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RESEARCH ARTICLE

Limited recovery of soil organic carbon and soil biophysical functions after old field restoration in an agricultural landscape

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Abstract

The conversion of woodland ecosystems to agricultural landscapes has led to unprecedented losses of biodiversity and ecosystem functioning globally. Unsustainable agricultural practices have contributed to the degradation of soil's physical and biogeochemical properties. Ecological restoration of unproductive agricultural land is imperative for reversing land degradation and ameliorating the degrading effects of agriculture on biodiversity and ecosystem functions. However, it is unclear to what extent common restoration activities, such as tree planting, can facilitate the recovery of ecosystem condition and in particular, improve soil physical, biogeochemical and biotic components. Here, we investigated how the cessation of cropping, followed by tree planting, affected soil carbon concentrations and key biophysical soil functions. Data were collected across 10 sites a decade after the replanting of woody species on old fields in semi-arid Western Australia. We applied a chronosequence approach and measured soil functions in fallow cropland (restoration starting point), 10-year-old planted old fields and intact woodland reference sites (restoration target point). We stratified sampling between open areas and patches under trees in planted old fields and reference woodlands to account for inherent biophysical differences. Soils under planted trees recovered to some extent, having reduced soil compaction and higher soil penetration depth in comparison with the fallow cropland. However, soils under trees in planted old fields did not reach woodland reference conditions for these properties. Moreover, recovery was not evident for other soil physical, biogeochemical and biotic components such as soil organic carbon, soil moisture, leaf litter and woody debris decomposition rates. Limited recovery of soil functions may be at least partly explained by time lags associated with slow growth rates of planted trees in dry ecosystems. Our study shows that the legacy of cropping can persist over long timeframes in semi-arid regions, with modest signs of woodland recovery beginning to emerge 10 years after tree planting.

KEYWORDS

ecological restoration, *Eucalyptus loxophleba*, litter decomposition, soil biophysical functions, soil bulk density, soil organic carbon, temperate eucalypt woodland, tree planting, York gum

INTRODUCTION

Native ecosystems across the globe face an increasing decline in biodiversity and ecosystem functioning. In the latest Global Outlook Report, the United Nations estimates that 40% of terrestrial land is degraded (United Nations, 2022), and clearing of intact native vegetation followed by agricultural use is one of the biggest drivers of land degradation (Curtis et al., 2018; Dudley & Alexander, 2017; Potapov et al., 2022). Large scale intensive farming methods such as repeated cultivation, fertilization and chemical application have been particularly detrimental to biodiversity and ecosystem functioning (Dudley & Alexander, 2017). These typically lead to soil degradation, including eutrophication, erosion, chemical contamination, acidification, salinization, compaction, and loss of soil carbon (Guo & Gifford, 2002; Kopittke et al., 2019; Lal, 2011; Singh et al., 2017). There is an imperative to reverse degradation in agricultural landscapes for food security and climate change mitigation, as well as for biodiversity conservation.

Global ecosystem restoration targets have been set to combat and reverse the degradation of 30% of terrestrial, freshwater and marine ecosystems by 2030 (Leadley et al., 2022). Ecological restoration of degraded agricultural land has the potential to recover biodiversity loss and degraded soil properties, which underpin key ecosystem functions (Bardgett & van der Putten, 2014; Crouzeilles et al., 2016; Parkhurst, Prober, Hobbs, et al., 2021). Cessation of agricultural practices followed by the re-establishment of native vegetation either through passive (regeneration) or active (e.g., tree planting) restoration are typically the first steps to reverse the degrading effects and to reinstate functioning ecosystems (Bullock et al., 2011).

Re-establishment of native vegetation can increase plant species diversity, vegetation cover and structural complexity (Benayas et al., 2009; Crouzeilles et al., 2016; Parkhurst, Prober, & Standish, 2021). In turn, vegetation can help restore soil physical, biogeochemical and biotic attributes (Ehrenfeld et al., 2005). Soil structure, texture and biological activity are influenced by the quantity and quality of litter and woody debris input as well as root zone activity (Porazinska et al., 2003; Putten et al., 2013; Wardle et al., 1998). Leaf litter reduces surface run-off and evaporation and provides organic matter (Cadisch & Giller, 1997). Plant roots increase soil organic matter and soil porosity, which can enhance soil water holding capacity (Eldridge & Freudenberger, 2005; Prober et al., 2014b). Increased soil moisture and organic matter content in turn enhance the activity of soil microbes and soil invertebrates (Prober et al., 2014a), potentially resulting in further improvements in soil structure and texture via decomposition processes and associated nutrient cycling (Ehrenfeld et al., 2005). In particular, soils under trees have been found to have increased carbon content, moisture and abundance of microorganisms due to below- and above-ground organic matter accumulation and associated interception of water (Ochoa-Hueso et al., 2018; Prober et al., 2014a; Schlesinger & Pilmanis, 1998).

