CHAPTER 2 Forests

2.1 Scope of the assessment

When considering the *hydrological functions* associated with forests (Section 2.4) and the resultant impact on the delivery of *ecosystem services* (Section 2.6) it is often the activities that take place (or do not take place) within closed forests (or open woodland) rather than the impact of individual trees that require assessment. Thus the focus of this synthesis of hydrological functions and ecosystem services strictly should relate to 'Forestlands' (cf. 'Wetlands') rather than 'Forests', to capture both the effects of individual trees and the impacts of management practices on soils, water and microclimates within forested areas. The interaction of hydrological functions with forest functions for carbon capture and retention will be discussed separately in Section 2.6.

This assessment covers all global forests along the latitudinal gradient from boreal forest (50-60^{o} N) to temperate forest and then tropical forest (Foley *et al*., 2005). Tropical forests include small areas of Tropical Montane Cloud Forest.

2.2 Global extent of forests

Forest and woodland areas with more than 10 percent tree cover currently extend over 4 billion hectares or 31 percent of the land area of the globe (Fig. 2.1). FAO (2010) hase estimated that 65 percent of these forests are, however, in a disturbed state. Hansen *et al* (2008) suggests that this figure may be even higher for lowland evergreen rain forest in the tropics. Further disturbance is expected, given that some 30 percent of the world's forests are classified as Production (rather than Protection) Forest where commercial forestry operations predominate; plus a further 16 percent of the world's forests are unclassified (FAO, 2010) and likely to be subject to disturbances. Within some tropical regions, notably Asia, tree planting is off-setting the rate of forest loss. Within this region, newly forested areas now exceeded 120 million hectares (FAO, 2010). The global rate of reforestation and afforestation cannot, however, offset the net loss of 7-11 million km^2 (0.7-1.1 billion hectares) of closed forests over the last 300 years (Foley *et al*., 2005); this includes 2.4 million km^2 and 3.1 million km^2 lost from North America and Europe, respectively (Bryant *et al*., 1997). Indeed, Drigo (2004) calculated a ratio of 18-24: 1 for the balance between closed forest destruction to forest planting. Consequently, it is essential that that this synthesis properly quantifies the significance of findings pertinent to the globally extensive *disturbed natural forests* in addition to those studies from undisturbed natural forests and plantations.

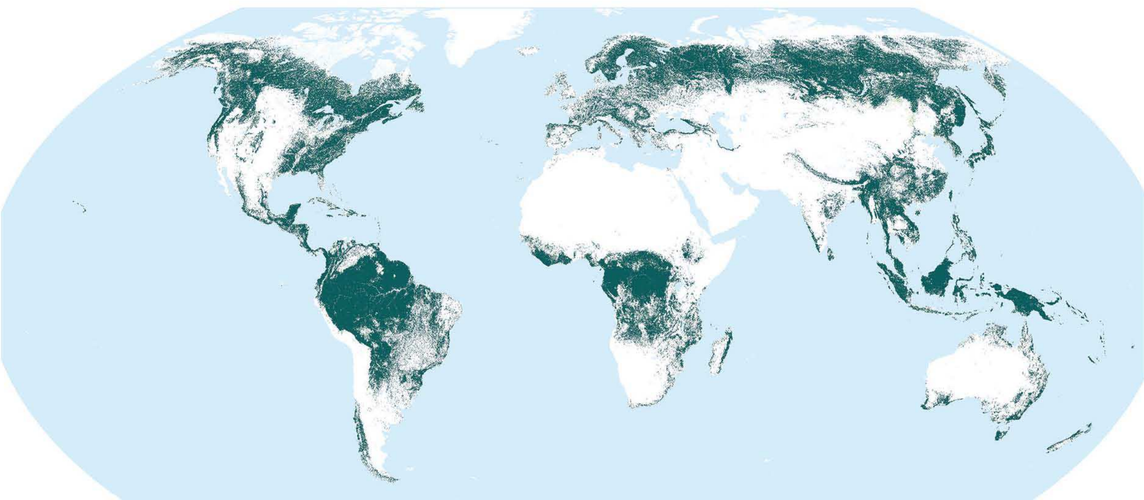

Fig. 2.1 Extent of global forested area (> 10 percent tree cover) from FAO (2010).

2.3 Hydrological processes within forests

The subject of the interaction between forests and water is plagued by myths, misinterpretations and too hasty generalisations (Andréssian, 2004; Chappell, 2005; Tognetti *et al*., 2005). This problem is a century old, with Pinchot (1905) noting "…it is unfortunate that so many of the writing & talking upon this branch of forestry has had little definite fact or trustworthy observation behind it. The friend & the enemies of the forest have both said more than they could prove…" (cited in Andréassian, 2004). Part of the misconceptions and debate about the interaction between forests and water globally is due to the ambiguous or even incorrect use of hydrological terms. It is therefore essential that the section on the evidence for the hydrological functions of forests is preceded by a *precise scientific definition* of the *hydrological pathways* underpinning the hydrological functions of forests. *Unless the hydrological pathways are defined correctly, accurately quantified and not confused, then the hydrological functions of forests are likely to be grossly misinterpreted*. Hydrological pathways are also called 'water-paths', 'runoff pathways' and 'streamflow generation pathways' when referring to the pathways of water penetrating the forest canopy to travel on or beneath the forest floor towards a stream channel. Within this synthesis, the hydrological pathways within the forest canopy (i.e., rainfall and snowfall reaching the forest canopy, cloud water interception, wet-canopy evaporation, throughfall, stemflow and transpiration) are discussed in addition to the runoff pathways.

The hydrological pathways that may be present within a forest environment are shown diagrammatically within Fig. 2.2. The hydrological pathways shown are: A = rainfall and/or snowfall, B = horizontal (occult) precipitation capture, C = wet-canopy evaporation (or interception loss), D = transpiration, E = throughfall and stemflow, F = infiltration-excess overland flow, G = infiltration, H = lateral subsurface flow in soil strata, I = lateral subsurface flow in regolith and/or rock, J = saturation overland flow (including recharge by return flow), and K = riverflow (or channel flow).

Rainfall and/or snowfall (Path A): Rainfall is defined here as precipitation in liquid state received in a raingauge located at the top of the canopy forest canopy (or a canopy gap) and with a funnel facing vertically (as separated from an occult precipitation gauge). Within this study, use of the term 'rainfall' without any qualifiers only refers to 'gross rainfall', i.e., the rainfall received above any vegetation canopies, and not to 'net rainfall', which is the rainfall received beneath vegetation canopies (i.e., throughfall and stemflow combined). Snowfall is the depth of precipitation collected using a snow pillow or by the melting of snowfall into funnel facing vertically. Clearly this depth may be different to that preferentially trapped by a forest canopy and is particularly important in boreal forests (Suzuki and Nakai, 2008).

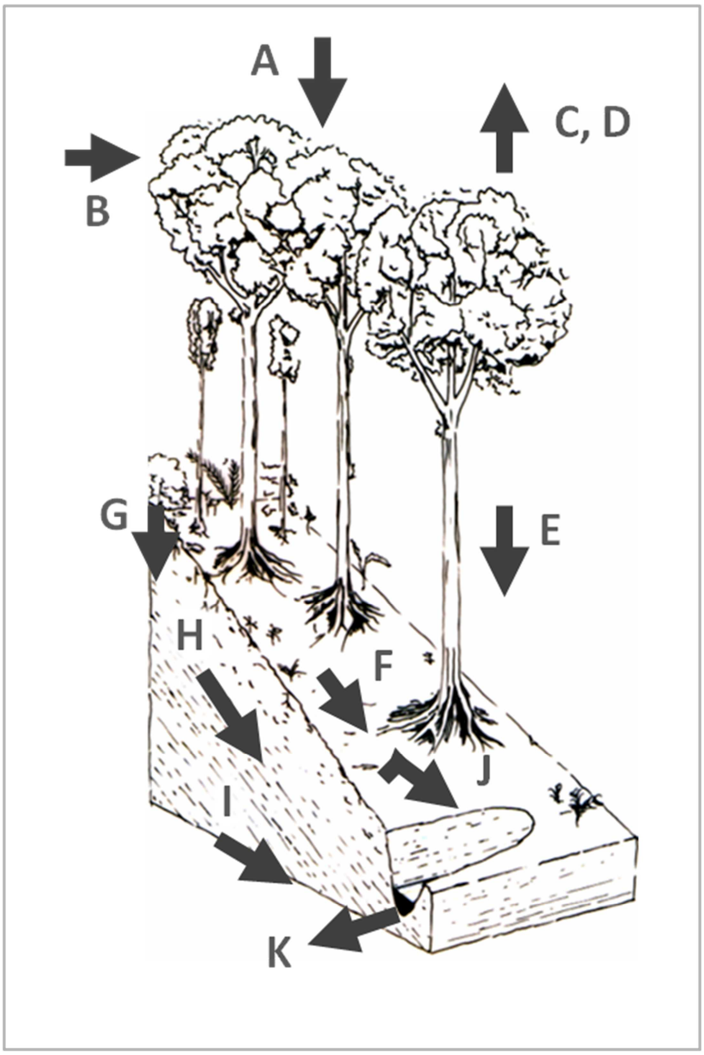

Fig. 2.2. Hydrological pathways are shown within a forested hillslope schematic, but present at scales from 0.1 km^2 experimental basins to international basins covering millions of square kilometres. Adapted by NA Chappell from the original diagram by Nick Scarle (with permission) published in Douglas (1977) Humid Landforms. MIT Press.

Horizontal (occult) precipitation capture (Path B): Horizontal or occult precipitation is that component of the precipitation measured using interceptor meshes that can capture occult precipitation (i.e., mist, fog etc: see Bruijnzeel *et al*., 2010).

Wet-canopy evaporation (Path C): Wet-canopy or wetted-canopy evaporation (Stewart, 1977) is the depth of water evaporated to the atmosphere from wetted parts of vegetation surfaces (i.e., leaves, branches and stems). This includes a forest canopy and a forest understory. Note that term wet-canopy evaporation is used in preference to 'interception loss' as 'interception loss' can be confused with 'interception', which means the water intercepted by a vegetation canopy, some of which will penetrate and reach the ground as throughfall and stemflow (see below), while some will leave the canopy as wet-canopy evaporation, and some stored temporarily on vegetation surfaces. Volumetrically, wet-canopy evaporation is most important in areas of low rainfall intensity, high rainfall totals, high wind run and forest canopies with a high leaf area index (Molchanov, 1960; Calder, 1990; Roberts *et al*., 2004).

Transpiration (Path D): Transpiration is the evaporation of water from within plant stomata into the atmosphere. This process is supported by water abstracted from the soil by plant roots and transported to the stomata within plant xylem.

Throughfall and stemflow (Path E): Throughfall is the component of the 'gross rainfall' (sometimes with some occult precipitation) that penetrates a vegetation canopy by either falling through gaps between branches and leaves or by hitting a branch or leaf before then falling to the ground. Stemflow is the component of the 'gross rainfall' that reaches a branch and then travels along to a plant stem on its way down to the ground surface. After integration of several days of data, the gross rainfall minus the net rainfall is equal to the wet-canopy evaporation.

