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Forest Land Conversion, Ecosystem Services, and Economic Issues for Policy: A Review

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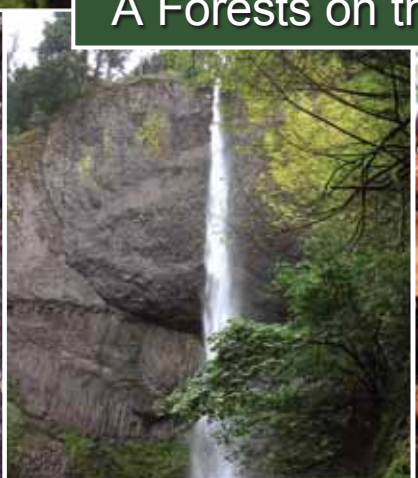
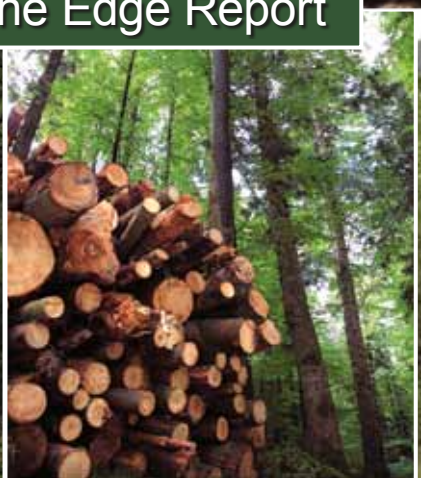
A Forests on the Edge Report





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Abstract

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The continued conversion and development of forest land pose a serious threat to the ecosystem services derived from forested landscapes. We argue that developing an understanding of the full range of consequences from forest conversion requires understanding the effects of such conversion on both components of ecosystem services: products and processes. However, there are unavoidable challenges involved in quantifying the threats from forest conversion and their related costs to human well-being. First, most attempts to quantify the costs of forest conversion on ecosystem services will necessarily rely on specific ecological science that is often emerging, changing, or simply nonexistent. For example, the role that many species play in ecosystem processes is poorly understood. Second, given the interconnected nature of ecosystem products and processes, any attempt to quantify the effects of forest conversion must grapple with jointness in production. For example, the cost of losing a species from forest conversion must account for that species' role as both (1) a product that directly contributes to human well-being, and (2) as a component in an ecosystem process. Finally, the ecology and the human dimensions of ecosystems are highly specific to spatial-temporal circumstances. Consequently, the effects of forest conversion in one spatial-temporal context are likely to be quite different than effects elsewhere.

Keywords: Ecosystem services, economic analysis, forest conversion, habitat fragmentation.

Introduction

The worldwide conversion of forest land to commercial and residential use is increasingly affecting the ability of ecosystems to provide basic services to humankind (Foley et al. 2005). This is particularly true for private rural lands in the United States (Alig et al. 2004). Conversion of forest land affects both private and public forest ownerships; for example, Stein et al. (2005) documented pressures on U.S. national forests from development on neighboring private forest lands, especially in counties that have experienced significant population increases in recent years (Garber-Yonts 2004, Johnson and Stewart 2007, USDA FS 2006). Predicted increases in population growth over the coming decades are expected to result in steadily increasing fragmentation of currently cohesive forest lands (Plantinga et al. 2007, Stein et al. 2007). This development can potentially reduce the goods and services derived from both public and privately owned forests. However, the extent to which the goods and services produced by forests are compromised by development is difficult to predict because human impacts on natural systems are not fully understood and are rarely straightforward.

In recent years, a multidisciplinary research tradition has emerged to address the complex dependence of humankind on services generated by functional ecosystems. In its broadest form, ecosystem service research focuses on what goods and services are produced by ecosystems, how they are valued, how human activities impact their production, and how to design policies to enhance their provision. Given this focus, ecosystem service research provides a useful conceptual framework for beginning to understand the effects of forest conversion in the United States on human well-being as derived from ecosystems. Furthermore, its multidisciplinary approach makes ecosystem service research well suited to understanding the potential for designing policies to conserve functional forest ecosystems.

Our approach emphasizes an application of the ecosystem services framework that differentiates the effects of forest conversion on both ecosystem *products* and *processes*.

The purpose of this report is to use the ecosystem service conceptual framework as a basis for understanding the ecological effects of forest-land conversion, and as a basis for understanding the economic issues that arise in designing policies to conserve forest ecosystems. Ecosystem services generally comprise two distinct attributes: (1) the direct products produced by ecosystems, and (2) the processes that produce the products. Rather than presenting a catalog treatment of the literature, our approach emphasizes an application of the ecosystem services framework that differentiates the effects of forest conversion on both ecosystem **products** and **processes**. We argue that such specificity is necessary for conceptually unpacking the various effects of forest conversion on human well-being, and we review the general state of knowledge in this area. Further, our analysis highlights the relevant state of knowledge associated with the economics of policy interventions

to enhance ecosystem services provision. We emphasize future research needs that can improve our understanding of policy interventions.

The report begins with a short background on ecosystem service research as well as some of its conceptual challenges. The need for specificity when adopting the ecosystem services framework to examine forest conversion is highlighted. Context is provided by reviewing the projections of forest conversion rates generated by the Forests-on-the-Edge project of the USDA Forest Service. Next, we present a literature review of previous ecosystem service research relevant to four services provided by forests in the United States: timber production, water quality, wildlife, and carbon sequestration. This review is notable for explicitly differentiating the effects of forest conversion on ecosystem products and processes. Further, we discuss the various policy issues associated with ecosystem services provision and how economics research can help support policy analyses, and, finally, concluding thoughts.

Ecosystem Services as a Concept

Broadly speaking, ecosystem services are the natural processes that sustain human life (Daily 1997). In this sense, understanding ecosystem services has been a cornerstone impulsion and justification for the study of ecology since George Perkins Marsh's seminal 1864 *Man and Nature*. However, it was not until the 1970s that ecosystem services were identified as a formal topic of study (Mooney and Ehrlich 1997). Drawing on many disciplines ranging from ecology to economics, the study of ecosystem services never gained prominence within any single field, although ecosystem services research gained significant academic traction in the late 1990s, resulting in the publication of several high-profile articles and books (Costanza et al. 1997, Daily 1997, Dissmeyer 2000).