Given these well-established pathways associated with plant-soil feedbacks and interactions, ecological restoration of native vegetation and associated debris would be expected to improve soil biophysical condition (Figure 1). However, whilst the changes in vegetation cover and structure after ecological restoration have been studied widely (Young et al., 2005), research on the effects of re-establishing native vegetation on soil biophysical condition and functioning is less well characterized (Barral et al., 2015; Hua et al., 2022). A recent meta-analysis by the authors revealed that very few studies worldwide have investigated the effects of re-established native vegetation on the recovery of soil properties, and only six studies have been conducted in the southern hemisphere (Parkhurst, Prober, Hobbs, et al., 2021). The lack of research investigating the recovery of soil carbon

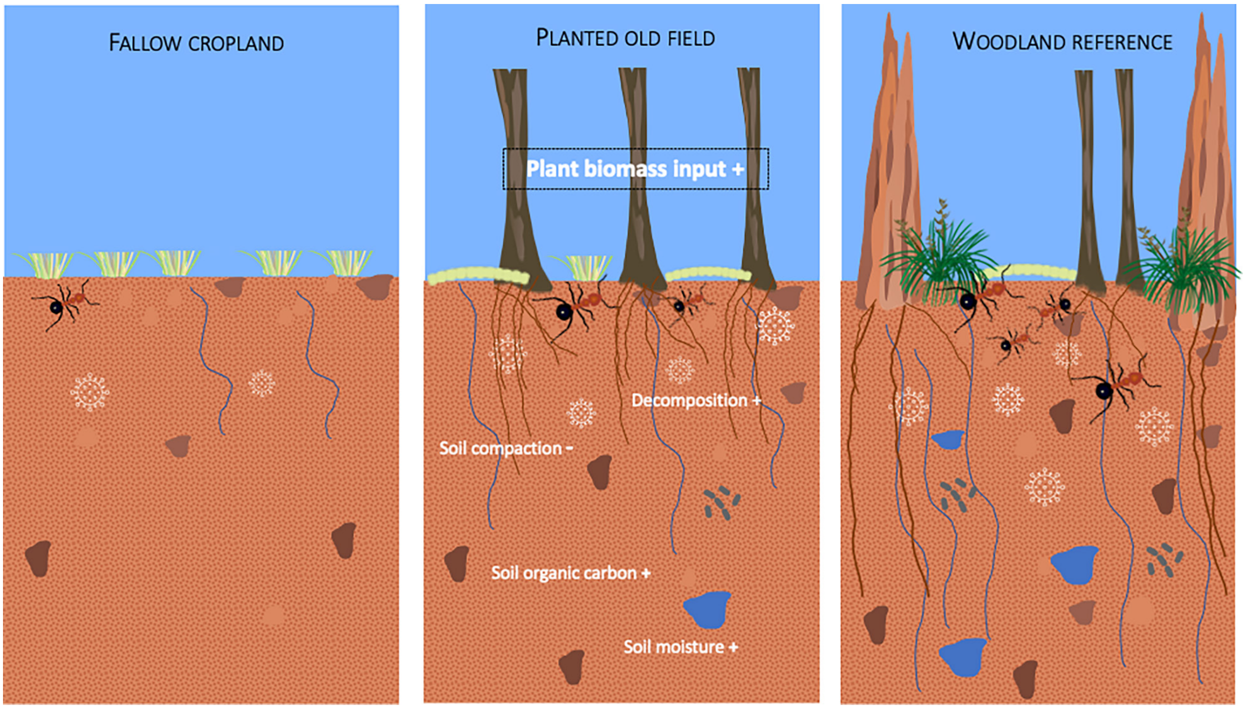

FIGURE 1 Schema of hypothesized changes in soil biophysical functions in old fields after planting perennial woody vegetation in comparison to the restoration starting point (fallow cropland) and restoration target (woodland reference).

and biophysical functions following the restoration of native vegetation on old fields is alarming and calls for further research, given that degraded soil functions (compaction, reduced soil carbon, reduced input of plant biomass and soil biological activity) limit the capacity of ecosystem recovery and consequently the likelihood of meeting ambitious global restoration targets.

Here, we investigated changes in soil organic carbon and soil biophysical functions a decade after tree planting on ex-agricultural land in a semi-arid landscape. This landscape is typical of those identified for global restoration efforts because of the potential to fund restoration with a carbon market (Strassburg et al., 2010). As illustrated in Figure 1, we hypothesized that:

1. Fallow cropland soils will be degraded compared with reference woodlands, showing low soil carbon and water holding capacity, high soil compaction and reduced decomposition activity.
2. Tree planting leads to higher soil organic carbon, reduced soil compaction, higher water availability and litter decomposition rates due to plant-soil feedbacks (plant litter accumulation, root development, pore creation, and reduced soil-surface exposure).
3. Soils under planted trees will recover more quickly than soils in open areas (inter-rows) owing to the positive influence of roots, shade, interception of water runoff and litter accumulation.

