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Abstract

Land-use change due to the widespread practice of swidden agriculture affects the supply of ecosystem services. However, there is comparatively little understanding of how the hydrological functioning of soils, which affects rainfall infiltration and therefore flood risk, dry-season flows and surface erosion, is affected by repeated vegetation clearing and burning, the extent to which this can recover following land abandonment and vegetation regrowth, and whether active restoration speeds up recovery. We used interviews with local land users and indicator plant species to reconstruct the land-use history of 19 different sites in upland eastern Madagascar that represent four different land-use categories: semi-mature forests that were never burnt but were influenced by manual logging until 15–20 years ago; fallows that were actively reforested 6–9 years ago; 2–10 year old naturally regenerating fallows; and highly degraded fire-climax grassland sites. Surface- and near-surface (down to 30 cm depth) saturated soil hydraulic conductivities (K_{sat}), as well as the dominant flow pathways for infiltration and percolation were determined for each land-cover type. Surface K_{sat} in the forest sites was very high (median: 724 mm h^{-1}) and infiltration was dominated by flow along roots and other preferential flow pathways (macropores), whereas K_{sat} in the degraded land was low (median: 45 mm h^{-1}) with infiltration being dominated by near-surface matrix flow. The total area of blue-dye stains was inversely correlated to the K_{sat}. Both surface- and near-surface K_{sat} had increased significantly after 6–9 years of forest regeneration (median values of 203 and 161 mm h^{-1} for reforestation and natural regeneration, respectively). Additional observations are needed to more fully understand the rates at which soil hydrological functioning can be rebuilt and whether active replanting decreases the time required to restore soil hydrological functioning or not.

Keywords: forest regeneration, reforestation, preferential flow pathways, saturated hydraulic conductivity, runoff generation, swidden agriculture.

Introduction

Large areas of the agricultural-forest frontier in tropical countries are dominated by swidden cultivation (also known as shifting cultivation or slash-and-burn agriculture; Brady, 1996; Van Vliet et al., 2012). Swidden agriculture typically results in a mosaic of land uses, including naturally regenerating fallows. Where population pressure is high and rotation cycles have shortened, it also results in extensive patches of highly degraded land that are no longer included in the agricultural rotation (Kleinman et al., 1995; Malmer et al., 2005, Bai et al., 2008). The ecological and soil fertility values of land in the various phases of the swidden agricultural cycle, and the extent to which they improve during forest regrowth, have received significant attention (Szott et al., 1999; Chazdon, 2014; Mukul and Herbohn, 2016). However, despite the importance of water for rural communities and ecosystems, our understanding of how the hydrological functioning of tropical soils is impacted by repeated forest clearance and burning followed by vegetation regrowth is still rather limited (e.g. Toky and Ramakrishnan, 1981; Gafur et al., 2003; Ziegler et al., 2004). Also, evidence concerning the degree to which soil hydrological functions may be restored by assisted regeneration (cf. Dugan, 2000) as opposed to full-blown reforestation seems largely absent (Scott et al., 2005; Ilstedt et al., 2007).

The fire used in swidden agriculture can decrease soil organic carbon content, reduce rooting density and depth, and decrease soil biotic activity (Fragoso et al., 1997; Lavelle et al., 2001). Such changes can lead, in turn, to decreases in soil infiltration capacity and thus increased surface runoff (Toky and Ramakrishnan, 1981; Ziegler et al., 2004). Excess surface runoff generation in the case of advanced soil degradation may even impair soil- and groundwater recharge, which can have negative impacts on dry-season streamflow and community water resources (Bruijnzeel, 2004; Forsyth and Walker, 2008).

In response to such problems, and to promote carbon sequestration, biodiversity, and rural livelihoods, several major international initiatives (e.g. the Global Partnership on Forest Landscape Restoration/IUCN, 2011; UN, 2015; cf. Aronson and Alexander, 2013; Lamb, 2014) have committed to restoring large areas of the world's degraded and deforested land. However, the exact hydrological implications of such efforts, especially with respect to changes in the streamflow regime, are under debate (Jackson et al., 2005; Scott et al., 2005; Malmer et al., 2009). There are indications that the water use of vigorously regenerating vegetation can exceed that of old-growth forest (Giambelluca et al., 2000; cf. Ford et al., 2011) causing a reduction in streamflow during at least part of the succession (Swank et al., 2001; Lacombe et al., 2015). On the other hand, total streamflow and streamflow responses to rainfall are also influenced by soil hydrological functioning (Bonell, 2005), which has been shown to improve during natural forest regeneration (Ziegler et al., 2004; Zimmermann et al., 2010; Hassler et al., 2011) and after tree planting on degraded soils (Bonell et al., 2010; Benegas et al., 2014). This improvement is thought to reflect increases in soil organic matter, rooting density and depth, and, especially, soil faunal activity and the development of preferential flow pathways (macropores) during vegetation maturation (Colloff et al., 2010). As such, the quantification of rainfall infiltration and related soil hydrological characteristics, and changes therein during forest regeneration are important for understanding the hydrological effects of tropical land-use change.

