller decline in viability from 93% to 68%, recorded after three years (Chia et al. 2016). In contrast, four *Persoonia* species (two rare and two common) experienced a significant decline in viability to around 30% following a 1-2 year soil burial (McKenna 2007). Interestingly, viability decline also significantly varied among populations, which could indicate local variation due to genetic or environmental factors. However, such a decline in viability might not adversely affect recruitment success where annual fruiting events produce an accumulating seedbank. For example, 476 *Persoonia mollis* subsp. *nectens* seedlings emerged following a wildfire from a population of 25 adult plants (Ayre et al. 2009). An additional 381 seedlings emerged following a second wildfire four years later, and before the first seedling cohort reached reproductive maturity (Ayre et al. 2009). These results suggest that the population had a large and persistent soil seedbank, capable of withstanding multiple fires. Auld et al. (2007) estimated that although the number of *Persoonia lanceolata* seedlings that emerged post-fire was 6-7 times greater than the pre-fire adult numbers, there was at least 72% of available soil seedbank that did not germinate.

Natural recruitment: the role of fire and smoke

It was originally reported that seedling recruitment of *Persoonia* is most likely to occur following a disturbance such as fire (McIntyre, 1969). Fire was thought to be a crucial factor for the recruitment of *Persoonia elliptica* seedlings in jarrah woodlands in Western Australia, with only one seedling observed in plots that had not been burnt (Nield et al. 2015). With fire comes the risk of seeds being destroyed by the combustion (Chia et al., 2015); fire intensity is likely

to be an important factor influencing seed survival, but has not yet been investigated for *Persoonia*.

Auld et al. (2007) remarked that since most *Persoonia lanceolata* fruits were found in the top 5 cm of soil, contact with smoke during or shortly after a fire was a distinct possibility. However, smoke has not been commonly used in *Persoonia* germination experiments or shown to have a positive benefit on germination success. In a comprehensive study of the effect of smoke water on the germination of plants in Western Australia, *Persoonia longifolia* seeds did not germinate when treated with smoke water (Dixon et al., 1995). However, this study used fresh, non-aged seed which would not reflect the ecological priming of soil-stored seed in nature and the post-fire germination found in this species. A subsequent study also found no change in germination success of *Persoonia longifolia* when smoke was applied at different times prior to sowing; however, again non-aged seed was used (Mullins et al., 2002). Similarly, there was no additive effect when smoke water was applied with GA₃. This could be due to both smoke and GA₃ having a similar mode of catalysing germination, through opening the respiration pathway by stimulating the conversion of oxygen to superoxide in the seed (Sunmonu et al., 2016). Furthermore, it is known that smoke catalyses the production of enzymes such as amylase, mobilising starch compounds from the endosperm to other parts of the seed (Cembrowska-Lech & Kępczyński, 2016; Sunmonu et al., 2016). Whether the application of smoke stimulates other mechanisms, particularly for species with non-endospermic seeds, remains unknown. However, this might explain why using smoke as a pre-treatment has no additive effect on the germination of *Persoonia* as the seeds lack an endosperm at maturity (Strohschen 1986).

The variability in germination success reported in *Persoonia* species means that further testing is required to determine the environmental and population factors needed for optimal seed collection, storage and germination conditions. We consider that while a disturbance such as fire may be required to break the mechanical dormancy in *Persoonia* seeds (i.e. the woody endocarp), smoke water provides no greater benefit for overall germination success of fresh seed than GA₃. However, if the role of smoke on germination is to be examined in an ecologically relevant manner, then trialling a possible interactive effect on soil-aged or seasonally-stratified seeds would be an appropriate future study.

Discussion

Much of our knowledge of *Persoonia* seed biology is derived from *ex situ* propagation research, be it for seed germination, dormancy status or viability. However, we also need to understand how the plants interact with their local environment *in situ*, i.e. their ecology. Once the main factors driving fruit set and dormancy are determined, we can use this knowledge to include particular species in propagation and restoration programs. An example demonstrating the relative benefits to ‘progress’ upon changes from *ex situ* germination testing to *in situ* ecological requirement studies

can be seen in our synopsis of *Persoonia longifolia* research (Fig. 3). Following on from this and other *Persoonia* species research efforts, we outline the major knowledge gaps for restoration practice and, therefore, the research priorities for future work on this taxon.

Climate and phenology

A major omission in many *Persoonia* studies to date is a quantifiable link between phenological events and climatic factors, namely temperature and rainfall. For example, changes in the timing and duration of flowering can have flow-on effects for other phenological events. In several Proteaceae species, a decline in mean daily germination (due to enforced seed dormancy) correlating with an increase of 1.4°C and 3.5°C during seed incubation has been reported (Arnolds et al. 2015). Below-average rainfall was postulated to cause the mortality of several *Persoonia* species following fire (McKenna, 2007). If the flowering phenology of *Persoonia* can be linked with climate, then this could provide better predictions of the species niche, as well as determine appropriate growing conditions and the adaptive timing for fruit set and maturation. Since *Persoonia* species have a breeding system that preferences outcrossing and are predominately pollinated by native and exotic bee species, the effect of the timing of flowering on interactions with pollinators warrants further investigation.

Recent work on *Persoonia longifolia* illustrated the importance of climatic events on both *in situ* and *ex situ* seed burial. For example, brief wet events over summer, such as a thunderstorm, were reported to greatly improve overall germination by breaking mechanical dormancy in the endocarp (Chia et al. 2016). As *Persoonia longifolia* seeds are physiologically dormant, this means that an interactive effect, in the sense of warm and cold stratification, is also required to alleviate dormancy in the seed. Furthermore, as post-fire germination is not always immediate, it is possible that heat exposure, rather than smoke, weakens or cracks the endocarp allowing germination to commence sooner (McKenna, 2007; Chia et al., 2015).

Rethinking the role of fire

If endocarp degradation commences in the soil after fruit drop and follows wet and dry cycles (Chia et al., 2016), then the timing of fire could have a significant effect on seed germination and viability. For example, seeds from populations of *Persoonia glaucescens* and *Persoonia bargoensis* (both obligate seeders) that had been burnt, declined in viability over 12 months (McKenna 2007). By contrast, seed viability in unburnt populations did not vary over the same time. Auld et al. (2000) reported that seed viability in *Persoonia pinifolia* (an obligate seeder) declined to 28–40% following 2 years of soil burial, and suggested that a prescribed burn midway through the experiment may have contributed to seed death. However, seed death did not significantly increase post-burning when compared with pre-burning, and it was thought that seed ageing was the main factor contributing to viability loss (Auld et al., 2000).

Persoonia endocarps are water-permeable, and poorly-timed fires could essentially pressure-cook any partially imbibed seeds (Norman & Koch, 2008; Chia et al., 2016). While fire may hasten the relaxing of mechanical dormancy, if a portion of the soil seedbank has already experienced some level of degradation, a fire could then scorch and kill the more water-permeable seeds. This outcome could be the underlying reason for significantly higher germination of *Persoonia mollis* subsp. *nectens* seeds following a medium-intensity burn, compared to a high-intensity burn (McKenna, 2007). Chia et al. (2015) suggested that fire had killed *Persoonia longifolia* seeds in the seedbank, as recruitment events only occurred following post-fire fruit set. Taken together, these results suggest a fine balance for endocarp degradation through wet and dry cycles and fire. This raises the question of whether endocarps require a fire and, if so, when should a fire occur relative to fruit drop? It is also plausible that a low-intensity burn following seed-sowing might lead to a shorter time to recruitment.

To determine whether either the prescribed burning of *Persoonia* soil seedbanks or *ex situ* ‘priming’ of seeds by burning is likely to be important, the *in situ* seed longevity must be known. Previous evidence indicates that seed longevity varies among *Persoonia* species (Auld et al. 2000; Norman & Koch, 2008) and suggests that seed longevity and endocarp degradation might be intimately linked. If seeds lose viability before the endocarp breaks down in the seedbank sufficiently to allow germination to commence, an early controlled burn could shorten the time for endocarp weakening.

However, some obligate-seeding *Persoonia* species require an interval of at least 8 years between fires to allow juvenile plants to reach reproductive maturity (Weston, 2003). This part of the life-cycle is still poorly understood; the length of the primary juvenile period for only six *Persoonia* species is known (Appendix 1).

Seed production areas

The goal for any species being re-introduced to an ecosystem is to produce self-sustaining populations. Genetic variation in local provenances is also an important consideration for restoration practices. For example, non-locally sourced material could have negative consequences for persistence due to factors such as maladaptation, where the non-local material is selected against local genotypes, leading to higher mortality rates (Bischoff et al., 2010). However, there may be occasions when non-local material is required, such as providing sufficient genetically-diverse material to buffer rare species from future environmental change (Broadhurst et al., 2008). It is not our intention to discuss the various merits of local vs. non-local material. Rather, we describe a more immediate requirement of generating a source of high quality seed.

Seed is often sourced from the wild in large quantities for restoration projects (Broadhurst et al. 2015) but issues including reproductive failure, low abundance, plant age and phenological variation can hamper the availability of large seed collections. Rare species often have life-history

traits that can create barriers to fecundity and survival (Abeli & Dixon, 2016; Reiter et al., 2016). Rare or threatened *Persoonia* species, including *Persoonia pauciflora* and *Persoonia bargoensis*, have poor seed-production years. Successful translocations of rare species (and, indeed, other restoration practices) rely on an understanding of the ecological requirements of the species (Abeli & Dixon 2016). Tellingly, in an analysis of 249 plant species worldwide a lack of species biology knowledge was found to be a main cause of reintroduction failure (Godefroid et al., 2011). A lack of data on the pollination biology was concluded to have caused failure in previous orchid reintroduction attempts (Reiter et al. 2016) and there is strong evidence that an understanding of pollination ecology is also important for rare *Persoonia* species (Rymer et al., 2005).

Plants from known sources can be established in seed production ‘orchards’ to provide seed that is genetically diverse and representative of a robust population, and as an alternative to overharvesting wild populations (Nevill et al. 2016). This requires an agronomic approach for maintaining and harvesting and the ecological requirements of a species can be implemented to produce large quantities of high-quality seed for collection. In *Persoonia*, for example, manually hand-pollinating flowers to promote outcrossing could result in a higher fruit set of large quantities of seeds for restoration programs. Many situations may also require short- or long-term *ex situ* seed storage prior to restoration projects. In this regard, *ex situ* seedbanks provide an important supportive role. High-quality collections ensure that seeds are more robust for *ex situ* storage conditions.

Conclusions

There is great potential for *Persoonia* species to be successfully mass-propagated from seeds and included more widely in restoration and horticulture projects. Research to date has added several pieces to the puzzle; however, the focus on optimising germination success has meant that the ecological factors affecting this process have not been widely tested. We also stress that maximising germination does not necessarily translate into maximum seedling survival. Similarly, assessing one seed batch from a population does not provide any interpretation for that population’s health beyond the collection year. *Persoonia* propagation requires integrated collaboration between the restoration and horticulture industries with rigorous scientific research to achieve successful reintroduction and conservation practices. We have highlighted several key areas for future *Persoonia* seed research (summarised in Fig. 4). The ecological requirements of *Persoonia*, in terms of climate, plant-pollinator interactions, and seed biology, are important for obtaining sufficient quotas of high-quality seed to meet the growing needs of conservation, restoration and horticulture.

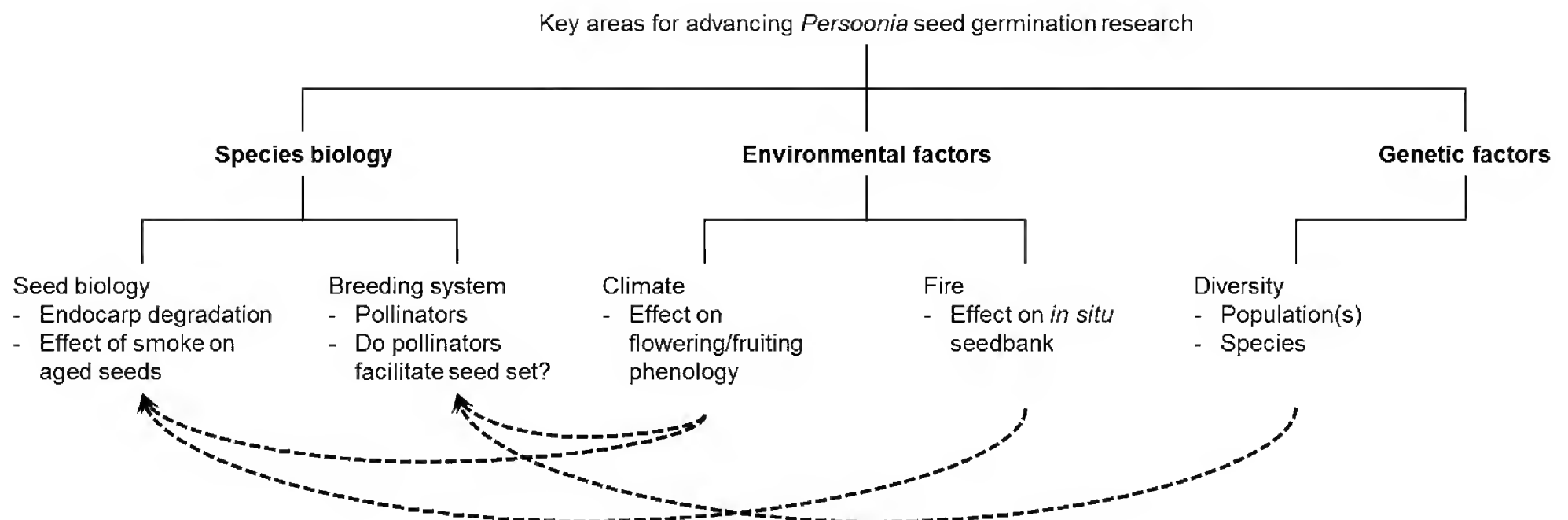


Fig. 4. Key areas that must be addressed for progressing *Persoonia* seed research. This requires an integrative approach, where data from environmental and genetic factors help to inform key questions surrounding seed biology and the breeding system.

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APPENDIX 1
Database table of biological and ecological species characteristics of 115 taxa (including all 99 species) of *Persoonia*.

