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Discussions

The impact of forest regeneration on streamflow in 12 meso-scale humid tropical catchments

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Abstract

Although regenerating forests make up an increasingly large portion of humid tropical landscapes, comparatively little is known of their water use and effects on streamflow (Q). Since the 1950s the island of Puerto Rico has experienced widespread abandonment of pastures and agricultural lands, followed by forest regeneration. This paper examines the possible impacts of forest regeneration on several Q metrics for 12 meso-scale catchments (23–346 km^2; mean precipitation 1720–3422 $mm\,yr^{-1}$) with long (33–51 yr) and simultaneous records for Q, precipitation (P), potential evapotranspiration (PET), and land cover. A simple spatially-lumped, conceptual rainfall-runoff model that uses daily P and PET time series as inputs (HBV-light) was used to simulate Q for each catchment. Annual time series of observed and simulated values of four Q metrics were calculated. A least-squares trend was fitted through annual time series of the residual difference between observed and simulated time series of each Q metric. From this the total cumulative change $\hat{A}$ was calculated, representing the change in each metric after controlling for climate variability and water storage carry-over effects between years. Negative values of $\hat{A}$ were found for most catchments and Q metrics, suggesting enhanced actual evapotranspiration overall following forest regeneration. However, correlations between changes in urban or forest area and values of $\hat{A}$ were insignificant ($p \geq 0.389$) for all Q metrics. This suggests there is no convincing evidence that changes in the chosen Q metrics in these Puerto Rican catchments can be ascribed to changes in urban or forest area. The present results are in line with previous studies of meso- and macro-scale (sub-)tropical catchments, which generally found no significant change in Q that can be attributed to changes in forest cover. Possible explanations for the apparent lack of a clear signal may include: errors in the land-cover, climate, Q, and/or catchment boundary data; changes in forest area occurring mainly in the less rainy lowlands; and heterogeneity in catchment response. Different results were obtained for different catchments, and using a smaller subset of catchments could have

led to very different conclusions. This highlights the importance of including multiple catchments in land-cover impact analysis at the meso scale.

1 Introduction

Tropical regions have experienced extensive changes in land use and land cover during the last few decades (Lepers et al., 2005). Continuously rising demands for crop land and timber have led to substantial deforestation in many regions (Drigo, 2005), and although the global tropical deforestation rate remains high at 13 Mha yr^{-1} (FAO, 2006), forest regrowth on abandoned agricultural land is increasing, particularly in Latin America (Aide and Grau, 2004; Hecht, 2010) and South-East Asia (cf. Fox et al., 2000; Xu et al., 1999). Because these "secondary forests" account for approximately one-third of the total tropical forest area (Brown and Lugo, 1990; Hölscher et al., 2005), understanding the impact of forest regrowth on water yield is important for water resources management and planning purposes (Giambelluca, 2002; Bruijnzeel, 2004) and the development of viable "Payments for Ecosystem Services" schemes (Landell-Mills and Porras, 2002; Lele, 2009). However, despite this recognized importance, little is known of the water use of secondary tropical forests, although there are indications of enhanced water use during the period of most active biomass accumulation (Giambelluca, 2002; Juhrbandt et al., 2004; Hölscher et al., 2005).

The relationship between forest cover and streamflow (Q) is subject to a long-standing and ongoing discussion (Andréassian, 2004), also in the tropics (Bruijnzeel, 2004; Calder, 2005). The influence of forest cover change on flooding is particularly contentious (e.g. FAO, 2005; Bradshaw et al., 2007; Van Dijk et al., 2009) whereas the effect of forestation on tropical dry-season flows is also under debate (Calder, 2005; Scott et al., 2005). The general contention is that the *net* effect of an increase in forest cover on dry-season flow depends on the "trade-off" between *increases* in Q due to enhanced soil water recharge on the one hand (as forestation generally increases soil macroporosity and infiltration characteristics; Ilstedt et al., 2007; Bonell et al., 2010;

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Zimmermann et al., 2006, 2010; Hassler et al., 2011), and *decreases* in soil water reserves and Q on the other hand due to the higher water use of trees compared to crops, pasture, or scrubs (Bruijnzeel, 1989; Bruijnzeel, 2004; Jackson et al., 2005; Scott et al., 2005). Reviews of micro-scale ($< 1\,km^2$) experimental catchment studies (e.g. Jackson et al., 2005; Brown et al., 2005) – mostly conducted outside the tropics and in non-degraded settings where soil infiltration characteristics are not likely to be improved substantially by forestation – suggest that the increase in vegetation water use is indeed more important, and thus an increase in forest cover commonly leads to a decrease in both total and dry-season Q. However, although direct experimental evidence of the "infiltration trade-off hypothesis" (Bruijnzeel, 1989; Bonell et al., 2010) is missing due to a lack of comprehensive studies, demonstrated reductions in amounts of headwater- or hillslope stormflow production after reforesting severely degraded land in various parts of the tropics (e.g. Chandler and Walter, 1998; Zhou et al., 2002; Zhang et al., 2004) should be large enough to overcome the associated increases in forest water use (Chandler, 2006; cf. Bruijnzeel, 2004; Scott et al., 2005; Zhou et al., 2010). Indeed, as long-term Q records for large, once degraded catchment areas are becoming available, evidence of improved baseflows (Q_{bf}) following large-scale land rehabilitation is beginning to be documented (Wilcox and Huang, 2010; Zhou et al., 2010).

The tropical island of Puerto Rico (PR) provides a solid opportunity to study the impacts of natural forest regeneration on Q at the meso-catchment scale. This is as relatively high-quality long-term hydro-climatic records and sequential land-cover data are available. Since the 1950s, PR has seen widespread secondary forest regrowth on abandoned pastures, agricultural land (mostly sugar cane), and coffee plantations (Thomlinson et al., 1996; Aide et al., 2000; cf. Del Mar López et al., 1998). Although previous work on the relationship between land-cover change and Q using lumped meso- and macro-scale catchment data has experienced some difficulty demonstrating unequivocal results (e.g. Van Dijk et al., 2012), possibly stronger conclusions may be obtained by selecting catchments within similar physiographic units (in this case

Central PR; cf. Peña-Arancibia et al., 2010). On the whole, one would expect marked drops in total Q and Q_{bf} during forest recovery in areas where the general extent of soil surface degradation before land abandonment was limited and soil structural characteristics (and thus infiltration opportunities) therefore remained relatively unaffected by forest regeneration (cf. Aide et al., 1996; Zou and Gonzalez, 1997). For catchments that experienced advanced soil degradation prior to agricultural abandonment, major declines in the volumes of both total Q and quickflow (Q_{qf}) would be expected during forest regrowth due to much improved infiltration and retention capacities (cf. Chandler and Walter, 1998; Zhou et al., 2002). The direction and magnitude of the change in Q_{bf} will depend on the trade-off between the changes in vegetation water use and infiltration associated with forest regeneration (Bruijnzeel, 1989; Scott et al., 2005).

The following hypotheses are tested here: (1) there is a negative relationship between the area under regenerating forest and the change in total Q (i.e. $Q_{qf} + Q_{bf}$); (2) Q_{qf} shows a negative relationship with area under regenerating forest and a positive one with area under urbanization; and (3) depending on the trade-off between the changes in vegetation water use and infiltration associated with forest regrowth, Q_{bf} shows either a negative, no, or a positive relationship with the area under regenerating forest. Specific objectives are to quantify the effects of forest regeneration and urbanization on total Q, Q_{qf}, and Q_{qf}.

2 Study area

Puerto Rico (PR) is an island with a tropical maritime climate, located in the north-eastern Caribbean occupying $\sim 8870\,km^2$. The geology is dominated by the volcanic Cordillera Central, with a few major outcrops of plutonic rock (mostly granodiorite), and karstic limestones towards the far north and south (Olcott, 1999). Soils developed in the volcanic substrates are largely clayey Ultisols with a rapidly diminishing saturated hydraulic conductivity (K_s) with depth (Schellekens et al., 2004) whereas the granodiorites produce less clayey Ultisols with a less pronounced K_s profile (Kurtz et al., 2011).

Island-wide mean annual P is ~1700 mm yr^{-1} (Daly et al., 1994, 2003). There is a moderate P seasonality, with the three driest months of the year (January–March) receiving on average ~1200 mm yr^{-1} and the three wettest months of the year (September–November) receiving on average ~3100 mm yr^{-1} (Daly et al., 1994, 2003). The northern and eastern portions of the island receive ~30 % more P due to the rising of the moisture-bearing trade-winds against the slopes of the central mountain range (Calvesbert, 1970; García-Martinó et al., 1996; Daly et al., 2003).

During the second half of the 20th century, socio-economic changes in PR led to migration from (upland) rural areas and (lowland) ubanization (Dietz, 1986). The associated abandonment of pastures and agricultural fields allowed secondary forests to develop over increasingly large areas as time progressed (Thomlinson et al., 1996; Grau et al., 2003; Helmer, 2004; Parés-Ramos et al., 2008; Fig. 1). The dynamics of this forest recovery are well-documented, both in terms of its expansion over time, and forest composition and structure (Aide et al., 1996, 2000; Chinea and Helmer, 2003; Grau et al., 2003). Tree density typically reaches a peak between 25 and 35 years after abandonment whereas species richness and forest structure resemble those of old-growth forest after ca. 40 yr of regeneration (Aide et al., 2000). Total actual evapotranspiration (ET_a) of mature upland forest in the maritime tropical climate of PR is high compared to tropical continental sites (Schellekens et al., 2000), mostly because of enhanced wet-canopy evaporation rates (Holwerda et al., 2006, 2012), and the same may well apply to the island's secondary forests (cf. Giambelluca, 2002).

The locations of the 12 catchments examined here and the respective changes in land cover over the period 1951–2000 (see also Sect. 3.1) are shown in Fig. 1. The size of the catchments ranges from 23 to 346 km^2 (median size 42 km^2), mean annual P varies between 1720 and 3422 mm yr^{-1} (median value 2021 mm yr^{-1}), and the length of simultaneous P, PET, and Q records between 33 and 51 yr (median duration 44 yr; Table 1).

