ECOLOGICAL ECONOMICS 64 (2007) 286–296

available at www.sciencedirect.com

ScienceDirect

www.elsevier.com/locate/ecolecon

Measures of the effects of agricultural practices on ecosystem services

Virginia H. Dale[a,*], Stephen Polasky[b]

[a]Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, TN, 37831-6036 USA
[b]Department of Applied Economics, and Department of Ecology, Evolution and Behavior, University of Minnesota, St Paul, MN 55108 USA

ARTICLE INFO

Article history:
Received 31 May 2006
Received in revised form
19 April 2007
Accepted 22 May 2007
Available online 13 July 2007

Keywords:
Agriculture
Chemical
Erosion
Land use
Ecosystem services

ABSTRACT

Agriculture produces more than just crops. Agricultural practices have environmental impacts that affect a wide range of ecosystem services, including water quality, pollination, nutrient cycling, soil retention, carbon sequestration, and biodiversity conservation. In turn, ecosystem services affect agricultural productivity. Understanding the contribution of various agricultural practices to the range of ecosystem services would help inform choices about the most beneficial agricultural practices. To accomplish this, however, we must overcome a big challenge in measuring the impact of alternative agricultural practices on ecosystem services and of ecosystem services on agricultural production. A framework is presented in which such indicators can be interpreted as well as the criteria for selection of indicators. The relationship between agricultural practices and land-use change and erosion impact on chemical use is also discussed. Together these ideas form the basis for identifying useful indicators for quantifying the costs and benefits of agricultural systems for the range of ecosystem services interrelated to agriculture.

1. Introduction

Ecological systems both contribute to and are affected by the production of goods and services that are of value to people. We refer to the contribution of ecosystems to human well-being in short-hand notation as "ecosystem services." Understanding how agriculture impacts ecosystem services, which in turn affect agricultural productivity, is of particular importance because of agriculture is a dominant form of land management. Globally, it is estimated that 38% of land is in agricultural uses (FAO, 2004), and excluding boreal lands, desert, rock and ice, this amount rises to 50% (Tilman et al., 2001). As Tilman et al. (2002) state: "Agriculturalists are the *de facto* managers of the most productive lands on Earth. Sustainable agriculture will require that society appropriately rewards ranchers, farmers and other agriculturalists for the production of both food and ecosystem services." But appropriately rewarding ranchers, farmers and other agriculturalists will require the ability to accurately measure ecosystem services in a verifiable quantitative manner.

Agriculture and ecosystem services are interrelated in at least three ways: (1) agro-ecosystems generate beneficial ecosystem services such as soil retention, food production, and aesthetics; (2) agro-ecosystems receive beneficial ecosystem services from other ecosystems such as pollination from non-agricultural ecosystems; and (3) ecosystem services from non-agricultural systems may be impacted by agricultural practices. In some cases, tracing the interrelationships between agriculture and ecosystem services is fairly direct as when pollinators increase agricultural crop yields or conservation easements on agricultural lands provide habitat for bird species enjoyed by bird-watchers. In other cases, the contribution may be more indirect or complex, as for example when wetlands reduce the load of nitrogen in surface water originating from agricultural fields and destined for a coastal estuary where eutrophication causes hypoxic conditions and reduced fish productivity.

* *Corresponding author*. Tel.: +1 865 576 8043; fax: +1 865 576 8543.
E-mail address: dalevh@ornl.gov (V.H. Dale).

0921-8009/$ - see front matter © 2007 Elsevier B.V. All rights reserved.
doi:10.1016/j.ecolecon.2007.05.009

To make the concept of ecosystem services operational with respect to agro-ecosystems, it requires a way of measuring ecosystem services. To be really useful in management and policy discussions, however, there must be a way to measure how ecosystem services change as a function of changing agricultural practices. This requires a thorough understanding how ecological systems function, both under current conditions and how these functions might change with different management regimes. For example, if pollinators were removed from the system, how would crop yields change? If a wetland is filled, how does this action alter nutrient flows, hydrology and habitat conditions that might ultimately affect local bird watching, nitrate levels in local groundwater, flood potential downstream, and fish productivity in coastal estuaries? Tracing through the full array of consequences can present great challenges.

At present, we lack ways to measure the quantities of many ecological services in a manner similar to measures of marketed goods and services in the economy. Accurate measures of goods and services in the economy arose because such accounting was a necessary condition for a market economy to function. Trade requires verifiable information on the quantity and quality of items being traded. Measures of goods are typically easy to define and monitor. A well-managed farm accounts for how many crops of various kinds are produced on the farm in a given year (bushels of corn, soybeans, etc.) and amounts of various inputs used (fuel, seeds, labor, etc.). Though often more difficult, firms producing services (e.g., legal, financial, or insurance firms) can also define the amounts of various services they produce and what inputs they use. Service providers track various measures such as billable hours or policies issued, but tracking the quality of the service, which may matter as much or more than the quantity, can be difficult. To illustrate some of the difficulties of measuring services, think about how to accurately measure the service provided by academics. If measuring the quality of the services provided was easy, then tenure and promotion decisions should be quite simple (ignoring college or university politics, of course).

Because ecosystem services typically have not been traded in markets, there has not been the same type of systematic effort devoted to defining operational and verifiable measures for ecosystem services. Measures of ecosystem services still need further development in many circumstances (Boyd and Banzhaf, 2006). Further complicating matters, the concept of scale often is important in ecological service because benefits may only be measurable over a large area or after a long time period. A major task in moving ecosystem services from the realm of being an interesting idea to being a practical reality is to define operational and verifiable measures.

Crop and livestock production are the best quantified services from agriculture. These production benefits are typically measured as the yield per area of per effort expended. Increases in agricultural production are clear from examination of production data at regional and global scales. Over the 40-year period from 1960 to 2000, global food production increased by 2.5 times, more than outpacing human population growth, which approximately doubled over the same period (Millennium Ecosystem Assessment, 2005). In addition, in marine and freshwater systems, farmed fish and shellfish have increased to be one third of all fish and shellfish production. The dramatic increase in crop and livestock production was partly the result of increasing the amount of land devoted to agriculture, the development of high-yielding varieties, and advancements in the integration of management. However, much of the increase in production came from increasing yields through a vast increase in application of chemical fertilizers and pesticides and water from irrigation systems (Tilman et al., 2002).

The Millennium Ecosystem Assessment (2005) found that several ecosystem services that relate to agriculture are in decline. Particularly noticeable are the worldwide declines in wild fish and fresh water. In many cases, declines in wild-fish stocks can be traced to over-harvesting (Jackson et al., 2001; Myers and Worm, 2003). Decreases in supply and quality of fresh water in many parts of the world can be traced to increasingly intensive agriculture, both in terms of withdrawal of water from rivers for irrigation, and lower water quality from the flow of nutrients, sediments, and dissolved salts from agricultural lands. The global increase in crop production may also account for declines in air quality regulation, climate regulation, erosion regulation, pest regulation, and pollination (Millennium Ecosystem Assessment, 2005). A major concern is that the increased agricultural production over the past 50 years has come at the cost of the ecological suatainability that will be necessary to maintain productivity in the future.

