1 **Title: Recovery potential of micro-wetlands from agricultural land uses**

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Summary

Many ecosystems located within agricultural landscapes are in decline, particularly woodlands, grasslands and wetlands. Surviving remnants are generally fragmented and unrepresentative of pre-disturbance states. Here, we investigate the potential for recovery of ecosystem function in a grassy woodland-wetland mosaic in south-eastern Australia. We focus on the Plains Woodland/Herb-rich Gilgai Wetland Mosaics which have declined in extent by 85%. The gilgai soils form a distinctive micro-relief of mounds and depressions which become seasonally waterlogged, providing important habitat for a large range of aquatic and dryland species. We surveyed 10 remnants subject to agricultural intensification and 7 remnants subject to passive restoration (4 with cessation of cultivation and 3 with livestock removal). Gilgai micro-relief was homogenized by cultivation, showed some recovery after release from cultivation, and was insensitive to grazing pressure. Floristic diversity, assessed through indicator species, was vulnerable to grazing. Indicator species were more prevalent in previously grazed sites, but further study is required to determine whether this reflects recovery or differing overall management history. We conclude that passive restoration is possible for recovery of wetland function and some biodiversity values. These conservation actions should be encouraged given the important role these micro-wetlands play in landscape connectivity and as drought refugia.

Key words: restoration; cracking clay soils; cultivation; grazing pressure; grassy woodland.

1. Introduction

Natural ecosystems located principally within agricultural environments – notably grasslands, woodlands and wetlands - have been subject to intensive loss and degradation across most of the globe as a result of agricultural activities (Neumann *et al.* 2016; Yates *et al.* 2000). Remnants are typically fragmented and have generally been transformed by cultivation, grazing, pasture improvement and drainage, resulting in highly modified environments which are unrepresentative of pre-disturbance states (McIntyre and Hobbs 1999) and support depauperate biotic assemblages (Neumann *et al.* 2016).

In Australia, many studies have documented positive state and transition responses of vegetation attributes of woodland and grassland systems as a result of changes in grazing regime and cultivation regime, including positive responses to grazing exclusion, reduced grazing and, in some circumstances, time since cultivation (Prober *et al.* 2009). Many studies have also documented positive ecological responses to reduced grazing pressure or restoration of natural hydrological regimes in larger, more recognisable wetland systems (Erwin, 2009). However, few studies have investigated the ecological impacts of grazing and cultivation on micro-wetland systems dispersed within predominantly dryland environments; nor the potential of these micro-wetlands to recover from such disturbances. Given the importance of all wetland systems as ecological refugia, especially in the context of climate change, it is critical that we understand the capacity of these agricultural wetland mosaics to recover from agricultural impacts and restore ecosystem and biodiversity functionality within that landscape. Here, we examine the effects of differing land management regimes on a grassy woodland-wetland mosaic in south-eastern Australia, particularly in relation to soil function and floristic richness. Our aims were to determine (i) whether the micro-relief recovered following release from cultivation; (ii) whether grazing intensity affected micro-relief; (iii) how grazing intensity affected floristic diversity; and (iv) the capacity of the system for ecological recovery following decreases in grazing intensity or cessation of cultivation.

2. Methods

2.1 Plains Woodland/Herb-rich Gilgai Wetland Mosaics

Our study focussed on a single micro-wetland ecosystem type which once occurred widely across lowland plains and lower slopes in northern Victoria. 'Plains Woodland/Herb-rich Gilgai Wetland Mosaic' are an ecological vegetation class defined by the Victorian Department of Environment, Land, Water and Planning, and are defined as open woodlands to 15 m tall on broad alluvial plains and along ephemeral drainage lines (<https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks>; accessed 03/10/2017). Soils are generally poorly drained heavy clays which form distinctive "gilgai" mounds and depressions in a fine-scale mosaic. Water accumulates seasonally

in the depressions, which creates micro-wetlands. The understorey consists of few, if any, shrubs while the ground layer is made up of a combination of “dryland” forbs/grasses and amphibious forbs/grasses tolerant of seasonal inundation (Casanova & Casanova). This micro-wetland ecosystem once occurred over an estimated 104 000 ha across Victoria, and has now been reduced to an estimated 16 000 ha – a decline of 85% (VEAC 2017). Further, intact examples of this ecosystem type support nationally threatened and listed migratory species including the Floodplain Rustyhood orchid (*Pterostylis cheraphila* D.L.Jones & M.A.Clem.), Ridged Water-milfoil (*Myriophyllum porcatum* Orchard), Mueller Daisy (*Brachyscome muelleroides* G.L. Davis) and Latham’s Snipe (*Gallinago hardwickii*).

