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Complex land cover change, water and sediment yield in a degraded Andean environment

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SUMMARY

Rapid land use/-cover change has increasingly transformed the hydrological functioning of tropical Andean ecosystems. The hydrological response to forest cover change strongly depends on the initial state of the ecosystem. Relatively little is known about human-disturbed ecosystems where forest plantations have been established on highly degraded land. In this paper, we analyze the impact of forest change on water and sediment fluxes for a highly degraded Andean catchment. Different pathways of land cover change (1963–2007) are observed in the Jadan catchment, with deforestation taking place in remote uplands and recovery and reforestation in the middle and lower parts where agricultural and bare lands are prevalent. Time series analyses of streamflow and rainfall data (1979/1982–2005/2007) show significant shifts in the distribution of rainfall and flow data. Changes in discharge are not resulting from changes in precipitation, as the direction of change is opposite. The removal of native forest for rangeland or croplands (by -20 km^2) is likely to have contributed to the increase in total annual water yield, through an increase in annual baseflow by 25 mm. The observed changes in peakflow are important as the 1st percentile highest flow rates were 54% lower, while the 1st percentile rainfall amounts increased by 52%. The observed decrease in peakflow cannot be explained by clearcut of native forest, but is likely to be related to reforestation of degraded lands as well as spontaneous recovery of vegetation on remaining grazing lands. Over the same time period, a major decrease in specific sediment yields and suspended sediment loads was observed. Although deforestation in the upper parts led to increased landslide activity, this change is not reflected in an increased sediment yield. Small upland rivers are often nearly completely blocked by landslide material, thereby reducing their potential to transport sediment. In contrast, the reduction in estimated erosion is likely to be caused by the reduction of the degraded areas in areal extent as well as to the (partial) recovery of the vegetation in these areas.

1. Introduction

The Andean headwater basins function as important regulators of the water and nutrient supply to the downstream reaches of the Amazon River. Interactions between human activities (e.g. infrastructure, agriculture and land use change) and the physical environment have increasingly transformed the hydrological functioning of Andean ecosystems (Vanacker et al., 2003a; Podwojewski et al., 2002). In these human-modified landscapes, land use/-cover change may have a profound effect on riverine water and sediment fluxes (Harden, 2006; Vanacker and Govers, 2007a; Little et al., 2009). The hydrological impacts of land use/-cover change are diverse, as changes in vegetation affect the various components of the hydrological cycle including evapotranspiration, infiltration and surface runoff (Costa et al., 2003; Bruijnzeel, 2004).

The effect of forest cover on water yields has been demonstrated for small (mostly experimental) catchments where the natural vegetation was removed and/or replaced by plantation forests (Buytaert et al., 2006; Bosch and Hewlett, 1982; Bruijnzeel, 1990, 2004). Previous studies have shown an increase in streamflow after massive removal of tall vegetation on the catchment slopes, and a decrease in streamflow after afforestation. Ruprecht and Schofield (1989) showed that the conversion of native forest into agricultural land increased streamflow as result of decreased transpiration and interception. Exotic tree plantations on natural grasslands (such as afforestation with *Eucalyptus gl.* in Sikka et al. (2003)), or conversions from native forest to exotic plantations (such as *Pinus radiata* in Little et al. (2009)) were reported to have severe impacts on water yields with strong reduction of low flows and summer runoff. However, Scott and Prinsloo (2008) suggested that the

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longer-term effects of exotic plantations on water yield are not necessarily as harmful as predicted from shorter-term studies. Their data show that the streamflow can be reversed to pre-afforestation levels when the plantations reach maturity.

Various authors suggested that the impact of forest cover change strongly depends on the initial state of the ecosystem (Scott et al., 2005; Chazdon, 2008; Hofstede, 2011). For the paramo ecosystem (high altitudinal grasslands), there exists detailed information from short-term experimental studies on the impact of forest plantations on the hydrological response. Buytaert et al. (2006, 2007) suggested that *Pinus patula* afforestation reduced the water yield by about 50% (or an average of 242 mm y^{-1}) based on (1–3 years) of hydrological data from four experimental catchments. Farley et al. (2004) showed that conversion of natural grasslands to Pinus plantation on volcanic soils significantly reduced the water retention capacity of Andosols. In contrast, relatively little is known about disturbed ecosystems where forest plantations have been established on highly degraded land (Wilk et al., 2001; Bruijnzeel, 2004).

Some field-based information exists from plot and controlled flow experiments in reforested gully systems. Rainfall-runoff experiments at experimental plots (ranging from a few cm^2 to 1 m^2) in the tropical Andes showed that land use strongly controls hydrological processes, with significantly higher cumulative runoff coefficients on abandoned and fallow land compared to recently plowed cultivated fields (Harden, 1996; and confirmed by Molina et al. (2007)). When controlling for land use type, the surface vegetation cover determines runoff generation, and a sharp exponential decrease of overland flow generation was observed after revegetation of abandoned and fallow land (Molina et al., 2007). Garcia-Ruiz et al. (2005) showed that the hydrologic response of a highly degraded experimental catchment (284 ha) in the Central Spanish Pyrenees is highly variable, due to the existence of different runoff-generating areas (related to different land cover types and covers). Particularly, the reforestation of gully areas can have major impacts on the hydrological connectivity of degraded catchments. Controlled flow experiments in steep gully channels in the tropical Andes showed that the transfer of overland flow and sediment from the slopes towards the river system highly decreases with the presence of vegetation in the gully channels (Molina et al., 2009b).

While these data provided valuable information on how revegetation may affect water fluxes, they are insufficient to predict the hydrological and sediment response at larger scales. An important reason for this is that changes in forest cover over large areas are often not unidirectional. A prime example of the complexity of change can be seen in many areas in the Andes, where there was a clear loss of native forest vegetation at high altitudes over the last decades while at the same time arable land and grazing land at low altitudes was gradually replaced by plantations of exotic species (Wunder, 1996; Hofstede et al., 2002). Similar complex changes were observed in many areas in the world (for a comprehensive overview, see Rudel et al., 2005). Secondly, the relative importance of processes in a catchment changes with scale, depending on how various land cover units are positioned and interconnected (Cerdan et al., 2004) making it difficult to predict how the entire system will respond from localized observations.

In this paper, we analyze the hydrological response to complex forest cover change for a highly degraded Andean catchment. The Jadan catchment (ca. 300 km^2) is located in the tropical Andes, and is characterized by strong changes in forest cover since the 1960s. As is the case in many Andean catchments, large areas of upland native forests were cleared for expansion of the agricultural land. However, there was not only forest loss as fast-growing forest plantations were established on areas, which were not suitable or economically not profitable for agricultural production, such as deeply intersected gullies or river valleys and badlands (Vanacker et al., 2003a). Exotic forest plantations in the area are characterized by dense and diverse understory vegetation, with presence of grassy and shrubby vegetation (see Molina et al., 2009a). In addition to these small-scale reforestation projects, the decline of grazing pressure favored natural colonization by herbs and shrubs of degraded land. Forest plantations were to some extent driven by the desire to reduce the intense soil erosion occurring in the Jadan catchment, and were promoted by extension workers. In the areas that were most affected by rill and gully erosion, UMACPA (Unidad de Manejo y Conservacion de la Cuenca del Rio Paute) directly implemented soil and water conservation schemes including plantations with exotic rapid-growing species and construction of check-dams in active gully systems (White and Maldonado, 1991). Here, we investigate if these changes in forest cover are associated with changes in the hydrological and sediment regime by combining information from previous small-scale studies with new measurements of water and sediment yield at the catchment outlet.

