Forest Ecosystems 14 (2025) 100376

Contents lists available at ScienceDirect

Forest Ecosystems

journal homepage: www.keaipublishing.com/cn/journals/forest-ecosystems

# Potential for improved groundwater recharge and dry-season flows through forest landscape restoration on degraded lands in the tropics

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ARTICLE INFO

*Keywords:*
Afforestation
Baseflow
Deforestation
Forest landscape restoration (FLR)
Infiltration trade-off
Reforestation
Soil degradation

ABSTRACT

As interest in tropical forest restoration accelerates, understanding its hydrological implications is increasingly urgent. While concerns persist that reforestation will reduce annual water yields—particularly in drier climates—we highlight conditions under which forest landscape restoration (FLR) can improve seasonal water availability, especially during the dry season. We examine the trade-off between increased vegetation water use ("pumping") and enhanced infiltration and subsurface retention ("sponging") following forestation of degraded lands, the recovery of vegetation's ability to capture "occult" precipitation (fog) in specific coastal and montane settings, and the role of forest cover in enhancing moisture recycling and transport at multiple scales. A pan-tropical sensitivity analysis shows that in degraded landscapes with deep soils and pronounced rainfall seasonality, infiltration gains following forestation can offset or exceed evaporative losses, thereby supporting groundwater recharge and increasing dry-season flows in approximately 10% of cases, with an additional 8% showing near-neutral (slightly negative) outcomes. These findings challenge the assumption that forestation uniformly reduces water availability and underscore the need to prioritize dry-season flow recovery—rather than annual water yield—as a central hydrological goal of FLR. We call for trans-disciplinary research and long-term monitoring to inform forest restoration strategies, particularly in seasonally dry regions where water scarcity is most acute.

## 1. The tropics: a forest landscape restoration (FLR) "hot spot"

In response to the loss and degradation of the world's forests and soils, forest restoration is being scaled up rapidly for the UN Decade on Ecosystem Restoration (Wuepper et al., 2020; Feng et al., 2021; FAO, 2022; UNEP & FAO, 2023). A total of 350 million ha of forest are to be restored by 2030 according to pledges made through the Bonn Challenge alone; other initiatives are similarly ambitious (Stanturf and Mansourian, 2020). The tropics assume a central position in pledged restoration efforts as not only is more than 50% of the world's population projected to live in the tropics by 2050 (Gerland et al., 2014), but the humid tropics also have the greatest potential for tree growth and carbon sequestration (Houghton et al., 2015; Cook-Patton et al., 2020). Furthermore, the dry tropics are amongst the most vulnerable regions

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Peer review under the responsibility of Editorial Office of Forest Ecosystems.

https://doi.org/10.1016/j.fecs.2025.100376
Received 14 March 2025; Received in revised form 20 June 2025; Accepted 24 July 2025

worldwide in terms of vegetation loss, soil degradation, and the number of people living on degraded and deforested land (Gibbs and Salmon, 2015; Barbier and Hochard, 2018; Song et al., 2018). Last but not least, past forest removal in the tropics has caused global temperature increases (Lawrence et al., 2022) and regional precipitation declines (Smith et al., 2023). Looking ahead, major increases in tropical forest cover will be required to address the combined needs for restoring biodiversity, capturing carbon to help mitigate global warming, increasing resilience to climatic change/variability, and supplying sufficient high-quality water to ever-growing urban populations (Bastin et al., 2019; Lenton et al., 2019; Gleeson et al., 2020; Folke et al., 2021).

An adequate and reliable supply of clean freshwater is essential for domestic uses, food production, energy security, and maintaining aquatic and riparian ecosystems and their biodiversity (Vörösmarty et al., 2010; Liu et al., 2021; Huggins et al., 2022; Oliveira et al., 2025). The major changes in forest cover proposed within the global framework of FLR (Stanturf and Mansourian, 2020) will have important consequences for the integrity of the flow of water through landscapes, affecting water availability at multiple scales (Jones et al., 2022; Hoek van Dijke et al., 2022; Tuinenburg et al., 2022). We call attention, however, to the discrepancy between the repeatedly expressed need for quantitative and objective information on the hydrological impacts of reforestation and FLR (Filoso et al., 2017; Jones et al., 2022; Marshall et al., 2022; Dib et al., 2023; van Meerveld and Seibert, 2024), and the limited body of pertinent findings as summarized to date for the tropics (Bruijnzeel, 2004; Scott et al., 2005; Bonnesoeur et al., 2019; cf. global reviews by Jarvis et al., 2013; Hua et al., 2022; Lalonde et al., 2024).

In this paper, we address a long-standing discrepancy in forest hydrology: while concerns persist that forestation reduces water yield, emerging evidence suggests that FLR can, under certain conditions, enhance water availability—particularly during dry seasons. We explore the potential for FLR to restore hydrological functioning of degraded tropical landscapes, drawing on both recent advances and foundational studies, while recognizing that most areas targeted for restoration have experienced some degree of soil degradation affecting hillslope and catchment hydrological response (Bruijnzeel, 1989; Sandström, 1998; Recha et al., 2012). Although our focus is on the tropics, insights from other regions are included. We consider "baseflow" in the broad sense of the sustained flow of water in streams and rivers between rainfall events (Price, 2011). However, given the methodological variability in baseflow estimation and the fact that baseflow encompasses a range of delayed flow pathways that tend to vary seasonally and spatially (Nathan and McMahon, 1990; Price, 2011), we use dry-season streamflow ("true" baseflow; Brutsaert, 2005) as a practical proxy instead, also because it is the most observable and relevant indicator of hydrological recovery under FLR. "Hydrological functioning" refers to streamflow response to precipitation, including the timing and magnitude of seasonal baseflows as well as the flow peaks associated with distinct precipitation events (Ward, 1984; Table S1 for a definition of terms and further discussion of their usage).

## 2. Forests and streamflow: a seemingly settled debate

One long-established view of the hydrological role of forests is that their complex of trees, understorey vegetation, surface litter, organic matter, roots, and soil acts as a "sponge" absorbing rainfall during wet periods and gradually releasing the stored water during subsequent dry periods (Eckholm, 1975; Myers, 1983; Sandström, 1998). Following forest removal, this "sponge effect" tends to diminish or may be lost altogether, causing springs and streams in seasonally dry climates to carry less water or even desiccate during extended dry periods; meanwhile, flooding is typically exacerbated during periods with high and intensive rainfall producing strong increases in surface runoff (Van Dijk and Vogelzang, 1948; Pereira, 1959; Daniel and Kulasingam, 1974; Bartarya, 1989; de la Paix et al., 2013).

The "forest as sponge" metaphor and related thinking came under serious scrutiny after Bosch and Hewlett (1982) summarized the changes in annual streamflow totals ("water yield") associated with vegetation change (deforestation or afforestation) for 94 so-called "paired-catchment" studies conducted mostly under temperate climate conditions. Although the variation in results was deemed "extreme", Bosch and Hewlett (1982) concluded that "no experiments in deliberately reducing vegetation cover caused reductions in water yield, nor have any deliberate increases in cover caused increases in yield". In a companion review of the flood-mitigating capacity of forests, Hewlett (1982) concluded that an undisturbed forest cover generally moderated peak discharges and stormflow as the "sponge" metaphor implied, although the effect decreased as the size of the rainfall event and catchment wetness level increased. Further, the influence of the forest on the magnitude of the largest events ("floods") was considered marginal (Hewlett, 1982). In short, the "forest sponge" was seen to have limitations, breaking down for extreme rainfall events and very wet soils.

Hamilton and King (1983), working contemporaneously, were among the first to realize the implications of these findings for the tropics—the hydrological impacts of forest conversion in particular. Seeing that forest removal leads to increased streamflow, they surmised that trees might be more appropriately labelled "pumps", raising water from the soil profile (and/or groundwater in specific settings) and returning it to the atmosphere. Further, they concluded from the fact that the flows associated with the largest storm events were not affected much by the presence or absence of forest cover: "Major floods occur due to too much precipitation falling in too short a time or over too long a time, beyond the capacity of the soil mantle to store it, or the stream channel to handle it". In provocative articles targeting the "four M's of myth, misunderstanding, misinformation, and misinterpretation" regarding the hydrological role of tropical forests—including the "myth of the forest sponge"—Hamilton (1985, 1987) called for "greater accuracy and realism". In his footsteps, many subsequent "hydrological myth-busters" highlighted the high water consumption of trees and forests and their inability to prevent extreme flooding (e.g., Forsyth, 1996; Calder, 2005; FAO–CIFOR, 2005; Kaimowitz, 2005). The value of a "good" forest cover for maintaining other aspects of hydrological functioning, such as sustaining dry-season flows and high-quality water, was downplayed or largely neglected in these publications.

Both Hamilton and King (1983) and Hewlett (1982) did recognize that the impact of "deforestation" on the hydrological functioning of a catchment could be significantly altered if widespread soil degradation were associated with forest clearance, either during (e.g., through the use of heavy machinery) or after (e.g., conversion to poorly managed agricultural cropping or grazing). However, these pioneers were not aware of any experimental evidence supporting such intuitions. In the words of Hamilton (1987): "Suggestions implying tropical reforestation or afforestation of non-forested lands, including extensive grasslands, will cause higher groundwater well levels, renewed spring flows, and increased low flows in streams are not supported by evidence from temperate zone research that indicates the reverse". Likewise, although Hamilton and King (1983) and later Bruijnzeel (1986) acknowledged various anecdotal reports of renewed springs and more reliable streamflow following tropical forestation, sound scientific data from the region were lacking at the time (Bruijnzeel, 1989; Hamilton and Pearce, 1987).

The initial conclusions of Bosch and Hewlett (1982) have been broadly echoed by successive reviews of the gradually expanding global literature on land-cover change effects on annual water yield (or groundwater levels; Smerdon et al., 2009), with the strongest relative changes in annual yields following the gain or loss of forest cover noted for sub-humid rainfall conditions (Sahin and Hall, 1996; Brown et al., 2005; Jackson et al., 2005; Zhou et al., 2015; M. Zhang et al., 2017; Bentley and Coomes, 2020; Yu et al., 2022; Hou et al., 2023). Similar conclusions about changes in annual water yield (i.e., lower yields under forest) were reached for the much smaller humid tropical dataset (Bruijnzeel, 1990; Malmer, 1992; Waterloo et al., 1999; Scott et al., 2005; Filoso et al., 2017; Bonnesoeur et al., 2019).

