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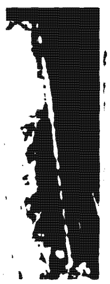
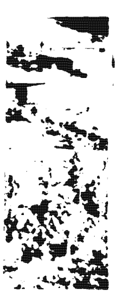


"Integrated Management of Environmental Services in Human-Dominated Tropical Landscapes"

"Manejo Integrado de Servicios Ambientales en Paisajes Tropicales Intervenidos"



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Abstracts

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Sesión 4

Ecosystem Services of Tropical Landscapes

Jeffrey Sayer[1]

Abstract

Environmental services have value to society; however the cost of providing them often falls on local land managers whilst the benefits are public goods. It is necessary that societies provide incentives or apply regulations to maintain these environmental benefits. Methodologies are proposed that can help in understanding the complexity of the mosaic landscapes that provide these services in the tropics. These approaches help to develop shared goals and are a necessary foundation for any scheme to compensate providers of services and to do this in ways that are equitable and contribute to the livelihoods of the poor. Integrated approaches to the management of large complex landscapes are costly and time consuming but it is important that they are undertaken in some representative tropical environments. R&D in these areas may yield insights that have widespread application. Success requires new approaches to the organisation of science and new relations between scientists and local resource users.

The problem of improving livelihoods and providing environmental services

All land uses, even monocultures of commodity crops, have impacts on the flow of environmental services. Throughout the world land has always been managed to moderate negative impacts and enhance positive impacts on the environment. Problems arise because the beneficiaries of environmental services are often not those who incur the cost of ensuring their provision. Land is usually managed for private benefit and environmental services are generally a public good. Regulations and incentives therefore have to be used to achieve an optimal balance between private and public benefits.

This problem is particularly severe in many tropical countries and especially in areas with high levels of poverty. Important environmental benefits are often provided by land where tenure and access rights are unclear or where institutions and regulatory frameworks are weak. The opportunity costs to poor people of maintaining a land cover that is optimal for the environment may be very high. These problems are most severe in the areas of lower agricultural potential where many poor people live. The hillsides of Central America are a notable example.

Much has been written about the potential of environmental service payments to be used to provide an incentive for farmers in the tropics to retain or replant forests and commit to other forms of land management that enhance the flow of environmental benefits. Such payments are seen as a contribution to alleviating the poverty of the rural poor. Governments and prosperous urban people appear to be willing to pay for such environmental services and the rural poor might reasonably be expected to be willing sellers. However it has been extremely difficult to put into place operational schemes that allow payments to be made in a sustainable way. The major problems appear to be both the difficulty of measuring and valuing the services and of assessing the cost incurred by rural people in providing them. The basic requirements for a market in environmental services are rarely present.

[1]Senior Associate – WWF – International

The obstacles to PES schemes in poor tropical areas are a special case of the problems encountered by many approaches to integrating conservation and development. Integrated conservation and development initiatives have always underestimated the difficulty of determining how much conservation and how much development is desirable in any particular situation. There has been a reluctance to confront the reality that conservation almost invariably imposes local economic costs and development often harms the environment. There are always gainers and losers in any intervention. A remarkably high level of investment has been made over several decades in development assistance projects that hid behind the myth that some idealistic solution existed where everyone would gain. There may be situations where this is true but if so one might expect that markets and societies would have found them.

Setting goals: The process

A lot of scientific attention has been given recently to designing mechanisms for environmental service payments and for establishing the value of the environmental payments concerned. Most attempts have been based upon macro-level estimates of services such as carbon sequestration, potential earnings from eco-tourism, conservation of water resources or from pharmaceutical products that might be derived from wild plants. There is another, and perhaps greater scientific challenge – that of measuring the flows of development and environmental benefits at smaller scales. Many environmental benefits flow from mosaic landscapes where the combined impacts of large numbers of individual actors combine to influence the performance of catchments or the habitat of species of conservation concern.

The need to deal with environmental and developmental issues in the context of complex landscape mosaics has led the conservation and the development communities on what have been described as "landscape" or "ecosystem" approaches. These share many characteristics with earlier attempts at integrated natural resources management and integrated catchment or river basin management. The weakness of all of these integrated approaches has been the difficulty of understanding and measuring the ways in which the actions of large numbers of actors operating within the cells of a matrix combine to impact on public goods environmental services.

A number of people have now attempted to tackle these problems and this presentation will present some promising lines of enquiry. I will particularly focus on the understanding and measurement of livelihood and environmental outcomes in what are commonly called "landscapes". For the purpose of this work we have adopted two definitions or explanations of what we mean by the term landscape.

A "Landscape" is a geographical construct that includes not only the biophysical components of an area but also social, political, psychological etc components of that system.

"A landscape is a geographical space in which the process or object of interest is completely expressed, observed or functions."

It is worth noting that the definition of the "ecosystem approach" adopted by the Convention on Biological Diversity in 2000 gives heavy emphasis to social and economic issues and is close to the definitions of landscapes given above. Earlier definitions of ecosystem management had tended to focus on purely biophysical components of systems – for instance the Ecological Society of America.

Building a shared vision

The challenge then is to measure attributes of a system – the landscape mosaic – that are manifest at the landscape scale. This cannot simply be based upon expert judgements or assessments. If one expects the different actors within a landscape to act in a coherent way to achieve landscape-scale outcomes then it is highly desirable that they share a common understanding of what the landscape is and, to the extent possible, what the desirable outcomes are. The participatory rural appraisal literature is rich in examples of techniques that can be used to build such a shared understanding of the "landscape". Participatory mapping is commonly used. We have adopted the approach of "rich pictures" described by Bell and Morse (1999 and 2003). In this approach the actors in a landscape draw their understanding of the landscape on a flip chart. The discussion that this generates can be a valuable way of getting people to be explicit about their assumptions and to share their understanding of landscape linkages.

It is often difficult to make people think about the major external influences that impact upon a landscape. A useful way of generating debate on this topic is by a participatory exercise in conducting a historical trends analysis. This is simply an exercise in which local actors record on a flip chart the events that they consider to have been important in influencing the historical changes in the landscape. A historical trends analysis can provide a good basis for a participatory process of developing scenarios for possible future landscapes – and seeking to understand the major influences that will determine which scenario will materialise.

Understanding causal relations

These participatory exercises reveal the extreme complexity of the processes that determine both the developmental and environmental flows from landscape mosaics. To further understand the way in which the landscape performs as a system we have found it very useful to build simple simulation models of the landscape. The problem with such models is that many people are intimidated by their complexity and by the mathematical skills that they believe are needed to operate any computer model. We have however found that building such models in a participatory way together with local actors can be a powerful tool in sharing ideas, assumptions and building shared understanding. Even people with no previous experience of modelling and computers can take part in, and benefit from this process. We have succeeded is building models over a period of 3-5 days in rural locations with the full involvement of local officials, farmers and other actors. Participants generally report that their understanding of the issues and options is greatly enriched by this process. When we have built such models in locations where environmental service payments are being discussed we have found that the models reveal just how difficult the management of such payment systems would be. The diversity of actors and of interventions in the system is so high that the number of transactions needed to make appropriate environmental payments would be very high. Measurement of these transactions and the interventions that they would mediate poses serious technical problems.

Measuring performance of landscape mosaics

Large numbers of initiatives are still attempting to reconcile conservation and development outcomes at the landscape scale. In an attempt to better understand how they might function better and what the implications of environmental payment schemes might be we have been experimenting with the development of simple measures of the performance of the landscape. We have called our approach a "landscape tracking tool". Our assumption has been that it should be possible to negotiate with local actors a limited set of attributes of the landscape that could be readily measured and that would over time give some indication of the trends in the ability of the landscape to deliver environmental and development outcomes. We have used the capital assets framework (Carney et al, 1999, Sayer and Campbell, 2004) to measure changes in people's livelihoods in the landscape. Our approach has been to negotiate a set of 4 – 6 indicators for each capital asset with the local stakeholders. Each indicator is scored on a 1 – 5 Likert scale and the scores for each capital asset are summed. The results can be presented on radar diagrams and can be the subject of periodic negotiations amongst actors. The indicator sets are different at different locations but the asset framework provides for a limited degree of inter-site comparison.

We have measured environmental outcomes differently. In general the locations where we have worked have been selected because they had some environmental value of special interest. We have therefore sought to work with local environmental stakeholders to establish indicators that would be sensitive to changes in the desired environmental outcomes for the site.

Our experience with the landscape tracking tool has been that, even in relatively simple landscapes with few stakeholders and a low diversity of land cover types, the development of an indicator set is quite a difficult and time-consuming task. It requires several iterations with the local actors before agreement can be reached on a credible and acceptable set of indicators. Measuring these indicators needs dedicated staff over several months. In general we have been surprised to find that the indicators of success built into project log frames have limited value in assessing landscape scale outcomes. They usually focus too much on response indicators and project deliverables or outputs and say little about overall outcomes. Pressure and response indicators are much easier to develop than state indicators.

When are these approaches needed?

Our overall conclusion from our work to date is that intervening to achieve conservation and development outcomes at a landscape scale is difficult. If measures are not put into place to negotiate the desired outcomes, understand the interlinkages and the relative costs and benefits to different actors and measure the outcomes at a landscape scale then the use of the term "landscape approach" is empty rhetoric. This suggests to us that one should only take on problems at the landscape scale if it is genuinely necessary to do so. The incremental benefits of getting into more and more levels of complexity have to be greater than the incremental costs of doing so. In many cases it is probably more effective to focus on the management unit of primary concern – a protected area, a managed forest or a farm – and not pretend that one has the capacity to manage an entire complex system. However there are many situations where conservation and development outcomes have to be sought at larger spatial scales. The hillsides of Central and South America are a good example.

To tackle these complex problems requires very different approaches to those of conventional natural resources research and management. It requires long term commitments, significant resources and a major investment in developing relations with all local actors. It requires that the research is conducted on real life situations, on farms and with farmers. It implies that the researcher will be seeking to detect patterns and causality in the outcomes of individual farmer's actions and to aggregate these up to determine how the landscape as a whole is performing. All management thus becomes experimental and all research is action research at real life scales. Rich pictures, historical trends analysis, participatory modelling and performance tracking are then all contributing to the process of shared experimentation and learning. This is obviously a major undertaking and the results may or may not have application beyond the landscape under study. We do however believe that natural resource research institutes such as CATIE and CIFOR should have at least a small number of high priority sites where they do make a long-term commitment of significant resources to tackle these complex issues. Payments for environmental services in poor countries with weak institutions will not be possible without the scientific understanding and social learning that such major studies can provide.

Intermediate solutions; Forest landscape restoration

One approach that is an interesting example of an intermediate between the fully integrated natural resource management model and the segregated management unit approach is forest Landscape Restoration – FLR.

Simply put, forest landscape restoration brings people together to identify, negotiate and put in place practices that optimise the environmental, social and economic benefits of forests and trees within a broader pattern of land uses.

FLR has to be based on a holistic understanding of the land use system – the landscape – but it is implemented through the tactical planting or management of trees in ways that favour particular targeted environmental and livelihood outcomes. It seeks to optimise benefits at the scale of the landscape and to do so at minimum cost. The concept of FLR and some guidelines for implementing it in the field are given in a review by Mansourian et al (2005). FLR is now the subject of a Global Partnership and is a rallying ground for forestry agencies striving to integrate their activities with the rest of the "landscape" in order to ensure the sustainable delivery of environmental benefits.

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Ecosystem Services: if they're so important, why do we continue to abuse them?

Dr. Robert Costanza[1]

Contrary to conventional economic wisdom, the environment is not a luxury good. The services provided by intact and functioning ecosystems contribute to human welfare and survival in innumerable ways, both directly and indirectly. We are only beginning to understand the pathways and magnitudes of these contribution, and advances in ecosystem science and integrated ecological economic modeling will help to further clarify and quantify these connections. We are also beginning to get a better handle on the magnitude of ecosystem services contribution to human welfare. For example, we have estimated that globally, ecosystem services contribute at least $33 Trillion/yr - larger than global GNP. But ecosystem services are largely public goods, and their value exists outside the market. Because their value is not perceived in conventional economic transactions and decision-making, ecosystem services, and the natural capital stocks that produce them, have been depleted and degraded by human actions to the point that the sustainability of the system is threatened. The solution is not, however, to privatize ecosystem services and set up conventional markets for them. Public goods are not well handled by private markets. The solution is to recognize the value of these public goods and modify market and other incentives to communicate that value to private decision-makers. Systems such as ecological taxes and subsidies, government mediated payment systems for ecosystem services, protected areas, and conservation easement and concession systems are some of the tools that are useful for incorporating the value of ecosystem services into private decision-making.

[1] Gund Professor of Ecological Economics and Director, Gund Institute of Ecological Economics Rubenstein School of Environment and Natural Resources The University of Vermont Burlington, VT 05405-1708 Telephone: 802.656.2974 Fax: 802.656.2995 email: Robert.Costanza@uvm.edu http://www.uvm.edu/giee

Consideraciones económicas y sociales en el desarrollo de un esquema de pago por servicios ambientales (PSA): teoría y práctica.

Alpízar, F; Madrigal, R.[1]

Resumen ejecutivo

Existe una alta demanda por el diseño e implementación de esquemas de PSA que incluyan activamente la participación local y la sostenibilidad financiera de las acciones a ejecutar. Para satisfacer estas necesidades se requiere de un marco metodológico integral de referencia, el cual debería incluir al menos los siguientes componentes: Diagnóstico de la problemática, análisis biofísico de la oferta actual y potencial de servicios ambientales (SA), medición de los costos de proveer SA, identificación y medición de la demanda de SA por parte de beneficiarios potenciales y por último, creación del marco operativo apropiado para la escala de intervención seleccionada.

Este marco metodológico ha sido validado para el caso del servicio ambiental hídrico en varios sitios piloto localizados en Centroamérica. La validación, realizada en diferentes grados, demuestra la efectividad de este marco para contribuir con la evaluación del potencial de operación de un esquema de PSA hídrico. Los casos de estudio que se presentan son representativos de la aplicación de alguno de los componentes de la metodología. Los casos de aplicación escogidos, bajo el criterio de representatividad geográfica, entorno socioeconómico e interés metodológico, son: la construcción de un índice de usos del suelo relacionados con la provisión hídrica, la definición de áreas prioritarias e inversiones necesarias a realizar en la subcuenca del río Jucuapa (Nicaragua), la estimación de la valoración de demanda por un programa de inversiones que promete beneficios ambientales hídricos para el cantón de Esparza (Costa Rica) y por último, el diseño del marco operativo e institucionalidad general alrededor de la implementación de un esquema de PSA hídrico en Copán Ruinas (Honduras).

El índice de usos del suelo relacionados con provisión hídrica (Alpízar, F; Madrigal, R. 2005. Proyecto GEF-FAO: Insumo para PSA hídrico en Nicaragua, Colombia y Costa Rica) constituye un esfuerzo por superar la incertidumbre asociada a la provisión hídrica y busca promover, en última instancia, un pago más acorde a la compensación por ofrecer servicios ambientales. Adicionalmente, el índice envía señales más precisas acerca de cuáles son los cambios más deseables a nivel de paisaje para incrementar la oferta hídrica. La construcción del índice, que incluye combinaciones de más de 20 usos y prácticas de manejo, se realizó gracias al trabajo conjunto en un taller de expertos internacionales, con representantes de varias organizaciones tales como FONAFIFO, MINAE, SINAC, PASOLAC, CIPAV, UNA, RUTA-BM, UCR, SNV, NITLAPAN, CATIE, entre otros.

En relación con la estimación de costos y priorización de áreas de intervención en la subcuenca del Río Jucuapa (Baltodano, MA; Alpízar, F; Madrigal, R. 2005. FOCUENCAS), se realizó la georeferenciación y caracterización de 23 fuentes de agua, se seleccionó un conjunto de áreas críticas con base en criterios de uso de la tierra y pendiente, y finalmente se estimaron y validaron los costos de implementar cambios en el

[1] 2005. SEBSA, CATIE

uso de la tierra y los costos de la adopción de prácticas de conservación de suelos y agua. Estos últimos van desde la opción más barata 78$/hec de inversión inicial y 16$/hec de mantenimiento en los años subsiguientes hasta la opción que incluye más obras físicas de conservación, con un costo de 243$/hec y 31$/hec de inversión y mantenimiento, respectivamente.

Por otra parte, la valoración de demanda por servicios ambientales hídricos en Esparza (Alpízar, F; Madrigal, R. 2005. Proyecto Enfoques Silvopastoriles Integrados para el Manejo Sostenible de Ecosistemas) encontró que la población del casco urbano de este cantón está dispuesta a pagar, en promedio y por encima del pago actual del servicio de agua, la suma de 1.09 $ mensuales para financiar un proyecto de inversiones en las zonas críticas de provisión de agua para el cantón. Con base en este monto, se estima que la población estima sus beneficios en un máximo de 54,262 US $. En un escenario de recaudación conservador, donde se cobraría el 50% de la disponibilidad de pago promedio, el monto posible de recaudación asciende a 27,131 US $.

En el caso de Copán Ruinas (Cisneros, J; Alpízar, F; Madrigal, R. 2005. FOCUENCAS), el estudio de la institucionalidad y el diseño del marco operativo para el PSA, muestra la importancia de la participación activa y la apropiación de la comunidad de los principios que dan sustento al PSA. La legalidad existente también es un factor positivo a considerar, además del diseño de una estructura operativa del manejo del PSA que se fundamente en las estructuras organizacionales ya existentes. De forma preliminar, se sugiere que el marco operativo del PSA incluya una Junta Directiva, representativa y con facultades efectivas para aprobar planes de trabajo y presupuestos, entre otras funciones esenciales. Adicionalmente se plantea la necesidad de que esta estructura contenga una rama operativa (seleccionar áreas prioritarias, hacer planes a nivel de finca, proponer tecnologías etc), una financiera (pagar oferentes, elaborara informes financieros, realizar transferencias) y por último, una rama encargada de las labores de monitoreo (revisar cumplimiento de planes, asignación de fondos, etc).

