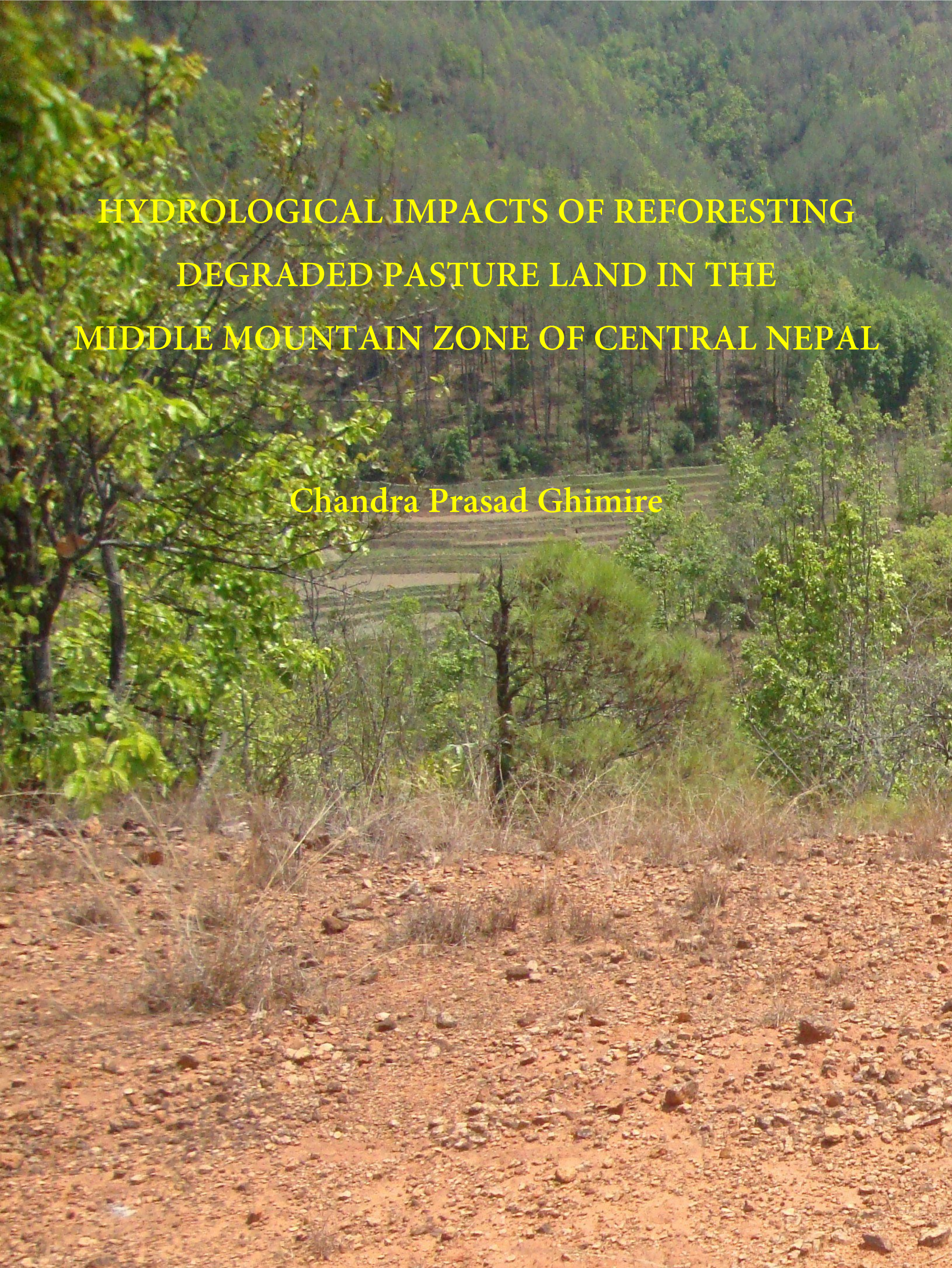
HYDROLOGICAL IMPACTS OF REFORESTING DEGRADED PASTURE LAND IN THE MIDDLE MOUNTAIN ZONE OF CENTRAL NEPAL

Chandra Prasad Ghimire



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VRIJE UNIVERSITEIT

Hydrological impacts of reforesting degraded pasture land in the Middle Mountain Zone of Central Nepal

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Dedicated to those who made this study possible

and also to those who opened my eyes

"All search for happiness is misery and leads to more misery."

- Shri Nisargadattta Maharaj

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Chapter 1

General introduction

1.1 Forest and water

The once widely held belief that the presence of a good forest cover *invariably* ensures a steady flow of water during prolonged rainless spells due to the forest 'sponge' effect in which wet season rainfall is absorbed and stored for subsequent gradual release during the dry season came under severe scrutiny in the early 1980s. Bosch and Hewlett (1982) reviewed the results from nearly one hundred paired catchment experiments around the globe and concluded that 'no experiments in deliberately reducing vegetation cover caused reductions in water yield, nor have any deliberate increases in cover caused increases in yield'. As such, the removal of a dense forest cover was seen to lead to *higher* streamflow totals, and reforestation of open lands to a *decline* in overall streamflow. These initial results have been confirmed by several subsequent reviews, both for the (warm-) temperate zone (Stednick, 1996; Brown et al., 2005; Jackson et al., 2005) and the humid tropics (Bruijnzeel, 1990; Grip et al., 2005; Scott et al., 2005). The fact that the bulk of the change in streamflow associated with such experiments was observed during conditions of baseflow (Bosch and Hewlett, 1982; Bruijnzeel, 1989; Farley et al., 2005) at first sight contradicted the reality of the forest sponge concept, and its very existence became questioned (Hamilton and King, 1982; Calder, 2005). Indeed, since the early reviews of Bosch and Hewlett (1982) and Hamilton and King (1982), many have emphasised the more 'negative' aspects of forests, such as their higher water use (Figure 1.1) or inability to prevent extreme flooding (e.g. Calder, 2005; FAO-CIFOR, 2005; Jackson et al., 2005; Kaimowitz, 2005) rather than focus on the positive functions of a good forest cover, including the marked reduction of surface erosion and shallow landslip occurrence (Sidle et al., 2006), improved water quality (Bruijnzeel, 2004; Wohl et al., 2012), moderation of all but the largest peak flows (Roa-García et al., 2011; Ogden et al., 2013) or carbon sequestration (Farley et al., 2005; Malmer et al., 2010).

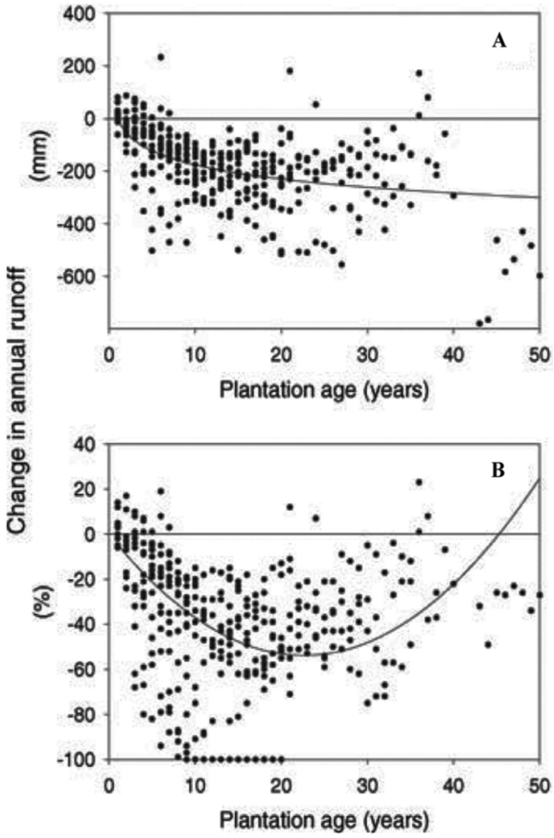

Figure 1.1: Changes in streamflow in mm: (A) and proportion (%) (B) as a function of plantation age (Source: Jackson et al., 2005).

At the same time, there is ample evidence of severe and widespread soil degradation after tropical forest conversion to unsustainable forms of land use (Figure 1.2) (Oldeman et al., 1991; cf. Bai et al., 2008). This is often accompanied by strongly increased stormflow volumes during times of rainfall (Bruijnzeel and Bremmer, 1989; Fritsch, 1993; Chandler and Walter, 1998; Zhou et al., 2002) and shortages of water during extended dry periods (Bartarya, 1989; Madduma Bandara, 1997; Bruijnzeel, 2004; Tiwari et al., 2011). This effectively reflects the loss of the former ‘forest sponge’ (cf. Roa-García et al., 2011; Krishnaswamy et al., 2012; Ogden et al., 2013) through critically reduced replenishment of soil water and groundwater reserves due to lost surface infiltration opportunities, despite the fact that the lower water use of the post-forest vegetation should have produced higher stream flows throughout the year (Bruijnzeel, 1986, 1989).

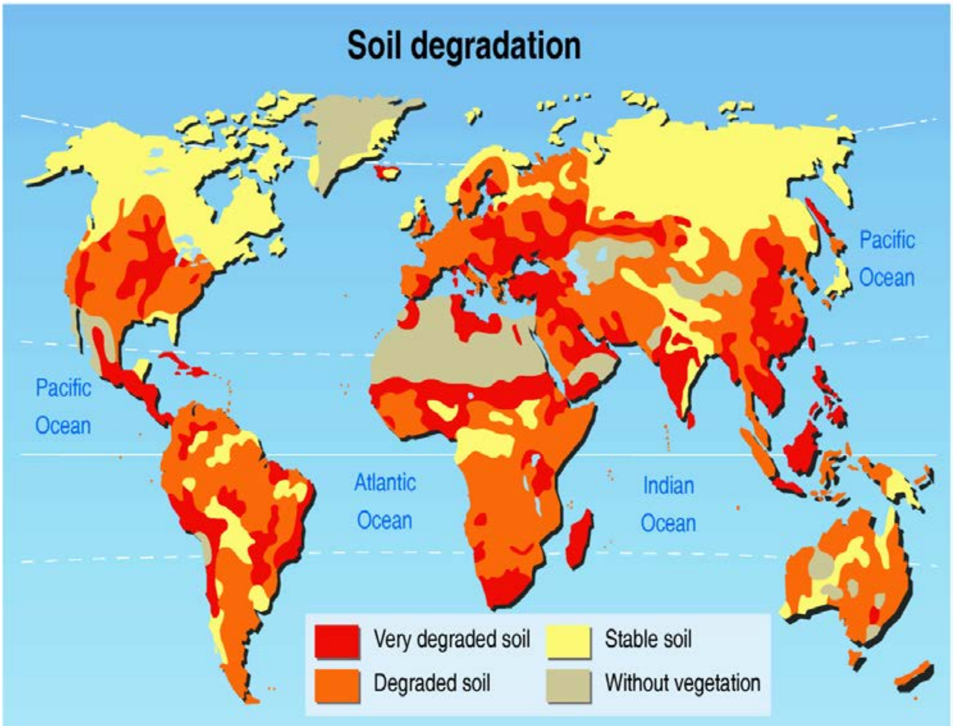

Figure 1.2: Global status of soil degradation (Source: UNEP/GRID-Arendal, 1997)

Reforestation of degraded land in the tropics is often conducted in the expectation that disturbed streamflow regimes (commonly referred to as the 'too little – too much syndrome': Bartarya, 1989; Schreier et al., 2006) will be restored by the increased rainfall absorption afforded by soil improvement after tree planting (Scott et al., 2005; cf. Ilstedt et al., 2007). At the same time, the water use of fast-growing tree plantations tends to be (much) higher than that of the degraded vegetation they typically replace – particularly where the trees have access to groundwater (Calder, 1992; Kallarackal and Somen, 1997; cf. Figure 1.1). Furthermore, major improvements in the infiltration capacity of severely degraded soils after tree planting may easily take several decades of undisturbed forest development to fully materialise (Figure 1.3) (Gilmour et al., 1987; Scott et al., 2005; Bonell et al., 2010). As such, reforesting degraded pasture or shrub land may well cause already diminished dry season flows to become reduced even further, depending on the net balance between increases in soil water reserves afforded by improved infiltration versus decreases caused by the higher plant water uptake (the so-called 'infiltration trade-off' hypothesis; Bruijnzeel, 1986, 1989).

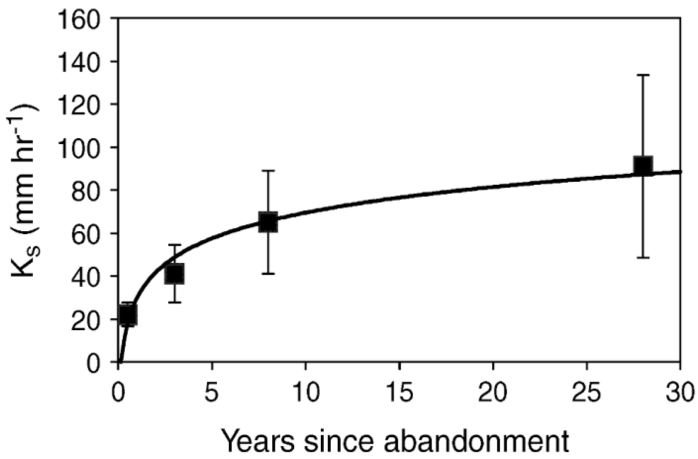

Figure 1.3: Rebuilding of soil saturated hydraulic conductivity (K_s) under secondary vegetation as a function of years since land abandonment (Source: Giambelluca, 2002).

Although the overwhelming majority of paired catchment experiments have shown major decreases in baseflow volumes after the establishment of a tree cover on non-forested land (Farley et al., 2005; cf. Figure 1.1), this does not disprove the possibility of enhanced dry season flows after reforestation. As pointed out by Bruijnzeel (2004) and Malmer et al. (2010), only three out of the 26 paired catchment studies of the hydrological impacts of reforestation reviewed by Jackson et al. (2005) and Farley et al. (2005) were located in the tropics (where soil degradation tends to be more acute; Oldeman et al., 1991; Figure 1.2) while none of the experiments represented degraded soil conditions. In other words, no positive effects of reforestation on soil water intake capacity could become manifest and the observed reductions in water yield simply reflected the higher water use of the trees compared to the grasses and scrubs they replaced (cf. Trimble et al., 1987; Waterloo et al., 1999; Scott and Prinsloo, 2008).

Nevertheless, although direct evidence for the 'infiltration trade-off' hypothesis of Bruijnzeel (1986) seemed to be lacking until recently (see below), several hillslope-scale or small-basin studies of the hydrological impacts of reforesting severely degraded land (discussed in some detail by Scott et al., 2005) observed reductions in stormflow volumes that were likely to exceed the estimated corresponding increases in vegetation water use (cf. Zhang et al., 2004; Sun et al., 2006). Unfortunately, as the catchments involved in these experiments were either too small or too leaky to sustain perennial streamflow, the expected net positive effect of the forestation on dry season flows could not be ascertained in these cases (Scott et al., 2005; cf. Chandler, 2006). However, recently published concurrent reductions in stormflow response to rainfall, and positive trends in baseflow over time since reforesting severely degraded land in South China (Zhou et al., 2010) and South Korea (Choi and Kim, 2013), or along a forest degradation–recovery gradient in the western Ghats of Southwest India (Krishnaswamy et al., 2012, 2013) suggest the infiltration trade-off mechanism to be at work under conditions of advanced land degradation and high rainfall.

1.2 Land degradation, reforestation and dry season flows in the Lesser Himalaya of Nepal

Traditionally, the Middle Mountain Zone (or Lesser Himalaya) constitutes the most densely populated part of the Himalaya, both in Nepal (Hrabovszky and Miyan, 1987) and India (Singh et al., 1984). High population pressure and heavy demands for forest products have led to widespread forest removal (Dobremez, 1976; Tucker, 1987; Singh and Singh, 1992), subsequent land degradation (Mahat et al., 1986; Mahat et al., 1987; Tiwari, 1988) and deterioration of streamflow regimes (Bartarya, 1989). Until comparatively recently, it was widely assumed that deforestation and overgrazing in the Himalayas were primarily responsible for the large-scale flooding and sedimentation experienced in the plains of northern India and Bangladesh (e.g., Eckholm, 1976; Nautiyal and Babor, 1985; Myers, 1986). Whilst this view is no longer tenable in the light of subsequent scientific evidence demonstrating the comparatively limited influence of land use on these *large-scale* hydrological phenomena (Bruijnzeel and Bremmer, 1989; Ives and Messerli, 1989; Hofer, 1993; cf. Gardner and Gerrard, 2003; Hofer and Messerli, 2006), the *local* adverse hydrological effects of advanced land degradation, such as accelerated erosion, enhanced peak discharges and reduced dry season flows (Bartarya, 1989; Bruijnzeel and Bremmer, 1989) required remedial action (cf. Negi et al., 1998). Land surface degradation in the zone has often progressed to a point where rainfall infiltration has become seriously impaired and overland flow is rampant (Bruijnzeel and Bremmer, 1989; Gerrard and Gardner, 2002; cf.

Ghimire et al., 2013a, b), with reduced dry season flows as a sad result (Bartarya, 1989; Tiwari et al., 2011; Tyagi et al., 2013). Moreover, the rivers originating in this part of the mountain range are mostly rain-fed and thus do not benefit from increased water yields from melting glaciers under a changing climate scenario (Bookhagen and Burbank, 2010; Andermann et al., 2012; Immerzeel et al., 2013). At the same time, forest land had to provide a variety of goods and services (such as timber and fuelwood, plus non-timber products including water, fodder and litter for animal bedding) to local communities in support of their subsistence farming systems (Campbell and Mahat, 1975; Mahat et al., 1987). A major reforestation programme was initiated in the early 1980s in the Middle Mountains of Central Nepal (Shepherd and Griffin, 1984) to address this multitude of issues. By the year 2000, ~23000 ha of degraded pasture and shrub land in the Districts of Sindhupalchok and Kabhre Palanchok (Central Nepal) had been converted to evergreen plantation forests (mainly *Pinus roxburghii* and *P. patula*) (source: District Forest Offices at Kabhre Palanchok and Sindhupalchok, unpublished data, 2010). The original vegetation of these areas consisted of forests of semi-evergreen broad-leaved species and pines (mainly *P. roxburghii*) (Dobremez, 1976). In addition, on-farm tree planting and natural regrowth on abandoned fields increased during this period (Gilmour and Nurse, 1991; Paudel et al., 2012) such that a recent survey reported a marked increase in both forest area and quality across Nepal's Middle Mountains over the last two decades (HURDEC Nepal and Hobley, 2012). Nevertheless and as also reported for other parts of the world (Trimble et al., 1987; Waterloo et al., 2000; Jackson et al., 2005; Scott et al., 2005), farmers in Central Nepal have expressed concerns about diminishing streamflow volumes following the large-scale planting of the pines (República, 2012). Such considerations assume added importance under the strongly seasonal climatic conditions prevailing in the Middle Mountain Zone where ~80% of the annual rainfall is delivered during the main monsoon (June–September; Merz, 2004; Bookhagen and Burbank, 2010) and where water during the dry season is already at a premium (Merz et al., 2003). However, there is a dearth of sound experiments in the Himalayan region to ascertain the hydrological effects of land use change (reviewed by Bruijnzeel and Bremmer, 1989; Negi, 2002) and indeed for the tropics in general with respect to the extent to which reforestation of degraded land can indeed improve or even restore diminished dry season flows (Scott et al., 2005; Zhou et al., 2010; Krishnaswamy et al., 2013; Choi and Kim, 2013).

1.3 Research objectives

The above considerations formed the rationale for a process-based study on the impacts of reforesting severely degraded grassland on dry season flows near Dhulikhel in Central Nepal, which is the first non-watershed-scale initiative of its kind in the Lesser Himalaya and indeed in the (sub-)tropics in general (Scott et al., 2005; cf. Krishnaswamy et al., 2012, 2013; Choi and Kim, 2013). The main objective of this study, therefore, is two-fold. Firstly, to describe and quantify the dominant hydrological processes (rainfall interception, transpiration, runoff generation, rainfall infiltration and percolation) operating under three contrasting land-cover types subject to gradually increased levels of anthropogenic pressure, viz: (i) little disturbed natural broad-leaved forest; (ii) a heavily used mature pine plantation established on former pasture land, and (iii) a severely degraded and overgrazed grassland. The second objective is to investigate the trade-off between changes in vegetation water use and soil infiltration capacity after the reforestation. In view of the difficulty of identifying catchments with a single dominant land cover type (e.g., forest or grassland) that are in addition large enough to support perennial flows (required for the evaluation of the change in baseflows during the extended dry season) in this rugged and spatially highly variable terrain, the present study opted for the ‘space for time substitution approach’ in which experimental plots with contrasting land-cover types are studied in terms of their hydrological processes. In order to achieve the above objectives the following five specific objectives have been defined:

1. To estimate the total annual interception loss (wet-canopy evaporation) from (semi-evergreen) natural broad-leaved forest and evergreen planted pine forest and to investigate the probable causes of the observed difference (if any) in interception loss between the two forests.

2. To examine the diurnal and seasonal variations in tree transpiration (dry-canopy evaporation), canopy conductance, and decoupling coefficient (*sensu* Jarvis and McNaugthon, 1986) for the natural broad-leaved forest and the planted pine forest and to quantify the total annual soil water uptake by the respective forests.

3. To investigate the effects of reforesting severely degraded grassland on field-saturated soil hydraulic conductivity, both at the surface and with depth, as well as quantify the production of overland flow during the main monsoon season for the three land-cover types.

4. To investigate the effects of long-term forest use on the development of field-saturated soil hydraulic conductivity.

5. To investigate the net trade-off between changes in vegetation water use and soil infiltration after reforesting degraded pasture land.

1.4 Outline of the thesis

This thesis consists essentially of five papers. In Chapter 2, the results of measurements and modelling of the rainfall interception loss from the natural forest and the planted pine forest are presented. The revised version of Gash's analytical model was calibrated and validated for the two contrasting forest types, representing a first for montane sub-tropical monsoonal conditions. The model is subsequently used to calculate the annual interception loss from the two forests. Moreover, the model is used to further investigate the observed difference in rainfall interception between the two forests. Chapter 3 presents the diurnal and seasonal variations of tree transpiration, canopy conductance, and decoupling coefficient for the same natural and planted forest plots using sap flow measurements and concurrent climatic and soil water observations. In addition, the transpiration totals are adjusted to allow for the inclusion of understory transpiration (where present, i.e. only in the broad-leaved forest) and combined with the annual interception losses estimated in Chapter 2 to give annual evapotranspiration totals for the two forests. Next, Chapter 4 deals with the effects of reforesting severely degraded grassland on field-saturated soil hydraulic conductivity and overland flow production. This Chapter also verifies the inferred processes of runoff generation by comparing the latter against the measured overland flow totals generated on the pasture and in the two contrasting forest types. Chapter 5 looks at the effects of multi-decadal intensive plantation forest use in the form of litter and fuelwood collection and grazing on field-saturated soil hydraulic conductivity at the hillslope scale. This work was conducted at a different site (Chautara) located some 20 km from the main research site at Dhulikhel. The Chautara pine plantations were subjected to similar soil physical research by Gilmour et al. (1987) and by re-visiting the same locations some 25 years later, a unique view of changes in surface infiltration capacity and inferred runoff response to rainfall during plantation development can be obtained in 'real time'. The various components of the site water budgets for the three land cover types are integrated in Chapter 6 which explores the trade-off between increases in vegetation water use and rain water infiltration afforded by soil improvement after reforesting severely degraded grassland. Finally, in Chapter 7 the main conclusions emerging from the current work are summarised, followed by various recommendations for future research in the area.

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Chapter 2

Rainfall interception by natural and planted forests in the Middle Mountains of Central Nepal[1]

Abstract. Measurements of gross rainfall, throughfall and stemflow were made during the 2011 rainy season in an evergreen natural forest and in a planted pine forest located in complex topography in the Middle Mountains of Central Nepal. For the period of observation (20 June to 9 September 2011), measured throughfall, stemflow and derived interception loss in the natural forest were 76.2%, 1.4% and 22.4% of incident rainfall, respectively. Corresponding values for the pine plantation were 83.0%, 0.5% and 16.5%. The revised version of Gash's analytical model of rainfall interception was calibrated and validated for the two stands, representing a first for montane sub-tropical monsoonal conditions. The results of the modeling corresponded well with observed values, provided an optimised value was used for the average wet-canopy evaporation rate. The model was subsequently used to calculate annual interception losses from the two forests. Modeled estimates of annual interception loss for the planted and natural forest were 291 mm and 319 mm and represented 19.4% and 22.6% of incident rainfall, respectively. The high interception loss from the planted forest is considered a contributory factor towards the observed decline in dry season streamflow following the reforestation of degraded hillsides in the Middle Mountains of Nepal.

[1]This chapter is based on: **Ghimire CP**, Bruijnzeel LA, Lubczynski MW, Bonell M (2012) Rainfall interception by natural and planted forests in the Middle Mountains of Central Nepal. *Journal of Hydrology*, 475, 270–280.

2.1 Introduction

Water scarcity, especially during the pre-monsoon period, is a major problem in the Middle Mountain zone (or Lesser Himalaya) of Nepal (Merz et al., 2003) and North-West India (Bartarya, 1989; Negi, 2001). The combination of steep topography, rapidly draining soils and a strongly seasonal climate delivering ca. 80% of the rainfall during the summer monsoon months renders the natural ecosystems of the region particularly fragile and thus vulnerable to disturbance (Karki and Chalise, 1995). Over time, continued population pressure in the Middle Mountains (Hrabovszky and Miyan, 1987) has led to widespread forest removal (Dobremez, 1976; Tucker, 1987)) and land degradation (Mahat et al., 1986; Mahat et al., 1987). For many years, it was widely assumed that deforestation and overgrazing in the Himalayas were responsible for the major flooding and sedimentation experienced in the Gangetic plain in northern India (e.g. Eckholm, 1976; Myers, 1986). Although it was realised increasingly during the 1980s that the dramatic tectonic, climatic and geomorphological processes responsible for the very existence of the Himalaya and its neighbouring plains were sufficiently impressive to render the *large-scale* effects of human activity insignificant in comparison, the *local* adverse hydrological effects of land degradation did need to be addressed (Bruijnzeel and Bremmer, 1989; Hamilton, 1987; Negi, 2002). Thus, an intensified effort in forest restoration in the Middle Mountains of Central Nepal was initiated in the early 1980s, which had converted some 23,000 ha of degraded pasture land and shrubland (that once carried various semi-evergreen to deciduous natural forests; Dobremez, 1976) to fast-growing evergreen plantation forest(mainly *Pinus roxburghii*) by the beginning of the 21st century. Reforestation of degraded land was expected to provide more stable streamflows due to the enhanced rainfall infiltration afforded by soil improvement after tree planting (cf. Gilmour et al., 1987). However, due to the higher water use of these fast-growing plantations compared to the degraded vegetation they typically replaced and due to the fact that major improvements in topsoil infiltration capacity may take several decades of undisturbed forest development (Gilmour et al., 1987; Scott et al., 2005; Bonell et al., 2010), reforesting degraded pasture- or shrub land may well cause already diminished dry season flows to be reduced even further (Bruijnzeel, 1989; Bruijnzeel, 2004; cf. Malmer et al., 2010). This appears to be the case in the Middle Mountains of Central Nepal where according to local farmers in the Jikhu Khola Catchment several springs have dried up completely and the yields of others have been declining since the reforestation programme (República, 2012).

Thus, quantifying the impact of the conversion of severely degraded pasture land to evergreen (pine) forests on water resources is a critical research topic for the region as well as for other regions that have undergone a similar land-cover transformation (cf. Zhou et al., 2002; Bruijnzeel, 2004; Scott et al., 2005; Krishnaswamy et al., 2012). As a first step towards quantifying the water budgets of natural and planted forests vs. degraded pasture land in the Middle Mountains of Central Nepal, this paper determines and compares the rainfall interception component of total evapotranspiration for the two forest types. To gain insight into the causes underlying any differences in interception loss between natural and planted forest, the revised version of Gash's analytical interception model (Gash et al., 1995) was applied for the first time under the monsoonal mountainous conditions prevailing in Central Nepal.

2.2 Study area

The Middle Mountain region or Lesser Himalaya, which occupies about 30% of Nepal, is home to ca. 45% of the total population (based on the 2011 population census). The region is characterised by a complex geology which has resulted in equally complex topographic, climatic and vegetation patterns (Dobremez, 1976). The geology consists chiefly of phyllites, schists and quartzites, with elevations ranging from 800–2400 m above mean sea level (a.m.s.l). Depending on elevation the climate is humid sub-tropical to warm-temperate and strongly seasonal with most of the rain falling between June and September. Rainfall varies with elevation and exposure to the prevailing moist monsoonal air masses (Merz, 2004). At higher elevations (> 2000 m a.m.s.l.), pines and broad-leaf trees are mixed, whereas at lower elevations (<1000 m a.m.s.l.) the forest is dominated by deciduous *Shorea robusta*. At intermediate elevations (1000–2000 m a.m.s.l.) a largely evergreen mixed broad-leaf forest dominated by *Schima wallichii* and various chestnuts and oaks (*Castanopsis spp.*, *Quercus spp.*) is found, with admixtures of *Rhododendron arboreum* above ca. 1500 m a.m.s.l. Due to the high population pressure, much of this species-rich forest has disappeared (<10% remaining), except on slopes that are too steep for agricultural activity (Dobremez, 1976; Merz, 2004).

The present study was conducted in the Jikhu Khola Catchment (JKC). The 111.4 km^2 JKC (27^0 35’–27^0 41’ N ; 85^0 32’ E) is situated approximately 45 km east of Kathmandu (the Capital of Nepal) along the Araniko Highway in the Kabhre district between 796 and 2019 m elevation (Figure 2.1). The general aspect of the catchment is southeast, extending from southeast to northwest. The topography ranges from flat valleys to steep upland slopes. The geology consists of sedimentary rocks that have been affected by various degrees of metamorphism. These meta-sediments include phyllites, quartzite, schists and mica-schist (Nakarmi, 2000). In general, soils in the JKC are of loamy texture and moderately well to rapidly drained (Maharjan, 1991).

The climate of the JKC is largely humid sub-tropical, grading to warm-temperate above 2000 m a.m.s.l. Mean (±SD) annual rainfall measured at mid-elevation (1560 m a.m.s.l.) for the period 1993–1998 was 1487 (±155) mm (Mertz, 2004). The main seasons experienced are the monsoon (June to September), the post-monsoon period (October to November), winter (December to February), and the pre-monsoon period (March to May). The rainy season (monsoon) begins early June and ends by late September, during which period about 80% of the total annual precipitation falls. In general, July is the wettest month with about 27% of the annual rainfall. The driest months are November to February, each accounting for about 1% of the annual rainfall (Merz, 2004). Average monthly temperatures measured at 1560 m a.m.s.l. range from 7.7 ^{0}C in January to 22.6 ^{0}C in June (Merz, 2004) whereas average monthly relative humidity varies from 55% in March to 95% in September. Strong winds are common during thunderstorms before the onset of the main monsoon, but these are momentary and average monthly wind speeds are always less than 2 m s^{-1} with slight seasonal variation. Vegetation cover in the catchment consists of 30% forest (both natural and planted), 7% shrubland and 6% grassland, with the remaining 57% largely under agriculture (Merz, 2004). The JKC was subjected to active reforestation until 2004 as part of the Nepal-Australian Forestry Project. Two forest plots, one in natural forest (NF) and the other in mature planted pine forest (PF), were selected at comparable elevations within the JKC for the current study. The NF was located on a north-west facing slope and the PF on a south-west facing slope. Both plots were situated on steep slopes (20–25^o) in complex topography and were ca. 2200 m apart (Figure 2.1).

The NF plot had a surface area of 0.25 ha (50 m x 50 m) and was located at an elevation of 1500 m a.m.s.l. near the centre of an area of remaining natural forest of ca. 20 ha. The NF had a well-developed understory and was floristically highly diverse. An inventory of trees made in May 2011 within a 25 m x 28 m sub-plot (where throughfall and stemflow measurements were concentrated) counted 123 trees with a diameter at breast height (DBH, measured at 1.30 m) ≥ 5 cm. The corresponding stem density, mean DBH and basal area were 1869 trees ha^{-1}, 13.6 ± 4.4 cm, and 27.1 m^2 ha^{-1}. Values of stem density and basal area were corrected for slope gradient to allow the proper calculation of stemflow volumes later on (see Section 2.3.3).

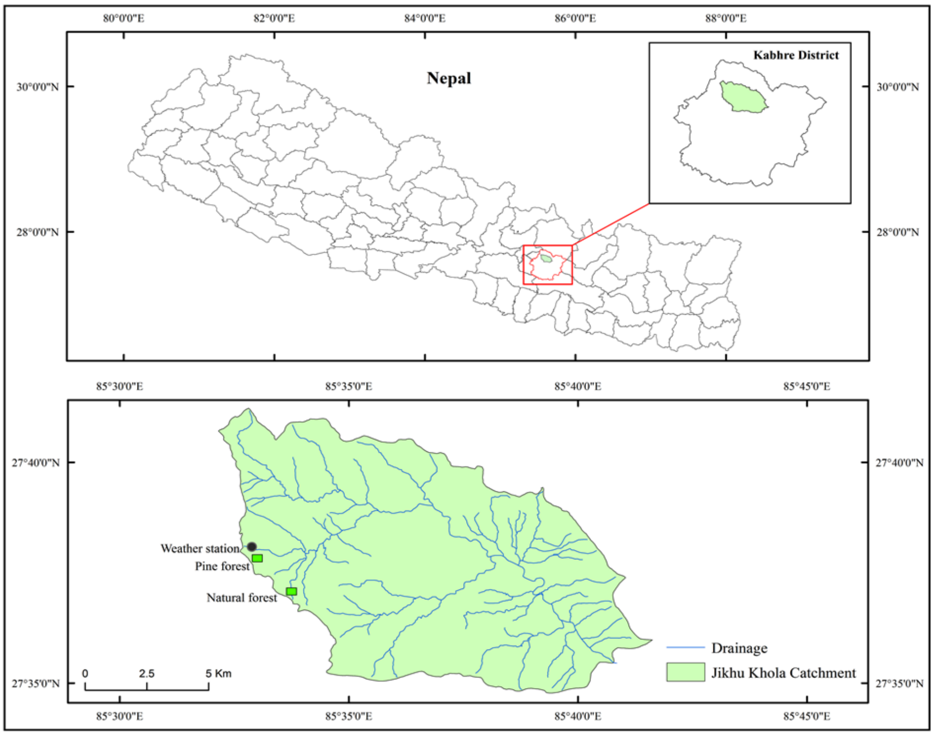

Figure 2.1: Location of the study sites in the Jikhu Khola Catchment in the Middle Mountains of Central Nepal.

Within the NF, more than half of the trees were *Castanopsis tribuloides* (65%) followed by *Schima wallichii* (16%), *Myrica esculenta* (6%), *Rhododendron arboreum* (5%), *Quercus lamellosa* (4%) and various other species (4%). The average tree height was determined at 14.0 ± 2.2 m. Although largely evergreen, the NF sheds a small proportion of its leaves towards the end of the dry season (February–March) but recovers quickly thereafter. For example, the maximum measured leaf area index (LAI, using a Licor LAI 2000 Plant Canopy Analyzer) was 5.43 ± 0.12 (S.D.) in September 2011 (i.e., at the end of the rainy season), while corresponding values determined during the pre-monsoon period in March, April and early June were 4.52±0.2, 5.14±0.1, and 5.32±0.1, respectively.

The PF plot had a surface area of 0.36 ha (60 m x 60 m) and was located at an elevation of 1580 m a.m.s.l. The PF site formerly consisted of degraded pasture and is part of an area of 25 ha that was reforested with *Pinus roxburghii* in 1986. The pine trees were planted randomly (i.e., not in rows) and 25 years old at the start of the throughfall measurements in June 2011. No other tree species were recorded in the PF plot. An understory was largely absent as grazing by cattle is common. In addition, the local population collects litter for animal bedding and regularly harvests the grassy herb layer (personal observation, C.P. Ghimire). Pruning of trees for fuelwood, and felling for timber are also common in the pine forests of the JKC (personal observation, C.P. Ghimire) although the research plots themselves remained free from such major disturbance throughout the present investigation. Like the NF, evergreen *P. roxburghii* sheds a small proportion of its needles towards the end of the dry season but also recovers quickly thereafter. The LAI of the PF was estimated at 2.21 ± 0.10 in September 2011 whereas corresponding values during the pre-monsoon period in March, April and early June were 2.05 ± 0.14 and 2.15 ± 0.12, and 2.17 ± 0.11 respectively. The average (±SD) height for all 85 trees present in the plot was determined at 16.3± 3.82 m in May 2011. The corresponding stem density, the mean DBH and basal area were 853 trees ha-1, 23.65±3.8 cm and 37.6 m2 ha^{-1}, respectively. Again stem density and basal area were corrected for slope gradient.

2.3 Methods

2.3.1 Rainfall

Incident rainfall (*P*, mm) for each forest plot was recorded in a nearby clearing at a distance of ca. 100 m using a tipping bucket rain gauge (Rain Collector II, Davis Instruments, USA; 0.2 mm per tip) backed up by a manual gauge (100 cm^2) between October 2010 and September 2011.

The orifices of all rain gauges were positioned at 50 cm above the ground to avoid ground splash effects. The automatically recorded data were stored at 5-min intervals. The manual gauge was read daily at approximately 8:45 AM local time, the Department of Hydrology and Meteorology's (DHM) standard time for morning meteorological readings in Nepal. Rainfall data were also collected at a weather station operated by the Kathmandu University (KU, at ca. 750 m distance from the PF). On 30 June and 1 July 2011 both the recording and the manual rain gauge at the PF plot were disturbed and no rainfall data were obtained. The data gap was filled using a linear regression equation linking daily rainfall totals at the PF plot and at the KU weather station ($R^2 = 0.96$, $n = 36$).

2.3.2 Throughfall

Throughfall (Tf, mm) in the NF and PF was measured daily using 15 and 10 randomly positioned funnel-type collectors (154 cm^2 orifice area), respectively. The higher number of funnel-type gauges deployed in the NF reflects the greater spatial variability of Tf expected for the NF compared to the PF as the NF was more floristically diverse. Thus, the gauges were grouped into three (NF) and two (PF) sub-groups of five gauges each that were randomly relocated once a week to minimise the effects of spatial variability on the magnitude of average Tf (Lloyd and Marques-Filho, 1988). In addition, Tf was recorded continuously using three tilted stainless steel gutters in each plot (200 cm x 30 cm each). The gutters were set up ca. 50 cm above the forest floor to avoid ground splash effects and at an angle of 25–30^o to the horizontal to facilitate drainage. Volumes were corrected for gutter inclination. Each gutter was equipped with a tipping bucket (50 ml per tip) plus data-logger device (Tinytec Plus TGPR-1201, Gemini Data Loggers Ltd., U.K.). Further, as no apparent alignment of the trees was observed in the planted stand, no specific orientation of the throughfall gutters was considered. The gutters were cleaned once a week and regularly treated with silicon solution to prevent clogging by organic debris and to minimise wetting losses, respectively. The records obtained with the gutters were converted to areal averages every time the manual gauges were emptied, using a weighting procedure that took the relative areas of the two types of gauges into account. On several instances not all the funnel-type collectors could be read due to disturbance by people or animals, and only the remaining collectors were taken into account for areal averaging.

The throughfall measurements were carried out from 20 June to 9 September 2011 (81 days), thereby covering the bulk of the 2011 rainy season.

2.3.3 Stemflow

Stemflow (Sf, mm) was measured on 10 trees which were representative of the dominant species in the NF plot and similarly on 8 trees in the PF plot. The sampled tree species in the NF were *Castanopsis tribuloides* (*n* = 4), *Schima wallichii* (*n* = 2), *Myrica esculenta* (*n* = 2), *Rhododendron arboreum* (*n* = 1) and *Quercus lamellosa* (*n* = 1). Stemflow was collected using 10-litre buckets connected to flexible polythene tubing fitted around the circumference of the trunk in a spiral fashion at 1m from the ground. Bark material was removed to provide a clean and smooth surface upon fitting the tube. Any remaining gaps between stem and tubing were closed using silicon sealant. The buckets were emptied once a week. Stemflow proved to be only a minor component of the wet-canopy water balance, and the number of sampled trees was insufficient to derive Sf vs. DBH relationships for up-scaling to the stand level (Manfroi et al., 2004; Holwerda et al., 2006). For this reason, Sf at the plot level was estimated by multiplying the average stemflow per species times the corresponding density of the trees (after correction for slope gradient). Sf measurements were carried out for 65 days between 28 July and 1 October, 2010. During this 65-day measurement period, a total of 509 mm of rain was received at the NF plot, *versus* a total of 452 mm at the PF plot. Sf was not measured during the 2011 rainy season. Instead, the average values derived for the 2010 rainy season (expressed as a fraction of incident rainfall) were used when estimating and modeling interception losses for 2011.

2.3.4 Additional meteorological measurements

Additional meteorological measurements were made using an automatic weather station located in a degraded pasture at 1620 m a.m.s.l and at a distance of 400 to 2700 m from the PF and NF plots, respectively (cf. Figure 2.1). Incoming solar radiation was measured using an SKS110-pyranometer (Skye Instruments, U.K.). Air temperature (T, ^{o}C) and relative humidity (RH, percentage of saturation) were measured at 2 m using a Vaisala HMP45C probe protected against direct sunlight and precipitation by a radiation shield. Wind speed and wind direction were measured at 3 m height, using an A100R digital anemometer (Vector Instruments, UK) and W200P potentiometer (Vector Instruments, U.K.) respectively. All measurements were recorded at 5-min intervals by a Campbell Scientific Ltd. 23X data-logger.

2.3.5 Modelling rainfall interception

The revised version of Gash's analytical rainfall interception model (Gash et al., 1995) was used to model annual interception losses at the NF and PF plots. Although an adaptation of the Gash model exists that takes seasonal changes in vegetation LAI into account (van Dijk and Bruijnzeel, 2001a; van Dijk and Bruijnzeel, 2001b), the presently determined variations in LAI – and corresponding rainfall amounts during the pre-monsoonal months – were too small to have a significant effect on overall annual interception loss. The revised Gash model assumes rainfall to occur as a series of discrete events. Events were defined in the present study as being separated by at least 3 h without rain to allow the complete drying of the canopy before the next event (cf. Schellekens et al., 1999). Each rainfall event of sufficient magnitude to fully wet the canopy is then sub-divided into three phases: (i) a wetting-up phase during which gross rainfall (P) is less than the amount required to fully saturate the canopy (P'), (ii) a saturated phase, when rainfall intensity (R) exceeds the evaporation rate from the wet canopy (E), and (iii) a drying phase after all drip from the canopy has ceased. Table 2.1 summarises the equations that are used in the model to calculate the interception loss associated with the respective phases. The amount of water needed to completely saturate the canopy is calculated after Gash et al. (1995) from:

$$P' = -{}^{\bar{R}}\!/\!_{\bar{E}_c}\, S_c In\left(1 - {}^{\bar{E}_c}\!/\!_{\bar{R}}\right) \qquad (2.1)$$

where $\bar{R}$ denotes the average rainfall intensity falling onto a saturated canopy. The canopy capacity per unit area of cover $S_{c,}$ is obtained by dividing the canopy storage capacity (S) by the canopy cover fraction (c). Similarly the wet canopy evaporation rate per unit area of land ($\bar{E}$) during rainfall is scaled down in proportion to the canopy cover to obtain the evaporation rate per unit area of cover, $\bar{E}_c$. In applying the analytical model, saturated canopy conditions are generally assumed to occur whenever the hourly rainfall exceeds 0.5 mm (Gash, 1979; Schellekens et al., 1999). The free throughfall coefficient (p) is the proportion of incident rainfall that falls directly to the forest floor without hitting the canopy. Evaporation from the trunk is specified in terms of trunk storage capacity, S_t, and the proportion of rain that is diverted to stemflow, p_t.

Gash (1979) has shown that the slope of a regression equation of observed interception loss on rainfall (on a per storm basis) should equal $\bar{E}/\bar{R}$, assuming that both $\bar{E}$ and $\bar{R}$ are constant for all storms. Thus $\bar{E}$ can be estimated in this way from $\bar{R}$ in the absence of above-canopy climatic observations (Bruijnzeel and Wiersum, 1987; Dykes, 1997; Holwerda et al., 2012; Schellekens et al., 1999).

Table 2.1: The five terms in the revised form of the analytical model of rainfall interception (after Gash et al., 1995).

Component of interception loss	Formulation
m small storms insufficient to saturate the canopy ($P < P'$)	$c\sum^{m} P_j$
Wetting up the canopy in *n* large storms ($P' \geq P$)	$ncP' - ncS_c$
Evaporation from saturated canopy until throughfall ceases	$\left(c\frac{\bar{E}_c}{\bar{R}}\right)\sum^{n}(P_j - P')$
Evaporation after throughfall has ceased	ncS_c
Evaporation from trunks; *q* storms with $P > S_t/p_t$, which saturate the canopy	$qS_t + p_t\sum_{j=1}^{n-q} P_j$

2.3.6 Derivation of canopy parameters

The canopy storage capacity S of the two study forests was estimated with the method of Jackson (1975). In this approach S is the difference between P and Tf at the inflection point in the P *versus* Tf graph. The inflection point is defined as the intersection of the two regressions between P and Tf for storms that do, and those that do not, fill the canopy storage (Jackson, 1975). The method also allows the derivation of p from the slope of the regression between P and Tf for storms that are too small to fill the canopy storage capacity. Finally, canopy cover fraction c is assumed to be equal to (1-p).

The stemflow parameters S_t (trunk storage capacity) and p_t (stemflow fraction) were derived following the method of Gash and Morton (1978) as the negative intercept and the slope of the linear regression line between Sf and *P*, respectively.

2.3.7 Wet-canopy evaporation

The mean rate of wet canopy evaporation $\bar{E}$ was derived from the value of $\bar{E}/\bar{R}$ as obtained from the linear regression of P against measured interception loss (hereafter referred to as the Tf-based average wet canopy evaporation rate ETF). $\bar{E}$ was also estimated using climatic date as measured at the degraded pasture site in the Penman-Monteith equation (hereafter referred to as the Penman-Monteith based average wet canopy evaporation E_{PM}) with the canopy resistance (r_s) set to zero (Monteith, 1965) and the aerodynamic resistance (r_a) estimated initially according to Thom (1975). To obtain values of net radiant energy above the respective forest canopies, albedo values of 0.12 (Oguntoyinbo, 1970) and 0.10 (Waterloo et al., 1999) were used for the NF and PF, respectively. It is recognised that the resulting estimates of E_{PM} may be too low due to the overestimation of r_a in complex terrain by the Thom (1975) approach (see Holwerda et al. (2012) for details).

2.4 Results

2.4.1 Rainfall characteristics

Between October 2010 and September 2011, a total of 1411 mm of rain was received at the NF plot, whereas the corresponding value for the PF plot was 1501 mm. The seasonal variation of rainfall amounts at the NF and the PF is shown in Figure 2.5 below. During the 81-day period of wet season measurements of throughfall and meteorological conditions in 2011, a total of 709 mm of rain was received at the NF plot, distributed over 58 rainy days (defined as having ≥ 0.4 mm of rainfall) and 76 events, whereas a total of 878 mm was received at the PF plot distributed over 57 days and 71 events, respectively. During the 2011 wet season study period, about 55% of the total rainfall (at both NF and PF) occurred between 18:00 h and 06:00 h. The highest rainfall intensity values (derived from 5-min periods) were 76.8 mm h^{-1} and 88.8 mm h^{-1} at the NF and PF, respectively.

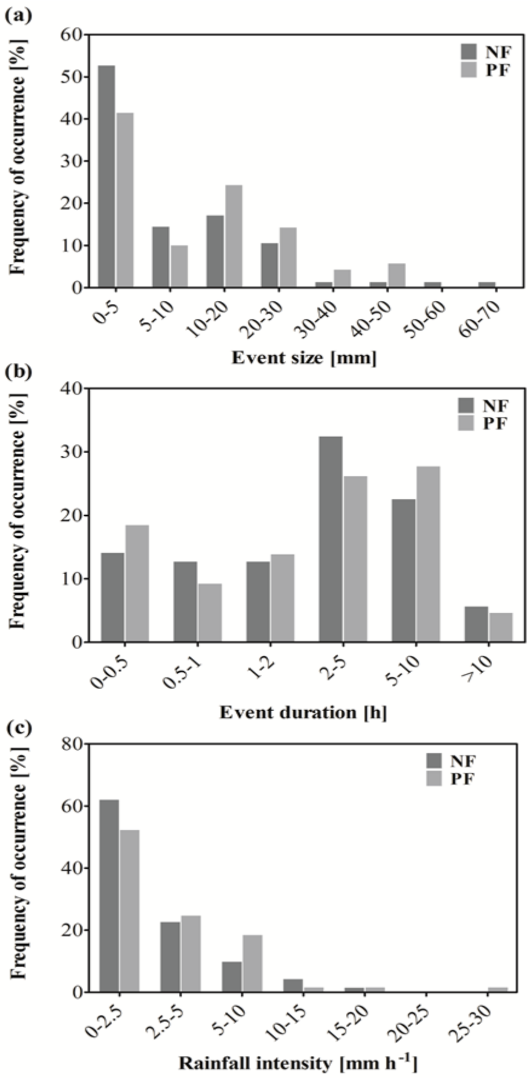

Figure 2.2: Frequency distributions of: (a) amount, (b) duration, and (c) intensity of rainfall events at the study plots. NF denotes the natural forest site and PF the pine forest site.

The average (and median) event-based amount duration and intensity of P at the NF plot were 8.4 (4.0) mm, 3hr44min (2hr20min), and 3.0 (2.0) mm h^{-1}, respectively (Table 2.2). Corresponding values for the PF plot were 12.1 (7.0) mm, 3hr55min (2hr55min), and 3.6 (2.4) mm h^{-1}, respectively. The frequency distributions of event size, intensity, and duration for the study period are shown in Figure 2.2. The frequency distributions of event size were highly positively skewed for both sites

(Figure 2.2) indicating low-magnitude rainfall to be much more frequent than high-magnitude rainfall. Therefore, median values were added to Table 2.2. As expected in view of the comparatively short distance between the two sites there were no significant differences between the two forests in terms of the frequency distributions for rainfall amount and rainfall intensity (Figure 2.2). Most events at the two sites delivered rainfall amounts ≤ 5 mm and at an average intensity ≤ 2.5 mm h^{-1} (Figure 2.2, cf. Merz et al., 2006b).

2.4.2 Throughfall, stemflow and derived interception loss

Measured throughfall (Tf) totals for the 2011 wet-season study period amounted to 540 and 729 mm in the NF and PF plots, respectively and represented 76.2% and 83.0 % of the corresponding incident rainfall, respectively. The average standard errors (SE) of mean Tf for individual sampling were 6.2% (i.e. 33.4 mm or 4.7% of P) and 4.9% (i.e. 35.7 mm or 4.0% of P) for the NF and PF, respectively. The higher Tf fraction in the PF plot relative to the NF plot is in line with the lower stem density and LAI of the pine plantation. Table 2.3 lists the basic characteristics of throughfall events at the NF and PF plots during the study period whereas Figure 2.3 shows the frequency distributions of measured Tf/P. Whilst the distribution is skewed towards the right for the NF, that for the PF tends to be bimodal with a secondary maximum for very low Tf fractions (Figure 2.3).

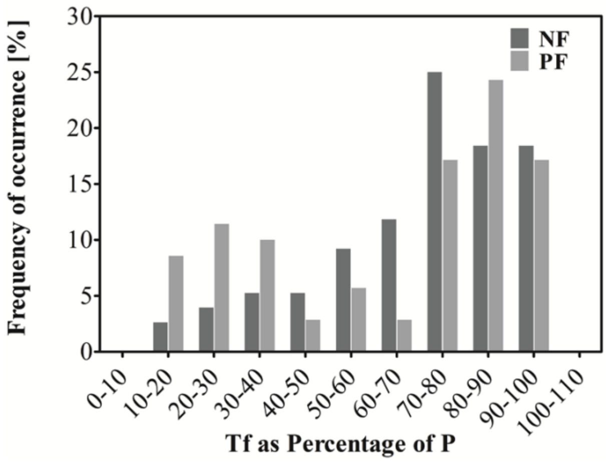

Figure 2.3: Frequency distribution of throughfall Tf, expressed as a percentage of rainfall (P) at the natural forest (NF) and pine forest (PF) plots.

Average stemflow (Sf) for different dominant tree species in the NF was estimated at 0.7% of *P* for *Castanopsis tribuloides*, 3.1% for Schima wallichii, 3.3% for *Myrica esculenta* and 0.9% for other species. Similarly, average Sf per tree in the PF was only 0.6%. Plot-level estimates of Sf in the NF and PF were 1.4% and 0.5% of incident rainfall, respectively, and included a correction for slope gradient. The SE values of the Sf estimate were 44.4% and 21.4% for the NF and PF, respectively.

Based on the wet-canopy water balance equation ($P = E_i$+Tf+Sf) for the wet season study period, derived rainfall interception losses (E_i) were estimated to be 159 and 145 mm for the NF and PF, respectively. These totals represent 22.4% and 16.5% of the corresponding *P*, respectively (Table 2.4). Adding the SE's of the Tf and Sf quadratically would suggest SE values of 33.6 mm (i.e. 4.7% of *P*) and 35.8 mm (i.e. 4.1% of *P*) for E_i for the NF and PF, respectively. Because stemflow values are very small, and the frequency distributions of rainfall amount and rainfall intensity did not differ significantly between the two forests (Figure 2.2), the higher interception loss from the NF compared to that from the NF primarily reflects the differences in Td between the two forest types and is in line with the difference in tree density and LAI (both much higher in the NF, cf. Section 2.2)

Table 2.2: Statistical parameters of rainfall events during the 2011 monsoon measuring campaign (n = 81 days) at the natural forest (NF) and the pine forest (PF) sites.

Parameters	Rainfall (*P*, mm)		Duration (day: hour: min)		Intensity (mm h^{-1})	
	NF	PF	NF	PF	NF	PF
Mean	8.4	12.1	00:03:44	00:04:00	3.0	3.6
Range	0.2-60.2	0.2-7.8	00:10:00-00:21:35	00:00:05-00:13:35	0.15-17.8	0.15-5.0
Median	4.0	7.0	00:02:20	00:02:55	2.0	2.4
Total	709	878	11:01:41	10:19:12		

2.4.3 Canopy parameters

The derivation of the canopy capacity *S* and free throughfall coefficient *p* for the two study forests using the method of Jackson (1975) is presented in Figure 2.4. The following values were obtained for the NF and PF, respectively: *S*, 0.89 and 0.67 mm; *p*, 0.19 and 0.27, and thus for canopy cover *c*, 0.81 and 0.73. The trunk water storage capacity S_t and the proportion of rain diverted to stemflow p_t following the method of Gash and Morton (1978) were determined at 0.02 and 0.033 mm (S_t), and at 0.017 and 0.007 (p_t) for the NF and PF, respectively.

2.4.4 Interception model application

The total number of rain events measured at the two forest plots during the rainy season of 2011 was divided into two sub-sets, i.e. sub-set 1 (NF, n = 41; PF, n = 39) for model calibration and sub-set 2 (NF, n = 35; PF, n = 32) for model validation. Rain events occurring in June, August and September (data-set 1) were used for the calibration of the model whereas the calibrated model was validated using data for the month of July (data-set 2). Together, the storms of data-set 1 produced 471 and 572 mm of rainfall at the NF and PF, respectively. Corresponding amounts for data-set 2 were 234 and 306 mm, respectively.

The analytical interception model requires the relative evaporation rate $\bar{E}/\bar{R}$, the ratio of the mean evaporation rate to the mean rainfall rate for hours when the canopy is saturated (i.e. $R \geq 0.5$ mm h^{-1}). An estimate of $\bar{E}/\bar{R}$ for each site was obtained firstly from the linear regression of interception loss *versus* incident rainfall (Gash, 1979). This yielded $\bar{E}/\bar{R}$ values of 0.23 and 0.14 for the NF and PF, respectively. The median rainfall intensity falling onto a saturated canopy as calculated from data-set 1 amounted to 1.73 and 2.46 mm h^{-1} for the NF and PF, respectively. Combining the latter values with the corresponding $\bar{E}/\bar{R}$ values gave Tf-based average wet-canopy evaporation rates E_{Tf} of 0.39 mm h^{-1} and 0.34 mm h^{-1} for the NF and PF, respectively. $\bar{E}$ values were also obtained using the Penman-Monteith equation with the canopy resistance (r_s) set to zero, giving wet-canopy evaporation rates E_{PM} of 0.12 and 0.11 mm h^{-1} for the NF and PF, respectively. Tf-based estimates of $\bar{E}$ are usually (much) higher than estimates using the Penman-Monteith equation (Holwerda et al., 2012) and the present results are no exception. Possible reasons for this discrepancy are discussed later.

The revised version of the Gash model was run on an event basis using the respective parameter for each forest as listed in Table 2.5. Several runs were conducted. The observed and predicted throughfall totals and interception losses for the respective runs are compared in Table 2.6. In the first run (Model 1) E_{PM} was used to characterise wet-canopy evaporation. Predicted throughfall amounts were ~ 8 % higher than observed totals for both forest plots, resulting in a relative underestimation of the interception loss by ~32 % for the NF and by 44 % for the PF. Predicted throughfall amounts were closer to observed totals in the case of the PF (but not the NF) when using the Tf-based average wet-canopy evaporation E_{Tf} (Model 2).

