Early response of soil properties and function to riparian rainforest restoration

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Abstract

Reforestation of riparian zones is increasingly practiced in many regions to achieve multiple objectives including biodiversity conservation, bank stabilisation, and improvement in water quality. Despite this, actual benefits of reforestation for recovering underlying soil properties and function remain poorly understood. Here we compare remnant riparian rainforest, pasture and reforestation plantings aged 2-20 years in an Australian subtropical catchment on ferrosols to determine the extent to which reforestation restores these soil properties. Of the nine soil attributes measured (total nitrogen, nitrate and ammonium concentrations, net nitrification and ammonification rates, organic carbon, bulk density, fine root biomass and water infiltration rates), only infiltration rates were significantly lower in pasture than remnant riparian rainforest. Within reforestation plantings, bulk density decreased up to 1.4-fold and infiltration rates increased up to 60-fold with time post-reforestation. Our results suggest that the main outcome of belowground processes of early reforestation is the recovery of the soils' physical structure, with potential beneficial ecosystem services to include reduced runoff, erosion and associated sediment and nutrient loads in waterways. We also demonstrate differential impacts of two commonly planted tree species on a subset of soil properties suggesting that preferential planting of select species could accelerate progress on specific restoration objectives.

Keywords

Below-ground processes, forest restoration, ecosystem services, infiltration rates, soil function, subtropical rainforest

1. Introduction

Globally, riparian zones are widely recognized for their critical role in water regulation (Meynendonckx et al., 2006, Orzetti et al., 2010, Connor et al., 2012, Russo et al., 2012, Vidon, 2012), and conservation of biodiversity (Charron et al., 2008, Griscom et al., 2009, Scott et al., 2010, Rodrigues et al., 2011). Clearing of vegetation from stream banks can cause increased soil erosion and decreased water quality through the loss of filtration services, in addition to the loss of critical species habitat (Naiman et al., 2005). In direct response, degraded stream-banks are increasingly being restored to reinstate ecological functions (Sweeney et al., 2004). Riparian zones are also regarded as foci of restoration opportunity as they are often preferentially renounced from agricultural production over agronomically more valuable areas (Rodrigues et al., 2011).

Soil physical, chemical and functional properties change with succession from pasture to secondary forest including reforestation plantings in the tropics, subtropics and temperate regions (Guo and Gifford, 2002, Peñuela and Drew, 2004, Maloney et al., 2008, Paul et al., 2010b, Zhu et al., 2010). Soil properties such as bulk density, soil water content, fine root biomass, porosity, stocks of nitrogen, carbon and phosphorus, nitrogen cycling, availability of macro- and micronutrients, as well as toxins (metals, salts), have been studied in context of land use change (Schoenholtz et al., 2000, Murty et al., 2002, Bronick and Lal, 2005). However, most studies focus on mid- to long-term effects of reforestation with less consideration of the short-term effects (<10-20 years) of reforestation plantings, and associated reinstatement of soil functions that might motivate landholders to become actively engaged in restoration. Another knowledge gap is the relative contribution of tree species to outcomes of riparian restoration plantings, although such information could inform decisions on species selection and reforestation goals.

In Australia, riparian reforestation is a major focus of habitat restoration in the tropics and subtropics; over 70% of rainforest restoration projects have targeted banks of rivers and streams (Catterall and Harrison, 2006). However, much evaluation of the success of reforestation plantings has focused on aboveground attributes (Kanowski et al., 2003, Catterall et al., 2004, Kanowski and

Catterall, 2010), with less regard to belowground processes including soil function (but see Paul et al. (2010b) for the recovery of soils via different rainforest restoration pathways). Understanding below ground processes is important to quantify the benefits of reforestation to recovering belowground processes that are directly relevant to ecosystem functions.

Soil properties and functions might be expected to change through time with reforestation to a state that more closely resembles the intact rainforest ecosystems. Studying riparian soil under remnant rainforest, pasture, and within 20 years post reforestation, we addressed three questions (i) how do soil attributes differ between pasture and remnant riparian rainforest, (ii) how do soils change with time following reforestation, and (iii) do tree species affect soils differentially following reforestation?

The study was motivated by the need for restoration of forest cover in water catchments due to the improvements in water quality that can be achieved (Wickham et al., 2011). The region targeted for our study is located in the Australian subtropics and has experienced large-scale clearing of rainforest since the mid 19^{th} century. The region is an important water catchment that supplies drinking water to more than 100000 people in the region (ABS, 2011), and numerous land owners and stakeholder holders are committed to ongoing restoration of riparian zones. Our research aimed to generate knowledge of soil properties to direct future reforestation activities.

2. Materials and methods

2.1. Study site

This study focused on riparian areas in the Maleny region in southeastern Queensland, Australia (26˚45'S, 152˚48'E). The climate is classified as subtropical with an annual rainfall of 1900 mm and annual daily mean temperature maximum and minimum of 23.2 ˚C and 14.2 ˚C, respectively. The soils in the region are classified as permeable red clay-loam ferrosols (Ellison and Coaldrake, 1954, Willmott, 2007) or Nitisol in the US soil taxonomy. Prior to European settlement in the mid-1800s and subsequent land clearing for timber production and agriculture (especially dairy production), the region was dominated by cool-subtropical rainforest classified as complex notophyll vine forest (Smith, 2001, Nakamura et al., 2007).

2.1. Sampling design

Field measurements and soil sampling were carried out between May and June 2013. The sampling design was based around riparian reforestation projects established in the region by the Lake Baroon Catchment Care Group (LBCCG). Reforestation sites were grouped into five spatial clusters across the Maleny Plateau (elevational range 196 to 441 m a.s.l.) to capture broad variation in land form and land use history. A subset of one to three reforestation sites ranging in age between 2 and 20 years were then selected from each cluster for a total of 10 sites. Sites were also established in remnant riparian rainforest where available (2 of 5 clusters) and samples were collected from pasture (adjacent to plantings or remnant forest) at all sites in each cluster.

