

The Value of Forest Ecosystem Services to Developing Economies

Katrina Mullan

Abstract

This paper assesses the scale of the potential co-benefits for residents of developing countries of protecting forest ecosystems in order to mitigate climate change. The objective is to improve understanding among development practitioners of the ways in which services provided by forest ecosystems can also make important contributions to achieving development objectives such as improvements to health and safety, and maintenance of food and energy security. This is achieved by reviewing empirical studies that estimate the value of specific ecosystem services derived from forests in order to evaluate and describe the current state of knowledge on how the wellbeing of local people is likely to be affected by the introduction of global mechanisms for avoided deforestation in developing countries. There are four main ways in which wellbeing can be affected: 1) forests provide soil protection and water regulation services, which in turn reduce waterborne diseases, maintain irrigation water supply, and mitigate risks of natural disaster; 2) forests provide habitat for birds, fish, mammals and insects that affect human health and income generation opportunities; 3) clearing forest through use of fire can lead to respiratory illness and property damage, particularly if the fires spread accidentally; and 4) tropical forests are particularly high in biodiversity, making them important locally as well as globally as a potential source of genetic material for new crop varieties and pharmaceuticals. Evidence on the size of these benefits suggests that while they are highly variable, households in or near forests and poor households benefit most from forest ecosystem services.

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The Value of Forest Ecosystem Services to Developing Economies

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Foreword

This paper is one of more than 20 analyses being produced under CGD’s Initiative on Tropical Forests for Climate and Development. The purpose of the Initiative is to help mobilize substantial additional finance from high-income countries to conserve tropical forests as a means of reducing carbon emissions, and thus slowing climate change.

The analyses will feed into a book entitled *Why Forests? Why Now? The Science, Economics, and Politics of Tropical Forests and Climate Change*. Co-authored by senior fellow Frances Seymour and research fellow Jonah Busch, the book will show that tropical forests are essential for both climate stability and sustainable development, that now is the time for action on tropical forests, and that payment-for-performance finance for reducing emissions from deforestation and forest degradation (REDD+) represents a course of action with great potential for success.

Commissioned background papers also support the activities of a working group convened by CGD and co-chaired by Nancy Birdsall and Pedro Pablo Kuczynski to identify practical ways to accelerate performance-based finance for tropical forests in the lead up to UNFCCC COP21 in Paris in 2015.

This paper, “The Value of Ecosystem Services to Developing Economies” by Katrina Mullan was commissioned by CGD to summarize the literature on the economic valuation of ecosystem services from tropical forests, demonstrating that the rationale for reducing deforestation extends beyond mitigation of climate change. The paper is intended to provide an up-to-date review accessible to the non-specialist that characterizes the literature, explains valuation methods, and synthesizes the findings of studies that rigorously apply such methods to monetize the benefits of forest conservation and the costs of forest loss.

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Key Findings

Forests in developing countries contribute to the health, safety, food and energy security, and income of local people through ecosystem services. There are four main ways in which wellbeing can be affected:

1) Forests provide soil protection and water regulation services. This affects health through access to water and mitigation of waterborne diseases, physical safety through reductions in flood and landslide risk, energy security through improved functioning of hydroelectric facilities, and food security through regulation of water for irrigation, particularly in periods of drought. Forests also physically influence wellbeing through the storm protection provided by mangroves.

2) Forests provide habitat for birds, fish, mammals and insects that contribute to food, income and health. For example, forests affect income by providing nursery grounds for commercially important fish and shellfish stocks, and habitat for birds and mammals that attract eco-tourists.

3) Clearing forest through use of fire can reduce wellbeing, particularly if the fires spread accidentally. In most years, these fires generate some local air quality problems and damage to property. However, in extreme years, weather conditions result in much more extensive fires with serious health impacts.

4) Tropical forests are particularly high in biodiversity, making them important locally as well as globally as a potential source of genetic material for new crop varieties and pharmaceuticals.

Evidence on the size of these benefits suggests that households in or near forests gain most. They benefit in multiple ways, in particular through reductions in waterborne, insect-transmitted and respiratory disease and improvements in agricultural productivity. Households in downstream areas of forested watersheds can also benefit considerably through improvements in health and safety and increased energy security.

Poor households benefit most from forest ecosystem services because they are often dependent on agriculture or fishing (where forests increase productivity) and vulnerable to the negative health effects and increased risks of natural disasters that can result from deforestation.

As these benefits are generally non-rival goods, the values vary substantially with the size of the affected population, being higher overall when there are more potential beneficiaries. As a result, the total value of ecosystem services will be greatest for forests in densely populated regions, particularly where households are dependent on agriculture or fishing, incomes are low, and/or risks of natural disasters are high.

1. Importance of ecosystems for human wellbeing

1.1 The benefits of forest protection

The scale of the Indian Ocean earthquake and tsunami in December 2004 was almost unseen in human history, with over 200,000 deaths in 14 countries. In those regions closest to the epicenter, little could have prevented catastrophic destruction. Further away, however, healthy coastal ecosystems substantially mitigated the impacts. Along one, otherwise homogenous, stretch of coast in Cuddalore District in Tamil Nadu, India, three villages unprotected by mangroves were entirely destroyed, while three protected villages were unscathed, and five within coastal tree plantations were only partially damaged (Danielsen et al. 2005). Similarly, in Phang-na, Thailand, large mangrove forests significantly reduced the impact of the tsunami in the north and south of the province, leaving protected villages undamaged (UNEP, 2005).

This is a particularly dramatic example of how forest ecosystems can directly affect human wellbeing, but other examples abound. In addition to their storm protection function, mangroves provide habitat for fish and shellfish. In the Mexican community of Campeche, a 3% reduction in mangrove area between 1980 and 1990 reduced annual revenue from shrimp harvests by \$279,000 (Barbier and Strand 1998). In Costa Rica, protection of forest fragments in the vicinity of a single large coffee farm increases profits by \$62,000 by providing habitat for bees that pollinate coffee plants (Ricketts et al. 2004). In southeast Asia, the haze from severe forest fires driven by forest clearing generated costs of \$674-799 million in medical treatment costs and losses to businesses in 1997-8 (Tacconi 2003).

Increasing recognition of the ways in which people are affected by the condition of ecosystems has led to programs to compensate those responsible for conservation through Payments for Ecosystem Services. These operate from the level of the individual watershed level to the global level. In the Rupa Lake watershed, Nepal, individual water users have set up the Rupa Lake Restoration and Fishery Cooperative to voluntarily make direct payments of \$3,400 per year to upstream villages in return for forest and land management that

reduces erosion and sedimentation (Pradhan et al. 2010). At the same time in China, policies for ecosystem protection and restoration have been introduced, with a total budget of \$40 million. (Bennett 2008).

Forest conservation is easy to view as a trade-off between local income generation, through clearing land for agriculture or logging for timber, and global environmental benefits such as biodiversity protection and carbon sequestration. Modern development has been based historically on the destruction and exploitation of natural ecosystems, from conversion of wild land for agriculture to urban construction and overfishing. For tropical forest countries, where millions struggle to meet their basic needs, it is hard to justify prioritizing conservation over improvements to living standards, particularly when wealthier countries did not. However, as the experiences in India, Mexico and Indonesia demonstrate, deforestation can also compromise development objectives such as health and physical safety; while forest protection can contribute to food security and income generation. As human populations grow and natural ecosystems become more scarce, the importance of these ecosystem benefits increases (e.g. Koch et al. 2009; Ghermandi et al. 2008). In addition, as will be shown below, the impacts of ecosystem degradation are felt most strongly in rural areas, where people are more directly reliant on functioning ecosystems for livelihoods, health and physical safety; and by the poor, who have fewer opportunities to substitute for the ecosystem benefits with alternatives such as stronger building materials, water filtration systems, or formal insurance markets.

Policies for conservation of tropical forests place limits on the use of forest land for economic activities such as logging or agricultural conversion. International transfer payment mechanisms for Reducing Emissions from Deforestation and Forest Degradation (REDD+) aim to compensate for economic opportunities forgone due to forest conservation. This compensation is important to mitigate potential impacts of forest conservation policies on economic development and poverty in a context where the primary beneficiaries of deforestation are residents of developing countries, while the benefits of reductions in global carbon emissions (or preservation of biodiversity) accrue globally. To the extent that residents of developing countries also receive benefits from the tropical forests in the form of ecosystem services, REDD+ will provide valuable co-benefits.

This paper assesses the scale of these potential co-benefits from forest protection for individuals and communities in developing countries. The objective is to improve understanding among development practitioners of the ways in which services provided by

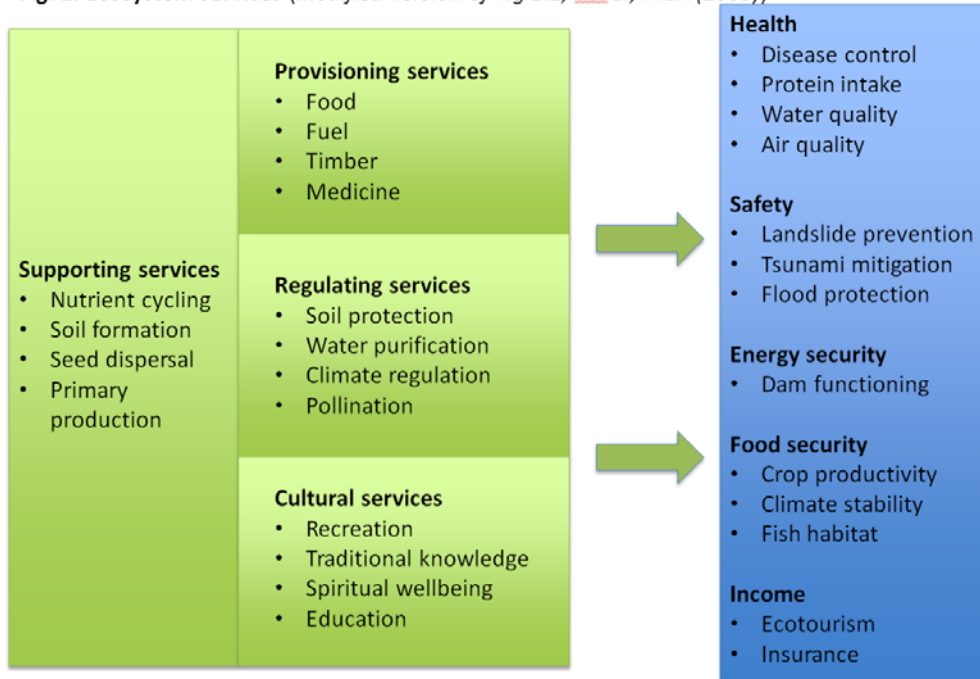
forest ecosystems can also make important contributions to achieving development objectives such as improvements to health and safety, and maintenance of food and energy security. This is achieved by reviewing empirical studies that estimate the value of specific ecosystem services derived from forests in order to evaluate and describe the current state of knowledge on how the wellbeing of local people is likely to be affected by the introduction of global mechanisms for avoided deforestation in developing countries.

The remainder of Section 1 and Section 2 introduce the concept of ecosystem services; describe alternative approaches to assigning monetary values to the non-market benefits of ecosystem services; and discuss the challenges associated with this type of valuation. Section 3 presents estimated values for different categories of ecosystem services from individual studies. The selected studies are those that use primary data; apply appropriate methods; and make explicit connection between a specified ecosystem change and a change in human wellbeing. The review only covers ecosystem services that accrue locally, although 'local' may be adjacent to the forest, downstream from the forest, or within the same region. It does not include benefits of forests for residents of developed countries such as ecotourism visits or the existence of endangered species unless these translate to income generation opportunities for local communities. It also excludes global benefits of forest protection such as reduced carbon emissions or protection of biodiversity. Sections 4 and 5 summarize the overall findings, and draw conclusions about the implications of policies such as REDD+ for local ecosystem service provision.

