

Understanding Spatiotemporal Lags in Ecosystem Services to Improve Incentives

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Ecosystem-service production is strongly influenced by the landscape configuration of natural and human systems. Ecosystem services are not only produced and consumed locally but can be transferred within and among ecosystems. The time and distance between the producer and the consumer of ecosystem services can be considered lags in ecosystem-service provisioning. Incorporation of heterogeneity and lag effects into conservation incentives helps identify appropriate governance systems and incentive mechanisms for effective ecosystem-service management. These spatiotemporal dimensions are particularly apparent in river–riparian systems, which provide a suite of important ecosystem services and promote biodiversity conservation at multiple scales, including habitat protection and functional connectivity. Management of ecosystem services with spatiotemporal lags requires an interdisciplinary consideration of both the biophysical landscape features that produce services and the human actors that control and benefit from the creation of those services.

Keywords: connectivity, conservation, ecosystem function, ecosystem service, payment for ecosystem services, riparian areas

Many human-dominated landscapes have lost or decreased their ability to provide key ecosystem services, such as clean air and water, agricultural pollination and pest predation, and carbon storage (Costanza et al. 1997, Vörösmarty et al. 2010). Although technological solutions may exist or may be developed to reverse this trend, many are unsustainable, can negatively affect human communities, and require large initial investments, such as the construction of dams to store water or filtration facilities to improve water quality (Gleick 2003). In particular, technological solutions also often lack the adaptability inherent in ecological systems, an attribute of increasing interest in the face of impacts from changing climates and land use (Levin 1999). In response, policymakers increasingly search for effective management solutions that foster safe land-use practices and protect livelihoods concurrently with the conservation of biodiversity and a wide range of ecosystem services (DeClerck et al. 2006).

In recent years, a conceptual shift in land management options has occurred from a focus primarily on provisioning services (food and fiber) to a more comprehensive approach that includes regulating, supporting, and cultural services provided by ecosystems, such as carbon sequestration

(Lal 2004), pest and disease regulation (Cardinale et al. 2003), soil erosion control (Estrada Carmona and DeClerck 2011), and a clean air and water supply (Stanton et al. 2010). Broadening the services we aim to protect recognizes that many services are currently unprotected and that their degradation can have significant impacts on human livelihoods and well-being (Karieva and Marvier 2007). Furthermore, researchers and policymakers increasingly recognize the spatial nature of ecosystem-service provisioning (Ricketts 2004, Chan et al. 2006) and aim to design policies to protect those services that are transferred among actors or across landscapes. Some services (e.g., soil nutrient cycling) are provided and consumed locally (a short temporal and spatial lag) and can therefore be considered *private services*. Other services (water quality, carbon sequestration) have a long lag between production and consumption among different areas on the landscape and can be referred to as *public or common-pool ecosystem services* (*sensu* Ostrom et al. 1999).

Recognition of the multiple scales of ecosystem services and their lags will improve the design of effective management strategies, including scale-appropriate incentive mechanisms that properly identify and protect suites of key ecosystem services at their relevant scales (Jack et al.

2008) without sacrificing livelihoods, public safety, and biodiversity. Building incentive mechanisms for both private and common-pool services requires quantifying and governing ecosystem services that accumulate over large areas and over long time steps and that can be transferred across governance structures, such as from landowner to landowner or from country to country. For this reason, researchers have increasingly sought to provide spatially explicit decision-making tools for managing ecosystem-service incentive mechanisms (Nelson et al. 2009, Villa et al. 2009). However, a crucial but often missing perspective in the success of ecosystem-service incentive mechanisms, including payment for ecosystem services (PES), lies in understanding the ecological drivers of ecosystem-service provision over local and global scales, the motivating forces that drive landowners to manage for the provision of ecosystem services, and the value of specific ecosystem services to consumer groups—all of which vary in heterogeneous landscapes. Addressing these crucial unknowns in ecosystem-service management will require integration of knowledge bases across disciplines and scales.

Here, we outline a scale-based ecosystem-service framework that integrates biophysical and social drivers for managing

ecosystem services (figure 1). We argue that effective management of ecosystem services and their incentive mechanisms requires quantifying the geography of social-ecological systems; this includes (a) how socioenvironmental heterogeneity varies with spatiotemporal scales, (b) understanding the spatiotemporal lag between landscape features that produce ecosystem services and the consumers of those services, (c) recognizing the considerable variation in the human valuation of each service, and (d) matching the spatial scale at which ecosystem services are generated and managed to improve the design of policies and incentive mechanisms for effective ecosystem-service management. In the present article, we outline this framework and use river-riparian systems as an example to improve ecosystem-service management.

Targeting ecosystem services: Location, location, location

Ecosystem services are the subset of ecosystem processes that provide direct benefits to humans; they have been defined as the conditions and processes through which natural and human-modified ecosystems and the species that constitute them support and sustain human needs (Daily 1997). The Millennium Ecosystem Assessment (MA) classifies