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3 Data and methods

3.1 Land cover

Land-cover maps were obtained for 1951, 1978, 1991, and 2000 (Fig. 1). The maps for 1951 (Brockmann, 1952; Kennaway and Helmer, 2007) and 1978 (Ramos and Lugo, 1994) were derived from aerial photography. Although the maps for 1951 and 1978 were rasterized at a resolution of ∼30 and ∼11 m, respectively, the actual mapping resolution is lower due to the minimum mapping unit used by the photo interpreters. The 1991 and 2000 maps[1] (∼30-m resolution; Helmer and Ruefenacht, 2005) were derived from Landsat data using regression-tree modelling and histogram matching. All land-cover maps were reprojected to a common 0.0001° (∼11 m) geographical grid by nearest-neighbour interpolation. To accommodate the different classification schemes used in the respective mapping exercises, each land-cover class was assigned to a generalized class (Table 2). For each catchment the net changes in urban and forest areas from the start of the simultaneous P, PET, and Q records (Table 1) until 2000 were calculated by linear interpolation of urban- and forest-area time series.

3.2 Streamflow (Q)

All available daily Q records were downloaded from the US Geological Survey (USGS) National Water Information System[2] in December 2012, resulting in an initial dataset of 111 gauging sites. Catchment areas were derived for each site using the PCRaster software (Wesseling et al., 1996) and the USGS National Elevation Dataset (NED) digital elevation model (∼30-m resolution). The following criteria for inclusion here were applied: (1) the USGS published estimate of catchment area deviated by < 10 % from our computed catchment area; (2) the length of the Q record between 1950 and 2005 was > 30 yr; and (3) the catchment was not subject to flow regulation or affected by

[1] Downloaded from https://www.sas.upenn.edu/lczodata/ in June 2011.

[2] Accessible at http://waterdata.usgs.gov/nwis.

major anthropogenic water extraction. The latter was assessed using annual USGS Water-Data Reports[3] and a map of water supply intakes in the Luquillo Experimental Forest (Crook et al., 2007; an island-wide map of intakes is lacking). The final dataset comprised 12 catchments (Fig. 1 and Table 1). For the conversion of measured discharge [$\text{feet}^3\,\text{s}^{-1}$] to areal mean Q [$\text{mm}\,\text{d}^{-1}$] the computed catchment area was used. The following four Q metrics were calculated, on an annual basis, to study changes in Q through time: (1) the annual 95th percentile (percent time not-exceeded) daily Q (Q_{p95} [$\text{mm}\,\text{d}^{-1}$]; indicative of peak flows); (2) the annual mean Q (Q_{tot} [$\text{mm}\,\text{yr}^{-1}$]; indicative of total water yield); (3) the annual 5th percentile daily Q (Q_{p5} [$\text{mm}\,\text{d}^{-1}$]; indicative of low flows); and (4) the annual mean dry-season (January–March) flow (Q_{dry} [$\text{mm}\,\text{yr}^{-1}$]).

For four catchments the daily Q strongly exceeded the daily P (see Sect. 3.5) on one or more days, indicating errors in the Q and/or P data. To prevent such errors from biasing the results, for some catchments parts of the Q record were excluded from the analysis. Specifically, for the Bauta catchment data for 1996–1998 were excluded, for the Grande de Loíza catchment data prior to 1961 were excluded, for the Fajardo catchment data for 1989 were excluded, and for the Inabón catchment data for 1975 were excluded.

3.3 Precipitation (P)

Daily P data from the Global Historical Climatology Network-Daily (GHCN-D) database[4] (Gleason, 2002) and from the El Verde station in north-eastern PR[5]. After quality checks, The entire record from the Cerro Maravilla station (included in the GHCN-D database) and 1991–1994 data from the El Verde station were excluded from the analysis. Stations with a record > 20 yr were selected from the GHCN-D database, resulting in a dataset comprising 70 stations (including the El Verde station). Figure 2

[3]Available at http://wdr.water.usgs.gov/.

[4]Downloaded from ftp://ftp.ncdc.noaa.gov/pub/data/ghcn/daily/ in December 2010.

[5]Downloaded from http://luq.lternet.edu/data/lterdb14/data/evrain.htm in July 2011.

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presents the number of P observations available for each day between 1955 and 2010. In addition, a map[6] of mean annual P derived using the Precipitation-elevation Regressions on Independent Slopes Model (PRISM) method was used (based on data from 1963–1995; Daly et al., 1994, 2003) to ensure reliable long-term, elevation-corrected mean annual P for the catchments. The method used to obtain time series of daily P for each catchment is described in Sect. 3.5.

3.4 Potential evapotranspiration (PET)

The present study used the empirical Hargreaves equation (Hargreaves and Samani, 1985) according to guidelines set by the UN Food and Agriculture Organization (FAO; Allen et al., 1998) to assess long-term changes in the evaporative situation of the study catchments. The Hargreaves equation (Hargreaves and Samani, 1985) reads:

$$\mathrm{PET} = 0.0023\sqrt{T_{\max} - T_{\min}} \times \left(\frac{T_{\max} + T_{\min}}{2} + 17.8\right) R_a, \quad (1)$$

where $T_{\min}$ and $T_{\max}$ are the daily minimum and maximum air temperature [°C], respectively, and R_a is the extraterrestrial radiation [mm d^{-1}]. R_a was computed as described by Allen et al. (1998).

The choice for the empirical Hargreaves method was motivated on the one hand by the lack of available long-term data for the climatic variables required for more physically based methods such as the Penman-Monteith equation (Monteith, 1965), and on the other hand by the better availability of $T_{\min}$ and $T_{\max}$ data. Daily $T_{\min}$ and $T_{\max}$ time series (> 20 yr) are available for 21 stations in PR within the GHCN-D database (Gleason, 2002). Figure 2 presents the number of daily $T_{\min}$ and $T_{\max}$ observations between 1955 and 2010. Maps of mean annual $T_{\min}$ and $T_{\max}$ from the WorldClim dataset[7]

[6] Downloaded from http://www.wcc.nrcs.usda.gov/ftpref/support/climate/prism/ in September 2011.

[7] Downloaded from http://www.worldclim.org in September 2011.

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(~1-km resolution; Hijmans et al., 2005) were also used to ensure reliable mean annual values. The approach followed to obtain catchment-wide daily mean time series of T_{min} and T_{max} is outlined in Sect. 3.5. The Hargreaves method has been evaluated successfully against PET based on the Penman–Monteith equation (Trajkovic, 2007) and various other equations (Lu et al., 2005; Sperna Weiland et al., 2012).

3.5 Spatio-temporal interpolation and rescaling of climatic variables

To calculate catchment-wide daily mean time series of the climatic variables P, T_{min}, and T_{max} from 1955 (marking the start of many records) to 2010 an approach was developed that: (1) removed the linear trend from the time series for each station and variable; (2) spatially interpolated the trend and daily-irregular components; and (3) reunited them on a per-pixel basis; after which (4) the time series were rescaled. More specifically, the following steps were carried out for each variable (where X denotes the variable in question):

1. T_{min} and T_{max} data were converted from °C to K. For each station with record length > 20 yr trends were calculated from annual mean X time series using the non-parametric Mann–Kendall statistical test (Kendall, 1975; Mann, 1945) with Sen's estimate of slope (Sen, 1968).

2. For each station daily X time series were de-trended and divided by the station mean such that the new mean is unity.

3. Daily maps with a resolution of 0.02° (~2 km) were computed from time series of X trend (having a constant value for each station) and from time series of de-trended unity X values using spatial interpolation (see next paragraph).

4. On a per-pixel basis the cumulative integral of the trend was calculated and offset such that the mean is unity, resulting in time series of net X change.

5. By multiplying the time series of net X change by time series of de-trended unity X values, time series of unity X values were obtained.

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6. The time series of unity X values were multiplied times a map of elevation-corrected long-term mean X (PRISM for P, and WorldClim for T_{min} and T_{max}; see Sects. 3.3 and 3.4, respectively).

7. Finally, daily X time series were calculated for each catchment by averaging over all the pixels comprising each catchment, and the T_{min} and T_{max} time series were re-converted from K to °C.

The most common techniques used for spatial interpolation (Hartkamp et al., 1999) are Thiessen polygons (Thiessen, 1911), inverse-distance weighting (IDW; Shepard, 1968; Dirks et al., 1998), kriging (Krige, 1951), and splining (Tait et al., 2006). Given the large number of maps that required interpolation, here we employed the computationally efficient IDW technique. IDW computes a value at an unsampled location (X_0) from the weighted neighbourhood observations (X_j) as follows:

$$X_0 = \frac{\sum_{j=1}^{R} X_j v_j}{\sum_{j=1}^{R} v_j}, \tag{2}$$

where j denotes the j-th observation, v_j are the weights given to neighbouring observations, while the summation is over $j = 1, 2, ..., R$. The weights were calculated according to:

$$v_j = \frac{1}{g_j^c}, \tag{3}$$

where g_j is the distance between X_j and X_0, and the exponent c determines the distance-decay relationship (i.e. a higher value of c indicates more weight is given to nearby stations). Although it is recognized that c varies per location (e.g. Lu and Wong, 2007), following recommendations by Garcia et al. (2008), and to reduce computational time, here a constant value of 3 for c was assumed.

The present approach has two advantages over the customary approach of using the nearest station with the longest record. Firstly, information from all nearby stations with records > 20 yr is incorporated. Secondly, our approach ensures unbiased mean annual values by rescaling against elevation-corrected long-term means. On the other hand, there are two caveats. First of all, portions of a time series may originate from different stations, thereby introducing spurious signals if, for instance, they have different seasonal patterns. Furthermore, for P, step six of the approach merely changes the intensity of the series, and does not correct for storm events unsampled by the isolated "point" network of meteorological stations.

