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Policy implications of a pan-tropic assessment of the simultaneous hydrological and biodiversity impacts of deforestation

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Abstract Tropical deforestation has many consequences, amongst which alteration of the hydrological cycle and loss of habitat and biodiversity are the focus of much public interest and scientific research. Here we examine the potential biodiversity and hydrological impacts of an extreme deforestation scenario – the loss of all tropical forest areas currently identified by the World Wildlife Fund as being threatened. Existing tropical forest areas are first classified according to two categories of biological distinctiveness – high and low – using indicators developed by the WWF. We apply the tropical deforestation scenario to a macro-scale hydrologic model, keeping track of the share of change in basin runoff that originates from the deforestation of areas of high versus low biological distinctiveness and where that change could impact human populations. Of particular interest are those basins where loss of the most threatened tropical forest areas would give rise to significant biodiversity loss *and* to potentially large hydrological impacts. In such cases it is conceivable that biodiversity conservation could "free-ride" on the concerns of resident populations to maintain the forests for the purpose of minimizing hydrological change. Where such an outcome seems likely, biodiversity conservation efforts might be better targeted elsewhere, perhaps to basins where the loss of forest areas with high biological distinctiveness would have less population impacts, hence requiring an alliance between biological and hydrological interests to gain sufficient social and financial support for conservation.

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1. Introduction

The world is rapidly losing its tropical forests, placing a significant fraction of terrestrial biodiversity at risk of extinction. Between 1990 and 2000 alone, 73.5 million hectares (Mha) of closed tropical forest were destroyed (FAO, 2001). Of these once forested areas, about 70% have been completely converted to other land cover types (largely pasture) and to agriculture. Only about 23% has some semblance of tree cover remaining, either in the form of fragmented forests (predominantly forest/agricultural mosaics) or timber and tree crops. In general, there are two main driving forces behind tropical deforestation: (1) *unplanned*, generally more gradual, forest degradation due to rural population pressure and its corresponding subsistence and energy needs, and (2) *planned* conversion of forest to other land uses as part of government-driven programmes to stimulate resettlement, cattle ranching and permanent agriculture, as well as commercial plantations (Bruijnzeel *et al.*, 2005). It is now widely appreciated that environmental costs are not taken into account when tropical forests are converted to cropland or pasture. In the face of tremendous social and economic pressures, conservation of these richly biodiverse landscapes is hampered by many operational challenges.

In principle, society can use regulation, taxes, or environmental service payments to harmonize forestholders' incentives with social goals of maintaining environmental service flows from the forest. (Millennium Ecosystem Asssessment, 2005). In practice, however, there has to be social and political support to impose regulations, levy taxes, or raise funds for environmental service payments (World Bank, 2002). Unfortunately, there is often little local support (Tomich *et al.*, 2004) and inadequate global financing for conservation of globally significant biodiversity. Hence, attention has turned to the role of forests in providing hydrological services. Because flood prevention and sediment mitigation are thought to be highly valued local services, conservationists hope to use these forest functions to motivate the conservation of biodiversity-rich forests (Dudley and Stolton, 2003). The efficacy of this strategy, however, rests on a largely untested and contentious assumption: that protection of biodiverse upstream forests would in fact yield benefits palpable enough to attract the interest of a large and comparatively wealthy downstream urban population. Although policymakers and the general public believe that upland deforestation causes downstream flooding in large river basins, contemporary hydrological science casts doubt on this assumption (Chomitz and Kumari, 1998; Calder, 2005; Bruijnzeel, 2004; FAO/CIFOR, 2005). In large basins, according to a plausible argument, flood conditions will result only when rainfall is so widespread and persistent that land cover is irrelevant to flows. Salient empirical data is however scarce. Due to the economic and environmental costs that would be incurred in performing large-scale forest conversion experiments, the only feasible method for evaluating large-scale hydrologic impacts is through the use of macro-scale hydrological models (Bruijnzeel, 2004; Costa, 2005).

In this paper, we present a macro-scale approach to assessing the potential hydrologic changes, and the consequent impacts on human vulnerability, due to a hypothetical but realistic scenario of future tropical forest conversion to agricultural land uses. We hypothesize that where large populations face the potential for marked change in hydrological regime as a consequence of land cover/use change, there will likely be a larger constituency for forest conservation. We highlight regions where this might be the case.

2. The pan-tropics, forest biomes, basins and biodiversity

The first step in this study was to define the geographical scope of the search for synergy between biodiversity and hydrological function. A focus on tropical forests was born out of a comfort level with the state of evidence-based knowledge on the biodiversity and hydrology of the tropics (the humid tropics in particular), as well as a sense of urgency about the need for new knowledge to support tropical biodiversity conservation efforts. It was then essential to identify reliable sources of information for discriminating the nature and status of biodiversity within tropical forest boundaries. Furthermore, in thinking about the hydrological impacts of tropical deforestation, a broader geographic domain than just the forests must be considered, since basins containing significant tropical forest areas may ultimately discharge into distant, non-tropical areas (e.g. the Nile basin). Thus, the extent and characterization of the study domain was determined by the intersection of a number of spatially-explicit variables: the boundaries of tropical forest areas; patterns of biodiversity within those boundaries; and the larger river basins in which tropical forests were found.

2.1. The pan-tropical domain

A review of the limited sources of biodiversity data that spanned the tropics in a consistent manner identified the mapping and characterization of the world's terrestrial ecoregions undertaken by the World Wildlife Fund[1] (WWF, Olson *et al.*, 2001) as most compatible with the needs of the study. Ecoregions are spatial units made up of complex plant, animal, and microorganism communities and the nonliving environment within which these communities function (CBD, 2002). The WWF map and database of global terrestrial ecoregions, was developed as a result of collaboration amongst over 1,000 scientists from around the world, and employs a 3-tier hierarchical classification system. At the highest level are six broad biogeographical *realms* of which four are found in the tropics. Distributed across the realms are 14 *biomes* – generalized global groupings of ecoregions. The biomes include 3 tropical and sub-tropical forest categories: moist forests (19.6 million km^2), dry forest (3.6 million km^2) and, for the sake of completeness, coniferous forests (0.7 million km^2). These three biomes were selected as the principal means of delineating the tropical forest areas of interest to the study. To complete the WWF hierarchy, biomes are further broken down into 828 unique *ecoregions* globally.

The hydrological context of the selected forest biomes was initially defined by delineating the set of river basins across the tropics in which at least one of these biomes was found. The overall pan-tropic boundary was established by overlaying the three WWF tropical forest biomes (converted to 2-minute resolution gridded fields) on to basins delineated from a 30-minute (0.5 decimal degree) simulated topological network. The minimum area unit of the hydrological model was determined by the availability of a validated and coherent grid of water/river flow paths for the pan-tropics (STN-30, Fekete *et al.*, 2001). Figure 1 shows the resulting pan-tropical boundary that initially defined 1,443 basins ranging in size from 5.9 million km^2 (Amazon) to 2,600 km^2 (single grid cell basins). Many of the world's major river basins, such as the Amazon, the Congo, and the Ganges, fell within the general purview of the study, but so too did many less obvious ones, such as the Nile, by virtue of a relatively small area of tropical forest (when expressed as a share of total basin area) in their headwaters.