A large proportion of Madagascar's renowned rain forest biome is now covered by a mosaic of land uses representing different stages of the swidden agricultural cycle, including highly degraded grasslands (Styger et al., 2007; Harper et al., 2007). Both governmental and conservation organisations have attempted to slow or stop the practice of swidden agriculture (Scales, 2014) and there have been occasional attempts at active reforestation (Portela et al., 2012; Busch et al., 2012). Recently, the Malagasy Government made a commitment to the United Nations Framework Convention of Climate Change to reforest 270,000 ha with native species, and greatly reduce the national rate of deforestation (Government of Madagascar, 2015). There have been claims that forest restoration is important in terms of hydrological regulation (Portela et al., 2012) but empirical studies of the effects of deforestation, forest regeneration or reforestation on soil hydrological functioning in Madagascar are scarce and concern observations made more than half a century ago (Bailly et al., 1974).

This study is part of a larger effort investigating the net hydrological impacts and ecosystem services of various land-cover types associated with swidden agriculture and forest regeneration in upland eastern Madagascar. The study region has experienced a long history of swidden cultivation, as well as various conservation interventions aimed at reducing forest clearance and burning, and, since 2005, active reforestation (Portela et al., 2012). Interviews with local people, and plant indicator species were used to identify 19 sites, which represented four widely occurring land-cover types (semi-mature forest, actively reforested fallows, young naturally regenerating fallows, and degraded grass- and shrub land). At these sites we investigated: (i) the differences in top-soil infiltration rates and associated soil physical properties, and (ii) differences in preferential flow pathways in order to determine how soil hydrological functioning is affected by land cover and whether active reforestation using native tree species results in a faster recovery of soil hydrological functioning after land abandonment than natural forest regrowth.

Materials and Methods

Research area

The research was carried out in two communities (Andasibe and Ambatovola) in the southern part of the Ankeniheny Zahamena Corridor (CAZ), which is a newly established protected area and REDD+ pilot project area. The CAZ is widely recognised for its extraordinary biodiversity (Le Saout et al., 2013). Swidden agriculture, in a system locally known as *tavy*, has been practiced in the region for many generations and is considered a major driver of deforestation (Styger et al., 2007; Clausen et al., 2013). The cutting and burning of forest (primary or secondary) is typically followed by one or two seasons of rice cultivation and a root crop the next season. The land is then left for natural fallow regrowth, until the recovering vegetation is cleared again (Styger et al., 2007). Due to rapid population growth, the length of the fallow cycle has decreased from 8–15 years in the 1970’s to as little as 3–5 years, resulting in degraded areas dominated by shrubs, ferns and grasses (Styger et al., 2007). Some of the degraded fallows in the research area were actively replanted with native tree species as part of the TAMS (Tetik Asa Mampody Savoka) reforestation project, which started in 2005 and planted more than 120 native species in more than 300 ha of degraded agricultural and forest land (Conservation International, 2011). The reforested sites did not receive regular follow-up maintenance (e.g. weeding of invasive species), and in most sites the trees are still relatively small (<5 m height) and do not yet provide closed canopy conditions.

The study area is characterized by steep slopes (>20°) and broad valleys; elevations range between 300 and 1800 m a.s.l. The area is underlain by Precambrian metamorphic and igneous basement rocks (granites, migmatites and schists) in which Oxisols and Ultisols have developed (Hervieu and Randrianaridera, 1956; Du Puy and Moat, 1996). Based on soil textural data down to 100 cm depth from the study area (Andriamananjara et al., 2016), the soils in our study sites are classified as Tropudults and typically show an increase in clay content at a depth of 60–70 cm. The climate is tropical monsoonal (Köeppen-type Am) with an average temperature at an elevation of 950 m a.s.l. (site number 8) of 15°C during the dry season (April to October) and 22°C during the wet season (November to March). Mean annual rainfall at Andasibe (990 m a.s.l.) was 1625 mm yr^{-1} for the 1983–2013 period (Météo Madagascar, unpublished data, 2013). Total rainfall measured with a tipping-bucket rain gauge (Rain Collector II, Davis Instruments, Hayward, USA; 0.2 mm per tip) near Andasibe at site number 8 (Figure 1) between October 2014 and September 2015 was 1650 mm. The median, 95th percentile and maximum 5-min rainfall intensities during this period were 3.0, 20 and 150 mm h^{-1}, respectively, with corresponding values of 1.0, 8.6, and 95 mm h^{-1} for the 15-min rainfall intensities. Almost a third of the annual rainfall (531 mm) fell at a 5-min intensity > 20 mm h^{-1}, while 41% (677 mm) occurred at a 15-min intensity > 8.6 mm h^{-1}.

<<Figure 1>>

Site selection

We wanted to sample sites that represented the four focal land-cover categories: (*i*) semi-mature forests that experienced heavy manual in the past but were never totally cleared and burned. These forests contain mostly small trees with a few larger trees (diameter at breast height ≥20 cm). It is likely that the latter represent remnant individuals that were considered too small to be harvested at the time of the latest harvesting (F); (*ii*) reforested shrub/tree fallows (RF), where endemic trees were actively replanted between 2005 and 2013 as part of the TAMS project; (*iii*) natural fallows (NF) dominated by shrubs and/or trees of natural succession on abandoned agricultural land; and (*iv*) highly degraded abandoned agricultural fields (DL) covered by scattered shrubs and grasses (fire-climax). Undisturbed mature forests that have not been influenced by illegal logging, and older fallows (>15 years) do not exist in the study area. We used a combination of previous work that describes the plant species composition associated with different land-cover stages (see Table 1), shape files showing reforested areas, and interviews with local leaders and land users to identify the areas that represented our four focal land-cover types. Within each land-cover category, we selected sites that were at least 120 m long and 75 m wide, without any major visual changes in vegetation or slope to minimise edge effects. A total of 19 sites were selected (Figure 1 and Supplementary Material 1). At each site measurements were taken at five locations at 15 m intervals along a 60 m transect. The transects were located along the hillslope gradient to avoid bias by only including upslope or downslope measurements (cf. Sobieraj et al., 2004; Ghimire et al., 2013).

