Chapter 3
Ecosystem Services in Agricultural Landscapes

Sean M. Smukler, Stacy M. Philpott, Louise E. Jackson, Alexandra-Maria Klein, Fabrice DeClerck, Leigh Winowiecki, and Cheryl A. Palm

Introduction

There is a tenuous relationship between the world's rural poor, their agriculture, and their surrounding environment. People reliant on farming for their livelihood can no longer focus on current food production without considering the ecosystem processes that ensure long-term production and provide other essential resources required for their well-being. Farmers are now expected to not only produce food, but also steward the landscape to ensure the provisioning of drinking water, wood products for construction and cooking, the availability of animal fodder, the capacity for flood attenuation, the continuity of pollination, and much more. Farmer stewardship of the landscape helps ensure ecological functions that, when beneficial to human well-being, are referred to as ecosystem services. Human activities strongly affect ecosystem services and there is often a resulting trade-off among their availability, which frequently results in the loss of many at the expense of few, most notably when producing food (Foley et al. 2005).

Ecosystem services have been categorized into four basic groups which include supporting services such as soil formation, photosynthesis, and nutrient cycling that are essential for providing provisioning services such as fodder, food, water, timber, and fiber production; regulating services that affect agricultural pests, climate, floods, disease, wastes, and water quality; and cultural services that provide recreational, aesthetic, and spiritual benefits (Fig. 3.1) (MA 2005a; Daily and Matson 2008). While provisioning services provided by agriculture have been and promise to remain a primary gateway for overcoming poverty, the dynamics of how to achieve poverty alleviation are changing rapidly with the declining availability of ecosystem services (MA 2005b; Witcover et al. 2006; Sachs 2008).

S.M. Smukler (✉)
Tropical Agriculture Program, The Earth Institute, Columbia University,
61 Route 9W, Lamont Hall, Room 2H, Palisades, NY 10964-8000, USA
e-mail: sas2242@columbia.edu

J.C. Ingram et al. (eds.), *Integrating Ecology and Poverty Reduction: Ecological Dimensions*, DOI 10.1007/978-1-4419-0633-5_3,

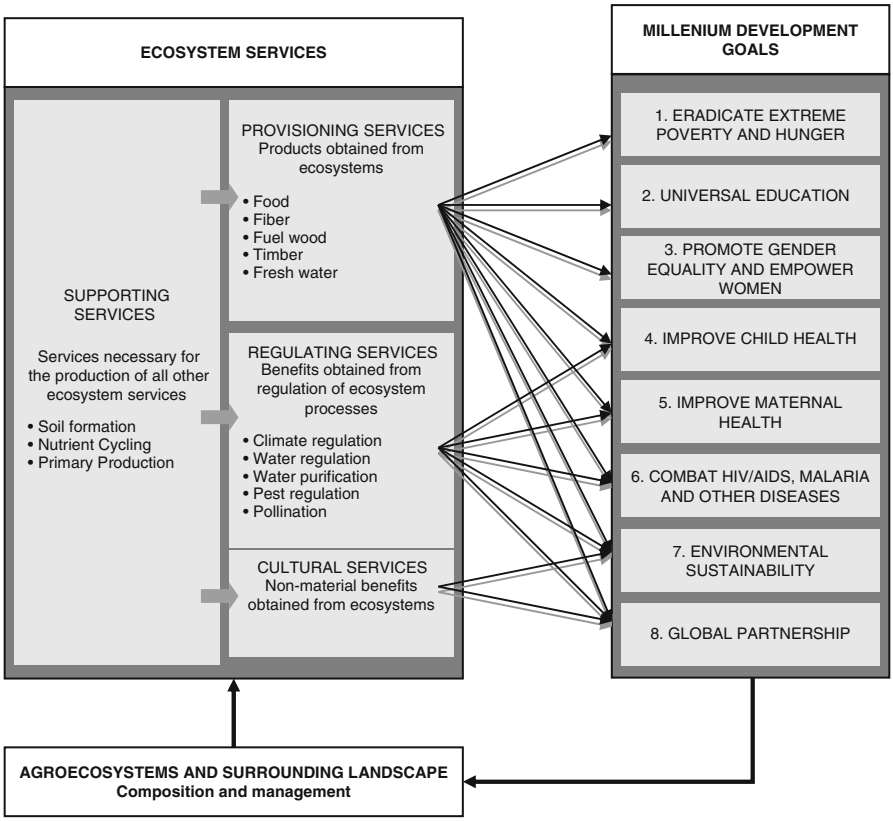

Fig. 3.1 Ecosystem services provided by agricultural landscapes and their connection to the Millennium Development Goals (Modified from the MA 2005a, b)

The strategy of using agriculture as the primary means for economic development has been successful primarily because of the availability and consumption of ecosystem services, many of which were supplied by natural ecosystems (Tilman et al. 2002; Evenson and Gollin 2003; Semwal et al. 2004; Robertson and Swinton 2005). For example, a farm's current yield may be largely dependent on nutrients supplied by thousands of years of plant decomposition and cycling (supporting services) prior to agricultural production and irrigation using water originating in natural watersheds (provisioning services). The availability of ecosystem services has both direct and indirect links to economic development and can be considered a form of capital. The processes by which this "natural capital" can be sustainably used to support the livelihoods of impoverished people requires an understanding of ecological and socioeconomic processes, particularly its links to other forms of capital, such as: (1) human capital: skills, knowledge, health, ability to labor and to pursue livelihood strategies; (2) financial capital: savings, livestock, supplies of credit, remittances; (3) physical capital: basic infrastructure (e.g., transport, energy, market

access, communication, irrigation); and (4) social capital: membership of groups, networks, access to wider institutions of society (Bebbington 1999). These forms of capital are intrinsically interconnected in how they contribute to management of agricultural landscapes, but here we focus mainly on natural capital and more specifically on the ecosystem services that define this capital.

It is clear that around the world natural capital is diminishing and the availability of ecosystem services provided by natural ecosystems is becoming more uncertain as populations and urban development expand, and farmers seek new or better land for agriculture to meet increasing food demands (Bennett and Balvanera 2007; Srinivasan et al. 2008). Already farmers in many regions of the world are no longer able to expand into forests, wetlands, and savannahs to produce or gather food and fiber, and many can no longer expect to rely on the resources provided by the remaining intact natural ecosystems (DeFries et al. 2007). To exacerbate the situation, climate change is adding to the variability in availability of ecosystem services and will dramatically impact natural capital over the coming century (Schmitz et al. 2003; Rosenzweig et al. 2004). While those with financial means are able to purchase necessities such as cooking fuel, timber or even clean water produced in distant locations, the most impoverished do not have this option. They are restricted to utilizing what they can, where they are. With recent loss of natural ecosystems, looming threats of climate change, and rapid expansion and intensification of agricultural production using non-renewable resources, ecosystem services must be better managed within farmed landscapes to ensure the well-being of the poor.

Land management decisions are often framed as a dilemma that involves a trade-off between agricultural productivity and the preservation of natural ecosystems (Balmford et al. 2005; Green et al. 2005). However, the apparent trade-off between agricultural productivity and other ecosystem services is an over-simplification of management choices available; farmers may be able to manage for both (Foley et al. 2005; Perfecto and Vandermeer 2008). Some have questioned the validity of achieving win–win situations for both conservation and agricultural benefits and argue that the best way to preserve nature is to intensify or maximize agricultural productivity (Balmford et al. 2005; Green et al. 2005). Intensification may spare land from agricultural conversion but it by no means ensures the preservation of ecosystem services, especially when it is based on inputs of synthetic fertilizers and pesticides that have adverse effects on biota and resources in adjoining natural areas (Perrings et al. 2006). It is therefore critical that any effort towards agricultural intensification include goals beyond crop and livestock yields.

The objective of this chapter is to give agricultural, environmental, and development practitioners a basic understanding of the many ecosystem services provided by agricultural landscapes, their relationship to poverty alleviation, and a brief introduction to some of the ways to manage for optimum availability using a variety of strategies at multiple scales. In addition, the long-term consequences of environmental degradation for the rural poor will be highlighted (Dasgupta 2007). We have selected a series of examples that demonstrate the relationship between ecosystem services and poverty alleviation in agricultural landscapes. We also recognize that there is much yet to learn about this relationship and try to illustrate the areas in need of more research.

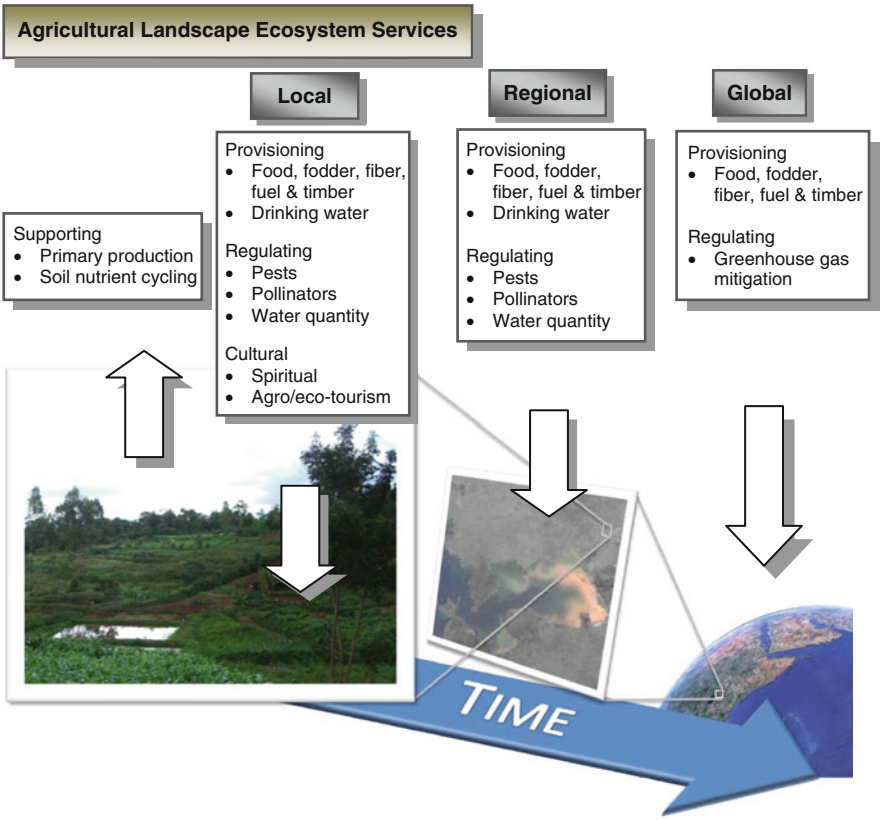

Fig. 3.2 Farm management when aggregated across a landscape can impact not only the ecosystem services available at a local scale but also the services available at regional and global scales. Here runoff from a farm in a watershed of Lake Victoria in western Kenya can negatively affect water quality and fisheries (image provide by Keith Shepherd). Practices such as agroforestry, if adopted widely, could reduce erosion, improve water quality, and mitigate greenhouse gas emissions but at time scales that may be beyond the farmer's season to season financial horizon

As agricultural landscapes are managed by numerous farmers, the overall supply of ecosystem services provided is dependent on aggregated practices, thus we discuss management decision made at scales ranging from the farm field, to the farmscape (the area of land managed by a farmer including the non-production areas), and the landscape. The aggregated results of farmers' management decisions impact the availability of ecosystem services not only for themselves but for people in distant places or in the distant future (Fig. 3.2). Our intent is to illustrate the origins and main beneficiaries of ecosystem services to show the various potential links of management decisions to poverty alleviation. In the first section of the chapter we discuss how agroecological management strategies at the field and farmscape scales can directly improve human well-being through increased agricultural production

and other ecosystem services such as soil health, nutrient cycling, pest regulation and crop pollination. In the second part we describe how many of the same management practices designed to increase local benefits when scaled up to the landscape can benefit regional water quality and global greenhouse gas (GHG) mitigation. Because the adoption of services with regional or global impacts may not have immediate economic benefits we introduce the role of payments for ecosystem services (PES) for poverty alleviation. Finally, we discuss the need for conserving biodiversity and assessing the potential trade-offs and synergies of ecosystem services across scales.

Modes of Agricultural Intensification

There is growing recognition that agricultural intensification must entail ecologically sustainable management. This goes beyond past efforts to improve the living standard of the world's rural poor, e.g., by increasing agricultural productivity using improved crop varieties and inputs such as inorganic fertilizers, biocides and irrigation (Matson et al. 1997; Tilman et al. 2002). While agricultural intensification strategies reliant on high inputs increased food production in Latin America and Asia (Evenson and Gollin 2003), this "Green Revolution" strategy was often at a cost to other ecosystem services and their societal impacts continue to be debated (Kiers et al. 2008). Moreover, these strategies were not universally successful and in many regions, such as sub-Saharan Africa, they were largely not adopted (Smaling 1993; Smaling et al. 1993; Stoorvogel et al. 1993; Sanchez et al. 2009).

