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The Costs and Benefits of Forest Protected Areas for Local Livelihoods: a review of the current literature

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Abstract

Protected areas could play a significant role in the implementation of schemes to reduce emissions from deforestation and degradation (REDD) in developing countries, through either the strengthening of the existing protected area network, or designation of new areas. Many rural poor people rely on forest resources, and may experience positive or negative changes to their livelihoods as a result of REDD. This review aims to assess the livelihood implications of the existing protected area network in order to inform future REDD policy.

The costs and benefits of individual protected areas for community livelihoods have been well documented. Costs can range from displacement of local communities to crop damage by wildlife, and sometimes include restricted access to resources and changes in land tenure. Benefits can include direct revenue from environmental protection, and the maintenance of ecosystem services such as watershed protection. The nature of these costs and benefits depends largely upon the protected area's status and governance, as well as its history of use.

The net livelihood impacts of protected areas are less easy to discern, as there is a lack of standardised assessment methodologies. The effect on livelihoods of differing governance types within and between IUCN protected area management categories is rarely assessed in the literature, and requires further research. However, general patterns can be observed. The livelihood impacts of protected areas vary with protected area status, management strategies and community involvement in governance. Strictly protected areas with top-down management structures (generally associated with IUCN management categories I-II) can result in major livelihood costs and cause conflict between local communities and protected area management. Community management schemes, and protected area management allowing sustainable use of forest resources (more often associated with IUCN management categories V-VI), can provide tangible benefits. However, significant costs can still be incurred by communities if management and institutional capacity is lacking, and issues of governance and tenure are not resolved.

Inequitable distribution of livelihood costs and benefits is an obvious problem that is often yet to be adequately addressed in protected area management. These issues need careful consideration as REDD policy develops. An analysis of livelihood costs and benefits in existing forest carbon markets has identified issues similar to those for protected areas; including lack of established tenure and the inequitable distribution of resources, particularly affecting the landless members of society. Involving local communities in the planning and implementation of REDD, and ensuring that financial or other benefits are shared, is likely to result in a more sustainable solution to deforestation than are less participative strategies.

1. Introduction

The effectiveness of protected areas has long been discussed in terms of their ability to reduce deforestation and conserve biodiversity. It is only relatively recently that the social impacts of such conservation measures have come under scrutiny. The establishment of forested protected areas can place restrictions on the use of resources within large areas of forest that had been freely available to local and indigenous communities. Whilst these areas provide important ecosystem services at the global, national and local scale (Table 1), there is concern that the costs are mostly incurred by the local people who rely on forest resources for their livelihoods.

Table 1: The ecosystem services provided by forests (Millennium Ecosystem Assessment, 2005)

Provisioning	Supporting	Regulating	Cultural
Food	Nutrient cycling	Climate regulation	Aesthetic
Fresh Water	Soil formation	Flood regulation	Spiritual
Fuelwood and fibre	Primary production	Disease regulation	Educational
		Water purification	Recreational

It has been suggested that carefully managed protected areas could help to alleviate poverty; conserving biological resources whilst providing developmental benefits to marginalised communities (WWF, in press). However, it has also been suggested that protecting areas of forest can increase poverty and marginalisation, resulting in lost livelihoods and dislocation of communities (Pimbert & Pretty, 1995), raising ethical moral, and practical questions regarding protected area management (Kaimowitz, 2003; Salafsky & Wollenberg, 2000). One ethical position is that as a minimum, protected areas should be managed such that their creation 'does no harm' to those living within and around them.

In the context of the targets to extend the protected area network, set by the Convention on Biological Diversity (CBD), and of the current discussions on reduced emissions from deforestation and degradation (REDD) in developing countries as a climate change mitigation option, the potential positive and negative impacts on livelihoods and poverty take on added importance. This paper reviews the current state of knowledge on the impact of protected area management on local communities with the aim of informing future protected area and REDD policy.

2. Forest resources and local livelihoods

2.1. Definitions of poverty and livelihoods

Whilst natural resource management decisions are increasingly discussed in relation to poverty and livelihoods, the definitions of these terms are not always clearly stated. 'Livelihoods' represent the means of living, and 'poverty' is typically an outcome-based measure of livelihood performance (Sunderlin *et al.*, 2005). Traditionally, poverty has often been measured in terms of absolute income, with a common indicator defining the 'poor' as those who earn less than US$1 per day (Anglesen & Wunder, 2003). The Human Development Index (HDI), developed by the United Nations Development Programme (UNDP), also includes health and education parameters. There are now various poverty assessment frameworks, which like the HDI recognise that poverty is not a matter of income alone. These tend to incorporate natural, human, social and physical capital, using indicators ranging from income, access to resources and basic infrastructure, to the vulnerability of populations to shock, and level of community organisation.

The Organisation for Economic Co-operation and Development (OECD, 2001) and World Health Organisation (WHO, 1997) have developed similar indicators. UNEP has taken this concept even further to identify indicators of 'well being' (UNEP, 2004), incorporating traditional, cultural and spiritual practices and the ability to make decisions on the sustainable management of resources. It has also been suggested that political capital should be added to the framework (Baumann, 2002), stressing the relevance of governance to livelihood and poverty issues. The widely accepted 'asset-based indicators of poverty' include measures for each type of capital (Table 2).

Table 2: Asset-based indicators of poverty (adapted from the sustainable livelihoods framework reported in Dubois, 2002)

Natural capital:	Land, forests, water, wildlife
Physical capital:	(a) privately-owned assets (e.g. farm, animals) (b) publicly-owned economic infrastructure (e.g. roads) (c) social infrastructure (e.g. schools, hospitals).
Financial capital:	Cash (income and savings) and readily convertible liquid capital.
Human capital:	Health, nutritional levels, education
Social capital:	Social relationships, cultural/spiritual
Political capital:	Empowerment, access rights and tenure, governance

The term ‘livelihood’ often refers to the access of individuals to these various types of capital, opportunities and services (Ellis, 2000), but has also been defined as comprising the capabilities, assets and activities required for a means of living (Carney, 1998; Sunderlin *et al.*, 2005). Livelihoods can be improved, for example, if natural capital is managed sustainably, and vulnerability to changes in the environment or market is lowered (Kaimowitz, 2003).

All these factors are considered in the following investigation of the social and economic impacts of protected areas under different forms of management and governance.

2.2. Forests and poverty

‘Forest’ is also defined differently by different actors. The Food and Agriculture Organization of the United Nations (FAO) considers forest to be land with a tree canopy cover of more than 10%, which has a larger area than 0.5 ha and is not specifically under a non-forest land use (FAO, 2001). Moreover, it includes clear-felled land that is destined for re-planting. Other classification systems have used higher canopy cover thresholds, for example defining coverage of 10-30% as ‘sparse trees and parkland’ (UNEP-WCMC, 2000). In reporting to the UN Framework Convention on Climate Change (UNFCCC), countries use their own national forest classification system within the thresholds set by guidance from the Intergovernmental Panel on Climate Change (IPCC) (Penman *et al.*, 2003). This is the guidance agreed by the UNFCCC for use in the ‘demonstration’ (pilot) phase of REDD.

Forests can be simultaneously recognised as a ‘poverty trap’ and a ‘safety net’ for the rural dwellers who use their resources (Angelsen & Wunder, 2003). There is a distinction to be made here between poverty reduction and mitigation, often bundled together as ‘poverty alleviation’. Poverty reduction refers to a successful improvement of livelihoods, whereas poverty mitigation refers to prevention of increased deprivation (Sunderlin *et al.*, 2003).

Table 3: Forest resource use and livelihood benefits. Adapted from Kaimowitz (2003)

Forest Resource	**Livelihood benefits**	**User groups**
Direct use		
Timber	Direct consumption (subsistence*):	Indigenous peoples and forest communities
NTFPs: fuelwood, resins, fibre, bushmeat, fish, fodder, berries, roots, medicines	Construction, food, medicine, fuel	Rural poor on forest margins
	Income source (commercial):	Smallholder farmers
	Large forest industry employment	Artisans and employees of small or large scale forestry
Source of new agricultural land (slash/burn/swidden cultivation)	Employment and income from small scale informal forestry markets (can be seasonal and supplementary)	
	Inputs for non-forest income generating activities	
	Indirect benefits:	
	Third party involvement – improved infrastructure, health benefit, skill development	
Indirect use		
Capital asset: Opportunity to alter land use for financial gain/subsistence needs	Diversified resource/asset base Security	Indigenous peoples and forest communities
		Rural poor on forest margins
Watershed protection (e.g. reduced soil erosion)	Improved agricultural, fisheries productivity. Adaptation to climate change. Improved water quality	Smallholder farmers
Carbon storage	Reduced climate change impacts**	
Existence	Cultural/spiritual values Religious values Ecotourism	

* Economies are increasingly cash based, so that 'subsistence' often involves some cash element.
**Whilst 'climate change mitigation' can also be construed as a global benefit, the 'safety net' function of forests is likely to become more important to local communities as agriculture in some climate zones becomes marginalised. Forest retention may therefore be viewed also as a means of adaptation to the impacts of climate change, such as an increasing uncertainty in agricultural yields, on the rural poor.

In providing a diversified income stream and resource base that can be relied upon in times of stress, forests can contribute to poverty reduction. However, an abundance of natural resources has long been associated with limited economic growth and development, with marginalised communities having little access to markets or other income streams, and often suffering growing restrictions on the use of their natural capital. Whilst the potential for forests to contribute to poverty reduction is often doubted (Angelsen & Wunder, 2003; USAID, 2006), forest resources have traditionally supported the subsistence of indigenous peoples. Forests can also contribute to well-being through ecosystem services such as flood and erosion control. Finally, where there is local control over forests, the option remains to clear them for other uses, such as farmland (Anderson *et al.*, 2006).

2.3. Livelihoods and forest resource use

It is estimated that 90% of the world's poor depend on forests for at least a portion of their income (World Bank, 2000; Scherl *et al.*, 2004; USAID, 2006). In Africa, 600 million people have been estimated to rely on forests and woodlands for their livelihoods (Anderson *et al.*, 2006), and in India, 50 million people are estimated to directly depend on forests for subsistence alone. Kaimowitz (2003) reviews the importance of forest resources to local communities. The benefits derived from forests are outlined in Table 3.

The users of forest products include forest dwellers, nearby farmers, commercial users (including small traders, producers and employees) and the urban poor. Timber, non-timber forest products (NTFPs) and animal protein are all used by the rural poor for subsistence, and also as a source of income and employment (Angelsen & Wunder, 2003). Depending on circumstances, forest products may offer both a 'daily net' and a 'safety net'. The 'daily net' describes everyday use, with products meeting current household needs, offering a reliable source of income to purchase agricultural inputs (Shackleton & Shackleton, 2004), or fodder for livestock herds. A 'safety net' comes into play when other sources of household income (e.g. plantations) fail to meet dietary shortfalls, or whenever a quick cash option is required (McSweeney, 2003). In Brazil, for example, the sale of one palm species supports over two million people and is most important during agricultural difficulties (WWF, unpublished).

NTFPs are a key resource for many poor communities (Sunderlin *et al.*, 2005). In West Africa, for example, bushmeat provides 25% of protein requirements, and can be the principal source for some indigenous groups (Bennett, 2000). NTFPs are often open-access resources, and require little processing or the use of low cost (often traditional) techniques. An overview of case studies indicates that forest products contribute between 20% and 40% of total household income in forest areas, and that poor households tend to be disproportionately dependent on forest resources (especially fuel wood and fodder) (Vedeld *et al.*, 2007). Based on this type of finding, investment in NTFP use has often been proposed as a method of poverty alleviation (Brown & Williams, 2003). Although NTFP sales often supplement income, it has been suggested that the same open-access characteristics that make them available to poor households in the first place make them poor candidates for poverty reduction schemes (Arnold & Perez, 2001; Belcher, 2005).