1.2 Ecosystem services

The concept of ecosystem services, i.e. that functioning ecosystems generate valuable flows of benefits to humans, was given increased public recognition through the Millennium Ecosystem Assessment (MEA). Launched in 2001 by the UN Secretary General and completed in 2005, the MEA brought together 1,360 experts to assess the consequences of ecosystem change for human wellbeing (Millenium Ecosystem Assessment 2005) A conceptual framework was developed to highlight the real impacts on human health, security, social relations and physical wellbeing.

Fig. 1: Ecosystem Services (modified version of Fig 1.1, Vol 1., MEA (2005))



Within the framework, ecosystem services are organized into four categories based on the type of benefits they provide. Specifically, provisioning services directly meet physical needs such as food, fresh water and fuel; regulating services indirectly contribute to health and safety through regulation of climate and disease, air and water purification, and prevention of soil erosion; cultural services provide nonmaterial benefits such as spiritual enrichment, cognitive development and recreation; and supporting services such as the production of oxygen and soil formation are necessary for the maintenance of all other services.

There are two important advantages to using the concept of ecosystem services. First, it focuses attention on the ways in which ecosystems help to meet basic human needs such as clean water, fuel, disease prevention or risk mitigation. It therefore makes explicit that conservation is not a question of people vs. nature, but rather involves trade-offs between different groups of people or different basic needs. For example, forest clearing may provide income for upstream households through the sale of agricultural products, but also increase the risk of waterborne illness for downstream households due to deterioration of water quality. Second, it not only highlights the ways in which ecosystem loss can have tangible costs, but also provides a relevant framework for quantifying those costs. Rather than trying to value the full range of benefits provided by a lake or a forest, the ecosystem services framework encourages the identification of changes in the flows of particular services as that ecosystem improves or deteriorates; and quantification of the resulting impacts on human

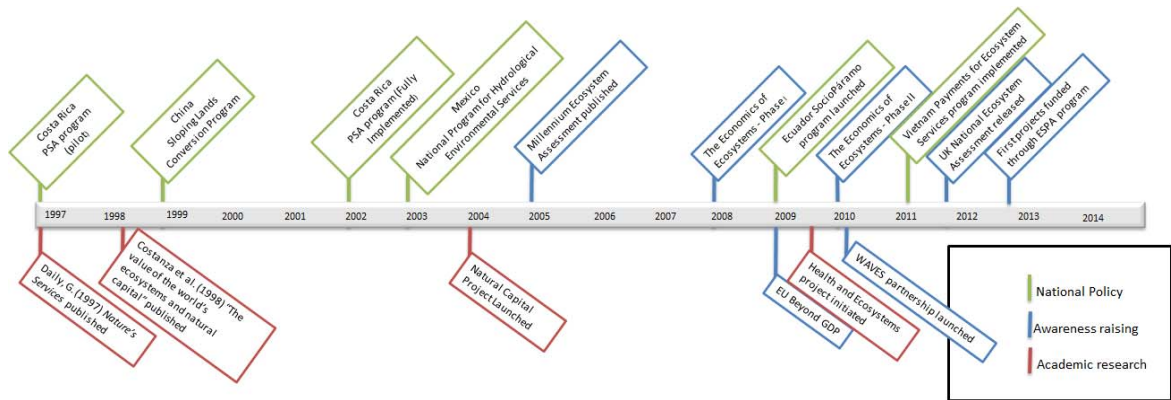
welfare. This is directly relevant to understanding the relative costs and benefits in terms of basic needs as ecosystems change.

Following early publications that primarily drew attention to the potential for assigning monetary values to ecosystem services (e.g. Daily 1997; Costanza et al. 1998), there are now a number of large interdisciplinary projects that aim to quantify the impacts of ecosystem change on human wellbeing. For example, the Natural Capital Project is developing methods to systematically quantify the flows of services from a given ecosystem, and how those flows would be affected by changes in the condition of the natural capital. Those impacts can then be compared with the gains from, for example, road construction or wetland development. The Health and Ecosystems project, a consortium of conservation and public health organizations and universities, is undertaking similar activities with a focus on better understanding the links between ecosystems and aspects of human health such as nutrition, disease, and vulnerability to natural disasters. The UK government is also funding research on the importance of ecosystem services in developing countries through its Ecosystem Services for Poverty Alleviation (ESPA) program. In addition to research activities, global policy initiatives such as The Economics of Ecosystem Services and Biodiversity (TEEB) and the Wealth Accounting and the Valuation of Ecosystem Services (WAVES) partnership are using and publicizing the latest research on the values of ecosystem services to influence decision-making at the national and global level. TEEB publications demonstrate the significance of ecosystems for global policy agendas including the CBD and UNFCCC; for local and regional policy making and urban planning; and for the private sector, and the WAVES partnership develops methodologies for ecosystem accounting, and works with individual countries to implement natural capital accounts.

As a result of increased awareness of the real costs associated with the loss of natural ecosystems, national governments have begun to implement policies with the explicit goals of protecting and maintaining ecosystem services. For example, the Costa Rican 'Pagos por Servicios Ambientales' (PSA), or 'Payment for Ecosystem Services', compensates individual landowners for reforestation and forest conservation. Although it grew out of an existing institutional structure for forest management, payments are now justified and targeted specifically to produce ecosystem services rather than to support the timber industry (Sanchez-Azofeifa et al. 2007). The largest reforestation program in the world, the Chinese Sloping Land Conversion Program, was introduced in 1999 with the dual objectives of watershed protection and poverty reduction following major flooding and drought in the

Yangtze and Yellow River basins that had been attributed to deforestation. Farm households receive payment in the form of cash, grain and seedlings in return for converting cropland to forest or grassland, with the intention to convert around 15 million hectares of land (Xu et al. 2010). More recently, among numerous others, Ecuador introduced the SocioPáramo program, which aims to both protect biodiversity, carbon and water, and alleviate poverty (Bremer et al. 2014) and Vietnam launched the Payments for Ecosystem Services program for watershed protection (Suhardiman et al. 2013).

Fig 2: Timeline of key events relating to ecosystem services



This widespread recognition of the importance of ecosystem services has led to diverse research efforts aimed at valuing the benefits of ecosystem conservation in different locations. These values are necessary for comparing the costs of ecosystem losses with income generation or development benefits associated with investments such as agricultural land clearing or road construction. They are also crucial to underpin policies such as payments for ecosystem services that aim to protect natural ecosystems. However there are substantial challenges associated with estimating these values. In particular, estimation requires knowledge from multiple disciplines to model both the relationships between management choices and ecosystem condition, between ecosystem condition and ecosystem service provision, and finally between ecosystem services provision and human wellbeing. All of these relationships are complex, frequently non-linear, and highly location specific. The following section describes the methods that are used for economic valuation of non-market services along with the key challenges. The subsequent sections review the best-available evidence on the contribution of forest ecosystems to different aspects of human wellbeing, focusing specifically on the benefits for those in developing countries.

2. Methods for valuing ecosystem services

2.1 Alternative approaches

The ecosystem services framework provides a useful way to think about the value of the environment, as it focuses directly on the relationship with human wellbeing. However, since ecosystem services are generally not traded in markets, their values are not expressed as market prices and therefore need to be estimated by some means if they are to be compared to other values in monetary terms. Methods that have been developed for the estimation of non-market values include direct market value methods, revealed-preference methods, and stated-preference methods. These differ in the source of the information used to infer the size of the welfare changes experienced by individuals or the changes in profits accruing to firms.

Direct market value methods combine information on the impacts of changes in ecosystem services on production or consumption of goods with market prices of those goods. The most widely used of these methods is the change in productivity, or production function, approach, whereby a theoretical model or regression analysis is used to estimate the physical effects of changes in ecosystem services on economic activity, and the corresponding value of the resulting changes in economic output. For example, Pattanayak and Kramer (2001b) estimate the impact of changes in availability of water for irrigation on profits from crop production in Indonesia, while Pattanayak and Wendland (2007) estimate the impact of changes in water quality on human health. Similar models are used to estimate avoided morbidity or mortality, e.g. Frankenberg et al. (2005) use panel regression to estimate differences in the prevalence of respiratory problems in regions and time periods with and without forest fires. These impacts may be valued based on treatment costs or work-days lost, or presented in non-monetary terms as risk of illness or numbers affected. An advantage of this method is that it directly values the changes in wellbeing, and can be straightforward to implement if the production function is understood. However, quantitative information on the relationship between ecosystem condition and human activity or outcomes is frequently absent.

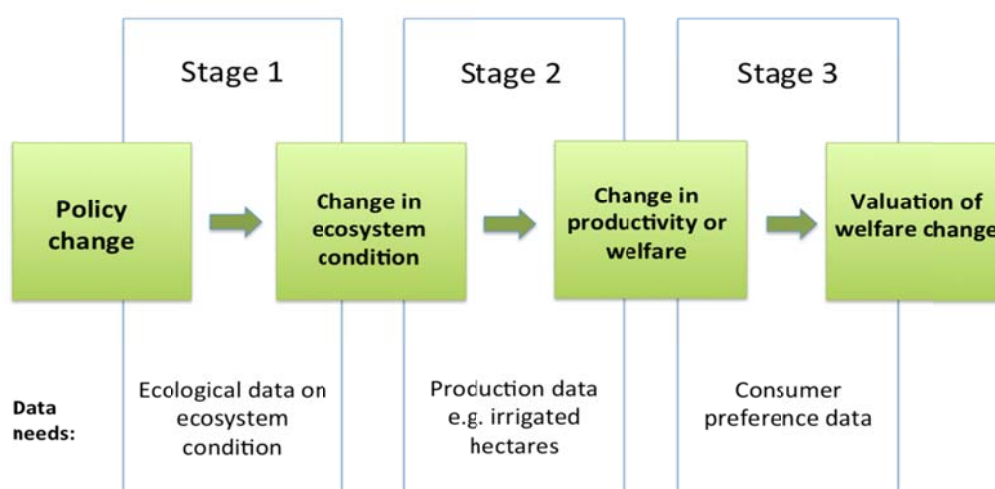
Revealed-preference methods infer the values held by individuals for non-market goods based on observations of their choices in other existing markets. Travel cost methods use variation in visits to a recreation site as travel costs (or travel time) increase to construct a demand curve showing how the marginal benefits provided by the site vary with the number

of visits. The area under the demand curve can then be used to estimate the total benefits visitors obtain from the site. Accurate estimation may be difficult if trips are made for multiple purposes, or if those with the strongest preferences for sites of a particular type choose to live near to those sites. Hedonic price methods use regression analysis to estimate how prices of market goods vary as environmental attributes associated with those market goods vary. The most common application is estimation of the impacts of environmental quality, e.g. pollution (Harrison Jr and Rubinfeld 1978) or access to open space (Irwin 2002), on house prices. Both of these revealed-preference methods are valuable because they use actual behavior to infer values. The key drawback is that they can only be used to estimate the value of environmental amenities that are consumed in conjunction with market goods or services.

