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Assessing changes in structural vegetation and soil properties following riparian restoration

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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 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), once beforehand, and then biennually after restoration. Selected soil properties (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. A mega-drought occurred throughout much of the study 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 positively 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). Our study is the first to comprehensively and experimentally test how vegetation, litter layer and surface soil properties respond following riparian restoration, and will help guide the development and implementation of other monitoring programmes.

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 surface water 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.

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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 lead 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 (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 factors such as 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 to a small increase in soil C from 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 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 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 conditions that occurred throughout this period were extreme, with rainfall 40% below the long-term average (~500 mm/year) during drought. Higher rainfall (100–150 mm above average) caused severe flooding at all sites when the drought broke (Supplementary Fig. 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) caused by 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 streams: Middle (−37.139, 143.913) and Joyces (−37.127, 143.962) creeks in the Loddon River catchment, and Faithfull 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 zones along these streams are dominated by river red gum (*Eucalyptus camaldulensis*), 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