2. Materials and methods

2.1. Jadan catchment

The Jadan catchment is located in the Ecuadorian Andes, and encompasses a drainage area of 296 km^2 (Fig. 1). It is located in the vicinity of the city of Cuenca, which is the third largest city in Ecuador. The landscape developed on late Tertiary volcanoclastic and sedimentary rocks that are often poorly consolidated and deeply weathered (Hungerbühler et al., 2002). The slope morphology is characterized by moderate to steep slopes, with about 1/3 of the area having slopes steeper than 40%. In most of the basin, the main valley floor of the Jadan River is relatively wide and flat, and a stack of alluvial terraces are present. The region has a tropical mountain climate (Dercon et al., 1998). The average monthly air temperatures show little seasonal variation, and the bimodal rainfall regime registers between 600 and 1100 mm of annual precipitation. The first rainy season occurs from January to May, and the second one from October to December. A marked dry season is generally observed from June to September (Celleri et al., 2007). However, interannual variability is high. The maximum 24 h rainfall intensity for a 5-year return period is about 42 mm (Cochapamba-Quingeo station, 2710 m; Baculima et al., 1999).

2.2. Time series of land cover data: 1963, 1995 and 2007

A time series of land cover data was established from panchromatic aerial photographs of 1963 and 1995 (Instituto Geográfico-Militar, Ecuador) and an ASTER satellite image of 2007. The ASTER images capture information in 15 bands of the electromagnetic spectrum: four in the visible and near infrared regions (VNIR, 0.5–1.0 μm) with a spatial resolution of 15 m; six in the infrared short wave region (SWIR, 1.0–2.5 μm) with 30 m resolution, and five in the thermal infrared region (TIR, 8–12 μm) with 90 m resolution. Five land cover categories were identified: (1) grassland and shrubland, (2) bare land with very poor vegetation cover, (3) exotic forest plantation, dominated by *Eucalyptus gl.* species, (4) agricultural land consisting mainly of rangeland and cropland, and (5) native forest. The interpretation of the land cover types on the aerial photographs of 1963 and 1995 was carried out manually using a Wild (R) mirror stereoscope with four times magnification. Land cover maps were then generated in the ILWIS 3.3 software (Integrated Land and Water Information System 2005) using the orthorectified aerial photographs as the basis for accurate on-screen digitising of the land cover categories. The land cover data of

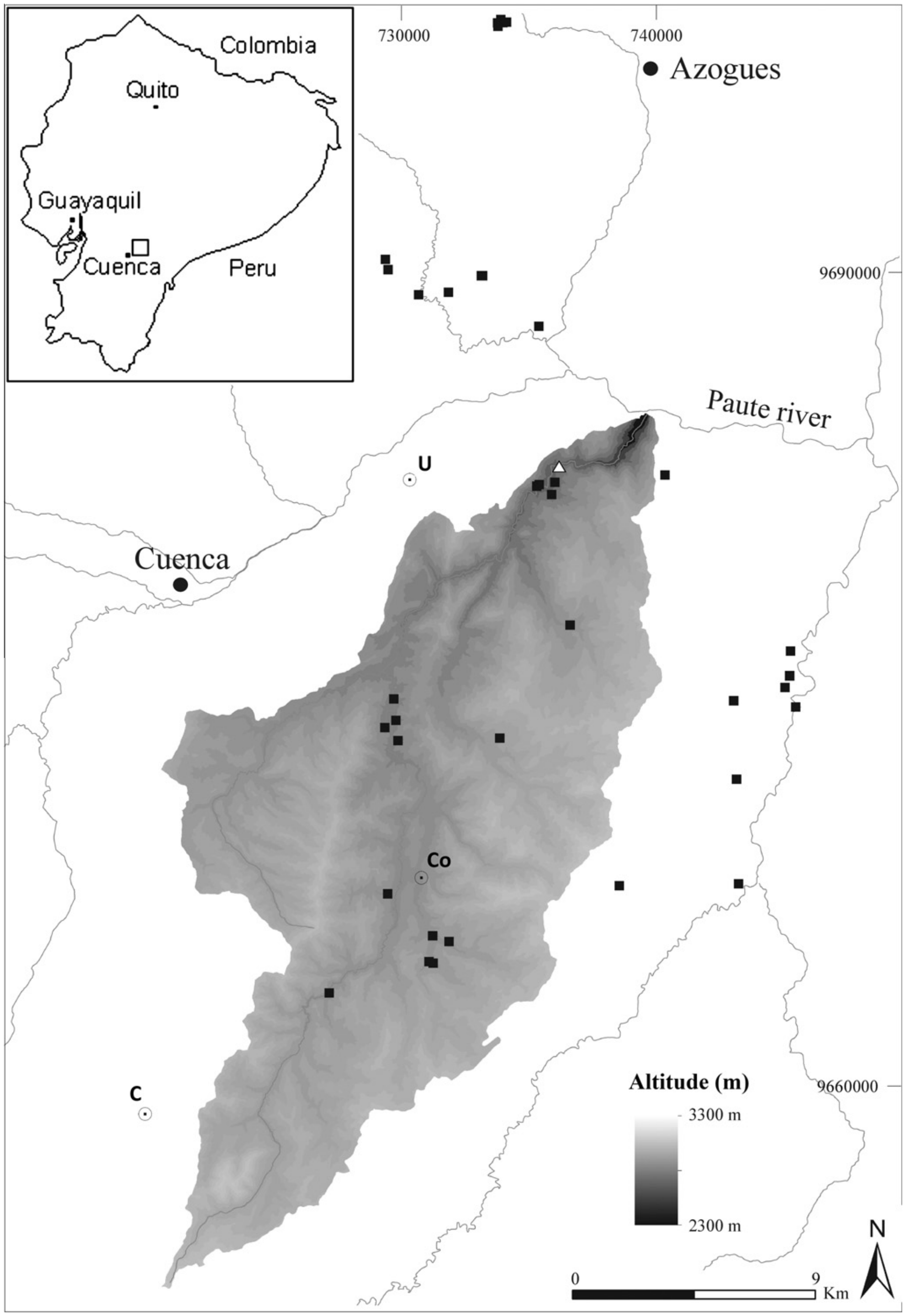

Fig. 1. Situation of the Jadan catchment in the Southern Ecuadorian Andes, with location of the rainfall stations (U: Ucubamba; Co: Cochapamba; C: Cumbe), the Jadan A.J. Paute gauging station (white triangle) and the 37 small catchments monitored for sediment fluxes (Black squares; Molina et al., 2008).

2007 were obtained from Aster satellite images which were geometrically, atmospherically and topographically corrected using PCI and ArcGis software. A supervised classification was then performed using field data from 2007 on the vegetation types and densities.