Stated-preference methods estimate values using responses to questions about hypothetical markets or scenarios. Contingent valuation involves directly asking survey respondents about their willingness to pay (WTP) for an environmental benefit (or willingness to accept for its loss). The question may be open-ended, or presented as a choice of whether or not to pay a fixed amount. The environmental resource is described in detail, along with a payment mechanism such as a tax increase or donation to pay for its protection. Choice modeling is similar, but survey respondents are provided with a series of choices with varying environmental attributes and different associated prices. The researcher can then estimate the change in willingness to pay as the environmental attributes change. The main advantage is that these methods can be used to value any environmental good or service, and not only those that affect markets directly or indirectly. However, there are significant issues relating to potential hypothetical bias (where responses differ from true preferences *because* the choice is hypothetical rather than actually made), insensitivity to the scope of the environmental change, sensitivity to the method of questioning and payment mechanism, and possibly unfamiliarity with the goods in question. Despite these, in the aftermath of the 1989 Exxon Valdez oil spill, a “blue-ribbon” panel of experts, convened by the National Oceanic and Atmospheric Administration and including two Nobel prize-winning economists, concluded that “CV studies can produce estimates reliable enough to be the starting point for a judicial or administrative determination of natural resource damages - including lost passive-use value” (Carson et al. 2003). They also specified guidelines for producing high-quality contingent valuation estimates, including in-person interviews; binary discrete-choice questioning; and careful description of the good and its available substitutes.

A frequently used alternative to any of the direct valuation methods described is benefit transfer. This involves applying the values estimated at one site (using any method) to a new site. Simple unit values (e.g. \$ per hectare) may be transferred, but “benefit function transfer” has greater theoretical support. If the values at the initial site are estimated as a function of the characteristics of the ecosystem, the characteristics of the beneficiaries, and the availability of substitutes, the full function can be transferred to account for differences in these variables at the new site. Benefit transfer is generally faster and cheaper than obtaining new value estimates, but there is significant possibility of bias in the valuation of the new site, especially if the relevant characteristics differ from the original site.

Fig 3: Stages in valuation of ecosystem services



Regardless of the valuation method used, there are three stages necessary for the estimation of the impacts of ecosystem services on wellbeing (Pattanayak 2004). As shown in Figure 3, which generalizes figures from Kramer et al (1997) and Pattanayak (2004), the first stage is to model the impact of a change in policy on the ecosystem under analysis. For example, assessing the impact of a protected area or PES program on deforestation requires both observed or predicted rates of deforestation with the policy, and a counterfactual rate of deforestation in the absence of the policy. Stage 2 relates the change in the ecosystem to impacts on humans, for example the impact of a change in forest area on the incidence of malaria or waterborne illnesses. The final stage is to value the change in human wellbeing resulting from the impacts on their health, production, or consumption. Each of these stages

is data-intensive, and the knowledge needed to accurately model each of the relationships draws on multiple disciplines.

2.2 Key issues in valuing ecosystem services

A highly cited study by Costanza et al (1997) concluded that the annual value of the world's ecosystem services was US\$33 trillion, arrived at by multiplying per hectare estimates of the benefits from 17 ecosystem services by the areas of 16 types of ecosystem or biome. The numerous critiques of this work (e.g. Bockstael et al. 2000; Toman 1998; Pearce 1998) highlight some of the major challenges associated with ecosystem service valuation, and the necessary features of a valid valuation exercise. Key points raised include: 1) The concept of economic value measures the difference between wellbeing in one state of the world relative to wellbeing in another state of the world, or alternatively, how much an individual or society would give up (or require compensation) to move from one state of the world to another (Bockstael et al. 2000). This is meaningful in relation to marginal changes, such as losing 100 hectares of forest in return for the profits from an oil palm plantation, but not in relation to losing all of the world's forests where it is unclear what the alternative state of wellbeing would be. 2) Estimated values were transferred from the (local, specific) contexts of the original studies to all hectares of a given biome. As discussed further below, the values of ecosystem services are fundamentally dependent on the characteristics of the ecosystem, the characteristics of the local population, and the availability of substitutes. 3) As any commodity becomes more scarce, we expect its value to rise. As a result, the last hectare of an ecosystem will be worth considerably more than the first, so multiplication of ecosystem area by a single unit value will lead to serious errors (Toman 1998). These issues are discussed in more detail below.

The first point illustrates that only valuation of marginal changes is useful or meaningful. There are numerous reasons for valuing ecosystem services, including cost-benefit analysis of policy decisions; measuring trends in wellbeing; creating markets such as Payments for Ecosystem Services; or simply drawing attention to the costs of ecosystem loss. However, to the extent that valuation is used to inform economic decision-making, it must relate to an exchange, or a trade-off, and to a defined shift from one situation to another. In other words, the relevant information is how *changes* in wellbeing relate to *changes* in the condition of the natural environment rather than the absolute value of a given state of the world.

In addition to the necessity of quantifying the welfare impacts of a defined change in ecosystem extent or condition, another reason for studies to focus on marginal changes is that value of a resource or service is unlikely to be constant across time or space. First, as expressed in point 3) above, the marginal value will depend on the total quantity available. The marginal value is expected to increase as the resource becomes more scarce, or as the availability of substitute sites and services declines. In separate meta-analyses of wetland valuation studies, Brander et al. (2006) find evidence of diminishing returns to scale in ecosystem service delivery, while Ghermandi et al. (2008) find that abundance of wetlands within 50km is negatively related to the value of ecosystem services from an individual wetland site. Second, attempts to value ecosystem services face the challenge that there may be thresholds below which services provided by an ecosystem decline dramatically or the ecosystem ceases to function altogether. Standard economic methods for non-market valuation do not offer good solutions to this. Typically, economic values are assumed to be relevant for marginal changes in a non-critical range, while alternative decision rules such as the Precautionary Principle, or Safe Minimum Standards would be more appropriate if an ecosystem is close to the threshold. However, a small number of studies have attempted to explicitly account for non-linearities in the value of ecosystem services (e.g. Barbier et al. 2008; Koch et al. 2009).

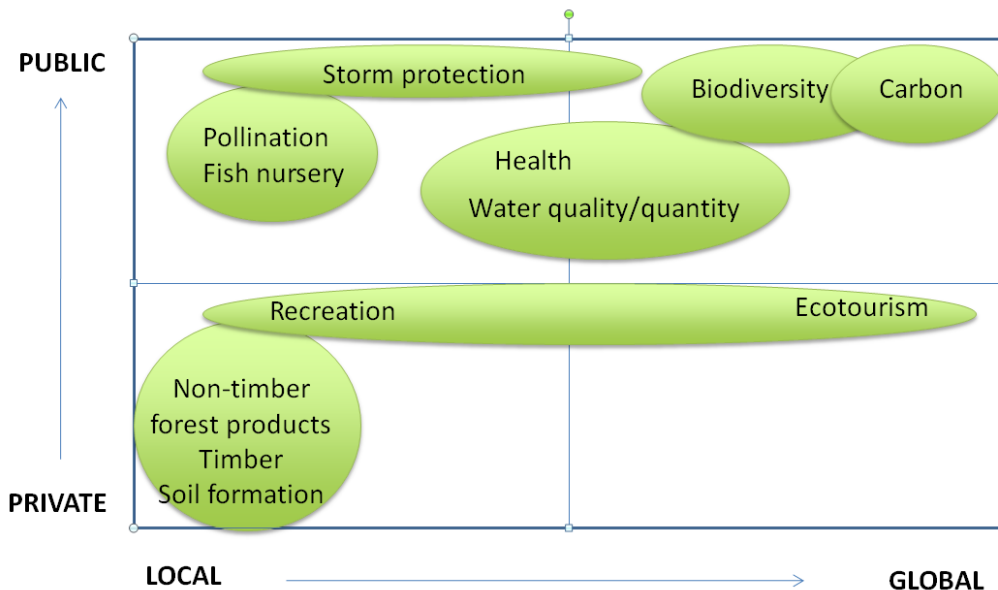
In addition to the role of relative scarcity, values of ecosystem services vary depending on other characteristics of the time and place in which they are valued. For example, 100 hectares of intact tropical forest may provide very little direct value to humans if it is remote from population, in a country where political instability deters tourists. However, if it were located in the upper-reaches of a densely populated watershed, or in a country attractive to eco-tourists with good road access, the value would be considerably higher. More precisely, the anthropocentric nature of ecosystem services means that their value depends not only on the services provided, but also crucially on the presence of people to benefit from those services. Fisher et al. (2009) emphasize this with their distinction between ecosystem *services* and ecosystem *benefits*. The services are the ecological phenomena such as climate regulation or water purification; they may or may not be used directly. Benefits are the uses to which humans put the ecosystem services in order to increase wellbeing. These include the use of clean water for drinking or recreation; the use of water flows for hydroelectric power generation or irrigation; or the benefits of stable or amenable climate in terms of increased crop production. The size of the benefits clearly depends on the extent to which humans are using the services. Furthermore, obtaining the benefits will often depend on the availability

of capital or labor inputs that must be used in combination with the ecosystem service to generate welfare improvements.

Although the value of ecosystem services depends on the presence of people who benefit, different services vary considerably in terms of where those beneficiaries are located relative to the resource. One type of service only generates benefits for those physically using the land, for example, soil formation services increase productivity only for crops planted directly in that soil. Similarly, recreational use of a forested area can only be carried out in that location, although beneficiaries may not be local residents if the site attracts domestic or foreign tourists. A second type benefits those located near to the ecosystem (with the exact definition of ‘near’ ranging from adjacent to in a neighboring country). For example, hydrological services such as water regulation or purification are likely to benefit those downstream from a forested area; while urban impacts of air pollution from dust storms or forest fires will be driven by ecosystem change in rural areas. The third type of service is produced in one location, but has global impacts. An important example is carbon storage, which benefits the entire global population, and another is biodiversity, to the extent that it provides natural materials for crop or pharmaceutical development or resilience within the global ecological system. Existence values – improvements in wellbeing that people obtain from the knowledge that a species, wilderness area or cultural resource exists – also fall into this category.

As shown in Figure 4, on-site ecosystem services are generally easier to capture privately. If the services can only be used on the land where they are generated, the beneficiary and the landowner may be the same, in which case they can manage the land to maximize their total benefits. Alternatively, there may be other beneficiaries, as in the case of some non-timber forest product harvests or recreational activities. However, the on-site characteristic of the benefits means that those who do not pay the landowner can potentially be excluded from enjoying the benefits. This becomes more challenging as the distance between landowner and ecosystem service increases and the number of beneficiaries expands, so that these ecosystem services are more likely to be underprovided. However, new policy mechanisms such as PES and carbon credits aim to internalize the benefits of forest protection for the landowner in order to increase provision.

Fig. 4: Characteristics of forest ecosystem services



There may be trade-offs in the supply of different services, which can lead to tension between the beneficiaries of those services, particularly if they are from significantly different populations. For example, prohibition of fuelwood harvesting for the purposes of carbon sequestration in a tropical region is likely to negatively affect poor rural households, while the benefits are spread over the global population. Alternatively, clearing forest for oil-palm plantations or mining may benefit multinational corporations, while generating negative health impacts for residents of local cities. Hein et al. (2006) cite the specific case of a South Asian mangrove forest that provides (i) wood and shellfish at the local level; (ii) protection from floods at the local to national level; (iii) nursery services for fish at the local to national level; and (iv) conservation of biodiversity, with benefits at the national to global level. To the extent that these services are not fully complementary, local residents prefer management rules that allow for wood and shellfish harvests while maintaining the regulation services of flood protection and fish nurseries. In contrast, global stakeholders are more concerned about biodiversity losses.

In sum, the value of ecosystem services will vary across space and time as a function of 1) the type of service; 2) the characteristics of the ecosystem and landscape e.g. species diversity, topography; 3) the characteristics of the beneficiaries, e.g. income level, demographics, economic structure, culture; and 4) the context, in particular the availability of substitutes and complementary sites. These differences also demonstrate that it is generally

not appropriate to multiply unit values (e.g. \$ per hectare) by the total observed area of a given ecosystem. It also makes it extremely challenging to transfer values from one site to another, particularly as differences between sites increase. For sites with similar characteristics, both in terms of ecosystems and human populations, it may be reasonable to transfer benefits functions, controlling for differences that do exist. However, it will not generally be meaningful to transfer values between sites that are very different, for example between temperate and tropical, or developed and developing countries.