MATERIALS AND METHODS

Study location

The study sites were located in the northern wheatbelt of south-western Australia, approximately 350 km northeast of Perth (Lat −29.248967°, Long 116.353483° to Lat −30.066424°, Long 116.215847°, Appendix: Figure S1). A semi-arid to Mediterranean-type climate is typical for the study area.

Across the study sites, long-term mean annual rainfall averaged 340 mm and mean monthly maximum temperatures ranged from 18.0°C (July) to 37.4°C (January; Bureau of Meteorology, 2020). However, climatic conditions, in particular rainfall, can vary greatly across years (Hobbs, 1993a). Approximately 90% of the native vegetation in the study region has been cleared for mixed farming since the early 1900s, and the remaining patches are small and highly fragmented (Hobbs, 1993b). The landscape is ancient, and soils are nutrient poor, in particular phosphorus deficient, which is typical for landscapes without recent glaciation or volcanic eruptions (Lambers, 2014). Prior to clearing for agriculture, woodlands were extensive and dominated by *Eucalyptus loxophleba* (York gum) and *Acacia acuminata* (Jam), on broad flat, alluvial plains with grey-brown to red sandy loams (Anand & Paine, 2002; Lambers, 2014; McArthur, 1991).

Site selection and sampling design

We selected 10 sites, each comprising three York gum woodland states: (a) fallow cropland – restoration starting point; (b) planted old field – 10 years old; and (c) woodland reference – restoration target. Sites were selected based on: (a) the best available representation of historical reference and (b) the best match across all three with regards to soil type, topographic position and rainfall. Reference woodlands had low exotic plant abundance and little or no history of livestock grazing, clearing or fertilization, and hence were assumed to best represent soil and vegetation prior to the introduction of cropping and grazing to these landscapes. All planted old field plots had previously been cropped and/or grazed and were planted with York gum as the main overstorey species 10–13 years prior to sampling, with the aim of recovering lost aspects of biodiversity, providing fauna habitat and/or providing ecosystem services (i.e., carbon sequestration). Plots in nearby fallow cropland (measuring 20 × 50 m; one per cropland and 10 m from edge habitat) were established where cropping and/or grazing had not occurred for 1–3 years.

In each of the planted old field and woodland reference states, one plot measuring 20 × 50 m was placed 10 m from the edge habitat and marked with posts at each corner. Soil sampling and decomposition measurements were stratified by patches that were beneath dominant overstorey vegetation (York gums) or in open habitat (i.e., planting rows (trees)/inter-rows (open) for old field plots and beneath York gums/gaps for reference plots) to capture known tree-driven patchiness in soil nutrients and functions (Eldridge & Freudenberger, 2005). Interrow spacing averaged three metres and created sufficiently open areas in the planted old fields, comparable with the woodland reference sites. Detailed vegetation cover and floristic data are presented in Parkhurst, Prober, and Standish (2021).

Sampling for soil organic carbon, volumetric water content, penetration resistance and depth, as well as decomposition rates, occurred at 20 points across each plot, with 10 samples randomly collected from locations beneath York gums and a further 10 samples collected in gaps. We sampled soil bulk density (g/cm^3) at six points across each plot (three random points under trees and three in gaps; $n = 10$ plots × 3 for fallow cropland and 20 plots × 3 × 2 for planted old field and woodland reference states).

We sampled soil organic carbon at 0–10 cm depth in November 2017 and stored samples at 4°C in plastic zip-lock bags until delivery to the CSBP Limited (Bibra Lake, Western Australia) laboratories. Soil samples were dried at 80°C, ground and sieved to 2 mm. Organic carbon was measured using the Walkley and Black method, 6A1 (Walkley & Black, 1934). We measured volumetric topsoil water content at 12 cm depth using a hand-held soil moisture probe in August 2017 (HydroSense™, Campbell Scientific Australia Pty Ltd).

To minimize variation of soil moisture due to time of day, soil moisture for each plot was measured during the early morning hours. We measured the penetration resistance and depth of the soil using a penetrologger with a 1 cm^2- and a 60°-top angle cone in November 2017 (Eijkelkamp Agrisearch Equipment, Giesbeek, the Netherlands). The device was pushed vertically into the soil at a constant speed of 2 cm/s. Penetration depth was recorded in centimetres and the resistance to penetration was stored as force (Newton, MPa) for every centimetre of depth up to a depth of 80 cm.

