Saturated hydraulic conductivity in the humid tropics: Sources of variability, implications for monitoring and effects on near-surface hydrological flow paths

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Abstract

Large areas in the humid tropics are currently undergoing land-use change. The decrease of tropical rainforest, which is felled for land clearing and timber production, is countered by increasing areas of tree plantations and secondary forests, emerging on formerly agricultural areas which were abandoned in the process of urbanization. These changes are known to affect the regional water cycle as a result of plant-specific water demand and by influencing key soil properties which determine hydrological flow paths.

One of these key properties sensitive to land-use change is the saturated hydraulic conductivity (K_s) as it governs vertical percolation of water within the soil profile. Low values of K_s in a certain soil depth can form an impeding layer and lead to perched water tables, diminished groundwater recharge and the development of predominantly lateral flow paths. Overland flow is one example of these lateral flow paths and can be induced by saturation of the topsoil above such impeding layers, as return flow when water re-emerges from perched water tables or pipes, or by infiltration excess when rainfall intensities exceed K_s at the soil surface. Enhanced water flow at the soil surface at the expense of groundwater recharge can cause nutrient redistribution, erosion and soil degradation and thus affects ecosystem services and human livelihoods dependent on discharge quality and quantity. Due to its sensitivity to land-use change, K_s is commonly used to assess the associated changes in hydrological flow paths. However, both K_s and the occurrence of overland flow have been shown to exhibit high spatial variability, and land-use dependent process relationships between both variables are still unclear.

The objective of this dissertation was to assess sources of K_s variability, evaluate implications for monitoring of K_s changes and examine K_s effects on near-surface hydrological flow paths in the context of land-use change. The research area was located in central Panama, a country widely experiencing the abovementioned changes in land use. The tropical climate provides high amounts of annual rainfall, therefore changes in water flow paths will immediately affect both ecosystem services and human livelihoods.

Understanding the sources of variability and the changes in K_s can allow prediction of these changes based on soil maps or other more readily available soil data, which is especially important in data-scarce regions such as the humid tropics. Both static, soil-inherent properties such as particle size and clay mineralogy and dynamic, land use-dependent properties such as organic carbon content influence soil structure and thereby K_s. By conducting a pair of studies with one of these influences held constant in each, the importance of static and dynamic properties for K_s was assessed. Applying a space-for-time approach to sample K_s under secondary forests of different age classes on comparable soils, a recovery of K_s from the former pasture use was shown to require more than eight years. The process was limited to the 0-6 cm sampling depth and showed large variability among replicates. A wavelet analysis of a K_s transect crossing different soil map units under comparable land cover, old-growth tropical rainforest, showed large small-scale variability, which was attributed to biotic influences, as well as a possible but non-conclusive influence of soil types. The two results highlight the importance of dynamic,

land use-dependent influences on K_s, visible both in the small-scale variability on the transect and in the variability among replicates in the succession study. Additionally, the change of these influences during secondary succession were shown to result in a subsequent change in K_s.

Monitoring studies can help to quantify land use-induced change of K_s, but there is a variety of sampling designs which differ in efficiency of estimating mean K_s. A comparative study of four designs and their suitability for K_s monitoring is used to give recommendations about designing a K_s monitoring scheme. Quantifying changes in spatial means of K_s for small catchments with a rotational stratified sampling design did not prove to be more efficient than Simple Random Sampling due to the spatial properties of K_s. The lack of large-scale spatial structure prevented beneficial stratification, and large small-scale variability resulting from local biotic processes and artificial effects of destructive sampling caused a lack of temporal consistency in the re-sampling of locations, which is part of the rotational design. Consequently, pilot studies that specifically address spatial structure of K_s were recommended to improve decisions on sampling design for a given K_s monitoring task.

The effect of K_s on near-surface hydrological flow paths is of critical importance when assessing the consequences of land-use change in the humid tropics. Flow paths are commonly inferred using either the "pedological approach", which entails a comparison of detailed K_s surveys to rainfall intensities, or the "hydrological approach", by directly measuring water volumes of the different flow paths or using natural tracers to separate contributions of flow paths to stream flow. Rarely are both approaches combined in a spatially explicit way. Therefore, the last part of this dissertation aimed at disclosing spatial relationships between K_s and overland flow as influenced by different land cover types. The effects of K_s on overland-flow generation were spatially variable, different between planar plots and incised flowlines and strongly influenced by land-cover characteristics. A simple comparison of K_s values and rainfall intensities was insufficient to describe the observed pattern of overland flow. Likewise, event flow in the stream was apparently not directly related to overland flow response patterns within the catchments. Although not conclusive, the dataset highlights the limitations of the two approaches and emphasizes the combination of pedological, hydrological, meteorological and botanical measurements to comprehensively understand the land use-driven change in hydrological flow paths.

In summary, K_s proved to be a suitable parameter for assessing the influence of land-use change on soils and hydrological processes. The results illustrated the importance of land cover and spatial variability of K_s for decisions on sampling designs and for interpreting overland-flow generation. As relationships between K_s and overland flow were shown to be complex and dependent on land cover, an interdisciplinary approach is required to comprehensively understand the effects of land-use change on soils and near-surface hydrological flow paths in the humid tropics.

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All of the work presented in this dissertation was carried out within the Agua Salud Project (ASP). I am very grateful for having been part of this project which provided me with the opportunity of getting to know many regions in Panama, of working in a variety of fascinating tropical ecosystems and of meeting many fantastic people. Apart from funding my work, the ASP also exposed me to the challenges of interdisciplinary research in the collaboration of different groups, and I want to especially thank Fred Ogden and Jefferson Hall for stimulating discussion beyond my own focus are.

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Contents

List of Figures

List of Tables

Chapter 1

Introduction

In the humid tropics, large areas are subject to land use change. On the one hand tropical rainforest is still being felled for timber production or land clearing, on the other hand urbanization leads to abandonment of agricultural land which is subsequently converted to timber plantations or is subject to secondary succession. As land-use change is known to affect soil properties and thereby hydrological processes, understanding its consequences is important for estimating changes in ecosystem services and possible implications for human livelihoods. Saturated hydraulic conductivity (hereafter abbreviated as K_s) is one of these key soil parameters shaping hydrological flow paths, and is sensitive to land use change.

K_s governs the movement of water through soil under saturated conditions. Its role in the hydrological cycle is fundamental: Rainfall reaching the soil surface, either directly or as throughfall when passing a vegetation canopy, is partitioned onto different hydrological flow paths because its vertical percolation within the soil is controlled by K_s. When high rainfall intensities exceed very low values of K_s at the soil surface or a low infiltration capacity, Hortonian overland flow (HOF) is generated. When low K_s values occur in a shallow soil depth, the soil profile above this impeding layer might be saturated during rainfall events. The resulting perched water tables can induce saturated excess overland flow (SOF) when reaching the soil surface or return flow in the case of lateral water movement and topographical forcing. However, enhanced water flow at the soil surface at the expense of groundwater recharge can cause nutrient redistribution, erosion and soil degradation and affects ecosystem services and human livelihoods dependent on discharge quality and quantity. Due to its sensitivity to land-use change, K_s is commonly used to assess the associated changes in hydrological flow paths. However, both K_s and the occurrence of overland flow have been shown to exhibit high spatial variability, and land-use dependent process relationships between both variables are still unclear.

The objective of this dissertation was to assess sources of K_s variability, evaluate implications for monitoring of K_s changes and examine K_s effects on near-surface hydrological flow paths in the context of land-use change.

1.1 Sources of variability of K_s

Saturated hydraulic conductivity of the soil is directly related to its effective porosity (Ahuja et al., 1984) and is thus determined by soil structure. Factors influencing soil structure and hence K_s can be roughly differentiated into static, soil-inherent properties such as clay mineralogy and particle size distribution and dynamic properties such as organic carbon content or biotic activity; the latter are greatly influenced by land use and land cover.

Knowledge about the importance of each of these influences on K_s is especially relevant for hydrological modelling because in data-scarce regions, information on K_s is frequently lacking as extensive K_s measurements are very time-consuming. However, soil maps depicting soil types or texture classes and land use maps might be available. When attempting to deduce K_s from such more readily available soil data using pedo-transfer functions, those functions which incorporated soil structure measures proved most successful in predicting K_s (Wösten et al 2001), hinting at a combined influence of soil type and land use. Yet, it remains to be clarified whether information on static soil-inherent properties or on dynamic properties are more useful for these predictions in tropical soilscapes.

Of the static properties, clay content and clay mineralogy have been shown to affect soil structure (Alekseeva, 2007; Senarath et al., 2010; West et al., 2008), whereas particle size has had no direct influence on K_s in a catena study in Amazonia (Sobieraj et al., 2002). The importance of the dynamic influence has been highlighted in various studies on the effect of land-use changes on K_s in the humid tropics. For example, the decrease in K_s after rainforest is cleared for pasture establishment has been documented in Honduras, Amazonia, and Ecuador (Germer et al., 2010; Hanson et al., 2004; Lal, 1996; de Moraes et al., 2006; Zimmermann et al., 2006) and is largely attributable to changes in soil structure by cattle treading (Martinez & Zinck, 2004; de Moraes et al., 2006). Vice versa, there are several reports of K_s recovery after pastures are abandoned (de Moraes et al., 2006; Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2010a), pointing towards the reversibility of soil structure changes.

The first aim of this dissertation was to evaluate both static, soil-inherent and dynamic, land use-dependent influences affecting K_s in soils of our research area in central Panama. A pair of studies was conducted to assess each of these influences, while the other remained constant.

1.2 Implications for monitoring K_s

Quantifying shifts in K_s resulting from land-use change requires measurement of K_s in space and time, a resource- and time-consuming task. In most studies this problem is avoided by using a space-for-time approach, assuming that the temporal trend of a process at certain location can be described by examining different stages of this process at various locations. For instance, the change of K_s under reforestation in the humid tropics has been mainly described with this approach (Hassler et al., 2011b; Nyberg et al., 2012; Peng et al., 2012; Zimmermann & Elsenbeer, 2008). However, in many cases the underlying assumption that the examined processes can be regarded as random in space might not be valid. Therefore, true chronosequences are necessary to capture exclusively the temporal shifts in K_s during land-use change.

When designing a sampling scheme for K_s monitoring, several crucial decisions must be taken, such as choosing between a design-based or model-based approach (Brus & de Gruijter, 1997, 1993; Papritz & Webster, 1995) and establishing the details on sampling frequency in space and time (Black et al., 2008; de Gruijter et al., 2006; Lark, 2009; Papritz & Webster, 1995). Although there are many theoretical studies and books about these questions (e.g. de Gruijter et al., 2006), finding the most appropriate design and conducting the necessary preliminary surveys still prove to be considerable challenges.

Thus, the second aim of this thesis was to compare different K_s monitoring schemes regarding their efficiency in estimating mean K_s and to provide recommendations about how to find a suitable design for a particular K_s monitoring problem.

1.3 Effects of K_s on near-surface hydrological flow paths

K_s affects hydrological flow paths mainly by supporting or impeding vertical percolation of water.

In the literature of tropical soils, especially the effect of land use on K_s and hydrological flow paths has been focus of many studies in the last decades (Chandler & Walter, 1998; Chaves et al., 2008; Germer et al., 2010; Hanson et al., 2004; Hassler et al., 2011b; Zimmermann et al., 2006). Most of these studies employ one of the following approaches: From a pedological perspective, K_s is used as target variable because of its sensitivity to land use change. Assessment of change in flow paths is then based on comparisons of K_s profiles with rainfall characteristics (Hassler et al., 2011b; Martinez & Zinck, 2004; Ziegler et al., 2004; Zimmermann et al., 2006). From a hydrological perspective, analysis of a variety of hydrometeorological measurements and discharge behaviour is used to estimate the contribution of the different flow paths, frequently coupled with an examination of solute or stable isotope composition (Biggs et al., 2006; Chandler & Walter, 1998; Chaves et al., 2008; Mun oz Villers & McDonnell, 2012; Neill et al., 2011).

Rarely are both approaches combined in sufficient detail to elucidate the spatial variability in hydrological flow paths although evidence of considerable spatial variability in K_s suggests more complicated processes (e.g. Miyata et al., 2010; Sobieraj et al., 2002; Zimmermann & Elsenbeer, 2008). Similarly, spatial heterogeneity in overland flow generation has been reported (Germer et al., 2010; Loos & Elsenbeer, 2011) and partly attributed to corresponding differences in K_s (Godsey et al., 2004; Zimmermann et al., 2013). Land-use change can alter these patterns and effects considerably (Germer et al., 2010; Zimmermann & Elsenbeer, 2008). Additionally, spatially variable overland flow generation within a catchment can affect soil structure, nutrient redistribution and erosion processes locally without necessarily affecting discharge response in the same magnitude (Miyata et al., 2010). Therefore, a combination of both pedological and hydrological approaches with a focus on spatial variability could considerably contribute to process understanding of land-use effects on hydrological flow paths.

This dissertation initially focuses on a more detailed understanding of processes related to K_s variability. However, studying the effects of this variability on hydrological flow paths is the logical consequence from the previous work. Therefore the last study aims at describing the relations between K_s and overland flow generation and differences therein with respect to land use.

1.4 Research area

The research area of this dissertation is situated in central Panama. The tropical climate provides high annual rainfall amounts (2300–2600 mm), therefore land use-dependent changes in K_s and hydrological flow paths, which will affect regional water budgets and may promote soil degradation, are of key importance in these regions.

One focus area of this dissertation is Barro Colorado Island (BCI), a former mountain top which became an island when the Panama Canal was flooded in 1914. Having been declared a National Monument and conservation area in 1946, it stimulated a multitude of research projects under the governance of the Smithsonian Tropical Research Institute. Besides research on fauna and flora of the island, its soils have been focus of various studies (e.g. Barthold et al., 2008; Daws et al., 2002; Grimm et al., 2008; Vincent et al., 2010), one outcome being a very detailed soil map (Baillie et al., 2007). This amount of information is exceptional when contrasted with the data scarcity typical for tropical soils (Hodnett & Tomasella, 2002). And as the range of soil types on the island roughly covers those in the Panama Canal Zone (Turner & Engelbrecht, 2011), study results can be transferred. Coupled with easy accessibility and excellent infrastructure, this site provides an ideal basis for a sampling-intensive study of soil-type effects on K_s variability while the influence of land cover, old-growth tropical forest, remains constant.

The second focus area of the dissertation is the project area of the Agua Salud Project (ASP). This project ”seeks to understand and quantify the ecological, social, and economic services provided by tropical forests in the Panama Canal Watershed” (http://www.ctfs.si.edu/aguasalud/). The ASP project area is underlain by a pre-Tertiary basalt plateau and its geology and soils are relatively homogeneous. Land cover, however, varies greatly: there is a range of secondary forests of different ages across the region, and old-growth forest in the neighbouring Soberanía National Park. It includes actively used pastures, catchments with mixed land cover and several newly established plantations of native species and teak. The area is particularly suited to studying the effects of land cover change on soil hydraulic properties of similar soil types, to observing soil changes during the first years of reforestation and to examining the effects of these changes on near-surface hydraulic flow paths.

The scope for cooperation within the project facilitates a comprehensive understanding of the ecosystem processes. For soil parameters such as K_s, it is particularly useful to consider studies of other project members on, for instance, throughfall as the actual water input into the soil, vegetation changes as an integral influence and discharge dynamics as the integral of water flow transport from the catchment.

1.5 Overview

This dissertation encompasses four studies about saturated hydraulic conductivity: Two studies focus on the sources of variability and influencing factors for K_s, another highlights issues of sampling design with respect to K_s measurements and monitoring, and the last aims at elucidating the consequences of changes in K_s for near-surface hydrological flow paths. A visual overview is illustrated in Figure 1.1.

The chapters are organised as follows:

- Chapter 1: *Introduction*

  A general introduction about the role of K_s in the context of the water cycle is followed by the research questions and objectives which originated the four following studies. Relevant aspects of the research area are mentioned and the chapter closes with a summary of the aims and methodological approaches of the different studies.

- Chapter 2: *Recovery of saturated hydraulic conductivity under secondary succession on former pasture in the humid tropics*

  The influence of land cover on K_s is examined by using a space-for-time approach. Pasture sites and secondary forests of different ages span the process of K_s recovery under secondary succession from pasture to old-growth tropical rainforest, while soil type remains constant.

- Chapter 3: *Exploring the variation in saturated hydraulic conductivity under a tropical rainforest using the wavelet transform*

  The influence of soil type on K_s, while land cover (old-growth tropical rainforest) remains constant, is investigated in this transect study across contrasting soil map units. The sampling design facilitates wavelet analysis which elucidates changes in variability of K_s across the transect.

- Chapter 4: *Which sampling design to monitor saturated hydraulic conductivity?*

  Monitoring K_s requires decisions about appropriate sampling in space and time. This chapter compares a rotational stratified design to simpler designs, with respect to efficiency in predicting mean K_s for three study catchments. It closes with practical recommendations for selecting a suitable sampling design for a given soil variable.

- Chapter 5: *Land use-specific overland flow generation in the humid tropics*

  The combination of K_s, rainfall and overland flow measurements coupled with observations of ancillary parameters of vegetation and soil, discharge and dye tracer experiments, aims at disclosing possible relations between K_s and near-surface hydrological flow paths.

- Chapter 6: *Summary and Conclusions*

  The findings of the four studies with regard to the overall research aims are summarised, limitations are mentioned and implications for future research are included.

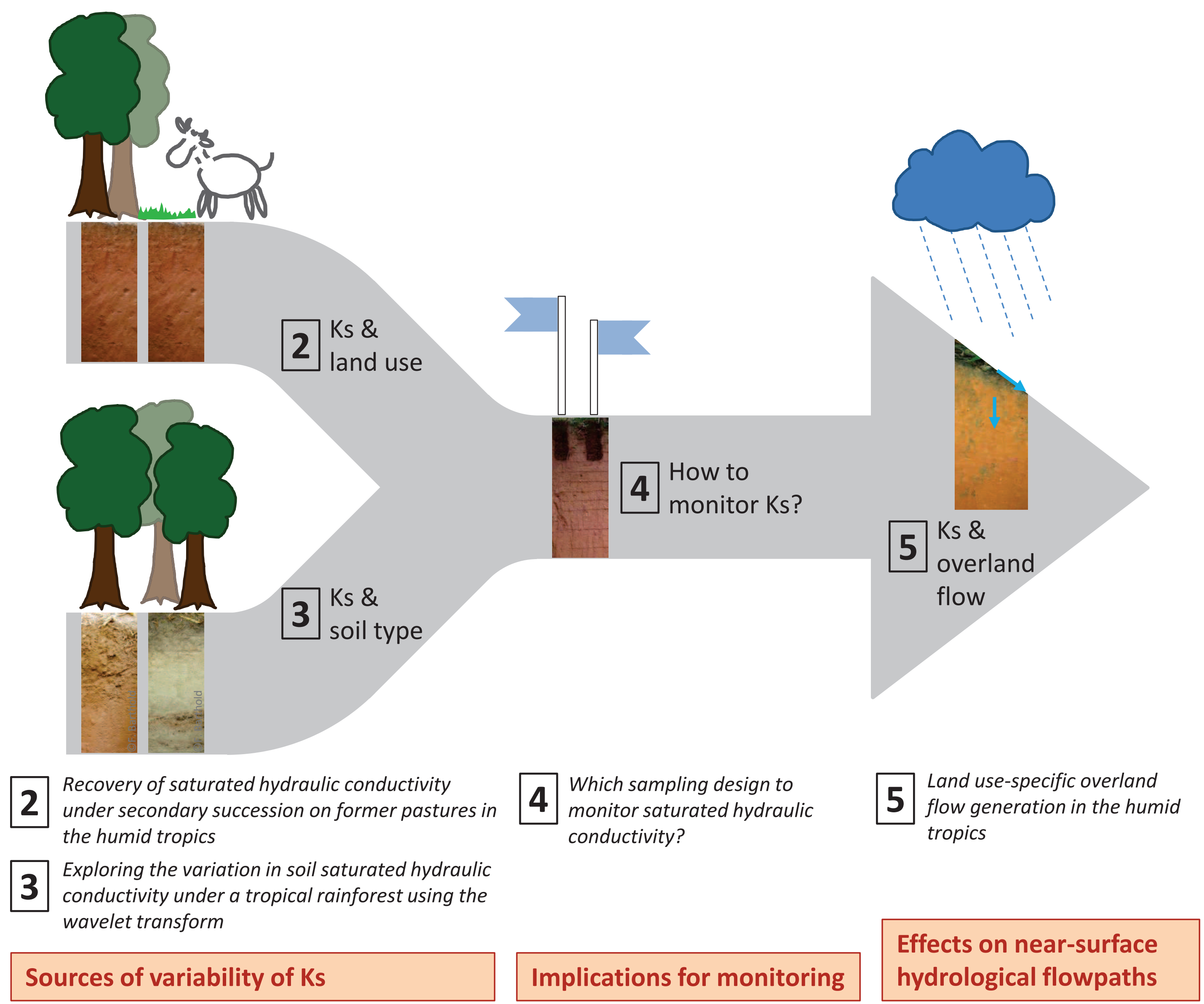

Figure 1.1: Visual overview of the thesis. Chapter numbers are shown in the boxes.

Chapter 2

Recovery of saturated hydraulic conductivity under secondary succession on former pastures in the humid tropics *

Abstract

Landscapes in the humid tropics are undergoing a continuous change in land use. Deforestation is still taking its toll on forested areas, but at the same time more and more secondary forests emerge where formerly agricultural lands and pastures are being abandoned. Regarding soil hydrology, the extent to which secondary succession can recover soil hydrological properties disturbed by antecedent deforestation and pasture use is yet poorly understood. We investigated the effect of secondary succession on saturated hydraulic conductivity (K_s) at two soil depths (0–6 and 6–12 cm) using a space-for-time approach in a landscape mosaic in central Panama. The following four land-use classes were studied: pasture (P), secondary forest of 5–8 years of age (SF5), secondary forest of 12–15 years of age (SF12) and secondary forest of more than 100 years of age (SF100), each replicated altogether four times in different micro-catchments across the study region. The hydrological implications of differences in K_s in response to land-use change with land use, especially regarding overland flow generation, were assessed via comparisons with rainfall intensities.

Recovery of K_s could be detected in the 0–6 cm depth after 12 years of secondary succession: P and SF5 held similar K_s values, but differed significantly ($\alpha = 0.05$) from SF12 and SF100 which in turn were indistinguishable. Variability within the land cover classes was large but, due to sufficient replication in the study, K_s recovery could be detected nonetheless. K_s in the 6–12 cm depth did not show any differences between the land cover classes; only K_s of the uppermost soil layer was affected by land-use changes. Overland flow – as inferred from comparisons of K_s with rainfall intensities – is more likely on P and SF5 sites compared to SF12 and SF100 for the upper sample depth; however, generally low values at the 6–12 cm depth are likely to impede vertical percolation during high rainfall intensities regardless of land use.

We conclude that K_s can recover from pasture use under secondary succession up to pre-pasture levels, but the process may take more than 8 years. In order to gain comprehensive understanding of K_s change with land use and its hydrological implications, more studies with detailed land-use histories and combined measurements of K_s, overland flow, precipitation and throughfall are essential.

*Hassler, S. K., Zimmermann, B., van Breugel, M., Hall, J. S., Elsenbeer, H. (2011), *Forest Ecology and Management*, 261 (10), 1634–1642, http://dx.doi.org/10.1016/j.foreco.2010.06.031.

2.1 Introduction

Landscapes in the humid tropics are undergoing a continuous change in land use, with two processes occurring simultaneously. On the one hand, ongoing deforestation still reduces primary forest in favour of creating pastures and agricultural land (Lindsey, 2007). On the other hand, as a consequence of social shifts in rural communities resulting in migration to cities, pastures and agricultural lands are being abandoned and secondary forests are rapidly spreading in tropical regions (Chazdon, 2008; Lugo, 2009), partly balancing primary forest loss. At the same time, active reforestation gains increasing importance (Malmer et al., 2010). In Panama, the location of our study area, natural forest decreased by 1.3 % and secondary forests increased by 7 % between 1992 and 2000 (Wright & Samaniego, 2008). However, the extent to which these secondary forests or active plantations provide the same ecosystem services as the former old-growth forests they ultimately replace has yet to be identified. Aspects that need to be evaluated include conservation of biodiversity, carbon sequestration, protection against erosion, impacts on local climate and alterations of water and nutrient cycles (e.g. Biggs et al., 2004; Bruijnzeel, 2004; Silver et al., 2000). Additionally, tradeoffs between these different ecosystem services are important for management decisions, for instance, negative implications for the water cycle, for example reduced baseflow because of enhanced water uptake by fast-growing trees, might in some cases outweigh benefits for carbon sequestration or nutrient cycling (Bennett et al., 2009; Jackson et al., 2005; Raudsepp-Hearne et al., 2010).

With respect to the hydrological cycle in the humid tropics, land-use changes have been shown to alter discharge (Brown et al., 2005), nutrient losses (Germer et al., 2009) and hydrological flowpaths (Chaves et al., 2008; Germer et al., 2010). Precipitation can take different paths once reaching the soil surface in sloping terrain: it may chiefly percolate vertically towards the groundwater; or it may run off atop the soil surface as overland flow; it may infiltrate to a shallow impeding layer and move laterally downslope (subsurface flow) or be forced back to the surface further downhill, resulting in return flow (Elsenbeer, 2001). A crucial parameter in the partitioning of incoming precipitation into these hydrological flowpaths is the soil saturated hydraulic conductivity (K_s), a property determining the rate with which water can move though the soil matrix under saturated conditions. Low values for K_s at the surface or in a shallow soil depth hinder vertical percolation of the water during high-intensity rainfall events, leading to a domination of lateral flowpaths, to perched water tables and overland flow (Bonell & Gilmour, 1978; Chappell & Sherlock, 2005; Elsenbeer, 2001; Germer et al., 2010). In turn, overland flow may aggravate erosion processes and thus nutrient depletion and land degradation. Due to its direct link to macroporosity (Ahuja et al., 1984), K_s has been found to be sensitive to soil disturbance (Alegre & Cassel, 1996; Ziegler et al., 2004; Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2006) and can therefore serve as indicator for the effects of land-use changes on soil hydrology. Subsequent comparison of K_s values to prevailing rainfall intensities further yields rough estimates of potential overland flow generation (de Moraes et al., 2006; Ziegler et al., 2004, 2006; Zimmermann & Elsenbeer, 2009; Zimmermann et al., 2006).

The soil hydrological effect of deforestation and establishment of pasture and agricultural land has been studied to some extent (Alegre & Cassel, 1996; Arevalo et al., 1998; Ghuman et al., 1991; Gijsman & Thomas, 1996; Martinez & Zinck, 2004). Especially pasture use has detrimental impacts on near-surface hydrological properties through treading stress leading to destruction of pores and to soil compaction (Drewry & Paton, 2000; Drewry et al., 2004; Singleton & Addison, 1999). Several studies found a decline of infiltrability, the entrance of water into the soil, after pasture establishment (Alegre & Cassel, 1996; Hanson et al., 2004; Malmer, 1996; Martinez & Zinck, 2004; Ziegler et al., 2004). K_s values have shown to be similarly reduced: In a temperate region, for instance, McDowell et al. (2003) showed a decrease in K_s because of cattle treading. Studies in the humid tropics found lower K_s in soils under pastures than in forest soils in Honduras, Ecuador and Eastern Amazonia (Godsey & Elsenbeer, 2002; Hanson et al., 2004; de Moraes et al., 2006; Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2006).

Yet very few studies have examined the influence of secondary succession on K_s recovery after pasture use: Nie et al. (1997) found increasing K_s values in a pastoral fallow compared to an active pasture in New Zealand. However, a recovery of K_s under secondary succession to pre-pasture levels might take more than one decade (de Moraes et al., 2006; Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2010a). Further studies on a wider range of soil types are still needed in order to clarify how K_s recovery is mediated by both land-use factors and soil-inherent abiotic properties.