These sources of variation in ecosystem service values also emphasize the importance of the selection of the population over which they should be aggregated. The beneficiaries of changes in on-site or local services such as soil protection and water flow regulation, which occur as land use changes within a single watershed, will be relatively straightforward to identify. However, for ecosystem services with off-site or global beneficiaries such as air quality impacts or species protection, aggregation will be more complex. First, values will vary across individuals with different characteristics and preferences; second, off-site benefits will vary with distance; and third, the availability of substitutes may significantly alter the values of services provided by a single site.

3. Values of ecosystem services to the poor

3.1 Selection of studies

The remainder of this paper reviews and summarizes studies that have attempted to quantify the human benefits from forest ecosystem services, particularly in monetary terms. As discussed above, values of ecosystem services can vary considerably depending on the characteristics of the ecosystem and the affected population. The focus in this case is on non-market benefits that would accrue to local households within rural or urban areas of developing countries if global mechanisms for avoided deforestation result in increased protection of forest ecosystems. The benefits of reduced carbon emissions themselves are omitted, as the objective of this paper is to understand the extent to which there are additional local benefits that would arise alongside the protection of forests for climate change mitigation. For the same reason, the potential local gains from international financial transfers for forest protection through REDD+ or similar mechanisms are not within the scope of this paper, although they could be highly significant in some cases. The benefits of tropical forest protection for mitigation of climate change are reviewed in detail by Goodman and Herold (2014).

Only the benefits provided by forest ecosystems that are compatible with forest protection for carbon storage and sequestration are included, which means that timber harvests are omitted. Non-timber forest products may be compatible with carbon storage if harvested sustainably, and therefore would fit within the scope of this study in principle. However, they have been the subject of extensive review elsewhere (for a recent example, see Wunder et al (2014) and other articles in the same World Development Special Issue). The benefits of forests in developing countries that accrue to residents of developed countries, such as enjoyment of forests through ecotourism or existence values, are also excluded. They can benefit those in developing countries, but will often be less important for low-income households. For example, in the context of wetlands, Nam Do and Bennett (2009) find that Vietnamese households do value higher levels of biodiversity, but that these benefits primarily accrue to urban households with higher incomes and levels of education.

Studies were identified based on the author's knowledge, databases of non-market values such as EVRI (evri.ca) and the TEEB-database (teebweb.org), and Google Scholar searches. They are drawn from peer-reviewed books and journals, reports published by government and non-government agencies, and "grey" literature where appropriate. Only those that meet three criteria are included: 1) The estimates are based on primary analysis that relates ecosystem benefits to the specific biophysical and socio-economic characteristics of a defined ecosystem. There are a large number of studies that transfer estimates of impacts of ecosystem change on ecosystem services and/or impacts of ecosystem service change on human welfare from other study sites (e.g. Yaron 2001; Adger et al. 1995). These studies were not included within this review. 2) The welfare impacts of changes in ecosystem services are estimated using standard economic methods such as those described in Section 2. 3) The study specifies both the relationship between a given policy scenario or ecosystem change and the resulting change in ecosystem service; and the relationship between the change in ecosystem service and a change in human wellbeing. There are very few studies that explicitly quantify both of these stages using state-of-the-art methods, so for the purposes of this review the second stage is emphasized provided there is some discussion of both elements. In particular, while the majority of studies reviewed here do focus on marginal changes in forest area, some do not describe the policy change explicitly, and others assume the 'change' is from complete protection to complete deforestation of a given forest area. The change in human wellbeing may be assigned a monetary value, or it may be expressed in physical terms e.g. avoided illness or reduced risk of storm damage. The aim for the paper is to include all studies that (i) meet these criteria to a reasonable degree and (ii)

focus on local benefits in developing countries. However, while it is likely that the majority of this category has been captured, it is of course possible that some studies may have been omitted (along with many that don't meet the criteria).

3.2 Health impacts

Forests can directly influence health through improvements in nutrition for those living adjacent to the forest (Pierce Colfer et al. 2006). Forests can also affect the spread of disease, through changes in the quality or quantity of water available for drinking and household uses, which in turn affect prevalence of waterborne diseases (Pattanayak and Wendland 2007); or changes in the habitat or vectorial capacity of insects and birds that transfer infectious diseases such as malaria or West Nile virus (Afrane et al. 2008; Pongsiri et al. 2009; Keesing et al. 2010). Respiratory problems and physical wellbeing more generally are affected by air quality. Urban trees can improve local air quality by removing pollutants, and urban residents may also be indirectly affected by changes in land use in rural areas. For example, in early-2013, air quality in Beijing reached 'dangerous' levels due to sandstorms resulting from soil erosion in Inner Mongolia (BBC 2013), although there are hopes that this will be avoided in future as a result of reforestation under the 'Desertification Combating Program around Beijing and Tianjin' (Yin and Yin 2010). In Indonesia in 1997-8, extensive forest fires initiated by forest clearing resulted in increased incidence of respiratory problems throughout the region in addition to income losses from reduced visibility for aircraft and fishing vessels, fewer tourist visits, and direct damage to property (Glover and Jessup 2006). Avoidance of forest fires is not an ecosystem service provided by forests. However, it is included here because forest protection aimed at mitigating climate change would benefit local residents through a reduction in the fires that are used to clear forest land.

The most common method of valuing health impacts of forests or forest fires is to use regression analysis to estimate the prevalence of illness, whether respiratory problems, malaria, or diarrhea, as a function of forest area or forest fires, and other physical, economic and demographic characteristics. This is then translated into an estimate of number of avoided cases of illness, reduced risk of illness, or a monetary value based on cost of treatment and lost work days. Hahn et al. (2013) use municipality-level data to estimate the relationship between deforestation and malaria, controlling for other municipality characteristics. This type of analysis can be subject to measurement issues, as the identification and reporting of malaria cases is unlikely to be random but will depend on the presence of medical staff and health centers, which may in turn be correlated with

unobserved variables that drive deforestation. Other studies estimate the relationship between forest area and disease prevalence using household-level data. For example, Saha et al. (2011) estimate the prevalence of respiratory problems as a function of forest clearing related to mining, controlling for other household characteristics. In particular, they use an instrumental variables strategy to control for whether a household member is employed in the mines, as that would clearly also be correlated with respiratory illness.

The quantified health benefits of water quality and quantity include impacts on diarrhea and typhoid, and, conversely, the availability of clean water for drinking and household uses. Sanglimsuwan et al. (2014) find that mining-induced deforestation raises the risk of diarrhea and typhoid, while Pattanayak and Wendland (2007) estimate the benefits of increased water availability as 2,600 fewer cases of diarrhea per year across a population of 13,700 households. This translates to total medical cost savings of \$5,900 per year. The role of forests in providing drinking water in the Valdivia region of Chile is valued at \$15.40 per household in summer, and \$5.80 the rest of the year, given the average annual production in the region. This translates to \$61.2-\$162.4 per hectare of native forest (Núñez, Nahuelhual, and Oyarzún 2006). In the Ruteng Park catchment in Indonesia, maintenance of baseflow by protected forests save the average household \$0.40-\$1.20 per year in collection costs. However, as with regulation of water for agricultural uses, there are cases when forests reduce available baseflow relative to other vegetation types (Pattanayak 2004).

Findings on the impacts of forests on malaria are also somewhat mixed. In the Brazilian Amazon, Olson et al. (2010) and Pattanayak et al. (2009) find a positive relationship between deforestation and malaria risk, and Hahn et al. (2007) find that a 0.7% increase in deforestation raises malaria risk by 21% in timber producing municipalities in Brazil, but has no impact after controlling for road density across the Legal Amazon as a whole. Pattanayak et al. (2009) find that in Indonesia, primary forests reduce malaria risks only for young children; however, secondary or disturbed forests are associated with higher risks. In contrast, Valle and Clark (2013) find evidence that forest cover increases malaria risk, and that in particular, cities near protected areas have higher incidence of the disease. It appears that the relationship between deforestation and malaria is non-linear and related to the presence of population, i.e. when forest cover is high, and population is low, the potential for malaria transmission is also low; when forests are disturbed and population increases, habitat is well suited to mosquitos and the potential for transmission to humans increases;

finally, as forests are cleared altogether, mosquito habitat is reduced, which can again reduce malaria risk.

A relatively large number of studies have estimated the costs of respiratory problems due to forest fires. These may be intentional fires aimed at clearing land for agriculture or mining, or accidental fires that have spread from the initial location. Mendonça et al. (2004) estimate that 9,346 people were affected by respiratory ailments per year due to forest fires in Brazil between 1996 and 1999, valued at \$7.4 million in WTP to avoid illness. Weather conditions significantly affect the costs of the fires, for example, Naidoo et al. (2009) estimate the total costs of fires used to clear land for oil palm plantations in Northern Borneo at \$70,000-\$5.7 million in a 'normal year' and \$209,000-\$17.2 million in an El Niño year. 1997-8 was a particularly bad season for forest fires due to El Niño, with up to 25 million hectares affected worldwide (Tacconi 2003). The most severe fires, in Sumatra and Kalimantan, Indonesia, have attracted considerable research attention. Respiratory problems are estimated to have increased by 8-9% within Indonesia (Frankenberg, McKee, and Thomas 2005), and have led to 15,600 infant, child and fetal deaths (Jayachandran 2009). The increased costs of illness during the period have been valued at \$295 million in Indonesia, and \$12.4-\$20.9 million in Malaysia and Singapore.

Overall, the relationships between deforestation and disease are not well understood, particularly impacts that operate through changes in insect habitat. Impacts on nutrition have not been quantified, although qualitative studies suggest an important relationship, especially for the poorest households and those experiencing economic hardship or illness (Ahenkan and Boon 2011; Sheil and Wunder 2002; Barany et al. 2001). Air quality impacts are better studied, and can be extremely high in years when forest fires spread easily. Additionally, within urban areas, there is evidence that tree planting can improve local air quality at lower cost than alternative pollution control measures (Escobedo et al. 2008). The Health & Ecosystems initiative (wcs-heal.org) has been developed in recognition of the lack of evidence on the relationships between ecosystem services and health. Experts in land use change and nutrition, waterborne disease, insect-transmitted disease, cardio-pulmonary disease and community wellbeing are currently implementing studies to improve understanding of these links.