Since its recent emergence, the study of ecosystem services has been increasingly taken up by both governmental and nongovernmental agencies as well as many research institutions. For instance, there are scores of researchers devoted to the study of ecosystem services in many American agencies, such as the USDA Forest Service, U.S. Fish and Wildlife Service, and the Environmental Protection Agency. In addition, several far-reaching, collaborative projects such as the Millennium Ecosystem Assessment (MA) and the Natural Capital Project (NCP) have been organized to bring together researchers from different regions and academic backgrounds around the study of ecosystem services. The goal of the MA (2008) was to understand the effects of anthropogenic ecosystem change, and it produced several technical volumes and synthesis reports aimed at grasping the current state

and trends in the production of ecosystem services. By comparison, the NCP (2008) is more focused on developing scientific and financial tools to "motivate and finance conservation." Finally, research related to ecosystem services can be found in many academic journals dedicated to the natural or social aspects of environmental science. Taken together, these efforts have done much to advance the understanding of ecosystem services, particularly in the field of valuation (Kremen 2005). Consequently, ecosystem service research has provided a framework by which the consequences of ecosystem change for human welfare can be more fully understood.

It is important to stress the utilitarian aspect of the ecosystem services framework as illustrated, for example, in the MA's linkage of ecosystem services and human well-being. The MA differentiated ecosystem services into provisioning, regulating, cultural, and supporting services and emphasized that each service can affect multiple aspects of human well-being. As such, changes in ecosystems are considered to be of concern because of the impacts of such changes on human well-being. The utilitarian aspect of the ecosystem services framework provides the basis for integrating the discipline of economics with natural science disciplines.

Considering its broad applicability and interdisciplinary appeal, it should not be surprising that several challenges have been associated with the study of ecosystem services. First, the concept of ecosystem services has been defined with diverse and sometimes contradictory descriptions (DeGroot et al. 2002, Kline 2007). When the literature is considered as a whole, it is difficult to imagine what aspect of nature, if anything, might not be considered an ecosystem service. By defining ecosystem services too broadly, there is the danger of diminishing the meaning of the concept (Ghazoul 2007). Second, much work focusing on ecosystem service valuation relies on ecological metrics that are loosely defined, not universally accepted, or simply nonexistent (Kline 2007, Kremen 2005). As a consequence, ecosystem service research will only be as strong as the ecological metrics on which it is based. Finally, challenges of scale and complexity are difficult to overcome within ecology alone (Allen and Hoeskstra 1992, Shugart 2004), and are particularly problematic when considered within the context of human value systems (Kline 2007, Reid et al. 2006, Tallis et al. 2008). As a result, the value and importance of ecosystem services is highly relative to the parameters of the individual study, leaving the absolute value and importance of ecosystem services as open questions (Costanza et al. 1997, Daily 1997, Kline 2007, Limburg et al. 2002, Maler et al. 2008). Although these challenges are significant, they represent potentially fertile ground for future research as well as opportunities to surmount inherent difficulties in bridging natural and social sciences (Chan et al. 2007, Reid et al. 2006).

The Need for Specificity With Ecosystem Services

The concept of ecosystem services is often used to simultaneously describe two interdependent yet discretely different things: the direct products produced by ecosystems and the processes that produce them (Boyd and Banzhaf 2006, Brown et al. 2007, de Groot et al. 2002, Kline 2007). By failing to distinguish between these two components, ecosystem service assessments run the risk of double counting services (Boyd and Banzhaf 2006) or ignoring important complexity (De Groot et al. 2002, Limburg et al. 2002). What then is the difference between ecosystem products and ecosystem processes? To de Groot and co-authors (2002), ecosystem products are end products produced by ecosystem processes. Kline (2007) considers the products as end services of ecosystems and the processes as intermediate services. Examples of ecosystem products include food, timber, fish, game, flood protection, and pollination. Conversely, examples of ecosystem processes are nutrient cycling, weathering of rock, population control, habitat provision, and conversion of solar energy into biomass. To stress this point, **ecosystem processes produce ecosystem products** (Boyd and Banzhaf 2006). In economic terms, ecosystem products are equivalent to output, whereas ecosystem processes are equivalent to the underlying technology that forms the production function. Nonmarket valuation studies quantify the effects of ecosystem products (not processes) on measures of human well-being, like the economic concept of willingness-to-pay.

At certain levels of analysis, the distinction between products and processes may be pedantic and unnecessary. However, when attempting to understand the long-term **costs** of land-use change on human well-being, this distinction is paramount because the danger of forest conversion is not only the one-time loss of an ecosystem product, but its presumed capacity to damage or disrupt ecosystem processes, thereby diminishing the capacity for the production of future products (Defries et al. 2004, Radeloff et al. 2005, Satake and Rudell 2007, Saunders et al. 1991, Tchir et al. 2004, Tscharntke et al. 2005, Vitousek and Mooney 1997). In analogous words, it's possible that the conversion of forest land to development reduces both the product coming off the assembly line, and the assembly line itself (fig. 1).

It is possible that the conversion of forest land to development reduces both the product coming off the assembly line, and the assembly line itself.

In addition to difficulties arising from separating product from process, the ecosystem service construct is also complicated by the inability to completely separate different ecosystem products and processes from one another (Daily 1997). For instance, sustainable human consumption of ecosystem products such as timber or wildlife is a major justification for ecosystem management. However, products such as wildlife and timber are key components in the production of other

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Figure 1—Forest conversion can cause loss of ecosystem products but also can damage or disrupt ecosystem processes.

ecosystem products and the consumption of one may affect the production of the other (G. Nelson et al. 2006, E. Nelson et al. 2008). For instance, trees produce timber—an ecosystem product—while simultaneously serving as a component of the process that sequesters carbon. Although harvesting timber is the realization of one valuable ecosystem service, it can come at the expense of the amount of carbon sequestered in a forest. Similarly, Jackson et al. (2005) found that strategies for the promotion of carbon sequestration could negatively affect the amount of clean water produced by forests, and a variety of researchers have found that carbon sequestration strategies may not necessarily be compatible with biodiversity conservation among plant and bird populations (Huston and Marland 2003, Matthews et al. 2002, Nelson et al. 2008).

Difficulties in differentiating ecosystem products and processes are conceptually unavoidable, necessitating the incorporation of complexity into the study of systems providing ecosystem services (Limburg et al. 2002). This means that any attempt to understand the effects of forest conversion on ecosystem services must clearly define the specific social, spatial, and temporal frames in which the study takes place (Pentilla et al. 2006, Symstad et al. 2003). As Defries et al. (2004: 252) indicated, "ecosystem responses to land use vary in space and time, and assessing

trade-offs associated with land-use decisions requires explicit recognition of the scale of analysis." Furthermore, embracing complexity requires explicit consideration of the tradeoffs involved when promoting any ecosystem service. As Ghazoul (2007) suggested, failing to transparently consider the complex tradeoffs between different ecosystem services seriously limits the conclusions of any research project, in addition to diminishing the credibility of ecosystem services as a concept. Although this may detract from the value of some ecosystem services, "a detailed understanding of the benefits of conservation will only guide policy in conjunction with an equally detailed understanding of the costs" (Chan et al. 2007: 62).