Infiltration-excess overland flow (Path F): When the rainfall intensity (e.g., mm/15-mins or mm/hr) exceeds the saturated hydraulic conductivity of the ground surface (equal exactly to the infiltration capacity, and also in mm/15-mins or mm/hr) then infiltration-excess overland flow will occur either on (Horton, 1933) or laterally within the forest litter layer (Hewlett, 1982). This hydrological pathway has been considered by engineers (civil, agricultural and hydrological) for the last 80 years to be the dominant pathway of water to rivers during rainstorms. Experimental evidence collected over the same period (by experimental hydrologists, forest hydrologists, hillslope hydrologists and scientific hydrologists) does, however, show that this pathway is a <u>volumetrically insignificant component of the river hydrograph</u> (Hursh and Brater 1941; Dubreuil, 1985; Chappell *et al*., 2006), except for a few isolated locations. Simply, the saturated hydraulic conductivity of the ground surface beneath most vegetated surfaces (forest, grass or crops) is far in excess of the <u>dominant</u> rainfall intensity at most locations. The exceptions occur in areas of slowly permeable soils (e.g., FAO Gleysols, FAO Vertisols), particularly where they coincide with areas of very high rainfall intensity (e.g., areas beneath the tracks of tropical cyclones). Intense compaction of topsoil by vehicles (Ziegler *et al*., 2007) or livestock trampling (Bonell *et al*. 2010) can also locally reduce the infiltration capacity sufficiently to give locally significant volumes of overland flow. While the infiltration-excess overland flow pathway may not transport most of the water that reaches the most rivers, it is of fundamental importance to the transport of soil particles (and bound chemicals such as phosphorus or pesticides) during the process of soil erosion (see section 2.6 and 2.8).

Infiltration (Path G): The movement of water into the topsoil (or ground surface where soil development is absent) is defined as the infiltration (cf. Hewlett, 1982 definition of infiltration-excess overland flow). In most areas of the globe at most times, rainfall (gross or net) is able to infiltrate the topsoil.

Lateral subsurface flow in soil strata (Path H): Once water has entered the soil by infiltration, it may then percolate vertically into underlying strata of unconsolidated rock (e.g., saprolite, glacial till) or a solid rock aquifer (i.e., a rock with both a high saturated hydraulic conductivity and porosity), where either are present. Alternatively, all or a proportion of the percolation may be lateral (i.e., downslope) within the A and B soil horizons to emerge in a river channel (or prior to a channel via 'return flow': Cook, 1946).

Lateral subsurface flow in unconsolidated rock and/or solid rock (Path I): Where deep strata of unconsolidated rock are present (e.g., granite saprolite), and are between a permeable A and B soil horizon and a impermeable rock strata, then lateral flow towards a river can take place with this layer. If the solid rock has a high saturated hydraulic conductivity and porosity (a rock aquifer by definition) and lies beneath permeable overlying horizons, then the dominant lateral flow towards the river will be within the rock. These deeper hydrological pathways tend to have a slower response to rainfall in comparison to the shallower pathways in the A and B soil horizons. Lateral flows within unconsolidated rock and/or solid rock aquifers can be described as 'groundwater', though care is needed, as hydrogeologists use this term to describe only flow within the permanently saturated strata. The role of these deeper pathways in streamflow generation (Hursh and Brater, 1941), have been incorrectly ignored by many studies (Bonell, 2004).

Saturation overland flow (including recharge by return flow) (Path J): Where subsurface flow (within Path H and/or I) emerges from the ground prior to reaching a channel ('return flow') then it will flow over the surface as saturation overland flow. In these 'wetland' areas, overland flow may be present where the prevailing rainfall intensity is less than the local saturated hydraulic conductivity. Any rainfall falling onto these saturated topsoils with their upward return flow will not be able infiltrate, and so add to the volume of saturation overland flow travelling towards the nearest river channel. Because subsurface flows tend to converge on channels, the riverside (or 'riparian' or valley bottom) soils have a greater likelihood of generating saturation overland flow (Kirkby, 1976).

Riverflow (or channel flow) (Path K): Once water from overland and subsurface pathways (Paths F, H, I and J) enters a defined river channel it then becomes riverflow. This hydrological pathway is responsible for the transport of water, particles and solutes over long distances within landscapes whether covered by forest or other land-uses. Strictly, the term *runoff* is the river discharge per unit basin area (e.g., units of mm/hr), particularly within water budget and modelling studies. Use of this term is, however, avoided because of the ambiguity arising from its alternative use to described rapid overland (Paths F and J) and rapid subsurface pathways (Path H and sometimes Path I also).

2.4 Observed evidence for the hydrological functions of forests

Any review of the observed evidence for the hydrological functions of forests has to manage the huge wealth of literature on certain topics, in addition to managing the problem of myths and misinterpretations noted earlier. Some topics, notably the effects of forest on the available water resources in rivers ('annual water yield') have received much study, while the effects of forested areas (forestlands) on water quality (relative to that of other land uses) have received comparatively little study (Chappell *et al*., 2007). Given these issues, several guiding principles have been established to structure the review and synthesis of the findings from boreal, temperate and tropical environments.

2.4.1 Guiding principles for reviewing the observed evidence of the hydrological functions of forests

The synthesis attempts to identify all hydrology-mediated processes operating in natural forests (of boreal, temperate and tropical environments) and plantations. Given the political significance of carbon capture and retention globally, and its link to the hydrological functions of forests, this additional forest function will be incorporated within the overarching perceptual model or schematic diagram (Fig. 2.4) and in a separate discussion (Section 2.5).

All of the hydrological functions to be identified *must be capable of being linked unambiguously to specific hydrological pathways* (Section 2.3), and consideration of the relative importance of each hydrological function globally must be consistent with the relative importance of each hydrological pathway involved.

This *synthesis is explicitly based on rigorous observational evidence* (i.e., well-designed field studies where observed data and findings have been subject to peer review). Ideas, views, concepts and models are only discussed where they relate to the observed evidence, or its absence.

Much observed evidence of the hydrological function of forests has come from direct comparison of the behaviour of forested basins (in response to tree cutting or planting) with that of adjacent non-forest basins. These so called *paired-catchment studies* (or paired-watershed studies in the USA) have been and remain important to the study of forest-water interactions (Swank and Crossley, 1988; Webb *et al*., 2012).

Some reviews of the hydrological function of forests focused only a limited number of beneficial effects to ecosystem services, while others have focused on a similarly limited number of negative effects to ecosystem services (e.g., Hayward, 2005). It is important that a *balanced approach* presenting the *dominant mode of behaviour of all hydrological functions of forest*, whether resulting in positive or negative effects to ecosystem services, is given (Bruijnzeel, 1990; Chappell, 2006; Chappell *et al*., 2007). Notable exceptions (often location specific) to the dominant mode of behaviour should be presented, particularly given that the decision to designate a dominant mode of behaviour is partly subjective. Additionally, these exceptions (or atypical responses of a hydrological function) need to be known where new management options (Section 2.8) or policies (Section 2.9) are to be advocated.

To facilitate a balanced view of the observed evidence for the range of hydrological functions within forests, a ‘traffic light schematic’ is used to summarise the findings for each hydrological function. The schematic diagram (Fig. 2.3) shows: (1) whether the forest impact on a specific hydrological function is broadly positive (green circle) or negative (red circle) for the dominant ecosystem service delivered, (2) the strength of evidence of the observed impact globally (i.e., from one circle for very few rigorous studies to three circles for numerous rigorous studies in different global forests), and (3) the global extent of the impact. Where the forest impact on a function is specific to a small area of globe (e.g., cloud forests), then this is shown with small circles. Where the studies indicate that the impact is widespread across the globe, then this is shown with large circles (Fig. 2.3).

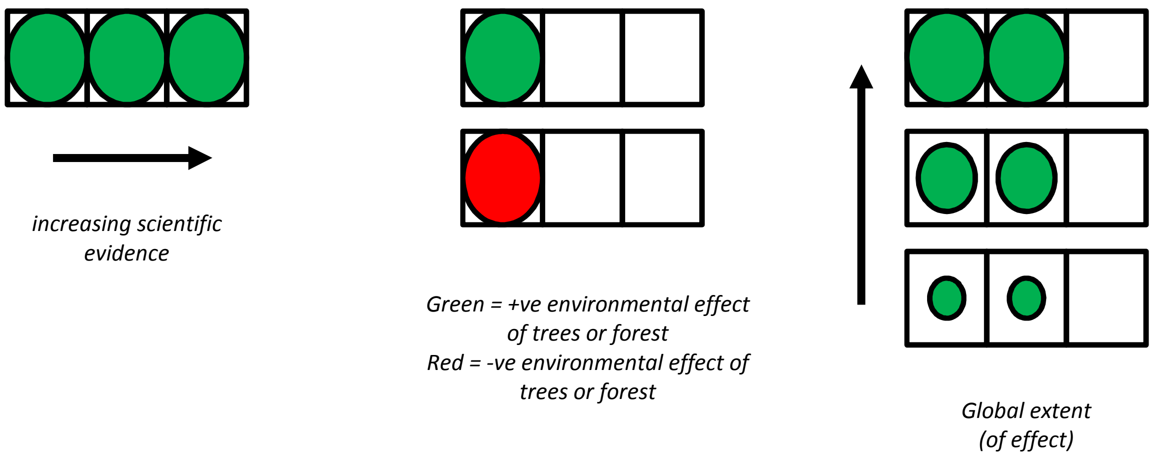

Fig. 2.3 A traffic light system for summarising the observed evidence for the effect of forests on hydrological functions

2.4.2 Summary schematic of the hydrological functions of forests

The schematic diagram illustrating the possible hydrological functions of forests (but not their magnitude, level of evidence or spatial extent) is given in Fig.2 .4. These functions are shown in a way that illustrates their linkage to the underlying hydrological pathways (Fig. 2.2). The direct hydrological functions shown are the: water availability via evaporation function (F1), precipitation capture function (F3), microclimate function (F4), enhanced infiltration (and reduced overland flow on slopes) function (F5), reduced slope erosion function (F6), exclusion of pollutant inputs function (F7), downslope utilisation of leached nutrients function (F8), downslope (and coastal) physical function (F9), peak-flow damping and low-flow enhancement function (F10), and enhancement of river water quality function (F11).

The carbon dioxide capture function (F2), and aquatic carbon source function (F12) is also shown to allow this schematic to illustrate the links to the hydrological pathways and functions, but is discussed separately in Section 2.5.

The observed evidence for each of the hydrological functions of forests will be discussed in the following sub-sections.