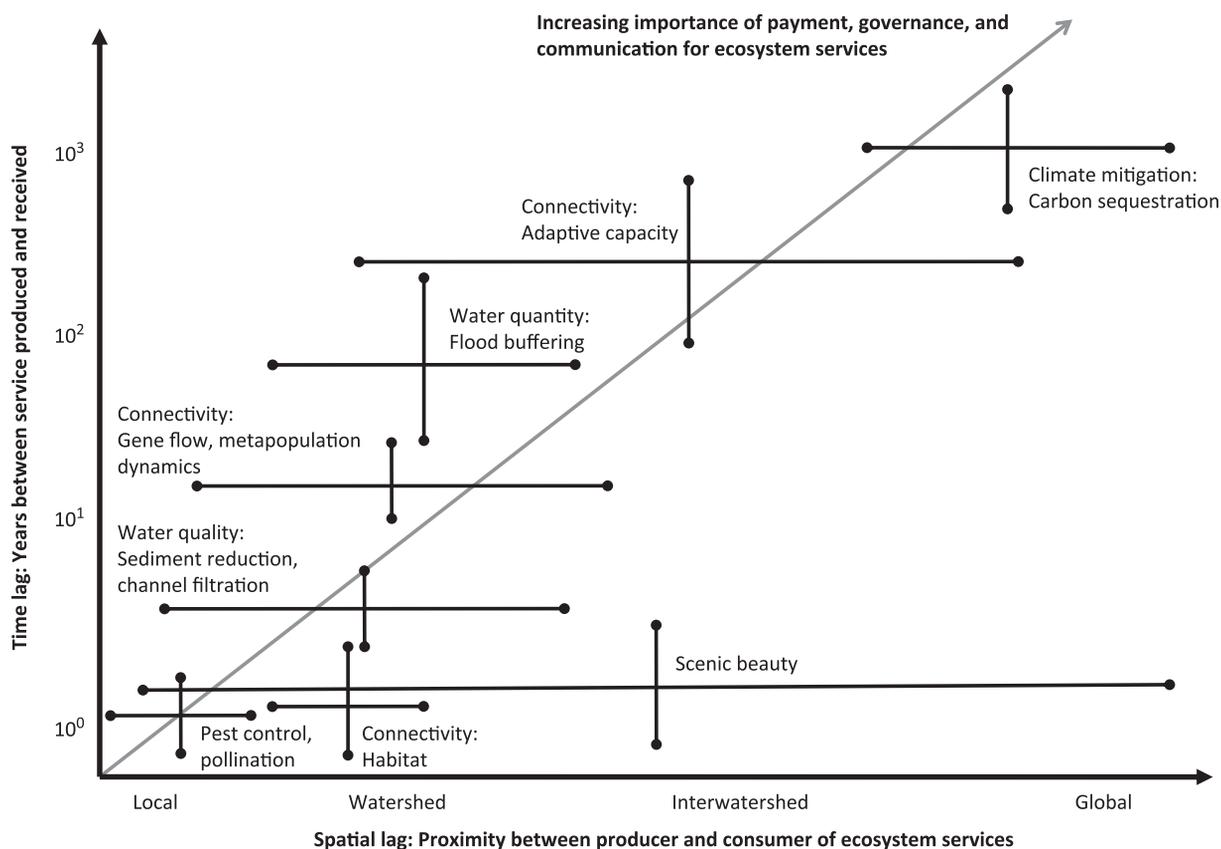


Figure 1. Effective management of ecosystem services requires an understanding of the lags between production and consumption, particularly across well-connected landscape features, such as within river-riparian systems. The gray line illustrates the increasing importance of management or payment for ecosystem services schemes and of matching the scale of the services with that of the organization.

these into four broad categories: (1) provisioning services, (2) regulating services, (3) support services, and (4) cultural services (MA 2005). *Provisioning services* are those goods and benefits provided by ecosystems, including the production of crops, timber, textiles, energy, water, and pharmaceuticals. *Regulating services* include the regulation of climate, hydrology, pest and disease cycles, and decomposition and detoxification rates. *Support services* include plant pollination and nutrient cycling. *Cultural services* include less tangible but equally important services, such as the spiritual, cultural, and recreational benefits that humans receive from ecosystems.

The influence of biophysical and social heterogeneity on ecosystem-service provisioning. The types and delivery rates of ecosystem services vary across a landscape because of the underlying biophysical characteristics, such as geology, soil type, climate, topography, and species composition, as well as their ecosystem landscape configuration and temporal variability (Chan et al. 2006). As an example, soil erosion is determined by several factors within a basin, such as cover management; slope steepness; slope length; soil characteristics; and the amount, seasonality, and frequency of precipitation events; among other factors (Renard et al. 1997). The magnitude of vegetation's impact on soil loss reduction varies by soil erosivity factors and topographic position. Therefore, any policy for improving erosion control should acknowledge these interacting effects and the possibility that a blanket policy for reforestation might not be as effective as one that prioritizes highly erosive portions or uses of the landscape.

For example, the US Department of Agriculture's Conservation Reserve Enhancement Program has a specific protocol for prioritizing lands for biological, hydrological, and soil conservation. This voluntary program has protected millions of square kilometers, with a clear enhancement of ecosystem-service benefits—mainly soil-loss protection and improved downstream water quality (Van Buskirk and Willi 2004). Understanding the spatial dimension of variables that drive ecosystem-service provision has been the basis of much of the ongoing work in developing maps of ecosystem services, such as those accessed using InVEST (Nelson et al. 2009) and ARIES (Villa et al. 2009). Identifying the drivers of ecosystem-service variability and quantifying the rates and scales at which the variability is relevant to service quantification and resource policy will help scientists and managers prioritize locations for payments, particularly when services are bundled (Kareiva and Marvier 2007, Pijanowski et al. 2010). However, the social landscape on which these services are provided needs to be understood as well in order to facilitate the appropriate policy formulation and implementation.

Just as the composition and configuration of a landscape's biophysical features have a direct impact on the type and rate of ecosystem-service provisioning, social heterogeneity is an important driving force in determining the adoption of conservation incentives for ecosystem services. The human

geography of a landscape or how distinct stakeholders (composition) are distributed (configuration) determines the spatial distribution of both those who have the capacity to manage an ecosystem to provide specific services and those who consume, value, or benefit from those services. This information is essential in the design of incentive mechanisms (Daily et al. 2009).

