Robin Hale, Paul Reich, Tom Daniel, Philip S. Lake, Timothy R. Cavagnaro

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Assessing changes in structural vegetation and soil properties following riparian restoration

Running head: Vegetation and edaphic responses to riparian restoration

Robin Hale[1,3,*], Paul Reich[1,2], Tom Daniel[1], Philip S. Lake[1] & Timothy R Cavagnaro[1,4]

[1]School of Biological Sciences, Monash University, Clayton, Victoria, Australia, 3800

[2]Arthur Rylah Insitute, Department of Environment, Land, Water and Planning, Heidelberg, Victoria, Australia, 3084

[3]Current address: School of BioSciences, University of Melbourne, Parkville, Australia 3010

[4]The Waite Research Institute and The School of Agriculture, Food and Wine, University of Adelaide, Waite Campus, PMB 1 Glen Osmond, South Australia, Australia, 5064

***Author for correspondence:**

Dr. Robin Hale
School of BioSciences
University of Melbourne
Parkville
Victoria, Australia, 3010
Ph: +61 437 194 515
Email: robin.hale@unimelb.edu.au

Abstract

Efforts are underway in many areas to restore riparian zones to arrest and/or reverse their degradation and the subsequent loss of the ecosystem services they provide. Despite strong links between edaphic conditions and riparian zone function, limited research has tested how soil properties respond to restoration, especially in an experimental context. With this important knowledge gap in mind, we established a field experiment to asssess structural vegetation and soil responses in the eight years following ~~to~~ livestock exclusion and replanting in lowland streams in south-eastern Australia. On three streams, paired restored and control sites were experimentally established and we monitored vegetation (stem density, cover of bare ground and tree canopy, and loadings of organic matter), ~~s~~ once ~~.~~beforehand, and then biennually after~~w~~ restoration. Selected soil properties~~oils~~ (total carbon, total nitrogen, plant-available phosphorus) were sampled once shortly after restoration, then after another five years. Significant changes in structural vegetation occurred (e.g. decreased bare ground, increased plant stem density, organic matter, and canopy cover. In contrast, those soil properties did not respond. ~~resulting in subsequent changes in soil properties (e.g. increases in soil carbon concentration). While vegetation changed significantly following restoration, soil responses did not.~~ A mega-drought occurred throughout much of the study period which was immediately followed by severe flooding. The floods redistributed organic matter at our study sites, with this effect mediated by vegetation structure: the probability of organic matter retention was positivley correlated with groundcover and stem density of plants. The timing of flooding was also correlated with increased soil carbon and nitrogen, which could be due to increased productivity in these systems (for the former), or potentially due to increased fertiliser inputs or increased fixation (for the latter). ~~Temporal changes in soil properties occurred that coincided with flooding and the redistribution of organic matter. Variability in this redistribution was mediated by vegetation structure – with the probability of retention increasing where groundcover and stem density of plants was highest.~~ Our study ~~provides~~ is the first to comprehensive and experimentally test ~~comprehensive experimental study examining~~ how vegetation, litter layer and surface soil properties respond following riparian restoration, and will help guide the development and implementation of other monitoring programmes.

Key words: agricultural landscapes, Australia, edaphic, land-use, livestock removal, revegetation

1. Introduction

Riparian zones act as the interface between aquatic and terrestrial ecosystems, and are often among the most productive and biodiverse areas in landscapes (Naiman *et al.*, 2005). Riparian zones provide a range of important ecosystem services, for example as habitat for flora and fauna (Naiman *et al.*, 2005), and carbon sequestration (Smukler *et al.*, 2010; Smith *et al.*, 2012). One of the most important roles that riparian zones play is to regulate the transfer of nutrients and sediments into waterways (Likens *et al.*, 1970), reducing the risk of eutrophication and biodiversity loss in aquatic environments (Naiman *et al.*, 2005). This is especially important in highly modified agricultural landscapes where riparian vegetation is often in poor condition, and nutrient inputs, as well as rates of erosion and surfacewater runoff, are typically high (Lovett and Price, 1999).

Despite their valuable ecosystem services, in many areas of the world riparian zones are highly degraded, and the pressures upon them are likely to increase under climate change, as they remain relatively more fertile and moist while upland areas become hotter and drier (James *et al.*, 2016). There is, however, an increasing recognition of the need to undertake management activities that attempt to return these ecosystem services (Naiman *et al.*, 2005), generally by excluding livestock from the riparian zone and replanting native vegetation. While monitoring is critical to evaluate the success of these activites, it is rarely undertaken effectively, if at all. Consequently data required to demonstrate responses are rare and urgently needed (e.g. Brooks and Lake, 2007; Reich *et al.*, 2016). Typically, when monitoring is undertaken, the emphasis is on assessing changes in structural measures (e.g. vegetation cover) rather than changes in ecosystem function (Palmer and Febria, 2012).

Most nutrients entering waterways either pass through or over the soil surface depending upon their mobility in the soil environment (Likens *et al.*, 1970). Edaphic conditions can strongly influence the ability of riparian zones to filter nutrients, for example, through their key role in regulating plant growth and development. The processing of nutrients and carbon in the soil is often extremely complex and dynamic, and strongly influenced by characteristics of the soil, for example, organic matter composition and soil microbial community composition (Smukler *et al.*, 2010; Mackay *et al.*, 2016). The transformation of nutrients in the soil, which is largely driven by microbial processes (Sathya *et al.*, 2016), can ultimately determine whether or not nutrients reach waterways (e.g. Gift *et al.*, 2010). Given the pervasive links between soil processes and the overall functionality of riparian zones, it is important to not only consider soil properties (e.g. soil nutrients and C) as drivers of change, but also as valuable measures of restoration success.