Species	Distribution (state)	Bioregion ¹	Growth form	Longev (yrs) ²	Juv Period (yrs) ²	Habitat ³	Phylogeny ⁴	Rarity ⁵	Fire resp ⁶	Breed Syst ⁷	Fruit set ⁸	Ovule num ⁹	Germ ¹⁰
<i>Persoonia acerosa</i>	NSW	SEH, SYB	spreading shrub			DSF	Lanceolata	Vu*†	1			1	
<i>Persoonia acicularis</i>	WA	GES, YAL	erect shrub			MH	Quinquenervis	NL	2			1	
<i>Persoonia acuminata</i>	NSW	NET, NNC, SEH	prostrate to spreading shrub			WSF	Lanceolata	NL				1	
<i>Persoonia adenantha</i>	QLD, NSW	SEQ	erect shrub or small tree			DSF, H, RF, WSF	Lanceolata	NL	3			1	
<i>Persoonia amaliae</i>	QLD	BBN, CMC, MGD, SEQ	erect shrub to tree			DSF, VF	Lanceolata	NL	3			2	
<i>Persoonia angustiflora</i>	WA	AVW, COO, GES, JAF, SWA	erect shrub			DW, H	Quinquenervis	NL	2			1	
<i>Persoonia arborea</i>	VIC	AUA, NET, SEH	erect shrub or small tree			WSF	Arborea	NL	1			1	
<i>Persoonia asperula</i>	NSW, VIC	AUA, SEC, SEH	prostrate to erect shrub			H, WSF	Lanceolata	Th*	1			1	
<i>Persoonia baeckeoides</i>	WA	MAL	erect shrub			H	Lanceolata	NL				1	
<i>Persoonia bargoensis</i>	NSW	SYB	erect shrub			DSF, DW	Lanceolata	En*, Vu†	1	SC		1	
<i>Persoonia biglandulosa</i>	WA	GES, YAL	erect to decumbent shrub			H	Teretifolia	NL	2			2	
<i>Persoonia bowgada</i>	WA	CAR, GES, YAL	erect to spreading shrub			DW, H	Quinquenervis	NL				1	
<i>Persoonia brachystylis</i>	WA	GES	erect to spreading shrub			H	Teretifolia	NL				2	
<i>Persoonia brevifolia</i>	NSW, VIC	SEC, SEH	erect shrub			DSF, WSF	Lanceolata	NL				2	
<i>Persoonia brevirhachis</i>	WA	ESP, MAL	erect to spreading shrub			H	Rufiflora	NL				2	
<i>Persoonia chamaepeuce</i>	NSW, VIC	AUA, NET, NNC, NSS, RIV, SCP, SEC, SEH, SYB, SVP, VIM	prostrate shrub		4	DSF	Lanceolata	NL	2			1	
<i>Persoonia chamaepitys</i>	NSW	SCP, SEH, SYB	prostrate shrub			DSF, H	Lanceolata	NL	1			1	
<i>Persoonia chapmaniana</i>	WA	AVW, GES, SWA	erect shrub			DSF	Chapmaniana	NL				2	
<i>Persoonia comata</i>	WA	AVW, GES, JAF, SWA	erect to spreading shrub			DSF, DW, H	Teretifolia	NL	2			2	
<i>Persoonia confertiflora</i>	NSW, VIC	AUA, NSS, RIV, SCP, SEC, SEH	erect to decumbent shrub			DW, WSF	Laurina	NL	2			1	
<i>Persoonia conjuncta</i>	NSW	NNC	erect shrub or small tree			DSF, WSF	Lanceolata	NL	3			1	
<i>Persoonia cordifolia</i>	WA	MAL	erect shrub			H	Dillwynioides	NL	1			2	

Species	Distribution (state)	Bioregion ¹	Growth form	Longev (yrs) ²	Juv Period (yrs) ²	Habitat ³	Phylogeny ⁴	Rarity ⁵	Fire resp ⁶	Breed Syst ⁷	Fruit set ⁸	Ovule num ⁹	Germ ¹⁰
<i>Persoonia coriacea</i>	WA	AVW, COO, GES, GVD, JAF, MAL, MUR, YAL	erect to spreading shrub			H, MH	Lanceolata	NL	2			2	
<i>Persoonia cornifolia</i>	QLD, NSW	BBS, NAN, NET, NNC, SEQ, SYB	erect to spreading shrub			DSF, DW	Lanceolata	NL	2			1	
<i>Persoonia curvifolia</i>	NSW	BBS, COP, NET, NSS, RIV, SEH, SYB	erect to spreading shrub			DSF, DW	Lanceolata	NL	3			3	
<i>Persoonia cuspidifera</i>	NSW	BBS	erect shrub			DSF, DW	Lanceolata	NL	1			1	
<i>Persoonia cymbifolia</i>	WA	ESP, MAL	spreading shrub			H, RO	Lanceolata	NL	2			2	
<i>Persoonia daphnoides</i>	NSW	NET	prostrate shrub			DSF, DW	Lanceolata	NL	1			1	
<i>Persoonia dillwynioides</i>	WA	ESP	erect to spreading shrub			H	Dillwynioides	NL	2			2	
<i>Persoonia elliptica</i>	WA	ESP, JAF, SWA, WAR	erect shrub to small tree			DSF, DW	Lanceolata	NL	3		39%	2	
<i>Persoonia falcata</i>	QLD, NT, WA	ARC, ARP, BBN, BBS, CEA, CEK, CMC, CYP, DAB, DAC, DAL, DEU, EIU, GFU, GSD, GUC, GUP, MGD, NOK, OVP, PCK, TIW, VIB, WET	erect shrub to small tree			DSF, DW, H, RO	Teretifolia	NL	3			2	
<i>Persoonia fastigiata</i>	NSW	BBS, NET	erect to spreading shrub			DSF, DW	Lanceolata	NL	2			2	
<i>Persoonia filiformis</i>	WA	GES, SWA	erect shrub			H	Quinquenervis	NL	2			1	
<i>Persoonia flexifolia</i>	WA	ESP	erect to spreading shrub			H	Dillwynioides	NL	2			2	
<i>Persoonia glaucescens</i>	NSW	SYB	erect shrub			DSF	Lanceolata	En*, Vu†	1	NC	86%	1	
<i>Persoonia graminea</i>	WA	JAF, WAR	erect to decumbent shrub			DSF, H	Graminea	NL	1			2	
<i>Persoonia gunnii</i>	TAS	BEL, TCH, TSR, TWE	erect shrub			AH, RF, WSF	Gunnii	NL	2			2	
<i>Persoonia hakeiformis</i>	WA	AVW, MAL	erect to spreading shrub			DSF, H, MH	Teretifolia	NL	2			2	
<i>Persoonia helix</i>	WA	COO, ESP, MAL	erect to spreading shrub			DW, H, MH	Lanceolata	NL	2			2	
<i>Persoonia hexagona</i>	WA	AVW, GES, YAL	erect to spreading shrub			DW	Quinquenervis	NL	1			1	
<i>Persoonia hindii</i>	NSW	SB	small suckering shrub	indefinite		DSF	Lanceolata	En*	2			1	
<i>Persoonia hirsuta</i> subsp. <i>evoluta</i>	NSW	NSS, SB	spreading decumbent shrub			DSF	Lanceolata	En*†	1			1	

Species	Distribution (state)	Bioregion ¹	Growth form	Longev (yrs) ²	Juv Period (yrs) ²	Habitat ³	Phylogeny ⁴	Rarity ⁵	Fire resp ⁶	Breed Syst ⁷	Fruit set ⁸	Ovule num ⁹	Germ ¹⁰
<i>Persoonia hirsuta</i> subsp. <i>hirsuta</i>	NSW	NSS, SB	spreading decumbant shrub			DSF	Lanceolata	En*†	1			1	
<i>Persoonia inconspicua</i>	WA	AVW, COO	erect to spreading shrub			H, MH	Ruffiflora	NL				2	
<i>Persoonia iogyna</i>	QLD	SEQ	erect shrub to small tree			DSF, WSF	Lanceolata	NL				1	
<i>Persoonia isophylla</i>	NSW	SYB	erect to spreading shrub			DW	Lanceolata	NL	1			1	
<i>Persoonia juniperina</i>	NSW, SA, TAS, VIC	BEL, FLB, FUR, KAN, KIN, MDD, NCP, SCP, SEC, SEH, SVP, TCH, TNM, TNS, TSE, TSR, TWE, VIM	erect to spreading shrub			DSF, H	Lanceolata	NL	2	SC	83%	2	
<i>Persoonia kararae</i>	WA	YAL	erect shrub			S	Teretifolia	NL				2	
<i>Persoonia katera</i>	NSW	NNC	erect shrub or small tree			DSF, H	Lanceolata	NL					
<i>Persoonia lanceolata</i>	NSW	NET, NNC, SEC, SEH, SEQ, SYB	erect to spreading shrub	25—60	6—8	DSF	Lanceolata	NL	1	NC	97%	2	~10%
<i>Persoonia laurina</i> subsp. <i>intermedia</i>	NSW	SEH, SYB	erect to spreading shrub	25—60		DSF	Lanceolata	NL	2			1	
<i>Persoonia laurina</i> subsp. <i>laurina</i>	NSW	NNC, NSS, SEH, SYB	erect to spreading shrub	25—60		DSF	Lanceolata	NL	2			1	
<i>Persoonia laurina</i> subsp. <i>leiogyna</i>	NSW	SEH, SYB	erect to spreading shrub	50—100		DSF	Laurina	NL	2			1	
<i>Persoonia laxa</i>	NSW	SYB	ground-dwelling shrub			DSF	Lanceolata	NL				2	
<i>Persoonia leucopogon</i>	WA	COO, MUR	erect to decumbent shrub			H	Lanceolata	NL				2	
<i>Persoonia levis</i>	NSW, VIC	NNC, SEC, SEH, SEQ, SYB	erect shrub or small tree	>60		DSF, DW	Lanceolata	NL	3	NC	55%	2	50%
<i>Persoonia linearis</i>	NSW, VIC	BBS, NET, NNC, NSS, SEC, SEH, SEQ, SYB	erect shrub or small tree			DSF, DW, WSF,	Lanceolata	NL	3			2	
<i>Persoonia longifolia</i>	WA	ESP, JAF, SWA, WAR	erect shrub or small tree			DSF, DW	Longifolia	NL	3		98%	2	95%
<i>Persoonia manotricha</i>	WA	AVW, MUR, YAL	erect shrub or small tree			RO, S	Quinquenervis	NL				2	
<i>Persoonia marginata</i>	NSW	SEH, MDD	spreading to decumbent shrub			DSF	Lanceolata	NL	2			2	
<i>Persoonia media</i>	QLD, NSW	NET, NNC, SEQ	erect to spreading shrub	20	6	RF, WSF	Lanceolata	NL	2			1	

Species	Distribution (state)	Bioregion ¹	Growth form	Longev (yrs) ²	Juv Period (yrs) ²	Habitat ³	Phylogeny ⁴	Rarity ⁵	Fire resp ⁶	Breed Syst ⁷	Fruit set ⁸	Ovule num ⁹	Germ ¹⁰
<i>Persoonia micranthera</i>	WA	ESP	prostrate to decumbent shrub			H, RO	Graminea	Cr* En†	1			2	
<i>Persoonia microphylla</i>	NSW	SEC, SEH, SYB	prostrate to erect shrub			DSF	Lanceolata	NL				1	
<i>Persoonia mollis</i> subsp. <i>caleyi</i>	NSW	SEC, SEH, SYB	prostrate to erect shrub			DSF	Lanceolata	NL	1			2	
<i>Persoonia mollis</i> subsp. <i>ledifolia</i>	NSW	SEC, SEH, SYB	erect shrub			DSF	Lanceolata	NL	1			2	
<i>Persoonia mollis</i> subsp. <i>leptophylla</i>	NSW	SEH, SYB	erect shrub			DW	Lanceolata	NL	1			2	
<i>Persoonia mollis</i> subsp. <i>Ivrens</i>	NSW	SEH, SYB	erect shrub			DSF, DW	Lanceolata	NL	1			2	
<i>Persoonia mollis</i> subsp. <i>maxima</i>	NSW	SYB	erect shrub		10	DSF	Lanceolata	En*†	1	NC	89%	2	
<i>Persoonia mollis</i> subsp. <i>mollis</i>	NSW	SEH, SYB	prostrate to erect shrub			DSF	Lanceolata	NL	1			2	
<i>Persoonia mollis</i> subsp. <i>nectens</i>	NSW	SYB	prostrate to erect shrub		>4	DSF	Lanceolata	NL	1	NC	91%	2	~40%
<i>Persoonia mollis</i> subsp. <i>revoluta</i>	NSW	SEC, SEH, SYB	prostrate to decumbent shrub			DSF	Lanceolata	NL	1			2	
<i>Persoonia moscalii</i>	TAS	TWE	prostrate shrub			AH, RF, WSF	Gunnii	NL				2	
<i>Persoonia muelleri</i> subsp. <i>angustifolia</i>	TAS	TCH, TNS, TSR, TWE	erect shrub			AH, RF	Gunnii	NL				2	
<i>Persoonia muelleri</i> subsp. <i>densifolia</i>	TAS	TSR, TWE	erect shrub to small tree			H	Gunnii	NL				2	
<i>Persoonia muelleri</i> subsp. <i>muelleri</i>	TAS	BEL, TCH, TNS, TSE, TSR, TWE	erect shrub to small tree			AH, WSF	Gunnii	NL				2	
<i>Persoonia myrtilloides</i> subsp. <i>cunninghamii</i>	NSW	NSS, SYD	erect to spreading shrub			DSF, DW	Lanceolata	NL	1	NC	70%	2	
<i>Persoonia myrtilloides</i> subsp. <i>myrtilloides</i>	NSW	SEC, SEH, SYD	erect to spreading shrub			DSF, DW, H	Lanceolata	NL	1	NC		2	
<i>Persoonia nutans</i>	NSW	SYB	erect to spreading shrub			DW	Lanceolata	En*†	1			2	
<i>Persoonia oblongata</i>	NSW	NSS, SEH, SYB	erect to spreading shrub			DSF, DW, RO	Lanceolata	NL	1			2	
<i>Persoonia oleoides</i>	NSW	NET, NNC	erect to spreading shrub			DSF, WSF	Lanceolata	NL	2			1	
<i>Persoonia oxycoccoides</i>	NSW	NSS, SEC, SEH, SYB	prostrate to spreading shrub			H, DSF	Lanceolata	NL	2			1	