3.6 HBV-light model

The HBV-light model (Seibert, 2005) was used to simulate Q. HBV-light is a spatially-lumped, conceptual rainfall-runoff model based on the HBV model (Bergström, 1976). HBV-light runs at a daily time step, has two groundwater stores and one unsaturated-zone store, and uses daily time series of P, PET, and T as inputs. T was not relevant in the present case since it is only used to drive the snow model subroutine. Rainfall interception is not estimated explicitly in HBV-light but is implicit in the BETA, FC, and UZL parameters. Although the model has been used predominantly in temperate-zone catchments (e.g. te Linde et al., 2008; Steele-Dunne et al., 2008; Driessen et al., 2010), it has also been used successfully in tropical settings and for a range of catchment sizes, e.g. in Fiji (0.63 km^2; Waterloo et al., 2007), Southern Ecuador (75 km^2; Plesca et al., 2012), and Thailand (12 100 km^2; Wilk et al., 2001). For in-depth discussion of the model, see Seibert (2005). Table 3 briefly describes the model parameters and lists the calibration ranges used. We were unable to close the water balance for several catchments after exhausting all possible parameter combinations, probably due to errors in the PRISM P map, Q data and/or catchment boundary data, (non-quantified) water extractions, and/or inter-basin groundwater transfers. Therefore, an additional

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parameter (PCORR) was introduced that scaled P as required to match observed and predicted Q. The influence of the P scaling on the model's results is limited because the simulated Q was only used to control for climate and storage carry-over effects (see Sect. 3.7).

Model parameters for each catchment were calibrated using Latin hypercube sampling (LHS; McKay et al., 1979). LHS is a more efficient alternative (Yu et al., 2001) to the commonly used Monte Carlo technique (Metropolis, 1987; Beven, 1993; Seibert, 1999). Using LHS the parameter space (Table 3) is split up in n equal intervals. Values for the parameters are generated by sampling each interval just once in a random manner. The model is run n times with random combinations of the parameter values from each interval for each parameter. Here $n = 30\,000$ model runs were used to ensure convergence of the performance criterion. The first 10 yr of the record (Table 1) were used as spin-up period to initialize the groundwater stores after which the model was run for the entire period (i.e. the first 10 yr were run twice). The split-sample procedure (Refsgaard, 1997) was used to calibrate the parameters against data from the second half of the period and validate the parameters against data from the first half of the period.

Parameters of hydrological models are commonly calibrated using a composite of objective functions (i.e. a summary statistic incorporating several measures of performance; e.g. Madsen, 2000). Here, three different objective functions that strike a balance between accurate representations of all portions of the hydrograph were used. The first objective function represents the Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970; NS [–]). The NS is defined as:

$$\mathrm{NS} = 1 - \frac{\sum_{t=1}^{k}\left(Q_\mathrm{o}^t - Q_\mathrm{s}^t\right)^2}{\sum_{t=1}^{k}\left(Q_\mathrm{o}^t - \overline{Q_\mathrm{o}}\right)^2}, \quad (4)$$

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where Q_s and Q_o represent 3-D mean simulated and observed Q, respectively [mm d^{-1}], t is the time step [–], and the summation is over $t = 1, 2, ..., k$. The second objective function (NS_{log} [–]) is the NS calculated from log-transformed Q_s and Q_o, thereby giving more weight to low Q values. NS and NS_{log} were calculated from 3-D mean Q to account for the flashy nature of the streams, which resulted in frequent mismatches between daily peaks of observed and simulated Q, thereby confounding the calibration. The third and final objective function represents the volume error (VE [%]):

$$VE = 100\frac{\overline{Q_s} - \overline{Q_o}}{\overline{Q_o}}. \tag{5}$$

For each run, the objective functions were combined to form a single score using the combined-rank method (Booij and Krol, 2010).

The main analysis was conducted using the 30 "best" Q simulations. Parameter uncertainty (e.g. Beven, 1993; Seibert, 1997) was quantified as described in Sect. 3.7. It should be noted that the K2 parameter in HBV-light was not calibrated, but was set to $1 - k_{bf}$ (where k_{bf} is the baseflow recession constant listed in Table 1). In addition, the MAXBAS parameter was not calibrated, but was set to one, since the travel time for the catchments under study is likely to be less than a day (confirmed by exploratory calibration efforts in all catchments). The PCORR parameter was calibrated only for catchments that had absolute VE values > 20 % for the calibration period when PCORR was set to one and parameter combinations were exhausted. For these catchments, the range of PCORR values used for the calibration was an initial estimate ±0.05, where the initial estimate was calculated based on the initially obtained VE values.

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3.7 Evaluation of the impacts of land-cover change on streamflows

For each catchment and Q metric, annual time series of the observations and the 30 best simulations were calculated, and annual time series of the deviations D between the two were calculated as:

$$D_d^x = 100\,\frac{Q_{\mathrm{obs}}^x - Q_{\mathrm{sim},d}^x}{\overline{Q_{\mathrm{obs}}^x}}, \tag{6}$$

where D are the deviation time series [%], Q_{obs} and Q_{sim} are annual time series of the observed and simulated Q metrics [mm yr^{-1}], respectively, x is the year [–], and $d = 1, 2, ..., 30$ are the simulations. D integrates the effects of P and PET variability and basin carry-over storage on the Q metrics, thereby isolating the impact of other factors, notably land-cover change.

For each catchment and Q metric, trend lines were fitted through time series of D prior to 2000, expressed by the following equation:

$$y = a\,x + b, \tag{7}$$

where y is the trend line [%] and a and b are fitted parameters [%]. The parameter a is the slope of the trend, indicating the change in D prior to 2000, when land-cover data is available. For each catchment and Q metric, the estimated value of a (termed $\hat{a}_{\mathrm{J}}$ [%]) and the associated standard error (termed $\hat{\sigma}_{\mathrm{J}}$ [%]) were calculated from the annual time series of deviation D using the non-parametric jackknife resampling technique (Quenouille, 1956; Tukey, 1958). From each annual time series of D consisting of M values, this procedure takes M subsamples of $M-1$ values by omitting a different value each time. The parameter values of Eq. (7) are re-computed for each subsample, using the least-squares method, thus obtaining for each Q metric a 30-by-M matrix of a and b estimates. The mean of the a estimates is the jackknife-estimated value of a ($\hat{a}_{\mathrm{J}}$). The total change ($\hat{A}$ [%]) in D prior to 2000 is calculated by:

$$\hat{A} = \hat{a}_{\mathrm{J}}\,L, \tag{8}$$

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where L [–] is the length of the record until 2000 (cf. Table 1). $\hat{A}$ can be interpreted as the cumulative residual change in annual observed values of the Q metric after accounting for the effects of climate variability and carry-over water storage on the Q metric.

The standard error of $\hat{a}_\mathrm{J}$ consists of two parts. The first ($\hat{\sigma}_\mathrm{s}$ [%]) was calculated from the mean dispersion of $\hat{a}$ within the time series of D, and is indicative of observational errors in P, PET, and Q data, possible defects in the HBV-light model structure, and faulty assumptions in the formulation of Eq. (7). $\hat{\sigma}_\mathrm{s}$ was calculated using the jackknife technique as follows:

$$\hat{\sigma}_\mathrm{s} = \frac{1}{30}\sum_{d=1}^{30}\sqrt{\frac{M-1}{M}\sum_{i=1}^{M}\left(\hat{a}_{di} - \overline{\hat{a}_d}\right)^2}, \quad (9)$$

where $\hat{a}_{di}$ [%] are the estimates of a for subsamples $i = 1, 2, ..., M$ of annual D time series $d = 1, 2, ..., 30$. The second part of the standard error ($\hat{\sigma}_\mathrm{p}$ [%]) was calculated from the dispersion of the mean a amongst the 30 time series of D, and is mainly indicative of parameter uncertainty. It was calculated as:

$$\hat{\sigma}_\mathrm{p} = \mathrm{std}\left(\overline{\hat{a}_{d=1}}, \overline{\hat{a}_{d=2}}, ..., \overline{\hat{a}_{d=30}}\right), \quad (10)$$

where std refers to the standard deviation. The two parts were combined to yield the standard error of $\hat{a}_\mathrm{J}$ ($\hat{\sigma}_\mathrm{J}$ [%]) as follows:

$$\hat{\sigma}_\mathrm{J} = \hat{\sigma}_\mathrm{s} + \hat{\sigma}_\mathrm{p}. \quad (11)$$

The standard error associated with $\hat{A}$ ($\hat{\sigma}_{\hat{A}}$ [%]) is given by:

$$\hat{\sigma}_{\hat{A}} = \hat{\sigma}_\mathrm{J} L. \quad (12)$$

To examine whether changes in forest and/or urban area influenced the four observed Q metrics, correlation coefficients were calculated between amounts of change

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in forest or urban area per catchment and corresponding values of $\hat{A}$. Conventionally a significance level of 0.05 is applied, but this level was adjusted for the number of inferences made (four in the present study) to 0.01 using the Bonferroni procedure (Bland and Altman, 1995).

4 Results

4.1 Changes in climatic variables

Using the new P dataset (1955–2010), which comprises the records from 70 stations, per-pixel trends for the island ranging from -0.25 to $+0.41\,\%\,\mathrm{yr}^{-1}$ (mean value $+0.02\,\%\,\mathrm{yr}^{-1}$) were found (Fig. 3a). In addition, a strong north-south disparity was observed, with positive P trends mainly identified to the south of the Cordillera Central, and negative trends north of it (Fig. 3a). Likewise, the new PET dataset (1955–2010), based on 21 stations, shows similar magnitudes of change, with per-pixel values ranging from -0.30 to $+0.17\,\%\,\mathrm{yr}^{-1}$ (mean value $-0.03\,\%\,\mathrm{yr}^{-1}$), but with a less clear spatial pattern compared to P (Fig. 3b).

4.2 HBV-light model performance

Table 4 lists the calibrated parameter values for the HBV-light model plus objective function scores for the calibration and validation periods for each catchment. Median Nash-Sutcliffe (NS) values of 0.64 and 0.63 (median of 12 values, where each value represents the mean NS value of the 30 best simulations for each catchment) were obtained for the calibration and validation periods, respectively (Table 4). The median absolute VE values were 3.1 and 7.2 % (median of 12 values, where each value represents the mean VE value of the 30 best simulations for each catchment) for the calibration and validation periods, respectively (Table 4). Note that strong land-cover effects would have degraded performance statistics for the validation period. Several

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catchments required optimization of the PCORR parameter (cf. Table 4), possibly due to biases in the PRISM P map, Q and/or catchment boundary data, water extractions, and/or inter-basin groundwater transfers, which combined may cancel out or amplify one another.