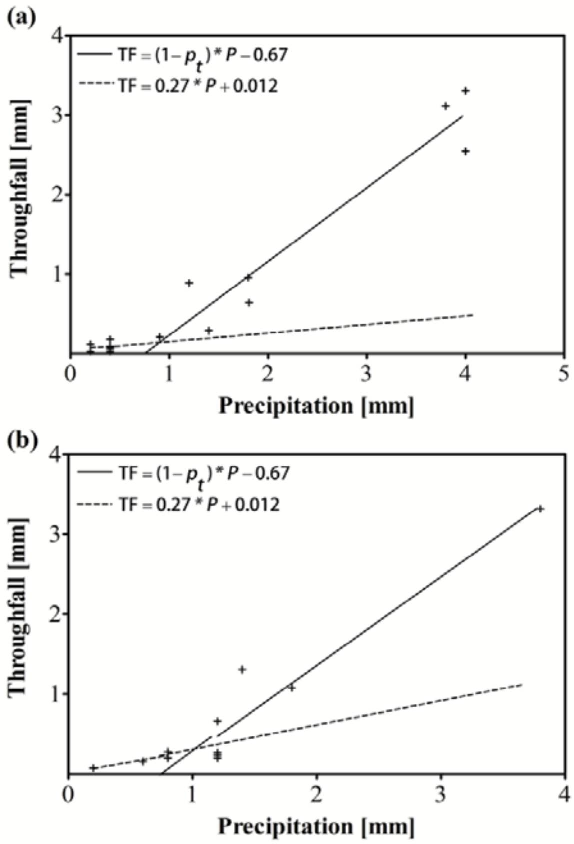

Figure 2.4: Estimation of canopy parameters *S* (canopy saturation value) and *p* (free throughfall fraction) using the method of Jackson (1975) for: (a) the natural forest and (b) the pine forest.

Model 2 underestimated Tf in the PF by ~ 2.5% but that in the NF by 11%, causing interception losses to be overestimated by 14% (PF) and 44% (NF). In the third run of the model (Model 3) $\bar{E}$ was optimised by minimising the root mean squared error (RMSE). This yielded average wet-canopy evaporation rates of 0.25 mm h^{-1} and 0.30 mm h^{-1} for the NF and PF, respectively. Using the latter, predicted interception losses agreed well with observed values for both forest plots (Table 2.6).

Table 2.3: Statistical parameters of throughfall events during the 2011 monsoon campaign (n = 81 days) at the natural forest (NF) and the pine forest (PF) sites

Parameters	Throughfall (Tf, mm)		Duration (day: hour: min)		Intensity (mm h^{-1})	
	NF	PF	NF	PF	NF	PF
Mean	6.4	10.0	00:05:24	00:04:55	1.1	1.8
Range	0.2-48.6	0.08-47.8	00:00:10-00:22:20	00:00:10-00:16:45	0.05-5.0	0.02-9.6
Median	3.0	5.1	00:04:05	00:03:05	0.7	1.1
Total	540	729	16:11:02	13:08:10		

Table 2.4: Amounts of rainfall (P), throughfall (Tf), stemflow (Sf), and derived estimates of rainfall interception loss (E_i) during the 2011 monsoon measuring campaign (n = 81 days) in the natural forest (NF) and pine forest (PF). Values between brackets represent percentages of incident rainfall.

Experimental plot	Gross rainfall (mm)	Throughfall (mm)	Stemflow (mm)[a]	Interception (mm)
NF	709	540 (76.2%)	10 (1.4%)	159 (22.4%)
PF	878	729 (83.0%)	4 (0.5%)	145 (16.5%)

[a] Based on measurements made during the 2010 rainy season.

The optimised model was then used to estimate the total interception losses from the two forests for the one-year period from October 2010 to September 2011 for which complete rainfall data were available. The associated rainfall totals recorded at the natural and pine forest plots were 1411 and 1501 mm, respectively. The corresponding model estimates of total interception loss were 319 mm for the NF and 291 mm for the PF, representing 22.6% and 19.4% of incident rainfall, respectively (compared to 22.4% and 16.5% based on measured interception losses for the wet-season study period only for the respective two forests). Figure 2.5 shows the seasonal variation of rainfall and interception loss at the NF and PF. Evaporation from a saturated canopy contributed 56.5% and 59.2% of total annual E_i in the NF and PF, respectively, followed by canopy drying (32.0% and 31.3%, respectively). Losses associated with canopy wetting and evaporation from the trunks were minor.

Table 2.5: Summary of wet-canopy evaporation rates E (Penman-Monteith based, Tf-based, and optimized) and forest structural parameter values for the two sites used in the modelling of rainfall interception.

Experimental plot	S (mm)	p	c	S_t (mm	p_t	E_{PM} (mm	E_{Tf} (mm	$E_{optimised}$ (mm h^{-1})
NF	0.89	0.19	0.81	0.02	0.017	0.12	0.39	0.25
PF	0.67	0.27	0.73	0.033	0.007	0.11	0.34	0.30

Table 2.6: A comparison of modelled and observed interception loss (mm) for rainfall data-sets 1 (June, August and September 2011, calibration) and 2 (July 2011, validation) in the natural and forest (NF) and pine forest (PF).

Component of interception loss	Calibration data						Validation data (set 2)	
	Model 1 (set 1)		Model 2 (set 1)		Model 3 (set 1)			
	NF	PF	NF	PF	NF	PF	NF	PF
Total gross rainfall (mm)	471	572	471	572	471	572	234	306
Total modeled interception loss (mm)	64.1	50.3	135.6	102.7	98.0	90.8	59.2	54.9
Total measured interception loss (mm)	94	90	94	90	94	90	65	55
Modeled – measured (% relative error)	31.8	44.1	44.2	14.1	4.3	0.9	8.9	0.2

2.5 Discussion

2.5.1 Rainfall characteristics

Total amounts of rainfall recorded at the NF (1411 mm) and PF (1501 mm) plots were slightly below and similar, respectively, to the long-term mean of 1487 mm as recorded close to the present study area at an elevation of 1560 m a.m.s.l. (1993–1998) (Merz, 2004).

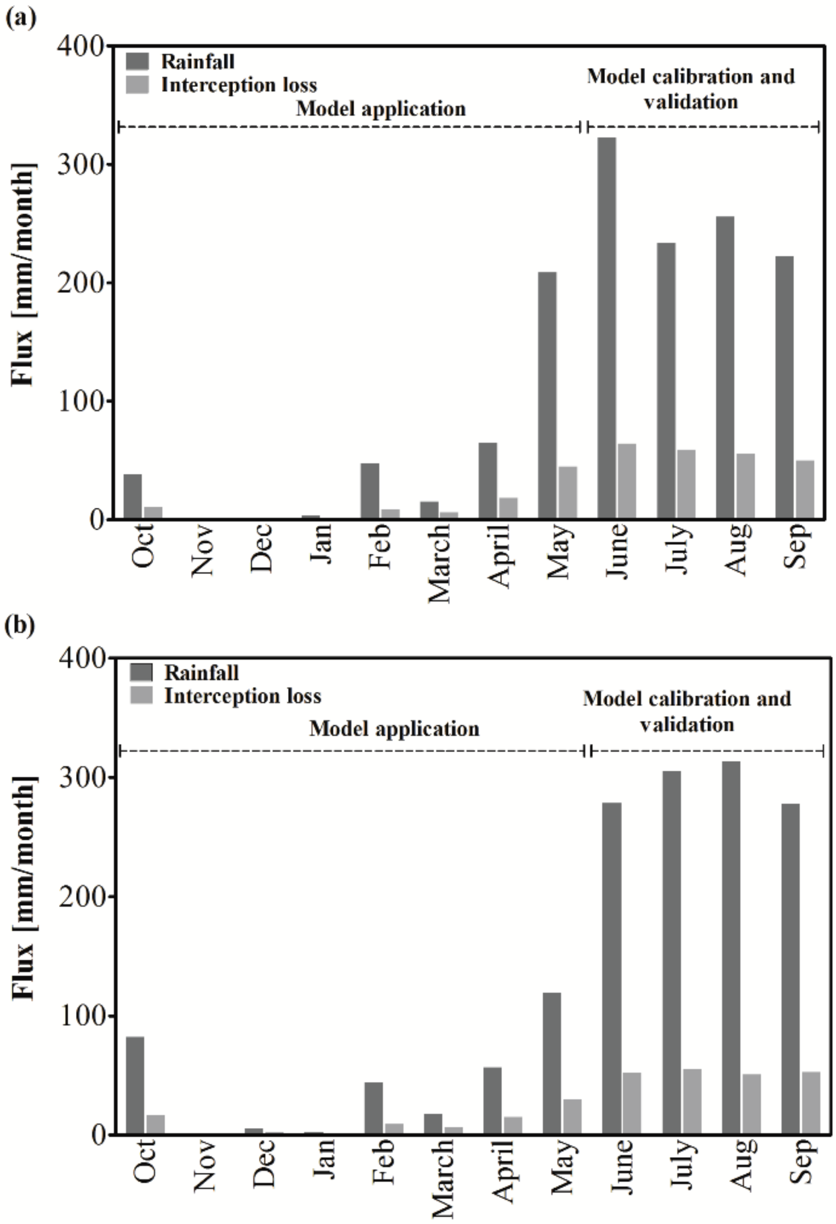

Figure 2.5: Seasonal variations of rainfall and interception loss for: (a) the natural forest and (b) the pine forest.

The presently observed difference of 90 mm in rainfall between the two forest sites that are located ca. 2200 m apart and differ 80 m in elevation is not surprising given the complex and rugged topography of the study area which may be expected to induce local rain shadows (leeward slopes) and precipitation hotspots (windward slope). Katiyar and Striffler (1984) identified slope orientation as the prime factor determining monsoonal precipitation amounts in a Middle Mountain Zone catchment in North-West India, followed by elevation. Local-scale rainfall variability in the Middle Mountains of Nepal is not well known nor well documented, mainly because of problems with access and the high costs of maintaining a dense rain gauge network in this type of terrain. Among the few local studies available, Merz (2004) found high spatial variability in rainfall within the Jikhu Khola catchment. Similarly, Lang and Barros (2002) observed very large spatial variability in precipitation (a roughly four-fold difference over ~10 km distance) elsewhere in Central Nepal. However, the elevation-rainfall relationships in the Middle Mountains of Nepal are far from clear (Lang and Barros, 2002; Merz, 2004). Future work might usefully employ an X-band local area Weather Radar (LAWR) that produces images at 500 m resolution and which has been shown to greatly improve understanding of the climatological and orographic influences of high mountains on rainfall distribution in similarly complex topography in the Ecuadorian Andes (Rollenbeck and Bendix, 2006; Rollenbeck et al., 2007).

There is equally little detailed information on rainfall characteristics (such as the nature of rainfall intensities) in the Middle Mountains of Nepal. During the 81-day period of intensive measurements in the 2011 rainy season, the highest 5-min rainfall intensity recorded at the PF site was 88.8 mm h^{-1} vs. 76.8 mm h^{-1} at the NF site. By comparison, Gilmour et al. (1987) reported a maximum 5-min intensity of 70 mm h^{-1} in Kathmandu (located 45 km west of the study area), whereas Merz et al. (2006a) reported a maximum 10-min intensity of 36 mm h^{-1} for large monsoon events in the JKC. However, as illustrated by Figure 2.2c, low intensities are far more common in the study area. Rainfall intensities ≤ 2.5 mm hr^{-1} occurred in 52% (PF) to 62% (NF) of the events. Such differences in rainfall intensity may contribute to differences in interception loss between the two forest sites (cf. Murakami, 2006) although a longer observation period would be required to allow more definitive conclusions. Nevertheless, the frequency distributions of rainfall amounts and intensities recorded by the present study are similar to those found by Merz (2004) elsewhere in the JKC over a longer time span. The presently found average rainfall intensities and frequency distributions are also very similar to those reported for two monsoon seasons at comparable elevations in the Kumaun Himalaya, north-west India (Pathak et al., 1984).

2.5.2 Canopy parameters

To the best of the authors' knowledge, there is no information available on the magnitude of the canopy saturation value *S* or free throughfall fraction *p* for Himalayan forests allowing meaningful comparisons to be made. Nevertheless, the presently derived values for the canopy parameters of the broad-leaved forest (S = 0.89 mm, p = 0.19; Table 2.5) are similar to estimates reported for some montane tropical forests (S = 0.89–1.1 mm; p = 0.23–0.24; Jackson, 1975; Schellekens et al., 1999). However, where montane forests become more epiphyte-rich (as occurs in Central Nepal at elevations well above the present study sites; Dobremez, 1976), values of *S* typically increase markedly (cf. Fleischbein et al., 2005; Holwerda et al., 2010).

The presently derived value of *S* for the mature *Pinus roxburghii* stand (0.67 mm) is intermediate between the 0.50–0.55 mm for an 18-year-old maritime pine stand in southern France (800 trees ha^{-1}, LAI 3.0; Loustau et al., 1992) and the 0.8–1.2 mm reported for well-stocked *Pinus caribaea* plantations in Fiji (Waterloo et al., 1999). In view of the intensive pruning of the pine forests of the study area and the correspondingly low value of the LAI of the PF plot (≤ 2.2, Section 2.2), the value of *S* for a less disturbed pine forest canopy is bound to be higher and may therefore approach that derived for the (much less disturbed) NF plot. Indeed, Waterloo (1994) found *S* in a mature pine plantation in Fiji to be reduced by 50% (from 1.2 to 0.6 mm) after the stand's LAI was reduced from 4.0 to 3.1 by cyclone damage. Likewise, mature plantations of *Pinus merkusii* in mountainous West Java (Indonesia) of similar stocking as the PF site exhibited canopy storage values of 0.6–1.0 mm (C.A. Bons, personal communication to the second author).

The somewhat lower value of the free throughfall coefficient (p) for the natural forest as compared to that for the pine forest (Table 2.5) is consistent with the more open character of the pine forests of the study area (as also reflected by the lower LAI value and tree density; Section 2.2). The value of *p* derived for the NF (0.19) is rather high considering the dense nature of the forest stand (LAI up to 5.4, Section 2.2). However, Tf-based estimates of *p* are often subject to considerable uncertainty (Jackson, 1975) because part of the rain hitting the canopy may splash off and so contribute to 'direct' throughfall before the canopy is fully saturated (Schellekens et al., 1999). Indeed, comparable results have also been obtained for similar montane tropical forests elsewhere (Jackson, 1975; Schellekens et al., 1999; cf. Table 2.6).

The very low stemflow fractions observed for the study forests (Table 2.4) render a comparison of the stemflow-related parameters S_t and p_t with other studies rather hazardous. However, the p_t-value obtained for the NF (0.017) is not dissimilar to that reported for montane forest of similar LAI in Puerto Rico (p_t = 0.023; Schellekens et al., 1999) but values for both S_t and p_t are roughly half those derived for more epiphyte-rich montane forest in Ecuador (Fleischbein et al., 2005). Likewise, the presently derived values for both p_t and S_t in the PF are half the values reported for a series of tropical pine plantations in Fiji (Waterloo et al., 1999). The higher trunk storage capacity S_t of the PF compared to that of the NF is in agreement with the much rougher bark of the pine trees compared to the oaks and chestnuts of the NF, and this apparently overrides the effect of the smaller stocking of the PF (Section 2.2).

2.5.3 Wet-canopy evaporation

The average wet-canopy evaporation rates obtained with the Penman-Monteith equation (E_{PM}) for the NF and PF during the interception model calibration phase (storm data-set 1 in Table 2.5) were about one-third of the corresponding Tf-based $\bar{E}$ values (Table 2.5). Several studies of rainfall interception loss from tropical forests have reported similarly large discrepancies between E_{PM} and E_{Tf} (e.g. Bruijnzeel and Wiersum, 1987; Dykes, 1997; Schellekens et al., 1999; Wallace and McJannet, 2008; Holwerda et al., 2012).

There are a number of possible reasons for this. Firstly, in the present study, the meteorological parameters that were used to calculate E_{PM} were collected in a nearby grassland where micrometeorological conditions (notably temperature, vapour pressure deficit and wind speeds) can be expected to differ from conditions prevailing above a forest canopy in such a way that evaporation during rainfall tends to be enhanced above the forest (cf. Pearce et al., 1980; McVicar et al., 2007; Wallace and McJannet, 2008).

Secondly, the formula used to derive the aerodynamic conductance (g_a) in the Penman-Monteith equation (Thom, 1975) is valid over uniform, flat surfaces where the effect of topography on wind flow patterns is minimal. Therefore, the application of Thom's equation in areas with rugged topography like the study area is questionable (Raupach and Finnigan, 1997) and tends to underestimate g_a and thus E_{PM} (Holwerda et al., 2012; cf. Monteith and Unsworth, 2008). This is because the presence of a complex topography distorts and deflects the airflow in such a way that g_a is enhanced. Holwerda et al. (2012) demonstrated for a broad-leaved evergreen forest in similarly mountainous terrain in Puerto Rico that E_{PM} and E_{Tf} did match after inserting *measured* g_a (derived from the friction velocity as inferred from eddy covariance measurements; cf. Gash et al., 1999; Van der Tol et al., 2003) into the Penman-Monteith equation instead of using the Thom-based g_a. Use of the latter produced values of E_{PM} that were much lower than E_{Tf} (Holwerda et al., 2012).

A third possible explanation of the observed discrepancy between E_{PM} and E_{Tf} relates to the fact that the micro-climate of a slope may be different from that on a ridge or in a valley (Raupach and Finnigan, 1997). In the present study, both forest plots were located on fairly steep slopes whereas the weather station was located on a ridge. Finally, there is the possibility of (large-scale) advection of warmer and drier air from areas downslope that remained dry during times of rainfall occurring higher up in landscape (cf. Calder, 1990) whereas enhanced evaporation of the splashed droplets that are produced when raindrops hit the forest canopy (Murakami, 2006) cannot be ruled out either.

2.5.4 Rainfall interception

The annual rainfall interception loss derived for the two studied forest by a combination of wet-season measurements and subsequent application of Gash's revised analytical model varied from 291 mm (19.4% of incident P) for the pine forest to 319 mm (22.6%) for the broad-leaf forest (Table 2.4). In doing so, fixed values were used for the main model parameters $\bar{R}$, S, $\bar{E}$ and p throughout the year. Although slight changes in the canopy-related parameters S and p are expected due to seasonal changes in LAI, and in $\bar{R}$ and $\bar{E}$ due to variations in seasonal weather patterns, the associated error in the model estimates of annual rainfall interception are considered to be minimal for two reasons (cf. Wallace and McJannet, 2008). Firstly, ~77% of the total annual rainfall in the area is recorded during the main monsoon season (Figure 2.5) when changes in LAI (and therefore in the canopy parameters used in the revised analytical model; cf. van Dijk and Bruijnzeel, 2001a) are negligible for both forests. Secondly, although the LAI of the two forests was temporarily reduced in response to partial leaf shedding in February and March, the rainfall during this period represented only about 4% of annual rainfall (Figure 2.5). In addition, LAI was shown to recover rapidly from mid-March onwards and reached stable values again before the onset of the main rainy season (see LAI values listed in Section 2.2).

To the best of the authors' knowledge there is no other work published to date on the magnitude of interception losses in the Nepalese Himalaya that would allow meaningful comparisons to be made. However, the present estimates are comparable with preliminary values reported for similar forest types in the Kumaon Himalaya, North-West India ((Pathak et al., 1985) although the latter are likely to be overestimated somewhat because of the non-random positioning of the throughfall gauges (gaps in the canopy were avoided) and the use of a fixed gauge arrangement that tends to underestimate the contributions by 'drip points' (Holwerda et al., 2006; cf. Lloyd and Marques-Filho, 1988).

The lower value of interception loss derived for the PF plot as compared to the NF plot is in accordance with what would be expected from the difference in stocking and LAI, as interception losses are usually higher in denser forest with higher LAI (Bruijnzeel, 2000; Fleischbein et al., 2005; Pypker et al., 2005; cf. Waterloo et al., 1999). Thus, considering that overall weather conditions in the two studied stands are likely quite similar due to the proximity of the sites (~2.2 km apart) and their comparable elevation, the higher interception loss derived for the NF is most probably largely attributable to the higher tree density and LAI of the NF compared to the PF (Section 2.2). Nevertheless, to rule out any effect of differences in rainfall intensity and event frequency distribution between the NF and PF plots (cf. Figure 2.2), the revised Gash model was also applied in a crossed manner for the 81-day measurement period, i.e. using the NF rainfall at the PF plot and *vice versa*. The effect of differences in rainfall intensity and event frequency distribution between the two sites on the difference in rainfall interception proved negligible, the differences being 0.9% in the NF and 0.7% in the PF.

Upon modeling interception losses from the two study forests, the best results were obtained after optimising the average wet-canopy evaporation rate (Tables 2.5 and 2.6). However, model performance for the NF was poorer than for the PF, both during the calibration and validation phase (Table 2.6). Although standard errors of mean Tf were not particularly high compared to values obtained by various other interception studies conducted in tropical broad-leaf forests that were both taller and more diverse than the present NF (cf. Dietz et al., 2006; Holwerda et al., 2006; Vernimmen et al., 2007), it cannot be excluded that the number of throughfall gauges deployed in the present study did not capture the full spatial variability in Tf. On the other hand, the joint surface area of throughfall gauges and gutter in the present study was not less than that used in these other studies whereas the difference between observed and simulated interception losses in the present study is (much) better compared with the results of various other modeling studies (e.g. Bruijnzeel and Wiersum, 1987; Calder et al., 1986; Lloyd et al., 1988; see also Schellekens et al., 1999).

2.6 Conclusion

This study, to the best of the authors' knowledge, is the first to provide a description of rainfall interception and corresponding canopy parameters for natural broad-leaf and planted pine forest in the Nepalese Himalaya. Measured throughfall, stemflow and derived estimates of interception loss over an 81-day period during the main rainy season were 76.2%, 1.4% and 22.4% of incident rainfall, respectively. Corresponding values for the nearby planted pine forest were 83.0%, 0.5% and 16.5%. Differences in interception loss between the natural and planted forest were expected, since there were significant differences in tree density and LAI values (both lower in the pine plantation). The revised analytical interception model (Gash et al., 1995) was used for the first time under the monsoonal montane conditions prevailing in the study area. There was good agreement between modeled and observed interception losses for the two contrasting forests, provided optimised values were used for the wet-canopy evaporation rate $\bar{E}$.Model estimates of annual interception losses for the natural forest and the pine forest were 319 mm (22.6% of annual rainfall) and 291 mm (19.4%), respectively. This considerable water loss from planted forest in the Middle Mountains of Nepal is likely to contribute to the observed reduction in streamflow amounts following the large-scale reforestation with pines of degraded pasture land in the study area. Pastures in the region are typically heavily overgrazed (cf. Gilmour et al., 1987) and their capacity to intercept rainfall must be considered to be minimal. However, needless to say, for a fuller assessment of the causes underlying these reductions in flows studies of the changes in dry-canopy evaporation rates (transpiration) of planted forest and in soil infiltration capacity as a function of time after reforestation are needed as well.

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Chapter 3

Transpiration, canopy conductance and decoupling coefficient of two contrasting forest types in the Lesser Himalaya of Central Nepal[1]

Abstract. Tree transpiration (E_t), canopy conductance (g_c) and decoupling coefficient (Ω) of a natural broad-leaved forest and a mature planted pine forest located in the Middle Mountain Zone of Central Nepal, were quantified using sap flow measurements and concurrent climatic and soil water observations. Estimated annual E_t totals were subsequently combined with corresponding interception losses (E_i) to calculate annual evapotranspiration totals (ET) for the two forests. E_t was strongly dependent on atmospheric vapour pressure deficit (VPD) but much less on short-wave radiation (R_s) while there was little evidence of any limitation by soil water deficits despite a strongly seasonal rainfall regime. Both forests transpired readily throughout the dry season, except during the period of maximum leaf fall (March–April). Annual E_t by the tree stratum amounted to 163 mm and 280 mm in the natural and planted forest, respectively, representing 12.2% and 19.6% of the corresponding incident rainfall totals. Estimated annual ET (including understory and litter evaporation) values were 524 mm and 577 mm, respectively (39.3% and 40.5% of rainfall). Maximum daily estimates of canopy conductance g_c ranged from 4.85 mm s $^{-1}$ in the natural forest to 11.46 mm s^{-1} in the planted forest and averaged 1.69 and 3.28 mm s^{-1}, respectively. A strong response of g_c to changes in VPD was detected in both forests. Daily average decoupling coefficient Ω was always less than 0.15 for both forests, with very low overall average values of 0.06 (natural forest) and 0.07 (planted forest) indicating that transpiration rates in the two forests were determined mostly by daily variations in VPD. Combined, the higher ET (particularly during the dry season), plus the heavy usage of the forest by the local population and correspondingly poor soil hydrological functioning of the planted forest, are considered to be important factors in the observed decline in dry season streamflow following large-scale reforestation of degraded grass- and scrubland with pines in the region.

[1]This chapter is based on: **Ghimire CP**, Lubczynski MW, Bruijnzeel LA, Chavarro-Rincón D (2014) Transpiration, canopy conductance and decoupling coefficient of two contrasting forest types in the Lesser Himalaya of Central Nepal. Provisionally accepted by *Agricultural and Forest Meteorology.*

3.1 Introduction

The Himalayas have been called the ‘water tower’ of Asia as more than 3 billion people are estimated to depend on the combined flows of the major rivers originating in the region (Bandhyopadyay, 2013). Although recent concerns about the sustainability of streamflow amounts from the area relate primarily to the state and fate of Himalayan glaciers under a changing climate (Immerzeel et al., 2009; Kargel et al., 2011; Scherler et al., 2011; Immerzeel et al., 2013), the rivers originating in the most densely populated part of the mountain range (the Middle Mountain Zone) are mostly rain-fed (Bookhagen and Burbank, 2010; Andermann et al., 2012). Under such conditions, land surface degradation after deforestation on steep unstable slopes may rapidly progress to a point where rainfall infiltration becomes seriously impaired, with reduced replenishment of soil water and groundwater reserves – and therefore streamflow volumes during the dry season – as a result (Bartarya, 1989; Bruijnzeel and Bremmer, 1989; Tiwari et al., 2011; Tyagi et al., 2013). Partly in response to the latter (in the hope to increase streamflows again through improved infiltration after tree planting (cf. Ilstedt et al., 2007)), some 23000 ha of severely degraded pastures and shrublands in the Middle Mountain Zone of Central Nepal were planted to fast-growing coniferous tree species (mainly *Pinus roxburghii* and *P. patula*) between 1980 and 2000 (District Forest Offices, Kabhre and Sindhupalchok, Nepal, unpublished data, 2010). Similar large-scale reforestation elsewhere in the world has been shown to have a profound negative effect on local and regional streamflow production because of strongly increased evapotranspiration (ET) (Trimble et al., 1987; Waterloo et al., 2000; Jackson et al., 2005; Scott et al., 2005; Scott and Prinsloo, 2008). Indeed, villagers and farmers in Central Nepal have expressed serious concern about diminishing dry season streamflows following the large-scale planting of the pines (República, 2012). Water is an increasingly scarce resource in this area (Merz et al., 2003; Schreier et al., 2006; Bandhyopadyay, 2013) which receives about 80% of its annual rainfall during the main monsoon season (June–September; Merz, 2004; cf. Bookhagen and Burbank, 2010). Detailed knowledge of the seasonal variation in vegetation water use and the factors affecting its regulation is, therefore, vital to understanding the impacts of reforestation on streamflow regime in this strongly seasonal environment (cf. Ghimire et al., 2013b; Krishnaswamy et al., 2013).

The magnitude of total ET in forested areas is strongly influenced by the tree transpiration component (E_t), *i.e.*, the water lost through the stomatal pores of the leaves (Bruijnzeel, 2000). The canopy conductance (g_c) regulates the evaporative exchange between plants and the atmosphere and may respond to several environmental variables, notably the vapour pressure deficit (VPD) of the atmosphere, radiation levels, and soil water status (Meinzer et al., 1997; Granier et al., 2000; Pataki and Oren, 2003; Kumagai et al., 2004; Herbst et al., 2008; García-Santos et al., 2009), with wind speed exerting an influence in some cases (Chu et al., 2009). The degree of canopy coupling with the overlying atmosphere (i.e., importance of biological and atmospheric controls of the rate of canopy transpiration) can be conveniently characterised using the decoupling coefficient introduced by Jarvis and McNaughton (1986).

Sapflow-based methods provide a simple, relatively inexpensive and increasingly robust means of continuous measurement of E_t from trees (Granier et al., 1996; Do and Rocheteau, 2002; Nadezhdina et al., 2002; Nadezhdina et al., 2012). Furthermore, measured sap flow rates have been used successfully to derive values of canopy conductance for a range of tropical and subtropical vegetation types by inverse application of the Penman-Monteith equation (e.g., Wullschleger et al., 2000; Motzer et al., 2005; Oguntunde et al., 2007; Gomez-Cardenas, 2008; García-Santos et al., 2009; Alvarado-Barrientos, 2013). The canopy conductance estimated in this way is helpful in better understanding the bulk behaviour of stomata in terms of regulating canopy-scale transpiration in response to changing environmental conditions. However, to the best of our knowledge these canopy processes have not yet been studied quantitatively in the forests of the Himalaya, although spot measurements of stomatal conductance, as well as pre-dawn and mid-day leaf water potentials for natural pine and oak forests in the Kumaun Himalaya (northwest India) have been reported by Zobel et al. (2001a, b) and for several oak species by Garkoti et al. (2000) and Tewari (2000). Similar porometer measurements for five broad-leaved species were conducted outside the monsoon season in the Middle Hills of Central Nepal by Poudyal et al. (2004) but sapflow-based techniques have not been applied in the region thus far.

The primary objective of the present study was, therefore, to examine the sapflow-based diurnal and seasonal variations of transpiration, canopy conductance and decoupling coefficient (*sensu* Jarvis and McNaughton, 1986) for a natural broad-leaved montane forest (dominated by *Castanopsis tribuloides*) and a nearby mature planted coniferous forest (*P. roxburghii*) in the Middle Mountain Zone of Central Nepal. In addition, the information on dry-canopy evaporation (transpiration) is combined with the corresponding wet-canopy evaporation totals (i.e., rainfall interception) obtained for the same sites by Ghimire et al. (2012) and estimates for evaporation from the understory (where present) and the litter layer, to quantify annual ET. Finally, the implications of the present findings for the likely changes in regional dry season streamflows are discussed briefly.

3.2 Study area

The Middle Mountain Zone or Lesser Himalaya, situated between ~800 and ~2400 m above mean sea level (a.m.s.l.) and occupying about 30% of Nepal, is home to ca. 45% of the country's population (based on the 2011 population census; http://cbs.gov.np/). The region is characterised by a complex geology which has resulted in equally complex topography, soil and vegetation patterns (Dobremez, 1976). The geology of the Central Nepalese Middle Mountains consists chiefly of phyllites, schists and quartzites (Stocklin and Bhattarai, 1977). Depending on elevation the climate is humid subtropical to warm-temperate and strongly seasonal with most of the rain falling between June and September. Rainfall varies with elevation and exposure to the prevailing monsoonal air masses (Merz, 2004; cf. Bookhagen and Burbank, 2006). At higher elevations (>2000 m a.m.s.l.) pines and broad-leaved trees are mixed whereas at lower elevations (<1000 m a.m.s.l.) the forest is dominated by deciduous *Shorea robusta*. At intermediate elevations (1000–2000 m a.m.s.l.) a largely evergreen mixed broad-leaved forest dominated by various chestnuts and oaks (*Castanopsis* spp., *Quercus* spp.) and *Schima wallichii* is found, with admixtures of *Rhododendron arboreum* above ca. 1500 m a.m.s.l. Due to the high population pressure, much of this species-rich forest has disappeared (<10% remaining), except on slopes that are too steep for agricultural activity (Dobremez, 1976; Merz, 2004).

The present study was conducted in the Jikhu Khola Catchment (JKC). The 111.4 km^2 JKC (27^{0}35' – 27^{0}41'N; 85^{0}32' – 85^{0}41'E) is situated approximately 45 km east of Kathmandu (the Capital of Nepal) along the Araniko Highway in the Kabhre district between 796 and 2019 m elevation (Figure 3.1). The general aspect of the catchment is southeast and the topography ranges from flat valleys to steep upland slopes (Maharjan, 1991). The geology consists of sedimentary rocks affected by various degrees of metamorphism and includes phyllites, quartzites, and various schists (Nakarmi, 2000). In general, soils in the upper half of the JKC are Cambisols (FAO, 2007) of loamy texture and moderately well to rapidly drained (Maharjan, 1991).

The climate of the JKC is largely humid subtropical, grading to warm-temperate around 2000 m a.m.s.l. Mean (±SD) annual rainfall as measured at mid-elevation (1560 m a.m.s.l.) for the period 1993–1998 was 1487 (±155) mm (Merz, 2004). The main seasons are the monsoon (June–September), the post-monsoon (October–November), winter (December–February), and the pre-monsoon (March–May). The rainy season (monsoon) begins early June and ends by late September. During the monsoon about 80% of the total annual precipitation is delivered. In general, July is the wettest month with about 27% of the annual rainfall. The driest months are November to February, each accounting for about 1% of the annual rainfall (Merz, 2004). The streamflow regime as monitored at the outlet of the entire JKC (1993–2000) is also highly seasonal with a maximum specific discharge of 270 l km^{-2} s^{-1} during the monsoon and a minimum of as little as 0.09 l km^{-2} s^{-1} during the late dry season (Merz, 2004). Average monthly temperatures measured at 1560 m a.m.s.l. range from 7.7 ^{o}C in January to 22.6 ^{o}C in June whereas average monthly relative humidity varies from 55% in March to 95% in September. Strong winds are common during thunderstorms before the onset of the main monsoon, but these are momentary and average monthly wind speeds are always less than 2 m s^{-1} with slight seasonal variation. Annual reference evaporation ET_o [Allen et al., 1998] for the period 1993–2000, was 1170 mm [Merz, 2004]. Vegetation cover in the catchment consists of ~30% forest (both natural and planted), 7% shrubland and 6% grassland, with the remaining 57% largely under agriculture (Merz, 2004). The JKC was subjected to active reforestation until 2004 as part of the Nepal-Australia Forestry Project.

Two forest plots, one in natural forest and the other in mature planted pine forest, were selected for the year-round measurement of sap flow at comparable elevations within the JKC for the current study. The natural forest plot was located on a northwest facing slope and the pine forest plot on a southwest facing slope. Both plots were situated on steep slopes (20–25^{o}) in complex topography and were ca. 2200 m apart (Figure 3.1). The natural forest plot had a surface area of 900 m^2 (30 m x 30 m) and was located at an elevation of 1500 m a.m.s.l. near the centre of an area of remaining natural forest of ca. 20 ha. The plot had a well-developed understory and was floristically highly diverse. Tree density as measured in May 2011 was 1869 trees ha^{-1} with a mean tree height of 14.0 ± 2.2 m (for trees with a diameter at breast height DBH ≥ 5 cm). Average DBH and basal area were 13.6 ±4.4 cm, and 27.1 m^2 ha^{-1}, respectively. More than half of the trees consisted of *Castanopsis tribuloides* (65%) followed by *Schima wallichii* (16%), *Myrica esculenta* (6%), *Rhododendron arboreum* (5%), *Quercus lamellosa* (4%) and various other species (4%). Although largely evergreen, the natural forest sheds a small proportion of its leaves towards the end of the dry season (February–March) but recovers quickly thereafter. For example, the maximum measured leaf area index (LAI, using a Licor 2000 Plant Canopy Analyzer) was 5.43±0.12 (SD) in September 2011 (i.e., at the end of the rainy season), while corresponding values measured during the pre-monsoon period in March, April and early June were 4.52±0.19, 5.14±0.09, and 5.32±0.10, respectively. The soil of the natural forest plot was classified as a Cambisol of clay loam texture. Clay content varied little with depth between 5 and 100 cm (26–29%) as did sand (24–26%) and silt (44–48%) contents. Soil organic carbon (SOC) declined from 4.10±0.25% at 5–15 cm depth to 0.72±0.13% between 50 and 100 cm depth. The topsoil was well covered with a litter layer, had low bulk density (0.93±0.08 g cm^{-3} at 5–15 cm) and high median field-saturated hydraulic conductivity (232 mm h^{-1} at the surface and 82 mm h^{-1} at 5–15 cm). Consequently, infiltration-excess overland flow occurrence was negligible (Ghimire et al., 2013b). During soil profile excavations and in road exposures roots were observed to penetrate into underlying (weathered) bedrock. Depth to bedrock within the plot amounted to ca. 2.3 m.

The pine forest plot also had a surface area of 900 m^2 (30 m x 30 m) and was located at an elevation of 1580 m a.m.s.l. The site formerly consisted of degraded pasture and is part of an area of 25 ha that was reforested with *P. roxburghii* in 1986. The pine trees were 25 years old at the start of sap flow measurements in June 2010. No other tree species were recorded in the plot. An understory was largely absent as grazing by cattle is common. In addition, the local population collects the litter for animal bedding and regularly harvests the grassy herb layer. Pruning of trees for fuelwood, and felling for timber are also common in the pine forests of the JKC (Schreier et al., 2006; cf. Ghimire et al., 2013a) although the research plots themselves remained free from such major disturbance throughout the present investigation (see Ghimire et al., 2013b). Like the natural forest, evergreen *P. roxburghii* sheds a proportion of its needles towards the end of the dry season but also recovers quickly thereafter. The LAI of the pine forest plot was estimated at 2.21±0.10 in September 2011 whereas corresponding values during the pre-monsoon period in March, April and early June were 2.05 ±0.14, 2.15±0.12, and 2.17±0.11, respectively.

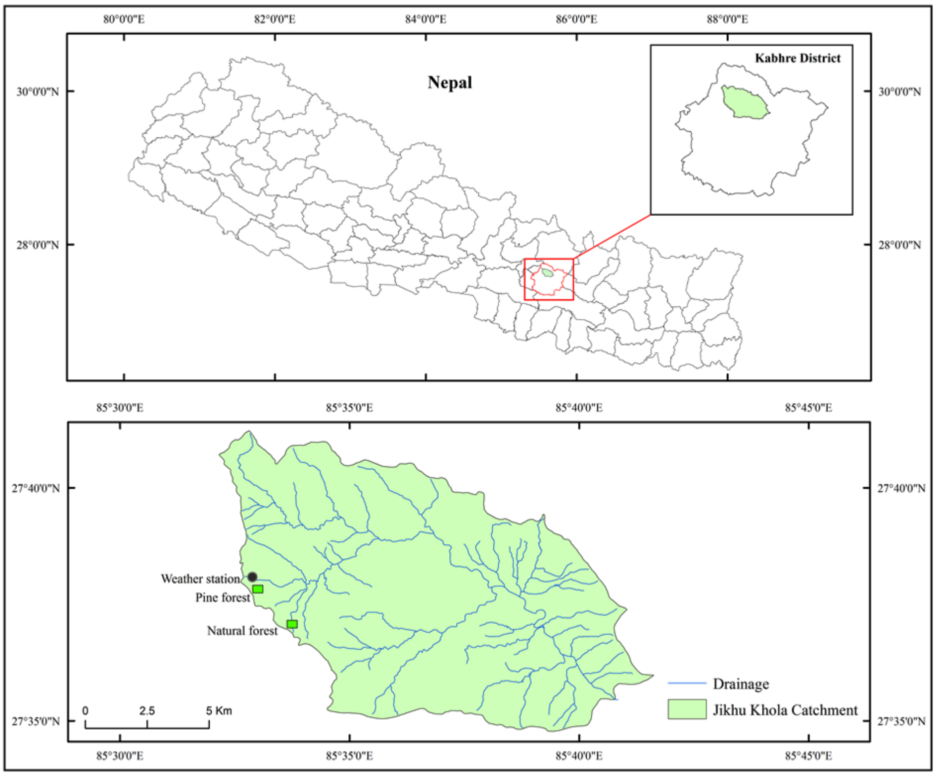

Figure 3.1: Location of study sites in the Jikhu Khola Catchment in the Middle Mountains of Central Nepal.

The stem density, mean DBH and basal area as measured in May 2011 were 853 trees ha^{-1}, 23.65±3.8 cm and 37.6 m^{2} ha^{-1} respectively. The average tree height was estimated at 16.3±3.82 m. The Cambisol underneath the pine forest plot had a silty clay loam texture, with a lower clay content (11–19%) and a much higher sand content (40–47%) than the soil of the natural forest. Because of the regular collection of litter and associated exposure of the soil surface to erosive canopy drip, topsoil carbon levels in the pine forest were much lower (1.7±0.3%) and bulk density significantly higher (1.24±0.10 g cm^{-3}) than in the natural forest plot. In addition, surface infiltration capacity was strongly reduced (median value of 24 mm h^{-1}) and infiltration-excess overland flow frequent (Ghimire et al., 2013b). Again, roots were observed to penetrate into the underlying (weathered) bedrock. Depth to bedrock within the plot was estimated at 1.5 m.

3.3 Methods

3.3.1 Environmental monitoring

Weather conditions were monitored by an automatic weather station located in a degraded pasture near the pine forest site (Figure 3.1). Incident rainfall (P, mm) for each forest plot was recorded in a clearing at a distance of ca. 100 m using a tipping bucket rain gauge (Rain Collector II, Davis Instruments, USA; 0.2 mm per tip) backed up by a manual gauge (100 cm^2 orifice) that was read daily at ca. 08:45 AM local time. Incoming short-wave radiation (R_s) was measured using an SKS 110-pyranometer (Skye Instruments, U.K.). Air temperature (T, ^{o}C) and relative humidity (RH, percentage of saturation) were measured at 2 m height using a Vaisala HMP45C probe protected against direct sunlight and precipitation by a radiation shield. Wind speed and wind direction were measured at 3 m height, using an A100R digital anemometer (Vector Instruments, U.K.) and W200P potentiometer (Vector Instruments, U.K.), respectively. All measurements were recorded at 5-min intervals by a Campbell Scientific Ltd. 23X data-logger. Soil water content was measured at depths of 10, 25, 50, and 75 cm in both forests using TDR (CS616, Campbell Scientific Ltd., type CS616) sensors at 30-min intervals. Values of R_s as measured at the climate station were adjusted for the difference in slope azimuth of the natural forest and pine forest sites using cosine correction methods (Liang, 2004).

3.3.2 Sap flow measurements

The quantification of tree transpiration is widely accomplished by *in situ* xylem sap flow rate (Q_t) measurements (Kumagai et al., 2004; Motzer et al., 2005; McJannet et al., 2007; Do et al., 2008; Chavarro-Rincon, 2009; García-Santos, 2012; Alvarado-Barientos et al., 2013; Reyes-Acosta and Lubczynski, 2013). Sap flow measurements on individual trees involves the measurement of xylem sap flux density (J_p) and sapwood area (A_x) as there is no direct method to estimate Q_t (Granier, 1985; Lu et al., 2004; Lubczynski, 2009):

$$Q_t = J_p \times A_x \tag{3.1}$$

where J_p is conventionally expressed in cm^3 cm^{-2} h^{-1} and A_x in cm^2. Sap flux density in the current study was measured using thermal dissipation probes (TDP), the heat field deformation method (HFD), and the heat ratio method (HRM), whereas sapwood area was estimated from wood cores extracted using an increment borer at breast height (Grissino-Mayor, 2003). The majority of the measurements were made using the TDP method because of its low cost, practical installation and simplicity. The HFD and HRM methods were used only for the purpose of deriving radial correction factors (see text below).

The TDP method deploys a pair of 20 mm long and 2 mm diameter probes which were inserted into the sapwood at a vertical distance of about 100 mm from each other following the procedure outlined by the manufacturer (UP GmbH, Munich, Germany). The upper (downstream) probe was heated continuously at 0.20 W using a constant-current power supply (UP GmbH, Munich, Germany) while the lower (upstream) non-heated probe measured the reference temperature of the sapwood (Granier, 1985). Sensors were always placed on the north side of the trunks to avoid sun-exposure, and were insulated using a radiation shield to prevent any unwanted externally induced heat influence. Given the dense nature of the two study forests, the natural thermal gradients (NTG) as measured along the specifically instrumented trunks (in three trees in the pine forest and four in the natural forest) were always much less than 0.2 ^{o}C (C.P. Ghimire, unpublished data). Therefore, the effect of NTG on the sap flow estimates was considered negligible (cf. Do and Rocheteau, 2002). The temperature difference between the upper heated probe and the lower reference probe was sampled every 30 s, and 5-min averages were recorded on a data-logger (CR23X, Campbell Scientific Ltd.). The recorded temperature difference was converted to sap flux density (J_p) as described by Granier (1985, 1987), subsequently revalidated by

Clearwater et al. (1999) and presented in Lubczynski et al. (2012) as follows:

$$J_p = 42.84\left(\frac{\Delta T_{max} - \Delta T}{\Delta T}\right)^{1.231} \qquad (3.2)$$

where ΔT is the temperature difference between the two probes and ΔT_{max} is the maximum value of ΔT recorded in the absence of transpiration, i.e., when J_p is zero or near zero.

Long-term monitoring of J_p was conducted on nine trees in the natural forest which were considered representative of the dominant species in the forest, and similarly on six trees in the pine forest plot between June 2010 and May 2011. A similar number of trees has been used in (sub)tropical montane forests in Australia (McJannet et al., 2007), Mexico (Alvarado-Barrientos, 2013) and the Canary Islands (García-Santos et al., 2009). A stratified sampling approach was used to select the trees for sap flow measurements, with the aim of representing the frequency of occurrence of tree species (natural forest plot only) and different tree sizes (both plots). In doing so, trees in the respective plots were first classified based on their DBH and subsequently the appropriate numbers of trees were selected from each class based on the frequency of occurrence of the tree in that class. The sampled trees in the natural forest were *C. tribuloides* (n = 4), *S. wallichii* (n = 2), *M. esculenta* (n = 1), *R. arboreum* (n = 1), and *Q. lamellosa* (n = 1). In addition, a sap flow campaign was held from March to May 2011 to capture the sap flow patterns of additional species that were not covered by the long-term monitoring. Sap flow data were collected in four additional plots (30 m x 30 m each) within the pine forest and in two additional plots (also 30 m x 30 m) within the natural forest using a single TDP sensor per tree. A total of 48 additional pine trees were monitored vs. 24 trees in the natural forest during this campaign.

To account for circumferential variation in J_p (cf. Nadezhdina et al., 2002; Alvarado-Barrientos et al., 2013), four TDP probes were inserted at different azimuths (North, South, East and West) in three *C. tribuloides*, two *S. wallichii* and two *P. roxburghii* trees, for at least three continuous days per season. A relationship was established between the sap flux density as measured on northern side of a tree and that on the other sides (i.e., South, East, and West), for each species. Next, the obtained relationships were used to estimate values of J_p at the other azimuths from the measured sap flux density measured on the northern side, for other trees where only the measurement at the northern side was done (cf. Lu et al., 2000). The sap flux density values obtained in this way at the four azimuths were averaged to derive the mean sap flux density which was used for further processing.

To further study radial variations in J_p the heat field deformation method (HFD; Nadezhdina et al., 2012) and the heat ratio method (HRM; Burgess et al., 2001) were used, employing sensors manufactured by ICT International (Armidale, NSW, Australia) in both cases. The HFD sensors consisted of a heater needle and three temperature sensing needles, all 1.6 mm in diameter. The first temperature sensing needle was installed at a distance of 15 mm above the heater (in the axial direction), the second at 5 mm next to the heater (tangential direction) and the third at 15 mm below the heater (axial direction). The HFD sensors have eight measuring points spaced 10 mm apart with the first measuring point located 5 mm inwards from the cambium (Nadezhdina et al., 2002). The HFD method is based on changes in the spatial variation of the heat field around a linear heater inserted into the sapwood. Two temperature differences measured around the heater have been found to describe this field, viz. the symmetrical (dT_{sym}) and the asymmetrical temperature difference (dT_{asym}), which allows the calculation of J_p ($cm^3\ cm^{-2}\ h^{-1}$) according to Nadezhdina et al. (1998, 2002, 2012):

$$J_p = 3600D \times \frac{K + dT_{sym} - dT_{asym}}{dT_{asym}} \times \frac{Z_{ax}}{Z_{tg}L_{sw}} \qquad (3.3)$$

where D is the thermal diffusivity of fresh sapwood, K is the absolute value of $dT_{sym} - dT_{asym}$ under conditions of zero sap flow (^oC), dT_{sym} is the temperature difference between the axial thermistor junctions (^oC), dT_{asym} the temperature difference between the tangential thermistor junctions (^oC), Z_{ax} is the axial distance between the upper or lower needle and the heater (15 mm), Z_{tg} the tangential distance between the heater and the tangential needle (5 mm) and L_{sw} is the sapwood depth (cm). The HRM sensors consisted of three 35 mm long needles. The top and bottom probes contain thermistors located at 7.5 mm and 22.5 mm from the tip of each probe. The third and centrally located probe consists of a line heater that runs along the full length of the probe to deliver a uniform pulse of heat through the sapwood (Burgess et al., 2001).

Thermal diffusivity (*D*) was determined experimentally for the dominant tree species in the natural forest (*C. tribuloides* and *S. wallichii*) and in the planted forest (*P. roxburghii*). For each tree species, three fresh core samples were taken at breast height. The samples were stored in a pre-weighed air-tight glass vessel to prevent weight loss due to evaporation and subsequently transported to the laboratory for the measurement of fresh weight and volume. Afterwards, the samples were oven dried at 60 ^{o}C to determine their dry weights. D (cm^2 s^{-1}) was then calculated using the method outlined by Marshall (1958):

$$D = \frac{K}{(c \times \rho)} \times 10000 \tag{3.4}$$

where *K* is the thermal conductivity of the wood (W m^{-1} K^{-1}), *c* is the specific heat capacity of the sapwood (J kg^{-1} K^{-1}) and ρ is the fresh wood density (kg m^{-3}). All these parameters were obtained from measured fresh weight, dry weight and fresh volume for each sample using the methods outlined in Vandegehuchte and Steppe (2012). The average values of *D* obtained in this way were 0.00254, 0.00256, and 0.00252 cm^2 s^{-1} for *C. tribuloides*, *S. wallichii* and *P. roxburghii*, respectively. The presently obtained *D* values are very similar to the commonly used *D* value for pine trees (0.0025 cm^2 s^{-1}; Marshall, 1958).

Radial sap flow patterns were measured using the HFD method in eight trees in the pine forest and in two *C. tribuloides* trees in the natural forest. Radial patterns for two other dominant species in the natural forest (*S. wallichii*, n = 2; *M. esculenta*, n = 1) were derived using the HRM. All measurements were carried out for at least three consecutive days per tree.

Because most of the sap flux density measurements in the present study were made using TDP sensors which had a measurement length of only 20 mm, the method described by Poyatos (2007) was used to obtain radial correction factors across the entire sapwood depth. First, a reference sap flux density ($J_{p,\ ref}$) was calculated in the outer 20 mm of sapwood from the average of the two shallower junctions of the HFD sensor (i.e., at 5 mm and 15 mm). According to the nominal measuring range of 5 mm around each measuring point along the probe, the first measuring point at 5 mm should be representative of the first 10 mm while the second point at 15 mm should account for the remaining 10 mm, so the first two external sensors cover the entire length of the

Granier probe (20 mm). For the trees that were monitored using HRM sensors, $J_{p,\,ref}$ was assumed to be equivalent to the value measured by the thermistors at 7.5 mm (see section 3.4.4). To account for the radial sap flux density profile in each tree sampled using TDP sensors, sap flux density ($J_{p,\,i}$) at the six measuring points (or one measuring point in the case of the HRM sensor) was related to $J_{p,\,ref}$ to yield a radial correction coefficient for each depth, $C_{rad,\,i}$ in which the subscript i indicates the corresponding depth (mm):

$$C_{rad,i} = \frac{J_{p,i}}{J_{p,ref}} \tag{3.5}$$

In the case of trees with a sapwood depth in excess of 8.0 cm (or 3.0 cm for the HRM sensor), the radial correction factor was extrapolated by fitting the linear regression line between $C_{rad,\,i}$ and sapwood depth. Next, the radial correction factors were used in combination with the TDP-based sap flux measurement (i.e., $J_{p,\,2cm}$) to obtain an estimate of the sap flux density for the entire sapwood depth according to:

$$J_{rad,i} = C_{rad,i} \times J_{p,2cm} \tag{3.6}$$

Finally, whole-tree sap flow rates were calculated assuming each measuring point sensed a length interval of 10 mm (or 15 mm for the HRM), centered around the measuring point (cf. Poyatos, 2007; Hatton et al. 1990):

$$Q_t = \sum_{i=1}^{i=n} J_{rad,i} \times A_i \tag{3.7}$$

where $J_{rad,i}$ is the sap flux density after applying the radial correction factors and A_i represents the area of the corresponding concentric ring of sapwood.

3.3.3 Modeling sap flow for gap filling

During the one year of measurements of sap flux density in the two forests, technical and logistical limitations caused gaps in the records. The two main problems encountered were power shortage and the resin produced by *P. roxburghii* which caused malfunctioning of TDP sensors on several occasions. Various methods have been proposed to simulate sap flow dynamics for gap-filling purposes, including multiple linear regression, empirical models, artificial neural networks (ANN), and the Penman-Monteith model (Cienciala et al., 2000; Liu et al., 2009; Whitley et al., 2009). In the current study, ANN were used to fill the gaps in the sap flow record because of its demonstrated better performance under a range of climatic conditions including montane ones (e.g., García-Santos et al., 2009; Whitey et al., 2009; Liu et al., 2009; Rey-Acosta and Lubczynski, 2013). ANN represents a purely statistically-based response to the meteorological forcing per time step (Abramowitz, 2005) and is essentially a black-box model consisting of highly non-linear relations between input and output variables (Schaap and Bouten, 1996). In this study, the ANN model included incoming short-wave radiation, vapour pressure deficit, temperature and wind speed – all at a 30-min time step. A three-layer feed-forward and back-propagation neural network trained by the Levenberg-Marquardt algorithm was used. The layers consisted of an input layer, one hidden layer and an output layer. The transfer functions used for the hidden and output layers were, respectively, a hyperbolic tangent sigmoid transfer function (tansig) and a logarithmic sigmoid transfer function (logsig), as described by Liu et al. (2009). The ANN model was trained using at least five days of sap flow data, while the model was subsequently tested against at least three days of independent sap flow data. The model performance was assessed using the Nash-Sutcliffe (1970) efficiency criterion. The ANN model was trained and tested for both dry and wet conditions and subsequently used to predict sap flow during the periods for which data were missing.

3.3.4 Tree transpiration at stand level

After correcting for radial variation and filling the gaps in the sap flow records, the water use of the sampled trees was scaled to the plot level to estimate the transpiration per unit area of land (E_t) in the natural and planted forest. This was done in five steps. First, simple least-squares regressions were derived between stem area (A_s) and sapwood area (A_x) using measurements of sap wood depth (L_{sw}) and tree DBH values. This was done for 54 trees in the pine forest, and for eight (*R. arboreum*) to seventeen (*C. tribuloides*) trees in the natural forest. This approach avoided the necessity to core every tree in the respective plots and determine their A_x directly. Second, values of A_x of all sampled trees in a plot were assigned to different classes based on their daily average J_p

(cm^3 cm^2 d^{-1}) as measured in the first 20 mm of sapwood. Third, to take into account the radial variability of J_p, the A_x values of all sampled trees in a stand were divided into successive concentric sapwood areas (A_{xi}) of 1 cm width each (cf. Reyes-Acosta and Lubczynski, 2013). Fourth, the sap flow (cm^3 h^{-1}) of individual trees in the stand was calculated by combining $J_{p,\,i}$ with corresponding A_{xi} values using Equation 3.7. Finally, the daily water use of all trees present in a plot (Q, cm^3 d^{-1}) was summed up and divided by the plot area (A, m^2) to determine the daily stand transpiration rate (E_t, mm d^{-1}):

$$E_t = \frac{Q}{A} \times 10^{-3} \tag{3.8}$$

Note that plot area was adjusted for slope angle so that transpiration was expressed per unit horizontal surface area.

3.3.5 Canopy conductance and decoupling coefficient

Canopy conductance (g_c, mm s^{-1}) was estimated from the actual value of canopy transpiration (derived from sap flow measurements) using the inverted Penman – Monteith equation as follows:

$$g_c = \frac{g_a \gamma \lambda E_t \times 1000}{\Delta R_n - \lambda E_t(\Delta + \gamma) + \rho_a c_p g_a (e_s - e_a)} \tag{3.9}$$

where g_a is the aerodynamic conductance (mm s^{-1}), γ the psychrometric constant (66.5 Pa K^{-1}), λ the latent heat of vaporisation (2.465 MJ kg^{-1}), E_t the measured stand transpiration (mm s^{-1}), Δ is the rate of change of saturation water vapour pressure with temperature (Pa K^{-1}), R_n the available energy (considered to be equal to net radiant energy, in W m^{-2}), ρ_a (kg m^{-3}) is the density of the air, c_p the specific heat of air at constant pressure (1.01 J g^{-1} K), e_s (Pa) is the saturation vapour pressure and e_a (Pa) the actual vapour pressure. The aerodynamic conductance was estimated according to the Thom (1975) formula despite its limitations for use in steep terrain (Holwerda et al., 2012). To obtain the values of net radiant energy (R_n) from incoming short-wave radiation for the respective forest canopies, albedo values of 0.12 (Oguntoyinbo, 1970) and 0.10 (Waterloo et al., 1999) were used for the natural forest and pine forest, respectively, in conjunction with the Brunt formula for the estimation of net long-wave radiation (Kijne, 1973).