Species	Distribution (state)	Bioregion ¹	Growth form	Longev (yrs) ²	Juv Period (yrs) ²	Habitat ³	Phylogeny ⁴	Rarity ⁵	Fire resp ⁶	Breed Syst ⁷	Fruit set ⁸	Ovule num ⁹	Germ ¹⁰
<i>Persoonia papillosa</i>	WA	GES	erect shrub			S	Quinquenervis	NL				1	
<i>Persoonia pauciflora</i>	NSW	SYB	spreading shrub			DSF, DW	Lanceolata	Cr*†	1			1	
<i>Persoonia pentasticha</i>	WA	AVW, GES, MUR, YAL	erect to spreading shrub			H	Chapmaniana	NL	2			2	
<i>Persoonia pertinax</i>	WA	COO, GVD	erect to spreading shrub			DW	Lanceolata	NL				2	
<i>Persoonia pinifolia</i>	NSW	SYB	erect shrub or small tree	25—60	5	DSF	Lanceolata	NL	1			1	
<i>Persoonia procumbens</i>	NSW	NET, NNC	prostrate shrub			DSF, DW	Lanceolata	NL				1	
<i>Persoonia prostrata</i>	QLD	SEQ	prostrate shrub			DSF, DW, H, S	Lanceolata	Ex					
<i>Persoonia pungens</i>	WA	AVW, GES	spreading to decumbent shrub			H	Lanceolata	NL	2			1	
<i>Persoonia quinquenervis</i>	WA	AVW, COO, ESP, GES, JAF, MAL, SWA	erect to spreading shrub			DSF, DW, H, MH	Quinquenervis	NL	2			1	
<i>Persoonia recedens</i>	NSW	SYB	prostrate to spreading shrub			DSF	Lanceolata	NL				1	
<i>Persoonia rigida</i>	NSW, VIC	AUA, BBS, NSS, SEC, SEH, SYB, RIV, VIM	erect to decumbent shrub			DSF	Lanceolata	NL	1	SC	67%	2	
<i>Persoonia rudis</i>	WA	AVW, GES, SWA	erect shrub			DSF	Quinquenervis	NL	1			1	
<i>Persoonia rufa</i>	NSW	NET, NNC	erect to spreading shrub			DSF, RO	Lanceolata	NL	1			1	
<i>Persoonia rufflora</i>	WA	AVW, GES, JAF, MAL, SWA	erect to spreading shrub			DW, H, MH	Rufflora	NL	2			2	
<i>Persoonia saccata</i>	WA	JAF, SWA, WAR	erect to spreading shrub			DSF, DW	Teretifolia	NL	2			2	
<i>Persoonia saundersiana</i>	WA	AVW, COO, GES, MAL, MUR, SWA, YAL	erect to spreading shrub			H, MH	Teretifolia	NL				2	
<i>Persoonia scabra</i>	WA	ESP, MAL	erect to spreading shrub			MH	Quinquenervis	NL	2			1	
<i>Persoonia sericea</i>	QLD, NSW	BBS, DRP, NAN, NET, NNC, NSS, SEQ, SYB	erect to spreading shrub			DSF, WSF	Lanceolata	NL	2			2	88%
<i>Persoonia silvatica</i>	NSW, VIC	SEC, SEH	erect shrub or small tree			DSF	Laurina	NL	2			1	
<i>Persoonia spathulata</i>	WA	ESP, MAL	erect to spreading shrub			H	Quinquenervis	NL	1			1	
<i>Persoonia stradbrokeensis</i>	QLD, NSW	NNC, SEQ	erect shrub to small tree			DSF	Lanceolata	NL	3			1	

Species	Distribution (state)	Bioregion ¹	Growth form	Longev (yrs) ²	Juv Period (yrs) ²	Habitat ³	Phylogeny ⁴	Rarity ⁵	Fire resp ⁶	Breed Syst ⁷	Fruit set ⁸	Ovule num ⁹	Germ ¹⁰
<i>Persoonia striata</i>	WA	AVW, COO, ESP, JAF, MAL	erect to spreading shrub			H, MH	Quinquenervis	NL	2			1	
<i>Persoonia stricta</i>	WA	AVW, GES, MUR, YAL	spreading shrub			DW	Teretifolia	NL				2	
<i>Persoonia subtilis</i>	NSW	BBN, BBS	spreading to decumbent shrub			DSF, DW	Lanceolata	NL	2			2	
<i>Persoonia subvelutina</i>	ACT, NSW, VIC	AUA, NSS, SEH	spreading shrub or small tree			DW, WSF	Arborea	NL				1	
<i>Persoonia sulcata</i>	WA	AVW, JAF, SWA	erect to spreading shrub			DW, RO	Quinquenervis	NL	2			2	
<i>Persoonia tenuifolia</i>	QLD, NSW	BBS, NAN, NET, NNC, SEQ	erect to decumbent shrub			DSF, H	Lanceolata	NL	2			2	
<i>Persoonia teretifolia</i>	WA	AVW, ESP, JAF, MAL	erect to spreading shrub			H, MH	Teretifolia	NL				2	
<i>Persoonia terminalis</i> subsp. <i>recurva</i>	QLD, NSW	BBS, NET	erect to spreading shrub			DSF	Lanceolata	NL	1			2	
<i>Persoonia terminalis</i> subsp. <i>terminalis</i>	QLD, NSW	NET	erect to spreading shrub			DSF	Lanceolata	NL	1			2	
<i>Persoonia trinervis</i>	WA	AVW, ESP, GES, JAF, MAL, SWA	erect to spreading shrub			DW, H, MH	Quinquenervis	NL	2			1	
<i>Persoonia tropica</i>	QLD	EIU, WET	erect shrub or small tree			DSF, WSF	Lanceolata	NL				2	
<i>Persoonia virgata</i>	QLD, NSW	BBN, BBS, CMC, NNC, SEQ	erect shrub			DSF	Lanceolata	NL	1	NC	49%	2	100%
<i>Persoonia volcanica</i>	QLD, NSW	SEQ	erect shrub			DSF, WSF, RF	Lanceolata	NL				1	

¹ Bioregions were extracted from Australia's Virtual Herbarium (avh.ala.org.au) on 23/04/2018; the list IBRA7 region codes and names are available from <http://www.environment.gov.au/land/nrs/science/ibra/ibra7-codes> (accessed on 23/04/2018)

² Obtained from Benson & McDougall (2000)

³ AH = alpine heath, DSF = dry sclerophyll forest, DW = dry woodland, H = heath, MH = mallee heath, RF = rainforest, RO = rocky outcrops, S = sandplains, WSF = wet sclerophyll forest, VF = vine forest

⁴ Obtained from Weston (2003)

⁵ Cr = critically endangered, En = endangered, Ex = extinct, NL = not listed under state or national legislation, Th = threatened, Vu = vulnerable; * threatened status listed under state/territory legislation; † threatened status listed under the national EPBC Act

⁶ Ability to resprout following fire: 1 = cannot resprout and reliant on seeds, 2 = can resprout from base only, 3 = can resprout from base and stems; Rymers (2006)

⁷ NC = non-compatible breeding system, SC = self-compatible breeding system; reference list available in Appendix 2

⁸ Highest percentage fruit set from open pollination as of 23/04/2018, reference list available in Appendix 2

⁹ Obtained from Rymers (2006)

¹⁰ Highest germination value reported in the literature as of 23/04/2018; reference list available in Appendix 2

Appendix 2

References for the data on *Persoonia* breeding system, fruit set and germination listed in Table 1 and Appendix 1.

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***Washingtonia robusta* (Mexican Fan Palm) as a coloniser in an artificial wetland at Albury, New South Wales**

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Abstract: *Washingtonia robusta* (Mexican Fan palm) is endemic to the semi-arid zone of California and northern Mexico. Dispersed globally by the horticultural trade, the species has demonstrated its ability to successfully invade disturbed areas and urban landscapes in warm temperate climates. Once established, the plant is extremely hardy. This paper presents the first documented instance of the successful establishment and growth of *Washingtonia robusta* in a pond in continually flooded wetlands at Albury, the first record of it naturalising in New South Wales.

Keywords: Weeds in wetlands—*Washingtonia robusta*—*Phoenix canariensis*—frugivory—dispersal of exotic palms—adaptation

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Introduction

Numerous plants have shown to be adaptable to changing and novel environmental conditions. Many of the exotics have become invasive at the expense of local vegetation. Birds in particular have often adapted to and taken advantage of these exotics to broaden their feeding habits. Ornamental palms such as *Phoenix canariensis*, *Washingtonia filifera* and *Washingtonia robusta* while initially confined to anthropogenic landscapes such as gardens, parks and streets, have been able to spread via birds and animals well beyond the confines of urban environments.

Introduced to the European nursery trade in the 1860s (Bailey, 1936, p. 63ff; Ishihata & Murata, 1971), and to Australia in the 1870s and 1880s, the Mexican fan palm or Desert Palm, *Washingtonia robusta* has become a major ornamental plant on a global scale, widely planted as a feature tree in private and public gardens, as well as an occasional street tree in many communities with a temperate climate (Spennemann, 2018b). Though much less invasive than other palm species, such as *Phoenix canariensis*, *Washingtonia robusta* is known to escape from horticultural settings and colonise natural areas. Endemic to semi-arid regions in northwest mainland Mexico and the southern Baja California (Cornett, 1989; McCurrach, 1960, p. 264f), *Washingtonia robusta* is regarded as naturalised not only in adjacent southern California and southern Arizona (Felger & Joyal, 1999), but also in subtropical areas such as southern Florida (Cornett, Stewart, & Glenn, 1986; Institute for Regional Conservation, 2016), Réunion (Indian Ocean) (Meyer, Lavergne, & Hodel, 2008), the North Island of New Zealand (Martin, 2009), and parts of Hawaii (Oppenheimer & Barlett, 2002). On the Australian continent it is regarded as naturalised in the Pilbara region, Western Australia (Pilbara Region, Keighery, 2010); evidence from Albury (NSW, Australia) presented in this paper demonstrates that it must also be regarded as naturalised in NSW.

Little is known about the dispersal potential of *Washingtonia robusta*. A review of dispersal vectors showed a number of vertebrate species feed on *Washingtonia* drupes, with some ingesting whole drupes and dispersing the seeds via regurgitation or defecation, in the Australian setting including Pied Currawongs (*Strepera graculina*) and fruit bats (*Pteropus poliocephalus*) (Spennemann, 2018c, 2018d). *Washingtonia robusta* have been noted as self-seeded plants in garden settings in Albury and Sydney (pers. obs.), but unlike *Phoenix canariensis*, *Washingtonia* do not seem to readily invade remnant bushland or agricultural areas. It is possible that their germination and initial establishment success, more so than that of *Phoenix canariensis*, relies on adequate moisture regularly provided in suburban gardens but which is seasonal in bushland settings. If such moisture is provided, it can establish in very marginal places such as cracks in the concrete pavements (Martin, 2009; Stein, 2010) and even in cavities in trees (Spennemann, 2018g). The long-term success of the seedlings is dependent on stable nutrient availability as well as lack of human intervention. Other factors that seem to influence establishment success in suburban gardens and peri-urban areas are the absence of grazing animals (primarily sheep and kangaroos), more

friable and less compacted clayey soil, and property owners who tend to be tolerant of such exotic adventitious species.

In its natural habitat, *Washingtonia robusta* is strictly a plant of the semi-arid zone confined to a small area of the Sonora desert and the southern Baja California (Bomhard, 1950; Cornett, 1989, p. 94; Felger & Joyal, 1999; McCurrach, 1960, p. 264f; Wiggins, 1964). It occurs naturally in annually or irregularly flooded riparian habitats and palm oases, with the plants concentrated closest to water sources, on occasions near, but not in, permanent water or wet soil (Felger & Joyal, 1999). In Southern California, it has colonised riparian strips and aided by its seed shadow has crowded out other plants. On occasion the palm thickets have become so dense that some grew at the very stream edge with their trunks just in the water (Kelly, 2007). *Washingtonia robusta* has not been recorded as growing in the middle of ponds, lakes or flowing bodies of water in California or Mexico. It was therefore surprising to find plants growing in 1.5m deep water in the middle of the eastern maturation pond of Albury's waste water treatment works with no evidence of such palms growing in the adjacent bushland. This paper describes the distribution and nature of these plants and the possible vectors and mechanisms facilitating colonisation success.

Biology of *Washingtonia robusta*

Washingtonia robusta Wendl. (Mexican Fan Palm) are monoecious, self-compatible plants that are solely propagated by seed (Barrow, 1998). They can have up to 30+ bright green, costapalmate fronds of 0.9 to 1.8 m with which are attached to the tree, each with a 1.2–1.5 m long plano-convex petiole. The red brown petiole exhibits curved spines on its entire length (Bailey, 1936, p. 63ff). A mature *Washingtonia robusta* will produce about 50 leaves annually (Brown & Brown, 2012). The trunk is usually covered with persistent dead fronds hanging from the crown, which, unless horticulturally removed, form a thick thatch ('skirt') surrounding the upper section of the trunk (Moran, 1978; Morton, 1998).

Washingtonia robusta typically grows to a height of 15–20 m with a trunk diameter of 0.6–1.2 m (up to 1.5m) (Broschat, 2017; Morton, 1998). The majority of palms will reach ages of less than 200 years, though specimens of 500 years have been estimated (Bullock & Heath, 2006). They reach maturity after they have reached at least 3m in height (Cornett cited in Martus, 2008, p. 25). The flower stalk ranges in length from 2 to 2.6 m, bearing numerous small, white bisexual flowers in compound clusters (Felger & Joyal, 1999). These result in infructescences (fruiting sprays) which ripen in autumn to early winter. Each drupe is small, black and ovoid-oblong to spherical fruit with a thin non-oily, carbohydrate rich pericarp and a single hemispherical seed (Brown & Brown, 2012). The drupes measure about 7–10mm in length with an average weight of 0.3g, while the seeds are about 4.7–6.5mm long and about 4.5–4.9mm thick with an average weight of about 0.1 g (Spennemann, 2018f). A single mature tree has been estimated to produce in excess of 300,000 drupes per year.

Washingtonia robusta seed germinate well within 14 days at soil temperatures of 25–35°C (Broschat, 2017; Brown & Brown, 2012; Mifsud, 1996). The fruit are eaten by a range of volant as well as terrestrial vectors. Passage through the gastro-intestinal tract enhances germination success of *Washingtonia filifera*, probably due to scarification by weak acids (Noto & Romano, 1987) and this is likely to be the case with *Washingtonia robusta* although no data was found on germination success. *Washingtonia filifera*, seeds and seedlings are allelopathic (Khan, 1982a, 1982b), giving the plant seedling a competitive edge over other vegetation, and this may also be the case with *Washingtonia robusta* although data has not been found. Once established, both species will do well with comparatively little water. *Washingtonia robusta* grows at the rate of about 0.6–0.9 m per year, but if well-watered, the palms reach annual growth rates of 1.8m, at least in the early stages (Morton, 1998; Muirhead, 1961, p. 41). Not well-watered *Washingtonia robusta* are on record as reaching 20m height under 30 years (Proschowsky, 1921).