2.3. Time series of rainfall and streamflow data (1979–1982, 2005–2007)

Time series of daily precipitation were obtained from the National Institute of Meteorology and Hydrology of Ecuador (INAMHI). The point rainfall measurements of the meteorological stations of Cochapamba, Ucubamba and Cumbe were extrapolated using Thiessen polygons as the rainfall amounts were strongly correlated (Fig. 1; Mejia et al., 1996). Given the strong topographic gradients in the basin, it is possible that this method leads to a systematic bias as rainfall amounts in the area are known to increase with altitude (Mora and Willems, 2012). However, even if this would be the case, the data can still be used to assess temporal trends in rainfall amounts.

Consistent and continuous series of daily streamflow were available for two time periods: from 01/01/1979 to 31/12/1982 and from 01/01/2005 to 31/12/2007. For the 1979–1982 period, daily streamflow data ($m^3 s^{-1}$) were obtained for the Jadan A.J. Paute station from the Ecuadorian Institute for Electrification (INECEL). The Jadan A.J. Paute hydrological station was reestablished in 2004. The measuring station was equipped with a full-size portable water sampler (ISCO 6712) and an acoustic distance sensor to measure water heights. Continuous discharge data are based on daily water stage (level) measurements at 30 min intervals. The daily stage readings for the 3 year record (2005–2007) were converted to mean daily streamflows based on a stage–discharge rating curve that was established through manual discharge measurements using a current meter at different water heights. Flow duration curves (FDCs) were constructed from the daily streamflow data for the two time periods: 1979–1982 and 2005–2007. The monthly streamflow data were then separated into baseflow and stormflow using the hydro-statistical toolkit WETSPRO. This model uses a continuous time series of streamflow data as input and calculates the quick and slow flows by means of subflow separation techniques based on a generalization of the original Chapman-filter. The technique is based on the general equation of a 'low pass filter', and assumes exponential recession for the hydrological subflows (Willems, 2009).

2.4. Estimates of catchment-wide erosion rates and suspended sediment loads

Two different approaches were used to constrain the evolution in erosion rates for the Jadan catchment. The first estimate of catchment-wide erosion is based on the application of the empirical model of mean annual sediment yield from Molina et al. (2008). This empirical model predicts specific sediment yield (SSY, Mg $km^{-2} y^{-1}$) in small catchments based on the fractional vegetation cover (C, dimensionless) and the lithology (L, dimensionless):

$$\mathrm{SSY} = a \times e^{cL+bC}$$

With $a = 2330$, $b = -4.91$ and $c = 2.73$.

The vegetation cover refers to the surface cover that is actually in contact with or very close to (<0.10 m) the soil surface and will therefore often be considerably lower than total vegetation cover. Land cover maps (1/10,000) were combined with data from vegetation plots to estimate the fractional surface cover (i.e. the fraction of the catchment covered by vegetation) for each catchment. The surface vegetation cover of each land cover class was characterized by measuring the surface vegetation cover for randomly selected vegetation plots of 1 by 1 m. In 2003 and 2004, 7–37 vegetation plots per land cover class were characterized. The fractional vegetation cover of each catchment was then calculated by taking the weighted average of the surface vegetation cover of all land cover types that are prevailing in the catchment, using the area of each land cover class in the catchment as the weighting factor. The lithology refers to the proportion of each catchment underlain by argillaceous rocks (argillaceous sandstone and so-called lutites by Litherland et al., 1994, including shales, siltstone, mudstone and claystone), and young pyroclastic deposits. This proportion (L, dimensionless) was estimated from 1/50,000 geological maps (DINAGE, 1974).

The empirical model parameters were derived from a comprehensive dataset on specific sediment yield data (Mg $km^{-2} y^{-1}$) for the southern Ecuadorian Andes (Molina et al., 2008). The 37 selected catchments represent the variability in lithology, topography, land cover, and soils within the degraded areas of the lower and middle part of the Cuenca intermontane basin. The mean annual specific sediment yield, SSY, was based on direct measurements of the accumulated sediment volumes behind 106 checkdams located within 37 catchments. Univariate and multivariate statistical analyses were performed to analyze the dependency of the mean annual sediment yield on topography, soil types, lithology and vegetation cover. For details on the catchment characteristics, we refer to Table 1 in Molina et al. (2008).

Table 1
Land cover change (1963–2007) in the Jadan catchment, with the relative extent of each land cover type obtained from 1/10,000 land cover maps (%), the fractional vegetation cover per land cover type (C, dimensionless) and indication of change (%) in land cover over the period 1963–2007. The fractional vegetation cover in 1963 was estimated based on indirect evidence from historical photographs. The fractional vegetation cover of 1995 and 2007 is derived from vegetation cover data from 1 m by 1 m vegetation plots that were realized in 2003 and 2004.

Land cover type	1963		1995		2007		Δ (1963–2007)
	(%)	C	(%)	C	(%)	C	%
Grassland and shrubland	5.1	0.19	4.9	0.57	4.6	0.57	−0.5
Bare land	11.3	0.03	6.5	0.09	7.1	0.09	−4.2
Exotic forest plantation	0.6	0.67	3.7	Id.	5.0	Id.	+4.4
Agricultural land	52.3	0.71	56.0	Id.	59.4	Id.	+7.1
Native forest	30.7	0.99	28.9	Id.	23.9	Id.	−6.8

The empirical model was able to explain more than 75% of the observed variance in SSY (Model R-Square = 0.80) in the region, whereby the vegetation cover alone explains 57% (Partial R-Square = 0.57, Molina et al., 2008). Lithology explains an additional 23% of the observed variance in ln(SSY). No significant association was found between vegetation cover and lithology based on our dataset of 37 catchments. Besides, no meaningful relationship between sediment yield and topography was found. An estimate of the specific sediment yield in the Jadan catchment for the years 1963, 1995 and 2007 was then obtained by applying the specific sediment yield model to the data in Table 1. The mean catchment-wide erosion rate for the Jadan catchment was then calculated by weighing the specific sediment yield of each land cover class by the surface area of the land cover class. The relative error on these sediment yield estimates typically ranges between 30% and 40% (Sougnez et al., 2011; Verstraeten and Poesen, 2002). There are two important limitations to this approach: (i) only erosion by overland flow (sheet, rill and gully erosion) is accounted for and (ii) deposition and remobilization of sediment during its transport within the river system is neglected.

There is no doubt that the overall vegetation cover of the poorly vegetated areas (such as grassland and shrubland, and bare land) changed strongly during the period 1960s – now. Historical photographs published in Liddle and Palmer (1941) provide clear evidence of the ubiquitous presence of highly degraded areas in 1940s. The photographs show that the vegetation cover on grasslands and bare lands was extremely low in 1940s, with only some agave plants and lupines present. In 1963, these areas were still subject to an intense grazing pressure resulting from the feudal land management system. Their extremely low vegetation cover is clear from the airphotos of 1963 (see also discussion in De Noni and Viennot (1993), Vanacker et al. (2003a)). Airphoto comparison shows that the vegetation cover of these degraded lands had increased strongly by 1995, about 30 years after the abandonment of the feudal system (Fig. 2). We have no quantitative information on the changes in vegetation cover through time, as direct measurements from vegetation plots are only available for 2003 and 2004 (see Appendix A of Vanacker et al., 2007). As a first approximation, we assume that (i) the vegetation cover that was measured in 2003 and 2004 is representative for the situation in 1995 and 2007, and (ii) the vegetation cover of the areas that were used for extensive grazing in the 1960s was at least three times lower in the 1960s compared to the situation nowadays.