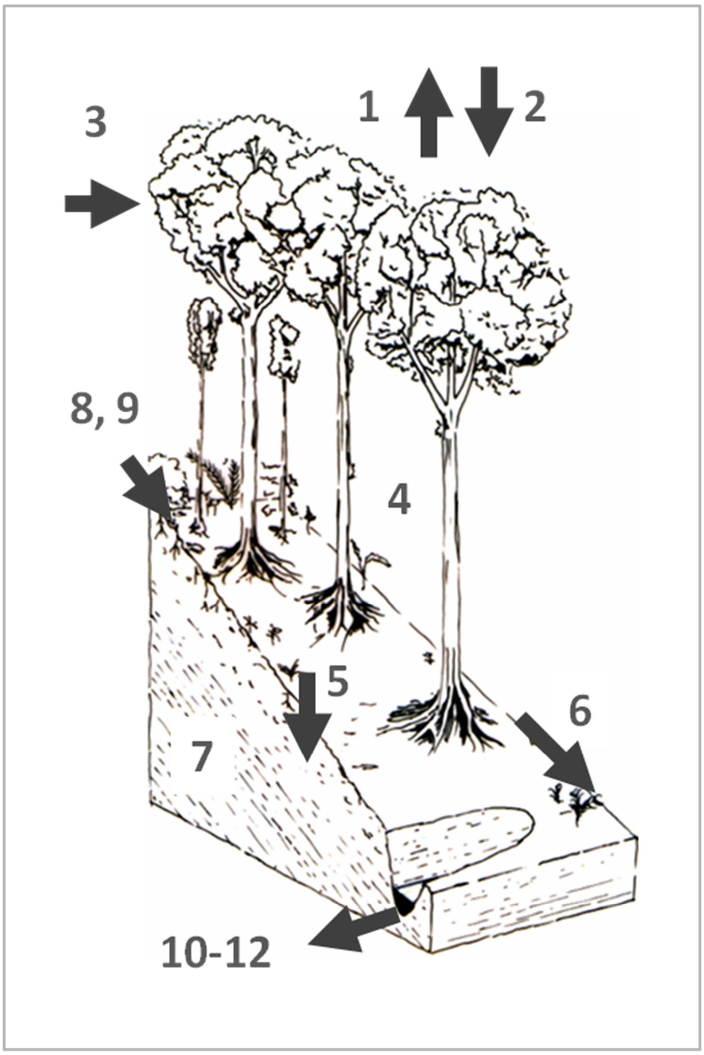

Fig. 2.4. Hydrological functions of forested areas, and include the important fluxes of gaseous, dissolved and particulate carbon that are transported by latent heat flux and riverflow. Each function is numbered where 1 = Water availability via evaporation function (-/+ve), 2 = Carbon dioxide capture function (+ve), 3 = Precipitation capture function (+ve), 4 = Microclimate function (+ve), 5 = Enhanced infiltration (& reduced overland flow on slopes) function (+ve), 6 = Reduced slope erosion function (+ve), 7 = Exclusion of pollutant inputs function (+ve), 8 = Downslope utilisation of leached nutrients function (+ve), 9 = Downslope (and coastal) physical function (+ve) 10 = Peak-flow damping and low-flow enhancement function (+ve), 11 = Enhancement of river water quality function (+ve), 12 = aquatic carbon source function (+ve).

2.4.3 Water availability via evaporation function

The presence of trees (rather than herbaceous vegetation, crops or bare ground) may affect the annual availability of water resources within deep groundwater (Path I) or rivers (Path K). The effect of trees on the provisioning ecosystem service of water supply (i.e., the water people abstract from the environment) is primarily via the evaporation function. Trees and forests affect the evaporation function via changes to the wet-canopy evaporation pathway (Path C) and/or transpiration pathway (Path D). The total evaporation is known by the American term ‘evapotranspiration’.

A huge number of paired-catchment studies have been used to examine the effects of removing trees from boreal, temperate and tropical forests and the effects of planting trees on the annual water yield of rivers. These studies have addressed forest clearance ('deforestation') and the selective logging practices characteristic of many tropical forests. Many studies show that natural forests and plantations have higher rates of evaporation than nearby herbaceous vegetation and so leave less water resources available in rivers. The higher evaporation relates to: (1) deeper tap roots that are able to continue to abstract water during dry seasons (Canadell *et al*., 1996), (2) a greater leaf area index, particularly with conifers, giving greater rates of wet-canopy evaporation (Calder, 1990), and (3) high growth rates and lower water use efficiency for young plantation trees (Vertessy *et al*., 2003). Rigorous reviews (e.g., Andréssian, 2004) of the available studies have however shown that the impact of the removal or addition of trees from the same catchment proportion gives very different changes in the annual river discharge per unit area (mm; Fig. 2.5). Some studies show large reductions of water yield in rivers, while others show only small or no changes.

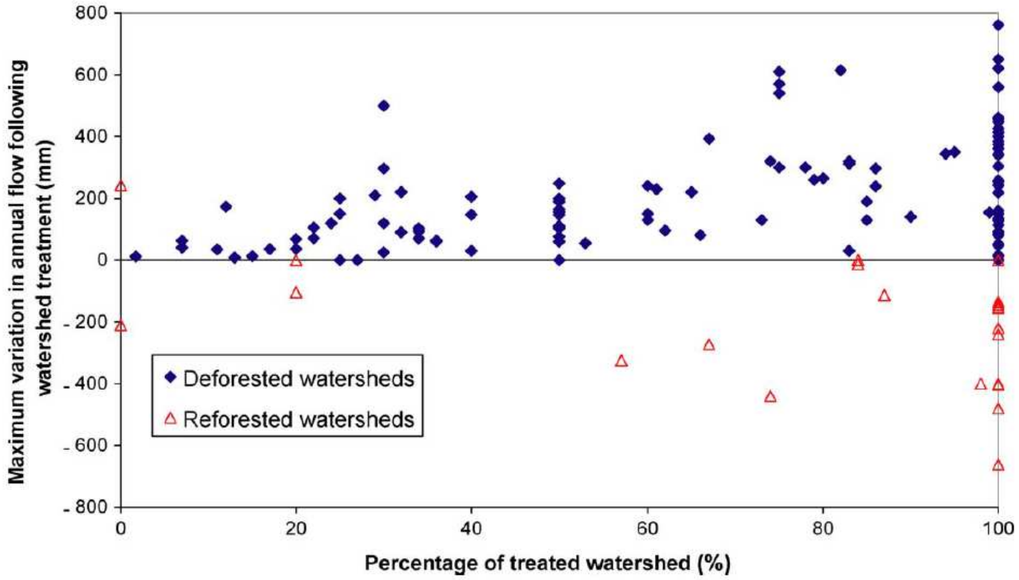

Fig. 2.5 Changes in the annual river discharge per unit area due to tree removal or planting, from Andréssian (2004 J. Hydrol. 291: 1-27)

There are indications that differences in 'tree type' affects evaporation. The review of Brown *et al*. (2005) that focused on forests in boreal and temperate locations, showed that conifers generally used (transpired) more water than hardwood trees (Fig. 2.6). They also noted that assessments were very sensitive to whether the 'peak change in water yield' or an 'average change over the duration of the study' was used.

Perhaps the clearest findings of the impact of trees on the evaporation function are from the study of Zhang *et al*. (2001). They reviewed 250 catchment-based, water balance studies from across the globe, including 35 from countries within the humid tropics. They demonstrated that the difference (mm) between water use by forests relative to that by grassland increases as the annual rainfall (mm) increases. Their model, fitted to the large number of data-sets with a correlation coefficient of 0.96, showed that forests typically have much greater evaporation rates than grasslands where rainfall exceeds 2000 mm/yr, but comparable rates where rainfall is less than 500 mm/yr (Fig. 2.7).

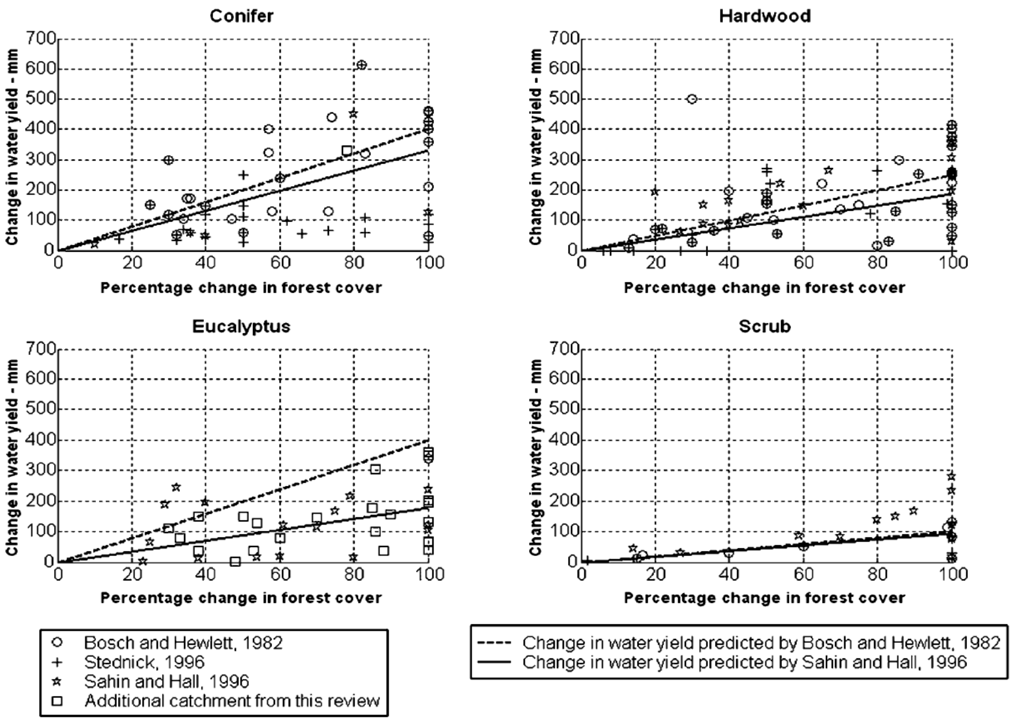

Fig. 2.6 Changes in the annual river discharge per unit area due to tree removal or planting from Brown *et al*. (2005 J. Hydrol. 310: 28-61)

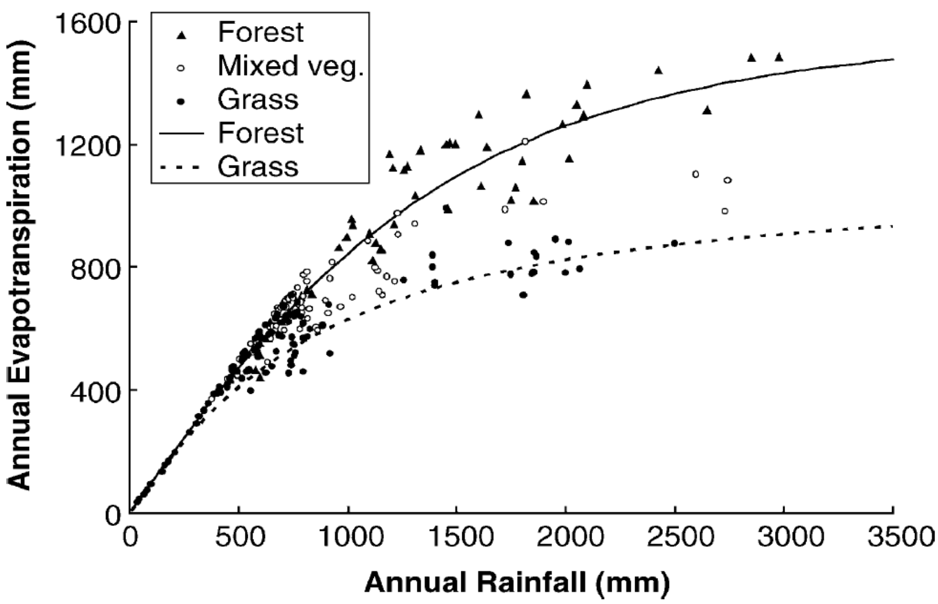

Fig. 2.7 Evaporation by forest and grassland basins (mm/yr) against annual rainfall from Zhang *et al* (2001 Water Resources Res 37: 701-708)

In contrast to the large number of studies that have examined forest impacts on annual river yield, very few observational studies have examined the impact on deep groundwater resources (Zhang and Schilling, 2006). There is also a lack of rigorous field studies that compare forest water use against that by irrigated crops and theoretically, irrigation in once forested areas may offset the effects of forest removal (see Ozdogen *et al*. 2010). This is particularly important in the seasonal tropics regions where rates of potential evaporation are very high.

As forest cutting may locally reduce the amount of water returned to the atmosphere, rainfall totals produced by local re-cycling may be affected. Extensive observational evidence for this effect is however lacking (Bruijnzeel, 2006). In part this is because a significant proportion if the rainfall over land is derived from ocean evaporation (Goessling and Reich, 2011) and partly because of the difficulty in attributing decadal changes in rainfall to land-cover change rather than the effects of natural climate dynamics (Chappell and Tych, 2012). The study of Lawton *et al*. (2001) does however, present observations to show that deforestation of lowland rainforest in Costa Rica has reduced local cloudiness. They then show by simulation, that this may affect rainfall (Path A) and horizontal precipitation (Path B) in downwind cloud forest. Many more observation-based studies such as Lawton *et al*. (2001) are however needed to show the true role of deforestation on local moisture recycling.

