Tropical forests in multi-functional landscapes

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Preface

Over the last decades, tropical forest landscapes have experienced drastic changes. Forest conversion, fragmentation and exploitation have created new types of landscapes in many tropical forest areas throughout the World. These "multi-functional" landscapes consist of mosaics of undisturbed, exploited, degraded and secondary forests combined with agro-forests and non-forest areas under agriculture. Human pressure in these landscapes is typically high.

In December 2002 and April 2003, the Prince Bernhard Centre of Utrecht University and the Dutch Association of Tropical Foresters organised two seminars in Utrecht to shed some light on the role of tropical forests in multi-functional landscapes. The two seminars included presentations on biodiversity conservation, carbon sequestration and hydrological functions of undisturbed and exploited tropical forests in the context of multi-functional landscapes (see Appendix for full list of presentations).

Although we have not succeeded in including papers for all presentations held at the two seminars, we believe that the 7 papers included in these proceedings provide an overview of the most important issues related to the potential services of tropical forests in multi-functional landscapes. The proceedings include papers on the value of multi-functional landscapes for biodiversity conservation (Zuidema & Sayer; Wiersum), for hydrological functions (Van Dam) and for carbon sequestration (Karjalainen *et al.*; Trines). Furthermore, the economic valuation of forest services is discussed (Van Beukering) as well as the options to deliver payments for these services to local forest users (Verweij).

We hope that these papers contribute to discussions on approaches to biodiversity conservation in tropical forests and on the potential of forest environmental services for their conservation. All papers are written in a non-technical style, thus making them also accessible to readers from outside the academic world.

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Tropical forests in multi-functional landscapes: the need for new approaches to conservation and research

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Changing landscapes in tropical forest regions

Tropical forest conservation emerged as a global environmental concern in the 1970s. At that time the main objective of the conservation movement was to establish very extensive undisturbed forests as protected areas. However in recent decades the landscapes of tropical forest countries have changed radically. Burgeoning populations and rapacious industries have opened-up large areas of forest. Forest clearance for agriculture and degradation caused by unsustainable logging, fires and grazing have combined to create landscapes consisting of mosaics of different types of forest and non-forest land cover. In many parts of the tropics we now have a landscape composed of patches forest that are little disturbed, forests allocated for timber production, forests managed by communities and a complex range of secondary and agro-forests all set in a matrix of pastures, croplands and plantations. Human pressure in these tropical forest landscapes continues to increase rapidly. The recent World Bank Forest Policy has argued that hundreds of millions of poor people in the tropics are dependent to a greater or lesser extent on natural tropical forests (World Bank 2003). Many of these people depend on severely modified, degraded or exploited forests for a living or as a subsistence 'safety net' (Scherr *et al.* 2003). Thus, tropical forests have been transformed from large, remote wilderness areas into mosaic landscapes in which agricultural, agroforestry and forestry land uses are mixed. Both conservation and development objectives have to be met from these landscape mosaics. The challenge facing researchers and managers is to reconcile the numerous and often conflicting demands upon the components of these mosaics in order to produce an optimal mix of goods and services. The narrow sector-based approach to management of our traditional natural resource institutions needs to change so that we can better understand and manage the flows and linkages that occur amongst and between the components of mosaics. We can no longer focus on just managing each cell in the matrix for a single product or service. Instead, we have to ensure that production and protection functions are optimised at the level of the landscape. This is the challenge of managing *multi-functional landscapes* (Sayer *et al.* 2003).

We have to be realistic about how much of the tropical landscape can be allocated to strictly protected areas. Only about 10% of tropical forest is currently given total protection and the quality of much of this protection is very poor (Brandon *et al.* 1998). As a consequence, the conservation of tropical forest biodiversity depends to a large extent on the >90% of tropical forest area outside parks and reserves (WWF 2002b). Much of that forest is exploited, degraded and embedded in multi-functional landscapes. Similarly, the environmental services provided by tropical forests to local communities (*e.g.* erosion control, water supply) and to the global community (*e.g.* carbon sink) will also increasingly depend on forests outside set-asides.

In short, the conservation of tropical forests including their biodiversity and their environmental services depends increasingly on the conservation of remaining forest areas

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in multi-functional landscapes. As a consequence, the management of the functions of tropical forests has to be considered in this context. Two central questions arise from this changed setting:

(1) What do unprotected forests in such multi-functional landscapes contribute to biodiversity conservation and other environmental services? And, how can this contribution be optimised? In this paper, we tackle these questions by providing an overview of the current knowledge and pointing at knowledge gaps.

(2) How can tropical forests in multi-functional landscapes be conserved? What is their economic value, and how could these values be transferred to local communities? This question is addressed by reviewing the successes and shortcomings of conservation practices so far, and deriving recommendations for shifts in approach of conservation activities.

The analysis presented here shows that the changed setting of tropical forests in multi-functional landscapes requires a large-scale approach to conservation (*cf.* WWF, 2002a). Furthermore, we argue that the economic, social and ecological interactions between various users, stakeholders and forest elements in the landscape involve increasing complexity in conservation-targeted research and management (*cf.* Sayer & Campbell 2003).

What do forests in multi-functional landscapes contribute to biodiversity conservation and other environmental services?

- Biodiversity conservation

A very general overview of current knowledge on biodiversity conservation in the different types of tropical forests that make up multi-functional landscapes is presented in Table 1. The Table does not directly answer the question raised in the title of this section. Rather it provides an overview of our present knowledge on the potential conservation of biodiversity in *elements* of these multi-functional landscapes, as most research has analysed units within the landscape instead of the entire landscape. What becomes clear from the Table is that there are many gaps in our knowledge, many of which are concerned with interactions within multi-functional landscapes or processes at the landscape level. Below the contents of Table 1 are briefly discussed.

Various studies and compilations have shown the consequences of (selective) logging on forest biodiversity, both the effects of changed environmental conditions on plant life (Ter Steege 2003); and those of disturbance and hunting on animal life (Fimbel *et al.* 2001). The negative effects are relatively small for tree diversity whereas they are variable and sometimes substantial for vertebrate diversity. Much depends on the type of logging intervention, with reduced impact logging (RIL) generally causing the least negative impact (Putz *et al.* 2001). Over the longer term, responses are less clear, although model simulations tend to show that tree diversity is maintained reasonably well (Ter Steege 2003, Putz *et al.* 2000, 2001; counter arguments by Bawa & Seidler 1998).

Recent literature shows that tropical forests are highly resilient to the effects of anthropogenic or natural disturbances, both from controlled logging and from complete forest removal for slash-and-burn agriculture or from tropical storms (Chazdon 1998; Ter Steege 2003). Much depends on the setting of disturbed forests in the larger landscape, and the proximity of other forest areas. In secondary forests, biodiversity recovers rapidly, although this also depends on the proximity to intact (primary) forest. Bird inventories have indicated that agro-forests may harbour substantial levels of diversity (Thiollay 1995). Even older tree plantations have been shown to have a significant biodiversity value

(Lamb 1998). What is lacking, though, are more insights into the landscape-scale retention of biodiversity in entire multi-functional landscapes that contain both forests with different use-intensities and non-forest areas.

Table 1. An overview of current knowledge and gaps in knowledge on the conservation of biodiversity in elements of multi-functional landscapes in tropical forest areas. Underlined are those topics that relate to interactions between components of multi-functional landscapes. Multi-functional landscapes are here defined as mosaic landscapes containing forest patches and experiencing strong human influence.

	Potential for biodiversity conservation	
Forest elements in tropical multi-functional landscapes	What we do know	What we don't know
Logged (or production) forests	Retention of (tree) diversity in selectively-logged forests[1]	Long-term retention of biodiversity[3]
	Diverse responses of vertebrates to logging[2]	
Degraded or secondary forests	Biodiversity recovery and high value in secondary forests[4]	Their role in conserving forest biodiversity at landscape-scale[5]
Agro-forests	Their high biodiversity value[6]	Their role in conserving forest biodiversity at landscape-scale[7]
Plantations	Their sometimes high biodiversity value[8]	Their role in conserving forest biodiversity at landscape-scale[9]
Forest fragments	Biodiversity loss/retention in (small) fragments[10]	Potential of (larger) fragments for biodiversity conservation[11]
		Their role in conserving forest biodiversity at landscape-scale[12]
Networks with corridors	The role of corridors in facilitating movement of (mainly large) vertebrates[13]	The role of corridors for flora and micro-fauna dispersal[14]
		The additional value of corridors in landscapes with (agro-)forests

response of biodiversity to the management of such landscapes. Furthermore, we lack knowledge on the factors that are important in determining this potential. This is strongly related to the relative extent, spatial distribution and connectivity of forest-elements in the landscape. To what extent secondary, degraded or agro-forests and plantations can serve as corridors between intact forest patches also remains unclear (Buck *et al.* 2003). However, it is clear that these landscape elements potentially may do so.

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[1] For selectively logged tropical forests: Putz *et al.* 2000, 2001; Ter Steege 2003
[2] Fimbel *et al.* 2001
[3] Sheil *et al.* 1999; Putz *et al.* 2000, 2001; Ter Steege *et al.* 2003
[4] Kammescheidt 2002; Kennard 2002; Wiersum 2003
[5] Kammescheidt *et al.* 2002: Kohler *et al.* 2003; Wiersum 2003: Vandermeer & Carvajal 2001
[6] E.g. Thiollay 1995; Garcia-Fernandez *et al.* 2003
[7] Vandermeer & Carvajal 2001
[8] Lamb 1998; and case studies in e.g. Perfecto *et al.* 2003; Raman & Sukumar 2002
[9] Wethered & Lawes 2003; Vandermeer & Carvajal 2001
[10] For forest fragments of 1-100 ha loss of biodiversity has been well-documented (see Zuidema *et al.* 1997; Laurance & Bierregaard 1997)
[11] For forest fragments >1000 ha in size, there is little information on biodiversity retention (Zuidema *et al.* 1997)
[12] Crome 1997; Bierregaard & Stouffer 1997; Martinez-Garza & Howe 2003; Melbourne *et al.* 2004
[13] Laurance & Laurance 1999; Haddad *et al.* 2003 (non-tropical forests)
[14] But see: Tewksbury *et al.* 2002

In spite of the widespread belief that forest fragments have little value for biodiversity conservation, this remains an untested hypothesis - one out of several that seem to persist in the conservation world. Forest fragmentation research has merely focused on small fragments, of 1-10 ha in size, for which the loss of diversity is rather evident (Zuidema *et al.* 1997). Extrapolating these results to larger fragments is impossible, and consequently little information is available on the potential for larger fragments (>1000 ha) to conserve significant biodiversity. More importantly, the total potential of landscapes containing forest patches for biodiversity conservation has not been evaluated (Crome 1997), although the landscape- or regional-level conservation potential in sets of fragments is clearly what counts for biodiversity conservation (WWF 2002a). Interactions between forest elements in networks with corridors have been advocated as a means of maintaining populations of animals and plants. Particular emphasis has been given to corridors that connect protected areas. Apart for a number of (larger) vertebrate species (*e.g.* Laurance & Laurance 1999), the effectiveness of corridors has not been demonstrated for most other (smaller) animals and plants (Simberloff *et al.*1992; but see Haddad *et al.* 2002). Also, the additional effect of corridors in landscapes with plantations, degraded forests and agro-forests has not been quantitatively assessed (Tewksbury *et al.* 2002).

- Other environmental services

Apart from biodiversity conservation, the most important other environmental services of tropical forests include carbon sequestration and soil and water conservation. Table 2 provides an overview of current knowledge and suggests some knowledge gaps. As in Table 1, the issues underlined relate to landscape-level processes or interactions at landscape-scale.

Table 2. An overview of current knowledge and gaps in knowledge on the environmental services of tropical forests. Underlined are those topics that relate to interactions between components of multi-functional landscapes. Multi-functional landscapes are here defined as mosaic landscapes containing forest patches and experiencing strong human influence.

Potential of environmental services for forest conservation:	What we do know	What we don't know
Carbon sequestration	Tropical forests are net carbon sinks[1] and their conversion causes large carbon emissions[2]	Carbon sequestration in production forests[3] To what extent the Clean Development Mechanism may contribute to forest conservation[4]
Soil and water conservation	(Well-managed) tropical forests have important hydrological functions at catchment-level[5]	The effect of forests on stream regulation and erosion, and how this is determined by the composition of the landscape[6] To what extent these services are maintained in fragmented and secondary forests

[1] Nobre 2002
[2] Houghton *et al.* 2000; Achard *et al.* 2002; DeFries *et al.* 2002
[3] De Jong 2001; Pinard & Cropper 2000
[4] Smith *et al.* 2000; Trines 2003
[5] Van Dam 2003
[6] There is some uncertainty as to the changes in hydrological functions in exploited forests and forests which have been converted into plantations (Calder 2002, Van Dam 2003): forest conversion leads to higher runoff, but may have divergent effects on dry-season flow regulation and erosion.

There is a growing belief that intact tropical forests are net carbon sinks and that their conversion leads to net fluxes of carbon into the atmosphere (Achard *et al.* 2002, DeFries *et al.* 2002, Nobre 2002). However, it is likely that this important role of intact forests cannot be made of use under the Clean Development Mechanism of the Kyoto protocol. The high initial expectations have been shown to be unrealistic (Smith *et al.* 2000) especially as forest conservation or sustainable use *per se* are not at present eligible (Trines 2003). Furthermore, we lack knowledge of the full carbon balance of production forests (Pinard & Cropper 2000).

Forest hydrological functions may have more direct links to forest conservation than carbon sequestration. In the latter case the benefits are at global (climate change) or national (carbon credits) level, whereas benefits from hydrological functions and soil conservation are at regional or local level (Verweij 2003). The general hydrological functions of tropical forests are apparent (Van Dam 2003), but how these functions are influenced by the setting of forests within a landscape and to what extent they are maintained in degraded or secondary forests remains largely unclear.

Consequences for the research agenda

The need to deal with forest conservation and management at a larger (landscape) scale of analysis requires that we focus on new topics. From Tables 1 and 2 it becomes clear that the majority of studies on biodiversity conservation and environmental services of tropical forests have dealt with units within the landscape, without considering their interactions with other units. Very few studies have dealt with the influence of the economic and social context of forest areas in multi-functional landscapes, or with the importance of the wider landscape context. This is why so many gaps in knowledge are highlighted in the two Tables. Those highlighted topics must be addressed if we are to achieve better forest conservation in tropical multi-functional landscapes. But equally essential are studies that truly integrate the processes across scales and across landscape units. That is, studies which deal with the entire multi-functional landscape. Such studies necessarily integrate across disciplines, and explicitly include different stakeholders, as this is essential to understand processes at work at the landscape-level (*e.g.* Wiersum 2003).

This landscape-level research requires a constant updating and management of knowledge and it may require new ways of organising to science (Sayer and Campbell 2003). If we are really to base the management of large-scale landscape mosaics on sound science then we need to make long-term commitments to priority areas. We need to explore future scenarios and develop long-term visions of desirable outcomes. We need to understand the processes that are driving change and we need to recognise that management is a question of societal choice, and that these choices will change with time and our visions and programmes will need to be continuously adapted. This means that much research will need to be large-scale action research in which researchers use real life management as their experimental framework. Continuing dialogue with resource managers and users is essential (*e.g.* Kates *et al.* 2001). Experimental management, structured learning and adaptation are fundamental. There are challenges in integrating across scales, across disciplines, across landscape components, across institutions and between actors (e.g. Van Noordwijk 2002). Ultimately this all requires that the institutions that manage these large landscapes have a strong inbuilt science capacity.

Some of the key questions that are in need of immediate scientific responses are:

- What is the potential of fragmented forest landscapes for biodiversity conservation?
- What is the role of non-forest areas in connecting remaining forests and maintaining biodiversity at landscape level?

- What is the potential of payments for environmental functions to conserve forests outside Meso-America?
- How can we maximise biodiversity conservation in landscapes in which managed forests predominate?
- How effective and how cost-effective are corridors, and how does this relate to their scale?
- What is the real nature of the trade-off between local livelihoods and biodiversity conservation in mosaic landscapes?

Consequences for approaches to conservation

As a prelude to this discussion it is necessary to evaluate what conservation activities have and have not achieved so far. Table 3 contains a brief overview of successes and shortcomings of conservation activities, and indicates again which of these are related to multi-functional landscapes (underlined). Much conservation action has not taken into account the broader developmental context within which it occurs. In particular the landscape matrix and human context are often given insufficient attention. Assumptions about the extent to which 'projects' can change development trajectories are often unrealistic (Sayer & Campbell 2003).

The new setting of tropical forests in multi-functional landscapes requires a different approach to conservation. Such an approach naturally includes a system of protected areas, which is an essential component of any conservation approach. But also for protected areas, the socio-economic context and landscape setting cannot be excluded (Wilshusen *et al.* 2002), and should thus be explicitly included in conservation programmes.

In addition to forest conservation in protected areas, a new conservation approach to tropical forests requires the explicit inclusion of forest areas outside protected areas. There is an enormous potential conservation gain in maintaining forest cover and integrity outside protected areas. Here the conservation challenges lie primarily in preventing large-scale forest conversion to other land uses, and furthermore in managing landscapes to maximise the conservation of forests, their biodiversity and their environmental services. Preventing forest conversion may be achieved by providing value through payment for environmental services (carbon offsets; hydrological functions). For this system to be successful in preventing forest conversion, payment schemes at local or regional level should be in place (Kremen *et al.* 2000). A major challenge lies in getting the payments into the hands of those stakeholders whose decisions impact on the forest (Verweij 2003).

This conservation approach clearly requires that managers have an overview of the landscape-level context in which protected areas are located. This does not mean that conservation managers have to have total control over all elements of a landscape mosaic. It does require that mechanisms be in place to ensure that the activities of those who control the non-protected or non-forest units in the landscape mosaic take conservation concerns into account. This goes beyond spatial planning at regional level and requires negotiation to ensure that values that are dependent upon landscape-level attributes are given adequate attention. This may require that forest use is intensified in certain areas in order to achieve better conservation in others (Fredericksen & Putz 2003). It also requires a change in the nature of conservation projects: these should shift their focus from managing landscape elements to negotiating at the scale of entire landscape mosaics (WWF 2002a).

Table 3. An overview of current achievements and shortcomings of tropical forest conservation activities, particularly in relation to the context of these forests in multi-functional landscapes. Underlined are those activities that take into account interactions between components of multi-functional landscapes and/or attempt to manage entire landscapes including their interactions. Multi-functional landscapes are here defined as mosaic landscapes containing forest patches and experiencing strong human influence.

Tropical forest conservation actions:	We succeeded in	We have not succeeded in:
Protected areas	Setting aside numerous (including many large) protected areas	Effectively protecting these areas[1] Sufficiently addressing problems outside parks and applying park management at a landscape scale[2]
Integrated conservation and development projects	Securing (additional) funds for protected areas) Recognising the significance of social development for forest conservation	Recognising the (sometimes) negative impact of economic growth on biodiversity conservation Addressing problems outside parks as well[3] Finding win-win solutions for improving livelihoods and conserving biodiversity[4]
Extractive reserves	Establishing several (large) sustainable use reserves (especially in Brazil)	Making forest extraction profitable without large subsidies; ensuring that all species are maintained
Communal/indigenous forests	Securing a conservation status to (often large) areas[5] Empowering communities and transferring assets to them[6]	Recognising and maximising the role of well-managed communal forests for biodiversity conservation, (at a landscape scale[7])
Conservation concessions	Setting aside and funding some additional protected areas[8]	Integrating conservation, sustainable use and regional development
Eco-tourism	Introducing eco-tourism as a vehicle for conservation in some limited areas	Recognising its very limited economic and conservation potential in many areas and most countries[9]
Payments for environmental services	Showing that such payments are possible, mainly in Central America[10]	Realistically assessing its potential at larger scale and in other tropical regions[11] Getting global values to benefit local people[12]
Establishing corridors	Securing (additional) funds to establish (very few) corridors[13]	Collecting empirical evidence for the cost-effectiveness of corridors[14]

[1] Brandon *et al.* 2003
[2] Brandon *et al.* 2003
[3] But see Browder (2002) for an example in which communities outside protected areas were involved
[4] Browder 2002; Sayer & Campbell 2003; Sayer & Wells *in press*
[5] Peres & Zimmerman 2001
[6] Peres & Zimmerman 2001; Zimmerman *er al.* 2001
[7] Wiersum 2003
[8] Rice *et al.* 1997
[9] Kaimowitz 2002
[10] Trines 2003; Verweij, 2003
[11] Smith *et al.* 2000
[12] Verweij, 2002, 2003
[13] Simberloff *et al.*(1992)
[14] More than a decade after this critique was put forward by Simberloff *et al.*(1992), empirical evidence is still lacking.