In view of the seemingly overwhelming evidence that “more forest implies less total streamflow (lower yield)”, studies demonstrating improved baseflows and dry-season water availability achieved through forestation in the tropics are generally considered exceptional (Bruijnzeel, 2004; Calder, 2005; Scott et al., 2005; Bonnesoeur et al., 2019; van Meerveld and Seibert, 2024). However, based on new work, we contend that the debate regarding tropical forest cover and “baseflows” is still alive. The dominating view that forestation can only reduce streamflow and water availability, diminishes opportunities to restore the hydrological functioning of degraded landscapes, and supply environmental and societal needs. Rather, our interpretation of a growing body of evidence indicates that the opportunities for FLR to improve dry-season flows and water availability can indeed be substantial and important in specific geographical settings.

## 3. Critical considerations regarding FLR and streamflow

Before detailing our perspective, we note five aspects that are important for judging the hydrological effectiveness of FLR, but are often neglected or inadequately emphasized in reviews and compilations.

1) *Catchment water yield, or total annual streamflow, differs from the fraction of streamflow that is useful to ecosystems or humans.* With few exceptions (Brown et al., 2005; Farley et al., 2005; Li et al., 2018; Crampe et al., 2021), the common focus on total streamflow (Jackson et al., 2005; Zhou et al., 2015; M. Zhang et al., 2017; Bentley and Coomes, 2020; Hou et al., 2023) neglects the importance of sustaining stable “baseflows” between rainstorms or across seasons. In seasonally dry areas, reliable dry-season flows are critical for supporting aquatic and riparian ecological systems and a host of human water uses (Sandström, 1998; Connolly and Pearson, 2005; Huggins et al., 2022; Oliveira et al., 2025). The other streamflow component, “stormflow”, is typically less useful for humans as it is often laden with sediment, and can be destructive because of flooding and siltation of reservoirs, irrigation channels, and river beds (Bathurst et al., 2011; Yin et al., 2019; Carriere et al., 2020). Although minor floods can have positive local ecological effects (e.g., increased primary productivity; Talbot et al., 2018), most floods are detrimental to impacted vegetation and their ecosystem services (LeRoy Poff, 2002; Talbot et al., 2018; Chen et al., 2023).
2) *Water use by vegetation is only one element influencing streamflow changes—and it is not always the most important.* Depending on catchment morphology (e.g., steep slopes with narrow valley bottoms versus gentle topography with wide valley bottoms), the presence of free-draining or poorly drained soils, depth of soil above the bedrock, permeability of the regolith, and surface conditions (e.g., related to level of soil degradation), the stormflow component of total streamflow may be large or small (Fritsch, 1993; Bonell, 2005; Zhang et al., 2018a; X.P. Zhang et al., 2022; Zwartendijk et al., 2023). Where large changes in stormflow following forest removal or addition occur (Dils, 1953; Mathys et al., 1996; Krishnaswamy et al., 2012; Qazi et al., 2017), conclusions that changes in vegetation water use alone produce the observed changes in total water yield are bound to be erroneous. For instance, large-scale soil conservation works (terracing and check dams) and vegetation restoration on the Chinese Loess Plateau have produced large reductions in total water and sediment yields (X.P. Zhang et al., 2022; Fang et al., 2023). However, the vast majority of these decreases reflect reductions in stormflow owing to increased infiltration and storage in the soil profile; conversely, dry-season (winter) baseflows for several larger catchments in the area gradually increased with time, stabilizing once the vegetation cover reached 60%–70% and had matured sufficiently (X.P. Zhang et al., 2022).
3) *The influence of land degradation and the subsequent recovery of soil hydrological processes should be considered explicitly when assessing streamflow changes related to FLR.* Global inventories of land degradation and soil erosion indicate extensive areas where hydrological functioning is likely to be affected adversely (Bai et al., 2008; Gibbs and Salmon, 2015; Wuepper et al., 2020)—with increased surface runoff during storms and possibly reduced groundwater recharge as a result of diminished soil infiltration capacity (Fig. 1b). However, none of the paired catchment studies referenced in the various global literature reviews cited above (Bosch and Hewlett, 1982; Jackson et al., 2005; M. Zhang et al., 2017; Hou et al., 2023) considered (widespread) degraded soil conditions explicitly. The same applies to several more regionally oriented reviews of (temperate-zone) forest management impacts on stream baseflows and stormflows (Jones and Post, 2004; Coble et al., 2020; Crampe et al., 2021). Such summaries are, therefore, not fully representative of typical FLR situations where the effects of soil degradation on the partitioning of rainfall into surface runoff, infiltration, and soil water storage cannot be ignored (Bruijnzeel, 2004; Scott et al., 2005; Bonnesoeur et al., 2019; Pandit et al., 2024).
4) *A distinction is needed between varying hydro-climatic regimes when assessing the hydrological potential of FLR.* Relative changes in water yield are more pronounced under drier (and sunnier) conditions (Zhou et al., 2015; M. Zhang et al., 2017). This response is reflected in the global finding of Hou et al. (2023), who reported a much larger relative change in annual water yield per unit forest cover change for 59 catchments undergoing forestation than for 197 catchments experiencing forest loss. Closer scrutiny of the data revealed that the difference between the two groups of catchments primarily reflected drier overall climatic conditions in the catchments receiving forestation. This finding has two important implications: (a) the global dataset on the impact of forestation on water yield is biased; and (b) reductions in water yield after forestation under more humid conditions (e.g., in the equatorial tropics) may be smaller than suggested by the average values presented by global reviews.
5) *Hydrological recovery related to vegetation gains following losses is often non-linear and hysteretic.* Soil hydrological properties can deteriorate rapidly with forest loss but take much longer to recover: soil degradation can be instantaneous when heavy machinery is used during forest clearing (Malmer and Grip, 1990; Schack-Kirchner et al., 2007; Suryatmojo, 2014), or take several years in the case of grazing or cropping (Dils, 1953; Zimmermann et al., 2006). Conversely, soil hydrological recovery may take up to several decades, depending on the initial level of degradation, plant moisture supply, etc. (Leite et al., 2018; Lozano-Baez et al., 2019a; Zhang et al., 2019; Qiu et al., 2022; Zwartendijk et al., 2024). Certain extreme forms of land degradation (e.g., extensive gullying) can have a lasting effect, even long after forest restoration (Chen et al., 2020). The same applies to the introduction of roads (e.g., in the context of timber harvesting), the effect of which on hillslope hydrological connectivity and therefore on the magnitude of stormflows will be more or less permanent (Crampe et al., 2021). In the context of restoration, however, impact assessments should also recognize the existence of several potential positive “feedbacks” on hydrological processes at various scales over time, including “trade-offs” between changes in vegetation water use and infiltration after foresting degraded land (Scott et al., 2005; Krishnaswamy et al., 2013). Also, the ability of vegetation to capture “occult” precipitation (water from passing fog and low cloud) in specific coastal and montane settings should eventually recover as the vegetation matures (Aboal et al., 2000; Juvik et al., 2011; Teixeira et al., 2021). Finally, the potential for moisture recycling, transport, and convergence at various scales increases,

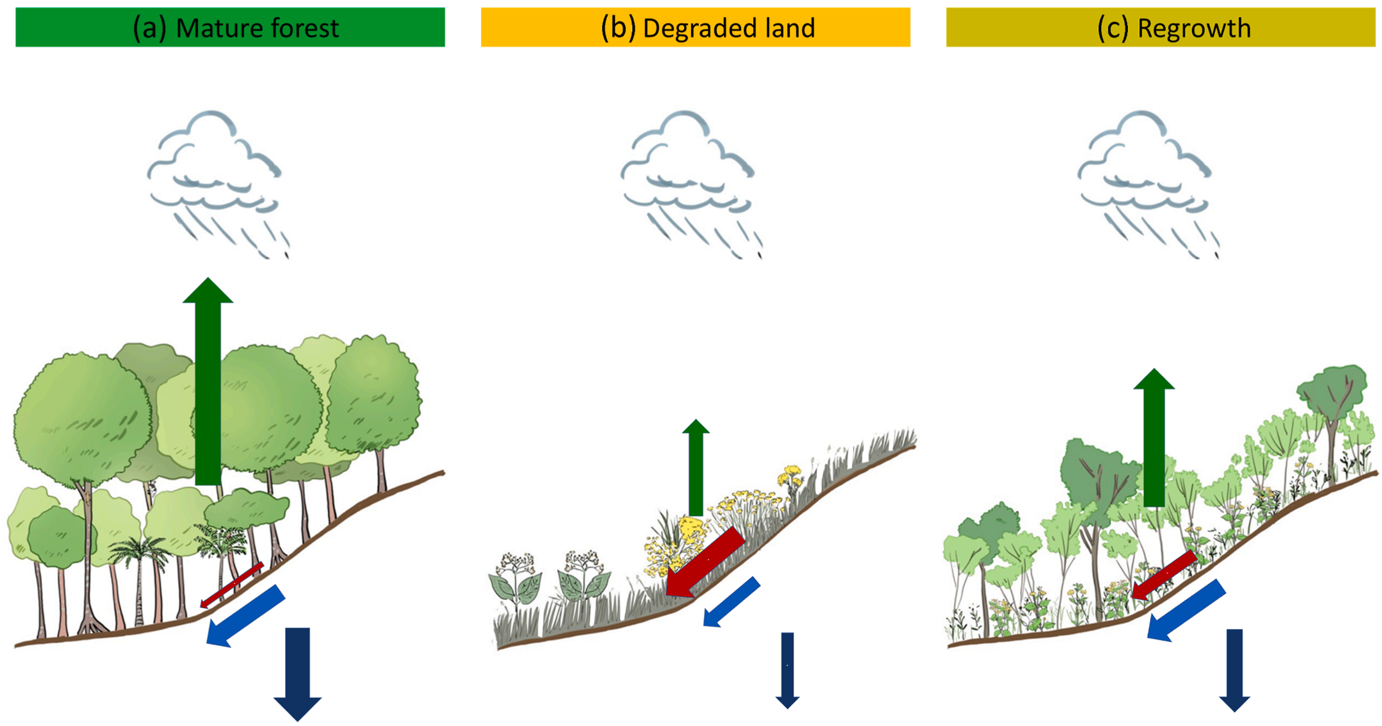


**Fig. 1.** Conceptual representation of the partitioning of precipitation into evapotranspiration, (near-)surface runoff, and groundwater recharge under (**a**) mature forest, (**b**) degraded land, and (**c**) natural regrowth to illustrate the "infiltration–evapotranspiration trade-off" mechanism governing changes in groundwater recharge following land-cover change (adapted from van Meerveld et al., 2021). Green arrows: evapotranspiration; red-brown arrows: surface runoff; lighter blue arrows: rapid subsurface stormflow; dark blue arrows: groundwater recharge. Arrow sizes indicate the relative magnitude of the respective fluxes for the three land covers.

thereby affecting patterns of precipitation (Hoek van Dijke et al., 2022; Makarieva et al., 2023; Theeuwen et al., 2023).