El diseño y validación de la metodología deja varias lecciones. Para empezar, la puesta en marcha de un esquema de PSA local depende de varios factores que van más allá de la correcta estimación de costos de inversiones, estimación de demanda por SA y marco operativo del esquema. Esto no resta importancia a la metodología sino que hace un fuerte llamado de atención para ubicar la misma dentro de un entorno político, socioeconómico y de posibilidades reales de superar los problemas de acción colectiva en las cuencas hidrográficas.

De esta forma, la viabilidad de los esquemas de PSA también depende del grado de empoderamiento de las comunidades en la toma de decisiones a nivel local, lo cual se traduce en un abandono de actitudes reactivas de las comunidades ante los dilemas de manejo de recursos naturales hacia una actitud más proactiva y comprometida con la solución de problemas que afectan sus formas de vida. Esto implica a su vez que existen canales adecuados de negociación y resolución de conflictos, derechos de propiedad bien establecidos y seguros, y normas de conducta y códigos sociales que promueven la conservación de los recursos naturales y el trabajo colectivo.

Adicionalmente, la experiencia ha demostrado la importancia de visualizar a los esquemas de PSA como procesos de negociación entre partes que rinden frutos sustanciales en el mediano y largo plazo. Para que este proceso se consolide, y rompa con los horizontes de planeamiento tradicionales, es necesario que se abran canales para el aprendizaje, la adaptación, la evolución y la divulgación de los resultados del programa. Para finalizar, la experiencia de trabajo de CATIE en el tema de PSA señala que la infraestructura de acueductos, y la búsqueda de aliados para el apalancamiento de fondos y desarrollo de proyectos, son variables que condicionan la generación de confianza alrededor de la puesta en marcha de esquemas de PSA.

¿Solucionando el Problema de Monitoreo? El Uso de un Indice Ecologico como Herramienta para Aplicar un Pago por Servicios Ambientales

J.Gobbi, M. Ibrahim, F.Casasola, E. Ramírez y E. Murgueitio

I. Introducción

Existe creciente evidencia indicando que los paisajes rurales manejados en forma amigable con el ambiente proveen una serie de servicios ambientales, entre ellos la generación de agua, la captura de carbono y la conservación de la biodiversidad. Por otro lado, existe un marcado interés por parte de la sociedad en asegurar la provisión de esos servicios, particularmente en tierras privadas. Una estrategia para alcanzar dicho objetivo es la compensación financiera a los propietarios rurales por los servicios ambientales originados a través de la incorporación de usos de la tierra amigables con el ambiente en sus fincas. Sin embargo, dicha estrategia enfrenta dos dificultades. Por un lado, los servicios ambientales mencionados son relativamente complejos de cuantificar en la práctica, variando el nivel de complejidad según el servicio ambiental de que se trate. Por ejemplo, la unidad de medida para el servicio de captura de carbono es ton de C, pero cuando se intenta cuantificar las ton de C generadas por un determinado tipo de uso de la tierra, digamos una pastura con alta densidad de árboles, se debe considerar sólo la biomasa aéra o también se debe incluir el C acumulado en el suelo? En el caso de biodiversidad es más complejo aún, ya que la misma se puede medir a diferentes escalas (por ejemplo, a nivel de especies o paisaje) y no tiene una unidad única de medida. Por el otro, las técnicas actuales de cuantificación y monitoreo de los servicios ambientales tienen costos relativamente elevados. Esto determina que si los proveedores de los servicios ambientales debieran internalizar dichos costos, la compensación financiera recibida por los mismos podría no ser lo suficientemente atractiva para asegurar la provisión de los servicios ambientales. Una de las maneras de remover las dificultades mencionadas anteriormente es por medio del diseño de un índice ecológico que permita (i) estimar la cantidad de servicio ambiental generado y (ii) servir de base para operacionalizar un esquema de pago en forma práctica y costo efectiva. En este trabajo se presenta un índice ecológico y la experiencia de su aplicación para implementar un pago por servicios ambientales en fincas ganaderas de Nicaragua, Colombia y Costa Rica.

2. El Indice Ecológico

El índice ecológico es una aproximación para estimar la cantidad de servicio ambiental generado en una finca cuando un uso de la tierra cambia de A a B, y puede ser usado para el monitoreo de los servicios ambientales y para definir la cantidad de pago por los mismos. El índice se construye listando los principales usos de la tierra presentes en el área de trabajo. Basados en la información disponible de la literatura, información de experimentos en el campo y "estimaciones educadas" de expertos, se asignan puntos a cada uno de los tipos de uso de la tierra de acuerdo a su capacidad para generar servicios ambientales (en este caso: biodiversidad y carbono). La Tabla 1 ejemplifica el índice ecológico utilizado por el proyecto GEF-Silvopastoril para operacionalizar el esquema de pago aplicado por el mismo. La forma de uso de la tierra a la cual se asigna el mayor valor es el bosque primario. Esto asegura su conservación y evita incentivos perversos que pudiesen conducir a su destrucción. A las tierras desnudas o degradadas se les asigna un puntaje igual a 0, ya que el aporte de servicios ambientales por parte de los mismos es nula. Los demás sistemas de uso de la tierra son calificados de 0 a 1 según su contribución a la conservación de la biodiversidad y de 0 a 1 según su capacidad de retención estable de carbono. Para estimar el aporte a la captura de carbono por cada uso de la tierra, se establece una equivalence de 1 punto del índice = 10 ton de Carbono. En el caso de biodiversidad, los usos de la tierra son ranquedos según su complejidad florística y estructural. El puntaje del índice ecológico para cada uso de la tierra resulta, entonces, de la suma del puntaje asignado por biodiversidad y por carbono.

Tabla 1. Ejemplo de índice ecológico utilizado por el proyecto GEF-Silvopastoril.

Uso de la Tierra	Biodiversidad	Carbono	[illegible]
	Puntaje	Puntaje	[illegible]
Cultivos Granos y Tubérculos	0	0	0
Pastura Degradada	0	0	0
Pastura Natural sin Árboles	0.1	0.1	0.2
Pastura Mejorada sin Árboles	0.1	0.4	0.5
Cultivos semi perennes (plátano café sin sombra)	0.3	0.2	0.5
Pastura Natural + baja densidad árboles 30 árboles/ha	0.3	0.3	0.6
Pastura Natural Enriquecida con árboles en baja densidad árboles 30 árboles/ha	0.3	0.3	0.6
Cercas vivas nuevas o con podas	0.3	0.3	0.6
Pastura Mejorada Enriquecida con árboles en baja densidad árboles 30 árboles/ha	0.3	0.4	0.7
Cultivos frutales monocultivo	0.3	0.4	0.7
Banco forrajero de gramíneas	0.3	0.5	0.8
Pastura mejorada + baja densidad de árboles	0.3	0.6	0.9
Banco forrajero con leñosas	0.4	0.5	0.9
Pastura Natural + alta densidad de árboles	0.5	0.5	1
Cultivos frutales poli cultivo	0.6	0.5	1.1
Cercas vivas multiestrato o cortinas rompe vientos	0.6	0.5	1.1
Banco forrajero diversificado	0.6	0.6	1.2
Plantaciones maderables Monocultivo	0.4	0.8	1.2
Café con Sombra	0.6	0.7	1.3
Pastura mejorada + alta densidad de árboles	0.6	0.7	1.3
Guadua o Bambú	0.5	0.8	1.3
Plantaciones maderables Diversificada	0.7	0.7	1.4
Tacotales	0.6	0.8	1.4
Bosque ripario	0.8	0.7	1.5
Silvopastoriles Intensivos	0.6	1	1.6
Bosque secundario Intervenido	0.8	0.9	1.7
Bosque secundario	0.9	1	1.9
Bosque primario	1	1	2

3. Operacionalización del Esquema de Pago

El principio rector del sistema de pagos basado en el índice ecológico es que el finquero provee servicios ambientales por medio de los cambios en el uso de la tierra en la finca, al pasar de monocultivos de pasturas naturales a sistemas de vegetación más complejos que incluye la integracion de arboles en las sistemas. Por lo tanto, cambios en los patrones de uso del suelo se toman como indicadores del volumen de los servicios ambientales provistos. Los pagos se efectúan en forma proporcional al incremento total en servicios ambientales medidos por medio de la aplicación del índice ecológico, en relación a una línea de base establecida al año 0. La aplicación del índice se realiza a nivel de finca para evitar problemas de "fuga", y el puntaje obtenido por la finca en su conjunto es la base para aplicar el pago. La cantidad a pagar por servicios ambientales surge de multipicar el puntaje obtenido por la finca por el valor monetario asignado al punto del índice. El valor monetario del punto del punto del índice se deriva de los precios de operatorias de pago por servicios ambientales existentes en el región. La Tabla 2 presenta un ejemplo de la operatoria aplicada por el proyecto GEF-Silvopastoril.

Tabla 2. Cómputo del pago por servicios ambientales. Proyecto GEF-Silvopastoril.

Uso suelo	Puntaje Indice	Linea Base		Primer año	
		Area	Area * Ptos	Area	Area * Ptos
Pastura natural sin árboles	0,1	15	1,5	3	0,3
Pastura natural baja densidad árboles	0,6	5	3,0	5	3,0
Pastura mejorada baja densidad árboles	0,9	0	0,0	10	9,0
Tacotal	1,4	0	0,0	2	2,8
		20 4,5 20		**Total**	15,1
				Puntaje incremental	**10,6**
Pago (puntaje incremental*US$ 75/punto)					**795,0**

4. Ventajas y Desventajas de un Indice Ecológico como Base para un Esquema de PSA

La aplicación de un índice ecológico como base para operacionalizar un esquema de pago por servicios ambientales posee las siguientes ventajas: (i) es barato de monitorear, comparado con las técnicas de cuantificación actuales de servicios ambientales, (ii) es fácil de entender por los proveedores de los servicios, y (iii) es perfectible, según se obtenga información de campo más confiable acerca de la cantidad de servicio ambiental generado por determinado uso de la tierra. Sin embargo, un esquema de pago por servicios ambientales basado en un índice ecológico requiere de (i) estudios de monitoreo para ajustar el puntaje asignado a los distintos usos de la tierra y (ii) de un marco contractual adicional que evite incentivos perversos.

2nd Session

Harnessing the Ecosystem Services of Tree Cover in Agricultural Landscapes Abstract

Sara J. Scherr, Ph.D.[1]

Agriculture (for domesticated crops, livestock, forest products and fish) has become the dominant terrestrial land use, and agricultural expansion and intensification have become the main drivers of biodiversity loss and ecosystem degradation globally, as recently confirmed the Millennium Ecosystem Assessment. In most populated rural landscapes, objectives for ecosystem services—conservation of biodiversity, protection of watersheds, climate regulation, and others—must be achieved within agricultural landscapes. This will entail modifications in both production and natural areas in landscape mosaics to achieve ecosystem benefits at a landscape scale, and economic and market innovations to ensure financial viability and secure livelihoods of farmers and their communities.

Extending and enriching tree cover can be a critical element in developing such "ecoagriculture" landscapes. Tree cover may be planted or managed in diverse configurations, including linear features along fields and streams, small and large patches of planted or natural or mixed forest, mixed intercropping of trees in lower densities within cropland or pastures. In ecosystems that are naturally in forest or woodland, agroforestry systems can be designed to mimic the structure and function of the natural ecosystem. By planting mosaics with patches of different tree species, or multi-strata mixtures or crop species, the structure of natural forest habitats can be imitated. Soil disturbance would be minimal compared to conventional crop cultivation or forest harvest, and the canopies, stems and root systems of perennial plants would provide habitat niches for many wild plant and animal species.

Ecosystem services of perennial trees and shrubs (and grasses) include soil and water conservation, control of water flow, soil nutrient cycling, win protection, and production of organic mulch in the formal of leaf litter, which reduces soil erosion. The actual contribution that trees make to these services depends on their temporal and spatial configurations, location relative to key landscape features (such as waterways or remnant natural forests), species choice and mixtures, density and management—all elements of landscape design.

One general approach is to strategically intergrate woody plants into conventional annual-crop based farms, through various agroforestry systems that also provide farm products or income. On sloping land, strips of economically productive perennials can be planted along the contour between strips of erodiable annual crops. Within a few years, natural processes can build up the level of soil behind such strips, forming terraces. Patches of natural vegetation can be encouraged to regenerate to protect waterways and alleviate severe gullying. Improved fallows can be used to regenerate soil fertility and other ecological characteristics, using methods that range from single- to mixed-species to enriched fallows. Multi-strata systems, such as agroforests or complex home gardens,

[1] President, Ecoagriculture Partners Director, Ecosystem Services, Forest Trends sscherr@ecoagriculture.org

can be developed by enrichment planting within logged or degraded forests, by planting perennials in cleared forest land; or by intercropping among existing plants of either upper-or middle-story species. Even monoculture tree plantations, if sited appropriately and managed to minimize agrochemical use and maintain ground cover, can provide far superior biodiversity values than most annual cropping systems.

Researchers have begun to document the biodiversity, watershed and carbon cycle impacts of tree cover in agricultural landscapes; this talk will highlight findings from several recent international reviews. Evidence of current and potential financial benefits of tree-growing for farmers and rural communities will also be reviewed, including for household consumption and production, product markets and payments for ecosystem services. Institutional innovations to promote tree-growing by individual farmers, farmer groups and community organizations will be noted.

Promoting tree-growing for ecoagriculture landscapes will require support for germplasm development, technical assistance for farmers and training for service providers and conservationists, monitoring of conservation and production, development of markets for tree products and support for enterprise development. New institutional mechanisms will often be needed to coordinate initiatives by farmers, businesses and conservation agencies for impact at a landscape scale.

Opportunities for conserving biodiversity within agricultural landscapes in Central America: lessons from the FRAGMENT project

Celia A. Harvey[1], Fergus L. Sinclair[2], Joel Sáenz[3], Muhammad Ibrahim[1], Cristóbal Villanueva[1], René Gómez[4], Marlon López[1], Jorge Montero[3], Arnulfo Medina[5], Dalia Sánchez[1], Sergio Vílchez[5] and Blas Hernández[5]

Introduction

In Central America, where large areas of land have been deforested and converted to cattle production, most landscapes now consist of mosaics of small forest patches interspersed within a matrix of pastures and crop fields (Harvey *et al.* 2005). Although agricultural landscapes are often seen as biological wastelands, they usually retain a conspicuous and abundant tree cover in the form of small forest patches, riparian areas, live fences and dispersed trees in fields (Harvey *et al.* 2004). This tree cover may play important roles in maintaining both local and regional biodiversity by serving as important habitat and resources for both plant and animal species, and by maintaining a certain degree of landscape connectivity. At the same time, this on-farm tree cover plays important roles in farm productivity, providing products and services to farmers. In order to manage tree cover in agricultural landscapes for both conservation and production, it is important to understand the existing patterns of on-farm tree cover within agricultural landscapes, their roles in maintaining farm productivity, and their importance for biodiversity conservation. It is also is critical to understand how farmers make decisions about on-farm tree cover, as these decisions determine the structure and composition of tree cover in agricultural landscapes, which in turn, influence their value for the conservation of biodiversity.

In this talk, I present an overview of the main results of research project 'Assessing the impacts of trees on farm productivity and biodiversity conservation in fragmented landscapes' (FRAGMENT project), in which we characterized on-farm tree cover, documented its value for farm production, explored farmer local knowledge and decision-making about tree cover, and assessed the role of on-farm tree cover for biodiversity conservation in 4 contrasting landscapes in Costa Rica and Nicaragua. On the basis of this information, I highlight the importance of on-farm tree cover, both for farm production and biodiversity conservation, and identify key opportunities for conserving biodiversity within agricultural landscapes in the region.

Methods

The research was conducted as part of the FRAGMENT project ("Developing methods and models for assessing the impacts of trees on farm productivity and regional biodiversity in fragmented landscapes"; INCO-Dev, ICA4-CT-2001-10099) whose overall goal was to assess the importance of tree cover in sustaining farm productivity and conserving regional biodiversity, and to develop innovative decision support tools for sustainable

[1] Department of Agriculture and Agroforestry, Centro Agronomico Tropical de Investigación y Enseñanza (CATIE), Turrialba, Costa Rica
[2] School of Agricultural and Forest Sciences, University of Wales, Bangor, UK
[3] Programa Regional de Manejo de Vida Silvestre, Universidad Nacional Autónoma, Heredia, Costa Rica
[4] Nitlapan, UCA, Managua, Nicaragua
[5] Fundacion Nicaraguense para la Conservacion, Managua, Nicaragua

landscape management. The FRAGMENT project was conducted in four landscapes - two in Costa Rica (Cañas and Río Frío) and two in Nicaragua (Rivas and Matiguás)– in which cattle production is the main economic activity and pastures dominate the landscape. These landscapes were chosen to represent the main types of cattle production landscapes present in Central America, with the Cañas and Rivas sites representing the cattle regions in the Pacific slope (Tropical Dry Forest), Río Frío representing the Atlantic slope (Tropical Wet Forest), and Matiguás being typical of the central highland cattle zones (Tropical Humid Forest). In each of these landscapes, an area of between 10,000 and 16,000 ha was selected for study. All field research activities were conducted within this area from Feb 2001 to June 2004.