In this study we assess the recovery of K_s under secondary succession on former pasture plots on a landscape scale. Conclusions of many of the existing plot-scale studies are difficult to extrapolate onto a larger scale because they generally lack replicates beyond plot or catchment borders. Thus we use four different land cover classes, pasture, secondary forest which is 5–8 years old, secondary forest of 12–15 years of age and secondary forest which is more than 100 years old, and we replicate each land cover class four times in different micro-catchments across the region to avoid pseudoreplication (Hurlbert, 1984). We address the following hypothesis: Decreased saturated hydraulic conductivity on former pasture sites can fully recover under secondary succession to the level of mature forests. We further evaluate hydrological implications of this recovery such as a decreasing possibility of overland flow generation in order to gain a more comprehensive understanding of ecosystem processes under changing land uses in the humid tropics.

2.2 Methods

2.2.1 Study site

Our study was conducted in central Panama in the watersheds of Río Agua Salud and Río Mendoza, which drain into the Panama canal (Fig. 1), partly covering the project area of the Agua Salud Project. The study area is characterised by a strongly dissected pretertiary basalt plateau (330 m amsl) with narrow interfluves, linear slopes averaging 42 % and narrow or no valley floors. Soils in the area vary little with respect to texture (silty clay to clay) due to the uniform parent material, with pH values (in water) ranging from 4.4 to 5.8 (Hall et al., unpublished data).

The climate of the study area is tropical with a distinct dry season from mid-December through April. Total annual rainfall in our research area averages 2300 mm (1998–2007, data from the Panama Canal Authority), the mean daily temperature on nearby Barro Colorado Island (BCI) is 27 °C and varies only little throughout the year (Windsor, 1990).

Land use in the area varies over short spatial and temporal scales. 8.3 % of this area is under active pasture, whereas 28.3 % is covered by young secondary forest (ACP & ANAM, 2006). This category, however, also includes "passive" pasture, i.e. pastures dedicated to a short fallow period followed by renewed clearing and grazing. The stocking density is about one animal per hectare. Old secondary forest covers about 50 % and mature forest 10 % (ACP & ANAM, 2006), most of which is situated within the Soberanía National Park.

2.2.2 Sampling design

In order to assess K_s variability related to secondary succession after pasture abandonment on a landscape scale we applied a space-for-time approach. Abiotic site parameters (geology, climate, soils) in the study area are homogeneous, so the plots are considered comparable regarding their initial conditions. Thus K_s variability within our study area is mainly attributable to land cover effects. We examined the following land cover classes: pasture (P), secondary forest of about 5–8 years of age on former pasture (SF5), secondary forest of about 12–15 years of age on former pasture (SF12), and secondary forest of 100 years or more of age (SF100) which serves as comparison with old forest conditions. These age classes were deduced from information gathered in interviews with the land owners regarding the plots' land-use history. SF5 and SF12 plots were already established as part of a larger study examining secondary forests on 50 transects across the landscape, so land-use history data and vegetation parameters could be used (van Breugel et al., unpublished data).

Stand structure differed clearly between the age classes (Table 2.1). Mean basal area increased from 13.6 $m^2\,ha^{-1}$ in SF5 to 23.5 $m^2\,ha^{-1}$ in SF12, which is close to the 26.3 $m^2\,ha^{-1}$ of nearby old secondary forest in Soberanía National Park (Table 2.1) and the 30.6 $m^2\,ha^{-1}$ of the 50 ha mature forest plot on BCI (Chave et al., 2004). Such a fast recovery of basal area early in succession is not uncommon (van Breugel et al., 2006). Stem density decreased from a mean of 9706 stems ha^{-1} in SF5, to 7990 stems ha^{-1} in SF12 and 3656 stems ha^{-1} in SF100, but differences were not significant due to large variation between sites (Table 2.1). The forest in the study area is classified as tropical moist forest according to the Holdridge life zone system. Studied pasture sites have been subject to cattle grazing for 20–30 years and secondary forest sites for 10–40 years prior to abandonment. Stocking densities on the former pastures were around one head of cattle per hectare which is typical for pasture management in the region.

Table 2.1: Stand structure characteristics of the three forest age classes, for trees with DBH $\geq$ 1 cm.

	Sites[1]	Basal area[2] ($m^2\,ha^{-1}$)	Stem density[2] (ha^{-1})
SF5	4	13.6 $\pm$ 2.3 a	9706 $\pm$ 1807 a
SF12	4	23.5 $\pm$ 1.1 b	7990 $\pm$ 831 a
SF100	2	26.1 / 26.4	3313 / 4000

[1] SF5 and SF12: data from two 0.1 ha plots per site (van Breugel and Hall, unpublished data); SF100: data from two one-ha plots in Soberanía National Park, within 2 km from the study area and very similar to the SF100 sample sites (Chave et al., 2004, Appendix B, plots 19 and 20).
[2] Different letters indicate significant differences between SF5 and SF12 (Student's t-test, $\alpha = 0.05$).

In each of the four land cover classes we sampled four plots, resulting in 16 plots altogether. Their width was constrained by topography and varied between 50 and 100 m; the length comprised the whole slope at the plot location, from watershed to stream. Sampling points were selected according to a two-stage sampling design: step 1 being the random selection of four plots out of a pool containing all possible slopes of the respective land cover class in different micro-catchments within the study area, thus avoiding pseudoreplication (Hurlbert, 1984). Step 2 comprised stratified random sampling within each plot with geographic slope stratification, leading to three strata per plot: upslope, midslope and downslope position. This was done to avoid clustering of sampling locations and to spread samples more evenly across the slope. The downslope stratum was excluded from sampling because of its overlap with gallery forest, a type of land cover not pertinent to our study. In each of the upslope and midslope strata, five sampling points were selected randomly. The locations of the sampled plots can be seen in the map provided in Figure 2.1.

2.2.3 K_s sampling and measurement

Sampling was undertaken in the rainy season from July to August 2009. Undisturbed soil cores with a diameter of 8.9 cm were taken at the ten sampling locations in each plot, in the depths of 0–6 cm and

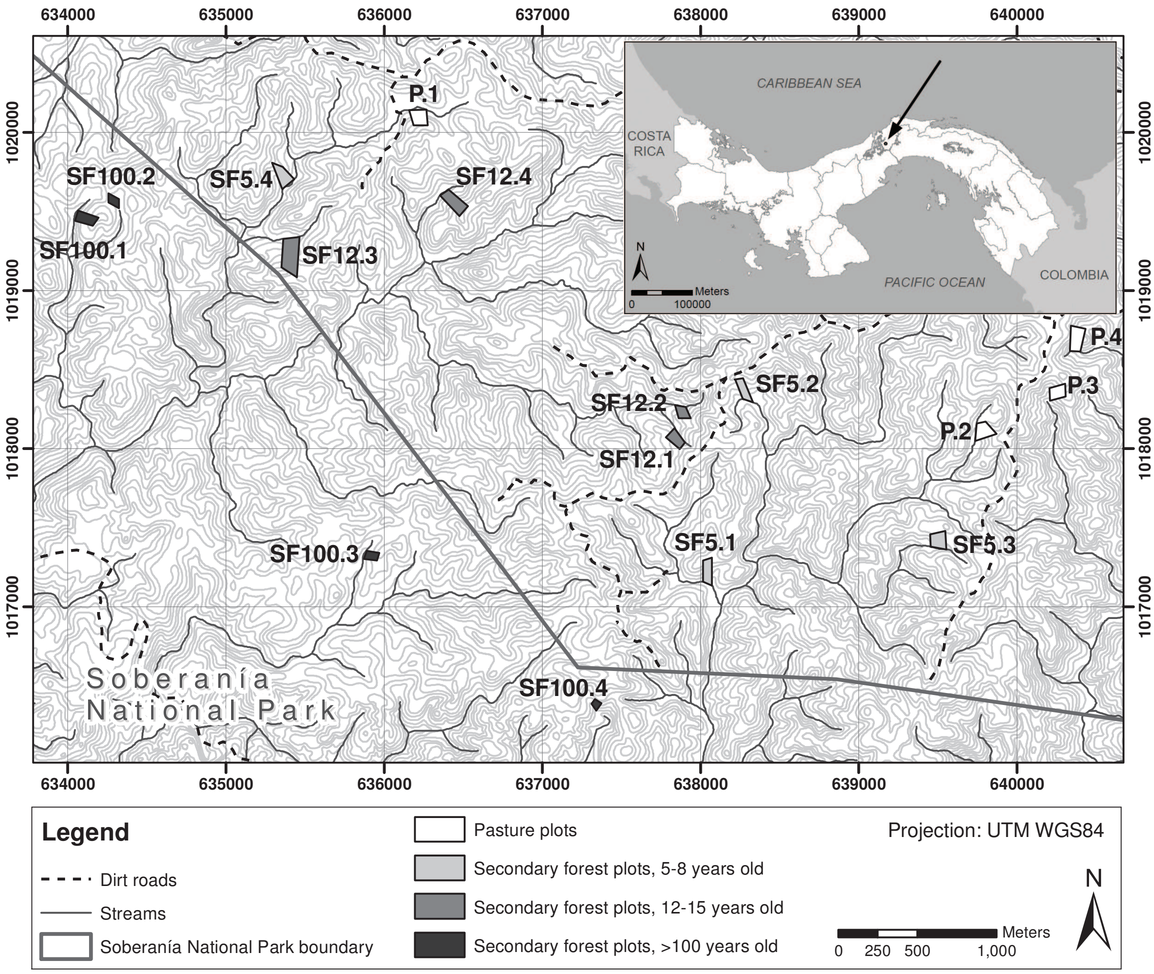

Figure 2.1: Map of the study site in Panama with the location of the sampling plots: P.1–P.4 are pasture plots, SF5.1–SF5.4 are secondary forest plots of 5–8 years of age, SF12.1–SF12.4 are secondary forest plots of 12–15 years of age and SF100.1–SF100.4 are 100 years-old secondary forest plots.

6–12 cm on levelled ground, employing a standard coring device (Soilmoisture Equipment Corporation, USA). We cut the core ends level with a sharp knife and slowly saturated the samples upside down over a time span of 64 h to prevent air entrapment. For measurement of saturated hydraulic conductivity we applied a constant water head and recorded water flow through the cores per time unit, following the methodology described by Reynolds et al. (2002). After a constant flow rate had been established, we took measurements of percolated water volume per time unit and calculated K_s according to Darcy's Equation for saturated conditions:

$$v = -K_s \frac{dh}{ds} \tag{2.1}$$

whereby v is the percolation velocity [m s^{-1}], K_s is the saturated hydraulic conductivity [m s^{-1}], $\frac{dh}{ds}$ is the hydraulic gradient; the percolation velocity can be expressed as $v = \frac{Q}{A}$ with Q being the water flux [m^3 s^{-1}] and A the cross-section the water flows through [m^2].

2.2.4 Ancillary variables

K_s is directly linked to soil macroporosity (Ahuja et al., 1984) which is likely being influenced by land-use changes through changed root channel formation and vegetation community-specific soil faunal associations, but may also partly be affected by soil-inherent factors such as texture and soil structure. We sampled several ancillary variables such as soil bulk density, soil organic carbon content (SOC) and texture in order to include possible factors influencing K_s and to ascertain that our prerequisite of homogenous abiotic conditions holds true, which should be visible in an absence of links between K_s and especially soil texture.

Soil bulk density was measured in each stratum on each plot at only one randomly selected point out of the five sampling points for K_s, both at 0–10 cm and 10–20 cm depth according to the compliant cavity method (Soil Survey Staff, 1999). At the same depths and locations, soil texture composition was determined using the pipette method (Gee & Or, 2002). According to catchment-scale pre-studies within the study area, both parameters are very uniform on this scale, thus two sampling points per plot but four replicates in each land cover class seem appropriate. SOC was measured at randomly selected two of the five sampling locations in each plot stratum at two depths, including the 0–10 cm depth (for further details see Neumann-Cosel et al., 2011). The measurements at 0–10 cm depth of bulk density, sand, silt and clay content and SOC concentrations were used for comparison with median K_s in the respective stratum at both the 0–6 cm and the 6–12 cm sampling depth.

2.2.5 Hydrological relevance

The hydrological relevance of the measured K_s values can be assessed using graphs of the cumulative distribution functions of K_s and of maximum rainfall intensities (Ziegler et al., 2004, 2006; Zimmermann & Elsenbeer, 2009; Zimmermann et al., 2006). Rainfall was continuously recorded with two Hobo® tipping bucket rain gauges (0.2 mm tip resolution) that were located in the northeastern part of our study area. As both devices agreed very well, the data of the gauge which was installed first were used for the purpose of this paper. Local variations in rainfall patterns are considered negligible for the quantification of rainfall intensities. Events were separated by at least 2 h without rain. We determined the maximum 10 min rainfall intensity distribution (I_{10max}) by calculating the maximum precipitation intensity over 10 min intervals for each event. Its cumulative frequency distribution was then contrasted with the cumulative frequency distributions of K_s for each land cover class at both depths. The procedure was repeated with the maximum 30 min rainfall intensities so as to account for less extreme rainfall intensities as well.

2.2.6 Statistical analyses

Due to non-normality of the K_s data that could not be alleviated with common transformations, we used non-parametric tests for statistical inference. Change of K_s with aging secondary forest was tested using the Kruskal-Wallis test (Kruskal & Wallis, 1952). For pairwise post-hoc comparisons we used the Mann-Whitney U-test with Bonferroni correction to account for the multiple comparisons (Holm, 1979; Mann & Whitney, 1947). Differences between K_s in the two soil depths were also examined employing the Mann-Whitney U-test. The importance of the ancillary variables regarding K_s was tested via correlation procedures. We used Kendall's Tau correlation because of the already mentioned non-normality of the K_s data and the small sample size ($n = 8$) for correlation (Kendall & Smith, 1939). Differences were taken to be significant with $p < 0.05$. All analyses were carried out in the language and environment R (R Development Core Team, 2009).

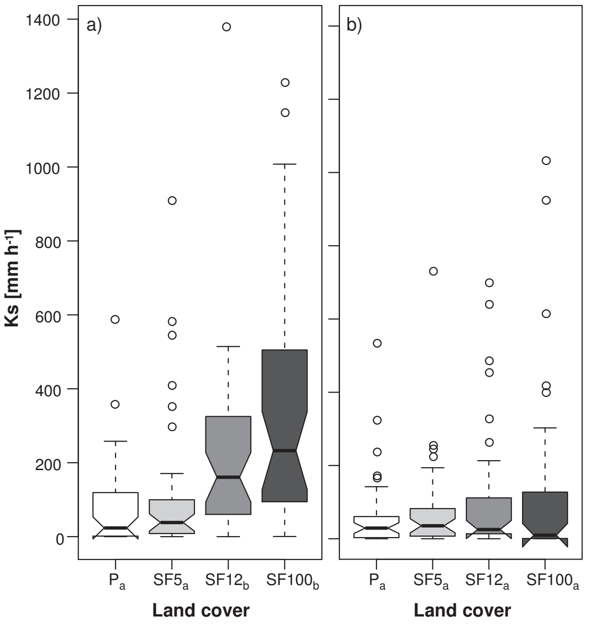

Figure 2.2: K_s dependent on land use at the (a) 0–6 cm depth and the (b) 6–12 cm depth. Land cover classes are abbreviated with P for pasture, SF5 for secondary forest of 5–8 years of age, SF12 for secondary forest of 12–15 years of age and SF100 for 100 years-old secondary forest. Subscript letters visualise the result of the Kruskal-Wallis test: different letters stand for significant differences between the groups ($p < 0.05$).

2.3 Results

Soil hydraulic conductivity values ranged between just above 0 mm h^{-1} up to a maximum of 1381 mm h^{-1} encountered in the SF12 class for the upper depth of 0–6 cm, the overall median being 87 mm h^{-1}. The 6–12 cm depth exhibited lower values: approximately 0 mm h^{-1} up to 1033 mm h^{-1} in the 100 years-old secondary forest, with an overall median of 26 mm h^{-1} (Table 2.2). Comparison of K_s between the two soil depths revealed a significant decrease in K_s from the upper to the lower depth. Slope position showed no significant effect on K_s.

A distinct shift in hydrologic conductivity was detected with age following pasture abandonment (Figure 2.2a): whereas pasture and 5–8 year-old secondary forest had similarly low values of K_s (medians of 23 and 38 mm h^{-1}, respectively, Table 2.2) for the 0–6 cm depth, K_s increased significantly by one order of magnitude between the secondary forests of 5–8 years and of 12–15 years, the latter in turn being similar to 100-year old secondary forest (medians of 161 and 233 mm h^{-1}, respectively, Table 2.2). The 0–6 cm K_s median under a mature tropical lowland forest on nearby Barro Colorado Island, sampled within the scope of a different study with the same method at identical depths (Hassler et al., 2011a), is within the same order of magnitude as the 100-year old forest in this study; therefore we consider SF100 K_s to be similar to K_s before forest conversion to pasture. The K_s values at the 6–12 cm depth did not vary significantly between the land cover classes, indicating a limited influence of land cover change – be it forest conversion to pasture or recovery – to primarily the uppermost soil layers (Figure 2.2b).

High variability in K_s was observed within all land cover classes at the 0–6 cm depth (Figure 2.3).

Table 2.2: Parameters of the distribution functions of K_s for (a) the 0–6 cm depth, (b) the 6–12 cm depth. Land cover classes are abbreviated with P for pasture, SF5 for secondary forest of 5–8 years of age, SF12 for secondary forest of 12–15 years of age and SF100 for 100 years-old secondary forest.

Measure	Unit	P	SF5	SF12	SF100	Total
(a)						
Minimum	[mm h^{-1}]	0	0	0	0	0
1st quartile	[mm h^{-1}]	1	9	60	95	14
Median	[mm h^{-1}]	23	38	161	233	87
Mean	[mm h^{-1}]	80	13	225	349	191
3rd quartile	[mm h^{-1}]	119	97	325	498	288
Maximum	[mm h^{-1}]	588	910	1381	1229	1381
(b)						
Minimum	[mm h^{-1}]	0	0	0	0	0
1st quartile	[mm h^{-1}]	5	7	14	1	4
Median	[mm h^{-1}]	29	35	25	10	26
Mean	[mm h^{-1}]	71	76	115	127	97
3rd quartile	[mm h^{-1}]	60	79	112	127	98
Maximum	[mm h^{-1}]	535	731	700	1033	1033

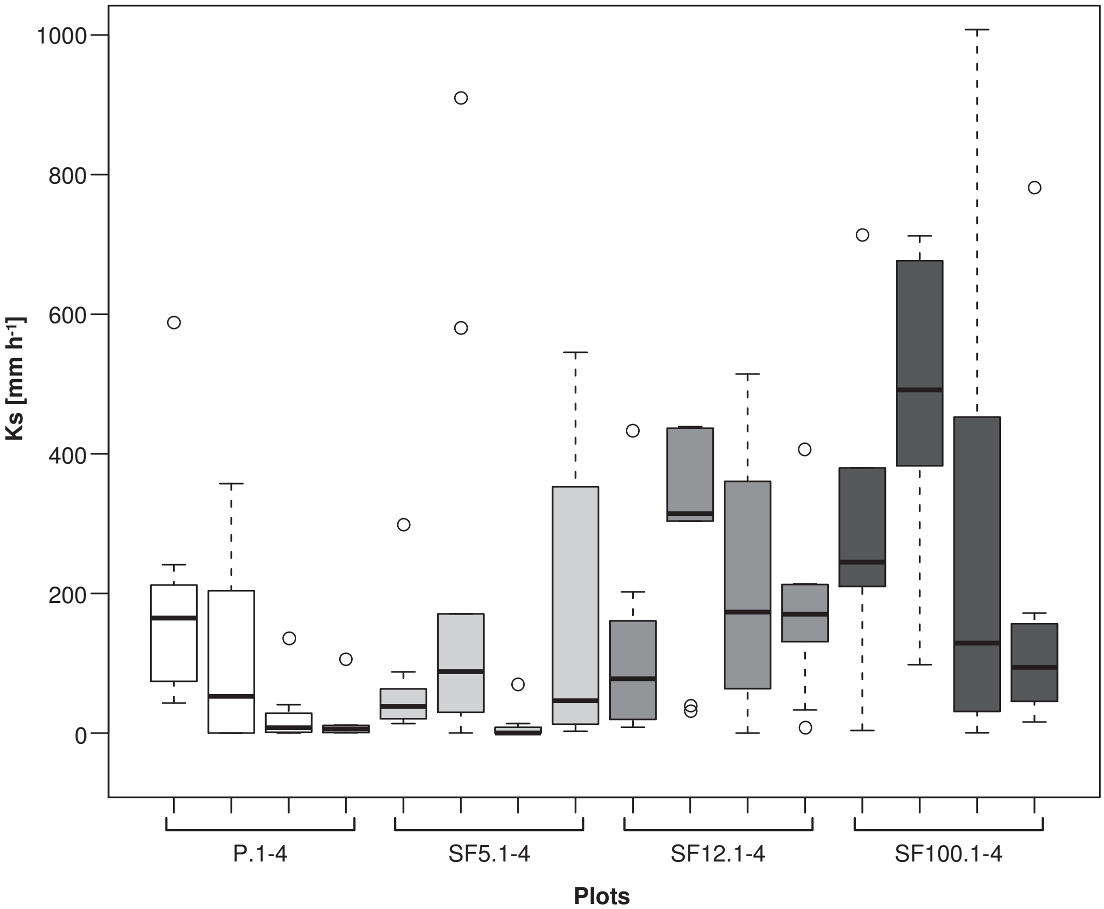

Figure 2.3: K_s on individual plots at the 0–6 cm depth. Land cover classes are abbreviated with P for pasture, SF5 for secondary forest of 5–8 years of age, SF12 for secondary forest of 12–15 years of age and SF100 for 100 years-old secondary forest, subscript numbers refer to plot numbers. Not included in the graph are three outliers (1381 in plot SF12.2, 1229 in plot SF100.1 and 1148 in plot SF100.2) for better view of the differences in box plots.

The 6–12 cm depth K_s also contained variability within the land cover classes, but with much smaller range, therefore it is not further discussed here.

Correlations of either depth's K_s values with the ancillary variables only yielded one significant result, a correlation between the 0–6 cm K_s data and soil bulk density (Table 2.3).

Assessing the hydrological relevance of these results with the help of cumulative distribution functions yielded the clearly visible difference between land uses for the 0–6 cm depth (Figure 2.4).

SF12 and SF100 held considerably higher K_s values than SF5 and P. Comparisons between K_s and the maximum 10 min rainfall intensities ($I_{10\max}$) showed that median K_s of secondary forest both 12–15 years and 100 years old were higher than even the maximum value of $I_{10\max}$. In contrast, the young secondary forest K_s median lay between the 75 % and 95 % quantile of $I_{10\max}$ and the pasture median was approximately the same as the 75 % quantile of $I_{10\max}$. In the 6–12 cm depth, K_s distributions of the land-use classes were very similar and values altogether were very low. Me-

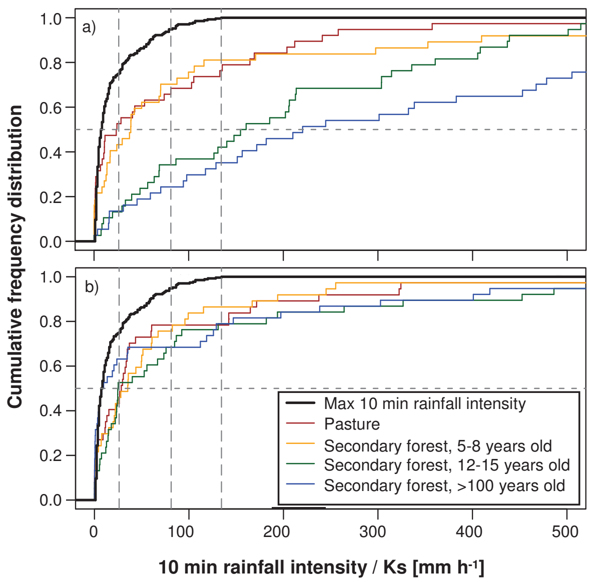

Figure 2.4: Cumulative distribution functions for the maximum 10 min rainfall intensities and (a) K_s at 0–6 cm depth and (b) K_s at 6–12 cm depth. The added (horizontal) short dashed line visualises the median line and the three (vertical) long dashed lines represent quantiles of the cumulative rainfall intensity curve. From left to right: 75 % quantile, 95 % quantile and maximum value.

dians lay around or below the upper quartile of $I_{10\max}$. We also computed the cumulative distribution functions using the maximum 30 min rainfall intensities. The comparisons with the K_s distributions were very similar to the $I_{10\max}$ results.

2.4 Discussion

2.4.1 Differences in saturated hydraulic conductivity among land cover classes

The decrease of K_s with depth has been similarly reported in several studies of soils undergoing land-use change in the humid tropics (Godsey & Elsenbeer, 2002; Malmer, 1996; Ziegler et al., 2004; Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2006). Such a decrease is apt to result in saturation overland flow during high-intensity rainfall events (Bonell & Gilmour, 1978; Germer et al., 2010; Godsey et al., 2004).

Increasing K_s during secondary succession (Figure 2.2a) implies a recovery process from pasture use, a process with a lag time of at least 8 years. Recovery of near-surface hydraulic properties has already been detected under decreasing or abandoned use of formerly intensively grazed pastures (Drewry et al., 2004; Greenwood et al., 1998; Nie et al., 1997). However, few studies have been concerned with the recovery of pasture K_s under secondary succession. In a lowland Amazonian region, Zimmermann et al. (2006) found K_s in 12.5 cm depth to recover after 15 years of pasture abandonment to pre-pasture levels, whereas, in a chronosequence study in the same region no recovery could be detected after 7 years of pasture abandonment (Zimmermann et al., 2010a). A different study in a tropical montane forest region in Ecuador showed that K_s in 12.5 cm depth on a former pasture had not recovered after 10 years, which might be due to a bracken infestation hindering natural secondary succession more inclined towards woody species (Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2010a). However, a nearby pine plantation of 25 years of age showed recovery of K_s after pasture use, hinting at the importance of woody vegetation involvement. Yet a time frame of more than a decade is likely to be necessary for recovery to pre-pasture values (Zimmermann & Elsenbeer, 2008). De Moraes et al. (2006) even concluded a recovery time of many decades after observing K_s values in 5–15 cm depth on a former pasture after 12 years of abandonment, that were still one order of magnitude lower than forest values and indistinguishable from pasture K_s on plinthic soils in Amazonia. Summing up, these studies and ours, covering the diversity of soils in the humid tropics, agree in that K_s recovery in the topsoil is possible but might take more than a decade to become hydrologically relevant.

Reconsidering ecosystem services, secondary

Table 2.3: Mean values of considered ancillary variables and correlations with K_s of the depth 0–6 cm (A) and 6–12 cm (B). Land cover classes are abbreviated with P for pasture, SF for secondary forest of 5–8 years of age, SF12 for secondary forest of 12–15 years of age and SF100 for 100 years-old secondary forest.

Variable	Unit	P	SF5	SF12	SF100	Total	Correlations	
							A	B
Bulk density	[$\mathrm{g\,cm^{-3}}$]	1.00	0.89	0.81	0.75	0.86	$^{***}\tau = -0.6$	n.s.
Sand content	[%]	12.3	9.9	13.4	9.6	11.2	n.s.	n.s.
Silt content	[%]	34.0	34.9	32.4	33.4	34.0	n.s.	n.s.
Clay content	[%]	53.6	55.4	54.1	57.0	54.8	n.s.	n.s.
SOC conc.	[%]	3.5	3.7	3.7	4.6	3.9	n.s.	n.s.

*** $p < 0.001$ for the correlation coefficient Kendall's τ.
n.s. symbolises not significant ($n = 8$).

forests older than 12 years seem to fulfil their role in the hydrological cycle similar to old, undisturbed forests as far as near-surface hydrology is concerned. However, with respect to trade-offs in ecosystem services, it is noteworthy that while K_s had recovered after 12 years of secondary succession, soil organic carbon, for instance, still showed pasture-level values within the same time frame (examined in a similar study on partly the same plots as ours, Neumann-Cosel et al., 2011).