<<Table 1>>

Field measurements

Soil physical characteristics

Soil cores (100 cm^3) were taken at two depths (12.5–17.5 cm and 22.5–27.5 cm) at each measurement point along a transect to determine porosity, moisture content at field capacity and bulk density. The moisture content at field capacity was defined as the volumetric moisture content after three days of gravity drainage rather than following the strict definition of the moisture content at a suction of 333 hPa (Koorevaar et al., 1983). The samples were saturated for 5 days, weighted, left to drain for 3 days and weighted again at the Andasibe field station. The samples were oven dried (24 h at 105 ^{o}C) at the Laboratoire des Radio Isotopes (University of Antananarivo) and weighted again. Porosity was determined following Klute (1986) by comparing the saturated weight and oven-dried weight of the samples. Samples for soil textural analysis were taken at the same depths as the cores and combined into one bulk sample per depth per transect. Particle size distributions were analysed at the VU University in Amsterdam using a QUIXEL Helium-Neon Laser Optical System (Sympatec GmbH, Clausthal-Zellerfeld, Germany). To ensure that there were no significant changes in soil texture along a transect, soil texture was determined at each measurement location in the field following Rowell (1994).

Saturated hydraulic conductivity

The saturated soil hydraulic conductivity (K_{sat}) describes the rate of steady-state infiltration (at the surface) or percolation (at depth). K_{sat} was measured at the soil surface, at 10–20 cm and at 20–30 cm depths. These measurement depths correspond with the main soil horizons in the study area and allow comparison of the results with other studies (e.g. Godsey and Elsenbeer, 2002; Zimmermann et al., 2006, 2010). Steady-state surface infiltration rates were determined using a portable double-ring infiltrometer (15 cm inner diameter, 21 cm outer diameter) that was inserted 9±3 cm into the soil, maintaining a constant head of 10±3 cm). These surface measurements were considered to represent the 0–10 cm layer. Values of sub-soil K_{sat} were measured using a constant-head field permeameter (Amoozegar, 1989). The diameter of the auger hole was 6.0±0.5 cm at 10–20 cm depth and 5.6±0.5 cm at 20–30 cm. The applied constant head was 17.5±1.5 cm.

Dye tracer experiments

Dye tracer experiments were carried out to characterise the soils of the investigated land-cover types in terms of their dominant infiltration and percolation patterns, i.e. matrix flow (through soil pores) vs. preferential pathways (along roots and through macropores) (Beven and Germann, 1982). Water with 2 g L^{-1} Brilliant Blue Dye (FCF C.I. 42090) was sprayed on a 1 m^2 plot at an average intensity of 20 mm h^{-1} in the middle of the transect at six study sites (forest, *n*=2; reforestation, *n*=1; natural fallow, *n*=1; and degraded land, *n*=2). Each plot was divided into two parts: the upper half received 20 mm of dye, the lower half received 40 mm. The irrigated plots were covered with a plastic sheet and the soil was excavated the next day. Six sections were excavated per plot (three per application rate), described qualitatively in the field and photographed for subsequent analysis (cf. Weiler and Fluhler, 2004), to determine: (i) the so-called volume density (i.e., the fraction of soil that contained blue dye, representing the fraction of the soil where water infiltrated), (ii) the fraction of blue stains narrower than 2 cm (indicating the dominance of preferential flow pathways with little interaction with the matrix), and (iii) the fraction of stains that were wider than 20 cm (indicating the dominance of preferential flow pathways with high interaction with the matrix or homogeneous matrix flow).

Data analysis

Differences in bulk density, porosity, soil moisture content at field capacity, sand, silt, and clay contents, as well as differences in K_{sat} between the respective land-cover types were tested for statistical significance by applying the Kruskal-Wallis analysis of ranks with Dunn’s method (Kruskal-Wallis, 1952). Differences were taken to be significant for values of $p < 0.05$. Spearman rank correlation (r_s) analysis was used to determine the correlation between K_{sat} and the other soil physical characteristics.

Median surface and subsurface K_{sat}-values for the different land-cover types were compared to the 95th percentiles of the 5-min and 15-min rainfall intensities as measured at Andasibe to infer the dominant runoff pathways (i.e. infiltration-excess overland flow occurrence, vertical percolation, lateral subsurface flow or saturated overland flow; cf. Bonell et al., 2010; Ghimire et al., 2014).

Results

Soil physical characteristics

Soil texture was either clay, clay loam, or sandy clay loam. Overall clay content at the various study sites varied between 23 and 66%, silt between 12 and 34%, and sand between 8 and 65%. Although clay contents were highest (and sand contents lowest) for the degraded land sites, the differences in sand, silt or clay contents between the land-cover types were not significant (Table 2). Sand, silt and clay contents did not differ significantly between the two depths intervals (12.5–17.5 cm and 22.5–27.5 cm) either.