Secondly, the mean annual suspended sediment loads (Mg y^{-1}) for the Jadan catchment for the periods 1979–1982 and 2005–2007 were estimated from two time series of daily streamflow records

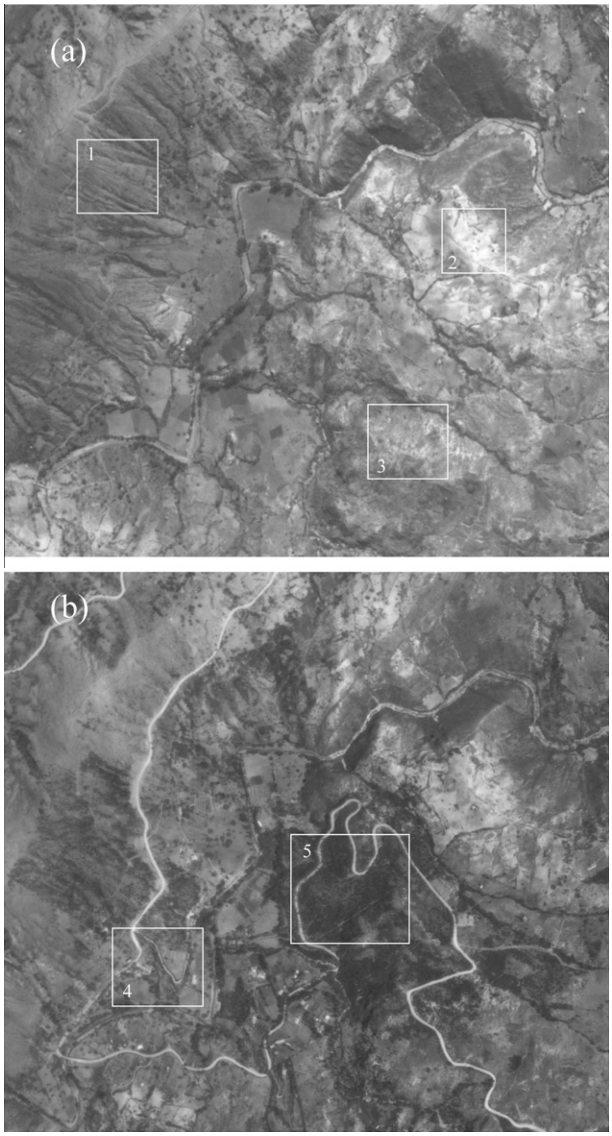

Fig. 2. Two aerial photographs (a: 1963; b: 1995) showing the increase in vegetation cover in grass- and shrublands and bare lands. In general, areas with very low vegetation cover have a high reflectance (and show up as white spots); whereas areas with dense tree cover show up as dark gray or black zones. As illustration, we characterize here five areas: (1) active gully systems with presence of active gully channels, (2) highly degraded land with very low vegetation cover and bedrock outcrops, (3) degraded land with presence of rills and small gully systems, (4) agricultural land bordered by trees, and (5) dense tree cover of Eucalyptus plantation with dense understory vegetation.

and a sediment rating curve. The relationship between suspended sediment concentration and stream discharge was established based on 101 instantaneous observations of discharge and suspended sediment concentration. From November 2004 till February 2007, samples of 1 L were taken for high, intermediate and low flow conditions using an ISCO automatic water sampler. The inlet of the intake hose was located at 1/4 of the distance across the channel, and at 15 cm above the channel bed to avoid blocking of the intake hose with coarse sediment or debris. Flow at the location of the sampler was strongly turbulent so that a near-vertical suspended sediment profile could be assumed. The suspended sediment concentration, Qs (g s^{-1}), was related to the instantaneous discharge, Q (l s^{-1}), by a power function (Fig. 3):

$$Q_s = aQ^b$$

With $a = 8.0 \times 10^{-5}$, $b = 2.22$.

The rating curve (established for the period 11/2004–02/2007) was then applied to the daily streamflow data to reconstruct the daily sediment loads for the period 2005–2007 as well as for the period 1979–1982. The annual sediment loads were then obtained by summing the daily sediment load estimates over the entire year. The reconstruction of the sediment loads of the 1979–1982 period might be biased as changes in suspended sediment transfer might have occurred over time: the implications of this will be discussed later.

3. Results

3.1. Land cover change (1963–2007)

The land cover analysis indicates complex dynamics with different pathways of change (Fig. 4). Three main processes were identified: deforestation, reforestation, and spontaneous recovery. Table 1 shows the land cover change matrix for the period 1963–2007. From the total catchment area of 296 km^2, 58% or ca. 172 km^2 did not show a change (47 km^2 of native forest, 115 km^2 of arable land and 9 km^2 of grassland). There was a net average loss of native forests of ca. 45.5 ha per year. About 22% of the total forest area in 1963 was cleared, and 74% of the total deforestation results from conversion from native forests to agricultural land. Today, only 24% of the total catchment is forested, with patches of native forest dispersed in protected areas or at remote locations at high altitudes.

As described by White and Maldonado (1991), deforestation rates were very high in the early 1960s resulting in a rapid expansion of the agricultural frontier due to agricultural reforms implemented by the government. Deforestation continued until recently, albeit at a slower rate (Vanacker et al., 2003a). In contrast to deforestation, the processes of recovery and reforestation were particularly important in the lower parts of the Jadan catchment (below 2900 m) where agricultural and bare lands are prevalent. The total area of 'bare land' decreased from 33.6 to 21.0 km^2 or a net decrease of 28 ha per year. Ca. 12.6 km^2 of the 'bare lands' that were extensively grazed by sheep and goats in the past were converted to either forest plantations (54%) or agricultural land (46%).

3.2. Changes in water discharge and seasonality of the flow (1979–1982, 2005–2007)

Fig. 5 shows a representative section of the measured rainfall and discharge of the Jadan catchment for the periods 1979–1982 and 2005–2007. The daily rainfall data show that most rainfall events are of low intensity, but that high-intensity storm events of short duration are also important (Fig. 6a). Maximum 24 h rainfall intensities of 41 mm day^{-1} and 69 mm day^{-1}were measured for the periods 1979–1982 (Cochapamba station) and 2005–2007 (Aguarongo station) respectively.

The combination of rainfall and flow duration curves for the periods 1979–1982 and 2005–2007 allows assessing changes in the hydrological response of the Jadan catchment through time (Fig. 6). Over the last three decades (1979–2007), the flow and rainfall regime changed significantly (Table 2). A chi-square analysis shows that both the shifts in discharge and rainfall distribution are significant (two-tailed p-value < 0.001). Changes in discharge are not resulting from changes in precipitation, as the direction of change is opposite. The flow rates of 2005–2007 are about 40% higher for moderate flows (10–90‰ flows) while the precipitation decreased slightly (Fig. 6a). Important changes were also observed for the 1% highest flow rates (Q1) that were about 54% lower while the highest rainfall amounts increased by 52% (Table 2, Fig. 6b). As a consequence of the decrease of peak flows, the total variation in