Despite the importance of soil properties to the function of riparian zones, few studies have examined how they might change following restoration. This can be in part be attributed to the difficulties associated with soil sampling, the large degree of spatial heterogeneity in some properties (e.g. Hale *et al.*, 2014) and the potentially long lag times in response to changed management (Post and Kwon, 2000; Burger *et al.*, 2010; Gift *et al.*, 2010; Matzek *et al.*, 2016). The exceptions have generally been observational rather than experimental (e.g. Burger *et al.*, 2010; Smukler *et al.*, 2010; Mackay *et al.*, 2016). Dedicated experiments are needed to properly characterise changes in soil properties, and to identify the underlying drivers of these responses. In addition, as knowledge improves of how soils respond to management, it may be possible to identify more easily measurable variables that can be used as proxies to assess changes in soil properties (e.g. using canopy cover to predict riparian soil carbon - Smith *et al.*, 2012).

Here, we present results from an experiment established at three riparian locations in south-eastern Australia to test how soils respond to livestock removal and replanting vegetation. We had two main aims: (1.) assess potential changes in structural vegetation properties following restoration and (2.) test if and when these responses led to subsequent changes in soil C, N and plant-avaliable P. Our first aim relates to the success of restoration implementation (i.e. do plants grow and survive), and how this development of replanted vegetation might change conditions within the riparian zone. While changes to soil properties might be predicted to be inevitable if restoration is successful, this assumption has not been tested, and it is also largely unknown when such changes are likely to occur. These two knowledge gaps were the basis for aim 2.

We initially developed a conceptual model outlining our predictions about when soil properties might change and the underlying drivers (Fig. 1). While a wide range of soil properties could change in response to replanting, we focussed on soil nitrogen, phosphorus and carbon. These are likely to be inherently less variable than some other parameters that could change following replanting (e.g. soil microbial community dynamics and mineralisation rates - Mackay *et al.*, 2016), and thus be more suitable for detecting responses in the medium- to long-term (Hale *et al.*, 2014). We hypothesized that soil nitrogen and phosphorus might decrease initially following livestock removal and thereafter through increased uptake as groundcover develops, based on evidence that soil physicochemial properties change following reforestation (Cunningham *et al.*, 2015), and soil nutrient levels often decrease following restoration due to a number of factors including a cessation of fertiliser inputs, increased nutrient demand with a shift to tree-based vegetation, reduced levels of biological nitrogen fixation from leguminous pasture species, and increased nutrient immobolisation (Hooker and Compton, 2003). Work in the study region (Burger *et al.*, 2010) has demonstrated that soil phosphorus in the riparian zone can be influenced by adjacent land use, especially fertilizer inputs, and we predicted therefore that this could override any response to restoration. We predicted that increases in soil carbon would occur in response to increased tree canopy cover (Post and Kwon, 2000; Burger *et al.*, 2010; Mackay *et al.*, 2016), which is unlikely in the study region for at least 10 years, based on the growth rate of the dominant riparian tree species in the study region, the river red gum *Eucalpytus camaldulensis* (CSIRO, 2004). However, we anticipated an increase in soil C:N ratios with time since restoration due a small increase in soil C due to increased plant inputs, and a decrease in soil N due to enhanced plant demand. There is some precedence for this with previous studies in riparian and non-riparian systems showing an increase in soil C:N with restoration (Cavagnaro, 2016; Cavagnaro *et al.*, 2016).

Monitoring has been undertaken for eight years following restoration. While to our knowledge this is one of only a very few, if not the only, studies/study to monitor reponses of vegetation and soil to experimental, riparian restoration, it still represents only the early days along the ultimate trajectory of response. However, such updates are vital, presenting an intermediate assessment upon which to update our potential predicted responses. Also our study began during the most peristent and severe drought in south-eastern Australia since instrumental records began (Timbal and Fawcett, 2013) and continued throughout the breaking of the drought. Environmental conditons that occurred throughout this period were extreme, with 40% below long-term average rainfallgenerally ~500 mm/year during drought and higher than average rainfall (100-150 mm above average) which caused severe flood events at all sites when breaking (Supplementary Figure S1). Such extreme events could potentially alter responses to restoration (Reich and Lake, 2015). As a consequence, we were able to address a third aim: (3) to test how floodplain inundation alters the quantity and distribution of surface organic matter. In particular, we were interested in testing how the probability of losing or retaining surface organic matter might vary as a function of structural elements on the floodplain (e.g. coarse wood, stem density of plants). We anticipated that vegetation structure would influence organic matter dynamics during inundation by governing the retentive capacity of the floodplain. Examining these relationships may shed light on temporal changes in soil properties (especially soil C) that relate to factors unrelated to changes in riparian management. For example, sites with less retentive capacity (e.g. without coarse wood, fewer plant stems) might lose more surface organic matter during flooding, and in turn be places where rates of soil carbon accumluation are reduced

2. Materials and Methods

2.1 Study sites and climate

We selected sites to be representative of typical, small lowland streams in the Murray-Darling Basin (MDB), south-eastern Australia, in an area where riparian restoration is becoming increasingly common (Brooks and Lake, 2007). Our sites met the following criteria: catchment size > 75 km^2, annual rainfall 450-850 mm, stream order 2-5, altitude <500 m, valley slope <1.2. Over ~34,000 km of the stream length of the MDB (~25%) has these characteristics, and our sites therefore reflect the types of sites that are commonly the focus of restoration efforts in the area.