Bulk density at 12.5–17.5 cm depth was significantly lower for the forest sites than for any of the other land-cover types, but the differences at a depth of 22.5–27.5 cm between the land cover types were small and not significant (Supplementary Materials 2 and 3a). Likewise, differences in porosity (for either depth interval) between the different land cover types were small and not significant. Although the differences in moisture content at field capacity between land cover types were larger than those for porosity, they were also not significant (Supplementary Materials 2, 3b and 4a). Drainable porosity (i.e., total porosity minus moisture content at field capacity) at 12.5–17.5 cm did not differ significantly between the different land cover types but at 22.5–27.5 cm the median value for the forest sites was significantly smaller than that for the degraded land sites (Supplementary Materials 2 and 4b). The results for drainable porosity as a fraction of total porosity were similar.

Saturated hydraulic conductivity

Values of saturated hydraulic conductivity (K_{sat}) were generally higher for the forest sites than for any other land cover (for all three measurement depths). However, the scatter in the individual measurements was such that only the difference between the median K_{sat} of the relatively undisturbed forest soils (724 mm h^{-1}) and that of the degraded land (45 mm h^{-1}) was statistically significant (Figure 2a and Table 2). At a depth of 10–20 cm, the median K_{sat} for the forest sites (87 mm h^{-1}) and reforestation sites (56 mm h^{-1}) were significantly higher than those for the natural fallows (14 mm h^{-1}) and the heavily degraded sites (20 mm h^{-1}) (Figure 2b and Table 2). At 20–30 cm depth, only the median K_{sat} of the forest (4.3 mm h^{-1}) and that for the soil at the degraded land sites (0.8 mm h^{-1}) differed significantly from each other (Figure 2c and Table 2).

<<Table 2>>

K_{sat} decreased quickly with depth at all sites (Figure 2 and Table 2). While the median surface K_{sat} exceeded the 95th percentiles of the 5- and 15-min rainfall intensities for all four land cover types (Figure 2a), K_{sat}-values at 20–30 cm depth were well below these intensities, regardless of land-cover type (Figure 2c). Median K_{sat} at 10–20 cm depth beneath the forest sites and reforestation sites (87 and 56 mm h^{-1}, respectively) was much larger than the 95th percentile of 5-min rainfall intensity (20 mm h^{-1}), while the median K_{sat}

for the young natural fallow sites (14 mm h^{-1}) and degraded land (20 mm h^{-1}) was similar to the 95th percentile of the 5-min rainfall intensity.

Median K_{sat} per transect at 10–20 and at 20–30 cm was not significantly correlated with the sand or clay contents. K_{sat} at 10–20 and at 20–30 cm was not significantly correlated with the bulk density or porosity either (not known for 0–10 cm). The K_{sat} at 10–20 cm depth was significantly correlated with moisture content at field capacity (r_s = 0.23), drainable porosity (r_s = -0.40), and the ratio of moisture content at field capacity and porosity (r_s = 0.40) at 12.5–17.5 cm depth. There was also a weak but statistically significant correlation between K_{sat} at 20–30 cm depth and the drainable porosity at 22.5–27.5 cm (r_s = -0.20). Taking the data for the reforestation and natural fallow sites together (Figure 3), surface K_{sat} appeared to increase with time since agricultural abandonment, although the relationship was not particularly strong (r_s = 0.42). The correlation improved (r_s = 0.68) when only considering the reforestation sites but was not significant for the natural fallow sites. Values of K_{sat} at 10–20 cm or 20–30 cm depth were not correlated with time since abandonment.

<<Figure 2>>

<<Figure 3>>

Dye infiltration patterns

There were no significant differences in the dye patterns or the maximum depth of dye infiltration between the 20 and 40 mm applications. Therefore, results for the two applications were analysed together for each land-cover type. For the semi-mature forest, as well as the reforestation/natural fallow sites, the infiltrated dye was located mainly along larger macropores (Figure 4a and 4b). The forest soils were mostly characterized by macropore flow with mixed interaction with the soil matrix (Figure 4a). The excavated soil sections of the degraded land plots showed a more or less homogeneously stained top layer (0–15 cm) where the fine roots were concentrated. The associated blue dye patterns were mainly characterized by matrix flow and occasional ‘fingering’ (Figure 4). Where macropore flow reached greater depth, it mainly occurred through worm holes or along old roots and was characterised by relatively limited interaction with the soil matrix. Infiltration patterns in the reforestation and young natural fallow sites varied considerably between sections and could not be characterized by a single dominant flow type. Because of this large variability and the small number of blue dye experiments, these sites were further analysed as one land-cover category (RF/NF).