2.4. Resource use inequalities

There is significant intra-community variation in the extent to which forest dwellers depend upon forest resources, and the income derived. Pyhala *et al.* (2006) estimated a difference in mean annual income per household of US$1 363 between the poorest and richest households across six communities in the Peruvian Amazon. Similarly, the value for economic production in the Peruvian Amazon was estimated at between US$425 000 and US$1 693 per household (Coomes *et al.*, 2004). Whilst the poorer members of a community rely more heavily on forest resources, the richer households often have the main share of resource use (DFID, 2002). In a community in the Brazilian Amazon, the three richest households were responsible for 24% of the total palm fruit harvest (Coomes *et al.*, 2004). The households receiving most income from bushmeat hunting in Gabon are from the richer part of the community (Coad, 2007).

Within rural forest communities there are also often gender differences in forest use and political power. In the Jaú National Park, Brazil, hunting and fishing is a predominantly male activity, whereas food preparation, collection of forest products to supplement diet, fuel wood gathering and agriculture is mainly women's work (Oliveira & Anderson, 1999). Similar differences in use have been shown in other forest communities (e.g. Ongugo, 2007). Some studies suggest that women are the primary users of forests; for example, in a study in Uttar Pradesh, India, women derived 33 to 45% of their income from forests and common land, whilst men derived only 13% (FAO, 2006). Whilst women have access to and substantial labour and management responsibilities for forest resources, they are much less likely to own land than men, and it is often men who control the use and marketing of the products and incomes (Lastarria-Comhiel, 1995; Rocheleau, 1997). Despite this lack of tenure and control, women's work and incomes can have a greater contribution to household welfare and security (FOA, 1996; IFAD, 1999). In response, development agencies are targeting household poverty reduction schemes at women (Nigenda & Gonzales-Robledo, 2005; Hoddinott & Skoufias, 2000).

When discussing forest poverty alleviation, it must be recognized that commercialisation and increased market access to forest resources does not necessarily provide opportunities for the poor, and may shift access and use towards the richer sections of a community (Arnold & Perez, 2001). In terms of poverty mitigation, forest resources are often vital to the poorest sections of the community.

3. Protected area management and community involvement

3.1. The global protected area network

Protected areas are defined by the World Conservation Union (IUCN) as areas of land or sea "dedicated to the protection and maintenance of biological diversity and of natural and associated cultural resources, managed through legal or other effective means". Protected areas may be further classified into six management categories (I-VI), reflecting the broad purpose of designation. The protected area network has grown rapidly since the 1970s, and there are currently over 120 000 designated areas recorded in the World Database on Protected Areas (WDPA), covering approximately 12% of the terrestrial surface (UNEP-WCMC, 2007).

3.2. The growth of community conservation

Until relatively recently, protected area management strategies have focussed upon the preservation of biodiversity through 'protectionist' approaches. Protected areas in many countries were for the most part state-owned, with no-take policies, and provided little access other than for tourism (Naughton-Treves *et al.*, 2005). In the 1970s - 80s, the rights and needs of local communities in the development and management of protected areas began to be recognised. Now, in line with the 'sustainable use' goals of the CBD, protected areas are expected to directly contribute to national development and poverty reduction (Naughton-Treves *et al.*, 2005), and as such form an indicator for the success of the Millennium Development Goals:

> *Target 9: 'Integrate the principles of sustainable development into country policies and programmes and reverse the loss of environmental resources.'*
>
> *Indicator 26: 'Ratio of area protected to maintain biological diversity to surface area'*

The level of community involvement varies greatly between individual protected areas, organizations and countries, and in relation to their management category and form of governance.

3.3. The IUCN protected area management categories

Originally, four categories of protected area were recognised: 'National park', 'Strict nature reserve', 'Fauna and flora reserve' and 'Reserve with prohibition for hunting and collecting' (Phillips, 2007). These reflected the conservation attitudes of the time, with little room in the classification system for community rights to tenure or resource use within the protected areas. In 1978, an IUCN protected area categorisation was produced that reflected a new emphasis on the sustainable management of natural resources by recognising 'resource reserves', 'anthropological reserves' and multiple use management areas'. This was followed in 1994 by the current IUCN protected area management category system (Table 4), which reflects an increase in the perceived importance of community involvement by including a new Category VI, which explicitly mentions community needs.

The categories reflect the rationale behind establishment of the protected area and do not determine protected area management and governance (Naughton-Treves *et al.*, 2005), which differ both within and between categories. In general, however, protected areas of IUCN category I-II are more restrictive of forest product use than are the lower categories (V-VI).

3.4. Recognising community conservation: Category VI

The number of category V and VI reserves have grown over the last decade, to represent 15% of the number, but 35% of the total area, of protected areas which have an IUCN category assigned (Feb 2008 edition of the WDPA). The new category VI received a mixed reception, with some suggestions that governments may use community forest reserves, whose main purpose is generally timber extraction for community profits, to make up the CBD 10% protected area coverage targets without increasing investment in conservation. Community-managed forests are a rapidly increasing means of forest conservation, but some feel that strict nature reserves (categories I-II) are still needed to truly protect endangered species and ecosystems. However, many authors have welcomed the change in the category system, on the grounds that with growth in both the protected area system and the human population, protected areas will be unable to fulfil their biodiversity goals without engaging local communities and their needs.

3.5. The IUCN governance matrix

The protected area management category does not indicate the ownership or management authority of a protected area. The land and resources in any of the six management categories can be owned and/or directly managed, alone or in combination, by government agencies, NGOs, communities and private parties (Borrini-Feyerabend, 2007). Distinctions can therefore be made between protected areas in terms of 'governance' type (Figure 1) and ownership. Globally, 77% of the world's forests are owned and administered by governments, 11% are reserved for or owned by local communities and 12% are owned by individuals (Anderson *et al.*, 2006). Forest ownership varies widely between countries; 80% of Mexico's forest area is community owned under the *eidjo* system (Barton-Bray, 2002, 2003).

The governance matrix neatly illustrates the range of ways in which communities and their needs can be, and have been, incorporated when designating and managing protected areas. At one end of the governance spectrum, the state has ownership of the area and may involve the surrounding communities in some decision-making through representation in stakeholder groups. At the other end, protected areas are owned and run by the communities themselves. The level of strict biodiversity protection, or conversely, the access that local communities have to local resources, can also vary from no-take enforcement (whether by the government or the community itself) to sustainable management of forest resources.

Table 4: The 1994 IUCN protected area management categories (IUCN, 1994)

CATEGORY Ia: Strict Nature Reserve: protected area managed mainly for science

Definition: Area of land and/or sea possessing some outstanding or representative ecosystems, geological or physiological features and/or species, available primarily for scientific research and/or environmental monitoring.

CATEGORY Ib Wilderness Area: protected area managed mainly for wilderness protection

Definition: Large area of unmodified or slightly modified land, and/or sea, retaining its natural character and influence, without permanent or significant habitation, which is protected and managed so as to preserve its natural condition.

CATEGORY II National Park: protected area managed mainly for ecosystem protection and recreation

Definition: Natural area of land and/or sea, designated to (a) protect the ecological integrity of one or more ecosystems for present and future generations, (b) exclude exploitation or occupation inimical to the purposes of designation of the area and (c) provide a foundation for spiritual, scientific, educational, recreational and visitor opportunities, all of which must be environmentally and culturally compatible.

CATEGORY III Natural Monument: protected area managed mainly for conservation of specific natural features

Definition: Area containing one, or more, specific natural or natural/cultural feature which is of outstanding or unique value because of its inherent rarity, representative or aesthetic qualities or cultural significance.

CATEGORY IV Habitat/Species Management Area: protected area managed mainly for conservation through management intervention

Definition: Area of land and/or sea subject to active intervention for management purposes so as to ensure the maintenance of habitats and/or to meet the requirements of specific species.

CATEGORY V Protected Landscape/Seascape: protected area managed mainly for landscape/seascape conservation and recreation

Definition: Area of land, with coast and sea as appropriate, where the interaction of people and nature over time has produced an area of distinct character with significant aesthetic, ecological and/or cultural value, and often with high biological diversity. Safeguarding the integrity of this traditional interaction is vital to the protection, maintenance and evolution of such an area.

CATEGORY VI Managed Resource Protected Area: protected area managed mainly for the sustainable use of natural ecosystems

Definition: Area containing predominantly unmodified natural systems, managed to ensure long term protection and maintenance of biological diversity, while providing at the same time a sustainable flow of natural products and services to meet community needs.

Governance type	A. Government Managed Protected Areas			B. Co-managed Protected Areas			C. Private Protected Areas			D. Community Conserved Areas	
IUCN Category (management objective)	Federal or national ministry or agency in charge	Local/ municipal ministry or agency in change	Government-delegated management (e.g. to an NGO)	Trans-boundary management	Collaborative management (various forms of pluralist influence)	Joint management (pluralist management board)	Declared and run by individual land-owner	…by non-profit organisations (e.g. NGOs, universities, etc.)	…by for profit organisations (e.g. individual or corporate land-owners)	Declared and run by Indigenous Peoples	Declared and run by Local communities
I - Strict Nature Reserve/ Wilderness Area											
II – National Park (ecosystem protection; protection of cultural values)											
III – Natural Monument											
IV – Habitat/ Species Management											
V – Protected Landscape/ Seascape											
VI – Managed Resource											

Type A: state governance—authority & responsibility by government through federal, national or sub-national agency	**Type B: shared governance**—authority & responsibility shared between governmental agencies and other entitled actors	**Type C & D: private & community governance**—authority & responsibility by landowners or communities with customary rights

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enforce rules and **repress violations**	**inform** and/or **consult** about management decisions	**seek consensus**, also through **benefit sharing**	**negotiate** (involve in decision making) & develop specific **agreements**	**formally share authority & responsibility** (e.g. via seats in a CM body)	**recognise** full private or customary rights **& assist** in management

Figure 1: Governance in protected areas

a) (above) The IUCN protected areas governance matrix, showing the different governance structures that can exist for each protected area management category

b) (left) Government involvement within governance types

Adapted from Borri-Fereyabend (2007)

There is some uncertainty in defining 'sustainable use of resources', and therefore over which community protected areas should be recognised as protected areas within Category VI. An example of such a situation would be where a community forestry reserve is set up to provide sustainable incomes for local communities and protect them from external exploitation of local forest by international logging companies. Here the emphasis would be on the needs of the community, rather than on a goal of retaining biodiversity, so whilst the area may conserve forest, it may not meet formal protected area definitions.

4. The costs of protected areas to local livelihoods

Some commentators believe that conservation measures should 'do no harm', whilst others feel that protected areas should also contribute to poverty reduction and development. This section reviews the evidence on the costs that protected areas impose upon local communities, in order to assess their ability to 'do no harm'. These costs can range from displacement of communities to loss of the infrastructure brought by logging concessions. The management categories and governance styles introduced above will obviously influence the effect of protected areas on local livelihoods, as discussed below.