At each reforestation or remnant site, five subplots were established. Each subplot was deliberately located under a different tree species. The five tree species targeted here were derived from a pool of species commonly used in local rainforest plantings, which in turn were selected to include early to late successional species, nitrogen fixing and/or waterlogging tolerant species. The tree species chosen to investigate species-specific effects on soil properties included *Acacia melanoxylon* R.Br. (Fabaceae, N_2 fixer, early succession species), *Ficus coronata* Spin (Moraceae, nitrophile, waterlogging tolerant, early succession species), *Flindersia schottiana* F. Muell. (Rutaceae, mid-late succession), *Homalanthus nutans* (G.Forst.) Guill. (Euphorbiaceae, early succession species) and *Podocarpus elatus* R.Br. ex Endl (Podocarpaceae, mid-late succession, gymnosperm). In the event that one of the chosen tree species was absent from a site, a closely related species was substituted or, if not available, subplots were located under a random tree within the site. Subplots were positioned under the canopy of each selected tree (mean distance from stem 89.5 cm, SD 45.2 cm).

At each reforestation or remnant forest site, we performed the following procedures: (1) collected a single 8 cm diameter core to a depth of 30 cm (hereafter "textural core") and divided it into 0-15 cm and 15-30 cm fractions for textural analysis; (2) performed an infiltration test; and, (3) collected two samples of the surface soil using a 12.5 cm diameter, 8.5 cm tall, 1043 cm^3 volume bulk density ring (hereafter "bulk density ring") from each of five subplots for analysis of physical and chemical properties of the soil. In addition, a single texture core and two to four bulk density ring subplots were collected from adjacent pasture and infiltration measurements performed. For the purposes of analysis, samples and infiltration measures from pasture were calculated as means for each cluster.

Overall, we sampled 10 reforestation sites, 2 remnant riparian rainforest sites and 5 pasture clusters comprising 24 cores for soil texture analysis, 150 cores for soil chemical and physical analyses and 24 infiltration tests.

2.3. Texture analysis

Changes in soil texture mostly occur due to weathering over very long time periods and as demonstrated by Maloney et al. (2008). Unless there is erosion or despostion of soil material, texture is unlikely to vary over time since reforestation. For this reason, the purpose of conducting a soil texture analysis at each site was not to determine changes due to reforestation, but to characterize and evaluate background variation in soil types among sites that might warrant differential treatments of sites in the analysis. Soil samples were not dispersed prior to analysis and as such, the calculated percentages represent the relative size of secondary particles (aggregates) present rather than primary particle size *sensu stricto*. This was considered a more relevant measure of soil function than particle size determined using standard methods, as it gives a relative measure of soil aggregation and structure. Furthermore, the ferrosols in this study are recognized as being subplastic, with clay-sized particles being primarily composed of sesquioxides that do not readily break down into primary particles (McIntyre, 1976). These analyses of soil aggregation were conducted on air-dried soil from 0-15 and 15-30 cm depth cores according to the hydrometer method for soil texture analysis (Robertson et al., 1999, DeForest, 2011). For each sample, 40 g of air-dried soil was ground and mixed with 100 ml 50 g l^{-1} Sodium hexametaphosphate solution and filled to 500 ml with deionized water. Samples were shaken overnight and then emptied into a 1 l cylinder for analysis. Cylinders were filled to 1 l with deionized water, mixed with a plunger and left to settle. Temperature and specific gravity readings were taken 40 seconds and 7 hours after plunging (Robertson et al., 1999). Using temperature and specific gravity readings, the fraction of sand, silt and clay sized particles in each soil sample was then calculated as a percentage.

2.4. Soil physical properties: bulk density and infiltration rates

Soil bulk density was determined from the second bulk density ring in each subplot. Fresh soil was oven dried to a constant weight at 105 ˚C. Soil bulk density is expressed as grams of dry soil per volume. To quantify fine root biomass, soil from one bulk density ring per subplot was passed through a sieve (8 mm mesh) and roots were manually removed. Roots were then oven dried at 65 ˚C to a constant weight, and results expressed as grams of fine roots per kilogram of dry soil.

Infiltration rates were measured using a ponded disk permeameter (Perroux and White, 1988). Soils were assumed to be at field capacity as the region had received in excess of 30 mm in the week preceding the first set of measurements and over 100 mm in the week prior to the second sampling visit. Infiltration tests were performed at a random, relatively flat area representative of the site. Tests in the adjacent pasture were performed at a random point beyond the farthest extent of the neighbouring forest canopy. Upon setting up of the permeameter, grass and other ground covers were clipped to just above the soil surface and the soil pre-moistened with rainwater to reduce the time needed to reach steady state infiltration. The permeameter was filled with rainwater collected from water tanks on local landholder properties to ensure that water quality was similar to that occurring under natural condition. Once the disk permeameter was in place, measurements of water level changes were recorded until the time interval between measurements was stable. As soils were close to field capacity at the start of measurements, steady stage was reached quickly. Infiltration rate was calculated as the average volume change per second and then converted into millimeters per hour.

2.5. Soil chemical properties: nitrogen and organic carbon

Total concentrations of soil nitrogen (TN) and organic carbon (SOC) were determined using a TruSpec CHN analyzer (LECO Australia Pty. Ltd., Castle Hill, NSW 2154, Australia), using dried, ground and sieved soil (2 mm sieve). Nitrate (NO_3^-) and ammonium (NH_4^+) were extracted with potassium chloride (KCl) and quantified via colourimetric assays. To remove NO_3^- and NH_4^+ from soil exchange sites, 15 g of fresh sieved soil was mixed with 1 M KCl in a 15 ml falcon tube and shaken on an orbital shaker for 1 hour. Samples were centrifuged at 4000 rpm for 4 minutes and 1 ml of the supernatant was frozen until analysis via colourimetric assays according to standard procedures (Kandeler and Gerber, 1988, Miranda et al., 2001). Net nitrification and ammonification rates were calculated from the difference in concentration between fresh soil samples and those incubated for 20 days at 28˚C. Briefly, 15 g of soil were weighed into 50 ml falcon tubes and incubated for 20 days. Soils were kept at field moisture by watering daily with deionized water. Seeds that germinated during the incubation period were removed. Concentrations of NO_3^- and NH_4^+ in the incubated soils following KCl extraction were quantified using colourimetric assays as described above. Net nitrification and ammonification rates are expressed as mg nitrate or ammonium generated per kg dry soil per day.