<<Figure 4 >>

The difference in median maximum volume density between the forest sites and RF/NF plots was significant, with the median value recorded for the forest (0.72) being much larger than the median for the younger regrowth (0.23; Table 3). The fraction of stains with a width larger than 20 cm was also greatest for the forest sites (median: 0.22) but the difference with the other land cover types was not statistically significant (median of 0.00 for the RF/NF sites vs. 0.10 for the degraded land). The fraction of stains smaller than 2 cm was lower for the forest soil sections (median: 0.37) than for the RF/NF (median: 0.49) and degraded land sections (median: 0.63). Even though these differences were not statistically significant, they do suggest a trend towards more and larger macropores and especially increased interaction with the soil matrix as vegetation regrows and the soil recovers (Table 3). Further support for the increased importance of preferential flow pathways in the sites with more mature vegetation comes from the fact that top-soil K_{sat} (0–10 cm) was inversely correlated ($r^2 = 0.72$) with the total blue-stained area (median of the 6 sections) per site; sites dominated by matrix flow had the largest blue dye stained area in the upper soil layers (Figure 5 and Supplementary Material 5).

<<Table 3>>

<<Figure 5>>

Discussion

Limitations of the space-for-time substitution approach

Although the limitations of using space-for-time substitutions are well recognised (Pickett, 1989), very few studies of changes in saturated soil hydraulic conductivity (K_{sat}) during tropical vegetation regrowth on degraded soils have taken measurements in (near-) real time after abandonment of agricultural land for cropping or grazing, or after tree planting (e.g. Zimmermann et al., 2010; Patin et al., 2012; Ghimire et al., 2014). Like we did here, the overwhelming majority of studies employed a space-for-time substitution (vegetation chrono-sequences) approach for practical reasons (e.g. Gilmour et al., 1987; Deuchars et al., 1999; Ziegler et al., 2004; Zimmermann et al., 2006; Hassler et al., 2011). An important challenge in using chrono-sequences when investigating the impact of land-cover change over time is to eliminate the influence of inherent differences in soil characteristics between sites and to reconstruct the land-use history at a particular location. The first obstacle can be largely overcome by carefully selecting sites that have the same soil type (Zimmermann et al. 2006). In this study, all plots were on the same metamorphic rock type, had a similar soil type and sub-soil textural differences between land-cover types were not statistically different. Further, K_{sat} was not strongly correlated with soil physical characteristics like bulk density or porosity, which suggests that differences in soil type or texture did not affect K_{sat} as much as land-cover type and that there was sufficient initial pedological homogeneity to allow the respective sites to be compared. Precise land-use history in the study area varies at a very fine scale across the landscape and finding sites with a truly identical history to group together is difficult or impossible. However, by using a combination of available secondary data (i.e. shape files from the TAMS reforestation project; Conservation International, 2011), indicator plant species, and key informant interviews we were able to identify sites falling into general categories of past land use. Detailed interviews with local people at each site gave additional information (such as time since abandonment), which allowed us to explore the impact of land-use history on K_{sat}.

Effect of land-cover type on saturated soil hydraulic conductivity

We observed much lower K_{sat}-values in the natural fallow sites and, especially, the degraded land sites than in the forest sites (Table 2 and Figure 2). The differences in K_{sat} between the degraded sites and the forest sites persisted to 30 cm depth. This suggests that repeated burning and cropping cycles, combined with shortening recovery periods (Styger et al., 2007), has a negative impact on soil hydrological properties down to a depth of at least 30 cm. This finding is similar to that of Ziegler et al. (2004), who found K_{sat} at 40–

70 cm depth beneath recently abandoned swidden fields and up to ~20 year old regenerating vegetation in northern Vietnam to be much lower than in the (disturbed) forest (median values of 15–45, 35–50, and 80–85, mm h^{-1}, respectively). Zimmermann et al. (2006), working in SW Brazil, found that clearance for swidden agriculture followed by a single season of cropping and 15 years of regrowth caused a hydrologically insignificant decrease in soil infiltration capacity (from 1690 to 940 mm h^{-1}, i.e. well above maximum rainfall intensities), but they found a more pronounced effect when clearance was followed by two years of cultivation and 20 years of grazed pasture (median K_{sat} of 113 mm h^{-1}). These effects were noticeable to at least 20 cm depth, although the magnitude of change diminished rapidly with depth (Zimmermann et al., 2006).

Our median K_{sat}-values for the top-soil are similar to those of Bailly et al. (1974), who worked in our study area during the 1960s and early 1970s (724 mm h^{-1} vs. 720 mm h^{-1} for the forest sites and 161 mm h^{-1} vs. 115 mm h^{-1} for the young fallows/'old bush' sites). However, literature values of median infiltrabilities for relatively young regenerating forests (<10 years) replacing grazed pasture or swidden agriculture elsewhere in the tropics are generally much lower (32–38 mm h^{-1}; Ziegler et al., 2004; Hassler et al., 2011) than we recorded for the 6–9 year-old reforestation sites (203 mm h^{-1}, Table 2). Corresponding published values for slightly older (12–20 years) successional vegetation range from approximately 65 mm h^{-1} (Ziegler et al., 2004) through 160 mm h^{-1} (Hassler et al., 2011) to 495–945 mm h^{-1} (Deuchars et al., 1999; Zimmermann et al., 2006). Our results for the semi-mature forest sites, which contain many trees that are 15-20 years old fall also in the higher range of these values, although they cannot be direct compared because they were never burned or cultivated. These differences in the median K_{sat}-values for our study sites and the literature values for other tropical areas likely reflect differences in the intensity of the disturbance regime and initial K_{sat} upon land abandonment (i.e. level of past soil degradation), as well as climatic (notably seasonality and length of dry season) and inherent edaphic factors (soil fertility) that affect the rate of regrowth and development of soil biological activity (Deuchars et al., 1999; Hairiah et al., 2006; Colloff et al., 2010).