## 4. "Pumps" and "sponges": a paradigm of hydrological "trade-offs"

To frame our message, we return to the analogy that well-developed forest ecosystems function both as "pumps" and "sponges". This conceptualization implies that vegetation has a drying effect as it intercepts rainfall and takes up water from the soil, releasing moisture back to the atmosphere via direct evaporation from the wetted canopy (Calder, 1990; Holwerda et al., 2012), the forest floor (beneath more open canopies; del Campo et al., 2022) and through transpiration (i.e., the "pump" side of things; Sinha, 2004). Meanwhile, the "sponge" effect pertains to the underlying soil absorbing, retaining, and moderating the passage of water through the catchment. If rainfall exceeds the soil's infiltration capacity and ponding occurs, surface runoff is generated that contributes to stormflow. Infiltrating water can be stored in the soil or transferred to the groundwater reservoir or move towards the stream system through subsurface flow pathways at variable rates and depths (faster and shallower during rain). "Baseflows", which are derived from deeper subsurface flows and discharging groundwater, represent the

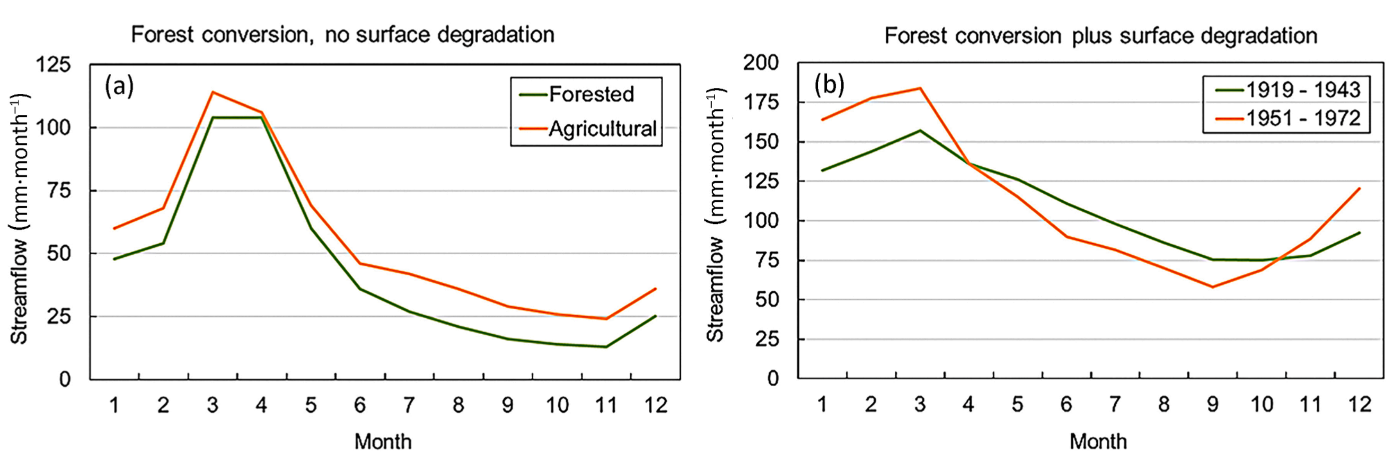


**Fig. 2.** Contrasting changes in monthly streamflow distribution following forest conversion under seasonal tropical conditions: (**a**) forest replaced by subsistence agriculture without soil degradation (Tanzania), (**b**) partial forest replacement by rainfed cropping and settlements with major changes in surface runoff generation (East Java, Indonesia). Streamflow from the agricultural catchment in (a) always exceeded that from the forested catchment, despite receiving ~265 mm less rainfall on average. Seasonal precipitation patterns for the two periods depicted in (b) were nearly identical. Based on original data in Edwards (1979) and RIN (1985), respectively.

equilibrium between water losses through evapotranspiration (ET), gains through infiltration, and the balance with storage (Fig. 1a and Table S1; Peña-Arancibia et al., 2019).

As long as the soil's infiltration capacity is more or less maintained, replacing forest by shorter-statured and more shallow-rooted vegetation types tends to increase "baseflows" year-round (more so during the dry season; Fig. 2a) and at a level largely proportional to the fractional changes in vegetation cover and water use (Bosch and Hewlett, 1982; Bruijnzeel, 2004). This response is observed because the associated changes in stormflow are generally small under non-degraded surface conditions (Hewlett and Bosch, 1984; Bruijnzeel, 1989; Farley et al., 2005). Also, peak discharges for forested and non-forested lands tend to converge as amounts of rainfall and soil wetness increase (Hewlett, 1982; Bathurst et al., 2011, 2020; Bonnesoeur et al., 2019). However, where major surface degradation has occurred, stormflows and peak stream discharges typically increase substantially (Dils, 1953; Mathys et al., 1996; Krishnaswamy et al., 2012; Tarigan, 2016; Zhang et al., 2018a). Moreover, these increases can be so large that the normally converging relations linking stormflow and precipitation amounts for forested and non-forested conditions diverge (Van der Weert, 1994; Mathys et al., 1996; Bruijnzeel, 2004; Scott et al., 2005). The extra surface runoff caused by the reduced infiltration no longer replenishes soil moisture reserves or the groundwater that sustains baseflow, and is thus effectively lost to the catchment ecosystem. In cases of greatly reduced soil infiltrability (e.g., due to heavy crusting or compaction), losses via surface runoff can lead to marked reductions in groundwater recharge (Fig. 1b) and dry-season flows (Fig. 2b), compared with the prior forested situation, despite the lower water use of the post-forest vegetation. In other words, the retentive "sponge" effect is diminished (Bruijnzeel, 1989; Sandström, 1998; Krishnaswamy et al., 2013; Qazi et al., 2017).

Moving up in scale, a recent spatially distributed hydrological sensitivity study by Peña-Arancibia et al. (2019) assessed the "trade-off" between annual and seasonal changes in the "pump" (ET) and "sponge" (infiltration) functions, respectively, after forest conversion to pasture across the tropics, both with and without imposed soil degradation (i.e., infiltration reduced by 50%). Overall, simply replacing forests by non-degraded pasture increased annual water yield by 18% because of the lower water use of the grass (reduced pumping and rainfall interception leading to higher flows throughout the year; cf. Fig. 2a). The greatest relative increases were found in water-limited regions while smaller changes were derived for the rainy humid equatorial regions where greater cloudiness limits ET. Overall annual water yields were indicated to increase by a further 8%–26% when forest conversion was followed by soil degradation, reflecting the associated increases in surface runoff and stormflows due to reduced infiltration. However, for nearly one-fifth of all grid cells (19%), a reduction in stream baseflow was inferred during one or more of the driest months in the soil degradation scenario, despite the lower water use of the pasture. In short, the change in infiltration had a greater effect on dry-season baseflows than did the change in ET (i.e., the "trade-off" between the two was negative; cf. Fig. 1b and 2b; Peña-Arancibia et al., 2019). It is worth noting that many of the areas for which decreases in dry-season flows were predicted coincide with areas targeted by the Atlas of Forest and Landscape Restoration Opportunities (Minnemeyer et al., 2014). Note also that any potentially negative changes in precipitation following widespread forest conversion (Smith et al., 2023) were not included in the modelling. At these larger scales, the indirect climatic effect (i.e., reduced rainfall) may offset the direct effect (i.e., increased runoff due to reduced ET) of deforestation (Ma et al., 2024).

As discussed more fully in Section 5.3, major improvements in soil infiltration capacity following the re-establishment of good vegetation cover on degraded soils have been observed over the course of time (Colloff et al., 2010; Lozano-Baez et al., 2019a; Zhang et al., 2019; Zwartendijk et al., 2024). Vegetation maturation is associated with the redevelopment of biologically mediated soil macropores, root channels, and animal burrows—some of which are typically lost during repeated slash-and-burn cycles, annual cropping, and grazing (Shougrakpam et al., 2010; Zwartendijk et al., 2017, 2024; Nespoulous et al., 2019; Qiu et al., 2023). Connected networks of macropores act as "preferential pathways" guiding infiltrating rainwater (typically within a few days and often less) through the root zone once a critical soil moisture storage threshold value is exceeded (Nimmo, 2016; Cheng et al., 2020; Illien et al., 2021), usually during times of ample rainfall (Jasechko and Taylor, 2015; Cheng et al., 2018; Jiang et al., 2020). This mechanism promotes the deep subsurface flows and groundwater recharge—especially during the latter half of the rainy season—that contribute to (dry-season) baseflows (Sandström, 1998; Owor et al., 2009; Cheng et al., 2020).

Increased macropore flow can also explain why stormflow responses of reforested headwater catchments can be reduced even under extreme rainfalls compared with nearby areas with greater surface degradation, as shown in various locations such as Panamá, the Philippines, Mediterranean SE France, and temperate South Korea (Mathys et al., 1996; Kim and Jeong, 2006; Ogden et al., 2013; Zhang et al., 2018b). Indeed, the largest decreases in storm runoff—and therefore the largest gains in infiltrated rainfall—have been observed after foresting heavily degraded areas subject to intense rainfall (Mathys et al., 1996; Chandler and Walter, 1998; Kim and Jeong, 2006; Zhang et al., 2018b). In such cases, increased macropore flow combines with other processes resulting from forestation, notably increased water storage potential in the soil due to pre-storm water use by the vegetation (Scott et al., 2005; Ogden et al., 2013; Zhang et al., 2018b).