This requires going beyond the traditional Integrated Conservation and Development Projects that have been the mainstay of many conservation organisations in recent decades. ICDPs often made unjustified assumptions about the convergence nature of conservation and development objectives. They assumed that with sufficient negotiation and technological innovation it was always possible to find win-win outcomes. Reality has now shown that this is rarely the case (Wells and Brandon 1992). Advocates of the conservation of global forest values have to recognise that their objectives will only be met at some cost to the livelihoods of local people. In the past these costs were imposed through laws and regulations - just as taxes are enforced through centrally mandated laws. Now we speak of payments for environmental services but although these have been widely used in the industrialised world this model has not really been tested in developing countries, especially not those that have weak institutions.

Progress in addressing this fundamental dilemma has been delayed by the fact that conservation organisations have been reluctant to accept the manifest reality that it is not possible to conserve all nature everywhere. We have to make choices and set priorities. The large- scale, ecoregional-planning exercises of WWF are a step in this direction. They recognise that priorities have to emerge from a multi-stakeholder negotiation process and cannot be imposed from outside. But as pressures on land mount and more and more of our biodiversity is dependent on mosaic landscapes we are going to have to be even more explicit about the balance that we want to achieve between conservation and development.

Conclusions

Emerging situations require new approaches to conservation: conservation of biodiversity cannot be adequately addressed uniquely in parks and large set-asides. Much more attention needs to be paid to the value of the forests elsewhere in the landscape - what sort of forests they should be and how they should be located to optimise production and conservation outcomes. Much more empirical evidence is needed on the impacts of different management techniques on biodiversity in fragmented and degraded forests. Strategies to conserve biodiversity as one element of a multifunctional landscape are lacking. The nature of such conservation programmes will be fundamentally different from those applied to set-asides. They will be focused on the regulation of exploitation, facilitation of forest recovery and forest restoration. They may require the establishment of networks of linked small protected areas. We will need land-use planning at a regional scale and effective mechanisms for making environmental service payments. Determining environmental values is only part of the challenge, ways have to be found to address the practical problems of making such payments in societies with weak institutions.

This new approach to conservation requires a new approach to research. This is required at the level of landscapes. We need to understand more about the connectivity and flows between different cells of the landscape matrix. Research on maximising biodiversity conservation in mosaics of exploited strictly protected and exploited forests and areas that people put to other uses. We need indicators of the performance of biodiversity in such complex landscapes so that managers can adapt their practices to achieve the conservation objectives that societies require. We emphasise that we are not implying any downgrading of the importance of protected areas as the core of conservation programmes. Well-designed protected areas, following clear criteria on diversity, endemism and complementarity are fundamental to securing conservation of ecosystems and biodiversity. What we wish to emphasise here is that in research and advocacy protected areas have dominated our thinking. There is now a need to be realistic and also look also at the potential of unprotected forests for conservation. Wilhusen *et al.* (2002) have done an excellent job of setting out the arguments for finding the middle ground between the hard line "neo-protectionist" approach to conservation and the soft, fuzzy world of Integrated Conservation and Development Projects. The next step is to implement the new approach to conservation that links conservation to development in the context of multi-functional landscapes and the new approach to science that provides the information necessary as a basis for such conservation activities.

References

Achard, F., H.D. Eva, H.-J. Stibig, P. Mayaux, J. Gallego, T. Richards, J.-P. Malingreau, 2002. Determination of deforestation rates of the World's humid tropical forests. Science 297: 999-1002.

Bawa, K.S. & R. Seidler, 1998. Natural forest management and conservation of biodiversity in tropical forests. Conservation Biology 12: 46-55.

Bierregaard & Stouffer 1997. Understory birds and dynamics habitat mosaics in Amazonian rainforests. In: Laurance, W.F. & R.O. Bierregaard (eds). Tropical forest remnants - ecology, management, and conservation of fragmented communities. University of Chicago Press, Chicago.

Brandon, K., Redford, K.H. and Sanderson, S.E. 1998 Parks in Peril: People, politics and Protected Areas. Island Press, Washington D.C.

Browder, J.O., 2002. Conservation and development projects in the Brazilian Amazon: lessons from the community initiative program in Rondônia. Environmental Management 29: 750-762..

Buck, A., Parotta, J. and Wolfrum, G. 2003 Building the future of the world's forests, planted forests and biodiversity. IUFRO Occasional Paper No. 15. Vienna, Austria.

Calder, I.R., 2002. Forest valuation and water - the need to reconcile public and science perceptions. In: P.A. Verweij (ed.) Understanding and capturing the multiple values of tropical forest. Tropenbos International, Wageningen, the Netherlands.

Chazdon, 1998. Tropical forests - log 'em or leave 'em? Science 281: 1295-1296.

Crome, F., 1997. Researching tropical forest fragmentation: shall we keep on doing what we're doing? In: Laurance, W.F. & R.O. Bierregaard (eds). Tropical forest remnants - ecology, management, and conservation of fragmented communities. University of Chicago Press, Chicago.

De Jong, B.H.J., 2001. Uncertainties in estimating the potential for carbon mitigation of forest management. Forest Ecology and Management 154: 85-104.

DeFries, R.S., R.A. Houghton, M.C. Hansen, C.B. Field, D. Skole, J. Townsend, 2002. Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 1990s. PNAS 99: 14256-14261.

Fimbel, R. A., A. Grajal & J.G. Robinson, 2001. The cutting edge - conserving wildlife in logged tropical forests. Columbia University Press, New York.

Fredericksen, T.S. & F.E. Putz, 2003. Silvicultural intensification for tropical forest conservation. Biodiversity and Conservation 12: 1445-1453.

Garcia-Fernandez C, Casado MA, Perez MR., 2003. Benzoin gardens in North Sumatra, Indonesia: Effects of management on tree diversity. Conservation Biology 17: 829-836.

Haddad NM, Bowne DR, Cunningham A, Danielson BJ, Levey DJ, Sargent S, Spira T, 2003. Corridor use by diverse taxa. Ecology 84: 609-615.

Houghton, R.A., D.L. Skole, C. A. Nobre, J.L. Hackler, K.T. Lawrence & W.H. Chomentowski, 2000. Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon. Nature 403: 301-304.

Kaimowitz, D., 2002. The role of forests in addressing global problems: what economic valuation methods won't tell us. In: P.A. Verweij (ed.) Understanding and capturing the multiple values of tropical forest. Tropenbos International, Wageningen, the Netherlands.

Kammesheidt, L., 2002. Perspectives on secondary forest management in tropical humid lowland America. Ambio 31: 243-250.

Kammesheidt, L., Kohler, P., Huth, A., 2002. Simulating logging scenarios in secondary forest embedded in a fragmented neotropical landscape. Forest Ecology and Management 170: 89-105.

Kates, R.W., W.C. Clark, R.. Corell *et al.*, 2001. Sustainability science. Science 292:641-642.

Kennard, D.K., 2002. Secondary forest succession in a tropical dry forest: patterns of development across a 50-year chronosequence in lowland Bolivia. Journal of Tropical Ecology 18: 53-66.

Kohler, P., Chave, J., Riera, B., Huth, A., 2003. Simulating the long-term response of tropical wet forests to fragmentation. Ecosystems 6: 114-128.

Kremen, C., J.O. Niles, M.G. Dalton, G.C. Daily, P.R. Ehrlich, J.P. Fay, D. Grewal, R.P. Guillery, 2000. Economic incentives for rain forest conservation across scales. Science 288: 1828-1832.

Lamb, D., 1998. Large-scale ecological restoration of degraded tropical forest lands: The potential role of timber plantations. Restoration Ecology 6: 271-279.

Laurance, S.G., Laurance, W.F., 1999. Tropical wildlife corridors: use of linear rainforest remnants by arboreal mammals. Biological Conservation. 91: 231-239.

Laurance, W.F. & R.O. Bierregaard (eds), 1997. Tropical forest remnants - ecology, management, and conservation of fragmented communities. University of Chicago Press, Chicago.

Martinez-Garza, C. & H. F. Howe, 2003. Restoring tropical diversity: beating the time tax on species loss. Journal of Applied Ecology 40: 423-429.

Melbourne,. B. A., K.F. Davies, C.R. Margules, D. B. Lindenmayer, D. A. Saunders, C. Wissel & K. Henle, 2004. Species survival in fragmented landscapes: where to from here? Biodiversity and Conservation 13: 275-284.

Nobre, C.A., 2002. Amazonian tropical forests: carbon source or sink? In: P.A. Verweij (ed.) Understanding and capturing the multiple values of tropical forest. Tropenbos International, Wageningen, the Netherlands.

Peres, C.A. & B. Zimmerman, 2001. Perils in parks or parks in peril? Reconciling conservation in Amazonian reserves with and without use. Conservation Biology 15: 793-797.

Perfecto, I., Mas, A., Dietsch, T., Vandermeer, J., 2003. Conservation of biodiversity in coffee agroecosystems: a tri-taxa comparison in southern Mexico. Biodiversity and Conservation 12: 1239-1252.

Pinard, M.A., Cropper, W.P., 2000. Simulated effects of logging on carbon storage in dipterocarp forest. Journal of Applied Ecology 37: 267-283.

Putz, F.E., G.M. Blate, K.H. Redford, R. Fimbel, J. Robinson, 2001. Tropical forest management and conservation of biodiversity: an overview. Conservation Biology 15:7-20.

Putz, F.E., K.H. Redford, J. Robinson, R. Fimbel, G.M. Blate, 2000. Biodiversity conservation in the context of tropical forest management. Environment Department Papers 75, The World Bank, Washington D.C.

Raman, T.R.S., Sukumar, R., 2002. Responses of tropical rainforest birds to abandoned plantations, edges and logged forest in the Western Ghats, India. Animal Conservation5: 201-216.

Rice, R., R. Gullison, J. Reed, 1997. Can sustainable management save tropical forests? Scientific American 276: 1246-1256.

Sayer, J, C. Elliott and S Maginnis. 2003. Protect, manage and restore: conserving forests in multi-functional landscapes. In: Proc. XII World Forestry Congress, Quebec, Canada, September 2003. C3a: 199-205.

Sayer, J.A. and B. Campbell, 2003 The Science of sustainable Development; Local livelihoods and the global environment. Cambridge university Press, Cambridge, UK.

Sayer, J.A. and B. Campbell. 2001. Research to integrate productivity enhancement, environmental protection and human development. Conservation Ecology 5: 32. http://www.consecol.org/vol15/iss2/art32.

Sayer, J.A. and M. Wells, *in press*. The Pathology of Projects. In T. McShane and M. Wells, Getting Biodiversity Projects to Work – Towards more effective conservation and Development. Colombia University Press, New York.

Scherr, S. J., A. White and D. Kaimowitz, 2003. A New Agenda for Forest Conservation and Poverty Reduction: Making Markets Work for Low-Income Producers. Forest Trends,Washington.

Sheil, D., Sayer, J.A., O'Brien, T., 1999. Tree diversity and conservation in logged rainforest. Science. 284: 1587.

Simberloff, D., J.A. Farr, J. Cox & D.W. Mehlman, 1992. Movement corridors: conservation bargains or poor investments? Conservation Biology 6: 493-504.

Smith, J., K. Mulongoy, R. Persson, J. Sayer, 2003. Harnessing carbon markets for tropical forest conservation: towards a more realistic assessment. Environmental Conservation 27: 300-311.

Ter Steege, H. 2003. Long-term changes in tropical tree diversity. Studies from the Guiana Shield, Africa Borneo and Melansesia. Tropenbos Series 22. Tropenbos, Wageningen, the Netherlands.

Tewksbury. J.J., Levey, D.J., Haddad, N.M., Sargent, S., Orrock, J.L., Weldon, A., Danielson, B.J., Brinkerhoff, J., Damschen, E.I., Townsend, P., 2002. Corridors affect plants, animals, and their interactions in fragmented landscapes. PNAS 99: 12923-12926.

Thiollay, J.M., 1995. The role of traditional agroforests in the conservation of rain-forest bird diversity in Sumatra. Conservation Biology 9: 335-353.

Trines, E., 2003. The Clean Development Mechanism: a flexible instrument for carbon offsets from afforestation and reforestation project activities. In: P.A. Zuidema (ed.). Tropical forests in multi-functional landscapes. Seminar proceedings. Prince Bernhard Centre, Utrecht University, Utrecht, the Netherlands.

Vandermeer, J. & R. Carvajal, 2001. Metapopulation dynamics and the quality of the matrix. American Naturalist, 158: 211-220.

Van Dam, O., 2003. Eco-hydrological functions and threats of tropical rain forests at different spatial scales. In: P.A. Zuidema (ed.). Tropical forests in multi-functional landscapes. Seminar proceedings. Prince Bernhard Centre, Utrecht University, Utrecht, the Netherlands.

Van Noordwijk, M., 2002. Scaling trade-offs between crop productivity, carbon stocks and biodiversity in shifting cultivation landscape mosaics: the FALLOW model. Ecological Modelling 149: 113-126.

Verweij, P.A., 2002. Innovative Financing Mechanisms for conservation and sustainable management of tropical forests – issues and perspectives. In: P.A. Verweij (ed.) Understanding and capturing the multiple values of tropical forest. Tropenbos International, Wageningen, the Netherlands.

Verweij, P.A., 2003. Getting a piece of the pie: profits of environmental services for local actors. In: P.A. Zuidema (ed.). Tropical forests in multi-functional landscapes. Seminar proceedings. Prince Bernhard Centre, Utrecht University, Utrecht, the Netherlands.

Wells, M. and Brandon, K. 1992 People and parks; Linking protected area management with local communities. The World Bank, Washington D.C.

Wethered R, Lawes MJ., 2003. Matrix effects on bird assemblages in fragmented Afromontane forests in South Africa. Biological Conservation 114: 327-340.

Wiersum, K.F., 1997. From natural forests to tree crops, co-domestication of forests and tree species, an overview. Netherlands Journal of Agricultural Science 45: 425-438.

Wiersum, K.F., 2003. Use and conservation of biodiversity in East African forested landscapes. In: P.A. Zuidema (ed.). Tropical forests in multi-functional landscapes. Seminar proceedings. Prince Bernhard Centre, Utrecht University, Utrecht, the Netherlands.

Wilhusen, P.R., S.R. Nrechin, C.L. Fortwangler, P.C. West, 2002. Reinventing a square wheel: critique of a resurgent "Protection Paradigm" in international biodiversity conservation. Society and natural resources 15: 17-40.

World Bank, 2003. Sustaining forestry - a World Bank Strategy. The World Bank, Washington D. C. Internet link: www.worldbank.org/forestry

WWF, 2002a. The Landscape Approach. Position paper, WWF, Gland.

WWF, 2002b. Forest management outside protected areas. Position paper, WWF, Gland.

Zimmerman, B., C.A. Peres, J.R. Malcolm & T. Turner, 2001. Conservation and development alliances with the Kayapó of south-eatsern Amazonia, a tropical forest indigenous people. Environmental Conservation 28: 10-22.

Zuidema, P.A., J.A. Sayer & W. Dijkman, 1997. Forest fragmentation and biodiversity: the case for intermediate size conservation areas. Environmental Conservation 23: 290-297.

Papers presented at first seminar (December 2002)

Conservation of tropical forest biodiversity in multifunctional landscapes

The economic value of tropical forest: A comparison between the Leuser National Park (Indonesia) and the Iwokrama Forest (Guyana)

Pieter van Beukering[1]

Introduction

Policymakers at various levels increasingly ask for estimates of the value of goods and services provided by tropical forests. They increasingly learn that deterioration of tropical forests doesn't pay off on the longer term. Local and national policymakers begin to agree that the value of natural resources depends not only on the market prices of its direct uses, but also on all other functions of the natural resources that generate value in its broadest sense. Especially now that policy discussions are heading for a better understanding of these 'hidden values', there is a need for the development of new simple tools to come to a 'fuller' valuation of these scarce resources to make better decisions. A method central to this effort is 'economic valuation'.

The objective of this paper is to explain the possibilities and limitations of using economic valuation in designing international nature conservation. The conceptual role of valuation is explained in determining the economic benefits of tropical rainforest at various scale levels in society. It is also described how the concept of benefit transfer can assist in applying economic values of an existing study to other areas or situations. To illustrate this aspect of economic valuation, two case studies in different regions are presented: Leuser National Park in Sumatra, Indonesia and the Iwokrama Forest, Guyana. To explain the danger in applying benefit transfer in a too simplified manner, the results of both case studies are compared and analysed and general conclusions and recommendations are formulated.

Role of economic valuation

In determining the TEV (Total Economic Value) of tropical rainforests, a distinction is often made between direct use values, indirect use values and non-use values (see Figure 1). The former relates to the values derived from direct use or interaction with a rainforests resources and services, whereas the second stems from the indirect support and protection provided to economic activity and property by the rainforests' natural functions, or regulatory 'environmental' services. A typical example of a direct use value of rainforests ecosystems is the provision of wood for housing or cooking. The classic example of an indirect use value as it relates to rainforest ecosystems is the water retention function that the forests support to downstream agricultural areas. Non-use values, amongst others, refers to an individual's willingness to pay (WTP) to secure the continued existence of, for example an endangered wildlife species, without ever actually seeing it in the wild (a 'use'). The classic example here is the contributions people make to actions that aim to preserve charismatic mega-fauna such as the tiger or the panda bear. If an individual is willing to pay $400 for preserving biodiversity in some rainforest area without any present or future use in mind (source of food, leisure hunting, wildlife viewing etc.) then this is his or her non-use value.

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Total Economic Value (TEV)

Use values | **Non-use values**

Direct use values

Outputs/services that can be consumed directly

- Extractive (timber, etc.)
- Non-extractive (tourism, research, etc.)

Indirect use values

Functional benefits enjoyed indirectly

- Biological support
- Water supply
- Fire prevention

Bequest, option and existence values

Functions that value either the future use, expected new information and based on moral convictions

- Endangered and charismatic species
- Threatened habitats
- Aesthetic forest landscapes
- 'Way of life' and traditional use

Figure 1. Composition of the Total Economic Value of rain forest

A common way to determine the use and non-use values is to pursue the sequence of underlying processes, starting with the cause of an impact, on to the physical impact and ending with the social and economic effects. The approach followed in this study proceeds in a series of methodological steps. Figure 2 provides an example of how the economic value derived from a rain forest by the agricultural sector is calculated. First, it is estimated what the ecological consequences are in terms of, for example, changes in water retention, erosion, and pest control. Next, these changes in the ecological services are translated into the physical impact for the agricultural sector. For example, the reduction of humus availability due to erosion may cause a decline in the overall agricultural output. Also, the reduced natural pest-control by birds and animals may cause an increased need for fertiliser and pesticides. Subsequently, these changes in the physical performance of the agricultural sector may cause a decline in the crop yield as well as an increase in the costs of production. This in turn can be translated into a change in the economic value of the Leuser National Park for the agricultural sector.

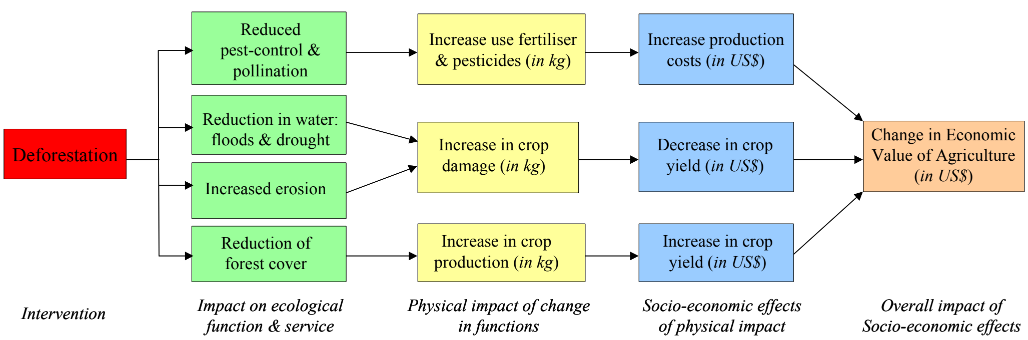

Figure 2 Overall approach applied to the agricultural sector

A growing number of studies apply economic valuation as the main analytical tool to compare the advantages and disadvantages of conservation of tropical forest. This rich basis of knowledge increasingly serves as a means to arrive at a balanced trade-off of economic costs and benefits in tropical forest management. The danger in doing so lies in the fact that the variation of values available varies enormously. Brander et al. (2003), for instance, show that the aggregated annual economic value of tropical wetlands ranges from less than $ 1 to just under $ 200 thousand per hectare. Similar conclusions can be drawn from studies investigating the value of rainforest conversion (see Table 1).