A total of 20 untreated *Pinus radiata* woodblocks (10 controls and 10 treatments) were installed on each of the 10 plots to follow the sampling design (fallow cropland (20 woodblocks), planted old field open area/tree canopy (40 woodblocks), woodland open area/tree canopy (40 woodblocks) = 1000 woodblocks) to determine rates of woody debris decomposition by termites following the Global Wood Blocks Protocol (Cheesman et al., 2017). Control and treatment woodblocks were deployed in June 2017 and left on the ground for 18 months (Appendix: Figure S2). Occasionally, woodblocks were moved by animals, probably kangaroos (*Macropus* spp.), and had to be repositioned. In August 2017, five green and five roiboos teabags were buried at 8 cm depth and at least 15 cm apart in each plot in line with the sampling design ($n = (5+5) \times 10$ for fallow cropland, $(5+5) \times 10 \times 2$ each for planted old field and woodland reference states = 500 bags). This method was applied to determine litter decomposition by microbes following the standardized Tea Bag Index (TBI) protocol developed by Keuskamp et al. (2013). After 3 months, in November 2017, teabags were retrieved and stored in paper bags for transport to the laboratory. After drying teabags for 48 h at 90°C, the dry weight (without cord and label) was recorded to three decimal points.

Six soil bulk density samples per plot (three under trees and three in gaps) were taken using steel rings at 0–10 cm depth in September 2017 following the TERN AusPlots method (White et al., 2012). Samples were placed in zip-lock bags and sealed for transport. At the laboratory, soil bulk density samples were then weighed and transferred to paper bags for oven drying (48 h at 105°C). The dry weight was measured at three decimal points. Bulk density was calculated by dividing the mass of the oven-dried soil (g) by the volume of the ring (cm^3). The gravimetric water content of the soil was determined by subtracting the weight of the dry soil from the weight of the moist soil and then dividing by the weight of the dry soil.

Data analysis

Woodblock weight loss data were analysed using a linear mixed effects model in the R n*lme* package (Pinheiro et al., 2021). This model included *Treatment* (fallow cropland, planted old field (open/canopy), reference woodland (open/canopy)) and *Termite* (termite-exposed, termite-excluded) as independent fixed factors, and *Site* as a random factor. All other response variables were analysed using ANOVA tests in the R *stats* package, R version 3.6.1 (R Core Team, 2019). Data were log transformed where appropriate to achieve normality of residuals.

RESULTS

Depleted soil condition in agricultural soils

As predicted, our data show that most biophysical properties of agricultural soils were highly depleted when compared with intact woodland reference sites, in particular with patches in woodland reference sites under trees.

This was particularly evident for soil organic carbon, gravimetric soil water content and decomposition rates. Soil organic carbon (mean 0.9%) and gravimetric water content (mean 2.17%) on planted old fields were at similar levels to the fallow cropland and open areas in woodland reference sites but significantly lower compared with areas under tree canopy in reference woodlands (Figure 2, Table S1). Soil organic carbon content in woodland reference sites was similar to data described in other York gum woodland reference sites and therefore representative of reference woodland soils in the study region (e.g., 0.65% to 1.2% (McArthur, 1991) and 0.93% to 1.45% (Prober & Wiehl, 2012)).

Similarly, decomposition rates of wood by termites were much lower in the planted old fields compared with the woodland reference sites. Woodblocks exposed to termites in the woodlands, both in open patches (11.47%) and under tree canopy (13.61%), lost a significantly higher amount of mass compared with woodblocks in the fallow cropland (7.83%) and planted old fields (open – 7.83%, canopy – 7.73%; Figure 3, Appendix: Table S2). There was no difference across treatments for unexposed blocks (mass loss ranged from 1.14% to 2.11%; Figure 3). Results for mass loss correspond with termite attack scores across all treatments (Appendix: Figure S3, Table S3). Whilst most woodblocks were not consumed by termites, the number of consumed wood blocks and their mass loss were highest in woodland reference sites (Appendix: Figure S3, Table S3).

Decomposition rates of tea litter were highest in fallow cropland and open areas in planted old fields and significantly lower in open areas in woodland reference sites (Figure 2, Appendix: Table S1). Decomposition rates under