Given that a reduction in surface-water or groundwater resources for water supply abstractions is a negative impact on this provisioning ecosystem service (and is particularly important in the dry season: Avila, 2011), then most observational evidence indicates that the impact should be considered as negative for high rainfall regions (Fig. 2.8). Given the number of studies collated by the reviews, this observed evidence is considered to be well attested for such regions. There is also no reason to believe that this phenomenon does not apply across large areas of the globe (i.e., humid tropical or humid temperate environments). The studies conducted in relatively dry regions of the globe and reviewed by Zhang *et al*. (2001), do however show no or little difference in water use by forest versus grassland. This finding indicates a neutral impact of forests on water resources ('an orange traffic light': Fig. 2.8). By incorporating the potential benefits of forests to local moisture recycling may indicate that the overall impact of forests on water availability is closer to neutral, than many reviews would suggest.

A further exception to the impacts summarised in Fig. 2.8 is the localised impact of forests within riparian zones (Sections 2.2.4.9 and 2.2.4.10) of high rainfall areas. Here the greater evaporation resulting from the presence of forest might be seen as a positive impact, as greater soil drying can reduce the amount of saturation overland flow (Path J) and hence reduce the transport of chemicals (Sections 2.2.4.9) and soil particles (Sections 2.4.10) to rivers (Hernandez-Santana *et al*. 2011).

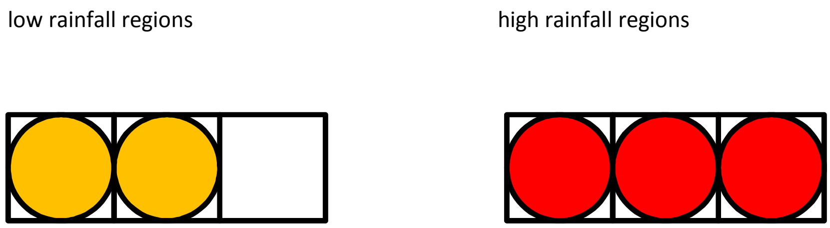

Fig. 2.8 Summary of the findings on *water availability via evaporation function* of forest (see Fig. 2.3 for key)

2.4.4 Precipitation capture function

Very high altitude areas are frequently in cloud. Forests within these areas are more efficient at capturing cloud water than low herbaceous vegetation. This means that forests are better at capturing water from within clouds. This process is now called *Cloud Water Interception* (CWI: Bruijnzeel *et al*. 2010), and formerly 'fog interception'. During CWI measurement, *Horizontal (wind-driven) Rainfall* is also captured (both give Path B of Section of 2.2.3). Locally, rates of CWI can be very high, for example Juvik and Nullet (1995) recorded throughfall beneath Tropical Montane Cloud Forest that was 120-180 percent of the open site (vertical) rainfall. The areas of Tropical Montane

Cloud Forest that are able to capture significant volumes of rainfall by this mechanism only occupy 215000 km^2 of the globe or 1.4 percent of tropical forest (Bruijnzeel *et al*. 2010).

Some boreal forests are in areas with a significant proportion of precipitation received as snowfall. In comparison to herbaceous vegetation, the higher canopies and leaf area index of forests can capture more horizontal and drifting snow.

Where the enhanced capture of precipitation by forested areas can be better utilised than if it fell elsewhere (e.g., at sea: Prada *et al*., 2010), then this function should be considered as a positive impact on the provisioning ecosystem service of water supply. The area of the globe where forests can enhance the capture of cloud water is however small (Fig. 2.9).

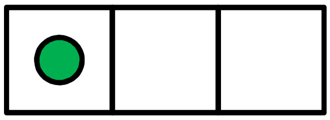

Fig. 2.9 Summary of the findings on the *precipitation capture function* of forest (see Fig. 2. 3 for key)

2.4.5 Microclimate function

Closed forests and open woodlands provide shade from direct solar radiation, shelter from rainfall and wind, plus regulate the humidity and temperature (soil and air) beneath their canopy (Gardiner *et al*., 2006; UK National Ecosystem Assessment, 2011). Most observational evidence comes from ecological studies along forest edges (e.g., Pinto *et al*., 2010; de Siqueira *et al*., 2004; Davies-Colley *et al*. 2000).

Tree shelterbelts (and forest edges) also affect the microclimate of adjacent land and can positively affect livestock production and the growth of crops through reduced evaporation (Delwaulle, 1977; Brenner, 1996) and increased soil moisture (Muthana *et al*. 1984; Ujah & Adeye, 1984). These changes positively affect the provisioning ecosystem service of food production.
The impact of trees on microclimate positively affect human comfort in villages and towns (Handley and Gill, 2009) and so the regulating (and associated cultural/recreational) ecosystem service of the mitigation of climate stress. Additionally, riparian trees regulate stream temperature (Studinski *et al*., 2012) and so enhance the supporting ecosystem service of aquatic habitat (see Section 2.4.10).
The findings from the observed evidence for the microclimate function of forest is positive and should be extensive, but have not been collated systematically (Fig. 2.10).

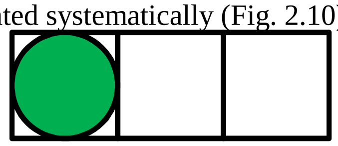

Fig. 2.10 Summary of the findings on the *microclimate function* of forest (see Fig. 2.3 for key)

2.4.6 Enhanced infiltration (& reduced overland flow on slopes) function

There is clear observational evidence that infiltration capacity of forest topsoil (equivalent to the saturated hydraulic conductivity of the topsoil: see Section 2.2.3) is typically greater than that of adjacent topsoil beneath grassland. This difference may be partly explained by the presence of a deeper litter layer, greater organic matter inputs to the topsoil and an absence of livestock trampling within most forests. Chandler and Chappell (2008) provide a table showing that the ratio of these two values is always larger than 1, and often considerably larger (Fig.2.11). More recently, Alvarenga *et al*. (2011) report a ratio of 5-15 for Cambisol soil beneath *Miconia sellowiana* trees relative to that beneath grassland.

With a higher infiltration capacity, there is an even greater likelihood that almost all net rainfall will infiltrate beneath forests, and minimise the production of infiltration-excess overland flow (Path F in Section 2.2.3). The greater infiltration will reduce the rate of soil erosion on slopes (having a mitigating impact on slope erosion: Section 2.4.7) and thereby enhance the regulating ecosystem services of reduced soil loss and enhanced water quality of rivers. These services have indirect impacts on provisioning services of food production and water supply, respectively.
If the presence of trees in a landscape with deep groundwater pathways (Path I; Section 2.3) can markedly reduce the proportion of the riverflow (Path K) that is generated by infiltration-excess overland flow (Path F), then the enhanced infiltration could add greater water resources to deeper groundwater reserves used for water supply or the slower hydrological pathways that maintain seasonal rivers during low-rainfall seasons (see the low-flow enhancement function in Section 2.4.10). This function would enhance the provisioning service of water supply. Few rigorous studies have addressed the water resource significance of the infiltration function, and new studies are needed, particularly within seasonally dry areas (Bruijnzeel 2004).

F/G	Soil type[a]	Tree type[b]	Reference
2.0	Luvisol	*Eucalyptus* spp.	Lorimer and Douglas (1995)
2.5	nk	*Eucalyptus* spp.	Burch et al. (1987)
3.4[c]	Gleysol	*Quercus robur*	This study
4.8	nk	*Pinus insularis*	Costalles (1979)
5.2	nk	*Pinus halepensis*	Berglund et al. (1981)
4.5–7.2	Cambisol	*Quercus robur*	Burt et al. (1983)
2.3–12	Ferralsol	*Eucalyptus/Gravillea* spp.	Wood (1977)
14	Nitisol	*Hibiscus elatus*	Ternan et al. (1987)
20	Andosol	Podocarp	Jackson (1973)
23–41	nk[d]	*Quercus* spp.	Molchanov (1960)
50[e]	Ultisol	*Quercus* spp.	Hoover (1949)
17–140	Cambisol	*Eucalyptus* spp.	Wood (1977)

F/G = ratio of the topsoil saturated hydraulic conductivity under trees to that under pasture (ranked by magnitude). (nk) not known.
[a] FAO-UNESCO classification.
[b] Dominant or representative tree species.
[c] At 3 m from Tree No. 1.
[d] Reported as 'dark grey soils'.
[e] 0.1 m depth.

Fig. 2.11 Ratio of the saturated hydraulic conductivity of the topsoil (A soil horizon) under forest compared to that under grassland for 12 studies reviewed by Chandler and Chappell (2008 For. Ecol. Manage. 256: 1222-1229; see this paper for the references cited therein).

Where soil infiltration capacity is very high beneath a pasture or cropland, so that virtually no infiltration-excess overland flow is produced, then the addition of trees may have a measureable impact on the infiltration capacity, but no measureable impact on the infiltration-excess overland flow (Gilmour *et al*., 1987). Equally, it should be noted that the beneficial effects of forests on infiltration are not as great, where there are is a high livestock density and hence marked soil trampling and compaction within forests (Bonell *et al*., 2010).
Despite these exceptions, there is ample observational evidence that the infiltration function of forests is clear, positive and extensive (Fig.2. 12).

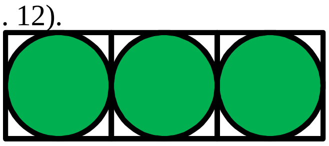

Fig. 2.12 Summary of the findings on the *infiltration function* of forest (see Fig.2.3 for key)

2.4.7 Reduced slope erosion function

There are numerous studies showing that undisturbed natural forest has a lower rate of slope erosion in comparison to cropland disturbed (tilled) on a regular basis. A recent study that demonstrates this reduced slope erosion function of forests is Liu *et al*. (2005) who used plot-scale measurements in Sichuan, China.

As a consequence, these forests maintain river water quality, notably lower turbidity and lower levels of pesticides and faecal coliforms that are transported with the soil particles (see Sections 2.4.8, 2.2.4.9 and 2.2.4.11). Elevated erosion also leads to accelerated losses of particulate carbon from slopes to rivers (Sections 2.5.2).

Studies have also reported localised reductions in slope erosion as a direct result of tree planting and growth via the beneficial impact on infiltration and soil stabilisation. One such study is that of Vasquez-Menandez *et al*. (2010) conducted in semi-arid Mexico.

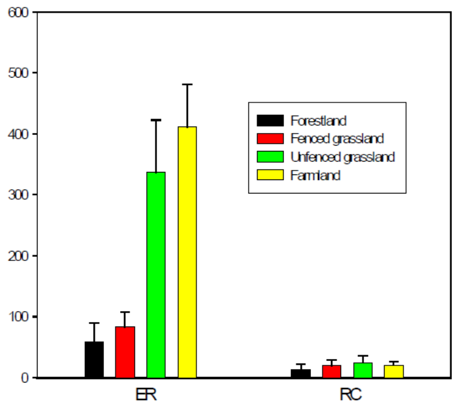

Fig. 2.13 Example of differences in slope erosion (ER) between undisturbed forest and cropland (labelled as 'farmland') by Liu *et al*. (2005 J. Mt. Sci-Engl: 2: 68-75). The runoff coefficient (RC) is also shown.