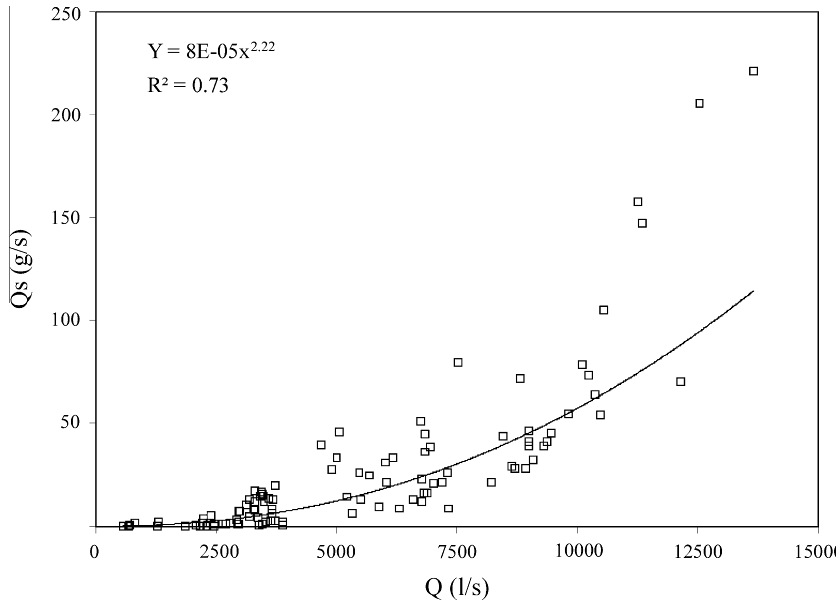

Fig. 3. Sediment rating curve established at the Jadan A.J Paute station based on river discharge and suspended sediment concentration measurements (2004–2006).

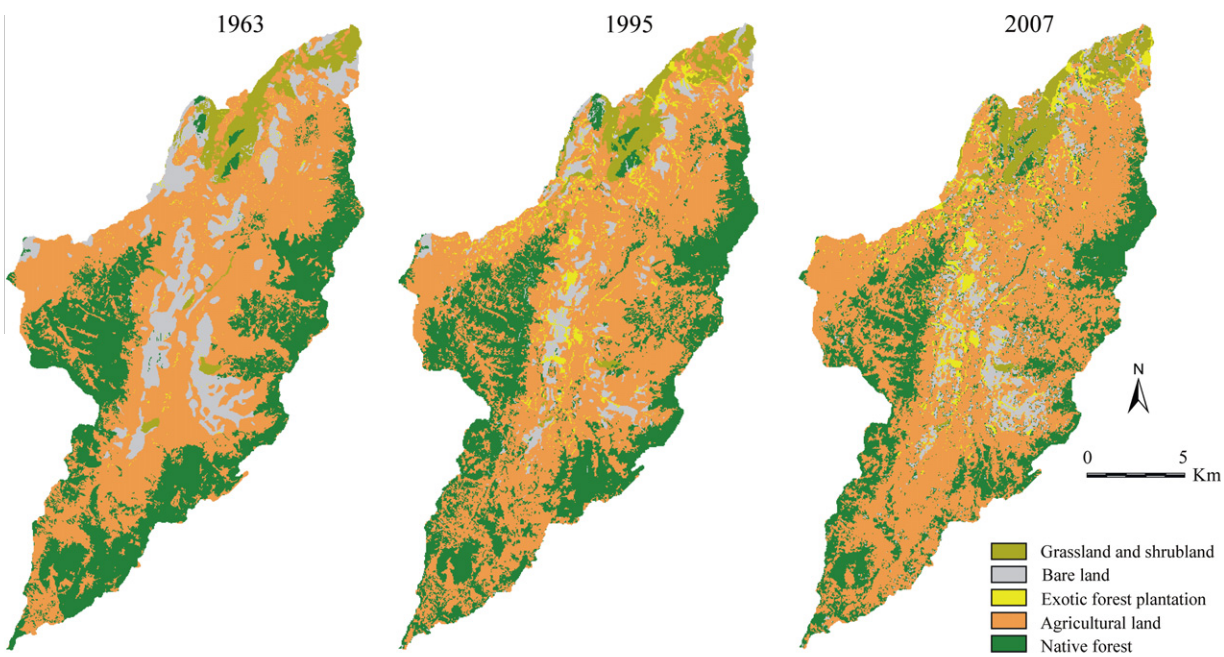

Fig. 4. Land cover maps of 1963, 1995 (based on aerial photographs) and 2007 (based on ASTER satellite image).

daily flow amounts (here measured as the difference between the highest and lowest flow values) strongly decreased from 10.62 to 5.05 mm day^{-1} while the total variation in daily rainfall increased by 108%.

The mean monthly streamflow was higher in the period 2005–2007 in comparison to the period 1979–1982, with exception of October and December (Fig. 7a). The baseflow, as calculated using WETSPRO, is systematically higher in the period 2005–2007: the increase in mean baseflow ranges from 0.6 to 4 mm month^{-1}with the largest increase during the wet season (January–May) followed by the dry season (June–September) and finally the second wet season (October–November, Fig. 7b).

3.3. Changes in catchment-wide erosion rates and suspended sediment loads (1970s–2000s)

The first estimate of catchment-wide erosion rates or specific sediment yields based on the empirical erosion model of Molina et al. (2008) shows a decrease in area-specific sediment yield from ca. 3700 ± 740 Mg km^{-2} y^{-1} in 1963 to ca. 1700 ± 340 and 1800 ± 360 Mg km^{-2} y^{-1} in 1995 and 2007 respectively. These values are based on the assumption that there is a threefold increase in vegetation cover in grass-, shrubland and bare land between the 1960s and the mid-1990s/2000s (Table 1). Even when we neglect these changes in vegetation cover completely, there exists a