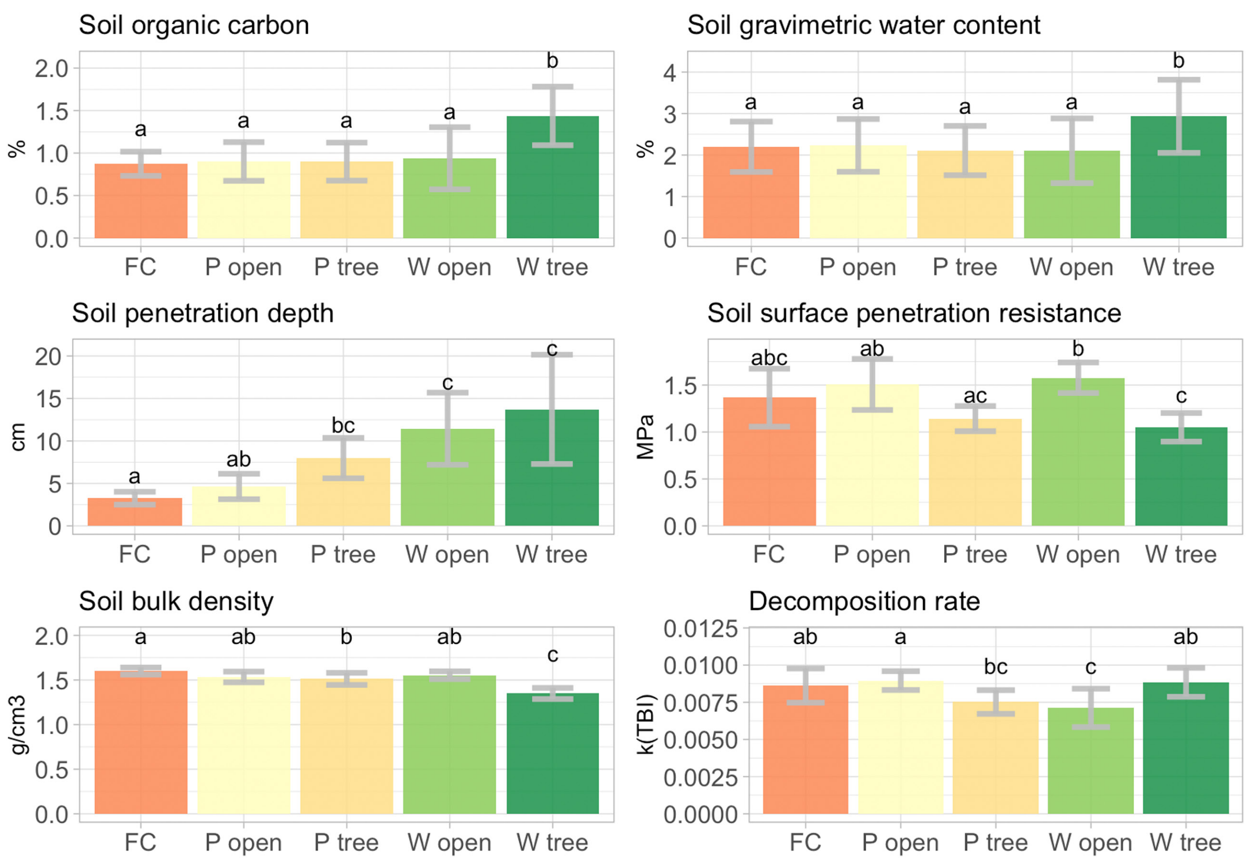

FIGURE 2 Bar plots (+/– 95% CI) showing soil organic carbon, soil gravimetric water, soil penetration depth, soil surface penetration resistance, soil bulk density and decomposition rates. Different letters indicate a significant difference ($p < 0.05$) between treatment groups (FC = fallow cropland, P open = planted old field open area, P tree = planted old field tree canopy, W open = woodland open area, W tree = woodland tree canopy).