Albury study location

The artificial wetlands under discussion, the Kremur Street Sewerage Treatment Plant of Albury consists of two artificially created maturation ponds and a series of downstream wetlands that now form part of the Horseshoe Lagoon billabong system, located on the northern side of the Murray River floodplain, some 2.5 km west of the centre of Albury. While Horseshoe Lagoon and associated billabongs are a natural system, the two maturation ponds are artificial water bodies created by erecting retainment embankments and flooding a paddock previously used for grazing.

After alienation from the Indigenous owners of the land in the mid-1830s, the area first formed part of the grazing lease ‘Bungowannah Station’ (run n° 40, Thomson, 1848), after which the area close to Albury was reserved as part of the Albury Temporary Common (notified on 26 June 1868, Wilson, 1868). The land covering both maturation ponds as well as the sewerage works was then set aside for police purposes (‘police paddock’) on 21 November 1871 (Reserve N° 853, Wilson, 1871) as revised on 6 October 1900 (Reserve N° R 31,592, Hassall, 1900a, 1900b, 1900c). Given the existence of the wetlands, the area was converted into an area reserved for the protection of birds in January 1916 (Black, 1916). The eastern section, which comprises the vast majority of the first maturation pond, forms part of land set aside in March 1917 for the Albury Sewerage Works (Department of Lands, 1927; Strickland, 1917). The Kremur Street sewerage works were opened in May 1919 (Anonymous, 1919), with the maturation ponds constructed in the 1950s or early 1960s (Johnson, 2018). The plant was modified into a biological nutrient removal plant in late 1984 or early 1985 (Johnson, 2018) with further expansion of the scheme in the late 1990s (Abbey, 1994).

Aerial imagery of the Kremur Street sewerage works, taken in May 1949, shows the area as pasture with scattered eucalypts primarily along a drainage line, as well as a market garden in the east (Adastra Airways, 1949; Spennemann,

2018e). Today, evidence for the previous land use abounds in the form of dead eucalypts in the maturation ponds. Judging from the state of preservation of the dead trees (proportion of large vs small limbs vs. small branches), some of the trees were long dead well before the area was converted into maturation ponds, but the majority died off due to permanently water-logged soil following the flooding.

Setting aside rain water and minimal surface run-off, the ponds are only supplied with treated, screened and filtered wastewater from the Kremur Street plant. The water levels in the maturation ponds, which have not been drained since their construction, are more or less constant as they are regulated by an outlet weir at the north-western end of maturation pond N° 2 (Johnson, 2018). Any fluctuations in water levels are limited to a variation caused by evaporation. The bottom sections of the ‘skirts’ of *Washingtonia robusta* exhibit a clean edge, confirming a more or less steady water level.

Results

Survey of Washingtonia robusta in the Albury wetlands

In total, seven *Washingtonia robusta* palms were identified in maturation pond n° 1, three in maturation pond n° 2 and a group of palms on the southern peninsula dividing the ponds (Figure 1) (see Spennemann, 2018e for photographic documentation). Reference specimens were collected from a cluster of self-seeded plants on the peninsula and deposited in the CSU Regional Herbarium (see Spennemann, 2018a for documentation).

The location and current appearance of the palms, as determined by ground survey, are summarised in Table 1. The plants range from an almost 11m tall mature plant (P2, Figure 6) on the peninsula, to small seedlings of 1m height on the peninsula (P4) and small established plants in the ponds (Figure 3–Figure 5). One dead palm (W2) was also encountered in the western maturation pond (Figure 7). The plants in the maturation ponds were documented and photographed from the shoreline from various directions (Spennemann, 2018e). The plants growing in the ponds range in height from 2m to about 5m (Table 1).

Presence and dimensions over time are based on the interpretation of historic aerial imagery of varied quality, primarily sourced from the commercial service Nearmap™, from Albury City’s Mapping Portal (which includes custom-shot aerial imagery) and from GoogleEarth™. The quality of the images depends on the resolution of the aerial or satellite imagery, and the presence of cloud cover or sun reflections off the waterbodies (Table 2).

The crown diameters of the palms were classed in quarter-metre intervals. The observed size fluctuations are due to the variations in the aerial images and the accuracy of the embedded scales. Identification of *Washingtonia* on the aerial images was carried out in two ways: i) backtracking the presently existing palms in time, noting their presence and any changes to diameter; and ii) the identification of additional *Washingtonia*. This was possible as the foliage of

Washingtonia stood out as a very light green tone compared to the other small vegetation islands in the pond, and the presence of a shadow. The latter differentiates *Washingtonia* from sedge islands. The data compiled in Table 2 are expressed as timelines in Figure 2.

The mature tree (P2) is by far the oldest. As it is growing among eucalypts, it is not always readily distinguishable on the aerial images, especially at its smaller sizes when it would have been sufficiently obscured by the canopy of the eucalypts. The palm shows above / in the canopy from about 2010 onwards, which suggests that by that time the plant would have been about 5m tall (compare Figure 8).

Washingtonia robusta reputedly grows at about 0.6–0.9 m per year, once established (at ca 2yrs of age). If well-watered, the palm can, at least in the early stages, reach annual

growth rates of 1.8m (Morton, 1998; Muirhead, 1961, p. 41; Proschowsky, 1921). Considering that the moisture regime at the site is not subject to climatic variation, but stable due the constantly flooded maturation ponds, the height gain of the plant between 2010 and 2018 suggests that plant P2 grew between 0.6 m and 0.7m per year. Projecting this growth rate back in time suggests that the plant seeded sometime in 2001 or 2002. The other six plants for which height data and aerial imagery exists (P1–P5, W1), grew at rates between 0.4 m and 0.55 m pa.



Fig. 1. Locations of the self-seeded palms shown on an aerial view of the maturation ponds of the Kremur Street Treatment Plant, Albury NSW. Aerial image flown 8 November 2017, image courtesy AlburyCity.

Table 1. Location, nature, dimensions and distance to source palm of the documented *Washingtonia* and *Phoenix* Palms at Kremur Street wetlands, Albury

Specimen	Status	Stem	Height (m)	Crown (m)	Setting	Distance (m)	Latitude	Longitude
<i>Washingtonia robusta</i>								
East Pond n° 1	immature	yes	3.5	2.5	ex perch	318	-36.085342	146.886768
East Pond n° 2	immature	yes	2.5	2.5	perch	259	-36.085306	146.88608
East Pond n° 3	immature	yes	3.5	2.5	perch	227	-36.084915	146.885858
East Pond n° 4a	immature	yes	4–5	2.5	low perch	195	-36.08478	146.885544
East Pond n° 4b	immature	no	2	2.5	—	194	-36.08478	146.885544
East Pond n° 5	immature	yes	3.5	2.5	perch	213	-36.085925	146.884853
East Pond n° 6a		not longer extant			perch	112	-36.084781	146.884594
East Pond n° 6b		not longer extant			perch	112	-36.084781	146.884594
East Pond n° 6c	immature	no	2.5	2.5	perch	112	-36.084781	146.884594
East Pond n° 7	immature	no	2.5	2.5	perch	123	-36.084781	146.884594
Peninsula n° 1a	immature	no	2	indet.	palm	4	-36.084411	146.883437
Peninsula n° 1b	immature	no	2	indet.	palm	4	-36.084411	146.883437
Peninsula n° 1c	immature	no	2	indet.	palm	3	-36.084411	146.883437
Peninsula n° 1d	immature	no	2	indet.	palm	2	-36.084411	146.883437
Peninsula n° 1e	immature	no	2	indet.	palm	2	-36.084411	146.883437
Peninsula n° 2	mature	yes	10-11	3-4	ex-perch	0	-36.084382	146.883500
Peninsula n° 3a	immature	no	1	indet.	palm	1	-36.084353	146.883416
Peninsula n° 3b	immature	no	1	indet.	palm	1	-36.084353	146.883416
Peninsula n° 3c	immature	no	2	indet.	palm	2	-36.084353	146.883416
Peninsula n° 3d	immature	no	2	indet.	palm	2	-36.084353	146.883416
Peninsula n° 3e	immature	no	2	indet.	palm	3	-36.084353	146.883416
Peninsula n° 3f	immature	no	2	indet.	palm	3	-36.084353	146.883416
Peninsula n° 3g	immature	no	2	indet.	palm	4	-36.084353	146.883416
Peninsula n° 4	immature	no	1	1.5	perch?	11	-36.084302	146.883496
West Pond n° 1	immature	yes	2	2.5	perch	135	-36.083452	146.88245
West Pond n° 2	dead	no	0.75	2	perch	232	-36.082616	146.882024
West Pond n° 3		not longer extant			perch?	123	-36.083744	146.882298
<i>Phoenix canariensis</i>								
Canariensis n° 1	mature, male	yes	4	5	no perch	n/a	-36.082963	146.886351
Canariensis n° 2	mature, male	yes	4	5	no perch	n/a	-36.082959	146.886461
Canariensis n° 3	immature	no	1	1	perch	n/a	-36.08282	146.888867
Canariensis n° 4	immature	no	1.5	2	perch	n/a	-36.082806	146.8888
Canariensis n° 5	immature	no	3	4	perch	n/a	-36.082242	146.88844

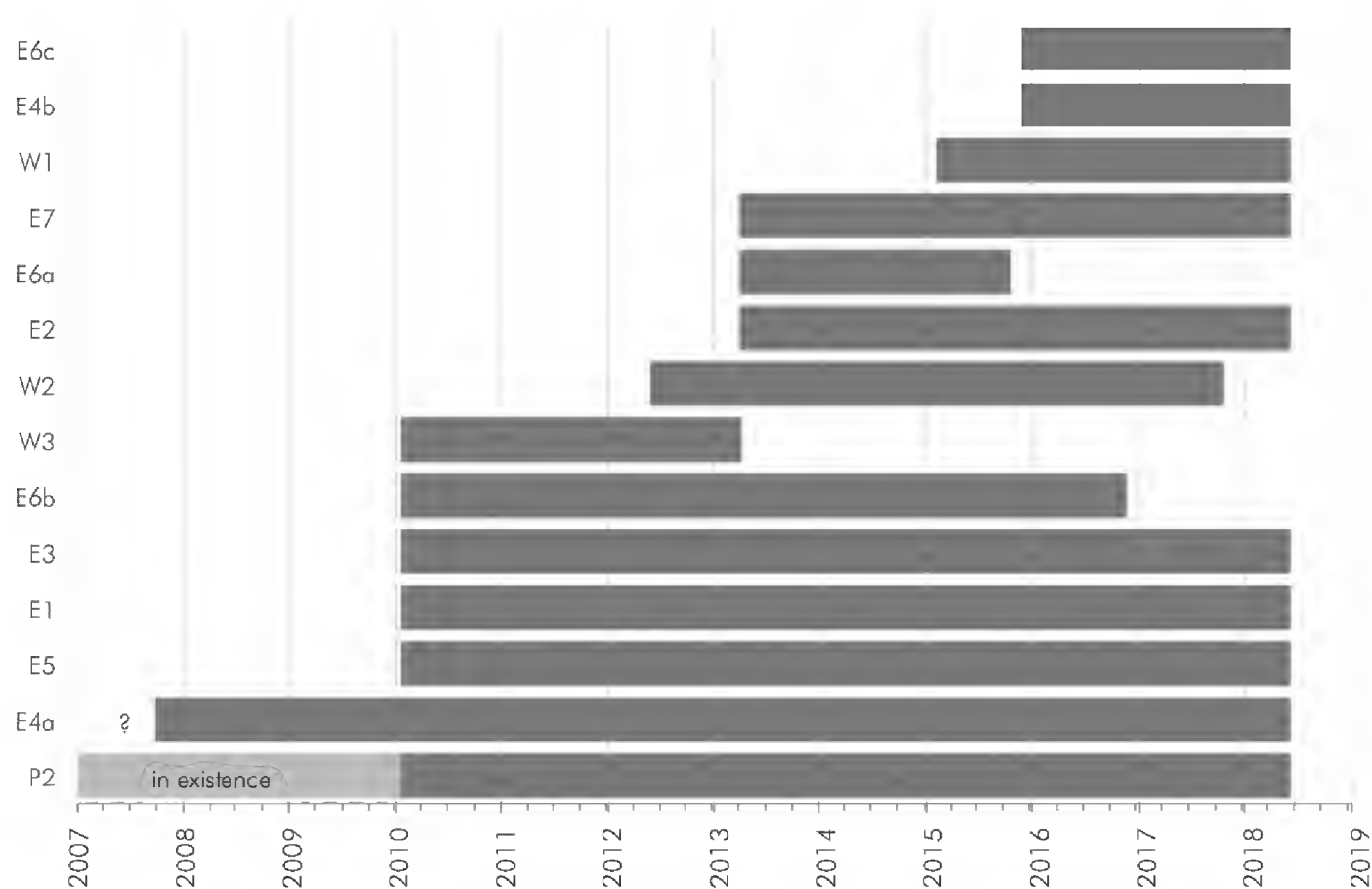


Fig. 2. Chronology of *Washingtonia* palms in the Kremur Street wetlands.

Table 2. Crown diameters (in m) of *Washingtonia* in Kremur Street wetlands based on the interpretation of aerial imagery and ground inspection (1 June 2018).