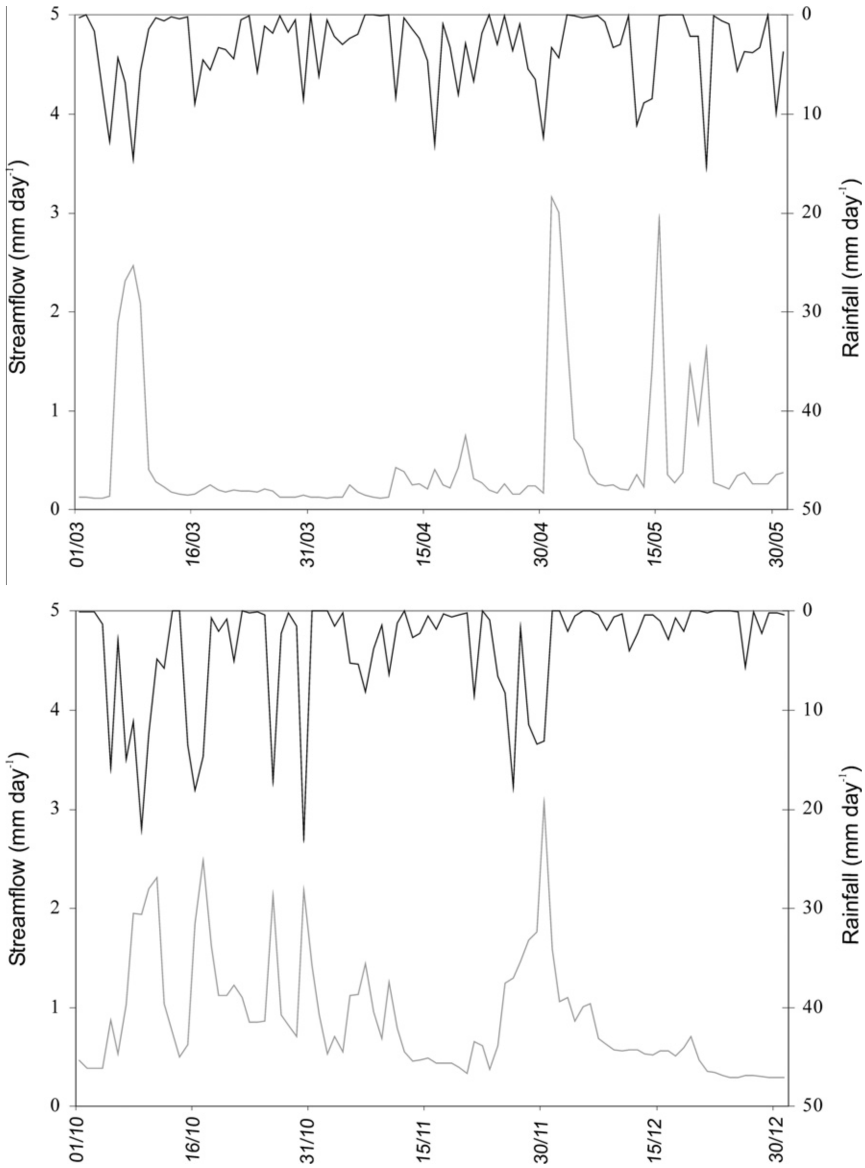

Fig. 5. A selected portion of the rainfall-runoff response at the Jadan A.J Paute station: (a) 01-03-1980 to 31-05-1980 and (b) 01-10-2005 to 31-12-2005.

measurable and strong decrease in specific sediment yields from ca. 3700 ± 740 Mg km^{-2} y^{-1} in 1963 to 2400 ± 480 and 2600 ± 520 Mg km^{-2} y^{-1} in 1995 and 2007, respectively The estimated changes in the total amount of sediment leaving the catchment are even larger: the estimated suspended sediment load drops from about $3.03 \times 10^6 \pm 0.43 \times 10^6$ Mg y^{-1} (corresponding to 10200 ± 1450 Mg km^{-2} y^{-1}) for the period 1979–1982 to a value of 'only' $0.72 \times 10^6 \pm 0.10 \times 10^6$ Mg y^{-1} (corresponding to 2400 ± 334 Mg km^{-2} y^{-1}) for 2005–2007. As the relationship between discharge and sediment concentration is assumed to be more or less quadratic (Fig. 3) the reduction in peak discharges results in a strong reduction in predicted suspended sediment loads.

4. Discussion

4.1. Land cover change controlling water yield

The removal of native forest for rangeland or croplands (by -20 km^2) is likely to have contributed to the increase in total annual water yield (Fig. 7a), through an increase in baseflow of about 25 mm (Fig. 7b). In rangelands, kikuyu grass (Pennisetum clandestinum) is dominant, with only limited establishment of other pasture grasses. This invasive grass species was introduced in the Ecuadorian Andes, and has a shallow root system compared to native bunch grasses such as Calamagrostis or Festuca. As a result of decreased evapotranspiration losses due to reduced water demand of the introduced grassy vegetation or agricultural crops in comparison to a full-grown native forest, both the total streamflow and the baseflow are expected to increase. Perennial grasses (such as kikuyu) and crops have a limited capacity to intercept and evaporate rainfall (Van Dijk and Bruijnzeel, 2001), and the shallow root system restricts the water uptake from the deeper soil layers (Calder, 1998). No data are available on rainfall interception and subsequent evaporation from the canopy and the litter layer. Based on interception measurements in similar environmental settings by Veneklaas and Vanek (1990), we estimate an interception loss of about 30% for Andean forest and 11% for grassland and cropland. In the upland areas, the removal of forest is likely to result in enhanced soil water infiltration and percolation in the well-drained and deep soils developed on volcanic deposits.

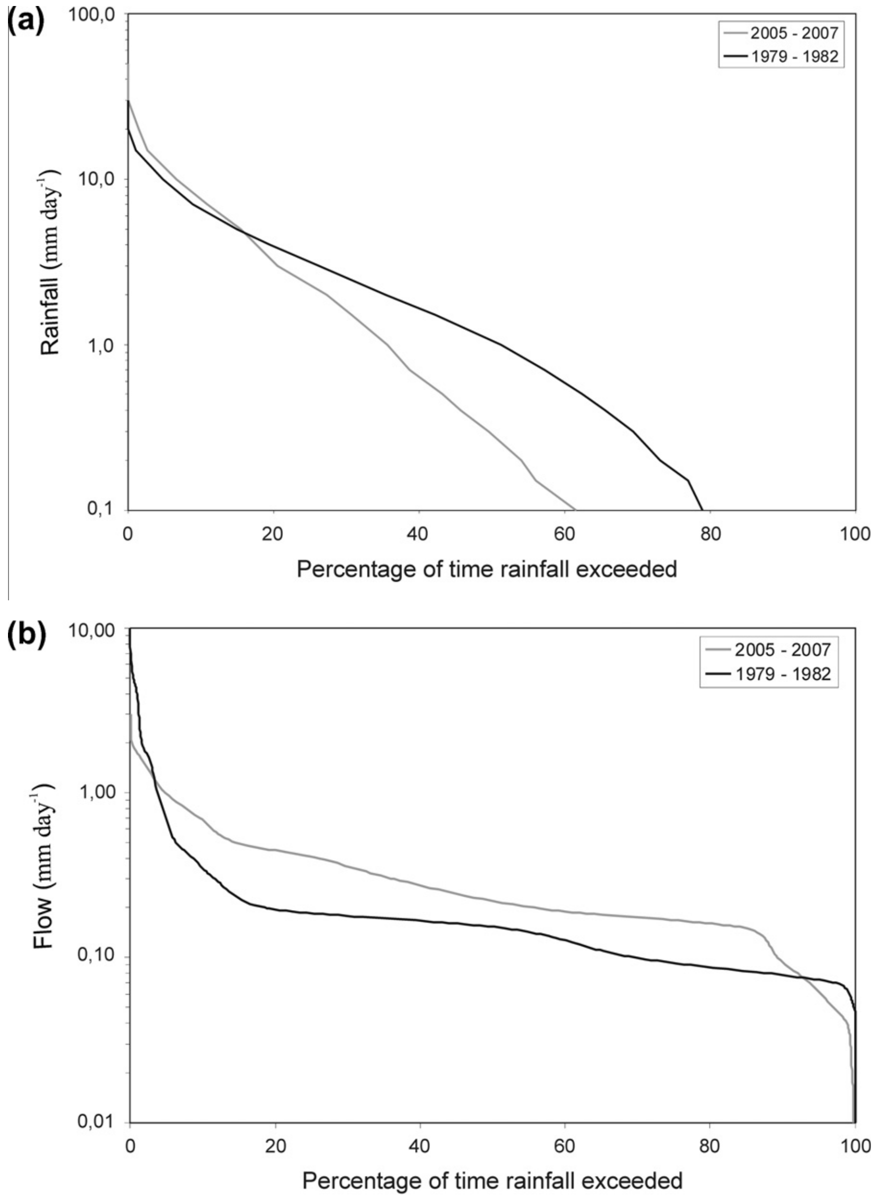

Fig. 6. (a) Rainfall duration curves for the period 1979–1982 (black line) and 2005–2007 (gray line) and (b) flow duration curves for the period 1979–1982 (black line) and 2005–2007 (gray line).