The adoption of changes in land-use practices will reflect the differences among landowner characteristics and motivations for conservation because of varying property size, location, land use, economic status, access to information, and trade-offs. For example, in Costa Rica, coffee farmers were aware of the ecosystem services provided by shade trees within their farms, but the trade-offs between specific ecosystem services and productivity drove management practices (Cerdán et al. 2012). Families on small farms, whose annual income is substantively dependent on production from the land, might also have fewer resources (monetary and land) and less access to information to assist them in adopting conservation practices than do owners of larger farms, who may have secondary sources of income and higher levels of education (Miranda et al. 2003, Vignola et al. 2012). Vignola and colleagues (2012) also found that the owners of smaller farms are aware of soil-loss risks but state that the cost of implementing conservation practices is too high. Short-term income needs can be of higher value than long-range planning and prioritization of values regarding ecosystem services, even though these same stakeholders scored higher on their understanding of the causes of soil erosion than did the wealthier owners of larger farms.

Stakeholder configuration in a landscape can also affect the success of ecosystem-service governance. The concentration of small farms in riparian zones of Brazilian landscapes led to the veto of large sections of Brazil's forest code in June of 2012. The agricultural lobby demonstrated that the protection of riparian zones, as was mandated by the code, would disproportionately harm the rural poor who owned small plots of land located entirely within the proposed protected river margins.

Environmental heterogeneity is widely recognized by ecologists, but this disciplinary group may be less prepared to understand social or economic heterogeneity, and social scientists may tend to ignore environmental heterogeneity at the expense of social heterogeneity. Understanding both types of heterogeneity and the interaction between them is key to building efficient policies for governing ecosystem-service production and consumption (Barrett et al. 2011). The origins, scales, rates, and valuation of ecosystem-service production and consumption are variable because of the heterogeneity of both the biophysical and the social landscapes. This heterogeneity challenges scientists and decisionmakers to prioritize areas of high importance and to incorporate the uncertainty of ecosystem-service provisioning into environmental policies (Ostrom 2005).

A failure to recognize both types of heterogeneity or to incorporate them into policy is evident in Costa Rica's

1996 forest law, which mandates 100-meter-wide riparian buffers in steeply sloped lands and 30-meter-wide buffers in flat lands. The law recognizes the generalizable soil erosion gradient whereby steep upland slopes generate sediment that is deposited in lowlands. However, it fails to recognize that the social geography of the country also follows the topographic gradient. Much of the country's vegetable production takes place on small or medium plots on steeply sloped higher-elevation plots where cooler climates predominate, as do smaller property holdings, in contrast to the larger landholdings found at lower elevations (mainly banana, pineapple, and pasture). In this case, the national law is seen as unfairly placing greater restrictions on small farmers with riparian areas on their farms because of the socioeconomic patterns across the landscape.

An example provided by Kareiva and Marvier (2007) demonstrates how the mapping of social, biophysical, and ecosystem characteristics can be used to proactively prioritize ecosystem-services-based interventions and management. Kareiva and Marvier (2007) mapped ecologically threatened wetlands, regions prone to flooding, and poverty hotspots along the Florida panhandle and suggested that conserving threatened wetland systems (ecological mapping) could mitigate the impact of storm-surge damage and flooding caused by hurricanes (biophysical mapping) in more vulnerable poor communities (social mapping). The combination of the three maps provides the geography of flood buffering as an ecosystem service by identifying who requires the service, which ecosystem provides the service, and where that service is provided. The combination of the three elements provides a crucial entry point for developing scale- and heterogeneity-appropriate incentive or governance mechanisms.

Ecological and sociological lags between producers of ecosystem services. A second fundamental component to managing the mechanics and governance of ecosystem services is the recognition that services are produced and consumed over different spatial and temporal scales and can be dissociated across spatiotemporal and political boundaries (Hein et al. 2006, Jackson et al. 2010, Thorp et al. 2010). Figure 1 illustrates the spatiotemporal lags between ecosystem-service providers and consumers specifically for river-riparian ecosystem services, although this conceptual framework is applicable to other systems. Borrowing from landscape ecology, we refer to this dissociation as the lag between the production and consumption of ecosystem services, such that a greater lag implies a greater spatial or temporal distance between the producer and the consumer of a specific ecosystem service.

At fine scales, on the order of hundreds of meters, forests or buffer strips of seminatural vegetation can provide important pest control and pollinator functions to adjacent fields (Ricketts 2004, Avelino et al. 2012). At this scale, the landowners implementing the intervention or their immediate neighbors are the direct beneficiaries of the intervention (the lower left corner of figure 1). At local

and regional scales (1–10 kilometers), intact headwater riparian areas filter sediment and excess nutrients, but the improved water quality is gained downstream (the center left of figure 1). This represents an increase in the spatio-temporal lag from the pollinator and pest control functions. At even higher lags, the carbon sequestered by large intact forests mitigates climate change, benefiting the entire global community, as is depicted by the upper righthand side of figure 1. Identifying lags between producers and consumers of ecosystem services will help policymakers match the type and scale of the incentive mechanism to the scale of the ecosystem service in question.