Date	E1	E2	E3	E4a	E4b	E5	E6a	E6b	E6c	E7	P1	P2	P3	P4	W1	W2	W3	Image Quality	Image
2018, June 1	3	3	3.5	3.5	2	3.5	—	—	2	2	2	3.5	2	0.5	3	dead	dead	./.	Site Visit
2018, May 1	3	3	3.5	3.5	2	3.25	—	—	2	2	invis	3.5	invis	invis	3	dead	dead	very good	NearMap
2018, Mar 11	3	3	3.5	3.5	2	3	—	—	2	1.75	invis	3.5	invis	invis	3	dead	dead	very good	NearMap
2018, Jan 20	3.5	3	3.5	3.5	1.5	3	—	—	2.5	1.75	invis	3.5	invis	invis	3	dead	dead	very good	NearMap
2017, Dec 22	obs	3	3.5	3.5	1	3	—	—	2	—	invis	3.5	invis	invis	obs	dead?	dead	partly obscured	GoogleEarth
2017, Nov 8	3.5	3.25	4.0	3.5	1	2.75	—	—	1	1.75	invis	3.5	invis	invis	3	dead?	dead	very good	AlburyCity
2017, Oct 10	3	3	3.5	3.5	1	2.75	—	—	2	1.75	invis	3.5	invis	invis	2.5	2	dead	partly obscured	NearMap
2017, Jun 13	3	3	3.5	3.5	1	2.75	—	—	1.5	1.75	invis	3.5	invis	invis	2.5	3	dead	very good	NearMap
2016, Nov 18	2.75	2.75	3.5	3.5	1	2.75	—	2	1.5	1.75	invis	pres	invis	invis	2.25	2.5	dead	very good	NearMap
2015, Nov 30	3	2.75	3.25	3.5	1	3	—	2	1.5	1.75	invis	pres	invis	invis	2	2.5	dead	partly obscured	NearMap
2015, Oct 15	3.25	2.5	3.25	3.5	—	3	1.75	pres?	—	pres	invis	pres	invis	invis	1.5	2.5	dead	very good	AlburyCity
2015, Feb 9	2.75	2.5	2.75	3.5	—	3	1.5	1.5	—	1.5	invis	pres	invis	invis	pres	2.5	dead	very good	NearMap
2014, Aug 20	2.5	2.5	2.75	3.5	—	2.75	2	2	—	1	invis	pres	invis	invis	—	2.5	dead?	very good	NearMap
2014, Feb 22	2.5	2.25	2.5	3.5	—	2.75	2.5	2.5	—	pres	invis	pres	invis	invis	—	2	dead?	very good	(LPI, 2014)
2014, Feb 2	2.5	2.5	2.5	3.25	—	2.75	2	2.5	—	1	invis	pres	invis	invis	—	2	dead?	very good	NearMap
2013, Apr 4	2.5	pres	2.5	3.25	—	2.75	pres?	2.5	—	pres	invis	pres	invis	invis	—	2	3	very good	NearMap
2012, Oct 5	2.5	—	2.0	3.25	—	2.75	—	2.5	—	—	invis	pres	invis	invis	—	2	3	very good	NearMap
2012, May 29	2.25	—	1.75	3.25	—	2.75	—	2.5	—	—	invis	pres	invis	invis	—	1.5	2.75	very good	NearMap
2011, Oct 2	1.75	—	1.5	3.25	—	3.0	—	2.25	—	—	invis	pres	invis	invis	—	—	2.5	good	AlburyCity
2011, Apr 16	1.75	—	1.25	3.25	—	2.75	—	2	—	—	invis	pres	invis	invis	—	—	2.5	very good	NearMap
2010, Nov 17	1.5	—	obs	2.75	—	2.5	—	pres	—	—	invis	pres	invis	invis	—	—	2.5	partly obscured	NearMap
2010, Jul 9	1	—	1.0	2.75	—	2.5	—	2.25	—	—	invis	pres	invis	invis	—	—	2	very good	NearMap
2010, Mar 13	obs	obs	obs	obs	obs	obs	obs	obs	obs	—	invis	pres	invis	invis	—	—	?	poor	GoogleEarth
2010, Jan 24	pres?	—	pres	2.5	—	1.5	—	2.25	—	—	invis	pres	invis	invis	—	—	1.5	very good	NearMap
2009, Jul 11	?	—	?	obs	obs	obs	?	?	?	—	invis	pres	invis	invis	—	—	?	very poor	GoogleEarth
2007, Oct	—	—	—	1?	—	—	—	—	—	—	—	?	—	—	—	—	?	poor	AlburyCity
2004, Nov	—	—	—	?	—	—	—	—	—	—	—	?	—	—	—	—	?	poor	AlburyCity
2003, Feb 19	—	—	—	?	—	—	—	—	—	—	—	?	—	—	—	—	?	very poor	GoogleEarth
2000, Nov 7	—	—	—	?	—	—	—	—	—	—	—	?	—	—	—	—	?	very poor	AlburyCity

Codes: invis—not visible on aerial, obscured by other vegetation; obs—obscured due to clouds or reflections; pres—present but not measurable;



Fig. 1. Plant East n°3 as seen from south.



Fig. 2. Plant West n° 1 as seen from the northwest.



Fig. 3. Plant East n°4 as seen from north.



Fig. 4. The source palm (P2) as seen from the south.



Fig. 5. Plant West n° 2 as seen from the northeast.



Fig. 6. The source palm (P2) as seen from the west. Note the low self-seeded clusters to the right (P1) and left (P3).

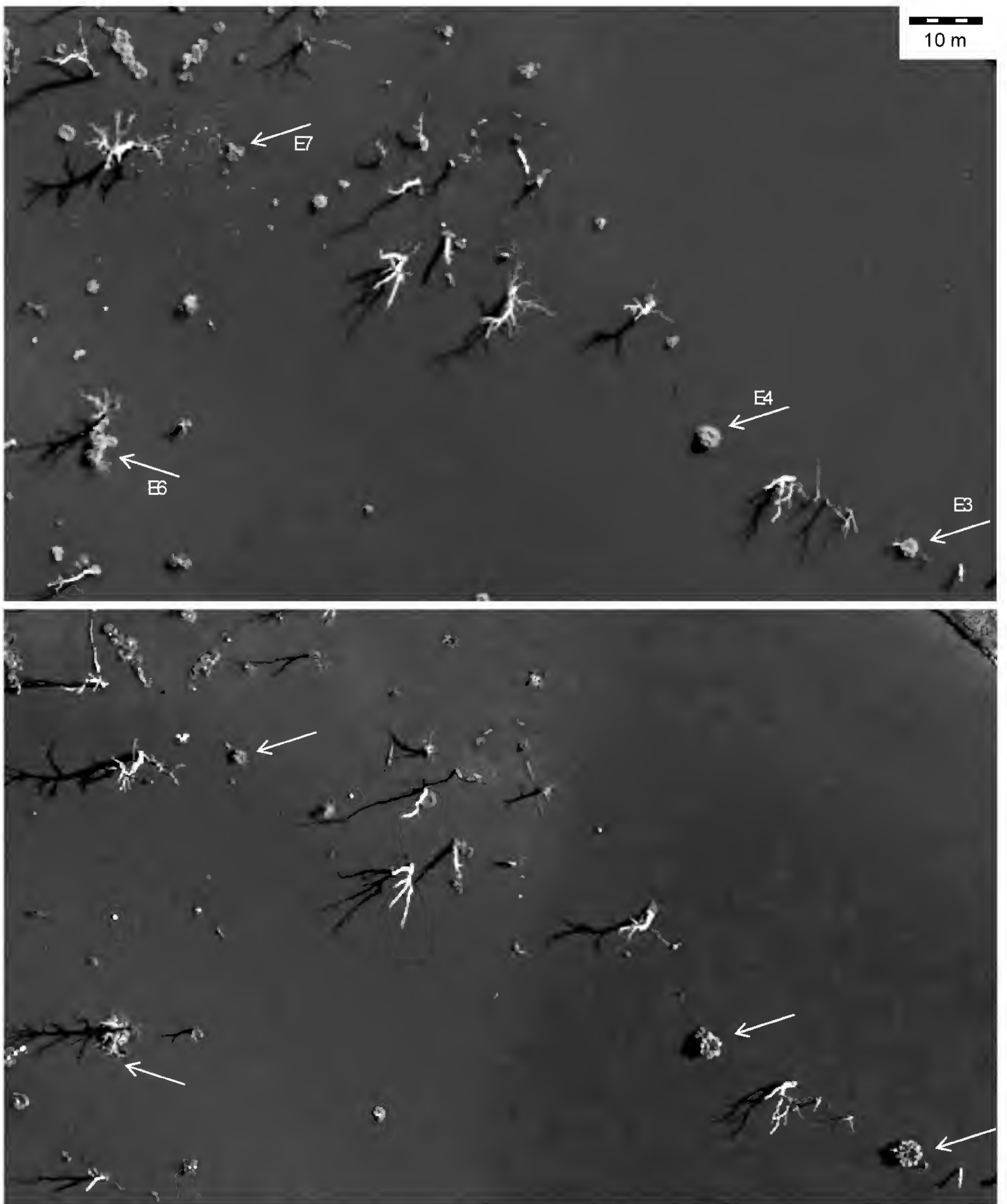


Fig. 7. Appearance of the self-seeded *Washingtonia* palms (East n°3, 4, 6–9) on 22 February 2014 (top) and 8 November 2017 (bottom).

Washingtonia dispersal and establishment processes

Washingtonia plants growing in the Kremur Street maturation ponds and on the peninsula are all self-seeded and started growing from 2000. Successful establishment was dependent on the micro-graphic setting and on vectors for dispersal of the plant seed.

The maturation pond is a flooded pastoral landscape studded with dead eucalypt trees, most of which are now largely branchless and act as perches for numerous bird species. Sedges and other littoral vegetation grow at the base of these tree trunks providing some modicum of substrate on which *Washingtonia* can germinate (Figure 1, Figure 2). Fallen trees and branches, resting semi-submerged in comparatively shallow water, act as a traps for floating leaf litter and develop into shallow vegetation islets. None of these islets, however, have *Washingtonia* growing on them, with the possible exception of plant E4, which may have grown on or next to a semi-submerged log (comparing the November 2004 and October 2007 imagery). Similarly, the temporary establishment of plant W3 may have occurred on a semi-submerged log.

As the plants are growing in the middle of the maturation pond, birds, alighting on the perches provided by the dead trees, are responsible for the deposition of the seeds. A common feature of the successfully established plants is that all grow in areas with little or no shading. Examination of

the shoreline, the peninsula and the larger vegetation islets showed the presence of other bird-dispersed weeds, such as blackberries (*Rubus* spp.) and figs (*Ficus* spp.) even in dense patches. *Washingtonia* seed does not appear to germinate or establish if it is choked by competing vegetation and concomitant lack of sunlight.

In California, *Washingtonia filifera* and *Washingtonia robusta* are also dispersed by fluvial activity (Talley, Nguyen, & Nguyen, 2012). Indeed, in the Albury setting, *Washingtonia robusta* seeds tend to end up in the street gutters and from there in the stormwater drains. As stormwater and sewerage are discharged and treated differently, however, none of these seeds are likely to end up in the sewerage treatment plant, let alone in the maturation ponds (as the outgoing water is screened and filtered). All seeds in the ponds would have been introduced by vertebrate vectors.

Vertebrate species, both volant and terrestrial feed on *Washingtonia* drupes and disperse their seeds. In Australia the only vectors on record are the Pied Currawong (*Strepera graculina*) and Grey-headed flying fox (*Pteropus poliocephalus*) (Spennemann, 2018c). Currawongs, ingest fruit as a whole and then fly to a nearby perch to digest the meal in the crop and regurgitate the indigestible elements such as seeds and part of the pericarp (Spennemann, sunpubl.).

Table 3. Potential volant vectors observed at the Kremur Street Parklands and adjacent Horseshoe Lagoon (HS)

Common name	Scientific name	consumption	perching	vector status
Greylag Goose (Domestic type)	<i>Anser anser</i>	probable	possible	unlikely
Spotted Dove	<i>Spilopelia chinensis</i>	possible	possible	possible
Crested Pigeon	<i>Ocyphaps lophotes</i>	possible	probable	possible
Pacific Koel (HS)	<i>Eudynamis orientalis</i>	probable	unlikely	possible
Noisy Miner	<i>Manorina melanocephala</i>	possible	likely	probable
Red Wattlebird	<i>Anthochaera carunculata</i>	possible	likely	probable
White-plumed Honeyeater	<i>Lichenostomus penicillatus</i>	probable	likely	probable
Blue-faced Honeyeater	<i>Entomyzon cyanotis</i>	probable	possible	possible
Little Friarbird	<i>Philemon citreogularis</i>	possible	likely	probable
Noisy Friarbird	<i>Philemon corniculatus</i>	possible	likely	probable
Australian Magpie	<i>Cracticus tibicen</i>	possible	likely	probable
Pied Currawong	<i>Strepera graculina</i>	documented	likely	likely
Black-faced Cuckooshrike	<i>Coracina novaehollandiae</i>	probable	likely	probable
White-bellied Cuckooshrike	<i>Coracina papuensis</i>	probable	unlikely	possible
White-winged Triller (HS)	<i>Lalage tricolor</i>	possible	probable	possible
Crested Shrike-tit	<i>Falcunculus frontatus</i>	possible	possible	possible
Olive-backed Oriole	<i>Oriolus sagittatus</i>	possible	possible	possible
Australian Raven	<i>Corvus coronoides</i>	documented	probable	probable
White-winged Chough (HS)	<i>Corcorax melanorhamphos</i>	possible	possible	possible
Common Blackbird	<i>Turdus merula</i>	documented	unlikely	unlikely
Common Starling	<i>Sturnus vulgaris</i>	documented	probable	probable
Common Myna (HS)	<i>Acridotheres tristis</i>	documented	possible	unlikely

Given the small size of the drupe (diameter 9–10mm), however, a much greater range of vectors can be inferred. In total 99 bird species are on record for the Kremur Street Parklands (Cornell Lab of Ornithology, 2018b), with an additional 26 species observed at the adjacent Horseshoe Lagoon (Cornell Lab of Ornithology, 2018a). Of these, 101 are pure waterbirds or exclusively insectivores, which can be ruled out as potential vectors for *Washingtonia* drupes/seeds.¹ The remainder have been classified in terms of their likelihood to act as effective vectors of the *Washingtonia* growing in the maturation pond based on whether they have been known to ingest *Washingtonia* or *Phoenix* drupes (Spennemann, unpubl. data), or like-sized, large seeded fruit (Barker & Vestjens, 1989) (Table 1 ‘consumption’) and whether they are likely to perch on isolated, very exposed dead trees or alight on small (<1 m²) vegetation patches in the maturation pond (Table 1 ‘perching’). The majority of these will alight on isolated trees or tree trunks with good visibility and some height to aid in emergency take off. There they process their meal and regurgitate the indigestible parts. The self-seeded palms on semi-submerged logs can be explained in that birds, that land on these to drink void indigestible elements by regurgitation before drinking. The domestic geese originate from a property near Horseshoe Lagoon and are spatially confined to that lagoon, away from the area where the *Washingtonia* occur.