Agricultural intensification that has focused on provisioning of agricultural products has often resulted in severe impacts to other ecosystem services because of inappropriate use of irrigation, tillage, and fossil fuel-based industrial farm inputs such as pesticides and fertilizers. Such intensification has contributed to degradation of 1.9 billion ha impacting some and 2.6 billion people, the withdrawal of 70% of global freshwater (2.45% of rainfall) for irrigation (IAASTD 2008), biocide effects on health of humans and other organisms (Maumbe and Swinton 2003; Atreya 2008), the massive coastal anoxic zones (Prepas and Charette 2007), and biodiversity loss that has exceeded historical rates (Tilman et al. 2001; Cassman et al. 2003; MA 2005b; Robertson and Swinton 2005). These impacts, while prominent in the developed world, are often exacerbated in the developing world where institutional recognition of environmental impacts and agricultural extension services are limited, safety equipment is prohibitively expensive and regulation is often minimal (Dasgupta et al. 2002; Atreya 2008). Furthermore, industrial inputs, most of which are reliant on fossil fuels for production and delivery, are susceptible to increasing fluctuations in availability and price, which could negatively impact cost–benefit ratios and thus farmers' livelihoods as seen in recent food crises (Gomiero et al. 2008).

There is a growing recognition that increasing agricultural productivity at the expense of other ecosystem services is not a sustainable means of alleviating poverty.

M.S. Swaminathan, one of the founders of the "Green Revolution," has called for a new "Ever-Green Revolution" that encompasses the principles of ecology, economics, and social and gender equality (Swaminathan 2005). The United Nations has taken up this charge and on the eve of the millennium, in their commitment to halve poverty by 2015, included environmental sustainability as one of the eight goals designed to be road markers for meeting their objective. The eight Millennium Development Goals (MDGs) are an interdisciplinary task list; each task carefully selected to help reduce poverty. While there is much debate as to the best strategies to achieve the MDGs, it is likely the inextricable link between each MDG and the availability of ecosystem services (Fig. 3.1) has been underestimated as their role is ignored in many of the current poverty alleviation approaches (MA 2005b). Lack of conclusive evidence linking ecosystem services to human well-being is often cited as a reason for this neglect particularly in regions where the poor are most reliant on them. Meanwhile 60% of the earth's ecosystem services continue to be degraded or used unsustainably (MA 2005b).

Increasing Ecosystem Services for On-Farm and Local Benefits

Many of the one billion people who earn less than a dollar a day are reliant on agriculture for survival. The needs of these farmers are largely dependant on the success of very specific farm practices. Farmers often grow food for personal consumption on small pieces of land, and are typically challenged by uncertain land tenure, little access to capital for investments, labor constraints, no available agricultural extension, and limited access to markets (Sachs 2005). In this context it may be difficult for farmers to think about managing resources beyond their farm. We will examine how utilizing ecological principles for management at multiple scales to intensify on-farm production can improve the availability of ecosystem services for farmers adopting such practices.

Agroecological Management from the Field to the Farmscape

The aim of agroecological management strategies on and around the farm is to focus beyond immediate crop yields to increase the production of ecosystem services through efficient use of resources to create a farming system that is profitable over time without compromising future well-being (Altieri 2002; Ananda and Herath 2003; Witcover et al. 2006). Ecological intensification of farm management can include incorporating a diversity of species to fill available openings or "niches" and can help ensure critical supporting and regulating services such as soil formation, pest control, and nutrient cycling required for sustaining provisioning services. Essentially this corresponds to the idea of increasing the agricultural use and conservation of biodiversity (Jackson et al. 2007).

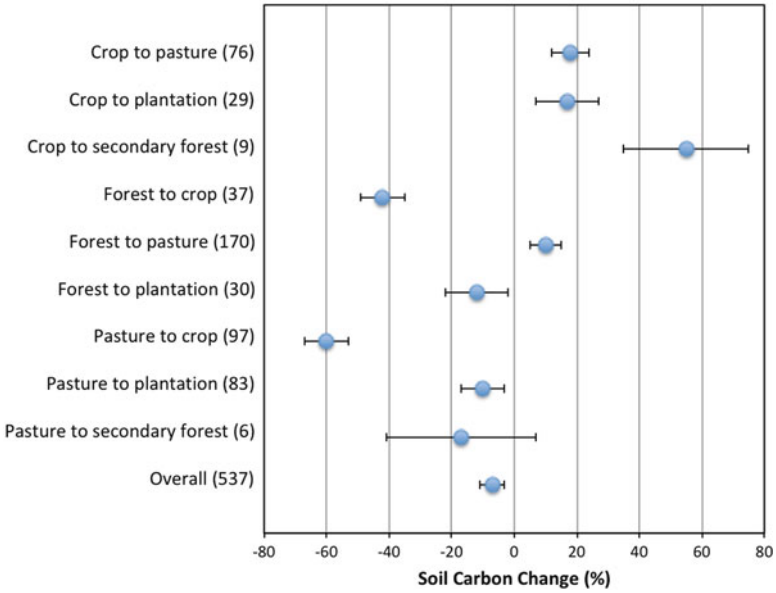

Fig. 3.3 Changes in soil organic carbon (SOC) an indicator of SOM resulting from changes in land management. This is from a study that reviewed 537 cases of land use changes. The bars indicate the 95% confidence intervals and the number of observations is shown in parentheses (Modified from Guo & Gifford 2002)

Supporting Services Example: Building Healthy Soil

Many of the ecosystem services provided by agricultural landscapes are contingent upon the status and management of soil, e.g., tillage negatively impacts soil aggregate stability affects and thus water infiltration rates (Cannell and Hawes 1994). Thus, collective management of on-farm soil properties can mediate the landscape's potential for supporting services underpinning numerous other ecosystem services (Fig. 3.1). Low soil fertility is a basic and immediate constraint limiting farmers' ability to meet consumption needs at the household level (Lal 2006; Sanchez et al. 2009). Identifying and improving the soil's ability to provide a favorable growing environment for crops is an important step in restoring ecological function and increasing food security, production, and income for smallholder farmers.

Converting natural ecosystems to simplified agroecosystems and maintaining a constant state of disturbance through annual cultivation and management causes drastic changes in soil conditions, including the decline of soil organic matter (SOM) (Fig. 3.3). Maintaining or increasing SOM may be the single most important management strategy designed to improve supporting services.

The addition of even small quantities of SOM can have substantial benefits for the biological, chemical, and physical properties of the soil (Stevenson and Cole

1999). Soil organic matter consists mainly of soil organic carbon (SOC), and provides the substrate required by bacteria to mineralize nitrogen and other nutrients stored in complex molecules (e.g., carbohydrates, amino acids, lignin) that would otherwise be unavailable to plants. SOM is a critical component of sustained nutrient cycling and has numerous other benefits such as increasing soil water holding capacity, infiltration rates, nutrient storage capacity, aggregation and reducing bulk density (Lal 2006). Continual cropping without sufficient organic matter inputs leads to depletion and unproductive soils resulting in reduced net primary productivity (NPP) or the rate an ecosystem accumulates energy or biomass. Eventually continued depletion causes a negative feedback loop where the landscape can no longer support the plants needed to provide the organic matter required for their growth (McDonagh et al. 2001).

Once soils have been degraded, improving their functions requires substantial active management and a clear understanding of the driving factors that led to degradation (Whisenant 1999). Using nitrogen fixing plants in agricultural fields can begin to rehabilitate soils. Alternatively inorganic fertilizers can be used to increase crop productivity and the amount of residue that can be returned to the soil (Palm et al. 2007). Incorporating residues can increase SOM resulting in a positive feedback loop, where yields continue to increase, and soil becomes responsive to further inputs. For example every 1 Mg ha^{-1} increase in the SOC pool in the rooting zone has been shown to increase wheat yields by 20–70 kg ha^{-1}, rice yields by 10–50 kg ha^{-1} and maize yields by 30–300 kg ha^{-1} over un-amended soils (Lal 2006).

Although inorganic fertilizer use may be an important tool for jump starting substantially degraded systems and increasing crop productivity, fertilizer use does not come without substantial trade-offs. The most important trade-off may be financial, as the use of inorganic fertilizer does not necessarily translate into increased livelihoods. Inorganic fertilizers are expensive, particularly for smallholder farmers, and their price may be increasingly volatile (Sogbedji et al. 2006; Gomiero et al. 2008; Huang et al. 2009). In 2008, for example, worldwide inorganic fertilizer prices rose as much as 200%. Furthermore, smallholder farmers living in impoverished regions may have limited access to inorganic fertilizers due to lack of infrastructure, capital or financing (Sachs 2005). The excessive or inappropriate use of inorganic fertilizers can also lead to substantial reductions in other ecosystem services. Therefore, developing locally derived sources of nutrients from plants and animals used instead of, or in combination with inorganic fertilizers may help farmers achieve greater economic returns over time and promote various ecosystem services.

Filling every available niche with carefully selected plants and animals can help balance nutrient losses from agricultural systems. Selecting appropriate plants for different niches can increase the availability of organic matter and retains nutrients otherwise lost through erosion and leaching (Fig. 3.4). To increase organic matter inputs, farmers may improve water management to increase NPP, incorporate woody biomass (trees or shrubs) within the cropping system (i.e., agroforestry), use cover crops to minimize erosion, green manures (nitrogen fixing legumes), mulch, or increase animal manure inputs. Once organic matter production is increased, combining inorganic fertilizers and organic materials can become even more effective in restoring fertility to nutrient-poor soils than either alone. Research in sub-Saharan

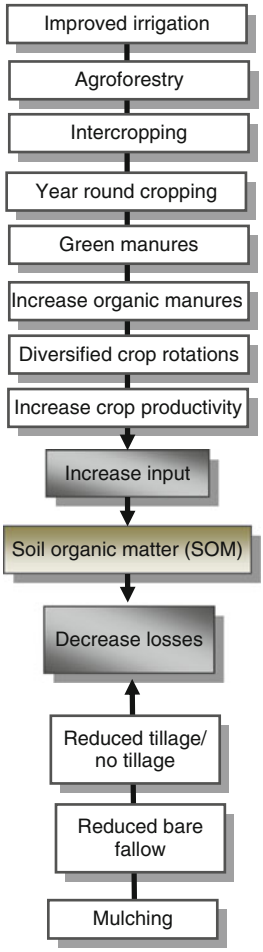

Fig. 3.4 Management practices that can either increase organic matter inputs across the agricultural landscape or reduce nutrient losses (Modified from Neider and Benbi 2008)

Africa shows that the application of inorganic fertilizers on soil amended with manure increased grain yields compared to plots to which just inorganic fertilizers or just manure was added (Kimani and Lekasi 2004). In another long-term study, maize yields in Kenya were compared along a climatic and amendment gradient; demonstrating the benefits of incorporating a combination of high-quality manure, crop residue, and inorganic fertilizers (Okalebo et al. 2004). Research on combining inorganic and organic soil amendments in smallholder systems is not new, but its current focus on the timing of nutrient availability, interactions between inorganic and organic sources, and the residual benefits to the system over the long-term is critical (Giller 2002; Vanlauwe et al. 2002; Drinkwater and Snapp 2007).

Utilizing plants that fix their own nitrogen (legumes) through symbiotic relationships with micro-organisms associated with their root system can also increase

overall nitrogen capital of the site. Legumes can be planted as annuals or perennial either in the field as a cover, relay crop, or intercrop or in the field margins as a hedgerow. Estimates for N fixation from legumes range from 58–120 kg N ha^{-1} $year^{-1}$ for annuals to 228 kg N ha^{-1} $year^{-1}$ for perennial legumes such as alfalfa (Nieder and Benbi 2008). A meta-analysis of over 94 studies of maize yields in sub-Saharan Africa reports that while yields increased most significantly using inorganic fertilizer applications (2.3 t ha^{-1}), yields also increased by 1.6 t ha^{-1} using coppiced woody legumes, 1.3 t ha^{-1} using non-coppiced woody legumes, and 0.8 t ha^{-1} using herbaceous green manure legumes (Sileshi et al. 2008). These authors concluded that the combination of inorganic fertilizer and organic matter inputs increased efficiency of fertilizer use, as there were no significant differences in crop yield between organic inputs combined with 50% vs. 100% of the recommended fertilization rate in 48 studies with paired treatments (Sileshi et al. 2008). This study suggests that the use of organic inputs could increase the efficiency of inorganic fertilizer use by at least 50%.

In addition to increasing organic matter inputs, management practices that reduce erosion can also have substantial benefits for soil nutrients. Clearing land for production has resulted in erosion rates 1–2 orders of magnitude greater than rates of erosion in natural ecosystems systems (Montgomery 2007). Generally, management that reduces the amount of soil disturbance or the impacts of disturbance, are likely to improve nutrient retention on the site (Lal 2004a). To decrease nutrient losses through erosion and leaching, farmers can eliminate or limit soil disturbance by reducing the amount of tillage. They could also fill the niche opened by tillage during land preparation or harvesting by mulching or planting non-production crops such as improved fallows, cover crops, or green manures. Mulching with crop or agroforestry residues can protect otherwise bare soil from dislodging due to rainfall impact or wind (Roose and Ndayizigiye 1997). Cover crops also protect soil and have been shown to reduce nitrate losses as the roots scavenge the soil for any residual nutrients which can then be incorporated and made available for subsequent production crops (Jackson et al. 1993; Wyland et al. 1996; Sileshi et al. 2008).