Recovery of soil hydrological functioning

We found that surface K_{sat} increased with time since agricultural abandonment when the results for the natural fallow and reforestation sites were combined but the trend was not significant when considering the NF sites only. Median values of K_{sat} at 20–30 cm depth did not differ significantly for the natural fallow and reforestation sites either (Table 2). Therefore, it is not clear from the measurements whether active reforestation decreases the time needed for hydrological restoration of the soil compared to natural regeneration. This could be due to the limited number of measurements (n = 30 for NF and n = 20 for RF) or that any difference is masked by differences in land-use history and the degree of degradation prior to agricultural abandonment. In addition, the time since planting (6–9 years) or land abandonment (2–10 years) was likely too short to distinguish between the two regenerative pathways. Unfortunately, older regenerating sites and reforested sites cannot be found in the area. We, therefore, suggest that our measurements should be repeated in the future (e.g. five and ten years and even further after the current measurements) as the effects of active reforestation versus natural regeneration may take longer to manifest than the age of sites available in this study.

We found no significant differences in K_{sat} at 10–20 cm or 20–30 cm below the surface between the natural fallow sites and the degraded grassland sites. Nor did we find a significant increase in sub-soil K_{sat} with time since land abandonment (Table 2). This suggests that the subsurface K_{sat} requires a (much) longer time to recover than surface K_{sat}.

Similar findings have been reported by Ziegler et al. (2004), Zimmermann et al. (2006), and Hassler et al. (2011).

An increase in top-soil saturated hydraulic conductivity with time following the cessation of agricultural activity has been reported in several other tropical studies (e.g., Deuchars et al., 1999; Ziegler et al., 2004) but certainly not by all (e.g. de Moraes et al., 2006; Zimmermann and Elsenbeer, 2008; Zimmermann et al., 2010). Patin et al. (2012) observed a nearly eight-fold increase in surface K_{sat} under fallow vegetation (no age given) during annual repeated measurements over a period of six years (from 23 to 176 mm h^{-1}) in Laos. Similarly, large contrasts in the recovery of surface K_{sat} have been reported after abandonment of grazing land. Hassler et al. (2011) found an initial improvement in median infiltrability after eight years of forest regeneration in Panamá (from 23 to 38 mm h^{-1}) followed by a rapid increase to 160 mm h^{-1} for 12–15 year-old regrowth, compared to 235 mm h^{-1} under old-growth forest. On the other hand, Zimmerman et al. (2010) measured infiltrability and near-surface K_{sat} for seven consecutive years during natural succession on an abandoned pasture in SW Brazil and found a slight but non-significant recovery during this period. A similar lack of change in surface K_{sat} has been reported by Zimmermann and Elsenbeer (2008) for 10-year-old regrowth in the Ecuadorian Andes. They attributed this to arrested regeneration because succession was dominated by bracken that prevented the establishment of pioneer tree species. While Colloff et al. (2010) showed a steady increase in surface K_{sat} and macropores with age of plantations of eucalypts and *Acacias,* the difference with nearby pastures only became significant after more than 11 years of growth.

Ilstedt et al. (2007) suggested in their review of the scant tropical literature that a three-fold increase in surface K_{sat} may be achieved, although they were hesitant to attach a time frame given the paucity of good-quality data. Ziegler et al. (2004) considered a natural succession period of at least 25 years to be necessary to recover most of the surface K_{sat} after 2–4 years of swidden cultivation in upland Vietnam. Extrapolation of the initial changes in infiltrability measured by Hassler et al. (2011) in Panamá suggests that full soil physical recovery there might be achieved within ca. 20 years. Ghimire et al. (2014), however, cautioned that reforestation *per se* does not guarantee an increase in K_{sat} and the restoration of the hydrological system if a site is not properly managed (e.g. fire disturbance or repeated harvesting of litter and branches for animal bedding and fuelwood). In such cases, surface K_{sat}-values may actually decrease again with time (Ghimire et al., 2014; Lacombe et al., 2015).

Effect of land use on preferential flow pathways

The infiltration capacity of clayey and silty soils is primarily affected by their organic matter content and the abundance and connectivity of preferential flow pathways (Deuchars et al., 1999; Zhou et al., 2008). The blue dye patterns observed in this study suggest that preferential flow caused the infiltration rates to be highest for the semi-mature forest sites and lowest for the degraded land. Generally, the infiltration pattern was more uniform in the higher-conductivity top layer which is relatively rich in organic matter, while percolation in the lower-conductivity and more clayey layer at around 7.5–15 cm depth, occurred mainly along preferential flow pathways (Figure 4). The preferential flow pathways were most abundant in the forest sites and less abundant in the reforestation, natural fallows and degraded land sites. There were fewer preferential flow pathways in the degraded sites, but where they occurred, they allowed water to move deeper than in the forest sites, in part because of the lower interaction with the matrix. While the dye experiments were useful to visualize the differences in the infiltration and percolation pathways, the number of experiments was too small to determine any statistically significant differences in the dye patterns between the land cover types. Additional experiments are thus needed to see if there are differences in the maximum depth of the dye or the volume density. These measurements could be combined with the device advanced by Mendoza and Steenhuis (2002) that allows the separate measurement of vertical and lateral fluxes.

