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OPINION

Carbon sequestration in tropical forests and water: a critical look at the basis for commonly used generalizations

ANDERS MALMER*, DANIEL MURDIYARSO†, L. A. (SAMPURNO) BRUIJNZEEL‡ and ULRIK ILSTEDT*†

*Department of Forest Ecology and Management, Swedish University of Agricultural Sciences, SE-901 83 Umeå, Sweden, †Center for International Forestry Research (CIFOR), PO Box 6596 JKPWB, Jakarta 10065, Indonesia, ‡Faculty of Earth and Life Sciences, VU University, De Boelelaan 1085, 1081 HV Amsterdam, The Netherlands

Abstract

Tree planting in the tropics is conducted for a number of reasons including carbon sequestration, but often competes with increasingly scarce water resources. The basics of forest and water relations are frequently said to be well understood but there is a pressing need to better understand and predict the hydrological effects of land-use and climate change in the complex and dynamic landscapes of the tropics. This will remain elusive without the empirical data required to feed hydrological process models. It is argued that the current state of knowledge is confused by too broad a use of the terms 'forest' and '(af)forestation', as well as by a bias towards using data generated mostly outside the tropics and for nondegraded soil conditions. Definitions of forest, afforestation and reforestation as used in the climate change community and their application by land and water managers need to be reconciled.

Keywords: carbon sequestration, dry-season streamflow, groundwater, hydrological model, REDD, transpiration, tropical forest, water budget

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Introduction

Trees and forests are being planted in the tropics for a broad range of (sometimes perceived) benefits, including erosion combatment, sustained soil fertility, improved quality and quantity of water supply, as well as socio-economic benefits ranging from enhanced livelihoods and poverty reduction to development and growth of national revenues. Lately potential benefits of carbon sequestration have added value to forest establishment. Win–win scenarios for environment, development and climate have been discussed (e.g., Lal *et al*., 1995; Wunder, 2007), and local examples are accumulating (Murdiyarso & Skutsch, 2006). Total areas of forest plantations can be expected to increase rapidly in the near future with carbon markets expanding and demands for bioenergy increasing (United Nations, 2008).

Correspondence: Anders Malmer, tel. +46 90 786 8416, fax +46 90 786 8163, e-mail: anders.malmer@sek.slu.se

Conflicts between afforestation and forest preservation vs. other land uses have long been on the scientific agenda (e.g., Morris, 1983; Hagberg, 2001; Urgessa, 2003). Today water is increasingly precious in many tropical regions, and often the poor are paying the highest price (Rockström *et al*., 2007). As competition for water is tightening, tree planting has been under increased scrutiny because a number of studies have shown strongly reduced streamflow after afforestation (Calder *et al*., 2004; Farley *et al*., 2005; Jackson *et al*., 2005; Kaimowitz, 2005).

In contrast, there is a widespread public perception that tropical forests act like 'sponges' providing dependable streamflow during the dry season. The underlying scientific argument is that a well-developed forest cover promotes high infiltrability and groundwater recharge during the wet season with a gradual release of water during the dry season. Once the 'sponge effect' is lost by mismanagement of the soil during postforest use, dry-season flows are often seen to decline (Bruijnzeel, 1989, 2004; Sandström, 1998) despite the fact that the new vegetation cover (crops, grassland) typically uses less water than the original forest (Zhang *et al.*, 2004). This 'sponge theory' has long been one of the cornerstones of promotion of forest conservation and reforestation of degraded lands, but while results with decreased dry-season flows after afforestation accumulate (cf. above) there is no rigorous study to show improved dry-season flows after planting trees on degraded tropical land.

One often hears that 'the basics of forest and water relations are well-known'. We argue that this assertion simply does not hold for the mixed land use patterns and dynamic landscapes found nowadays across the tropics, home of the majority of the world's poor. This paper presents arguments as to why generalizations of the forest and water relations in the tropics are weak and tend to be confusing. By critically evaluating the empirical basis for these generalizations, we intend to show that the prevailing confusion is caused in part by a lack of clarity as to what constitutes a 'forest'; in addition, global statements are often based on data that at best represent only a small and biased portion of the tropical afforestation and reforestation spectrum and at worst on data collected mostly outside of the tropics.