Ecosystem services that are produced and consumed locally (short lag) will probably require less outside management because of the proximity of the provider and the beneficiary and can be generalized as private ecosystem services (the bottom left of figure 1). That is, if local services are truly beneficial, an aware and well-informed land manager would have a strong incentive to recognize and protect them (Cerdán et al. 2012, Garbach et al. 2012). For example, if pollination and pest control services outweigh the perceived cost of alternative measures, the land manager has a clear incentive to manage and protect these riparian stands and the ecosystem services that they provide.

With an increasing lag and with increasing governance, producers may require outside incentives for ensuring ecosystem-service provision (figure 1). These long-lag ecosystem services can be considered common-pool resources and will require governance structures that treat them as such (Ostrom et al. 1999). *Common-pool resources* are characterized by their complexity, subtractability, and difficulty of exclusion. *Subtractability* occurs when the use of the resource by a single property owner reduces the availability or the quality of the resource available to other users. *Difficulty of exclusion* refers to the large scales at which these resources are delivered, whereby resource protection through landowner exclusion is practically impossible (Ostrom et al. 1999).

When the services provided by upstream landowners (or nonlocal landowners) have a recognized downstream value, the governance or incentive structure should match the scale of the lag, such as a local water tax (Villalobos and Solano Valverde 2007). For example, watersheds draining into the Panama Canal have, in part, been reforested by companies using the canal to mitigate changes in seasonal supplies of water flow back into the locks and to reduce sedimentation in the canal (Carse 2012). The city of Heredia in Costa Rica likewise found that residents were willing to pay an additional water tax of \$0.03 per cubic meter to protect the city's water source. Collective payments provided \$104 per hectare per year for forest conservation and \$577 per hectare per year for restoration (Villalobos and Solano Valverde 2007, DeClerck and Le Coq 2011).

At a similar lag, reforestation or forest protection to reduce soil loss can have downstream impacts on streams, reservoirs, and coastal estuaries. The Costa Rican Institute

of Electricity (ICE) manages the country's hydroelectric dams and provides financial and technical incentives to landowners within the watershed who adopt soil conservation practices, therefore reducing sediment transport to local reservoirs (Estrada Carmona and DeClerck 2011). Sedimentation affects dam operations through reservoir filling and increased maintenance costs, such as those associated with dredging or controlled releases of water. To minimize this problem, ICE makes payments or provides technological improvements to upstream landowners who implement soil conservation practices. Although there is much room for improvement and expansion of this program, this strategy helps build awareness and an incentive structure to aid upstream landowners in recouping the opportunity cost associated with many conservation interventions that benefit downstream communities.

At the global scale, REDD+ (Reducing Emissions from Deforestation and Forest Degradation) programs are intended to build an incentive structure for forest conservation and carbon sequestration, which are services provided to the global community. The large lag between the producer and the consumer of carbon sequestration services requires governance mechanisms at the same scale (the right top of figure 1). It also provides an opportunity to consider other social and ecological patterns that operate at the same scale—the increasing carbon storage, poverty density, and biodiversity threats associated with tropical regions, for instance. Venter and colleagues (2009) mapped priority areas for carbon storage and biodiversity conservation to demonstrate how global ecosystem-service incentive mechanisms can be bundled and targeted to increase the efficiency of meeting global targets for biodiversity conservation and climate change mitigation. Knowing what services are provided and valued at a specific scale and the economic and social trade-offs involved in managing these services will be key for designing appropriate incentive mechanisms for conservation and management in social-ecological landscapes.

In summary, we suggest that incentive mechanisms should be directly proportional to the spatiotemporal lag in the area, distance, or time between the production and the consumption of the ecosystem service in question. Alternatively stated, the motivation to conserve an ecosystem service is inversely proportional to its lag, which suggests that the most effective incentive structure is managed by the governance scale related to the lag between production and consumption of the service. Because, in reality, the menu of ecosystem services of interest to society are provided at multiple scales, multiscale coordination and a bundling of incentive mechanisms will be necessary.

More coordination is required through communication, governance, and policy among government agencies and landowners, as well as between neighboring countries and regional governing entities with increasing ecosystem-service lags (figure 1). This is particularly complicated when services are transferred across multiple jurisdictional

boundaries and scales, such as those between public and private lands, international borders, and socioeconomic clines.

Regardless of the scale of the lag, however, policy and management will be improved by a clearer understanding of the social contexts that motivate actors to protect particular services. For example, effective resource management groups have been shown to form around a shared resource of interest, not at political boundaries (Ostrom 2005). The degree of participation in watershed-scale groups (e.g., community-based drinking water organizations) is driven by the perceived value of the desired or managed ecosystem service—in this case, clean water. A direct dependency and trust between the producer and the consumer of the resource is still evident. Stakeholders retain sufficient interpersonal ties to develop locally specific access rules and to be capable of developing a shared vision for resource management (Ostrom et al. 1999). Personal investment in the management of the resource can contribute to a feeling of ownership, which can lead to higher organizational performance (Madrigal and Alpizar 2011). In this way, social organizations facilitate peer-to-peer interaction between the providers and the recipients of services—a crucial social function when midsize lags exist. In the case of global-scale lags (carbon sequestration), peer-to-peer interactions are impossible, and, therefore, global incentive mechanisms, such as REDD+, CDM (Clean Development Mechanism), WAVES (Wealth Accounting and the Valuation of Ecosystem Services); education; or markets will be key to ensuring ecosystem-service provision at these lags.

Although coordination among actors with shared resource interests improves success, individual levels of motivation, understanding, and the ability to act to conserve ecosystem services can be highly variable (table 1). Moreover, the difficulty of coordinating PES schemes among many small farms, including understanding the incentives and management protocols and access to payments, remains a barrier limiting their widespread implementation (Jack et al. 2008). In these cases, protecting ecosystem services is likely to require greater incentives and increased landowner participation in governance processes (Porrás et al. 2012).