To complement global measures of ecosystem services as reported in the Millennium Ecosystem Assessment (2005), measurements taken at the very local level of the agricultural field may be more useful in a practical management sense. Site-specific measures can better relate to particular farming practices. For example, Bockstaller et al. (1997) show how ten metrics relate to regulatory services provided by agriculture on 17 commercial arable farms. The services they consider are protection of ground water quality, surface water quality, air quality, soil quality, non-renewable resources, biodiversity, and landscape quality. Indicators measured in each field were nitrogen, phosphorus, pesticide, irrigation, organic matter, energy, crop diversity, soil structure, soil cover and ecological structures. The indicators relate to one or more of these services.

Ideally, it would be useful to have the ability to accurately measure the flow of ecosystem services from agro-ecosystems at several scales of resolution. These measures would allow documentation of the changes over time in ecosystem services from agriculture and how these ecosystem services have been affected by alterations in the agricultural sector at various resolutions. In part because of the challenges mentioned above, the set of ideal measures do not now currently exist. However, there has been extensive work on ecological indicators related to agriculture that can be used to quantify changes in ecological systems (e.g., Bockstaller et al., 1997; Pretty et al., 2000; Rigby et al., 2001; Boody et al., 2005). Developing a suite of indicators that are both measurable and tied to the provision of ecosystem services is one way to make progress on tracking changes in ecological systems and how this might affect the flow of ecosystem services.

This paper presents a framework in which such indicators can be interpreted as well as criteria for selection of indicators. The final section of the paper discusses the relationship between key changes in agricultural practices and land-use change, erosion, and chemical use and indicators of ecosystem services that might be affected. Together these ideas form the basis for identifying useful indicators for quantifying the costs and benefits of agricultural systems for the range of ecosystem services interrelated to agriculture.

2. A framework for interpreting indicators of ecosystem services

The decline in many important ecosystem services (Millennium Ecosystem Assessment, 2005) and the observation that some of these declines are related to the expansion of agriculture and the increased use of fertilizers and pesticides place a premium on finding effective ways to monitor changes in ecological systems and their impacts on ecosystem services. Ecological indicators have been used to quantify the magnitude of change, amount of exposure to change, or degree of response to the exposure (Hunsaker and Carpenter, 1990; Suter, 1993). The purposes of ecological indicators include assessing the condition of the environment, monitoring trends in conditions over time, providing an early warning signal of changes in the environment, and diagnosing the cause of an environmental problem (Cairns et al., 1993). Tradeoffs between desirable features, costs, and feasibility influence the choice of indicators (Dale and Beyeler, 2001). Because no one indicator can meet all of these goals, we anticipate that a set of indicators will be needed to capture key attributes of ecological systems of interest (Bockstaller et al., 1997; Dale et al., 2004). Yet multiple, interdependent ecosystem services and values present both conceptual and empirical research challenges (Turner et al., 2003).

Therefore we propose that a set of ecological indicators for ecosystem services both from and to agriculture should be considered as they relate to all pertinent spatial resolutions (Fig. 1). In addition to farm-level and global metrics, Pretty et al. (2000) point out that such measures are useful at the levels at which national and international policies are developed as well as the levels of particular programs and policies. Sometimes these intermediate levels of resolution are determined by topographic and ecological conditions. For example, the watershed of the Mississippi River basin, which drains 41% of the United States, is the relevant scale to address the contribution of riverine nitrogen to eutrophication that influences the recent increase in the size of the hypoxia zone in the Gulf of Mexico. While the actual metrics to be sampled will depend on the system and the specific ecosystem services being considered, Fig. 1 highlights the importance of the spatial resolution of the metrics.

Ecological indicators are meant to provide a simple and efficient means to examine the ecological composition, structure, and function of complex ecological systems (Karr, 1981). In addition to spatial scale, these ecological systems can be considered at various levels in the biological hierarchy: landscapes and regions, ecosystems and communities, and populations and species. Composition refers to such features as distribution and richness of patch types over a landscape or region; community diversity, life form distribution, or similarity; and species abundance, frequency, importance and cover. Structure includes spatial heterogeneity, patch size and shape, fragmentation and connectivity at the landscape or regional level; substrate and soil conditions, canopy openness, and gap characteristics at the ecosystem or community level; and dispersion, range, and morphological variability at the population or species level. Function refers to patch persistence, nutrient cycling, erosion, and disturbance regimes at the landscape or region level; productivity, decomposition, trophic dynamics, and succession at the ecosystem and community level; and demography, physiology, life history patterns and adaptation at the population and species level. The set of nested triangles in Fig. 2 illustrate how the components of composition, structure, and function vary with scale of the biological hierarchy ranging from populations and species (the smallest triangle) to

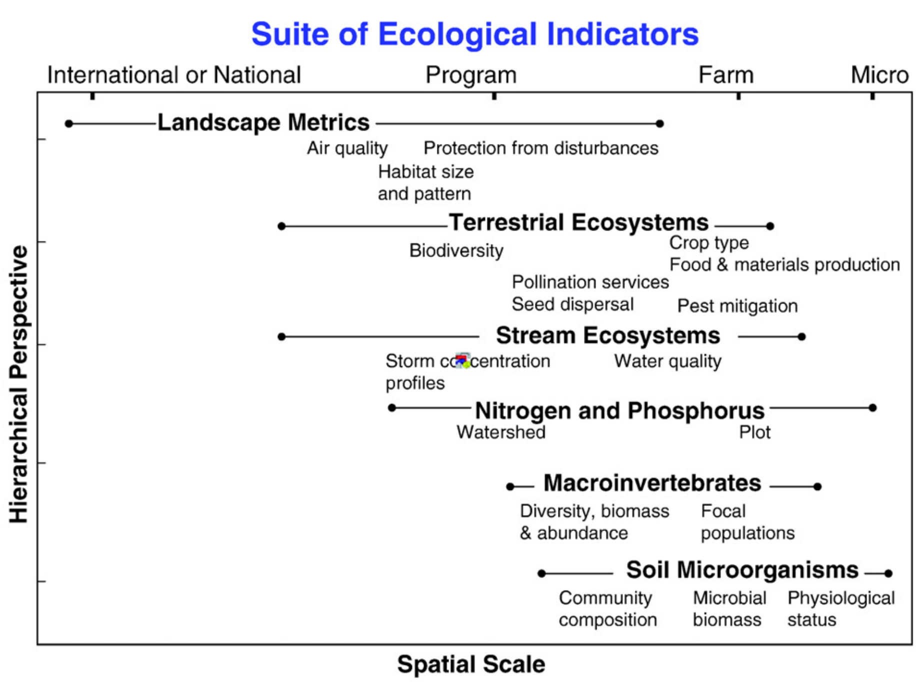

Fig. 1 – Spatial scales of metrics that relate to ecosystem services from agriculture. Spatial scales of metrics that relate to ecosystem services from agriculture.