[1] WWF-International changed its name to the World Wide Fund for Nature then simply to WWF. WWF-US, which published the Terrestrial Ecoregions of the World map, has retained the name World Wildlife Fund.

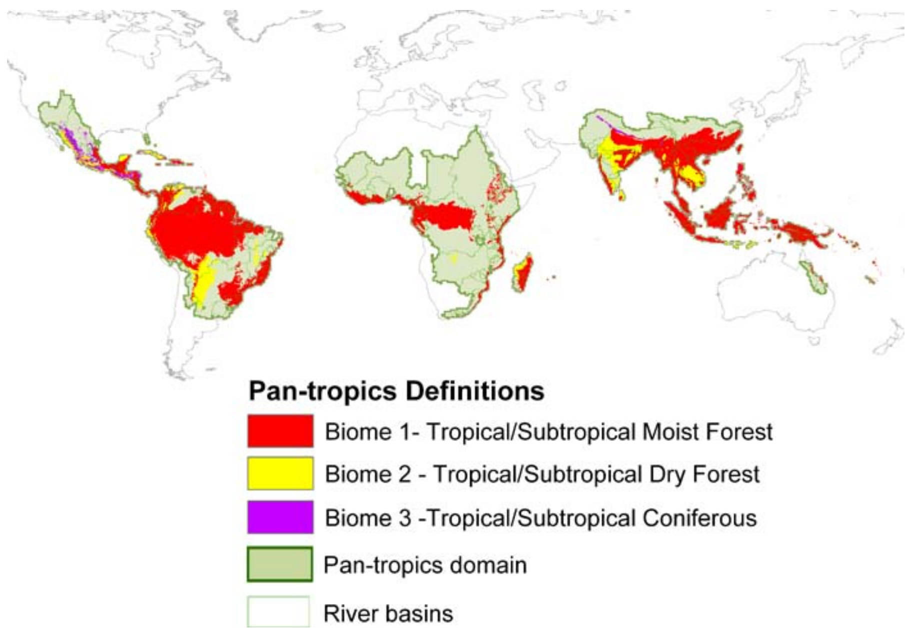

Fig. 1 Tropical forest biomes and the extent of the pan-tropical modeling domain. Forest biomes were extracted from the World Wildlife Fund (WWF) spatial database of terrestrial ecoregions (Olson *et al.*, 2001). The pan-tropical boundary includes river basins within which one or more of the three tropical forest biomes reside

While it was necessary to reduce the number of basins simply on pragmatic grounds, there were also good technical reasons for doing so. Analysis by Fekete *et al.* (2001) suggested that in order to reduce errors associated with representing small basins by a coarse gridded network, a minimum basin size of 30,000 km^2 is recommended. There were some 148 such basins, and these were further narrowed down to 108 focus basins by adopting an additional criterion; that at least 10% of basin precipitation must occur over the tropical forest biomes (Douglas *et al.*, 2005). Of the total area encompassed by this final pan-tropical domain (55 million km^2), the moist forest biome makes up 38%, the dry forest biome 7%, and the coniferous forest biome around 1.3% (Table 1)

2.2. Pan-tropical biodiversity

To obtain an understanding of the characteristics and distribution of biodiversity within tropical forest biomes it was necessary to utilize ecoregion-specific information. Ecoregions are mapped sub-components of biomes, and each ecoregion is associated with information on species richness and endemism (Olson *et al.*, 2001). The WWF defines ecoregions as ‘relatively large units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change’. Olson *et al.* argue that WWF ecoregions provide a better reflection of the distribution of species and communities than other global vegetation maps that rely heavily on modeling of specific biophysical features (e.g. Holdridge, 1967; Bailey, 1998; UNESCO, 1969; and Defries *et al.*, 1995). Furthermore, some of these efforts are defined at a scale considered more equivalent to WWF biomes rather than ecoregions (Mace, 2003). The WWF map includes over 800 ecoregions with an average size of approximately 150,000 km^2,

Table 1 Extent and distribution of tropical forest, protected areas, biological diversity, conservation threat and assumed deforestation by the World Wildlife Fund-delineated biomes and realms

	Total land		Forest area (contemporary)		Protected forest area		Hi-BDI in forest areas		Threatened forest areas		Converted forest areas		Converted high BDI forest areas	
Biomes and realms	Area (000 sq km)	Share (%)	Area (000 sq km)	Share (%)	Area (000 sq km)	Share of forest (%)	Area (000 sq km)	Share of forest (%)	Area (000 sq km)	Share of forest (%)	Area (000 sq km)	Share of forest (%)	Area (000 sq km)	Share of forest (%)
Biomes														
Tropical and subtropical moist Broadleaf forest	19,587	38.0	10,994	73.8	2,179	19.8	9,418	85.7	2,499	22.7	2,409	21.9	2,029	18.5
Tropical and subtropical dry Broadleaf forest	3,619	7.0	700	4.7	98	14.1	519	74.2	411	58.6	352	50.3	269	38.3
Tropical and subtropical coniferous forest	703	1.3	276	1.9	29	10.4	241	87.2	219	79.2	199	72.0	181	65.6
Other tropical biomes	31,016	53.6	2,935	19.7	378	12.9	1,855	63.2	818	27.9				
Total	**54,925**	**100.0**	**14,906**	**27.1**	**2,684**		**12,034**		**3,947**		**2,960**		**2,479**	
Share of forested land (in target biomes)				*100.0*		*18.0*		*80.7*		*26.5*		*19.9*		*16.6*
Realms														
Neotropical	16,923	30.8	6,933	46.5	1,790	25.8	5,250	75.7	1,490	21.5	983	14.2	684	9.9
Afrotropical	18,781	34.2	3,553	23.8	431	12.1	2,991	84.2	953	26.8	276	7.8	208	5.8
Indo-Malay	8,356	15.2	2,881	19.3	260	9.0	2,736	95.0	948	32.9	1,077	37.4	1,026	35.6
Australasia	1,756	3.2	666	4.5	81	12.2	654	98.2	356	53.5	301	45.1	299	44.8
Other realms (within study area)	9,109	16.6	873	5.9	121	13.9	403	46.1	200	23.0	324	37.1	262	30.1
Total	**54,925**	**100.0**	**14,906**	**27.1**	**2,684**		**12,034**		**3,947**		**2,960**		**2,479**	
Share of forested land				*100.0*		*18.0*		*80.7*		*26.5*		*19.9*		*16.6*

in comparison with the average unit size of over 740,000 km^2 found in the well-known global, biogeographical classification of Udvardy (1975). WWF used many of these coarser datasets to establish the first tiers of their hierarchical classification system – the 8 realms and 14 biomes, and then based the ecoregion delineation on expert opinion and other secondary and more local information (Olson *et al*., 2001; p. 934). As an example of the WWF hierarchy, in the Tropical and Sub-Tropical Moist Broadleaf Forest *Biome* within the Afro-Tropical *Realm*, there are 30 distinct forest *ecoregions*.