Sites were located on three small lowland streams in the Murray-Darling Basin in south-eastern Australia, Middle (-37.139, 143.913) and Joyces (-37.127, 143.962) creeks in the Loddon River catchment, and Faithful Creek (-36.619, 145.523) in the Goulburn River catchment (Fig. 2). This landscape is highly degraded as a result of the effects of a range of anthropogenic disturbances over the past century, in particular land clearance, mixed grazing and fertiliser application. The riparian zone along these streams are dominated by river red gum (*Eucalyptus camaldulensis* Dehnh), typically consisting of a strip only one or two trees wide with an understory of mainly exotic grasses (Williams *et al.*, 2008). Mean annual adjacent land use based on dry sheep equivalents (DSE) (Griffiths, 1998) was 9.64 (±0.24 se) DSE/days/ha at control, and 3.90 (±0.24 se) at treatment sites following restoration. Three pairs of sites were sampled, with a paired "treatment" (livestock removed and native tubestock planted) and "control" (unchanged management practices) site located on each of the three creeks. At each creek, sites were ~1 km long, with the control located approximately 3-4 km upstream to ensure independence from restoration activities. At all treatment sites, livestock were removed, native shrub and tree species planted as tubestock, and the site fenced to an average width of 20 m on both sides of the channel. Tubestock replantings were guided by modelled historical vegetation communities and local conditions (DSE, 2009). Livestock were initially removed from all sites in 2005, just prior to replanting. Sites were sprayed with a broad spectrum herbicide (glyphosate) and tubestock replanted in evenly spaced riplines (2-4 m apart) running parallel to the stream. Some additional replantings occured at Middle and Joyces Creek in 2006 and 2007 due to variabilty in tubestock survival between sites. For details on the temporal sequence of the study (i.e. when restoration occurred, when field sampling was undertaken), see Supplementary Fig. S2a, and for further details of restoration methods see Reich et al. (2009). We outline below the methods we used to sample a range of response variables within the riparian zone and adjacent floodplain and paddocks, see Supplementary Fig. S2b for a visual representation of the sample scheme.

2.2 Vegetation sampling

Vegetation was sampled in the austral summer, once before restoration (2005) and three times afterwards (biennually from 2009 onwards). Sampling was undertaken at six randomly selected permanent transects running perpendicular to the stream channel. Transects were located along the length of the site, separated by at least ~75 m, within the riparian zone (0.5-3.5 m from bank full height on the floodplain-herafter "Riparian"). Methods followed Williams et al. (2008) and Hale et al. (2015), with five randomly placed 1 m^2 quadrats used to estimate the percentage cover of bare ground, dead organic matter (e.g. dead unattached plant matter, leaf litter, fruiting material) and total plant cover. We supplemented our visual estimates with quantitative sampling of coarse particulate organic matter (organic particles > 10 mm - Cummins, 1974, hereafter "CPOM"). This involved the collection of all organic matter at the centre of each of the five quadrats within a circular (30 cm diameter) sampling frame. To examine whether CPOM differed with distance from the channel as a result of flooding, in 2009 and 2011 CPOM samples were collected at two additional locations at all sites- from within the area between the bank toe and bank full height (hereafter "Bank"), and from 9.5-12.5 m above bank full extending onto the floodplain (hereafter "Floodplain"). Samples were stored and transported to the laboratory in zip-locked plastic bags. In the laboratory, samples were transferred to a sieve and washed to separate the CPOM fraction from finer material, and large stones and gravel were removed. Each sample was then oven-dried at 70^o C to constant weight for 12-72 hours to determine dry weight, and burnt in a muffle furnace at 500^o C. Remaining ash was then weighed and ash free dry weight calculated as dry weight minus ash weight. Reliable relationships were detected between dry weight and ash free dry weight of CPOM for each site (linear regression $R^2>0.95$), so for some later samples, ash free dry weight was calculated ($AFDW/m^2$) using the linear regression equation of this relationship. Stem density was calculated as the total number of plants (number of planted tubestock plus naturally recruiting seedlings and number of trees) observed in each of the bank, riparian, and floodplain sampling zones at each transect (Fig. S2b). Canopy cover was estimated at Riparian and Floodplain locations at all sites in 2005 (i.e. before planting) and again in 2011 and 2013 using hemispherical photos which were taken in late summer-spring and images processed using Gap Analyser (v.2). Coarse wood (any organic matter >0.1 m diameter and >1 m length) was surveyed at all sites in 2013 using published methods (Webb and Erskine, 2003), whereby individual pieces and accumulations of coarse wood are measured and expressed as volume per unit area.

2.3 Soil sampling- field and laboratory methods

Soil sampling was undertaken at all sites in the austral winter of 2007 and again in 2012. The timing of these two samples was chosen based on the hypothesized timing of likely changes in soil properties identified in Figure 1. Sampling followed methods used in an early study at these sites (Hale *et al.*, 2014), with samples collected from the Riparian locations where vegetation sampling occurred. We also collected additional soil samples from adjacent paddocks (47-50 m from bank full height onto the floodplain) to examine potential links between phosphorus in riparian zones and inputs from nearby paddocks. Soil cores were taken from the 0-100 mm soil layer using a hand auger at ten randomly located positions along each transect. The cores from within each transect were combined, and a 2 kg sample of which was stored at 4^{o} C until return to the laboratory for further analysis, leaving a total of N = six soil samples per site on each sampling occasion. Before physicochemical analysis, soil samples were sieved (<2 mm) to remove rocks, coarse roots and other debris. Total Carbon and Total Nitrogen were measured by dry combustion with a Leco 2000 CNS analyser, and C/N ratios calculated (Schipper and Sparling, 2000). Plant-available phosphorus was determined using the Mehlich-3 extraction method (Carter and Gregorich, 2008).