The limitation of land-use influence on K_s to only the uppermost soil layer (Figure 2.2) has been observed by other authors as well: In lowland lowland Amazonia, land use was considered irrelevant for K_s values in depths below 20 cm (Godsey & Elsenbeer, 2002; Zimmermann et al., 2006), in Ecuador, K_s differences between pasture, secondary forest and forest vanished in 50 cm soil depth (Zimmermann & Elsenbeer, 2008). These studies suggest that land use-dependent factors directly influencing K_s occur close to the soil surface. These factors include root formation and soil faunal abundance and activity, which increase macroporosity and hence K_s (Ahuja et al., 1984). However, in our study the layer influenced by land cover is particularly thin, comprising only the upper 6 cm of the soil. The clay mineralogy of the soils in our study area may contribute to this apparently small effect of biotic activity on macroporosity. Swelling clay minerals – whose presence we inferred from soil cracks in the dry season and which do occur on similar soils on BCI – are apt to destroy any biogenic macroporosity at the onset of the wet season. Additionally, the high frequency of overland flow (Godsey et al., 2004) might hinder the establishment of a thick, structurally more stable topsoil horizon.

2.4.2 Variability within land cover classes

Although the robustness of the sampling design, i.e. the inclusion of a sufficient number of replicates, ensured the detection of significant effects of land cover change on K_s at the upper soil depth (Figure 2.2a), the within-class variability emphasizes the importance of local factors influencing K_s (Figure 2.3). For instance, it could be argued that the large variability within pasture plots as well as within the two secondary forest classes might result from differences in the duration of (actual or antecedent) pasture use. However, several studies found a rapid decrease of K_s (and infiltrability) directly after pasture establishment (Alegre & Cassel, 1996; Martinez & Zinck, 2004; Zimmermann et al., 2006, 2010a), thus pasture duration as main reason for within-class variability for pastures and secondary forests appears rather unlikely, as our pasture sites had been actively used for 10–

40 years prior to abandonment. A perhaps more plausible explanation relates to the fact that "pastures" in the study area exhibit large differences in the intensity of cattle grazing due to rotational use and short-term fallowing. Some of the variability could thus stem from this mosaic-like land-use pattern.

For the SF100 class, steepness of sample slopes might have caused variability in the data. Even in mature forest ecosystems in the tropics, overland flow results in erosion (Sidle et al., 2006; Zimmermann et al., 2010a). The soil surface loses its litter cover as the rainy season proceeds, and the understory is patchy at best in mature forests, both of which contribute to the soil surface being exposed to raindrops dripping from the canopy and to the full erosive impact of overland flow. Thus, especially in steep slope situations, a poorly developed A horizon would result, functionally, in a "subsoil" sample, even when taken at 0–6 cm depth.

Besides, the sampling protocol of levelling the ground before sampling could additionally favour subsoil samples in steep slope positions in all land cover classes. As a consequence, sampling in steeper plots might in some cases underestimate K_s values. However, comparison of slope angles from a digital elevation model with K_s values in the topsoil for all land cover classes did not show a consistent relation in this regard.

Therefore K_s values might vary from slope to slope due to a multitude of different reasons. However, by sampling sufficient true replicates we account for this variability within the land cover classes.

2.4.3 Correlations with ancillary variables

Bulk density is closely related to soil macroporosity which itself is a main determinant for K_s (Ahuja et al., 1984). Consequently, the drivers exerting the influence of land-use change on K_s are affecting bulk density similarly, examples being root channel abundance or activity of soil macrofauna, visible in the correlation between the two. A prerequisite of our sampling design to analyse changes of K_s due to land cover was a uniform study site with respect to abiotic parameters, with texture values not likely to be responsible for K_s variability. The absence of correlations between K_s and sand, silt or clay content supports this notion (Table 2.3). Further, although SOC could be suspected to influence K_s as it is linked to vegetation cover and thus land use, SOC concentrations did not prove to be an important direct determinant for K_s.

2.4.4 Hydrological relevance

The difference in the cumulative distribution functions of K_s between the land cover classes at 0–6 cm depth implies that for SF100 and SF12, the incoming rainfall can most likely infiltrate into the soil and percolate to deeper soil layers, their medians being higher than the maximum rainfall intensity values. In contrast, for P, 25 % of the recorded rainfall events and for SF5, between 5 and 25 % of the rainfall events have the potential to generate short-term overland flow as their maximum intensities over 10 min time intervals exceed the spatial median of K_s values (Figure 2.4).

The lower K_s values in the 6–12 cm depth may result in short-term overland flow generation by 25 % or more of the recorded rainfall events, using the rainfall intensities at the soil surface for these comparisons, assuming intensities would be transferred unimpeded though the upper 6 cm of soil.

The overall implications of comparisons with the 30 min maximum rainfall intensities were very similar to those of the $I_{10\max}$-K_s comparisons: at the upper depth, SF100 and SF12 are not likely to be prone to overland flow whereas P and SF5 are. For the 6–12 cm soil depth, all land cover classes could potentially generate overland flow.

Overland flow generation as a consequence of anisotropic K_s values and of K_s changes following land-use shifts in the humid tropics has been reported in the literature (Chandler, 2006; Germer et al., 2010; Hanson et al., 2004; de Moraes et al., 2006; Ziegler et al., 2004; Zimmermann et al., 2006). Related to overland flow are processes such as soil erosion, nutrient leaching and discharge fluctuations, therefore estimations of the overland flow volumes likely to be expected with land-use changes are necessary for understanding ecosystem processes and designing sustainable management options. As we did not measure overland flow in our study, we can only give rough estimates of potentially fulfilled conditions for overland flow generation. However, several factors are complicating these estimations: firstly, rainfall intensities will be redistributed in the vegetation layer before entering the soil, so actual values acting on the soil are likely to be different. Secondly, we only compare $I_{10\max}$ and $I_{30\max}$ to our K_s values, not accounting for other factors crucial for the generation of overland flow besides sufficiently high rainfall intensities. These could be, for instance, antecedent moisture conditions of the soil and longer-duration rainfall intensities determining possible slow saturation of the soil above an impeding layer (Germer et al., 2010; Mulholland et al., 1990). And thirdly, we use only spatial medians of K_s in the comparisons to the rainfall intensities although K_s is spatially very variable over short distances (Germer et al., 2010; Mulholland et al., 1990; Sobieraj et al., 2004; Zimmermann & Elsenbeer, 2008) and thus the mosaic of K_s values on a slope can buffer single locations prone to overland flow generation. Nevertheless, comparisons of K_s with rainfall intensities have been employed in several studies in the humid tropics (de Moraes et al., 2006; Ziegler et al., 2004, 2006; Zimmermann & Elsenbeer, 2009; Zimmermann et al., 2006), and as we did observe overland flow in the study sites on pasture and 5–8-year old secondary forest plots during short high-intensity rains in our field period, the method seems to be justified as a rough estimate of the possibility of overland flow occurrence.

On a catchment scale, partially deforested areas were inferred to being more prone to overland flow than fully forested areas by Kinner & Stallard (1999) and Ibanez et al. (2002) in a comparative study of hydrograph characteristics.

For our study, particularly the low K_s values already at the 6–12 cm depth might induce overland flow generation, regardless of land cover. Germer et al. (2010) for instance report considerable overland flow from pasture sites with the impeding layer being situated at 20 cm soil depth. Elucidating the link between K_s and overland flow requires further studies with sufficient measurements of both parameters, ultimately resulting in appropriate modelling approaches.

2.5 Conclusions

Saturated hydraulic conductivity did recover under secondary succession after pasture abandonment to a level similar to old forest. However, K_s recovery required more than 8 years to reach a pre-pasture level. Moreover, effects of secondary succession were restricted to the upper sampling depth of 0–6 cm, the significantly lower K_s at 6–12 cm soil depth did not vary among land cover classes. Our findings are in accordance with other studies in the humid tropics. The applied space-for-time sampling design including sufficient true replicates supports the notion that the observed K_s recovery is not a local phenomenon but can be occurring anywhere across this landscape, distinguishing the recovery effect from variability occurring on different slopes within a land-use class. Hydrological implications are potential generation of overland flow during high-intensity rainfall events, as the decrease of K_s with depth leads to an impeding soil layer. Already situated 6–12 cm below the soil surface irrespective of land cover, this layer possibly results in the formation of perched water tables. Considering K_s in the upper soil depth, overland flow would be more likely on pasture and younger secondary forest sites compared to older secondary forest and forest. In order to comprehensibly clarify the relation between K_s recovery and land use with its subsequent implications for hydrological pathways, ecosystem services and reforestation activities, further research should include variable and detailed land-use history and, more importantly, combined measurements of K_s, overland flow, precipitation and throughfall.

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Chapter 3

Exploring the variation in soil saturated hydraulic conductivity under a tropical rainforest using the wavelet transform *

Abstract

Saturated hydraulic conductivity (K_s) of the soil is a key variable in the water cycle. For the humid tropics, information about spatial scales of K_s and their relation to soil types deduced from soil map units is of interest, as soil maps are often the only available data source for modelling. We examined the influence of soil map units on the mean and variation in K_s along a transect in a tropical rainforest using undisturbed soil cores at 0–6 and 6–12 cm depth. The K_s means were estimated with a linear mixed model fitted by residual maximum likelihood (REML), and the spatial variation in K_s was investigated with the maximum overlap discrete wavelet packet transform (MODWPT). The mean values of K_s did not differ between soil map units. The best wavelet packet basis for K_s at 0–6 cm showed stationarity at high frequencies, suggesting uniform small-scale influences such as bioturbation. There were substantial contributions to wavelet packet variance over the range of spatial frequencies and a pronounced low frequency peak corresponding approximately to the scale of soil map units. However, in the relevant frequency intervals no significant changes in wavelet packet variance were detected. We conclude that near-surface K_s is not dominated by static, soil-inherent properties for the examined range of soils. Several indicators from the wavelet packet analysis hint at the more dominant dynamic influence of biotic processes, which should be kept in mind when modelling soil hydraulic properties on the basis of soil maps.

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3.1 Introduction

Soil saturated hydraulic conductivity (K_s) is a key variable in the water cycle. It co-determines how rainfall is partitioned into different water flow paths, such as groundwater recharge, lateral subsurface runoff and overland flow. Hence the estimation of K_s is crucial for hydrological modelling and for land management decisions, especially in regions with an intense water cycle such as the humid tropics.

Ample research has been carried out in order to predict K_s from readily available bulk soil data using the so-called pedotransfer functions (PTFs) for temperate soils. The PTFs that incorporate additional information on soil structure or pore distribution have proved more successful than those based solely on basic properties found in soil survey reports, such as particle size, bulk density and organic carbon content (Wösten et al., 2001), because of the direct link between K_s and effective porosity (Ahuja et al., 1984). Soil structure, and consequently soil porosity and K_s, is influenced by both static factors such as particle size distribution and clay mineralogy and dynamic factors such as organic carbon content and biotic activity, which are often linked to land use. Climatic conditions also influence soil structure (Lilly et al., 2008). Therefore, PTFs that predict K_s should not be transferred to 'soilscapes' outside the range of soil-forming factors for which they were designed (van den Berg et al., 1997; Wösten et al., 2001). Additionally, factors influencing K_s are typically scale-dependent, for instance, local biotic activity might lead to small-scale variability in K_s (McKenzie & Jacquier, 1997; Si & Zeleke, 2005). Hence, an assessment of the factors influencing K_s at different scales is essential for successful modelling of hydraulic processes of a given region.

For the humid tropics, information on soil hydraulic parameters is scarce. Tropical soils, especially under forest vegetation, are under-represented in soil databases (Hodnett & Tomasella, 2002) and in many regions only geological or soil maps with low spatial resolution (small cartographic scale) are available as a basis for estimating those parameters. Studies on the effect of land-use change on K_s have documented both the decrease of K_s after pasture establishment on former rainforest plots and the recovery of K_s after pasture abandonment (Martinez & Zinck, 2004; Zimmermann et al., 2010b). The observed changes result from an alteration in soil structure following land-use change. In contrast, the influence of static soil properties on soil structure, and thus on K_s, is poorly understood. In the Amazon region, Sobieraj et al. (2002, 2001) could not predict K_s with PTFs based on particle size, bulk density and saturated moisture content and found no influence of texture on mean K_s in a catena study. Clay content and mineralogy, however, affect K_s significantly because they influence soil structure (Alekseeva, 2007; Senarath et al., 2010; West et al., 2008).

In our study, the primary goal was to explore the relationship between K_s and soil map units in an area of uniform land cover, in this case old-growth tropical rainforest. We hypothesized that soil variability as expressed in soil map units would affect the observed variation in K_s. On Barro Colorado Island in Panama we measured K_s on soil cores taken from two depths (0–6 and 6–12 cm) at regular intervals on a transect. Using a linear mixed model, fitted by residual maximum likelihood (REML), we estimated the spatial means of K_s for the different soil map units to see if there were any significant differences. We applied a wavelet packet analysis, as outlined in the following section, to look for significant changes in the variation of K_s at various spatial frequencies and compared these with the map units. Wavelet analysis has been successfully used to investigate scale- and frequency-dependent variation and correlation of soil variables (Biswas et al., 2008; Lark, 2006), but its application to K_s data is limited (Si & Zeleke, 2005). In addition to these analyses, we computed wavelet correlations between K_s and elevation, as topography is a crude proxy for the geological influence on the island, independent from soil map units.

3.2 Theory

Wavelet transforms have proved to be a useful tool in soil science. They can help to identify important scales (or spatial frequencies) of variation for soil properties measured along a transect, as well as changes in variation that might be associated with changes in their controlling factors. Wavelet analysis requires that the variables have a finite variance and be sampled at regular intervals, but no further preconditions such as stationarity of variance are necessary. This is especially useful for the examination of soil characteristics because they are formed by a multitude of different processes operating at different spatial frequencies in complex ways. Therefore, assumptions such as an underlying stationary random variable are often not plausible (Lark & Webster, 1999). We do not describe the wavelet transformation in detail here, but refer the reader to the existing literature (Lark, 2006; Lark & Webster, 1999; Milne et al., 2009; Percival & Walden, 2000). In short, a wavelet is a function that oscillates over a compact support: a short interval in space over which it takes non-zero values. In the discrete wavelet transform (DWT) a 'mother' wavelet function is dilated by a scale parameter, commonly 2^j where j is an integer. Dilation changes the frequency of the wavelet's oscillations and the width of its support. The wavelet can also be translated in order to change its location. A set of dilations and translations of a mother wavelet provides a basis for a set of observations. That is to say, it comprises a set of mathematical building blocks from which observations can be represented by combining the wavelet functions with a set of coefficients. The values for the coefficients at different dilations of the wavelet show how the variation in the data depends on the spatial scale, and the values at different translations show how the variability changes in space.

If our data are a series $z(x)$, where $x = 1, ..., N$, sampled at intervals of x_0, then the largest spatial frequency that can be investigated is $\frac{1}{2}x_0$. In the DWT the frequency interval $[0, \frac{1}{2}x_0]$ is partitioned into predetermined intervals, which correspond to the successive dilations of the mother wavelet. In an alternative procedure, the discrete wavelet packet transform (DWPT), we derive from the mother wavelet a series of wavelet packet bases. These divide the interval $[0, \frac{1}{2}x_0]$ into a series of equal intervals (see Figure 3.1 for an example). In this instance the use of wavelet packet bases can divide the frequency interval into 2, 4, ... or 64 equal intervals. A complete basis for the data can be provided by the wavelet packets that correspond to any non-overlapping set of these packets that cover the whole interval. Two such bases are shown by the shaded grey cells in Figure 3.1. In this paper, we used the Maximal Overlap DWPT (MODWPT) which, as Percival & Walden (2000) explain, is preferable for statistical inference from data.

As explained by Percival & Walden (2000), the increase in resolution in the frequency domain when wavelet packet bases that divide the frequency range $[0, \frac{1}{2}x_0]$ into narrower intervals are used incurs a cost in spatial resolution because the support of the corresponding wavelet function becomes less compact. The best basis for a particular dataset will therefore be one that achieves good frequency resolution at those frequencies where the data appear to arise from a stationary process (and so there are no singular local features to resolve in space), and good spatial resolution where there are local transient features. Lark (2007) showed this formally. The best basis can be found as the one that minimizes some information criterion.

The MODWPT is invertible, that is to say, given a set of coefficients, it is possible to reconstruct the original signal. More importantly, it is also possible to construct components of the signal that correspond to each packet of a wavelet packet basis (Percival & Walden, 2000). This is a useful way

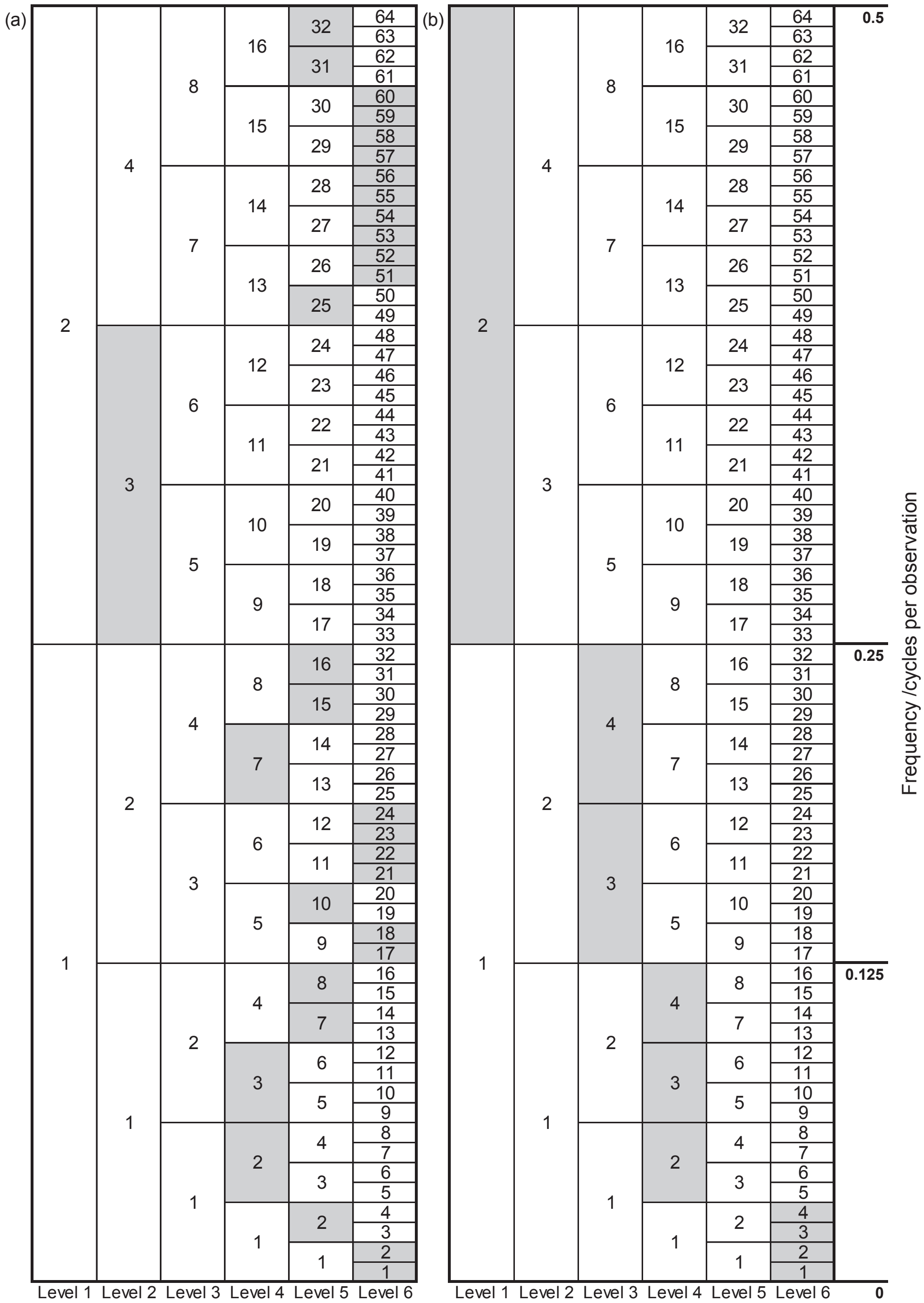

Figure 3.1: The best basis of the square-root transformed K_s data for (a) the 0–6 cm and (b) the 6–12 cm depths. The nominal partition of the frequency interval by the MODWPT to level 6 is shown with the best bases indicated by the shading. Packet $P_{k,i}$ is identified by packet number given in the cell and level number at the bottom. The frequencies increase from bottom to top.

of breaking down the signal so that frequency-specific behaviours can be seen, and is known as a multi-resolution analysis (MRA). The original signal is given by the sum of the components.

The MODWPT coefficients can be used to estimate the variance contribution associated with each packet or frequency interval defined by the MODWPT basis. This is known as the wavelet packet variance. By analogy, for two variables the coefficients can be used to estimate frequency-specific wavelet packet covariation and correlation: the formulae for this are described by Percival & Walden (2000) and Milne et al. (2009). Convention dictates that wavelet packet variances and correlations are discussed in terms of their respective frequency intervals. Spatial frequency is not always an intuitive concept. Therefore, throughout our analysis we describe our results in terms of frequency interval and, where helpful, interval of periods (the reciprocal of frequency).

Following Whitcher et al. (2000), Milne et al. (2009) described a method to detect significant changes in wavelet packet variance across the data series, and a similar method can be used to detect significant changes in wavelet packet correlation (Milne et al., 2009).

We used a MODWPT based on Daubechies's extremal phase wavelet filter with two vanishing moments (Daubechies, 1992). We selected this wavelet because it takes non-zero values over a narrow interval, $L = 4$, and so is particularly suitable for identifying localized features in the data. We found the best basis for our data using the cost function proposed by Constantine & Reinhall (2001), which is appropriate for the MODWPT. We followed the procedure proposed by Milne et al. (2009) to deal with the overlap of the wavelet function with the ends of the data series, by extending the data by symmetrical reflection and then discarding the coefficients that are significantly biased as a result.

3.3 Methods and analysis

3.3.1 Study site

The study was carried out on Barro Colorado Island (BCI), a 1500-ha island located in the Panama Canal (9° 09′ N, 79° 51′ W, 40–171 m amsl). Mean annual precipitation amounts to approximately 2600 mm, mean annual temperature is 27 °C, and the vegetation cover consists of old-growth lowland moist tropical forest (Leigh, 1999).

Soil variability on BCI derives primarily from lithology and secondarily from topography. The Bohio formation dates back to the early Oligocene and is located in the northwest and stretches through the centre of the island. The Caimito formation dates back to the late Oligocene and has a volcanic facies in the east and a marine facies mainly in the west of the island. The central plateau is formed by an andesite flow and forms the topographic high ground. These formations gave rise to 13 mapped soil types (Figure 3.2), which differ with respect to texture, clay mineralogy, depth and stoniness. The soil type classification of BCI is more detailed than Soil Taxonomy (Soil Survey Staff, 2006) or World Reference Base (IUSS Working Group WRB, 2006) because Baillie et al. (2007) took into account local peculiarities and pedologically important differences at the map scale and thus created a BCI-specific classification system, which we used as a basis for our soil type distinction. The map is semi-detailed with an approximate scale of 1:15000 and approximately one field observation per 2 ha (Baillie et al., 2007). The soil types on Barro Colorado Island fall well within the range of soils reported in other studies in Central Panama and according to Turner & Engelbrecht (2011) are likely to be representative for most soils in the Panama Canal zone.

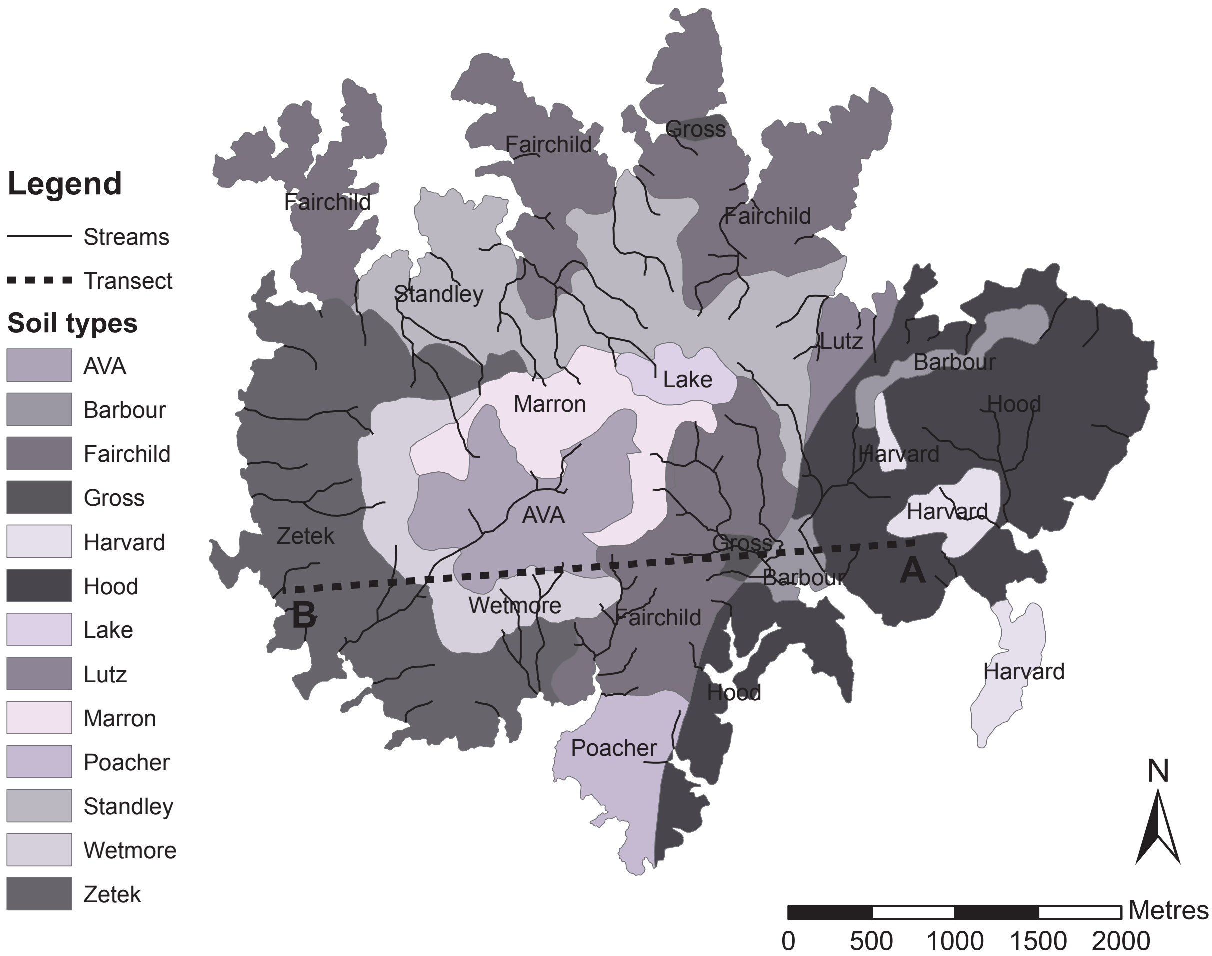

Figure 3.2: Map of the study site. The black line indicates the position of the transect (A–B), which runs approximately east-west.

3.3.2 Field and laboratory methods

We took undisturbed soil cores at depths of 0–6 and 6–12 cm at 256 locations along a straight transect stretching roughly in an east–west direction across the island (Figure 3.2). The transect was 3825-m long with a spacing of 15 m between the sampling locations. We sampled the most important soil types on the island so that they covered the different geological units, involved a large proportion of the Island's area and spanned a range of contrasting soil characteristics. These soil types comprise Ultisols, Inceptisols and Oxisols (Soil Survey Staff, 2006), or Cambisols, Acrisols, Gleysols and Ferralsols (IUSS Working Group WRB, 2006). Table 3.1 summarizes pertinent properties of the selected soil types (Baillie et al., 2007). The soil type for each sampling location was later deduced from intersecting the soil map with the transect points. The diameter of the soil cores was 8.9 cm and they were extracted with a standard core sampler on levelled ground.