The WWF ecoregion database links each mapped ecoregion unit within a biome to information on location, extent and various measures of biodiversity, as well as conservation status (to be discussed in Section 3). The biodiversity metric of specific relevance to this study was the "*Biological Distinctiveness Index*" (BDI), a scale-dependent attribute of biological richness based on 5 criteria: species richness; endemism; complexity of species distributions; uniqueness and rarity; and geographic uniqueness (e.g. areas that exemplify global rarity of their habitat type). The measures of richness and endemism used as BDI components were assessed for each ecoregion for birds, mammals and plants (Wikramanayake *et al*., 2002; Dinerstein *et al*., 1995). The BDI is premised on the assumption that, while all ecoregions are biologically distinct to some degree, some are exceptionally rich, complex or unusual. The WWF ranked ecoregions according to their BDI rating as Globally Outstanding, Regionally Outstanding, Bioregionally Outstanding, and Locally Important. Ecoregions are considered outstanding if they exemplify extraordinary levels of the first 4 criteria or if they meet the criterion for geographic uniqueness (Dinerstein *et al*., 1995). The BDI metric was derived independently from the level of "threat" to the ecoregion and as such is a "pure" metric of biodiversity (in other words, the BDI of an ecoregion is not higher if that ecoregion is under a greater threat of human exploitation). For the purposes of this study, all ecoregions classified as containing globally or regionally outstanding BDI were, arbitrarily, assumed to exhibit (medium-to-) "high" biodiversity value, and the remaining classes (bioregionally outstanding and locally important) were classified as "low"(-to-medium). Most tropical forest ecoregions fall into the "High BDI" category; over 85% of the area of both moist broadleaf and coniferous tropical forest biomes, and around 75% of the area of the dry broadleaf forest biome (see Table 1). The distribution of ecoregions of High BDI within the WWF tropical forest biomes and tropical realms is shown in Figure 2.

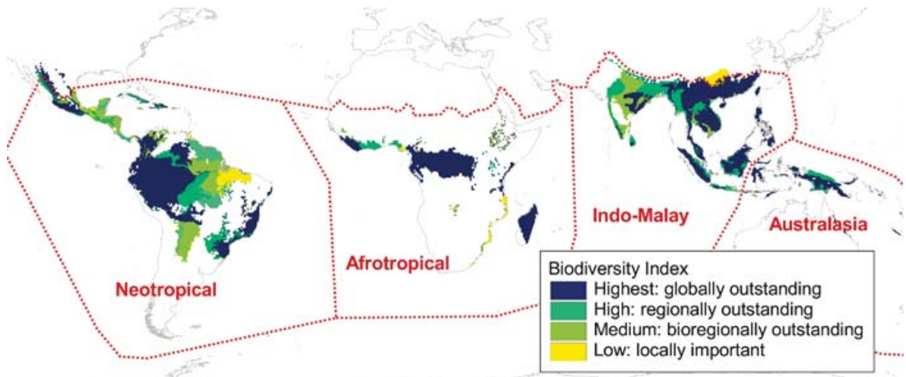

Fig. 2 Biodiversity richness as delineated by the World Wildlife Fund's Biological Distinctiveness Index (BDI) within the tropical forest biomes. The outlines show the major geographic realms that fall within the pan-tropical domain

3. Contemporary land cover and the land cover change scenario

Like other approaches to ecological mapping, WWF biomes and ecoregions provide assessments of untransformed land cover (variously called "potential" or "original" land cover). But a significant amount of transformation has already occurred in the 500 WWF ecoregions that fall within the pan-tropical boundary. A separate assessment of the loss of tropical forests from pre-industrial to contemporary times finds that forest conversion has reached around 23% in the moist broadleaf forest biome, 50% in the dry forerst biome, and 45% for the coniferous forest biome (Douglas *et al.*, 2005). This section describes the process of establishing the contemporary areal extent of remaining forest within the WWF forest biomes, how this contemporary land cover dataset was prepared for use in the hydrological model, and the design and definition of a deforestation scenario (taking the contemporary land use as a baseline).

3.1. Contemporary land cover

From the perspective of this study, potential change in biodiversity and hydrological function are both mediated through change in land cover. Biodiversity impact is assessed through change in forest areas of specific BDI classes. This is a pragmatic approximation that clearly does not address many key dimensions of biodiversity such as habitat fragmentation, species ranges, and sustainability thresholds. Hydrological impact is assessed in a more robust way since more is known (or can be assumed) about the potential above- and below-ground changes brought about by deforestation and how these changes affect hydrological processes. Thus, changes in land cover provide a trigger for changes in relevant parameters in a hydrological model that, in turn, generates changes in predicted river flow (see Section 4). A reliable contemporary land cover dataset, therefore, is necessary for two reasons: first, to calibrate the baseline biodiversity indicator (forest area by BDI class) by establishing to what extent intact forest areas remain within each WWF ecoregion; and second, to provide as reliable as possible an assessment of the actual mix of (all) land cover types within each basin, so that modelled runoff can reasonably be validated against actual gauged basin flows, where these are available. Runoff estimates generated using the contemporary landcover also establish a baseline against which post-deforestation runoff change is assessed.

Several criteria were used to help choose amongst available global land cover dataset options. First was the need to select a land cover data source whose legend would minimize re-classification and harmonization when used in combination with the two other primary spatial data sources: the WWF ecoregion map, which uses a WWF-specific ecoregion legend, and the "TEMVeg" global vegetation map developed by Melillo *et al.* (1993) used in defining land cover shares and associated hydrological parameters for hydrologic modeling. A second criterion was to use a land cover dataset that recognized the heterogeneity of land cover through the use of mosaic classes (i.e., that did not rely solely on a "majority-based" assignment of land cover, but that provided information on land cover sub-categories or shares within a single grid cell). A third criterion, related to the second, was that the land cover dataset be reliable in representing agricultural and urban land uses (since this is a clear indication that transformation of land use/cover has already taken place). In applying these criteria to available land cover datasets that covered the pan-tropical domain, (e.g., GLCCD, 2001; GLC, 2000; Wood *et al.*, 2000), a determination was made that none of them was entirely satisfactory, and that a new composite, contemporary land cover map was needed for the purposes of this study.

The base of the new land cover map was the GLCCD v2 (2001) global 1 km land cover map, integrated with the following components: the global cropland extent (IFPRI, 2002), the global irrigated area map (Döll and Siebert, 2000), global grazing lands (Ramankutty, 2003), and the global night-time lights database (Elvidge *et al.*, 2001). The process of integration is described in Sebastian *et al.* (2003). Although published finally at a coarse resolution (0.5 degree or ∼50 km) this database takes into account land cover shares (available in some of the component databases) below 1 km resolution, and is the first attempt to integrate these thematic land cover data sources into one comprehensive land cover product. The database includes, by grid cell, the majority land cover class and the share of each individual land class (e.g. forest and agriculture shares by grid cell). These data were translated into 0.5 degree grid cell shares of the 20 TEMVeg land cover classes before aggregating to the eight general land cover classes required by the hydrological model.