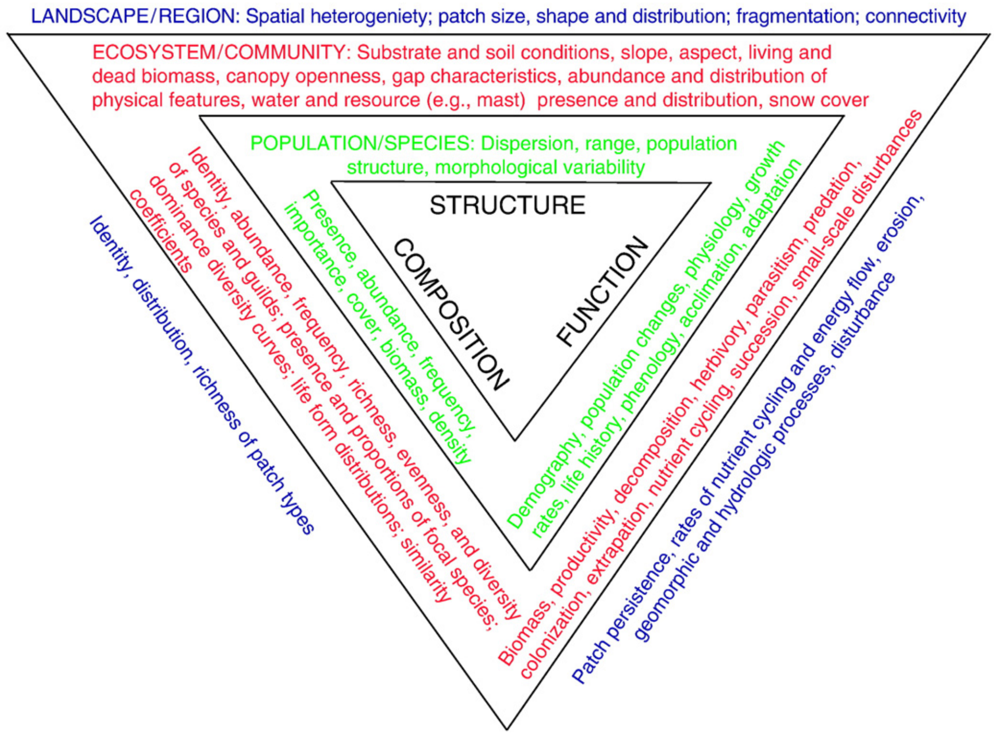

Fig. 2 – Indicators can be selected from the appropriate scale to represent composition, structure, or function of ecological systems (modified from Dale and Beyeler, Indicators can be selected from the appropriate scale to represent composition, structure, or function of ecological systems (modified from,Dale and Beyeler, 2001).

ecosystems and communities and finally to landscapes and regions.

The challenge for selecting ecological indicators for our purposes is to identify key features that represent the compositional, structural, and functional components of the system important in the provision of ecosystem services, for not everything can be monitored. Typically, structure and composition are easier to measure than function, and they often reveal information about function. For example, identifying a plant's size (structure) or species (composition) is easier than determining such functional attributes as the plant's influence on carbon sequestration, nutrient cycling, or enhancement of soil properties. Hence indicators often are structural or compositional attributes. For example, quantifying the amount of land in annual crops, perennials or forests of various ages via remote sensing may be an effective way to estimate carbon storage provided by a landscape. This approach of focusing on structure and composition is tractable and is valid as long as they accurately represent the functional attributes of the ecological systems related to the provision of ecosystem services (i.e., dominant vegetation types accurately reflect the amount of carbon storage). The use of ecological indicators assumes that changes in indicators reflect changes taking place in the ecological hierarchy — potentially from populations, to species, to ecosystems, or to entire regions (Noon et al., 1999). Landscape features can often be determined by using remotely sensed data and thus are less time consuming and expensive than field-based observations. Thus landscape-based metrics of structure and composition (shown in the outer triangle of Fig. 2) are often the most cost-effective indicators of ecological systems. The focus on landscape structure and composition leads to the question of what indicators are useful and necessary to supplement the landscape metrics as a way to capture key ecosystem services provided by and affected by agriculture.

3. Criteria

A challenge in developing and using a suite of ecological indicators for ecosystem services interrelated with agriculture is determining those indicators that adequately characterize the complexities of the entire system yet are simple enough to be effectively and efficiently monitored and modeled (Dale et al., 2004). Such indicators must be closely linked to, and predictive of, changes in ecosystem services. The scale of the service must be determined so that the indicator is at the appropriate resolution. For example, a service that exists for an entire watershed may be measured by an indicator sampled in the major stream, whereas more local effects might best be measured within a field or at the field edge. Furthermore the type of service and range of its conditions should be defined so that the indicator can appropriately measure the benefits achieved. Building upon analyses by Landres et al. (1988), Kelly and Harwell (1990), Cairns et al. (1993), Lorenz et al. (1999), and Dale and Beyeler (2001), we suggest the following criteria for ecological indicators of ecosystem services related to agriculture:

•*Be easily measured*. Ease of measurement involves several characteristics. Can the indicator be measured remotely? This aspect is important since sending a person out to collect data is often the most expensive aspect of collecting data. How often do the data need to be collected? What kind of expertise or equipment is necessary to obtain the measure? Does the sample need to be treated in a particular way (such as being placed on dry ice, kept dry, or identified in the field)? For example, remote sensing of the distribution of crops and buffers in relation to streams would be far cheaper than trying to measure the output of nutrients at field borders for assessing the amount of nutrient runoff from an agricultural landscape. Some amount of expensive field testing may be necessary at the beginning, however, to "ground truth" a more easily measured indicator.

•*Be sensitive to changes in the system*. Does the indicator respond to past or anticipated changes in the ecological system? For agriculture, these changes can include changes in crop type or crop rotations, amount and type of fertilizers or pesticides applied, timing of application, tillage practices, and type of farm equipment being used. All of these agricultural management decisions may have local or larger scale impacts on ecosystem services. Exogenous changes in precipitation or temperature caused by climate change may also have important impacts, and may interact in various ways with management decisions.

•*Respond to change in a predictable manner*. Does the indicator change in such a way that its value is indicative of the type and degree of the pattern of changes to the system? In other words is there a predictable relationship between the indicator and the amount or type of change. So, for example, are statistics on the change in land use toward or away from annual crops sufficient for predicting changes in carbon storage in an agricultural system, or are more detailed indicators required?

•*Be anticipatory, that is, signify an impending change in key characteristics of the ecological system*. Is the change in the indicator one that observers can anticipate under specific conditions and therefore monitor as an early warning device, such as a canary in a coal mine signifies poor air quality in time for miners to escape? For example, significant correlations of mid-square leaf $\delta^{15}N$ with late-season nutrient content and soil electrical conductivity (EC) suggest that the natural abundance of ^{15}N is a sensitive and early indicator of soil and plant nutrient status in a fertilized cotton field (Stamatiadis et al., 2006).

•*Predict changes that can be averted by management actions*. Does the value of the indicator change in response to management actions and hence allow the indicator to be used to monitor management as well as change? Often, a useful indicator is directly tied to management actions. For example, an indicator of water quality may be the amount of land in buffers along stream corridors.