In the laboratory we cut the core ends level with a sharp knife and slowly saturated the cores upside-down over 64 hours to prevent air entrapment. We measured K_s according to the constant-

Table 3.1: Characteristics of the soil types, representative for soil map units, on BCI that we examined in this study (after Baillie et al. (2007), and partly adapted from Grimm et al. (2008)).

	Soil types						
Characteristics	Hood (H)	Barbour (B)	Gross (G)	Fairchild (F)	Ava (A)	Wetmore (W)	Zetek (Z)
Geology	Caimito volcanic	Caimito volcanic	Bohio	Bohio	Andesite	Caimito marine	Caimito marine
Mineralogy	Kaol>mont>gibbs	Mont>kaol>(gibbs)	Mont>kaol>(gibbs)	Kaol>mont>gibbs	Kaol>gibbs	Kaol>mont>gibbs	Mont>kaol>(gibbs)
Mean depth[a] [m]	T: <0.05 S: <0.50	T: <0.05 S: >2.00	T: <0.15 S: <0.60	T: <0.04 S: <0.60	T: <0.04 S: >2.00	T: <0.10 S: <1.00	T: <0.05 S: >2.00
Mean texture[a,b]	cL, sicL cL, sicL	sicL C, siC	sicL C, siC	sicL siC	sicL siC, C	cL, sicL sicL	siC C, siC
Drainage	Freely drained	Poorly drained	Poorly drained	Freely drained	Mostly freely drained	Freely drained	Poorly drained
Stoniness	High, boulders	Low	Low	Low	Low	High, saprolite	Low
Structure topsoil	Med sub bl	Med sub bl	NA	Med sub bl	Med-fine sub bl	Med sub bl	Fine sub bl
Structure subsoil	Fine sub bl	Med sub bl	NA	Med and coarse sub bl	Med sub bl	Med sub bl	Fine sub bl
Soil taxonomy[c]	Lithic and Typic Eutrudept	Oxyaquic Paleudult and Typic Endoaquult	Oxyaquic Paleudult and Typic Endoaquult	Lithic and Typic Eutrudept	Typic Eutrudox, Hapludox and Kandiudox	Lithic and Typic Eutrudept	Oxyaquic Paleudult and Typic Endoaquult
World reference base[a,d]	Leptic and Eutric Cambisol	Vertic Luvisol and Acrisol and Vertic/Eutric/Alumic Gleysol	Vertic Luvisol and Acrisol and Vertic/Eutric/Alumic Gleysol	Leptic and Eutric Cambisol	Hypereutric and Haplic Ferralsol	Leptic and Eutric Cambisol	Vertic Luvisol and Acrisol and Vertic/Eutric/Alumic Gleysol
BCI area [%]	19.8	2.4	0.6	24.7	5.7	4.9	16.3

[a] After Grimm et al. (2008).
[b] FAO (2006).
[c] Soil Survey Staff (2006).
[d] IUSS Working Group WRB (2006).
Abbreviations are as follows. Geology: gibbs = gibbsite; kaol = kaolinite; mont = montmorillonite. Structure: med = medium; T = topsoil; S = subsoil; NA = not available; sub bl = subangular blocky.

head method (Reynolds et al., 2002). The values of K_s for the two sampling depths 0–6 and 6–12 cm along the transect and the borders of the sampled soil map units are shown in Figure 3.3 (a,b).

3.3.3 Data analysis

3.3.3.1 Spatial means of K_s for the different soil map units.

Spatial means of K_s for the different soil map units were estimated by residual maximum likelihood (REML). Our systematic sampling design required this model-based approach. Following Lark & Cullis (2004), we fitted a linear mixed model to the data with REML including the random variation as an autocorrelated random process. Fixed effects of the model were the spatial means of K_s for the soil map units. The model may be written as

$$\mathbf{y} = \mathbf{X}\boldsymbol{\beta} + \mathbf{u} + \boldsymbol{\epsilon}, \tag{3.1}$$

where $\mathbf{y}$ is an $n \times 1$ vector of observations of the variable of interest, $\mathbf{X}$ is an $n \times p$ design matrix that associates each of our n observations with one of the p levels of the fixed effects model (here a particular soil map unit), $\boldsymbol{\beta}$ is a $p \times 1$ vector of fixed effects coefficients (here mean values for the soil map units), $\mathbf{u}$ is a spatially correlated random variable and $\boldsymbol{\epsilon}$ is an independently and identically distributed (iid) random variable. These last two random components both have a mean of zero, and each has a finite variance. The variance of $\boldsymbol{\epsilon}$ is the nugget variance in geostatistical terminology, and the variance of $\mathbf{u}$ is the spatially correlated variance. Additional parameters describe the spatial correlation of $\mathbf{u}$. In this study we considered spherical and exponential covariance functions for $\mathbf{u}$ (Lark & Cullis, 2004), in which an additional distance parameter describes the spatial dependence. The variance parameters (the variances of $\boldsymbol{\epsilon}$ and $\mathbf{u}$ and the distance parameter for $\mathbf{u}$) were estimated by simulated annealing to find values that maximized the residual likelihood. Generalized least squares estimates of the map unit means (elements of $\boldsymbol{\beta}$) were then obtained (Lark & Cullis, 2004). To justify the assumption that variables $\boldsymbol{\epsilon}$ and $\mathbf{u}$ are Gaussian, the data were transformed to square roots before analysis. The Wald statistic was used to test for significant differences in the spatial means of K_s among the soil map units (Lark & Cullis, 2004).

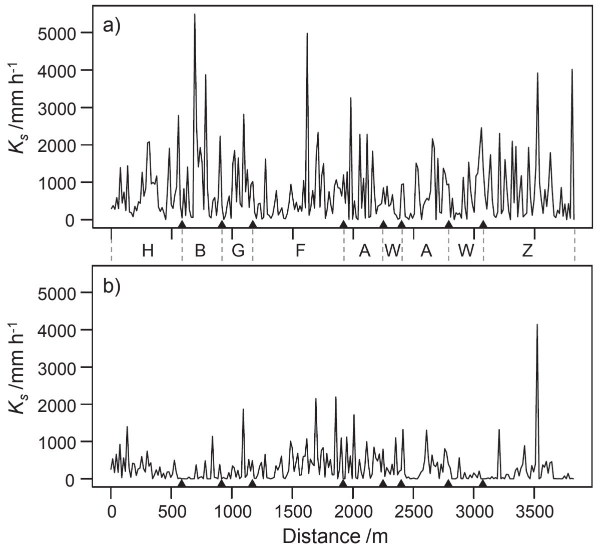

Figure 3.3: K_s data for (a) 0–6 cm and (b) 6–12 cm depths on the transect. The triangles on the x-axes symbolize soil map unit limits between the units H (Hood), B (Barbour), G (Gross), F (Fairchild), A (AVA), W (Wetmore) and Z (Zetek).

3.3.3.2 Variation in K_s along the transect.

The variation in K_s along the transect and the influence of soil type deduced from soil map units were assessed by wavelet packet analysis. As the variance and correlation change detection algorithms assume normality, as does REML, the square-root transformed data were also used in this analysis. We used the MODWPT (Lark, 2006; Milne et al., 2009) to analyse the variation in K_s. The best bases were estimated for both depths of K_s and these were used to assess the plausibility of the stationarity assumption. We estimated the wavelet packet variances for K_s at the two different sampling depths and for elevation, and looked for significant changes in K_s wavelet packet variance. The changes were then compared with map unit boundaries and with the elevation profile along the transect taken from a digital elevation model with a 10-m resolution. We also calculated the wavelet packet correlations between elevation and both sampling depths of K_s and looked for significant changes in correlation with the help of the change detection algorithm mentioned in the theory section; we included elevation in our analysis as an additional control because of its correlation with lithology, as pointed out earlier.

3.4 Results and discussion

3.4.1 Spatial means of K_s

Fitting the linear mixed model to the data with REML yielded the model parameters displayed in Table 3.2. The map unit means of the square-root transformed data were transformed back to the original scales of measurement for better interpretation. Because the transformation is non-linear, these back-transformed estimates are not estimates of the map unit means on the original scale but are best regarded as estimates of the map unit medians (given the assumption of normality after transformation) (Pawlowsky-Glahn & Olea, 2004). These vary between 416.9 $\mathrm{mm\,h^{-1}}$ (Wetmore) and 788.2 $\mathrm{mm\,h^{-1}}$ (Barbour) for the 0–6 cm depth and between 77.4 $\mathrm{mm\,h^{-1}}$ (Zetek) and 283.3 $\mathrm{mm\,h^{-1}}$ (Fairchild) for the 6–12 cm depth. However, neither the upper- nor the lower-depth K_s spatial means showed significant differences between the soil map units according to the Wald statistic (P-values of 0.813 and 0.331 for the 0–6 and 6–12 cm depths, respectively). Thus, on this transect, differences between soil types as expressed by the soil map units do not account for variations of K_s.

The small spatial dependence in the mixed model (0.08 and 0.20 for the 0–6 and the 6–12 cm depths, respectively) can be interpreted in two ways. Firstly, we consider small-scale factors affecting K_s. On the one hand, these include abiotic ones such as shrinking and swelling and short-range texture differences, and on the other hand, biotic ones such as root distribution and bioturbation, which would be more pronounced in the upper sampling depth. Secondly, measurement errors cannot be ruled out, or the sample volume could be too small to capture the representative elementary volume for clayey soils (Bouma, 1983; Lauren et al., 1988). The latter effect might result in an artificially small-scale variability in K_s, for example because a dead-end macropore in situ might form an open channel for rapid water flow in a sample core that partly intersects it. However, a sample support as large as that suggested by the above-mentioned authors would not permit a distinction between the, in many places very shallow, topsoils (<5 cm, see Table 3.1) and the subsequent sampling depth.

Table 3.2: Results for K_s values from the linear models fitted by REML for the 0–6 and 6–12 cm depths. The data were square-root transformed before the analysis. The map unit means were back-transformed to the original scales of measurement (so are best regarded as estimates of the map unit medians); the variance was multiplied by 100 for rescaling.

Variable	K_s (0–6 cm)	K_s (6–12 cm)
Back-transformed fixed effects coefficients (equivalent to map unit medians) $[\mathrm{mm\,h^{-1}}]$		
Hood	637.6	243.0
Barbour	788.2	79.5
Gross	724.0	124.7
Fairchild	440.4	283.3
Ava	530.6	211.0
Wetmore	416.9	130.1
Zetek	480.3	77.4
Variance parameters:		
Variance $[\mathrm{mm\,h^{-1}}]$	207.2	116.6
Model type	Spherical	Spherical
Spatial dependence	0.08	0.20
Distance parameter $[\mathrm{m^{-1}}]$	51.05	23.27
Negative log-likelihood	221.0	140.6
Wald statistic	3.0	6.9
P-value	0.813	0.331

3.4.2 Best bases for the wavelet analysis

The best bases for the K_s data for both sampling depths are shown in Figure 3.1. They are reasonably similar at low frequencies (<0.125 cycles per observation). At higher frequencies the best basis for the depth of 6–12 cm favours spatial resolution over frequency resolution, whereas the 0–6 cm depth does not. It is, therefore, more plausible to assume an underlying stationary random variable for the depth of 0–6 cm (Lark, 2007), which we attribute to the overriding influence of small-scale processes, such as bioturbation, on K_s in the topsoil, an influence prevalent equally along the transect. At the lower depth, other factors seem to contribute to, if not dominate, the variability of K_s, which is why we chose the best basis for the 6–12 cm depth for further analysis. The wavelet packets that make up this basis and their respective frequency intervals are given in Table 3.3. To facilitate the subsequent discussion, we have numbered these packets from one to ten, in line with increasing frequency.

Table 3.3: Nominal frequency intervals of the wavelet packet basis used in this study. The frequencies of the lower and upper bound of the intervals are calculated as cycles per observation (15 m).

		Frequency [m^{-1}]		Scale [m]	
Packet bound	Packet bound	Lower bound	Upper bound	Upper bound	Lower bound
1	$P_{6,1}$	0.000	0.008	3840	1920
2	$P_{6,2}$	0.008	0.016	1920	960
3	$P_{6,3}$	0.016	0.023	960	640
4	$P_{6,4}$	0.023	0.031	640	480
5	$P_{4,2}$	0.031	0.063	480	240
6	$P_{4,3}$	0.063	0.094	240	160
7	$P_{4,4}$	0.094	0.125	160	120
8	$P_{3,3}$	0.125	0.188	120	80
9	$P_{3,4}$	0.188	0.250	80	60
10	$P_{1,2}$	0.250	0.500	60	30

3.4.3 Multi-resolution analysis

The MRAs for each sampling depth are shown in Figure 3.4. There is much variation in the highest frequency interval (packet 10), possibly caused by small-scale processes such as bioturbation, which may have a larger effect at 0–6 cm (Figure 3.4a) than at 6–12 cm (Figure 3.4b), but the frequency interval corresponding to this packet is much larger than the other packet frequency intervals. The wavelet packet variance results (discussed below) show that when this is accounted for, the variation at higher frequencies is not, as the MRA figure suggests, as large as that of the lower frequency.

In the 6–12-cm signal (Figure 3.4b), the two lowest frequency components (packets 1 and 2) broadly reflect the pattern of elevation, suggesting some low-frequency correlation between the two variables, which might be related to the andesite flow in the centre of the island. This is not apparent in the 0–6-cm signal (Figure 3.4a). However, we can see that the variation in packets 5 and 6 is visibly smaller over the part of the transect that corresponds to the elevated plateau. Comparing this variation with the soil map units indicated in Figure 3.4, we see that no relationship is detectable visually.

3.4.4 Wavelet packet variance and change detection

The wavelet packet variance estimates show the partition of the signal variance across the frequency range for all packets. Figure 3.5 shows the partition of wavelet packet variance across the frequencies for each K_s series. As the frequency interval changes from packet to packet, depending on its original level of decomposition, a simple graph of wavelet packet variance against frequency can mislead. Therefore we plotted the wavelet packet variance divided by the associated frequency interval against the upper bound of the frequency interval. This quantity is a standardized wavelet packet variance for the frequency interval. For both the 0–6 and the 6–12 cm depth the wavelet packet variance distributions are very similar. This suggests

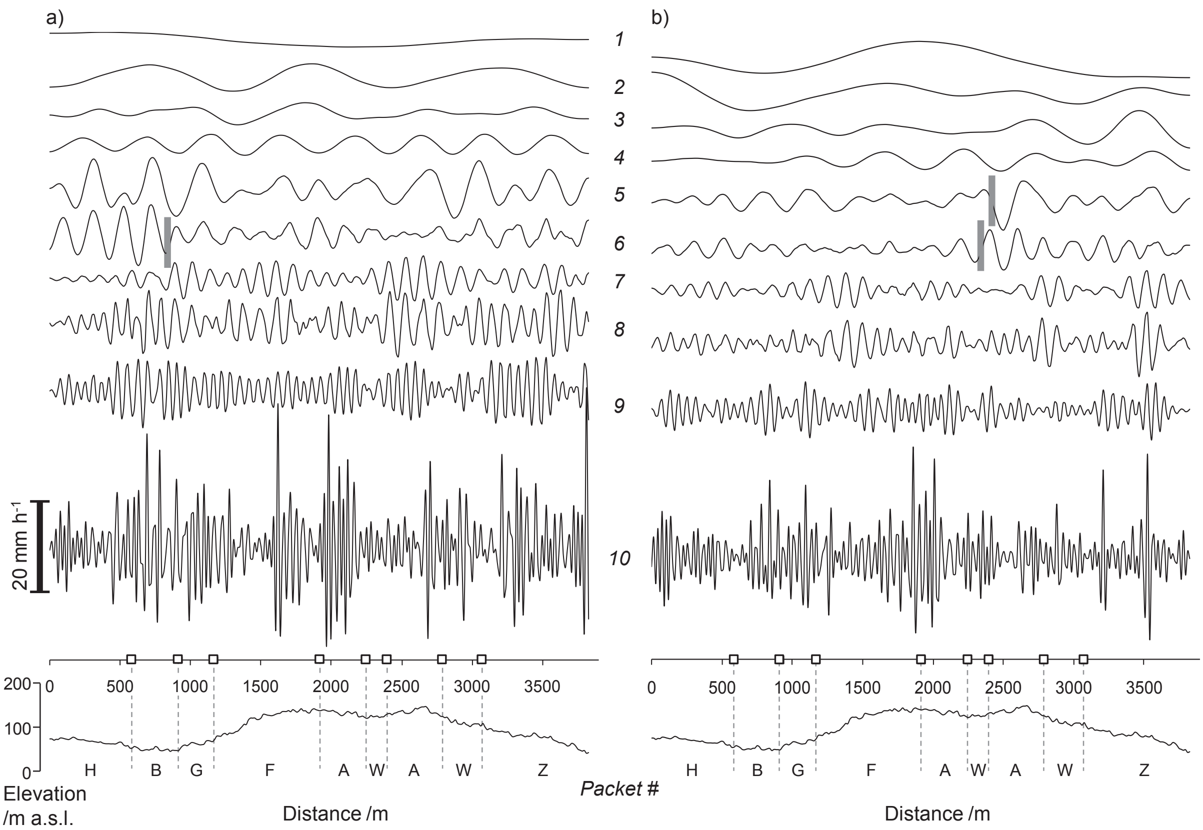

Figure 3.4: Multi-resolution analysis (MRA) of the square-root transformed K_s data for (a) the 0–6 cm depth and (b) the 6–12 cm depth on the best basis described in Table 3.2. In each case the highest frequency component is at the bottom of the figure and the top graph is the lowest frequency component. The scale bar on the left is equivalent to 20 mm h^{-1} in hydraulic conductivity and applicable to all 10 components. Below the x-axis the elevation profile for the transect is shown. The squares on the x-axes and their associated vertical lines stand for soil map unit limits between the units H (Hood), B (Barbour), G (Gross), F (Fairchild), A (AVA), W (Wetmore) and Z (Zetek). Grey bars in the MRA components indicate significant change points found by the wavelet variance change detection algorithm.

a common source of variation. The 0–6 cm depth signal generally indicates larger variation than the 6–12 cm depth signal, which may indicate a multitude of processes influencing the topsoil, leading to a more variable pattern in K_s than in the lower depth.

The largest contribution to variation is from frequency interval 0.023–0.031 cycles per observation (corresponding to a 480–640-m interval of periods; adjacent frequency intervals indicate large variation as well, especially for the 0–6 cm depth. The peak variance frequency interval coincides approximately with the scale of soil map units, so the influence of soil type on the variability of K_s cannot be ruled out.

The variance change detection for the 0–6-cm signal showed one significant change point in variation, in packet 6 (160–240-m scale) at location 840 m shown as a grey bar in Figure 3.4a). The closest map boundary is at about 75 m distance from this change point, the boundary separating map units Barbour and Gross. The prevalent soil types in these units are very similar in spite of different underlying lithology. Although map unit boundaries are idealizations with limited spatial accuracy, it seems unlikely that the one wavelet variance change point detected should relate to this rather non-spectacular change in soil types. Furthermore, none of the other map unit boundaries has a significant effect on the variation of the 0–6 cm depth data. Thus, no conclusive link between soil type and K_s variability can be seen in this depth.

For the 6–12 cm depth, two significant change points were detected, in packet 5 (240–480 m) at location 2415 m and in packet 6 (160–240 m) at location 2340 m. These are shown as grey bars in Figure 3.4b. At location 2400 m there is a pronounced change in soil type, from Wetmore to AVA (Table 3.1). Taking into account the location error of map unit boundaries, the change points in packets 5 and 6 may well reflect this change. However, it is again noteworthy that none of the other map unit borders were identified in the wavelet variance change detection. Additionally, none of the change points in wavelet packet variance in either depth occurred in the packets with the largest variance values. The wavelet packet variance shows a possible effect of the frequency interval of soil map units. However, we cannot conclude that soil type is the main determinant of K_s variability on this transect.

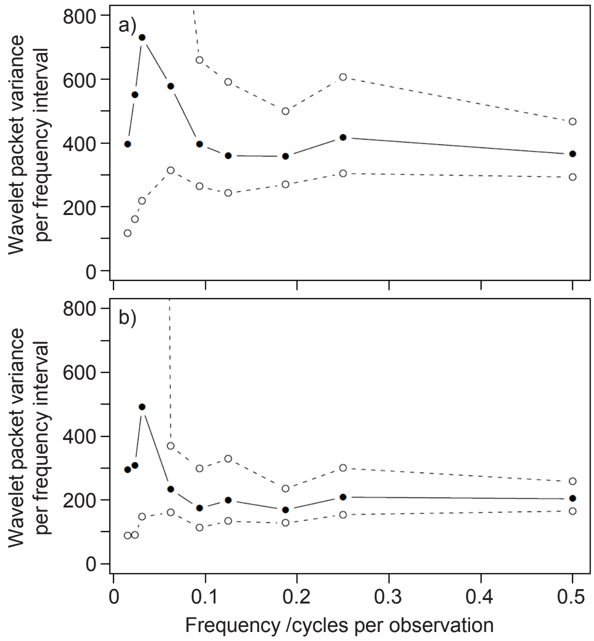

Figure 3.5: Standardized wavelet packet variances for (a) 0–6 cm and (b) 6–12 cm depths. The solid shapes indicate the wavelet packet variance values divided by their associated frequency interval and are plotted against the upper bound of the frequency interval they are associated with. The empty shapes show the 95 % confidence intervals.

3.4.5 Wavelet correlation and change detection

The wavelet correlations between the 0–6 cm K_s signal, the 6–12 cm K_s signal and elevation are shown in Figure 3.6. The correlation between the two depths of K_s is significantly different from zero for higher frequencies (>0.125 cycles per observation, or in terms of period <120 m), but relatively weak (<0.46). The variance plots for these two variables (shown in Figure 3.5) are similar, suggesting a common source of variation. This is supported by the significant wavelet packet correlations. However, these correlations are not particularly strong; thus this source of variation may not be as important as other factors that affect the variability of K_s locally. This again might indicate the influence of local small-scale processes such as bioturbation, which differs between the two sampled depths. It is also possible that the relatively small sampling volume produces an artificial small-scale variation, leading to wide confidence intervals at the higher frequencies and to the decoupling of the 0–6 and 6–12 cm sampling depths.

None of the correlations between the 0–6 cm depth of K_s and elevation is significantly different from zero and none is greater than 0.29. This is also true of the 6–12 cm depth and elevation correlations, with the exception of two frequency intervals, 0.008–0.016 and 0.063–0.094 (corresponding to the intervals of periods 240–480 and 960–1920 m). The low frequency (0.008–0.016) wavelet packet correlations show an interesting feature:

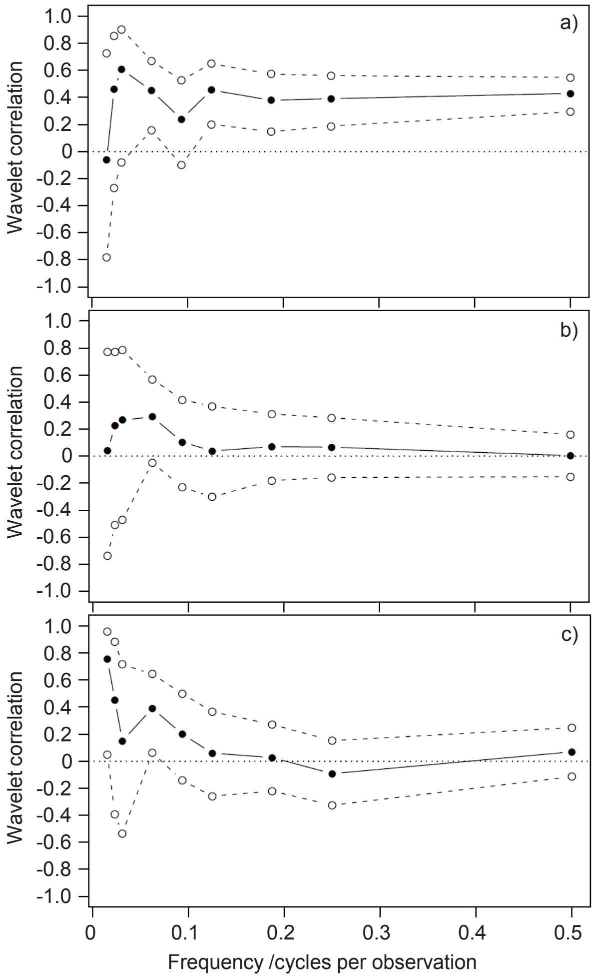

Figure 3.6: Wavelet packet correlations between (a) the 0–6 cm depth with the 6–12 cm depth square-root transformed K_s values, (b) the 0–6 cm depth square-root transformed K_s with elevation and (c) the 6–12 cm depth square-root transformed K_s with elevation, plotted against the upper bound of the frequency interval they are associated with. The solid shapes indicate the correlation values, the empty shapes show the 95 % confidence intervals.

the correlation of elevation with the 0–6 cm depth is weakly positive (0.04), whereas the correlation with the 6–12 cm depth is strong (0.75) and significantly different from zero, supporting the visible structure in those packets in the MRA. This indicates that factors related to elevation, or for which elevation is a proxy, affect low frequency variation in the 6–12 cm depth K_s.

There are no significant changes in correlation between the two sampling depths, and no significant changes in correlation between elevation and the 6–12 cm depth signal. There is one significant change point in the correlation between elevation and the 0–6 cm signal. It occurs in the highest frequency packet at location 3720 m. However, neither before, nor after, the change point is the wavelet packet correlation significantly different from zero. Thus, although the significant wavelet correlation at the 0–6 cm depth hints at an influence of factors related to elevation, which is a proxy for geology, there are no significant change points associated with that correlation, nor can we see this influence from the wavelet variance changes with those soil map units that are underlain by different geology.

3.4.6 Implications for the influence of soil type on K_s

The comparison of spatial means of K_s using REML did not show any differences between soil map units. The same applies to variability: neither the wavelet packet variance nor the wavelet packet correlation indicates a relationship between the variation in K_s and soil type on this transect. Dependence of K_s on soil structure has been shown in various PTF studies, where prediction of K_s has been more successful when including not only static bulk soil properties but also direct measures of structure, morphology or porosity (McKeague et al., 1982; McKenzie & Jacquier, 1997). For the humid tropics, Sobieraj et al. (2002) also failed to explain differences in K_s along a catena with texture and stressed the possibly dominating influence of biotic macropores. Although differing clay mineralogy has been shown to affect soil structure and thus K_s, this was not evident in our study as we did not detect any significant changes in K_s that we could confidently relate to soil map units with distinct clay mineralogy.

The fact that the best basis showed greater stationarity at higher frequencies in the 0–6 cm depth suggests that short-range processes such as bioturbation control K_s variability, especially because this greater stationarity is limited to the upper sampling depth. Nevertheless, measurement error and the effects of small sample volume cannot be ruled out. At our study site, the mosaic of different plants and associated soil faunal communities probably contribute strongly to this high-frequency biotic influence on K_s via macroporosity and soil structure (Angers & Caron, 1998; Oades, 1993). The lack of a relationship between soil type as deduced from soil map units and K_s even in the 6–12 cm depth, which should be a better indicator of static soil-inherent properties, points to the stronger influence of dynamic factors such as vegetation and soil faunal activity. Indeed, various studies dealing specifically with land-use change, and hence with changes in such dynamic factors, highlighted this marked influence on K_s (Hanson et al., 2004; Zimmermann et al., 2010b). Thus, attempts to model forested tropical regions with limited soil data should take into account the knowledge that soil hydrological properties such as K_s are affected more strongly by dynamic factors than by static soil properties as derived from soil surveys.

3.5 Conclusions

In our study we contribute to knowledge about the variation in K_s and its relationship with soil type in the humid tropics. The spatial means of K_s for the soil map units studied did not differ, nor did the wavelet analysis of the transect data reveal patterns in K_s variability that could be attributed to changes in soil type. The peak in wavelet variance at the approximate scale of soil map units points to a possible influence of soil type on K_s variability, but it was not verified in a change point coinciding with a map unit boundary. We conclude that near-surface K_s is probably not controlled by static, soil-inherent properties for the examined range of soil types, but by dynamic properties that may be related to biotic activity. For modelling purposes in such data-poor environments, these findings should be kept in mind when using soil maps as a basis for predicting soil hydrological characteristics.