In order to explore the patterns of tree cover, the importance of this tree cover for farm productivity, and the roles of this tree cover for biodiversity conservation, we conducted a series of interrelated studies. These included: 1) a general characterization of the structure and composition of each landscape, using aerial photographs and remote sensing images; 2) the characterization of on-farm tree cover (focused on isolated trees and live fences) using vegetation surveys; 3) the identification of factors affecting farmer decisions about tree cover by surveying farmers about tree use, and monitoring farms and tree use during a one-year period; and 4) the study of biodiversity (tree, birds, bats, dung beetles, butterflies and small mammals) in different types of tree cover present within the agricultural landscape. Details on the methodologies for the characterization of on-farm tree cover are found in Cárdenas (2002), Villacís *et al.* (2003), Chacón (2003), and López *et al.* (2004a). Methods for farmer decision-making and the recompilation of farmer knowledge are detailed in López *et al.* (2004b), Muñoz *et al.* (2003), Muñoz (2003), and Martínez (2003), whereas methods for biodiversity inventories are found in Cárdenas *et al.* (2003), Lang *et al.* (2003), Montero (2003) and Harvey *et al.* (in review).

Key results

Patterns of on-farm tree cover

In the four landscapes studied, pastures dominated the landscape accounting for between 48 and 68% of the total area; in contrast the remaining forest cover accounted for <20% and generally occurred as narrow small, isolated secondary forest patches or strips of riparian forests (Villacís *et al.* 2003, Villanueva *et al.* 2003, López *et al.* 2004; Harvey *et al.* 2005). All four landscapes retain a highly heterogeneous tree cover within pastures consisting of isolated trees and live fences. Isolated trees in pastures were common in all four sites, but occurred at low densities (ranging from a mean of 7 to 33 trees ha^{-1}). Most of these trees stem from natural regeneration or are relicts of the original forest; few are actively planted by farmers. At the landscape level the total species richness can be considerable (with between 72 and 106 tree species recorded as dispersed trees in each site), however individual farms generally have <35 tree species. The dispersed trees are dominated by only a handful of species, with the ten most abundant species accounting for >70% of all trees present in the pastures in each landscape. These common species tend to be timber species, firewood or forage trees that farmers have actively chosen to retain due to their utility such as *Guazuma ulmifolia* (a forage species), *Tabebuia* spp. (timber species), and *Cordia alliodora* (a timber species).

Live fences were also common elements in three landscapes, occurring on >87% of the farms with a mean density of 0.14 km ha^{-1} (Harvey *et al.* 2005b). Live fences generally consisted of a single row of trees, spaced 1 to 3 m apart, and located along the borders of pastures or farms that are planted directly by farmers to prevent animal movement.

Generally individual live fences have a low species richness (typically 1-7 tree species per live fence), but at the landscape level this overall species richness is higher due to the inclusion of the occasional remnant tree within the live fence. However, like the dispersed tree component, a handful of tree species dominate the live fences, mainly species that grow quickly, are easy to establish with stakes and are capable of resprouting following pollarding. In three of the four study sites, one species dominated the live fences in each area (*Bursera simaruba* in Cañas and Matiguás, *and Erythrina costaricensis* in Río Frío).

Although live fences cover a small physical area of the landscape, they have a disproportionate effect on the structure, composition and connectivity of landscapes, as they increase total tree cover, divide pastures into smaller areas, create rectilinear networks that cross the landscape and provide direct physical connections to forest patches (thereby potentially facilitating the movement of some animal species; Chacón and Harvey, in review).

Farmer management of on-farm tree cover

The existing patterns of on-farm tree cover reflect farmer decisions to plant, retain or remove trees on their farms (Harvey et al. in press). Farmers value on-farm tree cover for the products and services they provide. Interviews with farmers at each location showed that the majority of farmers maintained on-farm tree cover predominantly to serve as a source of timber for home construction, as well as a source of fence posts and fruits and forage for cattle production (Muñoz *et al.* 2003; Villacís 2003). Most farmers also valued the provision of shade by trees to cattle, particularly in the dry seasons when animal heat stress occurs. In contrast, few farmers mentioned or valued the role of trees in preventing soil erosion, providing organic inputs to soils, or providing habitat for wildlife.

However, at the same time that farmers were aware of the benefits of having on-farm tree cover, they also were aware that in pastures with high tree cover, pasture production is reduced. Farmers mentioned that forest patches, isolated trees and other forms of on-farm tree cover occupy space that could otherwise be used for crop or cattle production. Therefore, most farmers mentioned that they balanced their needs for forest products with their desire to maximize agricultural production by maintaining small forest patches, narrow strips of forest along riparian areas and low tree densities in pastures. According to farmers, this strategy ensures that sufficient products are provided for farm use, with minimal competition with agricultural production.

In all of the four study sites, farmers conduct a number of activities that affect on-farm tree cover. Activities that reduce on-farm tree cover include the weeding of pastures (and elimination of naturally regenerating saplings within the pastures), the pollarding of live fences, the harvesting of trees for timber, firewood and fence posts, and the removal of forest fallows or secondary forest to establish pastures or crop fields, whereas activities that increase farm tree cover include the planting of live fences and the establishment of fallow areas on previously cultivated land.The net impact of these activities on tree cover (and biodiversity conservation) depends on the frequency and intensity of these changes at the landscape level, as well as how these activities change tree diversity, density, and spatial arrangement. For example, the pollarding of live fences by farmers is likely to reduce the value of these elements for biodiversity conservation, as it reduces the availability of resources and habitat for wildlife, particularly for birds (Lang *et al.* 2003).

Importance of tree cover for biodiversity conservation

The agricultural landscapes studied appear to conserve a high proportion of the original biodiversity, despite being highly fragmented and deforested, and heavily impacted by cattle production. In each landscape, a total of 83-195 bird species., 48-64 butterfly species, 24-47 bat species and 32-37 dung beetle species were registered. This diversity represents a significant proportion of the original biodiversity. A comparison of the species richness in the agricultural landscape in Cañas with that of the National Park Santa Rosa, for example, showed that Cañas had 91% of the bat species, 52.5 % of the bird species, 74 % of the dung beetle species and 48% of the butterfly species richness present in the National Park.

Although these agricultural landscapes appear to harbour relatively high species richness, there are notable differences in the species composition between the communities in agricultural landscapes and forested landscapes in some of the taxa. In the agricultural landscapes, most of the organisms captured were generalist species, typical of open areas and most highly forest-dependent species were present only in low abundances. In addition, certain types of species that are highly vulnerable to fragmentation efforts- such as species that require intact forest understory or species that are vulnerable to hunting- were generally absent from the landscape. These results suggest that the transformation and modification of the landscapes has likely changed the composition and structure of animal communities, although it is difficult to know much these landscapes have changed due to the lack of historical or comparative data.

There were significant differences in species richness, abundance, diversity and composition of different organisms across habitats, but these differences varied both across taxa and across landscapes- as not all organisms respond similarly to the same landscape and not all landscapes are equal. However, a couple of key generalizations emerged from the comparison of species richness, abundance, and diversity across the different habitats. First, the high species richness within agricultural landscapes is due to the combination of the species richness within in individual habitats, rather than a particular habitat per se. No single habitat contains all of the species present in the landscape; instead the overall species richness of the landscape is due to the additive effect of species richness across different habitats.

Second, the patterns of species richness, abundance and diversity across the different habitats were variable both across taxa and across sites. Tree species richness was significantly different habitats in all four landscapes, with a general pattern of higher species richness in riparian forests and secondary forests, than the other habitats. Bird species richness, in contrast, showed no differences among habitats in two of the landscapes studied, whereas in the other two landscapes, there was a pattern of higher species richness in riparian forests, secondary forests, charrals and pastures with high tree cover than in the pastures with low tree cover and live fences. Bat species richness showed differences across all habitats in all 4 sites, with riparian forests (and sometimes live fences) hosting the greatest bat species richness. Dung beetle species richness and butterfly species richness, on the other hand, did not show any consistent differences across habitat types within the different landscapes.

The fact that the patterns of species richness and diversity vary across habitats and landscapes in different ways for different taxa means that conservation recommendations will need to be tailored to particular taxa and particular landscapes. However, the forested habitats –riparian forests, secondary forests and charrals- tended to have higher conservation value than the less forested habitats (live fences, pastures with high tree cover and pastures with low tree cover) for most taxa, due to the greater presence of forest species within these habitats and higher species richness of some taxa (e.g. birds, bats). Another important finding is that for many groups the presence of high tree densities within pastures appears to have a positive effect on species richness and diversity, compared to low tree densities.

Policy recommendations for conserving biodiversity in fragmented landscapes

Our results demonstrate that agricultural landscape that contain a diverse on-farm tree cover- including small forest patches, narrow riparian strips and dispersed trees and live fences in pastures- may support a high species richness and thereby play important roles in both local and regional conservation efforts. Riparian forests, secondary forests and charrals are the habitats of highest conservation value due to the fact that these habitats still harbour species from the original forest and maintain the highest tree, bat and bird species richness. Pastures with high levels of tree cover appear to hold greater biodiversity value than those with low tree cover, whereas live fences appear to play important roles in facilitating species movement (as witnessed by the high abundance of bats and birds caught in this habitat) and to lesser degree serving as habitat and resources.

Governments, land managers and policy makers need to explicitly recognize the value of the on-farm tree cover within agricultural landscapes for the conservation of biodiversity and actively promote the conservation of this tree cover for conservation purposes. Specific steps should be taken to ensure the adequate conservation and protection of existing forest patches and riparian forests, and to prevent their degradation by cattle entry and illegal logging. In addition, programs should be implemented to increase and diversify tree cover within pastures, with the goal of converting pastures with low tree cover to pastures with 15-25% tree cover (levels which are compatible with cattle production). Similarly, the establishment of live fences and the diversification of live fences with multiple strata and species that have value for wildlife species should be encouraged. Environmental payments, taxation schemes or other incentives should be explored as possible mechanisms for promoting these changes within cattle farms. Finally, government agencies should establish careful monitoring programs to determine the dynamics of plant and animal populations over time within these landscapes and when necessary, adapt land use practices to ensure biodiversity is conserved.

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Opportunities for carbon sequestration and conservation of water resources on landscapes dominated by cattle production in Central America

Muhammad Ibrahim[1], Mario Chacón[1], Jairo Mora[1], Sheyla Zamora[1], Jose Gobbi[1], Tangaxuan Llanderal [1], Celia A. Harvey[1] , Enrique Murgueitio[2], Francisco Casasola[1], Cristobal Villanueva[1] and Elias Ramirez[3]

Key words: environmental services, pastures, stable carbon, runoff, silvopastoral systems, water quality

Introduction

Conversion of native vegetation to different forms of land use has large implications for the energy, water and carbon exchange processes between soil, surface and atmosphere at local and regional levels. In Central America agriculture and livestock has had lead deforestation process through the expansion of the agricultural frontier at expense of natural habitat (Harvey et al. 1005a), in the region more than nine million ha of primary forest have been deforested for expansion of pasture and more than half of this area is degraded (Szott et al. 2000). Pasture degradation leads to a decline of the natural resource base (e.g., decreased biodiversity, soil and water quality), more rapid run off and hence higher peak flows and sedimentation of rivers, lower productivity, increased rural poverty and vulnerability and further land use pressure.

Promising options for reducing pasture and environmental degradation, and hence maintaining or increasing productivity, include: improvement of pasture and cattle management; more diverse silvopastoral or agrosilvopastoral systems, which include cattle together with other productive components; and systems based on perennial crops, or secondary or plantation forest management in which the importance of cattle is greatly reduced or gradually eliminated. The implementation of silvopastoral systems on cattle farms have resulted in significant improvements in animal productivity (> 30%) and generation of environmental services on landscapes dominated with cattle, but the lack of capital for investing in silvopastoral systems represents a major barrier for the adoption of these systems by cattle farmers in Central America (Ibrahim et al. 2003; Chagoya 2004).

The linkage of production activities with the marketing of environmental services could constitute a route to reconvert traditional cattle systems towards eco-friendly systems which integrate silvopastoral systems and this could represent one of the best strategies for poverty alleviation, ecological restoration, carbon sequestration, and conservation of water and biodiversity resources. This linkage allows the farmer to have the option of continuing to produce food, raw materials and services and at the same time provide benefits for society and global environment. This paper presents results on carbon sequestration in pasture, silvopastoral and forest systems that can be used for designing payment schemes for compensating farmers for carbon sequestration. Results are also presented on how these land use changes affect water storage and quality in cattle production systems.

[1] Livestock and environmental management group (GAMMA), CATIE, Costa Rica.
[2] Director of CIPAV and national coordinator of the GEF-SSP, Colombia
[3] NITLAPAN and National coordinator of GEF-SSP project, Nicaragua

Over the last four years CATIE has been implementing a GEF-Funded project "Integrated Silvopastoral Approaches for the Management of Ecosystems" in collaboration with CIPAV in Colombia, NITLAPAN in Nicaragua and the World Bank. This project has developed a novel approach for compensating livestock farmers for environmental services, and results are presented of how farmers make decisions on land use changes with payment for environmental services (PES). PES is based on the use of a land use index as a tool for monitoring land use changes, and this tool considers the potential of different land uses for C- sequestration and conservation of biodiversity, and a detailed description of the use of this tool is by presented by Gobbi et al (2005). The impacts of land use changes on biodiversity resources are considered in the paper by Harvey et al. (2005b).

Nutrient cycling and soil fertility

Depletion of soil quality and erosion are among the major reasons for pasture degradation. The establishment of grass-legume mixtures and multi-purpose trees has been shown to enhance nutrient cycling and hence increased productivity of pastures (Belsky et al. 1989). In the humid tropics legumes can fix more than 100 kg N ha yr^1 that is an important source for sustaining high productivity of the companion grass (Ibrahim and Mannetje, 1998). Experiences show that the integration of multipurpose trees and shrubs (e.g., *Erythrina* spp, *Gliricidia sepium*, *Acacia mangium*) in pastures established in humid and sub-humid zones resulted in significant improvements in soil organic matter, N and P (Bolivar et al. 1999). Shade tolerant grasses (*Panicum maximum* and *B. brizantha*) and legumes (e.g., *Arachis pintoi*) have been selected for these systems (Zelada and Ibrahim, 1997; Bustamante et al. 1998). Studies in the humid tropics showed that most improved grasses produced significantly higher DM yields (12 to 50%) in association with *Erythrina poeppigiana*, and under sub-humid conditions relatively high pasture yields can be maintained in traditional silvopastoral systems with diverse tree species (Esquivel unpublished data).

Water quantity and quality

The potential of AFS and silvopastoral systems to help secure water supplies (quantity and quality) is the least studied environmental service (Beer et al. 2003). Nepstad et al. (1994) found in Para, Brazil, that during severe dry season, plant available soil water from 2-8 m depth declined by 380 mm in the forest and 310 mm in the degraded pasture. Average daily rainfall was 0.6 mm and evapotranspiration was 3.6 mm for forest and 3.0 mm for degraded pasture. Less depletion of plant available soil water in the degraded pasture signifies that this ecosystem can store less rainfall than forest and may therefore produce more seepage to the ground water aquifer or sub-surface runoff to streams in the wet season. At the end of the dry season the forest could store an additional 770 mm of water in the upper 8 m of soil compared to 400 mm in the pasture which means that on landscapes with dominance of pastures water shortage may become a critical issue especially in semi-arid and arid regions. Recent studies conducted by the GEF-funded silvopastoral project showed that runoff of water was significantly higher in degraded pastures (42%) compared to fodder banks with woody perennials (3%), pastures with high tree densities (12%) and young secondary forest (6%). This means that land use changes with higher tree cover may be beneficial for water harvest depending on water use of tree species (Rios, unpublished data). The implementation of riparian forest on cattle farms and protection of water sources from cattle resulted in improvements in biological and chemical propertied of water quality (Chará 2004). The project is currently designing a system for PES for water resources in Esparza of Costa Rica and in Río-Blanco of Nicaragua. Preliminary results show that the community of Esparza is prepared to pay an ecological fee (10- 15%) above current water rates which will be used for creating a fund for payment for environmental services (Alpizar unpublished data).

Carbon sequestration and greenhouse gas effects

The interest of managing pastures and silvopastoral systems for C-sequestration has increased over the last years though there have been mixed results on the potential of tropical pastures to accumulate soil organic carbon (SOC). Neil *et al.* (1997) reported that eleven of fourteen pasture conversion sites studied in Brazil showed increase in soil carbon. All sites of pasture for at least 10 yrs showed increases with rates as high as 74.0g C m^2 yr^1 over 20 years. Veldkamp (1994) found a net loss of 2-18% of carbon stocks in the top 50 cm of forest equivalent soil after 25 years under pasture in lowland Costa Rica. The quality of management of tropical pastures is critical to the conclusions drawn about whether the soils under this land use represent a source or a sink of atmospheric carbon. In well-managed pastures in formerly forested areas, significant amounts of litter (roots and leaf litter) are recycled in the system that results in accumulation of SOC (Neil et al. 1997). However the root systems of grasses are generally concentrated in the upper soil layers (0- 35 cm depth) and there is little soil derived C of grasses in the deeper soil layers. Tarre *et al.* (2001) in the state of Bahia of Brazil showed that the establishment of *Brachiaria* pastures in deforested areas resulted in an accumulation of SOC (13.9 t ha^{-1}) over time. However samples taken to depth of 100 cm showed that below 40 cm depth there were no significant contributions of the *Brachiaria* derived C, and the integration of trees in pastures may increase the amount of C-stored at greater depths.