Table 1: Health impacts

Location	Change in ecosystem	Change in welfare	Method	Source
<u>Avoided respiratory disease</u>				
Brunei Darussalam	1997/8 Indonesian forest fires –10 unit increase in Pollution Standard Index	Total costs \$12,640/day for the 230,000 residents of Brunei-Muara District	Dose-response	Anaman and Ibrahim (2003)
Malaysia	1997/8 Indonesian forest fires – increase in particulate concentrations of 100 μm^3	7% increase in mortality risk across all age groups in Kuala Lumpur	Dose-response	Sastry (2002)
Malaysia, Singapore, Indonesia	1997/8 Indonesian forest fires	Total national costs of respiratory illness: Malaysia: \$8.41 million Singapore: \$4-12.5 million Indonesia: \$295 million	Dose-response	Glover and Jessup (2006)
Malaysia, Indonesia	Plantation development on 1.8 million hectares – 14.5% reduction in forest cover.	Total costs of respiratory disease in Heart of Borneo region: \$70,000 - \$5.7 million in ‘normal year’; \$209,000-\$17.2 million in El Niño year.	Dose-response	Naidoo et al. (2009)
Indonesia	1997/8 Indonesian forest fires	8-9% increase in respiratory problems	Dose-response	Frankenberg et al. (2005)
Indonesia	1997/8 Indonesian forest fires	15,600 infant, child and fetal deaths; 1.2 percentage point decrease in survival. Worse in poorer areas.	Dose-response	Jayachandran (2009)
India	Mining-induced deforestation	Living 1km closer to mines increases odds of respiratory infection by 2.7%.	Dose-response	Saha et al. (2011)
Brazil	1,800 km ² forest fires without El Niño /39,000 km ² with El Niño	Total costs in Legal Amazon region \$7.4 million/year	Dose-response	Mendonça et al. (2004)
Chile	Urban forest management	14.8-17.3g/ m ² /year removal of PM ₁₀ . Cost effective relative to alternative pollution control measures.	Dose-response	Escobedo et al. (2008)
China	Forest and grassland restoration	WTP \$0 for reduction in sandstorms (total WTP for reforestation program \$45-115/household per year)	Choice experiment	Wang et al. (2007)
<u>Waterborne illness reduction</u>				
Indonesia	Forest protection – 1% increase in baseflow (typical village has 1,002mm/year of baseflow)	Reduction of 2,600 diarrhea cases per year across 13,700 households	Production function	Pattanayak and Wendland (2007)
India	Mining-induced deforestation	Living near mines increases incidence of diarrhea and typhoid.	Dose-response	Sanglimsuwam et al. (2014)
<u>Insect-borne illness reduction</u>				
Indonesia	Forest protection	Reduction in probability of under-10 malaria with primary forest; increase in probability of under-5 malaria with secondary forest	Dose-response	Pattanayak et al. (2010)
Brazil	Forest restoration – 1 million hectare reduction in deforestation	Reduction of malaria rate by 2.7 per 1000 and dengue by 0.1 per 1000	Dose-response	Pattanayak et al. (2009)
Brazil	Forest protection – 0.7% reduction in deforestation across Legal Amazon region	0-24% reduction in malaria risk (average incidence 22 cases/1000 population)	Dose-response	Hahn et al. (2014)
Brazil	Forest fragmentation	Cities near protected areas have higher incidence of malaria	Dose-response	Valle and Clark (2013)

Brazil	Forest protection – 4.3% increase in deforestation (1 standard deviation) in one municipality	48% increase in malaria in region with average incidence of 1.16 cases/person	Dose-response	Olson et al. (2010)
<u>Water quality and quantity improvements</u>				
Chile	Protection of 1,117 hectare forest in Llancahue watershed – streamflow as an input to production of drinking water	\$15.4/household (summer); \$5.8/household (rest of year) for 33,000 households; total value of \$61.2-\$162.4/hectare native forest	Production function	Nunez et al. (2006)
Indonesia	Forest protection - Savings in water collection costs with 25% increase in forest cover in 56,000 hectare park buffer zone	Savings of \$0.40-\$1.20/household/year (some cases negative) for 13,700 households with mean incomes of \$350/year	Hedonic cost function	Pattanayak (2004)

3.3 Human safety

Forests are thought to play an important role in mitigating the impacts of extreme events such as storms, floods and landslides. To the extent that they do so, the benefits in terms of human safety can be extremely significant. However, the evidence on the relationship between forest cover and impacts of natural disasters is stronger for some types of events than others. Numerous studies have shown reductions in damages from storm surges in coastal areas protected by mangroves (Danielsen et al. 2005; UNEP 2005; Koch et al. 2009). Damages from landslides are frequently attributed to deforestation and forest degradation on hill slopes by policymakers, rural households and the media (MEA, 2005; Ahlheim et al. 2008; National Geographic 2010). Despite this, few studies have quantified the physical relationship, particularly in developing countries, although those that have find that deforestation is a significant predictor of landslide intensity and frequency (Kumar and Bhagavanulu 2008; Knapen et al. 2006; Kamp et al. 2008). The role of forests in preventing floods is highly contentious. Major reforestation and forest conservation programs have been introduced based on the belief that deforestation contributes to major flood events such as the 1998 Yangtze River floods that devastated large areas of central China (FAO 2005; Xu et al. 2010). Global analysis of the correlation between deforestation and flooding supports this view (Bradshaw et al. 2007; Laurance 2007). However, critics argue that while forest clearing may increase peak flows and therefore small, localized floods; it is unlikely to contribute directly to large-scale events (FAO 2005; van Dijk et al. 2009). A caveat to this is that large-scale floods may become more frequent and more damaging as a result of sedimentation caused by deforestation (van Dijk et al. 2009).

Valuation of the contributions of forests to human safety can be measured based on damages in locations with and without forests or mangroves when a natural disaster occurs

(Das and Vincent 2009; Barbier 2007). Alternatively, valuation of expected damages can be elicited using contingent valuation methods (Bann 1999; Badola and Hussain 2005; Ahlheim, Fror, and Sinphurmsukskul 2006). The latter has the advantage that it can capture households' perceptions about the risk and likely scale of potential damages, expressed as an annual monetary value. However, it may not represent the true physical linkages between forest area and the risk and scale of natural disasters.

The majority of the economic evidence on the role of forest ecosystems in human safety is focused on the benefits of mangroves in mitigating storm damage. Comparing villages protected by mangroves to differing degrees during the 1999 super-cyclone in Orissa, India, Das and Vincent (2009) estimate that in the absence of all existing mangroves in their study area, the cyclone would have resulted in an additional 1.72 deaths per village within 10km of the coast (there were 0.63 actual deaths per village on average). In the same context, Badola and Hussain (2005) estimate that damage costs averaged \$33 per household in one village with mangrove protection, and \$153 per household in a similar village without mangrove protection. Barbier (2007) estimates expected damages from all significant coastal storms for all coastal regions of Thailand at \$5,850 per year for a 'representative' hectare of mangrove deforestation. Barbier et al. (2008) extend this to show that the marginal effects increase as the total area of mangroves declines.

Evidence on the value of forests in reducing landslide and floods is relatively limited. Ahlheim et al. (2009) and Ahlheim et al. (2006) use contingent valuation to elicit values for landslide risk reduction among rural households in Vietnam and the Philippines. The values obtained, of \$3-5 per household per year suggest that households do perceive a risk-reduction benefit from forests, but the specific linkages between changes in deforestation and changes in risks are not made explicit. Kramer et al. (1997) do model all stages of the relationship between policy, deforestation, flood risks and economic impacts. They find that the designation of Mantadia National Park in Madagascar reduces local damages to cropland by \$126,700 over 20 years.

Table 2: Human safety

Location	Change in ecosystem	Change in welfare	Method	Source
<u>Storm Protection</u>				
India	Reductions in mangroves from 30,766 to 17,900 hectares, 1944-99.	Clearing remaining mangroves would have led to an increase in number of deaths from 0.63 to 1.72 per village (additional 256 deaths). 1ha of mangroves saved 0.0148 lives in 1999 Orissa super-cyclone.	Avoided mortality	Das and Vincent (2009)
Malaysia	Conservation of 1,690 hectares mangroves	WTP \$3.16/household/year for population of 12,650 with average incomes of \$450/year (37% below \$130/year)	Contingent valuation	Bann (1999)
India	Conservation of 145 km ² mangroves	WTP \$33 - \$154/household for 156 households with annual incomes of \$490/household	Contingent valuation	Badola and Hussain (2005)
Thailand	Mangrove loss of 3.44-18 km ² per year	Total storm damages \$5,580/year across 36,000-38,000 households	Avoided damages	Barbier (2007)
<u>Reduction in landslide risk</u>				
Vietnam	Reforestation – area not specified	WTP \$4.92/household/year for households with total assets of \$550.	Contingent valuation	Ahlheim et al. (2009)
Philippines	Introduction of ‘rainforest farming’	WTP \$4.94/household/year for population of 19,517 households with average annual incomes of \$2,800	Contingent valuation	Ahlheim et al. (2006)
<u>Reduction in flood risk</u>				
Madagascar	Primary forest protection - avoided damages to agricultural land	NPV \$126,700 (total for Madagascar, where 1991 GNP=\$207/person)	Production function	Kramer et al. (1997)

3.4 Energy security

Forests can provide energy directly to rural households in the form of firewood or charcoal, as well as a source of income from sales of fuel products. Although it is possible to harvest firewood sustainably, such harvests frequently lead to forest degradation, compromising other ecosystem services (Cooke, Kohlin, and Hyde 2008). An additional route through which forest ecosystems provide energy security, without compromising other services, is

through soil protection, which reduces the rate of sedimentation of hydroelectric dams, and water regulation. Sáenz and Mulligan (2013) model surface water balances in the watersheds of the 18,770 dams between the latitudes of 23.5N and 35.5S, which cover 32% of the tropical and subtropical land area. Although cloud forests only cover 4.4% of this area, they account for 21% of the surface water balances, suggesting that upstream cloud forests play an important role in maintaining water flows to dams.

Studies valuing hydrologic benefits use on-site empirical observations or Universal Soil Loss Equations to estimate how sedimentation and water dynamics vary with vegetation change on the land-types observed within the watershed of the dam. Veloz et al. (1985) estimate that the reduction in sedimentation resulting from a 25% reduction in soil loss (due to land management changes in the watershed) would increase the remaining lifespan of the Valdesia reservoir in the Dominican Republic from 19 years to 25 years. The net present value is estimated at \$2.7 million over the 25 years based on the costs of alternative electricity generation. Guo et al. (2007) value the reduction in sedimentation in the Three Gorges Hydroelectric Power Plant resulting from large scale reforestation and a ban on logging in the upper-watershed at \$15.1 million, based on the costs of clearing sediment. They also estimate the net present value of increased electricity production due to water flow regulation at \$21.9 million.

3.5 Food security

Forests can affect the wellbeing of rural households in developing countries through impacts on the production of food. One important pathway is the impact of forest cover on crop yields through maintenance of soil quality or pollination. Forests also have important impacts on the quality and quantity of water available to farmers: soil erosion and sedimentation are affected by forest disturbance; seasonal surface water flows, in particular dry season baseflow, and groundwater recharge may increase or decrease depending on the net effect of changes in evapotranspiration and infiltration; and regional precipitation can be affected (Aylward 2004). In coastal regions, mangroves provide important habitat for fish and shellfish, which in turn constitute a significant part of local diets, as well as a source of income. These benefits accrue to households and firms in close proximity to the forest ecosystem; either in the same location for the soil protection benefits of agroforestry or forest fallow; adjacent to one another, for the pollination and shade benefits; or within the same watershed for the hydrological benefits.

Table 3: Energy security

Location	Change in ecosystem	Change in welfare	Method	Source
Dominican Republic	Various improvements to land management in 85,090 hectare watershed, including agroforestry on 36-50% slopes and reforestation of >50% slopes (land use change on 11% of land)	NPV of additional electricity production \$2.7 million at 5% discount rate over 25 years.	Production function	Veloz et al. (1985)
China	Protection of 440,000 hectares of forest in three counties upstream from Three Gorges Hydroelectric Power Plant	Total water regulation benefits \$21.9 million/year; total benefits of reduced sedimentation \$15.1 million/year	Production function	Guo et al. (2007)

Many studies on the contribution of forests to food security use production function methods to value the impacts. There are two main approaches: the first is to estimate the production function using regression analysis, treating the forest ecosystem as an input to production while controlling for all other relevant inputs. Klemick (2011) estimates a model of agricultural revenue as a function of the area of on-site and upstream forest fallow (land set aside for temporary forest regeneration), and other inputs, while addressing spatial correlation and potential reverse impacts of farm productivity on land use decisions. The second approach is to model the physical relationship between forest cover and inputs or outputs, and then value the changes using market prices. Ricketts et al. (2004) conduct field experiments to assess how the productivity of coffee plants vary with distance from the forest edge as a result of pollination by bees from the forest. The profits associated with increased yields close to forest fragments are calculated using market prices for coffee and other inputs.