Defining one's forest

Too often all forests are 'bulked' in a single group during policy debates, and reference to forest establishment or postforest cultivation is often made without specification. The forest domain is also used to qualify the jurisdiction of particular agencies without specifically describing the vegetation under consideration. Terms like tree planting, reforestation and afforestation are used interchangeably. Forest establishment terminology is sometimes derived from FAO Forest Assessments (e.g., FAO, 2001), which define afforestation as a mechanism by which 'trees are planted to replace another land-use'. An internet search provided $>500\,000$ matches in which definitions of 'afforestation' range from 'converting areas that have not been forests before' to 'converting bare or cultivated areas that have not been forested for a long time'. The latter would include degraded forest land like poor secondary shrub and fire-climax grasslands on soils largely devoid of nutrients and organic matter (Perrolf & Sandström, 1995). For many degraded lands one would expect the hydrological effect of forestation to differ markedly from that of afforesting natural grassland (Scott *et al.*, 2005). Hence, use of the less well-defined term afforestation may be problematic for generalizations about hydrology, as will be increasingly the case for the emerging carbon trade (FAO, 2007). Carbon sequestration under the Kyoto Protocol's Clean Development Mechanism explicitly identifies the achievement of sustainable development as a central requirement. In this mechanism, in which developing countries can participate, forest, reforestation and afforestation are strictly defined for carbon accounting purposes (Noble *et al.*, 2000). Implications of the different definitions depend on the interpretation of the articles of the protocol. Land use may be defined in terms of administrative or cultural purposes and may have little to do with amounts of carbon stored. Similarly, at a certain stage of forest regeneration (e.g., after harvesting) canopy cover may be negligible while no substantial carbon is stored. Afforestation and reforestation are treated identically under the Kyoto Protocol in terms of carbon accounting. The only difference is the timing and sequencing of activities related to the establishment of forest. Afforestation refers to land that has been nonforested for a long period (usually 50 years) while reforestation takes into account the land status in the base year of the accounting rule of the Protocol (1990).

Arguments for scepticism about current generalizations with respect to the tropics

Degraded forest land vs. natural grassland

Land considered for afforestation is often extensively used grassland or shrubland from which the former forest has been absent for a long time. IPCC (2003) defines forest degradation and devegetation mostly in terms of loss of vegetative cover and changes in carbon stocks, which not necessarily equates soil degradation. Postforest soil degradation typically includes decreased water infiltrability due to progressive organic matter decline, exposure to raindrop impact and compaction by cattle or machinery (Perrolf & Sandström, 1995; Bruijnzeel, 2004; Ilstedt *et al.*, 2007). Not all long-term treeless areas are degraded; however, many natural grasslands in the (sub)tropics have high infiltrability and therefore their afforestation will not improve soil physical properties (Bruijnzeel, 2004).

A recent global review of the effects of afforestation on water yield (Farley *et al.*, 2005; Jackson *et al.*, 2005) summarized results from 26 sites between 45°S and 47°N and warned of intensified water shortages after afforestation of grasslands and shrublands, especially in drier regions. It is pertinent to note that only three sites were within 25° from the equator, and only one within 10°. Also, the tropical sites included only a single dry (<1000 mm rainfall) or wet (>2000 mm) location. The three (near-)tropical sites included indigenous mountain grassland in Malawi (Mwendera, 1994), fire-climax grassland and 'Shola savanna' in southern India (Sharda *et al.*, 1998) and natural grassland in South Africa (Smith & Scott, 1992). Given the likely differences in hydrological effects of foresting degraded and nondegraded lands, this attempt at global generalization clearly falls short of adequately representing the tropical situation.