Many of the practices just described are done so in the context of maintaining supporting services, mainly nutrient cycling required for primary productivity. Unfortunately, many of these practices generally introduce complexity and increased labor to the production system without necessarily increasing immediate economic returns. This creates an obvious draw towards the use of simpler methods relying solely on inorganic fertilizer inputs that promise immediate income even though the alternative integrated soil fertility management approach may have greater long-term financial benefits. In addition, it may not be apparent to those on such short time horizons that the soil provides the support services critical for the provisioning of many other ecosystem services, including ensuring a buffering capacity for environmental perturbations caused by climate change (e.g., resilience to drought through increased soil moisture capacity) (Pimentel et al. 2005). Therefore education, through demonstration and outreach or even short-term economic incentives, will help promote the adoption of soil management practices key to ensuring long-term agricultural productivity and well-being (Table 3.1).

Table 3.1 The importance, challenges, and strategies for securing provisioning ecosystem services key to rural well-being such as food, fiber, fuel, and timber production

Provisioning service	Importance	Challenges	Strategies
Food and fiber	Food security Independence Textiles Income	Biophysical constraints: water and nutrients Farm size Land tenure Labor	Identify most profitable agroecological crop management techniques Increase access to: markets, seeds, and soil amendments Diversify production Develop cooperatives
Fuel	Means to cook food Provide heat	Inefficiency of stoves Distance to available wood Labor	Improve stove efficiency Increase fuel resources Develop alternative fuel sources
Timber	Construction materials Carbon sequestration in biomass Litterfall inputs	Incorporation into farming system without impacting crops Land tenure Farm size Labor	Identify optimal design and management of agroforestry system

Provisioning Services Examples: Providing Food, Fiber, Fuel, and Timber

Small-holder farmers trying to escape poverty must find a way move beyond subsistence to produce goods and services that are profitable. For the rural poor their ability to profit is often constrained by limited availability of capital for investment in equipment, inputs, and land, limited access to markets both for sales and purchasing of inputs, and lack of infrastructure for irrigation or extension. While the focus of "Green Revolution" agricultural development on increasing the productivity of a few high yielding crops has had great success increasing the caloric output of the farm, it has not ensured profit, nutrition, or other provisioning services that the landscape may provide (Evenson and Gollin 2003; Kiers et al. 2008). These services can include timber, fodder, fuel wood, medicine, and many other products that are of direct use to farming communities or marketed for income generation. One approach for maximizing these services would be to fill all available niches with useful plant or animal species, capitalizing on synergistic interactions whenever possible.

Filling niches by managing farmland for multiple species or agroforestry has been promoted as a means of maximizing production potential for small landholders. Agroforestry systems improve soil conditions (Young 1997; Sanchez 1999), provide habitat for species (Somarriba et al. 2003; Schroth et al. 2004; Harvey and Villalobos 2007), mitigate against global climate change (Montagnini and Nair 2004;

Verchot et al. 2005), and provide alternative sources of income (Sanchez 1999; Alavalapati et al. 2004). Agroforestry systems is defined as multiple cropping of at least two plant species that interact biologically where one of the plant species is a woody perennial and at least one of the plant species is managed for forage, annual or perennial crop production and thus, can provide food, fuel, fiber, and timber resources (Somarriba 1992). Ecological interactions in agroforestry systems can have both positive (soil fertility improvement, improved nutrient cycling, carbon sequestration) and negative affects (shading, allelopathy) (Kohli et al. 2008); therefore, careful, ecologically based management that maximizes synergies is essential to ensure the greatest benefits at multiple scales.

There are numerous examples of the success of agroforestry systems to maintain ecosystem services from around the world, particularly in regions where there has been a strong infrastructure for research and extension (Graves et al. 2004). Remnant trees in otherwise treeless pastures in the highlands of Costa Rica provision food for birds and animals (90% of the tree species), timber (37%), firewood (36%), and fence posts (20%) (Harvey and Haber 1999). A spatial evaluation of agroforestry systems in Talamanca, Costa Rica demonstrated that cacao agroforestry systems can mimic the structural diversity of native tropical forests (Suatunce et al. 2003). Farmer interviews in Talamanca demonstrated that species incorporated into their agricultural landscapes had numerous ecosystem functions, for example fiber, timber, and fruit (Córdova et al. 2003; Somarriba and Harvey 2003). If farmers use their land as sources of fuel, fiber, and timber, the encroachment into native forests is minimized.

Agroecological Management from the Farmscape to the Landscape

Farmers can benefit directly from ecosystem services other than supporting and provisioning services by managing beyond the production field. There are a number of regulating services that farmers can manage that will directly enhance their well-being by maintaining or increasing crop productivity or the availability of other resources. These services can include the regulation of climate, water for drinking and irrigation, pollination and pest reduction. Most of these services require farmers to manage the farmscape and the surrounding landscape. Filling available niches around and between fields with a diversity of plants selected for multiple uses or maintaining already existing patches of natural vegetation can create windbreaks that generate favorable micro-climates for crops (Seck et al. 2005), maintain hydrologic processes that ensure water availability (e.g., percolation into ground water) (Rockström et al. 2004), reduce the prevalence of weeds (Bokenstrand et al. 2004), and provide habitat for pollinators and other beneficial organisms (Tscharntke et al. 2008). The management of areas of the agricultural landscape past the edge of the production field can result in a complex landscape mosaic of diverse vegetation types.

Regulating Services Example: Maintaining Pollinators

Recently, the importance of pollinators as a key element of agricultural diversity supporting human livelihoods has been increasingly recognized. The great majority of plant species benefit from pollination provided by bees, flies, birds, bats, and other taxa. For example, 90% of flowering species in tropical rainforests rely on animal pollination (Bawa 1990) and 75% of the most important crop species benefit from pollination services (Klein et al. 2007) accounting for €153 billion annually (Gallai et al. 2009). As much of the diversity of marketable horticultural products originates from developing countries (Aizen et al. 2008), crop diversification and pollinator-crop interactions could play a key role in poverty alleviation.

Agricultural intensification in many parts of the world has necessitated managed pollination services, as wild pollinators are not available in sufficient numbers to ensure crop yield (Potts et al. in press). At the global scale, populations of managed honey bees have not increased at the same rate as the proportion of agricultural crops that depend on pollinators (Aizen and Harder 2009). Hence, the demand for pollination services is likely to outstrip current honeybee hive numbers in many areas of the world (Aizen and Harder 2009). Because wild bees may be able to provide insurance against ongoing honeybee losses (Winfree et al. 2007), the demand for wild pollination is increasing. Concerns about the delivery of future pollination services are high for tropical countries as habitat isolation affects pollinators in tropical landscapes (Ricketts et al. 2008).

Managing pollination services solely reliant on the European honeybee is a risky strategy because it relies on single species management. Almond pollination in California relies on the European honey bee and oil palm pollinations in South-East Asia depend on a single imported African beetle (Kremen et al. 2002). Recent studies highlight the need to promote biodiversity to improve resilience in the provisioning of pollination services by buffering pollination against asynchronous annual fluctuations in bee abundance (Kremen et al. 2002; Winfree et al. 2007; Klein 2009). Different pollinating species also occupy different spatial, temporal, and conditional niches in which only a diversity of pollinator groups will lead to high quality and quantity services (Hoehn et al. 2008; Klein et al. 2008, 2009; Blüthgen and Klein 2011). These facts suggest pollinator diversity must be protected or restored across agricultural landscapes to ensure pollination services under various conditions and across space and time.

Local (farmscape) alterations that impact pollination services include changes in the abundance, diversity, distribution, and temporal continuity of floral resources (Klein et al. 2002; Greenleaf and Kremen 2006; Potts et al. 2006; Holzschuh et al. 2007; Williams and Kremen 2007) and availability of nesting sites and materials (Shuler et al. 2005). Therefore, local management practices must include planting and preservation of year-round foraging resources (e.g., understory blooming herb, flower strips), nesting resources (e.g., open ground, dead wood, and whole twigs), and should avoid agro-chemical use during pollinators' times of activity (Brittain et al. 2010).

Management recommendations should include possible risks and trade-offs or dis-services that might be promoted through the establishment of pollinator-friendly

management practices. These include the promotion of pest species or the repelling of pollinators from crop species by planting flower resources (pollinators attracted to the flowering resources planted rather than to the crop species of interest). It is especially important to collect information and train local people to shift human labor from providing hand pollination services (e.g., passion fruit pollination in Brazil) to the management of pollinator-friendly agricultural practices.

The conservation of natural or semi-natural habitats that provide additional pollinator resources is essential to promote pollination (Kremen et al. 2002; Ricketts et al. 2008). These natural or semi-natural habitats are often large in area, which seems to be important for many rare pollinating species. Therefore, pollination management practices should be promoted not only at the farmscape but also at landscape scales. These practices include natural habitat protection and creating stepping stones or corridors to connect agriculture and bee (semi- and natural) habitats of high quality. The complexity of managing habitats across a landscape suggests the coordination by local farmers and organizations may be critical for maintaining wild pollination services and sustaining the continued health pollinators.

Regulating Services Example: Managing Agricultural Pests

Many of the strategies used to promote soil and pollinator functions can also be employed to increase pest regulation services. The importance of local and landscape factors in pest regulation services has long been a topic of ecological studies. In theory, strategies that limit density and diversity of herbivores and increase density and diversity of predators should enhance pest regulation services, but relationships between farm management, pests, and predators are complex (Tscharntke et al. 2005; Clough et al. 2010). In fact, crop diversity, landscape complexity, and predator diversity all have important implications for provisioning of pest regulation services within agroecosystems.

Diversity of crop and non-crop plants within agroecosystems affects pest regulation. Theory on the relationship between crop diversity and pest regulation hypothesizes that natural enemies are more diverse in polycultures because alternative prey or other food resources are available ("enemies hypothesis") and that prey populations are lower in polycultures where locating host plants is difficult ("resource concentration hypothesis") (Root 1973). Subsequently Andow (1991) reviewed >200 studies on pests, predators, and crop diversity finding that the majority of herbivore species (52%) had larger population sizes in monocultures than polycultures; only a few herbivore species (15.3%) were more abundant in polycultures. A number of different strategies such as habitat diversification (e.g., intercropping, use of cover crops and trap crops, allowing weed growth, crop rotation, inclusion of perennial plants), agroforestry with a diverse, dense mix of trees as or in addition to crop plants, and organic management (e.g., elimination of pesticides, and organic matter addition) all contribute to maintenance of diversity and density of natural enemies within agroecosystems leading in some cases to enhanced pest regulation (e.g., Altieri 1999; Rao et al. 2000; Östman et al. 2001; Eilers and Klein 2009).

Increased vegetation complexity in agroforests, for example, can harbor greater abundance and diversity of insectivorous birds enhancing pest control services (Perfecto et al. 2004; Van Bael et al. 2008). However, plant diversity may not always benefit pest regulation if, for example, trees serve as alternate hosts for pests, compete with crops for water or nutrients making crops more susceptible to pest problems, or limit the movement of olfactory cues that attract natural enemies to pest species (Rao et al. 2000).

Landscape context is extremely important in determining the strength and persistence of pest regulation within agricultural systems. Placement of non-crop habitats such as hedgerows, windbreaks and early succession fallow areas at edges of crop fields can enhance predation by providing overwintering sites for natural enemies, facilitating dispersal, and providing habitat for alternative hosts of parasitoids of crop pests and necessary food resources like pollen, and nectar (Altieri 1999; Rao et al. 2000; Tscharntke et al. 2005; Bianchi et al. 2006; Zhang et al. 2007). Non-crop habitats act as sources for intensively managed agricultural systems to maintain biodiversity of natural enemies (Tscharntke et al. 2005; Eilers and Klein 2009). At larger spatial scales, maintaining a balance between the proportion of crop and non-crop vegetation may be critical as diversity and abundance of some natural enemies declines far away from natural habitats (Perfecto and Vandermeer 2002). Similarly, a greater degree of landscape complexity can enhance natural enemy density and specifically increase predator activity, fecundity, oviposition rate, predation area, and condition of natural enemies, and can reduce pest pressure (Bianchi et al. 2006). In some cases, complex habitats with smaller crop fields relate to slow establishment of pests (Östman et al. 2001); however, complex habitats and landscapes may also benefit populations of pests that subsequently invade crops (Bianchi et al. 2006). Finally, both vertical intensification (large numbers of trees in same area) and horizontal intensification (great expanses planted with the same species) may increase pest outbreaks (Rao et al. 2000). Thus, diversified landscapes can enhance pest control services when they result in higher natural enemy diversity and density of enemies that colonize crop fields, reduce pest densities, reduce damage levels, and increase crop yields keeping in mind possible trade-offs in maintaining complex landscapes (Bianchi et al. 2006).