Other studies use stated preference methods, in which local households are asked directly about the extent to which they benefit from ecosystem protection. For example, Rodriguez et al. (2006) (2006) ask households in Peru their WTP to avoid on-site soil erosion of *Opuntia* scrublands in Peru. The households collect fruit and cochineal insects, and erosion

is thought to increase production costs and reduce profits in the long term, although the effects are too slow to be observed by other means in the short term. The stated preference studies should in principle capture the true changes in consumer or producer surplus attributable to a given change in ecosystem services. However, the relationship between change in the ecosystem and the change in ecosystem service delivery can be imprecise. Rodriguez et al. (2006) cite studies on local soil erosion, but do not specify the level of soil erosion that would result from forest degradation. Barkmann et al. (2008) and Pattanayak and Kramer (2001a) develop their scenarios based on local hydrological models, but these are not fully integrated with the economic models.

The welfare impacts of reductions in availability of water for irrigation due to the drought mitigation services of forests largely fall in the range of a few dollars per household per year. Studying households near Lore Lindu National Park, Indonesia, Barkmann et al. (2008) estimate annual household WTP for one month less of water scarcity at \$4.06, while a similar population of households in the vicinity of Ruteng National Park, also in Indonesia, have a stated WTP of \$1.97 per year for forest protection (Pattanayak and Kramer, 2001a). Using a production function approach, the Pattanayak and Kramer (2001b b) find that an increase in forest cover of 25% in Ruteng Park would raise average agricultural profits by \$3-\$10 per household per year, although with a 75% increase in forest cover, this could increase to \$35, or 10% of annual farm profits, in some locations. Klemick (2011) also estimates the impact of baseflow regulation on net farm output within this range (based on a 10% increase in upstream forest fallow). Overall, the hydrological benefits for agricultural production are small, but can be significant for individual households. However, in some circumstances increased forest cover reduces water availability because runoff decreases (Pattanayak and Kramer 2001b; Lele et al. 2008), so the relationship is not necessarily straightforward.

On-site benefits on agricultural productivity can be substantial for the individual farmer. Households in Eastern Visayas, in the Philippines experienced a 6% increase in total income as a result of soil quality improvements due to agroforestry investments (Pattanayak and Evan Mercer 1998). Brazilian farmers who increased their fallow area by 10% experienced 1.4-1.6% increases in net output value (Klemick 2011). The benefits of pollination and shade provision for coffee and cacao plantations in Costa Rica and Indonesia are estimated at roughly 1-10% of annual profits (Barkmann et al. 2008; Priess et al. 2007; Ricketts et al., 2004).

Models of the role of mangroves as habitat for commercial fish and shellfish stocks have to specify the biological relationship between habitat and stock growth rates, as well as the technical and economic relationship between stocks and harvests. Barbier and Strand (2002) estimate that an average rate of mangrove loss of 2 km² per year between 1980 and 1990 resulted in lost revenues of \$280,000 for the Campeche shrimp fishery in Mexico. Barbier (2007) examines the role of mangroves as support for small coastal fisheries in Thailand, and finds that the net present value of mangroves lost between 1996 and 2004 in terms of habitat-fishery linkages is between \$708 and \$987 per hectare. The harvest reductions in the Mexican case study amount to only 0.4% of average annual harvests. However, the authors note that the marginal impacts are likely to rise as mangroves become increasingly scarce.

A potentially important benefit of forests for food security that has not been the subject of economic valuation is the *in situ* conservation of crop wild relatives. Genes from wild relatives are particularly important for improving the pest and disease resistance of commercially valuable crops such as wheat, rice and tomatoes (Hajjar and Hodgkin 2007). The availability of this genetic material is reduced by the loss of natural habitats including forests, threatening food security at the local and global level as crop yields decline (Stolton et al. 2006).

Table 4: Food security

Location	Change in ecosystem	Change in welfare	Method	Source
<u>Drought mitigation</u>				
India	Forest restoration – 20km ² in one degraded catchment	Reduction in profits of \$107/household/year	Production function	Lele et al. (2008)
Indonesia	Forest protection – one month less water scarcity from 7,230,000 hectare park	WTP \$4.06/household/year for 137,000 households with mean income of \$590/year	Choice experiment	Barkmann et al. (2008)
Indonesia	Forest protection - drought control from 32,000 hectare park	WTP \$1.97/household/year for 13,700 households with mean farm profits of \$350/year	Contingent valuation	Pattanayak and Kramer (2001a)
Indonesia	Forest protection - 25% increase in forest cover	Increase in profits \$3-10/household/year (1-3% of annual farm profits)	Production function	Pattanayak and Kramer (2001b)
<u>Soil quality and erosion control</u>				
Brazil	Forest fallow – 10% increase in on-site/upstream fallow	Increase in net revenue 1.4-1.6% (on-site)/ 2.5-4.6% (upstream).	Production function	Klemick (2011)
Philippines	Use of agroforestry practices on own property	Increase in profits of \$53/household /year (6% of total income)	Production function	Pattanayak and Mercer (1998)
Peru	Opuntia scrubland protection	Household WTP \$5.13/ha/year (3% of value of cochineal harvested – main economic activity)	Contingent valuation	Rodriguez et al. (2006)
Indonesia	Forest protection – 5% and 10% increase in baseflow and erosion control services from 32,000 hectare park	Increase in income \$9-11/household/ year (5% increase); \$19-24/household/year (10% increase) 13,700 households with mean farm profits of \$350/year	Production function	Pattanayak and Butry (2005)
<u>Shade</u>				
Indonesia	Forest protection – 10% increase in shade for cacao plantations	WTP \$4.70/household/year for 137,000 households with mean income of \$590/year	Choice experiment	Barkmann et al. (2008)
<u>Crop Pollination</u>				
Costa Rica	Protection of forest fragments within 100m of coffee plants	Increase in profits for single large coffee farm of \$62,000/year (7% of annual profits)	Production function	Ricketts et al. (2004)
Indonesia	Protection of 5659 hectares of forest within 1500m of coffee plantations (avoiding 0.2-2.5%/year deforestation).	Increase in profits for local coffee farmers \$63/hectare (0.3-13.8% of net revenues over 20 years).	Production function	Priess et al. (2007)
<u>Wildfire mitigation</u>				
Brazil	Forest fires – damage to 14,000 km ² pasture (2% of deforested area) and 16,000 km fences	Total costs in Legal Amazon region \$21-42 million/year	Avoided damages	Mendonça et al. (2004)
Brazil	Forest fires – damage to timber stocks from 1,800 km ² forest fires without El Niño /39,000 km ² with El	Total costs in Legal Amazon region \$1-13 million/year	Avoided damages	Mendonça et al. (2004)

Niño		<u>Fisheries habitat</u>		
Thailand	Mangrove loss of 3.44-18 km ² per year	NPV over period 1996-2004 \$708 - \$978/hectare for 36,000-38,000 coastal households	Production function	Barbier (2007)
Mexico	Mangrove protection – 3% reduction in area (2 km ² per year, 1980-1990)	\$279,000 reduction in revenue from shrimp harvests (0.4% of average annual harvests for 350 industrial and 5000 artisanal vessels)	Production function	Barbier and Strand (2002)
Malaysia	10% change in protected mangrove forest (currently 7,000 hectares)	WTP \$19 - \$21/household for households with mean annual incomes of approximately \$4,700	Choice experiment	Othman et al. (2004)
Micronesia	Protection of 1,562 hectares of mangroves	Total WTP \$1.08 million – \$1.26 million/year across population of 7,500 (of which crab and fish harvests account for 72%). Median WTP for mangrove protection is 2.1-2.5% of monthly household incomes.	Contingent valuation	Naylor and Drew (1998)

3.6 Income generation

The most obvious ways to generate income from forested land are by logging for timber or clearing for agriculture. However, in some settings there are income generation opportunities associated with intact forest land, particularly where the forest is healthy and diverse.

Multiple studies estimate the potential WTP by ecotourists to visit forest reserves, and in some cases the benefits are valued in millions of dollars across all visitors. These benefits can potentially be captured in the form of entrance fees, or through local expenditures by tourists on hotels, food etc. One challenge for using fees to capture the benefits is that, even if the consumer surplus experienced by visitors is substantial, the availability of substitute parks and reserves will mean that at very high entrance fees they are likely to go elsewhere. This will particularly be the case for foreign tourists, who have a large range of alternative options. Despite this, Ellingson and Seidl (2007) estimate the elasticities of demand for a particular forest reserve as hypothetical entrance fees rise, and they still find total WTP, for park entrance specifically, of around \$2 million.

Table 5: Income generation

Location	Ecosystem change	Welfare impacts	Method	Source
<u>Ecotourism</u>				
Costa Rica	Forest protection – 10,000 hectare Monteverde Cloud Forest Preserve	Aggregate WTP to protect park by all visitors: \$37,517,374	Contingent valuation	Echeverría et al. (1995)
Bolivia	Forest protection – 73,000 hectare Eduardo Avaroa Reserve	Total WTP in entrance fees: \$2.2m (CB); \$1.9m(CV)	Contingent valuation/behaviour	Ellingson and Seidl (2007)
Costa Rica	Forest protection – all Costa Rican rainforests	Total value of all ecotourist visits to Costa Rica by US residents: \$68 million	Travel cost	Menkhous and Lober (1996)
Uganda	Forest protection – 30,000 hectare Mabira Forest Reserve	Maximum total revenue from park fees with 20 bird species seen: \$18,032 Maximum total revenue from park fees with 80 bird species seen: \$40,423	Choice experiment	Naidoo and Adamowicz (2005)
Costa Rica	Forest protection – 10,000 hectare Monteverde Cloud Forest Preserve	WTP \$20-\$25/person	Contingent Valuation	Chase et al. (1998)
Costa Rica	Forest protection – 10,000 hectare Monteverde Cloud Forest Preserve	Annual consumer surplus from all domestic visits to reserve: \$97,500–\$116,200	Travel cost	Tobias and Mendelsohn (1991)
<u>Bioprospecting</u>				
Global	Forest protection – biodiversity hotspots	\$14 - \$65/hectare		Costello and Ward (2006)

Another potential source of income is the sale of contracts for bioprospecting. This involves payment by pharmaceutical companies to communities and/or national governments for access to plant materials from highly biodiverse forests. Estimated values are based on the amounts that could potentially be paid, using assumptions about the (small) likelihood of finding a useful compound and the (large) returns if one was discovered. Initial estimates varied widely (Rausser and Small 2000; Simpson, Sedjo, and Reid 1996), although Costello and Ward (2006) attempt to reconcile the values and arrive at estimates of \$14-\$65 per hectare across all global biodiversity hotspots. Although Merck made a \$1.2 million deal with Costa Rica in 1991 for bioprospecting and conservation, there has been little application of similar contracts since then due to a lack of trust between developing country governments and pharmaceutical companies, and difficulties reaching agreement on rules to govern intellectual property and benefit sharing in the case of a successful discovery (Dalton 2004).

Although both ecotourism and bioprospecting offer potential for large sources of incomes associated with forest protection, both can at best only apply to forests with particular characteristics. High biodiversity will be a pre-requisite for even the possibility of a bioprospecting agreement, while scenic beauty and easily viewable flagship species will be important for ecotourism. Income from ecotourism also depends on the country in question being politically stable, having sufficiently high-quality infrastructure, and having parks located close to major cities (R. Naidoo and Adamowicz 2005). Chase et al (1998) find that visitors are willing to pay more for forests that are located in parks with additional attractions such as volcanoes or beaches. As a result, these values cannot be generalized to the majority of tropical forest locations.