Current Projections of Forest Conversion in the United States

Over the last several decades, the demographic and ownership trends on and adjacent to U.S. forest lands have begun to change. For example, one recent study documents that in the Northeast and upper Great Lakes States, inmigration rates are significantly higher in counties with large amounts of public land—particularly national forests—than those counties with less public land (Lewis et al. 2002). Although changes in forest ownership have resulted from many different social and market forces, the end result has been the conversion of private forest land to smaller, more intensively used residential parcels (Johnson and Beale 2002, Johnson and Stewart 2007, Radeloff et al. 2005).

As a result of these changes, many efforts have been made to forecast where and to what extent forest conversion might occur. Some of these studies, such as those done for the Resource Planning Act (RPA) of 1974, have been undertaken to monitor the status of forest resources. Others, including Johnson and Stewart (2007), Hammer et al. (2004), and Radeloff et al. (2005) have used residential density patterns to demonstrate past and possible future increases in the development of rural forest land. Specifically, a recent study by Stein et al. (2005) demonstrated that if trends continue, substantial amounts of private forest land will shift from a low-density rural character to a more densely populated exurban categorization. Using fourth-order watersheds[1] as the scale of analysis, this research identified all lands in the contiguous United States with at least 10 percent forest cover and at least 50 percent private ownership. By applying population growth trends to these lands, the researchers found that many watersheds, particularly in the Northeast and Southeast could experience significant conversion from rural to exurban or

[1] The average size of fourth-order watersheds in the United States. is approximately 1 million acres (404 686 hectares).

exurban to urban uses. In particular, the Stein et al. (2005) results show that 65 of the approximately 2,100 fourth-order watersheds in the United States will see 10 to 20 percent of their forest area converted from a rural use to a more developed urban use by the year 2030.

The large expected increase in the development of forested watersheds found in the Stein et al. (2005) analysis could have substantial implications for the ecosystem services provided by U.S. forests, both public and privately owned. Although public land agencies can manage their own lands toward ecological outcomes, they rarely have the capacity to affect the management on private lands. Although this is not a major problem when neighboring lands are managed for similar uses, it can become very problematic when the use of public lands and neighboring private lands diverge. The conversion of forest land to residential use represents such a challenge and can potentially hinder the ability of the public land agencies to provide critical ecosystem services.

Forest-Based Ecosystem Services

The ecosystem services literature abounds with lists of products and processes that could be considered ecosystem services. Although many services are potentially provided for by U.S. forest ecosystems, four were chosen as illustrative of biotic and abiotic services typically found in the Nation's forests: provision of timber, clean water, and wildlife, and carbon sequestration. Each of these will be briefly considered as an ecosystem **product** as well as a component in other ecosystem **processes**. Furthermore, previous research identifying the potential effects of forest conversion on each will be briefly presented. We provide multiple references for those interested in more detail.

Timber stands often represent a critical part of forest ecosystems and the relation between timber and other components such as nontimber tree species, water quality, and wildlife is highly site specific.

Timber

Of all ecosystem products and processes on forest lands, timber is one of the most integral to market economies. The term timber generally refers to the portion of trees that can be used to produce market goods like furniture or paper. Not only is timber the primary market product of many forests, but the trees used to produce timber also provide a critical component in the production of nearly every other ecosystem product generated by forest land. For example, standing trees form the primary habitat for many migratory songbirds, in addition to serving as a storehouse of carbon. Furthermore, timber stands often represent a critical part of forest ecosystems and the relation between timber and other components such as nontimber tree species, water quality, and wildlife is highly site specific. Consequently, the realization of timber as an ecosystem product through harvesting trees

can potentially have dramatic positive or negative effects on ecosystem processes that produce other products. Recognition of the complex importance of timber both as a product and component in the production of other products was crucial to the establishment of the United States' first forest reserves (Dombeck et al. 2003, Steen 1991). Although timber production may not always be directly impacted by neighboring forest conversion, there are many indirect effects that can diminish a forested region's overall productivity.

Timber as an ecosystem service—

The use of timber as a forest ecosystem product has been a source of conflict as well as a driving component of human geography (Cronon 1992). Timber is one of the few, if not the only, major forest ecosystem products for which there are fully functional markets. Although the body of literature on timber as a product is voluminous, the recent trend toward ecosystem management has moved away from viewing trees only as a source of timber, similar to other crops. Much recent work has emphasized the most obvious tradeoff in producing timber—the cutting of trees.

In addition to their importance in producing the ecosystem product of timber, trees are a critical component in the production of many other ecosystem services. In fact, the recognized importance of standing trees in producing water quality (an ecosystem product) was crucial in the earliest movements to preserve large areas of forest. Protection of New York City's water supply was a major reason for the establishment of the Adirondack and Catskill Reserves in 1885 (Dombeck et al. 2003). Several years later, water protection and regulation were fundamental reasons for passage of the U.S. Forest Reserve Act of 1891, the precursor to the national forests (Steen 1991).

Shortly after the establishment of the first forest reserves, many researchers began to recognize the importance of forest stands to the production of wildlife (Meine 1991). Trees, as the critical component of many plant communities, are often the determining factor in the composition of animal communities. Understanding this relationship was crucial to the development of ecological science, and the research regarding the role of standing trees in the production of wildlife is vast. However, interest in the roles that trees play in providing habitat for wildlife peaked in the 1990s in an effort to understand the role of old-growth forest habitat as it pertained to endangered species such as the spotted owl (*Strix occidentalis caurina*) (Diaz and Haynes 2002). From this, a fuller understanding of wildlife's dependence on the quantity, age-class distribution, and spatial configuration of standing trees has emerged.

Finally, standing trees play a pivotal role in carbon sequestration. By definition, carbon sequestration is dependent on the retention of carbon within some medium such as water, soil, or vegetation. As it relates to forest land, standing trees are often the primary means of storing carbon or transferring carbon to the soil in the form of dead plant matter. Therefore, the conversion of forest land to other uses may directly reduce the potential to sequester carbon (Seely et al. 2002).

Given the immense importance of standing trees to the production of other ecosystem services, the consumption of timber can directly result in the diminished production of other services. The extent of the tradeoff between timber production and other ecosystem services is an empirical question that has only recently been addressed (e.g., see Polasky et al. 2008 for a recent example). Recognition of this tradeoff has been at the center of a shift in management of the U.S. national forests (Dombeck et al. 2003) resulting in the emerging perspective that standing trees cannot only be valued as timber, but also for their role in the process of producing other ecosystem products.