2.6. Statistical analysis

Statistical analyses were performed in R version 3.0.1 (R Core Team, 2013). To avoid non-independence among subsamples, we derived the aggregated mean of soil attributes for each site (reforestation or remnant rainforest) or cluster (pasture) and used these values as inputs in all analyses, excepting those determining species-specific effects which used values from a single subplot within each site. Pairwise correlation tests (Spearman's) were performed to examine associations between tested soil attributes. Soil attributes under both pasture and remnant riparian rainforest were directly compared using a one-way analysis of variance (ANOVA). Developmental trajectories of soil attributes within reforested sites (site level and separately for specific tree species within sites) were examined using linear regressions between either age or log-transformed age and individual soil attributes. The age term was log-transformed to account for potential non-linear developmental trajectory of reforestation plantings; often showing a rapid change immediately post planting followed by diminishing returns as the system reaches natural equilibrium. For the purposes of quantifying response trajectories with age, we assume that pasture represents 0 years post reforestation.

3. Results

Of the 56 subplots within sites, 47 (84%) were associated with one of the five target trees. *Homalanthus nutans* was the species most commonly absent, being present only in 50% of reforested sites. Soil aggregation showed only minor variation among sites and land uses so was not considered further in subsequent analyses (sand sized aggregates: mean=60%, range= 46-76%, clay sized aggregates: mean= 30%, range= 14-46%).

3.1. Correlations between variables

Infiltration rates were inversely correlated with bulk density (P=0.005) but were not correlated with soil particle size (percentage sand, silt and clay) at either 0-15 cm or 15-30 cm depth (P>0.5, not shown). Bulk density was correlated with both SOC and TN (P=0.02 and 0.01 respectively) (table 1). Pairwise correlations revealed strong associations between soil nitrogen-related variables (table 1).

Table 1: Pairwise correlations (Spearman's) between soil attributes across pasture, remnant rainforest and reforestation sites (n = 5, 2, 10 respectively). Only significant pairwise correlations (P< 0.05) are shown.

Variable	Variable	Spearman's correlation coefficient	P value
Bulk density	SOC	-0.554	0.023
Bulk density	TN	-0.615	0.010
Infiltration	Bulk density	-0.654	0.005
TN	SOC	0.784	<0.001
Ammonification rate	Nitrification rate	-0.664	0.005
Ammonification rate	Fine root biomass	0.662	0.005
Nitrate	Ammonification rate	-0.529	0.031
Nitrate	TN	0.495	0.045
Nitrate	SOC	0.630	0.008

3.2. Differences between pasture and remnant rainforest

There was no apparent difference between pasture and remnant riparian rainforest in the tested soil attributes except infiltration rates (table 2). Infiltration rates were significantly greater in the remnant rainforest (1421 ± 995 mm h^{-1}) than in pasture sites (220 ± 151 mm h^{-1}) ($F= 9.526$, $DF= 5$, $P= 0.027$, table 2).

Table 2: ANOVA results from a comparison of soil attributes between remnant riparian rainforest (n=2) and pasture (n=5). Bold text indicates significance at 0.05 level, NS not significant if P>0.05, SD standard deviation.

Variable	ANOVA P value	Mean pasture ± 1 SD	Mean remnant ± 1 SD	Contrast remnant (R) vs. pasture (P)
Fine root biomass (g kg^{-1})	0.09	6.64 ± 4.05	13.00 ± 0.38	NS
Bulk density (g cm^{-3})	0.36	0.69 ± 0.08	0.61 ±0.12	NS
SOC (%)	0.60	6.15 ± 1.83	7.31 ±4.15	NS
TN (%)	0.85	0.71 ± 0.14	0.74 ±0.26	NS
Nitrate (mg kg^{-1})	0.26	11.8 ±7.24	22.7 ± 17.9	NS
Nitrification rate (mg kg^{-1} soil d^{-1})	0.65	0.67 ± 0.75	0.39 ± 0.03	NS
Ammonium (mg kg^{-1})	0.46	4.36 ± 4.27	7.85 ± 8.10	NS
Ammonification rate (mg kg^{-1} soil d^{-1})	0.73	0.65 ± 0.98	0.65 ± 0.98	NS
Infiltration rate (mm h^{-1})	**0.03**	**220 ± 151**	**1421 ± 995**	**R>P**

3.3. Recovery of soil properties under reforestation plantings

Only bulk density and infiltration rates showed a relationship with age within 20 years post reforestation (table 3). In each case, models with a log-transformed age term explained more of the variation than models with an un-transformed age term (table 3). Bulk density of the soils ranged from 0.312 to 0.991 g cm^{-3}, and decreased significantly with time post reforestation (F= 6.312, DF= 13, P=0.026, figure 1). Conversely, infiltration rates significantly increased (F= 10.56, DF= 13, P= 0.006) within 20 years post reforestation (figure 1). Infiltration rates in reforestation sites ranged from 149 mm h^{-1} at a 6.5-year-old reforestation site to over 1800 mm h^{-1} at an 11-year-old reforestation site. Infiltration rates of pasture sites varied from 3 mm h^{-1} to >650 mm h^{-1}, although this variation was slightly less when data was aggregated by cluster.

Table 3: Relationship between tested soil attributes and the age of the reforestation sites (n=15). Pasture is assumed to be 0 years post reforestation. Results from linear models with both age and log-transformed age are shown. Bold P values are significant at the 0.05 level.