Table 1. Overview of NPV (Net Present Value) estimates for various land use options in different developing countries (in US$ per hectare).

Location/study	Result	Source
Peruvian Amazon	Selective forestry combined with fruit and latex: US$6,820 / ha	Peters, Gentry and Mendelsohn, 1989
	Clear cutting: $1,000 / ha	
	Timber & pulpwood plantation: US$3,184 / ha	
Malaysia	Forest production: US$2,455 / ha	Watson, 1988
	Intensive agriculture: US$217 / ha	
Cameroon	Cocoa conversion: US$1,248 / ha	Ruitenbeek, 1988
	Conservation: US$3,432 / ha	
General estimates	Agriculture: US$150-US$300 / ha	Pearce and Moran, 1994
	Sustainable forestry: US$200-500 / ha	
	Unsustainable forestry: US$1,000-2000 / ha	

Source: Combined from Beukering et al. 2001.

Because it is practically impossible to value at the respective time and place, it is inevitable to use data from previous studies that focus on a different region or time period. Therefore it is important to know when data from other studies can be used and under what conditions. For the transfer of monetary values, this practice is known as benefit transfer. The definition of benefit transfer is ‘an application of monetary values from a particular valuation study to an alternative or secondary policy decision setting, often in another geographical area than the one where the original study was performed’.

The most appropriate way to apply benefit transfer is by transferring the complete valuation function. Besides income, most studies find that other socio-economic and demographic factors have an influence on the valuation. Using statistical techniques (regression analysis) most valuation studies estimate a valuation function, that explains the value placed on a good or service as a function of these factors. This function can then be transferred to another site and by inserting the local values of the explanatory variables one can calculate an adjusted value. As opposed to wetlands, value functions for tropical rain forest have not been developed so far. The remainder of this paper will demonstrate how similar valuation approaches can lead to entirely different outcomes for different situations.

Economic Value of the Leuser National Park, Indonesia

The Leuser National Park in Northern Sumatra (Indonesia) covers 25,000 km2 and consists of a national park and a buffer zone. Deforestation in this ecosystem is widespread, despite its formally protected status (Van Schaik *et al.* 2001). This is believed to have severe ecological consequences, such as the probable local extinction of the Sumatran orang-utan, rhinoceros, tiger and elephant. In addition, the local economy could become structurally damaged as crucial ecological functions of the rainforest decline. Consequent damage caused by floods, erosion and loss of water supply can greatly exceed the revenues derived from timber extraction and land conversion.

The objectives of this study are to determine the Total Economic Value (TEV) of the Leuser National Park and evaluate the consequences of deforestation. The scenarios include: (1) the 'conservation' scenario, implying that protection of the rainforest is strictly enforced and that logging will be excluded as an economic activity; (2) the 'deforestation' scenario, implying a continuation of the current trend of clear-cutting; and (3) the 'selective use' scenario, in which logging of primary forest is substantially reduced and replanting of logged forest is assumed to be compulsory. The results are presented in terms of the types of benefits, the allocation of these benefits among stakeholders, and the regional distribution of these benefits.

Deforestation may be considered an easy way to generate fast cash. In the long term, however, the negative consequences will dominate. This is shown in Figure 3, which highlights the TEV in the three scenarios over time. In the deforestation scenario, ample revenues are generated in the first seven years. After the year 2006, revenues decline. The conservation scenario shows a steady increase in annual benefits throughout the 30-year period. By the year 2030, the annual benefits in the conservation scenario outweigh those of the deforestation scenario by a factor 4.

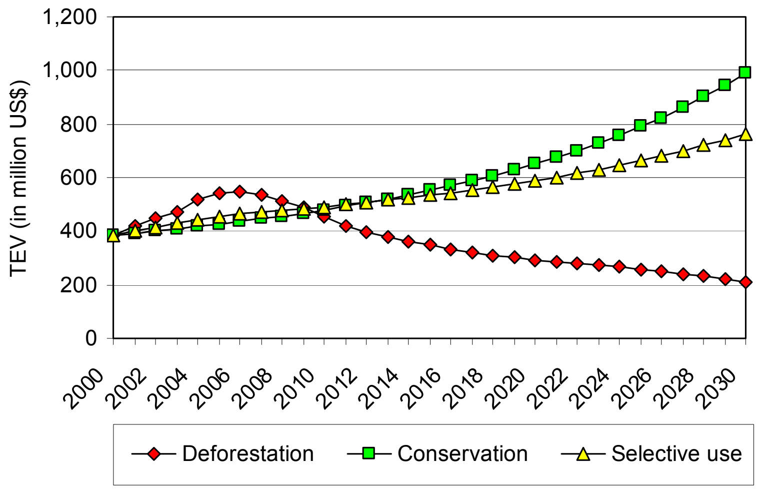

Figure 3. Net gains over time of Leuser National Park for the two scenarios.

By aggregating the annual gains over the 30-year period the overall TEV has been determined. The accumulated TEV at a zero discount rate of a deforested Leuser (US$ 11.3 billion) and of a conserved Leuser (US$ 18.5 billion) differ US$ 6.2 billion. This amount can be considered as the benefit of conservation (or the costs of deforestation).

The TEV is composed of numerous categories. These categories are shown in Table 2 for the three scenarios. The main contributors to the TEV are water supply, flood prevention, tourism and agriculture. Not surprisingly, timber revenues play an important role in the deforestation scenario.

Except for timber, NTFP and agriculture, the value of all benefits is higher in a scenario of conservation. Therefore, these categories are presented as benefits of conservation while timber, NTFP and agriculture are presented as the (opportunity) costs of conservation. The total aggregated benefits and costs of conservation amount to US$ 4.8 billion and US$ 2.2 billion, respectively. The main categories that gain from conservation are water supply, flood prevention, tourism and biodiversity. At the cost side of conservation, timber and agriculture are approximately of the same size. On balance, at a discount rate of 4%, the economy gains US$ 2.6 billion from conservation over a 30-year period.

Table 2. Distribution of TEV (Total Economic Value) among goods and services provided by the Leuser National Park over the period 2000-2030 (in million US$).

	Deforestation		Conservation		Selective use	
	Value	Proportion	Value	Proportion	Value	Proportion
Water supply	699	10%	2,419	25%	2,005	22%
Fisheries	557	8%	659	7%	674	7%
Flood prevention	1,223	18%	1,591	17%	1,396	15%
Agriculture	2,499	36%	1,642	17%	1,016	11%
Hydro-power	252	4%	898	9%	696	8%
Tourism	171	2%	828	9%	407	4%
Biodiversity	56	1%	492	5%	92	1%
Carbon sequestration	53	1%	200	2%	125	1%
Fire prevention	30	0%	715	7%	643	7%
NTFP	235	3%	94	1%	1,222	13%
Timber	1,184	17%	0	0%	825	9%
Total	**6,958**	**100%**	**9,538**	**100%**	**9,100**	**100%**

Note: for the period 2000 to 2030, at a discount rate of 4 %.

Besides the overall economic value of the Leuser National Park, it is important to be aware of the distribution of the TEV of deforestation, selective logging and conservation among the different stakeholders. Five groups of stakeholders have been identified in this study: (i) local communities; (ii) local government; (iii) elite logging and plantation industry; (iv) national government; and (v) international community. The distribution of the economic value among the stakeholders is presented in Table 3. Several typical features can be observed.

Contrary to popular belief, the local community is at present by far the main beneficiary of the Leuser National Park. Their share will grow in the conservation scenario. As expected, deforestation benefits the logging industry, mainly in the short run. In the long run, however, deforestation also harms the elite to a certain extent. As owners of large plantations and industries, they suffer negative consequences of reduced ecological services from the Leuser National Park. The local and the national government may also gain in the short term by collecting part of the rents of the harvested timber. In the long run, however, infra-structural damages increase while tax income decline. The international community only benefits from conservation of the Leuser National Park. Both the biodiversity value and the option of sequestration are important gains for developed countries.

A striking element is that the elite (logging) industry collects a much larger share of the total value in the deforestation scenario (23%). If the Leuser National Park would be strictly conserved, their share is only 11%. This reduction in value for the elite industry in the conservation scenario benefits predominantly the local and the international community. The power structure by the elite (logging and plantation) industry and the societal and spatial spread of the local and the international community, however, prevents the conservation scenario to emerge. For similar reasons, compensation of the latter by the former group is constrained.

Table 3. Distribution of NPV among stakeholders (in %).

Scenario	NPV (million US$)	Local Community	Local government	Elite industry	National government	International community
Deforestation	6,958	45%	11%	23%	7%	13%
Conservation	9,538	56%	9%	11%	5%	19%
Selective use	9,100	53%	10%	14%	5%	18%

Note: NPV over the period 2000-2030 at a discount rate of 4%

Economic value of the Iwokrama Forest, Guyana

At the 1989 meeting of the Commonwealth Heads of Government in Kuala Lumpur, the Honourable Desmond Hoyte, then President of Guyana, proposed Iwokrama as a laboratory to develop sustainable solutions for the economic, social, and environmental issues related to the management of tropical forests. As a consequence, the Iwokrama International Centre for Rain Forest Conservation and Development was established in 1996 and became fully operational in 1998.

Covering an area of some 360,000 hectares in central Guyana, the Iwokrama Forest provides an opportunity for research into, and development of, new approaches to sustainable forest management in tropical ecosystems. The Iwokrama Centre fully recognises that protected areas will only survive if they are seen to be of value, in the widest sense, to the nation as a whole and to local people in particular. Therefore, the programme conducted extensive studies on the options for sustainable utilisation of the Iwokrama Forest.

The main objective of the present study is to determine the Total Economic Value (TEV) of the Iwokrama Forest and evaluate the consequences of different forms of use for its main stakeholders. Economic valuation has been applied to evaluate the TEV of the Iwokrama Forest under two scenarios: (1) the '*sustainable utilisation*' scenario, implying that the rainforest, in particular the sustainable utilisation area (SUA), is utilised strictly following the conditions of sustainability with a strong role by the Iwokrama Centre; (2) the '*open access*' scenario, implying that the Iwokrama Centre will redraw from managing the forest leading to unsustainable practises such as high impact logging operations in various sectors. The benefits included in the economic valuation are among others: fisheries; agriculture; tourism; biodiversity; carbon sequestration; non-timber forest products; and timber.

Figure 4 shows the annual net benefits for the sustainable and open access scenarios over the period 2000-2030. Almost from the start, sustainable utilisation generates higher socio-economic benefits than the unsustainable practises. Only in the first few years, the costs invested in setting up the operations slightly exceed those of the costs made under unsustainable conditions. These additional costs, however, soon pay off in higher net-benefits. Conducting sustainable logging in combination with eco-tourism pays off better than following the route of 'open access'.

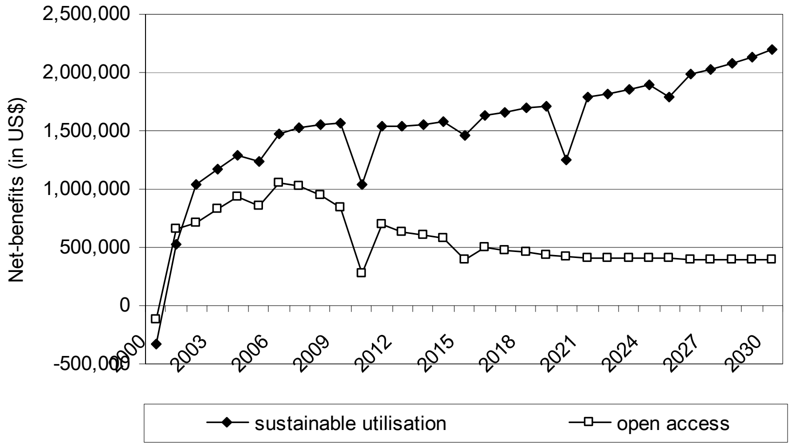

Figure 4. Net Benefits over time of Iwokrama Forest for the two scenarios.

If the annual stream of costs and benefits are not discounted, the TEV over 30 years of a sustainably managed Iwokrama Forest is calculated at over US$ 48 million. An unsustainably managed Iwokrama Forest is estimated to generate a value of US$ 17 million. Hence, the total loss due to improper management is roughly US$ 31 million. The latter can therefore be interpreted as the net losses due to deforestation vis-à-vis the conservation scenario.

The TEV presented above is composed of numerous benefits. The composition of these benefits is shown in Table 4 for the two scenarios. All benefit categories are higher in the sustainable utilisation scenario. Even the timber industry, which is usually the sector that benefits most from deforestation derives less net-benefits in the open access scenario. Under conditions of sustainability, NTFPs contributes most to the annual benefits (33%). Agricultural production (20%) and timber extraction (17%) are also significant sectors in the over value of the Iwokrama Forest. The three largest absolute differences between the two scenarios in individual benefits are timber, eco-tourism and agriculture. In relative terms benefit categories such as biodiversity values, carbon credits and services are practically absent in the open access scenario. Without the presence of a strong coordinating body such as the Iwokrama Conservation Centre, these benefits cannot be offered to the international community. Table 4 provides results, showing an average annual value of the sustainable utilisation scenario of US$ 843,421 compared to US$362,633 for the deforestation scenario.

Table 4 Distribution of average annual net-benefits to the different sectors.

	Sustainable utilisation		Open access	
	in US$	Proportion (%)	in US$	Proportion (%)
NTFP value	277,173	33%	104,215	29%
Agricultural value	167,364	20%	132,986	37%
Timber value	141,891	17%	81,446	22%
Biodiversity value	80,100	9%	5,325	1%
Carbon value	67,688	8%	-	0%
Recreational value	66,037	8%	34,122	9%
Service value	43,169	5%	4,541	1%
Total annual benefit	843,421	100%	362,633	100%

Note: for the period 2000 to 2030, at a discount rate of 4 percent.

Besides the overall economic value, the distribution of this value among the different stakeholders is important. The stakeholder distribution is shown is in Table 5. Five stakeholder groups have been identified: (i) local communities; (ii) the civil society; (iii) the business community; (iv) the Guyana government; and (v) the international community. In both scenarios, the local community is by far the main beneficiary of the Iwokrama Forest. They receive around half of the products and services provided by the forest. Another interesting observation is that the business community also gains from sustainable practices. Under sustainable conditions, more economic activities take place, thereby directly and indirectly supporting commercial ventures. Finally, although the international community receive more benefits under sustainable conditions, a larger share of the benefits stay within Guyana.

Table 5 Relative distribution of TEV among stakeholders (in million US$).

Scenario	Local community	Civil society	Business community	Guyana government	International community
Open access	5.5 (46%)	0.7 (12%)	2.2 (15%)	1.6 (15%)	1.0 (13%)
Sustainable utilisation	11.7 (51%)	3.1 (6%)	3.7 (20%)	3.7 (15%)	3.3 (9%)

Note: NPV over the period 2000-2030 at a discount rate of 4%

Conclusions and recommendations

Economic valuation has proven to be a strong and useful tool in analysing welfare changes for the different scenarios in the Leuser National Park and the Iwokrama Forest. In qualitative terms, both studies generate comparable results. Sustainable utilisation or conservation prevents damage and loss of income to society while uncontrolled deforestation only generates income on the short term and leads to severe loss of income on the longer term. Moreover, sustainable practises spread the benefits of the rain forest more equally among regions and stakeholders and thus prevent further social conflict. Uncontrolled deforestation, on the other hand, widens the income gap between societal groups and therefore may be an additional source of further conflict. This dependency may form a strong incentive for the regencies to develop and enforce a common plan.

Despite of these commonalities of both studies, and the similar methodology applied in both cases, several differences exist between the Leuser National Park and the Iwokrama Forest. The most typical difference is the value of the rainforest in both regions. Whereas the sustainable (conservation) and the unsustainable (deforestation) scenario in the Leuser National Park generate a total economic value of US$ 9,538 and US$ 6,958 per hectare, respectively, similar scenario's simulated for the Iwokrama Forest generate values of no more than US$ 270 and US$ 152 per hectare, respectively. This difference is caused by a number of ecological, geographical and economic factors. For example, the population density in the Leuser study area is approximately 2.64 people per hectare. In the Iwokrama the population density of the study area is not more than 0.01 people per hectare. The difference in population density for both areas has major consequences for the level and the composition of the economic value. While the benefits in Leuser lie mainly in the indirect use values (i.e. water supply, flood prevention), the benefits of Iwokrama are dominated by direct use and potentially captured non-use values (Figure 5).

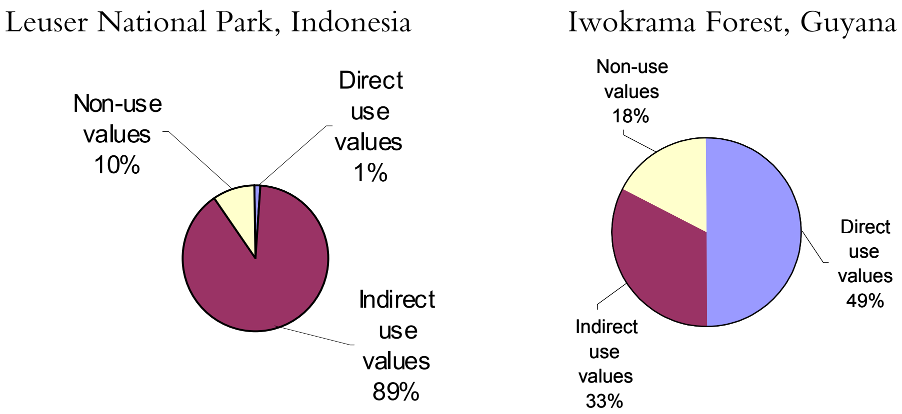

Figure 5. Composition of the Total Economic Value for the Leuser National Park and the Iwokrama Forest under conditions of sustainability.

As a result, consequences for policy makers differ between Leuser and Iwokrama.

Leuser National Park: Because of the high population density and subsequent importance of indirect use values, the efforts in Indonesia should mainly be focussed at protecting the Leuser National Park against encroachment and illegal logging. If the ecological integrity cannot be maintained, the negative economic consequences in terms of loss of water supply and increase of flooding and drought events can be enormous. Most of the benefits are already operational, despite the fact that they may be 'hidden'.

Iwokrama Forest: In Iwokrama, an opposite movement must be set in motion. Rather than franticly protecting the forest, sustainable ways should be found to materialise the potential values present in the forest. To realise this, a central organisation is essential to take initiative and establish the link between local supply and global demand for the ecological functions of the Iwokrama forest. Without the presence of the Iwokrama Centre different types of foreign grants and investments, such as carbon credits, bioprospecting and conservation grants, would most likely not be made.

References

Brander, L.M., R.J.G.M. Florax and J.E. Vermaat (2003) *The Empirics of Wetland Valuation: A Comprehensive Summary and Meta-Analysis of the Literature*. Working Paper. Institute for Environmental Studies (IVM), Free University, Amsterdam.

Van Beukering, P.J.H., H.Cesar and M.A. Janssen (2001) *Economic valuation of the Leuser Ecosystem on Sumatra, Indonesia: A stakeholder perspective*. Leuser Development Programme (LDP), Medan.

Van Beukering, P.J.H. and A.M van Heeren (2002), *The economic value of the Iwokrama Forest Reserve, Guyana*. Working Document. Iwokrama International Centre for Rain Forest Conservation and Development, Greater Georgetown, Guyana.

Van Schaik, C.P, K.A. Monk and J.M.Y. Robertson (2001) Dramatic decline in orang-utan numbers in Northern Sumatra, *Oryx* 35(1), 1-22.