Case study: Ecosystem-service-based management of riparian areas in Mesoamerica

Ecosystem-service-based management schemes have a relatively long history in Mesoamerica (Rapidel et al. 2011). Here, our goal is to adjust our scale-based framework to riparian areas and to underscore the importance of recognizing the interdisciplinary geography of ecosystem services and to provide an example of a scalable, multifunctional policy solution to ecosystem-service management.

A management focus on river-riparian areas recognizes that the dendritic landscape structure, high and unique biodiversity, and the capacity to recover after disturbance make them particularly well suited to provide ecological services and species conservation at multiple scales and lags (Chan et al. 2006, Thorp et al. 2010). Protecting riparian

Table 1. River–riparian ecosystem services across varying spatial scales.

Question	Spatial scale				
	Local	Watershed	Interwatershed	National	Global
What riparian ecosystem services are relevant at each scale?	Pest control, pollination, erosion control, forest products, scenic beauty, food production	Water quality, water quantity, scenery or recreation, climate regulation, hazard mitigation, erosion control	Habitat connectivity, gene flow, air quality	Carbon sequestration, hazard mitigation, scenic beauty, habitat connectivity, food production	Carbon sequestration, hazard mitigation, scenic beauty
Who is interested in conserving these ecosystem services?	Landowners, area residents	Hydroelectric agencies, drinking water groups, conservation organizations, tourism groups, fishermen, fish and agricultural farmers	Conservation organizations (e.g., biological corridor committees), public agencies	Ministries (e.g., of agriculture, environment, health), conservation organizations, consumers (interested in eco-socio-labeled products)	United Nations, World Bank, Millennium Development Goals, REDD+ (Reducing Emissions from Deforestation and Forest Degradation), Convention on Biological Diversity
Why are they motivated to conserve these ecosystem services?	Livelihood, aesthetics, regulatory penalties, well-being, ethical concerns	Energy production, social well-being, aquatic resources	Species conservation, to increase resilience or to mitigate climate change	Sustainable economic growth, human well-being, food and nutrition security, a commitment to global conventions and their monetary incentives	Sustainability, climate change mitigation, poverty reduction, food and nutrition security, biological conservation

forests provides natural corridors for species throughout human-dominated landscapes and between protected areas for a range of species' movements (e.g., figure 2; Hilty and Merenlender 2004, Sanfiorenzo et al. 2011). Riparian areas also make significant contributions to environmental management by acting as filtering sites between terrestrial and aquatic systems by increasing soil infiltration capacity (figure 3), mitigating flood damage, supplying clean water to households, improving in-stream supply, and helping to recharge groundwater in their floodplain (Brauman et al. 2007). Moreover, rivers and riparian forests have strong cultural value and are often the first areas delineated for protection (Tockner and Stanford 2002) or the last areas remaining with forest cover in many regions in Mesoamerica.

Riparian services across a nested set of spatial scales

Riparian forests may enhance pest control services by increasing landscape complexity and providing barriers to pest movement in agriculturally dominated areas (Avelino et al. 2012). For example, the movement of a coffee-boring beetle between coffee fields is most restricted by forests in the mixed-use landscapes of Costa Rica (Avelino et al. 2012), including narrow riparian forests. In addition, riparian forests provide dry-season fodder and water for grazing cattle (Harvey and Haber 1999), which enhances herd resistance to drought in the Pacific and central portions of Mesoamerica (Useche et al. 2011). Along riverbanks, undisturbed vegetation cover benefits in-stream water quality. Riparian trees maintain channel bank cohesion through the propagation of roots that strengthen soil structure and reduce bank erosion while facilitating the deposition of sediment (Gurnell 1997). Riparian forests also provide valuable fuel wood, natural medicines, and aesthetics to property owners.

When implementing conservation strategies at the watershed scale, scientists should assess the effect of upstream

conservation measures on the downstream transfer of ecosystem services. This scale of spatial lag applies to many public services, including flow regulation, flood buffering, sediment retention, pollution control, and scenic beauty (figure 1, table 1; Biggs et al. 2006, Stanton et al. 2010). These services typically benefit the public in general and downstream inhabitants in particular; however, some services are provided to others outside the watershed and region.

Water-based and watershed-scale PES schemes and management plans are booming in Latin America (Balvanera et al. 2006). These services are often quantified at the *watershed scale*, which is defined by an area of land within a bounded hydraulic system and typically encompassing more than one property. Once the spatiotemporal lag between the production and the consumption of the ecosystem service exceeds individual property boundaries, the service becomes a common-pool resource (*sensu* Ostrom et al. 1999). As an example, in the dry forest ecosystems of Nicaragua, a higher leaf-area index in riparian forests is correlated with significantly higher hydraulic conductivity (figure 3; Niemeyer 2011). Hydraulic conductivity governs infiltration, which is one of the primary mechanisms by which overland flow is decreased, and thereby reduces pollutant movement into streams and increases groundwater recharge (Chaves et al. 2008). By using remotely sensed data, it is possible to scale infiltration capacity to an entire watershed to better target a location for ecosystem-service payments (Chávez Huamán 2010). Overlaying these data with social data, such as parcel size and road networks in the watershed, will further improve PES effectiveness.