Highly productive AFS, including silvopastoral systems, can play an important role in C sequestration in soils and in the woody biomass (Beer et al. 2003). Well-managed silvopastoral systems can improve overall productivity (Bustamante et al. 1998; Bolivar et al. 1999), while sequestering C (López et al. 1999, Andrade 1999), a potential additional economic benefit for livestock farmers. Total C in silvopastoral systems varied between 68– 204 t ha^{-1}, with most C stored in the soil, while annual C increments varied between 1.8 to 5.2 t ha^{-1} (Table 1).

The amount of C fixed in silvopastoral systems is affected by the tree/shrub species, density and spatial distribution of trees, and shade tolerance of herbaceous species (Nyberg and Hogberg 1995; Jackson and Ash 1998). On the slopes of the Ecuadoran Andes, total soil C increased from 7.9% under open *Setaria sphacelata* pasture to 11.4% beneath the canopies of *Inga* sp but no differences were observed under *Psidium guajava*. Soils under *Inga* contained an additional 20 t C ha^{-1} in the upper 15 cm compared to open pasture (Rhoades et al. 1998). Measurements on carbon stocks in landscapes in the sub-humid tropics of Matiguas, Nicaragua and Esparza in Costa Rica showed the total amount of C stored (soil and above ground tree component) in secondary forest and silvopastoral systems were significantly higher than that of degraded pastures. In Esparza the mean total amount of stored carbon in pastures with high tree cover (25–35 adult trees/ha) is 132 t C ha^{-1} compared to 29.5 t C ha^{-1} for degraded pastures; a mean of 6.0% of total carbon is stored in the above ground tree cover of pastures with high tree cover. The pastures of Matiguas were characterised by higher tree densities than those of Esparza and hence higher amounts of C stored in the tree component (mean 11.8 vs. 7.5 t C ha^{-1}). In both sites more than 50% of carbon stored in the above ground tree cover is found in small diameter class of trees (5-15 cm) and a higher percentage is found in large diameter (> 30 cm) class in Matiguas compared to Esparza. Stable C-stocks measured on soils under silvopastoral systems (7.8 t C ha^{-1}) in the Quindio, Colombia, was higher than those measured under riparian forest (5.05 t C ha^{-1}) and degraded pastures (5.27 t C ha^{-1}) and this difference may be associated to higher inputs of recalcitrant carbon in the silvopastoral system which is important for permanent stocks of carbon (Ibrahim et al. in preparation).

In terms of greenhouse gases, the use of leguminous based pasture systems can offset the use of N fertilizers for sustaining pasture yields which contributes to a reduction in the emissions of N_2O and feeding better quality forages is involved in a reduction of CH_4 during rumen fermentation. Dairy farms that had a higher tree cover and use less inputs (e.g., concentrates, and N fertilizers) had greater C-budgets (e.g., less emissions of green house gases) compared to those farms that had lower tree cover and use more inputs (Mora 2001).

Impacts of PES on land use changes and C-sequestration

The results of the GEF-Funded silvopastoral project showed that PES/ farm increased significantly over time, and small cattle farms (< 25 ha) in Costa Rica received higher PES/ha compared to large (> 50 ha) farms (25.9 vs. 22.3 US/ha; PES in 2005) though the latter had higher PES/farm. These results demonstrate that small farmers can play an important role in the generation of environmental services and payment schemes can be designed to improve their livelihoods. As mentioned above the project developed a land use index as a tool for monitoring environmental services and we expressed the data as ecological points (EP) for evaluation of impacts of PES. A regression analysis showed that there was a negative linear relationship between incremental ecological points/ha (Y) and that of the base line/ha of cattle farms (X) and this means that those farmers who had good farming practices (baseline) did not benefit from the current payment scheme which is based on incremental EP and may be a disincentive for farmers to maintain good farming practices (Figure 1).

Over the three years of PES there was a significant increase in the percentage area of pastures with high tree densities (> 15%) and live fences (> 35%) on landscapes indicating that farmers made decisions to adopt land use practices that permit them to benefit from PES without suffering heavy losses in cattle production. The area under fodder banks and intensive silvopastoral systems also increased significantly and this was more striking for Colombian cattle farms that had in some cases more than 15 ha of intensive silvopastoral systems. There was a positive linear relationship between the area under forest (e.g., secondary succession, forest plantations; Y) and the area under improved pastures (Y) in livestock farms in Costa Rica ($Y = 4.61 + 0.36\ X$; $R^2 = 0.41$) and this may be associated to higher carrying capacities of improved compared to native pastures which permits farmers to liberate marginal lands for re-afforestation and environmental benefits. The project has been monitoring what are the impacts of land use changes on income of livestock farms and social indicators (e.g., labor use) to determine if PES is linked with improvement of well being of rural poor. The indicators showed that those farming practices which generated higher environmental services (e.g., secondary forest) had a lower demand for labor compared to management of intensive silvopastoral systems (e.g., fodder bank) (Ibrahim et al. in preparation).

The data on land use-changes were use to model to impacts of PES on C-sequestration using the CO_2-land model and the results showed that the amount of C-sequestered increased (> 20%) under PES compared to baseline data. It is important to note that the decisions made by farmers to increase the area of live fences (multi-strata) and of pastures of high tree cover is of much significance for the conservation of biodiversity in agricultural landscapes (Harvey et al 2005b).

Conclusions

It is concluded that silvopastoral systems in C-sequestration in agricultural landscapes while enhancing productivity of livestock farms and improvement of livelihoods of farmers and rural poor. There is need for more in-depth research to ascertain the role of silvopastoral systems in conservation of water though there are some results which show that these systems can enhance water harvest. The payment of environmental services in the adoption of good farming practices which include pastures with high tree cover and live fences that are of much significance for sustainability of livestock farms and conservation of biodiversity.

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Table 1. Carbon storage and carbon fixation in some silvopastoral and pasture systems.

Zone System (age in years)	Soil organic carbon[1] ($t\ ha^{-1}$)	Above-ground carbon[2] ($t\ ha^{-1}$)	Total carbon ($t\ ha^{-1}$)	Carbon fixation[3] ($t\ ha^{-1}\ yr^{-1}$)	Reference
Humid lowlands, Northern Region, Costa Rica					López et al. (1999)
Panicum maximum monoculture	233 ± 8		233		
P. maximum – Cordia alliodora (≤ 3)	177 ± 8	2.3	179		
P. maximum – C. alliodora (3-7)	196 ± 21	8.8	205		
P. maximum – C. alliodora (≥ 7)	175 ± 23	26.8	202		
Lower montane ecosystems, Ecuadorian Andes					Rhoades et al. 1998)
Setaria sphacelata pasture	69				
S. sphacelata – Inga sp.	87				
S. sphacelata – Psidium guajava	74				
Humid lowlands, Atlantic Zone, Costa Rica					Andrade (1999)
Brachiaria brizantha – Eucalyptus deglupta (2)		3.7		1.8	
B. decumbens – E. deglupta (2)		3.8		1.9	
P. maximum – E. deglupta (2)		4.7		2.3	
B. brizantha – Acacia mangium (2)		3.9		1.9	
B. decumbens – A. mangium (2)		3.9		1.9	
P. maximum – A. mangium (2)		4.2		2.1	
Humid lowlands, Atlantic Zone, Costa Rica					Avila (2000)
B. brizantha – A mangium (3)	87 ± 18	8.90 ± 0.03	96	2.2	
B. brizantha – E. deglupta (3)	87 ± 1	7.48 ± 0.26	95	1.8	
B. brizantha monoculture	66 ± 16	2.04 ± 0.16	68		
Ischaemum indicum monoculture	84 ± 11	0.12 ± 0.03	84		
Highlands, Volcanic Cordillera, Costa Rica					Mora (2001)
Pennisetum clandestinum monoculture	494 ± 35			5.16 ± 0.30	
P. clandestinum and trees	573 ± 30			5.14 ± 0.25	
Cynodon nlemfuensis monoculture	756 ± 54			4.79 ± 0.18	
C. nlemfuensis and trees	624 ± 65			4.91 ± 0.04	
Jhansi, India					Rai et al. (2001)
Mixed pasture[4]	0.47				
Mixed pasture – *Acacia nilotica* var. *cupressiformis*	0.67 ± 0.04				
Mixed pasture – *Dalbergia sissoo*	0.71 ± 0.04				
Mixed pasture – *Hardwickia binata*	0.71 ± 0.05				
Highlands, Volcanic Cordillera, Costa Rica					Villanueva (2001)
P. clandestinum monoculture	185 ± 32		185		
P. clandestinum – Alnus acuminata (2)	187 ± 46	1.1 ± 0.6	188		
P. clandestinum – A. acuminata (3)	196 ± 25	4.2 ± 1.7	200		
P. clandestinum – A. acuminata (4)	197 ± 9	6.2 ± 0.8	203		
Seasonally dry hillsides, Central Nicaragua					Ruiz (2002)
Naturalised grass monoculture	150 ± 15	1.4 ± 0.2	151 ± 16		
Naturalised grasses and trees	155 ± 13	9 ± 2.7	164 ± 14		
Improved grass monoculture	158 ± 15	1.6 ± 0.2	159 ± 16		
Improved grasses and trees	155 ± 15	15 ± 3.0	170 ± 16		

[1] Soil organic carbon values correspond to the following soil depths (cm): 0-50 (López et al 1999), 0-15 (Rhoades et al. 1998), 0-30 (Avila 2000), 0-100 (Mora 2001), 0-60 (Villanueva 2001) and 0-80 (Ruiz 2002).

[2] Above-ground carbon values were estimated from carbon stored in trees only (López et al 1999; Villanueva 2001) or in trees and pasture (Andrade 1999; Avila 2000; Ruiz 2002).

[3] Carbon fixation values correspond to carbon fixed in tree biomass (Avila 2000) and in soils (Mora 2001).

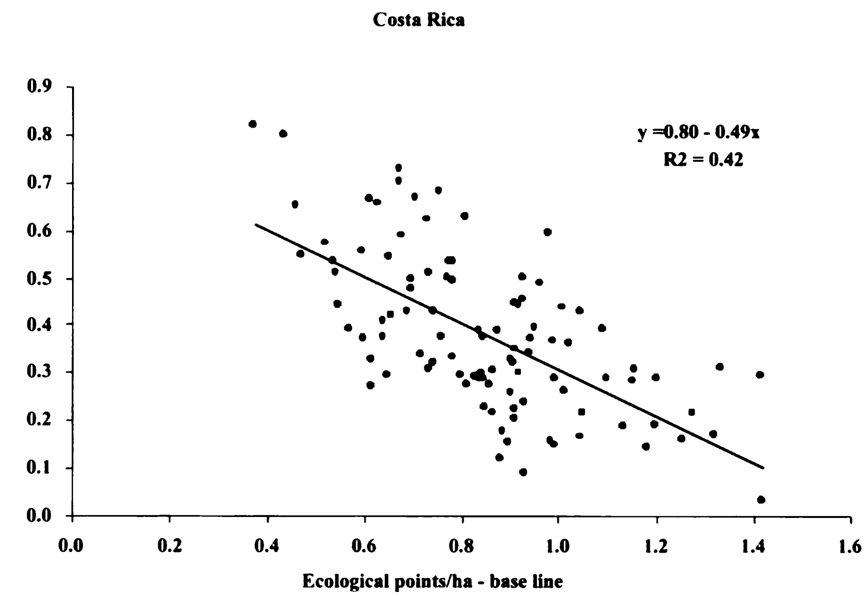

Figure 1. Relationship between ecological index (points ha^{-1} 2003, base line) and incremental ecological index (points ha^{-1}, difference between 2003 and 2005).

Environmental services of coffee agroforestry systems in Central America: a promising potential to improve the livelihoods of coffee farmers' communities

Philippe Vaast[1], John Beer[2], Celia Harvey[2], J M Harmand[3]

Key words: biodiversity, carbon sequestration, coffee, soil conservation, niche markets, sustainable systems, water quality.

Introduction

Coffee plantations have large-scale environmental impacts as they cover nearly 1,000,000 hectares of the Central American isthmus. In this region, coffee has been the main agricultural crop and source of export earnings over the last 100 years; indeed, the cultural and social patterns of Central America have developed on the basis of coffee production and export. Currently, coffee production sustains approximately 300 000 farmers in Central America and provides 1 700 000 permanent and seasonal jobs (Castro et al., 2004).

These coffee plantations are often situated in fragile mountainous ecosystems, mostly in the mid-upper watersheds that supply water to urban centers. Furthermore, this coffee region is one of the world hotspots for biodiversity as identified by Conservation International (www.conservation.org), and coffee agro-forests with shade trees interspersed amongst the coffee plants are often the only habitat with remaining tree cover within these areas, as almost all forests have been removed at the altitudes where coffee is grown. Therefore, coffee agro-forests are key habitats for many migratory birds in the Meso-American biological corridor, thus the promotion of diverse, multi strata agroforestry systems benefits national as well as international conservation efforts. Furthermore, trees in coffee plantations can provide continuity of tree cover and alternative sources of forest products in partially forested landscapes. In Central America, rural and often urban households relies on wood fuels derived from trees in coffee agro-forests for energy use, hence reducing pressure on surrounding natural forests.

Over the last 40 years, intensification of coffee cultivation in Central America has led to the loss of more than 50% of the tree cover on coffee farms, with native species of shade trees being the most severely reduced. This coffee intensification has resulted in progressive fragmentation of forest habitat, loss of landscape connectivity, increased pollution of rivers and aquifers by agro-chemicals and extensive loss of biodiversity in many coffee producing zones.

An increased awareness of the negative impacts of intensive coffee monoculture on the environment has contributed to the emergence of new markets for environmentally-friendly coffee in consuming countries. Locally, this has stimulated a shift in coffee management towards more environmentally-friendly practices that needs to be accompanied by revenue diversification (high quality coffee and wood products) and

[1] CIRAD-CATIE, Department of Agriculture and Agroforestry, Turrialba, Costa Rica.
[2] CATIE, Department of Agriculture and Agroforestry, Turrialba, Costa Rica.
[3] CIRAD, Departement Forêt, Montpellier, France.

new marketing opportunities to improve economic sustainability of local communities and reduce their dependence to coffee price volatility. Despite the multiplication of certified sustainable coffee seals (organic Coffee, Fair Trade Coffee, Organic and Fair Trade, Rainforest Alliance, Bird friendly, Starbucks Coffee-practices), these niche markets only represents 4% of the coffee exports of Central America in 2003 (Castro et al., 2004). As coffee agro-forests provide a great array of environmental services, they need to be recognized through local, national and international incentives to enhance rural livelihoods.

This article presents a brief review of the main environmental services that are provided by coffee agroforestry systems (AFS): 1) maintenance of soil fertility/reducing erosion through organic matter inputs to the soil and nitrogen (N) fixation and nutrient recycling by associate trees; 2) conservation of water (quantity and quality) through reduced runoff and greater infiltration; 3) reduced contamination of water courses; 4) carbon sequestration; and 5) biodiversity conservation in fragmented landscapes.

It also addresses the issues regarding: 1) indirect payment for environmental services of coffee AFS via sustainable coffee seals and niche markets; and 2) the need to develop local, national and international schemes to financially reward coffee farmers' communities and hence to promote a wider adoption of environmentally and biodiversity-friendly coffee cultivation practices.

Soil Fertility

Soil fertility improvement in coffee AFS is due to the reduction of soil erosion and inputs by the associated shade trees. Regarding soil erosion, recent studies indicate that shade trees can decrease runoff by more than half on coffee slopes (CASCA, unpublished data) via natural litter fall or pruning residues that cover the soil, reduced impact of raindrops, improved soil structure and enhanced infiltration (Beer et al. 1998; Snoeck and Vaast, 2004). Presence of deep-rooted trees and shrubs recycle plant nutrients from depth, build up soil organic matter and hence increase soil cationic exchangeable capacity and availability of nutrients in the top soil (Beer et al., 1998). Leguminous trees increase availability of N, the most limiting nutrient for coffee production, through biological fixation and enhanced N mineralization. In Central America, studies show that N fixation by leguminous shade trees can account to an N input to the coffee system up to 100 kg N ha^{-1} $year^{1}$ but is strongly reduced at high N fertilisation regimes (CASCA, unpublished data). N mineralization increases from 111 kg ha^{-1} $year^{1}$ in a full sun coffee monoculture to 145 kg N ha^{-1} $year^{1}$ in coffee AFS with the legume species *Erythrina poeppigiana* (Babbar and Zak, 1994).

Water quantity and quality

The potential of AFS to help secure water supplies (quantity and quality) is the least studied environmental service (Beer et al., 2003). According to recent studies by the CASCA project, the trees in AFS can greatly influence water cycling by increasing rain interception by 15-25% and up to 100% for low rainfall (<5 mm day^{-1}), transpiration by the coffee and shade trees, enhanced retention of water in the soil, reduced runoff, but increased infiltration. Reduced runoff results in less soil erosion and surface organic matter and nutrient leaching and hence better soil water recharge and water quality. Coffee AFS can reduce ground water contamination by nitrate and other substances that are harmful to the environment and human health. Over the last five years, CASCA has particularly studies nitrate losses and contamination of ground water and confirmed that nitrate leaching is higher from unshaded coffee plantations than from those containing shade trees in areas where high coffee yields have been achieved through large additions of N from chemical fertilizers (Harmand et al. 2004), due in a large part by higher rates of transpiration in the AFS (van Kanten and Vaast, 2005). In Costa Rica, legislation recognizes the benefits of AFS in terms of water. Several pilot projects have been undertaken by the public electrical company (ICE) to financially reward farmers implementing sustainable practices to avoid siltation of electrical dams.