3.2. Scenario for land use change

The trajectory of actual land use and land cover is difficult to project, and is influenced by a very wide range of economic, social, cultural and environmental factors, including land use policies and land tenure arrangements. In its *Terrestrial Ecoregions of the World* database, the WWF developed an indicator termed the *conservation status* to 'estimate the present and future capability of an ecoregion to meet three goals of biodiversity conservation: to maintain viable species populations and communities, sustain ecological processes, and respond effectively to short-and long-term environmental change' (Wikramanayake *et al.*, 2002; p. 41). The conservation status is determined at the landscape level and is based on an interpretation of evidence on loss of original habitat; number and size of habitat blocks; fragmentation/degradation; conversion rate and degree of protection, and is intended to provide a 30 year prediction of future conservation status given current conservation status trajectories. The classification of the conservation status was based on a quantitative assessment using available maps and current land cover data in conjunction with expert opinion on the region, according to the following categories (Dinerstein *et al.*, 1995):

- *Critical* – The remaining intact habitat is restricted to isolated small fragments with low probabilities of persistence over the next 5–10 years without immediate or continuing protection and restoration.
- *Endangered* – The remaining intact habitat is restricted to isolated fragments of varying size (a few large blocks may be present) with medium to low probabilities of persistence over the next 10–15 years without immediate or continuing protection or restoration.
- *Vulnerable* – The remaining intact habitat occurs in habitat blocks ranging from large to small; many intact clusters will likely persist over the next 15–20 years, especially if given adequate protection and moderate restoration.
- *Relatively stable* – Natural communities have been altered in certain areas, causing local declines in exploited populations and disruption of ecosystem processes.
- *Relatively intact* – Natural communities within an ecoregion are largely intact with species, populations, and ecosystem processes occurring within their natural ranges of variation.

WWF's ecoregion dataset includes both a 'snapshot' or current, and a 'global' or future conservation status, according to the above classes. The 'global' status reflects a 30-year prediction of future conservation status created by modifying the current status by estimates of future threat. The threat estimates were determined based on the cumulative impacts of habitat conversion, degradation, wildlife exploitation and exotic species (Ricketts *et al.*, 1999; Dinerstein *et al.*, 1995; Wikramanayake *et al.*, 2002).

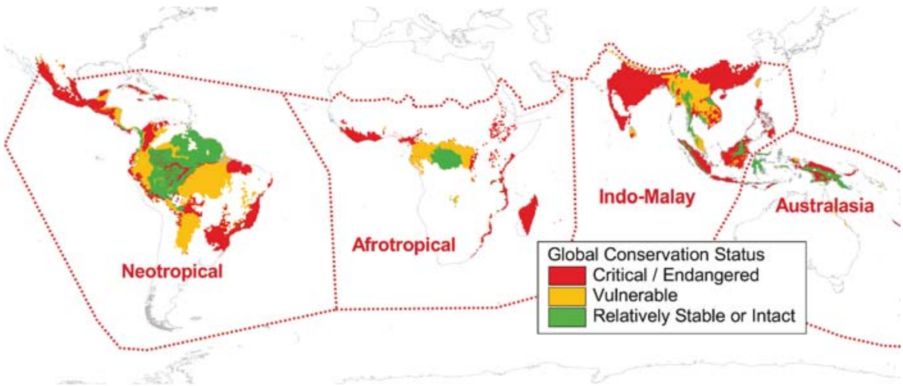

Fig. 3 Deforestation threat as defined by the World Wildlife Fund's Future Conservation Status within the tropical forest biomes. The outlines show the major geographic realms that fall within the pan-tropical domain

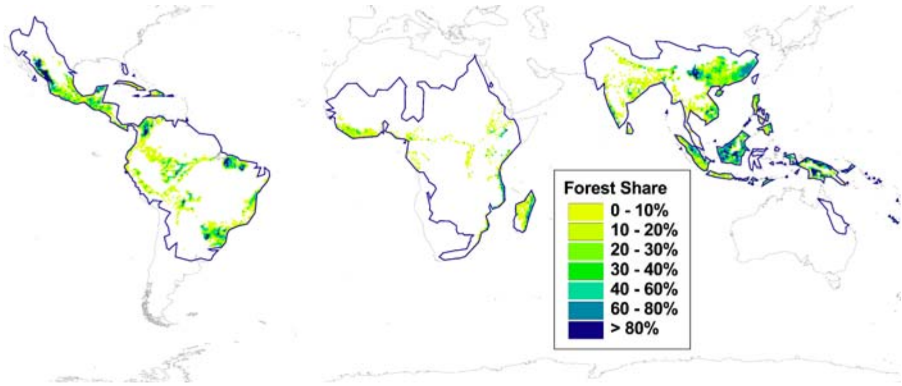

Fig. 4 Hypothetical forest converted (as fraction of grid cell), assuming that all of the most threatened forests are converted to agricultural land use, except for forests under active protection

For the purposes of this study, all ecoregions within the tropical forest biomes classified as *Critical* or *Endangered* according to the WWF's Global Conservation Status were considered to be threatened. This scenario assumes that all area within these ecoregions, *except for existing protected areas*, would be deforested within the overall 30 year time frame of the conservation status (and potentially within the next 15 years). The global protected areas database of the World Conservation Monitoring Centre (WCMC, 1992) was used to define the individual protected areas omitted from the deforestation scenario. Overall, approximately 18% of tropical forest biomes are protected, with the share of protected areas being greatest (26%) in the Neotropical realm (Latin America and the Caribbean), and least (9%) in the Indo-Malay realm (Table 1). Figure 3 shows which ecoregions within the tropical forest biomes were assigned to the most threatened (critical/endangered) categories.

The task in developing the hypothetical deforestation scenario was to identify the most-threatened ecoregions and convert whatever forest cover remained in them to agricultural land cover (the default deforested land use). Figure 4 shows the share of each 0.5 degree pixel within the most-threatened ecoregions that still contains forest, as identified in the contemporary land cover surface. It was these areas of forest that were lost (converted from forest to agriculture) under our hypothetical future land use change scenario.

The total forest area subject to conversion under this scenario is around 3 million km^2 (about 25% of the contemporary forests remaining in the tropical forest biomes) leaving only 9 million of the original (pre-industrial) 29 million km^2 tropical forests intact (Douglas *et al.*, 2005). Over 80% of the converted forest areas, some 2.4 million km^2, lie in the tropical and subtropical moist broadleaf forest biome, but regional differences are large. Around 37% of remaining forest area in the Indo-Malay realm would be converted, but only some 8% of the remaining forest areas in the Afro-Tropical realm (Table 1). After conversion of most-threatened topical forest areas, the proportion of agricultural area in the tropics increased from 13% to an estimated 18%. Interestingly, while this scenario was predicated on threats to biodiversity as a result of deforestation, the resulting 5% increase in agricultural area is within the range of land expansion needed to meet the growth in food demands over the next few decades (Bruinsma, 2003). In other words, while the land use change scenario developed for this study is hypothetical, the possibility of it becoming reality is quite high.