Finally, natural enemy diversity may actually enhance or hinder pest control, depending on the exact context examined. The combined effects of predators depend on two main groups of factors: (1) degree of complementarity in the assemblage of predators and parasitoids; and (2) the occurrence of predation or competition among predators and parasitoids of pests. In assemblages with a high degree of complementarity, there may be behavioral or diet differences among predator species that mean that multiple predator species are more effective than single predator species for controlling pests (Cardinale et al. 2003; Tscharntke et al. 2005; Snyder et al. 2006). For example, Cardinale et al. (2003) found synergistic effects of two ladybird beetles and one parasitoid species on aphid pests of alfalfa, and cascading effects on alfalfa yield, such that their effects were greater with all three natural enemies present than expectations based on their individual effects. Likewise, others have found additive effects of multiple predator species controlling cabbage aphids (Snyder et al. 2006). In other situations, however, multiple predators performed

worse then each predator alone due to the prevalence of intraguild predation (one predator consuming another instead of feeding on herbivore species among the predator assemblage) (Finke and Denno 2005). Thus, the exact relationship between natural enemy diversity and pest suppression and crop benefit may depend on predator and parsitoid species composition, and the type of agroecosystem examined.

Managing Agricultural Landscapes for Regional and Global Ecosystem Services

Many practices that maximize ecosystem services that directly benefit farmers have impacts far beyond farm borders. Soil management, irrigation, use of pesticides, inorganic fertilizers, fuel, and non-production areas all can affect ecosystem services at the regional or even global scale (Tilman et al. 2001; Prepas and Charette 2007; Smith et al. 2008). From the perspective of those living in distant locations the impact of individual farms may seem small and irrelevant but collectively can be dramatic. Although the impacts may not necessarily directly affect the farmers that manage them the impacts are indirectly connected to their livelihoods and the alleviation of poverty. We focus on two of the most obvious ecosystem services resulting from management of agricultural landscapes that have impacts at multiple scales: the regulation of water and GHG emissions. One example shows how agricultural practices regulate quality and quantity of water from upland to downstream users to the coastal communities (Fig. 3.1, Table 3.2) (Prepas and Charette 2007). Another example shows how management of nutrients across the agricultural landscape collectively results in GHG reductions on a scale that influences global atmospheric conditions. In both of these examples the impacts most likely will not be directly or immediately felt. Water scarcity and climate change will be chronic, slowly developing problems that will invariably disproportionally affect the poor (Parry et al. 2004; Srinivasan et al. 2008). It is important to recognize that the agroecological management practices that benefit regional and global population may be challenging for poor farmers to adopt for a number of reasons (van Noordwijk et al. 2002). It is, therefore, in the best interest of the recipients, particularly those in rich countries, to see themselves as stakeholders in the management of these potentially distant agricultural landscapes and support policy and actions that promote practices that maximize ecosystem services and alleviate poverty.

Regional Regulating Service Example: Protecting Water Quality and Availability

The amount of rainfall intercepted by agricultural landscapes and the immense volume of water applied as irrigation makes agricultural production one of the most important modifiers of water-related ecosystem services. More than 15% of the

Table 3.2 Summary of agroecological management practices and their associated ecosystem service

		Ecosystem services											
		Local benefits									Regional benefits		Global benefits
Management practices		Soil sustainability	Food and fiber	Fuelwood and timber	Drinking water quality	Genetic resources	Pollination	Pest regulation	Irrigation availability	Cultural	Water regulation	Water purification	Climate regulation
Crop management	Cover crops	X	X		X	X	X	X	X		X	X	X
	Relay crops	X	X		X	X	X		X		X	X	X
	Green manure	X	X		X	X	X		X		X	X	X
	Crop rotations	X	X		X	X	X	X	X		X	X	X
	Reduced tillage	X	X		X			X	X		X	X	X
	Perennial cropping	X	X	X	X	X	X		X		X	X	X
	Mulching	X	X		X			X	X		X	X	X
	Composting	X	X		X			X	X		X	X	X
Agroforestry	Hedgerows	X	X	X	X	X	X	X		X	X	X	X
	Intercropping	X	X	X	X	X	X	X		X	X	X	X
	Wood lots	X	X	X	X	X	X	X		X	X	X	X

(continued)

Table 3.2 (continued)

Management practices		Ecosystem services											
		Local benefits									Regional benefits		Global benefits
		Soil sustainability	Food and fiber	Fuelwood and timber	Drinking water quality	Genetic resources	Pollination	Pest regulation	Irrigation availability	Cultural	Water regulation	Water purification	Climate regulation
Livestock	Improved grazing management	X	X		X	X				X	X	X	X
	Restoration of grasslands	X			X	X				X	X	X	X
	Fire management	X	X		X			X			X	X	X
Infrastructure	Terracing	X	X	X	X				X		X	X	X
	Ponds	X	X						X		X	X	
	Biogas digesters	X		X									X
	Improve cook stoves	X		X	X					X			X

water that runs across the global terrestrial surface area flows in or through cultivated landscapes and agriculture accounts for 70% of global water withdrawals (MA 2005a, b). This interaction with agriculture affects a number of water-related ecosystem functions, including water availability, water quality and flood attenuation, all of which impact local human well-being and regional populations. Ensuring an adequate supply of clean water for both human consumption and the maintenance of natural habitat will be among the most significant global challenges of this century (MA 2005a, b). More than 1.1 billion people do not have access to clean water (WHO/UNICEF 2004), yet already 5% to possibly 25% of water use exceeds the long-term accessible supply of global freshwater (MA 2005a, b; see the chapters on water in this volume). As demand increases, water scarcity will inevitably result in problems for food production, human health, economic development, and biodiversity (Postel et al. 1996; Rockström et al. 2004). In addition, the degradation of water quality from agricultural runoff further threatens the viability of freshwater and coastal ecosystems and people dependant on coastal fisheries for their livelihood (see chapter on Fisheries, Vol. 2).

One aspect of human well-being that can be directly measured is human health, which is tightly linked in a number of ways to water availability and quality (see the chapters on Health, this volume). Nitrate losses from fields into waterways and aquifers may adversely impact human health. For example, high nitrate concentrations in drinking water have been attributed to methemoglobinemia (blue baby syndrome) and impaired immune response (Fewtrell 2004). Due to a lack of reliable data and difficulties of proper analysis, the extent of high nitrate levels in drinking water is unknown but suspected to surpass the 10 ppm nitrate limits set by the World Health Organization (WRI 1989) in many regions or the world. High nitrate levels can also cause toxic algal blooms (toxic cyanobacteria) associated with chronic disease such as liver cancer (WCD 2000). Irrigation ditches, canals, rice fields, and reservoirs harbor and spread vector-borne diseases, particularly in tropical regions where Rift Valley Fever and Japanese encephalitis occur (WCD 2000). For people with little access to medical care, poor water management can threaten their very survival.

Impaired water quality in aquatic habitats also indirectly affects human well-being. Movement of soil, nutrients, and pesticides from agricultural fields to adjacent waterways has already caused massive reduction of ecosystem services. The physical impact of rainfall hitting bare soil dislodges soil particles and can result in sealing and runoff of soil particles, nutrients, and pesticide residues. Likewise, irrigation can dislodge soil, nutrients, and pesticides affecting water quality and flow. Runoff of nitrogen from terrestrial to aquatic ecosystems has doubled from 111 million tons per year in pre-industrial times to between 223 and 268 million tons per year (Galloway et al. 2004). Inputs of nitrogen (N) and phosphorus (P) into aquatic ecosystems stimulate primary production including algal growth, thereby consuming much of the water's dissolved oxygen. This depletion of oxygen causes eutrophication, kills fish in streams, rivers, lakes, and coastal regions, and leads to substantial economic impacts to communities reliant on fisheries

(Prepas and Charette 2007). For example, in Lake Victoria, eutrophication threatens fisheries and the livelihoods of rural poor living around the lake (Prepas and Charette 2007).The impacts of pesticide runoff are not well understood as their release into the environment often creates synergistic or unanticipated effects. Pesticide runoff negatively affects humans (Rola et al 1993; Polidoro and Bosque-Perez 2007; Luo and Zhang 2009) and non-humans alike (Hayes et al. 2002; Jarrard et al. 2004).

Strategies to improve water-related ecosystem services in agricultural landscapes might be as simple as changing crop management practices (e.g., growing cover crops) or extremely involved (e.g., constructing detention ponds with irrigation return capabilities). In general, water-related ecosystem services can be improved by efficient use of inputs (i.e., nutrients, pesticides and irrigation), limiting bare soil, reducing the speed or volume of irrigation water or runoff, or by capturing and retaining runoff. Agroecological management strategies for increasing local ecosystem services also serve to increase water-related services that ensure the availability of clean water on a larger scale.

Beyond the agricultural fields or pastures, management of water-related ecosystem services may require more investment in time, labor, and money since strategies may not show obvious benefits for farm production, or may require longer time horizons or complex construction. However, basic practices could include protecting waterways by filling niches and covering the soil surrounding the farm fields with organisms that can utilize excess nutrients or pesticides or slow the movement of water. Planting vegetation along pathways that lead to water bodies is effective at capturing sediments and phosphorus (Uusi-Kamppa et al. 2000). Grassed waterways, once mature, can reduce the flow rate of runoff and cause sediment to drop out of suspension before reaching aquatic systems (Maass et al. 1988). One study found 92% of sediment, and 71% of nutrients contained in irrigation runoff was reduced in the first 4 m of a grass filter strip (Blanco-Canqui et al. 2004). Another showed that an 8 m buffer of grasses decreased nitrate loads by 28% and a 16 m buffer decreased loads by up to 42% (Bedard-Haughn et al. 2004). More complex buffer zones can be created by planting woody vegetation either along waterways or even adjacent to fields as hedgerows. The deep roots of hedgerow plantings of trees and shrubs reach niches in the soil where nutrients otherwise untapped by shallow rooted annual crops would leach from and enter water bodies via sub-surface flow (Wigington et al. 2003). Riparian buffers that included switch grass and woody plants removed 97% of the sediments, 94% of the total nitrogen, 85% of the nitrate, 91% of the total phosphorus, and 80% of the phosphate in runoff (Lee et al. 2003).

Alternatively, existing wetlands could be used to filter agricultural runoff and can remove as much as 59% of the total phosphorus, 38% of the total nitrogen, and 41% of the total organic carbon (Jordan et al. 2003). Practices such as planting and maintaining woody vegetation or infrastructure development may not be adopted without financial capital, as there is little incentive for farmers lacking financial capital to make the large investment required to ensure their success, and this will be discussed in more general terms below.

Global Regulating Service Example: Mitigating Climate Change

Global climate change resulting from GHG emissions is another major upcoming challenge that will inevitably and disproportionately impact the poor (Srinivasan et al. 2008). Impoverished people reliant on agricultural production will be highly susceptible to fluctuating temperatures and rainfall, movement of disease and pests, and many other anticipated results of climate change (MA 2005b). Some of the agroecological management strategies described here have potential to help farmers adapt to climate impacts and to mitigate climate change. Agriculture directly contributed 13.5% of total global GHG emissions in 2004, and when the additional 17.4% of total emissions from deforestation (much of which is for farming and grazing) are considered, managing to reduce these emissions represent a major pathway for mitigation (Smith et al. 2008).

Agriculture-related emissions result from deforestation caused by extensification, use of pesticides and inorganic fertilizers, livestock production, fuel use for mechanized production including irrigation pumping, and transportation of goods. While these practices may be a sizeable proportion of the global emissions they also represent ways that emissions could be reduced if alternative practices are adopted. Although policy discussions are largely focused on carbon dioxide, other more powerful GHGs can also be mitigated through changes in agricultural practices. Agriculture accounts for about 60% of the nitrous oxide and 50% of the methane emitted globally by anthropogenic sources in 2005 (Smith et al. 2007). Much of the emissions from agriculture landscapes, like other nutrient losses, indicate inefficiency in management practices.

Emissions from agricultural landscapes can be categorized into primary, secondary, or tertiary emissions (Lal 2004b). Primary emissions are those due to mobile operations (e.g., tillage, harvesting) that are not largely used in small-holder farming reliant on manual labor for most operations. Secondary emissions result from stationary sources (e.g., crop drying, irrigation pumping) that may be more important for small-holder operations. Tertiary emissions are a result of the equipment manufacture, construction of farm buildings and acquisition of raw materials (Lal 2004b). The adoption of agroecological management strategies may help limit emissions at each of these stages (Fig. 3.5).