Carbon sequestration and greenhouse gas effects

Within the framework of the Clean Development Mechanisms, also known as the Kyoto accord (in effect since mid-February 2005), carbon sequestration is one of the environmental services that could be rewarded financially when farmers shift from intensive, full sun coffee monoculture to coffee under shade.

For coffee trees, it is estimated that aerial carbon sequestration is in the range of 4-10 t C ha^{-1}. Estimations for shade trees are very variable depending on densities, age and species, and previous land uses as observed in surveys of over 200 coffee farms (CASCA, unpublished data). For example, mean aerial carbon sequestration for *Cordia alliodora* is 39 ± 27 t C ha^{-1} at a mean density of 184 trees ha^{-1} and an average age of 13 years; for *Terminalia amazonia*, this mean value is estimated 32 ± 16 t C ha^{-1} at a mean density of 373 trees ha^{-1} and an average age of 8 years; for *Eucalyptus deglupta*, this mean value is 14 ± 10 t C ha^{-1} at a mean density of 78 trees ha^{-1} and average age of 8 years. In the southern low zone of Costa Rica, a coffee AFS (14 year old) shaded by *Eucalyptus deglupta* (7 years old) increased the C stock in the aerial biomass by 17 t ha^{-1} compared to full sun coffee monoculture. Carbon sequestration in the aerial biomass is not the only compartment to consider as C accumulation in soil can account up to 220 t C ha^{-1}. Coffee agroforestry systems can greatly increase organic matter content of the top soil layer and contribute to an additional 5-15 t C ha^{-1} in the soil with an extra C accumulation in the litter layer of 3-5 t C ha^{-1}.

In terms of greenhouse gas effect, C accumulation in biomass and soil in coffee agroforestry systems can largerly offset N2O emission, whereas N2O emission may account for a major part of the accumulation of C in biomass and litter of full sun coffee"

Biodiversity conservation and climate change mitigation

Over the last 10 years, many studies have demonstrated that coffee AFS have positive impacts on the preservation of biodiversity in Central American and Mexico (Harvey et al. 2005). They have shown that "rustic" coffee systems, planted after clearing of the forest under-story, are better biodiversity-preservers than multi-story coffee systems with two or more shade tree species or coffee with one specialized shade species and, of course, full sun coffee monoculture in terms of floristic and structural diversity as well as faunal composition, species richness and abundance. Recent coffee AFS are mostly composed of fast growing, non-native tree species that contribute greatly to soil and water conservation but little to biodiversity conservation. There is a lack of management strategies, to be developed with farmers and extension agencies, promoting more diverse shade tree strata, embracing native tree species, and thresholds for tree cover that balance environmental services and coffee and shade tree productivity for different site and landscape contexts. Furthermore, most biodiversity studies have been conducted at limited spatial scales (e.g. plot or farm level) and there is a need to undertake studies at landscape level to assess their multiplier effect.

In medium and high lands, coffee agro-forests appear as the ideal land use in buffer zones around parks and protected areas contributing to enhance the protection and management of biodiversity (e.g. Central American biological corridor). Bird and mammal biodiversity (e.g. quetzals in Central America) are recognized as a vital resource for community tourism in the region. Scenic beauty of green, forested slopes and mountains is also vital for eco-tourism. Therefore, there is a need to design and start implementing mechanisms insuring the recognition of the value of biodiversity and its sustainable use through direct financial reward to rural communities and particularly coffee farmers.

Coffee farming communities are situated in fragile mountainous ecosystems vulnerable to climatic disasters (e.g. Hurricane Mitch in 1998 and Stan in 2005). Coffee agro-forests play an important role in decreasing flood landslides as they greatly reduce soil runoff and erosion. Cycles of low coffee prices often result in the elimination of coffee agro-forests (up to 20% during the last crisis in Central America) and the exploitation of new forest land or tree fallows to grow food crops on slopes prone to soil erosion. Exploitation of timber resources is often done without permits, and hence farmers sell standing shade trees at below market price to intermediaries who assume the risks of selling the uncertified timber. Therefore, there is an urgent need to improve the living conditions of coffee farming communities via a diversification of their revenues through legal and more profitable sales of timber and fuel wood and payment of incentives for flood control.

Niche markets

Despite the fact that the sustainable coffee market (organic Coffee, Fair Trade Coffee, Organic and Fair Trade, Rainforest Alliance, Bird friendly, Starbucks Coffee-Practices) is growing at a fast rate of 10-20% (Castro et al., 2004), these niche markets are still representing a low 4% of the coffee exports of Central America.

Due to the increasing complexity of environmentally-friendly guidelines developed by NGOs and the private coffee sector, it appears important to review their scientific basis and relevance in order to propose a harmonization and simplification that are needed to increase their acceptability by farmers' communities. To do so, dialogues have to be stimulated between farmers, cooperative's representatives, municipal and governmental authorities, and NGOs and the private sector involved in the marketing of eco-certified timber and coffee. Particularly, these consultations need to identify coffee farmers' constraints to maintaining and/or adopting environmentally and biodiversity-friendly coffee agro-forestry practices, and fulfill their expectations with respect to governmental support and market schemes that seek to promote these sustainable practices.

Conclusion

Coffee AFS provide an array of environmental benefits (biodiversity conservation, carbon sequestration, reduced pressure on natural forests, soil conservation, water quality and watershed protection) that are becoming more and more recognized and documented. Today, the implementation of environmentally and biodiversity-friendly coffee cultivation practices offer farmers the possibility to improve their revenues through diversification options (i.e. increased revenues from timber and fuel wood), lower production costs (i.e. reduced agrochemical inputs) and compensation options (premium paid for eco-certified products and/or high quality coffee). However, these practices affect farm productivity (e.g. decreasing coffee and other crops productivity with increasing tree cover above a certain threshold). Ultimately, the large-scale adoption of these sustainable practices depends on the economic benefits that they provided to farmers' communities in various socio-economic and ecological contexts.

The payment of environmental services, through local, national and international schemes, can greatly contribute to the wider adoption of these sustainable practices. However, there is a need to develop methodologies to value, directly or indirectly, biodiversity and other environmental services provided by these coffee AFS in order to assess their threshold values for economic profitability according to management systems, different economic scenarios and ecological contexts.

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Landscape mosaics: integrating forest management and environmental services in tropical landscapes

Markku Kanninen[1]

Abstract

To keep forest ecosystems resilient in the face of social and economic pressures and changing climates one must understand how ecological and social systems interact to generate particular land use patterns. Often there will be trade-offs between what is globally optimal and what is locally desirable. For instance, the need of conserving large areas in "hot spot" regions may not be compatible with the livelihood needs of local people living in those regions.

In fragmented landscape mosaics, forests and natural habitats can only be maintained if they can be managed in an integrated manner to generate benefits for local people and generate income trough a combination of products and ecosystem services. In this respect, there are several aspects that forest managers and land-use planner have to take into account. These include, i.e. a) local perceptions of the importance of forests, their products and services, b) the role of forests in managing livelihoods and environmental risks, c) existing local mechanisms for forest and ecosystem management, d) how to integrate environmental services into forest and ecosystem management at multiple scales, e) how to efficiently monitor the services produced, f) mechanisms for rewarding the production of environmental services, g) the role of markets, g) how to develop and manage multi-functions of forests for goods and services that are valued locally and to the wider community.

In the situation described above, the whole "landscape management process" becomes the focus element, rather than the production of individual goods or services as such. This "landscape management process" can be defined as a cycle consisting of various steps, e.g. a) "visioning and assessment" (learning processes geared towards defining management goals), b) "planning" (using existing planning mechanisms if available), c) "incentive assessment" (adapting the planning tools and incentive system), d) "implementation of plans" (adaptive ecosystem management, facilitation of learning processes) and e) "monitoring" (monitoring the progress).

When applying the "landscape management process" in practice we have several methods and tools either already available or that can be easily modified for such a situation. These include adaptive management of forests, multidisciplinary landscape assessment, participatory land-use mapping, and tools for developing future management scenarios. In other cases, e.g. with monitoring of the environmental services or assessment of vulnerability and risks, research is underway to develop these methods.

In the future, we have to able to identify what are the actions that can lead to a "negotiated, simple and adaptive" landscape management corresponding to local stakeholders' vision.

[1]Center for International Forestry Research (CIFOR)

Will markets for carbon sequestration affect human-dominated tropical landscapes?

Bruno Locatelli[1], Lucio Pedroni[2]

Abstract

Since the release of the United Nation Framework Convention on Climate Change in 1992, the forestry sector worldwide has been expecting an increased flow of national and foreign investments in carbon sequestration projects. Five years later, a number of industrialized countries acquired quantified emission reduction or limitation commitments under the Kyoto Protocol, and the Clean Development Mechanism (CDM) was defined. The CDM is a flexibility mechanism of the Kyoto Protocol that allows energy, industrial, waste management and forest projects in developing countries to sell Certified Emission Reductions (CERs) of greenhouse gas (GHGs) to the developed countries having emission reduction commitments under the Protocol (the so called Annex 1 countries). CDM projects have two objectives: to reduce or remove GHG emissions above business-as-usual levels, and to benefit to host country sustainable development.

In the first years following the definition of the CDM, many stakeholders of the forestry sector perceived the CDM as a way to receive payments for all carbon stored in any forest ecosystem. For this reason, many development projects were anticipating the possibility of marketing carbon credits for ensuring easy and long-term financial support.

Since 1997, several UN negotiation rounds have taken place to define the modalities and procedures of the CDM. For forest-based project activities, these negotiations concluded in December 2004. Now, potential CDM stakeholders are better aware of the real opportunities. For instance, only afforestation and reforestation project activity are eligible under the CDM for the first commitment period of the Kyoto Protocol (2008-2012). The eligibility of forest conservation, forest management, revegetation and other project activities in the Land Use, Land Use Change and Forestry sector will be discussed for subsequent commitment periods.

The design of CDM project activities must include a baseline definition, an estimation of GHG emissions in the without-project and in the with-project scenarios, the reasons why the project would not move forward without the CDM (additionality), and the impact of the project activity on GHG emissions outside the project boundary (leakage). The entities proposing a project activity under the CDM have to deal with a complex project cycle, the associated transaction costs, and the current low prices of carbon credits. Forestry projects have also to deal with the non-permanence of carbon removed from the atmosphere by forest ecosystems. In 2003, the Parties to the Kyoto Protocol decided in n Milan that forestry projects under the CDM would deliver non-permanent credits, namely tCER and lCER (temporary and long-term Certified Emissions Reductions). In the Kyoto protocol GHG accounting system tCERs and lCERs are the only one that expire, which makes them of lower market value.

[1] CIRAD-CATIE
[2] World Bank-CATIE

Forestry projects produce lower amounts of carbon credits than emission reduction projects do, and at a slower pace. As most buyers are interested in large amounts of credits for the first commitment period of the Kyoto Protocol (2008-2012), forestry projects will be disadvantaged. Moreover, as CDM modalities and procedures were defined later for forestry projects than for other projects, the delay may represent another impediment for forestry projects being able to issue significant amounts of carbon credits for the first period.

Non-permanent credits, high transaction costs and low credit prices make the participation of small-scale forestry projects more difficult, even if simplified modalities and procedures are being developed fore these projects. Market factors, such as transaction costs and credit prices, will affect strongly the scale of forest projects under the CDM. The issue of scale is associated with the project contribution to sustainable development. For instance, industrial plantation projects may have less positive impacts on sustainable development in rural areas than projects involving local communities in small-scale farm forestry and agroforestry activities. Moreover, small-scale forestry activities may diversify landscape mosaic and contribute to biodiversity conservation.

The impact of CDM on human-dominated tropical landscapes and its contribution to sustainable development will depend mainly on market factors, such as credit demand and prices. As the carbon market is emerging, some data are now available about forestry projects proposed to the CDM or to the World Bank Carbon Finance funds, especially regarding project types and anticipated impacts on landscape and sustainable development, as well as prices and transaction costs in some cases (Lecocq, 2005).
In 2004, more than 100 millions of tons of CO_{2e} were traded between GHG reduction or removal projects and carbon credit buyers and this market has an increasing trend (30 Mt in 2002, 80 Mt in 2003). Project-based transactions are the most important transactions in the global carbon market (95% in 2004), as transactions of emission allowances between entities with commitment, such as Annex-I countries or firms, started only recently under the European Emission Trading System (ETS). Under the Kyoto Protocol, the transaction of emission allowances will only start in January 2008, since the CDM is the only "early start" flexibility mechanism of the Kyoto Protocol

The global carbon market is divided into many segments. In 2000, the most important buyers were the voluntary buyers. These buyers do not have any obligation to reduce GHG emissions or buy carbon offsets but participate to the market for reasons such as public image, early positioning in the market, and others. In 2000, voluntary buyers purchased 16,3 Mt of CO_2e representing 95.3% of all project-based transactions. Some examples of these transactions are reforestation projects in Costa Rica supported by RTT (Reforest The Tropics), community plantations in India supported by US citizen associations, or protected area in Madagascar supported by the rock band Pearl Jam.

Between 2000 and 2004, the situation changed dramatically: in 2004, the entities with commitments were representing 98% of the total transactions (107 Mt) while the voluntary markets dropped down to 2,3 Mt. In 2005, with the start of the European ETS in January and the entering into force of the Kyoto Protocol in February, less and less market stakeholders are interested in purchasing voluntary credits. In 2004, Europe represented 60% (with 16% for the Netherlands) and Japan 21% of the demand for carbon credits. Future buyers may include Canada and New Zealand, among other.

The price of a ton of CO_2-equivalent (CO_2e) depends on many factors and is currently lower for project-based transactions than for allowances-based transactions. In project-based transactions a critical factor is the date of signature of the Emission Reduction Purchase Agreement (ERPA). The 2004 average price for non-forestry projects was 4,23 US$/$tCO_2e$ for ERPAs signed before project acceptance by the CDM Executive Board and 5,63 US$/$tCO_2e$ for ERPAs signed thereafter. The price difference is due to the risk of project rejection by the CDM Executive Board when a deal is signed before the project enters in the UN system. Prices may also depend on project specific risks, experience and reliability of the project entity, country risk, contract type (payment on delivery or anticipated payment), and socioeconomic or environmental impacts of the project. For forestry projects, there are very few market signals for estimating average prices. In 2004, forestry projects issued only 4% of the total carbon credits. The World Bank BioCarbon Fund (BioCF), which is currently the largest buyer of carbon credits from forest-based projects, offers a maximum price of US$ 4 per lCERs (or equivalent stream of tCERs) paid on delivery. The World Bank Funds purchase Verified Emission Reductions (VERs) and take the risk of project rejection by the UN system, which suggests that future market prices of CERs paids on delivery may be higher.

The World Bank created an instance called Carbon Finance Business (CFB) that manages an increasing number of carbon funds (currently 8) totaling an investment value of about one billion dollars. The Prototype Carbon Fund (PCF), which is already closed, was the first of theses funds. The BioCF was opened shortly thereafter and is so far the only one that deals with forestry projects in the CFB. The BioCF has three objectives: climate change mitigation in a cost-effective way, local environment and biodiversity conservation, contribution to local development. BioCF projects are mainly afforestation and reforestation projects eligible under the CDM, but some non-CDM projects, such as forest conservation, revegetation and soil management may also be accepted. Verified Emission Reductions (VER) issued by these latter project types will not be traded in Kyoto market. The BioCF received already around 140 Project Idea Notes (PINs) but needs an estimated of 20 projects to commit all the funds of its first tranche (about 53.8 million $). BioCF buyers may include private companies, governments, NGOs and development agencies. Currently, the buyers are: governments of Canada, Italy, Luxembourg and Spain, the French development agency AFD, Eco Carbone Okinawa Electric, Tokyo Electric, Japan Petroleum, Sumitomo Chemicals, among others.

As of September, 2005 the BioCF has retained 22 PINs for the preparation of Carbon Finance Documents (CFDs) and is issuing Letters of Intent (LoIs) for 17 of them, 9 of which have already been signed. No purchase agreements have been signed for forestry projects so far. The 22 retained PINs are located Latin America (9), Africa (9), Asia (2) and Eastern Europe (2). The main feature of these projects is that they all include a strong component of development and local environment. For instance, a project in Dominican Republic considers watershed protection and poverty reduction through ecosystem restoration and a project in Honduras aims at introducing agroforestry techniques to enhance soil conservation and to diversify economic activities.

Carbon markets are increasingly becoming a reality for forestry projects and, in many countries, new projects are being designed. As the projects already submitted integrate sustainable development priorities, the current qualitative trend shows that the impacts of carbon markets on landscapes and communities should be positive. Some stakeholders of the CDM play an important role for ensuring that CDM achieve its goal of sustainable development. Many NGOs (such as SinksWatch) control the CDM projects and inform the civil society or the buyers about any deviance. Other stakeholders, such as CCBA

(Climate Community Biodiversity Alliance) have developed standards and tools for helping CDM project developer to take into account sustainable development priorities and for allowing "good projects" to be certified.

Regarding the quantitative attributes of the carbon market for forestry projects, the emerging market is still very reduced. The supply is high, as it could be seen during the first call for proposals of the BioCF, but there is still a lot of uncertainties about the size of the demand side. The impact of markets for carbon sequestration on human-dominated tropical landscapes will greatly depend on the interest of buyers, especially European countries, and on the work of facilitating institutions, such as the World Bank Carbon Finance.