Where plantation development is accompanied by soil disturbances associated with artificial drainage, then the effects of forests in soil erosion may be negative. For example, artificial drainage prior to the planting of conifers in temperate Wales accelerated the rate of erosion within the studied Hafren and Tanllwyth basins in comparison to the pasture control – the Cyff basin (Fig. 2.14).

Catchment	Area (km^2)	Bed-load yield ($t\ km^{-2}\ y^{-1}$)	Suspended load ($t\ km^{-2}\ y^{-1}$)	Land use
Hore	3.08	11.8	24.4	Mature forest – first year of felling operations
Hafren	3.67	NA	35.3	Mature forest
Tanllwyth	0.89	38.4	12.1	Mature forest
Cyff	3.13	6.4	6.1	Pasture

NA = not available

Fig. 2.14 Bedload and suspended load resulting from erosion at the Plynlimon catchments, upland Wales (from Kirby *et al*. 1991 IoH Report 109).

Logging of forests (including the associated road construction in previously undisturbed natural forests) significantly accelerates erosion. Even selective harvesting of tropical natural forests gives increases in suspended sediment load that are between 4.3 and 52 fold larger than adjacent undisturbed forest basins (Chappell *et al*., 2004). During these harvesting periods the rates of erosion may be larger than those from nearby pastureland, though there is a dearth of such comparative studies.

The key message is that forests not subject to timber harvesting operations or artificial drainage have lower erosion rates than land covers subject to regular disturbance (e.g., tillage or livestock trampling), that has a beneficial impact on the regulating ecosystem services of reduced soil loss and enhanced water quality of rivers (Fig. 2.15). These services have indirect impacts on provisioning services of food production and water supply, respectively (as noted in Section 2.4.6). This beneficial hydrological function relates to the effect of: (1) greater soil surface protection, (2) enhanced infiltration (Path G) and (3) reduced infiltration-excess and saturation overland flow (Paths F and J), all via greater root development and litter-fall. If, however, the forest is subject to soil disturbances due to artificial drainage or harvesting, the presence of forests may have a negative impact on the soil erosion function (Fig. 2.15). The key issue may be the presence or absence of soil disturbance rather than trees.

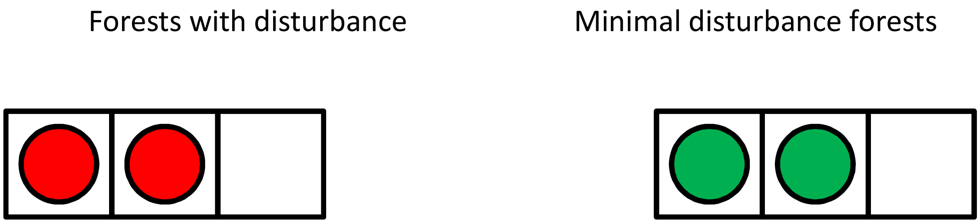

Fig. 2.15 Summary of the findings on the *reduced slope erosion function* of forest (see Fig. 2. 3 for key)

2.4.8 Exclusion of pollutant inputs function

Large global areas of cropland, grassland or urban development have large inputs of artificial chemicals (e.g., nitrates, phosphorus, pesticides) and/or artificial biochemicals (e.g., faecal coliforms associated with livestock or population centres), while these inputs tend to be absent from most areas of natural forests, and many plantations (Evans, 1982 p377; Chappell , 2005). This function clearly relates to the management practices within forest lands, rather than to the effects of individual trees (see Section 2.1). Liu *et al*. (2010) have shown that 136.60 trillion grams of nitrogen is added to the world's cropland each year; almost half as mineral nitrogen fertilizers. They also demonstrate that two fifths of this nitrogen is 'lost in ecosystems' – see Fig. 2.16 (i.e., stored or transported along overland, subsurface or river pathways: Paths F, H, I, J and K in Fig. 2. 2). Similarly, Macdonald *et al*. (2011) demonstrate that the agronomic input of phosphorus (P) in fertilizer amounts to 14.2 Tg P / yr globally, and a further 9.6 Tg P / yr is added as manure. They show that only 12.3 Tg P / yr are removed in crops, leaving a major imbalance and hence storage or transport along overland, subsurface or river pathways: Paths F, H, I, J and K in Fig. 2.2. Microbial contamination of water resources (e.g., faecal coliforms or cryptosporidium) is also an issue within non-forest areas. For example, Bolstad and Swank (1997) demonstrated how low levels of faecal streptococcus within the Coweeta forested catchment (USA) increased downstream, as urban development increased.
The absence or exclusion of large inputs of artificial chemicals or microbial contaminations within most natural forests means that groundwater (Path I) and river-water (Path K) draining from these forests *dilutes* the effects of contaminated drainage from the other land-uses. This exclusion of pollutant inputs function of forests consequently has a large positive impact on the provisioning ecosystem service of the supply of clean water suitable for abstraction and subsequent treatment for drinking water. It also has a positive impact on the regulating services of purifying soils and waters

and hence reducing human risk from contaminated waters, and the supporting service of providing river-water capable of sustaining life and biodiversity.

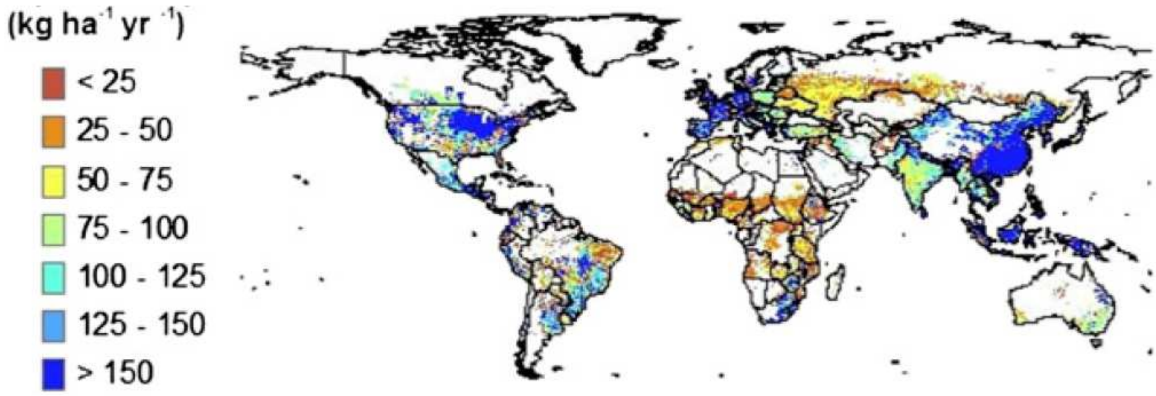

Fig. 2.16 Nitrogen outputs from cropland globally (Liu *et al*. 2010 PNAS 107: 8035-8040).

There are exceptions. Tree-crop plantations, e.g., oil palm, have high inputs of fertiliser and pesticides (Halimah *et al*., 2010). Similarly some commercial forests in the USA and elsewhere are treated with pesticides. Some agro-forestry systems in the tropics e.g., India, have high livestock densities and hence risks to water resources from microbial contamination. Indeed, many of the conifer plantations within the catchments of water supply reservoirs in the UK were added to exclude the risk of microbial contamination that was perceived to exist with the former land-use of grassland supporting cattle.

Given that the exclusion of pollutant inputs function applies to most natural forests (and these dominate globally: Section 2.2), its positive impact should be considered extensive globally (Fig. 2.17). The lack of rigorous studies that illustrate the effects on water quality of low-input forests versus high-input land-uses does however need to be highlighted. The observation that some forests, notably tree-crop plantations can have high chemical inputs also needs to be highlighted, even though they may be much less extensive (Fig. 2.17).

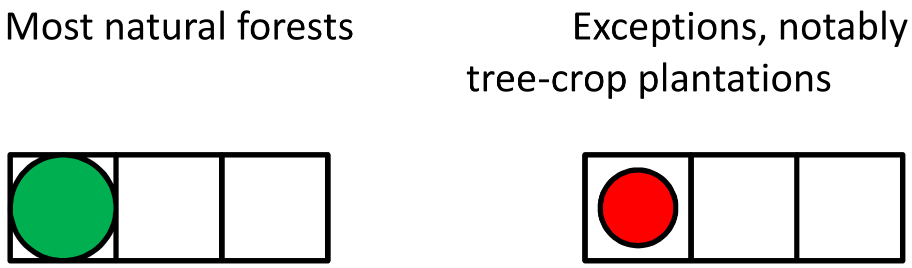

Fig. 2.17 Summary of the findings on the *exclusion of pollutant inputs function* of forest (see Fig. 2.3 for key)

2.4.9 Downslope utilisation of leached nutrients function

There has been considerable research into the value of trees in riverside or flood-plain areas (often called 'riparian areas') in mitigating water quality issues resulting from other land-uses upslope. Given the appropriate moisture regime and supply of carbon, trees within downslope areas can utilise nutrients leaching from upslope areas via overland or subsurface pathways (Paths F, H, I and J: Fig. 2. 3). High inputs to downslope areas often result because of the high artificial fertiliser inputs to cropland or improved pasture in the upslope areas. The widely cited study of Peterjohn and Correll (1984) undertaken in Maryland (USA) clearly demonstrates how downslope forest can remove dissolved nitrogen from overland flow (Paths F and/or J: that they called 'runoff': Fig. 2.18) and subsurface water (Paths H and/or I: that they called 'groundwater': Fig. 2. 18) draining from cropland.

Riparian forest has also been used successfully to reduce nitrate levels in contaminated rivers by diverting some of the riverflow into riparian forest via irrigation channels, to then return via drainage channels (Gumiero *et al*. 2011). Additionally, the negative effects on river nitrate loads of forestry drainage and logging operations with commercial forests have been reduced by drain blocking within riparian forest (Hynninen *et al*., 2011) or preventing harvesting of riparian forest (Clinton, 2011).
Given the importance of carbon to denitrification and to food webs, the enhanced litter-fall under riparian forest compared to other vegetation covers has been shown to be a beneficial function (Newham *et al*. 2011).
Critically, the effectiveness of riparian forest in the function of chemical removal from surface and subsurface waters is site specific, being dependent on: (1) the local biogeochemical conditions e.g., carbon availability and (2) the local hydrological conditions e.g., soil moisture content and hydrological pathways (Burt *et al*., 2010). Consequently, this hydrological function can have neutral or positive impacts on the same provisioning, regulating and supporting ecosystem services as the exclusion of pollutant inputs function (Section 2.4.8). Systematic global analysis of the extent of the conditions conducive to the effectiveness of this riparian forest function is however needed (Fig. 2.19).

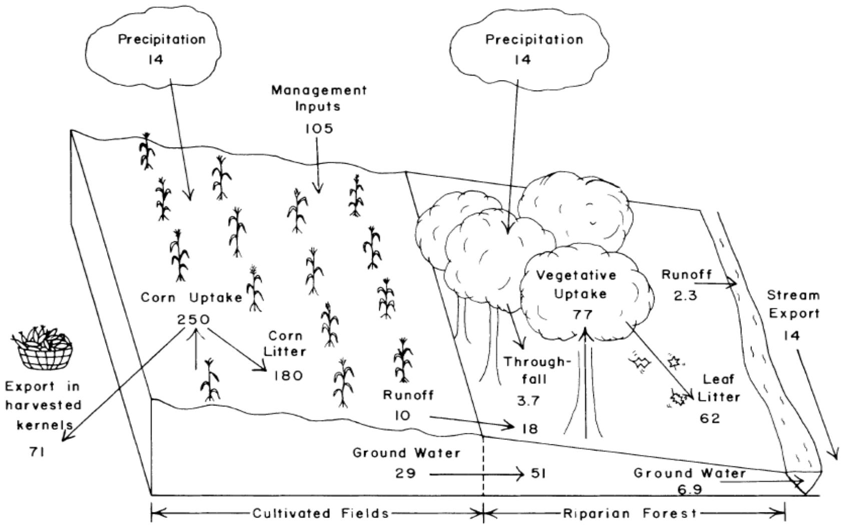

Fig. 2. 18 Total nitrogen flux from cropland to a river via a downslope forest (Peterjohn and Correll (1984 Ecology 65: 1466-1475).