4.1. Displacement

'Displacement' is often taken to mean the forced removal of local communities from their land. However, the World Bank's definition now includes displacement from resources without community movement:

> *"(i) relocation or loss of shelter; (ii) loss of assets or access to assets; or (iii) loss of income sources or means of livelihood, whether or not the affected persons must move to another location, or the involuntary restriction of access to legally designated parks and protected areas resulting in adverse impacts on the livelihoods of the displaced persons" (World Bank, 2002)*

Communities living in or around strictly protected areas, where resource restriction is incurred, could therefore now fall under the World Bank definition. This review adopts the traditional interpretation of displacement as the physical removal of communities from the land, with the costs of restricting resources for local community use being covered in Section 4.3. The most documented example of displacement is the removal of indigenous communities from Yellowstone National Park by the US army (Burnham, 2000). It has been suggested that Yellowstone served as a 'protectionist' model for the American West and then the rest of the world (Stevens, 1997). The displacement of local people from national parks is 'one of the most controversial and contested aspects of protected areas' (West & Brockington, 2006), and is often used to highlight the conflict between biodiversity conservation and poverty reduction (Nepal, 2002; Borgerhoff & Mulder, 2005; Brockington, 2004).

4.1.1. Displacement and international policy

A United Nations Declaration on the Rights of Indigenous Peoples was adopted in June 2006 by the UN Human Rights Council and then by the UN General Assembly in September 2007. The Declaration requires that states prevent any form of population transfer that has the aim or effect of violating or undermining rights of indigenous peoples (Lustig & Kingsbury, 2006):

> *'Indigenous peoples shall not be forcibly removed from their lands or territories. No relocation shall take place without the free and informed consent of the indigenous peoples concerned and after agreement on just and fair compensation and, where possible, with the option of return.' (Article 10)*
>
> *States [must] obtain the free and informed consent of the indigenous people prior to the approval of any project affecting their lands and other resources (Article 30)*

If displacement is planned during a development scheme, the World Bank requires that less drastic options should be explored before resettlement is used; and that resettlement should either improve the condition of the displaced communities or restore them to a situation no worse than before (World Bank, 2002).

The rights and wellbeing of displaced communities, and those facing resource restrictions due to protected areas, are now also recognized within the policy of major conservation bodies, including IUCN (Beltran, 2000), WCS (Redford & Fearn, 2007), and WWF (WWF, 2003).

4.1.2. How many people have been displaced from protected areas?

A review of approximately 250 published articles on 'conservation displacements' from 1970 onwards highlights some of the major findings (Brockington *et al.* 2006). Most articles documented case studies of evictions, but provided little quantitative information on the total number of protected areas where forced displacement (defined here as physical removal) has occurred, or the number of people that have been removed. Consequently, estimates range from 900 000 to 14.4 million people displaced (Geisler & de Souza, 2001; Geisler, 2003a, 2003b). For Central Africa, estimates of the number of people displaced from 12 parks (45% of the total for the region) were used to produce an overall estimate of 120 000 displaced to date, with the potential for another 170 000 if there are no changes in conservation policy (Cernea & Schmidt-Soldau 2006). The estimate is contested by NGOs in Central Africa, who suggest that 'the information on which it is based is poorly gathered and makes false assumptions' (Redford & Fearn, 2007).

There is a similar lack of information on the number of people currently living within protected areas. Brockington and Igoe (2006) provide details of studies in India, South America, Mongolia and Central Africa, which suggest that there are communities living within 56 to 85% of protected areas. Information on the numbers or densities of people living within these protected areas is not available. This not only demonstrates our poor knowledge of the scale of displacement events, long-term residency and migration into protected areas, but also restricts our ability to predict how many people may be affected by displacement in the future.

The lack of quantitative data does not prevent fierce debate. A current dispute involves the twelve new national parks in Gabon, jointly run by WWF and WCS in partnership with the Gabonese government, who have been accused (Brockington & Schmidt-Soltau, 2004) of under-reporting the number of people living within the parks and threatened with displacement. A recent unpublished study used rural population densities to estimate the number of those displaced in Gabon at 14 000 (Kramkimel 2005). This was rigorously refuted by the Gabonese government and Redford & Fearn (2007), who report that no displacements have taken place in Gabon and that Gabon's low rural population, combined with the 1940s national practice of relocating villages on main roads (*regroupements*), meant that it was possible to locate the parks in areas of extremely low population densities. These disputes, with few unbiased quantitative studies, make impartial assessments difficult.

4.1.3. When has displacement occurred?

Although much of the displacement literature has described historical events (articles published in 1990 describing 1970s events), a quarter of the papers reviewed document displacements after 1990. For example, 500 people were removed from the Nechasar National Park in southern Ethiopia in 2004 and resettled outside its borders (Pearce, 2005). This forced displacement was undertaken by the government before handing the park management over to a contracted Dutch-based organization, the African Parks Foundation (APF) (Adams & Hutton, 2007). Some case studies provide examples of a tightening, rather than a relaxing, of protected area laws. For instance, Nepal (2002) reports that 'the Thai cabinet has resolved to relocate hill tribes living in ecologically 'sensitive' areas, thus reversing the previous government's undertaking to respect the land rights of communities established before the protected areas were gazetted'. Similarly, Kothari (2004) claims that four million people face eviction in India as the result of the revision of conservation legislation. Despite these cases, Brockington and Igoe (2006) hypothesise that forced displacements are much less frequent or severe now than they were before 1980.

4.1.4. Where has displacement occurred?

Most records of displacement reviewed by Brockington and Igoe (2006) came from Africa, South and South East Asia and North America. There were relatively few reports for South and Central America, Australia, Europe or the former Soviet Union, or most of the Caribbean and Pacific although some authors suggest this represents a lack of reporting rather than of displacement (Poirier & Ostergren, 2002). The majority (69%) of recorded displacements reviewed were from protected areas in IUCN Category II, and 88% came from Categories I – IV, seen as more 'strictly protected' categories. Countries may be more likely to displace people from protected areas if their history has been one of strict government control. There is also some evidence of ulterior political motives for displacement. For example, the Tanzanian government used displacement from protected areas to resettle communities in collective villages, and evictions in South Africa were particularly vigorous during the Apartheid era (Koch, 1997).

4.1.5. The livelihood implications of displacement

The 'Impoverishment Risks and Reconstruction' framework outlines 8 major risks to displaced peoples (Cernea, 1997), many of which are also relevant to communities living around protected areas:

- Landlessness (expropriation of land assets and loss of access to land)
- Joblessness (even when the resettlement creates some temporary jobs)
- Homelessness (loss of physical houses, family homes and cultural space)
- Marginalisation (social, psychological and economic downward mobility)
- Food insecurity (malnourishment etc)
- Increased morbidity and mortality
- Loss of access to common property (forests, water, wasteland, cultural sites)
- Social disarticulation (disempowerment, disruption to social institutions)

Although case studies exist that describe the effects of displacement on livelihoods, few provide rigorous documentation (examples include Neumann, 1998; Saberwal *et al.*, 2000; McLean & Straede, 2003; Hitchcock, 2001; McCabe, 2002). Only a handful have used quantitative methods to measure the costs of displacement. The most well known study is that of McLean and Straede (2003) who conducted a 'before and after' study of forced displacement of 2 000 Tharu people from the Royal Chitwan National Park, Nepal. The displaced people were relocated onto areas with poorer soils, three hours away from water and forest resources. Brockington (2002) also showed that the removal of pastoralists from the Mkomazi game reserve led to a collapse in the local livestock market and economy.

Very few studies mention compensation for displacement, through land or money; those that do tend to provide examples of inadequate or absent compensation. The lack of detailed information on compensation mechanisms is not surprising, as most of these studies have been published to highlight the costs of displacement. Examples include the displacement of villagers from the Waza National Park, Cameroon, in 1998 (Bauer, 2003); of local people from the Mkomazi game reserve, Tanzania in 1988 (Igoe, 2003); and the Karrayu pastoral group from the Awash National Park, Ethiopia (Bassi, 2003). Impartial studies of the number of displaced people who receive compensation, and the effects of this compensation in mitigating the costs of displacement, are required.

4.1.6. Displacement and conservation success

Even the strongest opponents of displacement recognise that reducing human population densities within protected areas can reduce pressure on species and ecosystems (e.g. West & Brockington, 2006). However, displaced people often, unsurprisingly, hold negative attitudes towards conservation, which can result in biodiversity loss. In Uganda, the families that were allowed to resettle in the Lake Mburo National Park in 1986 after eviction in 1983 opted to slaughter the wildlife in an attempt to eliminate the area's conservation value and preclude the possibility of being re-evicted (Hulme, 1997).

Displacement of communities following protection of one area of land can also result in the transfer of land use to a nearby area. Intensified timber extraction in the landscape surrounding a protected area can occur after its designation (Oliviera & Anderson, 1999). When protected areas are designated, local communities outside as well as those inside the area are faced with a reduction in the land available for agriculture, grazing or extraction. This can increase pressure on and degradation of land bordering the reserve and available for local community use (Bassi, 2003). Land scarcity can also change local livelihood strategies, with farmers switching to more intensive agricultural techniques or crops. In Madagascar, subsistence 'tavy' (slash and burn) farming is often blamed for the country's deforestation and soil erosion problems, especially where land is scarce, rotational periods are short and slopes are steep. Similar criticisms are not typically applied to the cultivation of cash crops such as coffee (Jarosz, 1996). In the Ranomafana National Park, intensive irrigated rice paddies were used to replace forgone tavy yields (Ferraro, 2002). This approach can deliver high yields, but also may be more risky; for instance cyclone damage is much more prevalent in rice fields than on land used for tavy (Ferraro, 2002). Restrictions on available land can also result in a change in diet, as the quality of land under intensive farming may be reduced over time, so that communities switch to less nutrient-demanding crops such as manioc, which contains less protein and calcium than rice.

4.2. Changes in land tenure and community structures

The changes in land tenure rights that come with protected area designation have significant impacts on local land management. For centuries, the prevailing land tenure arrangements in Africa and Asia have involved significant communal control over land or resource use (WRI, 2005). Small forest communities often set strict laws concerning land use governed by traditional institutions that partition the ownership and use of local landscapes (Kajoba, 2003; Nguyen, 2006; Coad, 2007). Protected area designation often ignores these traditional systems and boundaries, removing the power of community institutions to control land use (Kaus, 1993; Bedunah & Schmidt, 2004; WRI, 2005). The loss of power and changes in landscape divisions can weaken local community institutions, traditional community structures and cultures. Conflict within the community may follow, as different groups fight for the control over natural resources (Aberkerli, 2001, Ostrom, 1990). Ethnic heterogeneity caused by migration or displacement also tends to dilute community solidarity (Ostrom, 1990) and may cause inter-ethnic conflicts in resource use.

If an official switch from community to state control is not accompanied by effective enforcement, this can lead to a situation in which neither old nor new rules are upheld, resulting in destructive land use and negative livelihood consequences (Bedunah & Schmidt, 2004). For example, the forest landscape of 60 km^2 surrounding a village in Gabon was traditionally split into 56 named areas, often delimited using rivers and hills (Coad, 2007). Separate village clans or families had owned and defended each named area, but reclassification of the land as 'government-owned forest' in the 1960s had weakened these traditions, resulting in a more open-access system of land use that community chiefs have little power to regulate.

4.3. Restricted access to resources

Almost by definition protected areas will result in resource restriction to local communities, with the level of restriction varying with the individual characteristics and management of each area. According to the World Bank, resource restriction is also a form of displacement. Even if communities are allowed to remain within or adjacent to protected areas, the loss of land use rights can produce many of the same risks outlined for displaced people by Cernea (1997). However, few studies have quantified the impact of restricted use on local communities. As with displacement, most literature details individual case studies, which describe, but do not quantify, the impacts of restricted access. This may partly result from the large number of costs associated with restricted access, some of which are intrinsically hard to quantify (such as social, cultural or health impacts). The lack of consensus on the methods used to quantify impacts may also be a contributing factor. This is considered further in a discussion document (Burgess, 2007), produced in conjunction with this review.