An aspect that remains unclear refers to the premium carbon price that may be awarded to projects with demonstrated positive impacts on sustainable development. It will be interesting to observe if the Annex-I countries that called for strong restrictions –for avoiding negative impacts- to forestry projects during the international negotiations will now show an higher willingness to pay for carbon from projects with positive impacts.

The Role of Model Forests for Managing Ecosystem Services at Landscape-Scales[1]

José Joaquín Campos[2] A. and Olga Corrales[3]

1. Importance of forest landscapes for the provision of ecosystem services

Forest landscapes are important providers of ecosystem services that are fundamental for the Earth's life support systems. Services include *provisioning services* such as food, water, timber and fiber; *regulating services* that affect climate, air quality, floods, diseases, wastes, and water quality; *cultural services* that provide recreational, educational, aesthetic, and spiritual benefits; and *supporting services* such as soil formation and maintenance of their fertility, photosynthesis, and nutrient cycling. Human well-being is fundamentally dependent on the flow of ecosystem services (Campos *et al.*, 2005; MEA, 2005). However, according to the Millennium Ecosystem Assessment (MEA 2005) approximately 60% (15 out of 24) of the world ecosystem services are being degraded or used unsustainably. With current trends these estimates could grow significantly during the first half of this century, becoming a barrier to achieving the Millennium Development Goals.

Unlike forest or agricultural products, most ecosystem services are not paid for by the beneficiaries. This means that too often, those who own, control or manage the natural resources where those services are generated do not capture the economic benefits that result from those services, thereby reducing the incentives to conserve these ecosystems, particularly the natural ones ((Nasi *et al.*, 2002). This is probably one of the most critical challenges to advance in the sustainable management of natural resources.

The development of market-based mechanisms such as the payment for ecosystem services (PES) in that respect could be a very powerful and innovative tool to provide the adequate incentives for the sustainable management of natural resources and a fair distribution of its benefits. According to Campos *et al.* (2005), implementing such schemes requires comprehensive intervention, including valuating the services, establishing sustainable financing mechanisms, designing and implementing effective payment systems, and providing the market with adequate institutional frameworks for transparent, accountable and verifiable decision making flow. To overcome the complexity of these challenges, research and management need to take into consideration larger temporal and spatial scales, as well as integrating the different bio-physical, social and institutional components in the landscape (actors, land uses, sectors and disciplines) and the policy and decision-making processes (Sayer and Campbell, 2003).

If PES markets are to ultimately produce welfare improvements, they need to be designed to provide the incentives that ensure a fair distribution of the benefits. This in turn requires a clearer understanding of the nature of the relationship between diverse land-uses and land management and the ecosystem services provided (Landell-Mills and Porras, 2002). But it also requires an understanding of human and ecosystem interactions (Rojas and Aylward, 2003), and the enabling institutional environment. Therefore, there is a need for

[1] Paper prepared for the IV International Wallace Conference "Integrated Management of Environmental Services in Human-Dominated Tropical Landscapes". CATIE, Costa Rica. November 1-3, 2005.

[2] Deputy Director General and Director of Natural Resources and Environment at CATIE. President Regional Network of Model Forests for Latin America and the Caribbean.

[3] General Manager of the Regional Network of Model Forests for Latin America and the Caribbean.

further research on the biophysical and economic characterization of ecosystem services (the evidence), and in identifying the enabling social framework for the development of sound institutions and policies for implementing payment systems (Campos *et al.*, 2005).

There are increasing examples of initiatives that aim at managing ecosystem services. However, we argue that, to ensure long-term sustainability, there is a need to encourage more initiatives that manage them at landscape-scales, based on systemic approaches, and constructed on participatory decision making mechanisms.

2. Integrated management of ecosystem services at landscape-scales

It is clear that humanity is highly dependent on the flow of ecosystem services, yet this flow is dependent on the way in which ecosystems are affected by human activities. Diverse uses of the natural resources, from preservation, hunting, gathering, to agricultural and forest management practices, have an important effect on ecological processes and therefore in the provision of ecosystem services within a landscape and beyond.

Physical, economic, or social impacts of ecosystem service degradation and improvement may cross political and geographic boundaries. For example, land degradation and associated dust storms or fires in one country can degrade air quality in other countries nearby (MEA, 2005), just as land improvements can affect water use in other countries in the low parts of a watershed.

These effects can more easily be appreciated at larger scales such as the landscape-scale (e.g. a watersheds, biosphere reserves, model forests or biological corridors), where the cumulative effects of interventions of human societies (farms, infrastructure, water management, urban environmental practices etc.), can be observed in polluted rivers, degraded forests and lands and the loss of biodiversity. A landscape based decision-making body with access to the broad vision of cause and effect of these actions, would acknowledge that these are issues of concern for the well-being within its community. In fact, ecosystem processes operate at different scales, but some are more dependent on processes at larger scales and therefore should be managed at such scales with representation of stakeholders within that scale.

Much attention has been given to the significant contribution that forest ecosystems play in the maintenance of ecosystem services. However, increasing attention is being given to the role that intervened landscapes with, for example, agriculture use, may play to the provision of ecosystem services. Incorporating trees in a planned way in live fences or as shade contributes to the fixation and storage of carbon, as well as providing habitats and corridors to many insect, bird and small mammal species that move between forest patches in the landscape. By combining trees with perennial crops, soil conservation measures and a restricted use of chemicals, agricultural systems may also contribute to the conservation of biodiversity and to maintaining water quality for consumptive use and reduce the risks of flooding. Trees, shrubs and non-commercial herbaceous plants in agricultural systems may also contribute to pollination and the reduction of crop damage due to pests and diseases. On the other hand, they may also be the cause for some damage (e.g. invasive species) and it is important to know which plants and animals can be tolerated and which not. It is key, therefore, to provide land users with readily available information and incentives that aim at improving their practices; this can be facilitated by a constant dialogue amongst research institutions and organized land users and decision makers within a landscapes. There are potential gains when the landscape matrix, in the hands of a stakeholders governance body, is considered to enhance the provision of ecosystem services.

From a scientific perspective, at larger scales, complexity and uncertainty could be overwhelming. Adaptive management and collaborative approaches with all relevant stakeholders are indispensable to address this complexity. As we understand more about the immensity of the challenges we face in order to revert the degrading trend regarding ecosystem's health, humbleness invade us and the conviction that we need all the resources and capacities of multiple stakeholders and multiple disciplines to revert this trend.
We propose that there is therefore a need to think and act at multiple scales (spatial and temporal), with multiple stakeholders, in multiple partnerships and in inter-disciplinary ways.

At the landscape-scales, social, economic and institutional issues become more relevant and in many situations could in fact turn to be the most important factors to sustainably manage natural resources. Trust and dialogue opportunity at this scale is particularly important for conflict prevention during designing and implementing mechanisms for managing ecosystem services (e.g. externalities and property rights).

Effective participatory governance and co-management mechanisms are key elements in achieving common visions and strategies to use natural resources that address stakeholders values. Experience has shown that in an informed economic context, these decision geared towards improving life quality without jeopardizing the quality of environmental services and providing the arrangements for an equitable distribution of the benefits.

At these larger scales, effective participatory governance and co-management mechanisms become crucial. In this respect, the innovative experiences being developed by several initiatives at the watershed level (watershed committees) and other territorial approaches such as model forests (board of directors) and biological corridors (management committees) should be further assessed to systematize and share the lessons that enable and encourage sustainable management of ecosystem services.

3. The role of model forests in the provision of ecosystem services

A model forest is a lot more than forest; it is a landscape-scale based decision-making process with multi-stakeholder participation. It is a board of users of ecosystem services with the common goal of sustainable development, based on an informed use of these ecosystems. Model forests operate through partnerships and networking through transparent and accountable governance, consensus based decision-making processes and community-based participatory action.

There are currently close to 40 model forests in development in most regions of the world, including developing regions, such as Latin America and the Caribbean, Asia and Africa. The innovative participation and management alternatives developed in each model forest are shared and analyzed through the regional and the global networks[4]. The Regional Network for Latin America and the Caribbean is the forum for model forests established in Argentina, Bolivia, Brazil, Chile, Costa Rica, the Dominican Republic; with other countries interested in joining the network.

The origin of model forest lies in a conflict between first nations of Canada and logging companies. Their dialogue platform proved effective in mitigating the conflict and enabling the sustainable use of the forest.

[4] For example, in December, 2004, representatives from all model forests in LAC met in Costa Rica (CATIE) to share and discuss the opportunities for implementing PES in their respective model forests.

In Latin America and the Caribbean, common features of these model forest boards include participation of central and local government, indigenous communities, private sector (such as agriculture chambers), research and academia and NGO's.

The landscape within which a model forest board is based typically consists of a wide range of ecosystems: from those relatively undisturbed, such as natural ecosystems in protected areas, to a mixture of patterns of more intense human use such as managed and modified ecosystems such as agroforestry systems, tree plantations and agricultural lands.

Model forest boards provide the platform for dialogue amongst providers and beneficiaries of environmental services, and can facilitate the identification of the financial mechanisms that can enable the system. As transparency, accountability and constant dialogue are the *modus operandi* of these boards, they provide a venue under which agreeing on principles and incentives that can enable the flow of ecosystem services is possible. Furthermore, as networking regionally, and globally, are also key elements of the identity of these boards, feedback and replicating models can also strengthen these mechanisms of effectively compensating for environmental services.

The effective management of ecosystem services at landscape-scales requires the support of sound knowledge in order to deal with complex issues such as the evidence that the quality and quantity of a particular ecosystem service is actually being provided by the agreed measures between the parties. In this respect, model forests and the model forest networks can play a very important role in facilitating the conditions for enhancing the innovation processes that would be required.

Model forests can serve as platforms for innovation, involving relevant stakeholders to develop proper innovation systems and processes (e.g. learning), linking also scientific research with traditional knowledge, as has been evidenced in practice and increasingly acknowledged by the UN technical bodies advising parties to environmental conventions.

Model forests can facilitate the conditions for effective management, involving participation of stakeholders in designing, negotiating and monitoring environmental services mechanisms. This approach can be relied upon to adjust research priorities to the heterogeneous demands of the rural stakeholders (participatory action-research).

LAC provides three examples of how a model forest enables a PES scheme: one in Chile, one in Argentina, and one in Costa Rica.

- In Chile, the Chiloé model forest board has provided the platform for reaching an agreement with the water provider in the design of the first PES scheme in Chile. As a matter of fact, the national government is anticipating Chiloé's experience as a precedent for their PES national policy.
- In Jujuy, the model forest manager, influencing policy making at the provincial level, launched a dialogue on PES that resulted at the issuance of the regulations that are enabling the model in Argentina.
- Finally, the Reventazon Model Forest in Costa Rica is providing research and advocacy support to the Turrialba-Jimenez Biological Corridor. Specific results have included identifying and agreeing on areas of critical conservation value that should be prioritized in the payment of ecosystem services.

It is worth acknowledging that working at landscape scales, with participatory approaches, increases complexity, and therefore result in higher transaction costs. However, we also believe that in most cases these efforts are worthwhile in the longer term. If the design of PES systems are based in mechanisms such as model forest boards, the likelihood of sound implementation and sustainability is significantly increased.

Experience has also indicated that an effectively facilitated approach to enable participatory mechanisms, attracts capable and motivated people and institutions. If this process succeeds in incorporating values regarding capacity to learn and adapt, adopting learning an intentional process, there is an increase willingness to accept change and an enhanced capacity for response. Belonging to a network with common values would promote collaboration in finding solutions for the challenges of joint decision making, effective participation, creating mutual confidence and respect.

The needs and demands could become so high that they might require even further and more effective communication and collaboration among the different stakeholders. Networking could become even more relevant to promote dialogue, exchange of experiences and information. In the case of model forests, networking between model forests in the same country, as well as between model forests within the same region or other regions in the world, is a strong attribute for achieving such collaboration. In the case of promoting a fair and effective flow of the benefits of ecosystem services, this seems to be pivotal for the global sustainability of resources.

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Keywords: co-management of natural resources; ecosystem services; integrated management of natural resources; landscape-scale management; model forests; networks; participatory decision-making; participatory governance; sustainable forest management.

Challenges in the use of environmental service payments for the achievement of biodiversity conservation goals

Bryan Finegan[1]

1. Introduction

The conservation of biodiversity (from here on, simply "conservation") is proving more complicated than conservationists once thought it was, and practical frameworks for conservation have increased in number, scope and complexity (Redford et al. 2003). Payments for environmental services (PES) are not an approach to conservation in themselves, but a tool to make possible the implementation of approaches to conservation such as those discussed by Redford et al. (2003).

PES is in its infancy and one of the major challenges regarding its future development as a tool to promote wise stewardship of natural resources is to increase its effectiveness in the achievement of conservation goals. Conservation goals for PES programmes are not usually clearly articulated, with corresponding room for improvement in priority-setting with respect to where and when to assign funds. The effective use of a tool that influences land use on relatively small private properties to contribute to conservation goals at multiple spatial and temporal scales is a major challenge. Increased effectiveness in the use of PES would mean an improved benefit-cost ratio for society, and it is a premise of this presentation that in the same way as conservationists are interested in the effectiveness of their management actions, society as a whole is interested in the benefits it obtains from financial instruments such as PES. As is unfortunately the case for conservation management in general, however, it is usually much easier to determine the monetary cost of a management action – in this case, a PES programme – and the area of land managed under the programme, than its actual benefits for biodiversity and society. This represents a challenge for conservation scientists and practitioners as well as managers of PES.

The overall objective of this presentation is to evaluate the concept and practice of PES for biodiversity from a conservation ecological viewpoint and set out proposals for increasing the effectiveness of the tool for conservation. In order to achieve this objective, I first attempt an overview of current thinking among conservation organisations regarding conservation targets, in order to set a clear picture of what, ultimately, PES programmes should help us conserve, and where. Then, I discuss the area of action of PES in relation to the development of tropical agricultural frontiers and propose specific conservation objectives for the human-dominated landscapes in which PES seems most likely to be effective as a conservation tool. I outline Costa Rica's PES programme in order to determine how the tool is being used for conservation in practice, and to facilitate the definition of suggestions for increased effectiveness of PES in general. Finally, I use case studies from Mesoamerican biological corridors for identifying additional challenges in the use of PES to support relatively high-resolution and spatially-explicit conservation proposals at the landscape scale.

[1] CATIE

2. Conservation goals: what to conserve, and where?

What? – functional conservation areas (FCAs) - The "what to conserve" question can be fairly clearly answered on the basis of the missions, objectives and approaches used by governmental and non-governmental conservation organizations. "Nature" was transformed into the explicitly multiple-scale, hierarchical and multi-faceted "biodiversity" during the 1980s. Systematic approaches to conservation have since emerged alongside wide acceptance that communities/ecosystems, along with species groups or individual species requiring individual attention, and *the ecological and evolutionary processes that generate and maintain these levels of biodiversity* are principal conservation targets (Scott et al. 1991, Noss 1996, Poiani et al. 2000, Redford et al. 2003). An attractive proposal for an integrated conservation objective or target is that of functional conservation sites, landscapes and networks for conservation (Poiani et al. 2000). In this approach, ecosystems and species are conserved on the basis of a land-use planning designed to ensure the maintenance of the ecological and evolutionary processes that unite the ecosystems and species to be conserved into an adaptive and therefore functional system – the system therefore becoming the higher-level conservation target. In this presentation I will use the collective term functional conservation area (FCA) to refer to this definition of the overall conservation goal.

Where? – apply priority-setting exercises– Not all biodiversity can be conserved everywhere, some communities or species are considered to be of higher priority than others, and the risk of biodiversity loss is acute in some places but currently low in others. Priority-setting exercises have therefore become a key feature of the conservation portfolio. The scale of these exercises tends to be large, from the global to the regional, they can have an essentially ecosystem/community focus (e.g. ecoregions, gap analysis) or a species focus (e.g. hotspots, Endemic Bird Areas). They will often set priorities by combining information on areas where the benefit/cost ratio of conservation actions is likely to be most favourable, such as those with high proportions of endemic species, with assessments of the degree of threat to conservation targets, either explicitly (ecoregions, hotspots) or implicitly (Endemic Bird Areas). Usually at smaller scales, proposals for biological corridors reflect priorities with respect to the maintenance of the functionality of conservation areas. The Mesoamerican Biological Corridor and the national and subnational corridors nested within it is an outstanding example.

3. Analysis: towards PES as an effective conservation tool

The development of agricultural frontiers, PES, and conservation targets

Certain social and institutional conditions are necessary for a PES programme to function. These conditions are not met everywhere, but develop as agricultural frontiers evolve. This means that the application of PES as a conservation tool is limited to human-dominated landscapes that have reached a certain point in their development following initial colonization. Spatial and temporal patterns of human impact on tropical forest landscapes can usefully be described by simple models like those of Henkel and Richards (Smith et al. 2001). An *Early Pioneer* stage advances into the forest, a stage during which the lack of definition of land tenure, poor infrastructure and weak institutional presence make the use of PES impractical. The partially deforested pioneer landscape evolves into the *Emerging Market Economy* stage of the frontier, in which farmers take advantage of improved infrastructure and access to markets and institutional interventions using tools like PES become possible. A large proportion of the natural forest cover will have been converted to agricultural use by this stage, and that which remains will be fragmented. This stage may evolve into a "zone of decay" as agricultural productivity declines and farmers abandon land and may migrate back into the early pioneer fringe. Substantial areas of naturally regenerated secondary vegetation may develop at this time. Land becomes concentrated into the hands of fewer property owners and the frontier may subsequently be "revitalised" as originally small plots are amalgamated for livestock production or higher-technology agriculture is introduced. This is the *Closing Frontier* stage in which availability of forest land for further colonisation is low or zero, though forest fragments or patches of secondary regrowth typically remain in the landscape, the former often divided up amongst more than one landowner - and there are additional improvements of infrastructure and farmer integration with markets. The application of PES is a practical proposition from the social and institutional points of view.