Effect of forest conversion on timber—

Three indirect consequences of forest conversion stand out as particularly problematic for managing forests for timber: (1) changes in regional timber economies, (2) introduction of nonnative species, and (3) increased fire risk. In many rural and forested areas, local economies are based largely on timber production from both public and private forest land. Some studies have documented strong negative effects of urbanization and commercial forestry operations (Munn et al. 2002, Wear et al. 1999). Because scale economies may be present in the production of timber (Romm et al. 1987), parcelization and the development of forested lands may cause private timberlands to become less productive (Gobster and Rickenbach 2004). This is thought to occur because parcelization and development can result in (1) a reduced timber tract size, and (2) a change in land ownership objectives resulting in reduced reinvestment in forest land (Kline et al. 2004). Consequently, reduced private forest-land productivity can increase pressure on public forest land and diminish the profitability of timber extractive industries (Luloff et al. 2000, Mehmood and Zhang 2001). Because timber extraction often occurs with low marginal profits, even small losses in timberland can have an impact on timber-dependent regional economies. Finally, increased residential development near public forests can bring increased opposition to any timber harvest that affects forest aesthetics (Ribe 2006). This loss of profitability and increased social conflict not only introduce new challenges to land mangers, but can potentially have broader effects on local economies dependent on timber as an ecosystem product.

In addition to social effects, one of the most consequential effects of forest conversion is increased forest edge and decreased core area. Generally speaking, this can lead to changes in forest composition, which can affect the biodiversity, stability, and complexity of the forest ecosystem (Bodin and Wiman 2007, Chapin et al. 2000, Diaz et al. 2006, Eiswerth and Haney 2001, Hansen et al. 2005, Kinzig et al. 2002, Pentilla et al. 2006, Quetier et al. 2007, Tilman 1982, Tilman et al. 2002). Such changes affect the ecosystem processes used to produce timber as an ecosystem product. More specifically, forest conversion can lead to the introduction of nonnative species and the favoring of species adapted to edge habitat (Dickens et al. 2005, Holway 2005, Yates et al. 2004). Nonnative pests can directly feed on or destroy valuable timber species. For example, in the Northeastern United States, hemlock woolly adelgid (*Adelges tsugae* Annaud), a small Asian, aphid-like insect that feeds on hemlock trees (*Tsuga* Carrière) has dispersed through the movement of infested nursery trees to woodland residences (USDA FS 2005). Similarly, the Great Lakes region of the United States has experienced the spread of the emerald ash borer (*Agrilus planipennis* Fairmaire), an invasive Asian beetle that feeds on and kills ash trees (*Fraxinus* L.) and (*Sorbus* L.). For this pest, a primary means of its long-range dispersion has been the movement of infested firewood to campgrounds and private woodlands (Bauer et al. 2003). A recent study in Ohio concludes that a widespread outbreak of emerald ash borer will result in economic damages between $1.8 billion and $7.6 billion (Sydnor et al. 2007).

In addition to tree-damaging pests, forest conversion can impact the ecosystem processes used to produce timber by helping to introduce and spread many nonnative plants that can outcompete and diminish native timber species. Introduction of these nonnative plants has occurred when ornamental plants such as English ivy (*Hedera helix* L.) and Japanese honeysuckle (*Lonicera japonica* Thunb.) escape domestic gardens and plantings. Alternatively, seeds of nonnative species such as leafy spurge (*Euphorbia esula* L.) and garlic mustard (*Alliaria petiolata* (M. Bieb.) Cavara & Grande) can be spread by contaminated roadside mowers that are more prevalent near residential and commercial developments (Czarpata 2005).

Last, forest conversion can have significant effects on timber production through changing forest fire regimes. The construction of housing and commercial development, along with the requisite road and utility construction, generally introduces more fire threats such as automobiles, campfires, and construction equipment. Consequently, the risk of forest fire is generally expected to increase with increased forest conversion (Haight et al. 2004). Alternatively, increased development in the wildland-urban interface can complicate efforts to reincorporate fire into the ecology of ecosystems that have adapted to frequent fires. For example, residents

often oppose the use of prescribed fire for fear that it will escape control (Winter and Fried 2000). In addition, many residents oppose the use of fire as a management tool because they do not trust the agencies, appreciate the outcomes, or perceive a personal benefit of nearby fire (Winter et al. 2002). Therefore, homeowner opposition often limits the use of fire as a timber management tool.

Clean Water

Water is a basic requirement for life, and there is a strong relationship between forest land and the filtration and provision of clean water. However, the availability of clean water has been poorly understood or taken for granted throughout much of human history (Dombeck et al. 2003, US EPA 2000). In fact, it was not until the 19th century that water treatment emerged as a field of scientific study. This early research, coupled with deforestation in the Eastern United States, ultimately led to an understanding of clean water as a product of forest ecosystems and provided justification for early attempts to conserve natural areas (Steen 1991). Along with developing an understanding of its role as an ecosystem product, research has long shown that water is also of great importance in the process that produces other ecosystem products such as timber and wildlife. Given its pivotal role as a product and process component, any changes to water cycling in an ecosystem can potentially have important effects on the production of other ecosystem products.

Two-thirds of the clean water supply in the United States is found in stream water from precipitation that is filtered through forests.

Clean water as an ecosystem service—
Clean water used for drinking or irrigation is an ecosystem product. Ecosystems provide clean water by holding and filtering water while regulating its flow to downstream locations (Dissmeyer 2000). Two-thirds of the clean water supply in the United States is found in stream water from precipitation that is filtered through forests (National Research Council 2000) (fig. 2). The holding and filtering of water is a particularly important function of private forests as 60 percent of the Nation's runoff flows from private lands (Stein et al. 2006). Ecology has long recognized that the quality, amount, timing, and regularity of water flows are fundamental parameters to which ecosystems adapt. Therefore, disruptions of these parameters can have potentially large effects on the processes that produce

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Figure 2—The majority of clean water supply in the United States is found in stream water from precipitation that is filtered through forest ecosystems.

other products. Although, some plants and animals are well adapted to a wide range of hydrological conditions, others can only survive in a very narrow range of conditions. This makes assessing the role of water as a process component highly dependent on the specific characteristics of particular ecosystems. Generally speaking, enhancing the stability of hydrological conditions is the critical role water plays as a process used to produce other ecosystem products (Graham and Smith 2004, Postel and Carpenter 1997, Sweeney et al. 2004).