Variable	Age				Log Age			
	Slope	R^2 value	F statistic	P value	Slope	R^2 value	F statistic	P value
Fine root biomass (g kg^{-1} soil)	0.02	0.001	0.02	0.90	1.14	0.03	0.34	0.57
Bulk density (g cm^{-3})	-0.009	0.28	5.16	**0.04**	-0.12	0.32	6.31	**0.03**
SOC (%)	-0.03	0.01	0.15	0.70	-0.03	<0.001	0.002	0.97
TN (%)	-0.001	0.004	0.05	0.83	0.005	<0.001	0.004	0.95
Nitrate (mg kg^{-1})	0.09	0.002	0.03	0.86	3.58	0.03	0.38	0.55
Nitrification rate (mg kg^{-1} d^{-1})	0.03	0.13	1.93	0.19	0.30	0.07	0.95	0.35
Ammonium (mg kg^{-1})	-0.09	0.04	0.51	0.49	-1.72	0.11	1.55	0.24
Ammonificatio n rate (mg kg^{-1} d^{-1})	-0.03	0.11	1.54	0.24	-0.47	0.15	2.25	0.16
Infiltration rate (mm h^{-1})	54.9	0.39	9.77	**0.008**	678	0.45	10.6	**0.006**

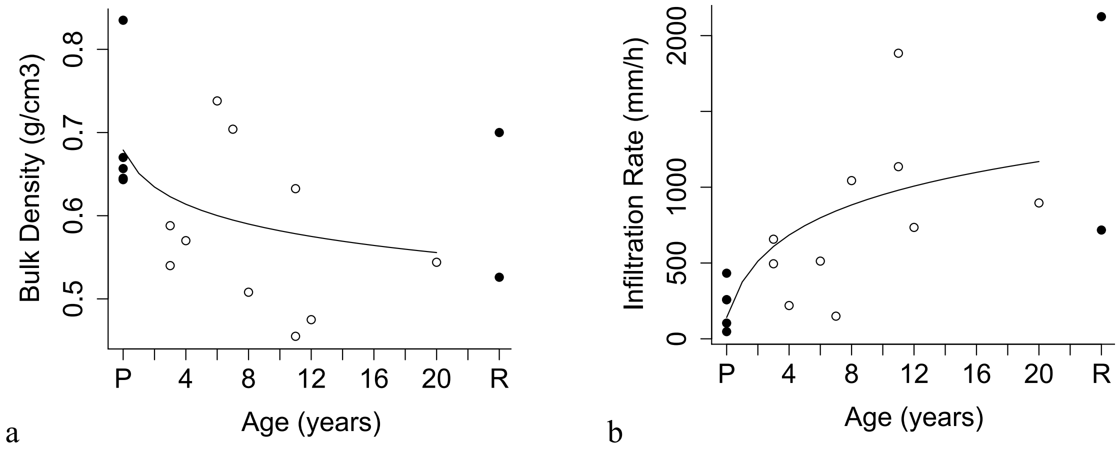

Figure 1: Differences in bulk density (a) and infiltration rates (b) among pasture (P), reforestation sites aged 2 to 20 years, and remnant rainforest (R) (n= 5, 10 and 2 respectively). Lines show fitted values of the modeled relationship between soil properties and log-transformed age of reforestation. Open circles show reforestation plantings, filled circles indicate pasture clusters and remnant riparian rainforest.

3.4. Species specific effects on the recovery of soils under reforestation plantings

Species-specific effects were apparent for two of five tree species when tested soil attributes were modeled against log-transformed age (figure 2). Ammonium concentrations in soil associated with *Flindersia schottiana* decreased with time post reforestation (F= 4.97, DF= 8, P= 0.056). Fine root biomass in soil associated with *Ficus coronata* decreased through time within 20 years post reforestation (F= 4.921, DF= 8, P= 0.057).

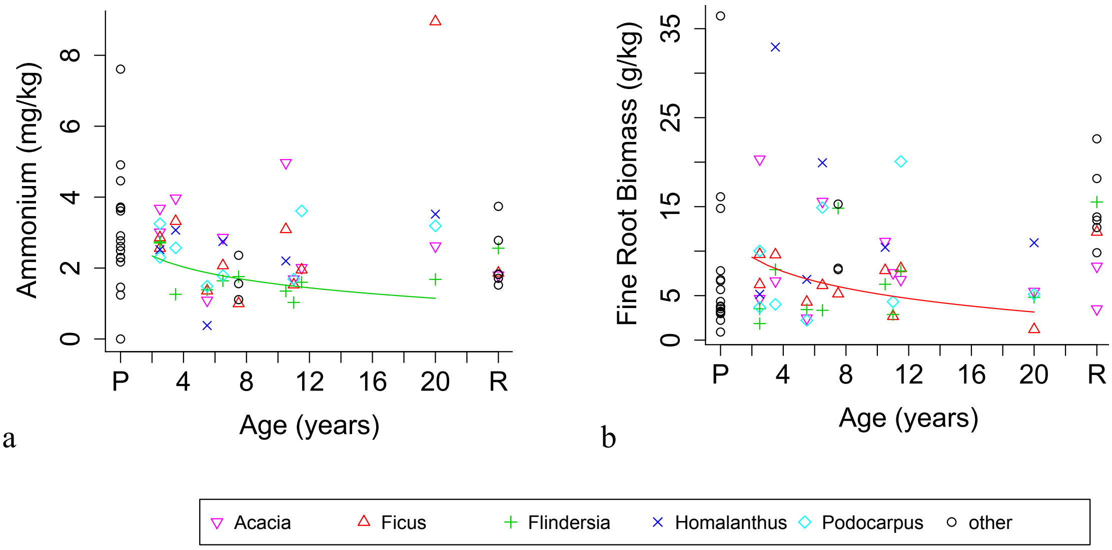

Figure 2: Differences in ammonium and fine root biomass over time under five tree species in pasture (P), reforestation sites aged 2 to 20 years, and remnant rainforest (R). Genus names refer to the selected tree species under which the soil samples were taken. “Other” refers to soil samples taken from non-specified tree species or pastoral sites. Lines show fitted values of the modeled relationship between individual soil properties and log-transformed age for species with significant models - *Flindersia schottiana* (a) and *Ficus coronata* (b).