2.2 Study sites

The Plains Woodland/Herb-rich Gilgai Wetland Mosaic remnants (hereafter referred to as “gilgai wetlands”), that were the subject of this investigation, are located in north-eastern Victoria (around 36°38'S, 145°41'E). The climate is cool-temperate with a mean minimum annual temperature of 9.4 °C and a mean maximum annual temperature of 21.0 °C. Rainfall is winter-dominated and averages 650 mm p.a. (Australian Bureau of Meteorology, 2016, Station 82016). Gilgai wetlands here have an open canopy of River Red Gum (*Eucalyptus camaldulensis* Dehnh.) or Grey Box (*Eucalyptus microcarpa* (Maiden) Maiden), with an understorey dominated by grasses, forbs and sedges. All existing remnants have been subject to disturbances ranging from light livestock grazing through to cultivation. After initial field assessments we identified 17 sites for inclusion in this study that were representative of our management treatments. Sites ranged in size from 5-200 ha, this size difference is unlikely to be a confounding factor because both smaller and larger sites occur in every category of land management. Sites were distributed over a geographic area of approximately 100 000 hectares. The sites were all considered remnant vegetation in various grades of condition which we assigned to one of the following land management categories: (1) currently cultivated; (2) previously cultivated (with no cultivation for > 10 years); (3) moderately-heavily grazed; (4) lightly grazed; and (5) previously grazed (but no grazing for ≥ 10 years). All grazed sites had no history of cultivation. There were three-four sites in each category (Table 1).

2.3 Micro-relief characterisation and plant diversity

Within each remnant the micro-topography was mapped along a 100 m transect, except for two cultivated sites which had shorter transects (85 m and 90 m) due to changes in slope across the site. The micro-topographic profile was surveyed at approximately 1 m intervals using a theodolite (Leica EDM Total Station Theodolite, Wetzlar, Germany). A linear regression was fit to the survey data to remove any broad-scale slope (Figure 1a). The residuals of the regressions were used to calculate

maximum depression depth, mound height and profile (maximum depth plus maximum height; Figure 1a) for each transect.

We assessed the effect of grazing intensity and livestock removal on plant diversity by recording the presence of indicator species within a 0.1 m² quadrat placed every 5 m along multiple 100 m transects. The number of transects was semi-proportional to the remnant area: <15 ha, 1 transect; 16-40 ha, 2 transects; > 200 ha, 5 transects. Indicator species were chosen because there are often low species densities in native grassy woodlands due to past grazing (Lunt 2003). The following seven species were selected following the methodology of Lunt (2003): Bulbine Lily (*Bulbine bulbosa* (R.Br.) Haw), Lemon Beauty Heads (*Calocephalus citreus* Less.), Swamp Billy-buttons (*Craspedia paludicola* J.Everett & Doust), Blue Devil (*Eryngium ovinum* A.Cunn.), Native Yam Daisy (*Microseris* sp. D.Don), Curved Rice-flower (*Pimelea curviflora* R.Br.) and Kangaroo Grass (*Themeda triandra* Forssk.). These species were chosen due to their ease of identification, their documented sensitivity to various intensities of grazing (Robinson 1998; Vesk and Westoby 2001; Lunt 2003); and their known presence within the study area (Ross and Robinson 2001).

Single-factor ANOVAs and Tukey's HSD *post-hoc* tests were used to test for significant differences between land management categories in micro-topographic features and diversity of plant indicator species. All analyses were performed in R 3.2.1.

3. Results

There were no significant differences in maximum depression depth across land management categories ($F_{4,12} = 1.73$, $P = 0.21$). Mean depression depth varied from 7.2-17.4 cm (Table 1). There were however significant differences in maximum mound height ($F_{4,12} = 12.67$, $P < 0.001$) and maximum profile ($F_{4,12} = 6.47$, $P = 0.005$) due to cultivation. Both currently and previously cultivated remnants had significantly lower mounds (13.4-20.6 cm) compared to remnants without any cultivation history (i.e. grazed sites: 29.0-33.0 cm). Currently cultivated remnants also had significantly lower profiles (13.4 $\pm$ 7.7 cm) compared to those which had never been cultivated (29.0-33.0 cm). However, sites released from cultivation, for at least 10 years, had mean profiles (20.6 $\pm$ 10.3 cm) which did not differ significantly to sites which had never been cultivated. Within grazed and previously grazed remnants there was no evidence of grazing pressure affecting maximum mound height or maximum profile ($P > 0.05$; Table 1).

Grazing pressure significantly affected the diversity of plant indicator species ($F_{2,7} = 4.96$, $P = 0.046$). Remnants with livestock removed, for at least 10 years, had significantly more indicator species compared to moderately-heavily grazed remnants ($P < 0.05$; average number of indicator species in previously grazed remnants: 5.0 $\pm$ 0.9; and in moderate-heavily grazed remnants: 0.5 $\pm$ 0.5). Lightly

grazed remnants had intermediate numbers of indicator species (3.0 ± 1.2), not significantly different from either of the other two classes of grazing intensity.