Given such trade-offs, it is premature to conclude that the "localized" flood-moderating effect of reforesting degraded headwaters is limited to small or intermediate rainfall events only, as suggested by Marshall et al. (2022) and others investigating catchments with limited soil degradation (Bathurst et al., 2011, 2020; Jones et al., 2022). However, it would be equally premature to extend such findings uncritically to much larger scales (such as large river basins) and expect upland forest restoration to eliminate all downstream flooding (Eckholm, 1975; Zhao et al., 2022). Large-scale flooding typically results from extensive and persistent rainfall fields of long duration and/or high intensity, often occurring when soils have been wetted up by previous rains (Hewlett, 1982; Hofer and Messerli, 2006; Bathurst et al., 2011; Wu et al., 2024). Flood risk further relates to the interplay of a host of additional factors, including degree of urbanization and hard surfaces, floodplain occupancy, wetland conversion, presence of storage reservoirs, and other infrastructural works, such as dikes and embankments, etc. (Mei et al., 2015; Adnan and Atkinson, 2018; Merten et al., 2020; Xue et al., 2022; Zhao et al., 2022).

## 5. Improved dry-season flows: a key measure for assessing the hydrological success of FLR

As stated earlier, annual streamflow totals associated with forested catchments are usually lower than those for non-forested catchments due to the generally higher water use of trees (Zhang et al., 2001; Farley et al., 2005; M. Zhang et al., 2017) and to the propensity of the whole "forest complex" to limit the generation of stormflows (Brown et al., 2005; Jones et al., 2022). However, the influence of restoring tree cover on dry-season baseflow generation remains enigmatic (Bruijnzeel, 1989; Scott et al., 2005; Yamamoto et al., 2020; Jones et al., 2022; van Meerveld and Seibert, 2024). Because dry-season flows have a particular practical importance, we argue that their recovery is a more suitable indicator of FLR success in hydrological terms (Bruijnzeel, 2004; X.P. Zhang et al., 2022), along with reductions in soil loss and catchment sediment yields (Wiersum, 1984; Ziegler and Giambelluca, 1998; Sidle et al., 2006; Vanacker et al., 2007; Fang et al., 2023). In the following subsections, we further explore the potential benefits of FLR with respect to dry-season flows.

### *5.1. Conditions where FLR decreases or increases dry-season flows*

Depending on the relative changes in both vegetation water use and infiltration related to forest restoration, baseflows will be reduced where losses through increased ET exceed the gains related to improved infiltration (Farley et al., 2005; Ribolzi et al., 2018). The net effect may also be near-neutral (Tennessee Valley Authority, 1961; Ghimire et al., 2014a), or positive where infiltration gains override the extra evaporation losses (Scott et al., 2005; Krishnaswamy et al., 2013, Fig. 1c). For large areas, the conditions and locations where FLR may be most effective hydrologically speaking are best shown through modelling studies. For example, Peña-Arancibia et al. (2019) inferred the largest negative changes in dry-season flows following deforestation and soil degradation for areas having strong seasonality in precipitation, a high rainfall surplus over ET during wet months, and deep soils. Therefore, upon restoration of the soil infiltration capacity after forestation, the largest absolute gains in groundwater recharge and dry-season flows may be expected in highly degraded areas under seasonally variable rainfall, and with sufficiently deep soils that are capable of storing the extra infiltrating water (Bruijnzeel, 1989; Scott et al., 2005; Chandler, 2006). Thus, using the same sensitivity modelling approach as Peña-Arancibia et al. (2019), but in an inverse manner (i.e., examining the impacts of forest addition instead of removal), degraded areas in mainland and maritime SE Asia (including Papua New Guinea), SE China, NE India, parts of West Africa, SE Brazil, and Central America were indicated to have the greatest likelihood of increased dry-season flows following forestation (Fig. 3). Out of a total of 3554 grid cells with a climatic potential for sustaining forest vegetation (1° resolution, ~100 km), a positive influence on groundwater recharge and baseflows (i.e., increased dry-season flows compared to the current situation) was predicted for 340 cells (~10%). These "bright spots" were again those with a high seasonal rainfall surplus, sufficiently deep soils, and significant initial surface degradation. A near-neutral change in "baseflows" (<2 mm decline per year) was indicated for an additional 292 grid cells (8%). Again, potential changes in precipitation after forestation (Hoek van Dijke et al., 2022) were not considered in the modelling (Section S1).

At the other end of the spectrum, the largest decreases in baseflows have been reported after planting fast-growing exotic tree species (often pines or eucalypts) in areas where average rainfall is insufficient to support (evergreen) forest naturally, and grassland or shrubland is the assumed natural baseline vegetation (Farley et al., 2005; M. Zhang et al., 2017). Similar decreases have also been observed after the introduction of exotic evergreen trees to fire-maintained grasslands in the seasonal tropics (Bailly et al., 1974; Waterloo et al., 1999). However, there are indications of gradually diminishing tree water use—and therefore partial streamflow recovery—as these planted forests mature, particularly over longer periods than the typical rotation duration for industrial

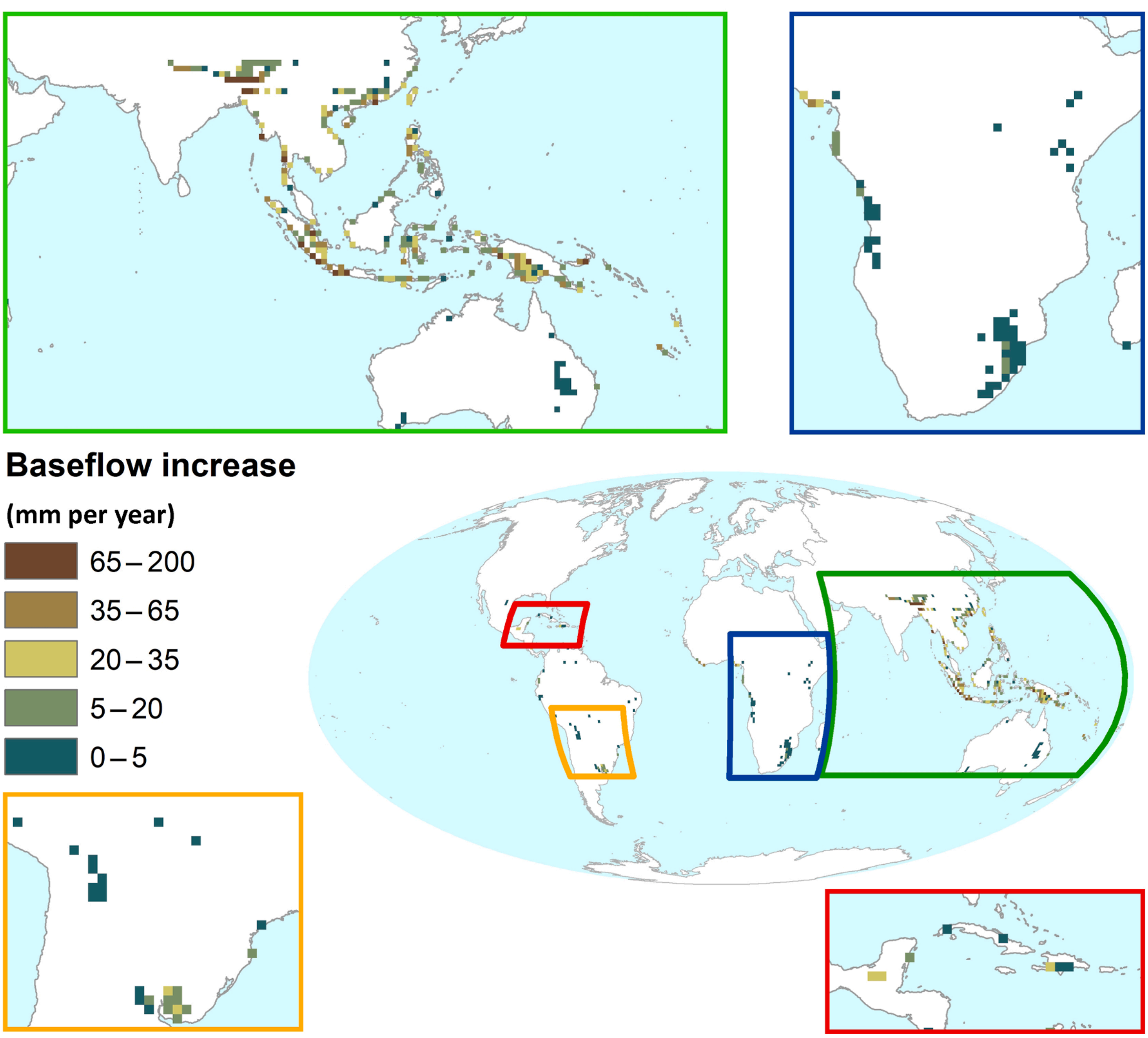


**Fig. 3.** Modelled increases in annual "baseflow" (mm·year$^{-1}$) after reforesting all degraded land that is climatically capable of carrying forest across the tropics. See Section S1 for background on the model and materials.

timber plantations in the tropics or subtropics (Scott and Prinsloo, 2008; Waterloo et al., 1999). Therefore, the longer-term effects of planted forests need not be as harmful to catchment water yield as shorter-term studies may appear to suggest (Hou et al., 2023). Recovery times for regaining hydrological functioning and low flows following forestation via natural regrowth and tree planting are discussed further in Section 5.3 below.

Similarly, rates of soil water uptake under given climatic conditions can vary more than two-fold between tropical tree species (Dierick and Hölscher, 2009; Kunert et al., 2010; Wang et al., 2020). This reflects different physiological and phenological responses to seasonal changes in climate and soil water availability that manifest themselves in the degree of stomatal control of evaporative losses exerted by different species (Roman et al., 2015; Konings and Gentine, 2017). Such differences may relate to a species' region of origin (e.g., humid vs. arid climates) or position within the ecosystem (e.g., tall and deep-rooted emergent vs. more shallow-rooted sub-canopy trees; Penha et al., 2024). In addition, stand-scale rainfall interception losses vary as a function of plant functional type (e.g., broad-leaved vs. coniferous, or evergreen vs. deciduous species; Magliano et al., 2019; Q. Zhang et al., 2022), stand density, and age (Bruijnzeel, 1988; Scott et al., 2005; Y. Zhang et al., 2023). Such differences provide opportunities for optimizing vegetation water use through management. We return to this important aspect in Section 5.4 below.