The degree of coupling of the canopy to the atmosphere was estimated in terms of the dimensionless decoupling coefficient, Ω, as introduced by McNaughton and Jarvis (1983):

$$\Omega = \frac{\left(\Delta/\gamma + 1\right)}{\left(\Delta/\gamma + 1 + g_a/g_c\right)} \tag{3.10}$$

The value of Ω can vary between zero (indicating perfect coupling with the atmosphere) and one (for complete isolation from the atmosphere). Therefore, stomatal control of transpiration becomes progressively weaker as Ω approaches unity.

3.3.6 Total evapotranspiration

In order to derive estimates for seasonal and annual evapotranspiration (ET) totals for the two forests, the presently obtained transpiration (E_t) totals were first combined with the corresponding interception losses as estimated using Gash's revised analytical model (Gash et al., 1995). The Gash model was run on a daily basis for the entire period between 1 June 2010 and 31 May 2011 using the forest structural and average evaporation model parameters established by Ghimire et al. (2012) for the same sites and daily rainfall values as input. To this should be added the contributions by understory evaporation (natural forest only in view of the absence of understory vegetation in the pine plantation; Ghimire et al., 2013b) and evaporation from the forest floor (cf. Walsh and Voight, 1977; Bruijnzeel, 2000). Neither component was measured directly and therefore estimated on the basis of findings obtained at other sites having comparable forest structural and climatic characteristics. Details are given in Section 3.5.

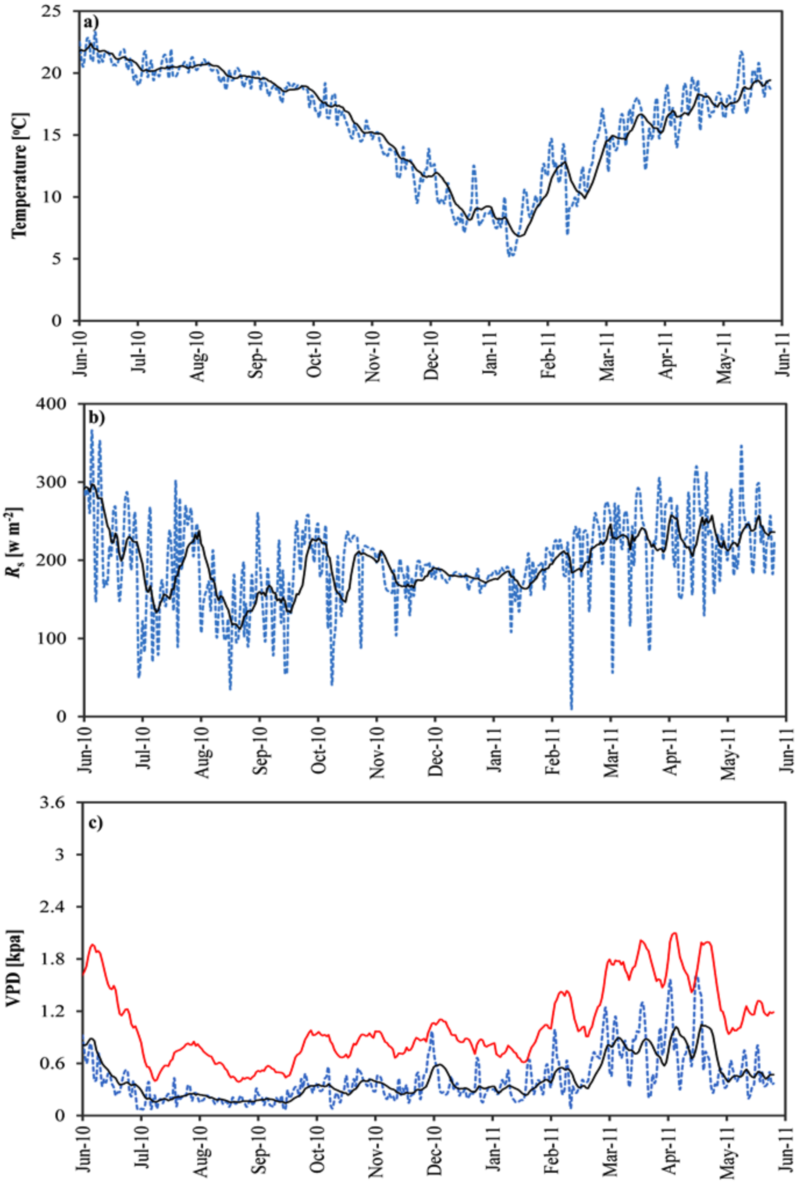

Figure 3.2: Daily climatic data as measured at 1620 m a.m.s.l. between 1 June 2010 and 31 May 2011 near Dhulikhel, Central Nepal: (a) temperature (^{o}C), (b) incoming short-wave radiation (R_s, W m^{-2}), and (c) vapour pressure deficit (VPD, kPa). Note the black lines indicate 10-day moving averages. Note also the red solid line indicating the 10-day moving averages for daily maximum VPD values.

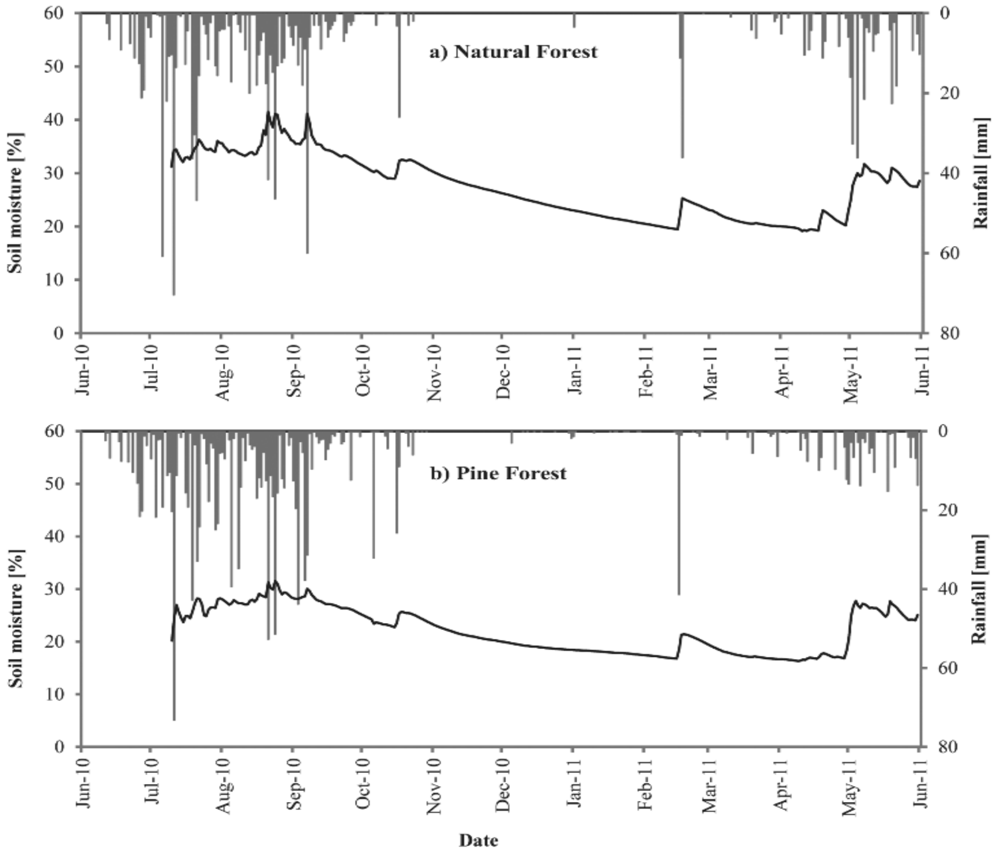

Figure 3.3: Daily rainfall (P, mm) and soil moisture (%) in the top 75 cm in natural and planted forest between 1 June 2010 and 31 May 2011 near Dhulikhel, Central Nepal: (a) Natural forest, and (b) Pine forest.

3.4 Results

3.4.1 Meteorological measurements

Between 1 June 2010 and 31 May 2011, a total of 1331 mm of rain was received at the pine forest site, whereas the corresponding amount at the natural forest site was 1423 mm. More than 70% of the annual rainfall was delivered during the main monsoon season at both sites, viz. 953 mm at the natural forest and 1084 mm at the pine forest (see also Table 3 below). Figure 3.2 further shows the seasonal variations in average daily temperatures, short-wave radiation R_s and VPD as measured at the degraded pasture site during the study year. The average daily temperature was 16.1 (± 4.4) ^{o}C with a minimum of 5.2 ^{o}C in January and a maximum of 24.4 ^{o}C in June. Average daily R_s was 198.8 (± 57.5) W m^{-2} with generally lower values during the monsoon months and higher values during the post-monsoon and pre-monsoon periods as expected. Likewise, average daily VPD was 0.42 (±0.29) kPa with highest monthly values (0.77–0.85) observed in March–April and minimum values (0.18–0.60) during the rainy season (Figure 2c). Values of maximum daytime VPD followed a similar patter ranging from 1.03 during the rainy season to ~3 kPa in March–April (Figure 3.2c). Wind speeds varied little seasonally and ranged from 1.44 m s^{-1} in August to 1.73 m s^{-1} in March (not shown).

3.4.2 Soil moisture

Both the natural and the planted forest plot remained relatively wet throughout the monsoon and late pre-monsoon season. There was little or no rainfall during the dry season and moisture depletion occurred continuously from early November to mid- February (Figure 3.3). The more clayey soil of the natural forest consistently exhibited higher volumetric moisture contents than the sandier pine forest soil, with estimated annual average values of 27.4(±5.8)% and 22.0(±4.3)%, respectively.

3.4.3 Sapwood area

All of the studied tree species in the natural and planted forest sites showed a strong (linear) relationship between their functional xylem area (A_x, cm^2) and stem cross-sectional area (A_s, cm^2), with coefficients of determination for the various relationships being consistently above 0.8 (Figure 3.4). Table 1 lists the average values of DBH, sapwood depth L_{sw} and A_x for each investigated species. The planted pine trees (25 years old at the time of measurement) were much larger (mean DBH 24.4 cm) than the main canopy trees in the natural forest plot (mean DBH for all species <15 cm). The average L_{sw} was highest for *P. roxburghii* (9.0 cm) and lowest for *S. wallichii* (3.1 cm). Sapwood depths for the other investigated species in the natural forest were much smaller (range: 3.1–4.6 cm) than the average L_{sw} for the pine. Average A_x was also highest for *P. roxburghii* (315.8 cm^2) and lowest for *R. arboreum* (47.2 cm^2).

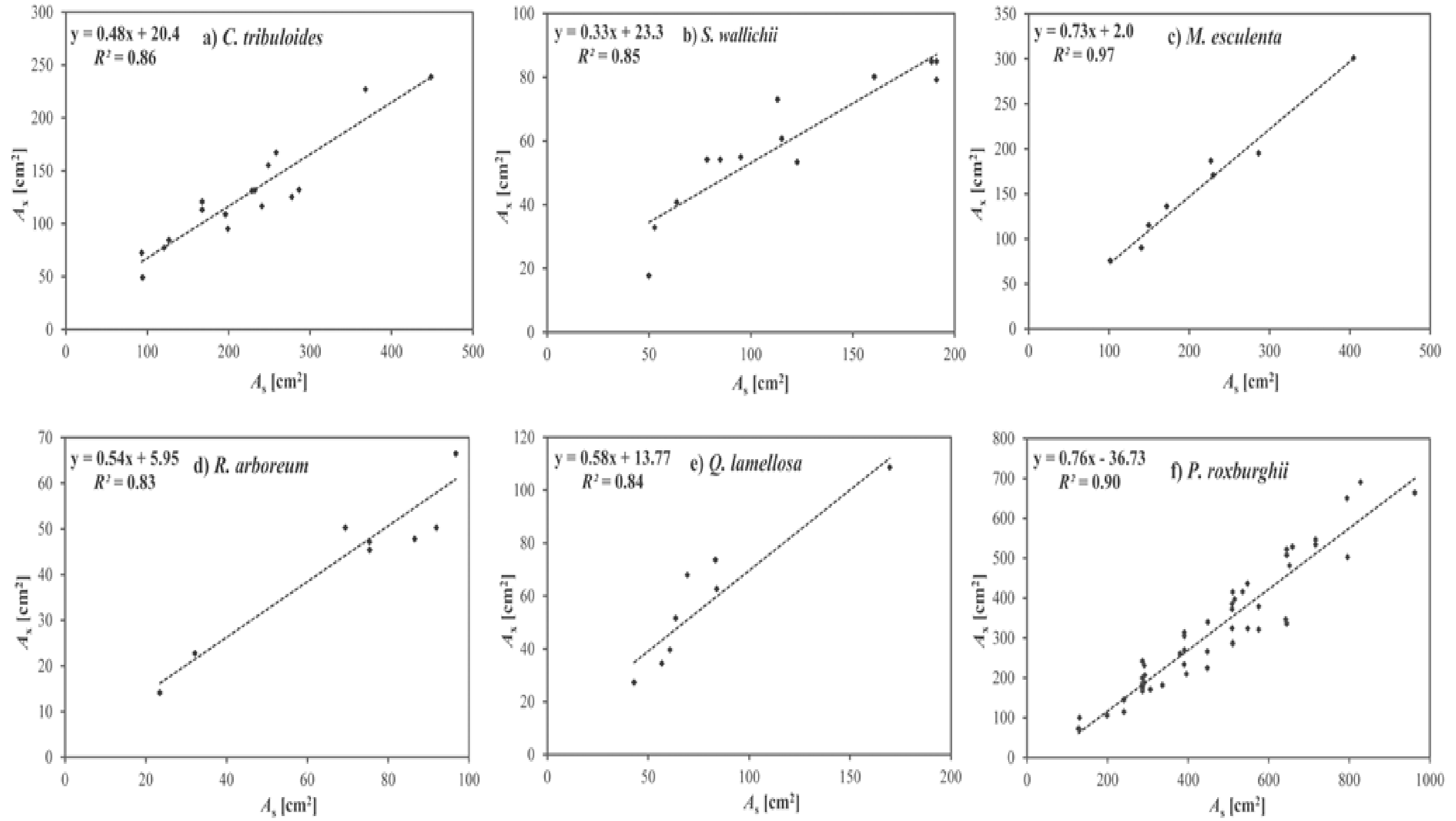

Figure 3.4: Relationship between stem area (A_x, cm^2) and sapwood area (A_s, cm^2) for several tree species near Dhulikhel: (a) *Castanopsis tribuloides*, (b) *Schima wallichii*, (c) *Myrica esculenta*, (d) *Rhododendron arboreum*, (e) *Quercus lamellosa*, and (f) *Pinus roxburghii* (25 years old).

Despite the similarity in L_{sw} among tree species in the natural forest, the variation in tree sapwood area was much larger due to the differences in tree size occurring in the plot (Table 3.1). There was a 2- to 2.5-fold difference in L_{sw} and A_x between the planted *P. roxburghii* and the dominant tree species in the natural forest (*C. tribuloides*) suggesting potentially large differences in transpiration between the two forest types.

Table 3.1: Diameter at breast height (DBH), sapwood depth (L_{sw}) and sapwood area (A_x) for selected tree species in natural broad-leaved forest and coniferous plantation forest Dhulikhel, Middle Mountains, Central Nepal. Values represent averages and their standard deviation whereas values in parentheses denote the sample size (*n*).

Species	DBH (cm)	L_{sw} (cm)	A_x (cm^2)
Castanopsis tribuloides	13.8 ± 4.7 (193)	4.6 ± 1.6 (17)	131 ± 49.06 (17)
Schima wallichii	9.8 ± 3.86 (67)	3.1 ± 0.81 (13)	60.7 ± 17.7 (13)
Myrica esculenta	14.5 ± 5.58 (20)	4.3 ± 0.95 (8)	170.6 ± 75.7 (8)
Rhododendron arboreum	8.4 ± 1.82 (22)	3.9 ± 0.54 (8)	47.2 ± 14.09 (8)
Quercus lamellosa	8.3 ± 2.35 (20)	4.0 ± 0.55 (8)	62.6 ± 27.3 (8)
Pinus roxburghii	24.4 ± 6.23 (370)	9.0 ± 2.55 (54)	315.8 ± 156.7 (54)

3.4.4 Diurnal and radial variations in sap flux density

Figure 3.5 shows the typical diurnal courses of 30-min values of R_s and VPD for two consecutive clear-sky days after the main monsoon (i.e., no soil water deficit; cf. Figure 3.3) plus corresponding sap flux densities (J_p) and sap flow rates (Q_t) for the pines and for the various hardwood species of the natural forest. Diurnal patterns of J_p were similar for all species studied (Figure 3.5b, e). Daily variations in J_p appeared to be mainly associated with the variations in VPD and R_s. Typically, J_p and hence Q_t increased rapidly in the morning as VPD and radiation increased and generally reached a maximum around mid-day, declining again during the afternoon (Figure 3.5). A more gradual decline in J_p was observed after sunset in all species studied. Quantitatively, J_p values were comparable among the species studied, except for *R. arboreum* which had very low sap flux density (maximum J_p < 2 cm^3 cm^{-2} h^{-1}). In line with the latter observation, *R. arboreum* also exhibited the lowest sap flow rates (< 100 cm^3 h^{-1}) (Figure 3.5c). Despite the similarity in J_p for most of the species, whole-tree water consumption rates (Q_t) of

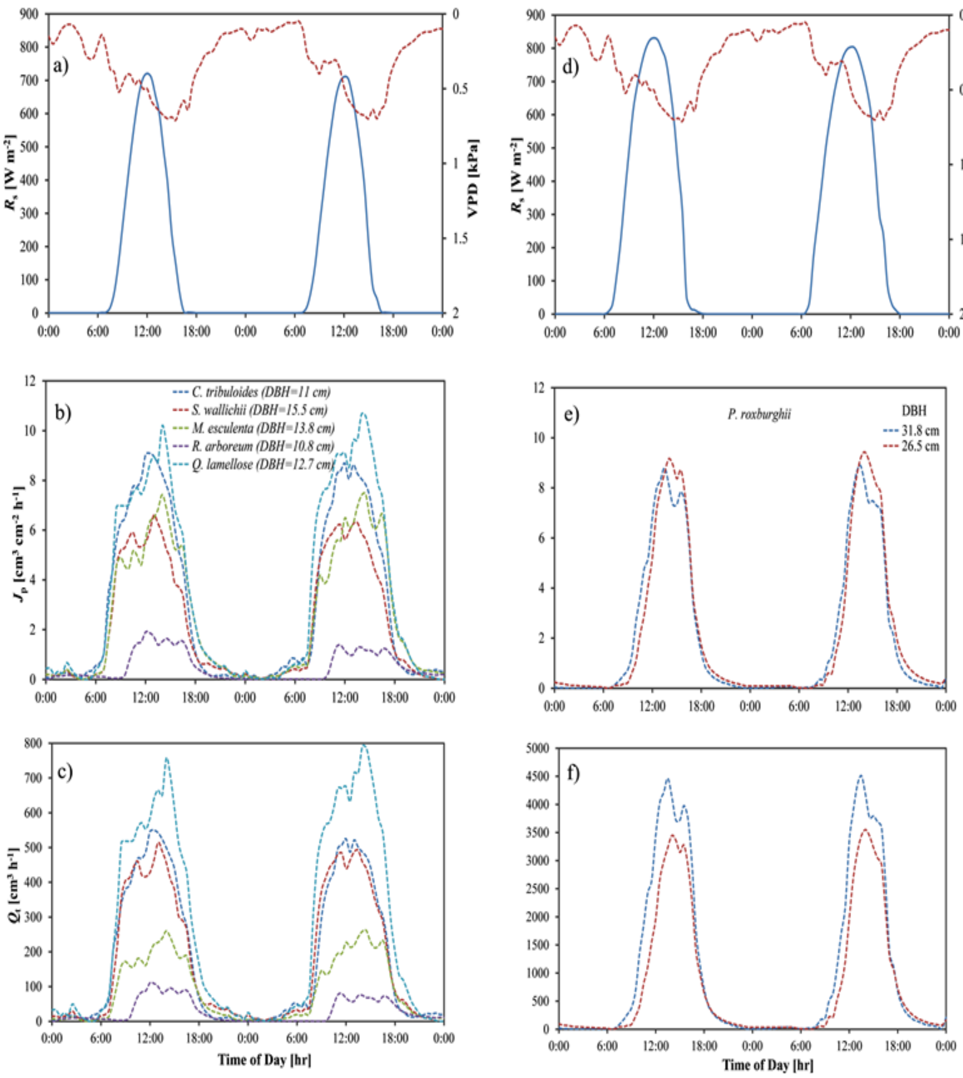

Figure 3.5: Diurnal course of short-wave radiation (R_s, W m^{-2}) and atmospheric vapour pressure deficit (VPD, kPa) and corresponding sap flux densities (J_p, cm^3 cm^{-2} h^{-1}) and whole-tree sap flow (Q_t, cm^3 h^{-1}) during two bright days (25 and 26 October 2010) for contrasting tree species near Dhulikhel, Central Nepal. Left-hand panels (a–c): natural broad-leaved forest, right-hand panels (d–f): *Pinus roxburghii* plantation. Note that values of R_s were adjusted to include the effect of different slope exposure. Note also the difference in scale for Q_t in the case of natural forest species (panel 3.5c) and *P. roxburghii* (panel 3.5f).

P. roxburghii were much higher than those of the natural forest species (Figure 3.5c, f) because of the much greater sapwood depth of *P. roxburghii* compared to the dominant natural forest species (cf. Table 3.1).

Figure 3.6 shows the radial patterns of J_p for the pines and three dominant species in the natural forest (*C. tribuloides, S. wallichii* and *M. Esculenta*). Radial patterns of J_p showed a certain similarity between species in that the highest flows were most commonly recorded at the outermost measuring point, i.e. 5 mm depth for the HFD measurements in *P. roxburghii* and *C. tribuloides*) and 7.5 mm for the HRM measurements in *S. wallichii* and *M. esculenta*. However, the rate of decline in J_p with depth was quite different between species, ranging from steep declines in the case of *C. tribuloides*, *S. wallichii* and *M. esculenta* to a (much) more gradual decline in *P. roxburghii* where the maximum J_p at a depth of 7.5 cm was still ~35% of the maximum value observed at the outermost measuring point (Figure 3.6d). In contrast, the radial pattern was (much) steeper in the natural forest species with the maximum J_p observed at 25 mm depth (or 22.5 mm, depending on the type of sensor used) already being reduced to ~15% (*S. wallichii* and *M. esculenta*) and up to 40% (*C. tribuloides*) of the maximum J_p at the outermost measuring point (Figure 3.6). Such contrasts in radial sap flow patterns between the pine forest and the natural forest species further contribute to differences in transpiration rates between the two forest types (see below). Moreover, the much steeper decline in sap flux density in the natural forest species also suggests that the TDP sensor alone can provide an estimate of sap flow with a reasonable accuracy for the investigated species. The ratio of sap flow across the outer 2 cm of sapwood depth (TDP equivalent) to that over the entire sapwood depth (radially corrected sap flow using HFD and HRM sensors) was greater than 0.8 for all natural forest species investigated. Additional support for this contention comes from the fact that the outermost 20 mm of the sapwood not only has the highest flow rates but by definition also represents the largest corresponding sapwood area and hence transfer the greatest sapflow volumes. In addition, the circumferential and axial variation in sap flux density in both the natural and the pine forest species were very small (results not shown).

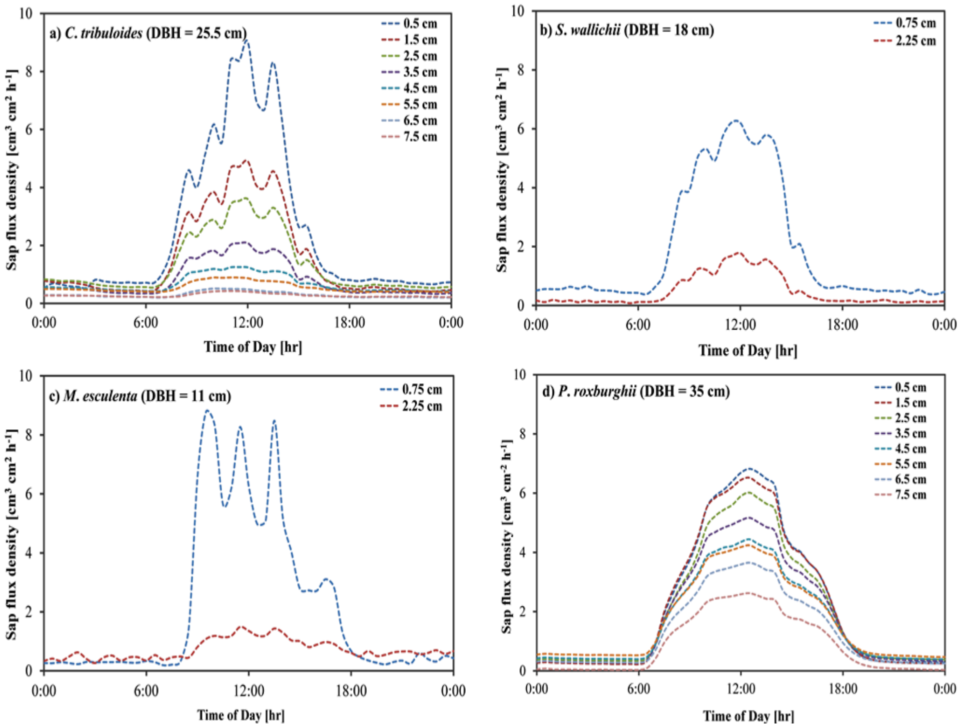

Figure 3.6: Radial sap flux density patterns for: (a) *Castanopsis tribuloides*, (b) *Schima wallichii*, (c) *Myrica Esculenta* trees, and (d) mature planted *Pinus roxburghii* trees near Dhulikhel, Central Nepal.

3.4.5 Daily, seasonal and annual stand transpiration

Using the up-scaling technique described in Section 3.3.4, time series of stand–level daily tree transpiration totals (E_t) were derived showing distinct seasonal variation in both the natural forest and the pine forest (Figure 3.7; Table 3.2). Although the seasonal patterns of E_t were similar, the transpiration rate of the pine forest was higher than that of the natural forest throughout the study period even when allowing for the inferred difference in short-wave radiation inputs (~25% lower at the natural forest site) due to the difference in site exposure (cf. Figure 3.1). Both sites showed higher transpiration rates during the dry season months (October–May) compared to the wet season months (June–September). Over the entire study period the highest monthly average daily E_t was recorded in May at both sites (natural forest, 0.58 mm d^{-1}; pine forest, 0.93 mm d^{-1}) while the lowest values were recorded in September (0.33 and 0.57 mm d^{-1}, respectively; Table 3.2).

The annual tree transpiration total for the natural forest was estimated at 163 mm of which 48 mm occurred during the wet season (June–September) and 115 mm during the remaining months. In contrast, estimated annual transpiration for the pine stand was much higher at 280 mm, with corresponding seasonal totals of 78 mm (wet season) and 202 mm (dry season). Total daily tree transpiration exhibited a plateau-shaped relationship with VPD at both sites. Transpiration rates increased sharply with VPD at low levels (VPD < ~0.4 kPa), but tended to level off at higher VPD values (Figure 3.8a, d). Despite considerable scatter in the data, increases in R_s up to values of ~150 W m^{-2} produced a reasonably linear increase in E_t, after which transpiration tended to level off for higher values of R_s (Figure 3.8b, e). Daily average R_s, however, had to exceed an apparent 'threshold' value of ca. 100 W m^{-2} before a transpiration rate greater than 0.2 mm d^{-1} was observed at either site. Stand transpiration was not clearly related to temperature (Figure 3.8c, f).

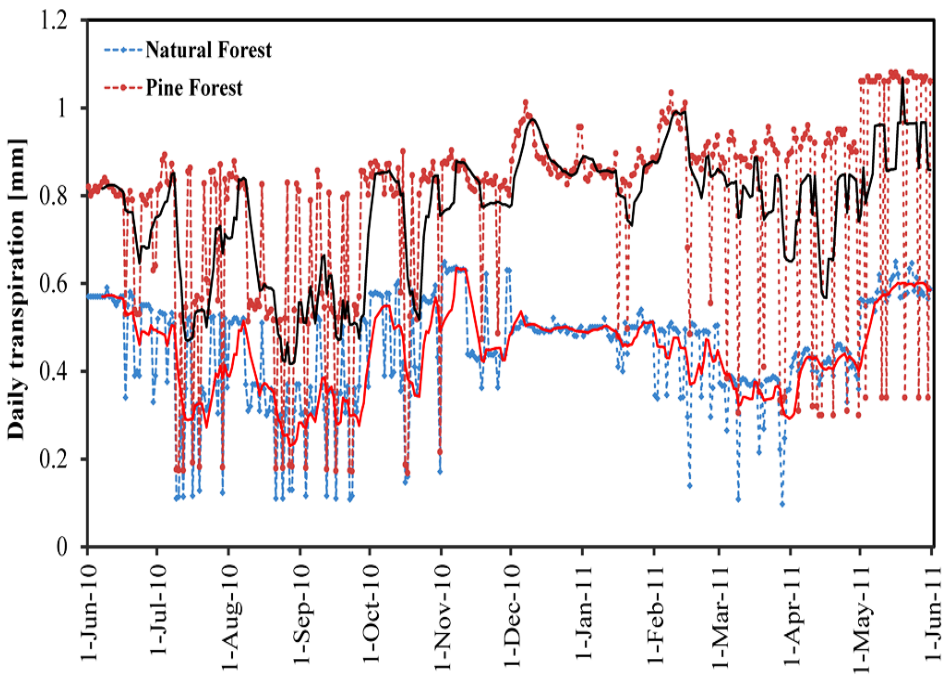

Figure 3.7: Daily transpiration totals (mm) between 1 June 2010 and 31 May 2011 in: (a) a natural, broad-leaved forest and (b) a planted coniferous forest near Dhulikhel, Central Nepal. Red and black solid lines indicate the 7-day moving averages for natural and pine forest transpiration totals, respectively.

3.4.6 Canopy conductance and decoupling coefficient

Figure 3.9 shows the diurnal patterns of 30-min estimates of R_s, VPD, canopy conductance as derived from the inverse application of the Penman-Monteith equation (g_c, mm s^{-1}), and the decoupling coefficient (Ω) following the method of McNaughton and Jarvis (1983) for *P. roxburghii* and two dominant species in the natural forest (*C. tribuloides* and *S. wallichii*) on a clear day.

Canopy conductances and decoupling coefficients for the species investigated followed a typical diurnal pattern; the peak values of both g_c and Ω were recorded around mid-day (natural forest species) to early afternoon (pines) and declined in the afternoon (natural forest species) to late afternoon (pines) (Figure 3.9). However, the morning rise and afternoon declines in g_c and Ω of the dominant species from the natural forest were much steeper while the maximum values were sustained for much longer compared to those observed in the pine forest. The patterns for g_c and Ω closely mirrored that for VPD in an inverse manner. A temporary decline of somewhat greater magnitude in g_c was observed in the early afternoon in response to an increase in VPD, for both forest types. Half-hourly estimates of Ω were less than 0.2 for most of the time (Figure 3.9), indicating (very) good coupling between the tree canopies and the atmosphere.

Considerable day-to-day and seasonal variation in g_c was observed in both the natural and the planted forest (Figure 3.10; cf. Table 3.2). Maximum daily estimates of g_c ranged from 4.85 mm s^{-1} in the natural forest to 11.46 mm s^{-1} in the planted forest (Figure 3.10), and averaged 1.69 and 3.28 mm s^{-1}, respectively. Values of g_c were highest during the post-monsoon period (November–February) and lowest during the driest part of the year (March–April) when some leaf shedding occurred in both forests. Similar seasonal variation was observed for daily Ω (Figure 3.10; Table 3.2). Daily Ω averaged 0.06 and 0.07 in the natural and planted forest, respectively. The analysis of the dependence of g_c on climatic variables showed only a poor relationship with R_s but the inverse relationship with VPD explained most of the daily variation in g_c in both the natural forest and the planted forest (Figure 3.11). Interestingly, with increasing VPD, canopy conductance did not fall to zero but stabilized at a small but constant positive value in both cases (Figure 3.11). Increases in average daily temperatures produced lower g_c values. Pertinently, g_c

was not clearly related to cumulative moisture content in the top 75 cm of the soil suggesting access to moisture contained in deeper soil layers and the (weathered portion of the) geological substrate.

Table 3.2: Monthly estimates of mean daily transpiration (E_t), canopy conductance (g_c) and decoupling coefficient (Ω) for a natural broad-leaved forest and a mature coniferous plantation forest near Dhulikhel, Central Nepal

Month	Natural forest			Pine forest		
	E_t [mm d^{-1}]	g_c [mm s^{-1}]	Ω	E_t [mm d^{-1}]	g_c [mm s^{-1}]	Ω
June 2010	0.51	1.13	0.05	0.74	1.92	0.05
July 2010	0.37	1.59	0.07	0.64	3.37	0.09
Aug 2010	0.35	1.65	0.08	0.61	3.41	0.09
Sep 2010	0.33	1.77	0.08	0.57	3.44	0.09
Oct 2010	0.48	1.83	0.07	0.74	3.03	0.07
Nov 2010	0.52	2.36	0.08	0.82	4.05	0.08
Dec2010	0.50	2.17	0.07	0.90	4.73	0.09
Jan 2011	0.50	2.72	0.07	0.83	4.76	0.07
Feb 2011	0.43	1.56	0.05	0.90	3.63	0.06
March 2011	0.34	0.67	0.02	0.77	1.61	0.03
April 2011	0.42	1.06	0.05	0.76	1.64	0.04
May 2011	0.58	1.64	0.06	0.93	3.43	0.07

3.5 Discussion

3.5.1 Control of transpiration

The present results provide the first quantitative information on transpiration rates in natural and planted forest in the Lesser Himalaya of Nepal. Strong seasonal variations in daily transpiration totals (Figure 3.7) and derived values of canopy conductance (Figure 3.10) in response to the marked seasonality in VPD, temperature and incoming solar radiation (Figure 3.2), were observed for both forest types (cf. Table 3.2).

As would be expected on the basis of prevailing radiation levels, the magnitude of VPD and the frequency of wet canopy conditions suppressing transpiration (cf. Rutter, 1967; Tanaka et al., 2003), average daily transpiration rates were lower during the rainy season than during most of the dry season (Figure 3.7). Both forests transpired readily during much of the dry season, except during the period of leaf senescence and partial defoliation (March–April) (Figure 3.7 and Table 3.2). Indeed, throughout the full canopy stage, there was little variation in tree water uptake and derived g_c despite the progressive depletion of soil water in the upper soil layers (Figure 3.3). Therefore, the roots must have had access to moisture present in deeper layers (almost certainly including the underlying saprolite given the observed presence of roots in the latter) during this time of the year. A similar conclusion was drawn by Poudyal et al. (2004) for *Castanopsis indica* and *Schima wallichii* trees at Phulchowki Hill, situated some 25 km to the West of the present sites, on the basis of a lack of response in stomatal conductance (g_s) during prolonged dry periods. The relative uniformity in pattern and peak transpiration rates at the Dhulikhel sites in the late dry season (May, Figure 3.7), was striking. Such findings support the results of other studies according to which tree water use may even increase during the dry season due to a combination of continued moisture availability, high radiant energy and absence of transpiration suppression by canopy wetting (O'Grady et al., 1999; Tanaka et al. 2003; Tanaka et al., 2008).

Throughout the study period the pine forest exhibited higher transpiration rates relative to the natural forest, with estimated annual totals of 163 (0.45 mm d^{-1}) and 280 mm (0.77 mm d^{-1}), respectively. The difference remained even after allowing for the higher insolation levels of the pine forest site (by 25%) and is thus most probably largely attributable to the greater sapwood depth and the much flatter radial sap flow pattern of the pines compared to those of the dominant hardwood species of the natural forest (cf. Figure 3.6; Table 3.1). To the best of our knowledge, there are no other data available on rates of vegetation water uptake in the Himalayan region although results for stomatal conductances for *P. roxburghii* and dominant hardwood species will be discussed in the next section.

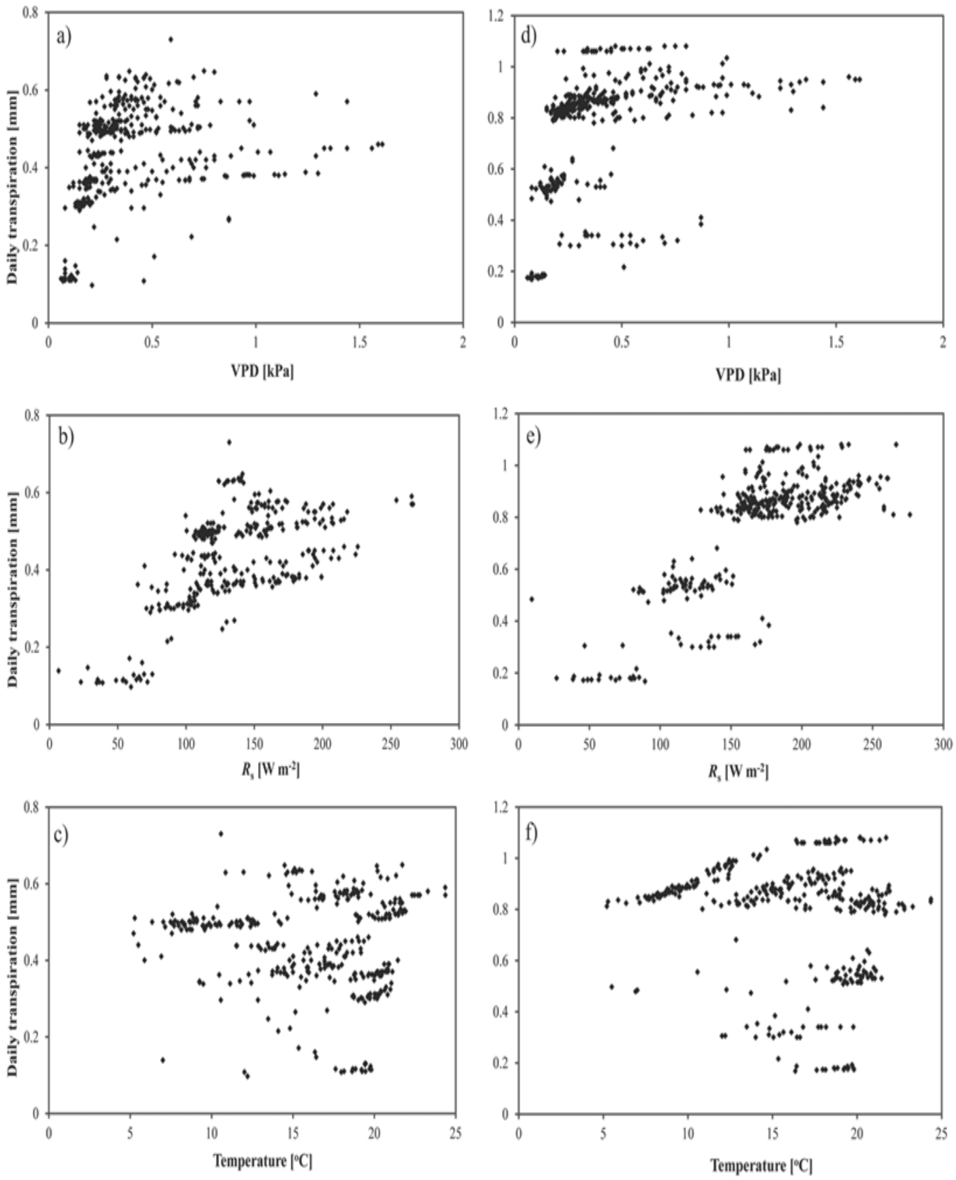

Figure 3.8: Relations between daily transpiration totals (mm) and atmospheric vapour pressure deficit (VPD, kPa), incoming short-wave radiation (R_s, W m^{-2}), and temperature (^oC), in natural broad-leaved forest (left-hand panels a–c) and planted coniferous forest (right-hand panels d–f) near Dhulikhel, Central Nepal. Note the differences in vertical scale between the two forests.

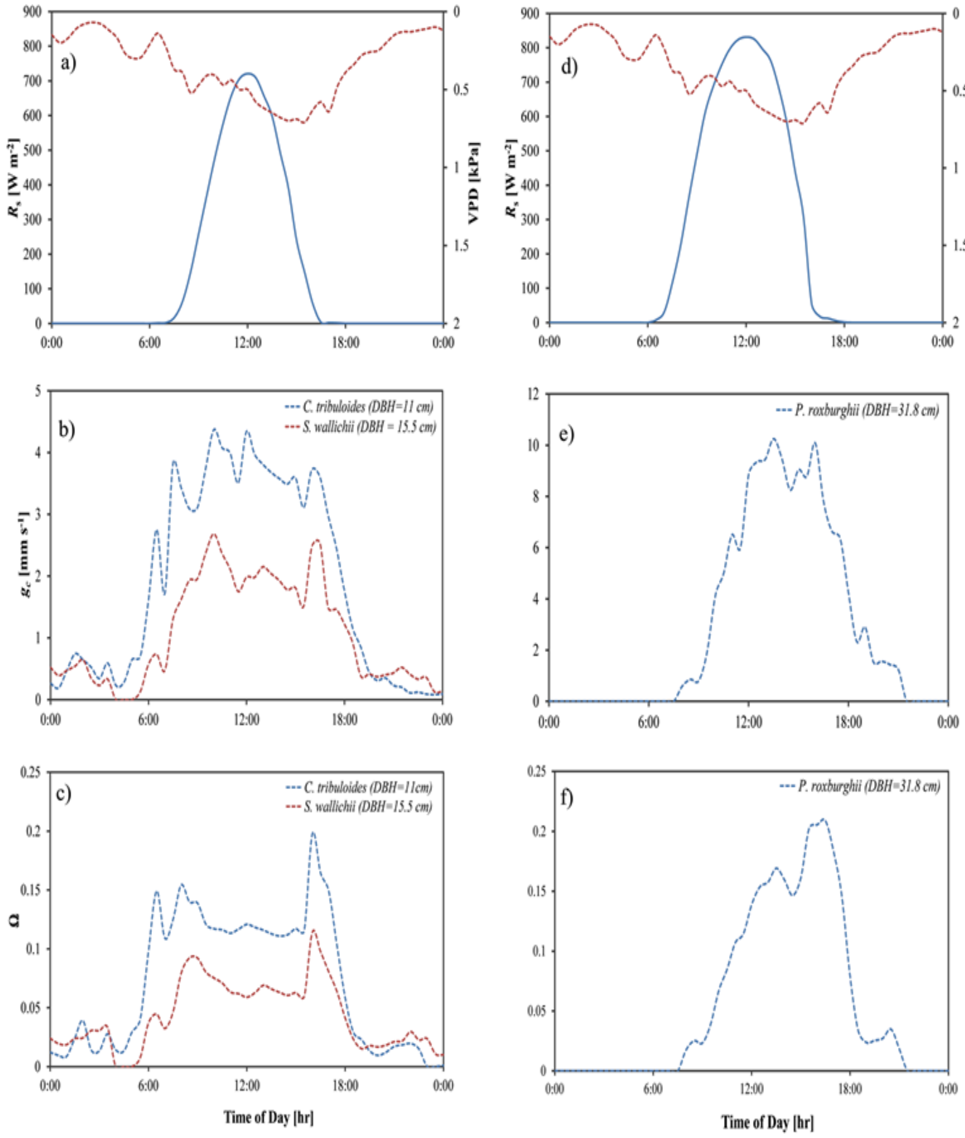

Figure 3.9: Diurnal course of canopy conductance (g_c) and decoupling coefficient (Ω) on a clear day (25 October 2010) for various tree species near Dhulikhel, Central Nepal. R_s is incoming short-wave radiation (W m^{-2}) (adjusted for site exposure) and VPD is vapour pressure deficit (kPa). Left-hand panels (a–c): natural broad-leaved forest, right-hand panels (d–f): *Pinus roxburghii* plantation. Note the differences in scale g_c between the two forests.

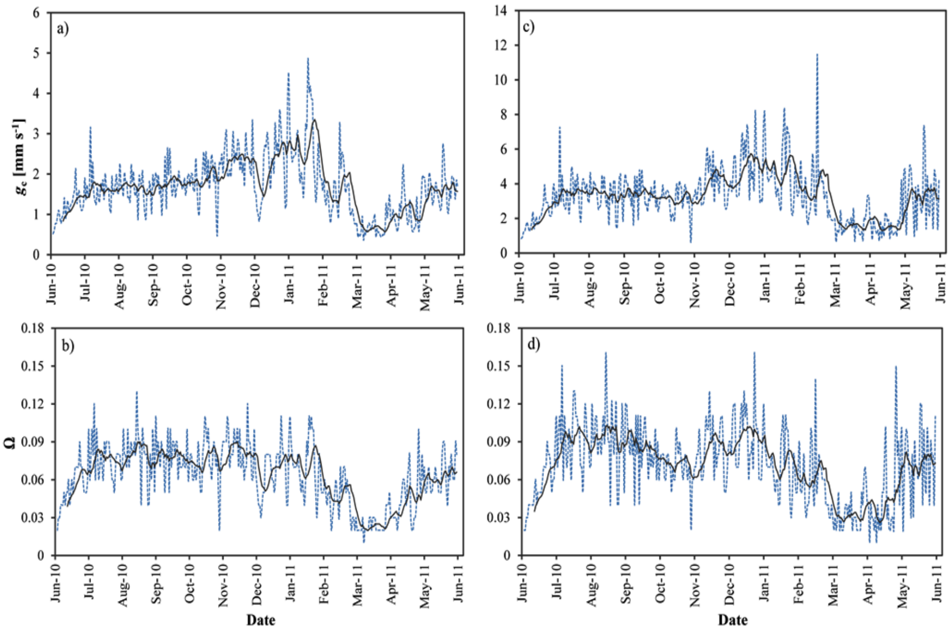

Figure 3.10: Average daily canopy conductance (g_c, mm s^{-1}) and decoupling coefficient (Ω) of a natural broad-leaved forest (left-hand panels a–b) and a planted coniferous forest (right-hand panels c–d) between 1 June 2010 and 31 May 2011 near Dhulikhel, Central Nepal. Black lines indicate 10-day moving averages. Note the difference in scale for g_c between the two forest types.

The presently obtained annual transpiration total for the natural forest (163 mm) appears to be very low compared to values reported for a range of tropical lower montane forests (646 ± 123 mm yr^{-1}; Bruijnzeel et al., 2011) even after allowing an extra 20% evaporation by the understory vegetation (based on the findings of Motzer et al. (2010) for a similarly sized and exposed lower montane forest in southern Ecuador) and normalizing the data for ambient evaporative conditions by dividing by some kind of reference evaporation (e.g., ET_o or Penman's open-water evaporation, E_o). For example, at 0.155 the value of E_t / E_o for the presently studied natural forest (including an added 20% for understory evaporation) is a mere fraction of that derived for a similarly sized (14 – 18 m tall, LAI 5.7) lower montane forest in South Ecuador (0.85, Motzer, 2003) or a somewhat taller lower montane forest in eastern Mexico (~0.85; based on data presented in Holwerda et al., 2013). The presently found low ratios suggest strong stomatal control in the Dhulikhel forest compared to the much wetter conditions prevailing in southern Ecuador or eastern Mexico (Motzer, 2003; Gomez-Cardenas, 2008; Motzer et al., 2010; Holwerda et al., 2013).

Although comparative data for other pine forests from the Himalaya appear to be lacking, the presently found annual E_t of 280 mm for the 25-year-old pine stand at Dhulikhel is very similar to the 252 mm reported by Luis et al. (2005) for a mature stand of *P. canariensis* (a close relative to *P. roxburghii*; Zobel et al., 2001a) on Tenerife, Canary Islands at a comparable latitude (28^o 35'N) and elevation (1650 m a.m.s.l.). It is also comparable to the 260 mm yr^{-1} obtained for a mature (~35 years) *P. patula* plantation in montane eastern Mexico using similar sapflow equipment and methods (Alvarado-Barrientos, 2013). Although the E_t / ET_o ratio differs somewhat between the Nepalese and Mexican stands (0.24 vs. 0.30, respectively), upon normalizing for forest LAI (2.2 vs. 3.0 m^2 m^{-2}, respectively) the difference effectively disappears (0.11 and 0.10, respectively). Interestingly, despite much higher absolute (645 mm yr^{-1}) and relative transpiration totals (E_t / ET_o = 0.80), water use for a nearby vigorous 10-year-old *P. patula* plantation at the Mexican site was essentially the same after normalising for canopy LAI (0.12; Alvarado-Barrientos, 2013).

Diurnal and seasonal variation in transpiration at the Dhulikhel sites appeared to be mainly associated with variations in VPD and to a lesser extent R_s (Figures 3.5 and 3.7). As such, it is not surprising that daily differences in VPD and R_s were mirrored in the daily variation in E_t. However, because VPD and R_s co-vary throughout the year (Figure 3.2), it is often difficult to separate the relative contributions of these two climatic variables to overall transpiration. Daily transpiration exhibited a plateau-shaped relationship with VPD (Figure 3.8) at both sites. Transpiration rate increased sharply with VPD at low levels of VPD (< 0.4 kPa), but tended to level off at higher VPD levels, indicating stomatal closure (cf. Figure 3.11). Similar relationships between transpiration rates and VPD have been observed elsewhere in the montane tropics (Motzer et al., 2005; McJannet et al., 2007). There also appears to be a reasonably linear relationship between daily transpiration and average daily R_s (Figure 3.8) as also found by others (Wullschleger et al., 2000; Motzer et al., 2005; McJannet et al., 2007). The radiation level needed to reach maximum transpiration was around 200 W m^{-2} for both forests. This partly supports the finding of higher transpiration rates during a major part of the dry season when days with clear-sky conditions were frequent

(Figure 3.2). As mentioned earlier, the comparative lack of response of g_c to changes in soil water content at the present and many other sites (e.g. Tanaka et al., 2003; Poudyal et al., 2004; Luis et al., 2005; Tanaka et al., 2008; García-Santos et al., 2009) must be due to continued access to moisture in deeper layers throughout the dry season.

3.5.2 Canopy conductance and decoupling coefficient

Although there is no other work published to date on canopy conductance values for Himalayan forest vegetation, it is of interest to convert the presently derived seasonal variations in g_c (Table 3.2) to corresponding approximate *stomatal* conductances (g_s) by dividing g_c by LAI to allow a preliminary comparison with existing information on seasonal change in g_s for various Himalayan tree species (e.g., Zobel et al., 2001a; Poudyal et al., 2004). Derived approximate values of g_s for the natural forest at Dhulikhel were lowest (0.2 mm s^{-1}) in April and highest (0.50 mm s^{-1}) in January. These values are very much lower than the overall mean g_s of 3.1–3.25 mm s^{-1} determined by porometry for *Castanopsis indica* or the 3.1–4.0 mm s^{-1} for *Schima wallichii* at 1400 m a.m.s.l. near Kathmandu during non-monsoon months (Poudyal et al., 2004; converting original values given in mmol m^{-2} s^{-1} to mm s^{-1} by dividing by 24; Luis et al., 2005). Corresponding values for the Ecuadorian and Mexican lower montane forests referred to earlier were ~6.9 m s^{-1} (Motzer et al., 2005) and 3.0–10.8 mm s^{-1} (Gotsch et al., 2013), respectively. Taking the Dhulikhel data at face value would suggest this forest to exert strong stomatal control at all times although the apparent contrast between the current sap flow-based estimates of g_s and earlier porometer-based estimates requires further work.

Derived approximate values for g_s in *Pinus roxburghii* at Dhulikhel were also lowest (0.76 mm s^{-1}) in April and highest (2.15 mm s^{-1}) in January. As such, values were distinctly higher than those derived for the nearby natural forest at all times of the year. Zobel et al. (2001a) obtained comparable values for *P. roxburghii* in the somewhat drier Indian Kumaun Himalaya during the post-monsoon period (1.2–1.7 mm s^{-1}) whereas Luis et al. (2005) reported values between 1.1 and 2.9 mm s^{-1} for g_c (corresponding to g_s values of 0.3–0.85 mm s^{-1}) in a mature Canary pine stand (LAI, 3.4 m^2 m^{-2}) on Tenerife at a comparable latitude and elevation to the Dhulikhel pine forest. Interestingly, Zobel et al. (2001a) considered g_s for their *P. roxburghii* trees to be at the lower end of published values for pines which lends some support for the presently inferred low stomatal values.

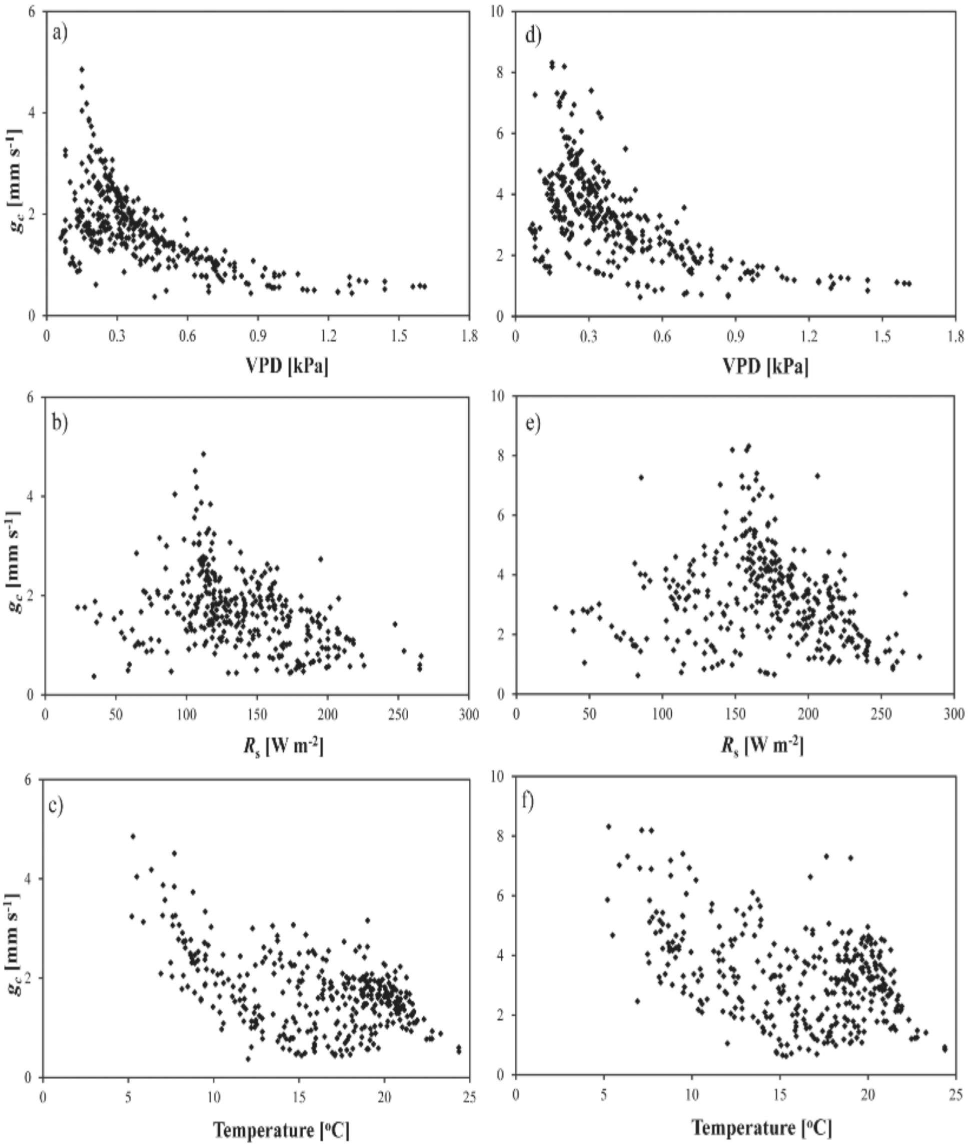

Figure 3.11: Relations between average daily canopy conductance (g_c, mm s^{-1}) and vapour pressure deficit (VPD, kPa), incoming short-wave radiation (R_s, W m^{-2}), and temperature (^{o}C), in natural broad-leaved forest (left-hand panels a–c) and planted coniferous forest (right-hand panels d–f) near Dhulikhel, Central Nepal. Note the differences in scale for g_c between the two forest types.

Characteristic exponential decreases in g_c with increasing VPD were observed at both forest sites, something also reported in similar studies of montane forests elsewhere (Tanaka et al., 2003; Luis et al., 2005; Motzer et al., 2005; García-Santos et al., 2009). However, with increasing VPD, canopy conductance did not decline entirely to zero but stabilised at a low value in both forests (Figure 3.11). Canopy conductance showed a poor relationship with incoming radiation at both forest sites. This behaviour may be attributed to a low threshold of saturation for light (~150 W m^{-2}), such that radiation was not limiting during most of the time (cf. Granier et al. 1996). The responses of canopy conductance to temperature were nearly identical for both forests, that is, an exponential decline in g_c with increase in temperature was found. Similar relationships between g_c and temperature have been observed elsewhere in the tropics (Sommer et al., 2002; García-Santos et al., 2009) although Poudyal et al. (2004) reported an *increase* in (stomatal) conductance with temperature for *Castanopsis* and *Schima* (but not in *Rhododendron* or various oaks) under similar conditions elsewhere in the Middle Mountains of Central Nepal. Zobel et al. (2001a) also found g_s of *P. roxburghii* in the Kumaun Himalaya to increase with temperature, again in contrast to the present findings.

A marked seasonality in g_c was observed in response to VPD (Figure 3.10). Higher canopy conductance was normally observed at low VPD values (< 0.4 kPa; Figure 3.11). Canopy conductance values reached a minimum in March (natural forest) and April (pine forest) in response to higher vapour pressure deficits (monthly average VPD >0.75 kPa; Figure 3.2). However, because maximum VPD, leaf senescence (Zobel et al., 2001a, b) and partial defoliation (cf. Section 3.2) all coincided, it is difficult to separate the relative importance of the respective factors.