3.7 Comparison of ecosystem service values at single sites

The values of different ecosystem services presented so far in this section are challenging to compare for two key reasons. First, they relate to very different ecosystem changes, from accidental forest fires generated by agricultural land clearing to forest preservation in national parks and the management of urban trees. These changes also occur at very different scales, from the protection of forest fragments around a single coffee farm (Ricketts et al., 2004) to the annual removal of mangroves in all coastal areas of Thailand (Barbier, 2007). The second reason why comparisons are difficult is that different benefits are aggregated over very different geographic and demographic scales. Some benefits are only obtained by those living on or adjacent to the forested land, while others are spread across multiple countries. The size and distribution of the various types of benefit are important to understand because there will often be trade-offs in their provision, either within different categories of benefit or with other land (and water) uses such as agriculture and aquaculture.

This section will examine the different estimates of ecosystem service values from two specific case study sites in order to compare the welfare outcomes from different ecosystem services, while keeping the site and the ecosystem change constant.

Case Study 1 – Coastal Thailand

Barbier (2007) presents the net present value per hectare of different ecosystem services generated by mangroves in coastal regions of Thailand between 1996 and 2004, as well as the alternative land-use of shrimp farming which requires clearing the mangroves. The values are shown in Table 6. Habitat-fishery linkages, storm protection, and forest product collection are all compatible with one another; if mangroves are protected, they can be obtained simultaneously. In contrast, the returns from shrimp farming are not compatible with the ecosystem services provided by mangroves; if mangroves are cleared, the other benefits are lost. The estimated values suggest that in this case study, the storm protection benefits of mangrove conservation are considerably larger than the other benefits, and also larger than the commercial returns from deforestation.

Table 6: Benefits and costs of mangrove protection (adapted from Barbier (2007))

Benefit/Cost	Net present value per hectare (10-15% discount rate)
Habitat-fishery linkage	\$708–\$987
Storm protection	\$8,966–\$10,821
Net income from forest products	\$484–\$584
Net economic returns from shrimp farming	\$1,078–\$1,220

This suggests that net social benefits are highest if mangroves are protected rather than cleared for shrimp farming. However, Barbier et al. (2008) extend this analysis to account for the nonlinear impact of mangrove area on wave attenuation during storms, and conclude that for large areas of mangroves, the storm protection benefits of an additional hectare are minimal. In such cases, conversion may be welfare improving overall. As total mangrove area declines, the marginal benefits of one additional hectare for storm protection increase and the net benefits of clearing diminish and then become negative.

Case Study 2 – Ruteng National Park, Indonesia

Pattanayak and co-authors have used multiple methods to value different ecosystem services benefitting households living close to a protected forest area in Indonesia. Ruteng National Park, in Manggarai region of Flores island was created in 1993. It consists of 32,000 hectares of protected forest and a buffer zone of 56,000 hectares in which sustainable forest product

collection, agroforestry, and small-scale agriculture are permitted. A set of studies use variation in forest cover in different watersheds within the region to estimate the welfare impacts of ecosystem services for the 13,700 households living in 48 villages within the buffer zone. These services are primarily related to the role of forest in drought mitigation, specifically in increasing baseflow, which is the non-episodic residual streamflow after rain leaves the system as runoff or evapotranspiration (Pattanayak and Kramer, 2001a).

The benefits valued are the impact of drought mitigation on agricultural profits; the impact on water collection costs for domestic uses; the impact of water quantity on diarrhea incidence; the impact of soil erosion on agricultural profits; and the impact of forest proximity on the risk of malaria. Butry and Pattanayak (2001) also estimate the costs of forest protection for the minority of households engaged in logging and forest product collection rather than farming. All of these ecosystem services are local, in the sense that they accrue to those living downstream from, but close to, the forested areas. As such, they all impact rural households with low levels of education and wealth (Pattanayak and Kramer, 2001a).

A 25% increase in forest cover is estimated to raise annual agricultural profits by \$3-\$10 per household as a result of increased water availability in the dry season (Pattanayak and Kramer, 2001b) and lower annual water collection costs by \$0.40-\$1.20 per household (Pattanayak, 2004). However, these benefits only occur in counties where higher forest cover is predicted to lead to higher levels of dry-season baseflow. In some counties, profits are lowered and collection costs are increased as more forest cover is associated with lower baseflow. Where baseflow does increase, it is also found to reduce cases of diarrhea (Pattanayak and Wendland, 2007), and proximity to primary forest appears to reduce the risk of malaria in children (Pattanayak et al., 2010). On the cost side, a 25% increase in forest cover is predicted to reduce annual incomes from logging and forest product harvest by \$52 per household for poor logging households (although the wealthiest households in the region gain by \$331 per household). While the benefits of forest exploitation are high compared with the ecosystem service benefits from forest protection, the former accrue to a smaller number of households. Of the 13,700 households in the region, only 2,000 are engaged in direct forest use, while almost all are engaged in agriculture and use water for domestic purposes.

These comparisons demonstrate that even within a small area, with a relatively homogenous population, gains and losses from forest protection are unevenly distributed, and both can be important at the household level.

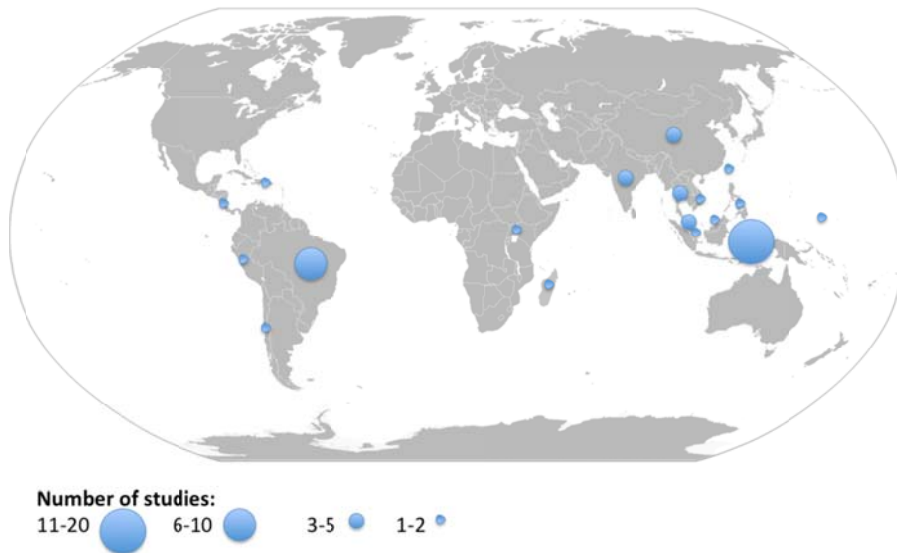
4 Scope of the ecosystem service valuation literature

4.1 Geographical coverage of studies

The tables in Section 3 demonstrate that there is considerable unevenness in both the geographic location of high-quality valuation studies, and the types of ecosystem services that are valued. Figure 5 shows the number of studies reviewed in this paper, by country. Only developing countries are included, by design. However, it is apparent that the majority of the estimated values in the ecosystem services literature are drawn from Asia, mainly Southeast Asia. There are a smaller number from Latin America, primarily Brazil, and only two from Africa. This is not to say that no research in this field is being conducted in Africa. However, existing studies either transfer values from elsewhere (Yaron 2001), estimate benefits to foreign tourists (Naidoo and Adamowicz 2005; Maille and Mendelsohn 1991), or focus on values of non-timber forests products (Schaafsma et al. 2013; Cavendish 2000; Shackleton et al. 2002). The literature on ecosystem services in China can be characterized in a similar way. This may change to some extent in future as the Natural Capital Project and collaborators have current initiatives to model ecosystem services in the Eastern Arc Mountains of Tanzania (Fisher et al. 2011), in the Primeiras e Segundas region of Mozambique, and at the national level in China.

The current geographic unevenness is in part related to the global distribution of forests. Indonesia and Brazil are the subject of many more valuation studies than any other countries, and are also in the top three countries in the world in terms of tropical forest area. The other country in the top three is the Democratic Republic of Congo, suggesting that the logistics of conducting primary research also influence the selection of case study locations. A third factor is that there are only a limited number of individuals conducting research within this field, for reasons discussed below. A result of this is that particular case study sites recur frequently. For example, nine of the ecosystem service estimates included in the tables in Section 3 come from the study sites of Ruteng National Park and Lore Lindu National Park, both in Indonesia.

Figure 5: Geographic distribution of valuation studies



The first implication of the relatively limited geographical scope is that there is very little information available on the values of ecosystem services in the lowest-income countries. It is likely that the monetary values of services such as air quality or water availability would be lower, but the impacts as a proportion of income may be larger, for example if households are more reliant on agriculture or less able to mitigate the health impacts of poor air or water quality or the impacts of natural disasters such as floods or landslides. The second implication is that as the existing studies have largely been conducted in populated regions of highly populated countries, the total benefits will be higher than average because there are more potential beneficiaries. There will be many areas of forest where the values of local ecosystem services are close to zero because they are remote from population. Therefore, when interpreting or transferring the values presented in this paper, they should only be considered to apply to locations where there are local human populations. In practice, however, deforestation pressures tend to be strongest where people have access to forests so such locations will be the focus of the majority of conservation planning and decision making.

Within each country the studies reviewed here concentrate, either implicitly or explicitly, on natural rather than plantation forests. This makes it more difficult to assess the local benefits of reforestation programs, as opposed to avoided deforestation, especially because planted forests are likely to provide some, but not all of the benefits of natural forests. For example,

to the extent that deforestation increases flood risks due to reduced soil infiltration or river sedimentation, reforestation will not necessarily reverse the risks (van Dijk et al 2009). More generally, the benefits of reforestation depend substantially on what type of forest is planted, and what it replaces. Chazdon (2008) presents a 'restoration staircase', in which commercial reforestation provides lower biodiversity and ecosystem services than reforestation with native trees or natural forest regeneration. Brockerhoff et al (2008) point out that the benefits will also depend on what the alternative land use would be, in that planted forests may provide improvements in biodiversity and wildlife habitat relative to agricultural land, but not relative to natural non-forested land.

4.2 Topical coverage of studies

As well as the uneven geographic distribution, there is also substantial variation in the degree to which different categories of ecosystem service have been studied. In some cases, this is due to the significance of the service. For example, more research has been conducted into the value of erosion control for agricultural production than the value of shade provision, because the former has wider applicability to a range of farming systems.

However, the primary reason for limited evidence on the values of some services is that they are very challenging to accurately quantify. As mentioned at the outset of this review, the data requirements and necessary knowledge to model all stages of the relationship between policy actions, changes in ecosystem condition, changes in production or consumption, and, ultimately, changes in human wellbeing are considerable. As a result, the services that are particularly complex, in terms of both the biophysical relationships and the human impacts, have so far not been extensively studied. Some services, such as the impacts of forests fires on incidence of respiratory illness are relatively straightforward at least to approximate. Others, such as the benefits of erosion control can be valued in a single stage using econometric methods that relate forest cover to agricultural revenues. In contrast, the contributions of forests in reducing downstream flooding or landslides, or the impacts of deforestation on malaria vector habitat, appear to be highly non-linear and to vary significantly depending on the context. In these cases, site-specific modeling of the hydrological or ecological relationships as well as the human responses to these is necessary.