Table 2
Probability distribution of streamflow and precipitation data, with indication of changes (%) in flow and precipitation between 1979–1982 and 2005–2007.

Percentile	1979–1982		2005–2007		Change (%)	
	Flow (mm)	Rain (mm)	Flow (mm)	Rain (mm)	Flow (mm)	Rain (mm)
Q95	0.11	0.0	0.09	0.0	−18	–
Q90	0.12	0.0	0.14	0.0	+17	–
Q50	0.20	1.1	0.34	0.3	+70	−73
Q10	0.44	6.6	0.95	7.3	+116	+11
Q5	0.99	9.9	1.50	12.0	+52	+21
Q1	5.32	15.1	2.43	23.0	−54	+52
Range (max–min)	10.62	20.8	5.06	43.2	−52	+108

Bruijnzeel (2004) collected a large dataset on the hydrological response to tropical forest clearing. Data show considerable scatter, but show, on average, an increase in annual streamflow (baseflow) by ca. 200 mm when 50% of the tropical forest cover in a catchment was cleared: this increase is most probably related to a decrease of plant transpiration (due to a decrease in leaf area index) and evaporation (due to a decrease in rain interception). The Jadan catchment lost 6.8% of its native forest cover during the observation period (1963–2007, Table 1). Assuming a linear relationship between increase in runoff yield and deforestation percentage, this would then correspond to an increase in annual baseflow of ca. 30 mm. This figure is indeed of the same order of magnitude than the observed increase in baseflow. For the period 1979/1982 till 2005/2007, an increase in annual baseflow of 25 mm was observed. The monthly baseflow increased by 0.6–4 mm month^{-1} with the largest increase during the wet season (January to May, Fig. 7b).

The effect of deforestation on surface runoff generation (and hence peak flows) can be expected to be limited. Freshly deforested areas are most often used as arable land for a limited period of time, after which grassland is established. The pasture grass (dominance of kikuyu) has a vegetation cover that is near 100%,

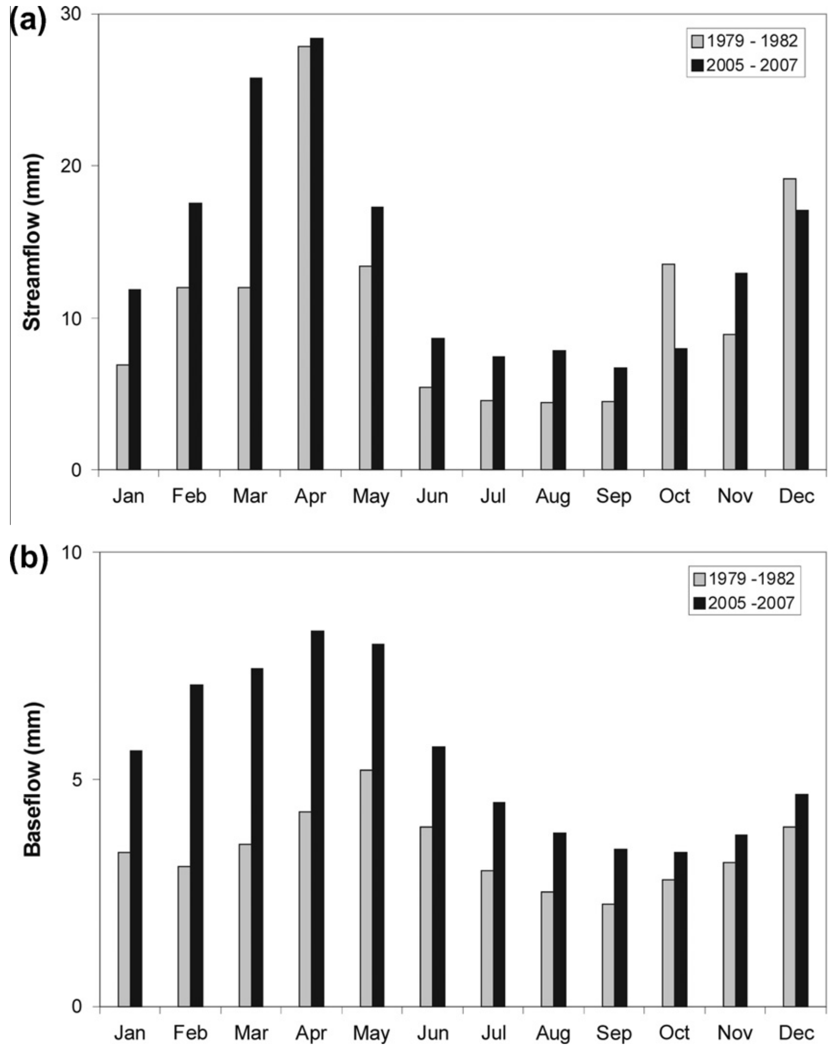

Fig. 7. Mean monthly streamflow (a) and baseflow (b) for the period 1979–1982 and 2005–2007.

resulting in very little surface runoff. Molina et al. (2007) showed that the cumulative runoff coefficient on arable land varies from 0% to 13%with an average value of 5% ± 4, and on rangeland from 0% to 8% with an average value of 4% ± 4. One may then wonder why the reforestation in the lower parts of the basin did not lead to an increased water use, thereby canceling out the gains in baseflow due to the removal of the native forest in the upper catchment areas. The main reason for this is, we believe, that these forests were planted on extremely degraded areas (badlands underlain by marls and consolidated clayey sandy materials) where no soil was left. Rainfall-runoff experiments (with an effective rainfall intensity of 41 ± 6 mm h^{-1}) indicated that degraded land has a large variability in runoff rates. Cumulative runoff coefficients range between 4% and 100%, with an average value of 47% ± 22 ($n = 36$). Molina et al. (2007) showed that the surface vegetation cover controls runoff generation: when the vegetation cover of the degraded lands is low (below 10%), the cumulative runoff coefficient is 58% ± 16 ($n = 19$). The absence of a soil layer able to store water during an event and gradually releasing it afterwards implies that these areas were not producing much baseflow before reforestation took place. Hence, planting these forests is not expected to lead to baseflow reduction at the catchment scale.

The removal of part of the original forest cover can also not explain the observed decrease in peak flows, which is of the order of −54% for the 1st percentile flows (Table 2). This reduction is, on the contrary, likely to be related to the reforestation of part of the degraded lands to exotic forest plantations as well as the spontaneous recovery of vegetation on bare land and the remaining grazing lands. Shrubs composed of *Cortaderia rudiuscula* (locally called zig-zal), *Spartium junceum* (locally called retama) and *Baccharis polyantha* (locally called chilca) are an important component of the understory vegetation in forest plantations, and they also colonize spontaneously abandoned grazing land. Underneath the shrubby canopy, stands of grass and weeds composed mainly of *Pennisetum clandestinum* (locally called kikuyu), *Holcus lanatus* (locally called grama), *Festuca megalura* (locally called pajilla) and *Cynodon dactylon* (locally called grama de la virgen) are established.