Use and conservation of biodiversity in East African forested landscapes

K.F. Wiersum[1]

Introduction

The dichotomy between wilderness areas as representations of nature and cultivated fields as cultural artefacts has been firmly established in Western thought since Classical times (Harris, 1996). This dichotomy is reflected in many studies which juxtapose forests and deforested areas. In most of those studies deforestation is equated with a drastic change in ecological conditions resulting in the loss of nature and biodiversity. Such a loss is assumed to be unavoidable, irrespective of the intensity of the process of forest change and the type of land-use following this process. However, several studies have indicated that human-induced changes in native forests and introduction of cultivation does not always involve deforestation in the sense of disappearance of a forest cover and loss of biodiversity. In several studies it has been found that the presumed deforestation process consisted in reality in a change from natural forests to anthropogenic forests which structurally still maintain many characteristics of the natural forest, but which have been changed in composition through the purposeful stimulation of species providing daily livelihood products for local people (Henkemans et al., 2000). In some cases it has even been shown that the assumed deforestation process did in reality not take place at all, and that instead of depleting forests local people were rather stimulating a forest extension (Fairhead & Leach, 1996). Such examples show that the human impact on forests should not be considered as always bringing with it a universal linear process of deforestation. Rather, it should be considered that it may involve a conscious change in forest structure and composition and that it forms part of a dynamic process of landscape transformation. Such landscape transformations may entail a gradual transformation of native forests into forested landscapes in which a variety of land-use systems ranging from natural forests, 'economically enriched' forests, agroforestry systems to mono-cropped fields co-exist (Wiersum, 1997a). Such indigenously developed forested landscapes have been described from all tropical continents (e.g. Campbell, 1996; Posey, 1985; Colfer et al., 1997). These forested landscapes illustrate the creativity of local communities in combining conservation with increased productivity of resources needed for daily livelihood needs (Wiersum, 1997b; Wiersum & Gomez Gonzalez, 2000). As a result of this process, rather than a dichotomy a continuum exists between nature and cultivated fields. Some of the intermediate vegetation types in this continuum still contain a high biodiversity. Although on the one hand some species of the natural vegetation may have been decreased, on the other hand the newly created anthropogenic forest types may have been enriched with species favoured for use by local communities.

The recognition that local communities are often actively engaged in using and maintaining biodiversity in forested landscapes, brings with it an understanding that two contrasting perspectives on the relevance of biodiversity are possible (Table 1). On the one hand, ecological scientists and conservation authorities mostly refer to the need to maintain biodiversity for use by future populations and they focus mainly on the need of conserving rare and endangered species occurring in wilderness areas. On the other hand, local communities value biodiversity as a resource for fulfilling livelihood needs. As a result, their biodiversity management activities focus on maintaining useful biodiversity

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and even enriching their environments with the biodiversity resources that have present value.

Table 1. Contrasting perspectives on biodiversity.

	Dominant perspectives of conservation authorities	Dominant perspectives of local communities
Major biodiversity value	Rare & endemic species	Species used for livelihood purposes
Main rationale for conservation	Maintain ecological integrity on basis of scientific criteria	Maintain instrumental and cultural values on basis of local criteria
Major objective for maintaining biodiversity	Preserving option and bequest values for future generations	Maintaining present use
Species considered	All taxonomically reasonably-known species	Resources: species which provide valuable products & services
Main conservation approach	In-situ preservation by prohibiting/limiting present use	Controlled utilisation and gradual domestication

The aim of this paper is to illustrate how indigenously developed forest landscapes offer good options for combining both professional and local perspectives and for integrated conservation of biodiversity for both present and future generations. The paper will focus on the following two questions:

- What are the features of an indigenously created forested landscape and how is it managed?
- Does the creation of forested landscapes contribute towards conservation of endangered species?

The discussion will be based on information of several recent studies in East Africa.

Forested landscapes as complexes of biodiversity use and conservation

Within the dichotomised view distinguishing forests and deforested areas, it is often considered that local people are one of the main driving forces between deforestation and loss of biodiversity. Consequently, it is proposed that forest and biodiversity conservation is best served by professional conservation agencies protecting natural forests. Such views are legitimated by data showing that local communities have often seriously degraded the natural vegetation. As illustrated in Table 2, it has often been found that clear differences in vegetation structure exist between forest areas under the control of professional conservation agencies and adjacent woodlands under the control of local communities. The data from Table 2 indicate that the original tree vegetation as still present in the forest reserve has in the village woodlands been replaced by a multi-stemmed bushlike vegetation structure. Moreover, in the village woodlands a loss of biodiversity has occurred notably of the species that are used by the local community. These data suggest that the village woodlands are degraded as a result of overexploitation.

Table 2. Vegetation structure in Chimaliro Forest Reserve and adjacent village woodland area in Malawi (Mangani-Kamoto, 1999).

	Forest Reserve	Village woodlands
Structure		
Trees per ha	533	3360
Basal area (m2/ha)	15	13
Biodiversity		
Species diversity (> 10 trees/ha)	36	34
No, of fruit tree species	15	12
No. of preferred firewood species	25	22

Table 3. Zonation of Chimaliro forested landscape, Malawi (Mangani-Kamoto, 1999).

Landscape types		Management objectives
Forest reserve	Natural forest	Watershed management, no wood harvesting
	Eucalyptus plantation	Wood production
Village lands	Village woodland	Collection of wood and non-timber forest products (ropes, mushrooms, medicines)
	Fallowed lands	Soil recuperation & additional collection of wild resources
		Production of fruits, poles
	Homesteads	Maintenance of cultural/religious values
	Graveyards	

However, at a closer look this dichotomised view can not be maintained and a more complex picture emerges. When considering the total landscape it appears village territory consists of a variety of land-use types. Rather than two forest types a variety of forested land-use types are present (Table 3). In the village area several patches of relatively undisturbed natural vegetation still occur. And in addition to the woodlands also homesteads with mixed tree growth are present. In these homesteads people have concentrated the cultivation of highly-valued tree species such as fruit trees. Consequently, the decrease in fruit and firewood species in the woodlands has been compensated by the cultivation of species with similar functions in the homesteads. As a result rather than the decline in forest product availability suggested by the data in Table 2, local villagers have experienced in increase in availability of several forest resources such as food and medicinal products (Table 4). Thus, the biodiversity of the original vegetation has been divided over different landscape units and a niche differentiation in biodiversity conservation has taken place. The vegetation in the different landscape units is under different degrees of management intensity ranging from controlled utilisation to tree cultivation (Table 5). Depending on their precise functions within the prevailing land-use systems, in the landscape niches specific species are purposefully maintained. And in some landscape niches even new species may be introduced. Consequently, even if species composition in some landscape niches may have declined in comparison to the natural vegetation, this does not automatically mean that also at the total landscape a similar loss of biodiversity has occurred. Thus, the creation of forested landscapes allowed specialization in the use and maintenance of different forms of biodiversity.

Table 4. Trends in forest product availability in Chimaliro region

Decreased availability	Increased availability
Game (incl. culinary caterpillar sp)	Fruits
Firewood	Medicines
Thatch grass	Rope fibre
Poles	Mushrooms

Table 5 Tree-related management practices

	Controlled utilisation	Stimulation of production	Stimulation of regeneration
Forest reserve	No wood cutting		
	Fire control		
Eucalyptus plantation	Harvesting by Forestry Department	Weed control in young plantations	Tree planting
		Thinning	
Graveyards	Strict control on harvesting by taboos		
Village woodland	Various forms of damage-controlled harvesting	Selection of coppice shoots	Coppicing & pollarding
	Individual claims on certain trees		
Homesteads	Harvesting by private owners	Manuring of trees	Planting of fruit trees and exotic timber trees
		Weeding around trees	
		Stimulating fruit production	

In East Africa several examples of such indigenously-created forested landscapes have been described both in the miombo woodland region (Fortmann & Nihra, 1992; Clarke, 1994; Campbell, 1996) as well as in other vegetation biomes (e.g. Kessy, 1998). The recognition of such forested landscapes brings with it an understanding that local communities may be actively engaged in a variety of indigenous biodiversity management activities (Fortmann & Nihra, 1992; Clarke, 1994; Wiersum, 1997b). The management practices as used by the local communities are often of a different nature than the management practices as practised by the professional conservation agencies (Table 5), and therefore they have often been overlooked. As illustrated in Table 3 the focus of the indigenous forest management practices is often directed at a more varied number of products and services and thus a larger number of species than the management practices by professional forestry agencies. Nonetheless, in both professional and local forest management the same basic principles are used for combining conservation and production. These principles include zonation in forest land-use zones, as well as the application of four main types of management practices along a gradient of increasing energy input per unit area, i.e. controlled utilization, conscious protection of valued species, stimulation of enhanced production of desired products, and active stimulation of regeneration (Wiersum, 1997b).

Role of forested landscapes in conservation of rare and endemic species

The creation of forested landscapes illustrates that the legal zonation in state forest reserves and village lands does not result in a landscape dichotomy of 'wild' forests and domesticated fields, but in a forested landscape in which different types of natural and anthropogenic woody vegetation types co-exist. The community activities do not just involve extraction of wild species from the native vegetation, but also creativity in developing resource-enriched forests. These forested landscapes illustrate the outcome of local communities efforts in maintaining and even enhancing biodiversity values, which are of local relevance and contribute to the local livelihoods of the present generations.

In respect to the conservation of biodiversity in the sense of protecting rare endemic or even endangered species, an important question is whether the enriched landscape niches created by rural communities are only repositories of relatively common and even exotic species, or whether they may also play a role in the conservation of rare and endemic species.

Table 6. Use of forest products in the East Usambara mountain region, Tanzania (no. species & no. of endemic species) (Kessy, 1998)

Type of products	Forests	Farmlands (incl. homegardens)
Different types of wood products	100 (12)	29 (6)
Ropes	11 (1)	-
Foods	28 (4)	28 (4)
Medicines	185 (n.a)	43 (8)
Home utensils	83 (2)	13 (1)
Shade & water protection		8 (1)

Table 7. Occurrence of *Dalbergia melanoxylon* in different land-use zones in Mozambique (after Albano, 2001)

Land-use system	Tree vegetation	Density of *D. melanoxylon*
Forests	Native vegetation	Total 32 stems/ha Seedlings & saplings 54%
Open savanna woodlands	Native vegetation	Total 156 stems/ha Seedlings & saplings 67%
Fallow lands	Purposefully retained & resprouting native species, domesticated species	Total 53 stems/ha Seedlings & saplings 78%
Crop lands	Purposefully retained native species, domesticated species	Total 30 stems/ha Seedlings & saplings 60%

This question cannot be answered unequivocally. In a study in Tanzania it was found, that local communities may consciously preserve and even cultivate endemic species on their village lands, because of their role to fulfil household needs (Table 6). In another study in Mozambique it was found that also endangered species may be consciously conserved in anthropological created forest-derived landscape niches. This study concerned the status of African ebony (*Dalbergia melanoxylon)*. This species is a favoured species for artisanal products and wood manufacturing. The high demand for the species is considered to threaten its natural population, and consequently it was proposed for listing on the CITES appendix 2. Such listing would involve restrictions on the international trade of this species. However, as a result of the great value of African ebony, rural people in Mozambique are actively protecting and sometimes even cultivating this species in several landscape niches (Table 7). By doing so, they are actively protecting this species rather than endangering it by ruthless overexploitation.

However, in yet another study in Tanzania on the status of several commercial timber species including *Dalbergia melanoxylon*, but also *Afzelia quanzensis*, *Khaya anthotheca*, *Milicia excelsa* and *Pterocarpus angolensis* it was found that local people hardly recognized the seeds and seedlings of these species. Seedlings were not protected and no stimulated regeneration of these trees was practised (Munyanziza & Wiersum, 1999). The difference between this finding and findings of other studies of local tree species being consciously protected and even cultivated was attributed to differences in regeneration strategies between species and hence in ease of propagation. Another factor may be differences in the legal status of the species. The difference in indigenous management practices for *D. melanoxylon* between Mozambique and Tanzania illustrate how such practices are often location-specific.

Conclusion

When considering biodiversity conservation, it is wrong to assume a dichotomy between 'pristine' nature and cultivated fields as a cultural artefact. Rather, it should be considered that as a result of the local people's needs for biodiversity resources and their creativity in transforming native forests to suit their resource needs, they often create forested landscapes in which a combination of land-use zones in a continuum from nature to cultivated fields co-exist. The creation of a forested landscape is the result of 'domiculture' (Chase, 1989). This term refers to the creation of a series of localized areas (domuses) of interaction between people and resources, each characterised by a specific set of resource technologies. Domiculture may include conservation of biodiversity through consciously preservation of valued species by restricting open-ended exploitation as well as cultivation of both local and exotic tree species in specific landscape niches. Consequently, forested landscapes offer combined options for '*in situ'* conservation of biodiversity in the protected native vegetations and '*in domo'* conservation of biodiversity in human-derived vegetations. The multiple roles of forested landscapes in preserving different types of biodiversity illustrates that conservation of biodiversity should not be only focused on limiting use of wild populations, but also on options for diversified management at landscape level.

For effective biodiversity conservation it should be considered that there exist various perspectives on the need to preserve biodiversity by different groups of people. In order to accommodate these different interests, biodiversity conservation should not just focus on conservation of wild vegetations. Rather, the attention should be focused on forested landscapes in which a mosaic of wild and human-influenced vegetation types co-exist. An approach which combines '*in situ'* and '*in domo*' conservation of biodiversity offers a good basis for niche specialisation in conserving different categories of biodiversity. Such a forested landscape approach offers options for involvement of various stakeholder groups in biodiversity conservation. This enables an integrated approach to biodiversity conservation, in which the concerns for both present and future generations are combined.

References

Albano, G., 2001. Indigenous management practices and conservation of *Dalbergia melanoxylon* (Guill. & Perr.) in Mozambique. MSc thesis Tropical forestry, Department Environmental Sciences, Wageningen University.

Campbell, B. (ed), 1996. The Miombo in transition: woodlands and welfare in Africa. Bogor, Indonesia, Center for International Forestry Research (CIFOR).

Chase, A.K., 1989. Domestication and domiculture in northern Australia: a social perspective. In: D.R. Harris & G.C. Hillman (eds), Foraging and farming, the evolution of plant extraction. Unwin Hyman, London, p. 42-54.

Fairhead, J. & Leach, M., 1996. Misreading the African landscape. Society and ecology in a forest-savanna mosaic. Cambridge University Press, UK.

Clarke, J., 1994. Building on indigenous natural resource management: forestry practices in Zimbabwe's communal lands. Harare, Zimbabwe, Forestry Commission.

Colfer, C., Peluso, N. & See Chung, C., 1997. Beyond slash and burn: building on indigenous management of Borneo's tropical rain forests. New York Botanical Garden, USA, Advances in Economic Botany Volume 11.

Fairhead, J. & Leach, M., 1996. Misreading the African landscape. Society and ecology in a forest-savanna mosaic. Cambridge University Press, UK.

Fortmann, L. & Nihra, C., 1992. Local management of trees and woodland resources in Zimbabwe: a tenurial niche approach. Oxford Forestry Institute, UK, O.F.I. Occasional Paper No. 43.

Harris, D.R., 1989. An evolutionary continuum of people-plant interaction. In: D.R. Harris & G.C. Hillman (eds), Foraging and farming, the evolution of plant exploitation. Unwin Hyman, London, p.11-26.

Henkemans, A.B., Persoon, G.A. & Wiersum, K.F. (2000). Landscape transformations of pioneer shifting cultivators at the forest fringe. In: K.F. Wiersum (ed), Tropical forest resource dynamics and conservation: from local to global issues. Wageningen University, Tropical Resource Management Papers No. 33, p. 53-69.

Kessy, J.F., 1998. Biodiversity management and local interests; alternatives for the East Usambara Forest reserve, Tanzania. Wageningen Agricultural University, Tropical Resource Management Paper No. 18.

Mangani-Kamoto, J.F., 1999. Indigenous silvicultural practices of miombo woodlands in Malawi. A case in five villages close to Chimaliro Forest Reserve. M.Sc. thesis Tropical forestry, Department of Forestry, Wageningen Agricultural University.

Munyanziza, E. & Wiersum, K.F., 1999. Indigenous knowledge of miombo trees in Morogoro, Tanzania. Indigenous Knowledge and Development Monitor 7(2): 10-13.

Posey, D.A., 1985. Indigenous management of tropical ecosystems: the case of the Kayapo Indians of the Brazilian Amazon. Agroforestry Systems 3: 139-158.

Wiersum, K.F., 1997a. From natural forests to tree crops, co-domestication of forests and tree species, an overview. Netherlands Journal of Agricultural Science 45: 425-438.

Wiersum, K.F., 1997b. Indigenous exploitation and management of tropical forest resources: an evolutionary continuum in forest-people interactions. Agriculture, Ecosystems & Environment 63: 1-16.

Wiersum, K.F. & Gomez Gonzalez, I.C., 2000. 'Intermediate' forest types as nature-human systems: characteristics and future potential. Paper International Workshop 'Cultivating (in) tropical forests, the evolution and sustainability of intermediate systems between extractivism and plantations'. Lofoten, Norway, June 28-July 1, 2000.

Papers presented at second seminar (April 2003)

Environmental services of tropical forests in multi-functional landscapes

Carbon sequestration in forests - the ten most frequently asked questions

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(The EFI carbon analysis team)

The number of publications on carbon sequestration in forests is overwhelming. In spite of this , there is a lot of confusion in this topic , especially now that discussions are mounting concerning the implications of the Kyoto Protocol. We try to clarify the situation by answering ten most frequently asked questions.

1. How does the principle of carbon sequestration in forests work?

Plants absorb carbon dioxide (CO_2, contains 12/44 proportion of carbon) from the air. With energy from sunlight, this carbon is converted into sugars (Gross Primary Production). Part of these sugars is used for maintenance (respiration), but the other part is used for the growth of plant organs (Net Primary Production). Only a small part of this growth remains as plant tissue for any length of time, because litterfall (dead plant material and its renewal) is the major part of the Net Primary Production. However, the process of net growth in trees can continue for a long period. Forests are thus the most effective land use type in this context, because a large amount of carbon is withdrawn from the atmosphere for a long time.

> The two terms ' carbon stock' and 'carbon flux' need to be separated (they are often mixed). Carbon stock is the storage of carbon in trees, ground vegetation, soil or products, usually expressed as tonnes of carbon per hectare. Carbon flux is the flow of carbon between the stocks or the atmosphere.

2. Isn't carbon released again after a while, undoing the sequestration process?

In the long run, the biomass in forests is indeed in equilibrium if the forest is not disturbed by e.g. a fire, insect or fungi damage or harvesting. In an unmanaged forests the trees will die and the carbon is released; in a managed forest trees are harvested and carbon is released through product oxidation (see Figure 1). It is also true that carbon is being continuously exchanged between forest, soil, products and atmosphere, but a certain amount of carbon is always stored. Carbon is cycling within the system. This situation differs from that of fossil fuels where their use and burning results in a one-way flux of carbon into the atmosphere.

> The terms 'carbon sink' and 'carbon source' are frequently used, 'sink' meaning that the net flux of carbon goes into the system from the atmosphere, while 'source' means that the net flux of carbon goes from the system into the atmosphere. Forests are typically a source of carbon when disturbed (as a consequence of fire, insect or fungi damage, harvesting) and a sink when recovering after disturbance.

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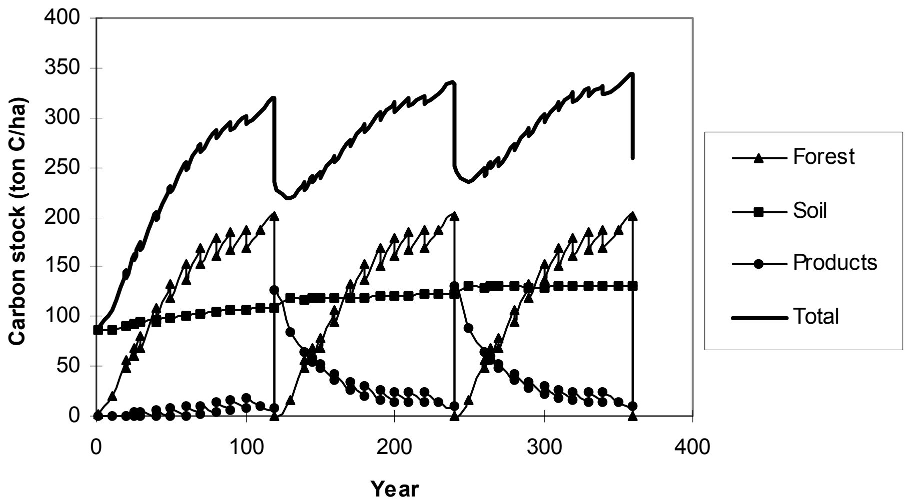

Figure 1. Carbon stocks in an oak forest in a rotation of 120 years. This figure was generated with the model CO2FIX. CO2FIX is a user-friendly forest carbon accounting model and can be downloaded from EFI's homepages at http://www.efi.fi/projects/casfor .

3. How much carbon do forests in Europe store?

The amount of carbon varies very much throughout Europe (like average growing stock), but on the average the carbon stock in trees amounts to 53.2 Mg (=tonnes) C ha^{-1}. Some statistics are given in Table 1.

Table 1. Carbon stocks and fluxes in the European forests. The total carbon emissions due to burning of fossil fuels amounts to 900 million ton C.