Fig. 2. 19 Summary of the findings on the *downslope utilisation of leached nutrients function* of forest (see Fig. 2.3 for key)

2.4.10 Downslope (and coastal) physical function

Riparian forest strips (sometimes known as ‘buffers’) have been used to trap sediments and any sediment-bound chemicals (e.g., phosphorous or pesticides) being transported in overland flow (Path F or J) from upslope cropland. Peterjohn and Correll (1984) cited within in the last section is a good example of where this can be effective. A more recent example is that of Santos and Sparovek (2011) who demonstrated the value of riparian forests in trapping sediment from upslope cotton farming at a site in central Brazil. In another recent example from Brazil, Bicalho *et al*. (2010) demonstrated that herbicides (i.e., Diuron, Haxazinone and Tebuthiuron) applied to sugar cane crops could be trapped by riparian forest. This water-quality related function has similar water-quality related provisioning, regulating and supporting ecosystem services as the previous downslope function (Section 2.4.9).

The presence of riparian trees can regulate the thermal regime of rivers (see Section 2.4.5). This function affects the regulating service of water quality, and the supporting services of aquatic habitat and biodiversity.

Additionally, closed forests and open woodland within river flood plains are known to reduce the speed of flood flows travelling across flood plains more than lower herbaceous vegetation (Straatsma and Baptist, 2008). This effect reduces the return of over-bank flows back to rivers (thereby mitigating downstream peak flows; Section 2.4.10), and also enhances flood plain infiltration (Section 2.4.6). A similar effect is afforded by mangrove forests that can better attenuate inland flooding by seawater in comparison to lower herbaceous vegetation (Gedan *et al*., 2011). These flood-related physical functions have the specific regulatory ecosystem service of mitigating flood hazard further downstream or further inland, respectively.

As with the *downslope utilisation function* the effectiveness of the sediment trapping function (known as the 'trap efficiency'), is seen to be site dependent whether beneath forest or other land-covers (Ziegler *et al*., 2006). This also probably applies to the flood attenuation potential of forests. Similarly, no systematic global analysis of the extent of the conditions conducive to the water or sediment trap efficiency of riparian forests has been undertaken, though it is known that *riparian and coastal-mangrove wetlands* (see Acreman – this volume) do cover relatively large areas of the globe (Fig. 2.19).

Fig. 2.20 Summary of the findings on the *downslope physical function* of forest (see Fig. 2.3 for key)

2.4.11 Peak-flow damping and low-flow enhancement function

The greater rates of wet-canopy evaporation (Path C: Fig. 2.3) from forests compared to those from herbaceous vegetation occur during rainy periods (Calder, 1990; Kume *et al*., 2008) and so should have a direct damping impact on river peak-flows in forest-covered basins. This effect is however moderated by the observation that catchment-average rates of wet-canopy evaporation (mm/hr) are typically much smaller e.g., one fifth of those of riverflow per unit area (mm/hr). Annual transpiration rates are often comparable to those for annual riverflow per unit area and so may have a larger impact on peak-flow if affected by a change of vegetation. Transpiration losses from catchment systems are however distributed over much longer periods than wet-canopy evaporation (Kume *et al*., 2008), so this may partially negate the beneficial effect on peak-flows inferred from greater long-term rates. The greater evaporation from forests may have an additional indirect impact on peak-flows. Greater evaporation will dry the soil more, and because of the inherent nonlinearities in catchment response (Young and Beven, 1994), this can have a disproportionately large mitigating effect on the rates of lateral subsurface flow in soil strata (Path H: Fig. 2.3) and so reduce peak-flows.

As noted in Section 2.3, infiltration-excess overland flow (Path F) does not produce more than a few percent of the riverflow in most vegetated areas (Dubreuil, 1985). Consequently, an enhancement of the infiltration capacity (Path G) through the planting of trees (Section 2.3), cannot remove any more than the few percent of the riverflow generated by infiltration-excess overland flow, and so cannot significantly effect on the peak-flows in rivers for most areas (see e.g., Chappell *et al*., 2006). Only in localised areas of very slowly permeable topsoil (e.g., FAO Gleysol, FAO Vertisol) that coincide with areas dominated by intense rainfall (e.g., areas below the tracks of tropical cyclones or extreme rainfall events in other areas of the globe), might the effect of trees on infiltration capacity affect river flows. Clear observational evidence of the effect of forests in these localised areas (Zimmerman *et al*., 2012) or during extreme events (e.g., 1 in 100 year rainstorms) is however lacking for most areas with humid climates.

Given these complex interactions, changes in peak-flow as a result of the presence of forests may be best examined by studying their integrated effects on peak-flow following a land-cover manipulation of forest cutting or planting. Guillemette *et al*. (2005) reviewed the impact of forest cutting in forests of boreal and temperate climates. They showed that 74 out of 75 studies showed either no change or an increase in peak flow with forest cutting. Most studies showed a 0 to 100 percent increase in peak-flow and a further four studies up to 170 percent increases (Fig. 2.21).

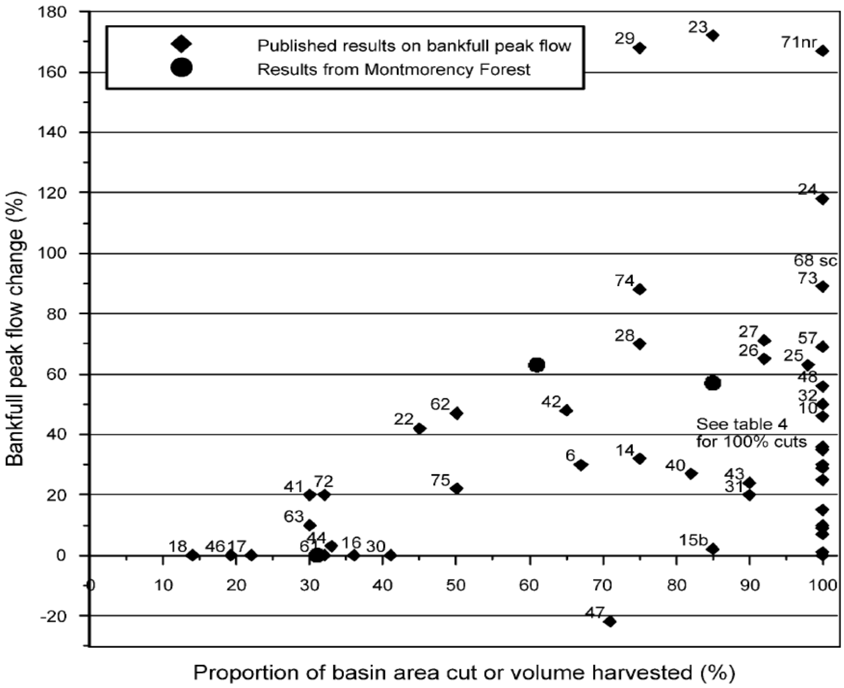

Fig. 2.20 A review of changes in river peak-flow following forest cutting in boreal and temperate regions by Guillemette *et al*. (2005 J. Hydrol. 302: 137-153).

Most studies from tropical climates similarly increases in peak-flow with forest cutting or reductions in peak-flow with forest planting (e.g., Fig. 2.21).

The notable exceptions to this general trend arise where the preparation of wetland sites for plantations involves cutting surface drainage channels, which can add new rapid pathways that can increase peak-flow (e.g., Fig. 2.22).

The dominance of a beneficial (i.e., reducing) effect of forests on peak-flows means that this function should be considered beneficial to the regulating ecosystem service of flood mitigation. However, the present inability to explain the wide range in the mitigation effect means that more work to strengthen the observational evidence is needed (Fig. 2.23). Moreover, all of the findings relate to small basins and should not be extrapolated to the behaviour of large rivers where the effects of channel routing dominate, where trees have reduced ability to mitigate channel velocities.

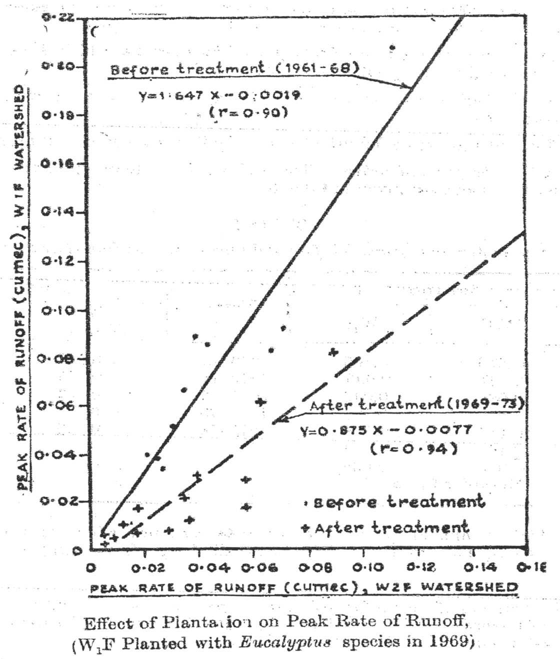

Fig. 2.21 Decreases in river peak-flow following tree planting shown by Mathur *et al*. (1976 Indian Forester 102: 219-226) in tropical India.

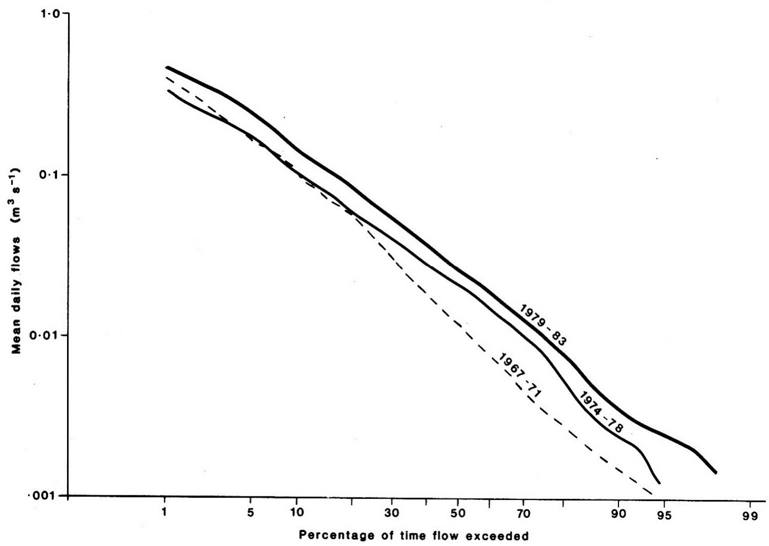

Fig. 2.22 Increases in river flows occurring for only 10 percent of the time (Q10) following the addition of forestry drainage channels to an upland wetland. The broken line is the 'flow duration curve' prior to drainage, and the solid lines the 'flow duration curves' for periods post drainage (Robinson *et al*. 1998 IoH Report 133).