Based on 55 rainfall-runoff simulation experiments, Molina et al. (2007) showed that the average intensities of the short-duration high-intensity rainfall events exceed the soil infiltration capacity of degraded and abandoned land (estimated at 22 ± 10 mm h^{-1}, $n = 36$). These areas generate surface runoff within a few minutes after the start of the rainfall event. In contrast, overland flow on well-vegetated surfaces (agricultural land, and native forest) is rare, as their soils are characterized by high infiltration rates (i.e. 38 ± 6 mm h^{-1} Molina et al., 2007). When using the experimental data on cumulative runoff coefficients, we can make a rough estimate of the change in total runoff amount for high-intensity rainfall events. Table 3 shows the cumulative runoff coefficient

Table 3
Cumulative runoff coefficients (RC,%) per land cover type based on 55 rainfall-runoff experiments conducted in the Jadan catchment by Molina et al. (2007).

Land cover type	RC	1963	2007
	%	Area (km^2)	Area (km^2)
Grassland and shrubland	56 ± 24	15.0	13.7
Bare land		33.6	21.0
With <10% cover	58 ± 16		
All	47 ± 22		
Exotic forest plantation	36	1.6	14.8
Agricultural land	4.5 ± 4	155.0	175.8
Native forest	0.5	91.0	71.0

per land cover class. The average cumulative runoff coefficient (weighted by the area of each land cover class) for the Jadan catchment is estimated at 12.1% for the year 1963 and at 11.3% for 2007. For one given high-intensity rainfall event of 40 mm, the observed land use changes are expected to result in an overall reduction of surface runoff by 6.7% (or an equivalent of runoff amount of 0.3 mm). An important increase in the vegetation cover in the areas that were used for grazing would lead to a further decrease of the total amount of surface runoff during high-intensity rainfall events.

The observed changes (between 1979/1982 and 2005/2007) in extreme events are important as the 1st percentile daily flow decreased by 54% or 2.89 mm; despite the fact that the 1st percentile daily rainfall increased by 52% or 7.9 mm (Table 2, Fig. 6). Tentatively, this stronger reduction may be explained by the spatial pattern of vegetation cover increase in areas that were previously used for grazing. The increase of vegetation cover is especially high in highly degraded land (here classified as grass-, shrubland and bare land), where active gullies were present in the 1960s (Fig. 2). Molina et al. (2009b) showed that the increase of the vegetation cover in gully channels reduces the transfer of runoff: part of the runoff that is generated upslope is infiltrating in the gully floors where, due to the establishment of vegetation, sediment is deposited. Field measurements on 138 gully segments located in 13 ephemeral steep gullies with different ground vegetation cover indicated that gully bed vegetation is the most important factor in promoting short-term (1–15 years) sediment deposition and gully stabilization. In well-vegetated gully systems ($\geqslant$30% of ground vegetation cover), 0.035 m^3 m^{-1} of sediment is deposited yearly in the gully bed (Molina et al., 2009a). This process then leads to the entrapment of more sediment and hence a further increase of infiltration, disconnecting the slopes from the rivers so that peak flows are further reduced.

4.2. Land cover change controlling specific sediment yield

Changes in specific sediment yield and suspended sediment loads were estimated based on two different datasets that were collected for the Jadan catchment. The results both point to a major decrease in specific sediment yields and suspended sediment loads over time. The specific sediment yield estimates from the empirical model show a decrease by a factor two (of −51%) between 1963 and 2007. The suspended sediment load estimates based on the sediment rating curve technique suggest an even sharper decrease (of −76%) over a much shorter period, i.e. between the 1979–1982 and 2005–2007 period. While a relatively large uncertainty is associated with the absolute values in both estimates, there is little reason to suspect that the relative magnitude of change is not correct. Furthermore, it is not surprising that both methods yield different results, both in terms of absolute values and with respect to the magnitude of change. The estimates based on suspended sediment loads are higher than those based on the empirical erosion model: this is especially true for the earlier estimates. This may be partly due to errors in our estimates: Verstraeten and Poesen (2002) have shown that estimates on sediment yields from volumetric data of reservoir infillings can easily surpass 30%.

It should be pointed out that deforestation in the upper parts of the basin has led to increased landslide activity (Vanacker et al., 2003b). This is not reflected in an increased sediment yield from the basin as the small upland rivers are often nearly completely blocked by the landslide material, thereby reducing their potential to transport sediment, a mechanism similar to the one described by Montgomery and Korup (2011) at a much larger scale. The reduction in estimated erosion is likely to be caused by the reduction of the degraded areas in areal extent as well as to the (partial) recovery of the vegetation in these areas (Table 1; Molina et al., 2009a). The efficiency of vegetation to trap sediment in active gully systems was already highlighted by Rey (2003) for forested marly catchments in France. The reduction rate is estimated to be most important between 1963 and 1995 as a result of the contraction of bare land of 14.4 km^2 (−4.8%) and the expansion of exotic forest plantation by about 9.3 km^2 (+3.1%; Table 1). Between 1995 and 2007, there is a slight increase in the area of bare land (by about 1.8 km^2) and an increase of the exotic forest plantation of 3.9 km^2 (Table 1).

5. Conclusions

This study provides new insights in the complex hydrological response of degraded mountain catchments. Land use (1963–2007) in these degraded Andean catchments changed rapidly through time as a response to changing socio-economic and demographic conditions. Three different pathways of forest cover change are observed: clearcut of native forest, plantations of exotic forests and spontaneous revegetation with native shrub and tree species. The forest cover changes occurred in spatially distinct regions, with massive deforestation in the upper (remote) parts and reforestation in the lower degraded parts of the catchments.

The combination of rainfall and flow duration curves for the periods 1979–1982 and 2005–2007 clearly show significant changes in flow and rainfall regime through time. Changes in discharge are not resulting from changes in precipitation, as the direction of change is opposite. Our data show that the shifts in the hydrological regime are associated with human-induced changes in vegetation type and density. The observed increase in baseflow (of about 25 mm over a 30 year period) is likely to be related to the conversion of native forests to agricultural land (−20 km^2 of native forests, 1963–2007). Reforestation in the lower part of the catchment not necessarily led to a reduction in total water yield, as the increase in forest cover occurred mainly on extremely degraded areas (so-called badlands) that did not produce any baseflow before restoration. The strong decrease in peakflows may be explained by the strong increase in vegetation cover in highly degraded lands, where active gully systems were active in the 1960s. The establishment of herbaceous and shrubby vegetation as understory vegetation of forest plantations gives rise to the formation of vegetated buffer zones in gully beds, which enhance sediment trapping and runoff infiltration. Vegetated buffer zones in degraded lands modify the connectivity of water and sediment fluxes by reducing the transport efficiency of gully systems, and by retarding surface runoff and enhancing runoff infiltration.

Our data suggest that the hydrological and sediment regime of degraded mountainous catchments can only be understood if the different pathways of forest cover change are analyzed in a spatially explicit way. In the Jadan catchment, the impact of deforestation is partially offset by the reforestation and spontaneous revegetation of degraded areas. Although vegetation restoration of badlands represents less than 10% of the total catchment area, its impact on the hydrological and sediment regime may be disproportionally large. This shows the potential of relatively small bioengineering works that are concentrated in extremely degraded areas to improve the erosion control and hydrological regulation of degraded mountainous catchments.

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