Acknowledgements

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Chapter 4

Which sampling design to monitor saturated hydraulic conductivity? *

Abstract

Soil in a changing world is subject to both anthropogenic and environmental stressors. Soil monitoring is essential to assess the magnitude of changes in soil variables and how they affect ecosystem processes and human livelihoods. But which sampling design is best for a given monitoring task?

We examined the efficiency of Rotational Stratified Simple Random Sampling (rotStRS) for the estimation of temporal changes in the spatial mean of saturated hydraulic conductivity (K_s) at three sites in central Panama. Weak regression characteristics in the rotational sampling and little benefit from stratification allowed us to also analyse the data ignoring stratification – Rotational Simple Random Sampling (rotSRS), and ignoring the rotational approach – Stratified Simple Random Sampling (StRS) and Simple Random Sampling (SRS). We then compared the estimated spatial means and variances across the four designs.

The poor performance of stratification and the weak predictive relationship between successive years yielded no advantage when using a sampling design more complex than SRS. The failure of stratification may be attributed to the lack of large-scale variability in K_s. Re-visiting previously sampled locations was not beneficial because of the large small-scale variability in combination with destructive sampling, resulting in poor consistency between re-visited samples.

We conclude that for our K_s monitoring scheme, repeated SRS is equally effective as rotStRS. Some problems of small-scale variability might be overcome by collecting several samples at close range to increase sample support. Finally, we give recommendations how to consider including stratification and rotation when designing a soil monitoring scheme.

*Hassler, S. K., Lark, R. M., Zimmermann, B., Elsenbeer, H. (submitted), *European Journal of Soil Science.*

4.1 Introduction

Soil in a changing world is subject to both anthropogenic and environmental stressors. Yet soil is providing the basis for food production and various ecosystem services. Changes in soil properties, their magnitude, velocity and associated processes, are thus becoming increasingly important for management of natural resources and human livelihoods. For example, in many regions undergoing land-use change, soil is increasingly susceptible to erosion, leading to a decrease in fertility of agricultural areas and larger sediment loads in rivers (e.g. Giertz et al., 2005; Huth et al., 2012). In order to assess changes in soil properties on relevant spatial and temporal scales, soil monitoring studies, the repeated measurement of soil properties, are essential (Arrouays et al., 2009).

When designing soil sampling schemes for monitoring purposes the first decision usually is whether to use a model-based or design-based approach (Brus & de Gruijter, 1997, 1993; Papritz & Webster, 1995). A model-based approach is based on the assumption that the values of a soil variable in the study area can be modeled as a stochastic process. Because the model is the source of randomness in the subsequent data analysis the sampling need not be randomized and is commonly performed on a grid, which distributes the samples regularly over the study area and is especially suited for constructing maps of the soil variable. Inferences from these data are based on the model, however, if the assumptions of the model are not met, statistical inference from this design is invalid (Arrouays et al., 2012; Brus & de Gruijter, 1997). Design-based methods, in contrast, do not assume an underlying model of the soil variable and base statistical inference solely on the inclusion probabilities of the sampling locations which are determined by the applied sampling design. They are often reported to be more suitable than model-based approaches for the determination of the spatial mean of an area and when only a small sample size is feasible (Brus & de Gruijter, 1997, 1993; Lark, 2009).

If the aim of a soil sampling scheme is to assess the spatial mean of a soil variable, the next step is to decide on the details of the sampling design. Two widely used designs are Simple Random Sampling (SRS) and Stratified Simple Random Sampling (StRS), described in depth by de Gruijter et al. (2006). Whereas SRS uses the whole study area to select random samples, in StRS the study area is first subdivided into strata before sampling randomly within the strata. Stratification can be based on previous knowledge of underlying processes influencing the target soil variable or simply by dividing up the study area into compact strata. For determining the average status and change of a soil variable over large regions, stratified designs have shown to be more efficient than SRS in various studies (Arrouays et al., 2012; Black et al., 2008; Papritz & Webster, 1995). However, an increase in efficiency depends on substantial variation of the soil variable being accounted for by the stratification, resulting in smaller within-stratum variances compared to the overall variance.

The aims of sampling and the options for design are more complex in the case of monitoring. One key design decision is whether or not to re-visit some or all previously sampled locations in order to form a set of direct observations of change between the two sampling times. This approach is generally most efficient if the primary objective is to estimate change (Lark, 2009). However, if we are also interested in the spatial means for each sampling time, as in this study, it may be advantageous to use a sampling design in which only a proportion of the sampling locations is re-visited and some additional locations are included in the second sampling time to increase the spatial extent of the sample (de Gruijter et al., 2006). This is termed a rotational design. The best sampling strategy depends, among other factors, on logistical constraints (maximum sample size, the challenges of re-locating sample sites and costs of repeated sampling campaigns) along with the spatiotemporal characteristics of the soil variable.

The target monitoring variable of this study is the saturated hydraulic conductivity (K_s) of the soil, a critical parameter in the water cycle. In the humid tropics, K_s is changing mainly due to shifts in land use. Converting tropical forest to pasture has been widely shown to affect topsoil soil hydrological properties including K_s (Alegre & Cassel, 1996; Martinez & Zinck, 2004). A consequence of this process can be the increased frequency of occurrence of overland flow and risk of topsoil erosion as the vertical water flow path is increasingly hindered by reduced K_s (Bonell & Gilmour, 1978; Germer et al., 2010; Hanson et al., 2004). In the last two decades, a different trend of land use change has been observed, pastures and fields are being actively replanted with timber species or recolonised by secondary succession. With one exception (Zimmermann et al., 2010a) the consequences of this reforestation for soil hydraulic properties have all been examined with space-for-time approaches which assume that soils at different sites under varying stages of reforestation can be seen as examples of the temporal trend in soil properties under reforestation at a fixed location (Hassler et al., 2011b; Nyberg et al., 2012; Peng et al., 2012; Zimmermann & Elsenbeer, 2008). However, this assumption relies on reforestation as being a random process across the landscape and might not be valid in many cases. In order to provide definitive information on how hydraulic properties change under reforestation, unconfounded with possible spatial variation and space-time interactions, it is essential to monitor variables such as K_s at reforestation sites. But which particular sampling design should be used for this task?

The aim of this study was to assess the efficiency of a Rotational Stratified Simple Random Sampling design (rotStRS) for the estimation of the temporal trend of the spatial mean in K_s on three reforestation sites in Central Panama. In this design a proportion of sampling location is revisited at consecutive sampling times while new locations are added as well. Furthermore, the random sampling is done within strata. We assess the consistency between re-visited samples and the efficiency of stratification and then compare spatial mean and variance as estimated from rotStRS to means and variances when analyzing the same dataset as if sampled with a rotational design based on Simple Random Sampling without stratification (rotSRS), a StRS design with compact geographical stratification and SRS.

4.2 Sampling designs

This section gives an overview of the sampling designs we considered, in addition to the rotStRS design used for sampling, and lists the equations to calculate their means and variances (adapted from de Gruijter et al., 2006). A schematic to visualize the differences between the four designs is shown in Figure 4.1.

4.2.1 Simple Random Sampling (SRS)

Sampling points are selected at random within the study area. Equations are adapted from de Gruijter et al. (2006), page 82ff. In this presentation the ith observation of the target variable is denoted by z_i.

With sample size n the estimated spatial mean for SRS across the study area is calculated by

$$\hat{\bar{z}} = \frac{1}{n}\sum_{i=1}^{n} z_i. \tag{4.1}$$

The sampling variance of the estimated spatial mean is given by

$$\hat{V}(\hat{\bar{z}}_{SRS}) = \frac{1}{n(n-1)}\sum_{i=1}^{n}(z_i - \hat{\bar{z}}_{SRS})^2 \tag{4.2}$$

and the spatial variance is estimated by

$$\widehat{S_{SRS}}^2(z) = \frac{1}{(n-1)}\sum_{i=1}^{n}(z_i - \hat{\bar{z}}_{SRS})^2. \tag{4.3}$$

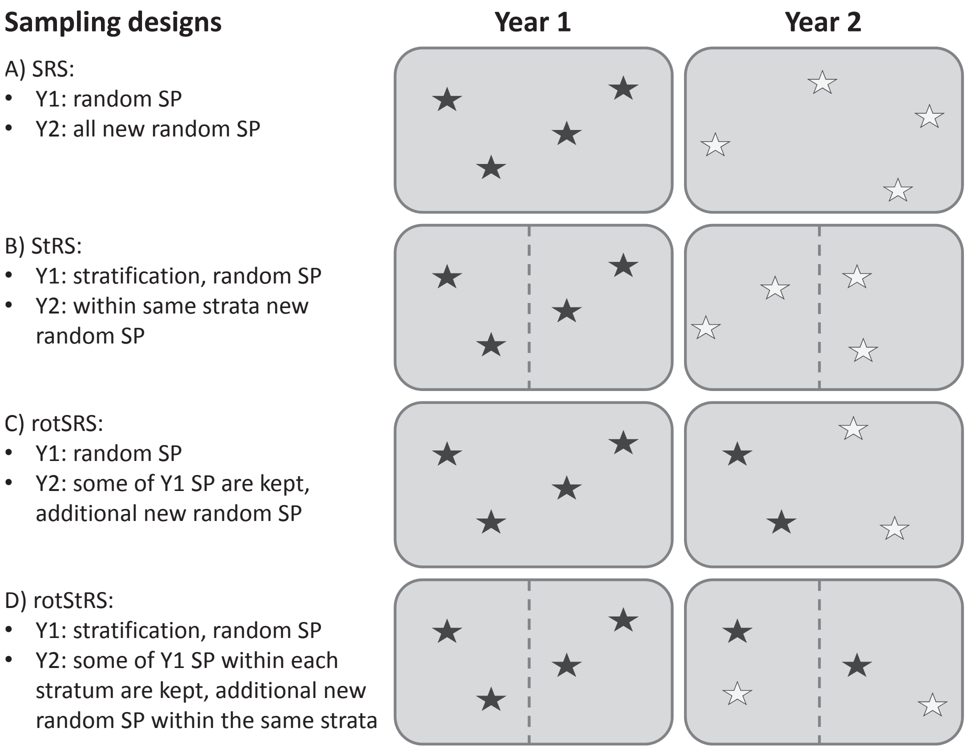

Figure 4.1: Schematic of the four sampling designs we compare in this study. Abbreviations are: SRS for Simple Random Sampling (A), StRS for Stratified Simple Random Sampling (B), rotSRS for Rotational Simple Random Sampling (C), rotStRS for Rotational Stratified Simple Random Sampling (D), SP for sampling points, Y1 and Y2 for Year 1 and Year 2.

4.2.2 Stratified Simple Random Sampling (StRS) with compact geographical stratification

The study area is divided into strata of equal size, random sampling is then done within the strata. Equations are adapted from de Gruijter et al. (2006), page 92ff:

For StRS the spatial mean can be estimated by

$$\hat{\hat{z}}_{StRS} = \sum_{h=1}^{H} a_h \hat{\hat{z}}_h, \tag{4.4}$$

where $\hat{\hat{z}}_h$ is the sample mean in stratum h, H is the number of strata and a_h is the relative area of stratum h.

The variance of $\hat{\hat{z}}_{StRS}$ can be estimated by

$$\hat{V}(\hat{\hat{z}}_{StRS}) = \sum_{h=1}^{H} a_h^2 \hat{V}(\hat{\hat{z}}_h) \tag{4.5}$$

with $\hat{V}(\hat{\hat{z}}_h)$ being the estimated variance of $\hat{\hat{z}}_h$ calculated as follows:

$$\hat{V}(\hat{\hat{z}}_h) = \frac{1}{n_h(n_h-1)} \sum_{i=1}^{n_h} (z_{hi} - \hat{\hat{z}}_h)^2 \tag{4.6}$$

Here n_h is the sample size in stratum h. The spatial variance can be estimated by

$$\widehat{S_{StRS}}^2(z) = \widehat{\widehat{z^2}}_{StRS} - (\hat{\hat{z}}_{StRS})^2 + \hat{V}(\hat{\hat{z}}_{StRS}) \tag{4.7}$$

where $\widehat{\widehat{z^2}}_{StRS}$ is the estimated mean of the target variable squared. It is calculated in the same way as $\hat{\hat{z}}_{StRS}$, but using squared values of the target variable.

4.2.3 Rotational Simple Random Sampling (rotSRS)

Rotational sampling is applied for soil monitoring, for example if the spatial mean of a target variable is estimated on successive sampling times. It includes re-visiting some sampling locations at consecutive sampling times, called the matched sample. If these observations are correlated, the efficiency of estimation of the spatial mean at the second sampling time can be increased by including a regression estimator gained from the matched sample. Not all sampling locations are re-visited, and at the successive sampling time, additional locations are established. The locations that are not re-visited and that are unique to one sampling time are called the unmatched sample. Equations for rotSRS are adapted from de Gruijter et al. (2006), page 226ff.

The spatial mean for the second sampling time is estimated by the composite estimator

$$\hat{\bar{z}}_{2c} = \hat{w}_1 \hat{\bar{z}}_{2gr}^{(m)} + \hat{w}_2 \hat{\bar{z}}_{2\pi}^{(u)}. \tag{4.8}$$

The second component of this estimator, $\hat{\bar{z}}_{2\pi}^{(u)}$, is the π-estimator for the mean of z_2 estimated only from the unmatched sample. The first component, $\hat{\bar{z}}_{2gr}^{(m)}$ is a regression estimator of the spatial mean of z_2. This is calculated by

$$\hat{\bar{z}}_{2gr}^{(m)} = \hat{\bar{z}}_{2\pi}^{(m)} + b(\hat{\bar{z}}_{1\pi} - \hat{\bar{z}}_{1\pi}^{(m)}), \tag{4.9}$$

where $\hat{\bar{z}}_{2\pi}^{(m)}$ is the π-estimator for the mean of z_2 estimated from the matched sample, b is the regression coefficient from the regression of the matched sample from the second sampling time on the matched sample from the first sampling time. The estimate $\hat{\bar{z}}_{1\pi}$ is the mean of the entire sample at the first sampling time, and $\hat{\bar{z}}_{1\pi}^{(m)}$ is the mean of only the matched sample at the first sampling time. The two separate estimates of the spatial mean at the second sampling time are combined in Equation 4.8 by weights that sum to one. These weights are calculated by

$$\hat{w}_1 = 1 - \hat{w}_2 = \frac{\hat{V}(\hat{\bar{z}}_{2\pi}^{(u)})}{\hat{V}(\hat{\bar{z}}_{2gr}^{(m)}) + \hat{V}(\hat{\bar{z}}_{2\pi}^{(u)})} \tag{4.10}$$

where $\hat{V}(\hat{\bar{z}}_{2\pi}^{(u)})$ is the estimated variance of the π-estimator for the mean of z_2 unmatched sample, $\hat{V}(\hat{\bar{z}}_{2gr}^{(m)})$ is the estimated variance of the regression estimator. When the rotational sampling is based on Simple Random Sampling with n total sampling points at each sampling time and m matched samples, the variance of the regression estimator is given by

$$\hat{V}(\hat{\bar{z}}_{2gr}^{(m)}) = \frac{\widehat{S^2}(e)}{m} + \frac{\widehat{S^2}(z_2) - \widehat{S^2}(e)}{n} \tag{4.11}$$

where $\widehat{S^2}(e)$ is the estimated variance of the regression residuals and $\widehat{S^2}(z_2)$ is the estimated variance of the whole sample at the second sampling time, and

$$\hat{V}(\hat{\bar{z}}_{2\pi}^{(u)}) = \frac{\widehat{S^2}(z_2)}{n-m} \tag{4.12}$$

Finally, the variance of the composite estimator is estimated by

$$\hat{V}(\hat{\bar{z}}_{2c}) = \frac{1 + 4\hat{w}_1\hat{w}_2(\frac{1}{m-1} + \frac{1}{n-m-1})}{\frac{1}{\hat{V}(\hat{\bar{z}}_{2gr}^{(m)})} + \frac{1}{\hat{V}(\hat{\bar{z}}_{2\pi}^{(u)})}} \tag{4.13}$$

4.2.4 Rotational Stratified Simple Random Sampling (rotStRS)

For calculation of the rotStRS we calculated first the means and variances of matched, unmatched and whole datasets for all years according to the stratified design. We then used the rotSRS equations to calculate estimates from the rotational design, substituting $\hat{\bar{z}}_{1\pi}$, $\hat{\bar{z}}_{1\pi}^{(m)}$, $\hat{\bar{z}}_{2\pi}^{(m)}$, $\hat{\bar{z}}_{2\pi}^{(u)}$ and $\widehat{S^2}(z_2)$ with their respective stratified counterparts. The regression coefficient b and the estimated variance of the residuals $\widehat{S^2}(e)$ were calculated without accounting for stratification, from the whole matched samples of two consecutive years.

4.3 Materials and methods

4.3.1 Study site

The study was conducted in central Panama in the watersheds of Río Agua Salud and Río Mendoza, which drain into the Panama Canal, partly covering the project area of the Agua Salud Project (Figure 4.2A). The study area is characterised by a strongly dissected pre-Tertiary basalt plateau (330 m amsl) with narrow interfluves, linear slopes averaging 42 % and narrow or no valley floors. Topsoil textures in the area vary from silty clay to clay, pH values (in water) range from 4.4 to 5.8 (Hall et al., unpublished data).

The climate of the study area is tropical with a distinct dry season from mid-December to April. Total annual rainfall in our research area averages 2300 mm (1998–2007, data from the Panama Canal Authority), the mean daily temperature on nearby Barro Colorado Island is 27 °C and varies only little throughout the year (Windsor, 1990).

Land use in the area varies over short spatial and temporal scales and includes pastures, timber plantations and secondary forests of different ages. This study was focussed on three catchments undergoing reforestation. The first site was a 34-ha plantation with native species, established in 2008. Formerly the catchment had been actively used as pasture, but included some larger trees. The second site was also a small former pasture catchment, covered by 5.7 ha of 3 year-old secondary succession. The third catchment holds a 10.9 ha teak plantation planted in 2008, formerly covered by a mixed land use: partly pasture and partly shrubland.

4.3.2 Sampling design

Each site was sampled to determine the spatial mean of K_s in the years 2009, 2010 and 2011 in a rotStRS design with compact geographical stratification. We first divided each of our catchments into twenty compact strata of equal area (19 in the case of the secondary-succession catchment) with a k-means clustering algorithm (Brus et al., 1999) from the R package SPCOSA (Walvoort et al., 2010). Within each stratum, we randomly selected two sampling locations in the first year (2009) and marked them after sampling. In the following year we kept one of these two points per stratum, discarded the other and randomly chose a new sampling point. For the third year campaign the sampling points in the matched sample for 2009 and 2010 were discarded, the unmatched sample points from 2010 were retained (now the matched set for 2010/2011) and a new sample point was randomly selected within each stratum to constitute the unmatched sample set for 2011; see Figure 4.2 for an example. The re-sampling of points in the matched set in any year took place within a distance of maximum one metre from the previous year's sampling point. Note that this initial sampling design is not appropriate for rotStRS because there is only one matched and one unmatched sample point per stratum in any year (which does not permit the calculation of a within-stratum variance). For this reason we merged adjacent strata so that each of the new strata contained (ideally) two matched and two unmatched points in any one year. In the case of the secondary-succession catchment (19 initial strata) one cluster of 3 strata were merged, and the remainder were merged in pairs.

4.3.3 Field sampling of saturated hydraulic conductivity (K_s)

The saturated hydraulic conductivity (K_s) was measured on undisturbed soil cores. Simultaneously, two soil cores of 8.9 cm diameter were extracted at depths 0–6 cm and 6–12 cm on levelled ground using a standard coring device (Soilmoisture Equipment Corporation, Santa Barbara, USA). Core ends were cut flat with a sharp knife and the samples were slowly saturated upside down over a period of 64 hours to prevent air entrapment. We measured K_s applying a constant

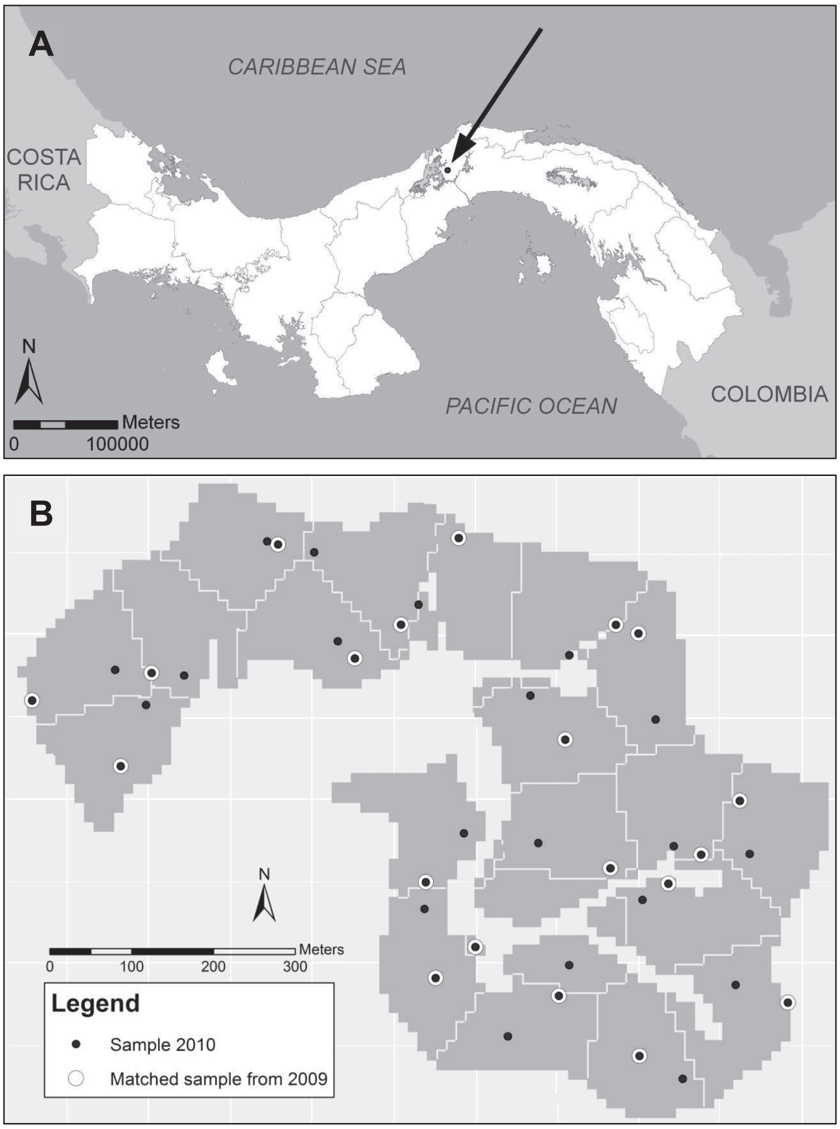

Figure 4.2: (A) Location of the study in Central Panama, (B) Map of the sampling design in the native-species catchment in 2010. Shown are the sampling points of 2010, the matched sample that was already sampled in 2009 and is re-sampled in 2010 and the stratification.

water head, following a simplified version of the methodology of Reynolds et al. (2002). After establishing a constant flow rate, K_s can be calculated according to Darcy's Equation for saturated conditions:

$$q = -K_s \frac{dh}{ds}, \tag{4.14}$$

where q is the flux density [$\mathrm{m\,s^{-1}}$], K_s is the saturated hydraulic conductivity [$\mathrm{m\,s^{-1}}$], $\frac{dh}{ds}$ is the hydraulic gradient. The flux density can be expressed as $q = \frac{Q}{A}$ with Q being the water flux [$\mathrm{m^3\,s^{-1}}$] and A the cross-section of the sample [$\mathrm{m^2}$].

4.3.4 Data analysis

Our data exhibited the well-known skewness for K_s. To obtain normally distributed datasets for the analysis of the different sampling design estimates, we performed a Box-Cox transformation (Box & Cox, 1964). A common Box-Cox exponent was estimated for all datasets (grouped by site, year and depth), and the BOXCOX procedure from the MASS package in R (Venables & Ripley, 2002) was used for estimation by maximum likelihood. The estimated value of the exponent was 0.16. Thus, all analyses were carried out using the transformed K_s values as follows:

$$z_{BC} = \frac{z^{0.16} - 1}{0.16} \qquad (4.15)$$

After estimating means and variances of the means according to the four different sampling designs, we calculated 95 % confidence intervals around the means. The backtransformation of the means and confidence interval limits was done by

$$z = (z_{BC} \cdot 0.16 + 1)^{1/0.16}. \qquad (4.16)$$

Because the transformation is non-linear the simple back-transformation of the sample mean yields a value which is a biased estimate of the mean on the original scale of measurement. However, assuming normality of the transformed data, the back-transformed mean can be regarded as an estimate of the median on the original scale of measurement (Pawlowsky-Glahn & Olea, 2004) since the mean and median of a normal variable are coincident, and order statistics can be back-transformed simply for a monotonic transformation such as ours. Because of this monotonic property the upper and lower confidence interval limits can also be back-transformed directly.

The calculations of means and variances for the transformed data for each sampling design were undertaken according to the equations cited in the Sampling Designs section. In some cases where missing data did not allow a complete analysis (since, for example, only one sample was available from a stratum), that stratum was discarded for purposes of the stratified analysis.

We found that stratification did not account for a substantial component of the variance of K_s in any catchment (see below) and thus also analysed the data ignoring stratification (rotSRS). As the consistency between matched samples was very weak, we also analysed the data without using the regression estimator, treating the samples of each year as if gathered by StRS or SRS. For this analysis we stuck to the original non-merged strata as to use the maximum available data, but strata with missing values were discarded. Since we were interested in comparing the performance of the different sample designs, we calculated the variance of the sample means for SRS and StRS by dividing their spatial variance by the sample size used for rotSRS/rotStRS.

It should be noted that the data were collected according to the rotStRS. The comparative analyses of all designs are merely justified because of the lack of a benefit from stratification from including the regression estimator. In order to express this quantitatively, we assess the efficiency of each design by comparing the width of the confidence intervals for the different years, catchments and depths. Information about within- and between-stratum variances for StRS from an ANOVA (Table 4.1) and about the regressions of the matched samples for rotSRS (Table 4.2) is presented to indicate how effective stratification and the inclusion of the regression estimator in the rotational designs can be with the given datasets. We examine the changes in K_s in the different catchments and depths over the years and conclude with remarks about how to consider including stratification and rotation when designing a soil monitoring scheme.

All statistical analyses were carried out in the language and environment R (R Development Core Team, 2012).

Table 4.1: ANOVA results for the analysis of the stratification effect within the different land uses, depths and years. The F ratio shows the quotient of mean-square of the between-stratum variance and the mean-square of the within-stratum variance. In the case of the between-stratum variance being zero, the expected value of the F ratio is one.

		2009		2010		2011	
Depth [cm]	Land use	F ratio	P value	F ratio	P value	F ratio	P value
0–6	Native species plantation	1.570	0.219	0.016	0.900	0.278	0.601
	Teak plantation	0.750	0.392	0.081	0.777	1.250	0.270
	Secondary succession	0.027	0.869	4.118	0.0499*	0.489	0.489
6–12	Native species plantation	0.024	0.879	1.237	0.237	2.764	0.105
	Teak plantation	0.156	0.695	0.131	0.720	1.203	0.280
	Secondary succession	1.460	0.234	0.037	0.848	0.449	0.507

* $p < 0.05$ for the correlation coefficient

Table 4.2: Regression parameters for rotational sampling (rotSRS and rotStRS) for the different years, sampling depths and land uses.