It is well documented that stomata exert high control on canopy transpiration when Ω approaches zero, and as a result transpiration becomes increasingly dependent on VPD and less dependent on the amount of radiant energy received (Magnani et al., 1998; Wullschleger et al., 2000; Kumagai et al., 2004). Over the entire study period, the average daily decoupling coefficient was always less than 0.16, with annual average values being as low as 0.06 (natural forest) and 0.07 (pine forest). This indicates a very high degree of canopy coupling to the atmosphere and thus strong stomatal control over transpiration (Jarvis and McNaughton, 1986). Decoupling coefficients of this magnitude also indicate that transpiration rates from the natural and planted forest are determined mostly by daily variations in VPD and much less by daily incoming solar radiation (Schulze et al., 1995). A similarly high degree of canopy coupling to the atmosphere has been observed for some tropical forests (Roberts et al., 1990; Granier et al., 1996; Vourlitis et al., 2002; McJannet et al., 2007) but not in others (range: 0.24–0.43; Kumagai et al., 2004; Motzer et al., 2005), possibly due to differences in seasonality, rooting depths and soil water storage capacity between sites (cf. Poudyal et al., 2004; Tanaka et al., 2008). Decoupling coefficients at Dhulikhel closely followed the course of VPD in an inverse manner, with slightly lower **Ω-** values during the dry season, suggesting greater coupling during that part of the year.

Table 3.3: Summary of rainfall and estimated evapotranspiration components for a mature planted pine forest and a natural broad-leaved forest near Dhulikhel, Middle Mountains, Central Nepal. See text for understory and forest floor evaporation estimates.

	Natural Forest			Pine Forest		
	Wet	Dry	Total	Wet	Dry	Total
Rainfall (*P*, mm)	953	378	1331	1084	338	1423
Transpiration (E_t, mm)	48	115	163	78	202	280
Interception (E_i, mm)	203	90	293	184	78	262
Understory evaporation (E_{us}, mm)	-	-	33	-	-	-
Litter evaporation (E_s, mm)	35	-	35	35	-	35
ET (mm)	286	205	524	297	280	577

3.5.3 Evapotranspiration and implication for dry season flows

Ultimately, the hydrological effect of vegetation in terms of site water yield is determined by its total evapotranspiration (ET). In order to derive annual ET totals for the Dhulikhel forests, the presently obtained transpiration totals were adjusted to allow for the inclusion of understory transpiration (where present) and combined with annual interception losses as estimated using the revised analytical rainfall interception model of Gash et al. (1995). In addition, evaporation from the forest floor was estimated.

Evaporation by the understory is determined largely by below-canopy levels of insolation and ventilation (and therefore by the degree of coupling with the atmosphere above the main canopy) as well as by understory stomatal conductances (Black and Kelliher, 1989; Motzer, 2005; Alvarado-Barrientos, 2013). Likewise, evaporation from the forest floor (litter layer) depends on these same atmospheric factors (Kelliher et al., 1992) plus the type and mass of litter, with pine litter typically exhibiting much lower water retention capacity than broad-leaved litter (Dabral et al., 1963; Walsh and Voight, 1977).

In view of the high LAI and percentage canopy cover of the natural forest (5.4 m^2 m^{-2} and 81%, respectively; Ghimire et al., 2012), the northwesterly exposition of the site and the low wind speeds prevailing in the study area, evaporative contributions by the understory would be expected to be modest Motzer et al. (2010), working in a lower montane forest of comparable stature (14–18 m) and LAI (5.7 m^2 m^{-2}) in Ecuador, determined the evaporative fraction contributed by the understory at ca. 20% of the transpiration by the overstory. Adopting this value for the Dhulikhel natural forest gave an estimated annual E_t of 196 mm (Table 3.3).

The analytical interception model was run on a daily basis for the entire period over which transpiration was measured (1 June 2010–31 May 2011) using the forest structural and evaporative model parameters established by Ghimire et al. (2012) for the same forest sites to estimate annual wet canopy evaporation losses as being 293 mm in the natural forest and 262 mm in the pine forest (Table 3.3).

Finally, evaporation from the forest floor (E_s) was also considered to be low for the natural forest for the reasons given above plus the strongly seasonal rainfall regime which causes most of the rain to fall within a four-month period and leaving the forest floor relatively dry for much of the remaining time (cf. Ghimire et al., 2012). Comparative measurements of E_s are rare but amounts determined in various tropical lowland rain forests range from 35–70 mm yr^{-1} (equivalent to 2–4% of corresponding E_0; Jordan and Heuveldop, 1981; Roche, 1982) to ~110 mm yr^{-1} (0.3 mm d^{-1}) in a well-watered New Zealand evergreen broad-leaved forest (Kelliher et al., 1992). Translating such findings to the Dhulikhel situation (E_0 = 1260 mm yr^{-1}) and assuming the bulk of E_s to take place during the rainy season (four months) gave estimated values of 17–35 mm yr^{-1}, respectively. Forest floor evaporation in the New Zealand forest constituted ca. 18% of transpiration (Kelliher et al., 1992). Applying a similar fraction to the E_t determined for the Dhulikhel forest (196 mm yr^{-1}) gave an estimated value for E_s of 36 mm yr^{-1}. Therefore, a value of 35 mm yr^{-1} was adopted as a first estimate for E_s in the natural forest, bringing its overall estimated ET to 524 mm yr^{-1}. Of this total, dry-canopy evaporation (E_t, including understory transpiration) made up an estimated 37% and wet-canopy evaporation (interception) ca. 56%, with the remaining 6–7% contributed by forest floor evaporation (Table 3.3).

Whilst absolute rainfall interception totals did not differ too much between the coniferous and the broad-leaved forest (31 mm yr^{-1} higher for the latter despite a lower rainfall total), both the seasonal and annual transpiration totals were distinctly higher for the pine forest even after allowing an extra 20% to be transpired by the understory in the broad-leaved forest (Table 3.3). Wet-season transpiration in the pine forest was some 20 mm higher than in the natural forest vs. 64 mm during the dry season. As for the estimation of E_s in the more open pine forest (LAI, 2.2 m^2 m^{-2}, canopy gap fraction 0.27; Ghimire et al., 2012), one would expect this to be somewhat higher than in the nearby natural forest. However, the pine litter is typically harvested immediately after the main leaf-shedding period in the dry season by the local population for use as animal bedding and composting (Ghimire et al., 2013a, b). Therefore, amounts of litter present on the forest floor (and therefore moisture retention capacity; Putuhena and Cordery, 2000) during the subsequent rainy season will be much reduced. In addition, a substantial fraction of the rainfall in this forest runs off as infiltration-excess overland flow (Ghimire et al., 2013b). Both factors will tend to reduce E_s. Waterloo et al. (1999) determined forest floor evaporation in a similarly stocked stand of *Pinus caribaea* in Fiji (LAI = 3.5 m^2 m^{-2}) to be 9% of E_0. Applying the same fraction and taking again an effective period of four rainy months yielded an estimated E_s for the Dhulikhel pine forest of 35 mm yr^{-1} which may be an over estimate in view of much smaller litter mass present in the Dhulikhel forest compared to the Fijian pine plantation. Combining the estimates for E_t, interception loss and E_s gives an estimated annual ET for the pine forest of 577 mm, which is 53 mm higher than ET for the nearby natural forest (Table 3.3). Wet canopy evaporation constituted ca. 45% of annual ET in the pine forest and transpiration ca. 48%.

The presently derived estimates of annual evapotranspiration and their components for the studied natural and pine forest at Dhulikhel constitute a first for the Himalayan region. As such, it is not possible to make meaningful comparison with other Himalayan ecosystems, also because any estimates of ET based on catchment water budgets (e.g. Rawat et al., 2000; Tiwari et al., 2011) are fraught with uncertainty due to the strongly spatially variable rainfall and the very real possibility of significant catchment leakage in this mountainous and heavily faulted terrain (Bruijnzeel and Bremmer, 1989; Andermann et al., 2012). There is a clear need for additional studies of Himalayan forest evaporation and its components using complementary plant physiological and hydrological techniques.

Although the present finding of a slightly higher ET (~10%, Table 3.3) for planted pine forest compared to natural broad-leaf forest is not going to have major hydrological consequences on an annual basis, the much higher water use of the pines during the dry season (Table 3.3) is likely to result in a corresponding reduction in water yield upon converting natural broad-leaf forest to pine plantations, especially during the more vigorous early growth stage (Bruijnzeel, 1997; Scott and Prinsloo, 2008; Alvarado-Barrientos, 2013). The contrast is even greater when comparing the presently established ET of this study against the estimated evaporation from degraded pasture in the same area (~200 mm yr^{-1}; Baral, 2012). Along with the over- intensive usage and correspondingly poor soil hydrological functioning of planted forests in the area (Ghimire et al., 2013a,b), such contrasts must be considered important when interpreting the regionally observed decline in baseflows (República, 2012) following the large-scale planting of pines on degraded land.

3.6 Conclusion

For the first time, tree transpiration, canopy conductances and decoupling coefficients were quantified and examined in a mature planted pine forest and in a natural broad-leaved forest in the Middle Mountain Zone of the Himalayan region. The average daily transpiration rate and canopy conductance were higher in the pine forest than in the natural forest. Transpiration of both forests was largely dominated by vapour pressure deficit as indicated by very low decoupling coefficients. Transpiration rates in both forests peaked in the dry season and exhibited little variation during much of the dry season, indicating the roots must have access to deeper soil layers and the weathered portion of the geological substrate. The higher evapotranspiration total obtained for the planted coniferous forest, particularly its higher transpiration during the dry season, is in line with the observed decline in dry season baseflows following the large-scale reforestation of degraded grass- and scrubland in the Middle Mountains of Central Nepal.

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Chapter 4

Reforesting severely degraded grassland in the Lesser Himalaya of Nepal: Effects on soil hydraulic conductivity and overland flow production[1]

Abstract. Severely degraded hillslopes in the Lesser Himalaya challenge local communities as a result of the frequent occurrence of overland flow and erosion during the rainy season, and water shortages during the dry season. Reforestation is often perceived as an effective way of restoring pre-disturbance hydrological conditions but heavy usage of reforested land in the region has been shown to hamper full recovery of soil hydraulic properties. This study investigates the effect of reforestation and forest usage on field-saturated soil hydraulic conductivities (K_{fs}) near Dhulikhel, Central Nepal, by comparing degraded pasture, a footpath within the pasture, a 25-year-old pine reforestation, and little disturbed natural forest. The hillslope hydrological implications of changes in K_{fs} with land-cover change were assessed via comparisons with measured rainfall intensities over different durations. High surface- and near-surface K_{fs} in natural forest (82–232 mm h^{-1}) rules out overland flow occurrence and favours vertical percolation. Conversely, corresponding K_{fs} for degraded pasture (18–39 mm h^{-1}) and footpath (12–26 mm h^{-1}) were conducive to overland flow generation during medium to high intensity storms and thus to local flash flooding. Pertinently, surface- and near-surface K_{fs} in the heavily used pine forest remained similar to those for degraded pasture. Estimated monsoonal overland flow totals for degraded pasture, pine forest and natural forest were 21.3%, 15.5% and 2.5% of incident rainfall, respectively, reflecting the relative ranking of surface K_{fs}. Along with high water use by the pines, this lack of recovery of soil hydraulic properties under pine reforestation is shown to be a critical factor in the regionally observed decline in baseflows following large-scale planting of pines and has important implications for regional forest management.

[1]This chapter is based on: **Ghimire CP**, Bonell M, Bruijnzeel LA, Coles NA, Lubczynski MW (2013) Reforesting severely degraded grassland in the Lesser Himalaya of Nepal: Effects on soil hydraulic conductivity and overland flow production. *Journal of Geophysical Research*: *Earth Surface*, 118, 2528 – 2545.

4.1 Introduction

The forests of the Himalaya provide a number of vital environmental functions, both to the people living in the area itself and those in the neighbouring alluvial plains (Ives and Messerli, 1989; Singh, 2007; Joshi and Negi, 2011). In the past, environmental scientists expressed serious concern about the rapid deterioration of the Himalayan environment. Until comparatively recently, it was widely assumed that deforestation and overgrazing in the Himalaya were primarily responsible for the large-scale flooding and sedimentation experienced in the plains of northern India and Bangladesh (e.g. Eckholm, 1976; Nautiyal and Babor, 1985; Myers, 1986). Whilst this view is no longer tenable in the light of subsequent scientific evidence demonstrating the comparatively limited influence of land use on these *large-scale* hydrological phenomena (Bruijnzeel and Bremmer, 1989; Ives and Messerli, 1989; Hofer, 1993; cf. Gardner and Gerrard, 2003; Hofer and Messerli, 2006), the *local* adverse hydrological effects of advanced land degradation, such as accelerated erosion, enhanced peak discharges and reduced dry-season flows (Bartarya, 1989; Bruijnzeel and Bremmer, 1989) required remedial action (Tiwari, 1995; Negi et al., 1998). At the same time, the continued provision of various goods traditionally supplied by forests to rural communities as part of their subsistence economy – such as timber and fuelwood, fodder, litter for animal bedding and composting, as well as a host of other minor products (Mahat et al., 1987; Singh and Singh, 1992; Singh and Sundriyal, 2009; Joshi and Negi, 2011) – had to be taken into account as well (Campbell and Mahat, 1975). Thus, as part of a major effort to reforest severely degraded pastures and shrublands in the Middle Mountains of Central Nepal (Shepherd and Griffin, 1984), some 23,000 ha were planted to fast-growing coniferous species (mainly *Pinus roxburghii* and *P. patula* at an initial planting density of ~1600 trees ha^{-1}) between 1980 and 2000 (District Forest Offices, Kabhre and Sindhupalchok, Nepal, unpublished data, 2010). In addition, on-farm tree planting and natural regrowth on abandoned fields increased during this period as well (Gilmour and Nurse, 1991; Paudel et al., 2012) such that a recent survey reported a marked increase in both forest area and quality across Nepal's Middle Mountains over the last two decades (HURDEC Nepal and Hobley, 2012). Nevertheless, and as also reported for other parts of the world (Trimble et al., 1987; Waterloo et al., 2000; Jackson et al., 2005; Scott et al., 2005), local farmers in Central Nepal have expressed concerns about diminishing streamflow volumes following the large-scale planting of the pines (República, 2012). Consequently, understanding the role of reforestation in regulating the flows of water in this complex and fragile environment is critical, even more so in view of the area's strongly seasonal climate (approximately 80% of the annual rainfall is received between June and September) (Merz, 2004; Ghimire et al., 2012). The pressure on the area's water resources is immense (Merz et al., 2003; Schreier et al., 2006) and is expected to increase markedly in the future due to a combination of climate change and socio-economic developments (Mishra and Herath, 2011; Bandyopadhyay, 2013).

Hillslope soil hydraulic conductivity, in combination with prevailing rainfall intensities have been shown to affect volumes of storm runoff generation, rates of surface erosion and dry season flows in many tropical- and subtropical uplands, including the Himalayas (e.g., Gilmour et al., 1987; Bruijnzeel and Bremmer, 1989; Gardner and Gerrard, 2002; Gerrard and Gardner, 2003; Ziegler et al., 2004; Zimmermann and Elsenbeer, 2009; Bonell et al., 2010; Krishnaswamy et al., 2012; Ghimire et al., 2013; Krishnaswamy et al., 2013). Therefore, restoring diminished soil hydraulic conductivities to help recover the replenishment of soil moisture and groundwater reserves and concurrently curtailing the high volumes of infiltration-excess overland flow (IOF) typically associated with advanced land degradation (e.g., Gilmour et al., 1987; Chandler and Walter, 1998; Zhou et al., 2002; Gerrard and Gardner, 2002; Cuo et al., 2008; Ghimire et al., 2013) are perceived by many in the Himalayan region to be the most effective path towards boosting diminished dry season flows and reducing flood-related hazards (cf. Batarya, 1989; Negi et al., 1998; Tambe et al., 2012). However, there is a dearth of quantitative information about the extent to which, and under what circumstances, reforestation in the heavily populated Middle Mountains (cf. Singh et al., 1984; Hrabovsky and Miyan, 1987) can indeed restore diminished soil hydraulic conductivities given the continued pressure on the forests (both remaining natural forests and newly planted stands; Singh and Sundriyal, 2009; Joshi and Negi, 2011). Gilmour et al. (1987) were amongst the first to present field-saturated soil hydraulic conductivity (K_{fs}) data for top- and subsoils under various land-cover types in the Chautara area of Central Nepal. They observed a trend of increasing surface K_{fs} with age for 5- to 12-year-old pine plantations compared to a very degraded pasture, although values were still much lower than the K_{fs} associated with largely undisturbed natural forest.

However, subsequent re-measurement at the same sites after more than 25 years (Ghimire et al., 2013) showed dramatic *reductions* in surface K_{fs} in both the planted and natural forests in comparison to those of Gilmour et al. (1987). A reduction in soil biological activity and macropore formation due to over-intensive usage of the forests was identified as the chief causal factor for this finding (Ghimire et al., 2013). Unfortunately, there is reason to believe that, despite the optimistic notion regarding the overall improved quality of the Lesser Himalayan forests expressed by HURDEC and Hobley (2012), the situation of over-intensive usage of forests and correspondingly poor soil hydrological functioning as described by Ghimire et al. (2013) is not limited to the Chautara area. For example, Gardner and Gerrard (2002) reported very high overland flow occurrence in degraded (broad-leaved) forests in the Likhu Khola catchment north of Kathmandu, whereas Tiwari et al. (2009) and Wester (2013) recently presented similar evidence for community-managed forests further West in Nepal.

As part of a larger venture investigating the ‘trade-off’ between the higher water use of tree plantations (relative to pasture) on the one hand, and possibly improved infiltration opportunities through organic matter build-up during plantation maturation on the other hand (cf. Bruijnzeel, 1989; Krishnaswamy et al., 2012; Krishnaswamy et al., 2013), this study aims to extend the work of Gilmour et al. (1987) and Ghimire et al. (2013) at a new location (the Jhikhu Khola Catchment near Dhulikhel). Apart from presenting extensive new K_{fs} data for heavily degraded pasture, little disturbed broad-leaved forest and an intensively used pine plantation, and extending the range of measurements to the hillslope scale, this study includes the first measurements of K_{fs} for a heavily frequented rural footpath in the Himalaya. With the aid of statistical analysis, K_{fs} values for the respective land-cover types are compared and combined with locally measured rainfall intensities of various temporal resolutions to infer the associated changes in the dominant hillslope stormflow pathways (as defined in Chappell et al., 2007). The latter, in turn, are compared against *measured* overland flow totals generated on the pasture and in the two contrasting forest types to verify the inferred processes of stormflow generation. In doing so, particular attention is paid to an evaluation of the impacts of reforesting the former degraded pasture and the sustained intensive use of the pine stand. Finally, the implications of the present findings on overland flow generation and infiltration for the evolution of regional dry season flows are discussed in terms of the ‘trade-off’ hypothesis referred to above. To this end, the new overland flow data are combined with recent information on forest and pasture water use in the area (Baral, 2012).

4.2 Study area

The field measurements were conducted between 1500 and 1620 m above mean sea level (a.m.s.l.) in the Jikhu Khola Catchment (JKC) (27^0 35’–27^0 41’ N; 85^0 32’–85^0 41’ E) close to Dhulikhel (the District Headquarters of Kabhre district) in the Middle Mountains of Central Nepal (Figure 4.1). The region has a complex geology that has resulted in equally complex spatial patterns of topography, soils and vegetation.

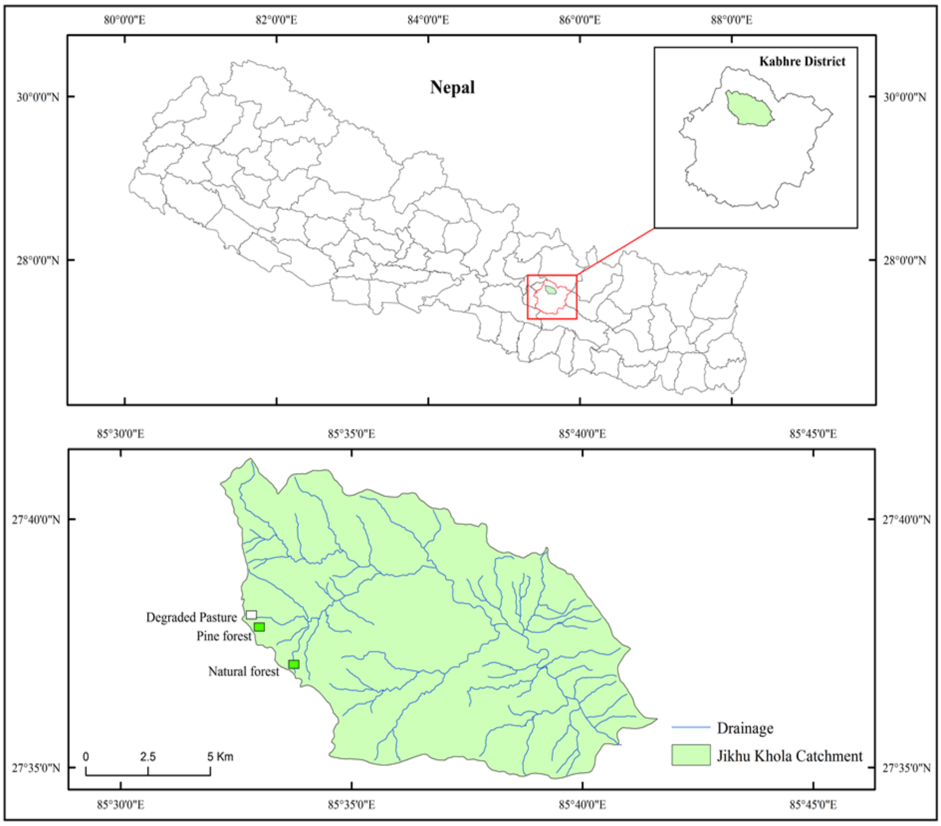

Figure 4.1: Location of study sites in the Jikhu Khola Catchment in the Middle Mountains of Central Nepal.

The geology includes phyllites, schists and quartzites on which Cambisols and Luvisols of silty to clay loam textures have developed (Maharjan, 1991). The climate of the JKC is largely humid subtropical, grading to warm-temperate above 2000 m a.m.s.l. Mean (±SD) annual rainfall, as measured at mid-elevation (1560 m a.m.s.l.) for the period 1993–1998, was 1487 (± 157 mm) (Merz, 2004). Similar amounts were recorded at 1580 m a.m.s.l. between October 2010 and September 2011 by Ghimire et al. (2012). Annual reference evaporation (following Allen et al., 1998) for the period 1993–2000, was 1170 mm (Merz, 2004). The main seasons are, respectively: the monsoon (June to September), the post-monsoon period (October to November), winter (December to February), and the pre-monsoon period (March to May). The rainy season brings about 80% of the total annual precipitation. In general, July is the wettest month with about 27% of the annual rainfall. The driest months are November to February, each accounting for about 1% of annual rainfall (Merz, 2004). The vegetation at elevations between 1000 and 2000 m a.m.s.l. consists of a largely evergreen mixed broad-leaved forest dominated by *Schima wallichii* and various chestnuts and oaks (*Castanopsis* spp., *Quercus* spp.), with admixtures of *Rhododendron arboreum* above 1500 m a.m.s.l. Due to the prevailing population pressure, much of the original species-rich forest has disappeared, with most of the remaining forest either occurring on slopes that are too steep for agricultural activity or being in various stages of degradation because of continued disturbance (Dobremez, 1976; Merz, 2004). Vegetation cover in the catchment consists of 30% forest (both natural and planted), 7% shrubland and 6% grassland, with the remaining 57% largely under agriculture (Merz, 2004). Parts of the catchment were deforested more than a century ago (Mahat et al., 1986). The JKC was subjected to active reforestation with various pines (notably *P. roxburghii* and *P. patula*) until 2004 as part of the Nepal-Australia Forestry Project. According to local farmers, several springs have dried up completely and the yields of others have been declining since the reforestation (República, 2012).

The respective measuring sites were selected so as to represent the various stages of anthropogenic pressure, ranging from a heavily compacted footpath through degraded pasture and an intensively used pine reforestation, to little disturbed natural broad- leaved forest. All sites were located within the headwater area of the JKC on slopes ranging from 20^o to 25^o. The land use at the respective research sites can be characterized as follows:

Degraded pasture: This site (south-east exposure, overall slope angle 18°) has been heavily grazed for more than 150 years [based on various personal communications]. Numerous patches of compacted or bare soil surface are evident (Figure 4.2a). The dominant grass and herb species are *Imperata cylindrica*, *Saccharum spontaneum* and *Ajuga macrosperma.* Little or no grass cover remains at the peak of the dry season (March–May). The numerous heavily compacted footpaths distributed along and across the hillslope have been in use for more than a century. In summary, the measurements at this site are considered to represent the combined effect of more than 150 years of continuous cattle grazing in addition to human trampling pressure.

Figure 4.2: (a) Cattle grazing at the degraded pasture site, (b) infiltration-excess overland flow on footpath during a medium-intensity rainfall event, (c) cattle grazing at the pine forest site, (d) little disturbed natural forest site, and (e) local women with litter material collected from the pine forest site.

Footpath: A heavily compacted footpath running along the slope within the degraded pasture that had been in use for more than 150 years. No vegetation existed on the heavily trampled surface (Figure 4.2b). Local people are using the footpath for two purposes mainly: (a) to access a pine forest close to the degraded pasture, and (b) to reach a small viewing tower located just above the degraded pasture which is a point of attraction for locals and tourists alike. On average, about 30 people pass this footpath twice a day.

Pine forest: This former degraded pasture located ca. 400 m from the studied degraded pasture on a 20^o slope of south-west exposure (Figure 4.1) was planted with *P. roxburghii* in 1986. At the time of the K_{fs} measurements (2011) the trees were 25 years old. An understory was largely absent as grazing by cattle is common (Figure 4.2c). In addition, litter is collected for animal bedding while the grassy herb layer is regularly harvested (Figure 4.2e). Pruning of trees for fuelwood, and felling for timber are also common. Combined, these activities have opened up the canopy and compacted the soil surface of the pine forest considerably.

Natural forest: This is a better protected site which consists of dense, largely evergreen forest facing little anthropogenic pressure (Figure 4.2d). It is located ~2700 m from the degraded pasture on a 24^o slope of north-west exposure (Figure 4.1). The tree stratum of the natural forest consists largely of a mixture of *Castanopsis tribuloides* and *Schima wallichii* with a few other minor species (cf. Ghimire et al., 2012). The soil surface is well covered with a broad-leaved understory as well as a litter layer. Grazing animals and litter collection are excluded but local people collect mushrooms during the rainy season.

4.3 Methods

4.3.1 Field measurements of soil hydraulic conductivity

A disc permeameter (Perroux and White, 1988; McKenzie et al., 2002) was used for the measurement of surface K_{fs} in the field. Although little is known about the effects of slope on measurements taken by the ponded version of the disc permeameter used here (Casanova, 1998; Joel and Messing, 2000; Bodhinayake et al., 2004), simulations undertaken elsewhere have shown that the effect of slope angles up to 20^o have only a marginal effect on the volume flux and thus on K_{fs} estimates (K.R.J. Smettem, personal communication, 2012). A steel ring was adapted for use on non-level land (slopes > 1^{o}). To obtain a uniform emplacement depth while maintaining an adequate water supply head across the soil surface within the ring (27 cm diameter), the lower ring's circumference was cut such that the ring wall was shallower on the upslope side than on the downslope side (i.e., 3 cm versus 7 cm height). For the natural forest site where the slope was greater than 20^{o}, the surface K_{fs} measurements were confined to somewhat flatter micro-topographic areas embedded within the overall slope. It is acknowledged that the resulting K_{fs} estimates may differ from estimates for the steeper slope (>20^{o}). However, given the previously noted fact that slope angles up to 20^{o} have only a marginal effect on K_{fs}, any such differences were considered to be minimal. Prior to measurement, any straw and stubble present were removed from the surface, with the least amount of surface disturbance. To improve the contact between the permeameter disc and the soil surface, a 5 mm thick layer of fine sand (size < 2 mm) was placed on the soil surface inside the ring after which the apparatus was placed on the ring. The rates of water discharge through the disc, as inferred from changes in the water levels in the storage tower of the apparatus, were recorded until steady-state flow rates were reached. In addition, two soil cores – one before the infiltration measurement commenced and the other after measurement was finished – were taken and weighed in the field. The cores were subsequently transported to the laboratory for the measurement of volumetric soil moisture content (based on gravimetric soil moisture content and dry bulk density). K_{fs} (mm h^{-1}) was then calculated using the method outlined by McKenzie et al.(2002):

$$K_{\mathrm{fs}} = I - \frac{4bS_o^2}{\pi r(\theta_o - \theta_n)} \tag{4.1}$$

where θ_n [m^3 m^{-3}] and θ_o [m^3 m^{-3}] are the *in situ* volumetric soil moisture contents before and after infiltration, respectively; b is a constant (0.55); S_o [mm $h^{-1/2}$], the sorptivity obtained by plotting cumulative infiltration volume versus the square root of time since the start of infiltration; r, the radius of the disc permeameter base (127.5 mm); and I [mm h^{-1}], the steady-state infiltration rate, calculated as:

$$I = \frac{q}{\pi r^2} \tag{4.2}$$

where q [mm^3 h^{-1}] is the slope of the plot of cumulative infiltration versus time after reaching steady–state conditions; with r as defined previously.

For the measurement of K_{fs} in deeper soil layers, a constant-head well permeameter (CHWP) was used (Talsma and Hallam, 1980). The use of the CHWP was restricted to the dry season to minimise errors from smearing of the auger hole walls (Chappell and Lancaster, 2007). The experimental procedure involved augering a cylindrical hole (with radius a = 4 cm) to the desired depth. Any sealing of pores in the column walls due to the augering was minimised by brushing the walls with a small metal brush. The hole was pre-wetted for 20 min before taking the measurement to achieve perimeter saturation as described by Talsma and Hallam (1980). The CHWP was then inserted to the required depth and the flow measured until a steady–state flow rate was reached. K_{fs} (mm h^{-1}) values were calculated from the measurements using equation 11 of Reynolds et al. (1983):

$$K_{\mathrm{fs}} = \frac{CQ_t}{2\pi H^2\left[1 + \frac{C}{2}\left(\frac{a}{H}\right)^2\right]} \tag{4.3}$$

Where Q_t [mm^3 h^{-1}] is steady state flow rate; H [mm], the constant height of ponded water in the well; a [mm], the radius of the well; and C, a dimensionless shape factor calculated as:

$$C = \sinh^{-1}\left(\frac{H}{a}\right) - \sqrt{\left(\left(\frac{a}{H}\right)^2 + 1\right)} + \frac{a}{H} \tag{4.4}$$

4.3.2 Experimental design

Field-saturated hydraulic conductivity (K_{fs}) was measured at the hillslope scale, both at the surface and at depths of 0.05–0.15 m, 0.15–0.25 m, 0.25–0.50 m, and 0.5–1.0 m. Except in the case of the footpath measurements, the locations of the measurement plots along the respective hillslopes were selected such that the plot's x-axis represented the contour line and the y-axis the slope (Figure 4.3). Plot size varied from 30 m x 50 m (natural forest) through 40 m x 65 m (pine forest) to 40 m x 115 m (degraded pasture) depending on local slope configuration.

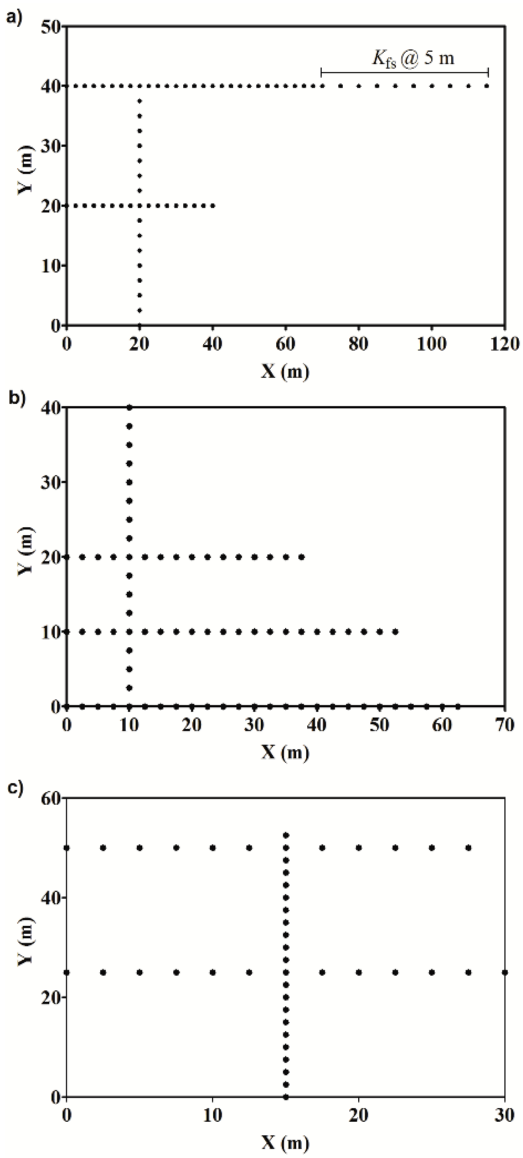

Figure 4.3: Field-saturated soil hydraulic conductivity sampling grid used at: (a) the degraded pasture site, (b) the pine reforestation, and (c) the natural forest in the headwaters of the Jikhu Khola Catchment, Central Nepal.

Next, several lines of different lengths along and perpendicular to the hillslope were superimposed (Figure 4.3) to avoid clustering of sampling points in any direction (cf. Ghimire et al., 2013). Further, at three sites (degraded pasture, pine forest and natural forest) the sub-surface K_{fs} was measured at 2.5 m intervals. An exception was made in the upper part of the degraded pasture site where the few measurements were made at 5 m intervals due to the obstruction caused by the presence of an automatic weather station (see Figure 4.3). In case of the 70 m long and 0.4 m wide footpath, K_{fs} was measured at 3 m intervals. For the measurement of surface K_{fs}, 6–17 replications were taken per site via random sampling using a disc permeameter (see Table 4.2 below). The number of surface K_{fs} measurements was much less than for the other depths (Table 4.2) because of the practical limitations of using a disc permeameter on steeply sloping land discussed earlier.

Table 4.1: Basic physical and chemical properties of the soil at different depths under the studied contrasting land covers in the headwaters of the Jikhu Khola Catchment, Central Nepal. Values listed are the means (±SD) of six samples.

Site	Depth [m]	Clay [%]	Sand [%]	Silt [%]	SOC [%]
Degraded Pasture	0.05–0.15	19.2 (±1.4)	34.0 (±3.14)	46.8 (±1.77)	2.23 (±0.27)
	0.15–0.25	16.0 (±1.95)	37.2 (±3.87)	46.8 (±2.46)	0.77 (±0.32)
	0.25–0.50	14.7 (±1.12)	38.5 (±1.21)	46.8 (±0.22)	0.66 (±0.11)
	0.50–1.0	12.4 (±0.64)	44.1 (±1.94)	43.5 (±1.40)	0.34 (±0.16)
Pine Forest	0.05–0.15	19.2 (±1.48)	40.0 (±2.0)	40.8 (±1.08)	1.69 (±0.32)
	0.15–0.25	17.1 (±1.48)	40.5 (±0.83)	42.4 (±0.99)	0.99 (±0.21)
	0.25–0.50	14.2 (±1.54)	43.0 (±1.3)	42.8 (±0.94)	0.47 (±0.19)
	0.50–1.0	11.5 (±1.19)	46.7 (±4.1)	41.8 (±2.99)	0.18 (±0.14)
Natural Forest	0.05–0.15	29.5 (±2.13)	24.0 (±1.5)	46.5 (±1.33)	4.1 (±0.25)
	0.15–0.25	29.2 (±2.57)	26.3 (±1.8)	44.5 (±1.69)	1.72 (±0.20)
	0.25–0.50	28.1 (±3.14)	26.1 (±1.1)	45.8 (±3.0)	1.43 (±0.20)
	0.50–1.0	26.0 (±5.81)	25.7 (±2.26)	48.3 (±3.78)	0.72 (±0.13)

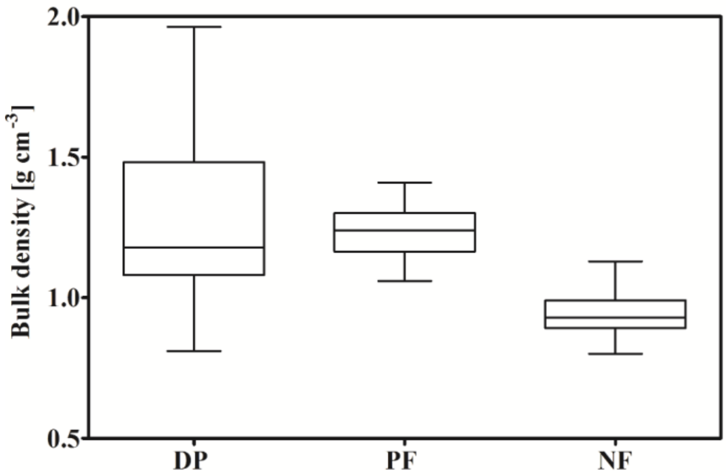

Figure 4.4: Topsoil (0 – 0.10 m) bulk density values at: (a) the degraded pasture (DP, n = 37), (b) the pine reforestation (PF, n = 18), and (c) the natural forest (NF, n = 21) site in the head-waters of the Jikhu Khola catchment, Central Nepal.

4.3.3 Rainfall intensity

The rainfall data that were used to infer the dominant hillslope hydrological pathways and changes therein as a function of the level of anthropogenic pressure, were recorded at the degraded pasture site during the respective monsoon periods (June–September) of 2010 and 2011.

Rainfall was recorded using a tipping-bucket rain gauge (Rain Collector II, Davis Instruments, USA; 0.2 mm per tip) at 5-min intervals. A rainfall event was defined as having at least 5 mm of rain in total [*Negishi et al.*, 2006] and was separated from a previous event by a dry period of at least 3 h. The maximum 5-min rainfall amounts (I_{5max}, expressed as equivalent hourly rainfall intensity) were determined by calculating the maximum precipitation over the corresponding time interval for each event. The maximum 10 min (I_{10max}), 15 min (I_{15max}), 30 min (I_{30max}) and 60 min (I_{60max}) rainfall intensities (all expressed as equivalent hourly rainfall intensities) were derived in a similar manner as well.

4.3.4 Overland flow

Overland flow at the degraded pasture, pine forest and natural forest sites was monitored between 20 June and 9 September 2011 (i.e., the bulk of the 2011 rainy season) using a single large (5 m x 15 m) runoff plot per land-cover type. Runoff was collected in a gutter system funneling the water to a first 180 l collector equipped with a 7-slot divider allowing 1/7th of the spill-over into a second 180 l drum, thereby bringing the total collector capacity to 1440 l (~20 mm). The water levels in the two collectors were measured continuously using a pressure transducer device (Keller, Germany) placed at the bottom. Collectors were emptied and cleaned after measuring the water level manually every day around 8:45 AM local time. Event runoff volume was calculated by converting the water levels to volumes using a calibrated relationship per drum and summing up to obtain total runoff volume. Measured overland flow volumes were corrected for direct rainfall inputs into the runoff collecting system. Overland flow volumes were divided by projected plot area to give overland flow in mm per event. Values were also expressed as the percentage of corresponding rainfall or throughfall (based on Ghimire et al., 2012) where appropriate.

4.3.5 Additional soil physical and chemical properties

Additional soil physical and chemical properties such as bulk density, texture and organic carbon content (SOC) were determined in order to quantify other factors that might affect K_{fs}. Soil bulk density was measured in each plot at randomly selected points (degraded pasture, n= 37; pine reforestation, n = 18; natural forest, n = 21) at 0–0.10 m depth using the core method (196 cm^{3}cores; Blake and Hartge, 1986). The chief soil textural components and SOC were determined at depths of 0.05–0.15 m, 0.15–0.25 m, 0.25–0.50 m, and 0.50–1.0 m using a Fritsch particle analyzer (Konert and Vandenberghe, 1997) and the Walkley and Black (1934) procedure, respectively.

4.3.6 Data analysis

4.3.6.1 Inferring the hydrological consequences of different levels of anthropogenic pressure

When considering hillslope hydrological response to rainfall, the key parameters controlling dominant stormflow pathways are the changes in K_{fs} with depth in combination with prevailing rainfall intensities (Bonell et al., 1983; Gilmour et al., 1987; Elsenbeer et al., 1999; Ziegler et al., 2006; Zimmermann et al., 2006; Bonell et al., 2010). To identify a potentially impeding soil layer to different rainfall intensities at each site, selected percentiles of maximum rainfall intensities I_{max} (e.g., over 5 min, I_{5max}) as measured at the degraded pasture site, were compared with the median K_{fs}-values for each site (i.e., land cover) and depth. The presence of an impeding layer at the surface or at a certain depth for the chosen rainfall intensity is indicated if the latter exceeds the median K_{fs}-value under consideration. In turn, dominant pathways of stormflow can then be inferred. Moreover, the disposition of rainfall delivered at the soil surface for the selected maximum rainfall intensity (I_{max}), was estimated by comparing that rainfall intensity and K_{fs}.

It is acknowledged that the K_{fs}-data presented in this work represent measurements that are biased towards dry season conditions and as such, they do not take into account any seasonal variability in K_{fs}. Wet season values of surface K_{fs} using the disc permeameter in monsoonal southwest India were shown to be reduced by up to an order of magnitude compared to dry season measurements taken over the same locations, possibly due to surface sealing by raindrop impact and blocking of larger pores by sediment carried by overland flow (Bonell et al., 2010; cf. McIntyre, 1958a,b). Thus, lower K_{fs}-values can be expected during the main monsoon season, and the presently inferred degree and frequency of IOF occurrence may therefore be under-estimated at such times.

4.3.6.2 Statistical analysis

For all statistical analyses, the language and environment of R, version 2.14.0 (R Development Core Team, 2011) was used. All data-sets were tested for normality using the Shapiro – Wilk *W* statistic (Shapiro and Wilk, 1965). In the case of non-normal distribution of the data, the raw data were transformed ($\log_e$ and square-root transformations) to achieve a normal or near-normal distribution. However, since it is not possible to compare differently transformed data, it was preferred to use non-transformed data and non-parametric statistical analysis. Differences in K_{fs} between sites were initially tested using the Kruskal-Wallis (1952) test. If the latter indicated a significant difference between medians, the Mann-Whitney U-test with Bonferroni correction was subsequently applied to account for multiple comparisons across sites and specific soil layers. In the case of comparisons between two sites only, differences in K_{fs} were statistically examined using the Mann-Whitney U-test. Differences in K_{fs} were assumed to be significant with $p<0.05$ for the Mann-Whitney U-test and $p<0.017$ for the Kruskal-Wallis test. For a rapid visual comparison among sites, box plots were computed as well.

4.3.6.3 Spatial analysis

To examine the possible effects of spatial correlation length on the above statistical analyses of K_{fs}, the spatial structure of the distribution of this parameter was also analysed using geostatistics. Where sample sizes were sufficient, new mean and median values for K_{fs} were calculated by including only every alternate, second or third K_{fs}-value from the original data-set, depending on the spatial correlation length. The statistical procedures were then repeated and the results were compared against the full data-sets.

The spatial analysis was limited to the degraded pasture, pine forest and natural forest sites as the number of pairs in the footpath was below 30 (a standard geostatistical rule; cf. Mohanty and Mousli, 2000). The sample sizes used in the geostatistical analysis are listed in Table 4.2. The semi-variance (γ) was determined according to Matheron (1962):

$$\gamma(h) = \frac{1}{2N(h)}\left\{\sum_{i=1}^{N(h)} [Z(x_i + h) - Z(x_i)]^2\right\} \tag{4.5}$$

where N (h) is the number of pairs separated by the lag distance h, and Z (x_i) is the K_{fs} measured at a point x_i. Experimental variograms were also estimated using the language and environment of R. Semi-variance was calculated for the residual of square root- (degraded pasture and pine forest sites) and $\log_e$-transformed data (natural forest site) to minimize the effect of outliers. Experimental variograms were fitted using a number of standard semi-variogram models: Gaussian, exponential and spherical, and the one with the smallest residual sum of squares was chosen as the best-fit model (Webster and Oliver, 2007). To compare the strength of the spatial correlation, the so-called nugget to sill ratio was used following the limits of Cambardella et al. (1994). The nugget stands for the variance at zero separation distance whereas the sill represents the variance at the range (Webster and Oliver, 2007). A variable is considered to be strongly spatially dependent if the nugget to sill ratio is ≤ 0.25; a ratio of 0.25–0.75 indicates a moderate dependence and a ratio of 0.75 or more suggests a weak spatial autocorrelation (Cambardella et al., 1994).

4.4 Results

4.4.1 Soil physical and chemical properties

Table 4.1 summarises the results of the soil textural and chemical analyses whereas the corresponding bulk densities are presented in Figure 4.4.

As expected on the basis of the intensity of usage, topsoil bulk density values at the degraded pasture (1.18 ± 0.33 g cm^{-3}) and pine forest (1.24 ±0.095 g cm^{-3}) sites were similar and significantly higher than those for the little disturbed natural forest (0.93 ± 0.082 g cm^{-3}) site (Figure 4.4). The pine forest soil had the highest sand percentage at all investigated depths in comparison to the soils of the degraded pasture and natural forest (Table 4.1). Such differences in sand content are likely to contribute to the observed differences in field -saturated hydraulic conductivity (K_{fs}) (Table 4.2 below). Topsoil organic carbon content was highest in the natural forest (4.10 ± 0.25%) and differed significantly from values obtained for the other two sites (Table 4.1), indicating the greater activity of soil micro-flora and fauna feeding on the litter layer in the natural forest.

Table 4.2: Descriptive statistics for field-saturated hydraulic conductivities (K_{fs}) associated with contrasting land cover types at different depth in the headwaters of the Jikhu Khola Catchment, Central Nepal.

	Surface	**(0.05 – 0.15 m)**	**(0.15 – 0.25 m)**	**(0.25 – 0.5 m)**	**(0.5 – 1.0 m)**
Degraded Pasture					
Mean	25	41	72	38	19
Median	18	39	68	35	19
Standard deviation	20	18	32	16	9
Minimum	4	7	25	14	6
Maximum	70	82	161	96	44
Sample size (*n*)	17	70[a]	70[a]	70[a]	70[a]
Footpath					
Mean	10	24	41	29	16
Median	12	26	39	30	19
Standard deviation	5	10	17	10	7
Minimum	4	7	13	14	7
Maximum	18	42	84	53	40
Sample size (*n*)	7	24	24	24	24
Pine Forest					
Mean	26	43	109	92	45
Median	24	39	107	92	45
Standard deviation	7	19	38	32	11
Minimum	18	11	29	25	24
Maximum	35	125	210	165	69
Sample size (*n*)	10	80[a]	80[a]	80[a]	80[a]
Natural Forest					
Mean	333	94	80	51	19
Median	232	82	64	47	18.8
Standard deviation	356	74	53	29	8
Minimum	32	7	22	11	5
Maximum	1256	389	285	152	36
Sample size (*n*)	11	45[a]	45[a]	45[a]	45[a]

[a]Data used for geostatistical analysis.

4.4.2 Exploratory analysis of field-saturated soil hydraulic conductivity

The descriptive statistics of K_{fs} for the various depth intervals at each site are presented in Table 4.2. As can be seen from the corresponding box plots (Figure 4.5), most of the K_{fs} data-sets showed non-Gaussian behaviour. Therefore, a global comparison of the results obtained for different sites and depths was based on median values.

As expected on the basis of the degree of anthropogenic pressure experienced by the respective sites, the median surface K_{fs} was lowest for the footpath (12 mm h^{-1}) and highest for the natural forest (232 mm h^{-1}), such that they differed by more than an order of magnitude (Table 4.2). Under the nearly undisturbed conditions prevailing in the natural forest, K_{fs} decreased with depth, particularly in the first 0.15 m, although the difference between 0.15 and 1m depth was still more than four-fold (Table 4.2 and Figure 4.5). The largest variance of K_{fs} at any depth within the natural forest was observed at 0.15–0.25 m depth (Figure 4.5c). A different pattern with depth was observed for the degraded pasture, footpath and pine forest sites – which all had very low values of surface K_{fs} – in that here the near-surface K_{fs} first *increased* down to a depth of 0.25 m and then started to decrease (Table 4.2). Arguably, the most striking feature of the current data-set is that the median K_{fs} at the surface and in the shallow soil layer in the 25-year-old pine forest had remained at the *same level* as those for the heavily grazed degraded pasture, suggesting the virtual absence of non-structural (i.e., biologically mediated) macropores within the pine forest's soil down to 0.15 m depth. This finding mirrors the pattern found earlier for topsoil (0–10 cm) bulk densities at the two sites (Figure 4.4).

A statistical comparison of K_{fs}-values measured at the surface and at a depth of 0.05–0.15 m between the pine forest, degraded pasture and natural forest sites supported the preceding inferences in that there was no significant difference ($p>0.017$) between the pine forest and degraded pasture, whereas both sites had significantly lower ($p<0.017$) values compared to the natural forest, as also evidenced by the much lower topsoil bulk density of the natural forest (Figure 4.4). Below 0.15 m depth, however, the K_{fs} in the pine forest was consistently higher ($p<0.017$) than values measured at any other site (Figures 4.5c–e), presumably because of a difference in soil texture (cf. Table 4.1).

Moreover, the comparison between the footpath and degraded pasture showed no significant difference in surface K_{fs}, indicating that surface conditions at the degraded pasture had already reached an extreme state of degradation. In contrast, K_{fs}-values between 0.15 and 0.25 m depth differed significantly between the footpath and degraded pasture. At 1.0 m depth, however, differences in K_{fs} between the respective land-cover types were mostly non-existent (except for the higher value beneath the pine forest already signaled), indicating the lack of influence exerted by cattle grazing and human trampling on the deeper soil layers.

4.4.3 Spatial structure

Experimental semi-variograms (cf. Equation 4.5) were developed to describe the spatial structure of the sub-surface K_{fs}-data using the residuals of a square-root transformation (degraded pasture and pine forest sites) or a $\log_e$ transformation (natural forest) of the data. The results of the spatial analysis are summarised in Table 4.3.

The spatial correlation length ('effective range') increased with depth throughout the profile underneath the degraded pasture but not in the pine forest or natural forest where the effective range first increased down to a depth of 0.25 m and then started to decrease (Table 4.3). At a depth of 0.05–0.15 m, all three land-cover types exhibited strong spatial dependence, with the correlation length ranging from 4.4 m (pine forest and natural forest) to 5.4 m (degraded pasture). For the 0.15–0.25 m depth interval, the most noticeable difference between sites was a much higher spatial correlation length under natural forest which, in addition, exhibited a weaker spatial dependence (as indicated by the higher nugget to sill ratio; Table 4.3). Spatial correlation lengths for the same depth interval in the degraded pasture and pine forest were similar and showed moderate (degraded pasture) to strong (pine forest) spatial dependence. At a depth of 0.25–0.50 m, both the pine forest and natural forest showed strong spatial dependence with correlation lengths ranging from 4.8 to 15.0 m, respectively, whereas the degraded pasture indicated moderate spatial dependency with an effective range of 10.5 m. Furthermore, between 0.50 and 1.0 m depth, the spatial correlation lengths in the degraded pasture and natural forest were similar and both showed moderate spatial dependence. As observed earlier for other depths, the spatial dependence in the deepest layer in the pine forest was very strong

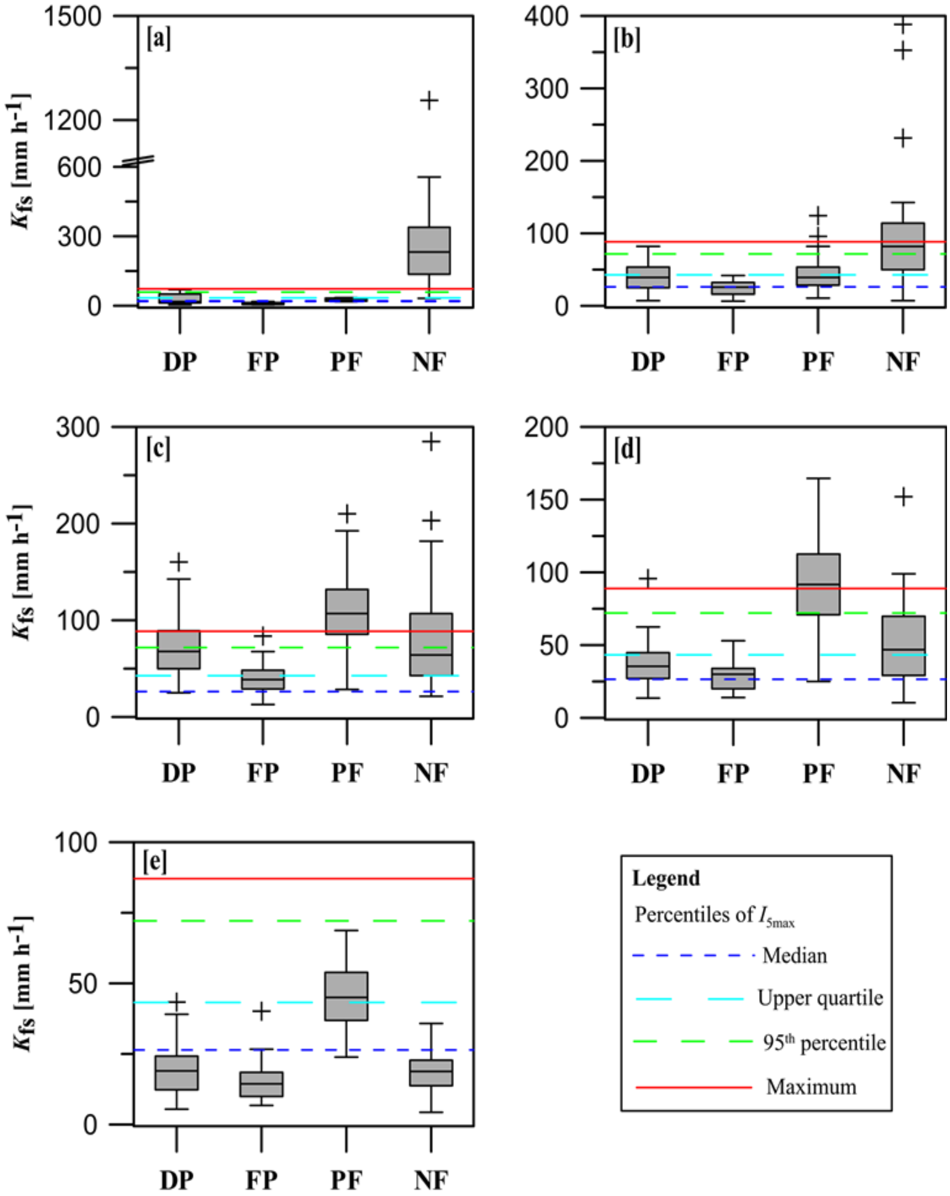

Figure 4.5: Field-saturated hydraulic conductivity K_{fs} as a function of land use and measurement depth in the head waters of the Jikhu Khola catchment, Central Nepal: (a) surface, (b) 0.05–0.15 m, (c) 0.15–0.25 m, (d) 0.25–0.50 m, and (e) 0.50–1.0 m. The solid horizontal line represents the maximum rainfall intensity over 5 min and various dashed lines median, 75% percentile, and 95% percentile of I_{5max} rainfall intensity. DP = degraded pasture, FP = footpath, PF = pine forest, and NF = natural forest.

but had a low spatial correlation length (4.1 m; Table 4.3). Overall, spatial correlations at or above the moderate category were detected at 0.05–015 m depth for all land covers and at greater depths for the degraded pasture and pine forest.

To examine the effect of spatial correlation length for near-surface (0.05–0.15 m) $K_{fs,}$ new mean and median values for K_{fs} were calculated by including only every alternate, second or third K_{fs}-value from the original data-set (Table 4.4). However, according to the Mann-Whitney U test the new median K_{fs} values did not differ significantly from the original ones.

Table 4.3: Parameter values for those semivariogram models that best described the observed spatial variability of the field-saturated hydraulic conductivity K_{fs} with depth as associated with the different land-cover types in the headwaters of the Jikhu Khola catchment, Central Nepal. DP = degraded pasture, PF = pine forest, and NF = natural forest.

	Soil depth (m)	**Model[a]**	**Nugget**	**Partial sill**	**Sill**	**Range (m)**	**Effective range (m)[b]**	**Nugget/ Sill**	**Sserr[c]**
DP	(0.05 – 0.15)	Exp	0	2.37	2	1.8	5.4	0	0.76
PF		Sph	0	2.3	2	4.4	4.4	0	5.66
NF		Sph	0	0.5	0	4.4	4.4	0	0.35
DP	(0.15 – 0.25)	Sph	1	1.68	3	9.2	9.2	0.47	0.65
PF		Sph	0	2.66	3	6.5	6.5	0.23	0.65
NF		Exp	0	0.038	0	6.5	19.5	0.9	0.01
DP	(0.25 – 0.50)	Sph	0	0.9	1	10.5	10.5	0.47	0.18
PF		Sph	0	2.67	2	4.8	4.8	0	2.87
NF		Sph	0	0.38	0	15.0	15.0	0.22	0.22
DP	(0.50 – 1.0)	Exp	0	0.41	0	4.5	13.5	0.57	0.028
PF		Sph	0	0.88	0	4.1	4.1	0	1.17
NF		Sph	0	0.18	0	14.8	14.8	0.42	0.023

[a]See text for explanation, [b]Model type: Exp = Exponential; Sph = Spherical, [c]For exponential model: Effective range = range*3, [d]Serr: Residual sum of squares

4.4.4 Rainfall intensity, overland flow, and hydrological consequences of anthropogenic pressure

A total of 99 storms events were recorded during the summer monsoons of 2010 and 2011 at the Dhulikhel rainfall station. The maximum equivalent hourly intensities (I_{5max} – I_{60max}) for the observed 5-, 10-, 15-, 30-, and 60-min rainfall classes for all storms over the two monsoon seasons were 88.8, 82.8, 62.4, 47.2, and 39.6 mm h^{-1}, respectively. The corresponding median values were 26.4, 19.2, 15.6, 10.6 and 7.6 mm h^{-1}.