Given its importance for human well-being, the value of water provided by ecosystems has been estimated by many studies. For example, the direct value of national forest water to humans has been estimated to exceed $27 billion per year (Brown 1992, Krieger 2001), although few studies have measured the value of clean water as it relates to the process of producing other ecosystem services.

Effect of forest conversion on clean water—
Conversion of forests to other uses, especially to developed uses with impervious surfaces, can have detrimental effects on water supplies (National Research Council 2000). Water quality can be degraded through increased water temperatures and sedimentation, and the risk of flooding in downstream areas can be increased with forest conversion. Other impacts can include the potential loss of aquatic habitat and related ecosystem services provided by streams and rivers. Because water's role in an ecosystem is highly specific to the characteristics of the particular ecosystem, the relationship between forest conversion and clean water is likewise spatially and temporally specific. Conceivably, some commercial and residential developments could have little effect on the production of clean water from neighboring forests. However, the construction of buildings, roads, and utilities can substantially increase surface runoff and add pollutants to hydrological systems. A large literature on this is summarized by several reviews (Forman and Alexander 1998, Trombulak and Frissell 2001).

The construction of impervious surfaces, channelization of streams, draining of wetlands, and installation of culverts can dramatically change the hydrology of a forest (Dissmeyer 2000, Zipperer 2002). For instance, the construction of roads, parking lots, and large buildings can speed the runoff of rainfall to local waterways, thereby intensifying the effects of storm events. This is particularly problematic in steep or mountainous areas, where even rudimentary logging roads can serve as drainage courses during heavy rains (Havlick 2002). Another hydrological challenge associated with development and road construction is the installation and maintenance of culverts. Roads in forested areas often cross numerous small streams, in which culverts are used to facilitate stream flow. Over time, these

culverts often become clogged or eroded so that water cannot flow through them, effectively creating small impoundments upstream and decreasing downstream flow (Tague and Band 2000).

In addition to the proliferation of roads, the conversion of forest land to residential and commercial development often involves disruption of wetlands, which provide many ecosystem processes used to produce other products. Although many regulations are in place to protect these valuable areas, development at their edge can remove protective buffers and interfere with the structural capacity of wetlands to store, clean, and cool water, especially in flood or drought periods (Burns et al. 2005). Consequently, these changes can alter the stability of the ecosystem by favoring certain species and marginalizing others. This is exemplified by the warming of riverine systems to favor warm-water species of fish at the expense of native cold-water fish such as trout (Havlick 2002).

Finally, the construction of residential and commercial developments in forest land can add pollutants to local watersheds with wide-ranging effects (Zedler 2003, Zipperer 2002). A primary source of residential pollution is the runoff of lawn fertilizer, which can promote aquatic plant growth in streams and lakes, thereby speeding eutrophication (Carpenter et al. 1998). Along with increased lawn fertilization, lawn pesticides associated with residential and commercial development can run off into neighboring streams and waterways, thereby upsetting ecological processes used to produce products such as sport fisheries (Overmyer et al. 2005). Of particular importance is the fact that the effects of nonpoint sources of pollution may not be clear until the sources have been in place for a number of years.

By speeding, slowing, or redirecting waterflow, forest conversion can potentially alter the hydrological conditions that regulate ecosystem functioning, and hence, impact important ecological processes. Not only can this alter the supply of clean water to humans (an ecosystem product), but it can destabilize the ecological processes that produce other products (e.g., sport fisheries) important to human well-being. Although many laws and policies for protecting water quality have been enacted over the last several decades, significant forest-land development is still generally allowed, along with the potential for a variety of effects on the production of clean water as an ecosystem service.

Wildlife

Prior to the last years of the 19th century, the concept of animal extinction was foreign to many people. However, firsthand experience demonstrated that overhunting and loss of habitat could devastate wildlife populations to the point of annihilation. In recent years, wildlife production has been seen as an ecological function, not

just a result of reproduction (Allen and Hoekstra 1992). In this sense, wildlife has been considered an ecosystem product for some time (fig. 3). However, we are just beginning to understand the role wildlife plays in maintaining ecological processes. As with timber and clean water, certain aspects of wildlife as an ecosystem service are particularly susceptible to forest conversion.

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Figure 3—Observation of wildlife is a popular recreational activity for many people and can be affected by land management and open space conservation that influences multiple ecosystem services.

Wildlife as an ecosystem service—

Much work in ecology has derived from the fact that wildlife is highly dependent on, and responsive to, ecological interactions and the structure of the plant and animal community (Allen and Hoeskstra 1992, Askins 2002, Herrmann et al. 2005, Tilman et al. 2002). Therefore, our understanding of wildlife as a product of ecosystem function is perhaps richer than any other ecosystem service. As a product, wildlife can be "consumed" by many different means, including hunting, observation, or simply knowledge of its existence. Apart from the occasional user fee, there are few if any markets available to capture the value of these products. Therefore, wildlife, as an ecosystem product, has initiated much work in nonmarket valuation. These valuations have included charismatic species such as the bald eagle (*Haliaeetus leucocephalus*) and timber wolf (*Canis lupus lycaon*), endangered species such as the whooping crane (*Grus americana*) and spotted owl, and popular game

species such as trout (*Salvelinus* and *Oncorhynchus* sp.) and big-horned sheep (*Ovis canadensis*) (Loomis and White 1996). Estimated nonmarket values for wildlife are highly variable across species.

Although wildlife represents a highly important ecosystem product, it is also a critical component of other ecosystem processes.

Although wildlife represents a highly important ecosystem product, it is also a critical component of other ecosystem processes. For instance, wildlife often serve to pollinate plants, disperse seeds, and control pest populations for timber species (Allan et al. 2003, Kremen et al. 2004, Pentilla et al. 2006, Straub and Snyder 2006). Often these relationships are such that plant and animal species are mutually adapted to each other through predator-prey relationships, competitive hierarchies, food webs, and trophic cascades. Given these often complicated interactions, evaluating the importance of wildlife in the process of producing other products—like timber—can be highly specific to species and locations. In addition to its role in grooming the plant community, wildlife often plays a role in maintaining hydrology by foraging on shoreline vegetation, wallowing in streams, or building dams (Dissmeyer 2000). This can result in extremely complex relationships, whereby aquatic species rely on the interaction between terrestrial species to maintain plants, which maintain hydrology that provide aquatic habitat. An examplar of this is the observation that wolves (*Canis* sp.), reintroduced to Yellowstone National Park, reduced the time elk (*Cervus* sp.) spend wallowing in streams, thereby improving trout habitat (Robbins 2004). Although fascinating, these complex relationships highlight the difficulty involved in evaluating the role of wildlife in the processes that produce other ecosystem products.