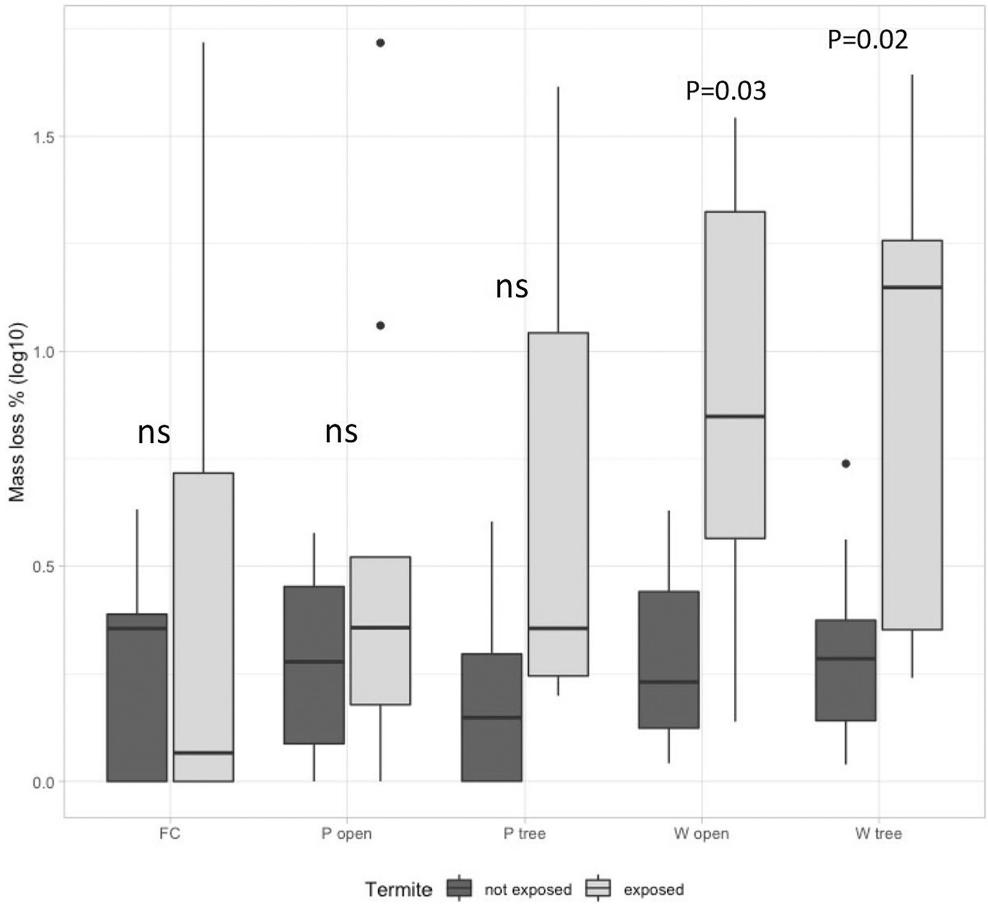

FIGURE 3 Mass loss (%) of woodblocks exposed (light grey) and not exposed (dark grey) to termites across the five restoration treatments (FC = fallow cropland, P open = planted old field open area, P tree = planted old field tree canopy, W open = woodland open area, W tree = woodland tree canopy).

the tree canopy for planted old fields and woodland reference sites did not differ compared with all other treatments (Figure 2, Appendix: Table S1). The stabilization factor across treatments did not differ (Appendix: Figure S4). Green tea mass loss averaged about 50% across all treatments, whereas Rooibos tea mass loss was much lower (14%–18%) and showed significant differences among treatments (Appendix: Figure S4).

Improved soil condition in plantings

Soil compaction data showed some evidence of improved soil conditions. Soil bulk density under trees in the planted old fields (mean 1.51 g/cm^3) was significantly lower compared with the fallow cropland (mean 1.6 g/cm^3), however, not quite as low as under tree canopy in woodland reference sites (mean 1.35 g/cm^3; Figure 2, Appendix: Table S1). There was no difference in soil bulk density in open areas in planted old fields (mean 1.53 g/cm^3), woodland reference sites (mean 1.53 g/cm^3) and fallow croplands (Figure 2, Appendix: Table S1).

Our soil penetration data showed a similar positive trend (Figures 2 and 4). Penetration depth under trees in planted old fields (mean 7.98 cm) was statistically similar to open areas and under trees in woodland reference sites (mean 11.44 and 13.72 cm, respectively) and significantly higher than the fallow cropland (mean 3.26 cm; Figure 2, Appendix: Table S1). However, this trend was not observed in the open areas in planted old fields, where penetration depth