For each rainfall intensity class, rainfall intensities were low for the majority of events. However, the higher-intensity storms contributed substantially to overall monsoon precipitation totals (Figure 4.6). The median, 75% percentile, 95% percentile and absolute maximum values of I_{5max} were selected as indices for inferring the dominant hillslope runoff pathways in combination with the results obtained for K_{fs} for the respective land covers (Figure 4.5).

At only 12 mm h^{-1}, the surface permeability of the heavily compacted footpath even remained below the median value of I_{5max} (Figure 4.5a), suggesting the frequent occurrence of infiltration-excess overland flow (IOF). Although not measured on the footpath, IOF was indeed observed to occur more rapidly and in greater quantities during medium to high intensity rainfall events than on the surrounding pasture (cf. Figure 4.2b). At the other extreme, the surface K_{fs} in the natural forest exceeded the maximum values of I_{5max}, suggesting IOF would never occur. Nevertheless, some overland flow was recorded at this site (see Figure 4.7 below) with a monsoonal seasonal total of 17.7 mm or 2.5% of incident rainfall and 3.3% of the corresponding amount of throughfall (Table 4.5). Given the much lower median K_{fs} derived for the 0.05–0.15 m depth interval in the natural forest (82 mm h^{-1}; Table 4.2) it cannot be excluded that at least some of the recorded overland flow was contributed by the saturation-excess type (SOF; Bonell, 2005; cf. Figure 4.8d below). In spite of the slightly higher surface K_{fs} in the degraded pasture and pine forest compared to the footpath, the upper quartile of I_{5max} still exceeded the median surface K_{fs} values for the grassland and pine reforestation (Figure 4.5 a), thereby indicating the occurrence of IOF during high-intensity rainfall (cf. Figures 4.7 and 4.8a, c). Indeed, overland flow at the degraded pasture site was typically generated after only 3–4 mm of rain (Figure 4.7) whereas the seasonal runoff total from the pasture amounted to 187 mm (21.3% of incident rainfall; Table 4.5). Corresponding values for the pine forest were comparable at 4.2 mm of rain before runoff would start (Figure 4.7), and a seasonal total of 136 mm (15.5% of rainfall and 18.6% of throughfall; Table 4.5). However, a noticeable shift from IOF-dominated runoff to less impeded infiltration occurs when considering the maximum rainfall over 60 min, i.e., moving from I_{5max} to I_{60max} (Figure 4.6).

Table 4.4: Selected statistical parameters of field-saturated hydraulic conductivity K_{fs} (mm h^{-1}) for the 0.05–0.15 m depth interval after including only every alternate, second, or third K_{fs} values from the original data set to remove the effect of spatial correlation length. Values in parentheses correspond to the results obtained using the original measurement interval.

	@5 m	@7.5 m	@10 m
Degraded Pasture			
Mean	39 (41)	40 (41)	36 (41)
Median	39 (39)	39 (39)	39 (39)
Pine Forest			
Mean	42 (43)	45 (43)	37 (43)
Median	36 (39)	43 (39)	32 (39)
Natural Forest			
Mean	91 (94)	82 (94)	101 (94)
Median	73 (82)	93 (82)	78 (82)

With regard to K_{fs} in the 0.05–0.15 m layer, the upper quartile of I_{5max} exceeded the median K_{fs}-values at the degraded pasture, footpath and pine forest (Figure 4.5b). At such rates, the rainfall that percolates to this depth would be capable of developing a perched water table and with it generate lateral subsurface stormflow (SSF; Figure 4.8a–c). This would be in addition to the concurrent occurrence of IOF at these sites. In contrast, the corresponding median K_{fs} at the natural forest is still above (or nearly equal to) most 5-min rainfall intensities (Figure 4.5b), thereby favouring mostly vertical percolation at this site (cf. Figure 4.8d).

For the 0.15–0.25 m and 0.25–0.50 m depth intervals, the K_{fs}-values in the degraded pasture, footpath and natural forest all indicate a similar hydrological response to rainfall, namely mostly lateral SSF and thus limited vertical percolation during high intensity rainfall (Figures 4.8 a, b and d). However, the much higher median values of K_{fs} between 0.15 m and 0.50 m depth at the pine forest site exceeded the maximum values of I_{5max} (Figures 4.5c and 4.5d; cf. Figure 4.8c) and thus rather favour vertical percolation. These high conductivities are likely to reflect the higher sand content of the soil beneath the pine forest (Table 4.1) and effectively rule out the frequent occurrence of a perched water table and SSF in this layer, in contrast to what was inferred for the other sites. The effect, however, must be counteracted to some extent by the low median surface K_{fs} in the pine forest which encourages IOF (Figures 4.5a, 4.7 and 4.8c) and restricts the amounts of water percolating to deeper layers. Finally, at 1.0 m depth, the differences in K_{fs} and inferred hydrological response to rainfall became insignificant between sites (Figure 4.5e).

4.5 Discussion

4.5.1 Impacts of land use/land cover on spatial correlation length and soil hydraulic conductivity

The spatial analysis has highlighted some important issues. The fact that the new median K_{fs}-values calculated by including only every alternate, second or third K_{fs} did not differ significantly from the original medians indicates that the spatial correlation length had little or no effect on the conclusions derived from the entire data-set or from the subsequently reduced sample populations. This suggests that the near-surface K_{fs}-values for the degraded pasture, pine forest and natural forest sites are more strongly associated with spatially independent random variation. On the other hand, the increase in spatial correlation length with depth throughout the profile beneath the degraded pasture is comparable to that reported for degraded pasture land in the mountains of south Ecuador (Zimmerman and Elsenbeer, 2008). Similarly, the currently found increase in spatial correlation length down to a depth of 0.25 m and the subsequent decrease with depth is also a known characteristic of little disturbed natural forest soils elsewhere (Zimmermann and Elsenbeer, 2008). Although the spatial correlation length in the pine forest first increased and then started to decrease, as was also observed in the natural forest, the pine forest is exceptional in that there was little variation in spatial correlation length with depth. In summary, no general depth-related spatial pattern was observed across all three study sites.

The present work has further highlighted a marked contrast in the changes in K_{fs} with depth between the natural forest and the other two land covers studied (pine forest and degraded pasture). The reported high K_{fs} in the surficial soil horizons (<0.25 m) of the natural forest and the subsequent decrease in K_{fs} with depth away from direct soil biological influences (cf. the diminishing SOC values with depth listed in Table 4.1) is a well-known characteristic of soils beneath little disturbed tropical forests (Elsenbeer, 2001; Bonell, 2005; Chappell et al., 2007). The data from the natural forest provide a 'baseline' for appreciating the marked reductions in $K_{fs,}$ especially at depths less than 0.25 m, within the soils of the intensely disturbed degraded pasture and pine forest due to multi-decadal human–induced pressures as also reported for a similar situation elsewhere in the Middle Mountains of Nepal by Ghimire et al. (2013).

Major decreases in surface- and near-surface K_{fs} following the conversion of forest to grazed pasture have been reported for a variety of tropical and sub-tropical settings (Alegre and Cassel, 1996; Tomassella and Hodnett, 1996; Deuchars et al., 1999; Zimmermann et al., 2006; Molina et al., 2007; Tobón et al., 2010) including the Himalaya (Patnaik and Virdi, 1962; Gerrard and Gardner, 2002; Ghimire et al., 2013). Moreover, the noted reversal in the trend of K_{fs} with depth (i.e., an initial increase in K_{fs} followed by a decrease) beneath the degraded pasture and pine forest is a characteristic in common with other reports where grazing and other soil compacting pressures have been sustained over decades (e.g., Tomassella and Hodnett, 1996; Godsey and Elsenbeer, 2002; Hamza and Anderson, 2005; Zimmermann et al., 2006). Furthermore, it is clear that the hydraulic conductivity of the pine forest soil 25 years after the trees were planted still reflects the influence of the previous degraded grassland that is further sustained by the continued human access and collection of forest products (cf. Singh and Sundriyal, 2009; Joshi and Negi, 2011; Ghimire et al., 2013). As most of the reduction in K_{fs} appears to be effected during the first few years of grazing (particularly in the case of mechanised forest conversion followed by intensive grazing) (Alegre and Cassel, 1996; Martinez and Zinck, 2004; cf. Zimmermann et al., 2010), one would expect comparatively little additional effect of multi-decadal grazing on infiltrability. Indeed, at 23–29 mm h^{-1}, the reported rates of surface K_{fs} after 20–30 years of grazing (Deuchars et al., 1999; Colloff et al., 2010; Hassler et al., 2011) are very similar to the values obtained for the very degraded Himalayan pastures of the current study

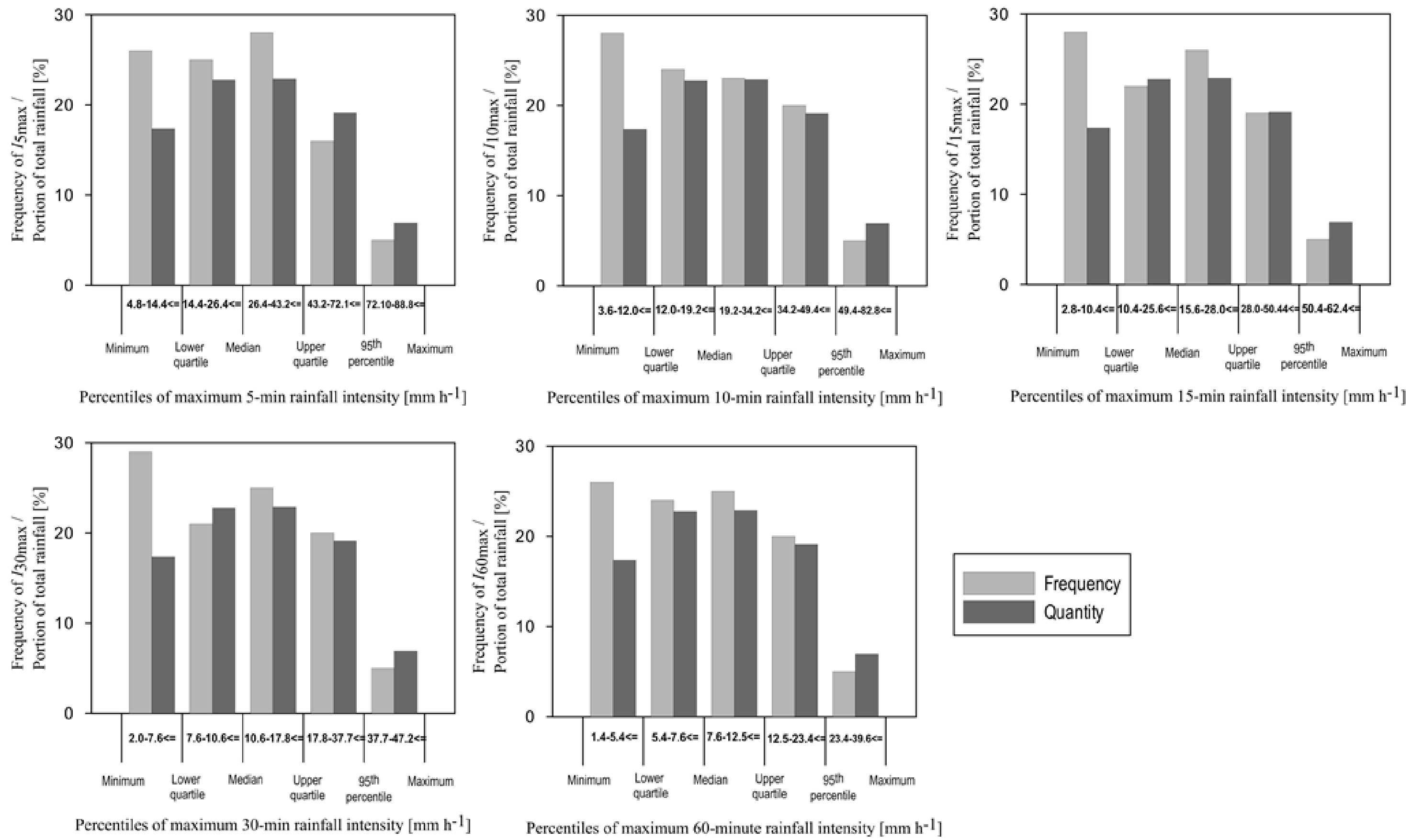

Figure 4.6: Frequency and quantity of maximum 5-, 10-, 30-, and 60-min rainfall intensities as recorded at the Dhulikhel rainfall station, Central Nepal, during the 2010 and 2011 monsoon periods.

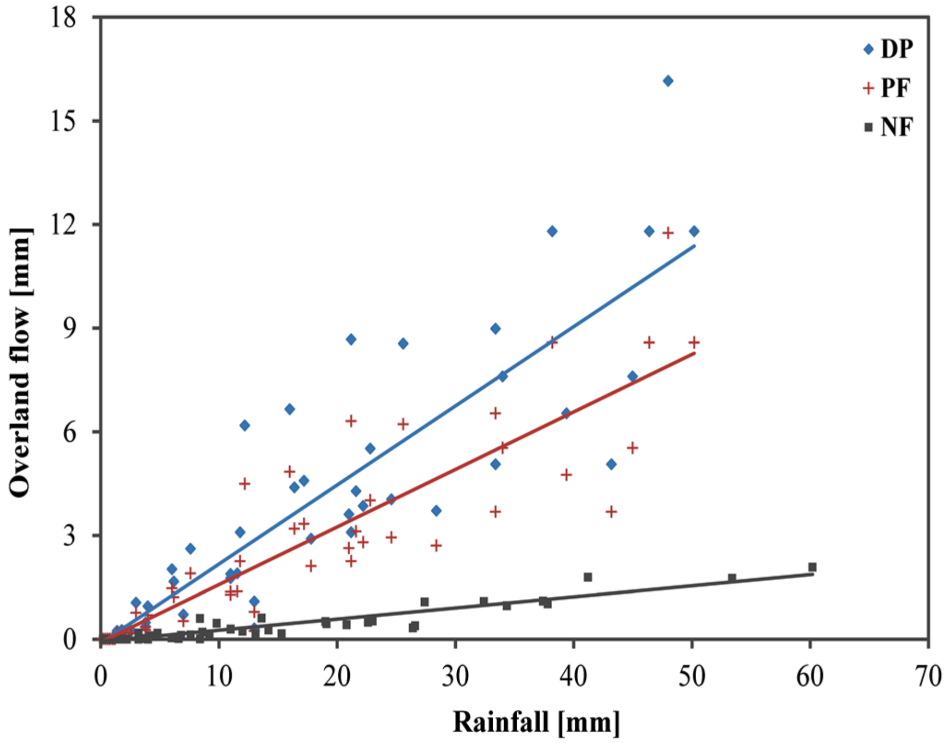

Figure 4.7: Relationship between daily rainfall (mm) and overland flow (mm) during the 2011 monsoon measuring campaign for degraded pasture (DP), planted forest (PF) and natural forest (NF) in the headwaters of the Jikhu Khola catchment, Central Nepal.

(median 18 mm h^{-1}, mean K_{fs} 25 mm h^{-1}; Table 4.2) and at Chautara (log-mean values of 33–39 mm h^{-1}; Gilmour et al., 1987; Ghimire et al., 2013). The radical reduction in surface- and near-surface K_{fs} under grazing conditions is primarily related to the destruction of macroporosity and by the trampling and strongly diminished soil biological activity after forest clearing and prolonged exposure of topsoil to the elements (McIntyre, 1958a,b; Lal, 1988; Deuchars et al., 1999; Colloff et al., 2010; cf. Bonell et al., 2010). Therefore, natural forest regrowth or reforestation on former degraded pasture land is considered by many to be an effective way of restoring diminished K_{fs}, although it may well take several decades of *uninterrupted* forest development before the infiltrability of highly degraded pasture reaches that associated with near-natural conditions (Gilmour et al., 1987; Godsey and Elsenbeer, 2002; Ziegler et al., 2004; Zimmermann et al., 2010; Hassler et al., 2011). However, the current results, and to a lesser extent the related work by Ghimire et al. (2013) in the Chautara area, suggest this one-dimensional view of restoring diminished surface- and near-surface K_{fs} requires some modification. This is particularly true in cases where anthropogenic pressures on both natural forests and tree plantations are as high as in the Middle Mountain Zone of the Himalaya (Singh and Singh, 1992; Mahat et al., 1987; Singh and Sundiyal, 2009; Joshi and Negi, 2011; cf. Gilmour and Shah, 2012). Indeed, one of the most striking findings of the present study is that the soil hydraulic conductivities at the surface and at a depth of 0.05–0.15 m in the 25-year-old pine reforestation did not differ significantly from those of the 150-year-old heavily degraded pasture, implying very similar hydrological response to rainfall (i.e., enhanced occurrence of IOF; cf. Figures 4.5a, b, 4.7 and 4.8c).

Table 4.5: Amounts of rainfall, throughfall and overland flow during the 2011 monsoon measuring campaign (20 June to 9 September) in the degraded pasture (DP), pine forest (PF), and natural forest (NF) in the headwaters of the Jikhu Khola Catchment, Central Nepal. Seasonal rainfall and throughfall total taken from Ghimire et al. (2012).

Experimental plot	Rainfall [mm]	Throughfall [mm]	Overland flow [mm]	Overland flow [%]	
				Rainfall	Throughfall
DP	878	-	187	21.3	-
PF	878	729	136	15.5	18.6
NF	709	540	17.7	2.5	3.2

4.5.2 K_{fs}, overland flow generation and forest management

It was previously acknowledged that the presently collected K_{fs} data are more likely to be representative of the higher end of the seasonal spectrum of this variable due to the fact that the measurements were undertaken during the latter part of a long dry season. Nonetheless, the rank order of measured seasonal overland flow totals (Table 4.5; cf. Figure 4.7) essentially reflects the *a priori* inferred stormflow pathways when comparing rainfall intensity with K_{fs} (Figures 4.5 and 4.8). For example, monsoonal totals of overland flow generated at the pine forest (15.5% of rainfall but 18.6% of throughfall) and the degraded pasture (21.3% of rainfall) were comparable (Table 4.5), as were their surface K_{fs}-values (Table 4.2). Likewise, the very low overland flow occurrence measured in the natural forest (Table 4.5, Figure 4.7) is also supported by the results of the K_{fs} survey, especially when it is taken into account that

K_{fs}-values during the main monsoon are likely to be lower than the ones determined during the dry season (Bonell et al., 2010; see also Vigiak et al. (2006) for a discussion of the representativity of different techniques to measure surface K_{fs}).High to very high incidence of overland flow on degraded pastures elsewhere in the Himalayas and South-east Asia have been reported previously (Impat, 1981; Chandler and Walter, 1998; Gerrard and Gardner, 2002; cf. Bruijnzeel and Bremmer, 1989) but **not** for *mature* (> 20 years old) pine plantations whose K_{fs}-values are usually sufficient to accommodate high rainfall intensities (Waterloo, 1994; cf. Ilstedt et al., 2007). It has been suggested that pine litter enhances surface soil water repellency and thus IOF generation relative to conditions encountered in broad-leaved forests, particularly for volumetric topsoil moisture levels (θ) below 0.2 $m^3\ m^{-3}$ and pH-values between 4 and 4.5 (Lebron et al., 2012). However, neither θ during monsoon conditions nor topsoil pH in the pine forest (5.42 ± 0.28, n = 9) could be considered conducive to soil water repellency in the presently studied pine forest. In addition, overland flow in undisturbed pine forest in the more seasonal (and thus likely to experience drier conditions) Kumaun Himalaya (north-west India) proved negligible (Pathak et al., 1984). Only where the trees are planted on heavy clay soils (as is often the case for teak) do K_{fs}-values not show any improvement during plantation maturation (Mapa, 1995; Bonell et al., 2010) and can overland flow be rampant (Coster, 1938; Bell, 1973; Wiersum, 1984). Likewise, where the development of a protective forest floor (litter layer, herb layer and understory) is interrupted repeatedly by fire, grazing or litter harvesting, the improvement of surface K_{fs} may be arrested (as observed in the pine forest, Table 4.2) or even be reversed as the plantation matures (Ghimire et al., 2013). However, even in the absence of trampling pressure, the combination of repeated litter removal, enhanced erosive power of crown drip (Wiersum, 1985; Hall and Calder, 1993), and reduced soil biotic activity (Ding et al., 1992; Hairiah et al., 2006) sets in motion a downward spiral towards a gradually diminishing infiltration capacity and a corresponding increase in overland flow production (Coster, 1938; Tsukamoto, 1975; Wiersum, 1984; Wiersum, 1985; Tiwari et al., 2009).

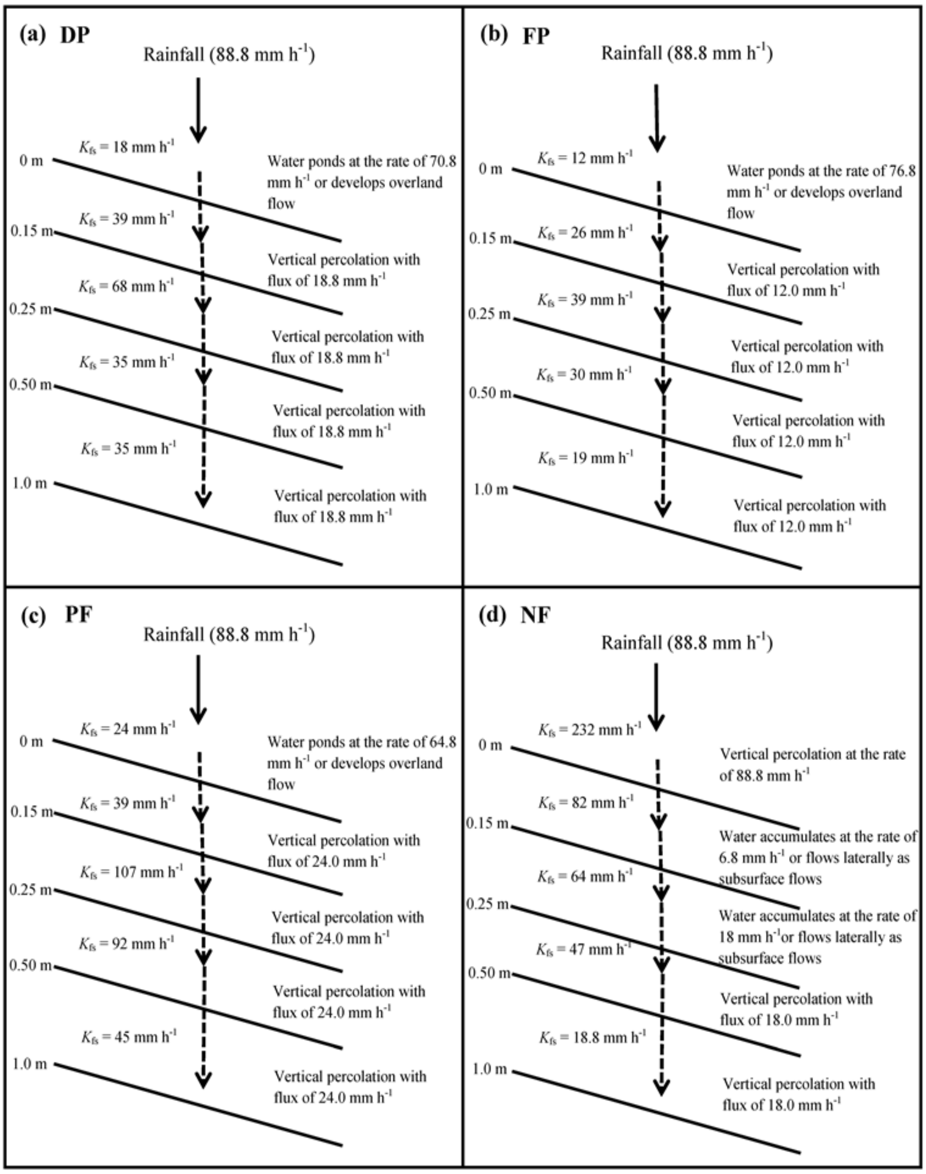

Figure 4.8: The percolation of rainfall delivered at the maximum 5 min intensity of 88.8 mm h^{-1} through various layers soil layers for contrasting land covers in the headwaters of the Jikhu Khola Catchment, Central Nepal: (a) degraded pasture (DP), (b) footpath (FP), (c) pine forest (PF), and (d) natural forest (NF). K_{fs} is field-saturated soil hydraulic conductivity.

4.5.3 Implications for dry season flows

The present finding of diminished infiltration opportunities in the pine forest and other, similarly heavily used forests (Gerrard and Gardner, 2002; Zhou et al., 2002; Tiwari et al., 2009; Ghimire et al., 2013) may prove critical when interpreting reports of declining stream baseflows following large-scale reforestation in the Middle Mountains of Central Nepal (República, 2012). Although reduced annual streamflow totals (and presumably baseflows as well) after afforesting *non-degraded* grasslands with fast-growing pines have been documented for many places (summarised by Jackson et al., 2005; Scott et al., 2005), dry-season flows may be expected to *increase* following reforestation of (heavily) degraded pasture land if the associated gains through improved rainfall infiltration override the extra evapotranspiration of the planted trees (the so-called 'infiltration trade-off' hypothesis) (Bruijnzeel, 1989; Bruijnzeel, 2004; cf. Wilcox and Huang, 2010; Zhou et al., 2010; Krishnaswamy et al., 2013).

As such, it is of interest to explore the possible consequences of the presently observed high overland flow volumes in the pine forest in the context of the 'infiltration trade-off' hypothesis more closely. Combining the above-mentioned overland flow percentages for the pine forest and degraded pasture with the mean annual site rainfall of 1500 mm (Merz, 2004), the difference in approximate annual overland flow production between the two land covers represents a gain in infiltration of ca. 90 mm year^{-1} under the pines relative to the grassland. Preliminary work on soil water uptake (transpiration) in the degraded pasture and pine forest (Baral, 2012), plus the rainfall interception losses from the pine forest and natural forest established by Ghimire et al. (2012) suggest a difference in annual evapotranspiration (ET) between the degraded pasture and pine forest of 360–400 mm, i.e., greatly in excess of the estimated gain in infiltration of 90 mm and in line with locally perceived declines in streamflow. Pertinently, even if the pine forest would have been well-managed and capable of absorbing even the highest rainfall intensities (i.e., no IOF occurrence and thus implying a maximum gain in infiltration equal to the observed overland flow total at the degraded pasture site of ~320 mm year^{-1}), this would still not have been sufficient to compensate the higher water use of the pines and thus prevent a decline in streamflow. Repeating the exercise for the natural forest (with an estimated annual ET close to 500 mm (Baral, 2012; Ghimire et al.,

2012) and very low overland flow production) would suggest the *ultimate* effect on dry-season flows to be near-neutral as the approximate gain in infiltration and the extra evaporative loss are very similar (ca. 300 mm year^{-1} each). However, in view of the widespread regeneration of vegetation on abandoned agricultural fields in the Middle Mountain Zone (Paudel et al., 2012) and because water use by vigorously regenerating forest tends to exceed that of old-growth forest like that of the natural forest site (cf. Giambelluca, 2002; Hölscher et al., 2005; Muñoz-Villers et al., 2012), it may be several decades before regional streamflows can be expected to stabilize or even rebound (cf. Wilcox et al., 2010; Zhou et al., 2010; Beck et al., 2013). Moreover, recent studies have reported an increasing trend in total monsoonal precipitation and extreme events in recent years for the Middle Mountain Zone and other parts of Nepal (Baidya et al., 2008; Lamichhane and Awasthi, 2009). Furthermore, application of a high resolution climate model elsewhere in the Middle Mountain Zone by Mishra and Herath (2011) indicated a significant increase in rainfall and rainfall intensity during the monsoon season and a decrease in dry-season rainfall. These scenarios suggest that there may be (much) more overland flow from planted forest sites in the future if the presently observed anthropogenic pressures on the forest do not decrease. Such circumstances will further hamper the replenishment of soil water and groundwater reserves, thereby reducing dry-season flows even further. On the other hand, the high surface- and near-surface K_{fs} observed in the little disturbed natural forest suggests that this type of forest may be able to cope with increased rainfall intensities in the future provided it does not become disturbed too much.

Overall, the present work has highlighted a ‘degradation’ in K_{fs} at the surface- and near-surface in the planted forest site. The potential benefits of reforestation in enhancing infiltration, and therefore the replenishment of soil water and groundwater reserves, are currently not realized by the continued need for access to the forest by the local population to obtain forest products (notably litter for animal bedding and subsequent composting), with sustained degradation of the forest floor as a result. The notion of forests established primarily for community needs *vis á vis* broader drainage basin functions is a point commonly ignored in forest hydrology where the focus tends to be placed more on the hydrological benefits of forests *per se* (i.e., without people) (e.g., Webb et al., 2012; Krishnaswamy et al., 2013). Such circumstances are common in the South Asian region.

4.5.4 Footpaths as source areas of storm runoff

Evidence is accumulating for various tropical and subtropical uplands – including the present study area–that rural footpaths, cattle trails within pastures, and rural yards can play a significant role in the generation of overall hillslope-scale IOF and accelerated erosion (Rijsdijk and Bruijnzeel, 1991; Giambelluca, 1996; Purwanto, 1999; Van Dijk, 2002; Ziegler et al., 2004a,b; Rijsdijk et al., 2007; Turkelboom et al., 2008; Tobón et al., 2010). These highly compacted surfaces can have extremely low surface infiltrabilities (< 10 mm h^{-1} in many of the examples cited above) and therefore they can be expected to exhibit a strong propensity towards producing IOF even during comparatively low rainfall intensities (cf. Figures 4.2b and 4.5a). For example, Rijsdijk et al. (2007) reported runoff coefficients of up to 70% for rural footpaths in the mountains of East Java, Indonesia. Indeed, in addition to the very low surface infiltrability of the presently studied footpath (Table 4.2) median K_{fs}-values at 0.05–0.15 m and 0.15–25 m depths beneath the footpath were close to the median value of I_{5max} and well below the 75% and 95% percentiles of I_{5max} (Figures 4.5b, c), suggesting frequent occurrence of IOF. Footpath-related IOF can be particularly harmful in that footpaths, despite their relatively small areal extent, can contribute disproportionally to catchment-scale storm runoff and sediment yield (Dunne and Dietrich, 1982; Ziegler and Giambelluca, 1997; Ziegler et al., 2004b; Cuo et al., 2006; Turkelboom et al., 2008). Moreover, the transmission rate of IOF from footpaths to streams can be (very) fast, owing to their connectivity with, and proximity to the streams. This is particularly true in the densely populated Middle Mountain Zone within the Himalaya where numerous footpaths typically run steeply uphill from the riparian zone. Footpath-related IOF is even more critical in the absence of downslope buffering zones with higher K_{fs} to dampen the flow (cf. Ziegler et al., 2004a, b; Turkelboom et al., 2008). This was also the case in the present study area where the footpath was surrounded by degraded pasture and poorly managed pine forest, both having low surface- and near-surface K_{fs} (Table 4.2). Collectively, such footpath-related IOF in combination with buffering zones of low infiltration capacity can be expected to elevate local-scale flash floods (see examples discussed by Bruijnzeel and Bremmer, 1989).

4.5.5 On the need to protect the remaining natural forests of the Middle Mountains

At the other end of the runoff-generation spectrum, the median surface- and near-surface K_{fs}-values in the little disturbed natural forest (Table 4.2) were such that even under conditions of intense rainfall, vertical percolation was indicated as the dominant initial hydrological pathway, only subsequently to become diverted laterally as SSF below 0.15 m depth during the most intense rainfalls (Figures 4.5c and 4.5d; Figure 4.8d). Therefore, conditions in the natural forest and similarly well-maintained forests elsewhere in the Himalaya (Pathak et al., 1984; Gardner and Gerrard, 2002) will encourage the replenishment of soil water and groundwater reserves through vertical percolation more than in any of the other land-cover types studied (Figure 4.5; Figure 4.8d), and thus better sustain baseflows during the long dry season for community water supply. The importance of the latter can hardly be overstated (Merz et al., 2003; Schreier et al., 2006; cf. Bandyopadhyay, 2013).

The abundance of macropores arising from the decomposing activity of soil microflora and fauna associated with the higher organic matter inputs to the forest floor in undisturbed forests (cf. Table 4.1) is considered a prime reason for the commonly observed high infiltration capacity and near-surface K_{fs} of such forest soils, together with a well-developed root system (Lal, 1988; Deuchars et al., 1999; Bonell, 2005; Zimmermann and Elsenbeer, 2008; Bonell et al., 2010). The results presently obtained for the natural forest and pine forest once more illustrate the importance of a well-developed litter layer and understory vegetation to hillslope hydrological functioning (cf. Pathak et al., 1984). Indeed, the presence of a dense shrub layer and a thick litter layer were considered to be the chief causal factors of the observed soil-protective and rainfall-absorbing role of degraded natural forest studied by Ghimire et al. (2013) in the Chautara area. Elsewhere in the Middle Mountain Zone, Tiwari et al. (2009) describe a case where overland flow and sediment production in a community-managed broad-leaved forest under a restricted tree pruning and grazing regime greatly exceeded that in nearby unmanaged forest. Although pruning and grazing were restricted in the latter forest, ground cover by undergrowth and litter was much better compared to the community-managed forest where litter and understorey vegetation were regularly harvested and left the forest floor unprotected (cf. Coster, 1938; Wiersum, 1985; Tiwari et al., 2009). The present results also underscore the hydrological importance of preserving the remaining natural forests of the Middle Mountain Zone (cf. Pathak et al., 1984; Negi et al., 1998; Singh, 2007; Ghimire et al., 2013).

4.6 Conclusions

To shed more light on the impact of reforesting degraded hillsides on surface- and sub-surface hydrological functioning, field-saturated hydraulic conductivities (K_{fs}) were measured in a little disturbed natural forest, a heavily degraded pasture, an intensively used footpath, and a mature planted pine forest subject to considerable anthropogenic pressure in rugged terrain near Dhulikhel, Central Nepal.

The high surface and near-surface K_{fs} observed in the natural forest effectively prevented the occurrence of large-scale infiltration excess overland flow (IOF) even for the most extreme rainfall events. Thus, natural forest favours largely vertical percolation and the replenishment of soil water and groundwater reserves. Conversely, very low surface- and near-surface K_{fs} were found in the degraded pasture (18–39 mm h^{-1}) and particularly for the footpath (12–26 mm h^{-1}) which encouraged the generation of IOF even during events with moderate rainfall intensities. The large volumes of overland flow generated on footpaths and degraded pastures in the study area can be expected to contribute disproportionally to local-scale stormflows due to the absence of well-developed footslope buffer zones of sufficiently high K_{fs} and the fact that many footpaths cross steep hillsides, thereby providing rapid transfer of IOF to the nearest stream.

Various human interventions (notably the regular collection of litter from the forest floor for animal bedding, plus understory removal by cattle grazing and fuelwood harvesting) in the planted pine forest had not allowed any improvement in K_{fs} over 25 years. In fact, surface and near-surface K_{fs} and IOF volumes in the pine forest had remained similar to those observed at the nearby degraded pasture site (i.e., the situation prior to reforestation). Reforestation of degraded hillslopes *per se*, therefore, does not guarantee the improvement of soil infiltration capacity and the restoration of hillslope hydrological functioning. The corresponding reduced infiltration must be considered a critical factor – together with the higher water use of the pine trees compared to old-growth broad-leaved forest – when interpreting the presently perceived decline in dry season flows in the Middle Mountains of Central Nepal.

The present results further illustrate the positive influence of a well-developed litter layer and understory vegetation on surface- and sub-surface hydrological functioning. They also bring out the hydrological importance of preserving the remaining old-growth forests of the Middle Mountain Zone of the Nepalese and Indian Himalaya. Continued degradation of the remaining old-growth vegetation and planted forests can be expected to lead to further increased storm runoff across the zone's river network during the main monsoon season due to the corresponding reduction of surface- and shallow subsurface K_{fs}, which may, in turn, have a further detrimental effect on already declining dry season flows because of further impaired replenishment of soil water and groundwater reserves.

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Chapter 5

The effects of sustained forest use on hillslope soil hydraulic conductivity in the Middle Mountains of Central Nepal[1]

Abstract. This work investigated the multi-decadal changes in field-saturated hydraulic conductivity, K_{fs}, beneath severely degraded pasture, natural forest and two mature planted *Pinus roxburghii* stands between two sets of measurements made in 1986 and 2011 at the same locations in the Middle Mountains of Central Nepal. Multiple measurements of K_{fs} were made at the four sites, both at the surface and at depths of 0.05–0.15, 0.15–0.25 and 0.25–0.50 m. The K_{fs} results were subsequently combined with rainfall intensities associated with different time intervals to infer multi-decadal changes in dominant hillslope stormflow pathways.

The widely assumed hydrological benefits of reforesting degraded land through the enhancement of near-surface permeability due to such factors as the incorporation of a greater amount of organic matter, formation of macropores, as well as root development were not observed in this study. Continued heavy use of the natural and planted forests of the Middle Mountains, particularly the removal of understory vegetation and leaf litter, and cattle grazing, are considered to be the chief causal factors of the presently observed deterioration in forest hydrological functioning. This situation is typical not only of the Middle Mountain Zone across the Himalaya, but is also observed in other densely populated parts of South- and South-East Asia.

The key conclusion of this work is that simply planting trees in degraded landscapes is not sufficient in itself to restore watershed hydrological functioning. Attention also needs to be given to on-going management of the reforested areas to balance product usage with watershed functions.

[1]This chapter is based on: **Ghimire CP**, Bruijnzeel LA, Bonell M, Coles NA, Lubczynski MW, Gilmour DA (2013) The effects of sustained forest use on hillslope soil hydraulic conductivity in the Middle Mountains of Central Nepal. *Ecohydrology*, **7**, 478–495.

5.1 Introduction

Traditionally, the Middle Mountain Zone (or Lesser Himalaya) constitutes the most densely populated part of the Himalaya, both in Nepal (Hrabovszky and Miyan, 1987) and India (Singh et al., 1984). High population pressure and heavy demands for forest products have led to widespread forest removal (Dobremez, 1976; Singh and Singh, 1992; Tucker, 1987), subsequent land degradation (Mahat et al., 1986; Mahat et al., 1987; Tiwari, 1988) and deterioration of streamflow regimes (Bartariya, 1989). Until comparatively recently, it was widely assumed that deforestation and overgrazing in the Himalaya were primarily responsible for the large-scale flooding and sedimentation experienced in the plains of northern India and Bangladesh (e.g., Eckholm, 1976; Nautiyal and Babor, 1985; Myers, 1986). Whilst this view is no longer tenable in the light of subsequent scientific evidence demonstrating the comparatively limited influence of land use on these *large-scale* hydrological phenomena (Bruijnzeel and Bremmer, 1989; Ives and Messerli, 1989; Hoefer, 1993; Hoefer and Messerli, 2006; cf. Gardner and Gerrard, 2003), the *local* adverse hydrological effects of advanced land degradation – such as accelerated erosion, enhanced peak discharges and reduced dry-season flows (Bartarya, 1989; Bruijnzeel and Bremmer, 1989) required remedial action (cf. Negi et al., 1998). At the same time, forest land had to provide a variety of goods and services (such as timber and fuelwood, plus non-timber products including water, fodder, and litter for animal bedding) to local communities in support of their subsistence farming systems (Campbell and Mahat, 1975; Mahat et al., 1987). A major reforestation programme was initiated in the early 1980s in the Middle Mountains of Central Nepal (Shepherd and Griffin, 1984) to address this multitude of issues. By the year 2000, ~23,000 ha of degraded pasture land and shrubland in the Districts of Sindhu Palchok and Kabhre Palanchok (Central Nepal) had been converted to evergreen plantation forests (mainly *Pinus roxburghii* and *P. patula*) (District Forest Offices at Kabhre Palanchok and Sindhu Palchok, unpublished data). The original vegetation of these areas consisted of forests of semi-evergreen broadleaf species and pines (mainly *P. roxburghii*) (Dobremez, 1976). These reforestation activities were carried out with the active participation of local communities and most of the resulting forests were formally handed over to Community Forest User Groups (CFUGs) in the early 1990s. Management of these forests has been in community hands since that time, with access and usage, including animal grazing and product harvesting, carried out in accordance with approved Operational Plans. Country-wide, more than 1.65 million ha of forest, both planted and natural, has been officially handed over to CFUGs (HURDEC and Hobley 2012). Numerous surveys have indicated that there has been a substantial increase in both forest area and forest quality across the Middle Mountains in the past two decades, thus reversing the earlier trends in deforestation in the zone (HURDEC and Hobley 2012).

Reforestation of degraded land in the tropics and sub-tropics is often conducted in the expectation that disturbed streamflow regimes (commonly referred to as the ‘too little – too much syndrome’: Eckholm, 1976; Bartarya, 1989) will be restored by the increased rainfall absorption afforded by soil improvement after tree planting (Scott et al., 2005; cf. Ilstedt et al., 2007). At the same time, the water use of fast-growing tree plantations tends to be (much) higher than that of the degraded vegetation they typically replace (Scott et al., 2005) – particularly where the tree roots have access to groundwater (Calder, 1992; Kallarackal and Somen, 1997). Furthermore, major improvements in the topsoil infiltration capacity of severely degraded land after tree planting may easily take several decades of *undisturbed* forest development to fully materialise (Gilmour et al., 1987; Scott et al., 2005; Bonell et al., 2010). As such, reforesting degraded pasture or shrub land may also cause already diminished dry season flows to become reduced even further, depending on the net balance between increases in soil water reserves afforded by improved infiltration *versus* decreases caused by the higher plant water uptake (Bruijnzeel, 1989; Bruijnzeel, 2004; cf. Malmer et al., 2010). Naturally, such considerations assume additional importance for regional water resources management under more seasonal climatic conditions, such as those prevailing in the Middle Mountain Zone where ~80% of the annual rainfall is delivered during the main monsoon (June–September; Merz, 2004).

There is increasing evidence that soil infiltrability (Hillel, 1980) increases with time during *natural forest regrowth* after abandonment of agricultural or pasture land in the tropics (e.g., Deuchars et al., 1999; Ziegler et al., 2004; Zimmermann *et al.*, 2006; Hassler et al., 2011). Conversely, information on the magnitude of infiltrability in tropical *tree plantations* is still scarce (see review by Ilstedt et al., 2007) and usually represents a single measurement in time, rather than showing the actual development of infiltrability as the plantation matures (e.g., Mapa, 1995;

Zimmermann et al., 2006; Bonell et al., 2010). A rare example of the latter was given by Gilmour et al. (1987) who reported on the contrasts of *in situ* field-saturated hydraulic conductivity (K_{fs}) (Bouwer, 1966; Talsma, 1987) in top- and sub-soils under various land-cover types in Central Nepal. The land covers investigated ranged from heavily grazed and trampled grassland, five- and twelve-year-old pine plantations, and a relatively undisturbed natural forest. Values of K_{fs} of the deepest layer did not differ significantly between sites, suggesting comparative geological homogeneity amongst the sites. However, variability in K_{fs} increased with proximity to the surface, with maximum differences occurring in the 0–0.1 m layer, thereby causing major differences in the propensity to generate infiltration-excess overland flow (*sensu* Horton, 1933) between land-cover types. Recovery of top-soil K_{fs} after reforestation was evident from the increases observed for the five- and twelve-year-old pine stands although values were still a long way from the infiltrability associated with the largely undisturbed natural forest. Gilmour et al. (1987) concluded that the observed improvements in K_{fs} relative to the degraded grassland site were due to a reduction in surface compaction by the exclusion of grazing animals (cf. Deuchars et al., 1999; Hassler et al., 2011), as well as improved soil aeration brought about by the enhanced activity of soil micro-flora and fauna feeding on the gradually accumulating litter layer after reforestation (cf. Lal, 1988; Zhou et al., 2002). Although there are only a few studies of the adverse impacts of multi-decadal forest use on K_{fs} (e.g., Patnaik and Virdi, 1962; Bonell et al., 2010), the recovery of K_{fs} following reforestation is likely to be inversely related to the intensity of forest usage. The forests (both natural and planted) of the Middle Mountain Zone along the length of the Himalayan chain are intensively used, with the lopping of branches for fuelwood, cutting of trees for timber, harvesting of understory vegetation and grasses for fodder, grazing by animals, and litter collection for animal bedding, composting and fuel all being widely practiced (Singh et al., 1984; Sing and Singh, 1992; Mahat et al., 1987; Gilmour and Fisher, 1991). As such, depending on the intensity of use of the replanted stands, the upward trend with time for K_{fs} envisaged by Gilmour et al. (1987) may continue, level off, or even decline. Even less is known about the time frame, the changes in dominant hillslope runoff generation processes, and catchment-scale stormflow production associated with the respective scenarios (cf. Bruijnzeel and Bremmer, 1989; Negi, 2002; Zimmermann and Elsenbeer, 2009; Bonell et al., 2010).

In the present study, the sites of Gilmour et al. (1987) were revisited after more than 25 years. It was expected that re-measuring K_{fs} at the same locations and using comparable measurement techniques would provide a direct estimate of the net effect of continued forest growth and litter layer development on the one hand, and forest use and disturbance on the other, over the 25-year time period between the two sets of measurements. To the best of our knowledge, this is the only study of this kind in the (sub)tropics. Next, the information on changes in K_{fs} over time was combined with regional rainfall intensity data of various temporal resolutions to infer likely changes in the dominant patterns and pathways of hillslope storm runoff response to rainfall (Chappell et al., 2007) after 1986. Finally, the implications of the present findings for likely changes in regional dry-season flows as well as for forest management are discussed in the context of land-use changes unfolding in the Middle Mountains of Nepal.

5.2 Study area

The study area (27^0 47’ 4” N, 85^0 42’ 33” E, 1660 m a.m.s.l.) is located in the Sindhu Palchok District close to the District Headquarter town of Chautara in the Middle Mountains of Nepal, about 40 km north-east of the capital Kathmandu. The region has a complex geology that has resulted in equally complex spatial patterns of topography, climate, and vegetation (Dobremez, 1976). The geology consists of phyllites, schists and quartzites in which Cambisols and Luvisols of a generally silty to clay loam have developed (Table 5.1; cf. Maharjan, 1991; Gardner and Gerrard, 2003). The elevation varies from 800–2400 m a.m.s.l. Depending on the elevation the climate is humid subtropical to warm-temperate. Generally, the Middle Mountains are well-watered, with annual rainfalls of 2000 mm or more, most of which (~80%) is delivered during the wet summer months (‘monsoon’, June–September). There is a distinct dry season that generally lasts from about October to May. However, rainfall varies with elevation and exposure to the prevailing moist air masses. The mean annual rainfall as measured at 1560 m a.m.s.l. in the nearby Jikhu Khola catchment between 1993 and 1998 amounted to 1487 (± 155 mm) (Merz, 2004). Similar amounts were recorded in the same area at 1580 m a.m.s.l. between October 2010 and September 2011 by the current lead author (Ghimire et al., 2012). Most rainfall events deliver less than 5 mm and at an average intensity ≤ 2.5 mm h^{-1} (Merz et al., 2006; Ghimire et al., 2012). Between 1000 and 2000 m a.m.s.l. the natural vegetation consists of a largely evergreen mixed broad-leaf forest dominated by *Schima wallichii* and various chestnuts and oaks (*Castanopsis* spp., *Quercus* spp.), patches of pine (mainly *Pinus roxburghii*) and admixtures of *Rhododendron arboreum* above ~1500 m a.m.s.l. (Dobremez, 1976). Due to the high population pressure and associated heavy usage of forest products by humans and animals, much of this species-rich forest had disappeared by the middle of the 20th Century (Mahat et al., 1986), with most of the remaining forest either occurring on slopes that were too steep for agricultural activity or in various stages of degradation (Dobremez, 1976; cf. Figure 5.4). However, substantial changes in land use began occurring in the 1970s and 1980s that changed the nature of the landscape across the Middle Mountains. Prior to the 1980s, farming families in the region tended to keep relatively large herds of animals (cattle, buffalo and goats) that were open-range grazed on common land. These circumstances contributed to the deforestation and degradation of steep hillslopes that was widely reported at the time in the context of a supposed “Himalayan environmental crisis” (Eckholm, 1976; Ives and Messerli, 1989). In the course of the 1970s, hill farmers began changing several key aspects of their farming practices. They changed their livestock management from open-range grazing of large herds to stall-feeding a small number of large animals (often one or two) and cutting and carrying fodder and animal bedding material from private and common land, a practice that still continues. A major consequence of this shift was that, particularly in some districts in Central Nepal, a large area of common land (officially Government forest land) became available for reforestation (Gilmour and Fisher, 1991). A second change commenced at about the same time when farmers began responding to a critical shortage of tree and forest products needed for their subsistence farming by planting trees. In addition, farmers commenced to allow naturally regenerated tree seedlings to remain on the periphery of their cultivated terraces and in the non-cultivated patches within the farming complex (Gilmour, 1988, 1995; Carter and Gilmour, 1989; Gilmour and Fisher, 1991; Gilmour and Nurse, 1991). The cumulative effect of these changes is that there are now substantially more trees and forests across the landscape of the Middle Mountains, on both private and common land, than there were 30 years ago (Gilmour and Fisher, 1991; HURDEC and Hobley, 2012).

The study sites were selected to represent the various stages of reforestation, ranging from heavily grazed and trampled grassland ('baseline') to a remnant broad-leaved forest whose tree layer was heavily disturbed ('semi-natural forest'). The oldest surviving reforestation stands in the area at the time the measurements were taken (February 2011) had an age of 36 years. The visual inspection of the greenness of the leaves and of the presence/absence of external diseases suggested that the planted forests were healthy enough at the study sites. Previously, the area had been subject to substantial deforestation dating back more than a century (Mahat et al., 1986).

Table 5.1: Selected physical and chemical properties of the soils at 0–0.15 m depth with the different land cover types under study near Chautara, Central Nepal. Values listed are the means (±SD) of six samples per site. Values sharing at least one same supescript are not significantly different from each other and *vice versa*.

Variable	Degraded Pasture [Site-1]	Pine Forest [Site-3]	Pine Forest [Site-4]	Semi-natural Forest [Site-5]
Bulk density [g cm^{-3}]	1.30$^{\mathbf{a}}$ (±0.05)	1.20$^{\mathbf{a}}$ (±0.1)	0.97$^{\mathbf{b}}$ (±0.06)	0.90$^{\mathbf{b}}$ (±0.18)
Clay [%]	34$^{\mathbf{c}}$ (±7)	30$^{\mathbf{c}}$ (±6)	39$^{\mathbf{c}}$ (±11)	20$^{\mathbf{c}}$ (±10)
Sand [%]	21$^{\mathbf{f,g}}$ (±6)	30$^{\mathbf{f}}$ (±5)	12$^{\mathbf{g}}$ (±7)	25$^{\mathbf{f}}$ (±12)
Silt [%]	45$^{\mathbf{h}}$ (±11)	40$^{\mathbf{h}}$ (±8)	49$^{\mathbf{h}}$ (±11)	55$^{\mathbf{h}}$ (±10)
SOC [%]	3.6$^{\mathbf{i}}$ (±0.77)	4.3$^{\mathbf{i}}$ (±0.98)	5.3$^{\mathbf{i,j}}$ (±1.2)	6.2$^{\mathbf{j}}$ (±1.1)

Of the five original sites of Gilmour et al. (1987), four were selected for re-measurement of K_{fs}. The soil surface of the second site (i.e., Site 2 in Table 1 of Gilmour et al., 1987) had numerous large stones, thereby rendering it difficult to take consistent measurements of K_{fs}. The four investigated sites (with site numbers in brackets corresponding with those of Gilmour et al., 1987) can be characterised as follows:

Degraded pasture land (Site-1): Originally deforested more than 125 years ago and already heavily grazed and trampled with widespread soil erosion in 1986. The current site condition differs only slightly from what it was in 1986 (Figure 5.1). Numerous foot paths are now distributed across the slope. The site is also more severely over-grazed and patches of soil surface are exposed. Little or no vegetation exists. The results of the re-measurements are considered to represent the net effect of 25 years of additional cattle grazing and trampling pressure by both cattle and humans.

Pine forest (Site-3): Planted with *P. roxburghii* in 1974 and aged 36 years in 2011. Site-3 was similar to Site-1 prior to reforestation but grazing had been excluded for 12 years by 1986, although litter was regularly collected. The surface condition in 2011 was quite different from that in 1986 (Figure 5.2). Since 1990, there has been an increase in cattle grazing. In addition, most of the understory has been removed and grass is cropped to ground level. The results of the re-measurements are considered to represent the net effect of more than 20 years of continuous heavy cattle grazing and trampling.

Pine forest (Site-4): Planted with *P. roxburghii* in 1974 and aged 36 years in 2011. Location adjacent to Site-1. The vegetation prior to planting consisted of low shrubs and herbs about 40 cm high plus a few scattered remnants of the original forest in 1986 (Figure 5.3). In 1990, the local people started to collect the litter layer for animal bedding. Nearly 50% of the planted trees were removed for timber and fuelwood between 1990 and 2010 in accordance with an approved community forest Operational Plan. However, grazing animals are generally excluded with the help of a forest watcher. The soil surface is well-covered with broad-leaved shrubs. The results of the re-measurements are considered to reflect the net effect of more than 20 years of litter collection and harvesting of understory vegetation.

Near-natural forest (Site-5): Back in 1986 this forest was considered to be in a near-natural state as harvesting was limited to some leafy materials for animal bedding and fodder. The current condition of this site is very different from what it was in 1986 (Figure 5.4). Extensive tree loss has occurred in the meantime although the soil surface is still well covered with a dense broad-leaved understory as well as a litter layer. Grazing animals and litter collection are excluded with the help of a forest watcher. The results of the re-measurements are considered to reflect the net effect of the gradual transformation of a nearly undisturbed natural forest to a shrubland with scattered remnant trees, particularly the effect of manual logging and ground-based removal of timber.

5.3 Materials and methods

5.3.1 Sampling design

Based on interviews with local people during the field campaign in February 2011, the sites can be considered to have been free from natural disturbances such as landslips. Hence, any improvement in K_{fs} within the reforested sites over the more than 25 years that have elapsed since the first round of measurements might be attributed largely to the effect of reforestation (i.e., the building up of the litter layer and incorporation of soil organic matter; cf. Lavelle and Spain, 2001), and any decrease in K_{fs} to site disturbance by humans and animals (e.g., trampling pressure, grazing, and litter collection) and thus to increased soil surface exposure to erosive crown drip (cf. Wiersum, 1985; Brandt, 1988; Hall and Calder, 1993).

Multiple measurements of field K_{fs} were made at the four Chautara sites, both at the surface and at depths of 0.05–0.15 m, 0.15–0.25 m, and 0.25–0.5 m. It was envisaged that the measurements made at the latter depth intervals would provide sufficient information to characterize the nature of water movement through the soil profiles on the prevailing steep slopes. For all depths except at the surface, a stratified sampling approach was used whereas measurements were made at the plot scale. First, two 45 m long lines were laid out within each plot along and perpendicular to the hillslope gradient. This was done to avoid the clustering of sampling in one direction only and to distribute the samples both along and across the hillslope. Further, K_{fs} was measured at 3 m intervals, which resulted in 16 measurements in each direction. For the measurement of surface $K_{fs,}$ 5–6 replications were taken per site via random sampling using a disc permeameter. The number of K_{fs} measurements made at the surface was much less than for the other depths because of the practical limitations of using a disc permeameter on steeply sloping land (K.R.J. Smettem, personal communication, 2012).