Effects of forest conversion on wildlife—

The effects of forest conversion on wildlife arise from the reduction and fragmentation of formerly contiguous habitat. In many circumstances, the areas bordering large sections of public and private forest lands serve as buffers, increasing the amount of core habitat available within the forest (Riitters et al. 2002). These buffers act as the forest edge and shelter core areas from edge effects that can arise from predators and nest parasites associated with nonforest habitat. A major concern is that the conversion of forests to residential and commercial use brings more roads and utilities, which effectively shrink or eliminate forest buffers. Consequently, forest conversion increases the proportion of edge and decreases the amount of core habitat within forested landscapes (Butler et al. 2004, Havlick 2002, Riitters et al. 2002). As with timber, much research has shown that an increased proportion of edge to core habitat can promote edge species and introduce nonnative wildlife, often at the expense of native species (Danielson et al. 1997, Deem et al. 2001, Lepczyk et al. 2003, Manolis et al. 2002, Radeloff et al. 2005, Riitters et al. 2002, Singleton et al. 2002).

Many species are negatively affected by the loss and fragmentation of forest habitat, including large mammals (Costa et al. 2005, Noss 1994) and Neotropical migratory songbirds (Askins 2002, Faaborg 2002). Songbirds have received particular interest in ecological science and are often considered indicators of ecosystem quality. Edge effects arise in conjunction with the boundary between natural environments and are particularly important for many songbird species (Askins 2002, Faaborg 2002). The breeding success of many bird species is affected by edge because of the increased proximity of nesting habitat to predators (e.g., house cats) and nest parasites (e.g., the brown-headed cowbird [*Molothrus ater*]). For forest-nesting birds, the ecological literature has shown that edge effects can extend from 50 m (160 ft) (Paton 1994) to 300 m (984 ft) (Van Horn et al. 1995) into forest patches. For many birds, breeding success is higher in core forest, defined as the interior area of a forest patch beyond the reach of edge effects (Askins 2002).

In addition to terrestrial species, forest conversion along lake shorelines can yield numerous effects on aquatic species associated with lakes. Converting forested shoreline to development can result in new residents clearing sunken logs along their shoreline property. Such sunken logs serve as habitat for a variety of aquatic species (Christensen et al. 1996). Shoreline forest conversion has also been shown to lead to a reduction in the growth rates of sport fish such as bluegills (*Lepomis macrochirus*) (Schindler et al. 2000) and potentially result in localized extinctions of amphibian species such as green frogs (*Rana clamitans melanota*) (Woodford and Meyer 2003). Other amphibians have also been shown to be affected by non-shoreline forest conversion (Kolozsvary and Swihart 1999, Lehtinen et al. 2003). Last, forested shoreline that is converted to development has been shown to coincide with an increase in aquatic species invasions arising from increased recreational use of lakes (Hrabik and Magnuson 1999).

Unlike plants, many animals are highly mobile and often rely on much larger areas of habitat, which can be broken up or eliminated by the construction of buildings and roads (Debinksi and Holt 2000, Rochelle et al. 1999). Requiring contiguous areas of habitat across both private and national forest land, many species populations are directly impacted by the conversion of forest to residential or commercial development. The research in this area is very extensive, showing that, in addition to nest parasites and increased predation by edge species, forest conversion can lead to potential population reductions through the direct loss of forage and breeding areas and increased roadside mortality (Debinksi and Holt 2002, Spellerberg 1998, Stephens et al. 2003). In addition, even limited fragmentation can act as a spatial barrier to species dispersion, thus isolating populations and threatening their long-term species viability (Askins 2002, Opdam et al. 1993).

As noted above, species can play multiple roles within animal communities, including predator-prey, pest suppression, and parasite-host. In this manner, species are often adapted to fit the unique geophysical characteristics of an area as well as adapted to fit within its unique collection of plant and animal species. Therefore, the decrease or loss of any one species in an area as a result of forest conversion to development can critically alter the stability, biodiversity, and complexity of the community as well as the relevant ecosystem (Aksins 2002, Armsworth and Roughgarden 2003, Luck et al. 2003). What is more troubling is that community roles are often complex, interdependent, and not intuitively obvious (Allen and Hoeskstra 1992). Therefore, the effects of forest conversion and development are rarely apparent or predictable and may not be observed or understood until population declines become irreversible.

By diminishing wildlife habitat, forest conversion can result in an immediate loss of certain species (an ecosystem product) and perhaps more consequentially, severely alter community relationships, thus constraining the ecosystem processes used to produce other critical services. The literature on the loss and fragmentation of forest habitat is voluminous and demonstrates a wide array of manners in which forest conversion can affect traditional ecological metrics such as biodiversity. This has been shown in several literature reviews (Saunders et al. 1991, Spellerberg 1998, Stephens et al. 2003). However, connections between land development and losses to particular ecosystem services associated with wildlife have rarely been made (Naidoo et al. 2008).

Carbon Sequestration

Because trees generally sequester carbon, and because carbon is a primary component in climate change, the possibility of using forests to reduce the consequences of climate change is a major recent policy objective (Alig 2003). Forestry mitigation alternatives to offset greenhouse gas emissions are appealing because they are often more cost-effective than many current options in other sectors (Lubowski et al. 2006).

Carbon sequestration as an ecosystem service—

Carbon sequestration is most clearly an ecosystem product. The basic process behind biological carbon sequestration is that carbon dioxide (CO_2) is taken up by plants through photosynthesis and the carbon is then stored in both living and recently dead plant biomass. Therefore, by increasing the production of plant biomass, it may be possible to mitigate some CO_2 emissions through tree planting, thus reducing the potential for catastrophic climate change (IPCC 2001).

For example, a recent study found that approximately one-third of the U.S. CO_2 reduction target under the Kyoto Protocol could be cost-effectively achieved with carbon sequestration policies in forests (Lubowski et al. 2006).

Despite the climate-control advantages, there are significant opportunity costs involved in sequestering carbon. Many studies have measured the economic costs of carbon sequestration, with results ranging from $25 to $90 per ton ($22.75 to $81.90/tonne) of carbon (Stavins and Richards 2005). In addition, carbon sequestration is an example of the opportunity costs that must be considered when promoting any ecosystem services. For example, fire is becoming increasingly recognized as a critical component of many healthy forest ecosystems, yet allowing forests to burn will result in the release of large amounts of carbon, thereby reversing the benefits of the sequestered carbon. Similarly, promoting carbon sequestration by encouraging afforestation can potentially reduce biological diversity and complexity, as well as impact hydrological stability (Bunker et al. 2005, Catovsky et al. 2002, Huston and Marland 2003, IPCC 2001, Jackson et al. 2005, Matthews et al. 2002, Nelson et al. 2008).