•*Are integrative: the full suite of indicators provide a measure of coverage of the key gradients across the ecological systems (e.g., gradients across soils, vegetation types, temperature, space, time, etc.)*. Does the set of indicators cover the major structural and compositional features of the ecological system (as shown in Fig. 2)?

•*Have known variability in response*. Is the variation in the indicator clear and within acceptable limits? Measures of carbon in soil, for example, have high variability that as yet are poorly understood, whereas above-ground carbon content is readily characterized by above-ground biomass.

Deriving a manageable set of indicators that meets all of these criteria simultaneously is difficult and often not possible. The set of ecosystem services under consideration should influence the choice of potential metrics and the criteria will determine those metrics that are suitable for this specific system (as we show in Fig. 3). The criteria can serve as goals and guides in the selection process. Often one criterion must be sacrificed and another emphasized. For example, resource managers agree that sometimes it is necessary to spend more time or money to obtain an indicator that is difficult to measure but that is highly sensitive to particular changes. The indicators as determined by the criteria are the appropriate measures of ecosystem services for the system being considered and hence could be added to the decision diagram of de Groot et al. (2002).

4. The relationship between changes and indicators

4.1. *Changes associated with agricultural use*

One way to address the challenge of identifying key indicators is to consider the major changes associated with agricultural use: land-cover change, erosion, and chemical and water use (Fig. 4). Land-cover change is common and often large-scale. Changes in land cover in agricultural systems are a direct result of management practices. Land-cover change includes both the conversion of land from different types of crops and change in the type of agriculture being practiced. For example, very different carbon sequestration and other environmental benefits accrue from row cropping of annual plants, growing perennial plants, or animal husbandry.

Information at the broadest spatial scale is often the most cost-effective in measuring land-cover change, for satellite imagery is relatively inexpensive and is very useful to understand changes in land cover and patterns over time, at least since

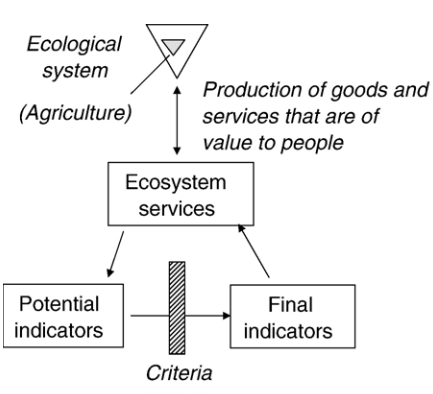

Fig. 3 – Relationship between the ecological system (of which agriculture is a part), ecosystems services, indicators, and criteria for the indicators. Relationship between the ecological system (of which agriculture is a part), ecosystems services, indicators, and criteria for the indicators.

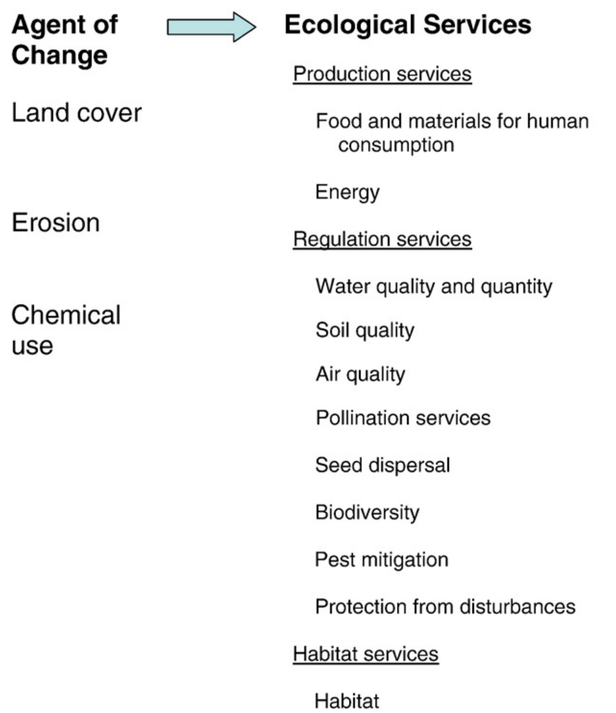

Fig. 4 – Changes from agriculture that affect and are affected by several ecosystem services. The services are organized according the typology of de Groot et al. (2002).

1972 when the first satellite monitoring system was established (Gallego, 2004). Land-cover patterns before 1972 can be discerned using imagery from aircraft (e.g., Scarpace and Quirk, 1980) or for times before human flight by using historical information such as witness trees (e.g., Foster et al., 2004). In some situations, it is possible to use remotely sensed information to quantify loss or degradation of habitats for particular species of special concern. In other cases, it is enough to know that a particular or common vegetation type is diminished, for certain habitats may occur only in one land-cover type. In still other cases, it is critical to know how structural features of vegetation types change.

Land-cover changes can also be used to predict erosion, which is tied to ecosystem services both in terms of water quality and future agricultural productivity. Erosion rates are largely a function of the proportion of bare ground and especially the amount of those bare lands that have slopes greater than 5% (Maloney et al., 2005). In agricultural systems, the location and proportion of bare areas can change greatly over a season, and, therefore, it is useful to quantify short-term changes in areas of bare ground — especially during periods of heaviest rainfall. Management practices that reduce the need for tillage can reduce erosion considerably as can the use of cover crops.

A third major change results from chemicals used in agriculture. Metrics of these effects include pesticide contamination, fossil-energy use, nitrogen (N) balance, phosphorus (P) balance, and nitrogen contamination risk (Viglizzo et al., 2003). Application of nitrogen fertilizer to agricultural systems is the leading source of the increase in reactive nitrogen in the environment (Galloway et al., 2003). Much of the nitrogen from agriculture derives from animal-production systems, both directly from nitrogen leakage to the atmosphere and waters from these systems, and indirectly from the demand for increased crop production for animal production (Howarth, 2004). For example, the use of mineral fertilizers and animal feed accounts for a high portion of the total energy use on specialized dairy and pig farms in Flanders (Meul et al., 2007). Intense use of the land can also increase chemicals in an agricultural setting. Non-methane volatile organic compounds (NMVOCs) emitted from livestock derive from sources such as dairy cattle slurry and manure and in the United Kingdom alone exceed 165 kt C/year (Hobbs et al., 2004). Besides affecting air quality, these NMVOCs may influence the cleansing capacity of the troposphere. These impacts can be quite significant. For example, worldwide about 1% of atmospheric emissions of methyl bromide and 5% of methyl iodide arise from rice fields (Redeker et al., 2000).

Nitrogen use and its consequent impacts on the environment provide an especially important and vivid example of the importance of measuring ecosystem services related to agriculture, and linking these measures to incentives of farmers (Tilman et al., 2002). Farmers receive the benefits of higher yields from fertilizer application but do not pay the environmental costs associated with nitrogen exports to ground or surface water, or via air emissions. Farmers, therefore, have little incentive to limit the use of nitrogen fertilizers. In many cases, use of fertilizer can be reduced significantly with highly beneficial environmental results and little or no loss of farm productivity. For example, conversion of specialized diary farms on sand soils in Germany from highly intensive conventional practices to organic farming significantly reduced nitrate input into groundwater while only slightly reducing yield (Taube et al., 2006). Also, organic and integrated farming systems can have higher potential denitrification rates, greater denitrification efficiency, higher organic matter, and greater microbial activity than conventionally farmed soils (Kramer et al., 2006). Furthermore, maximum yields with reduced input can be achieved with high-precision agriculture (which includes correct placement of fertilizers and other amendments in the soil or on or in the plant, timing of the input, relationships with other inputs to create proper balances, and correctly leveling, draining and contouring the land) (Wallace, 1994 and many subsequent studies).