In addition, two current PES schemes in the biological corridors of Costa Rica have recently been transformed to facilitate the management of hydrological services, although their original objective was ensuring functional connectivity for wild biodiversity. In the Volcánica Central

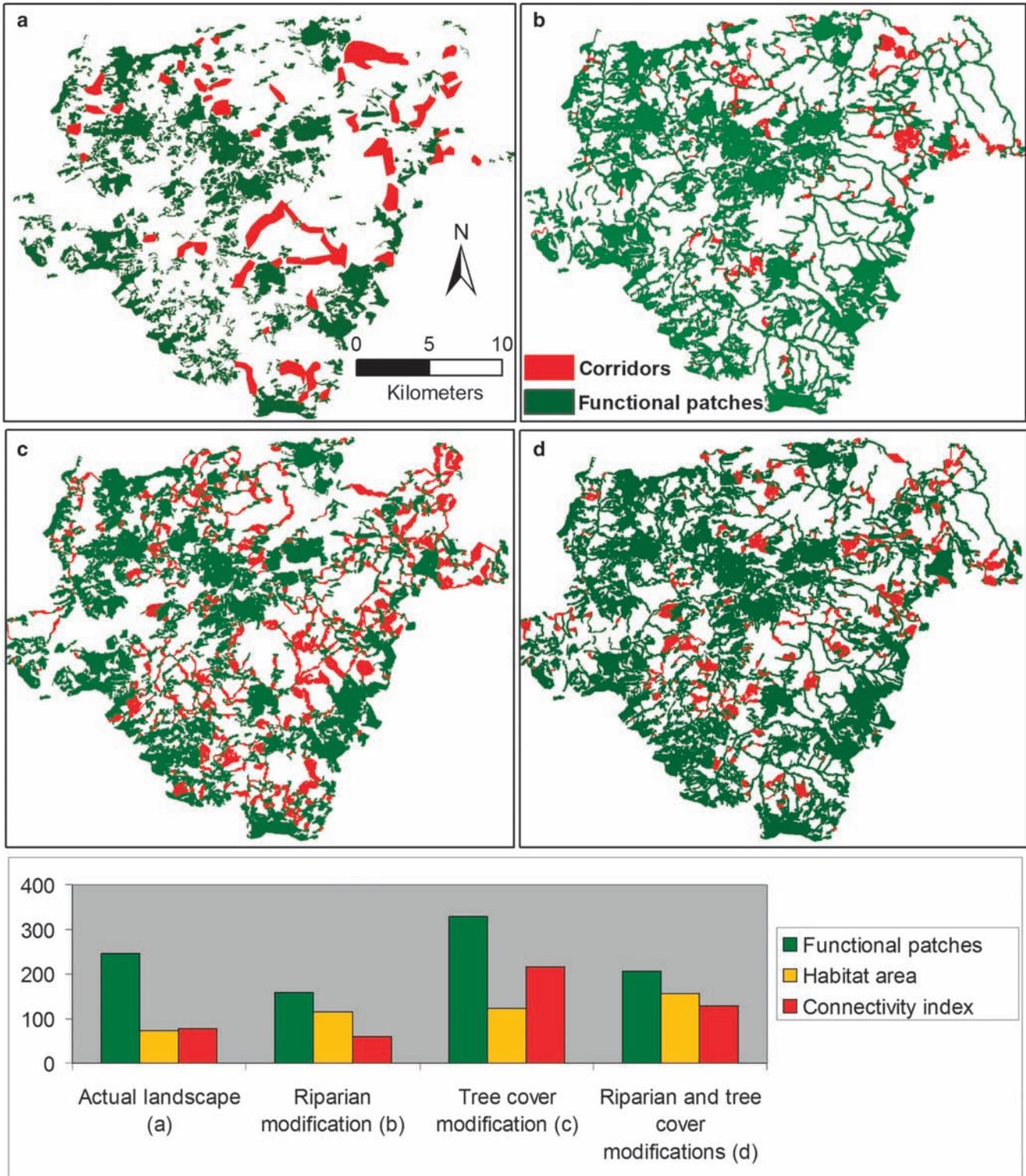


Figure 2. The impact of four land-cover change scenarios on landscape fragmentation and functional connectivity: (a) the actual landscape; (b) the inclusion of 50-meter riparian forests; (c) tree cover modifications with low tree density silvopastoral systems on inclines of less than 15%, high-density silvopastoral systems on inclines between 16%–29%, and forest regeneration on inclines steeper than 30%; (d) combination of riparian corridors and tree cover modifications (Sanfiorenzo et al. 2011). The results show that these changes will reduce the number of patches (the green bar in the bar graph), increase the amount of habitat available (the yellow bar, in square kilometers), and increase the connectivity of habitat (the red bar) for forest-dependent species, such as trogons.