		2010			2011		
Depth [cm]	Land use	Slope	P value	R^2	Slope	P value	R^2
0–6	Native species plantation	0.364	0.205	0.112	0.053	0.825	0.003
	Teak plantation	0.258	0.184	0.096	0.353	0.130	0.123
	Secondary succession	0.152	0.508	0.026	−0.309	0.289	0.066
6–12	Native species plantation	0.413	0.105	0.204	0.144	0.637	0.016
	Teak plantation	0.208	0.235	0.077	0.126	0.585	0.017
	Secondary succession	0.577	0.094	0.156	−0.232	0.330	0.063

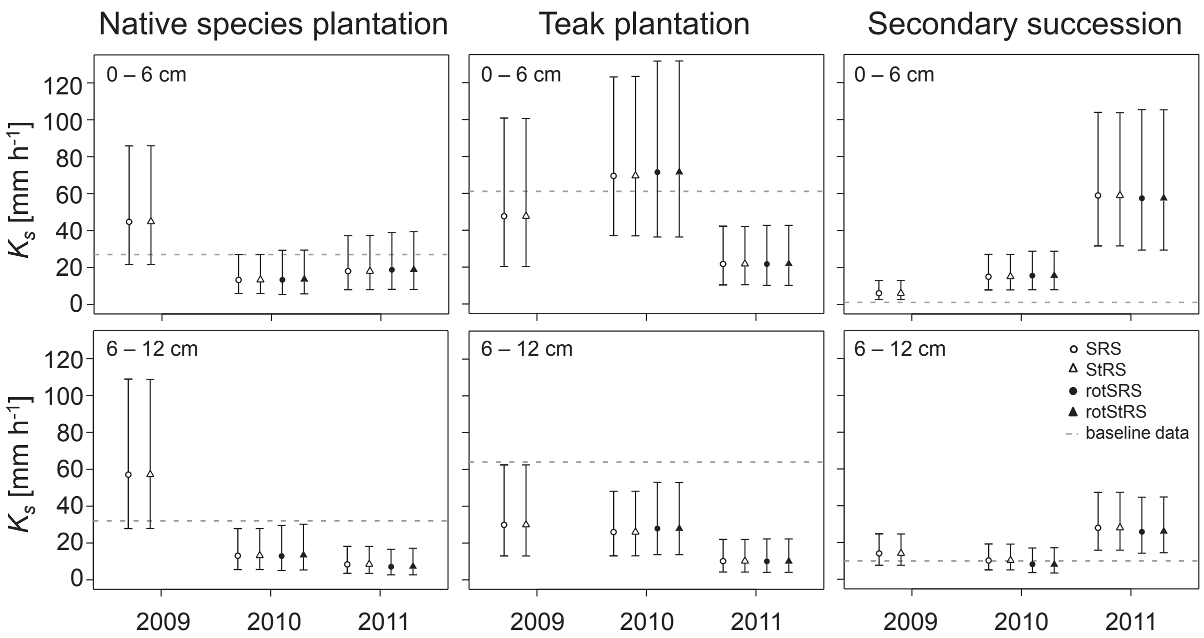

Figure 4.3: Means and confidence intervals of K_s calculated according to the four sampling designs SRS (Simple random Sampling), StRS (Stratified Simple Random Sampling), rotSRS (Rotational Simple Random Sampling), rotStRS (Rotational Stratified Simple Random Sampling). Shown are the comparisons between the three years 2009, 2010 and 2011, the three study sites, covered by native species, teak and secondary succession, and the two depths 0–6 cm and 6–12 cm.

4.4 Results

4.4.1 ANOVA of stratified data and analysis of regression parameters of matched samples

In StRS we use ANOVA to assess the contrast among within-stratum and between-stratum variance and consequently the potential benefit of stratification. The results of ANOVA show that the F ratios are generally small, indicating small contrasts in K_s between the strata, and only in one case, for the secondary-succession catchment in 2010 at the 0–6 cm depth, can we see a significant effect of stratification with a P value of 0.0499.

The rotational sampling is dependent on the regression of matched samples in consecutive years. Parameters of these regressions for the three catchments, two depths and three years are given in Table 4.2, namely the slopes of the regression, the P value for the null hypothesis that the slope is zero and the coefficient of determination R^2 of the regression. P values reach from 0.09 for the 6–12 cm depth for the secondary-succession site in 2010 to 0.83 for the 0–6 cm depth for the native-species site in 2011. The R^2 values are all very small, the largest being 0.2 for the 6–12 cm depth in the native-species catchment in 2010, pointing towards a weak relationship between the matched samples.

4.4.2 Comparison of the different sampling designs

We assess the efficiency of the different sampling designs by comparing the resulting estimated spatial means of K_s and their confidence intervals after back-transformation. The upper confidence interval limits of StRS and SRS (Figure 4.3) show only negligible differences, therefore no general increase in efficiency can be seen.

The results for rotSRS and SRS can be compared only for 2010 and 2011. The upper confidence interval limits of rotSRS are larger than of SRS in nine of the twelve datasets, with differences ranging from -2.6 to 8.6 mm h^{-1}. This again hints at no substantial gain, rather a decrease in efficiency of the rotational design over SRS. The regression estimator will have advantages over alternatives estimating the spatial mean from single year data when there is a strong regression of the matched samples. With only a weak relationship the regression estimator may perform poorly because of the substantial uncertainty in the regression coefficient.

Comparing the two rotational designs rotStRS and rotSRS for 2010 and 2011, the confidence interval limits are almost equal, with differences of between -0.1 and 0.7 mm h^{-1}. In eight of the comparisons cases rotStRS had larger confidence intervals compared to rotSRS. This may be due, in part, to the reduction in degrees of freedom for the standard error of the StRS mean relative to the SRS mean, since one degree of freedom is lost for each stratum mean that must be estimated.

4.4.3 Change of K_s in the three catchments from 2009 to 2011

The estimated means in the native-species catchment suggest a decline in K_s from 2009 to 2011 at both depths (Figure 4.3), with the largest change from 2009 to 2010. The differences are particularly pronounced for the 6–12 cm depth where the confidence intervals for the 2009 and 2011 estimates do not overlap. For the Teak catchment, at both depths any differences are small relative to the confidence intervals for the spatial mean in any one year. The secondary-succession catchment however shows an increase in K_s in the 0–6 cm depth which is large relative to the confidence intervals confidence intervals for 2009 and 2011. The 6–12 cm depth also shows an increase, but this is smaller.

4.5 Discussion

4.5.1 Efficiency of stratification for better estimate of variance

The comparisons of confidence interval widths for SRS with StRS and for rotSRS with rotStRS show that these are generally very similar, and in most cases the interval is slightly wider for the stratified analysis. The stratification did not increase efficiency because of the very small difference between within-stratum and between-stratum variance (see Table 4.1). As noted above, the benefits of stratification are seen when the strata are internally uniform with regard to the target soil variable but if there is contrast in between the strata. For K_s these differences could be due to land cover or marked changes in soil type. In our catchments, however, land cover and soil type were relatively uniform; consequently, we divided the catchment are into compact geographical strata. This type of stratification may nonetheless be beneficial, but only if the target soil variable exhibits spatial structure at larger scales, when the range of spatial autocorrelation is large (de Gruijter et al., 2006; Walvoort et al., 2010; Zimmermann et al., 2010b). However, K_s often fails to exhibit large-scale structure, it is usually characterized by high small-scale variability, partly because of biotic influences acting on this scale which determine soil structure, partly artificial when K_s is sampled with limited support like small soil cores (Bouma, 1983; Hassler et al., 2011a; Mallants et al., 1997; Sobieraj et al., 2004).

4.5.2 Efficiency of the rotational design

Rotational designs increase efficiency by incorporating knowledge about change of a soil property via the regression of matched samples. In our study, the regression estimator obviously did not improve the variance estimate much, as for most comparisons the confidence intervals of the rotational design were wider than for the nonrotational case (Figure 4.3).

The reason for this becomes clear when examining the parameters of the regressions of the matched samples (Table 4.2). The P values for the slopes show no evidence for slopes different from zero and the relations between the matched samples of all datasets are very weak, as is made clear by the very low R^2 values. Why is there such little temporal consistency of samples taken at the same location? The main problem is that we undertook destructive sampling by using soil cores. When resampling a sampling location, the core in the second year can be taken from no less than 50 cm from the previous year's sampling location in order to sample undisturbed conditions. It may be necessary to sample even further from the original location if the nearer sites are affected by compression or large roots etc., so cannot be sampled. Consequently, sometimes matched samples can be located about one metre apart from each other, and due to the abovementioned small sample support and large small-scale variability they might not actually qualify as matched samples at all (Goidts et al., 2009). To overcome this problem, taking several samples at the same location to account for K_s small-scale variability might be a suitable approach. There are some examples in the soil organic carbon literature that detected significant temporal changes by expanding their support from locations to larger areas (discussed in Arrouays et al., 2012).

4.5.3 Change of K_s in the three catchments

The suspected decrease in both depths' K_s data in the native-species catchment (Figure 4.3) could result from the consequences of rapid land cover change. In 2008, this catchment was an extensively managed pasture with some large trees, which were removed for reforestation with native species. During the felling and removing of the tree stumps, the soil was probably loosened to some extent, leading to an initial increase in K_s in 2009. The following decrease back to values close to the baseline data might suggest a settling of the soil after the initial disturbance.

In the Teak catchment any differences are small relative to the confidence intervals (Figure 4.3). The variation between the three years is probably also attributable to the rapid transformation of land cover, as here the formerly shrubby and very diverse vegetation was removed for the Teak plantation.

The catchment under secondary succession did not suffer from these severe changes; here cattle grazing was given up in the summer of 2006, after which secondary succession took over. The data exhibit a pronounced increase in the 0–6 cm depth and a weak increase in the 6–12 cm depth when compared to the baseline data. They show the recovery of topsoil K_s after abandoning pasture use and are consistent with other studies, conducted in the same area (Hassler et al., 2011b) and in other regions in the humid tropics (Peng et al., 2012; Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2010a).

4.5.4 Sampling design considerations for soil monitoring studies

Designing a sampling scheme for soil monitoring poses many decisions that can be difficult without knowing how the characteristics of the target soil property influence the potential efficiency of the possible designs. Therefore, some lessons learned from our study are listed below:

- In a design-based approach, stratification is a good way to spread out your samples across your study site. However, in terms of improving your variance estimate, stratification is only useful either if your strata show marked differences in factors influencing your target variable, or, in the case of compact geographical strata, if your target variable exhibits large-scale spatial structure. A pilot study can give insights into the spatial structure of your variable or potential strata.

- Rotational designs have proved helpful in estimating a temporal trend of a target variable in other circumstances. In order to take advantage of the regression estimator, there must be a strong consistency between repeated observations at the same location. An assessment of the 'best possible' consistency between re-visited samples could be done in a pilot study: If a set of exploratory sampling locations were sampled and then resampled, this would indicate how consistent matched observations can be in the absence of temporal change. If the consistency is poor then it would be clear that a rotational design has no advantages. Additionally, sampling more points at close range and thus increasing the support of the sample can be beneficial if sampling is destructive and therefore cannot target the same soil volume at consecutive sampling times.

- Judging how your target variable complies with the abovementioned conditions is the paramount step in deciding whether to include stratification or a rotational approach. If, as is the case for K_s sampling, the conditions are not fully met, choosing SRS over more complicated designs will barely affect the efficiency of your means and variances estimates. In some cases SRS might even improve the estimates. Thus, if other considerations such as potential difficulties to revisit the exact same sampling points for rotational sampling or straightforwardness of data analysis play a role, repeatedly applying SRS poses a very suitable design option.

4.6 Conclusion

The Rotational Stratified Simple Random Sampling design we used for our K_s monitoring studies did not yield the expected increase in efficiency compared to more simple designs such as Simple Random Sampling. Reasons for this were the small-scale variability and lack of large-scale structure in K_s so that the strata were no less internally variable than the study site as a whole. Including a regression estimator of the spatial mean in the rotational design also did not yield benefits because of the poor consistency of the matched samples. The lack of consistency is likely to be due to the large short-range variability of K_s. Thus, when destructively sampling K_s using soil cores, matched samples, albeit close in space, might be very different regarding K_s.

4.7 Acknowledgements

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Chapter 5

Land use-specific overland flow generation in the humid tropics *

Abstract

The partitioning of rainfall into different near-surface hydrological flow paths depends on the combination of soil properties and rainfall characteristics. As the relevant soil parameters such as the saturated hydraulic conductivity (K_s) are affected by land-use change, flow paths are expected to differ accordingly. However, there are few studies in the humid tropics which comprehensively assess the relations between K_s, rainfall parameters and overland flow (OF) generation, a specific K_s-dependent flow path, together with spatial variability in these processes and differences between land use types. We studied two headwater catchments covered by secondary forests of 5 and 25 years (SF5 and SF25). Measurements comprised OF presence-absence frequency on planar plots and within topographically incised flowlines, K_s values on the plots, rainfall characteristics and discharge events. Furthermore, we examined ancillary variables such as vegetation parameters and soil texture, and we conducted dye tracer experiments for visualisation of flow paths. Plot-based and event-based analyses revealed differences in overland flow-generating processes between the two forests which we attribute to the differences in vegetation influence and spatial structure of K_s values. In SF5 ground-covering vegetation promoting infiltration into the topsoil on plots contrasted with Hortonian overland flow in flowlines. SF25 was probably governed by a mosaic of areas with differing K_s generating a local pattern of overland flow. Event discharge response was comparable for the two catchments and was not directly related to OF. We conclude that for a comprehensive assessment of near-surface hydrological flow paths, a combined examination of soil hydraulic properties, rainfall and throughfall characteristics, OF and discharge measurements seems necessary. Especially spatial patterns of K_s and OF as well as vegetation characteristics should be included and complemented with dye tracer experiments.

*Hassler, S. K., Crouch, T. D., Ogden, F. L.,Blume, T., Elsenbeer, H. (in preparation).

5.1 Introduction

The activation of hydrological flow paths in tropical soils can be seen as result of an interplay between soil properties and rainfall characteristics. Possible flow paths include vertical percolation through the soil profile towards the groundwater if permitted by soil hydraulic properties, of which saturated hydraulic conductivity (K_s) is especially relevant. Predominantly lateral flow paths can imply different processes: Hortonian overland flow when infiltration capacity at the surface is exceeded by rainfall (Horton, 1933), saturation excess overland flow when the topsoil is saturated up to the surface (Dunne & Black, 1970), subsurface flow from perched water tables on impeding layers, and return flow when subsurface flow re-emerges at the surface, e.g. due to topographical forcing (Bonell & Gilmour, 1978). These lateral flow paths are frequently activated when high antecedent soil moisture or high-intensity rainfall events meet with a distinct anisotropy in K_s (Bonell & Gilmour, 1978; Elsenbeer & Vertessy, 2000; Germer et al., 2010; de Moraes et al., 2006).

Predominant flow paths in a landscape are altered by land-use change because land cover-dependent influences, e.g. organic carbon input or biotic activity, influence soil structure stability and thus K_s (Angers & Caron, 1998; Chappell et al., 1999; Hanson et al., 2004; Oades, 1993). The effects of land-use change on K_s have been the focus of various studies in the humid tropics. For example, a decrease in K_s after pasture establishment on former forest sites as well as a recovery of K_s after pasture abandonment has been reported (e.g. Hassler et al., 2011b; Martinez & Zinck, 2004; Ziegler et al., 2004; Zimmermann et al., 2006). Similarly, the effect of land-use change on hydrological flow paths has been shown. For example, diminished groundwater recharge in pasture compared to forest in the Philippines was attributed to a stronger decrease in K_s, and hence diminished vertical percolation (Chandler, 2006). In Amazonia, the contribution of overland flow to the annual quick discharge was shown to be considerably higher for a pasture site when compared to forest, due to a reduction of macropores and K_s (de Moraes et al., 2006).

Two main approaches have been used in the past to assess the effect of land use on hydrological flow paths. On the one hand, predominant flow paths have been inferred from analysis of extensive data sets of K_s (including anisotropy patterns) in comparison with rainfall characteristics (e.g. Martinez & Zinck, 2004; de Moraes et al., 2006; Ziegler et al., 2004; Zimmermann et al., 2006). This "pedological approach" is based on the assumption that K_s is the major process control and its sensitivity to land-use change. On the other hand, water volumes of surface runoff, subsurface flow and discharge have been measured directly to assess the contribution of the different flow paths to stream flow. This "hydrological approach" is frequently coupled with an assessment of solute or stable isotope composition as natural tracers in order to identify dominant flow paths and residence times (e.g. Biggs et al., 2006; Chaves et al., 2008; Mun oz Villers & McDonnell, 2012; Neill et al., 2011) Further insight into prevalent flow paths can be gained from dye tracer studies, in which active flow paths in the unsaturated zone can be revealed under controlled rainfall conditions (e.g. Hanson et al., 2004).

Several studies highlight the spatial variability of both K_s and overland flow (OF) occurrence, (Germer et al., 2010; Loos & Elsenbeer, 2011; Sobieraj et al., 2002; Zimmermann & Elsenbeer, 2008). However, only few studies combine both pedological and hydrological approaches to assess spatially explicit details of K_s influence on OF generation. Zimmermann et al. (2013) found that spatial structure in K_s which was related to the presence of flow lines determined patterns of OF in a tropical rainforest catchment in Panama. In the same area, Godsey et al. (2004) observed differences in OF generation linked to K_s on different lithologies, but under similar land cover – tropical rainforest. Although there is evidence for land-

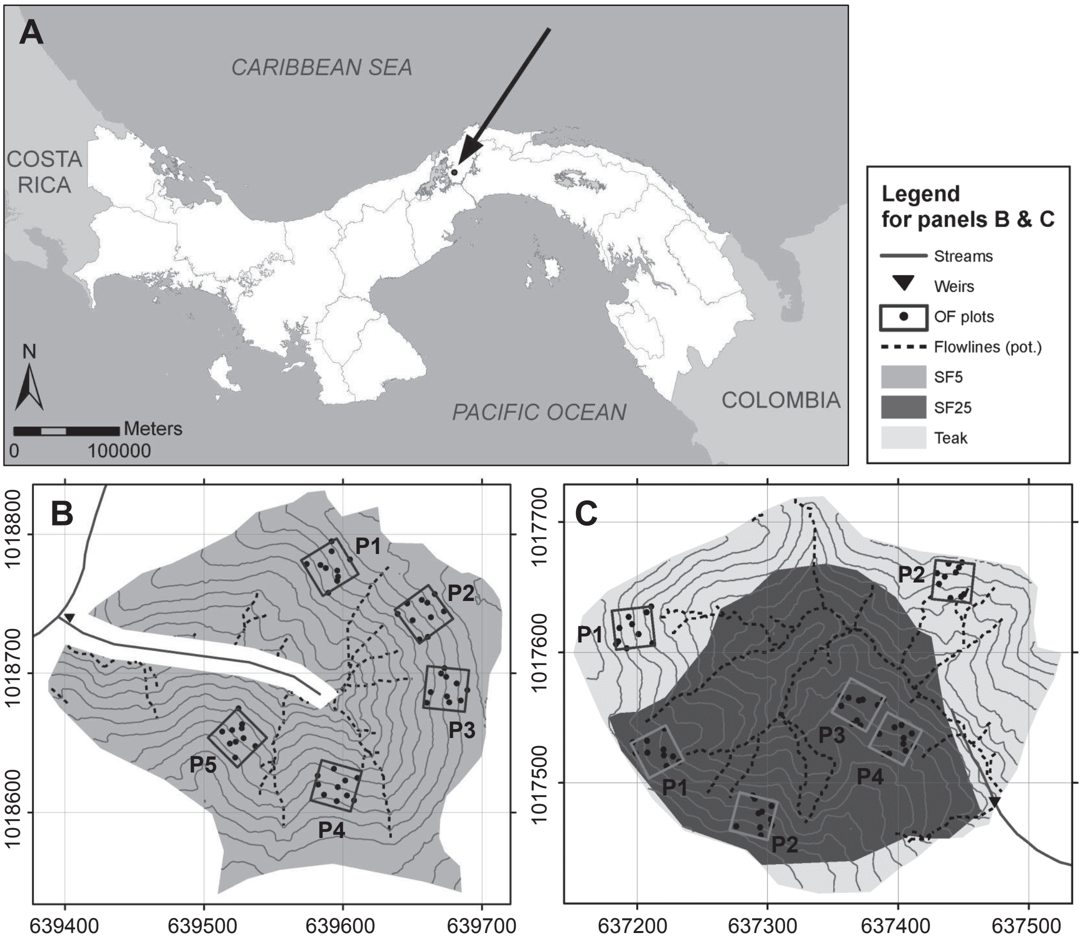

Figure 5.1: Location of the study area in Panama (A) and maps of the two catchments under 5 year-old secondary forest (B) and 25 year-old secondary forest in combination with a teak plantation (C). Shown are the overland flow (OF) plots and potential flowlines which were considered for OF detector installation. The strip along the stream in SF5 is covered by gallery forest and was not considered in this study.

use effects on spatial patterns of K_s and OF (Germer et al., 2010; Zimmermann & Elsenbeer, 2008), there is still a lack of comprehensive studies of the influence of K_s on near-surface hydrological flow paths that are spatially explicit and considering differences in land use.

The objective of this study was to contribute to a better understanding of overland flow generation in the humid tropics by combining the ”pedological” and ”hydrological approach” and conducting a broad set of measurements at two study sites under different land cover.

5.2 Methods

5.2.1 Study site

The study was conducted within the Agua Salud Project area in central Panama, in the watersheds of Río Agua Salud and Río Mendoza, which drain into the Panama canal (Figure 5.1A). The area is characterised by a strongly dissected pre-Tertiary basalt plateau (330 m amsl) with narrow interfluves, linear slopes averaging 42 % and narrow or no valley floors. Soils vary little with respect to texture (silty clay to clay) thanks to the uniform parent material, with pH values (in water) ranging from 4.4 to 5.8 (Hall et al., unpublished data).

The climate of the research area is tropical with a distinct dry season from mid-December through April. Total annual rainfall averages 2300 mm (1998–2007, Data from the Panama Canal Authority) and the mean daily temperature on nearby Barro Colorado Island (BCI) is 27 °C and varies only little throughout the year (Windsor, 1990).

Land use in the area varies over short spatial and temporal scales and includes pastures, timber plantations and secondary forests of different ages. This study focused on two secondary forests, 5 and 25 years old respectively (hereafter SF5 and SF25), located in two different headwater catchments. The younger forest covers most of a 5.7-ha catchment, excluding a strip of gallery forest that differs greatly from SF5 (Figure 5.1B). The older forest covers about 60 % of a catchment of 7.6 ha, with the remaining 40 % being a teak plantation (Figure 5.1C).

5.2.2 Field sampling

5.2.2.1 Overland flow (OF)

At each forest site, 30 m x 30 m planar plots with comparable slope and without incisions in topography were established. Within these plots we randomly chose 10 locations for overland flow detectors (OFDs). Additionally, incised flow lines in the catchments were equipped with OFDs in order to elucidate the spatial connection of OF to this preferential drainage network. OFD installation in flow lines started at the lower end of the flow lines, which is usually marked by a gully head with active signs of erosion, and continued linearly along the flow line at 5 metre intervals for the entire percievable topographic incision. Potential flowline positions were first determined from depressions in a digital elevation model (Figures 5.1B, 5.1C). On subsequent field surveys the gully head and possibilities for installation were assessed.

The OFDs we used resembled the design of those employed by Godsey et al. (2004): A partly perforated PVC tube attached to a T-piece reservoir is installed perpendicular to the slope on top of the mineral soil, perforation at the bottom, with the reservoir buried within the soil. Surface runoff flowing downslope enters the tube via the holes and water is collected in the reservoir, thereby providing a measure for presence or absence of overland flow between two measurement times. Rainfall in the humid tropics during the wet season predominantly happens as daily thunderstorms during the afternoons. OF occurrence can thus be assessed regularly in the dry hours between the rainfall events and attributed to the rainfall characteristics of the previous day. OF was noted when the reservoir was at least half full (approximately 50 ml). Figure 5.2 shows an OFD in dry conditions and during a heavy rainfall event as well as two OFDs in a flowline during the same event.

OF monitoring was conducted on plots and in flowlines in both forests in the wet season of 2010, and for SF5 monitoring was continued in the wet season of 2011 in order to provide insight into temporal dynamics. In SF5 we sampled five 30 m x 30 m plots (P1–P5), additionally five flowlines in 2010 and six flowlines in 2011. In SF25 four plots (P1–P4) and eight flowlines were equipped with OFDs. In 2011, OFDs in SF5 were installed at the same locations on the plots to facilitate continued monitoring. Flowline installation differed to some extent between the years as some flowlines could not be reused because of forest clearing activities in the catchment, and other flowlines were discovered. However, these differences were considered acceptable in the overall comparison of OF generation in flowlines and on plots. We monitored 80 OFDs in SF5 and 105 OFDs in SF25 in 2010. In 2011, we monitored 111 OFDs in SF5. In order to estimate the influence of the teak plantation adjacent to SF25 on catchment discharge, we separately monitored two plots and four flowlines with altogether 51 OFDs in the teak. As the influence on discharge was negligible and the teak area was sub-

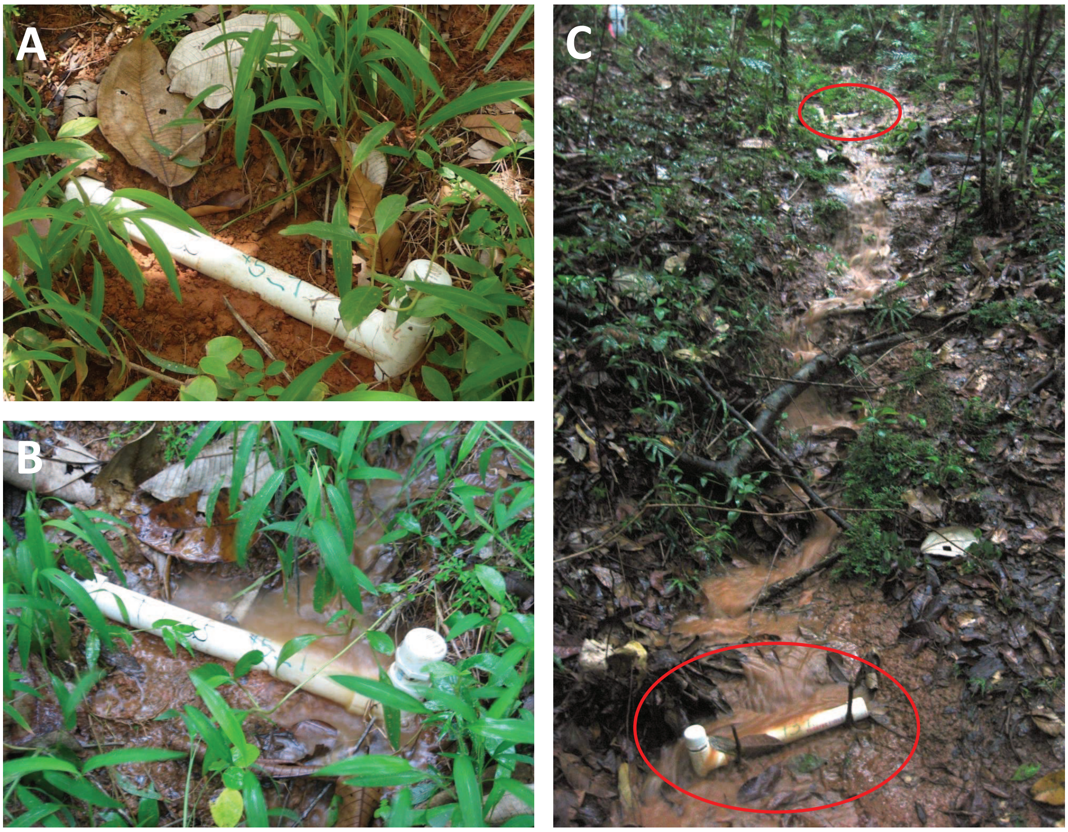

Figure 5.2: An overland flow detector (OFD) in dry (A) and wet (B) conditions. (C) shows connectivity in a flowline between two OFDs during a rainfall event.

ject to a land use history which differs greatly from the two forests, it is omitted from further analyses. Maps of the two catchments are shown in Figure 5.1.