The greatest uncertainties in understanding of the N budget at most scales are the rates of natural biological nitrogen fixation, the amount of reactive N storage in environmental reservoirs, and the production rates of non-reactive N_2 by denitrification (Gallowayet al., 2004). However methods to obtain accurate measures are still being developed for upland terrestrial sties where most agriculture occurs (Groffman et al., 2006).

4.2. *Categories of indicators of ecosystem services from agriculture*

The changes to the environment associated with agriculture affect a wide range of ecosystem services including food and materials for human consumption, water quality and quantity, soil quality, air quality, carbon sequestration, pollination services, seed dispersal, pest mitigation, biodiversity, habitat change and habitat degradation, and protection from disturbances (Fig. 3).

Food and materials for human consumption constitute a prime category of ecological indicators since this is the main purpose of agriculture. The service provided is usually measured as productivity (calculated as the weight of material per area in cultivation). In addition to food, crops are grown for energy, fiber, oils, fabrics, rope, and other such goods. Because the business of farming depends on productivity, we have exceptionally good records of this. It is the other ecosystem services for which reliable indicators are more problematic.

Water quality and quantity are important services that can be enhanced or degraded by agriculture. Agriculture has both a direct and indirect effect on water consumption and quality (e.g., Duarte et al., 2002). Irrigation for agriculture changes in-stream flows and infiltration patterns. Furthermore, chemical use and erosion significantly affect water quality in many areas. Metrics required to be measured under the Clean Water Act (CWA) in the United States are common indicators of water quality. Under the National Water Quality Inventory [305(b) of the CWA], information is provided biennially on the condition of all water bodies and the most common causes of water body impairment for both pollutants (chemicals, sediments, nutrients, metals, temperature, pH) and other conditions (altered flows, modification of the stream channel, introduction of exotic invasive species). The most common causes of water body impairment are sediments, pathogens, nutrients, metals, dissolved oxygen, and other habitat alteration. Agricultural practices are contributors to all of these metrics, and in some cases are the dominant contributors in many water bodies. The difference between the CWA and the ecosystem services approach is that the latter emphasizes the suite of benefits received to humans (not just water-related benefits).

Soil quality is also directly and indirectly affected by agricultural practices. Because soil properties are so variable over space and time, there is great interest in means to rapidly and remotely characterize soil quality. Non-invasive geophysical measurements of apparent soil electrical conductivity are proving effective (Jung et al., 2005; Corwin et al., 2006).

Furthermore, agricultural activity can affect sediment runoff. The proportion of bare ground and roads in watersheds on slopes greater than 5% can correlate to indicators of

- Stream suspended sediment concentrations (total and inorganic) during baseflow and storms (Houser et al., 2006)
- Baseflow concentrations of phosphate and stormflow concentrations of nitrate and phosphate (Houser et al., 2006)
- Daily amplitude and maximum deficit in dissolved oxygen concentrations (Mulholland et al., 2005)
- Habitat metrics such as benthic particulate organic matter, coarse woody debris, stream bed particle size, and bed stability (Maloney et al., 2005)
- Benthic macroinvertebrate richness and focal orders (Moore and Palmer, 2005; Maloney and Feminella, 2006)
- Fish focal species (Maloney et al., 2005).

These metrics largely relate to the sedimentation effects of particular land-use practices. However, measures of bare ground and roads on slopes are easier to quantify than detailed chemical or biological metrics.

Agricultural effects on air pollution include pesticides, odors, smoke, dust, allergenic pollens, and trash. Nitrogen compounds emitted from agricultural sources can affect air quality in two ways: ammonia (NH_3) emissions result from fertilizers and livestock, and nitrogen oxides (NOx) from fuel combustion in farm equipment. These effects are currently monitored in the US under the Clean Air Act.

Agricultural practices also affect net greenhouse gas emissions both through the burning of fossil fuels and through the release or storage of greenhouse gases in plant material and soils. The move toward no-till cropping provides some energy use efficiency. Though there is some support in the literature that no-till also increases carbon sequestration (Lal et al., 2007), there are also doubts that this is the case (Baker et al., 2007). Carbon sequestration on farm lands is a relative new and potential growing service of agricultural lands (US EPA, 2005). Moving to perennial crops or forestry can increase the amount of carbon stored on lands. While measuring above-ground biomass can yield good estimates of carbon sequestration, measuring carbon stored in soils is more difficult in part because soils are so variable. Feng (2005) found that differences in the location of the land influenced carbon sequestration. Agricultural practices can also result in emissions of N_2O, which is a powerful greenhouse gas, and methane.

Metrics of pollination services both to and from agriculture include the number of insects, the cost benefits to agriculture, or the cost of supplemental pollination. Crop pollination by wild insects is a valuable ecosystem service, but it is under increasing threat from agricultural intensification (Kremen and Ricketts, 2000; Ricketts et al., 2004). For example, pollination services by the nonnative honey bees (*Apis mellifera*) are critical for about 90 crops in the US and 300 crops worldwide, and value of this service in the US is estimated to be about 18 billion dollars (Sanford, 1998). Honey bees are extensively used in growing almonds, apples, cranberries, blueberries, kiwifruit, and cucurbit or vine crops and of minor importance to pollinate strawberries, peppers, peaches, pears, plums and citrus. Other bees that perform important pollination services include the nonnative alfalfa leafcutter bee (*Megachile rotundata*), native bumblebees (*Bombus* sp.), the imported hornfaced bee (*Osmia cornifrons*), and the domestic blue orchard bee, *Osmia lignaria*. Together they contribute more than several million dollars of value each year (Sanford, 1998). The economics of honey bee use in commercial pollination is affected by introduction of parasitic mites and contributes to the price of honey, which can cause instability in the supply of the pollination service. Because honey bees are usually the insect of choice in managed pollination circumstances, this service is often quantified by the cost of moving hives to crops. However that metric does not account for wild pollination that occurs as a benefit from agriculture. There is a clear need to determine how to appropriate value to pollination services both to and from agriculture.

Seed dispersal can be greatly affected by agricultural systems. An abundance of agriculture seeds can compete with native seeds. Furthermore, animals whose habitats and habits are affected by farms are important seed dispersers. Disturbances to coevolved interactions between plants and seed dispersers may leave areas of devoid of seedlings and thus impair the ability of plants to recover from human activities or natural disturbances that clear land (Daily, 1997). The most accurate measure of agricultural effects on seed dispersal will likely be a comparison of number and types of seeds dispersed per area with and without agriculture.