4. Discussion

Replanting riparian areas is driven by multiple objectives including increasing landscape connectivity for wildlife movements and biogeochemical functions such as trapping nutrients, sediment, pesticides, bank stabilisation, improved water quality and flood mitigation (Naiman et al., 2005). Here, we studied the trajectory of riparian reforestation to identify key changes in soil attributes through the transition from pasture to riparian tree plantings of increasing age and remnant rainforest. We showed that the major belowground outcomes within 3 to 20 years since planting are measures of the physical and associated functional attributes of soil rather than chemical properties.

4.1. Differences between pasture and remnant rainforest

Infiltration rate was the only soil variable studied here that differed significantly between pasture and remnant riparian rainforest. Pasture infiltration rates were ≈15% of those of riparian rainforest which is consistent with a previous study showing that the infiltration rate of pasture was 20% of that of remnant rainforest (Peñuela and Drew, 2004). We detected no statistically significant differences in other soil properties between pasture and remnant rainforest soils. Limited effects of land use on selected chemical properties is consistent with previous findings in Australian on ferrosol soils (Paul et al., 2010b). However, it was expected that attributes such as bulk density and fine roots would differ between land use types (Paul et al., 2010b). A possible explanation for this is large variability in relation to the range of measures for soil properties observed. Limited availability of reference remnant forest within sampling clusters, a consequence of extensive clearing of riparian rainforest in the region was also a contributing factor. Nevertheless, the data may also indicate that there is little degradation of soil with respect to N and C stock in the studied systems.

4.2. Recovery of soils under reforestation plantings

Bulk density and infiltration rates both changed significantly with time under the reforestation plantings. Bulk density showed a negative trend with age whereas infiltration increased with time in the reforestation plantings. The observed trend for decreasing bulk density is consistent with previous studies globally (Reiners et al., 1994, Paul et al., 2010b, Rodrigues Nogueira Jr et al., 2011, Neris et al., 2012). The higher bulk density and lower infiltration rates of the pastures may be partially attributed to trampling by livestock and/or machinery (Bell et al., 2011) Lower bulk densities are often associated with soil of forests and reforestation plantings because greater presence of roots and soil biota and higher concentrations of organic matter enhance soil porosity. In remnant forest and reforestation plantings, the high turnover of roots results in more open soil pores as roots grow and subsequently die (Beven and Germann, 1982). Additionally, pastures generally exhibit more shallow and non-woody roots than the larger range of fine to large woody roots characteristic of tree species, with large, woody roots leaving larger conduits in the soil post death (Jiménez et al., 2009, Finér et al., 2011, Powers and Perez-Aviles, 2012).

Higher densities and diversities of soil organisms in forests and reforestation plantings (Nakamura et al., 2003, Colloff et al., 2010) contribute to the aeration of soil and reduce bulk density. High soil organic matter is also associated with low bulk density (Murty et al., 2002), a trend seen in this study as bulk density was negatively correlated with SOC. This was expected as SOC is indicative of litter dynamics, turnover and decomposition rates and is responsible for providing low-density mass to the soil and improving soil structure (Murty et al., 2002, Peñuela and Drew, 2004, Neris et al., 2012, Laub et al., 2013). The bulk density values of the soils in our study (0.65 g cm^{-3}) are low in comparison to average soils (>1.2 g cm^{-3}). However, Paul et al. (2010b) reported low bulk densities (0.6-0.8 g cm^{-3}) for similar ferrosols in reforestation plantings and remnant rainforest, confirmation the low bulk density of this soil type. We attribute these low bulk density values to the friable nature of the ferrosols in the study region, high porosity and the high organic matter content in the topsoil. We also propose that the low bulk density contributes to the trend of increasing infiltration rates with time following reforestation as the two were inversely correlated.

Similar to prior studies, we found a significant increase in infiltration rates with reforestation age, with infiltration rates of the pastures being always lower than those of adjacent reforestation sites. We note a strong positive pattern with age within the first 5-10 years post reforestation, indicating that even young tree plantings are able to promote increased infiltration rates (Wood, 1977, Neris et al., 2012). The trajectory in infiltration rates from the plantings (up to 20 years old) indicate that increases of 182 mm h^{-1} can be expected for each year in the first decade since planting, with most gains seen within the first 10 years post-reforestation. This trend is consistent with Bharati et al. (2002) who found short term changes in infiltration rates under riparian buffers when compared to adjacent cultivated fields and pastures; noting a 5-fold increase in infiltration rates under riparian buffers after 6 growing seasons. Our finding that infiltration rates are strongly enhanced soon after reforestation has meaningful consequences for decision making on land use, water quality and conservation with reduced run-off, slowed surface erosion and reduced non-point source pollution as beneficial outcomes (Schultz et al., 2004).

Here, only two of the tested nine soil attributes varied between pasture and reforestation sites. One possible explanation is the recentness of the plantings. Reforestation sites were relatively recent compared to studies that focused on plantings of much greater age (up to 60 years) (Reiners et al., 1994, Maloney et al., 2008, Powers and Perez-Aviles, 2012) and this may have contributed to the uniformity between sites in our study. Age, however does not explain the common lack of differences between the pasture and remnant riparian rainforest sites, thus it is also possible that inherent high soil fertility may be a larger contributor to soil properties than land cover as has been suggested previously (Lamb, 1980, Powers and Perez-Aviles, 2012)

4.3. Species specific effects on the recovery of soils under reforestation plantings

In previous studies, species-specific effects were not taken into account when elucidating changes in soil properties post-reforestation (Paul et al., 2010a, b), although it is known that species-specific effects can modify soil properties (Bezemer et al., 2006). We detected two species-specific effects in the five tree species and eight variables examined. *Ficus coronata* and *Flindersia schottiana* showed negative relationships with time since reforestation for fine root biomass and soil ammonium levels respectively. We hypothesized that as reforested sites become more established, fine root biomass increases to access nutrients in the surface soil layer of rainforests (Claus and George, 2005, Finér et al., 2011, Powers and Perez-Aviles, 2012). The studied *F. coronata* specimens displayed the opposite trend to that expected, showing a negative fine root relationship with time post-reforestation. This decreasing trend exhibited by *F. coronata* may be explained by a need for greater structural support through large woody roots in which they invest carbon at the expense of fine surface roots, or the ability to access nutrients from the deeper soil, however greater sampling at depth would be needed to confirm these hypotheses. Nevertheless, if verified, an ability to shift root production into in deeper layers could be a useful trait in soils that provide physical impediment in the subsoil or if greater soil stability is a desired goal of reforestation.