2.4 *Statistical methods*

2.4.1 Analysing temporal changes in vegetation and soil properties

Linear mixed-effects models (lme function in the nlme package in R - R Development Core Team, 2009) were used to examine potential responses to changed management practices. For all variables (i.e. vegetation, CPOM, soils), models were run with Treatment (i.e. treatment or control), Year and Treatment*Year as fixed effects, with Creek (n = 3) as random, following the protocol outlined in Logan (2010). This approach provides analogous information to a repeated measures ANOVA. There were subtle differences in the analysis of different response variables. For vegetation variables sampled in 2005 (i.e. before livestock removal and replanting), we corrected data by subtracting the values for different variables observed in these initial samples (i.e. Value for time *x* – value prior to restoration). For these models, the Treatment term therefore describes whether a particular variable is different (i.e. a change that is statistically significant) after restoration, after accounting for values prior to restoration. For variables only sampled after 2005 (soil properties and CPOM), the Treatment term describes whether treatment and control sites are statistically different, but this could potentially be due to the continuation of pre-treatment differences between sites. In this case, the Treatment*Year term provides the test of potential responses to treatment. For analyses examining potential changes in riparian soil phosphorus, we included phosphorus concentrations in adjacent paddocks as a covariate, as adjacent land use can have a strong influence on riparian soil phosphorus in the region (Burger *et al.*, 2010).

2.4.2 Assessing changes in organic matter in response to flooding

We calculated the change in CPOM for each sampling location (n = 108, i.e. 6 sites x 3 lateral zones within each site x 6 permanent sampling locations within each zone) as the amount of CPOM in 2011 minus the amount of CPOM in 2009. We designated sampling locations where CPOM was lost as 1, and retained as 0 and modelled the probability of losing CPOM on two categorical factors (Creek and Lateral Zone) and three descriptors of structural vegetation (stem density, coarse wood, groundcover structure). Groundcover structure (calculated as the cover of plants and attached organic matter) and stem density were estimated in 2011. While measured in 2013, coarse wood loadings are unlikely to have changed appreciably as there were no major flooding events since 2011 when CPOM was sampled, and coarse wood can take several decades, even longer, to accumulate after replanting (Mac Nally and Horrocks, 2002). We initially ran a full model containing all factors, which indicated a significant Lateral Zone effect. We therefore ran separate models for each Lateral Zone initially including all factors then following a stepwise iterative approach where factors were removed based on changes in the Akaike Information Criteria (AIC). The fit and appropriateness of the model were evaluated with goodness-of-fit-tests following Logan (2010).

3. Results

3.1 Changes in structural vegetation properties

We detected significant changes in structural vegetation properties following replanting and livestock removal (Figure 3, Table S1). Stem density increased at treatment sites, due predominantly to planted tubestock (Figure 3a, Table S1 “Treatment” term $F_{1,10}$ = 11.93, p=0.01) although natural recruitment of woody species occurred at both treatment and control sites. Bare ground increased through time at control sites, but remained relatively constant, and low (<10%) cover at treatment sites (Figure 3b, Table S1 “Treatment” term $F_{1,10}$ = 5.12, p=0.04). Cover of dead organic matter, plants, and twigs were comparable between treatment and control sites (Table S1). Canopy cover was significantly higher on the floodplain at treatment sites (Figure 3c, Table S1), but not within the riparian zone. Overall, loadings of CPOM did not differ between treatment and control sites (Figure 3d, Table S1). However, the mean loading was higher at treatment sites in 2011 and 2013.

3.2 Loss and retention of organic matter after flooding

CPOM was most likely to be retained at locations with a greater degree of structure, particularly groundcover (Figure 4, Table S2), and this response was stronger in the riparian and floodplain zones than the Bank (Figure 4). The most parsimonious models for the Riparian and Floodplain zones based on AIC values also included stem density, and riparian coarse wood indicating that these structural elements may also be important, although their effects were not statistically significant.

3.3 Changes in soil properties

There was considerable temporal variability in all soil variables across both treatment and control sites i.e. changes through time not related to restoration (Table S3, “Year” term p<0.05 for all models). Neither total carbon nor total nitrogen concentration had responded to restoration, although overall increases in both variables were detected over time (Figure 5 a-b, Table S3). The C/N ratio of the soil decreased through time across both treatment and control sites, but remained significantly higher at restored sites (Figure 5c, Table S3). Concentrations of plant-available phosphorus in the soil were lower at treatment and control sites in 2012 compared to the earlier sampling time (Figure 5d). The concentration of plant-available phosphorus was also positively related to plant-available phosphorus in adjacent paddocks, although this relationship was not statistically significant (Table S3).

4. Discussion

Significant changes were detected in structural vegetation following replanting and livestock removal, with increased stem densities of plants, floodplain canopy cover and decreased bare ground. CPOM was not significantly different overall (although higher mean values were recorded at treatment sites in 2011 and 2013) and flooding led to a redistribution of organic matter at all sites. CPOM loss was greatest at locations with high percent cover of bare ground. In comparison, temporal changes in soil properties unrelated to restoration were observed (e.g. increases in soil carbon and nitrogen, decreases in plant-available phosphorus), but hypothesized responses to livestock removal and replanting did not occur. Interestingly, soil C/N ratios – a key driver of soil microbial activity – decreased over time, but were generally higher in restored sites. In summary, our results indicate that while significant changes in structural vegetation were observed, changes in soil properties may be slower to occur.

