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Asymmetric response to disturbance and recovery: Changes of soil permeability under forest–pasture–forest transitions

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ABSTRACT

In the humid tropics, continuing high deforestation rates are seen alongside an increasing expansion of secondary forests. In order to understand and model the consequences of these dynamic land-use changes for regional water cycles, the response of soil hydraulic properties to forest disturbance and recovery has to be quantified.

At a site in the Brazilian Amazônia, we annually monitored soil infiltrability and saturated hydraulic conductivity (K_s) at 12.5, 20 cm, and 50 cm soil depth after manual forest conversion to pasture (year zero to four after pasture establishment), and during secondary succession after pasture abandonment (year zero to seven after pasture abandonment). We evaluated the hydrological consequences of the detected changes by comparing the soil hydraulic properties with site-specific rainfall intensities and hydrometric observations. Within one year after grazing started, infiltrability and K_s at 12.5 and 20 cm depth decreased by up to one order of magnitude to levels which are typical for 20-year-old pasture. In the three subsequent monitoring years, infiltrability and K_s remained stable. Land use did not impact on subsoil permeability. Whereas infiltrability values are large enough to allow all rainwater to infiltrate even after the conversion, the sudden decline of near-surface K_s is of hydrological relevance as perched water tables and overland flow occur more often on pastures than in forests at our study site. After pasture abandonment and during secondary succession, seven years of recovery did not suffice to significantly increase infiltrability and K_s at 12.5 depth although a slight recovery is obvious. At 20 cm soil depth, we detected a positive linear increase within the seven-year time frame but annual means did not differ significantly. Although more than a doubling of infiltrability and K_s is still required to achieve pre-disturbance levels, which will presumably take more than a decade, the observed slight increases of K_s might already decrease the probability of perched water table generation and overland flow development well before complete recovery.

1. Introduction

The disappearance of tropical forests is mainly driven by the conversion of forests to cropland and pastures (Lindsey, 2007), and in the Brazilian Amazônia, the rate of conversion has picked up momentum again in recent years (WWF, 2009). In spite of continuing high deforestation rates, forest cover in 18 countries worldwide has begun to increase as a result of afforestation (tree plantations on previously unforested land) and natural regeneration (Chazdon, 2008). In the Brazilian Amazônia, for instance, secondary forests have reclaimed 31% of the once deforested land (Perz and Skole, 2003). On a pan-tropical scale, these developments already led to a predominance of secondary compared to old-growth forests (ITTO, 2002), which often differ in species composition and structure from the forests that they replace (Chazdon, 2003; Lugo, 2009). It has yet to be shown if these 'novel forests' (Lugo, 2009) restore ecosystem services, such as maintaining soil and watershed functions, to the level of their native counterparts.

There has been much debate about the hydrological role of a (new) forest cover (e.g. Andréassian, 2004; Brown et al., 2005; Calder et al., 2007; van Dijk et al., 2009). For instance, the 'low flow problem', i.e. the availability of sufficient dry-season flow, was identified as one of the most important 'watershed' issues requiring further research (Bruijnzeel, 2004). In case of forest clearing, the maintenance of soil surface characteristics at a level which allows continued infiltration of rainwater will increase dry season flow because of the reduced evapotranspiration associated with forest removal; in contrast, soil degradation and reduced infiltration after clearing and during subsequent non-forest land use will lead to diminished dry season flows (Bruijnzeel, 2004). If, subsequently, forest regeneration were to

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restore dry season flow, soil hydraulic recovery would have to outweigh the increasing amount of water that is transpired and intercepted by the growing trees. Clearly, former land use, soil type, climatic regime and type of regrowth (e.g. exotic tree plantations vs. natural succession) all influence this interplay between soil and vegetation. In this study, we attempt to shed light on the soil hydraulic aspect of forest disturbance and recovery. We focus on the ubiquitous conversion of tropical forests to pasture and on the common reclamation of pasture land by secondary forest in the course of natural succession.

The conversion of tropical forest to cattle pasture seems to inevitably affect soil hydraulic properties at or close to the soil surface (Alegre and Cassel, 1996; Martínez and Zinck, 2004; Zimmermann et al., 2006; Zimmermann and Elsenbeer, 2008), which is often attributed to soil compaction resulting from mechanized clearing (e.g. Ziegler et al., 2006), tillage (e.g. Lal, 1996), or the introduction of grazing animals. Regarding the latter, the influence of cattle treading on soil macroporosity and related properties such as the saturated hydraulic conductivity (hereafter K_s) was clearly demonstrated (e.g. McDowell et al., 2003; Pietola et al., 2005; Pande and Yamamoto, 2006). Several mechanisms were suggested to reverse its detrimental effect (listed in Drewry, 2006). Among those, wetting and drying cycles and subsequent soil cracking, the creation of biopores by soil macrofauna, and root penetration and decay are potentially relevant in tropical regions. Site-specific variations of the existence, strength, and timing of these mechanisms most likely influence the duration of soil rejuvenation. Additional variability may be introduced by different management practices and stocking rates during the period of active pasture use, which eventually produces a multitude of post-grazing initial conditions in terms of degradation severity; Scott et al. (2005) pointed out the critical role of initial conditions for the recovery process.