More generally, publication of studies that perform high-quality, original valuation of the impacts of ecosystem services on human wellbeing is fairly limited and appears to have declined in recent years. As discussed above, the geographical coverage is largely restricted to

a handful of countries, and for each category of service, there are at most a handful of studies, with few published in the past five years. Even within the studies reviewed here, it is generally the case that either the relationship between the ecosystem condition and the ecosystem service, or the relationship between changes in services and changes in wellbeing are well-understood, but not both. As this field develops, there is greater understanding of what constitute conceptually and empirically robust methods for valuation of ecosystem services. This has had three important impacts: 1) the quality of studies has increased over time, giving confidence in the accuracy of the estimated values for the setting in which they are studied; 2) benefit transfer has also improved so that more attention is generally paid to both biophysical and socioeconomic context when transferring values from one setting to another; but 3) it appears that fewer original studies are being conducted, primarily due to the need for interdisciplinary research in order to apply suitably state-of-the-art methods and models from multiple relevant fields to these questions.

Some interdisciplinary work is being conducted, for example Barbier and co-authors use non-linear bio-economic and wave attenuation models to estimate impacts of mangroves on coastal communities (Barbier et al. 2008). However, the inherent difficulties of this type of research, combined with a general lack of recognition within the economics profession (for example, little interdisciplinary research is published in the top economics journals), mean that the incentives for high quality, original academic research in this area are poor. The result is that significant expansion in the volume of original studies that would permit comparison of the relative importance of different ecosystem services in different settings seems unlikely.

An important caveat to this is that the focus of this paper on published research hides a significant body of work that is being conducted to aid land-use planning in specific locations. There is growing demand for *ex ante* estimation of the benefits and costs of potential management changes, as well as evaluation of those changes, from regional and national governments around the world. This is driving efforts such as the Natural Capital Project and the European Environment Agency's Land and Ecosystem Accounting project, that aim to meet this demand in part by improving methods for benefit transfer. These projects combine information from existing studies with socio-ecological models and GIS data in order to provide policy-makers with estimates of monetary values that are grounded both in theory and in the physical and socio-economic context in which the services occur. These are important contributions to the field, and directly contribute to improving land

management decisions. However, the outputs are less visible because their primary objectives are to produce decision support tools and information, rather than scientific publications, although the latter do also result, and may do so to a greater extent as programs are evaluated *ex post* using these tools.

5. Conclusions

Large-scale forest conservation policies, for example aimed at reducing carbon emissions, would limit the extent to which developing countries can generate income by clearing forest land for agriculture or timber. This could be perceived as a constraint on the development opportunities of poor rural households and communities in return for provision of a global public good. However, as the studies in this paper have demonstrated, while avoided deforestation programs may reduce income earning opportunities for local households and firms by prohibiting the use of land for agriculture and timber harvests, they also contribute to important development objectives such as improvements to health, physical safety, energy and food security, and in some cases alternative forms of income generation. Conversely, the importance of ecosystem services to physical wellbeing means that ongoing deforestation in developing countries brings both benefits and costs to local communities. As a result, the use of REDD+ mechanisms to incentivize forest conservation, and compensate for the the lost economic opportunities of forest conservation actions undertaken voluntarily by forest countries, is likely to have important co-benefits for communities located near to the protected forests.

There are four key ways that forests affect the wellbeing of local people, aside from direct use of the land or forest products:

- 1) Forests provide soil protection and water regulation services. This affects health through access to water and mitigation of waterborne diseases, physical safety through reductions in flood and landslide risk, energy security through improved functioning of hydroelectric facilities, and food security through regulation of water for irrigation, particularly in periods of drought. A related physical way in which forests affect wellbeing is through the storm protection provided by mangroves.
- 2) Forests provide habitat for birds, fish, mammals and insects that contribute directly or indirectly to food, income and health. For example, forests affect income by providing nursery grounds for commercially important fish and shellfish stocks and habitat for birds and mammals that can be important for attracting eco-tourists. They also indirectly

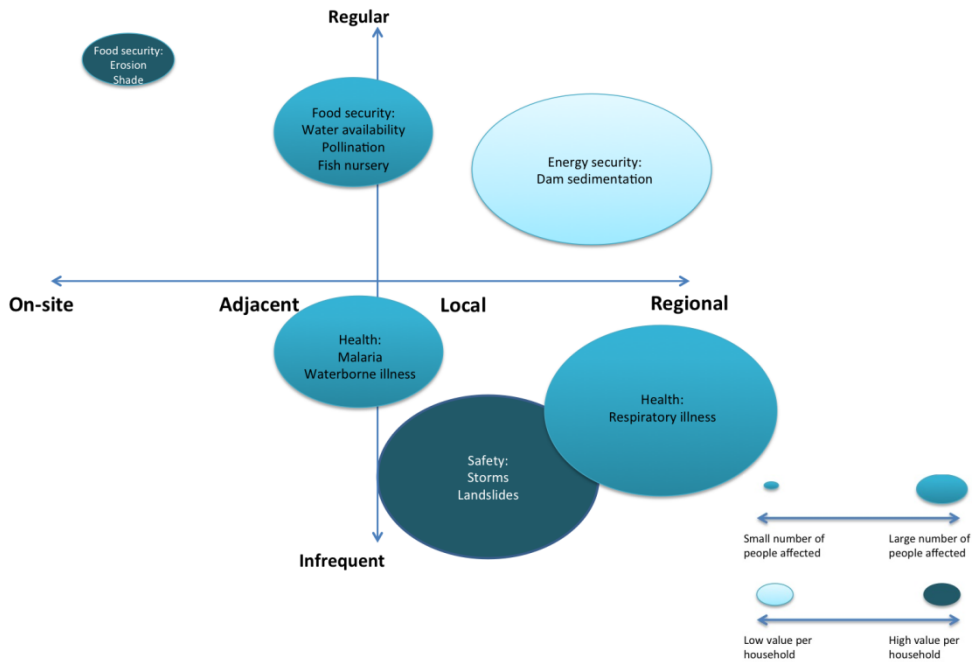
contribute to crop productivity through pollination services. The role of forests in providing habitat for insects that carry diseases such as malaria and dengue fever is not fully understood, but there is evidence that disease risks are higher near to disturbed forest than intact forest.

3) Clearing forest through use of fire can reduce wellbeing, particularly if the fires spread accidentally. In most years, these fires generate some local air quality problems, with implications for respiratory health, and some damage to pasture and timber stocks. However, in some years, most notably the 1997-8 season, weather conditions result in much more extensive fires with high costs over large geographical areas.

4) Tropical forests are particularly high in biodiversity. Of 24 high-priority terrestrial biodiversity hotspots identified by Mittermeier et al (1998), 14 are entirely tropical rain forest or tropical dry forest, and all but one contain some forest land. This diversity makes them important locally as well as globally as a potential source of genetic material that may have relevance for the development of pharmaceutical products or crop varieties.

Looking at the broad magnitude of the values of different ecosystem services from different study sites, some very general conclusions about the distribution of ecosystem benefits can be drawn. As discussed above, comparison of specific estimates is generally not useful because each relates to both ecosystem changes and beneficiary populations of very different scales and characteristics. However, looking across studies that value ecosystem services, some patterns can be observed. Figure 6 shows the location of different ecosystem benefits along the horizontal axis, and their frequency along the vertical axis. Darker shades indicate higher benefits per household, and larger ovals indicate greater numbers of people affected. The figure shows that 'common' benefits, i.e. those that are experienced regularly by many households, tend to be low for each individual household. Other benefits are large for individual households, but either accrue to only a small number of households (such as the on-site shade and soil erosion control benefits), or to many households infrequently (such as mitigation of natural disasters).

Figure 6: Size, distribution, scale, and frequency of ecosystem benefits



Most of the values reviewed in this paper are public goods, so consumption or use of the benefits by one individual or household does not reduce the potential benefits for others. As a result, the total value of the benefits is most strongly determined by the number of beneficiaries in proximity to the resource. For example, the largest total values arise from ecosystem services that affect large numbers of people, such as air quality and storm protection, as opposed to those that are very localized such as water provision and insect-borne disease regulation. This has implications for the transfer of values to new settings: as the majority of the existing studies are from relatively densely populated countries in South East Asia, the total value of many ecosystem services from forests in other parts of the world will be smaller (although the costs of forest protection may also be lower in less densely populated countries).

The geographic distribution of ecosystem services also affects which groups benefit. On-site and adjacent services will generally benefit rural households, while both rural and urban households will gain from local, and in particular, regional services. Overall, those living closest to the forested area will tend to benefit from multiple services, whereas those living further away will only be affected in specific ways, for example if air quality is affected for cities downwind of a deforested region. The result is that local households are likely to gain most from forest protection, both because they will benefit in multiple ways and because

impacts often diminish with distance. An exception to this is if local and more distant populations have different preferences, for example, urban residents may value a forest park for ecotourism opportunities while local rural households do not. Barbier et al. (2008) highlight which stakeholders each of the benefits of mangrove protection or conversion accrue to in Case Study 1 above. The economic profits from shrimp farming are largely earned by outside investors; the coastal community as a whole benefits from the storm protection functions of mangroves; and the local, mangrove-dependent coastal community benefits from storm protection as well as collection of wood productions and habitat-fishery linkages. Similar distributions of gains and losses can be observed for other ecosystem services.

Beyond geographic location, poorer households are likely to be disproportionately affected by losses of ecosystem services for a three main reasons. First, extensive tropical forest areas often coincide geographically with large numbers of poor people (Wunder 2001), so the adjacent and local households most affected by changes in ecosystem services will tend to be relatively poor. Second, many of the benefits amount to a few dollars per household per year, which will be negligible to richer households, but can be a significant proportion of annual agricultural profits for poor farm households. Third, the welfare impacts of ecosystem services such as disease prevention or mitigation of natural disasters will often be greater for poor households because their baseline vulnerability is higher. For example, the impacts of the Indonesian forest fires were most severe for the poor, the young and the elderly (Jayachandran 2009). Similarly, poor households were most severely affected by the 2004 Asian Tsunami (Rodriguez et al. 2006), as well as least able to recover (Sawada 2007; De Mel, McKenzie, and Woodruff 2012).

The results presented in this paper cannot be used in isolation to determine whether or where conservation of forest land should occur, as the benefits of forest clearing will also be location-specific. The finding that the estimated values tend to be either fairly small (impacts on agricultural productivity) or large but rare (impacts on health and physical safety) suggests that in at least some cases the economic benefits of deforestation will exceed the benefits of conservation. However, as many of the forest ecosystem services are public goods and the studies reviewed here show that they can be locally important, there will also be cases where the local benefits of ecosystem services exceed the economic returns to forest clearing even where deforestation is observed to occur. Where ecosystem service values exceed local economic returns to deforestation or where the beneficiaries of deforestation differ from the

beneficiaries of forest protection, the introduction of forest conservation policy will improve wellbeing for some groups. Even if the economic returns to deforestation are higher overall, the presence of local ecosystem services reduces the opportunity costs of forest protection and therefore the potential compensation required for avoided deforestation.

The studies reviewed in this paper indicate that the most important determinant of the overall value of the local ecosystem services is how many people are affected. Beyond that, many of the services benefit those engaged in agriculture or fishing, while those at risk of natural disasters can also be significantly affected, and as discussed above, the impacts are likely to be higher for poor households. It will therefore be most important to account for local ecosystem services in conservation planning when 1) rural population density is high, both adjacent to the forest and in the lower reaches of the same watershed; 2) in regions where risks of natural disaster are high; and 3) in cases where the income level of the affected population is low. These are likely to coincide with locations where risks of deforestation are also high, and therefore where there is a need for action to be taken. As a result, the locations where local ecosystem service values are most important, understanding their size and distribution will also be most policy-relevant.

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