4.3 Changes in land cover

Figure 4 shows the land-cover time series for each catchment. In all catchments forest and urban areas increased markedly between 1951 and 2000 at the expense of pasture and agricultural areas. During the period of simultaneous P, PET, and Q recording (Table 1) the net change in forest and urban areas for the study catchments ranged from +2 to +55 % (mean value +26 %) and from +2 to +11 % (mean value +7 %), respectively. The changes in pasture and agricultural areas ranged from −19 to +26 % (mean value −7 %) and from −63 to +9 % (mean value −26 %), respectively.

4.4 Changes in streamflow metrics

For each catchment and Q metric, Fig. 5 shows annual time series of the difference between observed and simulated Q as expressed by D (Eq. 6) and the corresponding fitted trend lines (y in Eq. 7), whereas Table 5 lists values of the total change $\hat{A}$ (Eq. 8) and the corresponding error $\hat{\sigma}_{\hat{A}}$ (Eq. 12). Catchments with low NS values generally gave higher $\hat{\sigma}_{\hat{A}}$ values (cf. Tables 4 and 5). In general D_{tot} shows the least temporal variability (Fig. 5), which is reflected by low $\hat{\sigma}_{\hat{A}}$ values for $\hat{A}_{\mathrm{tot}}$ (Table 5). The low temporal variability of Q_{tot} is likely due to the low degree of sampling error associated with the calculation of annual mean Q. Most catchments show progressive decreases over time in their time series of D (Fig. 5), as indicated by the mostly negative $\hat{A}$ values (Table 5), suggesting decreases in observed Q metrics that are unrelated to carry-over effects of water storage and climate variability. The most pronounced decreases in Q_{tot} were found for catchments B, C, and J, whereas a moderate increase was found for catchment D. Pronounced decreases in both Q metrics related to low flows (Q_{p5} and Q_{dry})

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were found for catchments A, C, F, G, H, J, K, and L, whereas a (moderate) increase was found only for catchment I. Clear decreases in the Q metric related to peak flows (Q_{p95}) were found for catchments B, C, I, and J, whereas strong increases were found for catchments D, H, and L.

4.5 Impacts of land-cover change on streamflows

Figure 6 shows regressions between the amount of urban and forest area change vs. the cumulative change in annual time series of D prior to 2000 (expressed by $\hat{A}$ in Eq. 8) for each Q metric. In spite of strong increases in forest and urban area, and pronounced changes in D over time for the Q metrics of several catchments, all correlations were insignificant ($p \geq 0.389$). Nonetheless, a weak (i.e. non-significant) positive relationship can be observed between changes in forest cover and changes in annual total streamflow Q_{tot}, when excluding catchments C and J (which appear to be outliers; Fig. 6c).

5 Discussion

5.1 Changes in climatic variables

Previous research on long-term changes in P in PR, mostly using a limited number of climate stations, has suggested progressive declines in long-term P over 1900–2000 (Larsen, 2000; Van der Molen, 2002), but to the authors' knowledge no comprehensive analysis has been conducted for the entire island. Using the new P dataset (1955–2010) a strong north-south disparity in terms of trend was observed (Fig. 3a), which may be attributed to changes in wind patterns induced in turn by changes in sea surface temperature (Comarazamy and González, 2011; cf. Van der Molen et al., 2006). Although trends in PET (1955–2010) were just as strong, they showed a less clear spatial pattern. Nevertheless, these findings reaffirm the importance of accounting for climate variability in studies assessing the effects of land-cover changes on Q.

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5.2 Impacts of land-cover change on streamflows

We were unable to accept the three hypotheses because no significant relationships ($p < 0.01$) were found between the change in forest cover or urban area, and the change in the investigated Q metrics across the 12 catchments (Fig. 6). This result agrees with previous meso- to macro-scale catchment studies in the tropics, subtropics, and warm-temperate regions (Table 6), which mostly failed to demonstrate a clear relationship between Q and change in forest area. Our use of multiple catchments proved important since many individual catchments showed pronounced changes in flow characteristics that might well have been attributed to land-cover change otherwise.

Nevertheless, a weak positive relationship was observed between the change in forest cover and the change in Q_{tot} (Fig. 6c), tentatively suggesting that regeneration of forests leads to increases in Q in PR. If true, this would imply that in the investigated catchments the increase in Q due to enhanced rainfall infiltration during forest regrowth overrides the decrease in Q associated with the greater water use of the aggrading forests (cf. Bruijnzeel, 1989; Scott et al., 2005). The soils beneath young secondary forests in PR show higher bulk density than beneath mature secondary forests (Weaver et al., 1987; Lugo and Scatena, 1995) and are less structured (Lugo and Helmer, 2004). Presumably, this reflects soil compaction by livestock and cropping prior to abandonment and subsequent secondary forest regeneration. However, the general level of soil degradation under pasture in PR (cf. Aide et al., 1996) is probably not sufficient to cause widespread occurrence of overland flow, although surface erosion under sugar cane and coffee plantations has been reported to be rampant in parts of the island (e.g. Del Mar López et al., 1998; Smith and Abruña, 1955). Unfortunately, to date, no unequivocal case of a positive relationship between changes in Q and reforested degraded area has been reported in the tropics (cf. Table 6), although demonstrated decreases in the amounts of headwater- or hillslope stormflow generation after reforesting severely degraded land (Chandler and Walter, 1998; Zhou

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et al., 2002; Zhang et al., 2004; Sun et al., 2006) must be considered large enough to overcome the associated increases in forest water use (Chandler, 2006; cf. Bruijnzeel, 2004; Scott et al., 2005). Nevertheless, long-term streamflow data for large, once degraded but subsequently rehabilitated catchments in sub-humid Texas (Wilcox and Huang, 2010) and the humid Red Soils region of South China (Zhou et al., 2010) have recently indicated gradually increased Q_{bf} over prolonged periods of time following large-scale land rehabilitation and regreening, suggesting soil improvement to be the dominant factor in these cases. Further process-based work is required to substantiate this contention. Similarly, pending the results of hydrological process studies in PR's regenerating forests (in particular, infiltration and soil water retention vs. forest water use) the presently obtained apparent positive relationships between the change in forest cover and the change in Q_{tot} may be spurious.

The present investigation is confounded somewhat by modest urbanization occurring simultaneously with forest regeneration in some of the investigated catchments (Fig. 4). Since urbanization typically increases the area of impervious surfaces, thereby enhancing the frequency and intensity of infiltration-excess overland flow, more Q_{qf} is produced (e.g. Harto et al., 1998; DeWalle et al., 2000; Ziegler et al., 2004; Rijsdijk et al., 2007). This, if progressing beyond a critical threshold, can even result in reductions in Q_{bf} (Van der Weert, 1994; Bruijnzeel, 1989; Bruijnzeel, 2004) on top of the reductions incurred already by the higher water use of the regenerating forest (Giambelluca, 2002). However, no significant relationships ($p < 0.01$) between the degree of urbanization and changes in any of the observed Q metrics in our catchments were identified (Fig. 6). This is probably due to insufficient amounts of urbanization occurring (+2 to +11 %, mean value of +7 %; Fig. 4). Similarly, studies of the "flashiness" (sensu Baker et al., 2007) of the runoff behaviour of urbanized and forested catchments in PR revealed no differences (Ramírez et al., 2009; Phillips and Scatena, 2010). However, Phillips and Scatena (2010) did detect significant differences ($p < 0.05$) in the frequency of stage change (cf. McMahon et al., 2003), possibly indicating locally enhanced Q_{qf} due to urbanization.

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Wu et al. (2007) reported a change in Q_{tot} of −25 % (corrected for changes in P using a single P station that was not included in the current study as it did not meet the necessary quality requirements), between 1973–1980 and 1988–1995 for the Río Fajardo catchment in north-eastern PR (here identified as catchment H). Wu et al. (2007) attributed this negative change in Q to the small amount of forest regeneration occurring during that period (estimated at 8 % based on the interpolated time series shown in Fig. 4). For the same catchment, here a similar change in Q_{tot} of −31 % (calculated using D_{tot}) was found between the respective periods. However, a much smaller estimated change of (-5 ± 4) % was obtained between the respective periods when taking into account the complete record (48 yr), using the calculated trend in D_{tot} prior to 2000 ($\hat{a}_J$) and the corresponding standard error ($\hat{\sigma}_J$, Eq. 11). Thus, Wu et al. (2007) may have overestimated the reduction in Q_{tot} for the Río Fajardo catchment because they only used part of the available record. Additionally, given that the Río Fajardo catchment appears to be an outlier in the present analysis (Figs. 6e and 6f), possibly due to non-quantified anthropogenic water extractions, any inferences for this catchment should be viewed with caution.