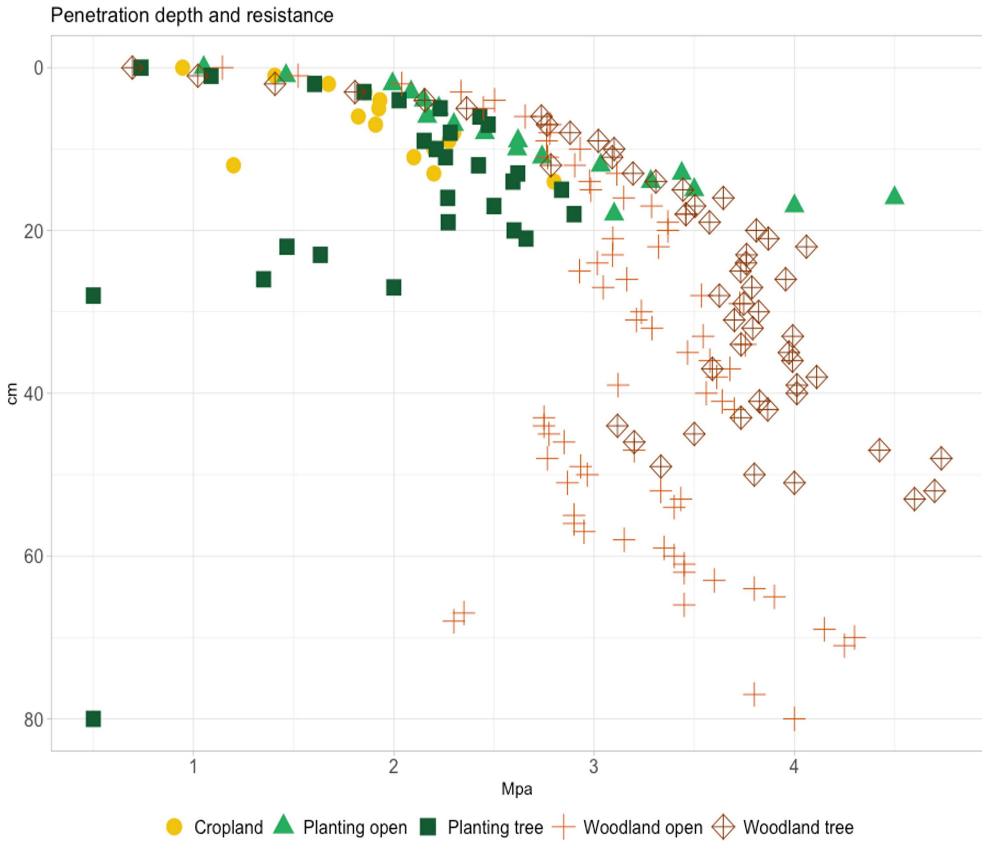

FIGURE 4 Mean penetration resistance (MPa) per cm depth across all treatments (yellow circles = fallow cropland, green triangles = planted old field open area, dark green squares = planted old field tree canopy, orange plus = woodland open area, brown diamond with cross = woodland tree canopy).

(mean 4.64 cm) was similar to the fallow cropland and significantly lower than the reference woodland sites (Figure 2, Appendix: Table S1).

We detected similar patterns in penetration resistance at the soil surface (0–3 cm) between the open and treed areas in the planted old field and woodland reference sites, with higher penetration resistance in open areas compared with treed areas; however, the difference was only statistically different between the open and treed areas in the woodland reference sites. All planted and woodland treatments were similar to the fallow cropland, reflecting high variability of soil surface hardness in the fallow cropland sites (Figure 2, Table S1).

DISCUSSION

As hypothesized (H1), most biophysical properties of soils in fallow croplands were degraded compared with reference sites. Specifically, these soils had lower soil organic carbon, soil moisture content and wood decomposition rates, as well as higher bulk density and lower soil penetration depth, particularly in comparison with soils beneath woodland trees. This pattern was also evident in the planted old fields. Therefore, the results only partially supported our second hypothesis, with tree planting and cessation of cropping leading to a reduction in soil surface compaction and higher soil penetration depth. Our data did not support our predictions for higher soil carbon concentration, soil moisture content and decomposition on old fields after tree planting and cessation of cropping.

Some of these changes were most prominent under trees in planted old fields, partially supporting the third hypothesis.

Lack of recovery of soil organic carbon, water content and decomposition

Whilst we observed lower soil compaction on planted old fields than in fallow cropland, soil organic carbon and water content, and decomposition of litter and woody debris did not differ. This contrasts with higher soil organic carbon, soil moisture and wood decomposition in woodland reference sites. Soil carbon is readily depleted by agricultural activities that reduce above- and below-ground plant biomass input and higher decomposition rates through disturbance activities and erosion (McLauchlan, 2006). To increase soil carbon, inputs of plant organic material need to be in large quantities to counteract the effects of decomposition and eventual respiration or leaching (Berthelin et al., 2022). In native reference ecosystems, plant litter, including root turnover, continuously replenishes below- and above-ground biomass input, therefore providing organic material for decomposition processes (Sayer, 2006). In particular, areas under trees have been found to have higher soil carbon content (consistent with our study) due to litter input from trees (Prober et al., 2014a). However, in a restoration context, soil organic matter may take several decades to replenish (Harper et al., 2012; Zethof et al., 2019). This might be due to initial time lags in litter accumulation (Parkhurst, Prober, & Standish, 2021) and subsequent slow soil organic carbon accumulation (Standish et al., 2022). Our study measured soil organic carbon content after 10 years; therefore, litter accumulation and root turnover may not yet have been sufficient to result in higher soil organic carbon, even in patches directly under trees.

Low soil organic carbon may also account for low soil moisture on the planted old fields, in contrast to the woodland reference sites. Soil organic matter inputs contribute to soil aggregation, structure and water retention, facilitating higher moisture content and the soil's ability to retain moisture (Ehrenfeld et al., 2005). In arid ecosystems, higher soil moisture is particularly evident under trees and shrubs compared with gaps, where soil organic carbon is also higher than in the open areas (Eldridge & Freudenberger, 2005; Tongway & Ludwig, 1990). Whilst this pattern matches the soil water content in the intact reference woodland in our study, higher soil moisture is not yet evident under trees in the planted old fields compared with fallow cropland.

Soil organic carbon and soil moisture availability, as well as temperature and litter quality, are also key factors driving the decomposition of litter and woody debris by microbiota and decomposer fauna (Bradford et al., 2017; Cheesman et al., 2017; Ehrenfeld et al., 2005; Wall et al., 2008). Soil microbes greatly influence soil carbon sequestration through respiration or deposition of microbial litter (Albornoz et al., 2022). In arid ecosystems, litter decomposition processes are generally slow (Keuskamp et al., 2013; Ochoa-Hueso et al., 2020), however, an increased metabolic response rate and abundance and diversity of microbiota (i.e., fungi and bacteria) are often present under plant canopies, possibly due to favourable conditions such as higher nutrient and moisture content (Ochoa-Hueso et al., 2018, 2020; Prober et al., 2014a). Differences in the abundance of microbial communities and their interactions (Albornoz et al., 2022) may contribute to a higher litter decomposition rate under the tree canopy in the woodland reference sites compared with the open, bare patches in our study. However, this pattern was not detectable in the planted old fields, possibly due to low soil carbon and moisture under canopies of planted trees and/or lower reduced microbial activity. In addition, photodegradation and photofacilitation can contribute to decomposition in arid ecosystems (Austin & Vivanco, 2006; Muñoz-Rojas et al., 2016), and may have contributed to decomposition in our study; however, we did not measure these processes.