Pest mitigation is an important service associated with agriculture. Although pests such as insects, rodents, fungi, snails, nematodes and viruses may destroy as much as 25 to 50% of the world's crops of food, timber, and fiber (Pimentel et al., 1989), about 99% of potential crop pests are controlled by natural enemies, such as birds, spiders, parasitic wasps and flies, lady bugs, fungi, viral diseases, and other organisms (DeBach, 1974). Natural enemies of pests have been shown to improve agricultural production (e.g., ground-living natural enemies of the bird cherry-oat aphid (*Rhopalosiphum padi* (L.)), reduced aphid abundance and increased yield of barley on ten farms in central Sweden (Ostman et al., 2003)). Natural biological control agents save farmers billions of dollars each year by protecting crops and reducing the need for chemical control (Naylor and Ehrlich, 1997). Quantifying this benefit requires estimates of the amount of crop receiving protection as well as information on costs of chemicals used on farms to control pests and their secondary implication in killing beneficial animals, instigating resistance in pests, and impairing human health. Pest mitigation provided by agriculture is sometimes quantified as the financial and human health benefits achieved by not using pesticides. Adoption of carefully designed site-specific alternative cropping systems along with appropriate methods of cultivation can minimize the occurrence or intensity of diseases, insect pests, and weeds (Gangwar and Prasad, 2005). Example new technologies include system-based integrated nutrient management, integrated pest management and resource conservation technologies such as, zero tillage, furrow-irrigated raised beds, and ridge or bed planting.

Biodiversity conservation is an important service, though one that is difficult to pin down precisely. Metrics of biodiversity typically refer to native species or systems and should reflect conditions at species, ecosystem, and landscape levels in order to capture changes in number of species, types of and interactions between ecological systems and how those systems are dispersed across space (Franklin, 1993). Natural biodiversity provides the genetic and biochemical resources that allow agricultural and pharmaceutical innovations. Use of the genetic diversity is estimated to contribute $1 billion in annual increases in crop productivity (NRC, 1992), but the value is much greater if the aesthetic and recreational value of diversity is included.

Habitat benefits that can be measured include the size, shape and integrity of ecological systems that provide essential conditions for particular species. Habitat can only be defined with a particular species or set of species in mind (e.g., free flowing streams for trout). The importance of habitat loss or degradation is typically proportional to the amount of habitat remaining for the species of concern. Measures of land-cover change or fragmentation over time reflect how agricultural practices affect those systems. Similarly, structural changes in vegetation or stream conditions can affect habitat (e.g., amount of downed wood in the forest or the stream). Landscape metrics that can capture changes in habitat conditions over time and space are well documented (Gustafson, 1998). Obtaining in situ measures of habitat quality is often time consuming and requires focus on the particular species of concern.

Protection from disturbances is a final major service of agricultural lands. This benefit can be achieved by protection of particular cover types (e.g., wetlands are known for their ability to reduce flooding) and land-use practices (e.g., use of perennial crops especially on slopes reduces erosion and subsequent flooding as streambeds become filled with silt). Increased frequency and severity of drought appears to be associated with broad-scale loss of vegetation cover. Quantifying the services provided to mitigating effects of disturbances requires that assumptions be made about future disturbances. It is not always possible to rely on past disturbance regimes because trends in frequency, severity, and extent may be altered along with climate change (Dale et al., 2000). Given certain disturbance regimes, protection can be estimated under various scenarios of land cover and land use.

5. Use of ecological indicators

We have presented indicators of ecosystem services as if they might be measured and interpreted independent of each other, but that is certainly not the case. The sixth criterion mentioned earlier that the metrics "be integrative." Hence the relations among metrics should be a major consideration in the selection of the set of indicators to be used for any location. The suite of indicators should capture the major ecosystem services of the system and be complimentary to each other.

In particular, ecological indicators should reveal cases where considerations of the suite of ecosystem services can increase the value of agricultural lands under alternative management as compared to current management. For example, remote agricultural lands in Wyoming that include wildlife habitat, angling opportunities and scenic vistas have higher prices per area than lands dominated by agricultural production (Bastian et al., 2002). Conservation buffers can benefit ecological condition by reducing erosion, improving water quality, increasing biodiversity, and expanding wildlife habitats. The comparison of ecological indicators should be useful to address questions about how best to implement and manage these buffers (Lovell and Sullivan, 2006). The challenge to land management is even greater when contrasting land uses abut each other. For example, streams and wetlands on non-farm lands may purify water with high nutrient content from adjacent agricultural lands. Natural habitat may provide valuable pollinator services to neighboring farms (Ricketts et al., 2004). In order to evaluate the effectiveness of management practices such as conservation buffers and alternative land uses, ecological indicators need to be identified that capture the diversity of amenities that can be provided.

Hence the set of indicators of ecosystem services should take into account the spatial context of the land under consideration and how the entire area can be affected by management decisions. In fact, the value of agriculture lands may best be viewed in a landscape context. Just as Mitsch and Gosselink (2000) document that wetlands are influenced by their agricultural and urban neighbors, the spatial context of agricultural lands affect their ecosystem services. Santelmann et al. (2004) and Boody et al. (2005) take a landscape perspective in documenting the effects of alternative management of agroecosystems in the Midwest U.S. on a range of ecosystem services. This broad-scale perspective is generally not taken because the valuation methodologies and/or data rarely exist to make this approach practical.

In a few cases, ecological indicators have been suggested that are based on ownership boundaries. For example, Rigby et al. (2001) developed a farm-level indicator of agricultural sustainability based on patterns of input use for 80 organic and 157 conventional producers in the United Kingdom. In other cases indicators focus on the entire human community and include ecological, economic and social dimensions of agriculture (e.g., Kammerbauer et al., 2001).

New challenges are raised by considering indicators of how agriculture systems affect and are affected by agriculture. The question of scale is at the forefront of these challenges. One of the first steps in measuring an ecosystem service is determining the appropriate spatial and temporal scale at which to obtain the measure. Broad-scale measures are useful to understand the overall effects on the system, and site-specific measures are useful to understand how particular management practices in a given field may affect services. Hence often measures at more than one scale are needed to obtain a comprehensive picture of the system. Then the question becomes how to relate the metrics obtained at different scales. Although this paper discusses how the issue has been addressed to some extent, understanding how ecosystem service metrics obtained at different resolutions affect interpretations is still a challenge.

A key challenge is to assemble the appropriate suite of indicators for ecosystem services relevant to a particular agro-ecosystem that captures key ways that agriculture activities can affect and are affected by ecosystem services. All of these types of indicators must be considered in order to have full understanding of agriculture's effects on ecological systems, but in the end only a select few will be able to be measured because of resource limitations.

By focusing on ecosystem services provided to and affected by agriculture, the relationship of agriculture practices to the ecosystems in which they occur will be made more clearly. Obviously people value the food provided by agriculture, but other benefits can be important as well. Furthermore, being able to quantify how agriculture can affect ecosystem services is necessary to perform a full accounting of the costs and benefits of agriculture both worldwide and in specific locations (Tilman et al., 2002). Understanding the benefits and costs of different types of management practices is necessary in order to be able to establish and maintain sustainable agro-ecosystems.