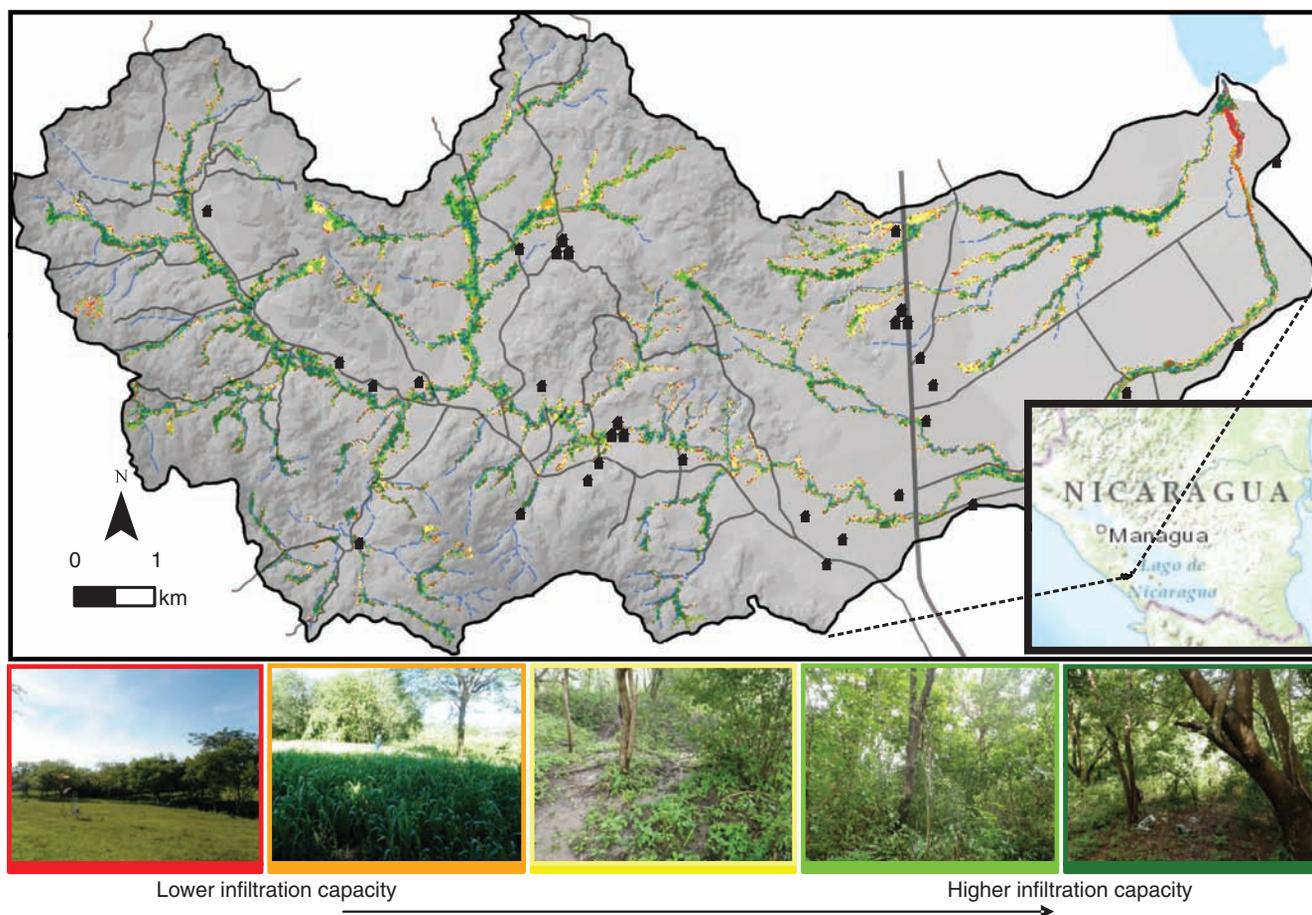


Figure 3. Water quality is improved through increased vegetation in riparian areas by increasing infiltration capacity. Using remotely sensed data, we are able to scale this ecosystem service to the watershed scale to target areas for ecosystem-service payment schemes (Chávez Huamán 2010, Niemeyer 2011). The photographs illustrate different vegetation classes based on leaf-area index, scaled to a watershed, using the Normalized Difference Vegetation Index. The black symbols are settlements and the color borders around the photos correspond to infiltration capacity on the map. Abbreviation: km, kilometers. Photographs: Ryan Niemeyer.

Talamanca Biological Corridor, payments are distributed for hydrological services such as sediment reduction and improved water quality by ICE, which was described above (and detailed in Estrada Carmona and DeClerck 2011), and payments are distributed for forest conservation in the Hojancha–Chorotega Biological Corridor. The effort is led by the local community as a means of increasing community resilience to El Niño–related drought events.

The conservation impact of riparian forest interventions can also largely be seen at watershed scales. Although riparian forests are important sources of habitat at the local scale, they provide crucial connectivity between forest fragments across the watershed in human-dominated landscapes (figure 2; Sanfiorenzo et al. 2011). Maximizing available habitat throughout the watershed, riparian forests ensure the continuity of ecological communities across elevation gradients and provide potential lateral connectivity in watersheds with high drainage densities (Brudvig et al. 2009). Sanfiorenzo and colleagues (2011) compared land-cover modifications in productive areas and the inclusion of

riparian forests in the Copán River watershed, Honduras, and demonstrated that, although both strategies independently conserve equal-size areas, only the riparian forest conservation strategy contributes to reducing the effects of fragmentation. Increased functional connectivity allows species to move to better habitats for different life-history needs and increases wildlife populations' resilience to short- and long-term environmental change.

Several riparian ecosystem services operate beyond the watershed scale, which might be called *national* or *global commons* (Ostrom et al. 1999). There are growing efforts to value these services in national and international accounting mechanisms, such as the World Bank's WAVES initiative. Carbon sequestration by riparian forests serves as one example; less obvious is the habitat connectivity within and between watersheds provided by riparian corridors. Galleries of riparian forests allow species to move across a landscape (Gillies et al. 2011). Habitat connectivity improves a species' ability to respond to environmental change, be that a press or pulse disturbance (Kostyack

et al. 2011). The scenic beauty offered by natural rivers and floodplains emerges at local scales with direct economic value at the national and international scales (figure 1). For example, Costa Rica's national parks generate an estimated \$1.5 billion in revenue per year (Moreno Díaz et al. 2010), largely driven by the scenic and recreational values embedded in forests, rivers, beaches, and wildlife viewing. Incentives for conserving ecosystem services at the inter-watershed and higher scales require governance structures that match these scales.