5.2.2.2 Saturated hydraulic conductivity (K_s)

Saturated hydraulic conductivity (K_s) was determined at 15 randomly selected locations on each of the OF plots. Flowlines were excluded from these measurements as they already represent preferential drainage lines within the catchment. Using a standard coring device (Soilmoisture Equipment Corporation, Santa Barbara, USA), we simultaneously extracted two soil cores of 8.9 cm diameter at the depths 0–6 cm and 6–12 cm on levelled ground at each location. After cutting the core ends flat with a sharp knife, the samples were slowly saturated upside down over a period of 64 hours to prevent air entrapment. We then measured K_s applying a constant water head, following a simplified version of the methodology of Reynolds et al. (2002). After establishing a constant flow rate, K_s can be calculated according to Darcy's Equation for saturated conditions:

$$q = -K_s \frac{dh}{ds} \tag{5.1}$$

where q is the flux density [$\mathrm{m\,s^{-1}}$], K_s is the saturated hydraulic conductivity [$\mathrm{m\,s^{-1}}$], $\frac{dh}{ds}$ is the hydraulic gradient. The flux density can be expressed as $q = \frac{Q}{A}$ with Q being the water flux [$\mathrm{m^3\,s^{-1}}$] and A the cross-section of the sample [$\mathrm{m^2}$].

5.2.2.3 Precipitation data

Precipitation was recorded in clearings within both catchments, in SF5 with an Onset tipping bucket with 0.2 mm resolution and in SF25 with a Davis tipping bucket with 0.254 mm resolution, monitored by the ASP Hydrology group of the University of Wyoming. A comparison of both types of tipping buckets within SF5 revealed compatible results.

5.2.2.4 Discharge data

Discharge data were measured and analysed by the ASP Hydrology group of the University of Wyoming. At both catchment outlets stream flow was recorded using pressure transducers (Level Troll 300 from In-Situ Inc.) in compound v-notch weirs.

5.2.2.5 Ancillary variables

Local slopes were measured at the K_s sampling points within each OF plot. Applying the "feel method", soil texture classes according to the German Soil Science Society (AG Boden, 1994) were determined from five soil cores, chosen at random from the 15 cores which had already been used to determine K_s on the overland flow plots. We derived clay and silt contents from the mean of the class limits of the obtained texture classes (Table 5.1).

Vegetation parameters were measured at three randomly chosen OFD locations within each OF plot. On a hexagon with 3 m side length around the OFD we determined basal area of the woody vegetation that was higher than 1.30 m. On the same hexagon we estimated cover proportions for the growth form categories trees, shrubs, grasses and herbs as well as the proportions of litter and bare soil (Table 5.1).

5.2.2.6 Dye tracer studies

In 2011 we conducted dye tracer experiments using Brilliant Blue in order to reveal water flow paths under the type of rainfall intensities that initiated overland flow in 2010. We used a pesticide pump sprayer to apply Brilliant Blue solution at a concentration of 8 $\mathrm{g\,l^{-1}}$ onto sites of one square metre within the OF plots. Two sites were located in SF5, on the plots P3 and P5, and two in SF25, on P2 and P4. The median rainfall intensity of the events producing OF was 40 $\mathrm{mm\,h^{-1}}$, thus we applied 20 l of Brilliant Blue solution in 30 minutes. Spraying was done during dry morning hours. Afterwards we covered the sprayed area in order to protect the staining from rainfall in the afternoon. On the following day we excavated the Brilliant Blue profiles according to the methodology suggested by van Schaik (2009), establishing three vertical profiles in 10, 25 and 40 cm distance from the downslope border. Uniform infiltration depth (UID), maximum depth (MD) of Brilliant Blue stains and total stained area (TSA) were noted for each profile. On the remaining half of the plot we excavated horizontal profiles in 2, 6 and 20 cm depth and also determined TSA.

5.2.3 Data analysis

OF response on plots was calculated as the fraction of the ten OFDs showing presence of OF. Response was only recorded if present in at least two of all plot OFDs in the respective catchment. We calculated flowline response as the fraction of "active" flowlines of all flowlines within the catchment; a flowline was termed "active" when at least two of its OFDs showed response.

Rainfall events were defined as rainfall periods separated by at least two hours without any precipitation. From this data we calculated rainfall amounts and intensities over various time intervals per event to explore the relationship between rainfall and OF. In the case of more than one event during the time between OF measurements, the higher rainfall intensity values were used in the analysis, and rainfall amounts were summed.

Discharge events were separated into baseflow and event flow with the two-parameter digital base-

Table 5.1: Ancillary variables measured on the overland flow plots (SF5: P1–P5, SF25: P1–P4), and averaged over all plots, for both the 5 year-old (SF5) and the 25 year-old (SF25) secondary forest.The + indicates that grasses were present, but did not measurably contribute to cover.

		SF5					SF25				Total	
Ancillary variable	Unit	P1	P2	P3	P4	P5	P1	P2	P3	P4	SF5	SF25
Slope	[°]	28	29	20	24	22	21	21	13	19	25	19
Soil texture												
Clay/silt in 0-6cm	[%]	35/63	45/42	40/53	32/68	30/70	30/68	35/63	30/70	32/68	36/58	31/67
Clay/silt in 6–12cm	[%]	32/68	42/49	35/63	30/68	30/70	32/68	35/63	30/70	32/68	34/63	32/67
Vegetation parameters												
Basal area	[m^2 ha^{-1}]	4.70	5.12	4.51	3.42	7.87	34.13	23.69	18.42	28.88	5.12	26.28
Tree cover	[%]	0	0	0	0	11	72	72	75	67	2	72
Shrub cover	[%]	67	50	75	50	44	28	19	22	25	57	24
Grass cover	[%]	4	8	11	8	+	0	0	0	0	8	0
Herb cover	[%]	19	42	33	83	86	11	14	11	11	53	12
Litter	[%]	50	44	50	15	11	81	78	78	78	34	78
Bare soil	[%]	28	22	6	2	3	8	8	11	11	12	10

flow filter (Boughton, 1993). Ancillary variables were averaged by OF plot and Brilliant Blue profile values were averaged by each site.

All data analysis was carried out in the R programming language and environment (R Development Core Team, 2012).

5.3 Results

5.3.1 Differences in OF response between land cover types

Median K_s values at the 0–6 cm depth were 31 and 89 mm h^{-1} for SF5 and SF25, respectively and their approximate 95 % confidence intervals did not overlap (Figure 5.3). At the 6–12 cm depth, K_s medians were 23 and 39 mm h^{-1} for SF5 and SF25, respectively, with overlapping intervals (Figure 5.3). The range of the OF response on SF25 plots reached 0.51 and exceeded the maximum on SF5 plots (0.33), however, there was no apparent difference in medians (0.14 and 0.16 for SF5 and SF25, respectively). The OF plot response for SF5 in 2011 showed a larger range with the maximum at 0.96, but the medians for SF5 in the two years did not differ (0.18 in 2011). In the flowlines, OF response under both land cover types and for both years were higher compared to the OF response on the plots, medians were 0.60, 0.63 and 0.83 for SF5 and SF25 in 2010 and SF5 in 2011, respectively (Fig 5.3).

5.3.2 Spatial variability within land cover types

K_s at the 0–6 cm depth varied between plots under both land cover types (Figure 5.4). At the 6–12 cm depth K_s exhibited more variability in SF25 than SF5 and median K_s values in SF25 at the two depths seemed to be correlated. For example, at both depths the range and median of K_s on plot P4 were largest among all the plots (Figure 5.4). For SF5, OF response was similar between plots in 2010 and in 2011, and was unrelated to K_s values on the plots. In contrast, OF response in SF25 differed between plots and appeared to be inversely correlated with K_s values, high OF response occurring on plots with low K_s and vice versa.

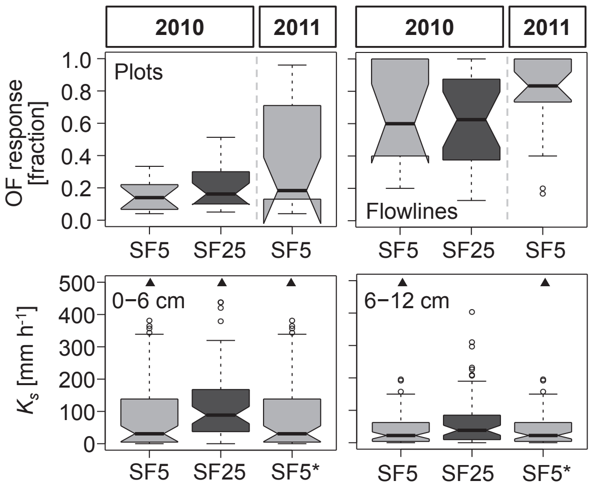

Figure 5.3: Saturated hydraulic conductivity (K_s) at the 0–6 cm and 6–12 cm depth and overland flow (OF) occurrence on plots and in flowlines for the 5 year-old (SF5) and 25 year-old (SF25) secondary forest in 2010 and for SF5 in 2011. SF5* is a repetition of the K_s values of 2010, re-plotted for comparison with the 2011 OF data. The triangles at the top margin of the K_s panels symbolise additional higher values which were left out of the plot for better visual comparison. The values are: at the 0–6 cm depth: for SF5, 583 mm h^{-1} and 795 mm h^{-1} and for SF25, 551 mm h^{-1}; at the 6–12 cm depth for SF5, 611 mm h^{-1}.

5.3.3 Event-based comparison of rainfall, OF, K_s and event discharge

Event-based analysis of rainfall, OF on plots and in flowlines, event discharge and K_s values can give insight into possible relations between the different variables. Figure 5.5 illustrates OF and discharge events during the respective measurement periods for each land cover type along the 10 minute-maximum rainfall intensity ($I_{10\max}$) of the corresponding rainfall events, from minimum to maximum intensities. The respective rainfall

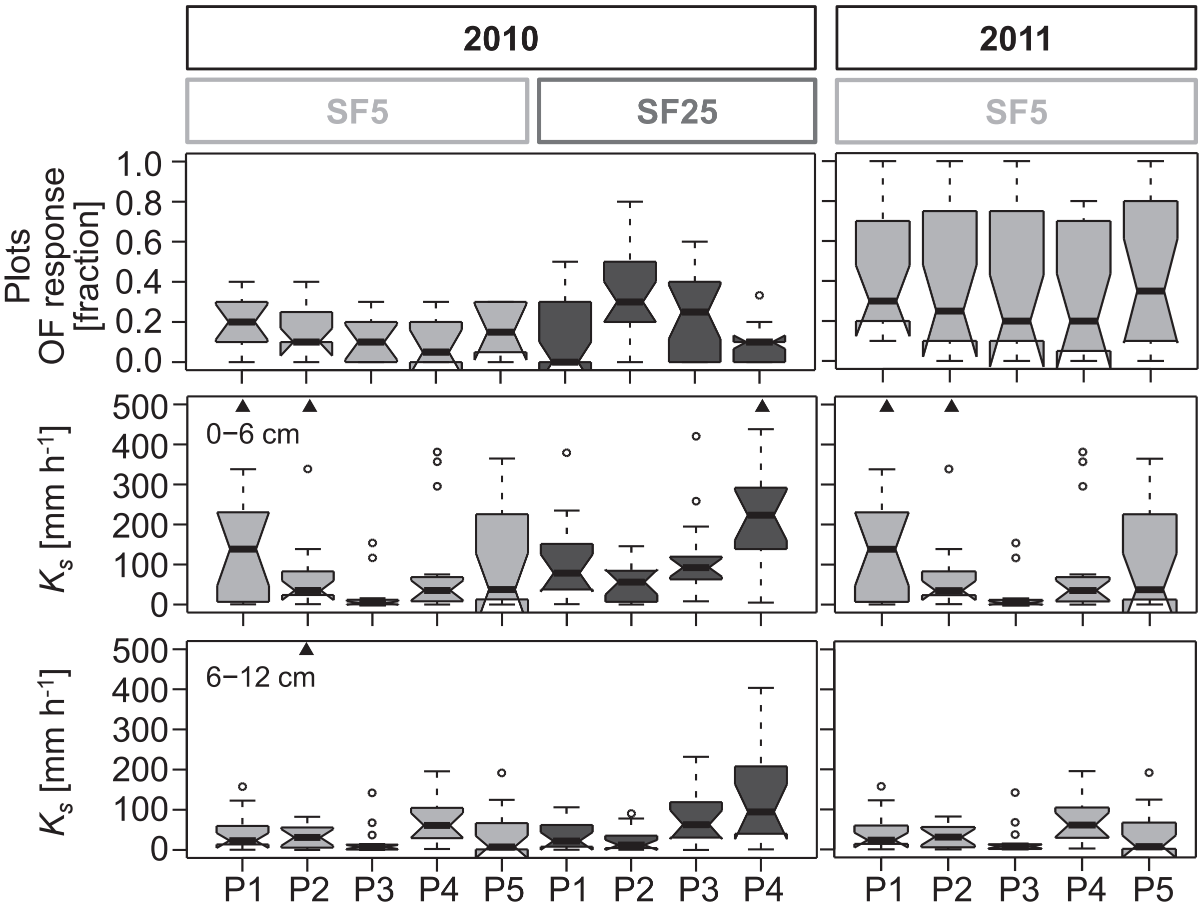

Figure 5.4: Saturated hydraulic conductivity (K_s) of the individual plots (SF5: P1-P5, SF25: P1-P4) of both the 5 year-old secondary forest (SF5) and the 25 year-old secondary forest (SF25) in 2010 and SF5 in 2011. To enable better visual comparison, the following points are not shown (but indicated with triangles): at the 0–6 cm depth: for SF5, 583 mm h^{-1} (P1) and 795 mm h^{-1} (P2) and for SF25, 551 mm h^{-1} (P4); and at 6-12 cm depth for SF5 at the 6-12 cm depth, 611 mm h^{-1} (P2).

amounts and cumulative frequency distributions for K_s are also included. In 2010, rainfall during the OF measurement period was somewhat higher in SF25, with events of larger rainfall depth and higher intensity, than in SF5. Rainfall in 2011 was characterized by many small rainfall events (Figure 5.6).

In SF5 in 2010 the medians of K_s at both depths plot very close together in the cumulative frequency distributions, at about 25 mm h^{-1}, which corresponded to approximately 60 % of the I_{10max} values (Figure 5.5). With one exception (in the flowlines), 25 mm h^{-1} was roughly the intensity at which the recorded OF events started to occur (Figure 5.5). This threshold was also apparent in the direct comparison of I_{10max} and OF in the lower panel of Figure 5.7 as the point where OF responses on plots and in flowlines set in. A rainfall intensity threshold is visible at about 60 mm h^{-1} after which flowline OF response mostly stayed high (Figure 5.5). In contrast, plot OF response did not show a clear threshold and appeared to be unrelated to rainfall intensity (Figure 5.5 and Figure 5.7). Finally, event discharge was mostly below

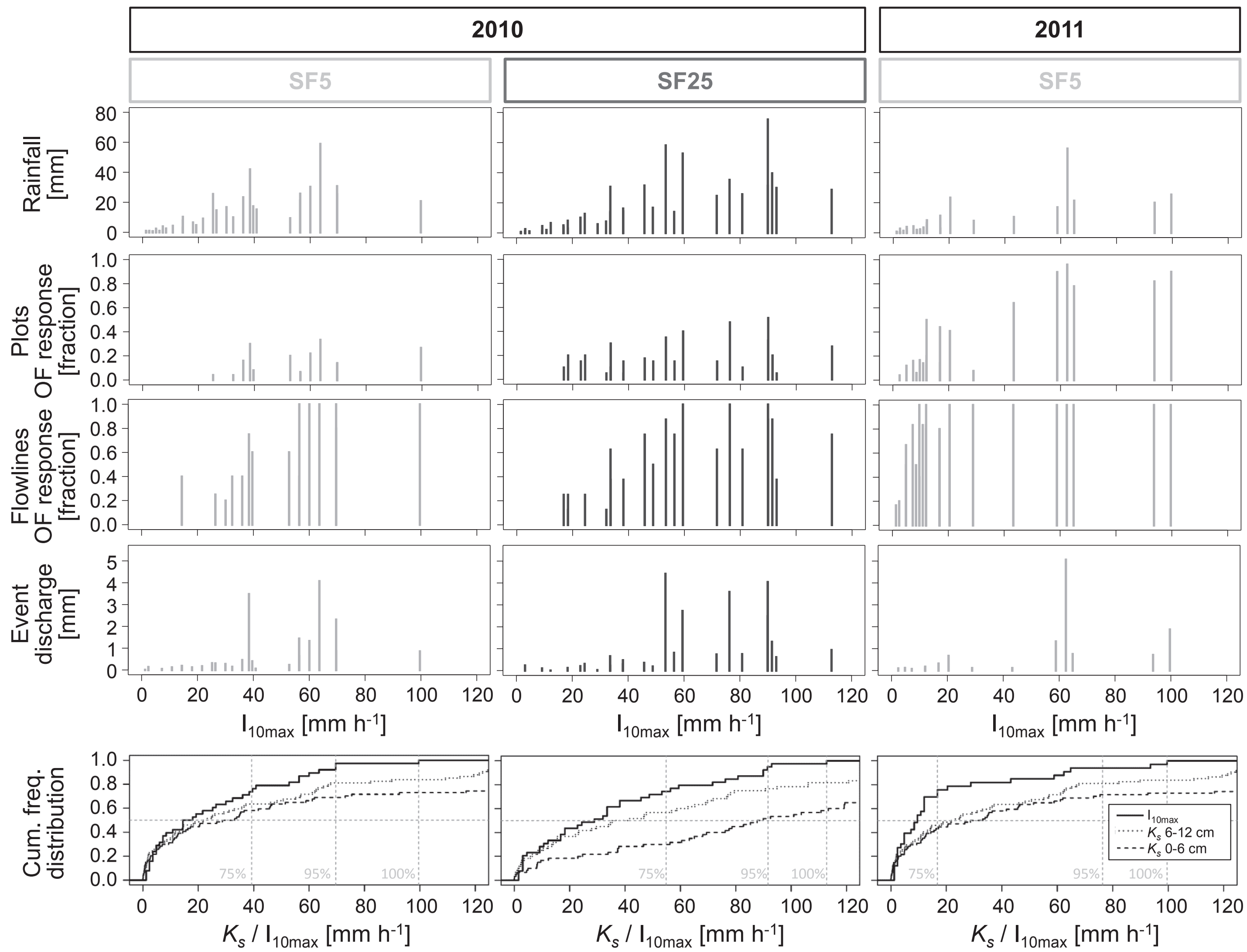

Figure 5.5: Event-based analysis of rainfall, overland flow (OF) on plots and in flowlines, event discharge and cumulative frequency curves of K_s and 10-minute maximum rainfall intensities (I_{10max}) for both secondary forests of 5 and 25 years of age (SF5 and SF25) in 2010 and also for SF5 in 2011. All parameters are plotted against their corresponding I_{10max} value. Note that as lines of events with the same I_{10max} are overlaid, only thresholds of the initial occurrence of large values can be compared in this figure. For direct comparison of all values of rainfall and OF, compare with Figure 5.7.

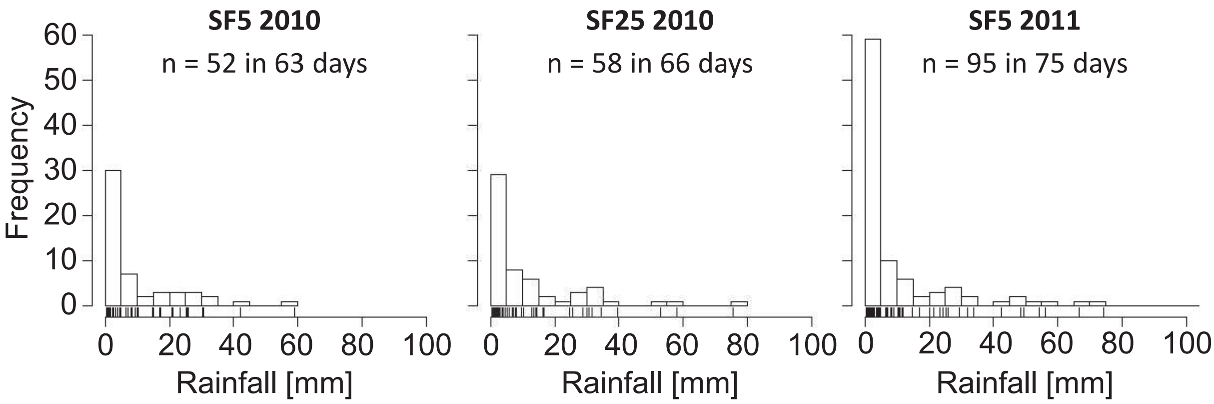

Figure 5.6: Rainfall events during the overland flow measurement periods for 5 year-old (SF5) and 25 year-old (SF25) secondary forest.

0.5 mm h^{-1} at low $I_{10\mathrm{max}}$ values. It had one high value at 40 mm h^{-1} and shows a threshold at nearly 55 mm h^{-1} after which the values were larger than 0.5 mm (Figure 5.5).

The older forest, SF25, showed different dependencies: median K_s in 0–6 cm was around 90 mm h^{-1}, coinciding with the 95 % quantile of the rainfall events. At the 6–12 cm depth, median K_s was considerably lower at around 40 mm h^{-1} and coincided approximately with the 60 % quantile of the rainfall events (Figure 5.5). There appeared to be a correlation between rainfall amount, but not so much intensity, and OF on plots and in flowlines (Figure 5.5 and Figure 5.7). Event discharge exhibited a threshold similar to the one in SF5, at about 55 mm h^{-1}.

For the 2011 measurement period in SF5, K_s medians at both depths roughly coincided with the 75 % quantile of the rainfall events. Compared to the previous year, OF responses both on plots and in flowlines were higher (Figure 5.5). OF response in flowlines exhibited an intensity-dependent threshold comparable to SF5 in 2010, but at 15 mm h^{-1}, which coincided roughly with the median of rainfall events (Figure 5.5). The direct comparisons between OF response and rainfall amounts and intensities point to an influence of both rainfall variables on OF occurrence (Figure 5.7). Event discharge exhibited the same threshold at 55 mm h^{-1}.

5.3.4 Ancillary variables and Brilliant Blue profiles

While soil texture varied only marginally between SF5 and SF25, vegetation parameters showed pronounced differences (Table 5.1). SF25 exhibited more tree and less shrub cover than SF5, but an important difference occured in proportion of non-woody vegetation and litter cover. SF25 generally had very little grasses and herbs but a litter layer covering almost 80 % of the measured plots. SF5 exhibited on average more than 50 % grasses and herbs whereas litter covered only 34 % of the soil. However, variability between the five plots in SF5 was quite high, for example Plot 1 had markedly less grass and herb cover and a higher proportion of litter (Table 5.1).

Vertical Brilliant Blue profiles showed deeper uniform infiltration in SF5 than in SF25, with 4 and 5.5 cm compared to 1.5 to 2 cm depth (Table 5.2 and Figure 5.8). This higher infiltration in the top 5 cm of the soil in SF5 was also visible in the horizontal profiles: 90 % of the area at 2 cm depth was stained blue whereas in SF25 the blue area amounted only to 70 %. However, in 6 cm depth SF25 exhibited 45 % stains, SF5 only 35 %.

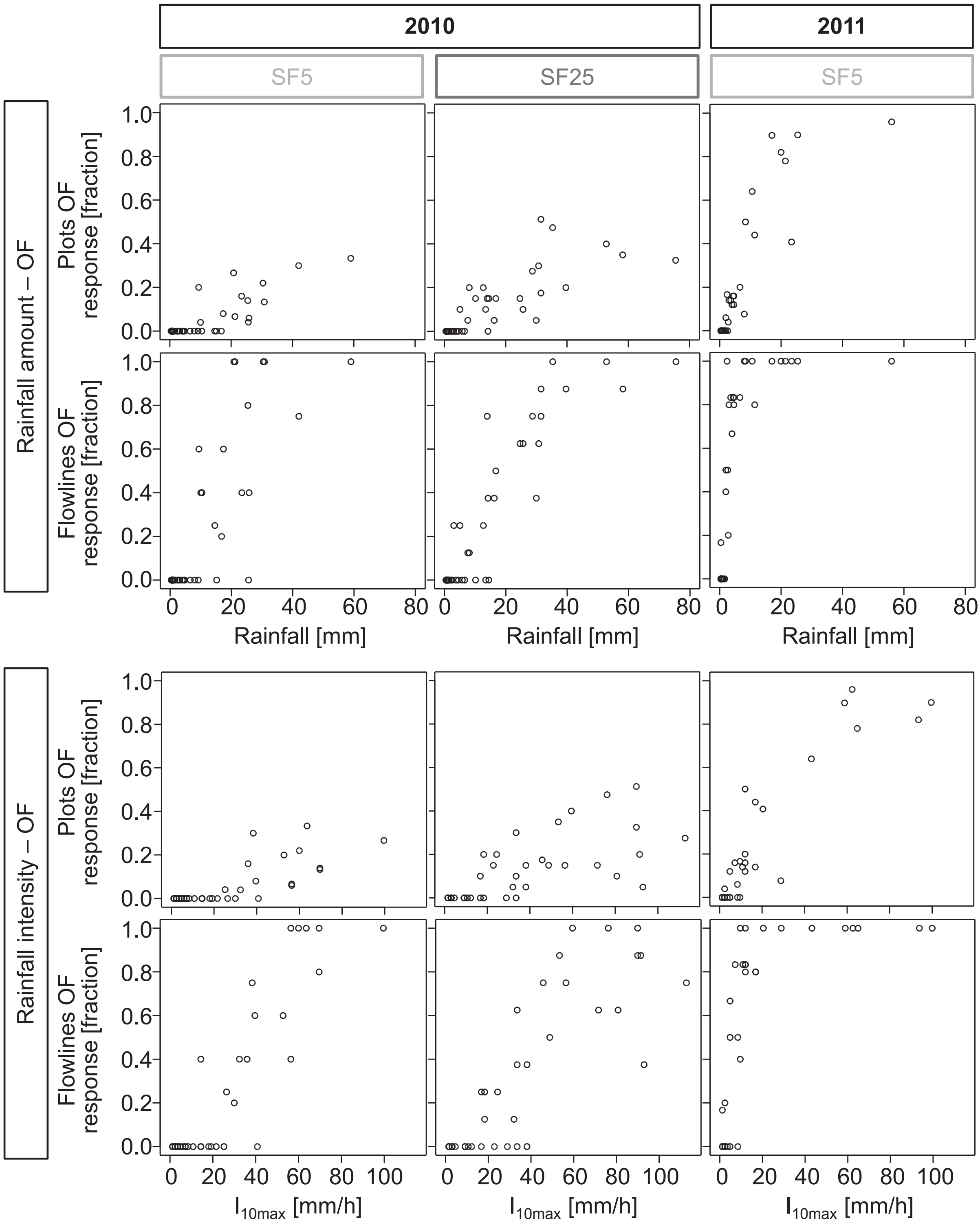

Figure 5.7: Overland flow (OF) response to rainfall amounts and 10-minute maximum rainfall intensities ($I_{10\max}$) on plots and in flowlines for both the 5 year-old (SF5) and the 25 year-old (SF25) forest in 2010 and SF5 in 2011.

Table 5.2: Results from the Brilliant Blue experiments in the 5 year-old and 25 year-old secondary forests (SF5 and SF25) on the individual plots P3, P5 (SF5), P2 and P4 (SF25). Further abbreviations are TSA for total stained area, V for vertical profiles and H for horizontal profiles in the given depths.