Nontropical data sets are likely to be biased also because of the predominance of more coarse-grained soils being used for nonagricultural uses, as forests on more fine-grained soils have been cleared long ago (cf. Lal *et al*., 1995). Coarse-grained soils are less prone to changes in their (high) infiltrability (and therefore streamflow regime) upon land cover change. Conversely, the mostly finer textured soils found in much of the tropics require continued addition of organic matter as well as bioturbation to maintain topsoil structure and porosity (Lal, 1987). The more rapid decomposition of organic matter in the tropics also implies faster loss of structure (soil degradation) than in other climates. Rehabilitation of more severely degraded tropical soils may take many decades (Lal, 1987; Scott *et al*., 2005), whereas for afforestation in general tree species-specific litter quality is probably one of several important factors in terms of its effect on infiltrability (Fig. 1).

In the most up-to-date review of tropical (af)forestation impacts, Scott *et al*. (2005) discussed the issue of changes in water yield in terms of a trade-off between high forest plantation water use on the one hand, and enhanced infiltration and groundwater recharge through the physical improvement of the soil by the trees on the other. They concluded as a working hypothesis that the increased water use of the trees is likely to override the benefit of increased infiltration unless soil degradation has progressed considerably (cf. Bruijnzeel, 2004; Chandler, 2006). Research needs to establish this critical threshold for different combinations of rainfall regime, soils and tree species.

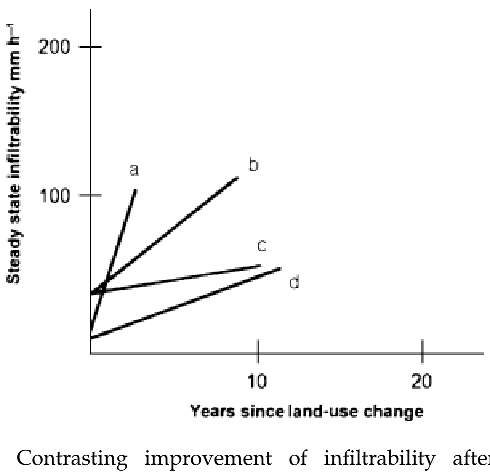

Fig. 1 Contrasting improvement of infiltrability after tree planting (a–c) illustrating the potential for improvement, but also difficulty to make broad generalizations about trees' effect on physical soil properties (after Malmer *et al*., 1998; Ilstedt *et al*., 2007). (a) Open land to *Sespania*, (b) open land to *Leucaena* agroforestry, (c) grassland to *Tectona* (teak) and (d) rehabilitation on tractor tracks under rainforest.

Temporal and spatial scales of forest structure

Forestation of degraded land is unlikely to reproduce the age structure and species mosaic of the former old-growth forest, nor the same spatial distribution of forests in the landscape. There is growing evidence that water use of young, regenerating tropical forest exceeds that of old-growth forest (Giambelluca, 2002; Fritzsche *et al*., 2006). Also, water use by vigorous, planted forest not only exceeds that of natural or fire-climax grasslands and cropland, but also that of secondary and old-growth forests (Scott *et al*., 2005). Scaling up to landscapes, high water use by forest in one area may be balanced by that of newly cut forest or less demanding vegetation elsewhere and such dynamics are likely to be more pronounced under the warmer and moisture conditions prevailing in the tropics (Fig. 2). A high water use is a price we pay for optimizing biomass production (Scott & Prinsloo, 2008) trough growing trees in even aged stands and reducing the rotation age to the most productive age of the trees. Clearly, over a natural life span, the tree species in Fig. 2 have a more neutral water use compared with the period with no trees (giving runoff increase) and the period with young forest (giving runoff decrease).

Hence it is problematic to extrapolate the results of the usually small-scale experimental catchment studies on streamflow (as reviewed by Farley *et al*., 2005; Jackson *et al*., 2005) to the policy-relevant landscape level. At larger scales, the flood-reducing effect of a good forest cover seen in small catchments typically disappears (e.g., Wilk & Hughes, 2002) whereas for baseflows, geology comes into play as well (Tague & Grant, 2004). Regarding both stormflows and low flows, land-use intensity and landscape-level vegetation water