4.1 Vegetation responses

It is well established that livestock can degrade riparian plant communities (Robertson and Rowling, 2000), and that grazed riparian zones generally have less tree regeneration, fewer shrubs, and less groundcover biomass than in ungrazed areas (Kauffman and Krueger, 1984; Robertson and Rowling, 2000). The changes in groundcover we observed following restoration were consistent with responses that have been observed in other studies (e.g. Robertson and Rowling, 2000; Wevill and Florentine, 2014). While some of these responses, for example, increases in stem density, are not surprising given they simply reflect planting of tubestock, reporting on them is still important, as it provides an indication that tubestock have successfully established and grown. It also provides useful baseline data for future studies of this nature.

We predicted (Fig. 1) that canopy cover may take several years to increase. The main tree species planted, river red gum (*Eucalyptus camaldulensis*), can reach >10 m height within several years of planting (CSIRO, 2004). Trees had not reached these heights during our study, especially within those replanted within the riparian zone, which may explain why we did not observe the increase in riparian canopy cover that occurred further out on the floodplain. It is likely that more pronounced increases in canopy cover will occur in the future as planted tubestock continue to grow and mature, especially if wetter conditions like those observed following 2011 continue.

No overall differences in CPOM were detected, despite the mean loading being higher at treatement sites in both 2011 and 2013. It was not unexpected that this response was more complex than for other groundcover and litter variables measured, given that the main contribution to this litter fraction is derived from trees and shrubs. The recovery post-restoration of woody vegetation is likely to be slower than that of the ground cover layer which responds relatively rapidly to livestock exclusion (Sarr, 2002; Hough-Snee *et al.*, 2013). It is likely therefore that significant increases in CPOM may take several more years to occur, as the canopy develops in the longer-term over the riparian zone.

The distribution of organic matter at study sites was greatly influenced by flooding in 2010 and 2011, which marked the end of a long period of record drought and resulted in floodwaters breaching the stream channel. Movement of floodwaters redistributed organic matter on the floodplain, and back into the stream channel as water levels receded. Locations with higher cover of bare ground, and thus limited structural vegetation, lost relatively the most CPOM, and this effect was most pronounced at sampling locations on the floodplain relative to those areas sampled closer to the stream channel. While previous studies have demonstrated that the retention of in-channel CPOM can be mediated by structural vegetation (e.g. coarse wood - Quinn *et al*., 2007), to our knowledge this is the first evidence of a similar phenomenon occurring on floodplains. The influence of flooding events on soil responses to riparian management warrants further attention. It should also be considered in the context of efforts seeking to build up accurate models of carbon stocks in revegetated riverine farmlands.

4.2 Soil Responses

Our results illustrate that plant-available phosphorus in the soil decreased at both treatment and control sites. Our earlier work in the region is consistent with our findings here that adjacent land use is likely to be a stronger influence on concentrations of riparian phosphorus than processes occuring in the riparian zone itself (Burger *et al.*, 2010). This relationship between riparian and adjacent paddock plant-available soil phosphorus can be attributed to the movement of phosphorus attached to soil particles entering the riparian zone via erosive processes, rather than in the soil solution (Lucas *et al.*, 2005; Naiman *et al.*, 2005). We predicted that restoration would decrease phosphorus concentrations through the combined of effects of livestock removal leading to reduced waste inputs, and replanting leading to increased plant uptake, but these responses did not occur. This highlights the need to manage riparian zones for phosphorus interception where streams are adjacent to sites where soil phosphorus is high, such as intensive livestock farming, and situations where phosphorus fertiliser use is high.

With the increase in vegetation following replanting and exclusion of livestock, we hypothesized that the concentration of soil carbon would increase at treatment sites. However, this was not apparent, most likely because of the slow rate at which carbon accumulates in soils following the implementation of restorative measures. For example, Burger et al. (2010) found a trend towards increased soil carbon in sites that had been restored for more than 12 years in the same geographic region as the present study. In comparison, soil carbon has been shown to increase following restoration in other areas of southern Australia with higher net primary productivity and rainfall (Cavagnaro, 2016). Interestingly, we observed comparable increases in total soil carbon and nitrogen concentrations through time at both treatment and control sites along the three creeks. For carbon, this may be due to a general increase in productivity at all sites, associated with a wetter period that followed the long drought in the region, and also the effects of flooding altering the spatial distribution of organic matter. In comparison, the increase in total nitrogen through time may be due to increased nutrient inputs, or greater levels of nitrogen fixation in the systems (Hoogmoed *et al.*, 2014). This, however, is speculative, and highlights the need for further studies of inter-annual variation in soil carbon and nitrogen. It would also be interesting to monitor not only changes in (plant-available) mineral forms of N in these soils, but N cycling processes (e.g. Potentially Mineralisable N) which have previous been found to respond to restoration activities where N pools did not (Smith *et al.*, 2012)

Contrary to expectations, the C/N ratio of the soils tended to decrease over time, and especially so, in the restored sites. With increasing plant cover it was expected that there would have been a drawdown of soil nitrogen as the plants grew, and an increase in soil carbon as litter inputs increased. The decrease in C/N we observed, however was small, and is unlikely to have a large effect on soil ecological process that are strongly driven by soil C/N ratios (Cavagnaro *et al.*, 2016; Mackay *et al.*, 2016). It will, however, be important to monitor changes in soil C/N ratios over the long term as they have important implications of soil nutrient and carbon cycling processes. Furthermore, we have focussed here on soil nutrients and carbon as they are of great interest due to eutrophication and potential carbon sequestration. While the emphasis here was on changes in soil properties that were expected to change over the medium to long term, more dynamic responses over the short term, such as mineralisation rates and microbial community dynamics, will also be important, as has been found in previous studies investigating riparian restoration in the region ((Mackay *et al.*, 2016)). Therefore, i~~I~~n future work, it will be important to broaden out the range of soil properties measured to include additional chemical, physical and biological variables.