6. Conclusions

Ecosystem indicators can be used for a variety of purposes. Indicators can help target key environmental problems and opportunities for improvement. Indicators can be helpful in designing effective farming systems and in monitoring them. They can also be used to predict efficiency of different farm practices (e.g., row cropping versus perennial plants) and to evaluate the repercussions of site-specific conditions (e.g., soils, slope) or place specific events (e.g., weather). In a landscape context, indicators can be useful in evaluating the overall ecosystem services provided by agricultural lands in contrast to other land uses.

Acknowledgments

The project was partially funded by a contract between the Strategic Environmental Research and Development Program (SERDP) Ecosystem Management Program (SEMP) and Oak Ridge National Laboratory (ORNL). Oak Ridge National Laboratory is managed by the UT-Battelle, LLC, for the U.S. Department of Energy under contract DE-AC05-00OR22725.

REFERENCES

Baker, J.M., Ochsner, T.E., Venterea, R.T., Griffis, T.J., 2007. Tillage and soil carbon sequestration — what do we really know? Agriculture, Ecosystems and Environment 118, 1–5.

Bastian, C.T., McLeod, D.M., Germino, M.J., Reiners, W.A., Blasko, B.J., 2002. Environmental amenities and agricultural land values: a hedonic model using geographic information systems data. Ecological Economics 40, 337–349.

Bockstaller, C., Girardin, P., van der Werf, H.M.G., 1997. Use of agro-ecological indicators for the evaluation of farming systems. European Journal of Agronomy 7, 261–270.

Boody, G., Vondracek, B., Andow, D.A., Krinke, M., Westra, J., Zimmerman, J., Welle, P., 2005. Multifunctional agriculture in the United States. Bioscience 55 (1), 27–38.

Boyd, J., Banzhaf, S., 2006. What are Ecosystem Services? Resources for the Future Discussion Paper 06-02. Resources for the Future, Washington, DC.

Cairns, J., McCormick, P.V., Niederlehner, B.R., 1993. A proposed framework for developing indicators of ecosystem health. Hydrobiologia 236, 1–44.

Corwin, D.L., Lesch, S.M., Oster, J.D., Kaffka, S.R., 2006. Monitoring management-induced spatio-temporal changes in soil quality through soil sampling directed by apparent electrical conductivity. Geoderma 131 (3–4), 369–387.

Daily, G., 1997. Nature's Services. Island Press, Washington, D.C.

Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecological Indicators 1, 3–10.

Dale, V.H., Joyce, L.A., McNulty, S., Neilson, R.P., 2000. The interplay between climate change, forests, and disturbances. The Science of the Total Environment 262, 201–204.

Dale, V.H., Mulholland, P., Olsen, L.M., Feminella, J., Maloney, K., White, D.C., Peacock, A., Foster, T., 2004. Selecting a suite of ecological indicators for resource management. In: Kapustka, L.A., Gilbraith, H., Luxon, M., Biddinger, G.R. (Eds.), Landscape Ecology and Wildlife Habitat Evaluation: Critical Information for Ecological Risk Assessment, Land-Use Management Activities and Biodiversity Enhancement Practices, ASTM STP 11813. ASTM International, West Conshohocken, PA, pp. 3–17.

DeBach, P., 1974. Biological Control by Natural Enemies Cambridge University Press, London.

de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics 41, 393–408.

Duarte, R., Sanchez-Choliz, J., Bielsa, J., 2002. Water use in the Spanish economy: an input–output approach. Ecological Economics 43, 71–85.

Feng, H.L., 2005. The dynamics of carbon sequestration and alternative carbon accounting, with an application to the upper Mississippi River Basin. Ecological Economics 54, 23–35.

Food and Agriculture Organization (FAO), 2004. Statistics from www.faostat.fao.org, updated February 2004.

Foster, T., Black, B., Abrams, M., 2004. A witness tree analysis of the effects of Native American Indians on the pre-European settlement forests in east central Alabama. Human Ecology 32, 27–47.

Franklin, J.F., 1993. Preserving biodiversity: species, ecosystems or landscapes. Ecological Applications 3, 202–205.

Gallego, F.J., 2004. Remote sensing and land cover area estimation. International Journal of Remote Sensing 25, 3019–3047.

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., 2003. The nitrogen cascade. Bioscience 53, 341–356.

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

Gangwar, B., Prasad, K., 2005. Cropping system management for mitigation of second-generation problems in agriculture. Indian Journal of Agricultural Sciences 75 (2), 65–78.

Groffman, P.M., Altabet, M.A., Bohlke, J.K., Butterbach-Bahl, K., David, M.B., Firestone, M.K., Giblin, A.E., Kana, T.M., Nielsen, L.P., Voytek, M.A., 2006. Methods for measuring denitrification: diverse approaches to a difficult problem. Ecological Applications 16, 2091–2122.

Gustafson, E.J., 1998. Quantifying landscape spatial pattern: what is the state of the art? Ecosystems 1, 143–156.

Hobbs, P.J., Webb, J., Mottram, T.T., Grant, B., Misselbrook, T.M., 2004. Emissions of volatile organic compounds originating from UK livestock agriculture. Journal of the Science of Food and Agriculture 84, 1414–1420.

Houser, J.N., Mulholland, P.J., Maloney, K., 2006. Stream chemistry indicators of disturbance on military reservations. Journal of Environmental Quality 35, 352–365.

Howarth, R.W., 2004. Human acceleration of the nitrogen cycle: drivers, consequences, and steps toward solutions. Water Science and Technology 49, 7–13.

Hunsaker, C.T., Carpenter, D.E., 1990. Environmental Monitoring and Assessment Program: Ecological Indicators. Office of Research and Development, United States Environmental Protection Agency, Research Triangle Park, North Carolina.

Jackson, J., Kirby, M., Berger, W., Bjorndal, K., Botsford, L., Bourque, B., Bradbury, R., Cooke, R., Erlandson, J., Estes, J., Hughes, T., Kidwell, S., Lange, C., Lenihan, H., Pandolfi, J., Peterson, C., Steneck, R., Tegner, M., Warner, R., 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293, 629–638.

Jung, W.K., Kitchen, N.R., Sudduth, K.A., Kremer, R.J., Motavalli, P.P., 2005. Relationship of apparent soil electrical conductivity to claypan soil properties. Soil Science Society of America Journal 69 (3), 883–892.

Karr, J.R., 1981. Assessment of biotic integrity using fish communitie. Fisheries 6, 21–27.

Kelly, J.R., Harwell, M.A., 1990. Indicators of ecosystem recovery. Environmental Management 14, 527–545.

Kammerbauer, J., Cordoba, B., Escolan, R., Flores, S., Ramirez, V., Zeledon, J., 2001. Identification of development indicators in tropical mountainous regions and some implications for natural resource policy designs: an integrated community case study. Ecological Economics 36, 45–60.

Kramer, S.B., Reganold, J.P., Glover, J.D., Bohannan, B.J.M., Mooney, H.A., 2006. Reduced nitrate leaching and enhanced denitrifier activity and efficiency in organically fertilized soils. Proceedings of the National Academy of Sciences of the United States of America 103, 4522–4527.