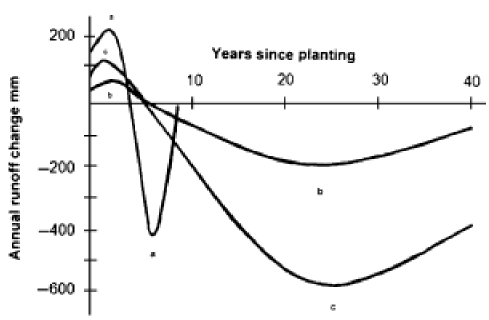

Fig. 2 Patterns of streamflow increase/decrease due to vigorously growing forest plantations. (a) *Acacia mangium*, Malaysia 5°N (Malmer *et al*., 2005), (b) global mean 'shrubland' to *Pinus*/*Eucalyptus* (Farley *et al*., 2005), (c) old-growth *Eucalyptus* to regenerating *Eucalyptus* 37°S (Kuczera, 1987).

use need to be balanced in terms of the proportion of land allocated to food production, timber harvesting (both from natural forest and plantations of various species and age) and old-growth protection forest.

Trade-offs between decreased streamflow and improved infiltration

The balance between rainfall intensity and soil infiltrability is crucial to the partitioning of rainwater into overland flow and soil water recharge. Adequate soil infiltrability becomes critical in the tropics, where the highest rainfall intensities occur (Jackson, 1989). In a recent meta-analysis of peer-reviewed literature on changes in infiltrability after afforestation and introduction of agroforestry in the tropics (Ilstedt *et al.*, 2007), only 14 experiments met reasonable statistical and methodological criteria. From these it was concluded that tree planting increased infiltrability across a wide range of rainfall conditions. The range of improved infiltrabilities were 37–119 $mm\,h^{-1}$ (change 24–106 $mm\,h^{-1}$) for afforestation and 6–172 $mm\,h^{-1}$ (change 2 −125 $mm\,h^{-1}$) for agroforestry. These improvements are very variable, but may be relevant compared with rain intensities in the tropics. However, also data and analysis of rain intensity in the tropics are severely lacking (Bonell *et al.*, 2005). This is the case especially for shorter duration intensities, which is more relevant to compare than more common 24 h data. Figure 3 show a typical intrastorm pattern relating to atmospheric processes during convective rains (McGregor & Nieuwolt, 1998). Relating that to real data, Bonell *et al.* (2005) gave a pantropical range of 50–138 $mm\,h^{-1}$ mean rain intensities of 30 min duration with 2 years return period. Notably in semiarid regions a fair portion of annual rainfall may occur in intense rainstorms. Rowntree (1988) reported 29% of annual rainfall with $>30\,mm\,h^{-1}$ intensity and 11% with $>50\,mm\,h^{-1}$.

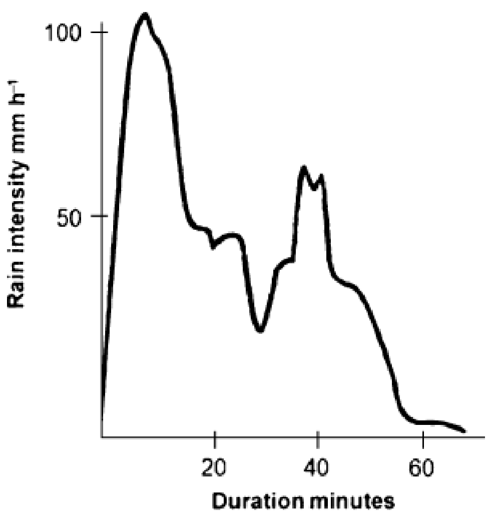

Fig. 3 Typical pattern of intrastorm rain intensity distribution (adapted from McGregor & Nieuwolt, 1998). Apart from this kind of temporal variation there are also spatial variations of rain intensity making it intricate to evaluate.

All individual studies considered in the meta-analysis showed the same positive trend in infiltrability, including those that had to be excluded on methodological grounds (Ilstedt *et al.*, 2007). Even after including the latter, the number of studies did not allow a stringent analysis of the role of factors like tree species, rainfall regime, initial soil type and soil quality or stand management for the degree of infiltrability improvement. However, from these afforestation studies (Fig. 1) and from agroforestry research (e.g., Palm, 1995), clearly tree litter quality is one important factor.