4. The hydrological model

Hydrological modeling was performed using the Water Balance Model (WBM) at a 0.5 degree (30 minute lat *x* long) spatial resolution (Vörösmarty *et al.*, 1998; Fekete *et al.*, 2001; Federer *et al.*, 2003). The WBM simulates monthly soil moisture variations, evapotranspiration, and runoff on single grid cells using biophysical data sets that include climatic drivers as well as vegetation and soil properties. The state variables are determined by interactions among time-varying precipitation, potential evaporation, and soil water content. Soil type and soil texture attributes were derived from the FAO Digital Soil Map of the World (1995; Global Soil Data Task, 2000). The soil type and texture attributes of the dominant soil type in each mapping unit were taken as representative of the entire mapping unit. Potential evapotranspiration (PET) was computed using a modification of the well-known Penman-Monteith surface dependent method (Monteith, 1965; Shuttleworth and Wallace, 1985; Federer *et al.*, 1996) which was found to perform best for global-scale land cover and climate modeling studies (Federer *et al.*, 1996; Vörösmarty *et al.*, 1998). This function requires temperature, net radiation, humidity, wind speed and cover type as inputs. The 30-min gridded datasets of climate variable time series (precipitation, temperature and diurnal temperature range, relative humidity, vapor pressure, and percent cloud cover) were developed by New *et al.* (1998). Operationally, for any time step in which rainfall exceeds the soil moisture deficit (the amount of soil moisture holding capacity currently unreplensihed), the excess is used to augment a rainfall-derived detention pool and to generate runoff. The soil infiltration rate is assumed to equal the rainfall rate; therefore overland flow is not explicitly simulated.

The necessary vegetation characteristics for WBM were derived from land cover inputs developed specifically for this study as described in Section 3. The major influence of vegetation type in WBM is in the computation of evapotranspiration. Broad vegetation classes are characterized using parameter values derived initially from the literature and subsequently refined through model application and validation (Table 2; see Federer *et al.*, 2003 for details). Two important vegetation parameters are leaf conductance and rooting depth. Maximum leaf conductance (GLmax) values in WBM are consistent with maximum leaf conductance observations for tropical forests and crops presented in Schulze *et al.* (1994). A sensitivity analysis showed that a 20% change in GLmax resulted in a change in annual runoff of 1.5% or less over the pan-tropics. Methods for determining appropriate rooting depths for global models can vary quite substantially (Kiedon and Heimann, 1998; Zeng, 2001). To improve the performance of the hydrological model in matching recorded basin flows, the following

Table 2 Vegetation parameters used in the Water Balance Model

Parameter (units)	Conifer forest	Broadleaf forest	Savannah/Pasture	Grassland	Cropland
Albedo (1)	0.14	0.18	0.18	0.2	0.22
Conopy height (m)	25	25	8	0.5	0.3
Max leaf area index (1)	6	6	3	3	3
GLmax[a] (m/s)	0.0053	0.0053	0.0053	0.008	0.011
Leaf width (m)	0.004	0.1	0.03	0.01	0.1
Zero-level roughness height (m)	0.02	0.02	0.02	0.01	0.005
98% root mass depth[b] (m)	1.61	1.01	1.0	1.0	0.49

Notes: [a]GLmax is maximum leaf conductance, the reciprocal of stomatal resistance
[b]Conifer and broadleaf rooting depths interpred from Jackson (1996)
Rooting depths for other cover types were values already established in previous model applications

adjustments were made in vegetation related parameters: forest rooting depths were modified to those published by Jackson *et al*. (1996), as recommended by Federer *et al*. (2003); pastures were assumed to be equivalent to savannahs; closed canopy woodlands were included in the dry forest class; mixed forests were split equally between the moist and dry forest classes; and both dry and moist tropical forests were modeled using the same broadleaf forest parameters. Hence, soil moisture availability rather than differences in vegetation characteristics was the major determinant of runoff generation in dry forests. To represent forest canopy interception (which is not explicitly modeled) the effective monthly rainfall utilized for simulating runoff from forest areas was computed off-line as 80% of observed monthly precipitation. A value of 20% canopy interception was selected because it was within the range of published values for annual canopy interception losses (Jackson, 1975; Calder, 1990; Bruinjzeel, 1990) and because it resulted in an average grid cell increase in annual runoff due the conversion of forest to agricultural land use similar to field observations (Oyebande, 1988; Bonell and Balek, 1993; Bruijnzeel, 1991; 1996). Average rates of wet canopy evaporation from forests can exceed those of shorter vegetation by two to five times (Calder, 1990; Bruijnzeel, 1990); hence interception for other vegetation types was not modeled. Irrigated croplands were simulated as having saturated soils. Irrigation withdrawals are not simulated in the current version of WBM, therefore, runoff from irrigated lands may be overestimated.

A common perception is that deforestation increases human vulnerability to extreme events such as floods. However, the most dramatic hydrologic effects of land use change are often short-lived, and have only been shown to impact smaller magnitude, higher frequency events, since the role of land cover decreases as the magnitude of the event increases (Bruijnzeel, 1996). The hydrologic analysis for this study was limited to modeling changes in the long-term average annual, monthly maximum and monthly minimum runoff as a first step in understanding the impacts of land use change on these events. Runoff was computed by first generating separate runoff estimates for all land cover types independently, and then summing the runoff derived from each cover type within each grid cell according to the share of grid cell area occupied by that cover type (using the land cover shares per pixel as estimated by the prior land cover and land cover change analysis). Increases in annual runoff due to the conversion of forest to agriculture are within the range of values observed from field studies (Oyebande, 1988; Bruijnzeel, 1996, 2004). The model performed well in matching long-term average annual flows in rivers across the pan-tropics (see Appendix A of Douglas *et al*., 2005 for a more detailed discussion). Annual changes in runoff were obtained by running the hydrologic model with long-term mean annual climate inputs, computed from New *et al.*, (1998), and comparing runoff generated from the contemporary land cover and the hypothetical future land cover datasets. Changes in maximum and minimum monthly runoff were obtained by running the hydrologic model with long-term mean monthly climate inputs and then selecting the maximum and minimum differences in runoff between the two land cover datasets. Annual and monthly river discharge (Q) was computed by accumulation of gridded runoff along a digital river network (STN-30, Fekete *et al.*, 2001). Flow impoundments were not represented in the model, hence the effects of hydroelectric power generation and reservoir siltation, which are important impacts in some pan-tropical basins (e.g., the Parana), were not investigated in this study.