Kremen, C., Ricketts, T., 2000. Global perspectives on pollination disruptions. Conservation Biology 14, 1226–1228.

Lal, R., Reicosky, D.L., Hanson, J.D., 2007. Evolution of the plow over 10,000 years and the rationale for no-till farming. Soil and Tillage Research 93, 1–12.

Landres, P.B., Verner, J., Thomas, J.W., 1988. Ecological uses of vertebrate indicator species: a critique. Conservation Biology 2, 316–328.

Lorenz, C.M., Gilbert, A.J., Cofino, W.P., 1999. Indicators for transboundary river basin management. In: Pykh, Y.A., Hyatt, D.E., Lenz, R.J.M. (Eds.), Environmental Indices: System Analysis Approach. EOLSS Publishers Co. Ltd., Oxford, UK, pp. 313–328.

Lovell, S.T., Sullivan, W.C., 2006. Environmental benefits of conservation buffers in the United States: evidence, promise, and open questions. Agriculture Ecosystems and Environment 112, 249–260.

Maloney, K.O., Feminella, J.W., 2006. Evaluation of single-and multi-metric benthic macroinvertebrate indicators of catchment disturbance at the Fort Benning Military Installation, Georgia, USA. Ecological Indicators 6, 469–484.

Maloney, K.O., Mulholland, P.J., Feminella, J.W., 2005. Influence of catchment-scale military land use on physicochemical conditions in small Southeastern Plains streams (USA). Environmental Management 35, 677–691.

Meul, M., Nevens, F., Rehe ul, D., Hofman, G., 2007. Energy use efficiency of specialised dairy, arable and pig farms in Flanders. Agriculture Ecosystems and Environment 119, 135–144.

Millennium Ecosystem Assessment, 2005. Living Beyond Our Means: Natural Assets and Human Well-Being. Island Press, Washington, D.C.

Mitsch, W.J., Gosselink, J.G., 2000. The value of wetlands: importance of scale and landscape setting. Ecological Economics 35, 25–33.

Moore, A.A., Palmer, M.A., 2005. Invertebrate biodiversity in agricultural and urban headwater streams: implications for conservation and management. Ecological Applications 15, 1169–1177.

Mulholland, P.J., Houser, J.N., Maloney, K.O., 2005. Stream diurnal dissolved oxygen profiles as indicators of in-stream metabolism and disturbance effects: Fort Benning as a case study. Ecological Indicators 5, 243–252.

Myers, R.A., Worm, B., 2003. Rapid worldwide depletion of predatory fish communities. Nature 423, 280–283.

National Research Council (NRC), 1992. Managing global genetic resources: The U.S. National Plant Germplasm System. National Academy Press, Washington, D.C.

Naylor, R., Ehrlich, P., 1997. The value of natural pest control services in agriculture. In: Daily, G. (Ed.), Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, D.C., pp. 151–174.

Noon, B.R., Spies, T.A., Raphael, M.G., 1999. Conceptual basis for designing an effectiveness monitoring program. In: Mulder, B.S., et al. (Ed.), The Strategy and Design of the Effectiveness Monitoring Program for the Northwest Forest Plan. US Department of Agriculture, Forest Service, Gen. Tech. Rep. PNW-GTR-437, Portland, Oregon, pp. 21–48.

Ostman, O., Ekbom, B., Bengtsson, J., 2003. Yield increase attributable to aphid predation by ground-living polyphagous natural enemies in spring barley in Sweden. Ecological Economics 45, 149–158.

Pimentel, D., McLaughlin, L., Zepp, A., Lakitan, B., Kraus, T., Kleinman, P., Vancini, F., Roach, W., Graap, E., Keeton, W., Selig, G., 1989. Environmental and economic impacts of reducing U.S. agricultural pesticide use. Handbook of Pest Management in Agriculture 4, 223–278.

Pretty, J.N., Brett, C., Gee, D., Hine, R.E., Mason, C.F., Morison, J.I.L., Raven, H., Rayment, M.D., van der Bijl, G., 2000. An assessment of the total external costs of UK agriculture. Agricultural Systems 65, 113–136.

Redeker, K.R., Wang, N.Y., Low, J.C., McMillan, A., Tyler, S.C., Cicerone, R.J., 2000. Emissions of methyl halides and methane from rice paddies. Science 290, 966–969.

Ricketts, T.H., Daily, G.C., Ehrlich, P.R., Michener, C.D., 2004. Economic value of tropical forests to coffee production. Proceedings of the National Academy of Sciences 101 (34), 12579–12582.

Rigby, D., Woodhouse, P., Young, T., Burton, M., 2001. Constructing a farm level indicator of sustainable agricultural practice. Ecological Economics 39, 463–478.

Sanford, M.T., 1998. Pollination, the forgotten agricultural input. In: Ferguson, J., et al. (Ed.), Proceedings of the Florida Agricultural Conference and Trade Show. University of Florida, Lakeland, FL, pp. 45–47.

Santelmann, M., White, D., Freemark, K., Nassauer, J.I., Eilers, J.M., Vache, K.B., Danielson, B.J., Corry, R.C., Clark, M.E., Polasky, S.,

Cruse, R.M., Sifneos, J., Rustigian, H., Coiner, C., Wu, J., Debinski, D., 2004. Assessing alternative futures for agriculture in the U.S. Corn Belt. Landscape Ecology 19 (4), 357–374.
Scarpace, F.L., Quirk, B.K., 1980. Land-cover classification using digital processing of aerial imagery. Photogrammetric Engineering and Remote Sensing 46, 1059–1065.
Stamatiadis, S., Christofides, C., Tsadilas, C., Samaras, V., Schepers, J.S., 2006. Natural abundance of foliar N-15 as an early indicator of nitrogen deficiency in fertilized cotton. Journal of Plant Nutrition 29 (1), 113–125.
Suter, G., 1993. Ecological Risk Assessment. Lewis Publishers, Ann Arbor, Michigan.
Taube, F., Kelm, M., Loges, R., Wachendorf, M., 2006. Resource efficiency as a regulation variable for the promotion of sustainable production systems: are there priority areas for organic farming? Berichte uber Lanbwirtschaft 84, 73–105.
Tilman, D., Farigione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D., Swackhamer, D., 2001. Forecasting agriculturally driven global environmental change. Science 292, 281–284.
Tilman, D., Cassman, K., Matson, P., Naylor, R., Polasky, S., 2002. Agricultural sustainability and the costs and benefits of intensive production practices. Nature 418, 671–677.
Turner, R.K., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiou, S., 2003. Valuing nature: lessons learned and future research directions. Ecological Economics 46, 493–510.
U.S. EPA. Office of Atmospheric Programs, 2005. Greenhouse Gas Mitigation Potential in U.S. Forestry and Agriculture. EPA 430-R-05-006.
Viglizzo, E.F., Pordomingo, A.J., Castro, M.G., Lertora, F.A., 2003. Environmental assessment of agriculture at a regional scale in the Pampas of Argentina. Environmental Monitoring and Assessment 87 (2), 169–195.
Wallace, A., 1994. High-precision agriculture is an excellent tool for conservation of natural resources. Communications in Soil Science and Plant Analysis 25, 45–49.