Another species-specific effect was observed with *F. schottiana* showing a decreasing relationship with ammonium concentrations and time post-reforestation. This result was not entirely unusual as ammonium is readily converted to nitrate in some soils and nitrate may be preferentially used by plants (Piccolo et al., 1994). However, we did not detect higher nitrate concentration in soil under *F. schottiana*, which would be expected if more ammonium was nitrified. This may be due to high N uptake and demand of *F. schottiana,* which was not investigated here. However, this finding conforms broadly to the results of prior studies (Reiners et al., 1994, Rasiah et al., 2004, Paul et al., 2010b), which note that ammonium levels are highest in pasture and lowest in forest.

It is noteworthy that the species-specific effects detected were not the same as those that were seen at a site level. At the site level when means were considered, only infiltration and bulk density were significant, however both *F. schottiana* and *F. coronata* showed significant variations through time for ammonium levels and fine root biomass respectively. This suggests that while individual species may be able to preferentially alter their immediate surroundings, their influence is diminished at a site level. A possible explanation is that the diverse mix of native species, representing a variety of plant physiotypes, and the individual species-specific effects on soil are less influential than the effects of the overall plantings on soil properties. The results for sites then may have been different and more closely resemble the species-specific effects seen had the trees in the reforestation planting been of a simpler species mix or the relative proportion of select species increased in the plots. This indicates that by explicitly selecting certain tree species for the reforestation plantings, particular reforestation outcomes may be able to be more easily achieved. Further studies have to determine the efficacy and benefits of a less species-rich reforestation species mixture.

5. Conclusion

In this study, reduced bulk density and enhanced infiltration rates were the key immediate outcomes resulting from riparian reforestation plantings. It is noteworthy that the most change occurred with just 5-10 years post-planting suggesting that this form of active restoration can have a rapid impact on reinstating important soil functions. Given the linkages between increased infiltration and various ecosystem services such as nutrient buffering, filtration of runoff and sediments, we conclude that to some degree the reforestation is, by extension, achieving the objective of improving water quality, regulation and soil health. This finding has consequences for land managers and conservationists interested in lowering runoff, non-point source erosion and limiting the loss of topsoil, all of which are major concerns and motivators for reforestation. Further, soil infiltration rates are intricately linked with various soil properties and are influenced by slight changes in soil structure well before changes are perceivable in other soil properties (Radke and Berry 1993). This may signify that the reforestation plantings are on a positive trajectory to restoring other soil properties and ecosystem functioning with infiltration rates the front-runner in these changes.

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References

AUSTRALIAN BUREAU OF STATISTICS. 2011. *Census* [Online]. Available: http://www.abs.gov.au [Accessed 5 April 2013].

BELL, L. W., KIRKEGAARD, J. A., SWAN, A., HUNT, J. R., HUTH, N. I. & FETTELL, N. A. 2011. Impacts of soil damage by grazing livestock on crop productivity. *Soil Biology and Biochemistry,* 113**,** 19-29.

BEVEN, K. & GERMANN, P. 1982. Macropores and water flow in soils. *Water Resources Research,* 18**,** 1311-1325.

BEZEMER, T. M., LAWSON, C. S., HEDLUND, K., EDWARDS, A. R., BROOK, A. J., IGUAL, J. M., MORTIMER, S. R. & VAN DER PUTTEN, W. H. 2006. Plant species and functional group effects on abiotic and microbial soil properties and plant–soil feedback responses in two grasslands. *Journal of Ecology,* 94**,** 893-904.

BHARATI, L., LEE, K.-H., ISENHART, T. M. & SCHULTZ, R. C. 2002. Soil-water infiltration under crops, pasture, and established riparian buffer in Midwestern USA. *Agroforestry Systems,* 56**,** 249-257.

BREMNER, J. M. & MULVANEY, C. S. 1982. Nitrogen—Total. *In:* PAGE, A. L. (ed.) *Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties.*

BRONICK, C. J. & LAL, R. 2005. Soil structure and management: a review. *Geoderma,* 124**,** 3-22.

CATTERALL, C. & HARRISON, D. A. 2006. Rainforest Restoration Activities in Australia's Tropics and Subtropics. Cairns, Australia: Cooperative Research Centre for Tropical Rainforest Ecology and Managemen, Rainforest CRC.

CATTERALL, C. P., KANOWSKI, J., WARDELL-JOHNSON, G. W., PROCTOR, H., REIS, T., HARRISON, D. & TUCKER, N. I. 2004. Quantifying the biodiversity values of reforestation: perspectives, design issues and outcomes in Australian rainforest landscapes. *Conservation of Australia's forest fauna,* 2**,** 359-393.

CHARRON, I., LALONDE, O., ROY, A. G., BOYER, C. & TURGEON, S. 2008. Changes in riparian habitats along five major tributaries of the saint Lawrence river, Québec, Canada: 1964–1997. *River Research and Applications,* 24**,** 617-631.

CLAUS, A. & GEORGE, E. 2005. Effect of stand age on fine-root biomass and biomass distribution in three European forest chronosequences. *Canadian Journal of Forest Research,* 35**,** 1617-1625.

COLLOFF, M. J., PULLEN, K. R. & CUNNINGHAM, S. A. 2010. Restoration of an ecosystem function to revegetation communities: The role of invertebrate mscropores in enhancing soil water infiltration. *Restoration Ecology,* 18**,** 65-72.