A review of 31 of these case studies suggests that there are general differences in approach between IUCN management categories. The nine category VI protected areas studied all allowed 'sustainable use' for local communities, whilst of the 25 category I-IV protected areas, only seven allowed extraction of forest products for livelihood support. A more comprehensive review of protected area restrictions, costs and benefits, over a range of management and governance strategies would help identify the factors that have the most impact on livelihoods.

4.3.1. Access to forest products

As discussed, many forest communities rely heavily on forest products for consumption and commercial use, both as a 'daily net' and a 'safety net'. In Central Africa, forest communities generate 67% of their total income from hunting and gathering, and only 33% from agriculture, labour and employment; which illustrates how vulnerable forest communities can be to changes in forest access (Cernea & Schmidt-Soltau, 2006). A recent survey of World Bank assisted projects identified 120 projects with restriction of access (GEF-ME, 2005). Numerous recent case studies have found that protected area designation results in restricted access to forest resources, including firewood, bushmeat, building materials, forest leaves fruits and vegetables. Examples include Barombi Mbo Forest Reserve, Cameroon (Ngome, 2006), Buxa Tiger reserve, India (Sharma *et al.*, 2004), Sarstoon-Temash National Park, Belize (Beltrán, 1998), Ranomafana National Park, Madagascar (Ferraro, 2002) and Annapurna Conservation Area, Nepal (Bajracharya *et al.*, 2006).

Firewood restrictions have been reported as being particularly problematic (Abbott & Mace, 1999; Vedeld *et al.*, 2007; Bajracharya *et al.*, 2006), as wood provides up to 70% of the energy consumed in Africa (Murray & Montalembert, 1992). In Malawi, 90% of the primary energy supply is provided from fuelwood, whilst there are restrictions on collection of firewood in 55% of woodlands and plantations (Kayambazinthu, 1988). The effects of fuelwood restrictions have been studied in the Mount Elgon National Park, Kenya (Ongugo, 2002). The Kenya Wildlife Service advocates a total ban on use of the forest's products, and members of the community must walk long distances to collect firewood from unprotected forest blocks. This has led to tension between the community and the Service and disregard of the National Park's regulations.

Restrictions in access can also cause significant changes in the diets of rural communities. Leaves, fruits and vegetables collected in the forest provide many people with vitamins and minerals (Foppes and Ketphanh, 2004), and bushmeat provides from 30 to 80% of the daily protein requirements of rural communities in the Congo Basin (Wilkie & Carpenter, 1996). It has been suggested that the establishment of the Ranomafana National Park in Madagascar may affect community health by restricting access to indigenous medical plants, reducing protein intake from wild crayfish and reducing the purchase of fat and oils, as a result of reduced household incomes from forest products (Ferraro, 2002). Conversely, in the Yucatan peninsula of Mexico, restriction of access to forest resources has increased Mayan dependency on purchased items, leading to a decline in overall nutrition (Leatherman & Goodman, 2005).

Reduced forest access can also remove the 'safety net' of alternative harvests when crops fail. This increases the impact of agricultural food or income losses, and the chance of households failing to achieve their minimum income requirement (Ferraro, 2002).

4.3.2. Access to land and employment

Forests surrounding villages are sometimes used for extensive grazing. Shifting cultivation, or 'slash and burn' agriculture, is also often practised by village communities. Forested land is clear-felled, used to grow crops, and then left fallow, allowing secondary forest regeneration and soil renewal before the forest is cut again on a multi-year rotational cycle (House, 1997; Coomes *et al.*, 2000). Individual households will often re-use the same areas after decades of regeneration. A farmer who knows that the land will pass to his or her children through the traditional familial system of agriculture is more likely to invest in the land than a farmer who knows that the land belongs to the state (Powell, 1998; WRI, 2005; Ngome, 2006).

Restrictions on forest use that reduce the amount of land available for agriculture can alter traditional land-use practices, reduce incomes from livestock and increase land degradation bordering the forest by concentrating cultivation or livestock in smaller areas (Bedunah & Schmidt, 2004). For example, when the Doi Inthanon National Park was established in Thailand, the traditional shifting cultivation system was perceived to be unsustainable and made illegal. Financial compensation was only awarded to some ethnic groups, and no alternative income generating projects were set up to compensate for the loss of livelihood (Nepal, 2002).

When protected areas are designated, extractive industry permits are also generally revoked. Where present, these industries are often the main form of formal employment for local communities, and their closure can therefore have a major impact on incomes. In 1998, logging of natural forest along the Yangtze River in China was banned, to reduce environmental degradation. This resulted in the loss of an estimated 1.1 million jobs locally, and of many social services in the region, such as education and health care, which were originally provided by state-owned forestry companies (Kaimowitz, 2003).

Where local people depend on forest land for subsistence or employment, but protected area or REDD goals necessitate restrictions upon use, a 'do no harm' ethic would imply a responsibility to ensure that suitable alternative livelihoods are developed.

4.3.3. Commercialization of forest products and services

Access restrictions on forest products can often result in an increased dependence on employment or commercial activities such as intensive agriculture, shifting communities towards a dependence on market economies. It may also result in these products becoming more valuable through their rarity, becoming privatised or commercialised within local communities. Tourism in protected areas can also put a commercial value on the forest ecosystem itself and the traditions and culture of communities (Rasker, 1993; Krech, 2005).

This commercialisation can change the way in which local communities perceive their environment (Pfeffer *et al.*, 2001). The income associated with ecotourism can change local perceptions of the value and use of their surroundings (Hough, 1988; Forbes, 1995; West *et al.*, 2006; West & Carrier, 2004), changing forests' value from cultural or spiritual to commercial (Brown, 1999), and increasing the perceived value of specific forest products (Vivanco, 2001; Merlin & Raynor, 2005). In a non-forest example, the sport hunting of ibex in Pakistan changed local hunting practices and beliefs, as commercial incomes became available to the local community (MacDonald, 2004).

As discussed, different sections of the community use forest products in different ways. As forest products become more commercialized, the richer sections of the community often have more ability to exploit them. This can increase differences in wealth within the community, even increasing poverty for marginalized groups (Arnold & Perez, 2001). Increasing commercialization, and the profits that ecotourism brings, can put pressure on local political systems, causing disputes within the community and producing new class systems of those who profit and those who do not (West *et al.*, 2006). Similarly, if forest resources are perceived to be valuable, it can encourage migration to the area and increased stress on resources. These issues will need to be addressed if markets are to be developed for the carbon storage capacity of forests through REDD.

4.3.4. Restricted access, community involvement and conservation success

As with displacement, reduced use of forest resources can result in conservation benefits. A recent review of the effectiveness of protected areas has suggested that more restrictive protected areas are more successful in reducing deforestation than those with less restrictive access (Clark *et al.*, 2008). However, displacement of forest use from within the protected area can add to degradation of land around its boundaries.

Where disempowered communities remain within or around the protected area, and forest laws are weakly enforced, compliance with restrictions on resource use is less likely (Seeland, 2000; Ongugo, 2002; Bedunah & Schmidt, 2004; Scherl, 2004). Accelerated extraction of resources have even been reported, where communities fear impending loss of forest use (e.g. in the northern Yunnan, China; Harkness, 1998), have lost local laws governing forest use due to changes in land tenure (Section 5.2), or are protesting against the protected area by undermining its conservation goals (Martyr & Nugraha, 2004). The recognition of these tensions has contributed to the growth of community involvement in protected area management.

4.4. Human-Wildlife Conflicts and Degradation of Resources

Human-wildlife conflict is increasingly emerging as an issue where there are increasing human populations, decreasing habitat for wild fauna and/or successful conservation practices leading to increased wildlife numbers (Saberwal *et al.*, 1994). The wildlife problems that can be encountered by local communities living close to forest protected areas fall into two main categories: damage to resources such as crop raiding and livestock predation, and threats to human life. Livestock may also face a risk of disease transfer from wild ungulates (Metcalfe, 2003). Conflicts with carnivores are more likely to fall into both categories, but herbivores such as mountain gorillas and elephants can also do so (Treves & Karanth, 2003; Hoare, 1999). Larger animals typically require larger home ranges and more food resources to sustain a viable population (Macdonald & Sillero-Zubiri, 2002), causing them to extend their range beyond the limits of small protected areas into neighbouring land, entering into conflict with local communities.

4.4.1. Crop raiding and livestock losses

Crop damage and predation of livestock have been studied in much more detail than other protected area livelihood impacts. The financial losses involved can be measured relatively easily compared with the many, and sometimes intangible, costs of displacement, resource restriction and social and cultural change.

Crop raiding can occur in farms inside and on the borders of forest protected areas that host the relevant wildlife species. Elephants are often mentioned as the most difficult and damaging species to defend against (e.g. Bauer, 2003; Madhusudan, 2003; Kideghsho *et al.*, 2007). Other species such as baboons, monkeys, civet cats, wild pigs, and other ungulates are also recorded frequently (Weladji & Tchamba, 2003; Bajracharya *et al.*, 2006). In forest areas, crop raiders are generally large-bodied species in forest areas, whereas in savannahs small-bodied species such as rats can cause just as much damage (Naughton-Treves, 1998). Large cats such as lions and tigers, and wild dogs, are the main species to prey upon livestock (Bedunah & Schmidt, 2004; Kideghsho, 2007; Bauer, 2003).

Case studies from Nepal, India, Indonesia, Cameroon and Uganda illustrate that human-wildlife conflict can be most problematic for communities living inside or close to protected areas, with 74 to 90% of farmers reporting crop raiding (Sekhar, 1998; Naughton-Treves, 1998; Weladji & Tchamba, 2003; Bajracharya *et al.*, 2006; Linkie *et al.*, 2007). Predation of livestock also has a significant livelihoods impact, although the effects may be felt by fewer farmers. Villagers living within the Bhadra Tiger Reserve, India, lost 12% of their livestock per year (Madhusudan, 2003), whilst 44% of farmers in and around the Annapurna reserve, Nepal, and 28% of farmers in the Benoue National Park, in northern Cameroon (Weladji & Tchamba, 2003) reported livestock predation to be a problem. Each household in Annapurna lost a quarter of their annual maize production to crop raiding (Bajracharya *et al.*, 2006). In India, on the border of the Sariska Tiger Reserve, 27% of the gram harvest was lost (Sekhar, 1998), and in Bhadra Tiger Reserve, crop raiding by elephants led to grain losses of 11% (Madhusudan, 2003).

These losses can result in serious reductions in agricultural incomes. In Benoue, 18% of livestock income was lost on average. Losses may not be spread evenly, with some farmers losing an entire year's crop to one night of elephant damage. The risks can be so high as to be prohibitive; Studsrød and Wedde (1995) found that some farmers around the Royal Bardia National Park, Nepal had opted to keep their fields fallow rather than risk crop raids. Such losses affect the food security of local communities directly through a reduction in staple food grains such as maize and millet (Banskota & Sharma, 1995), and indirectly through reduced household incomes (Weladji & Tchamba, 2003).

As wildlife populations increase within effective protected areas isolated within a rural matrix, the incidence of human-wildlife conflict is likely to rise. Increased conflicts may also result from increasing human populations along protected area boundaries, as a result of migration in search of new resources and opportunities (West *et al.*, 2006).