Similarly, we detected higher decomposition rates of woody debris by termites in the woodland reference sites, but no difference was evident between open and tree canopy patches. Moisture, as well as temperature dependency of termite activity have been reported elsewhere (e.g., Cheesman et al., 2017; Zanne et al., 2022); nevertheless, higher moisture content under tree canopies in the woodland reference sites was not matched with higher decomposition rates of woody debris by termites in our study. Potentially, the difference in moisture availability between the open areas and tree canopy patches did not impact termite activity. The lack of decomposition activity by termites in the planted old fields is consistent with the lack of recovery of other soil functions we measured, indicating that favourable conditions for these saprophytic biota have not yet been established at the restored sites (Sandström et al., 2019; Seibold et al., 2015). Indeed, other authors have observed low termite activity and diversity on abandoned farmland, even decades after abandonment (Abensperg-Traun, 1998; Abensperg-Traun & De Boer, 1990; de Bruyn & Conacher, 1990). Termites contribute to soil condition by improving water movement and organic carbon (Abensperg-Traun & De Boer, 1990), so their absence is likely contributing to our findings for these soil properties.

Partial recovery of soil compaction

The lower soil compaction, in particular in areas under trees, is likely attributable to higher organic litter input and the creation of pores by the fine and coarse roots of the planted trees, resulting in higher aeration and porosity (Ehrenfeld et al., 2005). Whilst soil compaction and penetration depth were favourable under trees in planted old fields, conditions were still less favourable than under trees in reference woodlands and may require more time to fully recover. Prolonged agricultural legacies (60+ years) of soil compaction, particularly in the sub-soil, have been evident elsewhere, limiting root growth and biomass accumulation (Piché & Kelting, 2015; Standish et al., 2006; Wen-Jie et al., 2011). Whilst the topsoil layer may recover more quickly due to its spatial proximity to plant litter input, deeper soil layers require much longer timeframes for plant-soil feedback mechanisms to take effect (Parkhurst, Prober, Hobbs, et al., 2021). Reduction of bulk density after restoration has been observed elsewhere, but soil conditions did not meet reference levels, even after 50+ years post-restoration action (Parkhurst, Prober, Hobbs, et al., 2021).

CONCLUSION

We have shown that the cessation of cropping, followed by planting native woody vegetaton on old fields in a semi-arid agricultural landscape has modest positive, albeit limited effects on favourable soil physical, biogeochemical and biotic conditions within a decade. Promisingly, soil compaction was lower on the planted old fields than crop fallows, in particular under tree canopies, indicating early signs of recovery following tree planting. However, improvements in other soil functions such as soil organic carbon, soil water content and decomposition of litter and woody debris were not detectable. The findings for organic carbon are consistent with other studies of tree planting efforts (Standish et al., 2022), which together may call into question the focus on increasing soil carbon for markets rather than potential biodiversity co-benefits (Macdonald et al., 2019; Moinet et al., 2023).

We conclude that the recovery of key soil functions after tree planting will take longer than a decade in semi-arid ecosystems due to climate constraints and time lags of above- and below-ground organic matter accumulation in young plantings. Whilst ecological time lags after restoration have been explored for biota and key features of their habitat (e.g., trees Vesk et al., 2008; Watts et al., 2020), there are few data on the recovery of soil physical properties. Further research and long-term monitoring are required to assess changes over extended time periods (e.g., several decades) and to identify restoration interventions that may hasten recovery.

AUTHOR CONTRIBUTIONS

Tina Parkhurst: Conceptualization (lead); data curation (lead); formal analysis (lead); funding acquisition (equal); investigation (lead); methodology (equal); project administration (lead); resources (equal); validation (lead); writing – original draft (lead); writing – review and editing (equal). **Suzanne M. Prober:** Conceptualization (equal); data curation (equal); formal analysis (supporting); funding acquisition (supporting); investigation (equal); methodology (equal); project administration (supporting); resources (supporting); supervision (lead); validation (supporting); visualization (supporting); writing – original draft (supporting); writing – review and editing (equal). **Rachel J. Standish:** Conceptualization (equal); data curation (supporting); formal analysis (supporting); funding acquisition (equal); investigation (equal); methodology (equal); project administration (supporting); supervision (lead); validation (equal); visualization (equal); writing – original draft (supporting); writing – review and editing (equal).

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data will be made available via TERN's data repository.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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