CONNOR, S., NELSON, P. N., ARMOUR, J. D. & HÉNAULT, C. 2012. Hydrology of a forested riparian zone in an agricultural landscape of the humid tropics. *Agriculture, Ecosystems & Environment*.

DEFOREST, J. L. 2011. Soil texture - Hydrometer method. *Lab protocols for testing soils* Ohio University, Athens, Ohio: The Department of Environmental and Plant Biology.

ELLISON, L. & COALDRAKE, J. E. 1954. Soil mantle movement in relation to forest clearing in Southeastern Queensland. *Ecology,* 35**,** 380-388.

FINÉR, L., OHASHI, M., MOGUCHI, K. & HIRANO, Y. 2011. Factors causing variation in fine root biomass in forest ecosytems. *Forest Ecology and Management,* 261**,** 265-277.

GRISCOM, H. P., GRISCOM, B. W. & ASHTON, M. S. 2009. Forest Regeneration from Pasture in the Dry Tropics of Panama: Effects of Cattle, Exotic Grass, and Forested Riparia. *Restoration Ecology,* 17**,** 117-126.

GUO, L. B. & GIFFORD, M. 2002. Soil carbon stocks and land use change: a meta analysis. *Global Change Biology,* 8**,** 345-360.

JIMÉNEZ, E. M., MORENO, F. H., PEÑUELA, M. C., PATIÑO, S. & LLOYD, J. 2009. Fine root dynamics for forests on contrasting soils in the Colombian Amazon. *Biogeosciencee,* 6**,** 2809-2827.

KANDELER, E. & GERBER, H. 1988. Short-term assay of soil urease activity using colorimetric determination of ammonium. . *Biology and Fertility of Soils,* 6**,** 68-72.

KANOWSKI, J., CATTERALL, C., WARDELL-JOHNSON, G., PROCTOR, H. & REIS, T. 2003. Development of forest structure on cleared rainforest land in eastern Australia under different styles of reforestation. *Forest Ecology and Management,* 183**,** 265-280.

KANOWSKI, J. & CATTERALL, C. P. 2010. Carbon stocks in above‐ground biomass of monoculture plantations, mixed species plantations and environmental restoration plantings in north‐east Australia. *Ecological Management & Restoration,* 11**,** 119-126.

KENZO, T., ICHIE, T., HATTORI, D., KENDAWANG, J. J., SAKURAI, K. & NINOMIYA, I. 2010. Changes in above- and below ground biomass in early successional tropical secondary forests after shifting cultivation in Sarawak, Malaysia. *Forest Ecology and Management,* 260**,** 875-882.

LAMB, D. 1980. Soil nitrogen mineralisation in a secondary rainforest succession. *Oecologia,* 47**,** 257-263.

LAUB, B. G., MCDONOUGH, O. T., NEEDELMAN, B. A. & PALMER, M. A. 2013. Comparison of designed channel restoration and riparian buffer restoration effects on riparian soils. *Restoration Ecology***,** 1-9.

LI, X. R., MA, F. Y., XIAO, H. L., WANG, X. P. & KIM, K. C. 2004. Long-term effects of revegetation on soil water content of sand dunes in arid region of Northern China. *Journal of Arid Environments,* 57**,** 1-16.

LOUMETO, J. J. & BERNHARD-REVERSAT, F. R. 2001. Soil fertility changes with eucalypt hybrids and plantation age: soil organic matter. *In:* BERNHARD-REVERSAT, F. R. (ed.) *Effect of exotic tree plantations on plant diversity and biological soil fertility in the Congo-Savanna: with special reference to eucalyptus.* Indonesia: Center for International Forestry Research.

MALONEY, K. O., GARTEN, C. T. & ASHWOOD, T. L. 2008. Changes in soil properties following 55 years of secondary forest succession at Fort Benning, Georgia, USA. *Restoration Ecology,* 16**,** 503-510.

MCINTYRE, D. 1976. Subplasticity in Australian soils. I. Description, occurrence, and some properties. *Australian Journal of Soil Research,* 14**,** 227-236.

MEYNENDONCKX, J., HEUVELMANS, G., MUYS, B. & FEYEN, J. 2006. Effects of watershed and riparian zone characteristics on nutrient concentrations in the River Scheldt Basin. *Hydrology and Earth System Sciences Discussions,* 3**,** 653-679.

MIRANDA, K. M., ESPEY, M. G. & WINK, D. A. 2001. A rapid, simple spectrophotometric method for simultaneous detection of nitrate and nitrite. *Nitric Oxide,* 5**,** 62-71.

MO, J., BROWN, S., PENG, S. & KONG, G. 2003. Nitrogen availabilty in disturbed, rehabilitated and mature forests of tropical China. *Forest Ecology and Management,* 175**,** 573-583.

MURTY, D., KIRSCHBAUM, M. U. F., MCMURTRIE, R. E. & MCGILVRAY, H. 2002. Does conversion of forest to agricultural land change soil carbon and nitrogen? a review of the literature. *Global Change Biology,* 8**,** 105-123.

NAIMAN, R. J., DÉCAMPS, H. & MCCLAIN, M. E. 2005. *Riparia: ecology, conservation and management of streamside communities.,* New York, Academic Press.

NAKAMURA, A., CATTERALL, C. P., HOUSE, A. P. N., KITCHING, R. L. & BURWELL, C. J. 2007. The use of ants and other soil and litter arthropods as bio-indicators of the inpacts of rainforest clearing and subsequent land use. *Journal of Insect Conservation,* 11**,** 177-186.

NAKAMURA, A., PROCTOR, H. & CATTERALL, C. P. 2003. Using soil and litter arthropods to assess the state of rainforest restoration. *Ecological Management & restoration,* 4**,** S20-S28.

NERIS, J., JIMÉNEZ, C., FUENTES, J., MORILLAS, G. & TEJEDOR, M. 2012. Vegetation and land-use effects on soil properties and water infiltration of Andisols in Tenerife (Canary Islands, Spain). *Catena,* 98**,** 55-62.