4.3 Integration and Conclusions

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There has been limited work examining how riparian restoration might lead to changes in soil properties, despite the obvious links between soil processes and the function of riparian zones, and this is one of the first attempts to experimentally test how soil properties respond to changed riparian land practices. Our results illustrate that riparian restoration led to clear changes in structural vegetation but subsequent changed in the soil properties measured here have largely not occurred. One this basis, ~~Thus,~~ vegetation properties appear to provide more sensitive indicators of early change following restoration when compared to changes in the soil properties measured here. However, it is important to note that while there were no clear changes in the soil properties measured here, it is likely that there were changes in other soil properties, such as water soluble carbon, carbon and nitrogen mineralisation, and microbial biomass, activity and diversity, and are worthy of further investigation. Our results represent an important, intermediate update eight years following restoration, and confirm our expectations that changes in some soil parameters, especially carbon, are not likely to occur within the first decade following replanting. Nevertheless, we expect soil properties to exibit changes in the longer term (Fig. 1), and for these changes to have a significant impact on the recovery of ecosystem processes.

Monitoring programs often fail because the underlying questions are not clearly defined and indicators are not justified (Lindenmayer and Likens, 2010). Based on relevant research from our study region and elsewhere (including non-riparian systems), we developed a conceptual model that outlines clear hypotheses about when different responses were expected to occur and the likely underlying mechanism. These hypotheses informed both the selection of indicators to monitor, and the temporal period over which monitoring needed to be conducted to test different hypotheses. While this approach in itself is not novel, many studies do not explicitly consider these elements, and the establishment and re-sampling of experimental sites to examine responses to restoration is not routine in the soil ecology literature. Using a similar approach to ours in other contexts, for example riparian zones in other areas of the world, or soil restoration undertaken in non-riparian systems, will provide a solid basis for beginning to test the potential generality of soil responses to restoration. Given the huge effort and level of replication involved in this study, it may be prudent in other similar studies to

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delay detailed attempts to characterise responses to restoration for longer time periods over which responses might occur (e.g. 20 year). It will also be important to consider the sensitivity of different indicators and their associated time and costs requirements (Hale *et al.*, 2014) to help guide the most cost-effective monitoring program.

The data presented here also highlight the importance of considering extreme climatic events (floods and droughts) when undertaking stream and ripararian restoration, underscoring the need for long-term, well-designed monitoring programs to assess and evaluate responses (Reich and Lake, 2015). This study has provided an invaluable insight into the likely short-term responses of soil properties to riparian management and continued monitoring will allow us to assess if responses predicted to occur over longer time scales (e.g. increased soil carbon) occur.

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Figure captions

Figure 1 Conceptual model outlining hypothesized responses to livestock removal and replanting in riparian zones in the southern Murray-Darling Basin, Australia. Clear boxes illustrate likely responses to livestock removal and vegetation replanting, and grey boxes how these effects may result in changes in riparian soil properties.

Figure 2 Map of study sites. The inset map shows Australia, and an outline of the Murray-Darling Basin (grey), with our sites located with the black box. The larger map shows the location of Middle Creek (triangle), Joyces Creek (square) and Faithfull Creek (circle).

Figure 3 Changes in vegetation properties at riparian sites in the southern-Murray Darling Basin 2005-2013. Paired treatment (livestock removal and tubestock replanting) and control (unchanged management practices) were located on three creeks. Livestock removal and tubestock replanting occurred in 2005. (a.) Stem density of planted tubestock and natural recruits, (b.) Bare ground, (c.) Canopy cover on the floodplain, and (d.) Coarse particulate organic matter (CPOM). Mean (± standard error) shown.

Figure 4 Results of logistic regression model showing the probability of losing coarse organic matter following flooding in relation to groundcover structure in the (a.) bank, (b.) riparian and (c.) floodplain zones moving out laterally from the stream channel. Data pooled across six locations on three creeks in the southern-Murray Darling Basin (see Fig 2 for details). Grey lines illustrate the predictions (and se) of the model for each predictor, with the other predictors held constant at their mean value. Values of 1 on the y-axis indicated locations where organic matter was lost following flooding, and values of 0 where it was retained.

Figure 5 Changes in soil properties at sites sampled two (2007) and seven (2012) years following livestock removal and replanting of native tubestock within the riparian zone (T) and at paired control (C – no change in land use or replanting) locations at three creeks in the southern Murray-Darling

Basin (a.) Total carbon, (b.) Total Nitrogen, (c.) C:N ratio, (d.) Phosphorus. For further details of study and figure description see Fig 3 caption.