Indeed, the available evidence of a recovery of soil hydraulic properties from cattle trampling differs widely in terms of recovery time, reaching from a few weeks (Warren et al., 1986) to some months (Nguyen et al., 1998) to many years (Braunack and Walker, 1985; Zimmermann and Elsenbeer, 2008). Most work on post-grazing recovery of soil hydraulic properties was done in New Zealand (Nie et al., 1997; Singleton and Addison, 1999; Drewry, 2006), and improvements were inferred from differences to controls without grazing or areas below fence lines.

In the humid tropics, the landscape mosaic typically includes old-growth forests, which offers the unique opportunity to compare the soil hydrology of abandoned pastures to that of those forests, which archive the original pre-disturbance conditions. There are too few studies available to date that would allow for more than a tentative guess about the time required for any tropical pasture soil to return to its pre-disturbance stage in terms of soil hydrology, but a decadal timeframe seems to be a reasonable working hypothesis (deduced from de Moraes et al., 2006; Zimmermann et al., 2006; Zimmermann and Elsenbeer, 2008).

As discussed above in connection with the low-flow problem, quantification of land use-induced changes of soil hydraulic properties is important because of their essential role in the hydrological cycle. Ilstedt et al. (2007) pointed out that any model of forest–water relations should include soil physical quality such as infiltration levels before and after reforestation. Their claim is supported, for instance, by the fact that hydrological catchment models behave very sensitively to soil parameterization in relation to land use (Bormann et al., 2007).

Apparently there is a mismatch between the need for land use-dependent soil hydraulic information on the one hand and the availability of that data on the other. In order to improve the understanding of deforestation and forest recovery on soil hydrology, we monitored changes in soil infiltrability and soil saturated hydraulic conductivity after forest conversion to pasture, and after pasture abandonment and subsequent natural succession at a site in the Brazilian Amazônia. The monitoring approach enabled us not only to quantify the rate but also the timing of changes following forest conversion and regeneration. In order to infer likely hydrological consequences, the soil hydraulic data were compared to prevailing rainfall intensities.

2. Study area

The study site, Rancho Grande (10°18′S, 62°52′W, 143 m a.s.l.), is a 1000-ha farmland in the Brazilian state of Rondônia. The area is part of a morphostructural unit known as “Southern Amazon Dissected Highlands” (Planalto Dissecado Sul da Amazônia, Peixoto de Melo et al., 1978), which is characterized by a pronounced topography with an altitudinal differential of up to 150 m. Remnant ridges of Precambrian basement rock, made up of gneisses and granites of the Xingu (Leal et al., 1978) or Jamari Complex (Isotta et al., 1978), are separated by flat valley floors of varying width. Soil orders associated with this morphostructural unit are mainly Ultisols (Soil Survey Staff, 1999) in valley floors, Inceptisols (Soil Survey Staff, 1999) on steep slopes and Entisols (Soil Survey Staff, 1999) along streams. The climate is tropical wet and dry (Köppen's Aw; Koeppen, 1931). From 1984–2003 mean annual temperature was 27 °C and mean annual precipitation was 2300 mm/a, with a marked dry period from July through September (Germer et al., 2006).

The undisturbed forest vegetation at Rancho Grande is predominantly terra firme open tropical rainforest (Floresta Ombrófila Aberta) with a large number of palm trees. Open tropical rainforest is a transition forest between the dense tropical rainforest of central Amazônia and the cerrado vegetation (savanna) of the southern Amazônia (IBGE, 2004; Germer et al., 2009). At our study site it covers all hills and portions of the lowland. The remaining area was cleared between 1979 and 1980, mainly by slash and burn. The cleared area was converted to pasture; in the meantime, 75 ha of the originally cleared land reverted to secondary forest. All pasture areas at Rancho Grande are grazed with one head of cattle per hectare throughout the whole year.

2.1. Site selection and monitoring approach

We monitored the change of soil hydraulic properties after forest conversion to pasture and during natural succession of secondary forest on former pasture. The disturbance sequence was recorded on a 22 ha area of the farmland, where a 15-year-old secondary forest was slashed and burned before a cattle pasture was established. These manual clearing operations did not affect soil hydrology as infiltrability and K_s at three soil depths of the cleared area were undistinguishable from nearby old-growth forest (Zimmermann et al., 2006). For the recovery time series, we fenced a plot in a ≈ 20-year-old cattle pasture to exclude livestock. Its long side bordered on old-growth forest, i.e. seed sources for secondary vegetation were in immediate vicinity. Both monitoring plots were selected according to slope and soil type, which we classified as Kandiudult (Soil Survey Staff, 1999); this classification implies inherently low fertility. Sand content was high across all plots and amounted to more than 70% in the first 30 cm of soil, with a constant silt content of around 10% throughout the upper 100 cm. Clay content increased up to 45% in the diagnostic subsoil horizon.

Plot sizes were 5000 m^2 for the deforestation and 7000 m^2 for the recovery plot, respectively.

Soil hydraulic measurements were taken annually in the month of July, i.e. in the dry season. We conducted five monitoring campaigns on the deforestation plot (year zero to four after pasture establishment) and eight on the recovery plot (year zero to seven after pasture abandonment). Infiltrability measurements on the recovery plot