ORZETTI, L. L., JONES, R. C. & MURPHY, R. F. 2010. Stream condition in Piedmont streams with restored riparian buffers in the Chesapeake Bay watershed. *Journal of the American Water Recources Association,* 46**,** 473-485.

PAUL, K., POLGLASE, P., NYAKUENGAMA, J. & KHANNA, P. 2002. Change in soil carbon following afforestation. *Forest Ecology and Management,* 168**,** 241-257.

PAUL, M., CATTERALL, C. P., POLLARD, P. C. & KANOWSKI, J. 2010a. Does soil variation between rainforest, pasture and different reforestation pathways affect the early growth of rainforest pioneer species? *Forest Ecology and Management,* 260**,** 370-377.

PAUL, M., CATTERALL, C. P., POLLARD, P. C. & KANOWSKI, J. 2010b. Recovery of soil properties and functions in different rainforest restoration pathways. *Forest Ecology and Management,* 259**,** 2083-2092.

PERROUX, K. M. & WHITE, I. 1988. Designs for Disc Permeameters. *Soil Science Society of America Journal,* 52**,** 1205-1215.

PEÑUELA, M. C. & DREW, A. P. 2004. A model to assess restoration of abandoned pasture in Costa Rica based on soil hydrologic features and forest structure. *Restoration Ecology,* 12**,** 516-524.

PICCOLO, M. C., NEILL, C. & CERRI, C. C. 1994. Net nitrogen mineralization and net nitrification along a topical forest-to-pasture chronosequence. *Plant and Soil,* 162**,** 61-70.

POWERS, J. S. & PEREZ-AVILES, D. 2012. Edaphic factors are a more important control on surface fine roots than stand age in secondary tropical dry forests. *Biotropica,* 45**,** 1-9.

R CORE TEAM. 2013. *R: A language and environment for statistical computing.* [Online]. Vienna, Austria: R Foundation for Statistical Computing. Available: http://www.R-project.org/.

RADKE, J. K. & BERRY, E. C. 1993. Infiltration as a tool for detecting soil changes due to cropping, tillage, and grazing livestock. *American Journal of Alternative Agriculture,* 8**,** 164-174.

RASIAH, V., FLORENTINE, S. K., WILLIAMS, B. L. & WESTBROOKE, M. E. 2004. The impact of deforestation and pasture abandonment on soil properties in the wet tropics of Australia. *Geoderma,* 120**,** 35-45.

REINERS, W. A., BOUWMAN, A. F., PARSONS, W. F. J. & KELLER, M. 1994. Tropical rain forest conversion to pasture: changes in vegetation and soil properties. *Ecological Applications,* 4**,** 363-377.

ROBERTSON, G. P., COLEMAN, D. C., BLEDSOE, C. S. & SOLLINS, P. 1999. *Standard soil methods for long-term ecological research,* New York, Oxford University Press.

RODRIGUES NOGUEIRA JR, L., DE MORAES GONCALVES, J. L., LEX ENGEL, V. & PARROTTA, J. 2011. Soil dynamics and carbon stocks 10 years after reforestation of degraded land using Atlantic Forest tree species. . *Forest Systems,* 20**,** 536-545.

RODRIGUES, R. R., GANDOLFI, S., NAVE, A. G., ARONSON, J., BARRETO, T. E., VIDAL, C. Y. & BRANCALION, P. H. 2011. Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. *Forest Ecology and Management,* 261**,** 1605-1613.

RUSSO, T. A., FISHER, A. T. & ROCHE, J. W. 2012. Improving riparian wetland conditions based on infiltration and drainage behavior during and after controlled flooding. *Journal of Hydrology,* 432**,** 98-111.

SCHOENHOLTZ, S. H., VAN MIEGROET, H. & BURGER, J. A. 2000. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. *Forest Ecology and Management,* 138**,** 335-356.

SCHULTZ, R., ISENHART, T., SIMPKINS, W. & COLLETTI, J. 2004. Riparian forest buffers in agroecosystems–lessons learned from the Bear Creek Watershed, central Iowa, USA. *Agroforestry Systems,* 61**,** 35-50.

SCOTT, S. J., MCLAREN, G., JONES, G. & HARRIS, S. 2010. The impact of riparian habitat quality on the foraging and activity of pipistrelle bats (Pipistrellus spp.). *Journal of Zoology,* 280**,** 371-378.

SMITH, I. 2001. Juggling-balancing Tourism and Biodiversity in a Small, Valuable Reserve - Mary Cairncross Scenic Reserve. *In:* BOYES, B. (ed.) *Biodiversity Conservation "From Vision to Reality".* Gatton, Queensland: Lockyer Watershed Management Association (LWMA) Inc. - Lockyer Landcare Group.

SWEENEY, B. W., BOTT, T. L., JACKSON, J. K., KAPLAN, L. A., NEWBOLD, J. D., STANDLEY, L. J., HESSION, W. C. & HORWITZ, R. J. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America,* 101**,** 14132-14137.

VIDON, P. 2012. Towards a better understanding of riparian zone water table response to precipitation: surface water infiltration, hillslope contribution or pressure wave processes? *Hydrological Processes,* 26**,** 3207-3215.

WICKHAM, J. D., WADE, T. G. & RIITTERS, K. H. 2011. An environmental assessment of United States drinking water watersheds. *Landscape Ecology,* 26**,** 605-616.

WILLMOTT, W. F. 2007. *Rocks and landscapes of the Sunshine Coast,* Brisbane, Geological Society of Australia, Queensland Division.

WOOD, H. B. 1977. Hydrologic differences between selected forested and agricultural soils in Hawaii. *Soil Science Society of America Journal,* 41**,** 132-136.

ZHU, B., LI, Z., LI, P., LIU, G. & XUE, S. 2010. Soil erodibility, microbial biomass, and physical-chemical property changes during long-term natural vegetation restoration: a case study in the Loess Plateau, China. *Ecological Research,* 25**,** 531-541.