Incentives for riparian forest conservation at a national scale may be present in the form of a country's commitment to biodiversity conservation or in the form of climate-change-related agreements, for example. At international scales, many countries in both the developed and the developing world are signatories to climate change protocols (e.g., Kyoto, REDD+), the Convention on Biological Diversity, or the United Nations' Millennium Development Goals, all of which can be classified as international drivers for conservation and ecosystem-service protection. Costa Rica's national interest in forest conservation, as another example, is in part driven by political interest and, more recently, by an active policy to be the first carbon-neutral country in the world. Although not a service specific to riparian areas, the protection of riparian forests aids in the effort to sequester carbon while simultaneously providing other economic and ecosystem-service-derived benefits. Recommendations by the National System of Conservation Areas specifically highlighted the protection of riparian areas for their conservation value in human-dominated landscapes of the country (SINAC 2007).

In the case of carbon, when compared with those of hydrological services, the spatial and temporal lags between the producer and the consumer are very high. Direct negotiation between the producers and the consumers of the service is difficult and is complicated by the complex relationship between carbon sequestration and global climate change. When spatiotemporal lags between producers and consumers are high, such as in the case of carbon sequestration, higher-order institutional arrangements are needed to incentivize ecosystem-service producers and to assure service delivery to consumers.

In another example of interwatershed-scale services, the impacts of high sediment loads not only decrease water quality and hydroelectric reservoir life span (watershed scale) but also have significant effects on coastal systems (coral reefs and estuaries), as is the case with the Mesoamerican Barrier Reef—the second-largest barrier reef in the world (Burke and Maidens 2004). Coordinated local- and watershed-scale efforts to conserve intact riparian forests and floodplains can help mitigate erosive land-use practices in multiple degraded watersheds draining into the Caribbean. Considering that the scale of the problem includes hundreds—if not thousands—of watersheds across Mesoamerica and affects services such as tourism and marine fisheries consumed at international, national, and

regional scales, incentives at these same scales are needed to ensure an effect. Although many organizations operating at these scales already exist, they could be further mobilized to focus on ecosystem-service investments and on monitoring riparian areas across national boundaries.

In addition to the services provided, riparian forests provide crucial habitat for the conservation of biodiversity at multiple scales. Conservation of even narrow belts of riparian forest combined with high densities of trees on adjacent lands can positively affect macroinvertebrate, fish, reptile, and amphibian diversity richness (Couceiro et al. 2006, Lorion and Kennedy 2009) and can lead to significant increases in reptile and amphibian diversity (Gómez et al. 2011).

The habitat provided by riparian forests mediates energy budgets across the aquatic–terrestrial interface (Sabo and Power 2002). This has unique implications for pest control services, because emergent aquatic insects constitute seasonally important food sources for insectivorous birds, bats, and spiders, which function as important predators of agricultural pests (Fukui et al. 2006, Marczak and Richardson 2007). The deforestation of riparian forests can result in significant changes to these subsidies (Chan et al. 2006), with potentially indirect effects limiting pest control services on adjacent agricultural lands. However, since most of the work on cross-system subsidies has been completed in temperate regions, an important goal for future research is to determine to what extent these subsidies exist in tropical regions.

Ecosystem services as an interdisciplinary framework

Considering that ecosystem-service-based management solutions require a detailed understanding of both the ecosystem and the socioeconomic forces at work locally, nationally, and regionally (Stanton et al. 2010), we argue—as others have—that management must increasingly foster interdisciplinary studies (Sabatier et al. 2005). That is, we begin to ask compound questions relating to the flows of ecosystem services and to the social landscapes on which they occur. For example, what are the types and amounts of ecosystem services provided by nondegraded ecosystems, and how are they valued at multiple scales? By matching the scales of ecosystem-service production with ecosystem-service consumption in social–ecological systems, we will be better able to develop incentive structures that account for spatial lags in ecosystem services (figure 1, table 1).

Effective ecosystem-service-based intervention depends on a clear understanding of the interactions among the biological, physical, and socioeconomic aspects both of the services that stakeholders at multiple scales are motivated to conserve and of the services available from the landscape. Application of multifunctional, landscape-scale conservation efforts must incorporate the social and economic heterogeneity of the people who inhabit the landscape, whose individual contexts and motivations will guide policy adoption and involvement in conservation interventions.

Moreover, it is crucial to understand consumer motivations for conservation and to quantify the spatiotemporal

lag between ecosystem-service provision and consumption. The ecological mechanisms that drive these services must also be clearly understood, including the location and rate of service provision. For example, in watershed-scale interventions, the services provided in headwater regions are distinct from those provided near the mouth of a river and are readily transferred to other ecosystems and thereby benefit others. The determination of the spatiotemporal patterns of ecosystem-service provision is therefore crucial in designing appropriate ecosystem-service-based management schemes. Furthermore, ecosystem-service approaches require an understanding of the complex interactions between the environmental constraints on a service and the added effect of ecological community diversity in a landscape shaped by multiple land managers and owners with varying motivations, goals, and constraints.

In summary, identifying policy and governance options to link producers and consumers to incentive mechanisms will require overlapping governance where multiple scales of authority interact with multiple ecological scales and a diverse set of human actors (Ostrom 2005, 2009). The most appropriate policy measure to support ecosystem-service-based management is dependent on the specific social-ecological scale and context of the ecosystem service in question. Therefore, we believe that explicit incorporation of scale, social context, and ecosystem processes is essential to the effective planning and implementation of ecosystem-service management programs.

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