While not a single rigorous study is available to demonstrate improved dry-season flows after reforesting degraded tropical land, it is pertinent that the greatest absolute reductions in stormflow have been observed under precisely these conditions. Some of these reductions should more than compensate the increase in vegetation water use following forestation (Scott *et al.*, 2005; Chandler, 2006).

Current gaps between development policies and research

Agroforestry is a good example of how innovative development of traditional uses of combinations of trees and agricultural crops has evolved to gain a largely positive image. In contrast, tropical forest management is getting a bad reputation despite decades of efforts to develop sustainable timber harvesting techniques (Cassells & Bruijzeel, 2005). We observe a general resistance among donor institutions against the kind of tropical biophysical field research described above. This is often based on arguments that such research is expensive and requires long-term commitment, but also a general feeling that the knowledge gaps are in the socioeconomic rather than biophysical fields. Soil erosion was globally identified as a major environmental problem in the 1970s (e.g., Eckholm, 1976), while famine in the Horn of Africa and the Sahel during the 1980s and 1990s resulted in massive research efforts to quantify soil degradation and promote soil conservation (cf. Hurni & Tato, 1992). As biophysical knowledge was accumulated but development problems continued to prevail, development research rightly shifted towards a more socioeconomic focus (e.g., Ghai & Vivian, 1992). However, the very small number of sufficiently rigorous studies confirming that reforestation actually does improve soils, and the equally small number of studies dealing with other natural or mediated soil rehabilitation processes in the tropics identified by Ilstedt *et al*. (2007) and Scott *et al*. (2005) does illustrate how physical field research is rapidly falling behind. Another example concerns the complete lack of research quantifying how forestation of seriously degraded land affects hydrological functioning at the landscape scale – despite repeated calls for such research (Bruijnzeel, 1989, 2004; Scott *et al*., 2005). Apparently, in the policy and donor arenas the presumably positive impacts of tree planting are simply taken for granted whereas for many researchers this type of long-term empirical data collection does not hold particular attraction.

Multilateral efforts to reduce carbon emissions from deforestation and forest degradation (e.g., REDD) in developing countries is top of the agenda today. On the other hand tropical landscapes already devoid of trees must remain on the international agenda because of associated problems for local residents in terms of livelihoods and environmental services that forests can provide. The present gap in knowledge regarding more intensively managed and spatially complex land uses is unsatisfactory given the current evolution of policies on forest and water relations described above. Available hydrological models, soil–vegetation–atmosphere transfer schemes, and ecosystem models are powerful tools for the spatially and temporally explicit modelling of water, carbon and nutrient fluxes through diverse tropical landscapes (e.g., Barnes & Bonell, 2005; Walker *et al*., 2007), but they lack empirical input data for validation, particularly with respect to the changes in soil physical attributes associated with land cover change and vegetation development (Bruijnzeel, 2004; Scott *et al*., 2005; Ilstedt *et al*., 2007).

We do share the concerns of Farley *et al*. (2005) and Jackson *et al*. (2005) about the need to clarify the environmental costs of tree plantations. With rightfully increasing demands for carbon storage and water, forest management definitions from different fields need to merge, or at least awareness and stringency in their use need to be boosted. Sound reviews and meta-analyses are essential to clarify research and inform policy. Useful generalizations must, however, be based on representative samples and careful analysis, recognizing shortcomings in knowledge. We need to broaden carbon and water management options to consider a multitude of forest and tree uses, not only plantations of fast-growing exotic species. Perhaps old-growth forests should be conserved not only for biodiversity and carbon storage, but also for their relatively low water use compared with regenerating forest. Thus, let managed forest landscapes be more diverse. Let them contain patches of agroforestry and wood source, as well as regenerating and older forests. Through silviculture it is possible to go from degraded grasslands, via forest plantations, to slow-growing indigenous forest (Otsamo, 2002), but concurrent hydrological experiments are sorely lacking. In the absence of strategically focused field research efforts the policy – research gap will only widen, with potentially disastrous hydrological consequences.

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