4.4.2. Threats to human life

Globally, wildlife is a minor cause of human death. Lions, tigers, leopards, pumas, hyenas and bears kill only a few hundred people each year (Macdonald & Sillero-Zubiri, 2002), but these casualties are often concentrated in relatively small regions. Where attacks are frequent, negative attitudes to carnivores are unsurprising. It is estimated that between 36 and 100 people are killed by tigers in the Sundarbans mangrove forests of Eastern India each year (Macdonald & Sillero-Zubiri, 2002). In the Gir forest in India, successful Asiatic lion conservation has led to an increase in attacks upon humans, as the protected area is too small to support their increasing numbers (Saberwal *et al.*, 1994). 193 attacks by Gir lions were reported between 1973 and 1991, averaging 14.8 attacks and 2.2 deaths annually (Saberwal *et al.*, 1994). These attacks can be attributed to a wide variety of triggers, ranging from predator defence of their kill, increased human contact as a result of lions being attracted to tourist areas using baits and the influence of climate (Macdonald & Sillero-Zubiri, 2002; Saberwal *et al.*, 1994; Treves & Karanth, 2003). Attacks increased dramatically in the period during and immediately following a major drought, because lions needed to roam further in search of prey; and because at night, communities began to keep their livestock much closer to, and even within villages in an attempt to ensure their survival (Saberwal *et al.*, 1994). This increased the likelihood of human contact with hunting lions. Attacks also increase during the wet monsoon season, when cooler daytime temperatures allow the lions to roam further and for longer periods, and denser vegetation cover increases the likelihood of accidental encounters (Saberwal *et al.*, 1994).

The effect of wildlife attacks upon local communities needs little explanation, and the economic impact on individual households can be extremely large, including the loss of labour and income (Tiffen *et al.*, 1994).

4.4.3. Wildlife damage and conservation success

Human-wildlife conflict can damage the relationship between local communities and protected area administration (Songorwa, 1999; Infield & Namara, 2001; Weladji & Tchamba, 2003; Gadd, 2005; Kideghsho, 2007; Linkie *et al.*, 2007). Examples include the strong community opposition to the conservation programme in the Selous Game Reserve, Tanzania (Songorwa, 1999), and the negative perception of wildlife conservation held by local communities in Laikipia, Kenya (Gadd, 2005). In the Kerinci Seblat National Park, Sumatra, farmers set snares in the farmland bordering the park, killing a range of wildlife, including tigers (Martyr & Nugraha, 2004). In Nepal, where livestock losses to the snow leopard and wolf populations amounted to between a quarter and a half of average annual income for affected Nepali farmers (Macdonald & Sillero-Zubiri, 2002), the most widely held opinion amongst livestock farmers was that eradication of the predators was the only option.

Methods to prevent human-wildlife, such as barriers and patrols, have been implemented in many protected areas, but are not always effective (Thouless & Sakwa, 1995; Linkie *et al.*, 2007). Treves & Karanth (2003) review the approaches to managing human-carnivore conflicts, including eradication, controlled harvest, preservation of wildlife and translocation. They suggest that future management should employ selective population control to remove persistently offending individuals only, and only for non-critically endangered species. Translocation is also advocated where it can be achieved successfully (relying upon territorial vacancies a significant distance from the removal site), but is considered to be unsuitable in many cases due to high mortality rates and large costs. Non-lethal deterrents, including aversive stimuli are economically infeasible for many of those affected. Interventions to modify human and livestock behaviour, for example introducing guard animals (llamas, dogs and donkeys) are being researched, and may provide an important mechanism by which local communities can feasibly reduce livestock predation and crop damage. In the case of threats to human life, education programmes are essential to increasing awareness of high-risk situations, although analysis of their effectiveness is urgently needed.

Where losses occur, there is often no or insufficient compensation (e.g. Sekhar, 1998; Bauer, 2003; Madhusudan, 2003; Weladji & Tchamba, 2003). Opponents to cash compensation schemes suggest that they may attract new residents to an area (Wells *et al.*, 1992). An alternative to cash compensation is to provide compensation 'in kind' by providing rights to fuelwood and fodder collection within the reserve (Studsrod & Wegge, 1995; Sekhar, 1998). A number of case studies have reported a positive impact on community attitudes to conservation, despite crop and livestock losses, where extraction rights have been granted (e.g. Studsrød & Wegge, 1995; Bajracharya *et al.*, 2006).

5. The benefits of protected areas for local livelihoods

Local livelihoods may be enhanced by diversifying sources of assets, or switching livelihood strategies to a singular but rewarding activity (Twyman, 2001). Diversification entails opening up the correct assembly of opportunities for a specific community (Salafsky & Wollenberg, 2000), which can be challenging to achieve. Despite the costs discussed above, protected areas can provide significant livelihood benefits to local communities. This section reviews the benefits of protected areas; both those provided by successful protection of forest ecosystem services, and those directly gained from the management structure of the protected area, ranging from direct income to provision of local amenities. Forest ecosystem services include supporting and regulating services, provisioning services, and cultural services, as defined in the Millennium Ecosystem Assessment (Figure 3).

It is sometimes difficult to recognise ecosystem services and to quantify them accurately, partly because they often provide indirect benefits, meaning that they remain poorly understood in relation to their importance (Myers, 1996). In 1997, Constanza *et al.* estimated the global value of biodiversity to be roughly $38 trillion, although this remains a highly controversial figure. Using a careful analysis of existing case studies, Balmford *et al.* (2002) found that the benefits of conversion of land (and subsequent loss of ecosystem services) were always outweighed by the costs. In each case, private benefits were accrued at the cost of social (community) benefits.

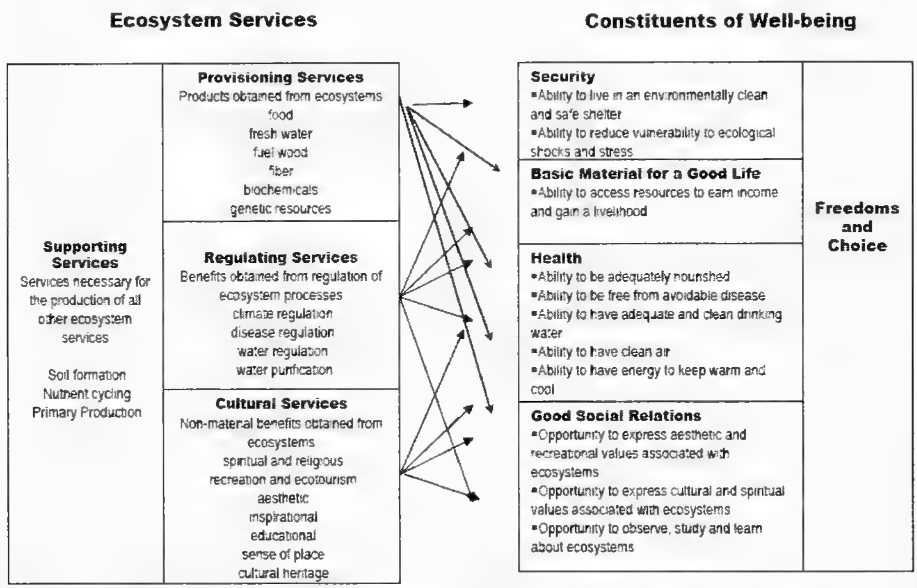

Figure 3: Millennium Ecosystem Assessment breakdown of benefits provided by ecosystems (Zakri, 2003). Although the diagram was not created specifically for forest ecosystems, all of the identified benefits are relevant.

5.1. The effectiveness of forest protected areas in biodiversity conservation

Forest protected areas and community conservation initiatives generally have lower deforestation rates than the surrounding non-protected areas (Clark *et al.* 2008). Less has been published, on the effectiveness of protected areas in conserving the animal and plant species contained within them, although the studies that have been carried out are often positive. WWF's analysis of over 200 forest protected areas suggested that biodiversity condition in protected areas was perceived to be good, and suggested that protected areas with an IUCN management category of I or II were likely to be more effective than less restrictive categories such as V or VI (Dudley *et al.*, 2004).

The benefit of biodiversity conservation is clear at the global scale. Intact ecosystems are thought to have more resilience to change, and to provide more ecosystem services (e.g. Cardinale *et al.*, 2006; Fox, 2006). However, the direct benefits to local livelihoods depend upon protected area management strategies: the inclusion or exclusion of those local communities and their livelihood activities, or the sharing of protected area benefits with surrounding communities.

5.2. Supporting and regulating services

Supporting and regulating services include generating and maintaining soils, primary production, sustaining hydrological cycles, runoff control, prevention of soil erosion and storing and cycling essential nutrients. For example, the forests of the Korup National Park, Cameroon provide flood control for agricultural land, and help to sustain downstream mangrove fisheries. The annual net benefit of these watershed functions has been estimated at US$85 per hectare of forest (Ruitenbeck, 1992; Myers, 1996).

Local communities may not recognise or value these services when their benefits accrue at the regional, national and global scales (Myers, 1996), especially given that the costs of protection are mainly incurred at the local scale (Balmford *et al.*, 2002). There may also be trade-offs between short-term and long-term benefits. An often-cited example of ecosystem service loss follows the conversion of forest land to rangeland for cattle ranching. Whilst economic returns can be high for the first few years, soil degradation and related nutrient depletion renders the land economically unviable and unable to regenerate as forest within a short amount of time, with corresponding long term impacts upon local livelihood options and security (Chomitz *et al.*, 2006).

Despite the problems surrounding the identification and distribution of benefits, many are recognised by local communities. In the Kerinci Seblat National Park, Indonesia, 94%, 88% and 66% percent of farmers, thought that forest loss would result in flooding, soil erosion and attacks from insect pests respectively (Linkie *et al.*, 2007). In the Annapurna community reserve, Nepal, communities have reported improved water resources after an increase in forest cover in the reserve (Bajracharya *et al.*, 2006). In Huertar Norte, Costa Rica, participants in a payment for ecosystem services scheme reported that reforestation in the area had improved soils and promoted tourism (Miranda *et al.*, 2002).

5.3. Provisioning services

It is often possible to identify and quantify the provisioning services provided by forest protected areas, as they are mostly direct benefits with visible economic impacts. The reliance of local communities on forest resources has already been highlighted, and it could be suggested that one of the biggest benefits of protected areas for local people is the protection of forest resources for future generations. Any analysis of the costs and benefits of resource restrictions must therefore be considered in the context of sustainability, and the livelihood costs that would result from the future loss of forest resources. There must be a balance between resource restriction and resource use if provisioning services are to be exploited by local communities today.

Brown *et al.* (2000) argue that the designation and sustainable use of protected areas can also lead to a more reliable resource base, whilst safeguarding the natural resources of a region for future use. The pattern of boom and bust in forest resource exploitation cycles can be replaced with a steadier economic base and the direction of benefits to local communities.

For these reasons, some communities have set up their own restrictions on forest use, citing the value of future use of forest resources as their primary motivation. Amongst several examples from Mexico, the community of La Trinidad has declared 29% of its forest as a biodiversity area and has begun reforestation in former agricultural plots (Bray *et al.*, 2003, Barton-Bray *et al.* 2003). In the USA, the Ute mountain reserve in Colorado was created by the Ute community for the preservation of cultural and natural resources (West *et al.*, 2006), and the native Americans of the Hoopa Reservation, California, have begun a programme of forest management and ecosystem enhancement, as part of a drive to rebuild community and culture (Baker, 2003). It is clearly possible for community management to deliver sustainable incomes and biodiversity protection at the local scale.