4. Discussion

Increasing agricultural intensification affected both micro-relief and floristic diversity of gilgai wetlands. As expected the spatial patterning of the micro-topographic features was homogenized by cultivation (Table 1). This is consistent with Thompson and Beckmann (1982) who found that cultivation transfers soil from the mounds to nearby depressions. However, the micro-relief showed some recovery after relief from cultivation, indicating that the gilgai soil micro-relief is resilient to intensification. This recovery of micro-relief may occur because the subsoil variation driving Gilgai formation can be > 1 m deep (Paton 1974), which is below the level of cultivation (Thompson and Beckmann 1982). Additionally, the swelling and shrinking associated with gilgai formation is a continuous process in response to repeated wetting and drying (Knight 1980). The micro-relief was also insensitive to grazing pressure. On some soils, grazing reduces micro-topographic variation and soils frequently become compacted (Yates *et al.* 2000). However, gilgai formations occur on vertisols which are considered to be less prone to deformation (Knight 1980).

While grazing-sensitive species were vulnerable to agricultural intensification, the diversity of indicator species was higher in sites where grazing pressure was reduced or removed for more than 10 years (Table 1), possibly indicating a degree of recovery. This is similar to findings in other types of grassy ecosystems in predominantly agricultural landscapes (Prober *et al.* 2002). These findings indicate that passive restoration is possible in such sites for recovery of micro-relief and some wetland function and biodiversity values following decreases in the intensity of cultivation and stock-grazing. However, we note that full recovery of these systems following major disturbance is unlikely, at least in relatively short time frames, and dependent on distance from propagules and colonisation potential of a given species (Moreno-Mateos *et al.* 2012). In the current context of highly modified agricultural landscapes, and climate change, however, these micro-wetlands play an important role in enhancing biodiversity, ecosystem function and landscape connectivity for a large range of aquatic and dryland species (Paton 1974, Céréghino *et al.* 2014, Casanova & Casanoca 2016; Calhoun *et al.* 2017). They also have an important role to play as drought refugia, even when dry, by providing shelter for small mammals and reptiles in the cracking soil (Waudby & Petit 2017), making their retention and restoration even more important in the context of a drying climate.

Our study showed that there is capacity for gilgai wetlands to recover from degrading land-uses such as over-grazing and soil cultivation. As in other parts of the world, however, these small, temporary wetlands have not been well recognised or studied, in comparison with other aquatic ecosystems

For this reason, and because of their preponderance on agricultural land, they are consequently very poorly conserved (Wood *et al.* 2003; Céréghino *et al.* 2014; VEAC 2017). The protection of micro-wetlands from destructive or degrading land-uses is therefore critical to enable their maintenance and restoration (Céréghino *et al.* 2014, Calhoun *et al.* 2017) and should be encouraged as a priority activity under agri-environment schemes wherever micro-wetlands occur. Without that active intervention and restoration, their loss is set to continue (Casanova & Casanova 2016; Calhoun *et al.* 2017).

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Tables

Table 1. Summary of micro-topographic attributes and diversity of plant indicator species in remnants of Gilgai grassy woodlands subject to varying land management. The maximum profile is

the sum of the maximum mound height and maximum depression depth. Within a column, different letters among land management categories indicate means were significantly different ($P < 0.05$).

Land management type	Number of sites	Micro-topographic attributes (cm; ± 1 SE)			Diversity of plant indicator species (± 1 SE)
		Max. mound height	Max. depression depth	Maximum profile	
Cultivated	3	6.1 $\pm$ 3.5 ^a	7.2 $\pm$ 4.2 ^a	13.4 $\pm$ 7.7 ^a	NA
Previously cultivated	4	7.8 $\pm$ 3.9 ^a	12.8 $\pm$ 6.4 ^a	20.6 $\pm$ 10.3 ^{ab}	NA
No history of cultivation					
Moderately-heavily grazed	4	15.6 $\pm$ 7.8 ^b	17.4 $\pm$ 8.7 ^a	33.0 $\pm$ 16.5 ^b	0.5 $\pm$ 0.5 ^b
Lightly grazed	3	14.1 $\pm$ 8.2 ^b	14.9 $\pm$ 8.6 ^a	29.0 ^b	3.0 $\pm$ 1.2 ^{ab}
Previously grazed	3	16.1 $\pm$ 9.3 ^b	14.9 $\pm$ 8.6 ^a	31.0 $\pm$ 17.9 ^b	5.0 $\pm$ 0.9 ^a

Figure legends

Figure 1 (a) Example of a micro-topographic survey showing height variation (cm) across a 100 m transect, with a linear regression fitted. Also shown is the maximum depression depth, maximum mound height and maximum profile height (sum of maximum depression depth and mound height).

(b) Example of micro-topographic profiles with broad-scale slope removed (i.e. the residuals from the fitted linear regression). Shown are example profiles of a currently cultivated remnant, previously cultivated remnant and a currently grazed remnant.

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