Agroforestry practices, i.e., woody perennial shrubs and trees in hedgerows significantly increase carbon storage across the landscape (Smith et al. 2008). The potential of agroforests to sequester carbon varies widely depending on soil type, climate, and rainfall. Values for biomass sequestration rates in agroforestry systems can range from 0.29 to 15.21 Mg C ha^{-1} $year^{-1}$ (Nair et al. 2009). Reforesting sites that are marginal for crop production could sequester between 0.8 and 18.8 Mg C ha^{-1} $year^{-1}$ depending on site conditions and management. (Richards and Stokes 2004) While both agroforestry and reforestation may substantially increase carbon storage, this may come at a cost to ecosystem services directly useful to the farmers (e.g., fuel wood, timber or food production). Alternatively, managing cropland soils for carbon may increase the mitigation potential and enhance crop production.

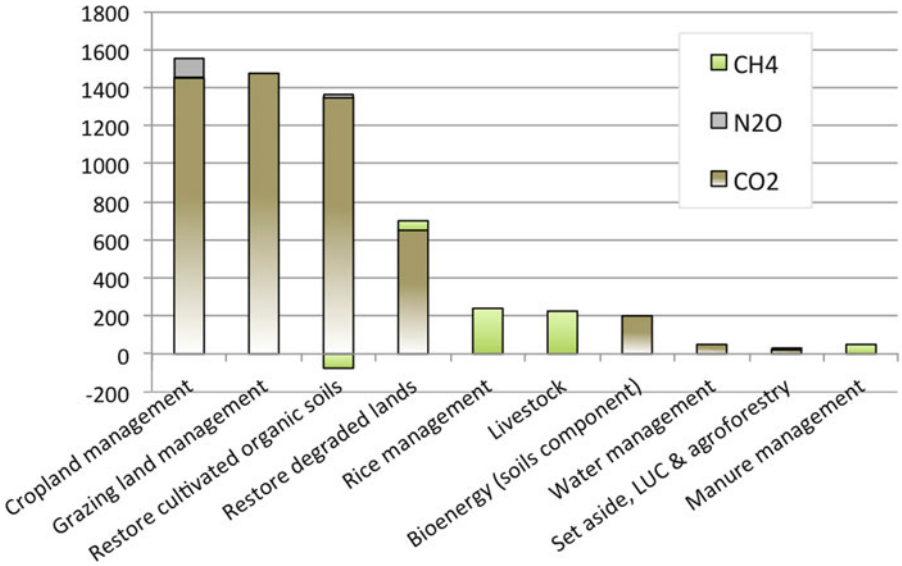

Fig. 3.5 The technical global mitigation potential of various agricultural management practices for nitrous oxide (N_2O), methane (CH_4), and carbon dioxide (CO_2) (Source Smith et al. 2007)

Reducing tillage and increasing residue inputs could sequester on average 0.2–0.7 Mg C ha^{-1} $year^{-1}$ or improved management of grazing lands could sequester 0.1–0.8 Mg C ha^{-1} $year^{-1}$ depending on climate zone (Smith et al. 2007). Managing soils for carbon sequestration, i.e., increasing SOM, may seem the most promising means of mitigating GHG emission in agricultural landscapes, but if additional benefits to farmers can materialize, then efforts to provide incentives should weigh the trade-offs in terms of labor and yield costs in the short-term.

Conservation of Biodiversity to Provide Ecosystem Services

Agrobiodiversity refers to the biological resources (genes, species, and habitats) that contribute to food, agriculture, and human well-being in agricultural landscapes (Qualset et al. 1995). It serves as a source of adaptation for crops and livestock, and in a larger context, for transformation to new production systems under unknown future environmental conditions. It also encompasses the biodiversity in natural ecosystem fragments and the organisms that move across the mosaic of ecosystems within agricultural landscapes (Jackson et al. 2007). For the poor, agrobiodiversity is one of the greatest sources of natural capital, especially in the form of traditional varieties of crops, medicinal plants, and useful wild species, e.g., that may be moved

into home gardens (Jarvis et al. 2008). Evaluating the actual value of agrobiodiversity is difficult to assess, given the lack of knowledge on ecosystem functions of most of the biodiversity in agricultural landscapes, and the discrepancy between private vs. social benefits derived from it (Pascual and Perrings 2007).

The conservation of biodiversity can be managed for at multiple scales across the landscape and is thought to be critical for many other services that can benefit poor farmers, besides the provisioning services generated by crop genetic resources. In Daily's (1997) pioneering work on ecosystem services, she defines these services as "the conditions and processes through which natural ecosystems and the species that make them up, sustain and fulfill human life." There is increasing evidence that ecosystem services are regulated by ecological communities yet global trends indicate that biodiversity losses are resulting in losses in ecosystem functioning (Flynn et al. 2009; Jackson et al. 2009). There is a growing recognition of a strong relationship between the biodiversity of an agricultural system or landscape and the provisioning of services (Chan et al. 2006), and the willingness of the some consumers (especially in the developed world) to pay for biodiversity friendly products (Wendland et al. in press). A small, though increasing number of studies demonstrate that conserving both natural and semi-natural elements in agricultural landscapes can make significant contributions to biodiversity conservation. Harvey et al. (2006) found that the abundance of bird, beetle, tree, and butterfly diversity could be conserved by increasing tree cover and that even minimal number of trees on pastures can enhance biodiversity (Daily and Ehrlich 1995; Harvey and Haber 1999; Ricketts 2004; Schroth et al. 2004). Likewise, maintaining forest cover in agricultural landscapes, and minimizing the distance between forest fragments can make significant contributions to conserving wild biodiversity and their related services (Daily and Ehrlich 1995; Ricketts 2004). Management of these patches can also contribute to the provisioning of ecosystem services, particularly pollination and pest control (Schroth et al. 2004), and carbon storage (Smukler et al. 2010).

Petit and Petit (2003) demonstrated that although many species are conserved in managed landscapes, few of these are species of conservation concern. Flynn et al. (2009) used a metric of functional diversity, where species were classified not by their taxonomy or evolutionary relationships, but rather were classified by their contributions to ecosystem functions (insectivores, frugiviores, omnivores, canopy species, leaf gleaners, etc.) and found that functional diversity was lost more rapidly than species richness. This work suggests that the loss of biodiversity may have important consequences for the provisioning of ecosystem services.

There are several overarching ecologically based management strategies that can be used to guide development practitioners and land managers. These are outlined by McNeely and Scherr (2003) who provide six general recommendations: (1) create biodiversity reserves that also benefit local farming communities; (2) develop habitat connectivity in non-farmed areas; (3) reduce or reverse the conversion of natural ecosystems to agriculture by increasing farm productivity; (4) minimize agricultural pollution; (5) modify management of soil, water, and vegetation resources; and (6) modify farming systems to mimic natural ecosystems.

Tradeoffs and Synergies

The current situation of diminishing natural areas and the loss of biodiversity with its associated ecosystem services is coupled with increased demands for food production. This forces farmers to make difficult management choices that can have serious long-term consequences for economic prosperity of their farming operations, as well as broader impacts on ecological and human well-being (Jordan et al. 2007; Scherr and McNeely 2008). The decisions farmers make will inevitably incur trade-offs in the availability of ecosystem services. At the same time some decisions could actually have synergistic effects on multiple ecosystem services (Robertson and Swinton 2005; Swallow et al. 2009). Converting forests to agriculture is obviously a trade-off between forest goods and services for food (Fig. 3.6). In a landscape where trees and food are produced together, such as in multistrata agroforestry systems, important synergies can be achieved. Planting trees in agricultural landscapes may enhance water quality and quantity, provide habitat for pollinators and beneficial insects, store carbon, and improve long-term agricultural productivity.

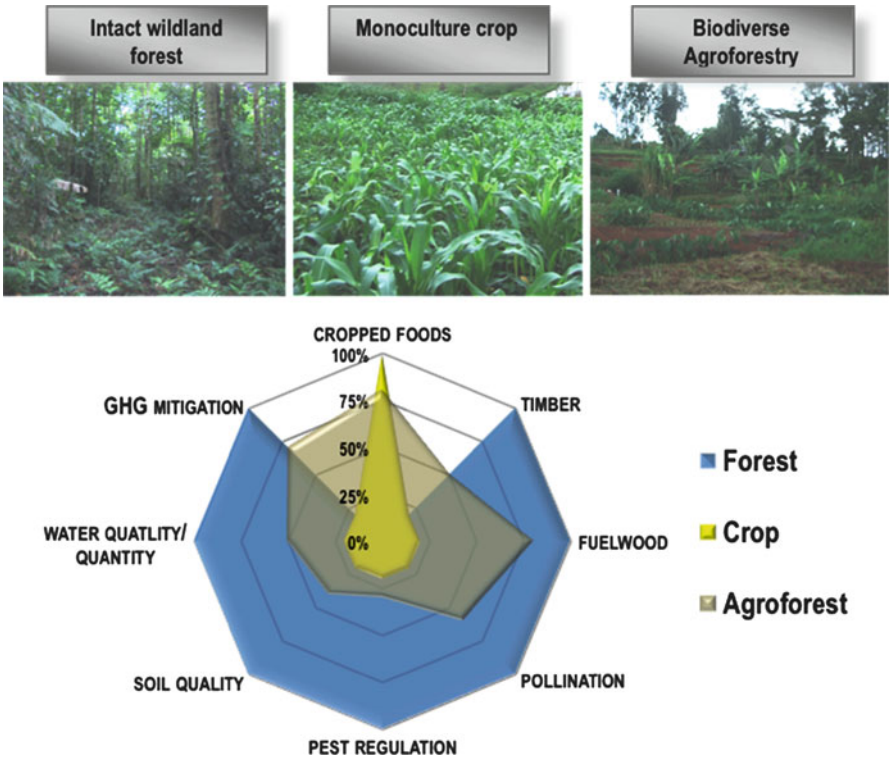

Fig. 3.6 A hypothesized comparison of the trade-offs in ecosystem services from a biodiverse intact wildland forest landscape, a monoculture farmed landscape, and a biodiverse farmed landscape (Modified from Foley 2005). The relative availability of ecosystem services could be compared as percentages of a reference landscape as indicated here

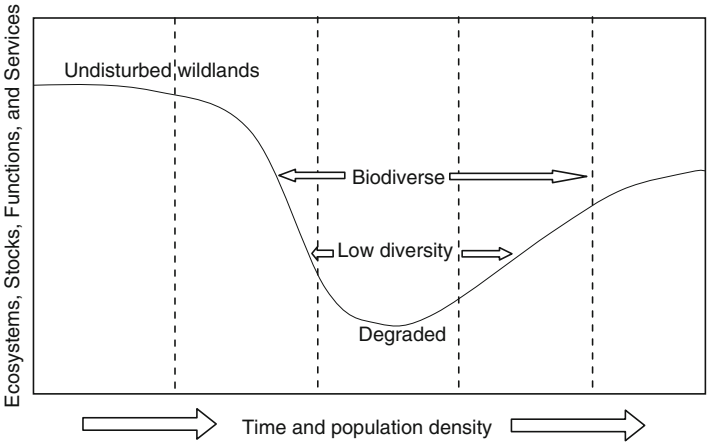

Fig. 3.7 A theoretical degradation and rehabilitation U-shaped trajectory curve. The possible relative degradation status of three categories of agricultural production, biodiverse, low diversity and degraded agricultural landscapes. Over time as agricultural production becomes more simplified (e.g. less biodiverse) due to intensification, ecological theory suggests that ecosystem stocks, functions and services are likely to decline. As biodiversity is restored to these systems, the stocks, functions and services may increase but never quite to the original state

Similarly planting trees may incur significant trade-offs in terms of a large short-term cash and labor investment to purchase seeds or seedlings and to ensure survival, or in some cases depending on the species, may actually reduce water availability (Farley et al. 2005).

Tradeoffs and synergies also occur among the communities that actually utilize the services. For example, people living at the top of a watershed may receive the benefits of food provisioning services from agricultural production while the people at the bottom of the watershed suffer from the loss of clean water that would have been provided by an intact forested watershed (Falkenmark and Folke 2002). This type of geographical disconnect between management practices and the availability of ecosystem services has lead to large-scale reduction in these services. Unless these two groups of stakeholders can interact to negotiate management strategies that have mutually beneficial outcomes, one group will continue to benefit from certain services and the trade-off will be loss of services to the other (Lant et al. 2008) Fig. 3.7.

These examples illustrate how evaluating the trade-offs and synergies for a particular management decision is extremely difficult given that many benefits may be felt in geographically or temporally distant places. Maximizing the benefits of ecosystem services requires an effective ecological assessment of the potential trade-offs and synergies of various agricultural land management options at a variety of scales that incorporate all those who could be potentially impacted. The need for an effective ecological assessment is clear but the methodologies for doing so are still in the development stage. To accurately assess the potential trade-offs and synergies requires simultaneous quantification of multiple ecosystem services over long periods of time. Given the difficulties of such research, most studies of ecosystem services are con-

fined to 2 or 3 years and one or two services. To date there are only few examples of projects that have assessed multiple services required to observe trade-offs and synergies let alone generate accurate quantification at a meaningful scale (Gamfeldt et al. 2008; Tallis et al. 2008, Smukler et al. 2010). Currently analysis must rely on complex biophysical process models that still need development and validation for agroecosystems (Tallis et al. 2008). A number of other models and planning tools are being employed to provide financial incentives directly to land managers for the provisioning of ecosystem services in developed countries (Eigenraam et al. 2006), but models directly relevant to farmers in developing countries seem a long way off. Furthermore, in order for assessments to actually influence management decisions, rural communities must be involved through a participatory or economic approach, evaluating and prioritizing their ecosystem service needs. The infrastructure required for such an approach in impoverished regions may be prohibitively expensive and require institutional intervention not currently in place, but the financial promise of PES may help initiate some form of such analysis in these regions.