Resource extraction from protected areas, including timber and non-timber forest products (NTFPs), has been cited by local communities as one of the greatest available benefits (Sekhar, 1998; Bauer, 2003; Holmes, 2003; Bajracharya *et al.*, 2006; Allendorf *et al.*, 2006). At Lake Mburo National Park, Uganda (a Category II protected area), 44% of respondents involved in community conservation programmes reported that the protected area was positive because it conserved wildlife, and other benefits including the provision of water, grazing and access to protected area resources were reported (Infield & Namara, 2001). Similarly, an attitudinal survey in three wildlife sanctuaries in Myanmar, Burma (Categories II and III), showed that 45% of residents were in favour of the protected area, with 63% citing the conservation of natural resources as the reason for their support, and another 16% citing the extraction of natural resources from the protected area. Some studies have shown that an increase in forest production through protection has benefited local communities. In the Annapurna Conservation Area, Nepal (Category VI), 72% of community respondents gave the sustainable use of resources as their main reason for becoming involved in conservation projects set up by the protected area, and reported an increase in fodder, fuelwood trees, forest cover, water resources and wildlife populations (Bajracharya *et al.*, 2006). Such positive attitudes towards resource provision have been shown to lead to support of protected areas even when significant costs are incurred.

Whilst not focusing specifically on protected areas, Belcher *et al.* (2005) review the use and management of NTFPs and their implications for livelihoods and conservation. NTFPs can include food, fibre, incense, medicinal plants or rubber. A small percentage of NTFPs enter local, regional or international markets, providing a cash income to producer households. They are more often consumed directly by the communities that extract them; in either case they may act as a daily net (e.g. providing food for subsistence or sale), or an infrequent safety-net. In the latter case, lives may depend upon NTFP availability, particularly as the poorest groups within a community make disproportionate use of NTFPs.

Hamilton (2004) further argues that medicinal plants can be key to including local people in conservation strategies. Protected areas may be designated because of these plants, and/or their harvest may be encouraged to support local livelihoods. As well as playing an important role in traditional healthcare, some forest medicines are sold into the expanding market in herbal remedies, or used as the basis for manufacture of modern drugs (Hamilton, 2004).

Finally, the protection of wildlife within forests can have spillover effects onto surrounding areas, providing a 'source' population of wildlife, which will then move towards the 'sink' areas outside the protected areas as their populations move towards carrying capacity (Joshi & Gadgil, 1991; Novaro *et al.*, 2000; Salas & Kim, 2002). This can lead to crop raiding and livestock predation, as already discussed, but can also increase hunting opportunities for communities surrounding the reserve. An analogous situation is seen in marine protected areas, where no fishing zones act as reservoirs for fish stocks (Roberts *et al.*, 2001; Shanks *et al.*, 2003), delivering significant benefits to the fishing industry and fishing communities. These source-sink dynamics are currently being trialled as a way of ensuring sustainable bushmeat harvesting around the Nouabalé-Ndoki National Park, Republic of Congo (WCS-Congo, no date).

Sustainability in a livelihoods context can be defined as the ability of the stakeholder to withstand short-term fluctuations in circumstances, and adapt to longer-term fluctuations (Scoones, 1998). NTFPs in particular can be regarded as a 'sustainable livelihoods gateway', diversifying sources of income and sometimes providing a stepping-stone to a non-poor life (Marshall *et al.*, 2006). There is an obvious economic potential to the sustainable harvest of the more valuable NTFPs such as medicinal plants from protected areas, but their potential to benefit local communities is often affected by unstable markets, poor infrastructure and market access, and lack of bargaining power and market information (Belcher *et al.*, 2005; Marshall *et al.*, 2006). To realise this potential, investment would be required not only in the sustainable harvest and processing of NTFPs by local communities, but also in the facilitation of market access. An analysis of the sustainable yields possible is essential to underpin the long-term viability of NTFP-based livelihoods.

5.4. Cultural services

Deforestation, whether by communities or external actors, can result in the loss of the cultural traditions and religions connected with the forest (Dearden *et al.*, 1998). Whilst the services discussed in the preceding sections are probably more easily quantifiable, the cultural and social benefits of forest protected areas are an intrinsic aspect of their role in local livelihoods. McNeely (1994) discusses the opportunities for social benefits of protected areas, and concludes that protected areas can play a crucial role in maintaining cultural identity, preserving traditional landscapes and empowering local knowledge. For example, attitudinal surveys undertaken in the Wolong Biosphere Reserve, in southwestern China, indicate that the principal social development benefit of the reserve is that of increased social stability and cultural identity (Lü *et al.*, 2003). These benefits may be less visible and tangible, but can be highly valued by local communities. The inclusion of local communities in planning stages and management decisions can help protected area managers to reach beyond socio-political factors, such as land tenure and resource access, to make local populations also stakeholders in conservation priorities.

NTFPs such as medicinal plants can be symbolically and culturally important, providing livelihood benefits through their social significance. Their value is not limited to that of a financial asset. Hamilton (2004, p.1482) describes how medicinal plants may be "held in special religious, nationalistic or ideological esteem", which he argues can be advantageous for conservation, by helping to establish culturally-based support for the value of flora and fauna. Various attempts are underway to link conservation projects with local livelihoods through medicinal plant agreements; at Shey Phoksundo National Park, WWF-Nepal is implementing a community-based system of sustainable harvesting of medicinal plants alongside the facilitation of customary medicinal practices in local communities, working with traditional Tibetan medicine practitioners (*amchis*; Hamilton, 2004). A*mchis* are respected members of the community, and key stakeholders in conservation initiatives to maintain healthy resource bases of medicinal plants.

Local knowledge of traditional medicinal practices and resources can be a source of employment opportunities for local communities, to serve local needs and sometimes through assistance to research projects. Its value is also intrinsic, however; providing a source of local empowerment and identity. A thorough assessment of local livelihoods would include these more difficult to quantify aspects of the relationship between local communities and the environment.

5.5. Benefits from protected area management and infrastructure

5.5.1. Ecotourism

Tourism in protected areas generates revenue directly, and has therefore been purported to be the ideal alternative income base on which to build sustainable conservation and developmental projects within protected areas (Metcalfe, 2003). Various studies document local benefits either through sale of goods and services to tourists, or through sharing of a portion of direct revenues such as entrance fees (Adams & Infield, 2003; Bedunah & Schmidt, 2004; Bajricharya, 2006). Naidoo & Adamowicz (2005) argue that tourism projects in protected areas need to embrace the market values of biodiversity attractions, including the tourist's willingness-to-pay in their pricing. This could substantially increase the revenue acquired, and would be a significant source of funds for local communities involved in the projects. These funds may be shared directly, or invested in community activities. For example, at the KwaZulu Natal National Park in South Africa, a Community Levy Fund has been established, levying charges to visitors for developmental and economic activities both within and outside the tourism areas (Luckett *et al.* 2003). Many tourism projects also yield significant non-financial benefits through the development of skills and increased access to information, credit and markets (Smith & Scherr, 2003). These benefits can diversify options for financial assets and income, including migration opportunities provided by new roads, as well as employment opportunities within the protected area.

In some cases tourism has stimulated environmental damage and around protected areas, through resource extraction and development of infrastructure (e.g. Liu *et al.* 2001; Nepal, 2002). Sport hunting is a particularly controversial form of tourism, often difficult in forested areas, and not always compatible with protected area goals (McKinnon, 2001). However, some local communities have accrued substantial benefits from trophy hunting around protected areas facilitated by integrated conservation and development programmes (ICDPs) such as CAMPFIRE in Zimbabwe (Hasler, 1999). In Uganda, 12% of the revenue generated goes directly to local communities (Scherl *et al.*, 2004).

Overall, tourism is rarely shown to generate significant benefits on a large scale or to deliver sustainable alternative livelihoods (Cernea & Schmidt-Soltau 2006; Hackel, 1999). Where it does so, there are associated risks: communities can become dependent upon the income from tourism and associated industries (West *et al.*, 2006), which can be problematic for an industry highly susceptible to outside influences ranging from armed conflict to fashion.

Benefits generated by ecotourism are not always equitably shared within communities (West *et al.*, 2006; Kiss, 2004), as illustrated in Belize (Belsky, 1999), Vietnam (Rugendyke & Son, 2005), Indonesia (Walpole & Goodwin, 2001) and Madagascar (Ferrero, 2002). A study of six ecotourism projects in Africa, Latin America and Asia concludes that whilst the benefits are not fairly distributed, being concentrated on the semi-skilled sectors of a community, they can lift the recipients out of poverty and deliver fairly secure livelihoods (Ashley *et al.*, 2001). The success of protected area tourism in supporting livelihoods is closely linked to the other governance issues discussed in this review, such as the scope for local people to participate at the planning stage, and the breadth of consultations undertaken.

5.5.2. Payments for ecosystem services

Direct payment schemes, whereby non-governmental organisations (NGOs) or government agencies directly pay local communities or private landowners for conservation of ecosystems, their services or species, have become the focus of debate in recent years (e.g. Ferraro & Kiss, 2002; Grieg-Gran *et al.*, 2005). Advocates of direct payment schemes cite them as examples of 'win-win' conservation; directly valuing biodiversity, compensating local people for protected area impacts, and thus efficiently delivering measurable conservation results. The poor can benefit from increased income, diversified livelihoods, formalised land tenure and strengthened social organisations (Grieg-Gran *et al.*, 2005).

Many direct payments relate to watershed services. In the Hoopa reserve, California, between 1994 and 1998, local communities were paid by the Bureau of Indian Affairs for enhancing ecosystem services by restoring four main watersheds within the protected area, with the aim of reducing the sediment load flowing downstream (Baker, 2003). By 1996, sedimentation levels had been significantly reduced, and communities benefited financially, as well as from the enhanced environmental assets. In Pimampiro, Ecuador, the local government pays groups of farmers whose land is in the headwaters of the town water supply to protect their forests. These payments constitute on average 30% of household incomes, and assistance for soil conservation and organic farming is also received (Echavarria *et al.*, 2004). The 'payment for ecosystem services' (PES) scheme in Costa Rica has now been in operation for 10 years (Grieg-Gran *et al.*, 2005), pays rural residents approximately $35USD annually per hectare of forest protected (Ferraro & Kiss, 2002), and participants have reported a strengthening of community associations through the program.

The carbon market offers increasing opportunities for payments for restoration and retention of forest carbon. The potential impacts of payment for avoided deforestation are discussed in Section 8 of this review. Direct payments for other services such as biodiversity conservation are less well developed, but examples exist such as the protection of corridors between protected areas in Kenya (Guillison *et al.*, 2000).

Although the number of direct payment schemes is growing, they still cover only a tiny fraction of protected areas and forest communities. Detractors point to the institutional complexity that may be required to allocate and monitor payment schemes, especially where countries have many small, remote rural communities (often true of forest-dwelling communities). There are also risks associated with dependence on payments that may themselves lack a sound long-term financing strategy; but this is also true for many development projects.

Like other development initiatives, direct payment schemes may negatively impact the livelihoods of those not involved in the scheme through increased land-use restrictions and loss of land tenure, and those excluded from these schemes may be the poorest members of the community who lack the capital for initial involvement and have few initial land-use rights (Grieg-Gran *et al.*, 2005). To provide benefits, land tenure or equivalent rights must be established, and communities be involved in the decision making process.

5.5.3. Sustainable resource management and development schemes

Integrated Conservation and Development Programs (ICDPs) have become increasingly popular over the last few decades, as the potential benefits of linking biodiversity conservation with social and economic development have been understood (Wells, 1999). ICDPs aim to encourage local support for conservation through the provision of benefits from protected area management, or though investment in alternative livelihoods including tourism (Wells, 1999; Scherl *et al.*, 2004; West *et al.*, 2006). Various reviews have examined the success of ICDPs, often questioning their ability to realise development and conservation benefits in practice (Brandon *et al.*, 1992; McShane & Wells, 2004).