5.3 Field-based estimates of vegetation water use

To explore the possible changes in total water yield that may be associated with a conversion from shaded coffee, sugar cane or pasture (the dominant land uses in PR in 1951, see Sect. 3.1) to (semi-)mature secondary forest (current situation), it is of interest to compare the typical amounts of total water use (ET_a) by the respective vegetation types. Field-based estimates of transpiration plus soil evaporation for shaded coffee and sugar cane in PR range between, respectively, 1.4–3.2 mm d^{-1} (Lin, 2010; Gutiérrez and Meinzer, 1994) and 2.5–4.6 mm d^{-1} (Vázquez, 1970; Fuhriman and Smith, 1951; Goyal and González-Fuentes, 1989; cf. Harmsen, 2003). The mean soil water uptake of various well-watered (lowland) pastures in northern PR was estimated at 2.8 mm d^{-1} (Van der Molen, 2002), whereas for mature upland (Tabonuco) forests, values converge around (3.0 ± 0.1) mm d^{-1} (Van der Molen, 2002; Wu et al.,

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2006). Thus, despite the deeper root systems of the forests (cf. Nepstad et al., 1994), mean soil water use for the agricultural crops under consideration, pasture, and forests is quite similar, possibly due to the relatively rainy climate prevailing in PR all year round (Calvesbert, 1970). To this, rainfall interception losses (higher for forest) should be added. The best estimates of rainfall interception for mature Tabonuco forest (∼ 21 % of mean annual P; Holwerda et al., 2006) translate to ca. 1.2 mm d^{-1} for an annual P of 2000 mm yr^{-1} (i.e. the approximate average rainfall for the 12 study catchments), vs. 0.9 mm d^{-1} for shaded coffee (Siles et al., 2010) and 0.2 mm d^{-1} for sugar cane (Leopoldo et al., 1981). Extrapolating the above-mentioned average values to a year, gives mean estimated ET_a totals for shaded coffee, sugar cane, and pasture of, respectively, ca. 1170, 1355, and 1020 mm yr^{-1} vs. ca. 1515 mm yr^{-1} for mature forest. Thus, the full conversion from pasture or crop land to mature forest could potentially change Q_{tot} by about −160 mm yr^{-1} (in the case of sugar cane) to −495 mm yr^{-1} (in the case of pasture).

For the catchments examined here a mean modeled change in Q_{tot} of −86 mm yr^{-1} (standard deviation ±124 mm yr^{-1}) was found (calculated from values of $\hat{A}_{tot}$ and Q_{tot} listed in Tables 5 and 1, respectively). If it is assumed that this change was solely due to the mean change in forest cover of +26 %, the full conversion from pasture or crop to mature secondary forest would have resulted in a change in Q_{tot} of about (−340 ± 480) mm yr^{-1}. Although the modeled change in Q_{tot} lies in between the above estimates based on local field studies of ET_a, the very high standard deviation of the estimated modeled change in Q_{tot} suggests that this agreement may be mere chance. Therefore, the question re-emerges as to why analysis of the impacts of land-cover change on Q in meso-scale tropical catchments does not confirm the findings of micro-scale experimental studies, which clearly demonstrate a relationship between change in forest area and change in Q (Bosch and Hewlett, 1982; Bruijnzeel, 1990; Sahin and Hall, 1996; Brown et al., 2005; Jackson et al., 2005).

5.4 Potential explanations for the lack of relationships

Upscaling in hydrology is an outstanding issue that has received considerable attention (e.g. Peterson, 2000; Rodriguez-Iturbe, 2000; McDonnell et al., 2007; Blöschl and Montanari, 2010). For many studies the lack of a clear relationship between deforestation and change in Q can be explained by the rapid regeneration of tropical forests, where ET_a quickly reverts back to its original level and possibly exceeds it for decades (Giambelluca, 2002; Bruijnzeel, 2004; Juhrbandt et al., 2004). However, this explanation does not apply to the present study, since the catchments were exploited as pastures or agricultural fields for a sustained period of time prior to their abandonment and subsequent forest regeneration. Another possible reason for the inconclusive results obtained by many studies is the use of seasonal or annual mean Q metrics only, whereas it is generally preferred to use metrics related to the frequency and magnitude of Q (Alila et al., 2009, 2010), particularly when evaluating the change in peak flows. Finally, the failure to correct for climatic variability by many studies may also have led to inconclusive, or even erroneous, outcomes.

The first among the potential explanations for the lack of relationship between land-cover change and Q metrics in PR is covariance between land cover and climate in space, due to landscape ecology and history (cf. Van Dijk et al., 2012), or in time, due to land-cover changes altering the climate (Van der Molen, 2002; Pielke et al., 2007). Several meso-scale atmospheric modelling studies have produced conflicting results regarding the existence of such a relationship for PR (Van der Molen et al., 2006; Comarazamy and González, 2011). The contrasting P trends reported here for different parts of the island (Fig. 3a) are also unable to provide evidence since similar magnitudes of forest regeneration occur across the island (Fig. 1). Also, because observed P and PET time series were input to the HBV-light model, most of the covariance between land-cover change, and P and PET should have been accounted for in the present analysis.

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A second potential explanation is that the land-cover, P, PET, Q, and catchment boundary data used here contain more errors than those used in small, commonly well instrumented, experimental studies. Using data for 278 Australian catchments Van Dijk et al. (2012) showed that the introduction of noise to long-term means of observed PET, P, and Q reduced the likelihood of detecting a land-cover signal. Here, as different methodologies were employed to derive the land-cover maps for the different years, and because the maps had different spatial resolutions and classification schemes (see Sect. 3.1), it is conceivable that the land-cover time series contained errors. For P a higher station density is probably desirable, particularly in tropical areas where the uncertainty in estimated catchment-wide P is exacerbated by the predominantly convective character of P and the limited spatial extent of individual storm cells (Nieuwolt, 1977; Hastenrath, 1991). Likewise, the PET data should also be viewed with caution, since net radiation, wind speed, and relative humidity, other dominant factors controlling PET magnitude and trends (see McVicar et al., 2012, and references therein), were not explicitly included in its calculation (cf. Eq. 1). The error in the PR Q data is about 3–6 % (Sauer and Meyer, 1992). The cumulative error in annual P, PET, and Q was quantified by $\hat{\sigma}_{\hat{A}}$ (Eq. 12). Even though $\hat{\sigma}_{\hat{A}}$ does not account for the error in the land-cover data or for drift errors in the P, PET, and/or Q data during the study period, values of $\hat{\sigma}_{\hat{A}}$ already constitute a substantial portion of the variance in $\hat{A}$ values (the estimated total change in the observed Q) for all Q metrics (cf. Table 5, and Fig. 6). This suggests that errors in land-cover, P, PET, and Q data may have restricted the detection of a relationship between land-cover change and Q metrics in PR. Kundzewicz and Robson (2004) and Chappell and Tych (2012) also suggested that a high degree of observational error may mask the identification of Q change.

A third potential explanation is that the amount of forest area change in the investigated catchments is less than that of most small catchment experimental studies. Based on small catchment experiments the critical threshold value for forest area change beyond which changes in Q can be detected is generally assumed to be ca. 20 % (e.g. Bosch and Hewlett, 1982). Increases in forest area for the investigated

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catchments were 2 to 55 % (mean 26 %), with increases exceeding 20 % in six of the twelve catchments. However, forest regeneration progressed from the headwaters to the lowlands (Grau et al., 2003) and by the time the records started (cf. Table 1) most of the headwaters would have been reforested already. Since most Q is produced in these rainier headwater areas (García-Martinó et al., 1996) the observed overall change in Q may be far less than expected on the basis of lumped representations of P and forest area.

A fourth potential explanation relates to anthropogenic water extractions from the investigated catchments. Irrigation on agricultural fields from local wells may have increased ET_a, thereby masking the effect of increasing forest cover on Q. However, irrigation of sugar cane and pineapple in PR occurs only on the drier southern coastal plains, and most irrigation water originates from lakes outside the study catchments (Molina-Rivera and Gómez-Gómez, 2008). Therefore, irrigation effects can probably be excluded as a possible explanation. However, water withdrawals associated with urbanization may have confounded the results, possibly causing reductions in Q_{bf} that are unrelated to changes in forest cover. Although catchments affected by major water extraction were excluded from the analysis (see Sect. 3.2), it is possible that some catchments contain undocumented water intakes.

A fifth potential explanation, proposed by Zhou et al. (2010) and applicable to humid tropical settings, is that ET_a under such conditions is constrained by PET (i.e. energy limited; Calder, 1998), and therefore the expected increases in ET_a due to forest regrowth are not evident in the Q record. However, the interception evaporation component of ET_a can be well in excess of PET, particularly on mountainous maritime islands (Schellekens et al., 1999; Roberts et al., 2005; Holwerda et al., 2006; McJannet et al., 2007; Giambelluca et al., 2009; Holwerda et al., 2012). Moreover, an insignificant correlation was obtained between forest cover change and mean Q during the dry season (January–March; Q_{dry}), when PET is not expected to be a limiting factor (Fig. 5g). Hence, this is not seen as a convincing explanation.

Finally, a sixth potential explanation concerns heterogeneity between the study catchments in terms of vegetation cover and land-use history prior to land abandonment (not accounted for by the semi-quantitative classification of the present study), morphology, geology, and soils (McDonnell et al., 2007), influencing the response of Q to changes in forest cover. In addition, the use of lumped values for the climate variables at the catchment scale may not be valid due to spatial heterogeneity of P–PET–Q relationships (e.g. García-Martinó et al., 1996; Blöschl et al., 2007; Donohue et al., 2011).

6 Conclusions

By and large, the present analysis found little evidence to support the hypothesis that there is a relationship between the degree of secondary forest regeneration and the change in Q for 12 meso-scale catchments in PR. Weak positive relationships were found between the change in forest cover, and the change in annual mean streamflow Q_{tot} and annual 5th percentile flow Q_{p5}, tentatively suggesting that regeneration of forests in PR leads to increased Q through improved rainfall infiltration conditions overriding the enhanced vegetation water use associated with forest regeneration. However, the corresponding drops in quickflow Q_{qf} production, and the required degree of land degradation prior to reforestation for this to happen are mostly lacking. As such, this apparent positive relationship must be considered spurious.

The most likely reasons for this lack of a clear hydrological signal include: (1) errors in the land-cover, climate, Q, and/or catchment boundary data used in the analysis; (2) Q is generated mostly in the rainier headwater areas that were already forested at the start of Q observations whereas subsequent changes in forest area mainly occurred in the drier lowlands; and (3) heterogeneity in the catchment response. Overall, it seems that the hydrological impacts of forest regeneration in PR are currently unpredictable at the meso scale, and further advances in our understanding are hampered by the quality, availability, and record length of the climatic and flow data. Our findings

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highlight the importance of catchment-scale analyses using multiple catchments but at the same time confirm the need for additional process studies at different stages of forest regeneration, notably of vegetation water use (both rainfall interception and transpiration) and changes in low-land hillslope hydrological response.

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Table 1. Background details and basic hydrological characterization of the catchments used in the present study. The catchments are identified by the letters A–L in the header.