### 5.2. Real-world examples of increased dry-season flows following forest restoration in degraded areas

Over the last two decades, an increasing number of studies have reported increases in dry-season flows following forestation of degraded land in the tropics and elsewhere across a range of catchment scales (from a few hectares up to 7,325 $km^2$). This list includes studies in the lowlands of the Philippines (Chandler, 2006; Zhang et al., 2018b), monsoonal SW India (Krishnaswamy et al., 2013), and South China (Hou et al., 2018), as well as the wet mountains of Costa Rica, Atlantic Brazil, and Puerto Rico (Krishnaswamy et al., 2018; Teixeira et al., 2021; Hall et al., 2022). Although none of these studies were part of a rigorous paired catchment experiment and some sites experienced a gradual increase in precipitation (by up to 18 mm·year$^{-1}$), they all had the large seasonal rainfall surplus and sufficiently deep soils that were identified by Peña-Arancibia et al. (2019) and Bruijnzeel (2019) as being conducive to increased dry-season flows after forest restoration. In all cases, infiltration improved. For the wet montane cases, increased capture of cloud water (fog) as the trees gained height and exposure to the prevailing winds likely contributed to improved dry-season flow as well (Ingwersen, 1985; Aboal et al., 2000; Rigg et al., 2002; Juvik et al., 2011).

Similar increases in dry-season flow have also been observed under non-tropical conditions (but sufficiently deep soils), such as in temperate South Korea, where ephemeral streams gradually became perennial again several decades after forestation of severely degraded (war-damaged) lands (Kim and Jeong, 2006). Other examples of improved hydrological flow regimes under drier conditions include Texas (shrub encroachment after heavy grazing ceased; Wilcox and Huang, 2010), montane Ethiopia (after implementing soil conservation measures coupled with a judicious use of native trees; Meaza et al., 2022), and the world's largest afforestation experiment: the semi-arid Loess Plateau in China (Z.L. Gao et al., 2015; X.P. Zhang et al., 2022). For the latter, the increased winter baseflows (during the cold dry season) observed for larger catchments some 20 years after the widespread introduction of soil conservation measures and vegetation on sloping land are enigmatic, however. Despite the near-complete elimination of the formerly rampant lateral surface runoff from agricultural fields, and the extra infiltration of rainfall that this elimination has afforded (Jin et al., 2020; Liu et al., 2020), the establishment of shrubs and trees under the prevailing low summer rainfall has generally produced an extensive dry layer within the soil that has been shown to hamper deep percolation and groundwater recharge (Gates et al., 2011; Deng et al., 2016; Jia et al., 2019). The application of stable isotope techniques suggests recharge in the area takes place through macropores during periods of high rainfall, whereas contributions by (very slow) matrix flow are negligible because of the extreme thickness of the loess deposits (Li et al., 2017). Although such macropores are found mostly in the upper soil layers beneath shrubs and trees (Ren et al., 2016; Qiu et al., 2023), most—if not all—of the infiltrated water is consumed again by the introduced vegetation (H. Zhang et al., 2017; Wei et al., 2019; G. Chen et al., 2023). As a result, the re-vegetated upper slopes and plateau areas have become effectively disconnected from the valley bottoms and the streams. It is likely, therefore, that groundwater recharge in the area occurs primarily on the new agricultural fields that have been created on eroded sediments trapped behind valley-bottom check dams, and on adjacent foot-slopes where the groundwater table is closest to the surface (Luo et al., 2020, 2023; Fang et al., 2023). Further work is necessary to separate the effects of forestation, soil conservation works, climatic variability, regional hydrogeological connectivity, and other anthropogenic factors (storage reservoirs, urbanization, groundwater extraction, etc.) on dry-season (winter) flow trends for the larger catchment areas that are typically studied in the area (Yan et al., 2023; Xia et al., 2025).

### 5.3. Recovery times for regaining hydrological functioning and baseflow integrity

Most studies suggest that a period of at least 10–15 years of soil infiltration capacity recovery is needed following forestation of intensively used lands (Fig. 4) before the surface runoff and stormflows generated by some of the more intensive rainstorms can be attenuated and a more normal level of hydrological functioning is achieved, including a potential recovery of dry-season baseflows (Chandler and Walter, 1998; Kim and Jeong, 2006; Colloff et al., 2010; Lozano-Baez et al., 2019a; Zhang et al., 2019). Even longer periods (20–35 years) may be required for forestation under semi-arid conditions (Zhao et al., 2014; Leite et al., 2018; Qiu et al., 2022; X.P. Zhang et al., 2022). This length of time is needed to rebuild the soil humus layer and dense understorey, which together eliminate the erosive power of raindrops reaching the forest floor, as well as promote infiltration through root network development, biologically mediated macropore formation, and slowed down surface runoff (Wiersum, 1985; Colloff et al., 2010; Perkins et al., 2014; Zhao et al., 2014; Song et al., 2020). Recovery time will also depend on initial soil and vegetation conditions at the time of forestation, the type and rate of vegetation development (Lozano-Baez et al., 2018; Mens et al., 2022; Qiu et al., 2022), and the prevailing rainfall intensities. No clear distinction between naturally regenerating or planted forests seems to exist in terms of their respective capacities to improve soil infiltrability following forestation of grazed land (Zwartendijk et al., 2024).

However, large increases in topsoil infiltration capacity associated with forestation will not have much of an effect on surface runoff and stormflow generation if rainfall intensities are mostly low to moderate, as is typically observed at higher elevations (Edwards, 1979; Gilmour et al., 1987; Zimmermann and Elsenbeer, 2009). Furthermore, not only the intensity of pre-forestation land use (notably grazing), but also post-forestation usage (e.g., harvesting of litter and fuelwood, grazing), exert a distinct influence on the rate and magnitude of soil hydraulic recovery (Ghimire et al., 2014b; Lozano-Baez et al., 2019a; Badu et al., 2022; Bargués-Tobella et al., 2024). Lastly, recovery in infiltrability can even be stalled more or less entirely in places with severe soil compaction, such as can result from mechanised timber extraction (Van der Plas and Bruijnzeel, 1993; Ziegler et al., 2006; Suryatmojo, 2014).

The net hydrological effect of improved infiltration vs. increasing water use by regrowing vegetation depends on forest structure and age. In humid tropical forests, both rainfall interception losses and soil water uptake (transpiration) increase rapidly during the first two decades of

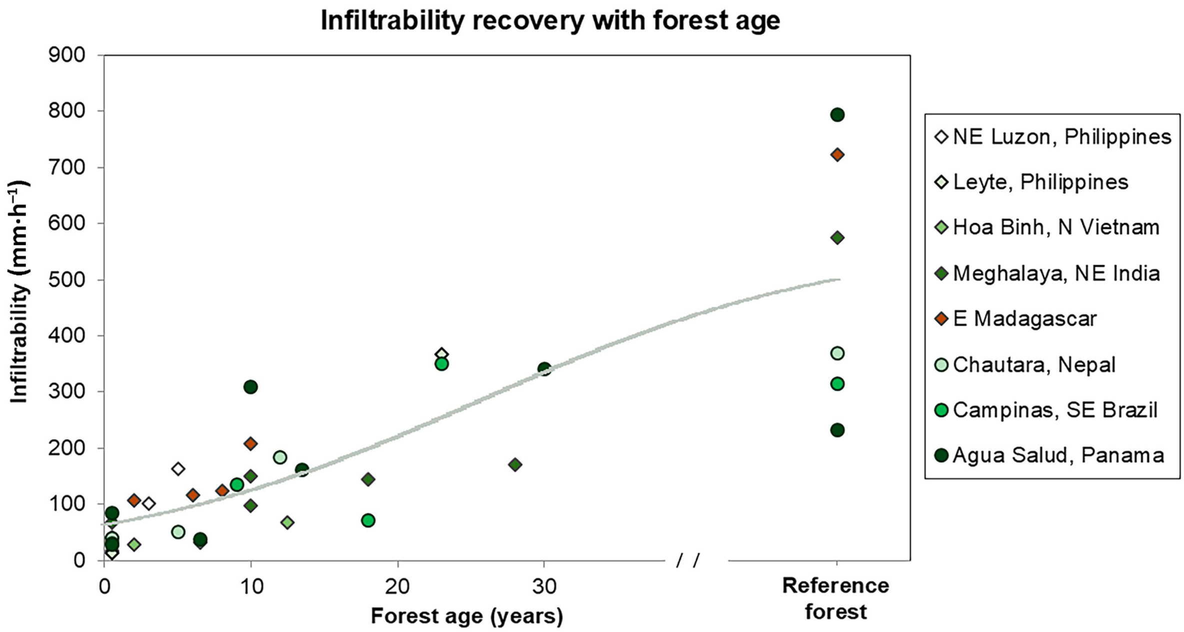


**Fig. 4.** Changes in soil infiltrability during forest maturation following abandonment of grazing or slash-and-burn cultivation at selected (sub)tropical sites. Original data: NE Luzon, Philippines (Snelder, 2001); Leite, Philippines (Zhang et al., 2019); Hoa Binh, N Vietnam (Ziegler et al., 2004); Chautara, Nepal (Gilmour et al., 1987); Meghalaya, NE India (Zwartendijk et al., 2024); E Madagascar (Zwartendijk et al., 2017); Campinas, SE Brazil (Lozano-Baez et al., 2018, 2019b; Alves Pereira et al., 2020); and Agua Salud, Panamá (Hassler et al., 2011; Litt et al., 2019; Bush et al., 2020).

regrowth, driven by rising leaf biomass. Transpiration typically stabilizes once the leaf area index reaches approximately 4–5 $m^2 \cdot m^{-2}$ (Zimmermann et al., 2013; Ghimire et al., 2022). There are some reports of reduced vegetation water use and recovering streamflow at advanced stages of natural regrowth (>35 years) under humid warm-temperate conditions (Vertessy et al., 2001; Hosoda et al., 2004), but the reverse (i.e., increased ET and reduced flows compared to old-growth forest) has been observed as well (Jackson et al., 2018; Crampe et al., 2021). Such contrasting findings reflect differences in species composition as forests mature, depending (amongst others) on the type of the preceding disturbance (e.g., logging, wildfire, and grazing; Hawthorne et al., 2013; Elliott et al., 2019; Mens et al., 2022). Comparable evidence for semi-mature regrowth from the tropics is both scant and contradictory: overall ET from vigorous ~20-year-old regrowth on former pasture land in central Amazonia was still about 20% greater than that from nearby old-growth forest (von Randow et al., 2020). The reverse (21% higher ET from old-growth forest) was observed in the case of ~20-year-old vegetation regenerating after a fire in the cloud forest zone of eastern Mexico (Muñoz-Villers et al., 2012). Likewise, similarly aged successional vegetation in upland Madagascar in an area subject to long-term slash-and-burn agriculture evaporated about 11% less annually than near-climax forest in years of comparable rainfall (Bailly et al., 1974; Ghimire et al., 2022). Under much drier conditions in montane Ethiopia, Descheemaecker et al. (2009) did not find increased deep percolation beneath 5- to 20-year-old natural regrowth following closure of steep, degraded slopes to grazing and cropping as the extra infiltrated rainfall was effectively consumed again by the aggrading vegetation (cf. the finding of Deng et al. (2016) in the equally dry Chinese Loess Plateau area). Additional measures would be needed, therefore, to boost deep soil- and groundwater reserves under such conditions (Meaza et al., 2022, Section 5.4 below).