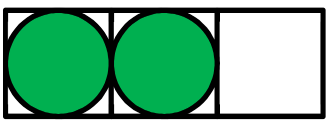

Fig. 2.23 Summary of the findings on the *peak-flow damping function* of forest (see Fig. 2.3 for key)

The observed evidence for the beneficial effects of forests on river low-flows does not match the perception of local farmers within the tropics (Pereira, 1959). Indeed, the review of comparative basin studies within the tropics by Bruijnzeel (1990) showed that forests are more likely to reduce river low-flows and thereby have a negative impact on the provisioning ecosystem service of water supply.

This negative effect could be attributed to the greater rates of transpiration from forest when compared with cleared land.

There may be circumstances where forests can enhance river low-flows. In areas of high rainfall intensity coincident with slowly permeable topsoils (e.g. FAO Gleysols, FAO Vertisols) a significant proportion (e.g., 50 percent) of the riverflow may be produced by infiltration-excess overland flow (Path F). If a significant proportion of the infiltration-excess overland flow can be diverted into the deep subsurface (Path I) via improvements to infiltration and easy vertical drainage thereafter, then river low-flows might be increased. However, to observed increases in low-flows following tree planting, the change in evaporation (mm/yr) must be a smaller than the change in infiltration-excess overland flow (mm/yr). This is the so called 'infiltration trade-off hypothesis', and clear evidence to support this hypothesis has not yet been collected (Bruijnzeel, 2004). Consequently, robust evidence for the low-flow enhancement effect of forest is not yet available (Fig. 2.24), and further research needed to support this function.

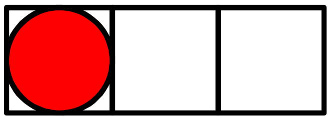

Fig. 2.24 Summary of the findings on the *low-flow enhancement function* of forest (see Fig. 2.3 for key)

2.4.12 Enhancement of river water quality function

Within forests, the *reduced slope erosion function* (Section 2.4.7), the *exclusion of pollutant inputs function* (Section 2.4.8), the *downslope utilisation of leached nutrients function* (Section 2.4.9) and the *downslope physical function* (Section 2.4.10) all mean that natural forests should enhance river water quality. A global assessment of the water quality of rivers only draining natural forest versus those draining cropland, improved pasture or urban areas has yet to be published. Some studies are however available that show the low nutrient contamination (from agricultural fertilisers: Section 2.4.8) of the largely forested headwaters of the Amazon basin relative to other rivers (Figueiredo *et al*., 2010). There are localised exceptions to these findings – certain tree-crop plantations, notably oil palm, have high chemical inputs that may leach (via Paths F, H, I and J) in to rivers (Halimah *et al*., 2010).

Agricultural productivity of croplands is sometimes supported by irrigation with alluvium-rich river-water. Where this process increases the rates of deposition of alluvium on the upstream flood plain, then the natural rates of sedimentation (that includes nutrients) on downstream river deltas may be reduced (in the same way that large dams reduced natural loads of alluvium). Where a forest cover discourages or excludes such irrigation activities it will enhance the provisioning service of downstream fisheries and the supporting services for deltaic habitat maintenance and associated biodiversity.

Overall the expected better water quality of rivers within natural forestlands, particularly due to the *exclusion of pollutant inputs function*, should benefit the provisioning service of clean river-water available for water supply abstractions, but more robust global data are needed to underpin specific policy recommendations (Fig. 2.25). Additionally, maintaining a natural nutrient cycle is supporting ecosystem service i.e., one that is essential for aquatic life.

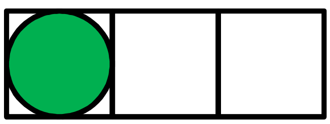

Fig. 2.25 Summary of the findings on the *enhancement of river water quality function* of forest (see Fig. 2.3 for key)

2.5 Related issues: carbon and water cycle interactions

The need to quantify the ability of different types of biome to capture, retain or lose carbon is major global issue (Yuan *et al*., 2009). These 'carbon pathways' are closely associated with the hydrological pathways. Whether a land-cover is capturing more carbon dioxide (i.e., downward flux of CO_2) or returning it to the atmosphere (i.e., upward flux) is measured directly from the direction of the vertical wind eddies and the associated concentration in the atmosphere, as is evaporation (Paths C and D). This balance is also affected by the moisture status in the soil (i.e., water within Path H; Cabral *et al*., 2011). The loss of carbon from soils into rivers, where it is then lost to atmosphere as CO_2 (Richley *et al*. 2002) or oceans in dissolved and particulate forms (Neu *et al*., 2011), is dependent on the surface and subsurface hydrological pathways (Paths F, H, I, J and K of Section 2.3).

2.5.1 Carbon dioxide capture function

Current evidence demonstrates that for the same temperate latitude, undisturbed forests capture more CO_2 than does grassland (Fig. 2.26; Valentini, 2007). Forests therefore contribute to the regulating environmental service of better carbon sequestration. However, some boreal deciduous forests and some temperate conifer forests have a net ecosystem exchange that shows they are losing more CO_2 than they are accumulating (Fig. 2.26). Undisturbed tropical forests tend to be accumulating CO_2, though not when they are disturbed and drained (Hirata *et al*. 2008).

2.5.2 Aquatic carbon source function

Very little observed data are available that can illustrate the differential effects of forest, grassland or crop land-covers on the release of dissolved and particulate carbon to rivers, particularly in tropical environments. Richey *et al*. (2002) controversially suggested that CO_2 degassing from rivers in Amazon basin could amount to 1.2 ± 0.3 Mg C / ha / yr, which is equivalent to the CO_2 losses from the forest canopy. As forest disturbance accelerates the loss of carbon to rivers (Schelker *et al*., 2012), the regulating environmental service of better carbon sequestration may apply to undisturbed natural forests. As the carbon naturally added to rivers provides food for the aquatic biota (Nystrom *et al*. 2003), then the maintenance of natural forests, particularly in riparian zones, helps maintain aquatic biodiversity, thereby providing a supporting ecosystem service.

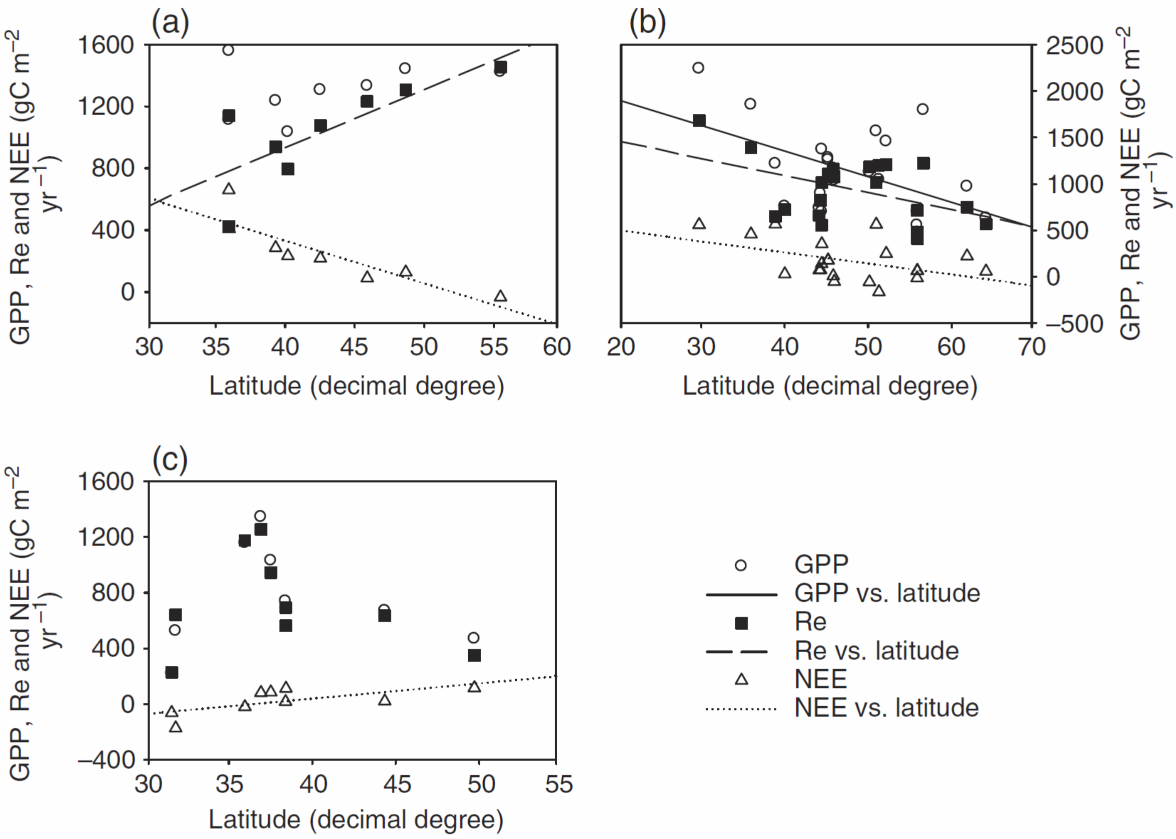

Fig. 2.26 The relationship between latitude and the accumulation of CO_2 (shown with a positive net ecosystem exchange, NEE) for: (a) deciduous forest, (b) conifer forest and (c) grassland biomes (from Yuan *et al*. 2009. Glob. Change Biol. 15: 2905-2920).

2.5.3 Hydrological co-benefits of enhancing the carbon function of forests

Of equal importance when considering carbon and water cycle interactions are the *co-benefits* to hydrological functions that should come from new schemes (e.g. United Nations 'Reducing Emissions from Deforestation and Forest Degradation' or REDD) to retain carbon within the landscape (Section 2.7). If the loss of forest-cover is reduced by REDD or forestry management enhanced under REDD+, then co-benefits to water-related ecosystem services should be produced (Strickler *et al*., 2009). However, scientific investigation is needed to quantify these hydrological co-benefits of carbon sequestration (Section 2.5), not a return to the myths or misinterpretations of the forest-water interactions (Malmer *et al*., 2010).

2.6 Economic values of water-related forest ecosystem services

Ecosystem services are humankind benefits that are supplied by natural ecosystems or natural capital (Kareiva *et al*., 2011). Those services specifically related to water are sometimes called *watershed services* (Stanton *et al*., 2010) or *water services* (Perrot-Maître and Davies, 2001). The assessment of observational evidence quantifying the hydrological functions of forests was presented in Section 2.5. This assessment shows that forests deliver a range of watershed or water-related ecosystem services. Six of the ten hydrological functions did however relate to the value of forests for delivering better quality water within rivers (Sections 2.4.6, 2.4.7, 2.4.8,.2.4.9, 2.4.10 and 2.4.12). This has direct and indirect impacts on the *ability of rivers (and any associated water supply reservoirs) to provide resources clean enough for water supply abstractions* (Shaw *et al*. 2010) Consequently, the regulating service of water quality (Millennium Ecosystem Assessment, 2005) is a fundamental control on the provisioning ecosystem service of water supply, and so the hydrological function of water quantity

(Section 2.4.3) should not be considered in isolation from the hydrological functions related to water quality (see above). This assessment of the hydrological functions of forests has also highlighted that not all management systems within the globe's forests are beneficial to ecosystem services. Forestry (within natural forests or plantations) that involves: (1) drainage, (2) the application of fertilisers and/or pesticides, or (3) intensive logging has a negative impact on various water quality related functions (see above) and peak-flows (Sections 2.4.10). The absence of drainage and chemical additions plus the need for reduced impact forms of timber harvesting (Section 2.7) may be required for the ecosystem service benefits to be seen.