Effects of forest conversion on carbon sequestration—
The effects of forest conversion on carbon sequestration are similar to the effects on timber production discussed earlier (fig. 4). For example, reduction in total forest productivity could result in a net loss in biomass, thus reducing the amount of carbon stored in a forest. However, the effects of forest conversion and development on carbon sequestration are extremely site specific. For instance, in mesic forests, human development might reduce total biomass through tree cutting, removal of leaf litter, and clearing and burning of brush. However, it is possible that forest conversion in more arid environments could actually increase total biomass as lawns and ornamental trees are planted and irrigated (Pouyat et al. 2006). Further, Cathcart et al. (2007) found cases where conversion of agricultural land to development can result in net increases in carbon sequestration owing to an increase in forest cover. In any case, the scale, scope, permanence, and persistence of anthropogenic change will strongly influence the effects of forest conversion on carbon sequestration.

Economic Issues for Ecosystem Service Policy

Research that quantifies the effects of forest conversion on ecosystem services has motivated an important environmental management challenge focused on how to improve the provision of ecosystem services from private forest land. Given that forest conversion is driven by the decisions of private landowners, the design of policy aimed at ecosystem services provision requires an improved understanding

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Figure 4—Timber harvesting and forest replanting can be important activities affecting the temporal provision of an ecosystem service such as carbon sequestration.

of private landowner decisionmaking. Because land is actively traded in markets and is subject to a variety of policy constraints, landowner decisionmaking must be examined within the context of local, regional, and national markets. As shown repeatedly in the economics literature of nonindustrial private forest owners, market drivers (e.g., commodity prices, tree planting costs, interest rates) are key determinants of private landowner behavior—e.g., Beach et al. (2005) provided a comprehensive review of the relevant econometric evidence. Further, examining the role of any potential land policy must account for the incentive structure that arises from the interaction of a particular policy mechanism with the relevant market where private land is traded.

Policies for Ecosystem Services

Decisions regarding the use and development of private forest land are generally driven by the private benefits and costs associated with a landowner's action. Ecosystem services produced by land are generally not considered as part of the private benefits and costs of a landowner's decision because the property rights to such services are not well-defined (fig. 5). For example, if a privately owned tract of forest provides habitat for wildlife species such as migratory birds, the landowner

Ecosystem services produced by land are generally not considered as part of the private benefits and costs of a landowner's decision because the property rights to such services are not well-defined.

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Figure 5—Some ecosystem services, such as scenic views, are public goods, and market failure can result if parties working in their own self interest are unable to provide these services in desired quantities.

cannot generally exclude others from the benefits derived by viewing the birds or from the benefits derived by simply knowing they exist. Without the ability to exclude others from deriving the benefits of ecosystem services, the provision of such services by private landowners will be inefficient because the benefits are external to the landowner—i.e., they are public goods. In the presence of externalities and public goods, land-use policies can be used to modify landowner incentives to achieve the provision of ecosystem services from private lands.

Three common approaches are used in land-use policy design. First, voluntary incentives can be offered to landowners in return for a defined change in the management of land—these approaches are now often referred to as "payments for ecosystem services" (Jack et al. 2008). A principal example of voluntary incentives is the U.S. Conservation Reserve Program, which provides payments to farmers for altering their agricultural production practices for a set amount of time (typically 15 years). Second, government agencies and nonprofit organizations can directly purchase land for conservation, or purchase the development rights of land as part of a conservation easement. Widespread purchase of conservation easements by local land trusts is a prominent example. Third, regulatory approaches such as zoning can be set to exploit the police power authority of local government agencies.

Although zoning is most commonly set by town governments, there are examples of statewide land-use planning programs. In particular, Oregon has a widely discussed program that emphasizes urban growth boundaries in an attempt to cluster residential development close to major areas of population.

Efficient Versus Cost-Effective Policies for Ecosystem Services

A policy is efficient in providing forest ecosystem services when the social net benefits from allocating land to forest and other uses are maximized. Although this principle is simple to state, it is extremely difficult to implement for at least two reasons. First, the social value of many relevant ecosystem services is composed of nonmarket values that are not readily observable as market prices. Second, the costs of providing ecosystem services will be heterogeneous across landowners owing to factors such as variation in the location of land relative to urban areas, soil quality, non-pecuniary values to landowners, and management skills.

If an ecosystem product is a marketable commodity, the social value of a change in the product can be obtained by calculating the sum of the changes in consumers' and producers' surplus in the related market (e.g., see Freeman 2003). If the ecosystem product is not a marketable commodity, obtaining information on the social values of the product is the domain of nonmarket valuation techniques. The concept of willingness to pay is the conceptual foundation for defining economic benefits of ecosystem services. Nonmarket valuation techniques generally comprise surveys that directly ask individuals' willingness to pay for hypothetical changes in ecosystem services (i.e., stated preference), and methods that use behavior from related markets to infer willingness to pay (i.e., revealed preference). Kline (2007) provided a survey of nonmarket valuation methods applied to forest ecosystem services. Estimating the economic value of an ecosystem process requires explict quantification of the link between the process and an end product that contributes to human well-being, including the potential for jointness in producing multiple ecosystem products. Freeman (2003) provided an example of a bird species that is valued both for pollination of a commercial fruit species and its role in controlling insects that damage other commerically valuable plants or tree species. The value of a species that assists in producing multiple ecosystem products is the sum of the value of the end products (Freeman 2003).

A policy can be designed to yield the efficient allocation of an ecosystem service if the marginal benefit and marginal cost curves associated with that service are explicitly quantified. However, the marginal benefit curve is typically unknown.

A policy can be designed to yield the efficient allocation of an ecosystem service if the marginal benefit and marginal cost curves associated with that service are explicitly quantified. However, a common informational constraint for most ecosystem services is the fact that the marginal benefit curve is typically unknown. Therefore, many economic analyses of ecosystem services emphasize cost-effective

conservation, whereby a policy is designed to achieve a predefined ecosystem service goal at minimum cost. Designing policy to account for heterogeneity in conservation costs across landowners is a defining feature of cost-effectiveness analysis. Example policy goals can include specifying tons of carbon sequestered by a policy, or specifying the number of species conserved. Importantly, the marginal benefits of ecosystem services do not need to be known to perform an analysis of cost-effective conservation, as the goal is taken as given. However, ecological research can be particularly useful in assisting policymakers with the definition of policy goals.