It may seem obvious that social and institutional factors will usually limit the scope of PES for conservation to human-dominated landscapes, but the exact implications of this fact with respect to conservation goals need to be clear. I would like to make three points:

1. Although large areas of natural habitat ("core areas" – national parks, for example) are the single most important components of FCAs, PES programmes have little to offer as a tool for their conservation, because forest dwellers and farmers in pioneer agricultural frontiers are out of the social domain and institutional reach of PES.

2. PES is a potential conservation tool for human-dominated landscapes and needs to be orientated in relation to likely conservation objectives for such landscapes; these are summed up in the following section.

3. The conservation community achieves a rare degree of consensus in accepting that large areas of undisturbed natural habitat are an effective conservation tool. On the other hand, the consequences for biodiversity of habitat destruction and fragmentation in human-dominated landscapes and the value to conservation of such landscapes are debated widely and deeply, partly because of the lack of information. Principles and guidelines to support conservation management in such landscapes (Lindenmayer and Franklin 2002, Bennett 2003) are only recently becoming widely referred to. Farmers who are within the social domain and institutional reach of PES therefore inhabit landscapes in which there is greater uncertainty regarding how to apply and evaluate PES for conservation.

Conservation in human-dominated tropical landscapes: what should society seek through PES programmes?

Likely conservation functions for human-dominated landscapes, which therefore represent a basis for guidelines for the application of PES, can be summed up as follows (adapted from Finegan et al. in revision; of course, two or more functions are likely to apply simultaneously in many situations):

1. *Contribution to regional priorities for species conservation* the landscape is part of a larger area that has been assigned high priority for conservation of biodiversity in a regional exercise because it is both rich in regional and local endemic species and is extensively disturbed by agricultural frontier expansion (examples are the montane forest ecoregions of Colombia's Cauca and Magdalena Valleys), so that conservation management should contribute to the conservation of these species.
2. *Representativity* the landscape contains natural community types, or significant proportions of the local distributions of individual species, that are

unrepresented in formally protected areas – conservation gaps (Scott et al. 1991). Conservation management should seek to improve representativity by conserving these community types and species. An example of such a landscape is Costa Rica's San Juan-La Selva Biological Corridor (see below) and in a biogeographically and environmentally complex area such as Mesoamerica it seems likely that all relatively large corridors can potentially fulfil this function.

3. *Connectivity* the landscape is a key area for the provision of ecological connectivity (again, Mesoamerica's corridors are an example), so that management should focus on the ecological processes grouped under the term connectivity.
4. *Buffering for protected areas* the fragmented landscape adjoins a protected area and management could seek to provide a buffer zone for the reduction of possible undesirable human impacts on the protected area. The fact that the human-dominated landscape may provide habitat for some organisms that complements that of the protected areas should be included in the buffering function.
5. *Sustainable agriculture* conservation management seeks to maintain or restore ecological processes dependent on areas of remnant natural habitat in the landscape that support agricultural production, as in the well-studied example of the pollination of coffee by native bees: probably relevant in most landscapes, though not the subject of this presentation.

The identification of conservation functions is a potential contribution to systematic conservation planning (Margules and Pressey 2000) for human-dominated landscapes, and to the effective use of PES. Functions 1 to 4 of those outlined above are key contributions to FCAs and a large-scale priority-setting exercise is necessary to determine where and when human-dominated landscapes have an important role in the consolidation of functional conservation areas at regional, national and subnational scales. A more detailed evaluation and priority-setting is then necessary for each landscape selected, in order to facilitate its own functionality – a landscape-scale FCA nested within subnational and national FCAs (see Section 2). Knowledge of landscape structure and composition is necessary at this stage. A process like this has been underway in Mesoamerica during the last ten years – the Mesoamerican Biological Corridor (MBC) - and has led to the delimitation of proposals for landscape-scale corridors on the basis of potential conservation functions using criteria like those identified above.

What to pay for and where, in practice: the Costa Rican PES programme

The evolution of Costa Rica's well-known PES programme should be traced from the country's first incentives for forestry during the 1970s, but in its its more immediate origins are in the 1996 Forestry Law and 1998 Biodiversity Law, which in turn responded to the 1992 Rio Summit (see Barrantes 2005). The programme specifically recognises environmental services in four categories: carbon storage and capture for the mitigation of climate change, water, conservation and scenic beauty. Costa Rica does not pay for conservation for the intrinsic value of biodiversity, but for the services that biodiversity provides, though these are not clearly identified in the official literature (e.g. Barrantes 2005). The scope and impacts of PES for conservation in Costa Rica are in many ways difficult to evaluate simply because the programme has been running for a relatively short time and has been accompanied by other major changes and innovations in land use policy (the prohibition of further clearance of primary natural forest, for example). However, it is straightforward to use the programme as an example of how PES can be applied for conservation in practice. What can we learn from Costa Rica's conservation goals and PES programme in relation to the objectives of this presentation? Specifically, what can we learn with respect to the scope of PES for facilitating the achievement of the conservation goals of tropical countries, and ways in which the effectiveness of PES can be increased?

Costa Rica attempts to direct the use of PES in the several ways, amongst which the following are most relevant to conservation goals (see *La Gaceta*, Friday March 5th 2004):

- *Environmental services from forests* At a first, straightforward level of orientation of PES for conservation, this is largely payment for the provision of environmental services *by forests*. For a time, the programme paid for environmental services from natural forests assigned to protection with no extractive use, natural forests under management for sustainable timber production, and forest plantations. Currently, however, timber production forests are excluded from PES. The qualification "largely" is used because the programme has now expanded to include payment for the establishment of trees in agroforestry systems.
- *In biological corridors* Through the Mesoamerican Biological Corridor initiative the countries of the region approved that priority should be given to properties in biological corridors for PES (though review of available literature gives the strong impression that payments made within the limits of corridors have often been assigned for hydrological functions, rather than conservation *per se*). Chassot *et al.* (2005) propose an expansion of the application of PES as a principal conservation instrument in Costa Rica's San Juan – La Selva Biological Corridor.
- *To certain social groups* Part of the programme's mission is to compensate medium and small farmers for the environmental services they provide and priority is given to PES in the country's less prosperous areas, as defined through an index of social development used by the Planning Ministry. Applicants must however fulfil a number of requirements including clear title to the land and the provision of a managementplan.
- *In very small to small areas over short time periods* PES for forest protection is paid on the basis of a five-year contract for forest areas between 2 ha and 300 ha, per landowner per conservation region.
- *With annual review and updating of guidelines Finally, official sources state that priorities for PES in the country will be reviewed and updated through executive decree on an annual basis.*

Besides the official guidelines, prestigious natural resource management NGO, FUNDECOR, has mapped its area of influence in the centre and north of the country in relation to priority for payment for carbon, water and conservation, including areas that are given priority for combinations of the three (go to http://www.fundecor.org).

The results of the application of the guidelines summarised above are clearly evident in maps showing the spatial distribution of properties receiving PES, which is strongly concentrated in some, but not all, of the country's biological corridors.

Case study: from geographical areas to FCAs in landscape-scale biological corridors

For the final part of this analysis I will use landscape-scale corridors as a case study, focusing on a relatively unexplored area – criteria for converting the corridor from a geographical area into an FCA. I will briefly introduce the landscape-scale corridor as it has involved under the MBC initiative, describe an approach to FCA development and discuss implications regarding PES for conservation at this scale.

The Costa Rican case shows that proposals for landscape-scale biological corridors (from here on, simply "corridors") vary widely in terms of the existence and clarity of conservation goals and the quantity and quality of ecological information on the areas delimited (Rojas and Chavarría 2005, L. Canet personal communication, 2005). For this presentation I have used information from three of the six Costa Rican corridors for which Canet (personal communication 2005) considers the situation to be optimal. PES for conservation in corridors that lack clearly-defined objectives seems unlikely to be effective or easy to evaluate.

Costa Rica's corridors are to a great extent human-dominated landscapes but vary widely in their size, inherent environmental heterogeneity, degree and pattern of habitat loss and fragmentation, and social context, among many other factors; while the connectivity function is common to all, therefore, other conservation functions are not. It is important to emphasise that these are not corridors in the landscape ecology textbook sense of linear areas of habitat that structurally connect two or more habitat patches, but multiple-use landscapes whose integrated management for conservation functions *including* connection is proposed.

Of course, the functioning of a corridor is not guaranteed by its geographical delimitation on maps, especially in the case of human-dominated landscapes. What tools are available to support the transformation of corridor proposals from geographical areas into FCAs? The complexity of the ecological systems under consideration in combination with the chronic lack of information suggests a coarse-filter approach based on relatively modest requirements for data regarding habitat structure and composition at landscape and local scales (Noss 1996; Scott et al. 1991; Lindenmayer and Franklin 2002). The enormous potential complexity of the connectivity function of corridors is dealt with in this type of approach from the straightforward structural point of view (Bennet 2004). Ramos and Finegan (2005) applied such an approach to the development of a proposal for the structural framework of an FCA within the lowland sector of the San Juan – La Selva corridor, adapting procedures successfully used in more information and resource-rich situations (Hoctor et al. 2000). An intermediate area of natural habitat still exists in this large corridor which should fulfil the representativity function for both natural community types and species, as well as the connectivity function. The four-step process used by Ramos and Finegan involved:

1. the delimitation and mapping of different forest types to facilitate a conservation gap analysis (the representativity function, for natural community types)
2. the identification of landscape-scale core forest areas for conservation, using criteria of ecological integrity, forest type and habitat suitability for the corridor's flagship species, the threatened green macaw (the representativity function, for both natural community types and species)
3. the modelling of potential routes for maintenance of structural connectivity between core areas in a GIS, on the basis of suitability values subjectively assigned to different cover types (the connectivity function)
4. the unification of core areas and connectivity routes in the proposed structural basis for the FCA (the overall goal – the corridor as an FCA)

This process used a digital map of the corridor based on a Landsat TM image, field sampling of vegetation characteristics using a dedicated protocol (this could be substituted by existing GIS maps of physical environments if necessary) and standard GIS software and procedures. This adaptive tool can be used to reevaluate the landscape assigning different weights to criteria or incorporating new information.

Costa Rica has succeeded in giving priority to this corridor for PES, but it is not clear whether this entirely voluntary programme orientated towards small forest areas could be more tightly focused to support a relatively high-resolution and highly spatially-explicit landscape-scale conservation proposal such as that produced by Ramos and Finegan's conservation ecological approach. With respect to the evaluation of effectiveness, this type of conservation proposal by its very nature provides clear sets of indicators, but does not help in the evaluation of PES if PES can not be focused to support it. It may therefore be the case that at the landscape scale, the conservation impact of a PES programme like Costa Rica's is likely to be diffuse and difficult to evaluate.

5. Conclusions

PES programmes are largely tools for conservation in human-dominated landscapes and at a certain, rather advanced point in the development of those landscapes, at which the loss, fragmentation and simplification of natural habitat are inevitably substantial. They are therefore tools to be used in environments in which there is a considerable and unavoidable degree of uncertainty regarding the status of biodiversity in the landscape and also therefore, regarding the necessary management actions to be supported by PES. On the other hand, widespread recognition of the key conservation functions that can and must be assumed in human-dominated landscapes means that PES potentially has a key role in the attainment of conservation goals. But in the overall framework of conservation goals, it is essential to recognise that PES has limited value to support the conservation of the single most important elements of functional conservation areas – large core areas of natural habitat.

In the context of human-dominated landscapes, biological corridors are key components of proposals for functional conservation areas in Mesoamerica. The attainment of functionality is a major challenge for conservation in such multiple-use landscapes and can be approached through the identification of priority areas of habitat and routes for structural connectivity. But it is not clear whether PES can be focused in such a way as to directly support this type of conservation proposal, so that its conservation impacts may remain diffuse and difficult to evaluate.

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La Conservación de la biodiversidad de la Reserva de la Biosfera Sierra de las Minas a través del desarrollo de un sistema de pagos del agua como un servicio ambiental de las cuencas del sistema Motagua/Polochic

Lic. Carlos Morales Cajas[1]

Resumen ejecutivo

El sistema Reserva de la Biosfera Sierra de las Minas, Motagua y Bocas del Polochic Guatemala es una de las áreas identificadas como de las más altas prioridades para la conservación de la biodiversidad de Mesoamérica Norte. Por sus características altitudinales, precipitación y posición geográfica, es considerada como un hábitat que alberga una gran cantidad de especies de flora y fauna con una evolución genética única y que además juega un papel clave en la conectividad de la región. En la cuenca alta de la reserva de la biosfera se originan más de 63 ríos permanentes que mantienen el caudal ecológico durante todo el año.

Por otro lado en su cuenca media y baja existe un "mercado" de usuarios del agua que van desde industrias grandes y medianas (licoreras, papeleras, embotelladoras, parques acuáticos), hasta usuarios domésticos. Aproximadamente medio millón de personas de al menos tres diferentes etnias, dependen de este recursos para su desarrollo económico y social. Finalmente estas cuencas tienen gran influencia, en el Sistema Arrecifal Mesoamericano, que ha sido declarado una prioridad de conservación para los gobiernos y otros actores claves de la región.

Las mayores amenazas reconocidas han sido la deforestación, la expansión ganadera, los incendios forestales y la expansión de la agricultura intensiva de monocultivo. Al mismo tiempo, no existe un mecanismo oficial, ni voluntario instaurado en el lugar para que los usuarios paguen por la extracción y consumo del agua, ni para la conservación y manejo de las cuencas.

Además durante los últimos años, estos usuarios han observado una reducción de la cantidad y calidad del agua, particularmente durante la época seca. Esto ha generado una mayor voluntad a colaborar en la búsqueda de garantizar el abastecimiento futuro de este recurso.

Con el propósito de proteger esta zona, promoviendo la sostenibilidad financiera de la conservación de las áreas protegidas y adecuado manejo de la cuenca en su parte alta y media, se impulsa un novedoso mecanismo de compensación e incentivo, centrado en el agua como un servicio ambiental (PSA) que el sitio provee a los usuarios de la cuenca media y baja y se conoce como **"El Fondo del Agua".** El mecanismo propuesto es totalmente voluntario y está diseñado para ofrecer alternativas de proyectos dirigidos a satisfacer, las necesidades de los usuarios, a corto y mediano plazo, relacionadas con el recurso hídrico.

En el documento inicial se resumen las generalidades de: organización, operación, tipos y ejecución de proyectos y factibilidad económica, en busca de ofrecer al lector interesado en el proyecto, un documento corto y práctico con los aspectos principales de esta iniciativa y sus ventajas.

[1] Oficial de Programa Agua Dulce WWF Centroamérica cmorales@wwfca.org www.wwfca.org

Experiencias y potencialidades del pago de servicios ambientales en cuencas hidrográficas en América Central

Francisco Jiménez[1], Jorge Faustino[2]

El pago de servicios ambientales (PSA) en cuencas hidrográficas es un tema de relevancia en foros locales, nacionales e internacionales en América Central. Para el manejo de cuencas hidrográficas, el PSA podría llegar a ser una opción importante para dar sostenibilidad a los programas que se están implementando. La Declaratoria de Arequipa, dada en el marco del III Congreso Latinoamericano de Manejo de Cuencas (2003), indica *que "Es tiempo de acción para adoptar o incrementar sistemas de pago por servicios ambientales en las cuencas";* además menciona que es *"una oportunidad realista de contribución al manejo integrado de los recursos hídricos con equidad dentro de las cuencas hidrográficas".*

1. Los servicios ambientales que pueden brindar las cuencas hidrográficas

Las cuencas hidrográficas pueden brindar múltiples servicios ambientales. Entre los más destacables están:

a) Los asociados al flujo hídrico: usos directos del agua (agricultura, industria, consumo humano, acuicultura, etc), protección de las fuentes productoras de agua, dilución de contaminantes, generación de hidroelectricidad, regulación de flujos y control de inundaciones, transporte de sedimentos, recarga de acuíferos, mantenimiento de la capacidad productiva del suelo, control de la erosión hídrica, control de la sedimentación de ríos y embalses, dispersión de semillas.

b) Los asociados a los ciclos biogeoquímicos: almacenamiento y liberación de sedimentos, almacenaje y reciclaje de nutrientes, almacenamiento y reciclaje de la materia orgánica, almacenamiento y fijación de carbono, liberación de oxígeno, destoxificación y absorción de contaminantes.

c) Los asociados a la protección y producción biológica: protección de ecosistemas, amortiguamiento contra perturbaciones, creación y mantenimiento de hábitat, mantenimiento de la vida silvestre, conservación de germoplasma, fertilización y formación de suelos, etc.

d) Los asociados a la belleza escénica natural o intervenida para fines recreativos, turísticos y científicos.