Case studies are available that demonstrate reforestation and improved forest management can so improve the water quality of rivers that the benefits to water supply economics outweigh the costs of the ecosystem service schemes. A good example comes from the temperate forests of the north-eastern USA. In order to improve the river water quality in the Catskill and Delaware catchments for water supply abstractions, the City of New York invested $1.0 to 1.5 billion in improved forest (and agricultural) management, including reforestation. This was financed by a 9 percent tax increase to water bills. Their only alternative was to construct a new raw water treatment plant that would have required a two fold increase in water bills (Perrot-Maître and Davies, 2001; Stanton *et al*., 2010).

To evaluate water treatment costs, the Source Water Protection Committee of the American Water Works Association conducted a survey in 2002 of approximately 40 water suppliers (Fig. 2.27).

% of Watershed Forested	Treatment and Chemical Costs *per mil gal*	% Change in Costs	Average Treatment Costs *per day at 22 mil gal*
10%	$115	19%	$2,530
20%	$93	20%	$2,046
30%	$73	21%	$1,606
40%	$58	21%	$1,276
50%	$46	21%	$1,012
60%	$37	19%	$814

Fig. 2.27 Water treatment (including chemical) costs based on percent of forested water supply catchment (Ernst *et al*., 2004).

Their survey results indicated that for every 10 percent increase in forest cover in the water supply catchments (up to about 60 percent forest cover), treatment costs decreased approximately 20 percent. They also found that 50-55 percent of the variation in the treatment costs could be explained by the percent forest cover in the water supply catchments (Ernst *et al*., 2004). The reasons for the beneficial effect of forests were not explained, though the role of the exclusion of pollutant inputs function of forestlands (Section 2.4.8) must be a significant factor.

Payments for Ecosystem Services (PES) are payments or exchange of credits between a buyer and seller to effect some improvement in the ecosystem service. There is a large potential for these payments to deliver water quality improvements given the current market value of water quality in the global environmental market (Fig. 2.28).

Environmental Market	Market Value (2008)
Regulated Carbon	$117,600,000,000
Water Quality	$9,250,000,000
Biodiversity	$2,900,000,000
Voluntary Carbon	$705,000,000
Forest Carbon	$37,100,000

Sources: World Bank. "State and Trends of the Carbon Markets: 2010." Ecosystem Marketplace Reports: "Building Bridges: State of the Voluntary Carbon Markets 2010" and "State of Biodiversity Markets: Offset and Compensation Programs Worldwide".

Fig. 2.28 Market value of environmental markets in 2008 (Stanton *et al*., 2010).

The markets for PES including *Water Quality Trading* (where water quality regulated organisations purchase and trade in offset credits to meet their obligations) are already established across the globe and continue to grow (Fig. 2.29).

	Programs Identified	Active Programs	Transactions 2008 (US$ Million)	Hectares Protected 2008 (million ha)	Historical Transactions through 2008 (US$ Million)	Hectares Protected Historically
Latin America	101	36	31	2.3	177.6	NA
Asia	33	9	1.8	0.1	91	0.2
China*	47	47	7,800	270	40,800	270
Europe	5	1	NA	NA	30	0.03
Africa	20	10	62.7	0.2	570	0.4
United States	10	10	1,350	16.4	8,355	2,970
Total PWS	216	113	9,245	289	50,048	3,240
Water Quality Trading	72	14	10.8	NA	52	NA
Totals	**288**	**127**	**9,256**	**289**	**50,100**	**3,240**

* Note: We separate China from the rest of Asia given the level of activity.

Fig. 2.29 Summary of PES transaction data for 2008 and historically (Stanton *et al*., 2010).

What is currently missing from PES analysis is a systematic economic valuation of each hydrological function of forestlands. Without this it is difficult to accumulate the financial benefits for a specific ecosystem service (e.g., provision of water supply) from the multiple hydrological functions. Equally, it is difficult to estimate the *trade-offs* between the beneficial and negative hydrological functions of forests at a particular location. Research on the economic valuation of each hydrological function of forests is needed.

2.7 Global and regional policies

A central aspect of the Convention on Wetlands ('Ramsar Convention') is the 'conservation and wise use of wetlands', under whatever land-cover, including forests. Given that riverine, lacustrine and palustrine ('bogs') wetlands typically receive their water from a much larger catchment area, then the land-cover on the surrounding catchment is also of fundamental importance to wetland conservation and management.

To achieve this mission, Ramsar recognise that better quantification of the ecosystem services delivered by wetlands is needed (Strategy 1.4ii of the *Ramsar Strategic Plan 2009-2015*) and underpinned by a robust understanding of the science, e.g., hydrological processes and pathways (Strategy 1.6; Section 2.3). Better scientific and financial evidence for wetland services should deliver greater cross-sectoral recognition of the significance of wetlands in decision-making. Quantification of the hydrological functions of forests (whether in or upslope of wetland areas) and the resultant assessment and valuation of ecosystem services delivered, is just one land-cover type associated with wetlands, and needs to be considered with the agricultural, grassland and urban land-covers. There is also an appreciation that many different hydrological functions affect a particular wetland and it may have many different users. Hence there is an appreciation the different functions and user needs must be assessed and managed together within an *Integrated Water Resources Management* approach (Strategy 1.7).

Ramsar is now working more closely with the Convention on Biological Diversity (CBD) to deliver its goals (Strategy 3.1). At the heart of CBD's Strategic Plan are 20 targets to be met by 2020, collectively known as the *Aichi Biodiversity Targets*. Several of these targets directly relate to both wetlands and forests (CBD, 2012). Notably:

Target 11: At least 17 per cent of terrestrial and inland water areas are conserved, and

Target 14: Ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded.

These strategies and targets are compatible with the desire of national and international forestry organisations to improve forestry management to increase the delivery of water-related ecosystem services. Most national forestry departments now have guidelines that seek to minimise the impact of forestry operations and improve the water service delivery. For example, management of the temperate forests (mostly plantations) in the United Kingdom is guided by the 'Forests and Water Guidelines' (Forestry Commission, 2003). Equally, management of the tropical natural forests in the State of Sabah (Malaysia) is guided by the 'RIL Operation Guide Book' (Sabah Forestry Department, 1998).

In forest blocks where forestry operations are subject to independent certification, guidelines can be replaced by very specific environmental criteria that <u>must</u> be met if a certificate of sustainable forest management is to be awarded and maintained thereafter. These certificates are particularly important within regions of tropical natural forests as some of the largest land-cover changes are taking place in this biome (Drigo, 2004), and some of the largest negative hydrological impacts seen (Chappell *et al*., 2004). Very few rigorous studies have attempted to quantify the effect of tropical forest certification on water services. Thang and Chappell (2004) have however shown that one example of such rules (i.e., the Malaysian Criteria and Indicators for forest management certification, or 'MC&I') are at least compatible with the delivery of water services. Forestry management outside of those forest blocks that are closely scrutinised by independent assessors, needs to adopt at least some strict rules (not simply 'guidelines') that are then shown to deliver water services locally (Chappell and Thang, 2007).

Some countries, notably India and China, are following policies of rapid and extensive reforestation with the aim of delivering water-related ecosystem services (Ravindranath and Murthy, 2010). The delivery of such services needs an equal level of scientific investigation and scrutiny by independent assessors, if others (e.g., FAO, 2005; Hayward, 2005; Calder and Aylward, 2006) are not to challenge the stated water services being delivered. Such an objective would be compatible with Strategy 1.6 of the *Ramsar Strategic Plan 2009-2015*, as noted earlier.

2.8 Management options

The management options to enhance the delivery of beneficial ecosystem services via changes to the hydrological functions of forests will be site specific, depending on 'forest type', local hydrological conditions and end-user requirements. Many wetlands or catchments have multiple land-covers (forest, cropland, grassland, urban), so the cumulative and net effects of the ecosystem services for each land-cover need to be considered together, thereby paralleling the policy approaches of *Integrated Water Resource Management* (IWRM).

2.9 Policy recommendations for future activities

The policy recommendations jointly to Ramsar and CBD resulting from this assessment of the observed evidence for the *hydrological functions of forestlands* (and the subsequent impact on water-related ecosystem services) are as follows:

1/ All assessments of the water-related ecosystem services from local to international scales will need to be based on sound hydrological science (notably the *dominant hydrological pathways*), and a thorough evidence-based (observational) understanding of the impacts of land-cover and associated management on the *hydrological functions*. This view is compatible with Strategy 1.6 of the *Ramsar Strategic Plan 2009-2015.*

2/ Assessment of the effect of forests (and associated management) on water-related ecosystem services will need to be *balanced*, namely it will need to cover both the physical (e.g., water quantity, river peak-flow, sediment trapping) and water quality related (e.g., exclusion of chemical inputs, soil conservation, nitrate utilisation) functions at local to international scales.

3/ The effect of forests on hydrological functions will need to quantify the effect of the different forms of forestry management, including plantation-related drainage, agro-forestry impacts of livestock or chemical additions, and timber harvesting impacts, where present.

4/ A systematic and in depth global review of hydrological functions of forests related to *water quality effects* will need to be undertaken to provide a clearer evidence base for ecosystem service valuation, management and to convince policy makers of the need to value the water services provided by forests.

5/ An equally rigorous assessment of the impact of forestry certification criteria and forestry management guidelines on hydrological functions will be needed.

6/ Some highly targeted experimental studies (with new observations) will be needed to quantify those hydrological functions of forests with a poor evidence base (sometimes linked to existing forestry management guidelines or rules), and the potential to have a significant global impact, and

7/ A systematic financial assessment of the impact of each forest hydrological function on the value of the ecosystem services delivered, will need to be undertaken; this would provide clearer financial evidence to convince policy makers of the need to value the water services provided by forests.

With a stronger evidence-base, policy makers may be more willing to make the financial investments necessary to deliver greater water services within forest-rich environments.

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CONFERENCE OF THE PARTIES TO THE
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Hyderabad, India, 8-19 October 2012
Item 13.3 of the provisional agenda*

REPORT OF THE WORK OF THE EXPERT GROUP ON MAINTAINING THE ABILITY OF BIODIVERSITY TO CONTINUE TO SUPPORT THE WATER CYCLE

Note by the Executive Secretary

1. The tenth meeting of the Conference of the Parties to the Convention on Biological Diversity, in decision X/28 paragraph 39, recognized the good synergies between the Convention on Biological Diversity and the Ramsar Convention on Wetlands and requested the Executive Secretary, and invited the Secretariat and Scientific and Technical Review Panel (STRP) of the Ramsar Convention, and other relevant partners, subject to the availability of financial resources, to establish an expert working group, building upon the relevant core expertise of the STRP, to review available information, and provide key policy relevant messages, on maintaining the ability of biodiversity to continue to support the water cycle. Progress with the work of the expert group was reported to the fifteenth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA). SBSTTA recommendation XV/5, section II subparagraph (b), requested the Executive Secretary of the Convention on Biological Diversity to make the report of the expert group available for the information of the eleventh meeting of the Conference of the Parties. Consequently, the Executive Secretary is hereby making the report of the expert group available.

2. This document is circulated in the form and languages in which it was received by the Secretariat of the Convention on Biological Diversity.

3. The summary report of the expert group is provided for the consideration of the eleventh meeting of the Conference of the Parties to the Convention on Biological Diversity as document UNEP/CBD/COP/11/30.

*UNEP/CBD/COP/11/1.

Report of the expert group on maintaining the ability of biodiversity to continue to support the water cycle

September 2012

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Contributors to the work of the expert group are acknowledged in Table 1.1 of this report.

TABLE OF CONTENTS

EXECUTIVE SUMMARY

For the current purpose, and that of the eleventh meeting of the Conference of the Parties to the Convention on Biological Diversity, the Executive Summary of this report is document UNEP/CBD/COP/11/30 (http://www.cbd.int/doc/?meeting=COP-11).

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