5.3.2 Measurements of soil hydraulic conductivity

A disc permeameter (Perroux and White, 1988; McKenzie et al., 2002) was used for the measurement of surface K_{fs}. Before conducting the measurements, any straw and stubble present were removed from the soil

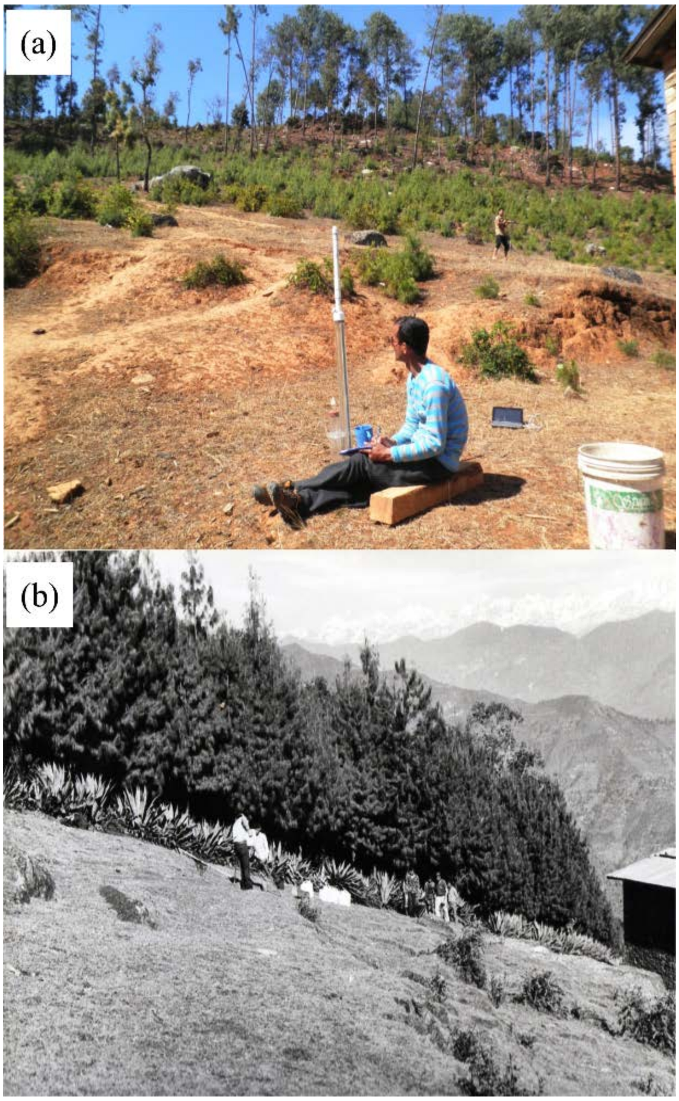

Figure 5.1: Degraded pasture (Site-1) near Chautara, Central Nepal in: (a) 2011 (Colour), and (b) 1986 (B/W)

surface with the least amount of surface disturbance. A steel ring of 25.5 cm diameter and 3 cm height was then inserted into the soil for about 4 mm. In order to improve the contact between the disc and the soil surface, a 5 mm thick layer of fine sand (size < 2 mm) was placed on the soil surface inside the ring. After preparing the soil surface in this way, the apparatus was placed on the ring. The rates of water discharge through the disc, as inferred from changes in the water levels in the storage tower of the apparatus, were recorded until steady-state flow was reached. K_{fs} was then calculated along the lines of McKenzie et al. (2002):

$$K_{\text{fs}} = I - \frac{4bS_o^2}{\pi r(\theta_o - \theta_n)} \tag{5.1}$$

where θ_n (m^3 m^{-3}) and θ_o (m^3 m^{-3}) are the *in situ* volumetric soil moisture contents before and after infiltration, respectively; *b* is a constant (0.55); S_o (mm $h^{-1/2}$), the sorptivity obtained by plotting cumulative infiltration volume versus the square root of time since the start of infiltration; *r*, the radius of the disc permeameter base (127.5 mm); and I (mm h^{-1}), the steady-state infiltration rate, calculated as:

$$I = \frac{q}{\pi r^2} \tag{5.2}$$

where q (mm^3 h^{-1}) is the slope of the plot of cumulative infiltration versus time after reaching steady–state conditions; with *r* as defined previously.

For the measurement of K_{fs} in deeper soil layers, a constant-head well permeameter (CHWP) was used (Talsma and Hallam, 1980). The use of the CHWP was restricted to the dry season to minimise errors from smearing of the auger hole walls (Chappell and Lancaster, 2007). The experimental procedure involved augering a cylindrical hole (with radius a = 4 cm) to the desired depth. Any sealing of pores in the column walls due to the augering was minimised by brushing the walls with a small metal brush. The hole was pre-wetted for 20 min before taking the measurement to achieve perimeter saturation as described by Talsma and Hallam (1980). The CHWP was then inserted to the required depth and the flow measured until a steady–state flow rate was reached. K_{fs} (mm h^{-1}) values were calculated from the measurements using equation 11 of Reynolds et al. (1983):

$$K_{\text{fs}} = \frac{CQ_t}{2\pi H^2\left[1 + \frac{C}{2}\left(\frac{a}{H}\right)^2\right]} \tag{5.3}$$

where Q_t (mm^3 h^{-1}) is the steady-state flow rate; *H* (mm), the constant height of ponded water in the well; *a* (mm), the radius of the well; and *C*, a dimensionless shape factor calculated as:

$$C = \sinh^{-1}\left(\frac{H}{a}\right) - \sqrt{\left(\left(\frac{a}{H}\right)^2 + 1\right)} + \frac{a}{H} \quad (5.4)$$

5.3.3 Inferring dominant hillslope hydrological pathways during rainfall

A growing body of hillslope hydrological and small catchment studies in the tropics and subtropics suggests that the most important parameters governing the way water runs off during and shortly after rainfall events (the so-called dominant stormflow pathways; Bonell, 2005) include the vertical distribution of K_{fs} and prevailing rainfall intensities (Bonell et al., 1983; Gilmour et al., 1987; Wenzel et al., 1998; Elsenbeer et al., 1999; Godsey and Elsenbeer, 2002; Ziegler et al., 2006; Zimmermann et al., 2006; Zimmermann and Elsenbeer, 2009; Bonell et al., 2010; Germer *et al.*, 2010; Hassler et al., 2011). In order to infer the dominant stormflow pathway at each site, selected percentiles of maximum rainfall intensities I_{max} (e.g., over 5 or 30 min: I_{5max} and I_{30max}), as measured at Dhulikhel (~22 km towards the south-east of the study sites, see below), were superimposed on the respective box plots of K_{fs}. Whilst short and intense storms can lead to the occurrence of infiltration-excess overland flow (IOF; Horton, 1933), longer-duration rainfall of low intensity can produce subsurface stormflow (SSF) and saturation-excess overland flow (SOF) (Bonell, 2005). The latter is usually generated in low-lying areas at footslopes, around streams and in depressions, but SOF may also become widespread on hillslopes where rainfall intensities exceed the transmission capacity of a soil layer at shallow depth (Bonell et al., 1983).

5.3.4 Rainfall measurements

The rainfall intensity data used in the characterisation of the dominant hillslope hydrological pathways in the Chautara plots and changes therein as a function of ecosystem degradation were recorded at Dhulikhel during the monsoon periods of 2010 and 2011. Although made at a location some 22 km towards the south-east, these data were considered to be typical of the rainfall characteristics in the Chautara area as they were collected at a similar elevation (1580 m a.s.l.versus 1660 m a.m.s.l.) and under the same environmental conditions. At any rate, they are likely to be more representative than the rainfall records for Kathmandu (being the only data available at the time) that were used in the previous analysis

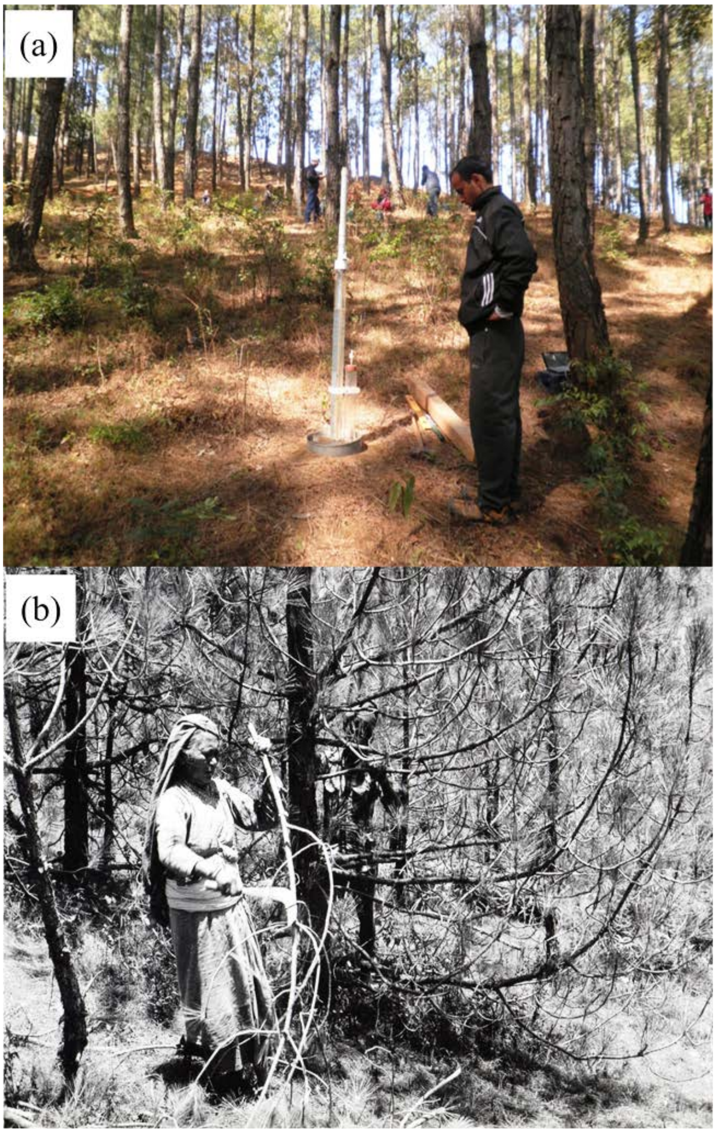

Figure 5.2: Pine reforestation site (Site-3) near Chautara, Central Nepal in: (a) 2011 (Colour), and (b) 1986 (B/W)

by Gilmour et al. (1987). Rainfall was recorded using a tipping-bucket rain gauge (Rain Collector II, Davis Instruments, USA; 0.2 mm per tip) at 5-min intervals. Rainfall events were defined as events totaling at least 5 mm (Negishi et al., 2006) and separated from each other by a dry period of at least 3 h. The maximum 5-min rainfall amounts (I_{5max}, expressed as equivalent hourly rainfall intensity) were determined by

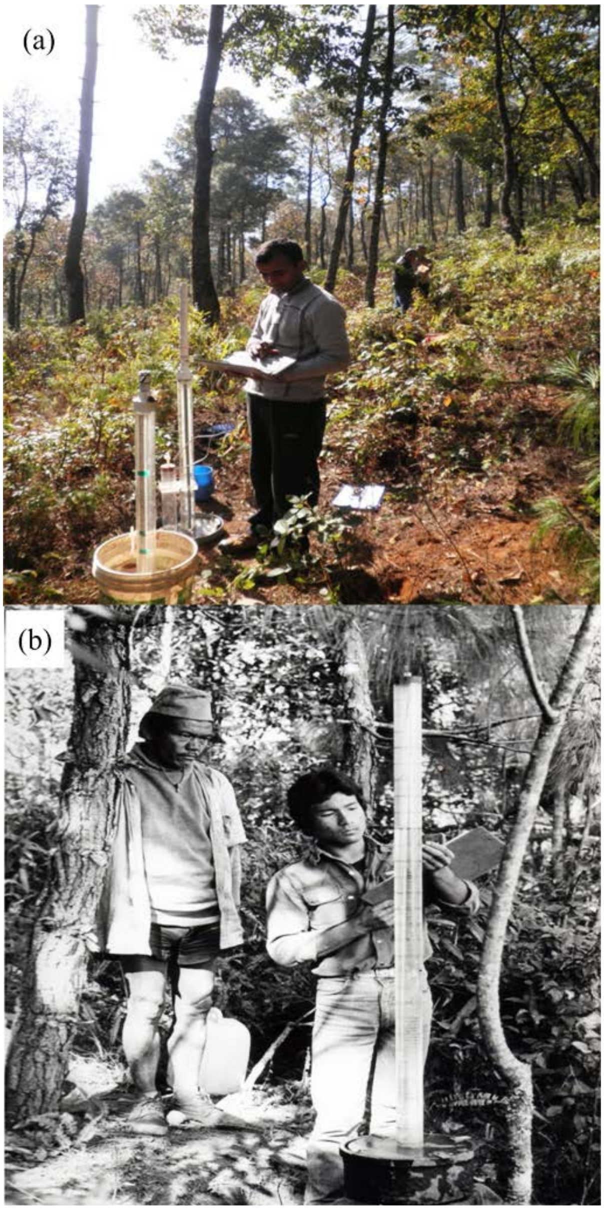

Figure 5.3: Pine reforestation site (Site-4) near Chautara, Central Nepal in: (a) 2011 (Colour), and (b) 1986 (B/W).

calculating the maximum precipitation over the corresponding time interval for each event. In addition, the maximum 10 min (I_{10max}), 15 min (I_{15max}), 30 min (I_{30max}) and 60 min (I_{60max}) rainfall intensities (all expressed as equivalent hourly rainfall intensities) were derived as well.

5.3.5 Additional soil physical and chemical properties

Additional basic soil physical and chemical properties such as dry bulk density, texture and soil organic carbon content (SOC) were determined in order to explore their possible influence on the magnitude of near-surface K_{fs} next to the influence of land use/forest management.

Soil bulk density was measured in each plot at six randomly selected points at a depth of 0–0.15 m on undisturbed soil cores (294 cm^3; Blake and Hartge, 1986). At the same depth and sampling points, soil textural composition was determined using the hydrometer method (Gee and Bauder, 1986). SOC was determined using the dry combustion method (Nelson and Sommers, 1982).

5.3.6 Comparison with the results obtained by Gilmour et al. (1987)

In order to quantify the net changes in field-saturated hydraulic conductivity caused by the multi-decadal use and disturbance of the Chautara forests, the presently obtained log-mean K_{fs} values and inferred dominant hillslope hydrological flow paths were compared with those for 1986 of Gilmour et al. (1987) for the selected soil layers. As the depth ranges used in the current field experiment were not entirely identical with those adopted by Gilmour et al. (1987), only those K_{fs} values representing soil layers at similar depth intervals were selected for comparison. K_{fs} measurements for the following depth intervals were used: (1) Surface; (2) 0.05–0.15 m; and (3) 0.25–0.50 m for comparison with Gilmour's (1) Surface; (2) 0.10–0.20 m; and (3) 0.20–0.50 m, respectively. For the deeper soil layers (i.e., > 0.2 m depth), as previously indicated, Gilmour et al. (1987) used the same instrument as the one used in the present study (i.e., the Talsma CHWP). However, the instrument used for the measurement of surface K_{fs} in the present study (i.e., the disc permeameter) differed from the ring infiltrometer (Talsma, 1987), as used by Gilmour et al. (1987). The implications of this will be discussed later.

5.3.7 Statistical analysis

Due to the non-normality of the K_{fs} data, non-parametric tests were applied for statistical analysis. Differences in K_{fs} between 1987 and 2011 for the selected layers were statistically examined using the Mann-Whitney U-test. Differences were taken to be significant when $p < 0.05$. The surface K_{fs} of all sites are excluded from statistical analysis because of the small size (n = 5–6 in 2011). For the statistical comparison of soil characteristics (bulk density, texture and SOC) between the sites, Kruskal-Wallis test (Kruskal and Wallis, 1952) was used. If the Kruskal-Wallis test indicated a significant difference, the Mann-Whitney U-test with Bonferroni correction was applied to account for the multiple comparisons across sites.

5.4 Results

5.4.1 Field-saturated soil hydraulic conductivity K_{fs}

The descriptive statistics for field-saturated hydraulic conductivity, K_{fs} for the various depth intervals at each site are presented in Table 5.2 and are plotted in Figure 5.5.

As expected, the median surface K_{fs} was lowest for the degraded pasture (Site-1) and highest for the degraded natural forest (Site-5), the median values being 32 and 228 mm h^{-1}, respectively, and so these differ by an order of magnitude (Table 5.2). K_{fs} decreased with depth throughout the profile underneath the natural forest but values in the two pine stands below a depth of 0.05 m did not vary much. K_{fs} values in the reforestation stands were slightly higher at 0.15–0.25 m depth compared to 0.05–0.15 m and attained their lowest values at a depth of 0.25–0.50 m, similar to the situation observed in the natural forest (Figure 5.5). Conversely, K_{fs} in the degraded pasture first increased to a depth of 0.25 m and then started to decrease. For the 0.05–0.15 m depth interval, the most noticeable difference between sites is the median K_{fs} value at Site-3 (mature pine forest) which is markedly lower than in any of the other sites (Table 5.2).

Furthermore, K_{fs} values in the 0.05–0.15 m and 0.15–0.25 m layers differed considerably between sites but less between the two depths at a given site, with no marked difference between the two depths at Site-3, a slight increase in K_{fs} with depth at Site-4, and a slight decrease with depth at Site-5 (Table 5.2 and Figure 5.5). At 0.50 m depth, however, the

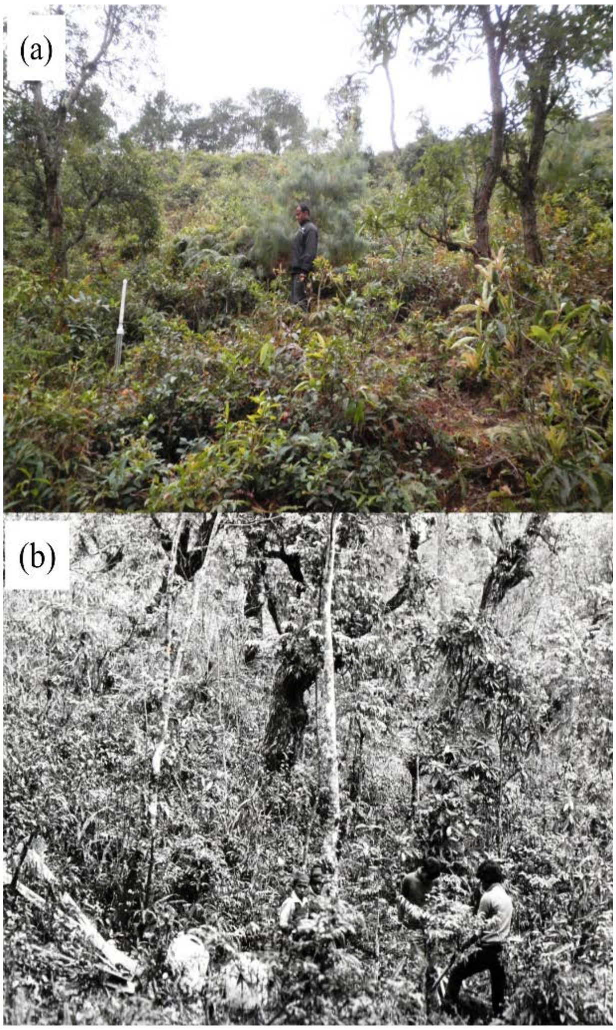

Figure 5.4: Near-natural forest site (Site-5) near Chautara, Central Nepal in: (a) 2011 (Colour), and (b) 1986 (B/W).

differences in K_{fs} between the respective land-cover types became smaller, indicating the limited influence of trampling (cattle grazing and human pressure) on the deeper soil layers. A comparison of the present log-mean K_{fs} values with those of Gilmour et al. (1987) is presented in Table 5.3. The data clearly show that only a small reduction in surface K_{fs} was detected at the degraded grassland site (Site-1) after the additional 25 years of human trampling and cattle grazing, in contrast to the relatively larger reduction (~40 mm h^{-1}) observed at a depth of 0.05–0.15 m. On the other hand, dramatic reductions in K_{fs} under both planted and natural forest were observed following increased forest use and disturbance since 1986. For example, reductions in K_{fs} of as much as ~200 mm h^{-1} were derived both for the surface and the shallow subsurface soil (<0.25 m) at Site-5. Similarly, an even more pronounced reduction at Site-4 (pine forest) of ~430 mm h^{-1} at the surface was observed *versus* ~130 m h^{-1}at shallower depth (< 0.25 m). Reductions at Site-3 (which had lower initial K_{fs} values to start with in 1986, presumably because of its more intensive land-use history prior to planting compared to Site-4; Gilmour et al., 1987) were less pronounced than those for Site-3, viz. ~90 mm h^{-1}, both at the surface and at shallow depth (<0.25 m). Values of K_{fs} at 0.25–0.50 m depth were lower in 2011 at all four sites, although the respective reductions are much smaller than those observed at depths <0.25 m (Table 5.3). Furthermore, K_{fs} for the respective soil layers, 0.05–0.15 m, and 0.25–0.5 m depths at the investigated sites differed significantly (p < 0.05) between the 1987 and 2011 data-sets (Table 5.3).

5.4.2 Rainfall intensities and inferred dominant hillslope runoff pathways

A total of 99 storm events were recorded at Dhulikhel during the summer monsoons of 2010 and 2011. The maximum equivalent hourly intensities for the observed 5-, 10-, 15-, 30-, and 60-min rainfall amounts (I_{5max}–I_{60max}) for all storms were 88.8, 82.8, 62.4, 47.2, and 39.6 mm h^{-1}, respectively. The corresponding median values were 26.4, 19.2, 15.6, 10.6, and 7.6 mm h^{-1}. For each rainfall intensity class, intensities were low for the majority of events (Figure 5.6). However, the higher-intensity storms contributed substantially to the bulk of overall monsoon precipitation totals (Figure 5.6). The median, 75% percentile, 95% percentile and maximum values of I_{5max} and I_{30max} were selected as indices for inferring the dominant hillslope runoff pathways during rainfall following Zimmermann and Elsenbeer (2009) and these have been plotted in conjunction with the results obtained for K_{fs} for the respective land covers in Figure 5.5.

It is evident from Figure 5.5a that for the degraded pasture site the upper quartile of I_{5max} exceeds the entire range of measured surface K_{fs} values and that infiltration-excess overland flow (IOF) is the inferred dominant stormflow pathway. Furthermore, 5-min rainfall totals between the 95th percentile and the maximum value continue to exceed the median K_{fs} for this site at all depths. Thus, vertical percolation will be limited for all

Table 5.2: Descriptive statistics for field-saturated soil hydraulic conductivities K_{fs} (mm h^{-1}) associated with contrasting land-cover types at different depths near Chautara, Central Nepal

	Surface	(0.05 – 0.15 m)	(0.15 – 0.25 m)	(0.25 – 0.50 m)
Degraded Pasture [Site-1]				
Mean	33	48	59	45
Median	32	43	50	43
Standard deviation	5	21	30	22
Minimum	28	14	21	13
Maximum	40	96	146	100
Sample size (*n*)	5	32	32	32
Pine Forest [Site-3]				
Mean	114	31	34	27
Median	83	25	33	21
Standard deviation	73	15	13	37
Minimum	32	11	18	9
Maximum	225	86	75	225
Sample size (*n*)	6	32	32	32
Pine Forest [Site-4]				
Mean	117	71	81	62
Median	94	64	77	55
Standard deviation	90	26	34	33
Minimum	33	29	32	6
Maximum	279	125	193	157
Sample size (*n*)	6	32	32	32
Semi-natural Forest [Site-5]				
Mean	204	147	128	68
Median	228	137	126	66
Standard deviation	109	104	53	24
Minimum	50	29	29	36
Maximum	340	570	214	133
Sample size (*n*)	6	32	32	32

rainfall events with these intensities (Figure 5.5a). However, a noticeable shift from IOF-dominated runoff to unimpeded infiltration occurs when considering the maximum rainfall intensity over 30 min, *i.e.* moving from I_{5max} to I_{30max} (Figure 5.5b). The presently inferred dominant stormflow pathway for the degraded pasture is similar to that suggested by the K_{fs} results obtained in 1986 for this site by Gilmour et al. (1987) (Figure 5.7a,b). This suggests that there has been only limited additional soil degradation at this site over the last 25 years.

Despite the lower topsoil K_{fs} values determined for the two planted forest sites compared to those of the degraded natural forest, the median surface K_{fs} values beneath the pines were still above (or nearly equal to) the most intense rainfalls (Figure 5.5c–e), indicating that IOF in the plantations must still considered a rare phenomenon. However, sub-surface median K_{fs} values at Site-3 between 0.05 m and 0.50 m are so much lower now than in 1986 (Table 5.3), that they are well below the upper quartile of I_{5max} (Figure 5.5c) and the 95th percentile of I_{30max} (Figure 5.5d). Hence, perched water tables at depths as shallow as 0.05–0.25 m are likely to be generated more frequently during rain events at Site-3. Depending on the depth of occurrence of such temporary groundwater tables, the water may run off in the form of shallow, subsurface stormflow (SSF) or even as hillside saturation-excess overland flow (SOF). Similar patterns were derived for Site-4, although in view of the generally higher K_{fs} at all depths compared to Site-3 due to its more benign land-use history (Table 5.2), the development of SSF at Site-4 is expected to be less frequent and SOF all but non-existent (Figure 5.5e).

Overall, when comparing the present situation with that in 1986 for the two reforestation stands (Figure 5.7c – f), a distinct change in hillside runoff pathways at these two sites is indicated. Such circumstances reflect the increased human trampling and cattle grazing pressure during the last 20–25 years. The 0.05–0.15 m soil layer at Site-3 is now more likely to develop a perched water table and thus SSF (occasionally supplemented by SOF) during most maximum 5-min rainfall intensities, whereas the same soil layer was transmissive to all maximum 5-min rainfall intensities 25 years ago (Figure 5.7c). Similarly, perched water tables and thus SSF (but not SOF) are likely to develop more frequently in the 0.05–0.15 and 0.25–0.50 m soil layers at Site-4 for which the historical data suggest they were (much) more permeable 25 years ago and thus capable of accommodating all 5-min rainfalls (Figure 5.7e).

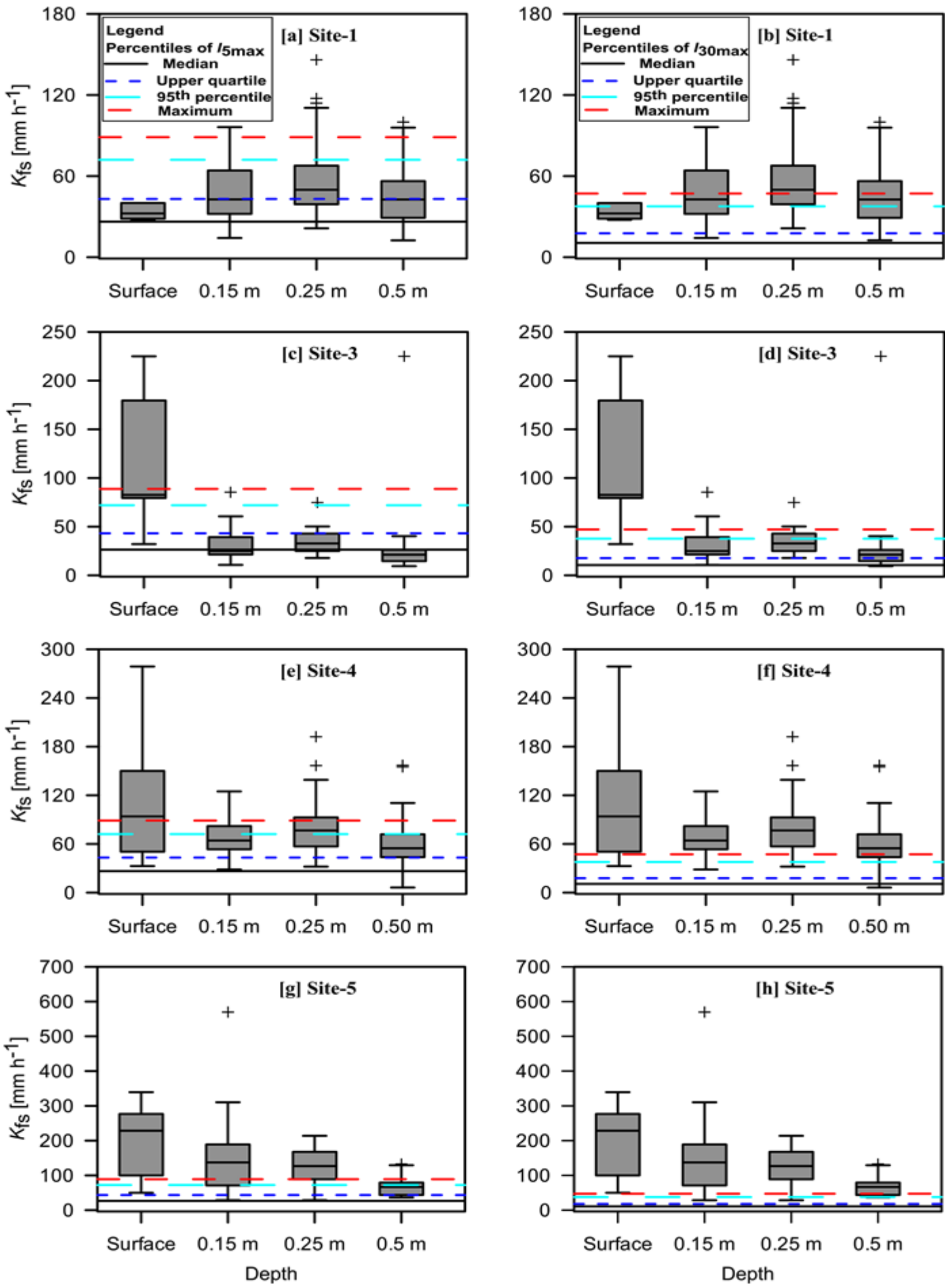

Figure 5.5: Box plots for: (a, b) degraded pasture (Site-1), (c, d) pine forest (Site-3), (e, f) pine forest (Site-4), and (g, h) natural forest (Site-5). The solid horizontal line represents the median rainfall intensity and dashed lines the 75% percentile, 95% percentile and maximum values of I_{5max} (one the left: a, c, d, g) and I_{30max} (on the right: b, d, f, h) rainfall intensities, respectively.

Table 5.3: Comparisons of the presently derived log-mean field-saturated hydraulic conductivities K_{fs} (mm h-1) for contrasting land-cover types near Chautara, Central Nepal, with the corresponding values of Gilmour et al. (1987). Values in parentheses are the measurement depths adopted by Gilmour et al. (1987).

	Log-mean K_{fs} (mm h^{-1})		Difference
	Present study	Gilmour et al. (1987)	
Surface (0 – 0.10 m)			
Degraded pasture (Site-1)	33	39	-
Pine forest (Site-3)	96	183	-
Pine forest (Site-4)	93	524	-
Natural forest (Site-5)	173	370	
0.05 – 0.15 m (0.10– 0.20 m)			
Degraded pasture (Site-1)	44	87	s.
Pine forest (Site-3)	29	117	s.
Pine forest (Site-4)	67	195	s.
Natural forest (Site-5)	130	310	s.
0.25 – 0.50 m (0.20– 0.50 m)			
Degraded pasture (Site-1)	40	63	s.
Pine forest (Site-3)	20	38	s.
Pine forest (Site-4)	54	100	s.
Natural forest (Site-5)	65	96	s.

s. symbolizes “significant”

By contrast, despite being gradually degraded in terms of tree biomass (including the manual removal of all larger trees) over the last 25 years (Figure 5.4), the (semi-)natural forest of Site-5 still showed considerably higher K_{fs} values down to 0.25–0.50 m. Five-minute maximum rainfall intensities did not exceed the median K_{fs} down to 0.15–0.25 m depth (Figure 5.5h). Consequently, the natural forest is still characterized by predominantly vertical percolation down to 0.15–0.25 m for all rainfall intensities. However, a perched water table is likely to develop at 0.25–0.50 m depth during the most intense rainfalls (Figure 5.5h). A comparison of inferred dominant hydrological flow paths based on the new K_{fs} data with those of Gilmour et al. (1987) suggests no significant change in the dominant stormflow pathways as yet in the (semi-) natural forest (Figure 5.7g,h), despite the observed major reductions in surface and subsurface K_{fs} (Table 5.3).

5.5 Discussion

5.5.1 Uncertainty in K_{fs}

Several studies have demonstrated that the use of different techniques for the determination of K_{fs} (for example, soil cores *versus* well-permeametry) may produce significantly different results (e.g., Talsma and Hallam, 1980; Davis et al., 1999; Chappell and Lancaster, 2007). Smearing of the auger hole, and thus pore closure and / or disruption of pore continuity as a result of augering, are reasons commonly given for the underestimation of hydraulic conductivity using well permeametry compared to core-based values (Talsma, 1987; Davis et al., 1999; Chappell and Lancaster, 2007). Because the K_{fs} data for the Chautara sites were collected during the dry season and the internal walls of the augered holes were brushed carefully, it was assumed that the smearing and blocking of soil pores and thus the effect on K_{fs} was minimised.

Values for the sub-soil K_{fs} were determined using similar instruments and sample sizes in the two studies. Therefore, it is expected that the presently obtained results for sub-soil K_{fs} are comparable with those of Gilmour et al. (1987). However, for the surface K_{fs}, the instrument used in 2011 (disc permeameter) differed from that used in 1986 (ring infiltrometer; Talsma, 1969; Dunin, 1976). Despite all care being taken during the 1986 survey, one cannot exclude a small risk of surface soil disturbance during ring insertion which may have had an effect on the results. Nevertheless, in view of the very similar measurement scales of the two instruments (ring diameters of 0.30 and 0.26 m, respectively), the respective measurements should be largely comparable. Some support for this contention comes from the fact that K_{fs} trends found in the 2011 survey for the surface and for the 0.05–0.15 m depth interval are similar to those of Gilmour et al. (1987) at all sites. Hence the K_{fs} measured at the surface may serve as an indicator of the change in condition despite the smaller sample size of the 2011 measurements.

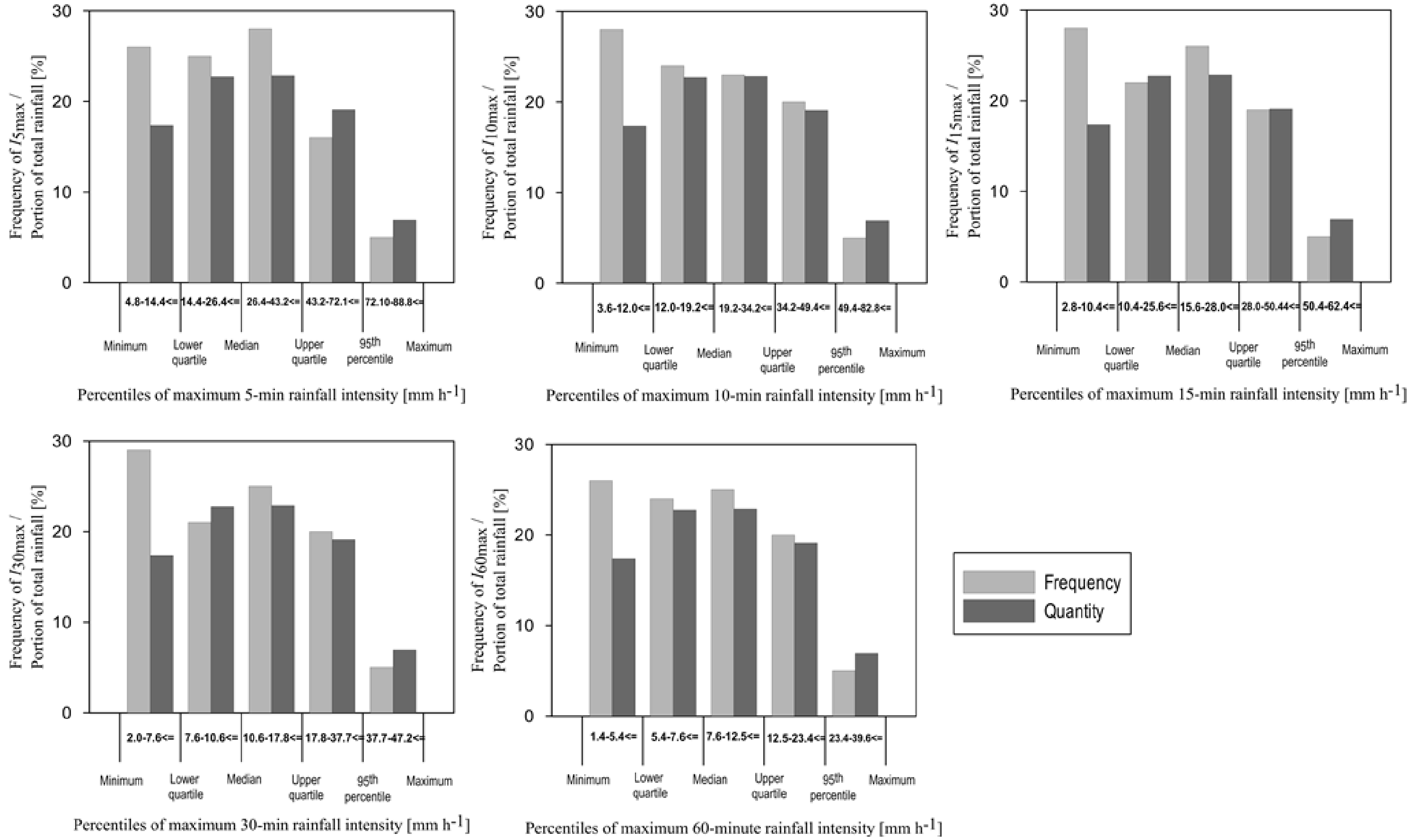

Figure 5.6: Frequency and quantity of maximum 5-, 10-, 15-, 30- and 60-min rainfall intensities as recorded at the Dhulikhel rainfall station, Central Nepal, during the 2010 and 2011 monsoon periods.

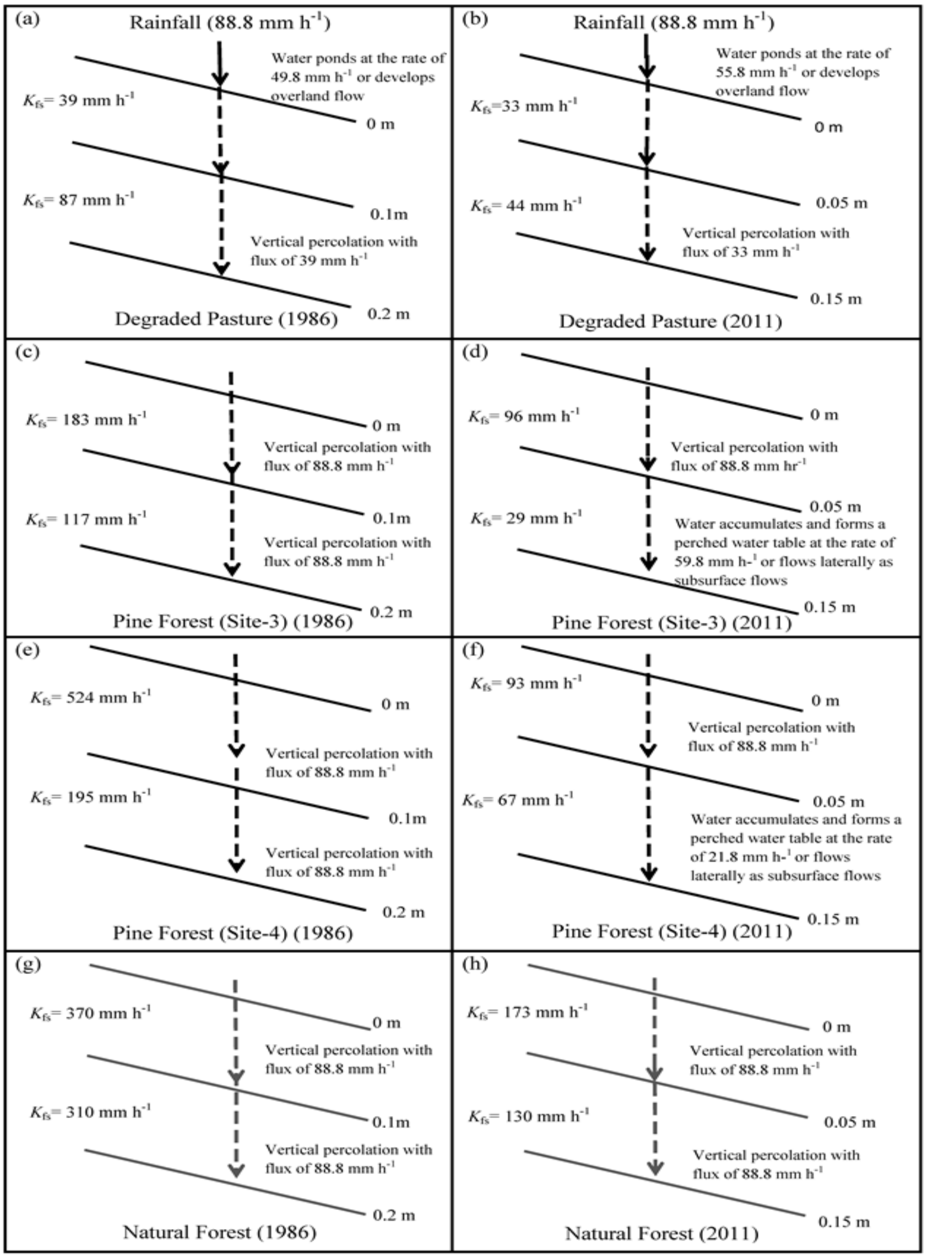

Figure 5.7: Comparaison of the dispersion of rainfall delivered at the maximum 5-min intensity of 88.8 mm h^{-1} through various soil layers contrasting land covers near Chautara, Central Nepal in (left) 1986 and (right) 2011. (a, b) degraded pasture (Site-1), (c, d) pine reforestation (Site-4) and (g, h) natural forest (Site-5). K_{fs} is field-saturated soil hydraulic conductivity. Data for 1986 after Gilmour et al. (1987).

5.5.2 Effects of multi-decadal land use on K_{fs}

5.5.2.1 Degraded pasture

Decreases in K_{fs} at shallow depths associated with the conversion of tropical forest to pasture are well-documented (e.g., Alegre and Cassel, 1996; Tomassella and Hodnett, 1996; Deuchars et al., 1999; Zimmermann et al., 2006; Tobón et al., 2010). Most of the often substantial reductions in K_{fs} appear to have occurred during the first three to five years after forest clearing (Alegre and Cassel, 1996; Martinez and Zinck, 2004) but the impact of continued multi-decadal compaction by grazing animals is less well-known. Gilmour et al. (1987) considered their degraded pasture site to have been in use for more than a century prior to the K_{fs} measurements in 1986. As such, by the time of the present study (2011) the site had experienced ~125 years of intensive grazing (cf. Figure 5.1). In view of the similarity in the log-mean values of surface K_{fs} determined in 1986 (39 mm h^{-1}) and 2011 (33 mm h^{-1}) it would seem as though the continued intensive use of the land over the last 25 years had limited additional impact on surface K_{fs}. The presently found values for surface K_{fs} are comparable to the 41 mm h^{-1} reported by Alegre and Cassel (1996) for an intensively grazed 5-year-old pasture in Amazonian Perú that had been created by mechanised forest clearing. Pertinently, they are also similar to the 15–20 mm h^{-1} and 40–45 mm h^{-1} observed in various Malaysian rain forests underlain by similar soils as the ones found in the Chautara area and subjected to mechanized harvesting using heavy tracked machinery (Kamaruzaman, 1991; Van der Plas and Bruijnzeel, 1993). Such similarities illustrate the advanced stage of surface soil compaction represented by the Chautara pasture site compared to, for example, the much less intensively used pastures in Amazonia investigated by Zimmermann et al. (2006) (median surface K_{fs} ~113 mm h^{-1}) or Tomassella and Hodnett(1996) (66 mm h^{-1}). Such major changes in soil surface K_{fs} after forest conversion and subsequent land degradation are bound to result in important changes in the dominant stormflow pathways over and within hillslopes. Using the data of Gilmour et al. (1987) as a baseline, the chief change in K_{fs} at the degraded pasture site was a nearly 50% reduction at 0.05–0.15 m depth. The relatively large and statistically significant reduction ($p < 0.05$) at 0.05–0.15 m suggests the continued trampling and cattle grazing to have a greater influence on the hydrological behaviour of the shallow sub-soil than at the soil surface where conditions have possibly reached an extreme stage already as discussed earlier. Infiltration-excess overland flow (IOF) is the inferred dominant stormflow pathway as also suggested previously by Gilmour et al. (1987) for this site. Indeed, very high IOF has been reported for similarly degraded bare soils in the nearby Likhu Khola area (57–64% of rainfall; Gerrard and Gardner, 2002), degraded grassland in West Central Nepal (26–47% of rainfall; Impat, 1981) and annually burned grassland in the Philippines (69%; Chandler and Walter, 1998). However, in view of the significant decreases in K_{fs} below the surface (Table 5.3), the generation of lateral SSF at increasingly shallow depths is likely to become more frequent too as soil degradation progresses.

5.5.2.2 Reforestation

Increases in K_{fs} at shallow depths have been reported by a number of studies in connection with improved soil physical characteristics during natural forest regrowth on former degraded pasture lands in Costa Rica (Deuchars et al., 1999), Amazonia (Godsey and Elsenbeer, 2002; Zimmermann et al., 2006), Ecuador (Zimmermann and Elsenbeer, 2009) and Panama (Hassler et al., 2011). However, comparable studies of changes in K_{fs} in planted forests are still exceedingly scarce (Scott et al., 2005; Ilstedt et al., 2007) and usually comprise a single measurement in time (i.e., at a given plantation age; Mapa, 1995; Zimmermann et al., 2006; Bonell et al., 2010) as opposed to documenting the gradual changes in K_{fs} over time since plantation establishment (cf. Gilmour et al., 1987). As such, the present comparison of K_{fs} values at two points in time separated by 25 years of gradually intensified and changing forest use is a first.

As can be seen from the comparative photographs taken in 1986 and 2011 (Figures 5.2 and 5.3), the surface conditions in the two reforestation stands (Site-3 and Site-4) have undergone dramatic changes over the last 25 years, and this is most likely the main reason for the observed radical reductions in surface- and near-surface K_{fs} (Table 5.3) and the associated change in inferred dominant hillslope stormflow pathways. Although values of surface K_{fs} in the two pine stands are still sufficiently high to accommodate maximum observed rainfall intensities (Figure 5.5), the significant reductions ($p < 0.05$) in K_{fs} observed at shallow depth (0.05–0.15 m) at Site-3 in particular (down to 29 mm h^{-1}; Table 5.3) suggests the occurrence of near-surface SSF will become much more frequent, even to the extent that widespread SOF may occur during prolonged events of sufficiently high intensity (cf. Bonell et al., 1983). SOF is less likely to be generated at Site-4 where K_{fs} at 0.05 m depth is still sufficiently high at 67 mm h^{-1} to accommodate all but the most intensive 5-min rains (Table 5.3 and Figure 5.5). However, the fact that K_{fs} was reduced about three-fold within less than three decades, does not bode well for the future unless protective measures can be taken (see also below). The relatively larger decadal reduction in K_{fs} at 0.05–0.15 m depth at Site-3 compared to that derived for Site-4 (Table 5.3) indicates the much more substantial and widespread compaction of the surface soil by human trampling and cattle grazing at Site-3 relative to Site-4, as also evidenced by the much greater bulk density of the topsoil at Site-3 (Table 5.1). Furthermore, in contrast to Site-3, the forest of Site-4 still had a reasonably developed understory whereas grazing was largely absent (cf. Figures 5.2 and 5.3). Forest disturbances such as the removal of understory vegetation for fodder or fuelwood, the regular collection of litter from the forest floor, and grazing by cattle all have an adverse effect on soil faunal activity and biomass (Lal, 1988; Ding et al., 1992) and therefore on the incorporation of organic matter (cf. Table 5.1), with adverse knock-on effects on soil aggregate stability, macro-porosity and pore connectivity (Lal, 1987; Hairiah et al., 2006). The removal of leaf litter is particularly harmful in that it also leads to greater exposure of the bared soil to the erosive force of drip from the canopy which is typically much greater than that of rainfall in the open, particularly in broad-leaved forests (Wiersum, 1985; Brandt, 1988; Hall and Calder, 1993). Further, infiltration capacities of soils from which litter has been removed repeatedly are generally much reduced whereas IOF production tends to increase dramatically (Tsukamoto, 1975; Wiersum, 1985; Zhou et al., 2002; cf. Gerrard and Gardner, 2002; Hairiah et al., 2006).

The present findings highlight the need for some form of protection of reforested areas to enable the forest soils to realise the enhanced infiltration / percolation benefits envisaged at the time of reforestation. Otherwise, any benefits from replanted forests and the added incorporation of organic matter into the soil (cf. Gilmour et al., 1987) are negated by the sustained biomass harvesting and surface deterioration, as shown by the dramatic decreases in near-surface K_{fs} over the last 25 years inferred by the present study. The human dimension is commonly less emphasised in the tropical forest hydrological literature (Tomich et al., 2004a; Bonell and Bruijnzeel, 2005) or in studies of the hydrological impacts of forestation (Scott et al., 2005; Ilstedt et al., 2007). However, in view of the intensive use of reforested lands in the study area (Figures 5.2 and 5.3) as well as in other densely populated uplands in Asia (e.g., Singh et al., 1984; Wiersum, 1985; Ding et al., 1992) it is clear that consideration needs to be given to striking a balance between utilization of tree and forest products and overall landscape rehabilitation. This is a challenge for local communities who need to manage their common land to optimise the provision of goods and services, and it requires a sophistication and maturity of local management that is not yet present in Nepal. Criticism is often voiced about the use of pines in reforestation programmes, as most farmers prefer products from broad-leaf species rather than pines for inputs into their farming systems (Gilmour and Fisher, 1991; cf. Singh and Singh, 1992). However, the ecological reality is that most broad-leaf species do not survive when planted on the degraded sites that are generally available for reforestation in the Nepalese Middle Mountains (Applegate and Gilmour, 1987; Gilmour, et al., 1990). Pines, as pioneer species, can survive and grow in such harsh conditions, which can then provide the site amelioration needed to allow broad-leaf species to grow as an understory (Singh and Singh, 1992) that can subsequently be managed to dominate the forest if the local Community Forest User Groups decide to manage their forests in this direction (Gilmour et al., 1990; Gilmour and Fisher, 1991). Conclusions of watershed-wide impacts of reforestation need to be drawn in the light of watershed-wide changes in land use over time, and in the case of the Nepalese Middle Mountains these changes have been in several directions. As mentioned in earlier sections, there has been a substantial increase in tree and forest cover on both private and common land across most of the Middle Mountain Zone during the past several decades as well as a substantial decrease in open-range grazing. A more recent phenomenon is the large scale and on-going abandonment of agricultural land – both irrigated and rain-fed terraces – as a direct result of the high levels of out-migration, particularly of the active male labour force, to urban areas and overseas (Kollmair and Hoermann, 2011; Gilmour and Shah, 2012). The extent of this agricultural abandonment is not yet clear, but indications are that as much as one-third of all agricultural land may have been abandoned in many districts (Paudel, et al., 2012). Some of this abandoned land is regenerating naturally to trees, further adding to the increasing tree cover across the region. The overall impact of these changes on the hydrological functioning of catchments in the Middle Mountains is as yet far from clear (see also Section 5.5.2.4 below). What is clear is that substantial changes have taken place in recent decades and changes are still underway. The situation is extremely dynamic and multi-faceted.

5.5.2.3 Gradually degrading natural forest

Even though the transformation between 1986 and 2011 of a nearly undisturbed natural forest to a shrubland with only scattered remnant trees (Figure 5.4) was demonstrated to greatly influence surface and subsurface K_{fs} (Table 5.3), vertical percolation still remained the dominant rainfall redistribution pathway at Site-5, as it was in 1986 (Gilmour et al., 1987; Figure 5.7g-h). Whilst the main tree canopy had largely disappeared, a dense shrub layer, as well as younger trees and an herbaceous layer remained in place (Figure 5.4). In addition, a thick litter layer was maintained due to the exclusion of cattle grazing and litter collection. The present results illustrate the overriding importance of a well-developed litter layer for soil surface protection compared to the role of the overstory (Wiersum, 1985). They further illustrate the important point made by Smiet (1987) that the margins for forest management with regard to soil protection are much broader than those associated with non-forest types of land use. The degraded natural forest is still capable of fulfilling its soil protective role despite having lost much of its main canopy whereas the severe disruption of the litter layer in the pine forest of Site-3 shows that here the margins have been exceeded (cf. Smiet, 1987). Nevertheless, the observed trend of significantly decreasing surface- (by 53% since 1986) and near-surface (by 58%) values of K_{fs} at Site-5 is alarming. As demonstrated elsewhere in the region (Patnaik and Virdi, 1962; Gerrard and Gardner, 2002; Bonell et al., 2010), the introduction of any extra pressure to this site in the form of cattle grazing and litter collection over the next few decades is likely to greatly reduce the surface soil's infiltration capacity and affect the dominant stormflow pathway. For instance, in the nearby Likhu Khola area, Gerrard and Gardner (2002) reported IOF to be negligible in largely undisturbed forest whereas IOF was rampant in degraded and grazed *Shorea* forest (see also various examples from other parts of the Himalaya discussed by Bruijnzeel and Bremmer, 1989). These results again underscore the need for some form of protection of the remaining natural forests of the study area if they are to maintain their original hydrological functions.

5.5.2 Implication of diminishing soil hydraulic conductivity for dry season flows

The adequate recharge of groundwater reserves during the monsoon period (June–September) is of critical importance for maintaining streamflow during the extended dry season in the Middle Mountains of Central Nepal. Reforestation of degraded pasture lands is often recommended as a means of restoring stable seasonal streamflow due to the enhanced infiltration afforded by improved soil porosity, structure and water retention capacity after tree planting (e.g., Bartarya, 1989; Negi et al., 1998). However, such a notion is conditional on the gain in groundwater recharge through increased rain infiltration being in excess of the increased plant water uptake after reforestation (Bruijnzeel, 2004; Chandler, 2006). Therefore, any reductions in site water losses via overland flow (and thus potential increases in soil moisture replenishment) will play a crucial role when trying to boost diminished dry season flows. In view of the very high runoff losses incurred via IOF reported for severely degraded grasslands in the region (Impat, 1981; Gerrard and Gardner, 2002), there is much to be gained by improving infiltration by reforestation of such land in the study area.

Even though rainfall intensities were relatively low for the majority of rainfall events during the investigation period (Figure 5.6), events having relatively high intensities (i.e., ranging from the median value until the absolute maximum) contributed nearly half of the total rainfall received. Furthermore, the presently observed radical and significant reductions in near-surface values of K_{fs} over the last 25 years for the two planted forests suggest that there may well be an equally radical shift in the dominant stormflow pathway at the hillslope scale over the next decades if these forests are not better managed. The chief inferred change in hillslope hydrological response to rainfall associated with continued forest degradation is that from subsurface stormflow-dominated flow (SSF) to a much more frequent occurrence of IOF. If continued long enough to reach a truly degraded state (as already observed at the pasture site), the extra volumes of water lost as IOF might well begin to critically reduce groundwater recharge during the rainy season (and thus dry season flows). Furthermore, the gain provided through enhanced infiltration after reforesting degraded pasture land is most likely to be (much) less if the soils are not allowed to recover. In addition, such diminished gains in infiltrated water will be outweighed more rapidly by the increased evaporative demand of the planted forest relative to that of the pasture (cf. Scott et al., 2005). Such a scenario of gradually diminishing infiltration opportunities through continued exploitation of the forest may, in turn, cause already diminished dry-season flows to be reduced even further (cf. Bruijnzeel, 1989). Thus, continued human usage and cattle grazing in the planted forests will most likely lead to a continued decline in K_{fs} which, in turn, will further contribute towards a reduction in dry season flows in the future. Madduma Bandara (1997) has described such a case in Sri Lanka, where over a period of 35 years, the gradual land degradation associated with small-holder cropping on former well-managed tea estates produced a reduction in dry season flows of ca. 33% in the Mahaweli River Basin whereas wet season flows increased by 45–50%. Assuming an estimated increase in annual water use of (mature) forest plantations over (non-degraded) grassland of 250–300 mm y^{-1} (based on a mean annual precipitation figure of 1500 mm and the generalised Zhang-curves for forest and grassland water use; Zhang et al., 2001), it follows that IOF under reforestation in the study area would need to be reduced (relative to IOF generated on degraded pasture) by an amount equal to at least 16–20% of the annual rainfall. Based on the presently determined values of surface and near-surface K_{fs} for the two pine stands *vis á vis* the degraded pasture such a situation still prevailed in 2011. However, in view of the observed decline in topsoil properties in the study area and in degraded forests elsewhere in the region (Gerrard and Gardner, 2002) it would appear to be only a matter of time before the balance will swing.

The present findings do point to the need for balancing the socio-economic and hydrological functions of forests – both planted and natural – in densely populated tropical uplands. Potentially promising developments in this regard include a greater emphasis on agro-forests that contain a variety of tree and crop species serving a range of uses as opposed to the mono-specific character of most planted forests (Hairiah and Van Noordwijk, 2005; Gilmour and Shah, 2012); and, although still at an initial stage, various approaches aiming at rewarding the upland poor for environmental services rendered (see the following for details: Van Noordwijk et al., 2001; Tomich et al., 2004b; Murdiyarso, 2005; Swallow et al., 2009; Van Noordwijk and Leimona, 2010).

5.6 Conclusions

The results of the present study concerning the effects of multi-decadal forest use on soil physical characteristics and inferred hillslope hydrological functioning in the Middle Mountains of Central Nepal can be summarised as follows:

1. Abrupt and significant decreases in field-saturated hydraulic conductivity (K_{fs}) were observed within the upper 0.25 m of the soil profiles under both natural and planted forest sites after 25 years of various forms of human activity. This reflects a loss of macro-porosity due to a reduced incorporation of soil organic matter after repeated collections of litter from the forest floor supplemented by continuous cattle grazing. Contrary to the forest sites, the studied degraded pasture site showed little change in surface K_{fs}, suggesting surface degradation had reached a ceiling. However, K_{fs} of the near-surface layer was affected negatively by the continued trampling and grazing.

2. Despite several decades of human activity, the natural forest still retained a “memory” at all investigated depths from the previous less disturbed state as recorded in 1986. Therefore, vertical percolation is still the dominant hydrological pathway. However, the continued exploitation of the forest, and the introduction of litter collection and animal grazing in particular, may change this unless some form of forest protection is realised in the future.

3. Rainfall events with relatively high intensities (i.e., above the median value) contributed nearly half of the total rainfall received during the two monsoon seasons covered by the investigation. Based on these rainfall characteristics, there are indications in the intensively used planted forests of a change from the formerly dominant subsurface hillslope stormflow pathway to more frequent occurrence of infiltration-excess overland flow. If the ongoing process of soil degradation continues over the next few decades it will most probably result in a further significant reduction in rainfall infiltration and, ultimately impaired groundwater recharge during the rainy season. Such circumstances are likely to impact dry-season flows adversely, also because the extra water use of the planted forest (relative to the degraded pasture that it replaces) is likely to be in excess of the (diminishing) gains through infiltration if forest degradation continues.