A clear increase in preferential flow pathways (relative to those in adjacent pastures and young tree plantations) was noted by Colloff et al. (2010) for 11–20-year-old tree plantations, with most of the macropores attributed to the activity of soil invertebrates like ants and termites. Hanson et al. (2004) showed for a site in Honduras that high surface infiltration rates and well-connected preferential flow channels in an aggregated clayey soil beneath primary forest resulted in rapid vertical infiltration to a depth of 35 cm. However, in a nearby degraded grassland site that had been subject to repeated slash and burn activity, infiltration rates were very low and excessive lateral flow occurred at and just beneath the surface and very little water infiltrated below 10 cm, even though macropores were present below this depth. These findings were explained in terms of blocking of near-surface macropores by fine sediment that was washed in from upslope (Hanson et al., 2004). Benegas et al. (2014) reported that in a mixed land-use setting (tree clumps within grazed pasture) in Costa Rica, preferential flow was only dominant close to mature trees, while matrix flow increased with distance from the trees. This effect was attributed to the combined action of tree root- and soil faunal activity beneath and in the vicinity of trees (Benegas et al., 2014). Bachmair et al. (2009) studied dye infiltration patterns in Germany in tilled and untilled farmland, pasture and deciduous forest and found large differences in maximum infiltration depth for the different land uses. They also found that larger rainfall applications resulted in deeper infiltration, except under forest. We did not find significant differences in the maximum depth of infiltration (or any other parameter describing the blue dye patterns) for the 20 and 40 mm applications (Table 3). Instead, the blue dye patterns and K_{sat} profiles with depth suggest that most of the infiltrated water stays in the top 30 cm of the soil or results in shallow lateral flow with very little water percolating through the denser clay layers below.

Implications for runoff generation processes

Whether rainfall infiltrates into a soil or flows along the surface depends largely on the magnitude of the surface K_{sat} relative to the prevailing rainfall intensity, and, in addition, on the change in K_{sat} with depth (Elsenbeer, 2001; Bonell, 2005). The relatively high surface K_{sat}-values exceed most rainfall intensities observed in the study area, suggesting that infiltration-excess overland flow is a rather rare phenomenon. However, K_{sat} decreased sharply with depth for all land-cover types (Figure 2 and Table 2), as was also reported for many other tropical studies (Godsey and Elsenbeer, 2002; Ziegler et al., 2004; Zimmermann et al., 2006, 2010; Hassler et al., 2011). Rain water will thus percolate vertically through the soil profile until meeting the first layer that has a lower K_{sat} than the incident precipitation rate, and will then start to accumulate above this layer (Figure 6). Depending on the magnitude of the lateral K_{sat} and the slope gradient, water will either flow laterally above this impeding layer or saturate the soil layers above it. For large rainfall events coinciding with high antecedent soil moisture conditions, this can lead to saturation-excess overland flow (Elsenbeer, 2001; Bonell, 2005). Because of the relatively low K_{sat}-values observed already at 10–20 cm depth in the young natural fallow sites and the degraded sites (Table 2), less water will be needed there to fully saturate the soil and generate saturation-excess overland flow at these sites than at the forest sites (Figure 6). This difference can reflect the removal of the top layer in the more degraded sites by surface erosion during past cultivation periods (cf. Ziegler et al., 2004). In fact, surface runoff was observed after 30 minutes of application of the blue dye at site number 15 (degraded land).

<<Figure 6>>

Enhanced surface runoff in the form of saturation- or infiltration-excess can lead to higher peak flows, more soil erosion and subsequent declines in soil fertility and water quality (Ziegler et al., 2009). It can also lead to decreased groundwater recharge and potentially lower streamflow during the dry season (Bruijnzeel, 2004). Bailly et al. (1974) conducted long-term catchment and erosion studies across Madagascar, including several sites located near the current field sites. The catchment of Bailly et al. (1974) that was classified as being under naturally regenerating vegetation (‘*brousse*’, no age given but presumably less than 10 years old) was characterized by greater volumes of surface runoff and higher peak flows compared to a nearby closed-canopy forest catchment. However, Lacombe et al. (2015) reported gradually diminishing streamflow during 12 years of natural regeneration in an area previously under swidden cultivation in Vietnam. Flows declined both during the wet and the dry season due to a combination of better infiltration and higher vegetation water use as the area under secondary forest expanded and matured over time. Conversely, Beck et al. (2013) did not find any statistically significant trends in long-term streamflow characteristics (high flows or low flows) when combining the results for twelve meso-scale catchments in Puerto Rico undergoing major changes in secondary forest cover. Different results were obtained for individual catchments, suggesting significant spatial heterogeneity and highlighting the importance of including multiple sites when analysing land-cover impacts on hydrological functioning of tropical catchments.