Conclusions

Increased agricultural productivity through "Green Revolution" technologies, including inorganic fertilizers, improved seed and small-scale irrigation projects will contribute towards meeting the United Nation's goal of halving global poverty by 2015 (Sachs 2005; Denning et al. 2009; Sanchez et al. 2009) but sustained poverty alleviation will require a diversity of interventions. We must manage agricultural landscapes for multiple ecosystem services, or else our efforts to alleviate rural poverty may in fact just transpose dire economic conditions to other regions (e.g., downstream), or even exacerbate poverty in coming generations.

Examining solutions to poverty through the lens of ecology means seeing the situation holistically both spatially and temporally. Ecological research illustrates a number of principles that can be applied to management of agricultural landscapes to ensure the availability of multiple ecosystem services. Management practices that increase soil organic matter, or maximize biodiversity by filling niches with a variety of plants that cycle nutrients and provide habitat for beneficial organisms, are examples of ways to increase natural capital across the landscape that have been effectively demonstrated. Given that many of these practices do not directly increase agricultural productivity or income immediately, novel approaches will be required to convince farmers to adopt them.

The plight of the rural poor cannot be isolated from either regional or global markets, and poverty has to be viewed as inextricably tied to regional and global ecosystem processes. Human well-being is reliant on a complex interconnection between the management of individual farms and the sustained availability of ecosystem services on a much broader scale. Those that recognize this relationship have a responsibility to help forge policy, markets, education, and community outreach required to plan for trade-offs and synergies that will empower impoverished rural

farmers to maximize the availability of ecosystem services that will have long-term benefits for society at large.

References

Aizen, M. A., L. A. Garibaldi, S. A. Cunningham, and A. M. Klein. 2008. Long-Term global trends in crop yield and production reveal no current pollination shortage but increasing pollinator dependency. Current Biology **18**:1572–1575.

Aizen, M. A. and L. D. Harder. 2009. The global stock of domesticated honey bees is growing slower than agricultural demand for pollination. Current Biology **19**:915–918.

Alavalapati, J. R. R., R. K. Shrestha, G. A. Stainback, and J. R. Matta. 2004. Agroforestry development: An environmental economic perspective. Agroforestry Systems **61**:299–310.

Altieri, M. A. 1999. The ecological role of biodiversity in agroecosystems. Agriculture Ecosystems & Environment **74**:19–31.

Altieri, M. A. 2002. Agroecology: the science of natural resource management for poor farmers in marginal environments. Agriculture Ecosystems & Environment **93**:1–24.

Ananda, J. and G. Herath. 2003. Soil erosion in developing countries: a socio-economic appraisal. Journal of Environmental Management **68**:343–353.

Andow, D. A. 1991. Vegetational diversity and arthropod population response. Annual Review of Entomology **36**:561–586.

Atreya, K. 2008. Health costs from short-term exposure to pesticides in Nepal. Social Science & Medicine **67**:511–519.

Balmford, A., R. E. Green, and J. P. W. Scharlemann. 2005. Sparing land for nature: exploring the potential impact of changes in agricultural yield on the area needed for crop production. Global Change Biology **11**:1594–1605.

Bawa, K. S. 1990. Plant-pollinator interactions in tropical rain-forests. Annual Review of Ecology and Systematics **21**:399–422.

Bebbington, A. 1999. Capitals and Capabilities: A Framework for Analyzing Peasant Viability, Rural Livelihoods and Poverty. World Development **27**:2021–2044.

Bedard-Haughn, A., K. W. Tate, and C. van Kessel. 2004. Using nitrogen-15 to quantify vegetative buffer effectiveness for sequestering nitrogen in runoff. Journal of Environmental Quality **33**:2252–2262.

Bennett, E. M. and P. Balvanera. 2007. The future of production systems in a globalized world. Frontiers in Ecology and the Environment **5**:191–198.

Bianchi, F. J. J. A., C. J. H. Booij, and T. Tscharntke. 2006. Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. Proceedings of the Royal Society B **273**:1715–1727.

Blanco-Canqui, H., C. J. Gantzer, S. H. Anderson, E. E. Alberts, and A. L. Thompson. 2004. Grass barrier and vegetative filter strip effectiveness in reducing runoff, sediment, nitrogen, and phosphorus loss. Soil Sci Soc Am J **68**:1670–1678.

Blüthgen, N. and A. M. Klein. 2011. Functional complementarity and specialization: the role of biodiversity in plant-pollinator interactions. Basic and Applied Ecology **12**(4):282–291.

Bokenstrand, A., J. Lagerlof, and P. R. Torstensson. 2004. Establishment of vegetation in broadened field boundaries in agricultural landscapes. Agriculture, Ecosystems & Environment **101**:21–29.

Brittain, C. A., M. Vighi, R. Bommarco, J. Settele, and S. G. Potts. 2010. Impacts of a pesticide on pollinator species richness at different spatial scales. Basic and Applied Ecology **11**:106–115.

Cannell, R. Q. and J. D. Hawes. 1994. Trends in tillage practices in relation to sustainable crop production with special reference to termperate climates Soil & Tillage Research **30**:245–282.

Cardinale, B., C. Harvey, K. Gross, and A. Ives. 2003. Biodiversity and biocontrol: emergent impacts of a multi-enemy assemblage on pest suppression and crop yield in an agroecosystem. Ecology Letters **6**:857–865.

Cassman, K. G., A. Dobermann, D. T. Walters, and H. Yang. 2003. Meeting cereal demand while protecting natural resources and improving environmental quality. Annual Review of Environment and Resources **28**:315–358.

Chan, K. M. A., M. R. Shaw, D. R. Cameron, E. C. Underwood, and G. C. Daily. 2006. Conservation planning for ecosystem services. PLoS Biology **4**:2138–2152.

Clough, Y., S. Abrahamczyk, A. Adams M-O, A , N. Ariyanti, L. Betz, D. Buchori, D. Cicuzza, K. Darras, D. Putra, B. Fiala, S. Gradstein, M. Kessler, A.-M. Klein, R. Pitopang, B. Sahari, C. Scherber, C. Schulze, Shahabuddin, S. Sporn, K. Stenchly, S. Tjitrosoedirdjo, T. Wanger, M. Weist, A. Wielgoss, and T. Tscharntke. 2010. Biodiversity patterns and trophic interactions in human-dominated tropical landscapes in Sulawesi (Indonesia): plants, arthropods and vertebrates. Pages 15–71 *in* T. Tscharntke, C. Leuschner, E. Veldkamp, H. Faust, E. Guhardja, and A. Bidin, editors. Tropical Rainforests and Agroforests under Global Change. Ecological and Socio-economic Valuations. Springer Verlag, Berlin.

Córdova, L. T., E. Somarriba, and C. A. Harvey. 2003. Plantas ütiles en las fincas cacaoteras de indígenas Bribri y Cabécar de Talamanca, Costa Rica. Agroforesteria en las Américas **10**:36–41.

Daily, G. C. 1997. Nature's services: societal dependence on natural ecosystems. Island Pr.

Daily, G. C. and P. R. Ehrlich. 1995. Preservation of biodiversity in small rain-forest patches rapid evaluations using butterfly trapping. Biodiversity and Conservation **4**:35–55.

Daily, G. C. and P. A. Matson. 2008. Ecosystem services: From theory to implementation. Proceedings of The National Academy of Sciences of The United States of America **105**:9455–9456.

Dasgupta, P. 2007. Nature and the economy. Journal of Applied Ecology:475–487.

Dasgupta, S., C. Meisner, D. Wheeler, and Y. H. Jin. 2002. Agricultural trade, development and toxic risk. World Development **30**:1401–1412.

DeFries, R., A. Hansen, B. L. Turner, R. Reid, and J. G. Liu. 2007. Land use change around protected areas: Management to balance human needs and ecological function. Ecological Applications **17**:1031–1038.

Denning, G., P. Kabambe, P. Sanchez, A. Malik, R. Flor, R. Harawa, P. Nkhoma, C. Zamba, C. Banda, C. Magombo, M. Keating, J. Wangila, and J. Sachs. 2009. Input Subsidies to improve smallholder maize productivity in Malawi: Toward an African Green Revolution. PLoS Biology **7**:2–10.

Drinkwater, L. E. and S. S. Snapp. 2007. Nutrients in agroecosystems: Rethinking the management paradigm. Advances in Agronomy **92**:163–186.

Eigenraam, M., L. Strappazzon, N. Lansdell, A. Ha, C. Beverly, and J. Todd. 2006. EcoTender: Auction for multiple environmental outcomes, Project Final report for National Action Plan for Salinity and Water Quality National Market Based Instruments Pilot Program. Department of Primary Industries, Victoria, Australia.

Eilers, E. J. and A. M. Klein. 2009. Landscape context and management effects on an important insect pest and its natural enemies in almond. Biological Control **51**:388–394.

Evenson, R. E. and D. Gollin. 2003. Assessing the impact of the Green Revolution, 1960 to 2000. Science **300**:758–762.

Falkenmark, M. and C. Folke. 2002. The ethics of socio-ecohydrological catchment management: towards hydrosolidarity. Hydrology and Earth System Sciences **6**:1–9.

Farley, K. A., E. G. Jobbagy, and R. B. Jackson. 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Global Change Biology **11**:1565–1576.

Fewtrell, L. 2004. Drinking-water nitrate, methemoglobinemia, and global burden of disease: A discussion. Environmental Health Perspectives **112**:1371–1374.

Finke, D. L. and R. F. Denno. 2005. Predator diversity and the functioning of ecosystems: the role of intraguild predation in dampening trophic cascades. Ecology Letters **8**:1299–1306.

Flynn, D. F. B., M. Gogol-Prokurat, T. Nogeire, N. Molinari, B. T. Richers, B. B. Lin, N. Simpson, M. M. Mayfield, and F. DeClerck. 2009. Loss of functional diversity under land use intensification across multiple taxa. Ecology Letters **12**:22–33.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global consequences of land use. Science **309**:570–574.

Gallai, N., J.-M. Salles, J. Settele, and B. E. VaissiËre. 2009. Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. Ecological Economics **68**:810–821.

Galloway, J. N., F. J. Dentener, D. G. Capone, E. W. Boyer, R. W. Howarth, S. P. Seitzinger, G. P. Asner, C. C. Cleveland, P. A. Green, E. A. Holland, D. M. Karl, A. F. Michaels, J. H. Porter, A. R. Townsend, and C. J. Vorosmarty. 2004. Nitrogen cycles: past, present, and future. Biogeochemistry **70**:153–226.

Gamfeldt, L., H. Hillebrand, and P. R. Jonsson. 2008. Multiple functions increase the importance of biodiversity for overall ecosystem functioning. Ecology **89**:1223–1231.

Giller, K. E. 2002. Targeting management of organic resources and mineral fertilizers: Can we match scientists' fantasies with farmers' realities? Pages 155–171 *in* B. Vanlauwe, J. Diels, N. Sanginga, and R. Merckx, editors. Integrated plant nutrient management in sub-Saharan Africa: From concept to practice. CABI, Wallingford.

Gomiero, T., M. G. Paoletti, and D. Pimentel. 2008. Energy and environmental issues in organic and conventional agriculture. Critical Reviews in Plant Sciences **27**:239–254.

Graves, A., R. Matthews, and K. Waldie. 2004. Low external input technologies for livelihood improvement in subsistence agriculture. Pages 473–555 Advances in Agronomy, Vol 82.

Green, R. E., S. J. Cornell, J. P. W. Scharlemann, and A. Balmford. 2005. Farming and the fate of wild nature. Science **307**:550–555.

Greenleaf, S. S. and C. Kremen. 2006. Wild bees enhance honey bees' pollination of hybrid sunflower. Proceedings of The National Academy of Sciences of The United States of America **103**:13890–13895.

Guo and Gifford. 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology **8**(4):345–360.

Harvey, C. A. and W. A. Haber. 1999. Remnant trees and the conservation of biodiversity in Costa Rican pastures. Agroforestry Systems **44**:37–68.

Harvey, C. A., A. Medina, D. M. Sanchez, S. Vilchez, B. Hernandez, J. C. Saenz, J. M. Maes, F. Casanoves, and F. L. Sinclair. 2006. Patterns of animal diversity in different forms of tree cover in agricultural landscapes. Ecological Applications **16**:1986–1999.

Harvey, C. A. and J. A. G. Villalobos. 2007. Agroforestry systems conserve species-rich but modified assemblages of tropical birds and bats. Biodiversity and Conservation **16**:2257–2292.