Stock or flux	Amount
Carbon stock in trees and soils of European forests	19,978 million tonnes C
Carbon stock in tree biomass	7,927 million tonnes C
Estimated net sequestration in European forest trees	101 million tonnes C per year
Estimated net sequestration in European forest soils	28 million tonnes C per year
Net sequestration in European forests (incl. Soils)	130 million tonnes C per year
Forest type with highest carbon stock in trees	Coniferous forest in Switzerland: 157 tonnes C per hectare

4. The contribution of forests in mitigating climate change is really minimal?

The increase rate of the carbon stock in tree biomass in temperate and boreal forests has been estimated to be 900 million tonnes C per year, and the increase rate of the carbon stock in the atmosphere to be 3200 million tonnes C per year. Without the increase in the biomass, the atmospheric CO_2 concentration would be increasing about 30% faster, i.e. 3200+900=4100 million tonnes C per year.

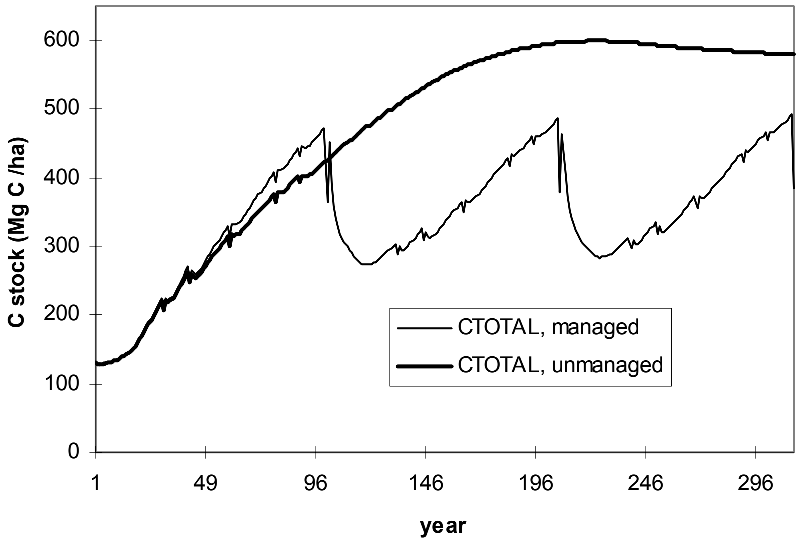

Figure 2. Comparison of the total carbon stock in managed forests (trees, products and soils) and unmanaged forest (trees and soils). A difference such as that given in this example will develop on sites where trees can gain a large standing biomass. This figure was generated with the model CO2FIX.

5. How many trees must I plant to compensate for my car use?

Suppose you drive for 40 years, 16,000 km per year. At a fuel efficiency of 11 km/litre of gasoline, and a C emission of 0.86 kg C/litre, you must compensate 50 tonnes C, or 1.25 tonnes C per year. If we assume a net sequestration of 1.4 tonne C/ha per year (average stem wood increment 3-4 m3/ha per year), you would have to plant 0.9 hectares of forest, assuming that carbon is stored at that rate and not released.

6. Is it wiser to invest in forestry projects in the tropics?

The cost efficiency of carbon sequestration in forests is a very complex matter. This is partly because in the very long term the net sequestration tends to be zero while the costs continue to exist. The costs vary from one project to another; e.g. in first afforestation projects the cheapest lands are used. The methodology of studies concerning cost efficiency varies. Some take into account the opportunity costs of the site, others do not .

However, it is clear that in the tropics the costs can be much lower than in Europe. In the tropics the costs are estimated at € 1-3 per tonne C, while in Europe the costs may rise to hundreds of € per tonne. On the other hand, the high costs in Europe are compensated by lower risks as well as other benefits generated.

7. Should I stop managing my forest, or continue the regular management?

In a managed forest, tree biomass has not (usually) reached its maximum when the final cut is carried out. The harvested wood is used for products that may last from 1 to more than 100 years. The amount of carbon in trees, products and soils in managed forests may in some cases be smaller than the amount of carbon in trees, soils and dead wood in unmanaged forests, if undisturbed (Figure 2). However, this depends on the site and the type of management executed.

However, wood products also have a carbon emission reduction effect, i.e. their production requires less energy than the production of aluminium, steel and concrete. Through the use of wood products (and thus managing the forest), a certain amount of carbon emission is prevented. In addition at the end of their life span, wood products can be used to generate energy, thus again saving fossil fuels.

8. Is it better to invest in a bio-energy short rotation plantation or in a long rotation forest type?

This depends on which carbon impacts are accounted for. When the effect of wood of reducing carbon emissions is taken into account, bio-energy plantations are more beneficial in the long term. On the other hand, long rotation forests provide other non-wood products and in some cases these other products may have a significant impact on the policy decision.

9. Do I as a forest owner benefit from all this?

Up to now forest owners have not gained anything from the carbon sequestration function of their forests. This may change, however. In the Kyoto Protocol, industrialised countries have committed themselves to reduce their carbon emissions. Also some countries have initiated a carbon emission tax. For instance in the Netherlands a carbon tax of € 19 is charged for emissions containing carbon which would be equivalent the amount of carbon in one cubic metre of wood (or 0.25 tonnes C sequestered). Where emissions are taxed, sequestration should be rewarded with the same amount.

10. What is a "Kyoto forest"?

The Kyoto Protocol states that some measures in forestry may be taken in order to achieve the agreed emissions reduction. It is yet unclear, however, how the terms "afforestation", "reforestation" and "deforestation" are going to be defined. The Protocol also mentions that further decisions are to be taken on additional activities in the agricultural and forestry sector. Also the accounting systems need to be agreed upon. For example, forests established on previously agricultural land since 1990 are accounted for. The stock change of these forests between 2008 and 2012 is then counted as a credit. On the other hand, conversion of a forest in 2008 into a shopping mall or highway would, as a consequence of this deforestation and loss of carbon be then counted as a debit.

The Clean Development Mechanism: a flexible instrument for carbon offsets from afforestation and reforestation project activities

Eveline Trines[1]

History of the Kyoto Protocol

The realization that emissions of greenhouse gases (GHG) from human activities impact on the global climate system, led to the drafting of a Framework Convention on Climate Change (FCCC) by the Intergovernmental Negotiating Committee established by the United Nations General Assembly in 1990. The FCCC was adopted on 9 May 1992 and was opened for signature at the "Earth Summit" that was held that same year in Rio de Janeiro, Brazil. The framework convention was signed by 154 countries (and the European Community) during the Summit and entered into force on 21 May 1994.

During the 1st Conference of the Parties (COP) to the FCCC that was held in 1995 the "Berlin Mandate" was adopted for new talks on strengthening the Convention. It was expected that COP3 (1997), in Kyoto, Japan would adopt a protocol that contained stronger commitments by developed country Parties for the 1st decade of the 21st century to meet the ultimate objective of the convention (source: introduction to the UNFCCC as published for the Climate Change Secretariat by UNEP's Information Unit for Conventions – no year of publication mentioned) which is to stabilize GHG concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. And so it happened that the Kyoto Protocol was formulated with country-specific quantified emission limitations and reduction commitments.

The Protocol was signed by most of the Parties to the FCCC. It contains a "double trigger" before it enters into force: 55 Parties to the Convention, together accounting for at least 55% of the total CO_2 emissions for 1990 of the Parties included in Annex I to the Convention need to deposit their instruments of ratification. Annex I lists roughly all industrialized countries: the OECD countries and the "economies in transition" or EITs (countries that are undergoing the process of transition to a market economy).

At present (2 December 2003) 120 Parties have ratified the Kyoto Protocol, together accounting for 44.2% of the total CO2 emissions for 1990. In practical terms, with the USA having withdrawn from the process, this means that the Russian Federation now holds the key to entry into force. Although Russia has expressed its commitment to ratification on the international political level at the World Summit on Sustainable Development in Johannesburg in 2002, on the domestic level a few procedures (and hurdles) still need to be taken before ratification is a fact. Currently the elections of a new president of the Russian Federation and Duma are looming on the horizon and it is not to be expected that Russian ratification is close by. Once the new Duma is in office (autumn 2004) the elections of the president of the United States will follow suit, and perhaps Russia will want to see to what extent the energy policy will alter as a consequence of the (re-)election. Therefore, it is hitherto unclear if and when the first meeting of the Parties to the Kyoto Protocol will be, if ever.

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The "Flexible Mechanisms" of the Kyoto Protocol

The first commitment period for which the "quantified emission limitations and reduction commitments" (QELRC) for Annex I Parties are listed in Annex B to the Kyoto Protocol, runs from 2008 to 2012: a 5 year period. The targets are set against the emission levels in 1990. Since the emissions have gone up significantly in most industrialized countries since 1990, the effort that countries will have to make to reach their targets is much higher, for some as high as 40% of their 2000 emissions.

For developing country Parties no quantified reduction targets have been set, even though some of these Parties will grow to be in the league of the largest emitters of Annex I right now within the next decade or two.

Three "flexible mechanisms" have been defined in the Kyoto Protocol articles 6, 12 and 17: respectively Joint Implementation (JI), the Clean Development Mechanism (CDM) and Emissions Trading.

Under JI activities can be undertaken by at least two Annex I Parties: e.g. the Netherlands financing the conversion of a coal-fired power station in the Czech Republic to using natural gas. This would be in exchange for the "emission reduction units" (ERUs) that are generated with such an activity. Such ERUs would be transferred from the accounts of the Czech Republic to the accounts of the Netherlands in exchange for the (technical and) financial assistance.

Under article 12 the CDM is defined whereby Annex I Parties can use "certified emission reductions" (CERs) generated by project activities that are undertaken in non-Annex I countries: countries without an emission reduction target. This could be projects undertaken by governments or the private sector or any other legal entity. There are a few prerequisites to the CERs, e.g.: emission reductions have to be independently verified by "designated operational entities" (DOEs): a type of licensed certifier; CERs have to be additional to emission reductions that would have taken place in the business as usual scenario; and the projects that generate the CERs have to contribute to sustainable development in the host country.

Article 17 enables the trade in emission reduction and removal units. Hence, Parties are flexible (to some degree) to generate emission reduction and removal units outside their own country or simply acquire emission reduction and removal units on an open trade market if that is more convenient. It is however, agreed that parties will undertake domestic activities to account for a *significant* part of their emission reduction target.

The land use, land-use change and forestry sector and the CDM

Since the inception of the Kyoto Protocol the subject of land use, land-use change and forestry (LULUCF) has yielded significant debate. And whether one is a protagonist or an antagonist with respect to the use of activities in the LULUCF sector to mitigate or adapt to climate change, this sector will continue to play an important role in developing countries for many reasons for many years to come, some of those reasons not related to climate change (e.g. improved livelihoods of people, natural resource planning, national conservation strategies, rehabilitation of degraded land, etc).

Due to the high degree of controversy associated with LULUCF activities under the CDM it was not possible to conclude the negotiations in its entirety to date on this subject, but it is anticipated that COP9 at the end of 2003 will conclude this phase. It has been agreed however, that for the 1st commitment period the eligible LULUCF activities are limited to afforestation and reforestation and that for possible next commitment periods new negotiations will determine the fate of LULUCF under the CDM.

Besides a limitation on the eligibility of activities, the amount of CERs that Annex I Parties can use to meet their emission reduction targets in the 1st budget period is limited to 1% of the entire amount of emissions that has been allocated to these Parties. In practice this boils down to approximately 32 Mt C yr^{-1}.[1]

The reason for this cap is twofold. First, many Parties (and some NGOs) are of the opinion that the emphasis should be on emission reductions and not on removals by sinks. Secondly, the high degree of scientific uncertainty related to the magnitude of the human-induced carbon fluxes as part of the global carbon budget. According to the Special Report on LULUCF from the Intergovernmental Panel on Climate Change (page 5, table 2, and page 32, table 1-2, IPCC, 2000) the fluxes are roughly (very roughly):

- 6 Gt C yr^{-1} due to fossil fuel combustion
- 1 Gt C yr^{-1} due to land-use change (mainly deforestation)
- uptake by oceans: 2 Gt C yr^{-1}
- increase of atmospheric CO_2 concentrations: 3 Gt C yr^{-1}

6 and 1 Gt C on one side and 2 plus 3 Gt C on the other side of the equation leaves a terrestrial uptake of approximately 2 Gt C yr^{-1}. However, rates and trends of carbon uptake in terrestrial ecosystems are quite uncertain. (IPCC, 2000)

Given that the total emission reduction target of the Kyoto Protocol of all Annex I Parties together is approximately 0.35 Gt C yr^{-1} it was considered unwise to allow unlimited crediting of LULUCF activities in the 1st budget period, which could potentially go up to as much as 2 Gt C yr^{-1} due to (natural) terrestrial uptake processes, which is approximately 5.7 times the total Kyoto target.

Rules, modalities and guidelines for afforestation and reforestation

Rules, modalities and guidelines (RMG) for the treatment of afforestation and reforestation project activities under the CDM are expected to be agreed by COP9 at the end of 2003. Having said that, a rough outline of the RMGs can already be drawn as they have been set for project activities in other sectors at COP7 and a first exchange of views has already taken place at previous negotiation session on RMGs for forestation project activities under the CDM. Only there where alterations are required will they be formulated for such LULUCF project activities.

One of the features that is likely to be equal at least – if not strengthened – is the project cycle, including 3rd party validation of the project design and verification of the emission reductions and removals by sinks. Figure 1 illustrates the various steps in the project cycle and the actors that are involved in the various steps.

[1] 1,000,000,000 tonne equals 1,000 Mt which equals 1 Gt, and 1 tC equals 3.67 tCO_2.

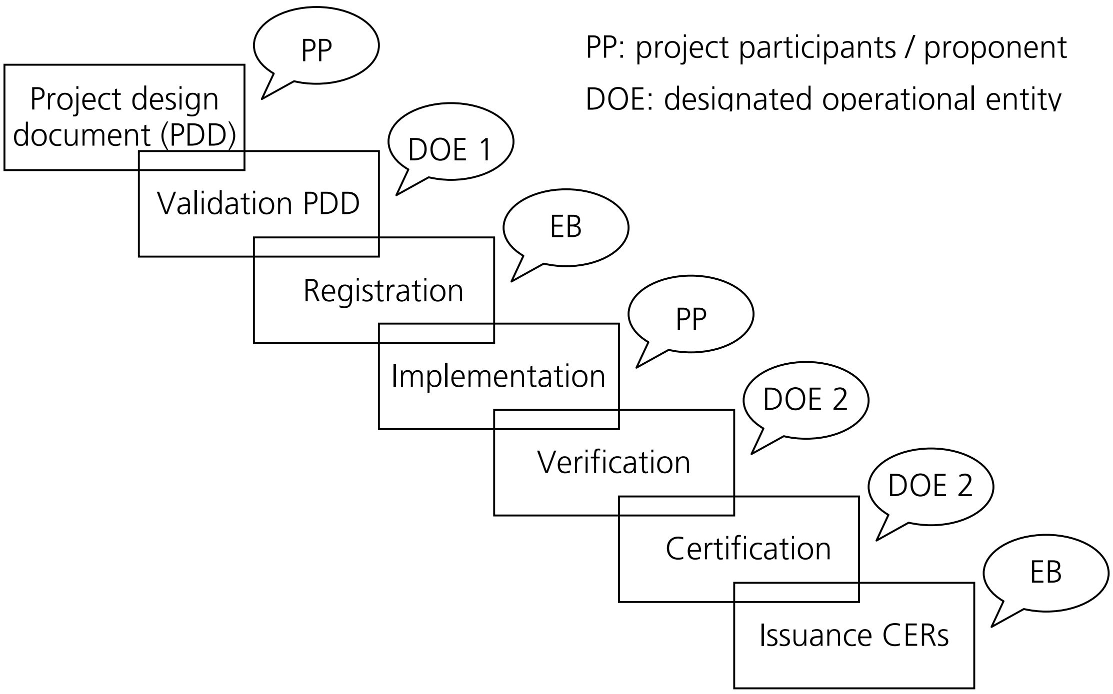

Figure 1. Steps in the Project Cycle of a CDM project.

The project proponent must prepare a project design document (PDD) that meets all requirements as set out in a special annex to the COP decision. A DOE has to validate this PDD and if it found to be adequate the project can be registered with the "Executive Board" (EB): the board that governs the CDM. The implementation phase should subsequently be consistent with the design of the project as documented in the PDD. This, and achieved emission reductions or removals by sinks can then be verified by another DOE. This DOE can certify the amount of truly additional carbon benefits of the project and if no objections occur from any stakeholder over a set period of time after announcing the certification, the EB can issue the CERs. The CERs are entered into the host country account at the registry of the EB and can – with the approval of the host country – be transferred to an acquiring Annex I Party or other legal entity.

Another feature that seems to gain common ground is the concept of "temporary CERs" or tCERs. This is a modality that is designed to take care of the reversibility of the sequestration of carbon in biomass: the CERs issued for forestation project activities will only be valid for one commitment period. If the carbon is still there after that period, the project can undergo another verification exercise and new tCERs can be issued for another 5 year period. Figure 2 illustrates the concept.

The vertical arrows indicate the time of 3rd party verification by DOEs and the numbers indicate the amount of tCERs that can be issued.

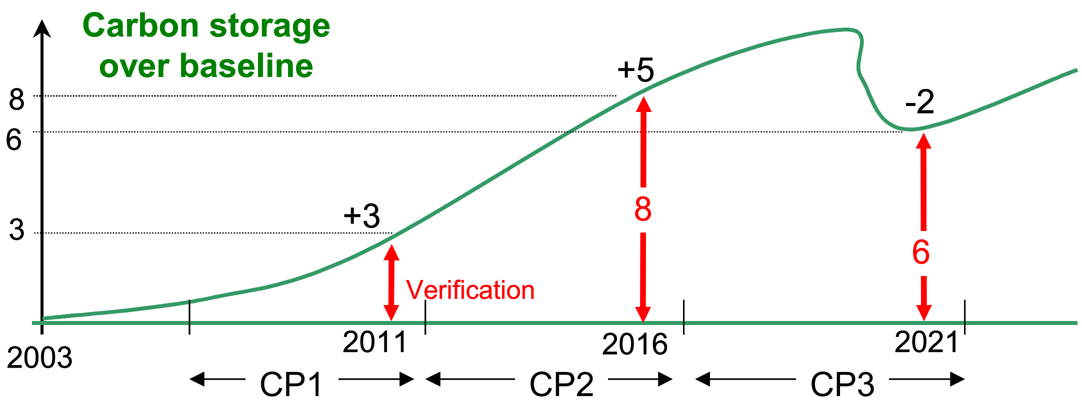

Figure 2. Carbon storage over the baseline in a forestation project.

Other features that are likely to be debated at COP9 are the requirement of a mandatory environmental impact assessment and/or social impact assessment if a 1st analysis of the impacts indicates that they may be significant. The European Union is in favour of a more rigorous assessment in such cases, while other Annex I Parties and the developing countries strongly object to any such requirement. The main reasons for objecting are that LULUCF project activities should not undergo more in-depth scrutiny compared to non-LULUCF project activities and the determination whether projects contribute to sustainable development is the prerogative of the host country and should not be prescribed in an international treaty. COP 9 will hopefully bring parties to a resolution.

Considerations for the future

As the climate change treaty only takes decisions by consensus, most decisions are negotiated hard and long. The outcome is often not entirely to the liking of everyone or even not to the liking of anybody: that's the consequence of reaching consensus in the spirit of compromise after many hours, days, and sometimes nights of negotiations. Although this might be perceived as a major disadvantage, it can also be considered a strength of the treaty: political agreement is reached and we move on with what we've got and any Party can take ownership of the result as it stands. Therefore, there is no need to go into any debate over the pro's and con's of the role of LULUCF under the CDM or the potential role it could have played. But, ….

Negotiations on the 2nd commitment period will start soon after the entry into force of the Kyoto Protocol (if ever) and policy makers are preparing their wish lists for that round. One of the items on the agenda will definitely be the eligibility of LULUCF activities under the CDM in future budget periods, and the modalities that will govern those activities. Hence, it is of utmost importance to get as much science and experience together before that time to reduce the amount of negotiating chips to the bare minimum.

Aspects to bare in mind in that run-up to the next negotiation round is the fact that CO_2 emissions (and other greenhouse gases) from deforestation contribute significantly to the emissions that the atmosphere experiences. Therefore, there is a great opportunity there to reduce rising levels of atmospheric CO_2 by reducing emissions from deforestation. Deforestation is often preceded by resource depleting use of forests. Hence, a transition from these practices to more sustainable types of forest management, with the objective to maintain continuous forest cover, is another area that should receive ample attention in the next few years.