4. This study has demonstrated that management of community forests in Nepal's Middle Mountains needs to include a consideration of hydrological functioning. Simply planting trees on degraded land is not sufficient in itself to restore the hydrological functions of degraded catchment areas. Attention needs to be given to on-going management of the areas to balance product usage with hydrological functioning. Of specific importance are: (i) balancing utilization (particularly the removal of litter, understory material and cattle grazing) with retention of organic matter and its incorporation into the surface soil, and (ii) limiting activities such as animal grazing that can lead to compaction of the soil surface. This represents an added dimension to the already complex process of community forest management and will pose a considerable challenge to both local communities and the Government.

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Chapter 6

Negative trade-off between changes in vegetation water use and soil infiltration recovery after reforesting degraded pasture land in the Nepalese Lesser Himalaya[1]

Abstract. This work investigates the trade-off between increases in vegetation water use and rain water infiltration afforded by soil improvement after reforesting severely degraded grassland in the Lesser Himalaya of Central Nepal. The hillslope hydrological functioning (surface- and sub-soil hydraulic conductivities and overland flow generation) and the evapotranspiration (rainfall interception and transpiration) of the following contrasting vegetation types were quantified and examined in detail: (i) a nearly undisturbed natural broad-leaved forest; (ii) a mature, intensively-used pine plantation; and (iii) a highly degraded pasture. Planting pines increased vegetation water use relative to the pasture and natural forest situation by 355 and 55 mm yr^{-1}, respectively. On balance, the limited amount of extra infiltration afforded by the pine plantation relative to the pasture (only 90 mm yr^{-1} due to continued soil degradation associated with regular harvesting of litter and understory vegetation in the plantation) proved insufficient to compensate the higher water use of the pines. As such, observed declines in dry season flows in the study area are thought to reflect the higher water use of the pines although the effect could be moderated by better forest and soil management promoting infiltration. In contrast, a comparison of the water use of the natural forest and degraded pasture suggests that replacing the latter by (mature) broad-leaved forest would (ultimately) have a near-neutral effect on dry season flows as the approximate gains in infiltration and evaporative losses were very similar (ca. 300 mm yr^{-1} each). The results of the present study underscore the need for proper forest management for optimum hydrological functioning as well as the importance of protecting the remaining natural forests in the region.

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[1]This chapter is based on: **Ghimire CP**, Bruijnzeel LA, Lubczynski MW, Bonell M (2014) Negative trade-off between changes in vegetation water use and soil infiltration recovery after reforesting degraded pasture land in the Nepalese Lesser Himalaya. *Hydrology and Earth System Science Discussion*, 11, 3437–3479.

6.1 Introduction

The once widely held belief that the presence of a good forest cover invariably ensures a steady flow of water during prolonged rainless spells due to the forest 'sponge' effect in which wet season rainfall is absorbed and stored for subsequent gradual release during the dry season came under severe scrutiny in the early 1980s. Bosch and Hewlett (1982) reviewed the results from nearly one hundred paired catchment experiments around the globe and concluded that 'no experiments in deliberately reducing vegetation cover caused reductions in water yield, nor have any deliberate increases in cover caused increases in yield'. As such, the removal of a dense forest cover was seen to lead to higher streamflow totals, and reforestation of open lands to a decline in overall streamflow. These initial results have been confirmed by several subsequent reviews, both for the (warm-) temperate zone (Stednick, 1996; Brown et al., 2005; Jackson et al., 2005) and the humid tropics (Bruijnzeel, 1990; Grip et al., 2005; Scott et al., 2005). The fact that the bulk of the change in streamflow associated with such experiments was observed during conditions of baseflow (Bosch and Hewlett, 1982; Bruijnzeel, 1989; Farley et al., 2005) at first sight contradicted the reality of the forest sponge concept, and its very existence became questioned (Hamilton and King, 1982; Calder, 2005). Indeed, since the early reviews of Bosch and Hewlett (1982) and Hamilton and King (1982), many have emphasised the more 'negative' aspects of forests, such as their higher water use or inability to prevent extreme flooding (e.g. Calder, 2005; FAO-CIFOR, 2005; Jackson et al., 2005; Kaimowitz, 2005) rather than focus on the positive functions of a good forest cover, including the marked reduction of surface erosion and shallow landslip occurrence (Sidle et al., 2006), improved water quality (Bruijnzeel, 2004; Wohl et al., 2012), moderation of all but the largest peak flows (Roa-García et al., 2011; Ogden et al., 2013) or carbon sequestration (Farley et al., 2005; Malmer et al., 2010).

At the same time, there is ample evidence of severe and widespread soil degradation after tropical forest conversion to unsustainable forms of land use (Oldeman et al., 1991; cf. Bai et al., 2008). This is often accompanied by strongly increased stormflow volumes during times of rainfall (Bruijnzeel and Bremmer, 1989; Fritsch, 1993; Chandler and Walter, 1998; Zhou et al., 2002) and shortages of water during extended dry periods (Bartarya, 1989; Madduma Bandara, 1997; Bruijnzeel, 2004;

Tiwari et al., 2011). This effectively reflects the loss of the former 'forest sponge' (cf. Roa-García et al., 2011; Krishnaswamy et al., 2012; Ogden et al., 2013) through critically reduced replenishment of soil water and groundwater reserves due to lost surface infiltration opportunities, despite the fact that the lower water use of the post-forest vegetation should have produced higher streamflows throughout the year (Bruijnzeel, 1986, 1989).

Reforestation of degraded land in the tropics is often conducted in the expectation that disturbed streamflow regimes (commonly referred to as the 'too little – too much syndrome': Bartarya, 1989; Schreier et al., 2006) will be restored by the increased rainfall absorption afforded by soil improvement after tree planting (Scott et al., 20005; cf. Ilstedt et al., 2007). At the same time, the water use of fast-growing tree plantations tends to be (much) higher than that of the degraded vegetation they typically replace – particularly where the trees have access to groundwater (Calder, 1992; Kallarackal and Somen, 1997). Furthermore, major improvements in the infiltration capacity of severely degraded soils after tree planting may easily take several decades of undisturbed forest development to fully materialise (Gilmour et al., 1987; Scott et al., 2005; Bonell et al., 2010). As such, reforesting degraded pasture or shrub land may well cause already diminished dry season flows to become reduced even further, depending on the net balance between increases in soil water reserves afforded by improved infiltration versus decreases caused by the higher plant water uptake (the so-called 'infiltration trade-off' hypothesis; Bruijnzeel, 1986; Bruijnzeel, 1989).

Although the overwhelming majority of paired catchment experiments have shown major decreases in baseflow volumes after the establishment of a tree cover on non-forested land (Farley et al., 2005) this does not disprove the possibility of enhanced dry season flows after reforestation. As pointed out by Bruijnzeel (2004) and Malmer et al. (2010), only three out of the 26 paired catchment studies of the hydrological impacts of reforestation reviewed by Jackson et al. (2005) and Farley et al. (2005) were located in the tropics (where soil degradation tends to be more acute; Oldeman et al., 1991) whereas none of the experiments represented degraded soil conditions. In other words, no positive effects of reforestation on soil water intake capacity could become manifest and the observed reductions in water yield simply reflected the higher water use of the trees compared to the grasses and scrubs they replaced (cf. Trimble et al., 1987; Waterloo et al., 1999; Scott and Prinsloo, 2008).

Nevertheless, although direct evidence for the 'infiltration trade-off' hypothesis seemed to be lacking until recently (see below), several hillslope-scale or small-basin studies of the hydrological impacts of reforesting severely degraded land (discussed in some detail by Scott et al., 2005) observed reductions in stormflow volumes that were likely to exceed the estimated corresponding increases in vegetation water use (cf. Zhang et al., 2004; Sun et al., 2006). Unfortunately, as the catchments involved in these experiments were either too small or leaky to sustain perennial streamflow, the expected net positive effect of the forestation on dry season flows could not be ascertained in these cases (Scott et al., 2005; Chandler, 2006). However, recently published concurrent reductions in stormflow response to rainfall, and positive trends in baseflow over time since reforesting severely degraded land in South China (Zhou et al., 2010) and South Korea (Choi and Kim, 2013), or along a forest degradation – recovery gradient in south-west India (Krishnaswamy et al., 2012, 2013) suggest a net positive outcome of the infiltration trade-off mechanism is possible under such conditions.

The Middle Mountain Zone of the Nepalese and Indian Himalayas provides another case in point regarding the need to restore diminished dry season flows. The rivers originating in this densely populated part of the mountain range (Singh et al. 1984; Hrabovsky and Miyan, 1987) are mostly rain-fed and thus do not benefit from increased water yields from melting glaciers under a changing climate scenario (Bookhagen and Burbank, 2010; Andermann et al., 2012; Immerzeel et al., 2013). Land surface degradation in the zone has often progressed to a point where rainfall infiltration has become seriously impaired and overland flow is rampant (Bruijnzeel and Bremmer, 1989; Gerrard and Gardner, 2002; cf. Ghimire et al., 2013a), with reduced dry season flows as a sad result (Bartarya, 1989; Tiwari et al., 2011; Tyagi et al., 2013). Partly in response to the latter, some 23000 ha of severely degraded pastures and shrublands in the Middle Mountain Zone of Central Nepal were planted to fast-growing coniferous tree species (mainly *Pinus roxburghii* and *P. patula*) between 1980 and 2000 (District Forest Offices, Kabhre and Sindhupalchok, unpublished data, 2010). However, villagers and farmers in Central Nepal have expressed serious concerns about diminishing dry season flows following the large-scale planting of the pines (República, 2012). Such considerations assume added importance under the strongly seasonal climatic conditions prevailing in the Middle Mountain Zone where ~80% of the annual rainfall is delivered during the main monsoon (June–September; Merz, 2004; Bookhagen and Burbank, 2010) and where water during the dry season is already at a premium (Merz et al., 2003).

There is a dearth of sound experiments in the Himalayan region to ascertain the hydrological effects of land use change (reviewed by Bruijnzeel and Bremmer, 1989; Negi, 2002) and indeed for the (sub)tropics in general with respect to the extent to which reforestation of degraded land can improve or even restore diminished dry season flows (Scott et al., 2005; Zhou et al., 2010; Krishnaswamy et al., 2013; Choi and Kim, 2013). In view of the difficulty of identifying catchments with a single dominant land cover type (e.g. forest or grassland) and which are sufficiently large to support perennial flows (required for the evaluation of changes in baseflows during the extended dry season) in this rugged and spatially highly variable terrain (Maharjan et al., 1991; Merz, 2004; Bookhagen and Burbank, 2006), the present study opted for the 'space for time substitution approach' in which experimental plots with contrasting land-cover types were studied in terms of their hydrological processes and taking the 'infiltration trade-off' hypothesis as a starting point. Thus, the primary objective of the present work was to investigate the trade-offs between the changes in water use going from a severely degraded pasture to a mature coniferous tree plantation or well-developed broad-leaved forest on the one hand, and the concurrent increases in soil water reserves afforded by improved rainfall infiltration after plantation or forest maturation on the other in a typically Middle Mountain Zone setting in Central Nepal. Total vegetation water use (evapotranspiration, ET), overland flow production and field-saturated soil hydraulic conductivity profiles with depth were determined in a little disturbed natural broad-leaved forest, a highly degraded pasture, and a mature planted pine forest, all situated near Dhulikhel. The resulting data were used to address the central question: 'Can successful reforestation of severely degraded hillslopes in the heavily populated Nepalese Middle Mountain Zone restore diminished dry season flows?'

6.2 Methods

6.2.1 Site descriptions

The Middle Mountain zone or Lesser Himalaya, situated between ~800 and ~2400 m above mean sea level (a.m.s.l.) and occupying about 30% of Nepal, is home to ca. 45% of the country's population (based on the 2011 population census; http://cbs.gov.np/). The zone is characterised by a complex geology which has resulted in equally complex topography, soil and vegetation patterns (Dobremez, 1976). The geology of the Central Nepalese Middle Mountains consists chiefly of phyllites, schists and quartzites (Stocklin and Bhattarai, 1977). Depending on elevation the climate is humid subtropical to warm-temperate and strongly seasonal, with most of the rain falling between June and September. Rainfall varies with elevation and exposure to the prevailing monsoonal air masses (Merz, 2004; cf. Bookhagen and Burbank, 2006). At higher elevations (>2000 m a.m.s.l.) a largely evergreen mixed broad-leaved forest dominated by various chestnuts and oaks (*Castanopsis* spp., *Quercus* spp.) and *Schima wallichii* occurs, with occasional *Rhododendron arboreum* above ca. 1500 m a.m.s.l. Due to the prevailing population pressure, much of this species-rich forest has disappeared (<10% remaining), except on slopes that are too steep for agricultural activity (Dobremez, 1976; Merz, 2004).

The present study was conducted in the Jikhu Khola Catchment (JKC). The 111.4 km^2 JKC ($27^0 35'$–$27^0 41'$N; $85^0 32'$–$85^0 41'$E) is situated approximately 45 km east of Kathmandu in the Kabhre District between 796 and 2019 m a.m.s.l (Figure 6.1). The general aspect of the catchment is southeast and the topography ranges from flat valleys to steep upland slopes (Maharjan, 1991). The geology consists of sedimentary rocks affected by various degrees of metamorphism and includes phyllites, quartzites, and various schists (Nakarmi, 2000). In general, soils in the upper half of the JKC are Cambisols (FAO, 2007) of loamy texture and moderately well to rapidly drained (Maharjan, 1991).The climate of the JKC is largely humid subtropical, grading to warm-temperate around 2000 m a.m.s.l. Mean (±SD) annual rainfall measured at mid-elevation 1560 m a.m.s.l.) for the period 1993–1998 was 1487 (±155) mm (Merz, 2004). The main seasons are the monsoon (June–September), the post-monsoon (October–November), winter (December–February), and the pre-monsoon (March–May). The rainy season (Monsoon) begins early

June and ends by late September. During the monsoon about 80% of the total annual precipitation is delivered. In general, July is the wettest month with about 27% of the annual rainfall. The driest months are November to February, each accounting for about 1% of the annual rainfall (Merz, 2004). Average monthly temperatures as measured at 1560 m a.m.s.l. range from 7.7 ^{o}C in January to 22.6 ^{o}C in June while the average monthly relative humidity varies from 55% in March to 95% in September. Strong winds are common during thunderstorms before the onset of the main monsoon, but these are momentary and average monthly wind speeds are always less than 2 m s^{-1} and show only slight seasonal variation. Annual reference evaporation ET_0 [Allen et al., 1998] for the period 1993–2000, was 1170 mm [Merz, 2004]. Vegetation cover in the catchment consists of ~30% forest (both natural and planted), 7% shrubland and 6% grassland, with the remaining 57% largely under agriculture (Merz, 2004). The JKC was subjected to active reforestation until 2004 as part of the Nepal-Australia Forestry Project.

We measured vegetation water use (wet- and dry-canopy evaporation), overland flow and saturated soil hydraulic conductivity (K_{fs}) in natural forest, degraded pasture and planted forest. The land use at the respective research sites can be characterised as follows:

Natural Forest: This site (elevation 1500 m a.m.s.l., northwest exposure, average slope angle 24^{o}) consisted of dense, largely evergreen forest facing little anthropogenic pressure. The 14.0 ± 2.2 m high forest had a well-developed understory and litter layer. Tree density as measured in May 2011 was 1869 trees ha^{-1}, whereas the average diameter at breast height (DBH) for trees with DBH ≥ 5 cm was 13.6 ± 4.4 cm and the corresponding basal area 27.1 m^2 ha^{-1}. The forest was floristically diverse. More than half of the trees consisted of *Castanopsis tribuloides* (65%) followed by *Schima wallichii* (16%), *Myrica esculenta* (6%), *Rhododendron arboreum* (5%), *Quercus lamellosa* (4%) and various other species (4%). Although largely evergreen, the natural forest sheds a small proportion of its leaves towards the end of the dry season (February –March) but recovers quickly thereafter. For example, the maximum measured leaf area index (LAI, using a Licor 2000 Plant Canopy Analyzer) was 5.43 ± 0.12 (SD) in September 2011 (i.e., at the end of the rainy season), while corresponding values measured during the pre-monsoon period in March, April and early June were 4.52 ± 0.19, 5.14 ± 0.09, and 5.32 ± 0.10, respectively. The soil was classified as a Cambisol of clay loam texture. Clay content varied little with depth between 5 and 100 cm (26–29 %) as did the sand (24–26%) and silt (44–48%) contents. Soil organic carbon (SOC) declined from 4.10 ± 0.25% at 5–15 cm depth to 0.72 ± 0.13% between 50 and 100 cm depth. The topsoil had a low bulk density (0.93 ± 0.08 g cm^{-3} at 5–15 cm). During soil profile excavations and in road exposures, roots were observed to penetrate into the underlying (weathered) bedrock. Depth to bedrock within the 50 m x 50 m plot was ca. 2.3 m.

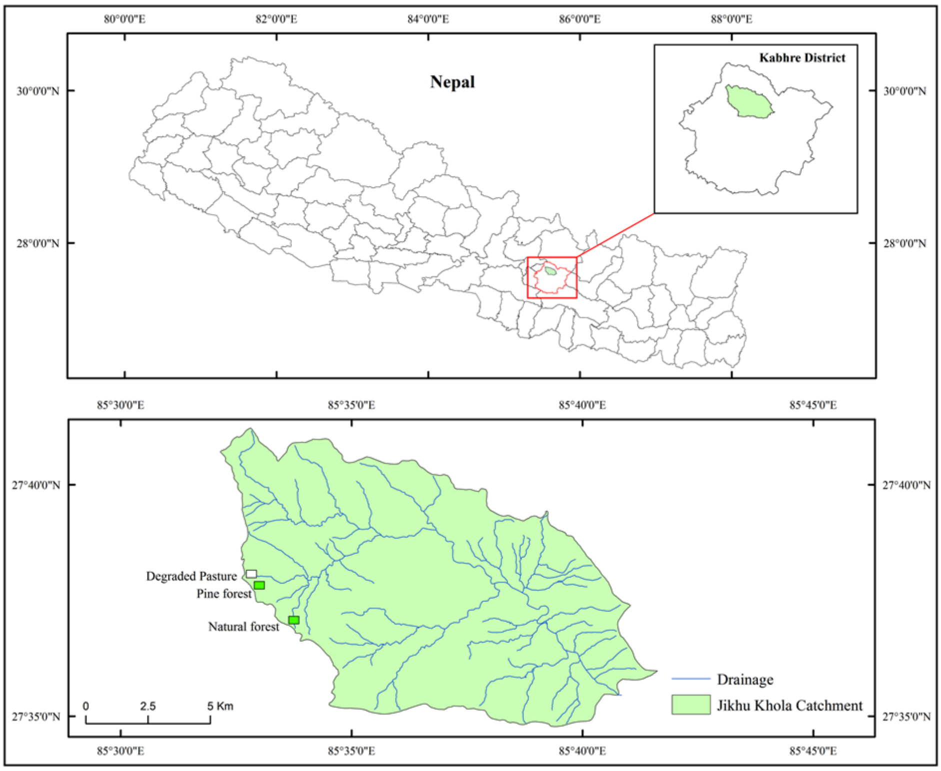

Figure 6.1: Location of the study sites in the Jikhu Khola Catchment in the Middle Mountains of Central Nepal.

Degraded Pasture: This site (1620 m a.m.s.l., southeasterly exposure, average slope angle 18^{o}) has been heavily grazed for more than 150 years according to various local sources and is crossed by numerous footpaths. It is located ~2700 m from the natural forest plot. Numerous patches of compacted or bare soil surface were evident. The dominant grass and herb species included *Imperata cylindrica*, *Saccharum spontaneum*, and

Ajuga macrosperma. Little or no grass cover remained at the peak of the dry season (March–May). The Cambisol underneath the degraded pasture plot had a silty clay texture, with a lower clay content (12–19%) and a higher sand content (34–44%) than the soil of the natural forest plot. Because of the intensive grazing and frequent human traffic, topsoil bulk density in the degraded pasture was significantly higher (1.18 ± 0.33 g cm^{-3}) than in the natural forest. Depth to bedrock within the plot was determined at 2.4 m.

Pine Forest: This former degraded pasture located approximately 400 m from the studied degraded pasture on a 20^o slope of southwesterly exposure was planted with *P. roxburghii* in 1986. The pine trees were 25 years old at the start of field work in June 2010. No other tree species were recorded in the plot. An understory was largely absent as grazing by cattle is common. In addition, the local population collects the litter for animal bedding and regularly harvests the grassy herb layer. Pruning of trees for fuelwood, and felling for timber are also common in the pine forests of the JKC (Schreier et al., 2006; cf. Ghimire et al., 2013a) although the research plots themselves remained free from such major disturbances throughout the present investigation. Like the dominant trees in the natural broad-leaved forest, the evergreen *P. roxburghii* trees shed a proportion of their needles towards the end of the dry season but also recovered their foliage fairly quickly thereafter. The LAI of the pine forest plot was estimated at 2.21 ± 0.10 in September 2011 whereas corresponding values during the pre-monsoon period in March, April and early June were 2.05 ± 0.14, 2.15 ± 0.12, and 2.17 ± 0.11, respectively. The stem density, mean DBH and basal area as measured in May 2011 were 853 trees ha^{-1}, 23.6 ± 3.8 cm and 37.6 m^2 ha^{-1}, respectively. The average tree height was estimated at 16.3 ± 3.8 m. The Cambisol underneath the pine forest plot had a silty clay texture, with a lower clay content (11–19%) and a (much) higher sand content (40–47%) than the soil of the natural forest or degraded pasture. Because of the regular collection of litter and associated exposure of the soil surface to erosive canopy drip, topsoil organic carbon in the pine forest was much lower (1.7 ± 0.3%) and bulk density significantly higher (1.24 ± 0.10 g cm^{-3}) than in the natural forest. Roots were observed to penetrate into the underlying (weathered) bedrock which occurred at a depth of ca. 1.5 m.

6.2.2 Field measurements

6.2.2.1 Environmental monitoring

Environmental conditions were monitored by an automatic weather station located in the degraded pasture site at a distance of 400 m from the pine forest plot (Figure 6.1). Incident rainfall (*P*, mm) for each plot was recorded using a tipping bucket rain gauge (Rain Collector II, Davis Instruments, USA; 0.2 mm per tip) backed up by a manual gauge (100 cm^2) that was read daily at ca. 08:45 AM local time. Incoming short-wave radiation (R_s) was measured using an SKS 110-pyranometer (Skye Instruments, U.K.). Air temperature (*T*, ^oC) and relative humidity (RH, percentage of saturation) were measured at 2 m height using a Vaisala HMP45C probe protected against direct sunlight and precipitation by a radiation shield. Wind speed and wind direction were measured at 3 m height, using an A100R digital anemometer (Vector Instruments, U.K.) and W200P potentiometer (Vector Instruments, U.K.), respectively. All measurements were recorded at 5-min intervals by a Campbell Scientific Ltd. 23X data-logger. Soil water content at each plot was measured at depths of 10, 25, 50, and 75 cm using TDR sensors (CS616, Campbell Scientific Ltd.) at 30-min intervals.

6.2.2.2 Forest hydrological measurements

Wet canopy evaporation or rainfall interception (E_i) was calculated as the difference between gross rainfall (*P*) and net rainfall (throughfall + stemflow). Throughfall (Tf, mm) in the natural and pine forest plots was measured daily using 10 (pine plot) or 15 (natural forest) funnel-type collectors (154 cm^2 orifice) that were regularly relocated in a random manner (cf. Holwerda et al., 2006). In addition, Tf was recorded continuously using three tilted stainless steel gutters in each plot (200 cm x 30 cm each). The throughfall measurements were carried out from 20 June to 9 September, 2011 (81 days), thereby covering the bulk of the 2011 rainy season. Stemflow (Sf, mm) was measured on 10 trees which were representative of the dominant species in the natural forest plot and similarly on eight trees in the pine forest plot. Stemflow was collected using 10-litre buckets connected to flexible polythene tubing fitted around the circumference of the trunk in a spiral fashion at 1 m from the ground. Stemflow measurements were carried out for 65 days between 28 July and 1 October, 2010. Sf was not measured during the 2011 rainy season. Instead, the average values derived for the 2010 rainy season (expressed as a fraction of P) were used when estimating and modelling interception losses for 2011. Annual interception losses were estimated using the revised analytical model of Gash et al. (1995) which was run on a daily basis for the year between 1 June 2010 and 31 May 2011 using the forest structural and average evaporation model parameters established by Ghimire et al. (2012) for the same sites and daily rainfall values as input. For a more detailed description of the measuring and modeling procedures used in the derivation of annual totals of Tf, Sf and E_i, the reader is referred to Ghimire et al. (2012).

The quantification of tree transpiration (E_t) was accomplished by *in situ* xylem sap flow rate measurements. Sap flow measurements on individual trees involve the measurements of xylem sap flux density and sap wood cross sectional area (Granier, 1985; Lu et al., 2004; Lubczynski, 2009). Sap flux density J_p was measured primarily using thermal dissipation probes (TDP; Granier, 1985) because of their low cost, easy installation and overall simplicity, while the heat field deformation method (HFD, Nadezdhina et al., 2012), and the heat ratio method (HRM, Burgess et al., 2001) were used in addition for the purpose of deriving radial sapflow patterns. Sapwood cross-sectional area was estimated from wood cores using an increment borer at breast height (Grissino-Mayor, 2003). Long-term monitoring of J_p was conducted on nine trees in the natural forest plot that were considered representative in terms of the dominant species present and overall size class distribution, and similarly on six trees in the pine forest plot between June 2010 and May 2011. Radial sap flow patterns were measured using the HFD method on eight trees in the pine forest and on two *C. tribuloides* trees in the natural forest for at least three consecutive days per tree. Similarly, radial sap flow patterns for two other dominant species (*S. wallichii*, n = 2; *M. esculenta*, n = 1) were derived using the HRM. In addition, a sap flow campaign was held from March to May 2011 to capture the sap flow of additional trees that were not covered by the long-term monitoring. A total of 48 additional trees were monitored in the pine forest (distributed over four additional 30 m x 30 m plots) vs. 24 trees in the natural forest (two additional 30 m x 30 m plots) during this campaign using a single TDP sensor per tree. Tree-scale measurements of J_p were scaled up to the plot level using least-squares regressions between total trunk cross-sectional area and corresponding sapwood area, relations between sapwood area and J_p, and information on radial changes in J_p for different tree species before summing the water uptake by all individual trees in a plot to give stand transpiration. For further details on the measurement of J_s, gap-filling and the scaling exercise the reader is referred to Chapter 3.

To obtain approximate annual evapotranspiration (ET) totals for the two forests, the respective values of wet- and dry canopy evaporation (i.e., E_i and E_t) were added and combined with estimates for evaporation from the understory (natural forest only given the absence of understory vegetation in the pine stand; cf. Ghimire et al., 2013b) and from the litter layer based on findings obtained at other sites having comparable forest structural and climatic characteristics. In view of the high LAI of the natural forest (5.4 m^2 m^{-2}), its northwesterly exposition and the generally low wind speeds, evaporative contributions by the understory were expected to be modest. Motzer et al. (2010) determined the evaporative fraction contributed by the understory at 20% of that by the overstory in a lower montane forest of comparable stature and LAI in Ecuador. This value was adopted for the Dhulikhel natural forest. Corresponding evaporation from the forest floor (E_s) was considered to be low for the reasons given above and in view of the strongly seasonal rainfall regime which causes the forest floor to be moist for four months only. Comparative measurements of E_s in sub-tropical broad-leaved forests are rare but Kelliher et al. (1992) determined a value of 0.3 mm d^{-1} in a well-watered New Zealand forest. Translating this finding to the Dhulikhel situation and assuming the bulk of E_s to take place during the rainy season (four months) gave an estimated value of 35 mm yr^{-1}. Effectively the same result (36 mm yr^{-1}) was obtained when applying the ratio of E_s to E_t derived for the same New Zealand forest (0.18; Kelliher et al., 1992) to the Dhulikhel forest. Thus, a value of 35 mm yr^{-1} was adopted as a first estimate for E_s in the natural forest,

For the more open pine forest (LAI, 2.2), one expects E_s to be somewhat higher than in the nearby natural forest. However, the pine litter is typically harvested after the main leaf-shedding period (Ghimire et al., 2013a, b), reducing amounts of litter present and thus its moisture retention capacity. Further, a substantial fraction of the rainfall in the pine forest runs off as overland flow (see Results below), reducing E_s even further. Waterloo et al. (1999) determined E_s in a similarly stocked stand of *Pinus caribaea* in Fiji at 9% of the Penman open water evaporation, E_0. Applying the same fraction and taking again an effective period of four rainy months yielded an estimated E_s for the Dhulikhel pine forest of ca. 35 mm yr^{-1}**.**

6.2.2.3 Soil hydraulic conductivity and inferred hillslope hydrological pathways

Field-saturated hydraulic conductivity (K_{fs}) in the respective plots was measured at the hillslope scale, both at the surface and at depths of 0.05–0.15 m, 0.15–0.25 m, 0.25–0.50 m, and 0.5–1.0 m. The K_{fs} at different depths were subsequently combined with selected percentiles of 5-min maximum rainfall intensities (RI_{5max}) to infer the dominant hillslope hydrological response during intense rainfall following Chappell et al. (2007). A disc permeameter (Perroux and White, 1988; Mckenzie et al., 2002) was used for the measurement of surface K_{fs} in the field and a constant-head well permeameter (CHWP; Talsma and Hallam, 1980) for the measurement of K_{fs} in deeper layers. Use of the CHWP was restricted to the dry season to minimize errors from smearing of the auger hole walls (Chappell and Lancaster, 2007). For a detailed description of the measuring procedures and sampling strategy, the reader is referred to Ghimire et al. (2013b).

6.2.2.4 Overland flow

Overland flow at the natural forest, degraded pasture and pine forest sites was monitored between 20 June and 9 September 2011 (i.e., the bulk of the 2011 rainy season) using a single large (5 m x 15 m) runoff plot per land-cover type. Runoff was collected in a gutter funneling the water to a first 180 l collector equipped with a seven-slot divider system allowing only 1/7th of the spill-over into a second 180 l drum, thereby bringing the total collector capacity to 1440 l (~20 mm). The water levels in the two collectors were measured continuously using a pressure transducer device (Keller, Germany) placed at the bottom. Collectors were emptied and cleaned after measuring the water level manually every day around 8:45 AM local time. Event runoff volume was calculated by converting the water levels to volumes using a pre-calibrated relationship per drum and summing up to obtain total runoff volume. Measured overland flow volumes were corrected for direct rainfall inputs into the runoff collecting system. Overland flow volumes were divided by the projected plot area to give overland flow in mm per event.

6.2.2.5 Grassland evaporation

The HYDRUS-1D model for one-dimensional soil water movement (Šimůnek et al., 2008) was used to estimate evaporation from the degraded pasture site. In doing so, ET was assumed to be equal to the evaporation from the unsaturated zone. Like most pastures in the area the site was heavily overgrazed (Gilmour et al., 1987; Ghimire et al., 2013a, b) and the capacity of the grass to intercept rainfall was considered negligible. The HYDRUS-1D model is based on the modified Richard's equation. In this study it was assumed that the gaseous phase plays an unimportant role in the overall transport of moisture while water flow due to thermal gradients is also neglected (Šimůnek et al., 2008).

The modeling was divided into three parts: (i) model calibration against measured soil moisture data (1 September–30 November 2010) using inverse modeling, (ii) model validation using data collected between 1 February and 31 March 2011, and (iii) complete simulation of soil water dynamics and evaporation for the entire annual period (1 June 2010–31 May 2011). The 1 m deep soil column was divided into four schematic layers as follows: (i) 0–0.15 m, (ii) 0.15–0.25 m, (iii) 0.25–0.50 m, and (iv) 0.50–1.0 m.

The soil physical parameters employed in HYDRUS-1D include θ_r for the residual water content, and θ_s for the saturated water content, together with the two parameters, α and n, describing the shape and range of the soil water retention curve and the derived relative hydraulic conductivity curve (Van Genuchten, 1980). Other model parameters include the saturated hydraulic conductivity (K_s), and I, a pore-connectivity parameter. All parameters were optimised using inverse methods except for K_s which was measured separately for each layer in the field using well permeametry as described above. The inverse method optimised the parameter values by fitting observed and modeled soil moisture values using the Marquardt-Levenberg optimisation algorithm. The model was run on an hourly basis. The boundary conditions used in the model were the atmospheric boundary (soil surface) with surface runoff occurrence, and free drainage for the bottom boundary.

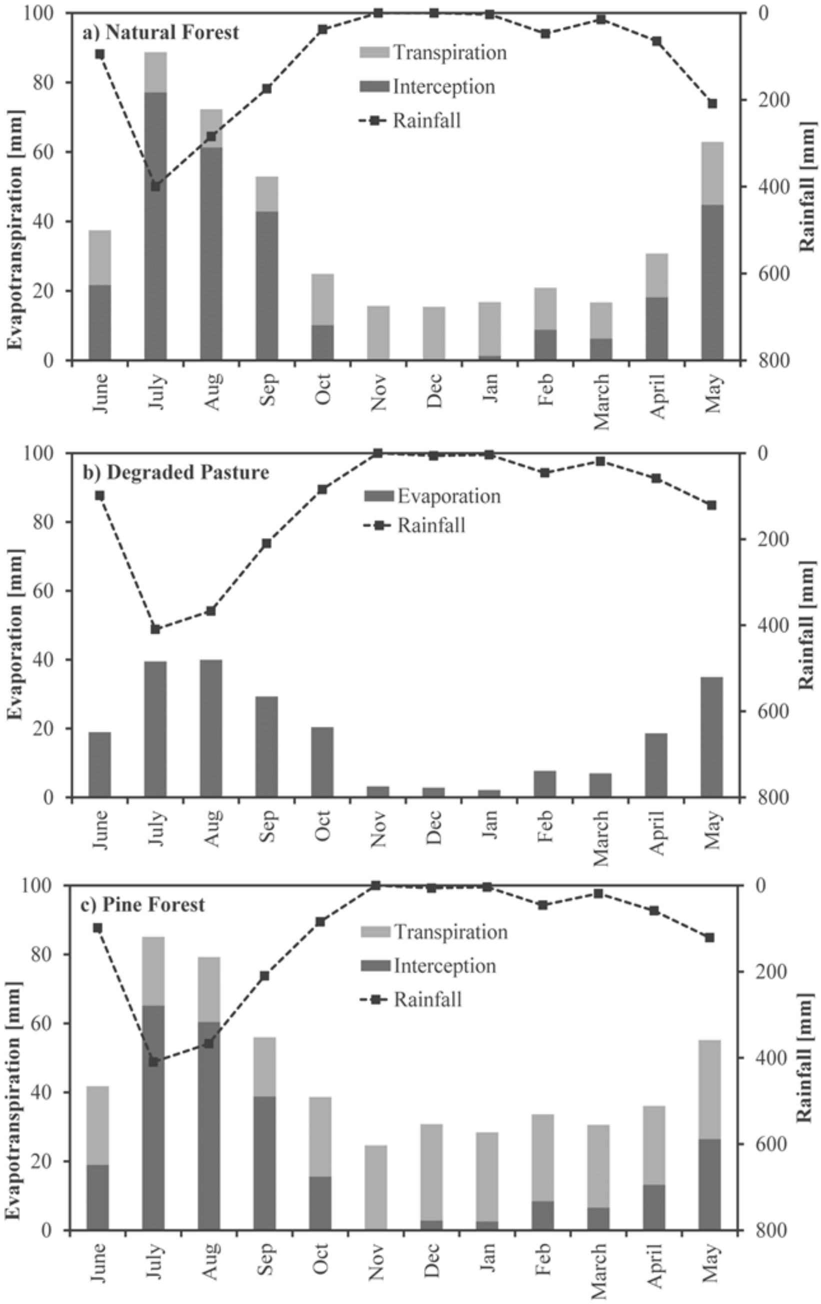

Figure 6.2: Monthly rainfall, interception and transpiration totals (mm) between 1 June 2010 and 31 May 2011 in: (a) natural broad-leaved forest, (b) degraded pasture, and (c) planted forest near Dhulikhel, Central Nepal.

6.3 Results

6.3.1 Evapotranspiration

Figure 6.2 shows the monthly variation in evapotranspiration (ET) and its two main components for the three land-cover types studied between 1 June 2010 and 31 May 2011 whereas the respective seasonal and annual evapotranspiration totals are presented in Table 6.1.

Although the seasonal patterns for ET were similar between vegetation types, monthly ET totals were generally highest for the planted pine forest and lowest in the degraded pasture throughout the monitoring period (Figure 6.2). All three sites showed higher ET rates and monthly totals during the wet season months (June–September) compared to the dry season months (October–April), with the transitional month of May (marking the first return of the rains; Figure 6.2a) showing intermediate values. Such findings can be attributed largely to the (much) higher frequency of wetting and subsequent drying (evaporation) during the wet season (cf. Table 6.1).

As found earlier for monthly ET totals, the annual ET for the planted forest was the highest of all three vegetation types studied (577 mm), being two and a half times larger (225 mm; Table 6.1). The annual ET for the natural forest was 524 mm, which is some 53 mm less than that for the nearby pine forest but 300 mm higher than the water use of the degraded pasture (Table 6.1). Whilst absolute rainfall interception totals did not differ too much between the natural and the planted forest (31 mm yr^{-1} higher in the broad-leaved forest despite a somewhat lower rainfall total), both the seasonal and annual transpiration totals were distinctly higher for the pine forest (Table 6.1). Wet-season transpiration in the pine forest was some 20 mm higher vs. 64 mm during the dry season.

6.3.2 Soil hydraulic conductivity, overland flow and subsurface flow paths

As expected on the basis of the degree of anthropogenic pressure experienced by the respective sites, the median surface K_{fs} was lowest for the degraded pasture (18 mm h^{-1}) and highest for the natural forest (232 mm h^{-1}), such that the two differed by more than an order of magnitude (Figure 6.3). The most striking feature of the K_{fs} data-set is that the median K_{fs} at the surface and in the shallow soil layer in the 25 year old pine forest had remained at the same level as the corresponding values for the heavily grazed pasture, suggesting the virtually complete absence of biologically mediated macropores in the pine forest soil down to 0.15 m depth. At 1.0 m depth, however, differences in K_{fs} between the respective land-cover types were mostly non-existent (except for the higher value beneath the pine forest reflecting the much higher sand content listed in the site description), illustrating the lack of influence

Table 6.1: Summary of rainfall and estimated evapotranspiration components for a natural broad-leaved forest, a degraded grassland, and a mature planted pine forest near Dhulikhel, Middle Mountains, Central Nepal.

	Natural Forest			Degraded Pasture			Pine Forest		
	Wet	Dry	Total	Wet	Dry	Total	Wet	Dry	Total
Rainfall (P, mm)	953	378	1331	1084	338	1423	1084	338	1423
Transpiration (E_t, mm)	48	115	163	-	-	-	78	202	280
Interception (E_i, mm)	203	90	293	-	-	-	184	78	262
Grassland evaporation	-	-	-	128	97	225	-	-	-
Understory evaporation (E_{us}, mm)	-	-	33	-	-	-	-	-	-
Litter evaporation (E_s, mm)	35	-	35	-	-	-	35	-	35
Total evapotranspiration (ET, mm)	286	205	524	128	97	225	297	280	577

exerted by cattle grazing and human trampling on the deeper soil layers. Importantly, the surface K_{fs} in the natural forest exceeded the maximum values of RI_{5max}, suggesting infiltration-excess overland flow (IOF) would never occur at this site. Nevertheless, some overland flow was recorded (Figure 6.4) with a monsoonal total of 18 mm (22 mm after normalising for the higher rainfall observed at the two other sites) or 2.5% of incident rainfall and 3.3% of the corresponding amount of throughfall. Given the much lower median K_{fs} derived for the 0.05–0.15 m depth interval in the natural forest (82 mm h^{-1}; Figure 6.3), it cannot be excluded that at least some of the recorded overland flow was contributed by the saturation-excess type (SOF) (Bonell, 2005). The median surface K_{fs} values of degraded pasture and pine forest were below the upper quartile of RI_{5max}, thereby indicating the frequent occurrence of IOF during high-intensity rainfall (Figure 6.2). Indeed, overland flow at the degraded pasture site was typically generated after 3–4 mm of rain whereas the seasonal runoff total from the pasture amounted to 187 mm (21.3% of incident rainfall; Figure 6.4). Corresponding values for the pine forest were comparable at 4.2 mm of rain before runoff would start and a seasonal total of 136 mm (15.5% of rainfall and 18.6% of throughfall; Figure 6.4).

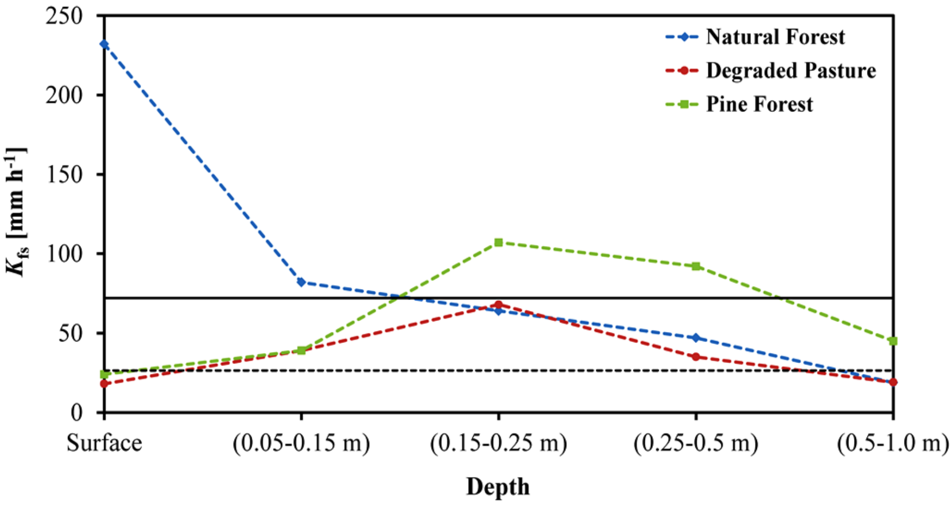

Figure 6.3: Changes in field-saturated hydraulic conductivity K_{fs} with depth as a function of land use near Dhulikhel, Central Nepal. The dashed and solid horizontal lines represent the median and 95% percentile of RI_{5max} rainfall intensity, respectively (after Ghimire et al., 2013a).

With regard to K_{fs} in the 0.05–0.15 m layer, the upper quartile of RI_{5max} exceeded the median K_{fs} values at both the degraded pasture and pine forest sites (Figure 6.3). At such rates, the rainfall that percolates to this depth would be capable of developing a perched water table and so generate lateral subsurface storm flow (SSF) at shallow depth. In contrast, the corresponding median K_{fs} at the natural forest is still above (or nearly equal to) most 5 min rainfall intensities, thereby favouring mostly vertical percolation at this site. For the 0.15–0.25 m and 0.25–0.50 m depth intervals, the K_{fs} values in the natural forest and degraded pasture indicated a similar hydrological response to rainfall, namely mostly lateral SSF and thus limited vertical percolation during high-intensity rainfall (cf. Figure 6.3). However, the much higher median values of K_{fs} between 0.15 and 0.50 m depth at the pine forest site exceeded the maximum values of RI_{5max} (Figure 6.3) and thus rather promote vertical percolation. These higher conductivities are likely to reflect the higher sand content of the soil beneath the pine forest (Ghimire et al., 2013b). The effect, however, must be counteracted to some extent by the low median surface K_{fs} in the pine forest which encourages IOF and restricts the amounts of water percolating to deeper soil layers. Finally, at 1.0 m depth, the differences in K_{fs} and inferred hydrological response to rainfall became insignificant between sites (Figure 6.3).

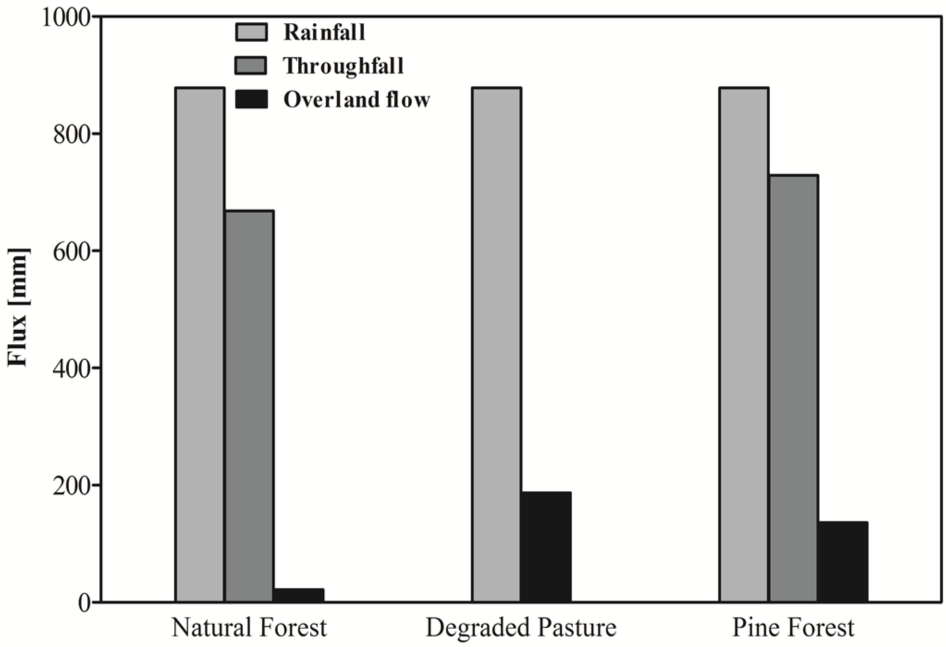

Figure 6.4: Amounts of rainfall, throughfall and overland flow (mm) during the 2011 monsoon measuring campaign (20 June – 9 September) in a natural forest, a degraded pasture, and a mature, intensively used pine plantation near Dhulikhel, Central Nepal. Note that values for the natural forest plot were normalized for rainfall amount to allow more direct comparisons.

6.3.3 Site water budgets

Combining the above mentioned IOF percentages for the pine forest and degraded pasture with a mean annual site rainfall of 1500 mm (Merz, 2004), the difference in approximate annual IOF between the two land covers represents a gain in infiltration of approximately 90 mm yr^{-1} under the planted pine forest relative to the degraded grass land (Table 6.2). On the other hand, the relative amounts of soil water uptake in the degraded pasture and the pine forest, plus the added rainfall interception losses from the pine forest suggest a difference in annual evapotranspiration of ~350 mm yr^{-1} after reforestation and stand maturation (Table 6.2).

Thus, the added loss through ET is greatly in excess of the estimated gain in infiltration of 90 mm after reforestation causing a net loss of ~260 mm. Repeating the exercise for the natural forest (with an estimated annual ET of ~525 mm and very low overland flow production) suggests the approximate gain in infiltration (285 mm yr^{-1}) and the extra evaporative loss (300 mm yr^{-1}) are both very similar (Table 6.2) implying no major change in moisture availability and dry season flow under mature forest conditions.

Table 6.2: Summary of changes in annual evapotranspiration (ET, mm) and overland flow (mm) and the resultant gains in infiltration (mm) and net evaporative losses (mm) when converting degraded pasture to (heavily used) planted pine forest or (little disturbed) natural broad-leaved forest near Dhulikhel, Central Nepal. Note that overland flow amounts were calculated for the mean annual site rainfall of 1500 mm (Merz, 2004). Note that all values are rounded off to the nearest 5 or 10 mm.

Experimental plot	ET (mm)	Overland flow (mm)	Infiltration gain (mm)	Net change in ET (mm)	Overall net effect (mm)
Degraded pasture	225	320	-	-	-
Pine forest	575	230	90	350	-260
Natural forest	525	35	285	300	-15

6.4 Discussion

6.4.1 Human impact on forest hydrological functioning – a under studied dimension

The results obtained for the respective water balance components suggested that soil water replenishment and retention during the monsoon are largely controlled by surface- and subsurface soil hydraulic conductivities and the resultant partitioning of rainfall into overland flow, lateral subsurface flow and deep percolation (Table 6.1, Figure 6.3). As long as rainfall intensities remain below the surface K_{fs} threshold for overland flow to occur, soil water reserves are being recharged. However, for intensities above this threshold a major proportion of the rain is re-directed laterally over the surface as overland flow and less water is available for soil moisture replenishment. The high surface- and near-surface K_{fs} in the natural forest (82–232 mm h^{-1}) ruled out IOF occurrence and favour vertical percolation. In contrast, the corresponding K_{fs}-values for the planted forest and degraded pasture were conducive to IOF generation during medium- to high-intensity storms which represents a net loss of moisture to these hillslopes (Figure 6.3, Table 6.2).

Marked reductions in surface- and near-surface K_{fs} after converting tropical forest to grazed pasture have been observed in many cases (Alegre and Cassel, 1996; Tomasella and Hodnett, 1996; Deuchars et al., 1999; Zimmerman et al., 2006; Molina et al., 2007; Tobón et al., 2010) and the Himalayas are no exception (Patnaik and Virdi, 1962; Gilmour et al., 1987; Gerrard and Gardner, 2002; Ghimire et al., 2013a, 2013b). The low surface and near-surface K_{fs} reported for grazing conditions is mostly the result of destroyed macroporosity through trampling by cattle and by the much diminished soil faunal activity after forest clearing and burning with the associated loss of topsoil organic matter and soil exposure to erosive precipitation of the topsoil to the elements (McIntyre, 1958a, 1958b; Lal, 1988; Deuchars et al., 1999; Colloff et al., 2010; Bonell et al., 2010). Whilst natural forest regrowth on degraded pasture or planting trees followed by uninterrupted plantation development can be expected to gradually improve the soil water intake capacity again (Gilmour et al., 1987; Ilstedt et al., 2007; Bonell et al., 2010; Colloff et al., 2010; Hassler et al., 2011; Perkins et al., 2012), the surface hydraulic conductivity of the intensively used pine forest showed little improvement even 25 years after the trees were planted (Figure 6.3). Clearly, the continued human access, grazing and collection of forest products (notably litter from the forest floor to be used for animal bedding and composting; Singh and Sundriyal 2009; Joshi and Negi, 2011) is having a profound negative effect on the stand’s hydrological functioning (cf. Ghimire et al., 2013a, 2013b). Thus, the general expectation of restored surface and near-surface K_{fs} with time after reforestation (cf. Gilmour et al., 1987; Ilstedt et al., 2007) is in need of modification under the conditions of high anthropogenic pressure that appear to be the rule rather than the exception in the Middle Mountain Zone (Singh et al., 1984; Mahat et al., 1987; Singh and Sundriyal, 2009; Joshi and Negi, 2011) despite claims to the contrary (HURDEC Nepal and Hobley, 2012). Indeed, if the potential benefits of reforestation such as enhanced infiltration, and therefore possibly improved replenishment of soil water and groundwater reserves, are to be realised, then a balance will need to be struck between the continued usage of the forests by uplanders whose livelihoods are at stake and sustained forest hydrological functioning (Ghimire et al., 2013a). Naturally, this holds for many other densely populated uplands as well (e.g., Ding et al., 1992; Van Noordwijk et al., 2001; Forsyth and Walker, 2008; Van Noordwijk and Leimona, 2010).

6.4.2 Trade-off between changes in vegetation water use and infiltration after capacity reforestation

The 'infiltration trade-off' hypothesis states that the ultimate hydrological effect of reforestation in terms of site water yield is determined by the net balance between increases in soil water reserves afforded by improved soil infiltration *versus* decreases caused by the higher plant water uptake (Bruijnzeel, 1986, 1989). In the absence of direct published evidence of improved dry season flows after reforestation (Jackson et al., 2005; Farley et al., 2005) at the time, Scott et al. (2005) discussed a number of tropical studies that had observed marked reductions in stormflow production at the hillslope (e.g., Zhang et al., 2004; Chandler and Walter, 1998) or small-basin scale (e.g., Zhou et al., 2002) after reforesting severely degraded land. They concluded that in a number of cases the increases in water retention should be more than enough to compensate the estimated corresponding increases in forest water use (not measured). It is unfortunate that the catchments involved did not sustain perennial flows and thus the presumed net positive effect of forestation on dry season flows could not be confirmed (Scott et al., 2005; cf. Chandler, 2006). Nevertheless, recently published evidence from South China (Zhou et al., 2010), South Korea (Choi and Kim, 2013) and Southwest India (Krishnaswamy et al., 2012, 2013) strongly suggests a net positive outcome of the infiltration trade-off mechanism is possible provided a sufficiently degraded initial situation and ample rainfall.

Such experimental catchment studies usually comprise measurement of the change in streamflow following reforestation as opposed to detailed

Figure 6.5: Conceptual diagram of the effect of land-cover transformation on annual total and dry season flows in the study area as well as other comparable regions with similar land-cover transformations.

process-based studies within the catchment undergoing the change (Farley et al., 2005; Scott et al., 2005). As such, the present process-based work which integrated the dominant hydrological processes (notably evaporation, infiltration and runoff generation) to quantify the net hydrological impact of reforestation is a first (cf. Chandler, 2006; Bonell et al., 2010; Krishnaswamy et al., 2013). Comparing the hydrological behaviour of the three contrasting land-cover types studied here (degraded pasture, mature near-undisturbed broad-leaved forest, and a heavily used mature pine plantation) within the context of the infiltration trade-off hypothesis showed that planting pines increased vegetation water use relative to the pasture situation by ~350 mm yr^{-1} (Table 6.1). On balance, the limited amount of extra infiltration afforded by the pine trees (~ 90 mm yr^{-1}) is clearly insufficient to compensate the much higher water use of the pines, giving a net negative balance of ~260 mm yr^{-1} (Table 6.2, Figure 6.4). Pertinently, the net effect would still have been negative (by ~120 mm yr^{-1}) even if all rainfall would have been accommodated by the pine forest soil through better forest and soil management promoting infiltration (cf. Wiersum, 1985). As such, the observed decline in dry season flows following reforestation in the study area (República, 2012) is likely to primarily reflect the higher water use of the pines (Tables 6.1 and 6.2; Figure 6.5).

If the degraded pasture were to revert to natural forest instead (with an estimated annual ET of 525 mm and very limited overland flow production) the ultimate effect on dry season flows would be expected to be near-neutral as the approximate gain in infiltration (~285 mm yr^{-1}) and the extra evaporative loss (~300 mm yr^{-1}) are very similar (Table 6.2). Effects might be more negative in case the water use of the young regenerating broad-leaved forest turned out to be enhanced compared to that of old-growth forest as found for various lowland tropical and warm-temperate forests (Vertessy et al., 2001; Giambelluca, 2002). Although the present finding of a slightly higher ET (~10%; Table 6.1) for planted forest compared to natural broad-leaved forest is not going to have major hydrological consequences on an annual basis, the much higher water use of the pines during the dry season (Table 6.1, Figure 6.1) is likely to result in a corresponding reduction in water yield upon converting natural broad-leaf forest to pine plantations (Figure 6.5), especially during the more vigorous early growth stage of the pines (Bruijnzeel, 1997; Scott and Prinsloo, 2008; Alvarado-Barrientos, 2013). The present results further illustrate that the conditions found in the nearly undisturbed natural broad-leaved forest and in similarly well-maintained forests elsewhere in the Himalaya (Pathak et al., 1984; Gerrard and Gardner, 2002) will encourage the replenishment of soil water and groundwater reserves through vertical percolation more than in any other land-cover type studied here (Figure 6.5; cf. Chuoi and Kim, 2013; Krishnaswamy et al., 2013) and so better sustain baseflows during the long dry season. The importance of the latter can hardly be overstated (Merz et al., 2003; Schreier et al., 2006; Bandyopadhyay, 2013).

6.4.3 Regional implications

The Himalayan river basins are home to about 1.3 billion people and supply water, food and energy to more than 3 billion people in total (Bandyopadhyay, 2013). Thus, large-scale changes in Himalayan land use and hydrology will have important regional consequences. For example, substantial decreases in dry season flows following advanced surface degradation (Bartarya, 1989; Madduma Bandara, 1997) or large-scale reforestation (cf. Trimble et al., 1987) would affect the availability of water for millions of people, both those depending directly on agriculture for their livelihoods and downstream city dwellers. Therefore, reforestation campaigns and the subsequent use of the planted forests must be based on a sound assessment of what is to be expected hydrologically (Peña-Arancibia et al., 2013).

The presently observed negative hydrological effect of an apparently long-term trend of gradual forest degradation in the Nepalese Middle Mountain Zone goes against the optimistic notion regarding the overall improved quality of Lesser Himalayan forests expressed by HURDEC and Hobley (2012). However, there is reason to believe that the situation of over-intensive use of forests and the correspondingly poor soil hydrological functioning are a rather more widespread phenomenon in Nepal’s Lesser Himalaya. For example, Gerrard and Gardner (2002) reported very high overland flow occurrence in degraded (broad-leaved) forests in the Likhu Khola catchment north of Kathmandu, whereas Tiwari et al. (2009) and Wester (2013) recently presented similar evidence for community-managed forests further west in Nepal. Although process-based hydrological evidence to this effect from the Indian Himalaya is scarce (Negi, 2002; Tiwari et al., 2011; Tyagi et al., 2013) the continued over-exploitation of its forest resources is well-documented (Singh et al., 1984; Singh and Sundriyal, 2009; Joshi and Negi, 2011). Such situations may significantly reduce the recharge of shallow groundwater reserves during the monsoon season, thereby potentially decreasing regional dry season flows (Bartarya, 1989; cf. Andermann et al., 2012).