	A: Tanamá	B: Bauta	C: Grande de Manatí	D: Cibuco	E: Grande de Loíza	F: Valenciano	G: Canóvanas	H: Fajardo	I: Grande de Patillas	J: Inabón	K: Portugués	L: Culebrinas
Catchment details												
USGS ID	028000	034000	035000	038320	055000	056400	061800	071000	092000	112500	115000	147800
Gauge latitude [° N]	18.301	18.236	18.323	18.352	18.242	18.217	18.318	18.296	18.035	18.086	18.078	18.360
Gauge longitude [° W]	66.782	66.455	66.459	66.336	66.010	65.925	65.889	65.692	66.034	66.562	66.634	67.089
Catchment area[a] [km^2]	47.1	43.5	345.9	39.8	233.0	41.5	26.5	39.3	46.3	24.5	22.9	177.1
Catchment mean elevation [m a.s.l.]	571	734	578	247	265	170	461	272	442	558	568	167
Catchment elongation ratio[b] [–]	0.66	0.57	0.69	0.84	0.73	0.72	0.71	0.69	0.76	0.61	0.52	0.62
Catchment mean slope [°]	16.2	21.5	19.6	13.1	13.3	8.9	14.4	16.0	20.1	24.1	22.8	10.0
Record[c] start year	1960	1970	1957	1970	1960	1971	1968	1962	1966	1965	1965	1968
Record[c] end year	2010	2010	2010	2006	2010	2010	2010	2009	2010	2010	1997	2010
Record[c] length [yr]	51	34	51	35	51	40	43	48	45	44	33	43
Dominant crop(s) in 1951[d]	c	c	c	c, sc, p	c, sc, f	c, sc, f	c, sc, f	sc	c, sc, f	c	c	c, sc
Dominant geology[e]	v, gd, ls	v	v, gd	v, gd	v, gd, q	gd	v	v, gd, q	v, gd	v, gd	v, gd	v, ls
Long-term catchment means of hydroclimatic parameters calculated for record[c] period												
Baseflow recession constant (k_{bf} [mm d^{-1}])	0.956	0.942	0.949	0.935	0.940	0.927	0.943	0.919	0.925	0.926	0.929	0.959
Precipitation (P [mm yr^{-1}])	2102	2038	2004	1827	1963	2039	3422	2875	1720	1781	1875	2178
Total streamflow (Q_{tot} [mm yr^{-1}])	953	812	653	642	845	1067	942	1529	1185	681	743	1509
Baseflow (Q_{bf} [mm yr^{-1}])	680	389	341	320	420	394	463	656	601	441	388	673
Quickflow (Q_{qf} [mm yr^{-1}])	273	423	312	322	426	673	480	872	584	241	354	836
$P-Q$ [mm yr^{-1}]	1149	1225	1351	1185	1118	972	2480	1347	535	1100	1132	669
Potential evapotranspiration (PET [mm yr^{-1}])	1487	1407	1456	1535	1542	1548	1399	1449	1514	1502	1519	1658
Runoff coefficient (Q_{tot}/P [–])	0.453	0.399	0.326	0.351	0.431	0.523	0.275	0.532	0.689	0.383	0.396	0.693
Aridity index (P/PET [–])	1.42	1.46	1.38	1.18	1.28	1.33	2.46	1.97	1.15	1.18	1.23	1.31
Baseflow index[f] (Q_{bf}/Q_{tot} [–])	0.714	0.479	0.522	0.499	0.496	0.369	0.491	0.429	0.507	0.647	0.522	0.446

[a] Computed catchment area; see Sect. 3.2. [b] Calculated by dividing the diameter of a circle having the same area as the catchment, by the maximum length of the catchment (Schumm, 1956). [c] Record is defined as the period of simultaneous P, PET, and Q data. Several catchments have gaps in their record. [d] After Brockmann (1952) and Kennaway and Helmer (2007). Explanation of abbreviations used: c, shade coffee; sc, sugar cane; f, fruits and orchards; p, pineapple. See Sect. 3.1 for details. [e] After Olcott (1999). Explanation of abbreviations used: v, mixed volcanic; gd, granodiorite; ls, limestone; q, Quaternary alluvium. See Sect. 2. [f] Separated using a forward- and backward-recursive filter Van Dijk, 2010.

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Table 2. Generalized land-cover classes as used here and their relation to the specific land-cover classes distinguished in the 1951, 1978, and 1991/2000 land-cover maps. The numbers correspond to the values for the classes used in the original land-cover maps.

Generalized classes	1951 map	1978 map	1991/2000 maps
Urban	1. Urban and coastal sand	12. Urban	1. High-medium density urban 2. Low-medium density urban
Bare ground		10. Rocky areas	20. Salt or mud flats 25. Quarries 26. Coastal sand and rock 27. Bare soil (including bulldozed land)
Pasture	4. Pasture and grass	3. Pasture	5. Pasture, hay or inactive agriculture 6. Pasture, hay or other grassy areas
Wetland	6. Non-forested wetlands	8. Mangroves	9. Non-forested wetland 19. Emergent wetlands including seasonally flooded pasture 21. Mangrove 23. Pterocarpus swamp
Agriculture	2. Herbaceous agriculture and hay 3. Coffee and mixed woody agriculture	2. Agriculture	3. Herbaceous agriculture 4. Active sun coffee and mixed woody agriculture
Forest	5. Forest, woodlands, shrublands, and forested wetlands 12. Unknown	4. Dense canopy forest 1 5. Dense canopy forest 2 6. Less dense canopy forest 7. Woodland and shrubland	7. Drought deciduous open woodland 8. Drought deciduous dense woodland 9. Deciduous, evergreen coastal and mixed forest or shrubland 10. Semi-deciduous and drought deciduous forest on alluvium and non-carbonate substrates 11. Semi-deciduous and drought deciduous forest on karst/limestone (includes semi-evergreen forest) 12. Drought deciduous, semi-deciduous and seasonal evergreen forest on serpentine 13. Seasonal evergreen and semi-deciduous forest on karst 14. Seasonal evergreen and evergreen forest 15. Seasonal evergreen forest with coconut palm 16. Seasonal evergreen and evergreen forest on karst 17. Evergreen forest on serpentine 18. Sierra palm, transitional and tall cloud forest 22. Seaonally flooded savannahs and woodlands 24. Tidally flooded evergreen dwarf-shrubland and forb vegetation 29. Elfin and sierra palm cloud forest
Water body	10. Water/other	11. Water body	0. Water 28. Water (permanent)

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Table 3. HBV-light model parameter units, descriptions, and ranges used for the calibration. The K2 and MAXBAS parameters were not calibrated, but were set to $1 - k_{bf}$ (cf. Table 1) and one, respectively. For the Valenciano, Canóvanas, Grande de Patillas, and Culebrinas catchments calibration ranges of 1.2–1.3, 0.7–0.8, 1.4–1.5, and 1.4–1.5, respectively, were used for PCORR. For the remaining eight catchments PCORR was set to one. See Sect. 3.6 for details.

Parameter	Units	Description	Minimum	Maximum
BETA	–	Shape coefficient of recharge function	1	6
FC	mm	Maximum soil moisture storage	50	750
K0	d^{-1}	Recession coefficient of upper zone	0.05	0.99
K1	d^{-1}	Recession coefficient of upper zone	0.01	0.50
K2	d^{-1}	Recession coefficient of lower zone	–	–
LP	–	Soil moisture value above which actual ET_a reaches PET	0.05	0.95
MAXBAS	d	Length of equilateral triangular weighting function	–	–
PERC	$mm\,d^{-1}$	Maximum percolation to lower zone	0	5
UZL	mm	Threshold parameter for extra outflow from upper zone	0	100
PCORR	–	P correction factor required to close water budget	–	–

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Table 4. Calibrated parameter values for the single best simulation by the HBV-light model, and objective-function scores (mean of the 30 best simulations) for the calibration and validation periods, computed from 3-day means. The PCORR parameter was only calibrated for four catchments.

	A: Tanamá	B: Bauta	C: Grande de Manatí	D: Cibuco	E: Grande de Loíza	F: Valenciano	G: Canóvanas	H: Fajardo	I: Grande de Patillas	J: Inabón	K: Portugués	L: Culebrinas
Calibrated parameter values for the simulation with the best performance												
BETA [–]	1.351	1.306	0.940	2.887	2.999	0.862	3.204	4.212	1.056	1.041	1.117	3.633
FC [mm]	995	927	651	493	874	394	918	889	514	987	918	893
K0 [d^{-1}]	0.934	0.521	0.666	0.756	0.611	0.699	0.778	0.752	0.462	0.514	0.337	0.508
K1 [d^{-1}]	0.284	0.353	0.366	0.210	0.441	0.274	0.449	0.270	0.489	0.246	0.323	0.492
LP [–]	0.685	0.328	0.146	0.854	0.888	0.144	0.443	0.901	0.525	0.443	0.420	0.851
PERC [mm d^{-1}]	4.645	2.571	1.697	1.807	1.757	1.490	1.832	2.706	3.303	3.065	4.524	3.077
UZL [mm]	80	3	4	0	7	4	8	0	7	65	2	19
PCORR [–]	1.000	1.000	1.000	1.000	1.000	1.215	0.712	1.000	1.415	1.000	1.000	1.400
Objective-function scores for the calibration period, computed from 3-day means												
NS [–]	0.62	0.63	0.63	0.65	0.77	0.72	0.75	0.65	0.78	0.58	0.46	0.37
NS_{log} [–]	0.60	0.71	0.76	0.42	0.76	0.60	0.56	0.58	0.68	0.56	0.53	0.60
VE [%]	−6.22	4.11	11.97	−1.63	−7.56	1.78	12.63	2.22	−0.69	−0.09	−13.26	−1.01
Objective-function scores for the validation period, computed from 3-day means												
NS [–]	0.50	0.64	0.68	0.71	0.73	0.68	0.63	0.54	0.77	0.62	0.40	0.52
NS_{log} [–]	0.60	0.68	0.60	0.63	0.74	0.63	0.50	0.53	0.67	0.67	0.60	0.77
VE [%]	−1.27	−16.91	−13.65	0.44	−8.83	−3.95	10.81	−7.57	−1.91	−15.64	−6.89	−4.79

Table 5. Cumulative change in the difference between observed and simulated Q metrics, as expressed by $\hat{A}$ (Eq. 8), and corresponding standard errors ($\hat{\sigma}_{\hat{A}}$; Eq. 12) for the 12 study catchments. The + and − signs indicate, respectively, increases and decreases in $\hat{A}$. The change covers the start of the record (cf. Table 1) until 2000.