5. Impacts of deforestation on biodiversity and hydrology

5.1. Biodiversity related impacts

Just over 80% of the tropical forest biomes are characterized as having High BDI, while of the contemporary forest areas in other tropical biomes, only around 63% are categorized as High BDI. The Australasia and Indo-Malay realms have the highest share of High BDI, around 98 and 95% respectively, and the Neotropical realm has the lowest, at around 76%. Of note is that forest protection is inversely related to the share of High BDI, e.g. only 9% of the more biodiversity-rich Indo-Malay tropical forest biomes are protected, in comparison to 26% of the Neotropical forest biomes (Table 1). With regard to conversion threat and projected forest conversion, around 26% of the tropical forest biomes are categorized as most-threatened, but this share falls to around 20% after subtracting out existing protected areas. The share of most-threatened forest areas differs considerably by biome. Around 23% of moist broadleaf forest, 59% of dry broadleaf forest, and 79% of coniferous forests are classified as most-threatened. Most-threatened status is most prevalent in Australasia (54%) and least prevalent in the Neotropics (22%). The share of High BDI areas within the converted forest areas also varies significantly by biome and realm. 84, 76 and 91% respectively of the most-threatened forest areas are classified as exhibiting High BDI for the moist broadleaf, dry broadleaf and coniferous forest biomes respectively. From a regional perspective, the share of High BDI areas that are converted are consistent with the overall share of High BDI in the forest areas of the Indo-Malay and Australasia realms, but in the Neotropical and Afrotropical realms they are somewhat less (e.g. a proportionately lesser share of High BDI areas are to be found in the most-threatened forest areas in those regions). The shares of total forest with High BDI and converted area with High BDI are 76 and 70%, respectively, in the Neotropics, and 84 and 75%, respectively, in the Afrotropical realm (Table 1). All the converted forest area in Australasia is classified as High BDI.

5.2. Hydrology related impacts

Figure 5 shows the change in average discharge (ΔQ, in km^3) relative to contemporary discharge (Q, in km^3) for long-term average annual flows (Figure 5a, from Douglas *et al.*, 2005) and long-term average maximum flows (Figure 5b). For annual discharge, the total hydrologic impact of the projected land cover change was less than 5% of contemporary Q, but the impacts were focused in southern China, western Mexico and the Yucatan peninsula, with more localized areas in Paraguay and Bolivia, and in Kenya. For the mean maximum monthly flows, a similar pattern emerged. Figure 5b shows a slight reduction in the area

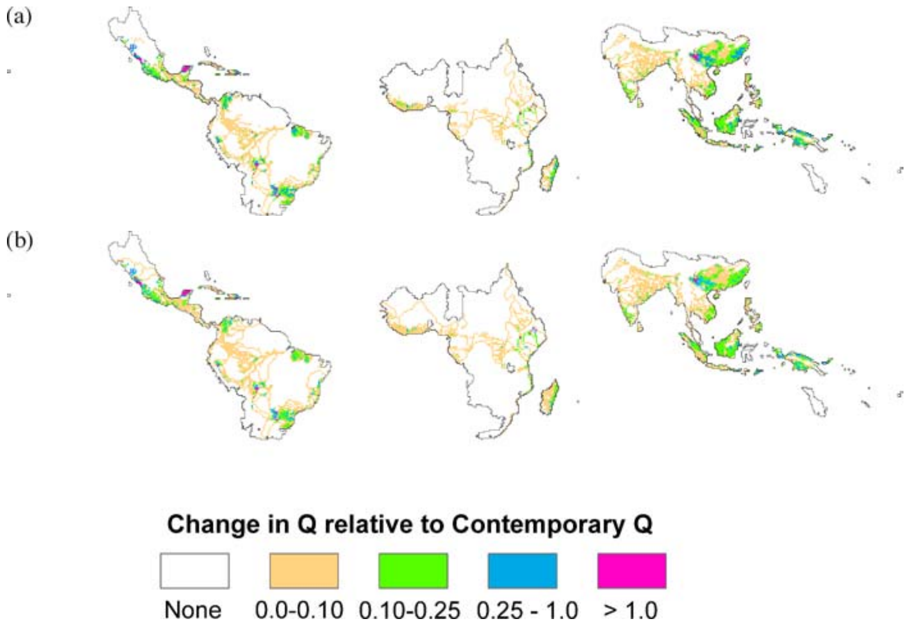

Fig. 5 Proportionate change in (a) mean annual and (b) monthly maximum flows arising from the removal of the most threatened tropical forests areas depicted in Figure t4

where $\Delta Q_{\max}/Q_{\max} \geq 0.25$ in Southeast Asia when compared to Figure 5a, whereas the patterns are essentially identical along the western coast of Central America. Annual rainfall in western coastal Central America ranges between 500 and 1000 mm, while over continental Southeast Asia annual rainfall exceeds 1000 mm and over Indonesia, it exceeds 2000 mm. The spatial differences between Figure 5a and b again support the assertion that deforestation results in greater relative hydrologic impacts in drier climates than in wet.

Table 3 summarizes the annual and maximum monthly discharge changes by biome, realm and basin size. Over the entire pan-tropics, the long-term average maximum monthly discharge increased by 159 km^3/mo or about 3% of the contemporary maximum monthly flows. The majority of this increase (138 km^3/mo) occurred in the moist broadleaf forest biome, but the coniferous forests experienced a greater percentage increase (11%) than either moist or dry forests, both of which had a 5% increase in maximum discharge on average. The Indo-Malay realm had the largest increase in magnitude, but the Australasia realm saw the largest percentage hydrologic impact for annual and maximum monthly increases. Forest conversion on the island of New Guinea accounted for all the hydrologic changes in this realm. While it is generally accepted that forest conversion results in increased annual flows and, as shown in this analysis, in increased high flows, the impact on low (base) flows is more complex and difficult to predict (Bruijnzeel, 1996). Aggregated over the entire pan-tropics, long-term mean minimum monthly flows were projected to increase by about 3%. Our hypothetical forest conversion in the moist and dry tropical forests resulted in a 10 and 2% increase in minimum flows, while a 53% decrease in minimum flows occurred in coniferous forests. However, the decrease in coniferous forest base flow only amounted to 1 km^3/month due to the small extent of the coniferous forest biome within the pan-tropics. Because of limited rainfall and greater seasonal variability in the dry and coniferous forest biomes, increased runoff from forest conversion may have been offset by greater soil moisture deficits, resulting

Table 3 Changes in long-term average annual, monthly maximum and monthly minimum discharge (Q) due to the hypothetical conversion of the most-threatened forests to agricultural land uses (cropland and pasture)

	Contemporary flows			Change in flows after deforestation					
Realms, biomes and basins	Avg Annual (km^3/yr)	Av. Max month (km^3/yr)	Av. Min month (km^3/yr)	ΔQ Annual (km^3/yr)	ΔQ Max (km^3/mon)	ΔQ Min (km^3/mon)	ΔQ Annual (%)	ΔQ Max (%)	ΔQ Min (%)
Change in Q by tropical forest biome									
Trop & Sub-trop Moist Broadleaf Forest	15,608	2,893	319	927	138	31	6	5	10
Trop & Sub-trop Dry Broadleaf Forest	1,052	298	5	63	16	0.1	6	5	2
Trop & Sub-trop Coniferous Forest	242	56	2	19	6	−1	8	11	−53
Change in Q by realm									
Neotropical (Latin America)	10,086	1,916	160	294	50	6	3	3	3
Afrotropical (Africa)	4,510	1,128	33	73	14	1	2	1.2	3
Indo-Malay (E, S, SE Asia)	5,765	1,225	96	411	63	13	7	5	13
Australasia	1,471	198	66	182	20	11	12	10	16
Other Realms (in study area)	873	198	11	49	12	0	6	6	−1.2
Change in Q by basin size									
Very Large Basins (>100,000 km^2)	12,351	2,567	157	325	58	6	3	2	4
Large Basins (>50,000–100,000 km^2)	1,970	360	49	201	29	7.3	10	8	15
Medium Basins (30,000 – 50,000 km^2)	969	164	28	66	9	3.0	7	5	11
Pan Tropics	**22,705**	**4,665**	**366**	**1,008**	**159**	**30**	**4**	**3**	**8**