A key message of the present work thus is the need to protect the remaining natural forests in headwater areas throughout the Middle Mountain Zone. The present findings further highlight the need for some form of protection of reforested areas that will enable the forest soils to realise the enhanced infiltration/ percolation benefits envisaged at the time of planting. Continued degradation of the remaining old-growth forests and planted forests that are now reaching maturity is likely to cause further increases in overland flow production during the monsoon season due to the corresponding decline in infiltration opportunities. This may, in turn, have a further negative effect on already declining dry season flows and will cause increased hardship to the rural populace (Merz et al., 2003; Schreier et al., 2006). Last, but not least, the present results point to the need for balancing the societal and hydrological functions of forests (both planted and natural) in densely populated uplands. Like elsewhere in Asia (e.g., Tomich et al., 2004; Hairiah and Van Noordwijk, 2005) agro-forests that contain a variety of tree and crop species serving a range of uses as opposed to the mono-specific character of most planted forests (Wallace et al., 2005) represent a viable alternative that is on the increase in Nepal (Gilmour and Shah, 2012).

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Chapter 7

Conclusions and Recommendations

This chapter constitutes the summary of the thesis **"Hydrological impacts of reforesting degraded pasture land in the Middle Mountain Zone of Central Nepal"** and lists the main conclusions emerging from the present research along with some recommendations for further work.

As soil degradation in the tropical parts of the world is on the increase (Oldeman et al., 1991; Bai et al., 2008) and streamflow regimes are increasingly disturbed with excess flows during the rainy season and water shortages during the dry season (Bartarya, 1989; Bruijnzeel, 2004; Roa-García et al., 2011; Ogden et al., 2013), the need for massive land rehabilitation is greater than ever (Global Partnership on Forest Landscape Restoration/IUCN, 2011). One widely applied rehabilitation strategy is the planting of trees in the hope that the addition of organic matter to the forest floor will result in a gradual improvement of the soil infiltration capacity and with it improved soil moisture replenishment (Malmer et al., 2010). However, several recent research reviews have stressed the fact that introducing tree plantations in non-forested areas mostly leads to greatly diminished water yields around the year compared to areas under agricultural crops, grassland or scrublands (e.g., Calder, 2005; Jackson et al., 2005; Kaimowitz, 2005). Such findings appear to be at odds with the traditional and widely held view which claims that a good forest cover provides higher and more stable seasonal flows. And whilst it is theoretically possible that the greater water use of the newly planted trees may be compensated by the enhanced infiltration afforded by soil improvement after planting trees on severely degraded land (the so-called "infiltration trade-off hypothesis"; Bruijnzeel, 1986) – thereby restoring the original dry-season flows – no single study has made the necessary hydrological process measurements in support of such a positive outcome. Recently, several catchment-scale studies have observed a positive trend in streamflow after reforesting highly degraded lands in South and Southeast Asia (Zhou et al., 2010; Krishnaswamy et al., 2013; Choi and Kim, 2013) but these studies remained essentially of the 'black box' type because of a lack of supporting process measurements.

The densely populated Middle Mountain Zone of the Nepalese and Indian Himalayas represents another area where much of the original forest cover is either degraded or disappeared because of strong anthropogenic pressure (Dobremez, 1976; Singh et al., 1984). In combination with a strongly seasonal rainfall regime (Bookagen and Burbank, 2006, 2010) the reduced capacity of the eroded soils to absorb monsoonal rains (Bruijnzeel and Bremmer, 1989; Gerrard and Gardner, 2002) has led to widespread shortages of water during the extended dry season (Bartarya, 1989; Merz et al., 2003; Schreier et al., 2006). In response to these pressures a major reforestation programme was launched in the Middle Mountains of Central Nepal in the 1980s but farmers have since complained about decreasing streamflow availability (República, 2012).

Thus, the main objective of the present study was two-fold. The first objective was to describe and quantify the dominant hydrological processes (rainfall interception, transpiration, runoff generation, rainfall infiltration and percolation) operating in three contrasting land-cover types subjected to gradually increasing levels of anthropogenic pressure: (i) nearly undisturbed natural broad-leaved forest, (ii) an intensively used mature pine plantation established on formerly degraded pasture land, and (iii) a severely degraded and overgrazed pasture, all situated near Dhulikhel, Central Nepal. The second objective of the study was to investigate the net effect (‘trade-off’) between the changes in vegetation water use and soil infiltration capacity after reforesting severely degraded pasture.

7.1 Summary of the research results

7.1.1 Wet-canopy evaporation

Chapter 2: As a first step, rainfall interception (wet-canopy evaporation) and the corresponding forest structural parameters governing rainfall partition at the canopy level were quantified for a natural broad-leaf forest and a 25-year-old planted pine forest. The results showed a higher interception loss from the semi-evergreen natural forest (22.6% of incident rainfall) as compared to that from the planted pine forest (19.4%). The higher interception loss derived for the natural forest was in accordance with what would be expected from the difference in forest stocking (1869 trees ha^{-1} in the natural forest vs. 885 trees ha^{-1} in the pine forest) and LAI (4.5–5.4 $m^2\ m^{-2}$ in the natural forest vs. 2.0–2.2 $m^2\ m^{-2}$ in the pine forest). The revised analytical interception model (Gash et al., 1995) was used for the first time under monsoonal montane conditions to derive these annual interception estimates. There was good agreement between modelled and observed interception losses for the two contrasting forests, provided optimized values were used for the wet canopy evaporation rate.

7.1.2 Dry-canopy evaporation

Chapter 3: For the first time, tree transpiration, canopy conductances and decoupling coefficients were quantified and examined in a natural broad-leaved forest and in a mature planted pine forest in the Middle Mountain Zone of the Himalayan region, using sap flow measurements and concurrent climatic and soil water observations. Tree transpiration rates in both forests were strongly dominated by atmospheric vapour pressure deficits, much less by amounts of energy available for evaporation, and least of all by soil water contents down to 75 cm depth. It was further found that transpiration rates in the two forests peaked during the dry season and exhibited little variation during much of the dry season, indicating the roots must have access to moisture present in deeper soil layers and the weathered portion of the geological substrate. Both the seasonal and annual transpiration totals were distinctly higher for the pine forest compared to the natural forest even after allowing for the difference in site exposition and thus radiation load.

7.1.3 Reforestation, soil hydraulic conductivity and hillslope hydrological response

Chapter 4: The effects of reforesting severely degraded grassland on field-saturated soil hydraulic conductivity (K_{fs}) and overland flow production were examined. Apart from presenting extensive new K_{fs} data for a heavily degraded pasture, a little disturbed broad-leaved forest, and an intensively used pine plantation and extending the range of measurements to the hillslope scale, this Chapter includes the first measurements of K_{fs} for a heavily frequented rural footpath in the Himalaya. The high surface and near-surface K_{fs} observed in the natural forest effectively prevented the occurrence of large-scale infiltration-excess overland flow (IOF) even for the most extreme rainfall events. Thus, intact natural forest favours vertical percolation and the replenishment of soil water and groundwater reserves through maximum infiltration. Conversely, very low surface and near-surface K_{fs} were found in the degraded pasture and particularly for the footpath sections which encouraged the generation of IOF even during events with moderate rainfall intensities. Pertinently, surface and near-surface K_{fs} in the heavily used pine forest had remained similar to those observed for the degraded pasture even after 25 years of forest development. This unexpected finding was attributed to the regular collection of litter material from the forest floor for animal bedding and composting, understory removal, cattle grazing, and fuelwood harvesting. The large volumes of overland flow generated on the footpath (not measured but inferred from a comparison of rainfall intensities and surface K_{fs}), in the degraded pasture (21.3% of incident rainfall) and the planted forest (18.6% of crown drip) in the study area can be expected to contribute disproportionally to local-scale stormflows. The results further illustrate the positive influence of a well-developed litter layer and understory vegetation (IOF in the natural forest only 2.5% of incident rainfall and 3.2% of crown drip) on surface and subsurface hydrology. They also bring out the hydrological importance of preserving the remaining old-growth natural headwater forests of the Middle Mountain Zone of the Nepalese and Indian Himalaya.

7.1.4 Sustained forest use and soil hydraulic conductivity

Chapter 5: To provide further support to the results for the Dhulikhel sites described in Chapter 4 and to obtain insight in 'real time' into the long-term changes in field-saturated soil hydraulic conductivity (K_{fs}) incurred by intensive land or forest usage, K_{fs} was also measured beneath a severely degraded pasture, a disturbed natural forest and in two intensively used mature planted *Pinus roxburghii* stands elsewhere in the Middle Mountains near Chautara (some 20 km from Dhulikhel). Here, a similar set of measurements was made in 1986 (Gilmour et al., 1987) and by repeating the measurements in 2011 at the same locations and using comparable techniques, a direct and unique comparison of K_{fs} values at two points in time separated by 25 years is obtained. Multiple measurements of K_{fs} were made at the above-mentioned four sites, both at the surface and down to 1 m depth. Abrupt and significant decreases in K_{fs} were observed within the upper 0.25 m of the soil profiles under both natural and planted forest after 25 years of various forms of human activity. These decreases essentially reflect the loss of macro-porosity due to the reduced incorporation of soil organic matter after repeated collections of litter from the forest floor supplemented by continuous cattle grazing. Contrary to the forest sites, the studied degraded pasture site showed little change in surface K_{fs} over the intervening 25 years suggesting surface degradation had reached a ceiling already in or before 1986. The results demonstrated that management of community forests in Nepal's Middle Mountain Zone needs to include a consideration of hydrological functioning. Simply planting trees on degraded land is not sufficient in itself to restore the hydrological functioning of degraded catchment areas. It is essential to give proper attention to the on-going management of the planted forest areas to balance forest usage by the local populace with adequate hydrological functioning.

7.1.5 Reforestation and dry season flows

Chapter 6: The trade-offs between changes in vegetation water use and soil infiltration after reforesting severely degraded pasture land with pine trees was investigated. In doing so, the hydrological process measurements presented in Chapters 2–5 were integrated in a water budget framework. Planting of the pines increased vegetation water use relative to the pasture by 355 mm yr^{-1}. On balance, the limited amount of extra infiltration afforded by the pine plantation relative to the degraded pasture (only 90 mm yr^{-1} due to continued soil degradation associated with regular harvesting of litter and understory vegetation in the plantation) proved insufficient to compensate the much higher water use of pines. In contrast, a comparison of the water use of the natural forest and degraded pasture suggests that replacing the latter by (mature) broad-leaved forest would ultimately have a near-neutral effect on dry season flows as the approximate gains in infiltration (285 mm yr^{-1}) and evaporative losses (300 mm yr^{-1}) were very similar. The results of the present study thus highlight once more the need for proper forest management (both natural and planted forests) for optimum hydrological functioning as well as underscore the importance of protecting the remaining headwater forests in the region.

7.2 Recommendations for further research

This research represented a detailed process-based study to quantify the hydrological impacts at the plot level of reforesting severely degraded pasture land with fast-growing pines in the Middle Mountain Zone of

Central Nepal. Due to logistical and financial constraints no replications per land-cover type were possible. Neither were replications in time, i.e. pine plantations of different age and biomass. Thirdly, although discharge measurements were made at the outlet of the 14 km^2 Dhulikhel river catchment within the framework of the current study, these data have not been included because the catchment represents a variety of land-cover types and water yield as determined at the catchment outlet cannot be linked directly to the plot-scale measurements (Blöschl et al., 2007). Thus, a range of future research topics would complement and advance the steps taken by the present research. Much of this possible future work can be listed under the following headings: (i) Vegetation water use (ET), (ii) Runoff generation, and (iii) Upscaling to the catchment level. Below, each set of questions is discussed briefly.

(i) Vegetation water use

This work quantified the evapotranspiration (rainfall interception or wet-canopy evaporation and transpiration or dry-canopy evaporation) from a mature pine plantation and an old-growth natural broad-leaved forest (Chapters 2 and 3). The overall water use determined for the respective forests was rather low (Chapter 3) which may be taken as an evolutionary adaptation of these forests to survive under the strongly seasonal conditions (Singh and Singh, 1992; Zobel et al., 2001) but in the absence of site or measurement replication or indeed differently determined data on tree water use in the Himalayas this contention remains to be confirmed and further plant physiological work is necessary. Further, in view of the widespread regeneration of vegetation on abandoned agricultural fields in the Middle Mountain Zone (Paudel et al., 2012) and the presence of differently aged pine plantations in the study area, future research in the area should focus on the measurement of ET and its components (including understory and litter layer evaporation which had to be estimated at the Dhulikhel sites; Chapter 3), both for naturally regenerating forests at different stages of growth (cf. Giambelluca, 2002; Hölscher et al., 2005) and differently aged pine forests. This will help to ascertain the effect of the presence of differently aged vegetations (both planted and naturally regenerating) on streamflow. This is particularly important because the water use of vigorously regenerating forests typically exceeds that of old-growth forest (cf. Giambelluca, 2002; Hölscher et al., 2005; Muñoz-Villers et al., 2012). Similarly, the water use of young pine trees can be much higher than that of mature pine trees

(Alvarado-Barrientos, 2013). In view of the steep and complex topography prevailing in the Himalayas and the generally limited spatial extent of the forests (Paudel et al., 2012; cf. Dobremez, 1976), micro-meteorological techniques are less than suitable (Baldocchi et al., 1988; Raupach and Finnigan, 1997; Holwerda, 2005). Likewise, soil water depletion methods (Ward and Robinson, 1990) are likely to produce underestimates of ET in view of the fact that the trees tend to root into the underlying weathered rock in which it is difficult to monitor changes in soil water content. As such, the presently used sap flow techniques (Chapter 3) would seem to be the preferred method of choice, possibly augmented with stable isotope measurements in xylem water and soil water to determine the depth of water uptake at different times of the year (Goldsmith et al., 2012).

(ii) Runoff generation

This work quantified the surface runoff (overland flow) from three contrasting land-cover types viz: (i) little disturbed natural broad-leaved forest; (ii) a heavily used mature pine plantation established on former pasture land, and (iii) a severely degraded and overgrazed grassland (Chapter 4). Based on measurements of field-saturated soil hydraulic conductivity (K_{fs}) at the surface and with depth and combining these with information on rainfall intensities occurring at different frequencies, dominant subsurface flow paths (i.e., predominantly vertical percolation or lateral flow) were inferred (Chapters 4 and 5). However, subsurface flows were not measured in this study. Thus, the proper quantification of subsurface flows should be an important focus of future work on stormflow generation in the area (cf. Bonell, 2005; Chappell et al., 2007). In addition, the spatial and temporal extent of saturation-excess overland flow (Bonell, 2005) needs to be investigated further because no overland flow should be recorded in the near-undisturbed broad-leaved forest on the basis of prevailing maximum rainfall intensities and surface- and near-surface values of K_{fs}. Yet, overland flow was measured occasionally (Chapter 4) which may represent a contribution of saturation-excess overland flow during particularly wet conditions or simply reflect an over-estimation of K_{fs} as the measurements were conducted during the dry seasons. Values of K_{fs} may be halved at the height of the wet season (when the overland flow was measured) (cf. Bonell et el., 2010). Hillslope saturation detectors (Elsenbeer and Lack, 1996) may be used in connection with geochemical and isotope tracers-based hydrograph separation to help ascertain the relative importance of the contributions to overall storm runoff by 'old', pre-event water (usually subsurface flow) and 'new', event-water (usually overland flow) (Buttle and McDonell, 2005; Muñoz-Villers and McDonnell, 2012).

(iii) Upscaling to the catchment level

This thesis work focused on the plot scale measurement of hydrological fluxes so there remains room for speculation as to the linkage between plot scale and catchment scale results (Blöschl et al., 2007). Using the improved (quantitative) insight into the prevailing runoff generation mechanisms (cf. item (ii) above) and dry- and wet-canopy evaporation rates associated with different land-cover types (recommendation (i) above), an integrated physically-based distributed hydrological model can be used for assessing the impact of reforesting degraded lands (especially on dry season flows) at the catchment scale – provided sufficient streamflow data are available for model calibration and validation (Barnes and Bonell (2005) and references therein).

In general, there is tremendous scope for additional ecohydrological research in the Himalayas using some of the more modern techniques outlined in the previous paragraphs. Most work carried out to date (see Bruijnzeel and Bremmer, 1989; Negi, 2002 for early reviews) concerns catchment-scale comparisons but lacks the within-catchment process-based research that would be required for in-depth understanding of the observations (cf. Bonell, 2005).

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Samenvatting: Conclusies en Aanbevelingen

Dit hoofdstuk vat het proefschrift '**Hydrological impacts of reforesting degraded pasture land in the Middle Mountain Zone of Central Nepal'** samen en geeft de belangrijkste conclusies, gevolgd door enkele aanbevelingen voor vervolgonderzoek.

Aangezien bodemdegradatie in de tropen en subtropen zich geleidelijk aan uitbreidt en de seizoenale rivierafvoerpatronen in deze streken steeds meer verslechteren – met een teveel aan water gedurende de regentijd en watertekorten in de droge tijd – is het niet verwonderlijk dat de roep om grootschalig landherstel luider klinkt dan ooit. Een veel toegepaste strategie is het planten van bomen in de hoop dat de toevoer van organische stof aan de bodem in de vorm van verterend bladafval de capaciteit van de bodem om regenwater te absorberen geleidelijk aan zo zal verhogen dat daarmee de bodemvocht-huishouding ook wordt verbeterd. Echter, een aantal recente overzichten van de literatuur op dit gebied legt de nadruk op het feit dat rivierafvoeren in niet-beboste gebieden typisch sterk afnemen na het planten van bomen. Dergelijke bevindingen lijken op het eerste gezicht in tegenspraak te zijn met de traditionele opvatting dat de aanwezigheid van een goed ontwikkeld bos het hele jaar door hogere en meer stabiele rivierafvoeren te zien geeft. Hoewel het in theorie mogelijk is dat het hogere watergebruik van nieuw aangeplante bomen gecompenseerd wordt door bovengenoemde verbetering in infiltratie (de zogenaamde 'infiltratie-compensatie hypothese') – waardoor rivierafvoeren tijdens de droge tijd verbeterd zouden worden door herbebossing – heeft geen enkele studie de nodige metingen gedaan aan de onderliggende hydrologische processen om een dergelijk effect te onderbouwen. Wel is het zo dat recentelijk verschillende studies uitgevoerd op stroomgebiedsschaal een positieve trend in afvoeren hebben waargenomen na herbebossing van sterk gedegradeerde bodems in Zuid China, Zuid Korea, en Zuidwest India. Echter, door het ontbreken van aanvullende hydrologische procesmetingen blijven de onderliggende redenen voor dergelijke observaties vooralsnog onduidelijk.

Het dicht bevolkte Middengebergte in de Nepalese en Indiase Himalaya vertegenwoordigt een gebied waar veel van het oorspronkelijk aanwezige natuurlijke bos ofwel vergaand is gedegradeerd of min of meer is verdwenen vanwege zware gebruiksdruk in de vorm van begrazing, kap, en verzamelen van bladstrooisel en brandhout. In combinatie met een sterk seizoenaal regenpatroon (meer dan 80% van de jaarlijkse regen valt in de zomermaanden juni – september) heeft het sterk afgenomen vermogen van de geërodeerde bodems om de moessonregens te absorberen geleid tot wijdverbreide schaarste aan water gedurende de droge tijd. In antwoord op dit soort problemen werd een groot herbebossings-initiatief gestart in Centraal Nepal in de jaren tachtig van de vorige eeuw maar boeren in het gebied hebben sindsdien geklaagd over afnemende bronnen en rivierafvoeren.

De belangrijkste doelstellingen van het huidige onderzoek waren: (1) het beschrijven en quantificeren van de belangrijkste hydrologische processen (onderschepping van de regen door de vegetatie, wateropname uit de bodem, het genereren van oppervlakkige afvoer tijdens regenbuien, infiltratie van regen en percolatie van bodemwater naar diepere lagen) voor drie verschillende typen van landgebruik, te weten een sterk gedegradeerd grasland, een vrijwel ongestoord natuurlijk loofbos, en een intensief gebruikte dennenaanplant; en (2) het bepalen van het netto-effect van de veranderingen in enerzijds het watergebruik van de vegetatie, en anderzijds in regeninfiltratie na herbebossing (met dennen) van gedegradeerd grasland. Alle experimentele gebieden zijn gelegen op ca. 1500 m hoogte rond het stadje Dhulikhel in Centraal Nepal, zo'n 45 km ten oosten van de hoofdstad Kathmandu.

In de volgende paragrafen worden de belangrijkste bevindingen van de studie kort samengevat.

Hoofdstuk 2: Als belangrijke component van de totale verdamping van bosvegetatie is allereerst de mate van onderschepping ('interceptie') van inkomende regen door het natuurlijke loofbos en door de dennenaanplant (25 jaar oud) onderzocht. De regeninterceptie op jaarbasis door het natuurlijke bos (22.6%) was iets hoger dan die voor het dennenbos (19.4%), in overeenstemming met de structurele verschillen tussen de twee bossen in termen van boomdichtheid (1869 bomen per hectare in het loofbos tegenover 885 in het dennenbos) en het totale geprojecteerde bladoppervlak ('leaf area index': 4.5–5.4 m^2 m^{-2} voor het loofbos vs. 2.0–2.2 m^2 m^{-2} voor het dennenbos). De geciteerde jaartotalen zijn verkregen met behulp van een interceptiemodel (het zogenaamde 'revised analytical model' van J.H.C. Gash) wat een goede overeenstemming te zien gaf met tijdens de moesson gemeten interceptie-totalen zo lang geoptimaliseerde waarden gebruikt werden voor de snelheid van verdamping van onderschept regenwater vanaf de natte boomkronen.

Hoofdstuk 3: De tweede hoofdcomponent van de verdamping door een bos betreft de wateropname uit de bodem door de wortels (‘transpiratie’). Voor het eerst zijn de transpiratiesnelheden gedurende verschillende seizoenen voor de belangrijkste boomsoorten in het natuurlijke loofbos van het Himalaya Middengebergte en voor (aangeplante) dennen gemeten met behulp van in de stam aangebrachte sapstroomsensoren, naast continue metingen van de weers-omstandigheden en de bodemvochtsituatie. De transpiratiesnelheid in beide bostypen werd vooral bepaald door de vochtigheid van de lucht (‘vapour pressure deficit’), in mindere mate door de hoeveelheid beschikbare energie voor verdamping (‘straling’), en vrijwel niet door de hoeveelheid beschikbaar vocht in de eerste 75 cm van het bodemprofiel. Transpiratiesnelheden gedurende de droge tijd waren het hoogst en vertoonden opvallend weinig variatie in de tijd, hetgeen suggereert dat de wortels gedurende die tijd nog steeds toegang hadden tot vocht in diepere lagen van het bodemprofiel en in het onderliggende verweerde gesteente. Wateropname door de dennen was hoger dan die van het loofbos gedurende het gehele jaar (ook na correctie voor verschillen in inkomende straling voor de twee bossen).

Hoofdstuk 4: De capaciteit van de bodem om regen te absorberen (‘infiltratiecapaciteit’) en door te laten (‘percolatie’) wordt in dit hoofdstuk vergeleken voor het sterk gedegradeerde grasland, het nagenoeg ongestoorde natuurlijke loofbos, en voor het aangeplante dennenbos dat gedurende 25 jaar door de lokale bevolking op intensieve wijze is gebruikt in de vorm van begrazing door vee en het verzamelen van strooisel en brandhout. Daarnaast werden metingen gedaan op een veel gebruikt voetpad binnen het grasland. De hoge doorlatendheden die gevonden werden voor de bodem onder het natuurbos voorkwamen het veelvuldig optreden van oppervlakkige afstroming, ook tijdens extreme regenintensiteiten. Derhalve vindt onder intact loofbos voornamelijk verticale percolatie plaats. Daarentegen trad in het grasland en vooral op het voetpad oppervlakkige afstroming al op bij relatief lage regenintensiteiten vanwege de bijbehorende sterk verminderde infiltratiecapaciteit en doorlatendheid. Een verrassend resultaat was verder dat de infiltratie en percolatie onder het dennenbos niet noemenswaardig afwijken van de situatie onder grasland, ondanks 25 jaar bosontwikkeling. Dit is hoogstwaarschijnlijk een effect van genoemde intensieve begrazing en verwijdering van strooisel. De hoge volumes aan oppervlakkige afvoer die gegenereerd worden op het gedegradeerde grasland (21.3% van de regen) en in het dennenbos (18.6% van de netto neerslag) dragen naar verwachting sterk bij aan lokaal verhoogde piekafvoeren. De huidige resultaten onderstrepen daamee het belang van een intacte bosvloer (strooisellaag en ondergroei) voor het optimaal hydrologisch functioneren van een bos (geplant of natuurlijk).

Hoofdstuk 5: Ter ondersteuning van de rond Dhulikhel verkregen resultaten en om inzicht te krijgen in de veranderingen op lange termijn in de infiltratiecapaciteit van (bos)bodems in relatie tot langdurig intensief gebruik werden aanvullende metingen verricht in het gebied rond Chautara, zo’n 22 km van Dhulikhel. Hier werden infiltratiesnelheden en doorlatendheden gemeten in hetzelfde (gedegradeerde) grasland, loofbos, en dennenplantages als die in 1986 bestudeerd werden door D.A. Gilmour en collega’s en met gebruik van dezelfde meettechnieken. Op deze wijze werd een unieke directe vergelijking mogelijk van de situatie in 1986 en 2011 waardoor de gezamenlijke effecten van intensief landgebruik over een periode van 25 jaar konden worden gequantificeerd. Abrupte afnames in de doorlatendheid werden vastgesteld voor de eerste 25 cm van het bodemprofiel onder zowel het natuurlijke bos als onder de dennenaanplant, maar niet onder het grasland waar de mate van bodemdegradatie klaarblijkelijk reeds een ‘plafond’ had bereikt in 1986. De bodemverslechtering in de onderzochte bostypes kon gerelateerd worden aan hetzelfde intensieve soort gebruik als eerder gesignaleerd in de bossen rond Dhulikhel. De voor het gebied rond Chautara verkregen resultaten bevestigen nogmaals het feit dat bij het beheer van bossen door lokale gemeenschappen in het Middengebergte hydrologische aspecten meer aandacht verdienen. Het simpelweg aanplanten van bomen op gedegradeerde gronden is op zich niet voldoende om het hydrologisch functioneren te herstellen, de aanplant dient ook duurzaam te worden beheerd.

Hoofdstuk 6: In dit hoofdstuk worden de bevindingen uit de eerdere hoofdstukken geïntegreerd om zo het netto effect van het herbebossen van gedegradeerd grasland met snelgroeiende dennen op de waterbalans te evalueren. Het aanplanten van de dennen deed het waterverbruik van de vegetatie toe nemen met 355 mm per jaar ten opzichte van het grasland. Tegelijkertijd was de winst in termen van de hoeveelheid extra geïnfiltreerd water onder dennenbos met 90 mm per jaar een stuk minder dan verwacht, zodat het netto resultaat van de aanplant negatief is en tot geringere grondwateraanvulling en rivierafvoeren zal leiden. Het huidige resultaat komt grotendeels voor rekening van de beperkte bodemverbetering onder dennen maar zelfs als alle netto neerslag had kunnen infiltreren (bij voorbeeld door een beter beheer van de aanplant) dan nog zou dit onvoldoende zijn geweest om de hoge verdamping van de dennen te compenseren. Daarentegen suggereert een vergelijking van het verschil in waterverbruik van het natuurlijke bos en het grasland (ca. 300 mm per jaar) met de geschatte winst aan infiltratie (ca. 285 mm per jaar) dat vervanging van het gedegradeerde grasland door een goed beheerd natuurlijk loofbos uiteindelijk een min of meer neutraal effect zal hebben op de afvoeren. De gevonden resultaten bevestigen wederom het belang van een goed beheer c.q. bescherming van de bossen van het Middengebergte voor het optimaal hydrologisch functioneren van het gebied. De grote uitdaging voor het beheer van de bossen in de streek ligt in het combineren van de belangen van de lokale bevolking met het garanderen van een optimaal ecologisch en hydrologisch functioneren van de bossen.

Aanbevelingen voor nader onderzoek

Het huidige onderzoek vertegenwoordigt een gedetailleerde studie naar de hydrologische gevolgen op lokale schaal van het aanplanten van snelgroeiende dennen op sterk gedegradeerde gronden in het Middengebergte van Centraal Nepal. Vanwege financiële en logistieke beperkingen was het niet mogelijk om dergelijke intensieve metingen op meerdere plekken te herhalen voor elk van de drie onderzochte landgebruikstypen. Ook herhalingen in de tijd (bij voorbeeld in dennenopstanden van verschillende leeftijd) waren niet haalbaar. Tenslotte, hoewel er afvoermetingen zijn uitgevoerd in het kader van het huidige project aan de monding van het Dhulikhel-stroomgebied (14 km^2), zijn deze gegevens niet gebruikt in bovenstaande analyse omdat het stroomgebied een groot aantal typen van landgebruik (ook vormen van landbouw) kent en de gemeten afvoeren niet direct aan de bevindingen op de schaal van de bemeten onderzoeksplots gekoppeld kunnen worden. Er is daarom volop ruimte voor aanvullend onderzoek in dit verband. Toekomstig werk kan handzaam worden ondergebracht onder de volgende drie noemers: (i) waterverbruik van verschillende vegetatietypes, (ii) het genereren van versnelde oppervlakkige en ondergrondse afvoer tijdens en vlak na regenbuien, en (iii) het opschalen en integreren van deelresultaten verkregen op plotniveau naar gehele stroomgebieden. In het volgende worden deze drie thema's kort besproken.

(i) Vegetatie en waterverbruik

Dit onderzoek heeft de totale verdamping (interceptieverdamping plus transpiratie) gequantificeerd voor een 36-jaar oude dennenopstand en een natuurlijk loofbos (climaxvegetatie). Het waterverbruik van beide bostypen was bescheiden (zie hoofdstuk 3), wat opgevat kan worden als een evolutionaire aanpassing van de onderzochte boomsoorten om te kunnen overleven onder de heersende sterk seizoenale klimaatsomstandigheden (lange droge tijd). Echter, zonder herhalende experimenten op andere plaatsen in de regio of bevestiging middels op andere wijze bepaalde verdampingswaarden moeten de huidige resultaten met enige voorzichtigheid worden benaderd. Aanvullende plantenfysiologische metingen zijn dan ook wenselijk, zowel voor de nu onderzochte vegetaties als voor regenererend bos op verlaten akkers en aangeplante bossen in verschillende groeistadia. Dergelijk werk is van belang omdat er aanwijzingen zijn dat het waterverbruik van snelgroeiende jonge vegetatie die van oudere bomen sterk kan overtreffen, met alle gevolgen van dien voor de aanvulling van grondwatervoorraden en derhalve voor de grootte van rivierafvoeren. Omdat de meeste bosopstanden in het sterk versneden terrein van de Himalaya slechts een beperkt oppervlak beslaan zijn micrometeorologische technieken om het waterverbruik te schatten minder geschikt vanwege een gebrek aan equilibrering van de passerende lucht (te korte 'fetch'). Ook de toepassing van bodemwaterbalanstechnieken is beperkt vanwege het feit dat de wortels van de vegetatie doordringen tot in diepere (gesteente)lagen waarin het niet mogelijk is om de bij onttrekking behorende veranderingen in de hoeveelheid vocht te volgen, met als mogelijk gevolg onderschatte verdampingssnelheden. Als zodanig zijn de in het huidige onderzoek gebruikte plantenfysiologische meettechnieken te verkiezen, eventueel in combinatie met de analyse van de gehaltes aan stabiele isotopen in bodemvocht en spinthout / xyleem om te bepalen van welke diepte het in de stam getransporteerde vocht afkomstig is gedurende de verschillende seizoenen.

(ii) Landgebruik en het ontstaan van piekafvoeren

In deze studie is de productie van oppervlakkige afvoer tijdens regenbuien bepaald voor ongestoord natuurlijk bos, een intensief gebruikte dennenopstand, en een sterk gedegradeerd grasland (hoofdstuk 4). Op basis van een vergelijking van voorkomende regenintensiteiten en doorlatendheden voor zowel oppervlakkige als diepere bodemlagen was het mogelijk om de richting van de meest voorkomende vorm van watertransport door de bodem af te leiden (d.w.z. vooral verticaal of juist meer lateraal) in elk van deze gevallen (zie hoofdstuk 4 en 5). Zo werd onder meer voorspeld dat in het geval van het vrijwel niet verstoorde loofbos de kans op laterale oppervlakkige afvoer miniem was. Echter, de ondergrondse patronen van de waterstroming tijdens en vlak na regenbuien zelf zijn niet gemeten. Derhalve vormt de quantificering van snelle ondergrondse afvoeren een belangrijk onderdeel van toekomstig onderzoek naar de relaties tussen regenval, landgebruik, bodemtype en de hoogte van piekafvoeren in het studiegebied. Daarnaast is het niet duidelijk of de waargenomen (geringe) hoeveelheid oppervlakkige afstroming in het ongestoorde bos het gevolg is van zogenaamde ‘saturation overland flow’ (verzadiging van onderaf in het bodemprofiel waarna de regen niet langer de grond in kan dringen en gedwongen wordt langs het oppervlak af te stromen). Het is ook mogelijk dat deze discrepantie tussen waarneming en voorspelling het gevolg is van een overschatting van de bodemdoorlatendheid, bij voorbeeld omdat deze metingen in het droge seizoen plaats vonden en de werkelijke waarden in de regentijd een stuk lager uitvallen. Het plaatsen van simpele verzadigingsdetectoren op verschillende posities langs de hellingen en het gebruik van hydrochemische parameters en stabiele isotopen als tracers om de relatieve bijdragen van de verschillende watertypen (zowel oppervlakkig als ondergronds) aan de piekafvoeren in de beken te bepalen met behulp van zogenaamde mengmodellen zou tot meer helderheid in deze moeten leiden.

(iii) Integratie op stroomgebiedsniveau

Aangezien het voorliggende onderzoek naar de grootte van de verschillende waterstromen in relatie tot landgebruikstype plaats vond op perceelsniveau, is er enige ruimte voor speculatie met betrekking tot de effecten van de onderzochte landgebruiksveranderingen op stroomgebiedsschaal. Een beter inzicht in de heersende mechanismen die de hoogte van de piekafvoeren bepalen (zie voorgaande paragraaf) en aanvullende informatie over het waterverbruik van andere dan de hier onderzochte vegetatietypes (bij voorbeeld regenererend bos, jongere dennenopstanden – zie voorgaande paragraaf) kunnen in de toekomst worden gebruikt in een ruimtelijk expliciet hydrologisch procesmodel om de effecten van herbebossing op verschillende schalen binnen een stroomgebied te evalueren (in combinatie met de voor model-ijking en validatie benodigde gegevens voor gebiedsneerslag en (trends in) rivierafvoeren).

In zijn algemeenheid kan gesteld worden dat het potentieel voor verder ecohydrologisch veldonderzoek in de Himalaya enorm is. Enerzijds omdat de discipline in de regio nog in de kinderschoenen staat en anderzijds omdat eerder werk zich vooral heeft geconcentreerd op de productie van water en sediment op stroomgebiedsschaal zonder zich al te veel te richten op de onderliggende ecologische, hydrologische en geomorfologische processen. Door deze processen expliciet in toekomstig onderzoek op te nemen kan naar verwachting snel vooruitgang worden geboekt met betrekking tot ons begrip van het hydrologisch functioneren van de belangrijkste ecosystemen in (met name) het dichtbevolkte Middengebergte van de Himalaya.

सारांश

विश्वका खास क्षेत्रहरूमा बढिरहेको भूक्षयीकरण र बर्खाका बेला सीमा नाघेर आउने बाढीका उत्पातहरू एवम् सुख्खा गर्मीयाममा बढिरहेको पानीको अभावका सन्दर्भमा जमिनको व्यापक पुनःस्थापनाको आवश्यकता पहिलेभन्दा झनै बढेर गएको छ । जमिनको पुनःस्थापनाका लागि व्यापक रूपमा प्रयोग हुँदै आएको एउटा रणनीति रुख रोप्नु (वृक्षारोपण) हो, जसबाट के आशा गरिन्छ भने यसले जङ्गलको सतहमा प्राकृतिक तत्वहरूको वृद्धि गर्छ, जसको परिणामस्वरूप माटोको पानी सोस्ने क्षमता (infiltration capacity) मा उल्लेख्य वृद्धि हुन्छ र सँगै माटोको आर्द्रता पुनःस्थापित हुने प्रक्रियामा सुधार आउँछ । यद्यपि केही पछिल्ला शोधहरूका समीक्षाले के तथ्यमा जोड दिएका छन् भने कृषिबाली लगाइएका क्षेत्र, घाँसे क्षेत्र र खाली जमिनअन्तर्गतका ठाउँहरूमा वर्षभरिको पानीको अल्पताको अवस्थाको तुलना गर्दा जङ्गल नभएका उजाड ठाउँहरूमा वृक्षारोपण गरेपछि पानीको उपलब्धतामा केही सकारात्मकता थपिएको पाइन्छ । यस्ता निष्कर्षहरू परम्परागत र व्यापक रूपमा मानिँदै आएका दृष्टिकोणसँगसँगै अनौठा कुरा प्रतीत हुन जान्छन्, जसले के दाबी गर्छन् भने घना जङ्गलको आवरणले मौसमचक्रको प्रवाहलाई अझ बढी स्थिर र उच्च बनाउँछ । र, जबकि यो कुरा सैद्धान्तिक रूपमा सम्भव छ कि भर्खरै रोपिएका रुखहरूमा माटोको सुधारमार्फत् उन्नत किसिमको माटोको पानी सोस्ने प्रक्रिया (Infiltration) द्वारा अवलम्बन गरिएको पानीको अधिक प्रयोग रुख रोपेपछि एकदमै उजाड जमिनमा सुख्खा मौसममा समेत सक्कली प्रवाह पुनर्भण्डारण गर्नका लागि क्षतिपूर्ति हुन सक्छ, तर यस विषयमा यस्ता किसिमका सकारात्मक उपलब्धिहरूलाई सहयोग पुर्‍याउने कुनै एकल अध्ययनले जलविज्ञानसम्बन्धी पद्धतिको मापनको आवश्यकतालाई निर्माण गर्न सकेको छैन । हालसालै केही फुटकर किसिमका अध्ययनहरूबाट दक्षिण र दक्षिणपूर्व एसियाका नाङ्गा जमिनहरूमा पुनः रुख रोपेपछि बाढीका सन्दर्भमा एकखाले सकारात्मक प्रवृत्तिको आँकलन गरिएको छ, तर यस्ता अध्ययनहरू सहयोगात्मक प्रणालीका मापदण्डहरूको कमीका कारण 'ब्ल्याक बक्स' शैलीका केही अवशेष मात्र हुन् ।

नेपाली र भारतीय हिमालय क्षेत्रको सघन जनघनत्व रहेको मध्यपहाडी भूभागले अर्को क्षेत्रको प्रतिनिधित्व गर्छ, जहाँ सक्कली जङ्गलको आवरणको अत्यधिक अंश तीव्र मानवीय दबाबका कारण या त नाङ्गो भैसकेको छ, या लोप भएको छ । मौसमी वर्षाका नियमहरूसँग माटोमा विद्यमान मौसमी वर्षालाई ग्रहण गर्ने क्षमता क्षय हुँदै गएको अवस्थालाई दृढतापूर्वक संयोजित गर्दा विस्तृत हुँदै गएको सुख्खायामको अवधिमा पानीको व्यापक भण्डारणको आवश्यकतातर्फ धकेलिरहेको देखिन्छ। यी दबाबहरूलाई सम्बोधन गर्ने सन्दर्भमा पुनःवृक्षारोपणको एउटा ठूलो कार्यक्रम सन् १९८० को दशकमा मध्य नेपालको मध्यपहाडी भूभागमा कार्यान्वयन गरिएको थियो, तर किसानहरूले अद्यापि बर्खामा मूल फुट्ने कुरा घटिरहेको भनेर गुनासो गरिरहेका छन् ।

यसरी प्रस्तुत अध्ययनको मुख्य उद्‌देश्य दुई तहको थियो । यसको पहिलो उद्‌देश्य प्रमुख भूआवरणका तीन खाले प्रतिकूलित तरिकाहरूमा उल्लेख्य रूपमा बढिरहेको मानवसम्बद्ध दबाबको स्तर : (१) लगभग समस्यारहित प्राकृतिक ठूलो पाते जङ्गल, (२) औपचारिक रूपमा जङ्गल फाँडिएको खुरिल्लो जमिनमा राम्ररी प्रयोग गरिएको परिपक्व pine (सल्लो) को वृक्षारोपण र (३) खुरिल्लै हुने गरी उजाड र अधिक चरन भएको जमिन, यी सबै धुलिखेल नजिकैको नेपालको मध्यभूभागमा अवस्थित छन् । विषयसँग सञ्चालित जलविज्ञानसम्बन्धी पद्धतिहरू (वर्षामा रुकावट, वायुसञ्चरण, ऊर्जाको आकस्मिक विच्छेद, वर्षामा माटोको पानी सोस्ने पद्धति र तरलता) को व्याख्या एवम् परिमाणको मापन थियो । यस

अध्ययनको दोस्रो उद्देश्य सामान्य वनस्पतिहरूमा पानीको प्रयोग र उजाड किसिमका खुरिल्ला जमिनमा पुनःवृक्षारोपणपछि माटोको पानी सोस्ने क्षमतामा आएको परिवर्तनबीचको ठोस भिन्नताको खोजी गर्नु थियो ।

अनुसन्धानका निष्कर्षहरूको सारांश

(क) आर्द्र-आवरणको वाष्पीभवन

अध्याय २ : पहिलो चरणका रूपमा वर्षात्को पानी रुखको पातमा अडिने प्रक्रिया (आर्द्र-आवरणको वाष्पीभवन) का साथसाथै जङ्गलको संरचनागत मापदण्डसँग मनसुनको स्तरमा वर्षात्को अंशदानलाई मिलाएर प्राकृतिक ठूलो पाते जङ्गल एवम् २५ वर्षअघि वृक्षारोपण गरिएको Pine (सल्लो) को जङ्गलमा आवरणको तह मापन गरिए । यी निष्कर्षहरूले सदाबहार प्राकृतिक जङ्गलबाट वातावरणको चिसोपन खिच्नसक्ने शक्ति ह्रास भएको (आकस्मिक वर्षात्को २२.६ प्रतिशत) देखाए, वृक्षारोपण गरिएको सल्लाको जङ्गलका तुलनामा (१९.४ प्रतिशत) । प्राकृतिक जङ्गलका सन्दर्भमा वर्षात्को पानी रुखको पातमा अडिने प्रक्रियामा आएको उच्च ह्रासले जङ्गलको जगेडापनको भिन्नता (प्राकृतिक जङ्गलमा प्रतिहेक्टर १८६९ रुखहरूका सट्टा सल्लाका जङ्गलमा प्रतिहेक्टर ८८५ रुखहरू) र LAI (प्राकृतिक जङ्गलमा ४.५-५.४ $m^2\ m^{-2}$ का सट्टा सल्लाको जङ्गलमा २.०-२.२ $m^2\ m^{-2}$) तर्फ डोर्‍यायो, जुन कुरा आशा गरेअनुसारकै थियो । वर्षात्को पानी रुखको पातमा अडिने प्रक्रियासम्बन्धी पुनरावलोकित विवेचनात्मक पद्धति लाई पहिलो पटक प्रयोग गरिएको थियो, मौसमी montane अवस्थाहरूअन्तर्गत वर्षात्को पानी रुखको पातमा अडिने प्रक्रियासम्बन्धी वार्षिक अनुमानहरू प्राप्त गर्नका लागि । यसमा दुई विपरीत जङ्गलहरूका लागि नमुनाकृत र परीक्षणीय पानी रुखको पातमा अडिने प्रक्रियाका ह्रासहरूका बीचमा राम्रो समझदारी थियो र आर्द्र आवरणको वाष्पीभवन दरका लागि उपलब्ध रहेका राम्रा सूत्रहरू प्रयोग गरिएका थिए ।

ख) सुख्खा-आवरण (मण्डल) को वाष्पीभवन

अध्याय ३ : पहिलो पटक वृक्षहरूको जल निष्कासन, आवरण वा खोलको आयोजन र पृथक् तुल्याउने गुणाङ्कहरू मात्रात्मक तुल्याइए र हिमाली क्षेत्रको मध्य पहाडी भूभागमा अवस्थित प्राकृतिक ठूला पात हुने जङ्गलका साथै वृक्षारोपण गरिएको परिपक्व सल्लाको जङ्गलमा परीक्षण गरियो, क्षयीकरणको श्रृङ्खलाका मापकहरू र तात्कालिक मौसमसम्बन्धी एवम् माटोको जलविज्ञानसम्बन्धी परीक्षणहरूलाई प्रयोग गरेर । दुवै जङ्गलहरूमा रुखको श्वासप्रश्वास प्रणाली (Transpiration) का दरहरूमाथि पर्यावरणीय वाष्प-चापको न्यूनताले जोडदार रूपमा प्रभुत्व जमाइएको थियो, वाष्पीभवनका लागि उपलब्ध ऊर्जाको राशिभन्दा निकै कम र माटोको जलविज्ञानसम्बन्धी अन्तर्वस्तुहरूभन्दा सर्वाधिक कम, अर्थात् ७५ से.मि.गहिरोभन्दा थोरै । पछि के कुरा पनि पत्ता लागेको थियो भने यी दुई जङ्गलहरूमा रुखको श्वासप्रश्वास प्रणाली (Transpiration) का दरहरू सुख्खा मौसममा चरम अवस्थामा पुगे र सुख्खा मौसमको अधिकांश समयमा सामान्य भिन्नता देखापर्‍यो, माटोको गहिरो तहमा विद्यमान आर्द्रता र भूमिको उर्वर तहको विनाशसँग सम्बन्धित अंशका निम्ति पुग्नै पर्ने मूल कुरालाई औँल्याउँदै । खुल्ला किसिमको र विकिरणभारको भिन्नतालाई समेत अनुमति दिँदा पनि प्राकृतिक

जङ्गलको तुलनामा सल्लाको जङ्गलमा मौसमी र वार्षिक गरी दुवै खालका रुखको श्वासप्रश्वास प्रणाली (Transpiration) को कूल समग्रता उल्लेख्य रूपमै उच्च देखियो ।

(ग) पुनः वृक्षारोपण, माटाको जलविज्ञानसम्बन्धी सुसञ्चालन र पहाडी भिरालो भूभागको जलविज्ञानसम्बन्धी प्रतिक्रिया

अध्याय ४ : खुरिल्लै हुने गरी उजाडभएको पूर्णतया खँदिलो घाँसेभूभागमा पुनः वृक्षारोपणको प्रभावका साथै जलविज्ञानसम्बन्धी सुसञ्चालन K_{fs} र सतहमाथिको प्रवाहले गर्ने उत्पादनको परीक्षण गरियो । अहिले विद्यमान पर्याप्त रूपमा नयाँ K_{fs} तथ्याङ्कबाहेक निकै उजाड चरिचरन, केही समस्याग्रस्त ठूला पात हुने जङ्गल र उल्लेख्य रूपमै प्रयुक्त सल्लाको वृक्षारोपण र पहाडी भिरालो भूभागको श्रृङ्खला-मापनको श्रेणीलाई विस्तृत तुल्याउँदै, यस अध्यायले हिमालयक्षेत्रमा एकदमै निर्बाध रूपमा प्रयोग भएको ग्रामीण पैदलबाटोको K_{fs} मा आधारित पहिलो मापनलाई समेत समाविष्ट गरेको छ । प्राकृतिक जङ्गलमा अवलोकन गरिएको उच्च सतही एवम् आसन्न सतही K_{fs} ले बृहत् किसिमका माटोको पानी सोस्ने पद्धतिसँग सम्बन्धित सतही प्रवाह (IOF) का घटनालाई, अझ अति सीमान्त वर्षाका घटनाहरूलाई समेत प्रभावकारी रूपमा अवरुद्ध गर्‍यो । यसरी अक्षुण्ण प्राकृतिक जङ्गलले माटोमा सिधै तल पानी छिर्ने प्रक्रियाको पक्षपोषण गर्छ र माटोमा रहेको पानीको शोधभर्ना गरी अत्यधिक माटोको पानी सोस्ने पद्धतिद्वारा जमिनमुनिको पानीलाई कायम राख्छ । यस विपरीत उजाड चरिचरन, खासगरी पैदल खण्डमा, मझौला खाले वर्षाका तीव्र घटनाहरूका अवधिमा समेत निकै कम सतही एवम् आसन्न सतही K_{fs} पाइएको थियो, जसले IOF को उत्पादनलाई उत्प्रेरित गर्छ । उपयुक्त रूपमा सल्लाका जङ्गलमा उच्च रूपमा प्रयोग गरिएको सतही र आसन्न सतही K_{fs} अवलोकन गरिएका खुरिल्ला चरिचरनहरूसँग २५ वर्षसम्मको जङ्गल विकासपछि पनि समान किसिमले स्थिर रहेको थियो । यस्तो अनपेक्षित निष्कर्षले सूचित गरेको थियो, जङ्गलको सतहबाट जनावरलाई ख्वाउन र कम्पोस्ट मल बनाउनका लागि सोत्तर आदिको नियमित सङ्कलन, तलेत्री उप्काउने काम, चौपायाको चरन र दाउरा खोजाइलाई । पैदलमार्गमा जमिनको सतहमाथि उत्पादित प्रवाहका ठूला परिमाणहरू (मापन नगरिएको, तर वर्षाको दर र सतहको K_{fs} को तुलनाद्वारा अनुमान गरिएको), खुरिल्लै पारिएको चरिचरन (आकस्मिक वर्षाको २१.३ प्रतिशत) मा र अध्ययन क्षेत्रको वृक्षारोपण गरिएको (Crown drip अर्थात् ठूला थोप्ले १८.६ प्रतिशत) क्षेत्रमा, जसबाट स्थानीय स्तरको खोलाको बहावमा नगण्य योगदानको आशा गर्न सकिन्छ । यी परिणामहरूले राम्ररी विकास भएका बाहिरी सतह र तलेत्रीका वनस्पतिको तहमा सकारात्मक हस्तक्षेपलाई पुनः व्याख्या गर्छन् (प्राकृतिक जङ्गलमा आकस्मिक वर्षाको केवल २.५ प्रतिशत IOF र ठूला ठूला थोपाको ३.२ प्रतिशत), सतह र उपसतहको जलविज्ञानमा । यिनीहरूले जलाधारस्रोतका रूपमा बाँकी रहेको नेपाल र भारतीय हिमाली क्षेत्रको मध्यपहाडी खण्डको पुरानो प्राकृतिक जङ्गलको संरक्षणको जलविज्ञानसम्बन्धी महत्वलाई समेत उजागर गर्दछन् ।

(घ) जङ्गलको दिगो प्रयोग र माटोमा विद्यमान जलांशको सुसञ्चालन

अध्याय ५ : अध्याय चारमा व्याख्या गरिएका धुलिखेल क्षेत्रसँग सम्बद्ध निष्कर्षहरूका लागि थप सहयोग उपलब्ध गराउन एवम् जमिन र जङ्गलको उल्लेख्य प्रयोगबाट अनुभव भएका पूरै भिजेको माटोमा जलको सुसञ्चालन (K_{fs}) मा दीर्घकालिक परिवर्तनहरूसँग सम्बद्ध कुराहरूबारे 'सही समय' मा यथार्थ ज्ञान प्राप्त गर्न, खुरिल्लै हुने गरी उजाड भएको चरिचरन तल, एउटा समस्याग्रस्त प्राकृतिक जङ्गलमा र अन्य उल्लेख्य रूपमै प्रयोग भएका अर्थात् एकदमै राम्रो गरी *Pinus Roxburghii* रोपिएका मध्य पहाडी भूभागको चौतारा (धुलिखेलबाट भण्डै २० कि.मि.) नजिकैबाट यताउता पर्ने दुई वटा ठाउँमा समेत K_{fs} नापिएको थियो । यहाँ १९८६ मा मापनहरूको उस्तै श्रेणी निर्माण गरिएको थियो र २०११ मा सोही ठाउँमा ती मापनहरूलाई पुनः दोहोर्‍याएर र तुलनात्मक पद्धतिलाई प्रयोग गरेर २५ वर्षको समयान्तरका दुई भिन्न समय-बिन्दुहरूको K_{fs} सम्बन्धी प्रत्यक्ष र नवीन तुलना प्राप्त गरिएको थियो । माथि उल्लेख गरिएका चार वटै ठाउँमा सतह र एक मिटर गहिराई तलको समेत गरी दुवै किसिमको K_{fs} मापनका विविध विधि बनाइएका थिए । प्राकृतिक र पछि रोपिएका जङ्गलहरूमा २५ वर्षसम्मको विविध मानवीय गतिविधिपछि माटोको माथिल्लो सतहको ०.२५ मिटरका K_{fs} विवरणहरूमा आएका आकस्मिक र उल्लेख्य न्यूनताहरू अवलोकन गरिएका थिए । यी न्यूनताहरूले स्थूल छिद्रहरूको ह्रास, विशेषगरी जङ्गलको सतहबाट पटक पटक हुने गरेको सोत्तर संकलन र निरन्तरको चौपाया चरनका कारण माटामा रहने जैविक तत्त्वमा आएको कमीलाई प्रतिबिम्बित गर्छन् । जङ्गल क्षेत्रको अर्कोपट्टि, अध्ययन गरिएका उजाड चरिचरन स्थलले हस्तक्षेप गरिँदै आएका २५ वर्षसम्म सतहको K_{fs} मा भएको सामान्य परिवर्तनलाई देखाए, जसबाट के देखियो भने सतहको विनाशको अधिकतम बिन्दुलाई १९८६ वा त्यो भन्दा पहिले नै छोइसकेको थियो भन्ने कुरा स्पष्ट हुन पुग्यो । यी निष्कर्षहरूले देखाए कि नेपालको मध्य पहाडी क्षेत्रको सामुदायिक वनहरूको व्यवस्थापनले जलविज्ञानसम्बन्धी क्रियाकलापलाई समाविष्ट गर्नु आवश्यक छ । खुरिल्ला जमिनमा वृक्षारोपण गर्नु मात्र उजाड भूभागमा जलविज्ञानसम्बन्धी क्रियाकलापलाई पुनः स्थापित गर्नका लागि स्वयंमा पर्याप्त काम हुन सक्दैन । वृक्षारोपण गरिएका जङ्गल क्षेत्रहरूमा पर्याप्त जलविज्ञानसम्बन्धी क्रियाकलापसहित स्थानीय जनताहरूद्वारा गरिने जङ्गलको प्रयोगलाई सन्तुलित तुल्याउन वर्तमान व्यवस्थापनका सम्बन्धमा यथोचित ध्यान दिनु आवश्यक छ ।

(ङ) पुनः वृक्षारोपण र सुख्खा मौसमको प्रवाह

अध्याय ६ : सल्लाका वृक्षहरू रहेको क्षेत्रका साथै खुरिल्लो र उजाड जमिनमा पुनः वृक्षारोपणपछि वनस्पतीय जलप्रयोग र माटोको पानी सोस्ने प्रक्रिया (Infiltration) बीचका परिवर्तनहरूको समग्र प्रभावको खोजी गरिएको थियो । यसो गर्दा २ देखि ५ अध्यायसम्म प्रस्तुत गरिएका जलविज्ञानसम्बन्धी प्रणालीका मापकहरू जलसन्तुलन (Water Budget) को ढाँचामा समाविष्ट गरिएका थिए । चरिचरनका सापेक्षतामा सल्लाको वृक्षारोपणले वनस्पतीय जलप्रयोगलाई ३५५ मि.मि. प्रतिवर्षका दरमा बढायो । सन्तुलित रूपमा, उजाड जमिनका सापेक्षतामा सल्लाको वृक्षारोपणद्वारा बहन गरिएको अतिरिक्त माटोको पानी सोस्ने प्रक्रियाको सीमित अंश (सोत्तरको नियमित संकलन र रोपिएका वृक्षमा तलेत्रीका वनस्पतिसँग जोडिएको र निरन्तर हुँदै आएको माटोको विनाशका कारण केवल ९०मि.मि. प्रतिवर्ष) सल्लाहरूमा हुने अत्यधिक जलप्रयोगको क्षतिपूर्ति गर्न अपर्याप्त हुने कुरा प्रमाणित गर्‍यो ।

विपरीत रूपमा, प्राकृतिक जङ्गल र खुरिल्लो जमिनमा जलप्रयोगको तुलनाबाट के सुझाव प्राप्त हुन्छ भने परिपक्व ठूला पाते जङ्गलद्वारा पछिल्लोलाई पुनःस्थापित गर्नुले अन्ततः सुख्खा मौसमको प्रवाहमा लगभग तटस्थ प्रभाव हासिल हुन सक्छ, जहाँ माटोको पानी सोस्ने प्रक्रियामा निकटस्थ प्राप्ति (२८५ मि.मि. प्रतिवर्ष) र वाष्पीभूत ह्रासहरू (३०० मि.मि. प्रतिवर्ष) निकै समान थिए । यसरी प्रस्तुत अध्ययनका निष्कर्षहरूले अधिकतम जलविज्ञानसम्बन्धी क्रियाकलापसँगै यस क्षेत्रका बाँकी बँचेका प्रमुख जलाधार जङ्गलहरूको संरक्षणको आवश्यकता र जङ्गलको यथोचित व्यवस्थापन (प्राकृतिक र वृक्षारोपण गरिएका दुवै खाले जङ्गल) को आवश्यकतालाई फेरि एक पटक उजागर गर्छन् ।

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