However, there are various examples of social and livelihood benefits provided by ICDPs (Barton Bray *et al.*, 2002; Bajricharya, 2006). In some cases, ICDPs have been vital for building local institutional capacity for strengthened protected area management, facilitating a greater role of local communities in conservation activities. Education and training for alternative livelihoods, and awareness campaigns have been cited as one of the major areas of success for ICDPs (Fomete & Vermaat, 2001; Mackinnon, 2001). Many ICDPs promote the sustainable harvesting of NTFPs (Mackinnon, 2001; Bajricharya, 2006). Other documented successes include the ICDPs associated with the Bwindi Impenetrable Forest and Lake Mburo National Park in Uganda, and the Amboro National Park in Bolivia, which have provided benefits from local forest management, improvement of local infrastructure, formalisation of traditional alternative livelihoods, control of crop damage and improved education and health services (Hughes & Flintan, 2001).

The majority of reviews do highlight complex issues with ICDPS that need to be resolved. It has been suggested that whilst the concept of linking local communities to resource management is sound, project design often assumes that attaching economic value to resources will achieve developmental objectives, without considering realistic social goals, establishment of tenure, and provision of mechanisms for equitable distribution of resources (Scherl *et al.*, 2004; West *et al.*, 2006; Ferrero, 2002). For example, in the Annapurna Conservation Area, Nepal (IUCN category VI) often documented as an example of success, there have been reports that many community members have yet to receive economic benefits. The otherwise successful ICDP in the Bwindi Impenetrable Forest in Uganda has not eradicated illegal use of forest resources (Scherl *et al.*, 2003). ICDPs often implement projects based on social units that differ from traditional norms, which can reduce local acceptability and equitable outcomes. There is often a failure to engage with multiple stakeholders and failure to include local communities in decision-making processes (McShane & Wells, 2004; Kideghesho, 2007).

There are also some concerns over the environmental and economic sustainability of alternative livelihood practices (Wells *et al.* 1992; Hughes & Flintan, 2001; Naughton-Treves, 2003; Scherl *et al.*, 2003;). It is clear that ICDPs have the potential to provide social benefits from protected area management in suitable areas, but that adequate institutional and capacity building arrangements are required for the distribution of social and economic benefits without undermining traditional systems. This requires local participation and consideration of the heterogeneous nature of community households to be considered in ICDP design.

5.5.4. Strengthened land tenure and protection from external threats

The legal provisions related to protected area designation can often provide local communities with formal protection that would otherwise be unavailable. This can protect traditional lands from external threats such as extractive industries or development. The benefits to local livelihoods may include preservation of the natural resource base, maintenance of access to traditional lands, and the opportunity for enhanced livelihood strategies in the future, which would be prevented by external ownership.

The designation of many protected areas has been driven by local communities to safeguard local resources (Laurence, 2000; Catton, 1997; De Lacy, 1994; Chapin, 2000; Colchester, 2000; Naughton-Treves, 1998; Schwartzmann *et al.*, 2005; Sohn, 2007). Whilst the direct benefits will depend upon protected area management strategy, designation is likely to be more favourable to local livelihoods than the transfer of land ownership to external companies. For example, when the Peruvian government declared that the Madre de Dios region of the Amazon was to be opened up to oil and gas exploration, locals and conservation groups objected to the plans (Chicchón, 2000). The issue was eventually resolved through the designation of the Bahuaja Sonene National Park in 1996, and an agreement that the exploration activities in adjacent regions, would return any land not desired for extraction programmes for inclusion in the protected area. Further examples include the Juna Tribal Park in Panama, created by the Kuna Indians to prevent encroachment by agriculturalists moving towards the coasts from the centre of the country (Chapin, 2000); the Arctic National Park in Alaska, set up as a collaboration between Inuits and conservationists in order to block an oil pipeline (West *et al.*, 2006); and the Kaa-Iya Park in Bolivia, managed by the Guarani-Izoceno people adjacent to their lands to buffer their territory from external threats (Naughton-Treves *et al.*, 2005).

When protected areas are established, there is an opportunity to formalise or transfer of land rights in the area or its buffer zone to local populations. This is less common within purely state-run protected areas, but has often been highlighted as a priority to facilitate successful conservation. Enabling sustainable resource extraction within a community that could face future relocation is very challenging (Roth & Haase, 1998). Land tenure rights not only secure financial assets for communities, but also increase physical assets (in the form of the land itself), and socio-political assets ('the power of legitimisation'; Winter, 1998). A legitimate claim upon the land should in theory increase motivation to conserve the natural resource base, though tenure rights alone will not resolve all conflicts with protected area goals.

6. The distribution of the costs and benefits of protected areas

The impact of protected area designation on an individual within a nearby forest community is likely to depend on his or her use of the forest, tenure rights and political power within the community. Those with high dependency on the forest, few land-tenure rights and little political influence will be most at risk from protected area designation, which in turn is likely to influence their attitude towards conservation. In semi-structured interviews to measure attitudes towards conservation in communities in and around protected areas, wealth, ethnicity, age, gender and occupation have all been shown to be important in predicting attitudes (Kideghesho, 2007; Infield, 1998; Infield and Namara 2001; McClanahan *et al.*, 2005; Allendorf *et al.*, 2006). Such variation within communities is often not reflected in protected area management, even where communities are involved in governance.

6.1. Wealth

Although richer members of forest communities are often the biggest harvesters of forest products, the poor can be more dependent on these resources, relying on the collection of forest products as a safety net during times of low-employment and food production. Forest restrictions can therefore have large impacts on the poorest sections of forest communities. In Ranomafana National Park, Madagascar, wild sources of food and income accounted for a larger share of household incomes among the poor, so that the establishment of the park was likely to affect these households the most, possibly increasing the size of loans during times of food deficit (Ferraro, 2002). Poorer households more frequently borrowed from richer household, reducing community harmony.

Land use restrictions can also increase the value of forest and forest products, increasing the pressure towards privatization of resources and therefore the value of land tenure within communities. This can further increase the wealth gap between land-owning and landless households; the latter may have to pay landowners for access. Compensation for loss of forest use or human-wildlife conflicts can be tied to land rights. For example, in the Bhadra Tiger Reserve, India, those claimants without evidence of land tenure were not compensated for livestock losses (Madhusudan, 2003).

Wealthy households and individuals often have more political influence within the community, which means that they are more likely to gain the benefits provided by protected areas than are the poor. In Nepal, community forest user groups can reflect existing hierarchies, and are biased in favour of wealthier, more powerful or higher caste individuals (Chakroborty *et al.*, 2001; Jones, 2007). Members of community institutions with political influence are in the best position to gain from ICDP and ecotourism benefits, which are often distributed using existing community institutions. Where benefits are distributed through local institutions, the rich elite are usually the main beneficiaries (Platteau and Gaspart, 2003). The same issue may also arise when protected areas are turned over to community management. In a Thai example, large areas of land originally designated as a forest reserve were given to a few wealthy individuals (Dearden, 1998).

Whilst there will be inertia in the system once benefits are distributed, The Lapande Reserve in Zambia recently changed the distribution of hunting concession revenues to local communities, after finding that distribution through community leaders resulted in benefits not reaching villagers most in need (Child & Dalal-Clayton, 2004).

6.2. Gender

Differences in forest use, tenure and power can mean that protected area designation has different impacts on men and women. Women often make more use of forest resources, but not necessarily the same resources that men use. Resource restrictions will therefore differentially affect the livelihoods of men and women: for instance, some protected areas allow NTFP and firewood collection, but ban hunting (e.g. Sekhar, 1998; Allendorf, 2006). The designation of the Ranomafana National Park resulted in a shift in agricultural activities from men to women, as the system changed from shifting cultivation (tavy) to irrigated rice paddies and gardens.

Traditional male tenure rights mean that many protected area managers ignore women when involving the community in protected area management (e.g. Sundberg, 2003). Payments for ecosystem services, and devolution of protected area management to local communities are often linked to existing land tenure rights, increasing the gap in political power between men and women. However, disruption of local institutions can sometimes favour women, creating opportunities for them to build new alliances and work outside of the community structure (West *et al.*, 2006). Handicraft production for tourist markets can provide women with new economic power (Vivanco, 2001). As development organizations increasingly recognize the importance of the role of women in household wellbeing, development projects around protected areas are also likely to increase the wealth and political power of women. Some protected areas are now using gender analyses within their management plans, to identify how men and women are likely to be affected by management decisions (Ongugo, 2002).

6.3. Ethnicity

Communities around protected areas often consist of multiple ethnic groups, which vary in their use of the forest, tenure rights and political power (Ngome, 2007). Compensation for protected area impacts is sometimes only offered to those ethnic groups that are 'indigenous' (Nepal, 2002; Ngome, 2007). The Doi Inthanon National Park, Thailand, provides an example of inconsistency in compensation due to ethnic differences. Two groups, the Karen and Hmong, live around the park and are restricted in their use of forest resources. However, Thai compensation policy supports opium growers over farmers, which has favoured the Hmong (Nepal, 2002).

If land is degazetted or otherwise returned to community control, the protected area resources and infrastructure can attract immigrants from different ethnic groups (often economic refugees), who have no existing tenure rights (Ngome, 2007). These recent immigrants can find themselves with even fewer access rights than in the original management system.

6.4. Age and education

Age is directly related to livelihood activities and forest dependency, with the young and old being particularly dependent on forest resources. In the Ranomafana National Park, old men are more likely to pursue shifting cultivation (tavy) than irrigated agriculture, because it requires less heavy labour. Similarly, households headed by young men were more dependent on tavy because they had not yet inherited land (Ferraro, 2002).

Where standards of living are rising, younger people may have had more access to formal education than older people. Education can provide increased employment opportunities, and therefore alternative livelihood strategies (Kideghesho, 2007). As gender is correlated with education, it can also be a factor behind gender differences in benefits. For example, within the Jaú National Park, Brazil, 61% of the literate population were men (Oliviera, 1999).

7. Quantifying the costs and benefits of protected areas

Very few studies have been able to assess whether protected areas have a net cost or a net benefit for local communities; which restricts comprehensive analyses as to whether protected areas generally help, harm or are neutral in their impact on local communities. This review found no studies that had directly measured the impact of protected areas on poverty, wealth or other variables that might indicate an individual or community's wellbeing. This is partly because these studies would require data from before and after a protected area has been established and must monitor these variables for sufficient time after establishment (5 – 10 years). The studies reviewed below that have considered overall costs and benefits have generally used either an economic cost-benefit analysis, or attitudinal surveys.

7.1. Economic costs benefit analyses

At least six papers have quantified costs and benefits for forest protected areas: three case studies from Madagascar (Kremen *et al.*, 2000; Ferraro, 2002; Carrett & Loyer., 2003), one from Kenya (Norton-Griffiths & Southey, 1995), one from Malaysia (Mohd Shahwahid *et al.*, 1999) and one from Cameroon (Yaron, 2001). Similar methods were used for each study: for the costs and benefits that could be quantified (generally spiritual, social and cultural value are not included), monetary values were estimated for a certain time period. These costs and benefits were then discounted for their future value, at discount rates of up to 20%. The use of such high discount rates to estimate the value of ecosystem services to future generations is increasingly controversial, especially where losses are irreversible, and can have a significant impact on the outcomes.

Kremen *et al.* (2000) looked at the net benefits of the Massola National Park, Madagascar, at global to local scales. The park was chosen because of its ICDP programme, which provided ample data for economic evaluation. The costs and benefits varied at the global, regional and local scale (Table 5). The protected area provided a net benefit over a ten-year timescale for local communities, mainly through the sustainable community forestry programme, and the use and protection from forestry of NTFPs. The costs were focussed at the national level, due to the loss of large-scale timber extraction, and a net global benefit was calculated, as a result of the carbon value of the forest protected from future logging activity.