Regarding planted forests, most studies of tree water use in tropical plantations concern relatively young trees (~3–16 years of age), hence the longer-lasting effects on water yield are still poorly known (Samraj et al., 1988; Scott et al., 2005; Kallarackal and Somen, 2008; Kunert et al., 2010; Ferraz et al., 2021). Plantation water use is often seen to decline upon maturation, with the timing of the onset varying greatly between species. For example, work has shown that for eucalypts growing under subtropical conditions, a decline in water use sets in after about five years, whereas for pines it varies between 15 and 25 years (Scott and Prinsloo, 2008; Aguilos et al., 2021; Ferraz et al., 2021). Again, where planted forests are allowed to grow over (much) longer periods than the typical industrial rotation period, streamflow appears to be tending towards pre-forestation levels (i.e., tall native grasslands). Examples include planted pines (>30 years of age) and eucalypts (>15 years) in sub-humid South Africa (Scott and Prinsloo, 2008) and various species of *Acacia* in tropical SE China (20–50 years: Ma et al., 2008; J. Gao et al., 2015); *Acacia* water use peaks around 10 years of age (Cienciala et al., 2000). However, no decline in tree water use was observed for mature *Schima* plantations in SE China between 26 and 35 years of age, possibly because transpiration had already stabilized (Li et al., 2023). Similarly, streamflow from catchments with mature stands (~40–50 years old) of white pine (*Pinus strobus*) in SE USA (Ford et al., 2011) and slash pine (*P. elliottii*) in SE Queensland (Bubb and Croton, 2002) showed no signs of recovery.

Summarizing, dry-season flow increases (if any) may take several decades or more in humid (sub)tropical areas (Krishnaswamy et al., 2012; Hou et al., 2018; Zhang et al., 2018b), and even longer in water-limited areas (Scott and Prinsloo, 2008; Descheemaecker et al., 2009; X.P. Zhang et al., 2022). There is a clear need for new dedicated work documenting the changes in water use (and the associated effects on groundwater recharge and streamflow) as different types of native and planted (sub)tropical forests and tree-species configurations mature (Kunert and Cárdenas, 2015; Amazonas et al., 2018; Bentley and Coomes, 2020; Ghimire et al., 2022; Hua et al., 2022). This type of information is essential for the development and validation of hydrological models and remote sensing products (Salazar-Martinez et al., 2022; Sun et al., 2023, 2024). It also facilitates decision-making in the context of forest restoration programmes in which potentially conflicting ecosystem services and interests between stakeholder groups need to be balanced (Wang et al., 2017; Asbjornsen et al., 2022). For instance, mixed tree plantations can promote biodiversity and tree growth (i.e., sequester more carbon), but can also use more water than do mono-specific plantations (Kunert et al., 2012; Germon et al., 2017). Likewise, as stated previously, both planted and naturally regenerating forests under wet-dry tropical conditions can use much more water than the degraded grasslands or croplands they replace (Waterloo et al., 1999; Scott et al., 2005; Ribolzi et al., 2018). Thus, where reduced dry-season flows following vegetation restoration cannot be avoided, this may be viewed as a necessary cost of achieving these other, non-hydrological benefits (notably improved livelihoods, increased biodiversity, and carbon sequestration as reviewed by Marshall et al., 2022; Edwards and Cerullo, 2024; Brancalion et al., 2025), next to affording protection against soil erosion, and improving stream water quality (Hua et al., 2022; Oliveira et al., 2025). Furthermore, the water that is released to the atmosphere through ET is not "lost" per se, as it will contribute to precipitation elsewhere (van der Ent et al., 2010; Hoek van Dijke et al., 2022; see Section 6 below).

### 5.4. *Forest management, groundwater recharge, and dry-season flows in seasonally-dry climates*

As much of the tropical world includes seasonally-dry climates (Chang and Lau, 1993; Huggins et al., 2022), it is sensible to avoid high tree water use in forestation in such water-limited areas (Deng et al., 2016; Jia et al., 2019), also in view of the seemingly more limited opportunities for boosting groundwater recharge and dry-season flows through forestation compared to wetter areas (Fig. 3).

Several precautions would help to limit excessive water use by planted vegetation: (a) choosing species judiciously (e.g., native, slow-growing, or deciduous rather than exotic, fast-growing evergreen trees; Kunert and Cárdenas, 2015; Lara et al., 2021), (b) maintaining a mosaic of vegetations of different ages and types (Ferraz et al., 2021), (c) avoiding short rotations, overly dense planting, and coppicing (Sharda et al., 1998; Kallarackal and Somen, 2008; Hakamada et al., 2020), and (d) optimizing tree cover in accordance with prevailing rainfall and soil conditions, also at the micro-scale (e.g., considering slope aspect and topographic position; Bucci et al., 2008; Wang et al., 2008; Giambelluca et al., 2009). In one example of the latter approach, maintaining an intermediate tree density with scattered tree cover in an agroforestry parkland setting in seasonally dry West Africa had notable hydrological benefits (Ilstedt et al., 2016). Here, this approach (sometimes referred to as the "optimum tree density" hypothesis; Fig. 5) resulted in improved groundwater recharge, with local increases in infiltration and reductions in surface runoff and soil evaporation beneath individual trees (Bargués-Tobella et al., 2014). The landscape-scale groundwater recharge in the optimum tree density scenario was found to be five to six times higher compared with areas without trees (Ilstedt et al., 2016). In addition, the trees supplied surrounding shallow-rooted crops with moisture through "hydraulic lift" during extended dry periods (Bayala et al., 2008). Reij and Garrity (2016) documented the rapid expansion of naturally regenerating trees on agricultural fields elsewhere in the West African Sahel, suggesting that such farmer-managed natural

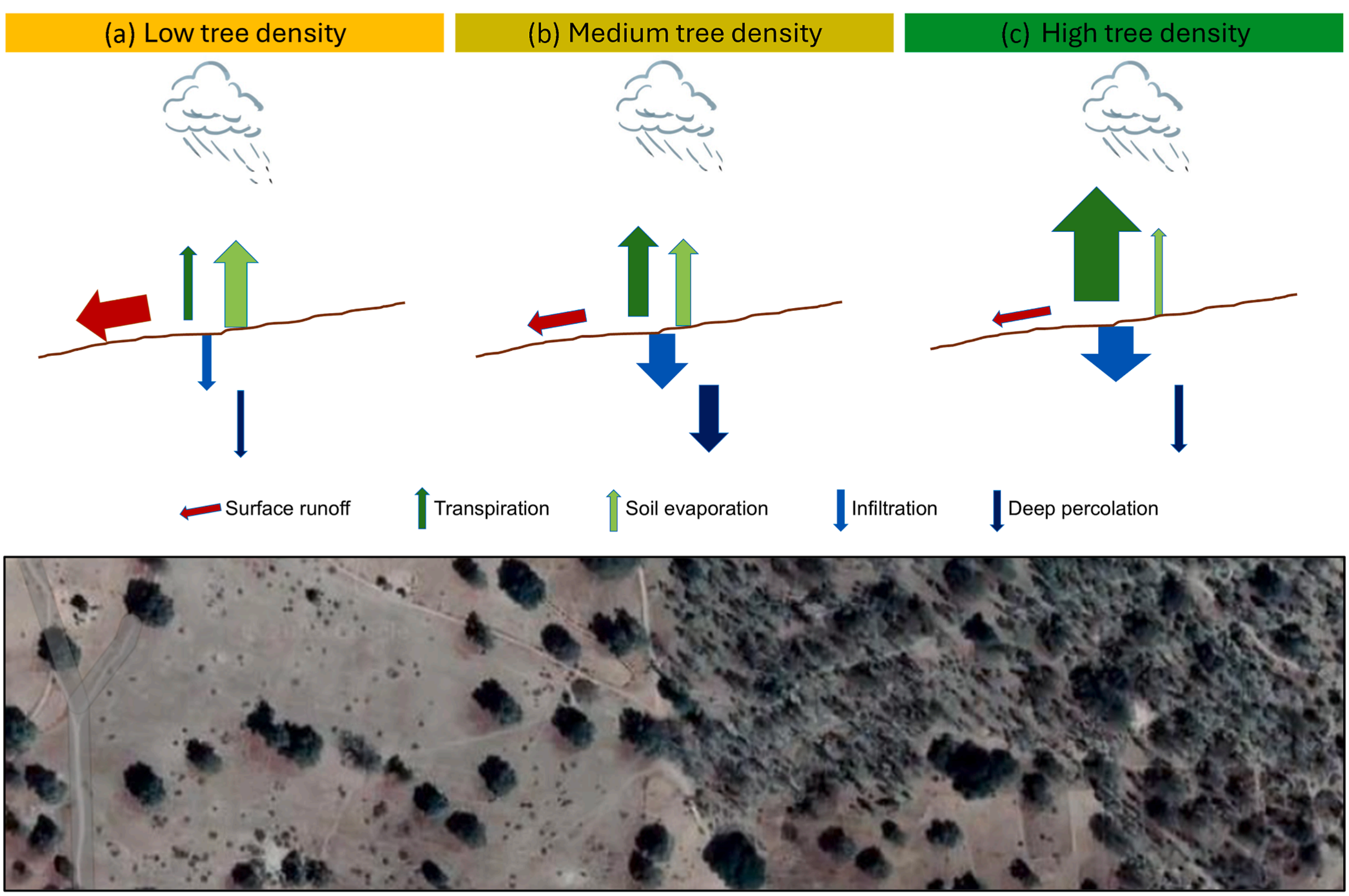


**Fig. 5.** Conceptual representation of the changes in magnitude of individual water budget components as a function of tree density under seasonally dry tropical conditions (adapted from Ilstedt et al., 2016). Optimum groundwater recharge (deep percolation) is associated with intermediate tree densities due to lower losses via surface runoff and soil evaporation compared to sparser vegetation (on the left), while transpiration and interception losses are moderate compared to those for denser woodland (on the right). Net recharge in the latter is lower despite higher near-surface infiltration. Satellite imagery: Google Maps Data, CNES/Airbus, Maxar Technologies, 2025. Retrieved on 22 May 2025.

regeneration is a low-cost approach to large-scale restoration of the region's open woodlands.