	A: Tanamá	B: Bauta	C: Grande de Manatí	D: Cibuco	E: Grande de Loíza	F: Valenciano	G: Canóvanas	H: Fajardo	I: Grande de Patillas	J: Inabón	K: Portugués	L: Culebrinas
$\hat{A}_{p95}$ [%]	−4 ± 13	−27 ± 24	−61 ± 31	+17 ± 22	+8 ± 15	−6 ± 22	−3 ± 31	+39 ± 21	−19 ± 22	−53 ± 19	+8 ± 29	+19 ± 16
$\hat{A}_{tot}$ [%]	−5 ± 8	−15 ± 12	−37 ± 14	+10 ± 12	−2 ± 7	−19 ± 15	−13 ± 13	−6 ± 13	−16 ± 9	−34 ± 12	+7 ± 18	+9 ± 10
$\hat{A}_{p5}$ [%]	−12 ± 17	−13 ± 23	−52 ± 18	−17 ± 22	−33 ± 19	−58 ± 39	−65 ± 35	−147 ± 34	+17 ± 19	−22 ± 19	−12 ± 22	−29 ± 22
$\hat{A}_{dry}$ [%]	−10 ± 16	+5 ± 33	−49 ± 21	+1 ± 16	+13 ± 19	−42 ± 33	−44 ± 23	−48 ± 20	+3 ± 21	−32 ± 17	−13 ± 28	−7 ± 24

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Table 6. Previous studies on tropical, subtropical, and warm-temperate meso- ($\geq 1\,km^2$) and macro-scale ($\geq 10\,000\,km^2$) catchments investigating the impact of changes in forest area on Q, listed alphabetically by country name. Only findings based on time-series analysis were listed. Present study included for completeness. In the "Q change" column "No change" also includes insignificant changes in flows. Where possible the reported change in Q was corrected for P variability. The "No. of gauges" column lists the number of stream gauges used in the analysis. The + and − signs indicate, respectively, increases and decreases in forest area or Q. The mean annual PET values represent reported estimates or Penman–Monteith estimates taken from Fisher et al. (2011) when no estimate was reported. Abbreviations: P, precipitation; PET, potential evapotranspiration; and CO, carry-over water storage.

Region (country)	No. of gauges	Area [km^2]	Mean annual P [$mm\,yr^{-1}$]	Mean annual PET [$mm\,yr^{-1}$]	Forest area change [%]	Time series length [yr]	Corrections applied	Q change [%]	Reference
Comet, Upper Burdekin (Australia)	2	16 440, 17 299	650, 690	1680, 1930	−45, −25	62, 13	P, PET	No change	Peña-Arancibia et al. (2012)
Tocantins (Brazil)	1	767 000	1600	1600	−19	40	P	+24	Costa et al. (2003)
Ji-Paraná (Brazil)	7	33 012	1875	1450	−50	23	P	No change	Linhares (2005) Rodriguez et al. (2010)
P. en Nirivilo, C. en El Arrayán (Chile)	2	253, 708	835, 718	1100, 1100	+5, +9[a]	20	P	−43, −32[b]	Little et al. (2009)
Guangdong Province (China)	32[c]	179 752	1770	1230	+33	51	P, PET	No change	Zhou et al. (2010)
Dakeng (China)	1	10	1530	1050	−19	27	P, PET	−22[d]	Sun et al. (2008)
Hainan Island (China)	12	7 to 729	1950	1350	−29	25	P	No change[e]	Qian (1983)
Upper Yangtze (China)	4	200 000	793	900	+8	41	P	+10[f]	Cheng (1999)
Citarum River (Indonesia)	1	4133	2500	1650	−50	16	P	+11[f]	Van der Weert (1994)
Galas, Kelantan (Malaysia)	2	13 100	2000	1500	−3, +6[g]	31	P	No change	Adnan and Atkinson (2011)
Río Fajardo (Puerto Rico)	1	40	2908	1500	+12	23	P	−25	Wu et al. (2007)
Various (Puerto Rico)	12	23 to 346	1720 to 3422	1393 to 1657	+2 to +55	33 to 51	P, PET, CO	No change[h]	Present study
Nilwala (Sri Lanka)	1	380	2188	1500	−35	58	P	No change	Elkaduwa and Sakthivadivel (1999)
Chao Phraya (Thailand)	2	14 522, 36 000	1150	1500	−45	18, 25	P	No change	Dyhr-Nielsen (1986)
Nam Pong (Thailand)	1	12 100	1350	1700	−63	36	P, PET, CO	No change	Wilk et al. (2001)
Upper Tacuarembó (Uruguay)	1	2097	1300	1230	+26	34	P	−38	Silveira and Alonso (2009)
Southern Piedmont (USA)	10	2820 to 19 450	1300	1050	+10 to +28	49	P	−4 to −21[h]	Trimble et al. (1987)

[a] Represents the net forest area change of native forest cover (−38 and −28 %, respectively) and forest plantations (+43 and +37 %, respectively). [b] Refers to changes in summer Q. [c] Mostly nested catchments. [d] The decrease in Q was explained by the higher ET_a of the regenerating vegetation. [e] Refers to changes in peak flows only. [f] Bruijnzeel (2004) argues that these increases in Q should be ascribed to urbanization rather than tree planting. [g] Represents the net forest area change of native forest cover (−9 and +2 %, respectively) and forest plantations (+6 and +4 %, respectively). [h] No relationship was found between the degree of forest-cover change and the change in Q.

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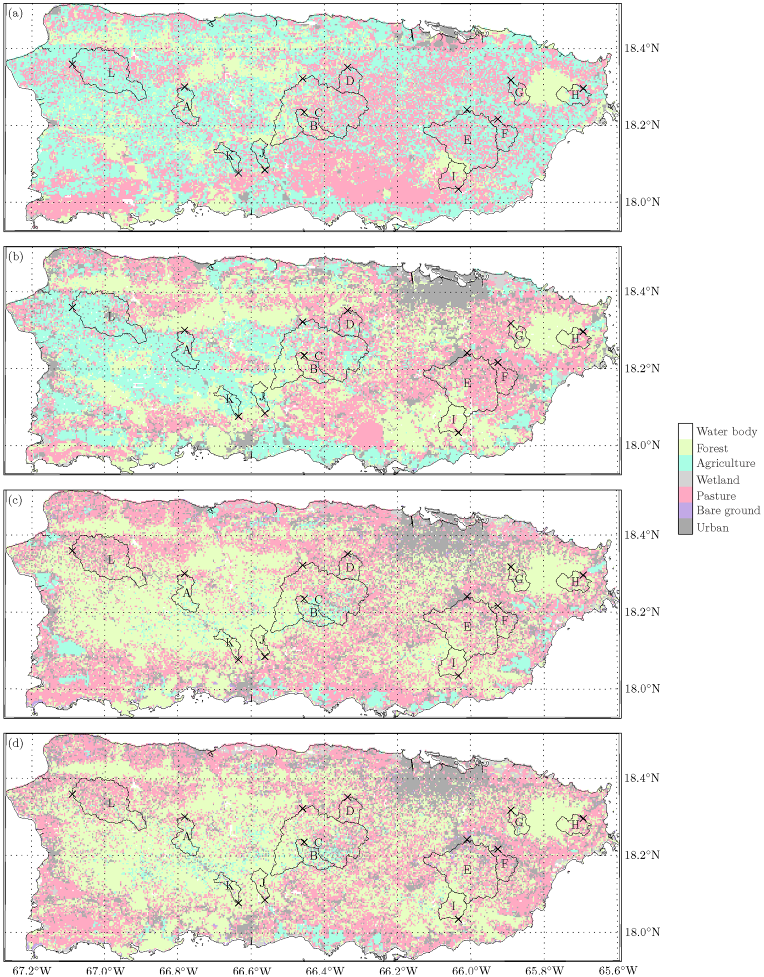

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Fig. 1. Maps showing the locations of the study catchments in PR with the generalized land-cover classifications for: **(a)** 1951; **(b)** 1978; **(c)** 1991; and **(d)** 2000. Maps for 1951 and 1978 courtesy of E. H. Helmer and O. M. Ramos (USDA-IITF), respectively. Catchments identified by letters and stream gauges by crosses. All maps in this paper are presented in the Albers equal area conic projection with latitudinal limits 17.93° N–18.52° N and longitudinal limits 67.27° W–65.59° W.

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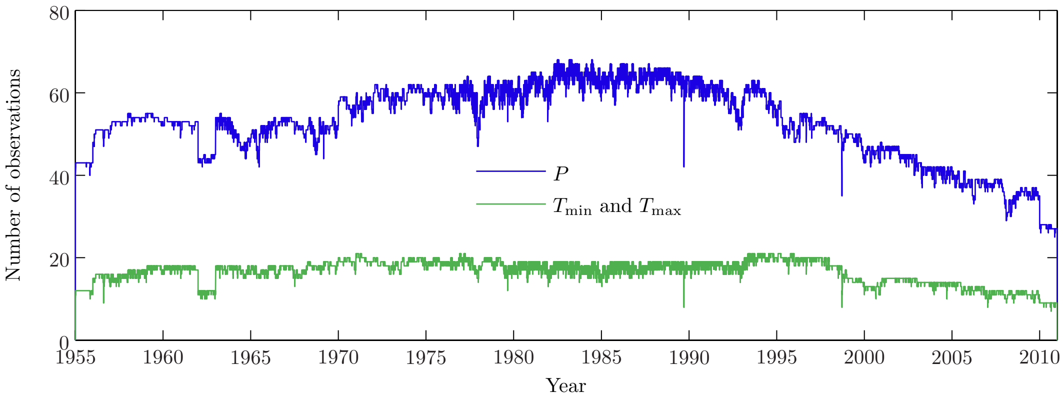

Fig. 2. Graph showing the number of daily observations for P, T_{max}, and T_{min}. Only stations with records > 20 yr were used. The x-axis marks 1 January of each fifth year.