Discussion

Long-term success

At the time of on-ground inspection one of the palms (W2) was dead (Figure 5). The remains showed at least 13 fronds, indicating it had thrived for some time. Based on aerial imagery, the plant existed from 2012 to 2017 and seems to have originally established on a semi-submerged trunk. The cause of the palm’s death is not evident as the terminal growth bud is well above the water. It is possible that the palm ran out of nutrients.

Another example of establishment and subsequent failure are palms East 6a and 6b. Both grew under the perch of a dead eucalypt. Plant 6b established in ca 2010 and was followed to the south by plant 6a in ca 2013. Both were thriving in 2014 (Figure 7 top) but by late 2015 both had died and another *Washingtonia robusta* (East 6c) had emerged closer to the base of the dead eucalypt tree. This palm is still extant today (Figure 7 bottom). Today nothing remains of plants 6a and 6B and it remains unclear why they died.

The plants on the peninsula are growing on clayey soil, ca. 0.5m above the water level, while the palms in the pond are growing on an unspecified substrate. It can be surmised that the initial establishment occurred on substrate accumulated on a semi-submerged log or on debris trapped at the base

of the trunk that serves as perch. Over time the roost must have reached the soft bottom of the maturation pond as the stability of the palms cannot be explained otherwise.

Worth noting is the differential growth rates between the palms growing on the peninsula (0.6–0.7m pa) and those growing in the maturation ponds (0.4–0.5m pa). This differential growth can be explained in terms of the availability of soil nutrients and the inhibiting factors of water-logged soil. *Washingtonia robusta* reputedly exhibited retarded growth when over-watered or planted in wet soils (Meerow, 1994). Clearly, the document growth rated of 1.8m per year among well-watered *Washingtonia robusta* (Morton, 1998; Muirhead, 1961, p. 41) are contingent on ample supplies of nutrients and soil substrate as water availability alone is not a factor.

The palms growing in the maturation ponds at the Albury Sewerage Works highlights the adaptability of the palm. The only other mention of *Washingtonia robusta* growing in water comes from Malta in the Mediterranean, where Mifsud (1995) observed two mature palms growing in a pond in the San Anton Gardens.

Other self-seeded palm species nearby

Two well-established, mature *Phoenix canariensis* are located in the grassed area north of maturation pond N° 1. Both are male specimens with similar size (5 m diameter, 4m height). Given that neither palms is located near any perch trees or overhead wires, they are likely to have grown from seeds that were either dropped mid-flight by avian vectors, or, more likely, deposited in scats by terrestrial vectors such as foxes (Spennemann, 2018d). Three additional, immature *Phoenix canariensis* were noted in the cluster of eucalypt trees to the north of maturation pond N° 1. Two of these palms were close to the southern edge of the wooded area, while the third was at the edge of an opening near the northern margin (Spennemann, 2018e).

The absence of *Washingtonia robusta* in the areas beyond the maturation pond is striking. The reasons are not self-evident, as both *Phoenix canariensis* and *Washingtonia robusta* are well represented as self-seeded plants in suburban Albury. A similar situation was observed in open agricultural settings at Alma Park (Southern Riverina of NSW) where *Phoenix canariensis*, originating from a number of source trees, widely established itself since the 1950s, whereas a *Washingtonia robusta*, planted in 1906 (at the same time as the homestead), produced no viable offspring.

Unlike *Phoenix canariensis*, *Washingtonia robusta* may have trouble germinating and establishing in areas of higher competition with groundcover grasses, especially those which have allelopathic capacity (Downer & Hodel, 2001). The same applies to seeds dropped under perch trees, as

¹ The Spiny-cheeked Honeyeater (*Acanthagenys rufogularis*), which was identified for Horseshoe Lagoon (Cornell Lab of Ornithology, 2018a) was omitted from the table as its presence is very unlikely (pers. comm. D Watson).

similar allelopathic potential is exhibited by several species of eucalypt (Chu et al., 2014; Zhang & Fu, 2009) as well as *Callitris* (Harris, Lamb, & Erskine, 2003).

No data exist for *Washingtonia robusta*, but as both seeds and seedlings of its congener, *Washingtonia filifera*, are allelopathic (Khan, 1982a, 1982b), it can be expected that this is the case for *Washingtonia robusta* as well. The allelopathic potential of a *Washingtonia robusta* seed and seedling appears to be outcompeted by that of other plants due to the small volume of the *Washingtonia robusta* seed and the small area of its seedling. In areas of non-competition it is also possible that *Washingtonia robusta* may germinate but fail to successfully establish on compacted clay soil, while *Phoenix canariensis* with its larger seed mass has that ability. Once successfully established as a seedling, the plant has a high chance of survival unless subjected to grazing.

Implications

Setting aside a discussion on the merits of novel ecosystems as ‘valid’ environmental states (Hobbs, Higgs, & Hall, 2013; Miller & Bestelmeyer, 2016; but see Murcia et al., 2014), it needs to be asked from a management perspective whether the colonisation of the wetland at Albury by *Washingtonia robusta* is beneficial or detrimental. Clearly, *Washingtonia robusta* do not rapidly invade the bushland surrounding the wetlands; the failure to establish in areas where other invaders (blackberry, fig) thrive, demonstrates this well. Establishment seems to be random (P) and only successful in the event of seed rain (i.e. underneath P2). The colonisation of spaces underneath perches in open water seems to be successful, albeit not on a large scale. No other plant seems to successfully establish at such locations, suggesting that *Washingtonia robusta* adds habitat variation rather than detract from it.

Unlike the dense crown of pinnate fronds of *Phoenix canariensis*, which provides roosting habitat for numerous birds, marsupials and rodents, the sharp spines that line the upper edges of the petioles of *Washingtonia robusta* act as a major faunal deterrent. Consequently, colonisation by *Washingtonia robusta* does not add to habitat until such time that the plants have developed a deposit of old, dry leaves that form the ‘skirt.’ Once this has occurred, *Washingtonia robusta* provides sheltered nesting habitat for several species.

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The current status of exotic freshwater vascular plants in Australia - a systematic description

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Abstract: Freshwater systems are considered particularly vulnerable to human impact, through habitat modification, changes to water regimes and quality, invasion by exotic species and climate change. Using various records, we conducted a descriptive analysis of the naturalised freshwater plant species in Australia. There are 63 freshwater plant species belonging to 45 genera and 26 families naturalised in Australia with the dominant families being Cyperaceae, Poaceae and Plantaginaceae. More than 40% of these species are categorised as either invasive or declared weeds, the majority being perennial wetland marginal plants. They originated from all the inhabited continents with most of the species being native to Europe, South America and North America. The greatest number of species are currently found in New South Wales (90%), Queensland (68%) and Victoria (65%); the ornamental aquarium plant trade was identified as the main introduction pathway. Most species are clonal plants with flexible modes of reproduction and multiple dispersal vectors. We conclude that exotic plant species are now an important component of Australia's freshwater systems and that ongoing monitoring of their status, distribution and impact should be a high priority in light of the increasing influence of anthropogenic factors including climate change.

Key words: aquatic; ecosystem; flora; invasive; native; naturalised; ornamental

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Introduction

Freshwater ecosystems are estimated to cover about 3% of the Earth's land surface area (Downing *et al.* 2006) but they provide habitat to a disproportionately high number of specialised plant and animal species (Balian *et al.* 2008). Globally, these ecosystems are experiencing severe declines in biodiversity due to a mix of human-mediated threats such as pollution, overexploitation, flow modification, habitat degradation and invasive species (Dudgeon *et al.* 2006). These declines in many cases are more pronounced compared to terrestrial ecosystems (Sala *et al.* 2000); it has been argued that freshwater ecosystems are the most threatened of global ecosystems (Saunders *et al.* 2002; Dudgeon *et al.* 2006). Despite freshwater ecosystems being extremely species-rich and harbouring many threatened species (Abell *et al.* 2008), they do not receive the same conservation efforts and research attention as terrestrial ecosystems (Brundu 2015). For example, there is comparatively little information on freshwater plants, insects, molluscs and crustaceans in most parts of the world (Revenga *et al.* 2005) possibly due to the difficulty of monitoring freshwater ecosystems (Brundu 2015).

One of the most significant threats to freshwater ecosystems is the widespread introduction of exotic plant species into new areas as a result of increased international human travel and trade (Mack *et al.* 2000). Most species are introduced deliberately for ornamental or agricultural purposes, others passively find their way to new regions as contaminants of ballast water or as hitchhikers on other species (Champion *et al.* 2010). Although strengthened pre-border biosecurity measures have slowed the rate of introductions to Australia, the process is ongoing (Weber *et al.* 2008; Dodd *et al.* 2016), and it is inevitable that a proportion of these introduced species will become naturalised or even problematic invaders. Currently Australia is estimated to have around 2700 naturalised plant species, about 12% of its total flora (Randall 2007; Dodd *et al.* 2015).

Non-invasive naturalised species are those that establish self-perpetuating populations in the wild without having profound negative effects on the ecosystem (Richardson *et al.* 2000). It is estimated that, with time, about 10% of naturalised species will overcome reproductive and dispersal barriers, and become *invasive* (Williamson & Fitter 1996; Williams & West 2000). In future this proportion may increase as ongoing environmental and global climatic changes provide ecological opportunities in some regions for some of these species to become invasive (Groves 2006; Scott *et al.* 2008; Duursma *et al.* 2013; Sorte *et al.* 2013; Leishman & Gallagher 2015). When this occurs, the highly connected nature and dynamic disturbance regimes of freshwater ecosystems will further facilitate the spread of these species through the landscape (Dudgeon *et al.* 2006).

Invasive exotic freshwater plant species can exert dramatic negative impacts on native communities and ecosystems similar to their terrestrial counterparts (Evangelista *et al.* 2014). For example, a more than 50% decline in species richness of co-occurring native freshwater plant species was observed with increasing abundance of the invasive exotic

Alternanthera philoxeroides in natural ponds (Chatterjee & Dewanji 2014), and *Myriophyllum spicatum* in Lake George, New York, USA (Boylen *et al.* 1999). Furthermore, this suppression of native plant communities by exotic plant species may modify trophic interactions (Richardson & van Wilgen 2004) by simplifying and rendering the native plant communities a poorer food source for herbivores and higher trophic level consumers (Havel *et al.* 2015). Thus, exotic plant invasions can have detrimental ecosystem-level effects on freshwater systems (Yarrow *et al.* 2009).

The naturalised flora in Australia is considered one of the most species rich in the world (Dodd *et al.* 2015) and a large effort has been made to establish a comprehensive inventory of the entire naturalised flora (Randall 2007). In addition, the Australian Virtual Herbarium (<http://avh.chah.org.au/>) has digitised occurrence records of extant plant species and created a publicly accessible online database (Haque *et al.* 2017). These records have been useful in assessing patterns of species endemism (Crisp *et al.* 2001), mapping species threats (Evans *et al.* 2011), predicting plant invasions (Duursma *et al.* 2013), analysing drivers responsible for patterns of naturalisation (Dodd *et al.* 2015), and identifying areas that have high richness of naturalised exotic species (Dodd *et al.* 2016). However, these outcomes are broad and generalised across different ecosystems. Therefore, it is important for ecosystem-level descriptions of naturalised non-invasive and invasive exotic species to be undertaken so ecosystem-specific monitoring and management practices can be devised. The aim of this study is to provide a systematic description of the distribution, origin and richness of naturalised non-invasive and invasive exotic freshwater plant species in Australia.

Methods

Compilation of species list

We searched ISI Web of Knowledge for information on naturalised plant species in freshwater ecosystems of Australia using the following combinations: (invasi*) OR (invader) OR (non-native) OR (exotic) OR (alien) OR (non-indigenous) OR (introduced) OR ("naturalised species") OR ("naturalized species") OR (biological invasion*) AND (plant) OR (macrophyte*) AND (freshwater) OR (aquatic) OR (river*) OR (pond*) OR (lake*) OR (dam*) OR ("farm dam") AND (Australia) OR ("New South Wales") OR ("NSW") OR (Queensland) OR ("Northern Territory") OR ("NT") OR ("Western Australia") OR ("WA") OR ("South Australia") OR ("SA") OR (Victoria) OR (Vic) OR (Tasmania) OR ("Australian Capital Territory") OR ("ACT"). In addition, a list of naturalised freshwater plant species in Australia was compiled from existing inventories and lists (e.g. Aston 1973; Sainty & Jacobs 2003; Randall 2007) and online databases (e.g. <http://weeds.dpi.nsw.gov.au/>; <http://plantnet.rbgsyd.nsw.gov.au/>; <https://keyserver.lucidcentral.org/weeds/data/media/Html/index.htm#A>; <https://www.business.qld.gov.au/industries/farms-fishing-forestry/agriculture/land-management/health-pests-weeds-diseases/weeds-diseases>; <https://nt.gov.au/environment/weeds/weeds-in-the-nt/A-Z->

list-of-weeds-in-the-NT; <https://florabase.dpaw.wa.gov.au/search/advanced?current=y&alien=y>; http://www.pir.sa.gov.au/biosecurity/weeds_and_pest_animals/weeds_in_sa; <http://agriculture.vic.gov.au/agriculture/pests-diseases-and-weeds>; <https://dpipwe.tas.gov.au/invasive-species/weeds/weeds-index/declared-weeds-index>).

We categorised plant species as ‘freshwater’ using the following definition: “closely bound to freshwater habitats whose vegetative parts actively grow either permanently or periodically (for at least several weeks each year) submerged below, floating on, or growing up through the water surface” (Lacoul & Friedman 2006; Chambers *et al.* 2008; Hussner *et al.* 2012). It should be noted that although we concentrate on *freshwater* species, many species in Australia have wide salinity tolerances and can occur across a wide part of the gradient from fresh to saline in variable habitats such as saltmarshes and estuaries.

Validity of the species names was checked using the Australian Plant Census website (<https://biodiversity.org.au/nsl/services/APC>) and species not found in the census or with unresolved nomenclature were excluded. Any species whose status as native or exotic was unclear according to the Australian Plant Census was also excluded from the analysis. We then checked the naturalisation status of each species using a comprehensive data set of the introduced flora of Australia - an updated version of Randall (2007) containing unpublished data, and excluded any that was not naturalised. We also excluded species that are associated more with saline water than fresh water.