Hayes, T. B., A. Collins, M. Lee, M. Mendoza, N. Noriega, A. A. Stuart, and A. Vonk. 2002. Hermaphroditic, demasculinized frogs after exposure to the herbicide atrazine at low ecologically relevant doses. Proceedings of the National Academy of Sciences of the United States of America **99**:5476–5480.

Hoehn, P., T. Tscharntke, J. M. Tylianakis, and I. Steffan-Dewenter. 2008. Functional group diversity of bee pollinators increases crop yield. Proceedings of the Royal Society B-Biological Sciences **275**:2283–2291.

Holzschuh, A., I. Steffan-Dewenter, D. Kleijn, and T. Tscharntke. 2007. Diversity of flower-visiting bees in cereal fields: effects of farming system, landscape composition and regional context. Journal of Applied Ecology **44**:41–49.

Huang, W. Y., W. McBride, and U. Vasavada. 2009. Recent volatility in U.S. fertilizer prices. Amber Waves : The Economics of Food, Farming, Natural Resources, and Rural America **March**.

International Assessment of Agricultural Science and Technology for Development (IAASTD). 2008. Island Press, Washington, DC.

Jackson, L., T. Rosenstock, M. Thomas, J. Wright, and A. Symstad. 2009. Managed ecosystems: biodiversity and ecosystem functions in landscapes modified by human use.*in* S. Naeem, D. E. Bunker, and A. Hector, editors. Biodiversity, Ecosystem Functioning, and Human Wellbeing: An Ecological and Economic Perspective. Oxford Univ Pr.

Jackson, L. E., U. Pascual, and T. Hodgkin. 2007. Utilizing and conserving agrobiodiversity in agricultural landscapes. Agriculture Ecosystems & Environment **121**:196–210.

Jackson, L. E., L. J. Wyland, and L. J. Stivers. 1993. Winter cover crops to minimize nitrate losses in intensive lettuce production. Journal of Agricultural Science **121**:55–62.

Jarrard, H. E., K. R. Delaney, and C. J. Kennedy. 2004. Impacts of carbamate pesticides on olfactory neurophysiology and cholinesterase activity in coho salmon (Oncorhynchus kisutch). Aquatic Toxicology **69**:133–148.

Jarvis, D. I., A. H. D. Brown, P. H. Cuong, L. Collado-Panduro, L. Latournerie-Moreno, S. Gyawali, T. Tanto, M. Sawadogo, I. Mar, M. Sadiki, N. T. N. Hue, L. Arias-Reyes, D. Balma, J. Bajracharya, F. Castillo, D. Rijal, L. Belqadi, R. Ranag, S. Saidi, J. Ouedraogo, R. Zangre, K. Rhrib, J. L. Chavez, D. Schoen, B. Sthapit, P. De Santis, C. Fadda, and T. Hodgkin. 2008. A global perspective of the richness and evenness of traditional crop-variety diversity maintained by farming communities. Proceedings of The National Academy of Sciences of The United States of America **105**:5326–5331.

Jordan, N., G. Boody, W. Broussard, J. D. Glover, D. Keeney, B. H. McCown, G. McIsaac, M. Muller, H. Murray, J. Neal, C. Pansing, R. E. Turner, K. Warner, and D. Wyse. 2007. Environment - Sustainable development of the agricultural bio-economy. Science **316**:1570–1571.

Jordan, T. E., D. F. Whigham, K. H. Hofmockel, and M. A. Pittek. 2003. Nutrient and sediment removal by a restored wetland receiving agricultural runoff. Journal of Environmental Quality **32**:1534–1547.

Kiers, E. T., R. R. B. Leakey, A. M. Izac, J. A. Heinemann, E. Rosenthal, D. Nathan, and J. Jiggins. 2008. Ecology - Agriculture at a crossroads. Science **320**:320–321.

Kimani, S. K. and J. K. Lekasi. 2004. Managing manures throughout their production cycle enhances their usefulness as fertilisers: A review. Pages 187–197 *in* A. Bationo, editor. Managing nutrient cyles to sustain soil fertility in sub-Saharan Africa. AfNet-CIAT, Nairobi.

Klein, A. M. 2009. Nearby rainforest promotes coffee pollination by increasing spatio-temporal stability in bee species richness. Forest Ecology and Management **258**:1838–1845.

Klein, A. M., S. A. Cunningham, M. Bos, and I. Steffan-Dewenter. 2008. Advances in pollination ecology from tropical plantation crops. Ecology **89**:935–943.

Klein, A. M., C. M. Mueller, P. Hoehn, and C. O. U. P. Kremen, pp. 195–208. 2009. Understanding the role of species richness for pollination services.*in* S. Naeem, D. E. Bunker, A. Hector, M. Loreau, and C. Perrings, editors. The consequences of changing biodiversity – solutions and scenarios.

Klein, A. M., I. Steffan-Dewenter, D. Buchori, and T. Tscharntke. 2002. Effects of land-use intensity in tropical agroforestry systems on coffee flower-visiting and trap-nesting bees and wasps. Conservation Biology **16**:1003–1014.

Klein, A. M., B. E. Vaissiere, J. H. Cane, I. Steffan-Dewenter, S. A. Cunningham, C. Kremen, and T. Tscharntke. 2007. Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society B-Biological Sciences **274**:303–313.

Kohli, R. K., H. P. Singh, D. R. Batish, S. Jose. 2008. Ecological Interactions in Agroforestry: An Overview. Page 382 *in* D. R. Batish, R. K. Kohli, S. Jose, H. P. Singh. Ecological Basis of Agroforestry. CRC Press, Boca Raton.

Kremen, C., N. M. Williams, and R. W. Thorp. 2002. Crop pollination from native bees at risk from agricultural intensification. Proceedings of The National Academy of Sciences of The United States of America **99**:16812–16816.

Lal, R. 2004a. Agricultural activities and the global carbon cycle. Nutrient Cycling in Agroecosystems **70**:103–116.

Lal, R. 2004b. Carbon emission from farm operations. Environment International **30**:981–990.

Lal, R. 2006. Enhancing crop yields in the developing countries through restoration of the soil organic carbon pool in agricultural lands. Pages 197–209.

Lant, C. L., J. B. Ruhl, and S. E. Kraft. 2008. The tragedy of ecosystem services. Bioscience **58**:969–974.

Lee, K. H., T. M. Isenhart, and R. C. Schultz. 2003. Sediment and nutrient removal in an established multi-species riparian buffer. Journal of Soil and Water Conservation **58**:1–8.

Luo and Zhang. 2009. Multimedia transport and risk assessment of organophosphate pesticides and a case study in the northern San Joaquin Valley of California. Chemosphere **75**(7):969–978.

Maass, J. M., C. F. Jordan, and J. Sarukhan. 1988. Soil-Erosion and nutrient losses in seasonal tropical agroecosystems under various management-techniques. Journal of Applied Ecology **25**:595–607.

Matson, P. A., W. J. Parton, A. G. Power, and M. J. Swift. 1997. Agricultural Intensification and Ecosystem Properties. Science **277**:504–509.

Maumbe, B. M. and S. M. Swinton. 2003. Hidden health costs of pesticide use in Zimbabwe's smallholder cotton growers. Social Science & Medicine **57**:1559–1571.

McDonagh, J. F., T. B. Thomsen, and J. Magid. 2001. Soil organic matter decline and compositional change associated with cereal cropping in southern Tanzania. Land Degradation & Development **12**:13–26.

McNeely, J. A. and S. J. Scherr. 2003. Ecoagriculture: strategies for feeding the world and conserving wild biodiversity. Island Press, Washington, DC.

Millennium Ecosystem Assessment. 2005a. Ecosystems and Human Well-Being: Global Assessment Reports Island Press, Washington, DC.

Millennium Ecosystem Assessment. 2005b. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC.

Montagnini, F. and P. K. R. Nair. 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. Agroforestry Systems **61**:281–295.

Montgomery, D. R. 2007. Soil erosion and agricultural sustainability. Proceedings of The National Academy of Sciences of The United States of America **104**:13268–13272.

Nair, P. K. R., B. M. Kumar, and V. D. Nair. 2009. Agroforestry as a strategy for carbon sequestration. Journal of Plant Nutrition and Soil Science-Zeitschrift Fur Pflanzenernahrung Und Bodenkunde **172**:10–23.

Nieder, R. and D. K. Benbi. 2008. Carbon and nitrogen in the terrestial environment. Springer.

Okalebo, J. R., C. Palm, J. K. Lekasi, C. O. Othieno, M. Waigwa, and K. W. Ndungu. 2004. Use of organic and inorganic resources to increase maize yields in some Kenyan infertile soils: A five-year experience. Pages 359–372 *in* A. Bationo, editor. Managing nutrient cycles to sustain soil fertility in sub-Saharan Africa. AfNet-CIAT, Nairobi.

Östman, Ö., B. Ekborn, and Bengtsson. 2001. Landscape heterogeneity and farming practice influence biological control. Basic and Applied Ecology **2**:365–371.

Palm, C., P. Sanchez, S. Ahamed, and A. Awiti. 2007. Soils: A contemporary perspective. Annual Review of Environmental Resources **32**:99–129.

Parry, M. L., C. Rosenzweig, A. Iglesias, M. Livermore, and G. Fischer. 2004. Effects of climate change on global food production under SRES emissions and socio-economic scenarios. Global Environmental Change-Human and Policy Dimensions **14**:53–67.

Pascual, U. and C. Perrings. 2007. Developing incentives and economic mechanisms for in situ biodiversity conservation in agricultural landscapes. Agriculture Ecosystems & Environment **121**:256–268.

Perfecto, I. and J. Vandermeer. 2002. Quality of agroecological matrix in a tropical montane landscape: Ants in coffee plantations in southern Mexico. Conservation Biology **16**:174–182.

Perfecto, I. and J. Vandermeer. 2008. Biodiversity conservation in tropical agroecosystems - A new conservation paradigm. Pages 173–200 Year in Ecology and Conservation Biology 2008.

Perfecto, I., J. H. Vandermeer, G. L. Bautista, G. I. Nunez, R. Greenberg, P. Bichier, and S. Langridge. 2004. Greater predation in shaded coffee farms: The role of resident neotropical birds. Ecology **85**:2677–2681.

Perrings, C., L. Jackson, K. Bawa, L. Brussaard, S. Brush, T. Gavin, R. Papa, U. Pascual, and P. De Ruiter. 2006. Biodiversity in agricultural landscapes: Saving natural capital without losing interest. Conservation Biology **20**:263–264.

Petit, L. J. and D. R. Petit. 2003. Evaluating the importance of human-modified lands for neotropical bird conservation. Conservation Biology **17**:687–694.

Pimentel, D., P. Hepperly, J. Hanson, D. Douds, and R. Seidel. 2005. Environmental, energetic, and economic comparisons of organic and conventional farming systems. Bioscience **55**:573–582.

Polidoro and Bosque-Perez. 2007. Incorporating livelihoods in biodiversity conservation: a case study of cacao agroforestry systems in Talamanca, Costa Rica. Biodiversity and Conservation **16**:2311–2333.

Postel, S. L., G. C. Daily, and P. R. Ehrlich. 1996. Human appropriation of renewable fresh water. Science **271**:785–788.

Potts, S. G., J. C. Biesmeijer, C. Kremen, P. Neumann, O. Schweiger, and W. E. Kunin. Global pollinator declines: trends, impacts and drivers. Trends in Ecology & Evolution **In Press, Corrected Proof**.

Potts, S. G., T. Petanidou, S. Roberts, C. O'Toole, A. Hulbert, and P. Willmer. 2006. Plant-pollinator biodiversity and pollination services in a complex Mediterranean landscape. Biological Conservation **129**:519–529.

Prepas, E. E. and T. Charette. 2007. Chapter 9.08 Worldwide Eutrophication of Water Bodies: Causes, Concerns, Controls. Pages 311–331 *in* B. S. Lollar, D. H. Heinrich, and K. T. Karl, editors. Treatise on Geochemistry. Elsevier, Oxford.

Qualset, C. O., P. E. McGuire, and M. L. Warburton. 1995. Agrobiodiversity key to agricultural productivity. California Agriculture:45–49.

Rao, M. R., M. P. Singh, and R. Day. 2000. Insect pest problems in tropical agroforestry systems: Contributory factors and strategies for management. Agroforestry Systems **50**:243–277.

Richards, K. R. and C. Stokes. 2004. A review of forest carbon sequestration cost studies: A dozen years of research. Climatic Change **63**:1–48.

Ricketts, T. H. 2004. Tropical forest fragments enhance pollinator activity in nearby coffee crops. Conservation Biology **18**:1262–1271.

Ricketts, T. H., J. Regetz, I. Steffan-Dewenter, S. A. Cunningham, C. Kremen, A. Bogdanski, B. Gemmill-Herren, S. S. Greenleaf, A. M. Klein, M. M. Mayfield, L. A. Morandin, A. Ochieng, and B. F. Viana. 2008. Landscape effects on crop pollination services: are there general patterns? Ecology Letters **11**:499–515.