Vegetation recovery and improved surface infiltration following closure of degraded steepland to grazing and cropping under much drier conditions in northern Ethiopia did not lead to increased deep percolation because of the associated increase in vegetation water use. However, groundwater recharge was boosted substantially in places where runoff from degraded upslope areas ("run-on") also infiltrated the vegetation exclosures (Descheemaecker et al., 2009). Foot-slope springs were re-activated as well in these areas (Nyssen et al., 2009). Indeed, banded natural vegetation patterns maintained by run-on from poorly or non-vegetated parts of the landscape are a well-known phenomenon across the semi-arid tropics and act as natural water-harvesting systems that can be more productive than tree plantations in the intermediate zones (Valentin et al., 1999). Such observations underscore the limitations of purely vegetation-based approaches in arid climates (Deng et al., 2016; G. Chen et al., 2023) as well as the importance of having additional soil and water conservation measures in place to maximize rainfall infiltration and water retention (WOCAT, 2007; H. Zhang et al., 2017; Wei et al., 2019; Meaza et al., 2022).

Thinning has been used successfully to mitigate ecosystem drought stress in dry areas experiencing climate-related forest dieback and growth decline (Sohn et al., 2016; del Campo et al., 2019a). However, although opening up the canopy through (drastic) thinning allows (much) greater quantities of rainfall to reach the forest floor (del Campo et al., 2022), a lasting net positive effect on subsoil moisture levels, groundwater recharge, and low flows is usually achieved only after removing a major portion (often ~50% or more) of the trees, coupled with (regular) removal of understorey vegetation through prescribed burning (McLaughlin et al., 2013; del Campo et al., 2019b). This is because the greater availability of light, soil moisture, and nutrients after thinning stimulates the growth and water use of understorey vegetation and remaining trees such that net increases in stored soil water are typically limited and transient (Stogsdill et al., 1992; Lesch and Scott, 1997; Hawthorne et al., 2013; del Campo et al., 2019b; Momiyama et al., 2021). Further, increased insolation after thinning increases evaporation losses from the forest floor and soil surface (Wang et al., 2008; del Campo et al., 2019a). However, heavy thinning and prescribed burning also pose a danger to near-surface hydrological functioning in that removal of understorey vegetation and litter by (repeated) fire can easily cause soil deterioration and accelerated surface erosion (Wiersum, 1985; Song et al., 2020; Wang et al., 2021). Nevertheless, the finding that strip-thinning tends to produce greater increases in water yield than equivalent vegetation removals in the form of patch-cutting or uniform thinning (Hawthorne et al., 2013) suggests there are opportunities for judicious vegetation management approaches in which surface runoff produced in more degraded parts is allowed to re-infiltrate downslope, thereby boosting both vegetation development and local groundwater recharge (Wang et al., 2008; Descheemaecker et al., 2009).

Summarizing, much more work is needed to characterize and develop suitable management strategies for diverse tropical settings to restore on-site hydrological functioning with potentially positive impacts on groundwater recharge and the flow regime of affected rivers and streams. Examples include approaches that involve the use of intermediate tree densities (including tailored agroforestry systems; Kaushal et al., 2021; Jinger et al., 2023) as well as areas that have been heavily disturbed or grazed (Simonneaux et al., 2015; Lulandala et al., 2022; Bargués-Tobella et al., 2024), have acidic or salt-affected soils (Marcar et al., 1997; Jinger et al., 2023), have very low or very high rainfall (Valentin et al., 1999; WOCAT, 2007; Descheemaecker et al., 2009; Wei et al., 2019; Meaza et al., 2022), or receive their primary water input via occult precipitation (Aboal et al., 2000; Hildebrandt and Eltahir, 2006; Verbrugghe and Khan, 2024).

## 6. Increased tree cover, moisture recycling, and precipitation

A potentially important positive feedback of FLR is that higher ET associated with forest increase may contribute to increased rain at downwind locations via moisture recycling and convergence. Considerable advances have been made in the last ~15 years in understanding the global water cycle and the role of trees and forests in this regard (van der Ent et al., 2010, 2014; Wang-Erlandsson et al., 2014, 2018; Tuinenburg et al., 2020; Wunderling et al., 2022; Theeuwen et al., 2023). Large-scale studies that track air flows, atmospheric moisture, and rainfall show that air passing over large tracts of forest captures more water and produces more rain downwind than does air that passes over sparse vegetation (Spracklen et al., 2012). For example, the high year-round ET from the largely forested Congo River Basin has been inferred to provide >30% of the precipitation falling over large adjacent river basins to the drier north (e.g., Lake Chad) and south (e.g., Zambezi) (te Wierik et al., 2022). Similarly, moisture evaporated from the forests of the wet mountainous Western Ghats in SW India has been estimated to contribute 25%–40% of average monsoon rainfall in the water-deficient eastern state of Tamil Nadu, and up to 50% during dry years (Paul et al., 2018). Likewise, a pan-tropical assessment of the impacts of forest loss between 2003 and 2017 on precipitation indicated clear reductions in observed precipitation at scales >50 km, with the greatest declines found at a scale of 200 km, which was the largest scale considered (Smith et al., 2023). Studies of deforestation effects on precipitation have tended to focus on annual effects, neglecting seasonal variation (Spracklen et al., 2018). Recently, Qin et al. (2025) demonstrated contrasting precipitation responses to deforestation in Amazonia between seasons. During the wet season, deforested lands (pastures) heat up more than surrounding forest vegetation, triggering upward air flows and creating areas of lowered atmospheric pressure. Moisture is drawn into these "heat lows" from upwind areas, thereby increasing cloud cover and rainfall over the deforested patches, at the expense of precipitation in the forested upwind areas (meso-scale effect). Conversely, the dominant effect during the dry season is reduced atmospheric moisture and rainfall over a wider area due to decreased evaporation, but particularly over the deforested areas (Qin et al., 2025).

Recent attention has also been focused on tree cover loss and its contribution to "tipping points", where local climates may reach a threshold and become unable to sustain the existing moisture regime (Lenton et al., 2019; Xu et al., 2022; Bochow and Boers, 2023; Singh et al., 2024). Drought-prone areas are also susceptible to reinforcing feedbacks, where upwind drought conditions can lead to significant reductions in precipitation (Schumacher et al., 2022).

A grand vision for forest restoration aims to reverse these processes, cooling and stabilizing the climate while contributing to increased moisture supplies to regions currently facing threats or diminished water availability (Alkala and Cestcatti, 2016; Ellison et al., 2017; Sheil, 2018; Cui et al., 2022; Ruijsch et al., 2025). Sheil (2018) and Makarieva et al. (2023) emphasized the potential benefits of increased tree cover, in general, including strategically placed forestations, possibly spanning across borders, to enhance rainfall and water availability downwind through intensified moisture convergence (Weng et al., 2019; Cui et al., 2022). Similarly, promoting the infiltration of rainfall through soil conservation measures in seasonally dry areas can elevate soil moisture during the rainy and immediate post-monsoon periods (WOCAT, 2007; Reij and Garrity, 2016; Meaza et al., 2022). This, in turn, has a favourable impact on air temperatures (reduced) and seasonal precipitation (Walker and Rowntree, 1977; Koster et al., 2004; Castelli et al., 2019; te Wierik et al., 2021; cf. Liu et al., 2023).

Given the bi-directional and highly non-linear nature of the underlying relationships, large-scale forestation may affect precipitation and water availability positively through moisture convergence once atmospheric moisture contents are high enough (Baudena et al., 2021; Liu et al., 2023; Makarieva et al., 2023). There is a need to subject these dependencies to critical evaluation (e.g., Haas et al., 2024; Ruijsch et al., 2025). However, predictions of the magnitude of the effects of land cover (change) on precipitation vary markedly depending on methodological choices (te Wierik et al., 2021; Luo et al., 2022; Spracklen and Coelho, 2023). Further, an unresolved aspect concerns the fraction of the rainfall generated by the large-scale recycling of evaporated moisture from forestation that is sufficiently intense to contribute to deep drainage and groundwater recharge (i.e., affecting baseflows). If most of the rain falls at relatively low intensities, much may then be intercepted, evaporated, and/or used by the vegetation during transpiration instead of contributing to groundwater recharge and baseflow (Owor et al., 2009; Li et al., 2017; Cheng et al., 2020). In the case of the Loess Plateau of China, total precipitation resulting from regional atmospheric moisture convergence increased in some areas following large-scale vegetation restoration and afforestation, but the amounts of "intense" precipitation (defined locally as $> 12$ mm·day$^{-1}$) decreased between 2000 and 2015 (B Zhang et al., 2023). Similarly, using a state-of-the-art water vapour tracker module embedded in a coupled regional atmospheric model to examine the effects of regional greening on precipitation over the Loess Plateau, Liu et al. (2023) derived an increase in precipitation over the growing season of 0.45 mm·d$^{-1}$, 15% of which was inferred to have resulted from increases in local ET. This increase in precipitation was deemed sufficient to not only compensate for the higher evaporative losses associated with regional greening but even increase water yields somewhat (Liu et al., 2023). However, the latter claim is questionable given the small volumes involved, the widespread occurrence of a dried soil layer, and the dominant role of macropore flow in the area's groundwater recharge mechanism (Section 5.2).

Although model predictions of the magnitude of increases in precipitation following large-scale forestation vary depending on model choice and parameterization (te Wierik et al., 2021; Luo et al., 2022), relatively strong effects are generally predicted for montane humid tropical locations (Hoek van Dijke et al., 2022; Teo et al., 2022; Tuinenburg et al., 2022). Globally, the fraction of evaporated moisture that precipitates again "locally" (i.e., at a distance <50 km from its source) is estimated at < 2% only; however, local "moisture recycling ratios" may reach values of 5%–7% in tropical mountain areas (Theeuwen et al., 2023), whereas much higher proportions of locally generated precipitation have been inferred for individual mountains at certain times of the year (e.g., Mt. Kilimanjaro (Schumacher et al., 2020)). Such model predictions are supported by observed increases in the height of the local cloud base following removal of forest in adjacent upwind areas on African mountains (Abera et al., 2024), and in humid tropical lowlands elsewhere (e.g., Costa Rica; Ray et al., 2006). They are also supported by the altitudinal movements of the local cloud base after forests have been impacted by hurricanes, leading to loss and subsequent recovery of foliage (e.g., Puerto Rico; Scholl et al., 2021).