Plant data collation

Data on the native regions of each species, introduction purpose, and their biology (growth habit, longevity and dispersal mechanisms) were compiled from multiple sources including regional floras, published literature and the online databases (e.g. <https://www.cabi.org/ISC/search>; <http://ausgrass2.myspecies.info/content/fact-sheets>; <https://npgsweb.ars-grin.gov/gringlobal/taxon/taxonomysimple.aspx>). Eight broad regions of origin were identified as follows: Europe, North America (including Mexico), Central America (including the Caribbean), South America, Sub-Saharan Africa (including Madagascar), North Africa, temperate Asia (including the Middle East), and southern and south eastern Asia. Multiple sources of origin were assigned where a species had a wide native geographical region. For example, *Alisma lanceolatum* is native to Europe, North Africa and temperate Asia and was counted as a species of each of these regions. The current economic uses of the species were used to assign their purpose of introduction (Weber *et al.*, 2008) where such information was not explicitly available. We also conducted internet searches to determine if each species is currently available for purchase from aquarium suppliers.

The Australian Virtual Herbarium (<http://avh.ala.org.au/>) was used to determine presence or absence of each species in each of the Australian states and territories. Randall’s (2007) list was used to categorise the species as naturalised non-invasive, invasive (*sensu* Richardson *et al.* 2000), or declared weeds.

A declared weed was defined as a plant species that has been identified for control, eradication or prevention of entry into an Australian jurisdiction by a legislation of that jurisdiction.

Results

Taxonomy and status

After screening 255 titles returned by the literature search, 42 papers that were studies of freshwater plants were reviewed for collation of the naturalised species list (Appendix 1), in addition to data derived from existing inventories and online data sources. In total, 63 exotic species of freshwater plants (belonging to 45 genera and 26 families) were identified as naturalised in Australia (Figure 1; Appendix 2). Of these, 40 species (63%) were classified as naturalised non-invasive, 13 species (21%) were designated as invasive and 10 (16%) as declared weeds (Figure 2; Table 1). The plant families with the highest number of exotic naturalised species were Poaceae (9 species), Cyperaceae (9 species) and Plantaginaceae (5 species). Fourteen of the 27 families were represented by only one species.

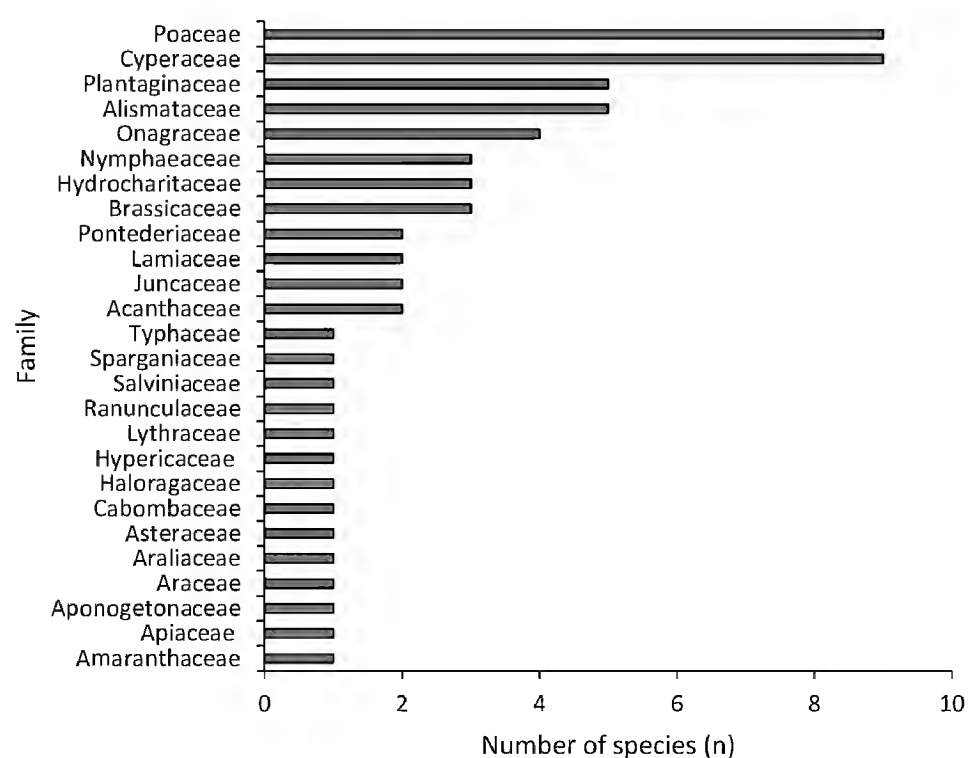


Fig. 1. Taxonomic diversity of the naturalised freshwater plant species in Australia.

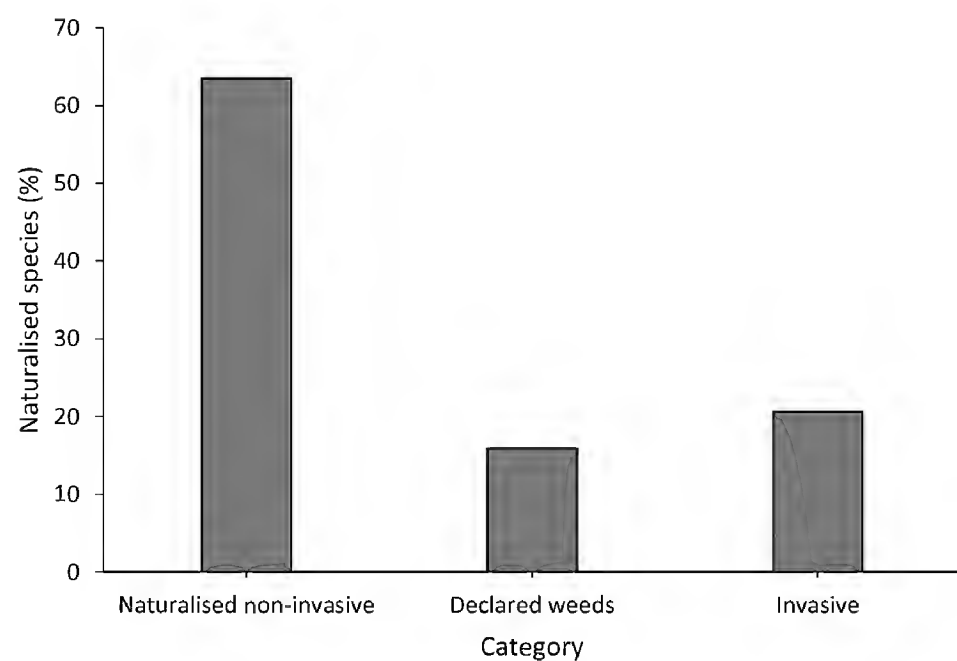


Fig. 2. The percentage of naturalised freshwater plant species in different categories of invasive status.

Table 1: Naturalised freshwater species that are considered invasive in Australia. An asterisk (*) indicates that the species is a Weed of National Significance (WONS) (<http://www.environment.gov.au/biodiversity/invasive/weeds/weeds/lists/wons.html>).

Species	Family
<i>Hygrophila costata</i> Nees	Acanthaceae
<i>Alternanthera philoxeroides</i> (Mart.) Griseb.*	Amaranthaceae
<i>Gymnocoronis spilanthoides</i> (D.Don ex Hook. & Arn.) DC.	Asteraceae
<i>Cabomba caroliniana</i> A.Gray*	Cabombaceae
<i>Myriophyllum aquaticum</i> (Vell.) Verdc.	Haloragaceae
<i>Egeria densa</i> Planch.	Hydrocharitaceae
<i>Juncus articulatus</i> L.	Juncaceae
<i>Ludwigia peruviana</i> (L.) H.Hara	Onagraceae
<i>Arundo donax</i> L.	Poaceae
<i>Hymenachne amplexicaulis</i> (Rudge) Nees*	Poaceae
<i>Eichhornia crassipes</i> (Mart.) Solms*	Pontederiaceae
<i>Salvinia molesta</i> D.S.Mitch.*	Salviniaceae
<i>Sagittaria platyphilla</i> (Engelm.) J.G.Sm.*	Alismataceae

Growth habit

The majority of the species (94%) were perennial; 3% were annual and the remaining 3% have annual stems but perennial rhizomes. Most of the species were emergent marginal wetland species (59%). The emergent plants that grow through the water column constituted 24% while the submerged (8%), floating leaved (6%) and free-floating (3%) species made up the remainder.

Region of origin

Exotic naturalised freshwater species originated from a variety of regions, with Europe, South America and North America being the most widely represented (Figure 3). Species that are native to southern and southeast Asia were the most poorly represented with only two reported as naturalised in Australia. Only eight species (12%) did not have multiple places of origin. Of the 13 species that are classified as invasive in Australia, 10 are native to South America.



Fig. 3. Regions of origin of the naturalised freshwater plant species in Australia.

Distribution in Australia

New South Wales (NSW) was the most species-rich state with 90% of the exotic naturalised freshwater species being present, followed by Queensland (68%) and Victoria (65%) (Figure 4). Northern Territory had the lowest number of naturalised freshwater plant species (14 of the 63 species or 22%). Species that were present in every state include *Cyperus eragrostis* (Cyperaceae), *Arundo donax* (Poaceae), and *Polypogon monspeliensis* (Poaceae).

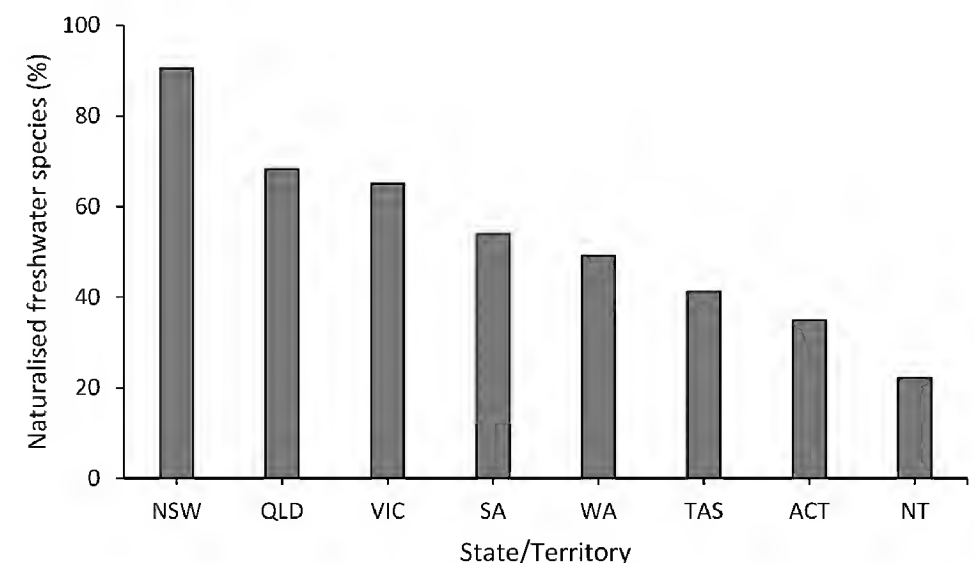


Fig. 4. Percentage of naturalised freshwater plant species present in the states and territories of Australia.

Introduction pathways

Almost two-thirds of the species (57%) were introduced for aquarium and ornamental water garden purposes while a further 25% were imported for agricultural purposes including as vegetables, for example *Alternanthera philoxeroides* and *Rorippa* spp., and pasture grasses. The introduction reason for the remaining 17% of species is unknown; there is no information available on their economic use (Figure 5). Currently 33% of the species are available for sale within Australia either by water garden nurseries or over the internet (Table 2).

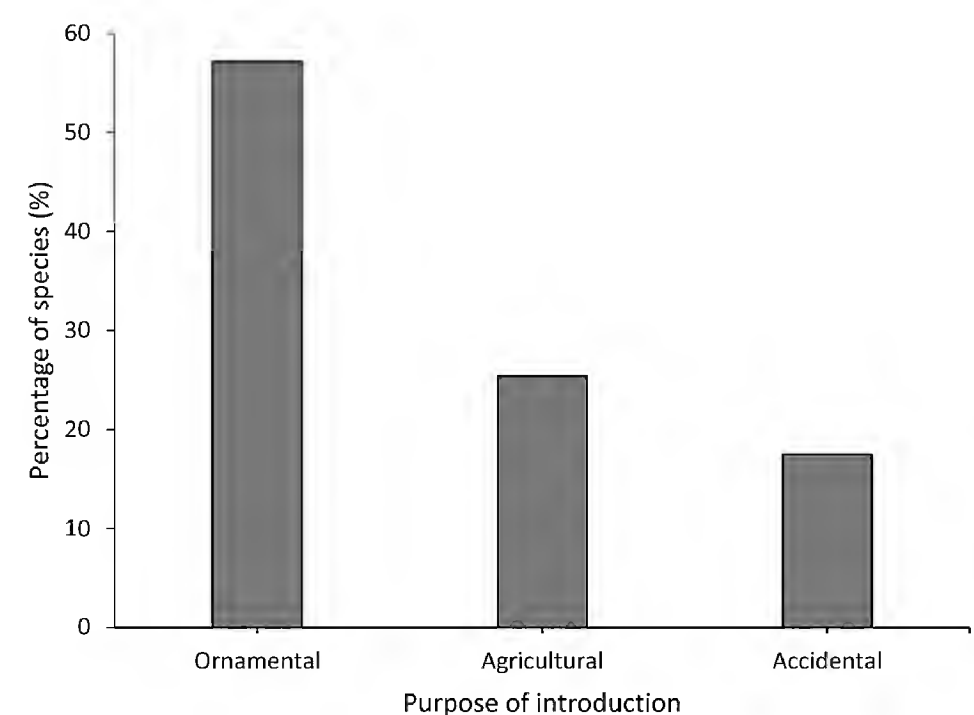


Fig. 5. Purpose of introduction of naturalised freshwater plant species

Table 2: Naturalised freshwater plant species available for sale in Australia. An asterisk (*) indicates that the species is a declared weed.