in less runoff available for base (low) flow. In the mountainous regions of western Mexico and Central America, a combination of low vapor pressure deficits and low modeled leaf conductance used in simulating coniferous forests (see Table 1) may have resulted in an underestimation of ET and an overestimation of runoff from these forests. The conversion of these forests, predominantly to pasture, resulted in the model predicting decreased rather than increased runoff due to deforestation in these areas. The location of these "negative runoff" effects roughly coincide with the locations of tropical montane cloud forests (TMCF) shown in Bruinjzeel (2005, Figure 18.2, pg. 465). Bruinjzeel (2005) and Bruinjzeel and Proctor (1995) note that little is known about the hydrological functioning of TMCF or about the hydrologic effects of converting these forests to cropland or pasture. Measured interception losses in tropical montane environments can be as high as 45% of precipitation (Bruinjzeel, 2005, Table 18.2, pg. 470), which is more than twice the amount that we assumed in our model. This could have led to our anomalous "negative runoff" results in these areas, the magnitude of which ultimately had a negligible impact on our results. Field studies are currently underway to better quantify evapotranspiration and interception processes along with appropriate vegetation parameters for TMCF in Central America (Sampurno Bruinjzeel, World Bank, personal communication, October 13, 2005), which may shed light on the modeled behavior in these areas.

Table 3 also summarizes hydrologic changes by basin size. Out of all 108 focus basins, 99 (92%) showed an increase in long-term mean maximum monthly flows while 8 had no change. For the long-term mean minimum monthly flows, 84 of the 108 focus basins (78%) showed an increase while 16 (15%) showed a decrease. The remaining 8 basins had no change. While the very large basins (basin area $>100{,}000\,km^2$) had the largest magnitude change in long-term mean annual and maximum flows (325 and $58\,km^3$, respectively), large basins (area between 50,000 and $100{,}000\,km^2$) had the largest change in mean minimum flows ($7.3\,km^3$ or 8%). The number of basins with increasing minimum flows was distributed fairly uniformly across the three size classes (29, 32, 23 respectively), but most of the basins experiencing decreased minimum flows were in the large and very large size classes. In relative terms, however, large basins (basin size between 50,000 and $100{,}000\,km^2$) had the largest increases in annual, maximum and minimum flows, followed by medium sized basins (basin size between 30,000 and $50{,}000\,km^2$).

5.2.1. Potential human vulnerability to hydrological change

A key factor in examining the effects on human welfare of changes in land use and hydrological function is recognition that the populations affected by change are topologically linked to disturbance through river networks. Thus affected populations could be living both in the areas where the land use change takes place and in areas downstream of these changes. The hydrological response can be propagated far downstream of the actual point of disturbance and become intensified or diluted depending on the characteristics of change in the influent tributaries (Douglas *et al.*, 2005). Of the 3.7 billion people who live within the pan-tropic domain, approximately 2.2 billion people – about 1/3 of the world's total – reside within the boundaries of the WWF tropical forest biomes (Table 4) with 1.7 billion (or 70%) living in areas classified as having High BDI. As previously noted, the WWF biomes delineate "potential" tropical forest areas, at least one-third of which have already been deforested (Douglas *et al.*, 2005). Approximately 570 million people (28% of the pan-tropical population) live in contemporary (circa 1992/3) tropical forest areas. Of these, nearly one-half (250 million) live in the most-threatened forest areas, highlighting the intense human pressure on the remaining forest (Table 4). Densely populated urban areas could be especially vulnerable to the effects

Table 4 Distribution of forest and population by biome, realm and basin size

Realms, biomes and basins	Total area (000 sq km)	Totol population (M. persons)	Total population in contemporary Forest area (M. persons)	Population in deforested pixels (M. persons)	Flood plain population (M. persons)	Population densities: Total pop/Total area person km^{-2}	Population densities: Forest pop/Forest Area person km^{-2}	Population densities: Floodplain pop/Total Area person km^{-2}
By tropical forest biome								
Trop & Sub-trop moist broadleaf forest	19,587	1,736	337	211	453	89	31	
Trop & Sub-trop dry broadleaf forest	3,619	406	48	26	55	112	68	
Trop & Sub-trop coniferous forest	703	65	22	13	1	93	78	
By Realm								
Neotropical (Latin America)	16,923	433	71	41	60	26	10	
Afrotropical (Africa)	18,781	625	72	14	90	33	20	
Indo-Malay (E, S, SE Asia)	8,356	1,962	299	162	502	235	104	
Australasia	1,756	25	11	3	1.1	14	16	
Other realms (in study area)	9,109	637	116	30	168	70	133	
By basin size (108 focus basins only)								
Very large basins (>100,000 km^2)	25498	1804	299	108	461	71	30	18
Large basins (>50,000–100,000 km^2)	2890	264	44	30	69	91	40	24
Medium basins (30,000–50,000 km^2)	1133	87	13	7	45	77	28	40
Pan tropics	**54,925**	**3,682**	**568**	**250**	**821**	**67**	**38**	

of hydrologic changes when located in floodplains along major rivers. At the time of this study, the Global Rural Urban Mapping Project (GRUMP, CIESIN, 2005) had not yet been completed and it was not possible to break the pan-tropical population into rural and urban inhabitants. However, according to this dataset (now completed), approximately 35% of the pan-tropical population lives in urban areas.

6. Implications – a strategic perspective

Douglas *et al.* (2005) used some elements of the analyses described above to identify "hydrological hotspots" where the following conditions applied at the pixel level: forest area within a tropical forest biome, high BDI (WWF globally or regionally outstanding), most-threatened (WWF critical or endangered conservation status), and $\geq$25% projected increase in annual runoff if the forest areas are converted to agriculture. Categories of hotspots for pixels that met these criteria were then assigned on the basis of floodplain population. About 104 million people (roughly 40% of the total number of people living within the most-threatened forest areas) are deemed to be potentially at risk from high levels of hydrological response to deforestation. More than three-quarters of these people (80 million) live on floodplains within or downstream of these highly responsive areas, which makes them particularly vulnerable to both immediate and long-term changes in hydrologic regime. Some areas with elevated risk of hydrological change and biodiversity loss according to this approach are found in east and southeast Asia, in particular in the Zhujiang, Menjiang, Chang Jiang, Fuchun Jiang, Hanjiang, Menjiang and Hong basins of southern China. Other areas include western Mexico where a series of smaller watersheds along the Pacific coast were highlighted as having increased vulnerability in terms of hydrology, biodiversity and populations. In South America, the Parana basin which covers parts of Argentina, Paraguay and southern Brazil also contains several hotspot areas. The Parana basin was targeted by the land use change scenario primarily due to the lack of protection and the degree to which forest areas are currently deemed to be under threat.