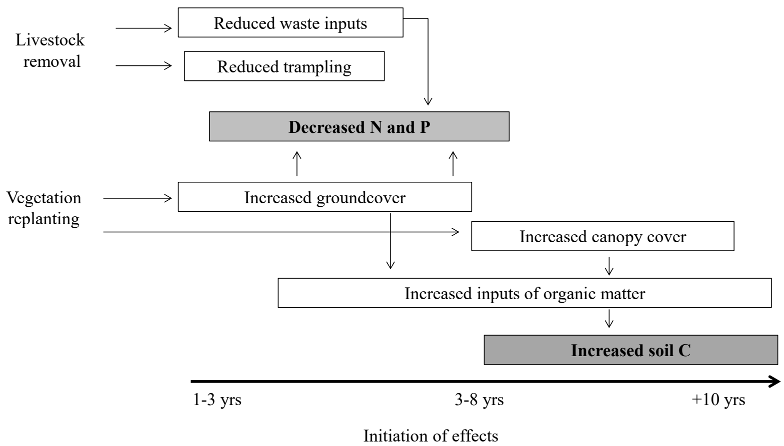

Livestock removal
Reduced waste inputs
Reduced trampling
Decreased N and P
Vegetation replanting
Increased groundcover
Increased canopy cover
Increased inputs of organic matter
Increased soil C
1-3 yrs
3-8 yrs
+10 yrs
Initiation of effects

Figure 1

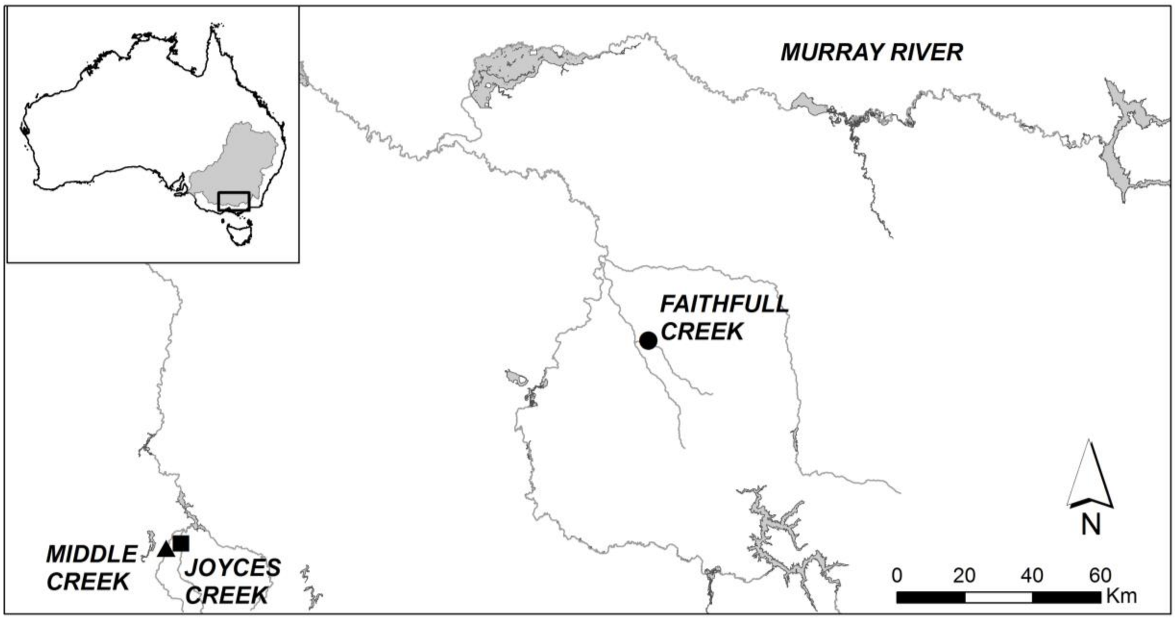
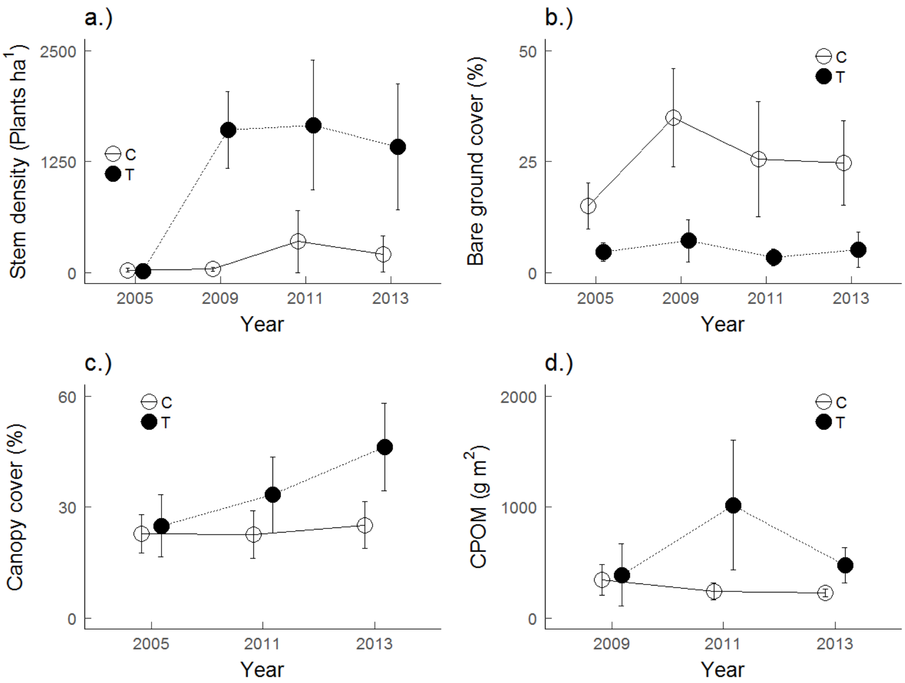