On the quantification side this paper has already highlighted the scientific uncertainty related to the contribution of the terrestrial biosphere to the global carbon budget. This should be another area of attention. What proportion is directly human-induced and what proportion is due to indirect human-induced effects or natural effects, such as elevated CO_2 concentrations, increased nitrogen deposition, higher temperatures, changed precipitation patterns, etc.? What targets should we be looking at if the LULUCF sector in its entirety can be used by countries such as Brazil, India, or China in future commitment periods. How does that affect the global emission reduction target that we need to set to still get closer to meeting the ultimate objective of the Convention: stabilize greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system. That will be – amongst other things – our challenge for the next years to come in the context of the Climate Change Treaty.

References

IPCC, 2000: Land Use, Land-Use Change, and Forestry. A Special Report of the Intergovernmental Panel on Climate Change, edited by Robert T. Watson, Ian R. Noble, Bert Bolin, N.H.Ravindranath, David J.Verardo, and David J.Dokken. Published for the IPCC by Cambridge University Press, Cambridge, UK, 2000.

Eco-hydrological functions and threats of tropical rain forests at different spatial scales

Oscar van Dam[1]

Introduction

Many authors turned poets in an attempt to describe the tropical rain forest, a term which was first introduced by Schimper (1889). His definition of an evergreen, at least 30 m high forest, rich in thick-stemmed lianas and trees with herbaceous epiphytes is still maintained by many scientists, although it gives only a poor idea of what a tropical rain forest really is (*sensu* Richards 1996). Even for a casual visitor, the extravagant vegetation and abundant wildlife, if sighted, are obvious characteristics of a tropical rain forest. Less obvious, but equally important, is the heterogeneity of the soils that bear the forests, which form the base of entangled webs of competition for nutrients and water (Baillie 1989, Bernard-Reversat 1975). In addition, the forests themselves create a mosaic of microclimatic environments of light, temperature and humidity caused by the tumbling over of single trees (Wayne and Bazzaz 1993, Whitmore *et al.* 1993). Sustaining the wide variety of life forms in tropical rain forests are only possible through an intriguing interaction between flora, soils and fauna, mediated by the ever presence of water. Water that is present in the atmosphere and in plants and soils. Water acting as transport agent of nutrients and energy. The discipline that studies these plant-water interactions and the hydrological processes related to plant growth is called eco-hydrology (Baird and Wilby 1999). Thus, a tropical rain forest is more than just trees.

Eco-hydrological processes in a tropical rain forests are one the most important aspects of ecosystem functioning. Disturbances to these processes with almost immediately have an impact on forest structure and biodiversity, but with a strong spatial element. Because water is the key agent of eco-hydrological processes and water flows downstream, even small-scaled disturbances to the forest stand can have its influence downstream. Maintaining tropical rain forest eco-hydrological functions means maintaining landscape ecological stability for a much larger region. In this paper I will highlight the most important eco-hydrological functions of a tropical rain forests and potential threats of these functions. It is important to realize that these functions and threats operate at various spatial scales, each with its only specific characteristics and impact on the forest ecosystem and its inhabitants. Eco-hydrological functions and threats will be grouped according to these spatial scales. The paper finalizes with putting the eco-hydrological functions and threats in a broader perspective of forest management and conservation.

Spatial scales of eco-Hydrological functions

Tropical rain forest eco-hydrological functions and threats can be viewed at three spatial scales; single tree, multiple trees (forest stand) or catchment. At the smallest and medium scale, disturbances to the eco-hydrological processes in a tropical rain forest are in the form of canopy gaps. These canopy gaps can have a natural origin, caused by the death of a tree, wind throw or small-scaled landslide. Selective logging usually causes human-induced canopy gaps. These small and medium-scale disturbances are in the range of 50 – 5000 m^2. Large landslides, clear felling, land conversion and forest fires cause large-scale

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disturbances, operating at the catchment level. These catchment-scale disturbances are usually larger than 5000 m^2. Microclimatic conditions and the water balance of tropical rain forests have been described at these spatial scales: single tree (Schuttleworth *et al.* 1984, Jetten 1994), catchment (Lesack 1993, Schellekens 2000) or regional (Leopoldo *et al.* 1987). The differences in spatial scales generate a variety of approaches to field studies and water balance modelling. Water balance studies at regional scales generally work with bulk vegetation, climate and soil data and are often incorporated into a geographical information system (GIS). Catchment studies usually work with average values of the vegetation, microclimate and soil, but additional data on discharge and run-off are necessary. Single tree or forest stand studies require a detailed data set of the vegetation, microclimate at various levels inside and above the forest and a thorough analysis of the underlying soil physical parameters. Logging induces changes to the water cycle of the forest. Depending on the extent of the logging activities, the changes in the water cycle can also be expressed at various spatial scales. At a regional scale, logging can increase the frequency and duration of flooding. Clear-felling operations accompanied by slash-and-burn practice during which (at least part of) the original forest is removed, are best studied at a catchment scale (Bruijnzeel 1990, Malmer & Grip 1990). Selective logging only removes one or a few trees (10 - 20) per ha. The gaps that are created by selective logging are within a range of 10 to approximately a maximum of 5000 m^2. At this scale, the effects of logging on the microclimate and the water cycle are studied at the single tree or forest stand scale. Also, changes in the water balance on a catchment scale due to selective logging cannot be noticed (Jetten 1994). Throughout this paper, the spatial scales form a focal point for assessing the functions of and threats to tropical rain forest eco-hydrology.

The forest hydrological cycle in a nutshell

The forest hydrological cycle is shown in Figure 1, in which the left section represents the undisturbed forest and the right section a logged forest with bare soil. Although the pathways of water in both systems are similar, the magnitudes of the fluxes are different, or even absent. The major fluxes of water can be summarized as follows, where the *italic* font refers to the fluxes as listed in Table 1. *Rainfall* enters the ecosystem at the top and is intercepted by the canopy trees (*interception*), falls directly on the soil litter or soil (*throughfall*) or drains to the soil via the stems of the trees or saplings (*stem flow*). Water that remains behind on the leaves evaporates (*evaporation*). Only a small portion of the *throughfall* remains behind on the soil litter and evaporates and the remainder infiltrates into the soil (*infiltration*). Ponding will occur when rainfall or through fall intensities exceeds the infiltration rate. When the ponding capacity is exceeded and the topography of the terrain is not flat, the ponding water is transported downhill to streams and creeks as *run-off*. In flat terrain, most ponding water will form small puddles from where the water slowly drains into the unsaturated zone of the soil or evaporates. Water in the unsaturated zone of the soil either flows through to deeper soil layers (*percolation*) or is lost from the topsoil trough direct soil *evaporation*. The vegetation extracts water from the soil through their roots (uptake) after which it is transpired (*transpiration*). Water leaves the forest system through *percolation* below the rooting depth of the vegetation and will drain to the groundwater.

It is interesting to see that the amount of intercepted water by the canopy in tropical rain forest throughout the world is on average 20% of the amount of rainfall with only a relatively narrow range of 11 – 30% (based on data from Richards 1996 and van Dam 2001). This is remarkable, since interception loss within a single forest stand can have large spatial and temporal variability. Sluiter and Smit (1999) measured throughfall and interception loss in a mixed tropical rain forest in Guyana and found an average throughfall of 84.6% (standard deviation=28.13, n=727, median=84.3), but with a range of 7.3 – 295.8%! Data from various tropical rain forests also suggest some consensus on relationships between rainfall, evapotranspiration and catchment discharge loss. Approximately 50% of annual rainfall is lost via evapotranspiration and 50% through

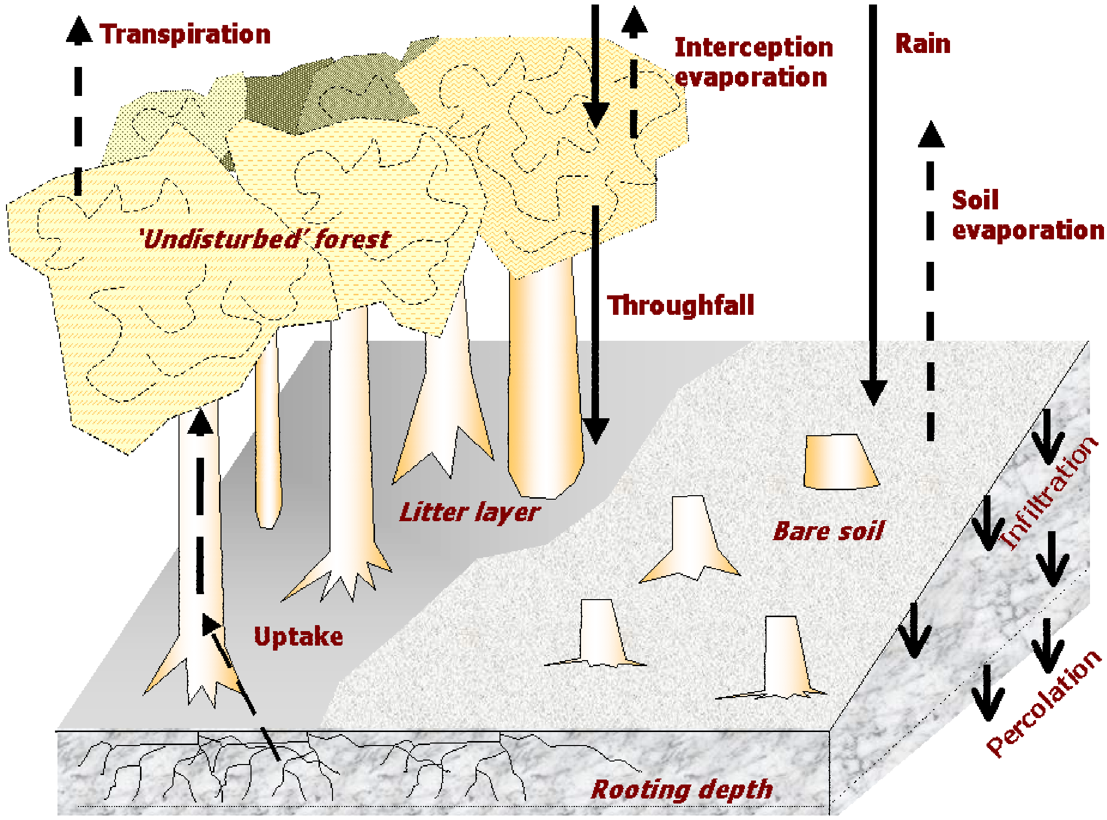

Figure 1: Hydrological cycle of an undisturbed forest and of bare soil (Source: adapted from van Dam 2001).

catchment discharge (based on data from Richards 1996 and Jetten 1996), although the ranges in these data are somewhat larger than for throughfall.

The general overview of the forest hydrological cycle as given above is true for most tropical rain forests, but not for cloud forests. The main input of water in cloud forest comes from moisture that is being stripped from passing clouds, especially in the dry season and can add up to hundreds of mm per year (Bruijnzeel and Hamilton 2000). In the absence of trees with branches loaded with epiphytes, the amount of water that is added to the ecosystem reduces substantially. Disturbances to cloud forests can have major implications on the water budget of the whole catchment where that forest is located and even influence the regional rainfall patterns.

Table 1. Hydrological cycle of undisturbed forest and disturbed forest with bare soil, in mm precipitation and as % of annual rainfall (2700 mm); Δ is the difference between the two scenarios in % of annual rainfall. (data from Mabura Hill, Guyana).

	Forest		Bare soil		Δ
Flux	mm	%	mm	%	
Rainfall	2700	100	2700	100	0
Throughfall	2230	83	2700	100	17
Stem flow	30	1	0	0	-1
Infiltration	2170	80	2605	96	16
Percolation	1090	40	1635	61	21
Runoff	90	3	95	4	1
Interception	440	16	0	0	-16
Transpiration	1060	39	0	0	-39
Evaporation	20	1	970	36	35

Source: adapted from Jetten 1994.

Eco-hydrological functions and threats of tropical rain forests

Water in any terrestrial ecosystem is vital for maintain forest ecosystem functioning, growth, exchange and transport of nutrients and soil biological processes. As such, plants survival and growth and water availability and the magnitude of hydrological processes are inextricably linked. These linkages are most evident at the scale of a single tree or a forest stand. However, at the scale of a catchment, trees and water are also co-dependent. Forests regulate catchment discharge through base-flow and peak-flow stabilization. Forests protect the downstream areas for excessive flooding or drought. Moreover, the purified groundwater under tropical rain forests can be used for drinking water. The discussion below focuses on the ecological functions and threats at different scales.

Functions and threats at the forest stand scale

The most prominent characteristic in disturbed forests, where most of the trees have been removed, is the amount of solar radiation that reaches the soil. This has striking effects on the hydrological cycle. Soil temperature increases as well as the amount of soil evaporation. The main differences between 'undisturbed' forest and bare soil are (Table 1):

1) The interception capacity of the vegetation that influences the amount of through fall and interception evaporation. Immediately after a disturbance like the creation of a logging gap, the remaining vegetation existing of small seedlings and saplings, defines the interception area in a gap. In the case of clear felling or slash-and-burn practices, only bare soil will be left behind.

2) The area of bare soil influences the amount of soil evaporation. Modelling the evaporation loss from bare soil in a logging gap in Guyana showed that soil evaporation loss could increase 25 times compared to the undisturbed forest (Jetten 1994).

3) The uptake from the soil by the vegetation and successive transpiration is less in a disturbed forest than in the undisturbed forest. The mature trees of the undisturbed forest have a larger leaf area available for transpiration and a better developed root system for

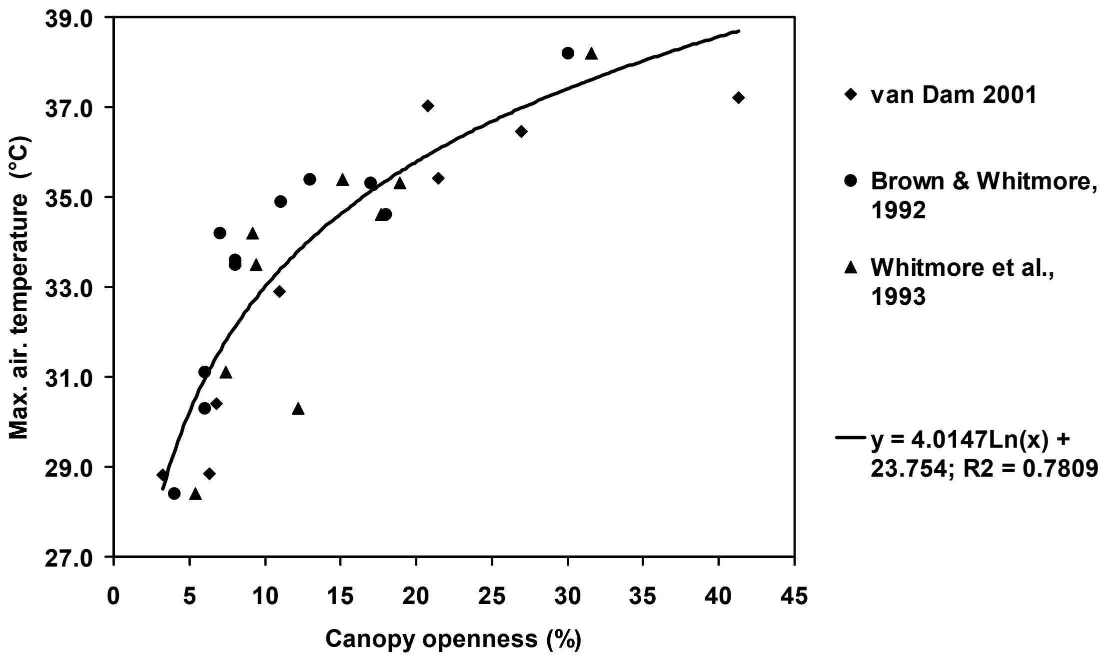

Figure 2. Relation between canopy openness and maximum air temperature in tropical rain forest gaps at 5° N (Source: van Dam 2001).

uptake than the smaller vegetation in the gaps. If all vegetation is removed, transpiration is even absent.

Besides these impacts on the water balance, other soil parameters are also affected. Soil physical parameters, like texture, bulk density, water retention, hydraulic conductivity and infiltration capacity influence the soil moisture fluxes. Soil physical characteristics can be influenced by logging due to the activity of the skidder and the dragging of the log (Henrison 1990, Jetten *et al.* 1993). These soil characteristics influence the infiltration rate, sorptivity and saturated hydraulic conductivity, as will be elaborated in detail below.

The size of a canopy gap notably influences air temperature. In experimental gaps in Guyana, the maximum air temperature in gap centres increased with increasing gap size from 28.5 °C in a 40m^2 gap to 36.6 °C in a 3200m^2 gap (van Dam 2001). Apparently, there is a logarithmic relation between gap size or canopy openness and maximum air temperature (Figure 2). With increasing gap size, the open area has in increasing influence on the dense forest surrounding the opening. Increased air temperatures were found up to 10 m into the forest of a 3200 m^2 gap (van Dam 2001). Williams-Linera (1990) reported a distance of 20m into the forest edge of a clearing, after which no changes in vapour pressure deficit (VPD) could be measured, while in the edge of a forest fragment, Kapos (1989) reported that at 60m into the forest, the clearing no longer influenced the VPD. In a later study Camargo and Kapos (1989) found no effect on VPD into the edge of the same forest fragment. These latter findings were attributed to the regenerating plants in the edge of a clearing, which created large local differences that changed over time.

Other microclimatic parameters that are directly influenced by the magnitude of a disturbance in the forest canopy are humidity, wind and soil temperature. The relative air humidity at 1 m from the forest floor is usually near saturation during night time and around 90% during mid-day in undisturbed forest. In a 3200m^2 gap, humidity can drop to 60% at mid-day, while in a 600m^2 gap humidity will not drop below 85%. While the average wind in a large opening can be 10 times higher than in the forest understory. Soil

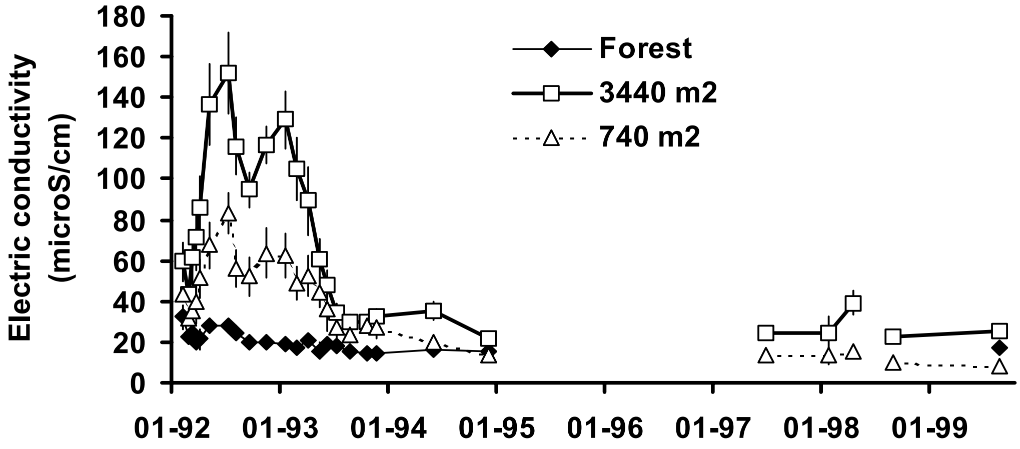

Figure 3: Course of the electric conductivity (EC in µS.cm-1) in the Forest Reserve Mabura Hill gaps and forest from February 1992 to August 1999. The error bars indicate 1 standard error, Nforest = 4, Nsmall gap = 8 and Nlarge gap = 13 (Source: Brouwer 1996, van Dam 2001).

temperature is not only influenced by gap size, but also strongly by soil cover, and thus has a large spatial variability. The highest soil temperatures are usually found in areas with bare soil, like skid trails and the lowest soil temperatures in areas with a large litter cover. These microclimatic parameters have large influences on the water balance of a site. Although soil evaporation will increase sharply in disturbed sites without forest cover, transpiration and interception evaporation losses are diminished, resulting in a 20% reduction of the evapotranspiration losses (Table 1). This decrease in evapotranspiration flux causes a 20% increase in percolation loss.