Table 5: Costs and benefits of Massola National Park at different scales (Kremen *et al.*, 2000)

Local (net benefit)		National (net cost)		Global (net benefit)	
Cost	Benefit	Cost	Benefit	Cost	Benefit
Hill rice farming opportunity cost Forestry opportunity cost	NTFPs Community forestry Ecotourism	Hill rice farming opportunity costs Forestry opportunity cost Park management cost	NTFPs Community forestry Ecotourism Watershed protection	Donor investment in ICDP	Carbon Conservation Development

Ferraro (2002) produced a similar analysis of the costs and benefits of the Ranomafana National Park in Madagascar for local communities, using household surveys and questionnaires on the use of the park's land for agriculture, NTFP collection and timber before and after its establishment in 1991. The net cost of the existence of the protected area was estimated at US$39/year/household. The total opportunity costs for local communities of the establishment of the park were estimated at US$3.37 million. Health, cultural and social costs to the communities were not included, being more problematic to quantify. The global and national scale benefits of the park's designation were estimated to far exceed these costs. This imbalance could have been addressed effectively if conservation funding had been allocated to local conservation support projects and strategies to offset the costs incurred by locals, but at the time of the study the park had no tourism or ICDP projects.

The third study based in Madagascar (Carrett & Loyer., 2003) assessed the overall costs and benefits of the entire protected area network. The existence of the parks system created a net overall benefit and the land under protection was valued at US$15.7 per hectare, with a 54% economic rate of return. These benefits from biodiversity conservation, ecotourism and watershed protection were mainly felt at the regional to global scale, whereas the management and opportunity costs were incurred at the regional to local scale. Similar results were seen for the protected area network of Kenya (Norton-Griffiths & Southey, 1995), with the main costs being the opportunity costs of land conversion, which were borne at a local and national level.

This pattern of local costs and regional to global benefits was also observed in Selangor, Malaysia, where it was more profitable at the local level to unsustainably harvest timber than protect the forest (Mohd Shahwahid *et al.*, 1999). At the global scale, forest protection provided net benefits through flood protection and conservation of carbon stocks and endangered species. The study from Mount Cameroon had similar results, with local net benefits of forest conversion for small-scale agriculture rather than for forestry (Yaron, 2001).

Overall, the literature suggests that although protected areas provide net global economic benefits, if community conservation programmes are absent, they can often have a net cost at the local and national level, with developing countries and the rural poor bearing many of these costs (Balmford *et al.*, 2002). These studies can help to measure the level of compensation, or alternatives that ICDP and other community programmes may need to provide, in order to fully recompense local communities.

7.2. Attitudinal surveys

The most common method used to assess the effect of protected areas on local people is attitudinal surveys to measure their perception of protected areas. This review has identified ten attitudinal surveys. Positive or negative attitudes are sometimes correlated with protected area costs and benefits (Allendorf, 2006), but communities may undervalue protected areas, as many of the benefits of protected areas (such as forest products and ecosystem services) are future use values, and may not be perceived to be under threat by the community. There are currently too few studies to draw any major conclusions on whether people support protected areas generally, but the interesting finding from a few studies is that even with high costs, communities can support protected areas (Sekhar, 1998), generally citing the forest use benefits that they receive from the protected area.

7.3. Direct impact studies

No completed studies were identified that have looked at the direct effect of protected areas on local livelihoods and wellbeing. However, a paper by Wilkie *et al.* (2007) describes a project currently underway to directly monitor the livelihood impacts of the protected area network in Gabon. This five year project uses quantitative household surveys, using the methods of the World Bank Living Standards Measurement Study, to look at household wealth, health, incomes and consumption of natural resources and other goods. It will monitor the impact of the new system of protected areas on a stratified sample of 1 000 households.

8. Protected areas and livelihoods in the context of REDD

The extent to which the benefits from protected areas are realised by local communities is greatly influenced by the wider political and economic climate. The studies reported above typically assess costs and benefits at site level; but there is also a need to consider the wider macro-economic scale to set the results in context. The current discussions on carbon finance for reducing emissions from deforestation and forest degradation (REDD) have the potential to have major impacts upon the livelihood costs and benefits associated with protected areas.

REDD is likely to affect the protected area network by influencing which forests exist, and where, and who benefits from the services that they provide (Smith & Scherr, 2003; Brown & Corbera, 2003). The potential benefits to local communities of a carefully managed carbon finance mechanism could be considerable. The protection and management of carbon storage areas, whether protected areas or other sustainably managed forests, can draw upon lessons learnt from previous experiences of 'people and parks' (Bass, *et al.*, 2000; Asquith *et al.*, 2002; Peskett, 2007). Some concerns have been expressed that reducing deforestation and degradation of forest carbon stocks may entail a reversion to the 'fences and fines' approach of forest management (Orlando *et al.*, 2002; Griffiths, 2007). The current review makes it clear that denial of access to forest resources has negative effects on the poorest members of local communities, and that managed access and/or compensation generally yields better results for both conservation and livelihoods.

The development of a market for the carbon storage services of standing forest could provide the opportunity to engage a considerable amount of governmental and private sector funds in forest conservation on a scale that has not previously been seen. Whilst REDD is likely to involve national-scale policy changes and planning, forest management changes will have to be implemented at a site scale. It has been suggested that the benefits of international environmental markets, like those of protected areas, are often realised globally, whilst most of the costs are incurred at the local level (May *et al.*, 2004; Peskett, 2007). Clear governance, including well-defined property rights, is critical for emerging international markets (Landell-Mills & Porris, 2002). Lessons from existing carbon projects could help to inform the development of an international forest carbon market that also yields multiple benefits for conservation, livelihoods and ecosystem services.

In existing carbon markets, the transaction costs of projects tend to favour large operators at the expense of small landholders (Pfaff *et al.*, 2007). Carbon forestry projects have not been overly successful in equitable distribution of resources and are particularly weighted against those whose livelihoods are dependent upon less formal rights to forest resources, such as poor or landless households and women (Brown *et al.*, 2004; Grieg-Gran *et al.*, 2005); leading to the capture of most of the benefits by elite groups (Brown and Corbera 2003). Carbon markets have favoured middle-income communities in Mexico, even with allocation of tenure rights (Brown *et al.*, 2004). Similar issues have been identified for a number of forest carbon forestry projects; including the best example of an avoided deforestation scheme to date, the Noel Kempff Mercado Climate Action Project in Bolivia (May *et al.*, 2004; Nelson & de Jong, 2003. Brown & Corbera, 2003 Brown *et al.*, 2004; Grieg-Gran *et al.*, 2005; Griffiths, 2007). There are also several examples of well-designed carbon finance projects have led to significant livelihood improvements, mostly as a result of good governance and equal allocation of benefits (Brown *et al.*, 2004; Jindal, 2006; Peskett, 2007).

The issues seen in carbon projects, such as governance, tenure, and inequitable distribution of costs and benefits, are often the same issues identified as barriers to the provision of protected area livelihood benefits. In both cases, the importance of including local stakeholders at all stages of decision-making has been highlighted, along with allocation of ownership of resources (Boyd *et al.*, 2005). The literature suggests that national governments often do not allocate land use rights in current protected area networks, and that more powerful stakeholders can reap the benefits at the expense of local communities. These factors therefore need serious consideration if carbon finance is to be created for forest protection, as increased land values may exacerbate these tendencies.

If the obvious issues can be addressed, avoided deforestation and other carbon storage schemes could provide much needed funds for conservation and development. Addressing the root causes of deforestation is likely to require improved governance of forest areas rather than heavy restrictions on the activities of local communities (Chomitz, 2006). REDD implementation could provide the incentive for governments to strengthen policies for forest protection and settle tenure issues. An increase in the economic value of standing forests could also have positive impacts on the livelihood benefits of protected areas.

9. Conclusions

A large number of the rural poor rely on forest resources. The potential costs and benefits of protected areas to community livelihoods have been well documented, and there are a number of case studies that assess these costs and benefits at an individual site level.

The benefits of protected areas can range from the ecosystem services protected within the forest area, to direct and indirect benefits from protected area management. In the case of the former, such benefits include watershed and soil erosion protection, and provision of forest resources such as NTFPs. The extent to which these resources can be used, however, is largely dependent upon the protection status and management strategy of the area. Where protected areas (usually categories I-II) have strong restrictions on resource use, these benefits are not always realised. Additional benefits from protected area management can include revenue from ecotourism, direct payments for conservation, development schemes, employment, secured land tenure and protection of resources from external threats. The provision of these benefits to local communities is again largely dependent upon the mechanisms in place for benefit-sharing through management structures, community involvement in governance or clearly allocated property rights.

The costs of protected areas can include: displacement of local communities, changes in traditional land tenure, denied or restricted access to resources, loss of employment, crop damage and livestock predation. Of these costs, displacement is arguably the most damaging to livelihoods, but is relatively infrequent. Almost 90% of displacements have occurred within the most restrictive IUCN management categories (I and II). Changes in tenure from traditional property rights systems to government owned land, can also have significant livelihood costs; particularly when communities are not involved in land use decisions.

Despite the many studies on this topic, few have attempted to assess the net livelihood costs and benefits, most likely due to the lack of a consistent methodological framework with which to do so, and the difficulties in placing a monetary value on some of the livelihood aspects investigated. However, one of the clearest conclusions arising from this review is that costs and benefits are not distributed evenly throughout local communities, with the more prominent members of society typically capturing most of the benefits whilst suffering less of the costs. This trend seems to be a combined result of pre-existing forest resource use patterns and inadequate distribution of benefits, and is often true regardless of the protected area status or the level of community involvement in governance.

Protected area management can provide direct benefits to communities; but can also restrict access to resources, alter local power structures, and change social/traditional values and behaviours. Some attitudinal studies have found positive community perceptions of protected areas, whilst other communities recognise the need for the protected area for its conservation value, but are negative towards the restrictive management structures. Whilst management strategies are not specific to protected area status, strictly protected areas lacking in community involvement can have major livelihood impacts that cause conflict between local communities and protected area management.

To further assess the costs and benefits of protected areas to local livelihoods, increased efforts are required into the standardisation of methodologies for social impact assessment. Further study is also required into the combined effects of protection status and governance on the costs and benefits of forest protection. Existing studies often record the IUCN management category of the area, without documenting the level of community involvement.

All management categories of protected area can involve communities in governance and establish clear property rights, but category V-VI areas with a focus on sustainable development are more likely to do so. Both community management schemes and protected area management allowing sustainable use of forest resources have met with varying degrees of success. Whilst tangible benefits are frequently reported, significant costs can still be incurred by communities if management and institutional capacity is lacking, and issues of governance and tenure are not resolved.

The costs and benefits of protected areas for local communities are therefore highly dependent upon protected area management and governance. If strict protection is implemented, local people need to be involved in management and compensated for losses, or the likely result will be that they will not cooperate with protected area management. Even for less restrictive protected areas, issues of tenure and equity may remain. Community-based forest management could have an important role to play in the protected area network, increasing the area and success of forest protection, but will require social investment and capacity building

These conclusions are particularly relevant in the context of REDD. An analysis of livelihood costs and benefits in existing forest carbon markets has identified issues similar to those in protected area management; including lack of established tenure and the inequitable distribution of resources; particularly for the landless members of society. Increased finance could exacerbate these issues, and there is the potential for the protection of carbon areas to positively or negatively affect local livelihoods. The potential exists for REDD mechanisms to reduce the large-scale drivers of deforestation, secure land tenure rights in forest areas, and increase the potential benefits to local communities from conservation through community management regimes. These past experiences indicates that involvement of local people in planning and implementation of REDD, and ensuring sharing of the benefits from REDD finance is likely to result in a more sustainable long-term solution to deforestation.

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