Robertson, G. P. and S. M. Swinton. 2005. Reconciling agricultural productivity and environmental integrity: a grand challenge for agriculture. Frontiers in Ecology and the Environment **3**:38–46.

Rockström, J., C. Folke, L. Gordon, N. Hatibu, G. Jewitt, F. Penning de Vries, F. Rwehumbiza, H. Sally, H. Savenije, and R. Schulze. 2004. A watershed approach to upgrade rainfed agriculture in water scarce regions through Water System Innovations: an integrated research initiative on water for food and rural livelihoods in balance with ecosystem functions. Physics and Chemistry of the Earth, Parts A/B/C **29**:1109–1118.

Rola et al. 1993. Pesticide use, rice productivity, and human health impacts in the Philippines. Agricultural Policy and Sustainability: Cases Studies from India, Chile, Philippines, and the United States pp. 47–62.

Roose, E. and F. Ndayizigiye. 1997. Agroforestry, water and soil fertility management to fight erosion in tropical mountains of Rwanda. Soil Technology **11**:109–119.

Root, R. B. 1973. Organization of a plant-arthropod association in simple and diverse habitats: The Fauna of collards (*Brassica Oleracea*). Ecology **43**:95–124.

Rosenzweig, C., K. M. Strzepek, D. C. Major, A. Iglesias, D. N. Yates, A. McCluskey, and D. Hillel. 2004. Water resources for agriculture in a changing climate: international case studies. Global Environmental Change-Human and Policy Dimensions **14**:345–360.

Sachs, J. 2005. The End of Poverty. Penguin, London.

Sachs, J. 2008. Common Wealth Economics for a Crowded Planet. Penguin Group, New York, NY.

Sanchez, P. 1999. Delivering on the promise of agroforestry. Environment,Development and Sustainability **1**:275–284.

Sanchez, P., G. Denning, and G. Nziguheba. 2009. The African Green Revolution moves forward. Food Security.

Scherr, S. J. and J. A. McNeely. 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. Philosophical Transactions of the Royal Society B-Biological Sciences **363**:477–494.

Schmitz, O. J., E. Post, C. E. Burns, and K. M. Johnston. 2003. Ecosystem responses to global climate change: Moving beyond color mapping. Bioscience **53**:1199–1205.

Schroth, G., G. A. B. da Fonseca, C. Harvey, C. Gascon, H. Vasconcelos, and A. N. Izac. 2004. Introduction: The role of agroforestry in biodiversity conservation in tropical landscape.*in* G. Schroth, G. A. B. da Fonseca, C. Harvey, C. Gascon, H. Vasconcelos, and A. N. Izac, editors. Agroforestry and Biodiversity Conservation in Tropical Landscapes. Island Press, Washington.

Seck, M., M. N. Abou Mamouda, and S. Wade. 2005. Case study 4: Senegal - Adaptation and mitigation through "produced environments": The case for agriculture intensification in Senegal. Ids Bulletin-Institute of Development Studies **36**:71-+.

Semwal, R. L., S. Nautiyal, K. K. Sen, U. Rana, R. K. Maikhuri, K. S. Rao, and K. G. Saxena. 2004. Patterns and ecological implications of agricultural land-use changes: a case study from central Himalaya, India. Agriculture, Ecosystems & Environment **102**:81–92.

Shuler, R. E., T. H. Roulston, and G. E. Farris. 2005. Farming practices influence wild pollinator populations on squash and pumpkin. Journal of Economic Entomology **98**:790–795.

Sileshi, G., F. K. Akinnifesi, O. C. Ajayi, and F. Place. 2008. Meta-analysis of maize yield response to woody and herbaceous legumes in sub-Saharan Africa. Plant and Soil **307**:1–19.

Smaling, E. M. A. 1993. Soil nutrient depletion in sub-Saharan Africa. Pages 53–67 *in* van Reuler and W. Prins, editors. The Role of Plant Nutrients for Sustainable Food Crop Production in sub-Saharan Africa. Dutch Association of Fertilizer Producers, Leidschendam, Netherlands.

Smaling, E. M. A., J. J. Stoorvogel, and P. N. Windmeijer. 1993. Calculating Soil Nutrient Balances in Africa at Different Scales .2. District Scale. Fertilizer Research **35**:237–250.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, O. Sirotenko, M. Howden, T. McAllister, G. Pan, V. Romanenkov, U. Schneider, S. Towprayoon, M. Wattenbach, and J. Smith. 2008. Greenhouse gas mitigation in agriculture. Philosophical Transactions of the Royal Society B-Biological Sciences **363**:789–813.

Smith, P., D. Martino, Z. Cai, D. Gwary, H. Janzen, P. Kumar, B. McCarl, S. Ogle, F. O'Mara, C. Rice, B. Scholes, and O. Sirotenko, editors. 2007. Agriculture. In Climate Change 2007: Mitigation. Cambridge University Press Cambridge, United Kingdom and New York, NY, USA.

Smukler, S., L. Jackson, S. Sánchez Moreno, S. Fonte, H. Ferris, K. Klonsky, A. O'Geen, K. Scow, and K. Steenwerth. 2010. Biodiversity and multiple ecosystem functions in an organic farmscape. Agriculture Ecosystems and Environment.

Snyder, W. E., G. B. Snyder, D. L. Finke, and C. S. Straub. 2006. Predator biodiversity strengthens herbivore suppression. Ecology Letters **9**:789–796.

Sogbedji, J. M., H. M. van Es, and A. K.L. 2006. Cover cropping and nutrient management strategies for maize production in western Africa. Agronomy Journal **98**:883–889.

Somarriba, E. 1992. Revisting the past: an essay on agroforestry definition. Agroforestry Systems **19**:233–240.

Somarriba, E. and C. Harvey. 2003. ¿Cómo integrar producción sostenible y conservación de biodiversidad en cacaotales orgánicos indígenas? Agroforesteria en las Américas **10**:12–17.

Somarriba, E., M. Trivelato, M. Villalobos, A. Súarez, P. Benavides, K. Moran, L. Orozco, and A. López. 2003. Diagnóstico agroforestal de pequeñas fincas cacaoteras orgánicas de indígenas Bribri y Cabécar de Talamanca, Costa Rica. Agroforesteria en las Américas **10**:24–30.

Srinivasan, U. T., S. P. Carey, E. Hallstein, P. A. T. Higgins, A. C. Kerr, L. E. Koteen, A. B. Smith, R. Watson, J. Harte, and R. B. Norgaard. 2008. The debt of nations and the distribution of ecological impacts from human activities. Proceedings of The National Academy of Sciences of The United States of America **105**:1768–1773.

Stevenson, F. J. and M. A. Cole. 1999. Cycles of soil: carbon, nitrogen, phosphorus, sulfur, micronutrients. John Wiley & Sons Inc.

Stoorvogel, J. J., E. M. A. Smaling, and B. H. Janssen. 1993. Calculating soil nutrient balances in Africa at different scales 1. Supra-National scale. Fertilizer Research **35**:227–235.

Suatunce, P., E. Somarriba, C. Harvey, and B. Finegan. 2003. Composición florística y estructura de bosques and cacaotales en los Territorios Indígenas de Talamanca, Costa Rica. Agroforesteria en las Américas **10**:31–35.

Swallow, B. M., J. K. Sang, M. Nyabenge, D. K. Bundotich, A. K. Duraiappah, and T. B. Yatich. 2009. Tradeoffs, synergies and traps among ecosystem services in the Lake Victoria basin of East Africa. Environmental Science & Policy **12**:504–519.

Swaminathan, M. 2005. Lessons from the Past and Policies for the Future: An Ever-green Revolution. An International Dialogue on Agricultural and Rural Development in the 21st Century, Beijing.

Tallis, H., P. Kareiva, M. Marvier, and A. Chang. 2008. An ecosystem services framework to support both practical conservation and economic development. Proceedings of The National Academy of Sciences of The United States of America **105**:9457–9464.

Tanner, C. C., M. L. Nguyen, and J. P. S. Sukias. 2005. Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. Agriculture, Ecosystems & Environment **105**:145–162.

Tilman, D., K. G. Cassman, P. A. Matson, R. Naylor, and S. Polasky. 2002. Agricultural sustainability and intensive production practices. Nature **418**.

Tilman, D., J. Fargione, B. Wolff, C. D'Antonio, A. Dobson, R. Howarth, D. Schindler, W. H. Schlesinger, D. Simberloff, and D. Swackhamer. 2001. Forecasting Agriculturally Driven Global Environmental Change. Science **292**:281–284.

Tscharntke, T., A. M. Klein, A. Kruess, I. Steffan-Dewenter, and C. Thies. 2005. Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. Ecology Letters **8**:857–874.

Tscharntke, T., C. H. Sekercioglu, T. V. Dietsch, N. S. Sodhi, P. Hoehn, and J. M. Tylianakis. 2008. Landscape constraints on functional diversity of birds and insects in tropical agroecosystems. Ecology **89**:944–951.

Uusi-Kamppa, J., B. Braskerud, H. Jansson, N. Syversen, and R. Uusitalo. 2000. Buffer zones and constructed wetlands as filters for agricultural phosphorus. Journal of Environmental Quality **29**:151–158.

Van Bael, S. A., S. M. Philpott, R. Greenberg, P. Bichier, N. A. Barber, K. A. Mooney, and D. S. Gruner. 2008. Birds as predators in tropical agroforestry systems. Ecology **89**:928–934.

van Noordwijk, M., T. P. Tomich, and B. Verbist. 2002. Negotiation support models for integrated natural resource management in tropical forest margins. Conservation Ecology **5**.

Vanlauwe, B., J. Diels, K. Aihou, E. N. O. Iwuafor, O. Lyasse, N. Sanginga, and R. Merckx. 2002. Direct interactions between N fertilizer and organic matter: Evidence for trials with ^{15}N-labelled fertilizer. Pages 173–183 *in* B. Vanlauwe, J. Diels, N. Sanginga, and R. Merckx, editors. Integrated plant nutrient management in sub-Saharan Africa: From concept to practice. CABI, Wallingford.

Verchot, L. V., J. Mackensen, S. Kandji, M. Van Noordwijk, T. Tomich, C. Ong, A. Albrecht, C. Bantilan, K. V. Anupama, and C. Palm. 2005. Opportunities for linking adaptation and mitigation in agroforestry systems.*in* C. Robledo, M. Kanninen, and L. Pedroni, editors. Tropical forests and adaptation to climate change: In search of synergies. Center for International Forestry Research (CIFOR), Bogor.

WCD (World Commission on Dams). 2000. Dams and Development: A New Framework for Decision-Making. Water and Environment Journal : The Journal. World Commission on Dams, Earthscan, London, UK.

Wendland, K. J., M. Honzák, R. Portela, B. Vitale, S. Rubinoff, and J. Randrianarisoa. Targeting and implementing payments for ecosystem services: Opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. Ecological Economics **In Press, Corrected Proof**.

Whisenant, S. G. 1999. Repairing Damaged Wildlands. Cambridge University Press, Cambridge.

WHO/UNICEF. 2004. Meeting the MDG Drinking Water and Sanitation Target: A Mid-term Assessment of Progress. Page 33. WHO, Geneva, Switzerland/ UNICEF, New York, NY.

Wigington, P. J., S. M. Griffith, J. A. Field, J. E. Baham, W. R. Horwath, J. Owen, J. H. Davis, S. C. Rain, and J. J. Steiner. 2003. Nitrate removal effectiveness of a riparian buffer along a small agricultural stream in Western Oregon. Journal of Environmental Quality **32**:162–170.

Williams, N. M. and C. Kremen. 2007. Resource distributions among habitats determine solitary bee offspring production in a mosaic landscape. Ecological Applications **17**:910–921.

Winfree, R., N. M. Williams, J. Dushoff, and C. Kremen. 2007. Native bees provide insurance against ongoing honey bee losses. Ecology Letters **10**:1105–1113.

Witcover, J., S. A. Vosti, C. L. Carpentier, and T. C. D. A. Gomes. 2006. Impacts of soil quality differences on deforestation, use of cleared land, and farm income. Environment and Development Economics **11**:343–370.

WRI (World Resources Institute). 1989. World Resources 1998–99: Environmental change and human health. WRI, UNEP, UNDP, and The World Bank, New York.

Wyland, L. J., L. E. Jackson, W. E. Chaney, K. Klonsky, S. T. Koike, and B. Kimple. 1996. Winter cover crops in a vegetable cropping system: Impacts on nitrate leaching, soil water, crop yield, pests and management costs. Agriculture Ecosystems & Environment **59**:1–17.

Young, A. 1997. Agroforestry for soil management. 2nd edition. CAB International, Wallingford.

Zhang, W., T. H. Ricketts, C. Kremen, K. Carney, and S. M. Swinton. 2007. Ecosystem services and dis-services to agriculture. Ecological Economics **64**:253–260.