The above analysis can be extended to examine a mix of possible biodiversity and hydrological outcomes from deforestation at the basin scale, and the potential strategic implications of each. To do this, the 108 focus basins were categorized according to:

(a) The proportion of High-BDI forest areas that would be lost, where "High BDI" areas are those originally occupied by globally or regionally outstanding forests (see Figure 2). Basins where this ratio exceeded 0.75 (the mean across basins) were categorized as "higher BDI loss"
(b) The ratio of threatened population to total forest area converted (TP/FC), where threatened populations are those with within "hydrologic hotspots", grid cells where the change in Q due to deforestation relative to contemporary $Q(\Delta Q/Q)$ was $\geq$25%. Basins where TP/FC >40 people/km^2 (the mean across basins) were categorized as 'higher potential downstream impact'.

The result is four categories based on relative biodiversity and hydrological impacts:

Category 1 (Low B-Low H): lower biodiversity loss, less population affected
Category 2 (Low B–High H): lower biodiversity loss; more population affected
Category 3 (High B–Low H): higher biodiversity loss; less population affected
Category 4 (High B–High H) higher biodiversity loss; more population affected

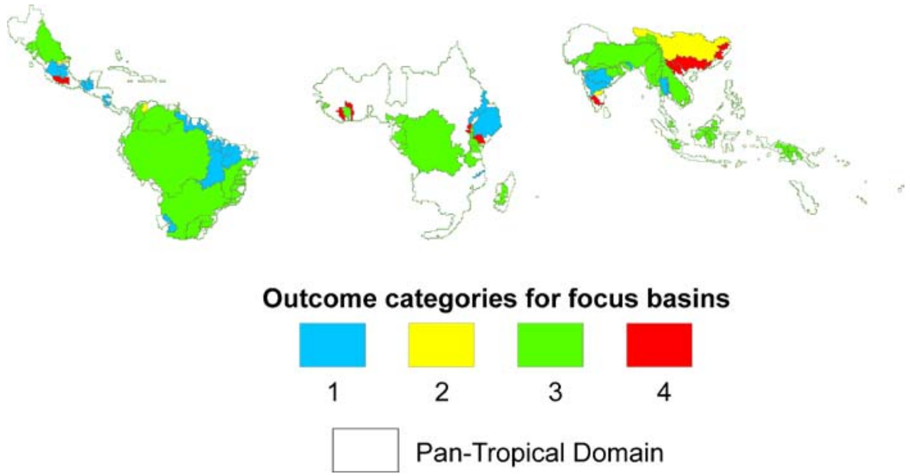

Fig. 6 Outcome categories (as defined in text) assigned to pan-tropical focus basins

Most basins (60 out of 108, or 56%), were classified as *High B-Low H* including the Ganges, Parana, Mekong, Amazon, Zaire, Orinoco, Rio Grande, Sao Francisco and Magdalena, basins. This category represents 59% of the total focus basin area. Thirty-one percent of focus basins were in the *Low B-Low H* (e.g., Krishna, Jubba, Godavari, Chao Phraya) and 4% were in the *Low B-High H* (e.g., Chang Jiang, Penner) categories. Ten percent of the focus basins were classified as *High B-High H*, including the Zhujiang and Hong rivers. Figure 6 shows the spatial distribution of basins by outcome category. Most of the basins in the Neotropics were classified as category 1 or 3, due to relatively lower population densities, whereas in the Indo-Malay region, all categories are represented. These rough categorizations can be used to assess the feasibility of devising strategies, assembling constituencies and raising funds for forest conservation. Support for conservation depends on beneficiaries' perception of its benefits relative to the costs of organizing, negotiating with forest dwellers, prohibiting illegal deforestation, and so on. On the hydrological side, forest dwellers' perceived benefit/cost ratio for supporting conservation may be proportional to the ratio of threatened population/converted forest area (TP/FC). For instance, potentially affected populations may be willing to pay \$10/person/year to avert a 25% increase in annual flows, to pick an arbitrary but plausible number. If TP/FC > 40, then forest conservation may be justified, and possible, if costs of conserving the forest are less than \$400/km^2/year. Mobilizing conservation support becomes more feasible with higher TP/FC, wealthier populations, and lower opportunity costs or implementation costs of forest maintenance. There are 13 basins with TP/FC > 40, and 9 with TP/FC > 100. The latter have a combined population of 42 million and a combined at-risk forest area of 229,000 km^2. In basins with the highest TP/FC, domestic hydrological concerns might play a leading role in driving forest conservation, with biodiversity conservation as a side benefit. Conversely, in category 1 and 3 basins, interest in maintaining globally important biodiversity will probably play the most important role. (Note however that there may well be important domestic benefits of biodiversity conservation in the lower-BDI forests.) In many ways category 4 (High Biodiversity, High Hydrological Impact) is the most interesting. In principle, conservation of these forests could be supported and financed either by biodiversity or hydrology beneficiaries. In practice, it is likely that an alliance between these two interests will be important to achieving social and financial support for conservation.

7. Concluding remarks

This study was predicated on the existence of significant tracts of tropical forest that provide both havens of biodiversity richness and socially beneficial watershed services. The goal was to identify the location and extent of such tracts within the tropics globally and to generate evidence of the nature and scale of the biodiversity and the hydrological services they deliver. Land cover surfaces were generated to represent two “snap shots” in time; a contemporary view, based on a combination of existing evidence representing the state of land cover/land use in the mid-1990s; and a hypothetical future land cover, representing conversion of the most-threatened tropical forest tracts to agriculture. The biodiversity and hydrological impacts and potential human threats of the hypothesized land cover changes were examined, some hydrological hotspots identified, and a prototype biodiversity conservation strategy schema developed. These results appear promising and point to the validity of future work in two main areas; further validation and elaboration of the empirical results in terms of hydrological hotspots and improved typologies of biodiversity conservation strategies, as well as the need for continued improvement in the underlying data and analytical approaches employed.

With regard to the measures used in the assessment, there are clearly some weaknesses. First is the disconnect between a relatively rich and consistent global characterization of biological distinctiveness developed by WWF for forest biomes and ecoregions that may in reality have already been significantly degraded – at least according to the contemporary land cover evidence. Second, the biodiversity grouping schema used here that treats only areas of global and regional distinctiveness as being of high biodiversity value gives perhaps insufficient weight to other, still important categories of biodiversity. But the approach does highlight where major challenges will likely be faced from a regional and pan-tropic perspective. Third, a better understanding of the actual magnitude of deforestation threat would represent an improvement over the simple threat ratings used here, and might significantly change the spatial pattern of prioritization. Finally, there are also shortcomings in the resolution (both temporally and spatially) and completeness of the climate, hydrological, and basin scale components of assessing (changes in) river flow. Ideally the hydrological modeling would have been capable of examining the impacts of deforestation on extreme runoff events (particularly high flows) at a higher temporal resolution than one month, and over a lengthy time series rather than through the use of a long-term average climatologies. Some improvements were made in the hydrological process model in terms of representation of interception storage in forest areas, but runoff modeling could still be improved through better representation of infiltration, delayed base-flow storage, and river flow routing. But there are also pragmatic analytical constraints when running a global hydrological model with multiple land cover types represented in each pixel that force analytical tradeoffs to be made. Work on developing a downstream indicator of human vulnerability is still in progress.

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