MURRAY RIVER
FAITHFULL
CREEK
MIDDLE
CREEK
JOYCES
CREEK
N
0
20
40
60
Km

Figure 2



a.)
Stem density (Plants ha-1)
2500
1250
0
C
T
2005
2009
2011
2013
Year
b.)
Bare ground cover (%)
50
25
0
C
T
2005
2009
2011
2013
Year
c.)
Canopy cover (%)
60
30
0
C
T
2005
2011
2013
Year
d.)
CPOM (g m2)
2000
1000
0
C
T
2009
2011
2013
Year

Figure 3

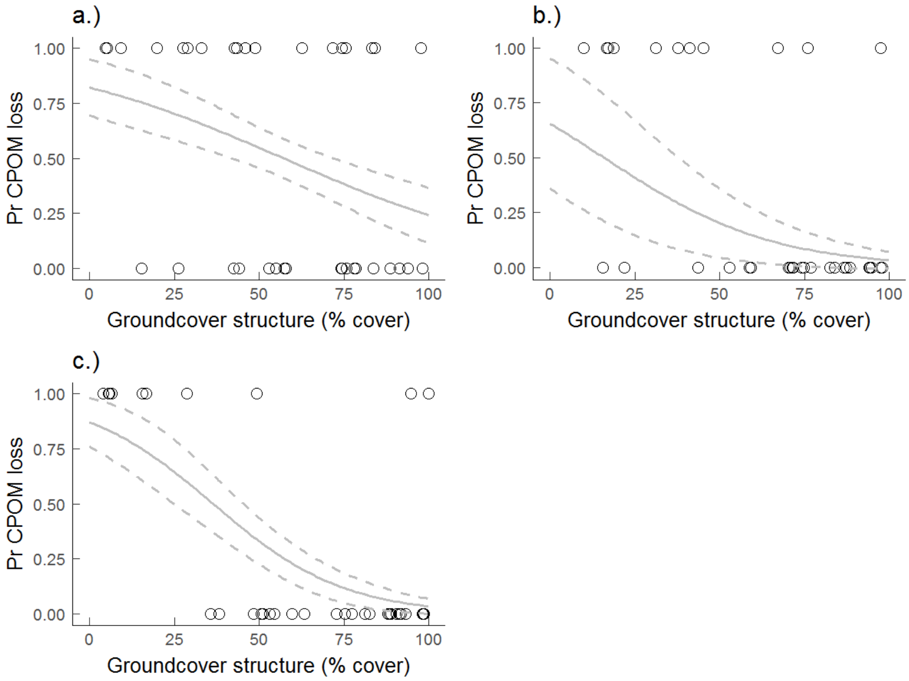

a.)
b.)
c.)
Pr CPOM loss
1.00
0.75
0.50
0.25
0.00
0
25
50
75
100
Groundcover structure (% cover)

Figure 4

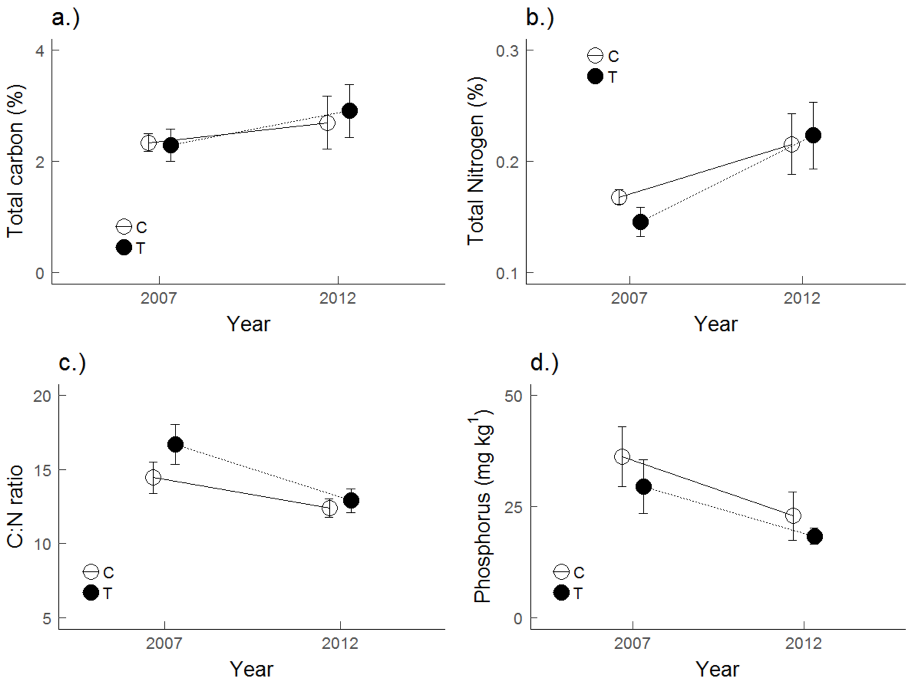

a.)
Total carbon (%)
C
T
Year
2007
2012
b.)
Total Nitrogen (%)
C
T
Year
2007
2012
c.)
C:N ratio
C
T
Year
2007
2012
d.)
Phosphorus (mg kg-1)
C
T
Year
2007
2012

Figure 5