Nutrients that are dissolved in percolating soil water are lost from the ecosystem. Brouwer (1996) and van Dam (2001) showed that with increasing gap size the amount of dissolved chemicals in percolating water increases. The bulk of the nutrients are lost within 1.5 years after logging, but leaching continues even after 7 years in large openings (3440 m^2) (Figure 3). The fate of cation leaching is strongly linked to the release of nitrogen from organic debris. Organically bound nitrogen is released as NH_4 (ammonification). Plants can take up NH4, but most NH_4 will be transformed to NO_3 (nitrification). Hereby, two protons are released, which increases the acidity of the soil and therewith the solubility of Al. Exchangeable bases in the soil are displaced by either Al or the protons and together with the nutrients that were released during decomposition are lost from the soil exchange complex. Unlike NH_4, NO_3 is mobile in the soil and will be transported with percolating water to deeper soil layers. NO_3 is accompanied by a cation, which increases the leaching of these bases (Van Breemen et al. 1983, Cahn et al. 1993, but see Brouwer 1996). The intensity of selective logging, expressed as gap size, can therefore have a strong effect on the eco-hydrological processes that influence the amount of nutrients that are leached from the gap (Brouwer and Riezebos 1998, Parker 1985).

Functions and threats at the catchment scale

Eco-hydrological functions of tropical rain forests at the catchment scale are: 1) regulating catchment discharge by limiting peak flow and maintaining base flow, 2) reducing downstream flooding magnitude and frequency and the occurrence of drought, 3) optimising rainfall infiltration through root-soil interaction, presence of a litter layer and protecting the soil against compaction, 4) minimizing the amount of surface run-off and therewith soil erosion, and 5) minimizing suspended sediment transport in catchment creeks and thus sedimentations problems downstream. Disturbances to forests at the catchment scale have a direct impact on the above-mentioned functions. Patterns of eco-hydrological functioning at the catchment scale can be somewhat different, depending on the magnitude of the disturbance. For example, the impact of selective logging practices at the outflow of a catchment can be negligible small, both in terms of water discharge (Jetten 1994) and dissolved nutrients (Brouwer 1996). A logging experiment with a logging intensity of 21 $m^3.ha^{-1}$, logging gaps evenly distributed and not exceeding 500 m^2, neither positioned on steep slopes, did not reveal any significant changes in discharge (Jetten 1994). However, logging caused a rather short lived increase of K concentrations in discharge water, a delayed increase in Na concentrations and a relatively clear increase of NO_3 concentrations (Brouwer 1996). Nevertheless, the remaining vegetation, especially along the creeks operated as buffer for most of the nutrient loss.

Clear felling or slash-and-burn practices have major implications for discharge, soil loss and the nutrient cycle (Malmer 1996, Bruijnzeel 1990). A forest cover protects the drainage area for excessive water discharge, because of the buffering capacity of the vegetation. The vegetation intercepts rainwater and extracts soil water. The litter layer under forests and the abundant root mat in the topsoil protects the soil against overland flow and thus erosion. Even in undisturbed forest ecosystems and tree plantations with a well-developed litter layer some erosion may occur. Reported rates in the early literature (summarized by Wiersum, 1984) range from 0.02 to a maximum of 6.2 $t.ha^{-1}.year^{-1}$ (median values of 0.3 and 0.6 t ha^{-1} $year^{-1}$ for forests and plantations, respectively; $n = 47$). Discharge characteristics in forested catchment have in general a steady base-flow, protecting the downstream area against droughts, and reduced peak-flow during and shortly after a rainstorm (Fig. 4). By contrast, unprotected soil will have more direct run-off to streams and thus create large peak-flows during and after rainstorms, thereby generating potential flooding risks downstream. During dry periods, since most of the rainwater has not been contained in the soil, base-flow will be low and drought related problems down stream might occur. These changes are the result of logging activities that changed topsoil properties (Malmer 1996, Jetten 1994) due to the use of heavy machinery. Skidders or tractor crawlers compact the soil, or remove the soil completely, thereby decreasing the infiltration capacity and porosity and increasing bulk density. All these changes have important implications for the amount of water that will infiltrate in the soil during a rainstorm and the total amount of water that can be stored. This disturbance in clear-felled forest can be considerably. Malmer (1996) reported 24% of a catchment with disturbed topsoil properties due to the use of crawler tractors. The use of these machines not only disturbs the topsoil, often it completely removes the top layer of the soil. Another problem lies in the fact that this type of disturbance has a very long recovery time, if the disturbances to the topsoil properties can recover at all. Malmer (1996) reported almost 1200 mm more runoff in three years in a logged catchment compared to a control catchment, which resulted in a significant increase of suspended sediment during storm flows for at least one year after logging. But not only the soil itself is lost, the nutrients that are contained in the soil organic matter or bounded to the soil matrix are lost as well (Malmer and Grip 1994), reducing the regeneration potential of the future forest because of lack of nutrients. For example, the paired catchment experiment of Malmer (1996) showed that 39.9 kg $N.ha^{-1}$ was lost within 33 months after felling and burning, while by comparison 118 kg $N.ha^{-1}$ was removed in the harvest itself.

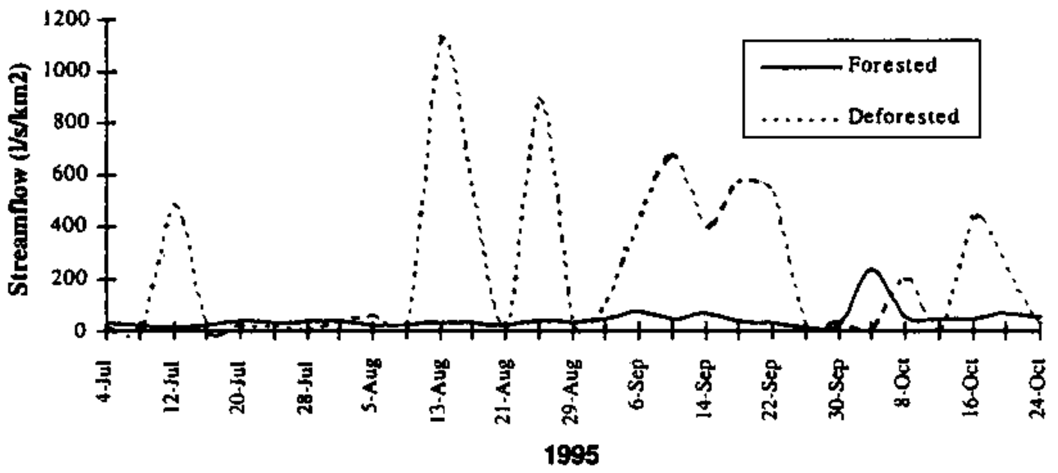

Figure 4: Catchment response after forest clearing, Honduras (Source: Bruijnzeel and Hamilton 2000)

Functions and threats at the regional scale

Cloud formation and rainfall generation depend on the concentration, the size and the type of Cloud Condensation Nuclei (CCN). In the predominantly clear atmosphere of the humid tropics, cloud formation processes are most sensitive at low amounts of CCN. Above tropical rain forest, typical CCN value is less than 300 cm^{-3}. These CCN's produce low, warm, precipitating clouds, which rain out while clouds are growing. Clear felling and burning influences the regional or mesoclimate in two ways. First, the lack of forest cover with large LAI values give rise to low amounts of evapotranspiration, thereby decreasing the amount of water vapour in the air. As such, logging directly influences the amount of moisture in the air that could possibly form clouds and produce rain. On the other hand, due to logging and subsequent burning the amount of CCN's increases and may be 10 times higher than normal. The large number of CCN gives smaller droplets in the air and produces more stable clouds with an enhanced lifetime. These clouds are well-developed but non-precipitating. Although the impact of burning is relatively short-lived, the reduced evapotranspiration and resulting reduced rainfall can last much longer. Clear felling also influences the overall air and soil temperature. O'Brian (2000) reported an increase of 1 to 4°C yearly average air temperature, which was not only the result of an increased solar radiation, but also an increased length of the dry season. These changes in mesoclimate result in an overall drier landscape in which the risks of the occurrence of a fires and the potential damage of fire is greatly enhanced.

The need for maintaining eco-hydrological functions of tropical rain forests

As outlined above, the eco-hydrological roles of tropical rain forests are versatile. The plant-water interactions ensure ecosystem stability at various spatial scales. Water and nutrient exchange between plants and soil are important processes in survival and growth of tropical rain forest species and biodiversity. Tropical soils are among the poorest soils in the world and forestry is often the only sustainable land use. Tropical forest products, whether timber or non-timber, should be harvested in a sustainable manner to ensure an optimised functioning of eco-hydrological processes. These processes are at the base of ecosystem stability. If eco-hydrological processes are disturbed, depending on the scale of disturbance, nutrients will be permanently lost, microclimate changes will occur that limit the establishment and survival of plant species, soil stability will be reduced, down-stream water will be contaminated with sediments, reduced availability of potable groundwater, increased occurrences of droughts and flooding and regional climate changes will be noticeable with increased fire risk. Obviously, a proper forest management is needed to prevent these threats of forest ecosystem functioning of occurring.

Actions that must be undertaken to maintain eco-hydrological functions are:

1) Minimizing large-scale deforestation. Sustainable and controlled selective logging is preferred, but in areas where clear-cutting is common practice and economically valuable, strict limitations to the size and location of the areas should be adhered to. An elaborate discussion on this is beyond the scope of this paper.

2) Limit access to forest areas. As soon as forests are being used for economic reasons, roads are being constructed to gain access. These roads are not only being used by the people that created them, but others will follow, including illegal loggers, poachers or legal hunters, miners is the area is rich in minerals and travellers in general. Roadside damage to the forest and thus eco-hydrological functions is a common characteristic, which includes the increased usage of the road itself and the need to maintain it. Unfortunately, carelessness results only too often in fires and environmental wastes of oil products. Closing the roads minimizes this threat.

3) Increasing awareness of the high value of undisturbed forest. This is an important aspect for maintaining forest eco-hydrological functions and it serves more than just that purpose. The multi-functional use of tropical rain forests should be explained, which could include eco-tourism, sustainable forest management, harvesting of non-timber forest products, use of the forest for carbon sequestration, potential medicinal use of flora and fauna, highly valued drinking water extraction and the regulating function for downstream flooding, sedimentation and droughts.

The three points of action for maintaining eco-hydrological functions of tropical rain forests must of coarse be seen a much broader context for tropical rain forest ecosystem management and conservation. This summation mainly serves to stress the strong linkages of eco-hydrological functions with all aspects of the tropical rain forest ecosystem. Minimizing potential threats to these functions is automatically beneficial for the whole forest community, flora, fauna and man.

References

Baillie, I.C., 1989. Soil characteristics and classification in relation to the mineral nutrition of tropical wooded ecosystems. *In: Mineral Nutrients in Tropical Forest and Savanna Ecosystems, J. Proctor (ed.). Blackwell Scientific Publications.*

Baird, A.J. and R.L. Wilby, 1999. Eco-Hydrology: Plants and Water in Terrestrial and Aquatic environments. *Routledge London.*

Bernhard-Reversat, F., 1975. Recherches sur l'ecosysteme de la foret subequatoriale de basse Cote-d'Ivoire. VI. Les cycles des macroelements. *Terre Vie, 29:229-254.*

Brouwer, L.C., 1996. Nutrient cycling in pristine and logged tropical rain forest. A study in Guyana. Thesis, Dept. Of Physical Geography, Utrecht University, the Netherlands. Tropenbos-Guyana Series 1. Tropenbos-Guyana Programme, Georgetown, Guyana.

Brouwer, L.C. and H.Th. Riezebos, 1998. Nutrient dynamics in intact and logged tropical rain forest in Guyana. In: Schultz and Ruhiyat. Proc. Soils of tropical forest ecosystems. Characteristics, ecology and management. Pp73-86. Springer-Verlag Berlin.

Bruijnzeel, L.A., 1990. Hydrology of moist tropical forests and effects of conversion a state of knowledge review *Free University Amsterdam, UNESCO int. hydr. progr., Humid Tropics Programme.*

Bruijnzeel, L.A. and L.S. Hamilton, 2000. Decision time for cloud forests. *UNESCO. IHP Humid Tropics Programme Series No. 13.*

Cahn, M.D., Bouldin, D. R., Cravo, M.S. and Bowen, W.T., 1993. Cation and Nitrate Leaching in an Oxisol of the Brazilian Amazon. *Agronomy Journal 85:334-340.*

Camargo, J.L.C. and V. Kapos, 1995. Complex edge effects on soil moisture and microclimate in central Amazonian forests. *J. Trop. Ecol. 11(2):205-220*

Hendrison, J., 1990. Damage-controled logging in managed tropical rain forest in Suriname. *Thesis, Agricultural University Wageningen, the Netherlands.*

Jetten, V.G., 1994. Modelling the effects of logging on the water balance of a tropical rain forest A study in Guyana Thesis. Dept. Of Physical Geography, University of Utrecht. Tropenbos Series 6, The Tropenbos Foundation, Wageningen, the Netherlands.

Jetten, V.G., H.Th. Riezebos, F. Hoefsloot and J. van Rossum, 1993. Spatial variability of infiltration and related properties of tropical soils *Earth Surface Processes and Landforms 18:477-488, John Wiley & Sons.*

Kapos, V., 1989. Effects of isolation on the water status of forest patches in the Brazilian Amazon. *J. Trop. Ecol 5:173-185.*

Leopoldo, P.R., W. Franken, E. Salati and M.N. Ribeiro, 1987. Towards a water balance in the central Amazonian region. *Experientia 43:222-233.*

Lesack, L.F.W., 1993. Water balance and hydrologic characteristics of a rain forest catchment in the central Amazon basin. *Water Resources Research 29:759-773.*

Malmer, A, 1993. Dynamics of hydrology and nutrients losses as response to establishment of forest plantation. A case study on tropical forest land in Sabah, Malaysia. *PhD thesis, Swedish University of Agricultural Sciences, Dept. of Forest Ecology, Umeå.*

Malmer, A., 1996. Hydrological effects and nutrient losses of forest plantation establishment on tropical rainforest land in Sabah, Malaysia. *J. Hydrol. 174(1-2): 129-148.*

Malmer, A. and Grip, H., 1990. Soil disturbance and loss of infiltrability caused by mechanised and manual extraction of tropical rainforest in Sabah, Malaysia. *Forest ecology and management 38:1-12.*

O'Brian, K., 2000. Upscaling tropical deforestation: Implications for climate change. *Climatic Change 44:311-329.*

Parker, G.G., 1985. The effect of disturbance on water and solute budgets of hillslope tropical rainforest in North-eastern Costa Rica. *Thesis, Athens, Georgia.*

Richards, P.W., 1996. The tropical rainforest. (Sec. Ed.) *Cambridge University Press.*

Schellekens, J., 2000. Hydrological processes in a humid tropical rain forest: a combined experimental and modelling approach. *Thesis Vrije Universiteit Amsterdam, the Netherlands.*

Schimper, A.W.F, 1898. Pflanzengeographie auf Physiologischer Grundlage. *Sec. Ed., G. Fischer, Jena.*

Shuttleworth, W. J., H.C. Gash, C.R. Lloyd, C.J. Moore, J. Roberts, A.De O. Marques Filho, G. Fisch, V. De Paula Silva Filho, M. De Nazaré Góes Ribeiro, L.C.B. Molion, L.D. De Abreu Sá, J.C.A. Nobre, O.M.R. Cabral, S.R. Patel and J.C. De Moraes, 1984. a Eddy correlation measurements of energy partition for Amazonian forest. *Quart. Jour. of the Roy. Meteo. Soc. 110:1143-1162.*

Sluiter, R. and N. Smit, 1999. Gap study in mixed tropical rainforest. Guyana 1998 MSc thesis. Dept. of Physical Geography, Utrecht University. Tropenbos Guyana-Programme, Georgetown, Guyana.

van Breemen, N., Mulder J. and Driscoll, C.T., 1983. Acidification and alkalinization of soils. *Plant and Soil 75:283-308.*

Van Dam, O., 2001. Forest filled with gaps. Effects of gap size on water and nutrient cycling in tropical rain forest. A study in Guyana. *PhD thesis Utrecht University. Tropenbos-Guyana Series 10, the Tropenbos-Guyana Programme, Georgetown, Guyana.*

Wayne, P.M. and F.A. Bazzaz, 1993. Morning vs afternoon sun patches in experimental forest gaps: Consequences of temporal incongruency of resources to birch regeneration. *Oecol. 94, pp 235-243.*

Whitmore, T.C., N.D. Brown, M.D. Swaine, D. Kennedy, C.I. Goodwin-Bailey and W.-K. Gong, 1993. Use of Hemispherical photographs in forest ecology: measurement of gap size and radiation totals in a Bornean tropical rain forest. *J. Trop. Ecol. 9:131-151.*

Wiersum, K.F. (ed), 1984. Strategies and designs for afforestation, reforestation and tree planting. Proceedings International symposium on the occasion of 100 years of forestry education and research in the Netherlands (19) Wageningen 1983 - 23-09-1983, Pudoc Wageningen the Netherlands.

Williams-Linera, G., 1990. Vegetation structure and environmental conditions of foret edges in Pananma. *J. Ecol. 78:356-373.*

Getting a piece of the pie: profits of environmental services for local actors[1]

Pita A. Verweij[2]

Introduction

Forest goods and services, such as carbon storage, regulation of the hydrological regime at different levels, soil protection, habitat for species and people, and landscape amenities, represent potential values. Deforestation and degradation of forests continue at a massive scale, in spite of the world-wide recognition of the potential values of forest products and services. Diffuse ownership and corruption often enable these activities. Underlying causes include poverty, population growth, market demand for forest products, and macroeconomic policies. A better insight in the multiple values of forest services may contribute to their maintenance by increasing investments into sustainable forest management and conservation. It is crucial to generate more financial revenues for sustainable forest management, in order to increase its competitiveness with alternative, non-sustainable land uses. Local communities must be given sufficient (financial) interest in managing the forests in a sustainable manner.

This is in brief the problem that innovative financing mechanisms seek to overcome. Innovative financing mechanisms can be a powerful instrument in support of sustainable forest management and in contributing to sustainable livelihoods of local forest-dependent communities, provided that criteria of social, ecological and economic sustainability are accounted for. This concerns the interests of both present and future generations. Financing mechanisms must be developed and implemented in combination with enforcement of policies that take into account the interests of forest dependent people and future generations. This paper presents some findings on innovative financing mechanisms (IFMs) for sustainable forest management in relation to local livelihoods. The question is, how local actors can benefit from profits of forest environmental services. The main focus is on carbon and hydrological services, provided by forests.

An innovative financing mechanism (IFM) is defined as an institutional arrangement that results in the transfer of new or increased financial resources from those willing to pay for sustainably produced goods and/or forest ecological services, to those willing to provide these goods and services in turn (Verweij, 2002). The overall goal of developing IFMs is to help forest managers add financial value to their forests based on the benefits they generate, thus increasing the incentives to conserve and restore forests. There are two different ways in which an IFM can be effective. Firstly, an IFM can 'capture' the non-market values of ecological services through some kind of economic transaction, thus creating new markets. On the other hand, the IFM can charge on the non-marketed portion of people's willingness to pay for forest goods, thereby increasing the market value of forest goods that are produced in a sustainable way.

The following factors are major constraints to the development of effective financing mechanisms:

- An important bottleneck to the development of IFMs in support of sustainable forest management is diffuse ownership. Forest use and property rights are often not clearly

[1] Parts of this paper are based on Verweij (2002) and Verweij (2003).

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defined. On the demand side, actors who pay for forest goods and services preferably deal with identified and legitimate parties. Land users without land titles often face difficulties in having access to markets for environmental services.

- Long learning trajectories are required to develop new marketable products and services. This is partly due to the complexity of institutional arrangements needed to internalise real costs and benefits into market prices.
- The enabling conditions to generate demand are frequently insufficient or lacking. Markets often originate from the establishment of (international) regulations and agreements. The negotiations in relation to the Kyoto protocol for example, have generated a demand for certified CO_2 emission reductions.

Local communities often do not participate in the development and implementation of IFMs. In case of payments for forest environmental services to other actors (e.g. government entities) in turn for restriction of access to forest resources, this can lead to impoverishment, or to intensified use of forest resources elsewhere.

IFMs aimed at capturing the value of specific services are usually associated to certain types of institutional arrangements (Verweij, 2002). Therefore, the evaluation of different institutional arrangements can best be done for each category of product, service, or a combination of services. Carbon sequestration and hydrological services are among the most promising categories according to the level of the benefits raised (Pearce & Pearce, 2001).