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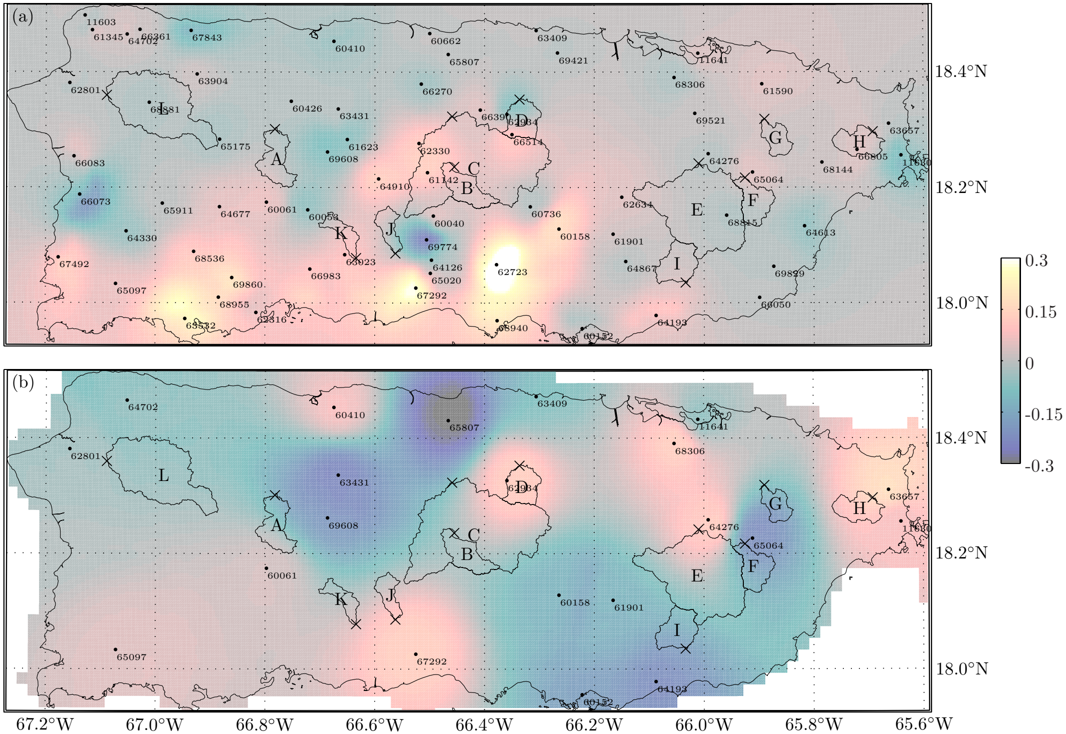

Fig. 3. The trend [% yr^{-1}] from 1955 to 2010 for annual mean **(a)** *P* and **(b)** PET. Significant as well as insignificant trends are shown. The points mark stations with > 20 yr time series of **(a)** *P*, and **(b)** T_{min} and T_{max} (used to derive PET, see Sect. 3.4). The numbers represent the last five digits of the GHCN-D station identification code. Catchments identified by letters and stream gauges by crosses.

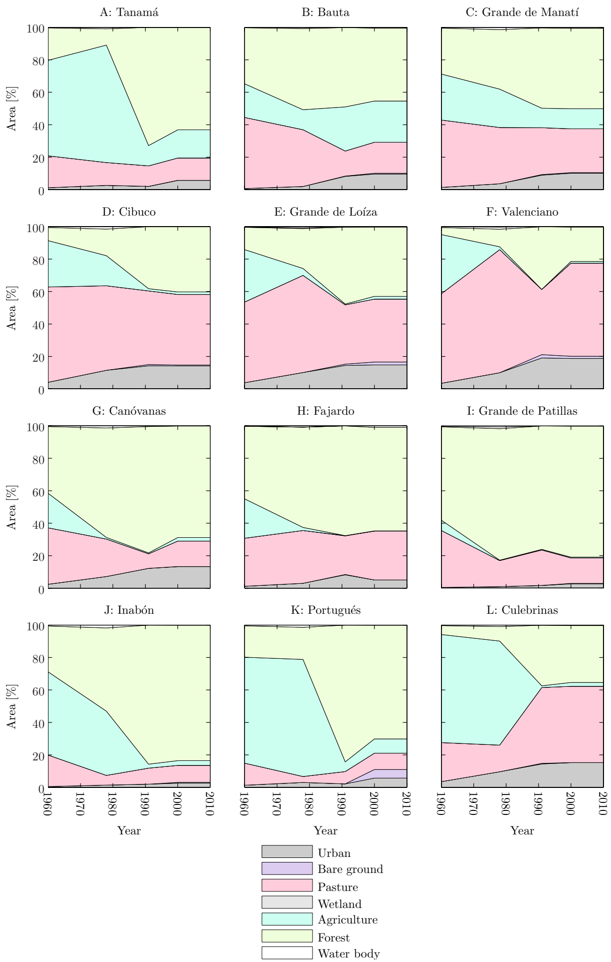

Fig. 4. Caption on next page.

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Fig. 4. Changes with time in area of the dominant land-cover types for the study catchments. Graphs are based on generalized (cf. Table 2) land-cover classification maps for the years 1951, 1978, 1991, and 2000.

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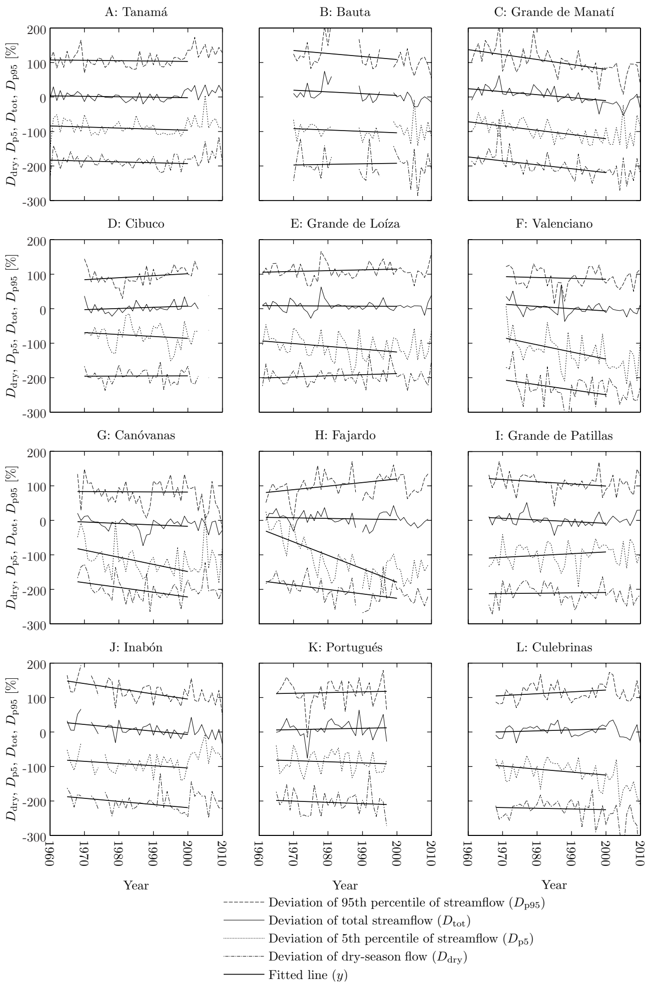

Fig. 5. Caption on next page.

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Fig. 5. Annual time series of D_{p95}, D_{tot}, D_{p5}, and D_{dry} (displayed, respectively, from top to bottom in each panel; calculated using Eq. 6) and fitted lines (Eq. 7). For clarity the D_{p95}, D_{p5}, and D_{dry} time series were offset by +100, −100, and −200 %, respectively.

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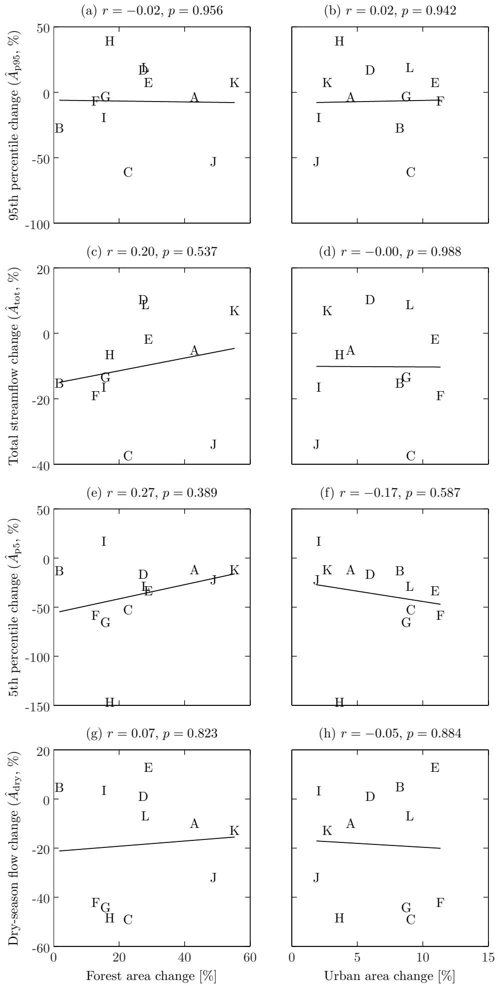

Fig. 6. Caption on next page.

Fig. 6. Net changes in forest and urban areas plotted against A_{p95} **(a–b)**, Q_{tot} **(c–d)**, Q_{p5} **(e–f)**, and Q_{dry} **(g–h)** values (Eq. 8; Table 5). Each letter in the scatter plots represents a different catchment. The change covers the start of the record (cf. Table 1) until 2000.

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