Species	Family
<i>Aponogeton distachyus</i> L.f.	Aponogetonaceae
<i>Berula erecta</i> (Huds.) Coville	Apiaceae
<i>Bacopa caroliniana</i> (Walter) B.L.Rob.	Plantaginaceae
<i>Cyperus papyrus</i> L.	Cyperaceae
<i>Cyperus prolifer</i> Lam.	Cyperaceae
<i>Hydrocleys nymphoides</i> (Humb. & Bonpl. ex Willd.) Buchenau	Limnocharitaceae
<i>Hygrophila polysperma</i> (Roxb.) T.Anderson	Acanthaceae
<i>Hypericum elodes</i> L.	Hypericaceae
<i>Ludwigia palustris</i> (L.) Elliott	Onagraceae
<i>Ludwigia repens</i> J.R.Forst.	Onagraceae
<i>Mentha aquatica</i> L.	Lamiaceae
<i>Mentha pulegium</i> L.*	Lamiaceae
<i>Nymphaea caerulea</i> Savigny	Nymphaeaceae
<i>Nymphaea mexicana</i> Zucc.	Nymphaeaceae
<i>Phalaris arundinacea</i> L.	Poaceae
<i>Pontederia cordata</i> L.*	Pontederiaceae
<i>Rorippa nasturtium-aquaticum</i> (L.) Hayek	Brassicaceae
<i>Rotala rotundifolia</i> (Buch.-Ham. ex Roxb.) Koehne	Lythraceae
<i>Typha latifolia</i> L.*	Typhaceae
<i>Veronica anagallis-aquatica</i> L.	Plantaginaceae
<i>Zantedeschia aethiopica</i> (L.) Spreng.*	Araceae

Reproduction and dispersal

Almost half (49%) of the 63 species reproduce both sexually and vegetatively (17 reproduce by both seeds and fragmentation, 14 by both seeds and rhizomes). 33% of the species (21 out of 63) reproduce exclusively by means of seeds. The remaining 17% reproduce exclusively by vegetative means. All the species that reproduce exclusively vegetatively, do so by stem fragmentation. Water currents, waterfowl, flood and watercraft were identified as the main dispersal agents of the seeds and stem fragments.

Discussion

Our search identified 63 exotic freshwater plant species that have become naturalised in Australia, a large proportion of which are perennial wetland marginal species. They belong to 26 families representing 16% of families of the naturalised flora. The majority of the species originated from Europe, South America and North America and are currently most widely distributed along the eastern coastal fringes of the country. They were mostly introduced for ornamental purposes via the aquarium and water garden plant trade. The majority of the species reproduce both sexually and vegetatively, with water currents, waterfowl and watercraft identified as their main dispersal vectors.

Given that there are about 2739 naturalised plant species in Australia (Randall 2007), freshwater plant species represent a very low proportion (slightly over 2%). However, despite their

seemingly small number, they may have disproportionately strong environmental impacts as exemplified by the fact that nearly 20% of the Weeds of National Significance are freshwater species (6 out of 32) (<http://www.environment.gov.au/biodiversity/invasive/weeds/weeds/lists/wons.html>). This may be partly attributed to the widespread geographic distribution of many of these naturalised freshwater species, with most species found in multiple states within Australia. More than 40% of the naturalised freshwater plant species we identified are categorised as either invasive or declared weeds in Australia. This proportion is much greater than for the naturalised terrestrial flora of which around 14% have become invasive (Leishman *et al.* 2017).

Of the naturalised freshwater species in our analysis, a large majority are in the Poaceae, Cyperaceae and Plantaginaceae families. These families are among the twenty most commonly represented in the naturalised Australian flora (Dodd *et al.* 2015) and reflect the Australasian (Jacobs & Wilson 1996) and worldwide (Chambers *et al.* 2008) trends where Poaceae and Cyperaceae are the most species-rich freshwater plant families. Many plants belonging to Poaceae and Cyperaceae are important pasture crops on which livestock production in Australia relies heavily (Cook and Dias 2006), which may further explain their dominance compared to the other families.

The majority of the naturalised species we identified are perennial, clonal plants with the ability to exploit heterogeneous habitats. Clonality may explain the invasion success of some of these species as it enhances persistence and spread of plants at local scales (Santamaría 2002). The largest proportion (57%) of the species in our analysis were emergent species that fringe the margins of water bodies. This may be due to water margins being suitable for species that can withstand periodic submergence as well as helophytes that can cope with periodic draw-downs (Lacoul & Freedman 2006). In contrast, open water bodies provide a narrower range of environmental conditions, resulting in less species being suited to that habitat. The overrepresentation of the marginal species in our analysis may also have resulted from study biases since these species are conspicuous and easier to sample and identify, in contrast to, for example, submerged species. In addition, emergent species that occur along water body margins may be able to disperse their propagules not only by water but also by wind, allowing them to colonise widely across the landscape (Soomers *et al.* 2013).

Many naturalised freshwater plant species in Australia have originated from Europe, South America and North America. This is largely due to historical and trade linkages between Australia and these continents. However, these regions of origin are likely to have shifted through time, with invasion success of plant species from Europe strongly linked with European settlement in Australia (Phillips *et al.* 2010) and more recent successful introductions originating from South America now contributing the largest proportion of naturalised freshwater plant species. The majority of the naturalised freshwater plant species in our study had multiple broad regions of origin. Many freshwater plant

species have broad distributions due to selective advantages provided by asexual reproduction and long distance dispersal of propagules (Santamaria 2002). Species with large native ranges tend to have broad environmental tolerances and thus may be effectively pre-adapted to their introduced range (Pyšek *et al.* 2009; Keller *et al.* 2011). This may explain why the naturalised freshwater species of Australia are small in number but a large proportion have spread extensively across the continent and are now considered as species of concern.

New South Wales, Queensland and Victoria, the most densely populated states (A.B.S. 2018), have the highest numbers of naturalised freshwater species. This is not surprising as there is a strong correlation between human population density and exotic species richness, due to humans being responsible for the deliberate or accidental introduction of exotic species (Weber *et al.* 2008; Dodd *et al.* 2016; Haque *et al.* 2017). Furthermore, a higher human population density also means a higher number of potential aquarium keepers, representing a greater propagule pressure (Hussner *et al.* 2010). Alternatively, biases in herbarium specimen collection may have painted a picture of relatively higher species numbers in the densely populated states than reality (Lavoie *et al.* 2012; Dodd *et al.* 2016; Haque *et al.* 2017). It has been observed that the intensity of herbarium specimen collection in Australia, on which our species regional distribution analysis relied, was higher in the densely populated areas (Dodd *et al.* 2016).

Our analysis revealed that almost 60% of the freshwater plant species naturalised in Australia were deliberately introduced for ornamental water garden and aquarium purposes. This is consistent with other studies globally reporting that importation and trade in ornamental plants is the most important pathway for freshwater plant introductions (Champion *et al.* 2010; Strayer 2010; Keller *et al.* 2011). Petroschevsky & Champion (2008) suggested that 85% of aquatic weeds in Australia were traded as aquarium or water garden plants. We found that a third of Australia's naturalised freshwater plant species are currently available by trade for either ornamental or agricultural purposes. Surprisingly, among these actively traded species are four declared weeds (*Mentha pulegium*, *Pontederia cordata*, *Typha latifolia* and *Zantedeschia aethiopica*). Some of these species may be traded with misspelled or incorrect scientific names that mask their exotic status (Brunel 2009). For example, we found that an aquarium supplier had listed *Eleocharis* for sale without specifying the species; if these species escape into the wild they could well remain undetected for a long time. There are also reports of aquarium plant dealers who, mostly due to ignorance, misrepresent exotic plants as similar-appearing native ones (Kay & Hoyle 2001). A more serious practice that may contribute to infestation of waterways is the deliberate cultivation of exotic ornamental plants in natural waterways, by aquarium traders in order to meet customer demands (Petroschevsky & Champion 2008).

Twenty-five percent of the naturalised freshwater species of Australia have been introduced deliberately for agricultural purposes. These include traditional vegetable species such as *Alternanthera philoxeroides* and *Rorippa* spp., and garden

herbs such as *Mentha aquatica*. However, the majority of the agricultural species are ponded pasture plants that were introduced for livestock grazing. Since commercial livestock production is a major contributor to the Australian economy, many state governments actively promoted introduction of exotic ponded pasture species through much of the 20th century (Cook & Dias 2006; Cook & Grice 2013). These species may have then spread across the broader landscape through natural dispersal mechanisms.

Almost a fifth of the naturalised freshwater plant species in Australia have no known economic uses and may have been introduced inadvertently in ballast water or as contaminants of other deliberately imported species, which is a common occurrence (Kay & Hoyle 2001). For example, Maki and Galatowitsch (2004) found that ten percent of freshwater plants that they obtained commercially contained exotic plant contaminants. Occasionally, some of these contaminants prove attractive and easy to grow and are therefore placed on the market. A good example is *Salvinia molesta*, which was introduced initially as a contaminant of other plants but was considered sufficiently attractive to be consequently traded as an ornamental species in Texas, USA for several years (Kay & Hoyle 2001).

Many of the naturalised plants in our analysis reproduce both sexually and vegetatively, and are easily dispersed by water currents and floods, wind, water birds and watercraft (Santamaria 2002). As vegetative spread and multiple dispersal vectors enhance establishment and therefore naturalisation success (Keller *et al.* 2011), these factors may also be drivers of invasion success of these naturalised freshwater plants.

From our study, we can conclude that although naturalised freshwater plant species form a very small proportion of the naturalised flora, they nevertheless are an important component of the Australian flora, being widespread across multiple regions. In spite of the existence of many statutory and regulatory measures to control trade in potential weeds in Australia at local, state and federal levels, a few declared weeds continue to be traded. A strict enforcement of these controls is therefore necessary through monitoring of the online aquarium market and periodically assessing compliance by nurseries through site visits. It is also important that we continue to assess the weed risk of naturalised species in light of ongoing environmental and climatic changes and to monitor potential spread of wild populations constantly (Champion *et al.* 2010). Finally, accessing information on naturalised freshwater plants ranging from the local to state level is difficult as data is contained within disparate sites. Therefore, a centralised system of storing data on ecology and management of naturalised freshwater plant species would be desirable for better knowledge sharing.

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Appendix 1: Titles reviewed for collation of the naturalised freshwater plant species list

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Appendix 2: List of the species included in the analysis of naturalised freshwater plants in Australia

Botanical name	Family
<i>Hygrophila costata</i> Nees	Acanthaceae
<i>Hygrophila polysperma</i> (Roxb.) T.Anderson	Acanthaceae
<i>Alisma lanceolatum</i> With.	Alismataceae
<i>Hydrocleys nymphoides</i> (Humb. & Bonpl. ex Willd.) Buchenau	Alismataceae
<i>Limncharis flava</i> (L.) Buchenau	Alismataceae
<i>Sagittaria calycina</i> Engelm.	Alismataceae
<i>Sagittaria platyphylla</i> (Engelm.) J.G.Sm.	Alismataceae
<i>Alternanthera philoxeroides</i> (Mart.) Griseb.	Amaranthaceae
<i>Berula erecta</i> (Huds.) Coville	Apiaceae
<i>Aponogeton distachyos</i> L.f.	Aponogetonaceae
<i>Zantedeschia aethiopica</i> (L.) Spreng.	Araceae
<i>Hydrocotyle ranunculoides</i> L.f.	Araliaceae
<i>Gymnocoronis spilanthoides</i> (D.Don ex Hook. & Arn.) DC.	Asteraceae
<i>Rorippa microphylla</i> (Boenn. ex Rchb.) Hyl.	Brassicaceae
<i>Rorippa nasturtium-aquaticum</i> (L.) Hayek	Brassicaceae
<i>Rorippa palustris</i> (L.) Besser	Brassicaceae
<i>Cabomba caroliniana</i> A.Gray	Cabombaceae
<i>Cyperus eragrostis</i> Lam.	Cyperaceae
<i>Cyperus involucratus</i> Rottb.	Cyperaceae
<i>Cyperus papyrus</i> L.	Cyperaceae
<i>Cyperus prolifer</i> Lam.	Cyperaceae
<i>Eleocharis minuta</i> Boeckeler	Cyperaceae
<i>Eleocharis pachycarpa</i> Desv.	Cyperaceae
<i>Eleocharis parodii</i> Barros	Cyperaceae
<i>Isolepis prolifera</i> (Rottb.) R.Br.	Cyperaceae
<i>Schoenoplectus californicus</i> (C.A.Mey.) Sojak	Cyperaceae
<i>Myriophyllum aquaticum</i> (Vell.) Verdc.	Haloragaceae
<i>Egeria densa</i> Planch.	Hydrocharitaceae
<i>Elodea canadensis</i> Michx.	Hydrocharitaceae
<i>Lagarosiphon major</i> (Ridl.) Moss	Hydrocharitaceae

Botanical name	Family
<i>Hypericum elodes</i> L.	Hypericaceae
<i>Juncus articulatus</i> L.	Juncaceae
<i>Juncus ensifolius</i> Wikstr.	Juncaceae
<i>Mentha aquatica</i> L.	Lamiaceae
<i>Mentha pulegium</i> L.	Lamiaceae
<i>Rotala rotundifolia</i> (Buch.-Ham. ex Roxb.) Koehne	Lythraceae
<i>Nymphaea alba</i> L.	Nymphaeaceae
<i>Nymphaea caerulea</i> Savigny	Nymphaeaceae
<i>Nymphaea mexicana</i> Zucc.	Nymphaeaceae
<i>Ludwigia longifolia</i> (DC.) H.Hara	Onagraceae
<i>Ludwigia palustris</i> (L.) Elliott	Onagraceae
<i>Ludwigia peruviana</i> (L.) H.Hara	Onagraceae
<i>Ludwigia repens</i> J.R.Forst.	Onagraceae
<i>Bacopa caroliniana</i> (Walter) B.L.Rob.	Plantaginaceae
<i>Callitriche brutia</i> Petagna	Plantaginaceae
<i>Callitriche stagnalis</i> Scop.	Plantaginaceae
<i>Veronica anagallis-aquatica</i> L.	Plantaginaceae
<i>Veronica catenata</i> Pennell	Plantaginaceae
<i>Alopecurus geniculatus</i> L.	Poaceae
<i>Arundo donax</i> L.	Poaceae
<i>Echinochloa polystachya</i> (Kunth) Hitchc.	Poaceae
<i>Echinochloa pyramidalis</i> (Lam.) Hitchc. & Chase	Poaceae
<i>Glyceria maxima</i> (Hartm.) Holmb.	Poaceae
<i>Hymenachne amplexicaulis</i> (Rudge) Nees	Poaceae
<i>Phalaris arundinacea</i> L.	Poaceae
<i>Polypogon monspeliensis</i> (L.) Desf.	Poaceae
<i>Urochloa mutica</i> (Forssk.) T.Q.Nguyen	Poaceae
<i>Eichhornia crassipes</i> (Mart.) Solms	Pontederiaceae
<i>Pontederia cordata</i> L.	Pontederiaceae
<i>Ranunculus sceleratus</i> L.	Ranunculaceae
<i>Salvinia molesta</i> D.S.Mitch.	Salviniaceae
<i>Sparganium erectum</i> L.	Sparganiaceae
<i>Typha latifolia</i> L.	Typhaceae