Conclusions

Swidden agriculture continues to be an important land-use practice in many tropical forest areas. Understanding its influences on important soil- and water-related ecosystem services is therefore important. Our study in eastern Madagascar shows that land degradation, which can arise from swidden agriculture with short fallow cycles, changes soil functioning in ways that reduce rainfall infiltration. Infiltration into the forest soil was dominated by preferential flow with a high interaction with the soil matrix, while infiltration in the degraded land was mainly due to matrix flow in the top soil layers. We found a sharp decline in soil hydraulic conductivity with depth and a low hydraulic conductivity relative to the prevailing rainfall intensities in the degraded sites, which suggest that saturated overland flow in the degraded land is common. Enhanced overland flow occurrence can result in progressive soil erosion and degradation and diminished rates of soil water- and groundwater recharge, which may ultimately impact dry-season flows in streams and rivers.

Our results, further, suggest that saturated soil hydraulic conductivity at the surface increased after several years of land abandonment and forest regrowth. However, we found no significant differences at 20–30 cm depth. Full hydrological recovery of degraded sites with vegetation regrowth may, therefore, take several decades. Due to differences in soil degradation before reforestation or natural regrowth, and the short time span since reforestation (< 10 years), it remains unclear whether active replanting decreases the time required for soil hydrological restoration. Given the interest in active forest restoration in Madagascar, as in many other areas of the tropics, further work is needed to more fully understand the rates at which soil hydrological functioning can be rebuilt and to quantify the extent to which active replanting, rather than passive regeneration, can contribute to more rapid rebuilding of soil- and water-related ecosystem services.

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Research ethics

The research was approved under the Bangor University Research Ethics framework. Soil pits were backfilled to minimise damage. Field work was conducted under permits provided by the Madagascar Ministry of Environment, Ecology, Sea and Forests

(050/14/MEF/SG/DGF/DCB.SAP/SCB) and with permission from the local authorities and local land owners. All informants were reassured that they would not be identified and that their participation was completely voluntary.

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Table 1: Indicator species and fallow succession stages based on Styger et al. (2007), Klanderud et al. (2010), and Schatz (2005).

Species	Family	Fallow stage	Cropping / fallow cycle
Solanum mauritianum Scop.	Solanaceae (tree)	Tree fallow	1
Clidemia hirta (L.) D. Don	Melastomataceae (tree/shrub)	Shrub fallow (not dominant)	-
Cryptocarya R. Br.	Lauraceae (tree)	-	-
Croton L. sp.	Euphorbiaceae (tree)	Tree/shrub fallow	1 – 2
Tambourissa Sonn.	Monimiaceae (tree)	Tree/shrub fallow	1 – 2
Trema orientalis (Blume)	Ulmaceae (tree)	Tree fallow	1 – 2
Harungana madagascariensis Lam. Ex Poir	Clusiaceae (tree/shrub)	Tree/shrub fallow	1 – 5
Psiadia altissima	Asteraceae (shrub/tree)	Tree/shrub fallow	1 – 6
Aframomum angustifolium (Sonn.) K. Schum	Zingiberaceae (shrub, perennial herbaceous)	Shrub fallow (not dominant)	2 – 6
Lantana camara L. (invasive)	Verbenaceae (shrub)	Shrub fallow	2 – 6
Rubus moluccanus L. (invasive)	Rosaceae (shrub)	Shrub fallow	2 – 6
Imperata cylindrica (L.) Raeusch.	Poaceae (herb/grass)	Shrub fallow / grassland	> 3
Pteridium aquilinum (L.) Kuhn	Dennstaedtiaceae (herbaceous, fern)	Shrub fallow	3 – 7
Sticherus flagellaris (Bory ex Willd.) Ching	Gleicheniacea (fern)	Shrub fallow	3 – 7
Aristida similis Steud	Poaceae (herb/grass)	Shrub fallow / grassland	> 6
Hyparrhenia rufa (Nees) Stapf	Poaceae (herb/grass)	Shrub fallow / grassland	> 6
Psorospermum Spach	Clusiaceae (shrub)	Shrub fallow / grassland	> 6

Table 2: Median soil hydraulic conductivity (K_{sat}) and sand, silt and clay fractions, per land-cover type. Different superscript letters denote statistically significant differences between the land cover types.

	F	RF	NF	DL
K_{sat} 0–10 cm [mm h^{-1}]	724^{a}	203^{ab}	161^{ab}	45^{b}
K_{sat} 10–20 cm [mm h^{-1}]	87^{a}	56^{a}	14^{b}	20^{b}
K_{sat} 20–30 cm [mm h^{-1}]	4.3^{a}	0.9^{ab}	0.9^{ab}	0.8^{b}
Sand [%]*	29.2	31.6	30.6	19.8
Silt [%]*	27.2	26.7	21.3	26.0
Clay [%]*	43.7	40.5	45.8	53.3

*No statistically significant differences found between land covers. Number of soil texture samples: F: 4; RF: 6; NF: 11; DL: 8.

Table 3: Median values for the characteristics describing dye tracer patterns. Different superscript letters denote statistically significant differences between land-cover types.

	F	RF/NF	DL
Maximum depth of infiltrated blue dye [cm]	31^{a}	25^{a}	35^{b}
Maximum dye volume density [-]	0.72^{a}	0.23^{b}	0.53^{ab}
Fraction of stains smaller than 2 cm [-]	0.37^{a}	0.49^{a}	0.63^{a}
Fraction of stains larger than 20 cm [-]	0.22^{a}	0.0^{a}	0.10^{a}
Size of the stained area [cm²]	364^{ab}	114^{a}	567^{b}