At these smaller scales, occult contributions via cloud water (fog) capture by (mostly taller) vegetation can be a crucial additional source of moisture in forests within coastal or montane cloud belts (Aboal et al., 2000; Bruijnzeel et al., 2011; Dominguez et al., 2017; Hughes et al., 2024). Such extra inputs assume particular importance in semi-arid regions (Hildebrandt and Eltahir, 2006; Klemm et al., 2012; Los et al., 2019), where fog has been shown to markedly increase groundwater recharge (Ingraham and Matthews, 1988; Friesen et al., 2018), but occult contributions can be substantial under windy, wet conditions as well (where they typically include a mixture of fog and wind-driven rain; Bruijnzeel et al., 2011; Frumau et al., 2011; Juvik et al., 2011). Occult inputs have also been reported to aid tree establishment under conditions where rainfall alone is insufficient (Calamini et al., 1998; Rigg et al., 2002; Macek et al., 2018). Nearly two-thirds of all montane tropical forests experience a significant incidence of fog and low cloud (Mulligan, 2010; Wilson and Jetz, 2016). Knowledge of "hot spots" with high fog interception in the (sub)tropics (Bruijnzeel et al., 2011; Klemm et al., 2012) may be used in conjunction with knowledge of local moisture recycling patterns (Theeuwen et al., 2023), to identify suitable areas for enhancing cloud water capture through up-wind FLR projects (cf. Weathers et al., 2020).

## 7. Research needs

Given the extent of tropical forest loss/degradation worldwide, we call for greater involvement of hydrologists, plant physiologists, soil and atmospheric scientists in the development and assessment of FLR initiatives (Dib et al., 2023). Only trans-disciplinary study teams simultaneously looking into the soil-vegetation-atmosphere-continuum might be able to develop the various lines of evidence needed to disentangle trade-offs of forest restoration impacts on hydrological processes, even more so considering climate change. Climate change brings more extreme weather (i.e., higher frequencies of severe meteorological drought, heat waves, and high-intensity storms), changes in atmospheric composition (notably rises in greenhouse gas concentrations), and has various direct and indirect impacts on land cover including shifts in forest species composition, structure, and plant water use efficiency, thereby further affecting such dominant hydrological processes like evaporation and infiltration. The latter, in turn, will affect amounts and timing of groundwater recharge and streamflow (Mileham et al., 2009; Bargués-Tobella et al., 2019; M. Zhang et al., 2022; Sun et al., 2024).

Long-term monitoring is required to understand the non-stationary nature of change (Birkinshaw et al., 2014), but securing funding for such endeavours is increasingly challenging. In particular, more (long-term) monitoring of changes in vegetation water use (all types and growth stages (Yi et al., 2024)) and the resulting changes in streamflow (components), root-zone soil water storage and groundwater levels, soil physical characteristics, surface runoff generation, and rainfall (if any) associated with FLR is needed to further improve our mechanistic understanding of the potential hydrological impacts of forestation initiatives through time. Ideally, such work should represent contrasting climatic and lithological/geomorphic conditions, as well as different stages of surface degradation (soil depth) and vegetation restoration approaches (e.g., natural regeneration vs. active planting), and preferably be part of a dedicated site network (Banwart et al., 2011; Pujari et al., 2020). Advantages of networks include shared measurement protocols for better comparability of results between sites (Banwart et al., 2012; Rebmann et al., 2018) and increased integration across disciplines (White et al., 2015; Guo and Lin, 2016)—in stark contrast to the monodisciplinary nature of most of the studies reviewed here. Given the central role of soil-vegetation-atmosphere interactions (including carbon dynamics) and the interlinking function of water flows, the proposed kind of integrated hydrological observations effectively align with "Critical Zone" research in which soil processes (in the broadest sense) assume a central position (Banwart et al., 2011; Guo and Lin, 2016). Soil degradation and its hydrological impacts already feature in several non-tropical Critical Zone Observatories (Banwart et al., 2012; Chen et al., 2020), whereas others have begun to include vegetation restoration effects (Jia et al., 2024). However, there is still considerable scope for an extension of the international network of Critical Zone Observatories towards the tropics (Guo and Lin, 2016; Pujari et al., 2020) and for the creation of a dedicated subprogramme on the hydrological impacts of land degradation and FLR.

Paired-catchment studies have long been a cornerstone of forest hydrology, but under conditions of rapid climate change, ongoing land degradation, and pronounced hydrological heterogeneity—even between adjacent catchments—additional lines of evidence are needed to reveal the inner working of catchments (notably subsurface dynamics) and evaluate future hydrological responses. Our capacity to measure, monitor, and model environmental change in situ is greater than ever nowadays, including the remote sensing of soil moisture (Dorigo et al., 2017; Peng et al., 2017; Vergopolan et al., 2022; Poehls et al., 2025) and vegetation water use (Baker et al., 2021; Salazar-Martinez et al., 2022). Further, sophisticated models and observations (which can be assimilated to update model variables) allow us to better describe soil–vegetation–atmosphere dynamics (Miguez-Macho and Fan, 2021; van Oorschot et al., 2021, 2024; Burton et al., 2024; Fan and Miguez-Macho, 2024; Schwemle et al., 2024) and seasonal changes in dominant atmospheric moisture transport pathways and precipitation source areas at larger scales (Section 6), while stable water isotope measurements of precipitation allow tracing of water sources (e.g., terrestrial vs. oceanic; Dar and Ghosh, 2017; Xia, 2023) and the dominant rainfall generating mechanism (e.g., convective vs. orographic; Durán-Quesada et al., 2017; Esquivel-Hernández et al., 2019). In situ monitoring of stable water isotopes in soil, xylem, and atmospheric vapour (Kühnhammer et al., 2021; Ring et al., 2023) is increasingly feasible at high-temporal resolution (hourly), albeit still limited to the plot scale. Upscaling such data and studies from plot to, eventually, catchment scales is crucial and likely requires a combination of remote sensing and eco-hydrological modelling (Arciniega-Esparza et al., 2023). However, advances are still needed to elucidate the effect of atmospheric moisture convergence on precipitation amounts and intensity, groundwater recharge, and dry-season flows (as opposed to total streamflow) by including rainfall partitioning at the vegetation and soil surface (into surface runoff, infiltration, evaporation, and transpiration) in such simulation models. Thus far, large-scale model applications have assumed that all precipitation arriving at the soil surface is infiltrating (Wang-Erlandsson et al., 2018; Weng et al., 2019; Hoek-van Dijke et al., 2022; Ma et al., 2024), while complex ecohydrological catchment models such as that of Neill et al. (2021) remain constrained to smaller spatial scales. More parsimonious conceptual isotope-enabled ecohydrological models that explicitly partition water into evaporation and transpiration for different vegetation stages may help bridge the scaling gap (Birkel et al., 2025). Such models could also effectively simulate paired-catchment studies in a virtual environment (Thyer et al., 2024).

## 8. Conclusions

Although a recent “landmark theme issue” comprised of 20 papers in *Philosophical Transactions of the Royal Society* B 378 on Forest Landscape Restoration addressed “knowledge gaps that need closing to advance restoration practice” (Marshall et al., 2022), it did not include a dedicated article on the linkage between forest landscape restoration and hydrology (which was their knowledge gap no. 10). Addressing that omission, we argue herein that healthy soils and a reliable supply of high-quality water are crucial for human and ecological well-being, making them essential considerations for any FLR project (Dib et al., 2023). This conviction is supported by the recent field and modelling studies discussed herein, which demonstrate that forest landscape restoration has the potential to restore the hydrological functioning of degraded tropical catchments disrupted by forest loss and disturbance. However, political will and numerous socio-economic factors aside (Stanturf and Mansourian, 2020; Toto et al., 2025), the realization of this potential also depends on various physical factors, notably local climate (such as seasonality and rainfall patterns), soil conditions (degree of surface degradation, soil depth, and site fertility), and the choice of vegetation (native vs. exotic, shrubs/grass vs. trees, etc.). The timing and extent of hydrological recovery also rely on the initial level of soil and vegetation disturbance and how the emerging vegetation affects the partitioning of precipitation into evaporative losses, surface runoff, and subsurface flow components.

We agree with Marshall et al. (2022) that “there has never been a more important time to deliver the scientific foundations for effective and long-lasting impacts of forest restoration that meets the needs and priorities of different stakeholders”, particularly in the tropics. The evidence we present in this paper challenges the often-repeated notion that “more forest invariably implies less water” by highlighting circumstances where more positive improvements in streamflow dynamics have been brought about via forestation and FLR. Regarding the hydrological goals of FLR, we advocate for prioritizing the recovery of dry-season baseflows, which is mainly achieved through improving soil infiltration capacity and water retention, rather than focusing solely on increasing annual water yields. Restoring this “high-quality” streamflow component also aligns with the need to address other essential ecosystem services, including carbon sequestration, biodiversity, habitat preservation, soil erosion prevention, and (non-reservoir-associated) human water uses (Edwards and Cerullo, 2024). Much work remains to be done, however, to help design suitable vegetation restoration and management strategies for optimum hydrological functioning under specific climatic and soil conditions.

## CRediT authorship contribution statement

**L. Adrian Bruijnzeel:** Writing – review & editing, Writing – original draft, Conceptualization. **Jorge L. Peña-Arancibia:** Writing – review & editing, Visualization. **Douglas Sheil:** Writing – review & editing, Writing – original draft, Conceptualization. **Alan D. Ziegler:** Writing – review & editing, Writing – original draft, Conceptualization. **Jun Zhang:** Writing – review & editing, Visualization. **Bob W. Zwartendijk:** Writing – review & editing, Visualization. **Christian Birkel:** Writing – review & editing. **Ge Sun:** Writing – review & editing. **Yanhui Wang:** Writing – review & editing. **Xiaoping Zhang:** Writing – review & editing.

## Data availability

Data will be made available on request.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Acknowledgements

We thank Dr. Ilja van Meerveld (University of Zurich) for help with Fig. 1. The paper benefited from constructive comments received from five anonymous reviewers.

## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.fecs.2025.100376.

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