Financing carbon sequestration

The Clean Development Mechanism (CDM) as defined in the Framework Convention on Climate Change and the Kyoto Protocol provides a possibility to capture the economic value of carbon sequestration services. Before the Conference of Parties in Bonn (COP6 bis), CDM was expected to become a substantial source of finance for tropical forest management. During the pilot phase to the implementation of CDM, a wide range of activities was funded, including forest conservation, restoration and management. The resulting carbon sequestration costs are highly variable, but far below any theoretical assessment of its economic value. Concrete carbon sequestration prices depend on the type of forest ecosystem, the proposed activities, and the methods used. Most of the carbon sequestration activities financed during the CDM pilot phase do not result in clear benefits for local communities. There are a few exceptions, however.

Miranda et al. (2002), describe the successful example of a bilateral agreement between Norway and Costa Rica in support of a major carbon sink project including the rehabilitation of a degraded watershed. They stress the importance of the social dimension of this kind of project, in terms of providing multiple sources of income to communities inside the project watershed and outside. Farmers are successfully involved in reforestation and forest protection activities and receive hectare-based compensation in turn. Local communities also participated in environmental education and other training activities as part of the implementation of the project.

May (2002) presents the experiences of the Brazilian NGO Pró-Natura with carbon sink projects in Mato Grosso. Pró-Natura has been engaged for more than a decade in this region in agroforestry demonstration and institutional capacity building in support of local producer associations and municipal governments. The French car manufacturer Peugeot-Citroën is presently investing into a major commercial carbon sink in Mato Grosso, in co-operation with Pró-Natura. This involves reforestation of 5,000 ha of degraded pasture and testing of large-scale reforestation with native species. Not only did these activities promote efforts to restore degraded sites, but also led to an increase in the perceived value of the standing forest by local communities.

The Bonn Conference however set important restrictions to the application potential of CDM in relation to tropical forests (Skutsch, 2002). CDM is limited to afforestation and reforestation projects, at least during the first commitment period (2008-2012). Currently, there is no legal framework operational or under construction that stimulates the development of financing mechanisms at global level that are explicitly aimed at capturing the value of carbon sequestration of natural forests.

Financing water services

Together with carbon sequestration benefits, hydrological services are among the most valuable of the wide range of ecosystem services provided by forests. As in other parts of the world, the majority of the population in tropical countries lives downstream of forested watersheds and therefore suffers from the effects of watershed degradation. Forest cover plays an important role in the maintenance of water quality and a stable water flow. Forest ecosystems slow the rate of run-off, resulting in decreased impacts of flooding and increased minimum stream flows during dry seasons. In comparison to agricultural land use systems, forests reduce soil erosion and sedimentation of waterways. Investments in sustainable watershed management may be substantially cheaper than investments in new water supply and treatment facilities (Johnson et al., 2001). In Vietnam, local communities acknowledge the major function of forest cover as a windbreak against natural disasters such as typhoons.

Of the wide variety of hydrological services, IFMs have been designed principally aimed at maintaining water quality and water flow for drinking water and electricity generation. Traditionally, funds for watershed management and protection have come from public revenues and are not based on the actual value of the water service provided by these areas. Due to ineffective tax systems and economic crises, governments are often facing drastic shortfalls in revenues. Moreover, subsidies and regulations aimed at the promotion of soil conservation techniques on private lands do not seem to be very effective. Therefore, there is a growing tendency to privatise public water and hydroelectric utilities.

At watershed level, there are examples of local-level private entities that have developed mechanisms to ensure water quality and a stable water flow. The principal activities benefited by these mechanisms are the provision of drinking water, hydroelectric power generation and irrigated agriculture. In The Philippines for example, a hydroelectric power company successfully provides incentives to local communities for reforestation activities (Mero, 2002).

An example of a private-public financing mechanism is the Working for Water program in South Africa. This initiative is aimed at the removal of alien tree species and restoration of native vegetation (Hope et al., 2002). The activities result in improved water supply at a fraction of the cost of water delivered through diversion or reservoir projects.

In continuation, some examples of payment systems for forest hydrological services in different stages of implementation are given for Latin America.

- Voluntary agreements and water fees, Costa Rica

Regarding payments for hydrological services, two types of voluntary agreements have been developed in Costa Rica (Reyes et al., 2002). On one hand, private agreements are established between NGOs and private companies. An example is the agreement since

1998 between the hydropower company La Esperanza and the conservation organisation Asociación Conservacionista Monteverde, where the company pays 10 US$/ha per year to the NGO for hydrological services of forests in the Peñas Blancas watershed. On the other hand, the National Forestry Finance Fund FONAFIFO established agreements with hydroelectric power companies. Payments are in the range of 10 US$/ha up to 47 US$/ha per year for the more recent agreements, while the period of commitment of the agreements varies from 5 to 10 years. Over the last few years, financial compensation has also been provided to local forest managers who do not possess official land titles.

There is another example from Costa Rica, which is not based on an institutional agreement with FONAFIFO or the Ministry of Environment. In 2000, the drinking water company of Heredia established a hydrological fee in three minor watersheds in the Central Valley of Costa Rica. The company collects a small fee on consumed water, to be reinvested in forest conservation and reforestation within the same region.

- Water funds, Ecuador

In Ecuador, the municipal water companies of Quito and Pimampiro created water funds by charging levies on drinking water (Echavarría & Granizo, 2000). In the case of Quito, the revenues of the water tax would have to be complemented by voluntary payments of major agricultural and industrial water consumers, and be invested into nature conservation activities in montane forests and páramo. In Pimampiro, an environmental NGO promoted the set-up of the municipal water fund, which results into direct payments to forest owners. International donors provided seed money to both water funds. The municipal water company of Cuenca, also in Ecuador, invested revenues from water into the purchase of upstream nature areas for strict conservation purposes (Hofstede & Alban, 2002).

- Watershed protection, Colombia

In Colombia, regulation provides for the payment of hydrological services at the regional level. Resources from a tax on hydroelectric energy (3% of the gross revenues) for example, are transferred to Regional Development Co-operations. These resources should be reinvested in the watersheds where the energy is generated. In practice however, this is not done efficiently due to the fact that management plans for the watersheds concerned are often lacking. Furthermore, an important share of the funds is spend on administrative costs or is channelled to other destinations.

In the Cauca Valley of Colombia, water users united in private entities of Associations of Water Users, decided to pay voluntarily an additional fee to the one paid to the Regional Development Co-operation (Echavarría, 2000; Perrot-Maître & Davis, 2001). An important rationale for this development was the fact that the agro-industrial water users in the Cauca valley had been facing continuous water shortages. The additional fee is collected at (sub-) watershed level and is reinvested into watershed management activities in the same area, which seems to work more effectively. In 2000, there were 12 Associations of Water Users, covering an area of more than 1 million hectares and benefiting approximately 97.000 families in upper watersheds (Echavarría, 2000).

- Sierra de las Minas, Guatemala

In Guatemala, the NGO Defensores de la Naturaleza is embarking on a major conservation and development program for the region of Sierra de las Minas. The main objective is the design and implementation of a water fund. The financial mechanism will be used to charge large (agro)industries for water use downstream, whereas the funds will be reinvested into upstream conservation of forest ecosystems within the national park of Sierra de las Minas and its buffer zone.

The Sierra de las Minas, a mountain chain 15-30 km wide extending 130 km, is important as a generator of orographic rainfall for the adjacent valleys. It gives rise to 62 permanent streams. The core area of the national park carrying the same name contains a large extension of cloud forest, and premontane and lower montane forests. In the north, farmers depend on the intercepted water for crop production and cattle farming. Smallholders are invading the steep forested upper slopes after having been displaced from lower elevations by large farms. To the south, both smallholders and commercial agro-industrial companies (e.g. melon, mango, grapes) in the valleys depend on irrigation water originating from the sierra. The buffer zone of the national park includes a hydro-electrical power plant. The dam and water reservoir are surrounded by native pine forests, in state of degradation due to impacts of frequent fires and grazing. Also light industry depends on a steady water supply, including a paper manufacturer, Coca Cola, Pepsi Cola, a beer brewery and a mineral water company. There is a striking contrast between the massive water consumption of the downstream (agro-) industries and the upstream local communities, who are sometimes still lacking the provision of drinking water.

The need for establishment of a water fund is also underlined by the fact that water flow has been reduced significantly during the last 10 years and the water table has dropped, probably because of a combination of the loss of natural vegetation and excessive water extraction.

The analysis of the case of the Sierra de las Minas region forms part of a PhD project on valuation of hydrological services and institutional aspects of (emerging) water markets, which receives support from the European Union. The NGO Defensores de la Naturaleza also supports this research, as it will provide an important scientific basis for the design of the financing mechanism. The research is expected to result in recommendations in relation to the institutional set-up of the mechanism and practical tools in support of the negotiations among governmental organisations, private companies that consume water, and forest managers providing hydrological services.

Financing a combination of services

In Costa Rica, the legal framework endorses the concept of paying for a combination of environmental services provided by natural forests and plantations. The latest Forest Law, No. 7575, supports payments to owners of forestland, or land in the process of reforestation, to compensate for the environmental services provided by their activities to society in general. Benefits in terms of regulation of hydrological cycles, scenic beauty, carbon sequestration, and biodiversity conservation are acknowledged. The implementation of Forest Law 7575 is done through a program of Payment of Environmental Services, which results in payments by the National Forestry Finance Fund FONAFIFO to private landowners. These payments represent monetary compensation to forest owners either for maintenance of primary forest, establishment of forestry plantations, or forest management. FONAFIFO counts with funds provided by the national government. Of the taxes collected from fossil fuels, a fixed percentage goes to FONAFIFO, whereas the voluntary agreements mentioned above provide additional financial resources.

From 1997 to end of 2002, the Payment for Environmental Services program included 310,134 hectares (6 % of Costa Rican territory). Of these, 261,500 ha correspond to forest protection, 21,360 ha to forestry plantations and 27,274 ha to forest management.

Contributions of IFMs to sustainable livelihoods

There is a broad spectrum of Innovative Financing Mechanisms, representing market-based instruments or mechanisms that create new markets. These have a strong potential to increase financial flows towards sustainable forest management. At the same time, it is important to notice that besides policy failures, market failures represent another important cause of forest destruction. Markets can be malfunctioning, distorted by subsidies or simply do not exist. Actors other than local forest owners and users may benefit mainly from the profits of forest environmental services.

How important are the described payment systems for forest environmental services in terms of their potential to contribute to sustainable livelihoods and poverty alleviation? Do target areas for payment of environmental services overlap with poverty? Do poor farmers participate in IFMs, and what are direct and indirect benefits for the rural poor? The use of market instruments to guarantee the maintenance of forest ecological services raises important questions on issues of equity. On the basis of a large number of literature cases, Landell-Mills & Porras (2002) conclude that market mechanisms may or may not contribute to poverty alleviation: there is no straightforward answer.

The situation regarding water services is relatively favourable in terms of participation by the rural poor and benefits they can receive. The explanation lies in the fact that water services in a specific region cannot be delivered by anybody: land users and landowners in upper watersheds necessarily provide these services. These land users are often poor. In Costa Rica, people without land titles have also received payments for the provision of hydrological services in the case of voluntary agreements with private companies. In several cases (e.g. in Ecuador), the implementation of IFMs in relation to hydrological services has led to the definition of land use rights and the assignation of land titles to formerly landless farmers. Thus, payment systems hydrological services have a clear potential in transferring financial resources to the rural poor.

This does not hold true for payment systems for carbon services. Those actors, who provide cheaper carbon services, have an advantage over others. Smallholders and farmers without land titles are dealing with higher transaction costs. Therefore, they are likely to supply carbon services at a higher cost than large landowners.

Institutional aspects

In Costa Rica, a national legal framework has promoted the implementation of payment systems for environmental services. In Colombia, regulation in support of the payment of hydrological services exists at the regional level.

Experiences from several other countries show however, that a legal framework either at national or regional scale is not indispensable. In Guatemala, a pilot programme of support to forest conservation activities (PPAFD) started in 2000, where landless farmers living at elevations above 1500 m will receive revenues of up to 55 US$/ha per year in a number of watersheds. Also in Nicaragua, Honduras and El Salvador, pilot projects of 'payments for hydrological services (PHS) started at municipal level in the framework of the PASOLAC programme ('Programa para la Agricultura Sostenible en Laderas de América Central'; Pérez, 2003). International donors support these pilot projects of payments for environmental services in Central America.

In comparison to other regions in the world, in Latin America a large number of payment systems for environmental services have been developed. Most of the systems aim at the maintenance of hydrological services. Although these IFMs correspond to institutional set-ups at a range of scale levels varying from national to sub-watershed level, they are mostly operational at the lower scale levels of watersheds and sub-watersheds. The complexity of the institutional set-ups decreases with lower scale levels and the administrative costs involved are correspondingly lower.

NGOs and private companies such as hydropower and drinking water companies have proved to play an important role in the set-up and implementation of effective payment systems for hydrological services at (sub-) watershed level. Whether legal frameworks are absent or in place, in many Latin American countries NGOs, private companies and international donors are increasingly taking their responsibility in creating proper mechanisms to transfer financial revenues to local communities for the provision of forest environmental services.

Recommendations

In order to increase the potential of financing mechanisms to contribute to sustainable livelihoods from forests, the following recommendations should in particular be taken into account:

- Integrate property and use rights in relation to forest resources in a proper legal framework as a condition to the implementation of IFMs aimed at sustainable forest management and conservation.
- Develop financing mechanisms that generate revenues for those stakeholders possessing property and use rights in relation to the forestland concerned.
- Ensure participation of local communities in the development of effective IFMs.
- Strengthen the capacity at the level of local communities to manage forests for the maintenance of ecological services and to increase access to financial revenues rewarding the corresponding local and global benefits.
- Evaluate the impacts of the current policy as regards forestry under the Clean Development Mechanism, in terms not only of carbon but also of sustainability in the broader sense. Attention should be paid to the question how payments for carbon sequestration can contribute to improving the livelihoods of local forest-depending communities.
- Support the development of financing mechanisms for the maintenance of hydrological functions, such as water funds with revenues collected from taxes on drinking water and payments by large water users.

International institutions (donors, multilateral banks and international NGOs) should enhance their role in catalysing the development of IFMs in different contexts, by developing policies to contribute to institutional capacity building, stimulating exchange of information, and stimulating an enabling environment (e.g. good governance, inter-sectoral policy co-ordination, proper legal and policy framework in countries and in international financial institutions). Furthermore, they can play an important role by bearing large part of the costs of pilot projects and initial implementation of IFMs.

Literature

Echavarría, M., 2000. Valuation of water-related services to downstream users in rural watersheds: Determining values for the use and protection of water resources. Land-Water Linkages in Rural Watersheds Electronic Workshop, Background Paper No. 4, 14 pp. FAO, Rome.

Echavarría, M. & T. Granizo, 2000. Valoración del agua en los páramos. Pp. 174-175 in: J. Recharte, J. Torres & G. Medina (Eds.), II Conferencia electrónica sobre usos sostenibles y conservación del ecosistema páramo en los Andes. CONDESAN, Mountain Forum.

Hofstede, R. & M. Albán, 2002. Payment for hydrological services in the Ecuadorian Andes: Water taxes and water funds at municipal level. ETFRN News 35: 45-47, Wageningen.

Hope, R., I.R. Calder, J.W. Gowing, N. Laurie, P.-J. Dixon, C.A. Sullivan, N.A. Jackson, G. von Maltitz, J. Bosch, D. LeMaitre, P. Dye, T. Netshiluvhi, N. Hatibu & G. Paterson, 2002. Saving the trees and the poor? Catchment Management and Poverty (CAMP). ETFRN News 35: 48-51, Wageningen.

Johnson, N., A. White & D. Perrot-Maître, 2001. Developing markets for water services from forests: Issues and lessons for innovators. Forest Trends, WRI & The Katoomba Group, 19 p.

Landell-Mills, N. & I. Porras, 2002. Silver bullet or fools' gold? A global review of markets for forest environmental services and their impacts for the poor. Pp. 89- 92 in: P.A. Verweij (ed.), Understanding and capturing the multiple values of tropical forest. Tropenbos International, Wageningen, The Netherlands.

May, P. 2002. CDM and sustainable land reform in the Brazilian Amazon. ETFRN News 35: 30-32, Wageningen.

Mero, D. 2002. Financing reforestation for improved watershed management in the Philippines. ETFRN News 35: 44-45, Wageningen.

Miranda, M., C. Dieperink & P. Glasbergen, 2002. The social meaning of carbon markets; Institutional capacity building for a green market in Costa Rica. ETFRN News 35: 33-35, Wageningen.

Pérez, C.J., 2003. Pagos por servicios hidrológicos a nivel municipal y su impacto en el desarrollo rural: la experiencia del PASOLAC. Pp. 33-34 in: Foro Regional Sistemas de Pago por Servicios Ambientales en Cuencas Hidrográficas, Informe Final. FAO, Oficina Regional para América Latina y el Caribe, Santiago.

Perrot-Maître, D. & P. Davis, 2001. Case studies: Developing markets for water services from forests. Forest Trends, Washington D.C. http://www.forest-trends.org.

Pearce, D.W. & Pearce, C.G., 2001. The value of forest ecosystems. Report to the Secretariat of the United Nations Convention on Biological Diversity. Montreal, Canada.

Reyes, V., O.Segura & P.A. Verweij, 2002. Valuation of hydrological services provided by forests in Costa Rica. Pp. 101-106 in: P.A. Verweij (ed.), Understanding and capturing the multiple values of tropical forest. Tropenbos International, Wageningen, The Netherlands.

Skutsch, M. M., 2002. Access to finance for community forest management under the UNFCCC and Kyoto Protocol. ETFRN News 35: 28-30, Wageningen.

Verweij, P.A., 2002. Innovative Financing Mechanisms for conservation and sustainable management of tropical forests – issues and perspectives. Pp. 107-117 in: P.A. Verweij (ed.), Understanding and capturing the multiple values of tropical forest. Tropenbos International, Wageningen, The Netherlands.

Verweij, P.A., 2003. Payments for forest hydrological services in Latin America: trends and perspectives. Paper presented at the International Congress on 'Globalization, Localization and Tropical Forest Management', Tropenbos International, Amsterdam, 22-23 October 2003.

Appendix: Seminar programmes

First seminar:

Tropical forests in multi-functional landscapes: biodiversity conservation

Date and venue:

2 December, 2002, 13:00 – 17:00 hours;
Room 017 at Hoogt 13, Utrecht (downtown).

Programme:

13.00 Opening - Pieter Zuidema (Prince Bernhard Centre, Utrecht University)
13.05 *Introduction:* Biodiversity conservation in tropical forests: the need and challenge - Jeffrey Sayer (Utrecht University, WWF, CIRAD)
13.15 Biodiversity conservation in protected forests in Costa Rica - Maarten Kappelle (Copernicus Institute, Utrecht University)
13.45 Potential for biodiversity conservation in production forests - Hans ter Steege (National Herbarium, Utrecht University)
14.15 Economic valuation of forest conservation in Indonesia and Guyana - Pieter van Beukering (IVM, Free University)
14.45 Coffee / tea break
15.15 Use and conservation of biodiversity in East African forested landscapes by Freerk Wiersum (Forestry Department, Wageningen University)
15.45 *Discussion and integration:* Biodiversity conservation in tropical forests: where are we and where are we going?" by Jeffrey Sayer
16.30 Closing - Wim Dijkman (VTB)

Second seminar:

Environmental services of tropical forests in multi-functional landscapes

Date and venue:

11 April, 2003, 13:00 – 17:00 hours
Room 017 at Hoogt 13, Utrecht (downtown).

Programme:

13.00 Opening - Pieter Zuidema (Prince Bernhard Centre, Utrecht University)
13.05 *Introduction:* Environmental services of tropical forests: which services and what significance? - Jeffrey Sayer (Utrecht University, WWF, CIRAD)
13.15 Tropical forests as sources or sinks: recent technological insights, interaction with forest management and uncertainties - Gert-Jan Nabuurs (Alterra, Wageningen UR)
13.45 Hydrological functions of tropical forests: importance inside and outside forests - Oscar van Dam (Environmental Geo-Sciences, Free University, Amsterdam)
14.15 Coffee / tea break
14.45 Getting a piece of the pie: profits from environmental services for local actors - Pita Verweij (Copernicus Institute, Utrecht University)
15.15 Instruments for carbon offsets: what works and how well? - Eveline Trines (Independent consultant - Sinks expert; Ministry LNV)
15.45 *Discussion and integration:* Environmental services of tropical forests: knowledge gaps, lack of regulations and difficulties in application - Jeffrey Sayer
16.30 Closing - Wim Dijkman (Dutch tropical forest association)