2. El pago de servicios ambientales en cuencas hidrográficas: de lo ideal a la realidad

Idealmente la visión de la cuenca hidrográfica como escenario para el pago de servicios ambientales debería partir del principio de la cuenca como un sistema, en el cual interactúan diferentes subsistemas biofísicos y socioeconómicos, con el agua como recurso

[1] Coordinador del Grupo Temático Manejo Integrado de Cuencas Hidrográficas del CATIE, fjimenez@catie.ac.cr1
[2] Especialista en Manejo y Gestion de Cuencas Hidrográficas del CATIE, faustino@catie.ac.cr
[3] Castro R. 2003. El agua: de bien gratuito a bien comerciable. In: Dia de las Américas. Tercer Foro Mundial del Agua. GWP-BID. p. 153-168.

integrador, un enfoque socioambiental (basado en las interrelaciones del ser humano con el ambiente) y una estrategia de la cuenca como unidad flexible de planificación, pero con unidades múltiples de intervención (la finca, la unidad de producción, el marco legal, el marco institucional, etc).

En América Central, el tema de PSA en cuencas hidrográficas no es nuevo en términos de su análisis y discusión. Sin embargo, las experiencias operativas de PSA con una visión integral de la cuenca como sistema son escasas. La mayoría de iniciativas son bastante localizadas y corresponden principalmente a la protección de las áreas de recarga hídrica o al manejo de microcuencas de abastecimiento hídrico para consumo humano o para hidroelectricidad (cuadro 1). El reto para encontrar el camino y *los canales que permiten trascender de lo local a lo meso (intermedio) o a lo nacional es enorme.* Un aspecto preocupante es la falta de estudios de valoración real e integral de los servicios; los elementos oferta, demanda y mercado apenas empiezan a estructurarse. También es crítica la frecuente falta de una buena identificación de los proveedores y usuarios de los SA, lo que crea conflictos y dificultades de aplicación de los sistemas de PSA.

Cuadro 1. Objetivo principal del PSA en las principales experiencias en microcuencas de América Central.

Nombre de la microcuenca	**Objetivo del PSA**
Río Volcán, Costa Rica	Protección y manejo de la cuenca para generación hidroeléctrica
Río San Fernando, Costa Rica	Protección y manejo de la cuenca para generación hidroeléctrica
Río Platanar, Costa Rica	Protección y manejo de la cuenca para generación hidroeléctrica
Río Aranjuez, Costa Rica	Protección y manejo de la cuenca para generación hidroeléctrica
Río Balsa, Costa Rica	Protección y manejo de la cuenca para generación hidroeléctrica
Lago Cote, Costa Rica	Protección y manejo de la cuenca para generación hidroeléctrica
Cuenca Alta Río Virilla, Costa Rica	Protección y manejo de la cuenca para secuestro de carbono
Río Segundo, Costa Rica	Protección y manejo de la cuenca para agua potable, industria
Río Ciruelas, Costa Rica	Protección y manejo de la cuenca para agua potable
Río Tibás, Costa Rica	Protección y manejo de la cuenca para agua potable
Río Bermúdez, Costa Rica	Protección y manejo de la cuenca para agua potable
Río Cumes, Honduras	Protección y manejo de la cuenca para agua potable
Río Las Amayas, Honduras	Protección y manejo de la cuenca para agua potable
Río Escondido, Honduras	Protección y manejo de la cuenca para agua potable
Río Las Colinas, El Salvador	Protección y manejo de la cuenca para agua potable
Río Las Pirámides, El Salvador	Protección y manejo de la cuenca para agua potable
Río Gualabo, El Salvador	Protección y manejo de la cuenca para agua potable
Río Cara Sucia, El Salvador	Protección y manejo de la cuenca para agua potable
Río Paso los Caballos, Nicaragua	Protección y manejo de la cuenca para agua potable
Río Estela, Nicaragua	Protección y manejo de la cuenca para agua potable

La situación degradada de la mayoría de las cuencas de América Central sugiere la necesidad e importancia que representa la valoración e implementación del cobro y pago de los servicios ambientales, como una forma de prevenir, mitigar, controlar y revertir los acelerados procesos de deterioro del ambiente, los recursos naturales y pérdida de la diversidad biológica, para mantener y mejorar la función de las cuencas, así como para innovar el financiamiento y dar sostenibilidad a los programas y planes de gestión que se están implementando. Sin embargo, en la práctica, hacer realidad este ideal es mucho más complejo de lo previsto. Algunos de los elementos que generan complejidad se indican en el acápite siguiente.

3. ¿Qué sabemos de las experiencias de PSA en cuencas hidrográficas en América Central?

Con base en las experiencias de PSA en cuencas hidrográficas en América Central, se pueden mencionar algunos aspectos que caracterizan la situación:

- Se ha creado demasiadas expectativas con el papel del PSA, como mecanismo de financiamiento de la gestión de cuencas. La mayoría de las cuencas de América Central no tienen recursos estratégicos tan sobresalientes que puedan justificar y viabilizar integralmente mecanismos de PSA. Por ahora el discurso de PSA más fuerte es el teórico y el académico; es más lo que se dice que lo que se hace.

- No hay un enfoque claro de la cuenca como sistema, en la estructuración y entendimiento de los procesos que definen la oferta y demanda de servicios ambientales; aún con el servicio ambienta hídrico, donde por ser la cuenca la unidad hidrológica natural, debería ser obvio este enfoque, no existe claridad en la planificación y menos en la implementación; por ejemplo, el ordenamiento territorial que es clave para valorar, potencial y priorizar los bienes y servicios ambientales de la cuenca, casi no se aplica.

- Hay ausencia o debilidad del marco institucional y legal generalizados sobre los cuales operar los mecanismos de PSA. Esto es más grave cuando existe una centralización de los servicios, como ocurre con frecuencia con el agua potable. En la mayoría de los casos, se ha tenido que recurrir a ordenanzas municipales para darle un poco de sustento legal y permitir la implementación de mecanismos locales de PSA. Por ejemplo, pocas veces es claro el marco organizativo, quién paga, cómo hacer los cobros, cómo se paga, quién certifica los SA?.

- Existe una necesidad creciente de información (investigación) en aspectos fundamentales como los siguientes:

    - o ¿Cuáles son los usos de la tierra y dónde se localizan, el tipo de cobertura vegetal y el manejo de los ecosistemas y agroecosistemas que más favorecen la generación de determinados servicios ambientales, como por ejemplo, la regulación hidrológica, la producción hídrica?.

    - o Integrar el *componente social*: ¿Cuáles son los servicios ambientales demandados, quién los demanda, cuál es la manera más eficiente de proveerlos?, o sea, se requiere entender los gustos y necesidades de la sociedad en términos de servicios ambientales, luego es necesario el análisis económico para medir la demanda.

    - o Existe muy poca información sobre *valoración socioeconómica real* de los servicios ambientales que prestan diferentes ecosistemas y agroecosistemas en las cuencas hidrográficas. Por ejemplo, es necesario investigar la forma en que los diferentes actores valoran los bosques y sus servicios ambientales y cuáles son los mecanismos y cantidades de compensación que puedan ser utilizadas. Este tipo de información es necesaria para el apoyo a la toma de decisiones que ayuden al desarrollo de políticas e instrumentos de mercado. En el caso de agroecosistemas es necesario la valoración económica, en términos del costo de oportunidad, de cambios de uso de la tierra, y la implementación de tecnologías y prácticas agrosilvopecuarias que ayudan a generación de servicios ambientales. La valoración debe ser integral, considerando los costos y beneficios económicos, sociales y ambientales, incluyendo los costos de transacción, que generalmente son muy altos, lo que se ha convertido en una limitación importante para la implementación de sistemas PSA en América Central.

   - *Monitoreo y línea base.* No existen los protocolos definidos de quién y cómo se debe monitorear la calidad ambiental, ni sobre cuál es la línea base que nos permite determinar si se están generando los servicios ambientales por los que se supone se está pagando, o se va a pagar.

- Parece demasiado optimista pensar en que el PSA tenga impacto en reducir la pobreza y contribuir a la equidad socioeconómica en las cuencas. Existe tendencia a que las iniciativas actuales benefician más a grandes propietarios de tierra y mucho menos a los pequeños propietarios y grupos indígenas.

- El problema de derechos de propiedad (muchos ocupantes de tierras sin un título) crea una dificultad real de operacionalizar sistemas de PSA. Por ejemplo la inseguridad de comunidades aguas arriba por los convenios de PSA que deben firmar, ya que con frecuencia existen conflictos por la tenencia y uso de las tierras.

- En términos económicos, la generación de servicios ambientales en la cuenca no garantiza la existencia de un mercado. Además de proveedores, es necesario tener demandantes de los servicios que estén dispuestos a pagar por ellos. Los mecanismos a utilizar dependerán de cada situación en particular y el monto del pago depende de la disponibilidad a pagar de los receptores de los servicios como de aceptación de los propietarios de las tierras que brindan los servicios ambientales en las cuencas. Evidentemente hay un desbalance entre el interés de los que quieren vender servicios ambientales y los que pueden pagarlos. La idea sería que en las cuencas rurales se produzcan servicios ambientales, pero no está nada claro quién los quiere y puede pagar. En la práctica funcionan mejor los mecanismos de compensación ambiental mediante arreglos entre organizaciones.

- También es un asunto de pobreza, cultura de pago y prioridades. La pobreza extrema de los habitantes en muchas microcuencas de la región, imposibilita implementar sistemas justos de compensación económica por parte de los que reciben los servicios ambientales (aunque tengan voluntad de pago), a los que generan esos servicios. También es cierto que si bien la mayoría de la población reconoce la importancia de los servicios ambientales, falta una cultura de pago, principalmente por servicios que no se perciben directamente, como belleza escénica, conservación de la biodiversidad, fijación de carbono, producción de oxígeno. La situación cambia en función de la prioridad del servicio ambiental; es obvio que cuando se trata de un recurso vital que afecta directamente los intereses públicos y privados, individuales y colectivos, como el caso del agua, los demandantes del servicio ambiental hídrico de buena calidad, cantidad y disponibilidad, están dispuestos a realizar sacrificios económicos.

- Aunque podría ser más eficiente o más barato proveer servicios ambientales a través de sistemas de producción y usos de la tierra ambientalmente positivos, que también podrían ser rentables en producción agrícola, casi no existen políticas nacionales que permitan ordenar y viabilizar esta alternativa. Esto es más relevante si se toma en cuenta que, con frecuencia, en América Central, la producción agropecuaria es el principal uso de la tierra y la principal fuente de ingresos en zonas de recarga hídrica.

4. ¿Dónde están las mayores potencialidades para el PSA en cuencas hidrográficas en América Central?

A la luz de la situación de las cuencas hidrográficas en la Región, de sus recursos naturales y el ambiente, de las características sociales y económicas de la población y del entorno nacional e internacional, es importante identificar dos escenarios donde el PSA tiene gran interés, expectativas y potencialidad en cuencas hidrográficas.

El primero es seguir la inercia del proceso que ya se ha venido dando, asociado al servicio ambiental hídrico en cuencas estratégicas, esto es: cuencas que utilizan el agua para producción hidroeléctrica, cuencas que abastecen de agua para consumo directo a la población, cuencas cuyas aguas se utilizan en la industria (por ejemplo agua embotellada y refrescos, beneficiado de café, etc.), cuencas cuyas aguas se utilizan para la irrigación (distritos de riego, grandes irrigantes, etc.) y que requiere fortalecerse e impulsarse vigorosamente; nadie discute que la cuenca es la unidad natural de planificación y gestión territorial para ese fin; tampoco hay dudas de la importancia creciente del agua como recurso estratégico y vital para la sociedad. Sin embargo, es necesario que:

a) Las empresas privadas y públicas que utilizan agua para generación hidroeléctrica (por ejemplo en Costa Rica hay más de 20 proyectos hidroeléctricos y en Nicaragua 9 identificados) paguen los montos justos, no solamente por la protección de las zonas de recarga aguas arriba de los embalses en las cuencas que utilizan, sino también por aquellos usos de tierra y prácticas agrosilvopecuarias en esos espacios territoriales, que favorecen la regulación hidrológica, la baja erosión y el bajo arrastre de sedimentos y contaminantes hasta los embalses y turbinas de generación eléctrica. Pero también es necesario no perder de vista que las interrelaciones de la cuenca como sistema, imponen también la necesidad de pensar y actuar en la cuenca, aguas abajo de los embalses; de lo contrario, es un abordaje parcial de una situación que requiere atención integral. Además, en algunos casos, los aportes reales de algunas de las pocas empresas que realizan algún pago para propietarios que brindan el servicio ambiental hídrico, es más una estrategia propagandística, que un aporte real y justo para compensar el valor que tiene para el flujo y la regulación hídrica el uso de la tierra por el que se está compensando.

b) Las empresas que utilizan el agua de las cuencas para la industria del agua embotellada (agua, refrescos, cerveza, etc.), una de las industrias de mayor crecimiento y más lucrativas actualmente en América Central, paguen montos justos, como PSA por la conservación, protección y manejo por estas cuencas (manantiales) de donde se abastecen. Por el momento, prácticamente no existe esta compensación y cuando la hay, es por montos risibles en comparación con las ganancias que genera esta actividad industrial. Por ejemplo, en Costa Rica, el precio de un litro de agua embotellada es más de 4400 veces mayor, que el de un litro de agua tomado del grifo o cañería, y es una de las industrias de mayor crecimiento, entre 15 y 30% anual (Castro[3], 2003).

c) Los beneficiarios directos (léase propietarios de fincas o parcelas) en los distritos de riego en la Región, así como grandes irrigantes o grupos de pequeños productores, aporten recursos económicos, bajo el mecanismo de PSA, para compensar a los propietarios de terrenos en las cuencas de las cuales se deriva el agua para riego o que alimentan embalses que se utilizan para riego, por la protección de la cobertura vegetal o por aplicar usos de la tierra conservacionistas que favorecen la recarga y regulación hídrica, que ayudan a la buena calidad del agua, y simultáneamente, a generar otros servicios ambientales complementarios al SA hídrico. En este caso puede ser importante considerar el tamaño de la cuenca, ya que en algunos casos, podría ser poco viable implementar mecanismos de PSA por uso de agua para riego en cuencas pequeñas.

d) La mayor voluntad y disponibilidad real de pago por el SA hídrico ocurre en las microcuencas que abastecen de agua para consumo humano. Sin embargo, la falta de políticas nacionales y de marcos legales, así como centralización de los servicios de agua de consumo humano, limitan la operacionalización de esquemas de PSA en estas microcuencas. Aún con esas limitaciones, son las cuencas con mayor potencial para establecer mecanismos de PSA.

El segundo escenario de potencialidad de PSA en cuencas en América Central, es muy importante y necesario, pero bastante idealista. Consiste en el PSA en microcuencas, principalmente rurales, con intervención antrópica, a través de la agricultura, la ganadería, la forestería y otros usos de la tierra, incluso las microcuencas semiurbanas y urbanas, cuando hay posibilidad de revegetación. La mayoría de cuencas de América Central están en este grupo. Un elemento fundamental de un mecanismo de pago por servicios ambientales en este contexto, es el que reconoce el esfuerzo que el productor realiza, a través de la implementación de tecnologías y prácticas de producción agrosilvopecuarias y de conservación y protección de los recursos naturales que producen o contribuyen a generar servicios ambientales. La gran interrogante es quién paga por esos servicios ambientales.

Para estas microcuencas no se evidencia, a corto plazo, una viabilidad fuerte de establecer mecanismos de PSA, puesto que sería principalmente las colectividades a diferentes niveles: municipios, estado central, cooperación internacional, empresas públicas, etc. los beneficiarios (demandantes) de esos servicios ambientales y consecuentemente, los que deberían de asumir el pago. Un caso particular es cuando estas microcuencas, además de ser agrícolas, su recursos hídricos tienen algún uso estratégico como los mencionados en el primer caso (hidroelectricidad, industria, etc.); en ese caso, es factible desarrollar mecanismos de PSA por actividades agrosilvopecuarias conservacionistas que contribuyan a mejorar principalmente el recurso hídrico, cuando el mismo tiene un uso estratégico que hace viable el cobro y el pago correspondiente.

Otra observación frecuente es que pagar por implementar prácticas y tecnologías amigables con el ambiente, la conservación y gestión adecuado de los recursos naturales, es un incentivo perverso, puesto que por principios, por valores, por actitudes, por su propio beneficio y el de sus descendientes, el productor debería hacerlo. Sin embargo, es evidente que por razones complejas, entre ellas la vulnerabilidad socioeconómica y biofísica de América Central, esto no ha sido posible y las cuencas continúan cada día aumentando su estada de degradación, por lo que se debería de recurrir a incentivar económicamente a los productores que realizan un manejo adecuado de los recursos naturales, aún si esto para algunos constituya un incentivo perverso.

Para poder motivar al proveedor de los servicios a conservar los recursos naturales, el PSA no puede ser concebido como un apoyo complementario (caso de Ley Forestal de Costa Rica), sino que el agricultor debe percibir el mismo nivel de ingreso que si produjera rentablemente su finca. De lo contrario no es viable para pequeños productores con fincas de 1 ó 2 ha, además que los costos de transacción son altos.

En conclusión, es evidente que no se pueden resolver todos los problemas de manejo de los recursos naturales mediante el mecanismo de PSA. En América Central las mayores expectativas en cuencas hidrográficas están asociadas al PSA de alcance más local, como es el servicio ambiental hídrico, pero es necesario orientarlo bajo un enfoque sistémico de la cuenca, donde se integre el manejo de otros recursos naturales como el suelo y la vegetación, que por una parte son necesarios para generar el SA hídrico, y por otra parte, permiten generar otros servicios de ámbito más global y de una escala geográfica más amplia o más flexible, tales como la belleza escénica, la fijación de carbono y la mitigación del cambio climático. Este enfoque permitiría también integrar beneficiarios, actores locales y globales en el pago de los servicios ambientales, así como diferentes niveles de amplitud geográfica: desde microcuencas hasta grandes cuencas.