Parameters	Unit	SF5		SF25	
		P3	P5	P2	P4
Uniform infiltration depth	[cm]	5.5	4	2	1.5
Maximum infiltration	[cm]	71.5	72.5	65	72
TSA-V	[%]	8.5	6	4	4.5
TSA-H (2 cm)	[%]	90	90	70	70
TSA-H (6 cm)	[%]	35	35	45	45
TSA-H (20 cm)	[%]	3	3	1	1

5.4 Discussion

5.4.1 Differences in OF response between land cover types

The lower values of K_s at the 0–6 cm depth in in the younger forest (Figure 5.3) are likely due its use as a pasture preceding the secondary succession. Cattle treading and suppression of larger woody vegetation probably caused compaction of the topsoil and reduced porosity. The reduction of K_s under pasture has been reported in various studies in the humid tropics (Godsey & Elsenbeer, 2002; Hanson et al., 2004; de Moraes et al., 2006; Zimmermann et al., 2006) and a recovery under secondary succession is believed to require more than 10 years (de Moraes et al., 2006; Zimmermann & Elsenbeer, 2008). Therefore SF5 probably still exhibited the pasture-typical low K_s values in 0–6 cm. The limited influence of land-use change on the lower soil depth is consistent with a study on K_s recovery on a regional scale in the same area (Hassler et al., 2011b) and with other studies in the humid tropics (Godsey & Elsenbeer, 2002; Zimmermann & Elsenbeer, 2008; Zimmermann et al., 2006). The higher spatial variability in SF25 K_s values points towards higher contrasts in processes affecting soil structure in the older forest.

From our finding that OF on plots does not vary with K_s when comparing the two forests, we conclude that, for our study sites, OF occurrence cannot be explained by K_s directly but seems to be more complex (Figure 5.3). The similar median but larger range in OF response on SF5 plots in 2011 points towards a local intensification of OF generation, while generally its land cover-specific controls remain. One reason for this change might be the higher antecedent soil moisture because of many small rainfall events during the measurement periods, however, this seems to affect only part of the OF generation process in SF5, causing the larger range of OF response (Figure 5.6).

The higher OF response in flowlines compared to plots, for both forests and both years (Figure 5.3) might originate in the development of flowlines. Flowlines are preferential drainage pathways in the catchment and they are incised into the surrounding slopes. Consequently, they are in fact located in deeper soil layers with lower K_s values. Water transport within flowlines may be additionally self-reinforcing by clogging of pores with suspended material. The contrast between flowline K_s and the surrounding areas has also been observed in an old-growth tropical rainforest in Panama where an increase in K_s with distance from flow lines resulted in larger OF response within and adjacent to flowlines (Zimmermann et al., 2013). Furthermore, flowlines form convergent structures within the catchment, thus little OF response on slopes might result in measurable response when collected in flowlines.

5.4.2 Spatial variability within land cover types

The large K_s variability within both forests (Figure 5.4) indicates a strong influence of small-scale processes governing K_s (Hassler et al., 2011a; Si & Zeleke, 2005; Zimmermann & Elsenbeer, 2008). Differences between plots in SF25 but not in SF5 point to a dominance of small-scale influences on soil structure in SF5, whereas SF25

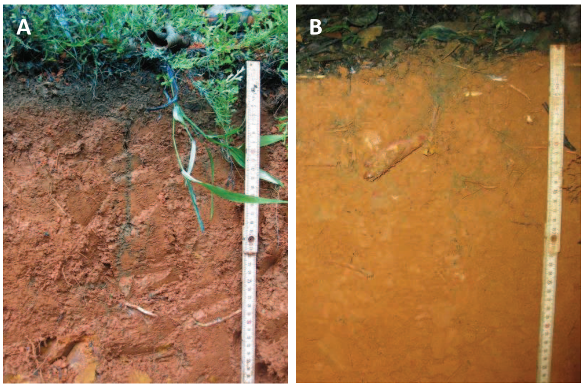

Figure 5.8: Exemplary Brilliant Blue profiles for SF5 (A) and SF25 (B). The photograph in (B) had to be taken using artificial lighting, therefore the colours differ from the true colours.

might show some larger-scale underlying pattern. A reason could be more differentiation within older forest stands in contrast to more uniform smaller-scale influences in younger forests. For example, the secondary forest in Chapter 2 (Hassler et al., 2011b) also showed an increase in variability between plots with age (Figure 2.3).

OF response on plots seems to be related to K_s values in SF25, however, this link is lacking in SF5. We attribute this to the presence of considerably more ground-covering vegetation in SF5 (Table 5.1). This vegetation layer, consisting of mainly moss of 20 cm in height (included in the 'herbs' category) together with some grasses and herbs, may promote water transport along stalks and roots into the very top layer of the soil which would resemble the infiltration depth of around 5 cm that we see in the Brilliant Blue profiles (Figure 5.8). Our method of K_s sampling vertically on levelled ground does not resemble true depths of 0–6 cm and 6–12 cm in sloping terrain as a certain sampling area needs to be cut into the slope to extract the soil cores. Consequently, we might miss most of the possible infiltration-enhancing influence of the ground-covering vegetation with our K_s sampling method, explaining the missing link between K_s and OF.

5.4.3 Event-based comparison of rainfall, OF, K_s and event discharge

Event-based analysis of the OF-generating processes is discussed for each forest and year separately whereas event discharge is jointly summarized afterwards due to a lack of land cover-specific or temporal differences.

In SF5 in 2010 the rainfall intensity-driven OF response in flowlines (Figure 5.5, Figure 5.7) is likely caused by HOF within the flowlines themselves as well as their adjacent areas. As already described above, flowlines cut into soil layers of lower K_s, accumulate water from the surrounding areas as a convergent structure and might be self-sustaining via clogging of pores with suspended particles. Even in undisturbed forest there is evidence of a gradient of increasing K_s between flowlines and surrounding areas (Zimmermann et al., 2013). Thus flowlines and their adjacent areas can act as a source of HOF when rainfall intensities exceed the intensity threshold given by the low K_s values. With intensities higher than this threshold the flowlines then would be always responding, similar to SF5 in 2010 at about 60 $\mathrm{mm\,h^{-1}}$.

OF response on plots is less pronounced than in the flowlines and does not seem to be conclusively correlated to either rainfall intensities or amounts (Figure 5.5, Figure 5.7). The processes governing OF generation are probably linked to the vegetation cover which is also responsible for the decoupling from K_s values. Water infiltration into the top soil layer is aided by the vertical structure of stalks and structure-forming effects of roots as seen in the Brilliant Blue profiles; however, K_s values in shallow soil depths are very small as a relic from the former pasture use. Consequently, the very top layer can become saturated and additional rainfall would lead to SOF irrespective of rainfall intensity. However, the ground-covering vegetation is also prone to intercepting rainfall (Miyata et al., 2009), thus a response might be delayed because of a combination of interception and wa-

Figure 5.9: Direct comparison of overland flow on plots and in flowlines of both the 5 year-old (SF5) and the 25 year-old (SF25) secondary forest in 2010, and for SF5 in 2011.

ter storage in the top soil layer. Lateral transport within this top layer might also be imaginable and could be another part of the generally low OF detection on the SF5 plots in 2010. As flowlines in SF5 are relatively short and active gullies, which are not sampled within this design, reach far upslope, there might be a direct contribution of these lateral flows from plots to gullies whereas the flowlines themselves are not as well connected to the plot-like areas and exhibit more local HOF generation. This could explain the partial independence of flowline response from plot response which one could infer from the occurrence of the intensity-dependent threshold for flowlines but not for plots (Figure 5.5) and from the direct comparison of OF response on plots and in flowlines as we did in Figure 5.9.

In SF25 OF response on plots seems more likely to be driven by rainfall amounts (Figure 5.5, Figure 5.7) than by rainfall intensities. This points towards SOF as the dominant process because a saturated topsoil layer would result in OF irrespective of rainfall intensity. The K_s cumulative frequency distribution supports this as OF response roughly starts at rainfall intensities close to the 6–12 cm K_s values. However, from the comparison between K_s and OF on the individual plots we see a relation between K_s and OF. If OF is generated by saturation, OF should not be dependent on K_s values. One explanation for this discrepancy might be the effect of the tree canopy in SF25 which is denser than the canopy in SF5. Rainfall amounts and intensities are altered when passing vegetation because of evaporation, interception and redistribution processes within the canopy (Keim et al., 2006a,b; Park & Cameron, 2008; Zimmermann et al., 2009). Our event analyses however are related to bulk precipitation and not throughfall, therefore a possible dependence on throughfall intensities cannot be determined with the available data set.

Brilliant Blue profiles in SF25 exhibited very shallow uniform infiltration (Table 5.2), seemingly contradictory to a) the assumption of SOF above the 6–12 cm soil depth as dominant process and b) the relation between OF and K_s as seen in the individual plot comparison (Figure 5.4). The following more detailed analysis of differences between the plots might yield possible solutions to this disagreement. The dependency of OF on rainfall amounts that we see in the direct comparison between these two parameters (Figure 5.7) is not equally pronounced for all plots as SF25 shows large spatial variability. In direct comparsions for individual plots (graphs not shown here), only P1 and P2 seem to show this relation, leading to the impression of a link between OF and rainfall amount whereas the relation for P3 and P4 is weak or non-existent. An interpretation of this could be viewing SF25 as a mosaic of areas with higher K_s values close to the surface, where water can temporarily infiltrate and would suppress OF, intermingled with areas of near-surface lower K_s with less uniform infiltration but possibly some infiltration via macropores which are prevalent in old forests. The observation of overland flow during Brilliant Blue application with an intensity of 40 mm h^{-1} in P2 (Figure 5.10) seems contradictory to the median K_s at roughly at the same value on this plot. As P2 shows large variability within the plot (Figure 5.4), we might have coincidentally targeted one of the lower K_s locations with the Brilliant Blue experiment. However, this hypothesis cannot be verified from our data because the K_s measurements for estimating mean K_s of the plots were conducted in the previous year and at different locations.

Figure 5.10: Observation of overland flow (red arrow) during Brilliant Blue spraying of a plot (red lines) in the 25 year-old forest. The blue colour upslope and to the sides of the plot represent the fuzzy margins of the spraying, however, the downslope blue pattern was not sprayed directly.

The infiltration depth at the Brillant Blue location in P2 in SF25 suggests that a saturation of the 0–6 cm layer resulting in SOF is unlikely; rather, saturation of only the top 1–2 cm, or something more comparable to HOF might have given rise to the Blue OF observation (Table 5.2). In SF25 the soil is covered to a large extent by litter which might additionally promote short-distance lateral transport. Such ”litter flow” has been reported in Japanese cypress forests (Gomi et al., 2008; Sidle et al., 2007), northern hardwood forests (Brown et al., 1999) and laboratory experiments (Guevara-Escobar et al., 2007). However, in the tropical forests of this study, the litter layer decays markedly during the course of the rainy season and, as the vegetation data shows, the litter cover is not continuous (Table 5.1). This cover might, however, initiate a lateral OF component or bridge high-K_s patches. Finally, an explanation for the discrepancy between the link of K_s and OF response and shallow Brilliant Blue infiltration might also originate in only having duplicated the Brilliant Blue experiment within each forest. If K_s and infiltration patterns are as highly spatially variable as the data suggests, we might have sampled two sites with low K_s close to the surface by coincidence, as for each plot we only sample one of its 900 square metres. P2 generally shows lower K_s values (Figure 5.4), so sampling a site at the lower end of the range seems probable. The other plot that was sampled, P4, has lower OF response but a very large range of K_s values (Figure 5.4). Generally, OF generation on these plots seems to be a complex interplay of different vegetation influences and spatial patterns of soil characteristics.

Flowlines in SF25 also appear to be driven by rainfall amounts and do not exhibit a clear intensity threshold as they do in SF5 (Figure 5.5). If the closed canopy in SF25 really smoothes or temporally and spatially alters rainfall intensities, this process would naturally affect OF response in flowlines similarly to the OF response on plots. An alternate interpretation of the differences in flowline OF response between SF5 and SF 25 lies in the spatial heterogeneity of SF25 and the flowline characteristics. Flowlines here are longer compared to SF5 and span a larger part of the slope because gullies are not as long and pronounced. This might induce a better connection between plot areas and flowlines, resulting in similar reactions to rainfall input. If the catchment exhibits the large spatial variability that we assume, the areas with more permeable topsoil and more macropores might exhibit deeper percolation of water and consequently less response in the corresponding flowline.

In 2011, OF response in SF5 was higher than in 2010, both on plots and in flowlines and possibly related to both rainfall amounts and totals (Figure 5.5 and Figure 5.7). Given the many small and low-intensity rainfall events during that year's measurement period (Figure 5.6) it seems likely that a generally higher soil moisture content lead to an intensification of both prevalent processes of OF generation within this catchment. On the one hand, it may have led to faster saturation on plots, inducing more SOF for otherwise similar rainfall events. On the other hand, higher soil moisture values could lead to more OF converging in the flowlines which in combination with the flowline-inherent HOF could lead to complete flowline response already at lower rainfall intensities. Additionally, in 2011 two long flowlines were sampled which would probably benefit more from OF generated on the plots.

Event discharge in the two catchments and for both years shows a threshold after which higher values (>0.5 mm) occur (Figure 5.5). One would expect a dependency of discharge on OF in flowlines as they are the preferential drainage lines leading to the gullies in the catchments. However, the discharge threshold does not relate to thresholds for OF in flowlines. As our criterion for a flowline to be active is not strict (only more than one OFD needing to respond), connectivity of flowlines might well be disrupted during the smaller rainfall events and a certain amount would be necessary to connect the whole flowline to the gully and the stream.

5.4.4 Implications for studies of near-surface hydrological flowpaths

The complex interplay of land cover-dependent spatial variability of soil parameters and vegetation influences resulted in a combination of OF-generating processes such as HOF and SOF, specific to each of the two studied forests. The accepted pedological approach of comparing K_s values to rainfall intensities (e.g. de Moraes et al., 2006; Ziegler et al., 2004; Zimmermann et al., 2006) did not prove to be suitable in this study because the actual OF generating processes were more complex. Similarly, from a hydrologist's perspective, analysis mainly based on stream flow might could describe the whole story of OF generation for our study sites because for many rainfall events, intermittent OF might occur without triggering stream flow response (Gomi et al., 2010; Miyata et al., 2010). Yet these small OF events might influence surface structure, sediment transport and nutrient redistribution within the catchment.

The spatial variability in K_s and in OF occurrence has been highlighted in previous studies (Germer et al., 2010; Loos & Elsenbeer, 2011; Miyata et al., 2010; Sobieraj et al., 2002; Zimmermann & Elsenbeer, 2008) and is also apparent in our data. However, few studies combine detailed assessment of K_s and OF (Godsey et al., 2004; Zimmermann et al., 2013), and especially in the hydrological approach, a spatially explicit examination of K_s is often lacking (Biggs et al., 2006; Chaves et al., 2008; Mun oz Villers & McDonnell, 2012). Our study supports the importance of spatial variability for OF-generating processes, but further data collection is required in order to determine the major processes and controls more accurately. Nevertheless, from the existing data, we were able to formulate process hypotheses, and to determine future data needs for a comprehensive process study.

A comprehensive study of land use-specific OF generation should include both the pedological and hydrological approach in a spatially explicit way, combined with measurement of vegetation characteristics. In order to further elucidate the processes within our study, several approaches would be useful. For instance, the effect of canopies of secondary forests of various ages on throughfall amounts and intensities is currently investigated by members of our working group and should give insight into true dependencies of OF on rainfall characteristics. The spatial variability of flow paths could be assessed with more Brilliant Blue infiltration experiments. Measuring infiltrability on both plots and flowlines with rainfall simulators or infiltrometers could elucidate flow path processes at the atmosphere-soil-interface. These processes are probably highly influenced by the ground-covering vegetation, therefore knowledge of infiltration processes would be especially suited to complement the K_s assessment for OF generation. Estimates of unsaturated hydraulic conductivity would help to explain infiltration and percolation during rainfall events, and lateral water movement in the subsoil could be estimated via trench studies. Finally, an assessment of connectivity in flowlines and its influence on stream flow would close the last gap in the assessment of flow path processes in these catchments.

5.5 Conclusions

Overland flow generation in the two examined forests are the result of a complex interplay of land cover-dependent spatial variability of soil parameters and vegetation influences and cannot be explained by K_s values and rainfall intensities alone. Processes generating OF differ between plots and flow lines and are specific to each land cover type and are spatially variable. For a truly comprehensive assessment of near-surface hydrological flow paths within a given land use, a combined examination of soil hydraulic properties, vegetation parameters, rainfall and throughfall characteristics and OF measurements seems necessary. Especially spatial patterns of K_s and OF should be carefully considered by including measurements of variability on different scales and also related to structures such as flowlines. Assessments of subsurface flow as well as discharge response and supporting dye tracer studies for flow path description would complement the picture.

5.6 Acknowledgements

The Agua Salud Project (ASP), in which this study was undertaken, is a collaboration among the Panama Canal Authority (ACP), the National Environmental Authority of Panama (ANAM), and the Smithsonian Tropical Research Institute (STRI). It is financially supported by the HSBC climate partnership (HCP), with additional funding from STRI, ACP, the Frank Levinson family foundation, and the Motta family foundation.

Discharge data were made available in a pre-analysed form by the Hydrology group of the Agua Salud Project, Fred Ogden, Trey Crouch and Guy Litt from the University of Wyoming. Finally, the field work of this study would not have been possible without the help of a number of interns and Bachelor students from the University of Potsdam, as well as local support by several ASP staff and people from the village El Giral.

Chapter 6

Summary and Conclusions

Land use change in the humid tropics has been shown to alter soil properties such as Ks and resulting water flow paths. The main research aim of this dissertation was to elucidate the mechanisms of this process by conducting studies of sources of variability of K_s, implications for K_s monitoring and the effects on near-surface hydrological flow paths. Within this chapter, I summarize the research results from the previous chapters, point out their limitations and give suggestions for further research within this topic.

6.1 Summary of research results

Specifically, I address the following questions: How do the different chapters relate to the overall research question? How suitable was the applied methodology? What was the outcome of the studies? And, how do the results fit into the bigger picture of what we learn about K_s?

6.1.1 Sources of variability of saturated hydraulic conductivity

Variability in K_s is strongly related to variability in soil structure. The two main influences on soil structure are the static, soil-inherent influence of, for example, soil texture and clay mineralogy and the dynamic, land use-dependent influence of, for instance, biotic activity and organic carbon content. The transect study in Chapter 3 examined K_s variability with regard to the static influences of soil types as deduced from soil map units on a range of spatial scales, applying a wavelet analysis. As the transect was located entirely within old-growth tropical rainforest, land cover was assumed constant for all soil map units. The results provide some evidence that K_s might vary between soil map units; however, small-scale effects of land cover-driven biotic influences within the forest seem dominant. The wavelet analysis proved appropriate for studying spatially dependent variability of K_s and emphasised the importance of considering dynamic influences especially for modelling purposes based on soil maps.

In contrast to studying static influences on various scales in Chapter 3, the dynamic influence of land cover on a regional scale was examined in Chapter 2. Soils in the area are believed to vary little with respect to texture; land cover, however, consists of a mosaic of pastures, secondary forests of different ages, agricultural lands and timber plantations. The study focused on the process of K_s recovery under secondary succession, by sampling replicates of the same forest age classes distributed across the region in a space-for-time approach. This design avoids pseudo-replication and thus provides better transferability of the results. K_s was shown to recover after pasture abandonment although variability between replicates was high. Results indicated a recovery time of more than 8 years, which is consistent with other studies, however, in this study, recovery was limited to the top 6 cm of the soil. By inference, vertical water percolation may be impaired throughout all examined the secondary forests, thus the extent to which the reported K_s recovery leads to a change in water flow paths remains unknown.

In summary of both Chapters 2 and 3, land use has been shown to affect K_s variability considerably at differing scales, small-scale biotic processes dominating K_s variability on the transect, and secondary succession recovering K_s values back to levels of old-growth forests. However, land use-dependent large spatial variability and anisotropy of K_s may lead to complex effects on near-surface hydrological flow paths.

6.1.2 Implications for monitoring

A drawback of many studies examining the effects of land use change on K_s is the application of a space-for-time approach to compare the stages of change. While this approach is convenient within the usual project finance timeframes and with sufficient replication also yields acceptable results, it cannot replace true chronosequences in addressing the mechanisms of change. However, monitoring change of a soil variable such as K_s requires careful thought about the most suitable sampling design in order to be able to detect changes both accurately and efficiently.

The case study of K_s monitoring in Chapter 4 assessed the efficiency of four different sampling designs to estimate the spatial mean of K_s. The applied Rotational Stratified Simple Random Sampling (rotStRS) did not provide a gain in efficiency compared to simpler designs. In some cases stratification and re-sampling of locations inherent in rotStRS even worsened estimations, suggesting Simple Random Sampling as preferred design option. The main reasons for the failure of stratification to improve efficiency were the large spatial variability of K_s at a small scale, probably representing the biotic influence, and the scarcity of large-scale structure in, for instance, vegetation or soil type, which could be represented in strata. The lack of temporal consistency between repeated samples which prevented re-sampling from yielding advantages is also attributable to high small-scale variability in K_s, in combination with the destructive sampling method. The original sampling design might have been suited to a study area with larger contrasts in land use influence which would be amenable to stratification. Similarly, if rotStRS was applied to different soil types, soil structure formation and thus K_s might be less dependent on vegetation influence and more on soil texture, possibly leading to less small-scale variability. In such soils re-sampling of locations might be beneficial for detecting a temporal trend in K_s.

The lessons learned from this study are transferable to other soil monitoring schemes. To select the most appropriate sampling design for K_s prior knowledge about the spatial variability and the temporal consistency is required. An efficient design then facilitates detection of temporal changes of mean K_s as expected during land-use change.

6.1.3 Effects on near-surface hydrological flow paths

The main goal in studying sources of variability of K_s and temporal changes is to assess alterations in near-surface hydrological flow paths which presumably occur during land-use change and will consequently affect regional water cycles, ecosystem processes and human livelihoods. One possibility to assess these flow paths, a ”pedological approach”, is to compare the observed changes and patterns in K_s with prevalent rainfall characteristics and to estimate possible consequences for infiltration and percolation from this comparison. A ”hydrological approach” is to estimate contributions from different flow paths to stream flow directly by measuring various hydrometeorological parameters, often supported by isotope composition assessment. Combining both, as we did in Chapter 5, can provide deeper insight into the relation between K_s and near-surface hydrological flow paths under different land use types.

In this study, overland flow (OF) was the target parameter for assessing flow paths. Additionally to an extensive K_s survey, rainfall measurements, stream flow data and ancillary variables were collected to facilitate process understanding. Comparison of K_s values and rainfall intensities did not describe the observed pattern of OF. Likewise, event flow in the stream was apparently unrelated to the OF response patterns within the catchments. OF-generating processes in both of the studied catchments were a complex interplay of K_s values, land-cover influence and rainfall characteristics. Especially the spatial variability of K_s, which differed between the two land cover types because of the contrasting vegetation, seemed to govern patterns and processes of OF generation. The study showed that in order to comprehensively understand the puzzle of near-surface hydrological flow paths many meteorological, pedological, botanical and hydrological pieces must be put together.

6.2 Limitations of the presented studies

Generalising from the presented studies can only be sensible when their limitations are considered. The first limitation is of pedological nature as the studies were conducted in a small range of soil types. Although the transect encompassed soils which were representative of most of the Panama Canal Zone, the texture of all studied soils was very clayey, which influences structure formation and porosity. A greater contribution of static soil properties may therefore be expected in studies with larger contrasts in soil type.

The second limitation relates to the method of sampling and measuring K_s using undisturbed soil cores. The measurement itself, applying a constant water head, supposedly holds a maximum possible error of 5 % (Elsenbeer et al., 1992). However, several other error sources arise during the sampling of soil cores.

Firstly, taking soil samples in sloped terrain requires levelling of the soil surface. An effect of this method is that the true depth of the sample within the soil continuum is a gradient from the target depth to deeper layers, depending on the slope angle. This effect could lead to an underestimation of K_s when there is large anisotropy in the soil, or simply to misjudging the processes in the top layer because they are not completely captured by the sample. An additional study to examine this possible methodological error did not find any underestimation resulting from slope. However, the OF study suggests that processes at the soil surface are mostly not explained by K_s values in the upper soil depth. This could be alleviated by including infiltrability measurements in order to cover the processes at the soil surface.

Secondly, K_s was only examined at a very limited soil depth of 0–12 cm. The survey assessing K_s under secondary succession (Chapter 2) suggests that, in this type of soil, the 6–12 cm soil depth shows very low K_s values independent of land cover. However, the deeper layers might still be important for overall water transport, either by allowing percolation or inducing formation of perched water tables. Recent selected soil profile measurements within the study area support an almost continuous decrease in K_s with depth, thus our results of the upper soil depths are probably covering most of the relevant processes.

Thirdly, throughout the studies in this dissertation, undisturbed small soil cores were used for K_s measurement. They do not always cover the representative elementary volume (REV) necessary for K_s determination, thus additional small-scale variability might be introduced artificially. This method was used because the clayey soil of the study site render in-situ methods which involve drilling of holes unsuitable, as the hole walls become easily smeared. Undisturbed sampling of larger cores would be nearly impossible in tropical forest because of the high root density and they would also fail to properly represent the mostly very shallow topsoils. Care was taken to collect appropriate sample sizes in order to detect differences in K_s even with possible additional small-scale variability.

6.3 Suggestions for further research

The studies in this dissertation raise some questions that should be addressed for a comprehensive understanding of land use-dependent changes of K_s and their effects on hydrological flow paths within this region. An investigation of infiltrability would complete the K_s assessment, help to explain processes at the soil surface, and may facilitate interpretation of OF patterns. A more detailed assessment of vegetation influences, such as the effects of roots on soil structure formation and porosity, might also help to solve the puzzle.

Each of the studies in this dissertation highlights the importance of understanding the spatial structure in K_s at various scales. In particular, clarifying K_s patterns in space and quantifying their variation during land-use change may provide a deeper understanding of hydrological flow path activation. This will require careful sampling design.

Assessing the effect of forest regrowth on K_s and OF distribution with respect to flowline position could be achieved by measuring K_s within flowlines, similar to the study by Zimmermann et al. (2013). A comparison of OF connectivity within the flowlines of our five year-old and 25 year-old forests and the old-growth forest examined in their study, and the possible connection to stream flow is already planned as a collaboration. This could give insight into the generation and the "aging" of flowlines during forest regrowth and the consequences on the catchments' water balances.

The impact of different OF-generating processes on stream flow has so far only been touched upon. A more detailed analysis of stream flow responses within the two headwater catchments where OF was measured is planned, in collaboration with the ASP Hydrology group from the University of Wyoming. These two headwater catchments are part of larger catchments with different land use. Stream flow will be compared to these larger catchments' responses and to other catchments with different land cover in the region. This could provide insight into land use-dependent stream flow response on various scales and possibly establish the link to the OF generation measured in the headwater catchments.

Bridging the gap between point-scale measurements of K_s and modelling hydrological response on the catchment scale requires some method of upscaling. The small support of soil cores, excluding large macropores, might yield insight into soil matrix K_s at distinct locations, but for a slope or across the expanse of a catchment, the effective K_s which determines water flow paths cannot not be easily derived from point-scale K_s measurements. The spatial context of land-use influences and microtopography are likely to considerably govern water movement on these scales. Combining point-scale K_s data with more integral approaches such as geophysical techniques or remote sensing might contribute to solving this upscaling problem and can help to increase the value of point-scale K_s data for larger-scale modelling purposes. However, further research on this topic is required.

Water transport via soil cracks is a possible flow path that might be additionally relevant to include in larger-scale hydrological models. Soil cracks are both large and abundant in the region and they can remain open for up to two months in the rainy season. Water transport through these structures might be considerable and thus important for the regional water budget. A combination of stream flow analysis, flow path assessment via hydrometerological measurements and tracer studies could help to quantify the effect of these soil cracks.

Land-use change is affecting large areas in the humid tropics. Its influence on key soil hydraulic properties changes hydrological flow paths and thus affects ecosystem functions and human livelihoods. This dissertation highlights the role of K_s

in the change of hydrological flow paths, its suitability for assessing land-use effects on soils but also its limitations for inferring changes in flow paths. An interdisciplinary approach is needed to comprehensively understand the complex relationships between the influences of soils and vegetation on the water cycle during land-use change in the humid tropics.

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