Challenges With Designing Cost-Effective Policies for Ecosystem Services

Although many challenges affect the design of cost-effective policies for ecosystem service provision, three deserve particular research attention. First, in addition to the aggregate amount of forest in a region, the spatial pattern of forest blocks can be critically important for ecosystem service provision because of the effects of habitat fragmentation (Armsworth et al. 2004, Askins 2002). Therefore, policy design must explicitly account for any effects of the policy on landscape pattern. Second, because the willingness of landowners to accept conservation payments is generally private information, policies that focus on voluntary decisions (e.g., incentives, land purchases, and conservation easements) will be unable to directly determine the spatial pattern of conservation (Lewis and Plantinga 2007, Nelson et al. 2008). Third, although it is generally cheaper to buy land that is more remote from population centers, such remote land is also generally less threatened by development. Therefore, there is a tradeoff that arises in conserving threatened parcels with high land costs, or conserving cheap parcels that have little likelihood of being converted to development (Costello and Polasky 2004, Newburn et al. 2006). Such a tradeoff is unaviodable when budgets are limited.

The body of literature on how to allocate the spatial pattern of conservation is quite large and dominated by reserve-site selection (systematic conservation planning) studies that arose in the conservation biology literature. Reserve-site selection studies examine how to optimally allocate conserved land—typically for species conservation—subject to an area or budget constraint (e.g., Camm et al. 1996, Church et al. 1996, Csuti et al. 1997, Margules and Pressey 2000). A common theme in most analyses is the importance of considering the spatial pattern of conservation efforts. A notable strength of this literature is the detailed modeling of the relationship between species distributions and landscape composition and configuration. The primary weakness of these studies is that the approach includes

an unrealistic policy mechanism, characterized by a regulator with the power to dictate land-use decisions, and hence, landscape pattern. Such approaches are generally untenable in societies dominated by private property rights, and relaxing this assumption is important for future research.

The literature that examines the use of voluntary incentive policies for ecosystem service provision has received far less attention than the reserve-site selection literature, and is generally found in studies by resource economists. Theoretical work has focused on mechanism design in the presence of landowners' private information (Smith and Shogren 2002), whereas experimental work has examined the potential for offering bonus payments to conserve clustered habitat (Parkhurst et al. 2002). Empirical work has examined the role of voluntary incentives in reducing forest fragmentation (Lewis and Plantinga 2007, Lewis et al., in press) and in jointly providing carbon sequestration and species conservation (Nelson et al. 2008). A common theme from these studies is that although voluntary incentives can be used to improve the provision of ecosystem services, the private information regarding the costs of landowner conservation limits the cost-effectiveness of using incentives compared to what is theoretically possible if agencies had full information. The primary problem is that policymakers cannot directly control the spatial configuration of conserved land with incentive policies (Lewis and Plantinga 2007) despite the importance of spatial pattern on ecological outcomes. Many policy designs considered in the voluntary incentives literature are relatively simple, and much work remains in developing a fuller understanding of how to design voluntary incentives to achieve particular patterns of conservation. Further research on auction mechanisms aimed at eliciting landowner costs of conservation would be extremely helpful in solving the relevant information problems (Lewis et al. 2009).

The literature that examines the tradeoff between land costs and development threats is even less extensive than the literature on voluntary incentives. In a theoretical analysis, Costello and Polasky (2004) showed that the timing of conservation decisions is critical in the presence of heterogeneous land costs and development threats. In an empirical implementation that combines models of land cost and the threat of land-use conversion, Newburn et al. (2006) showed how optimal consideration of the tradeoff between land costs and development threats alters the spatial conservation decision relative to the typical example from the reserve-site selection literature. Although this literature is sparse, improvements can be made by continuing the refinement of spatial econometric and simulation methods to improve the spatial forecasts of development threats and ecosystem change (Lewis 2009). As documented elsewhere, the literature developing spatial models of parcel-scale development decisions is extremely limited (Nilsson et al. 2003), and much work

remains to be done to develop accurate land-use forecasting models capable of incorporating realistic policy constraints and uncertainty associated with future market conditions.

Quantifying Interactions Among Policy, Markets, and Ecosystem Services

The research that explicitly quantifies the effects of market conditions or specific policies on current or future ecosystem services is limited, and provides another good avenue for research. Two specific research problems are relevant. First, how is the supply of ecosystem services expected to change in the future, given a range of uncertain market conditions? Second, how have ecosystem services been affected by past market conditions and policy decisions?

Forecasting the provision of ecosystem services has been generally labeled as "ecological forecasting" in the literature (e.g., see Clark et al. 2001). Some relevant research from ecology has focused on extrapolating past land-use trends as a means for forecasting changes in ecosystem services (e.g., Pimm and Raven 2000, Tilman et al. 2001). However, such approaches have generally been conducted at an aggregate scale and are unable to account for either the spatial pattern of land-use change, or the role of market conditions or policy in altering forecasts. Research using economic methods to generate ecological forecasts—although promising—has been minimal. Lewis (2009) developed an integrated econometric-simulation framework to forecast distributions of the spatial pattern of parcel-scale land-use change. The integrated framework is estimated from actual landowner decisions and is used to forecast extinction probabilities of an amphibian species across a large set of lake shorelines in the forested region of northern Wisconsin. In particular, the method can explicitly quantify the effects of alternative market conditions or zoning rules on changes in the extinction probabilities, providing an important quantitative link between policy, private landowner decisions, and expected changes in a specific ecosystem product.

Research documenting the effects of prior policy decisions on ecosystem services is also fairly limited, yet would provide important information in assessing the performance of past policies.

Research documenting the effects of prior policy decisions on ecosystem services is also fairly limited, yet would provide important information in assessing the performance of past policies. A notable example is that of Stavins and Jaffe (1990), who developed an economic land-use change model to conclude that 30 percent of the forested wetlands depleted in the lower Mississippi waterway in the mid 20th century resulted from federal flood control projects. A second example is the analysis of Chomitz and Gray (1996), who developed an economic land-use change model of deforestation in Belize, and found that road construction (much of which was induced by policy) contributed substantially to tropical deforestation. A

third example is by Andam et al. (2008), who used statistical matching methods to show that government-established protected areas had minimal effects on tropical deforestation rates in Costa Rica.

Conclusion

The continued conversion and development of forest land pose a serious threat to the provision of ecosystem services derived from forested landscapes. In this report, we argue that developing an understanding of the full range of consequences from forest conversion requires understanding the effects of such conversion on both components of ecosystem services: products **and** processes. However, there are unavoidable challenges involved in quantifying the threats from forest conversion and their related costs to human well-being. First, most attempts to quantify the costs of forest conversion on ecosystem services will necessarily rely on specific ecological science that is often emerging, changing, or simply nonexistent. For example, the role that many species play in ecosystem processes is poorly understood. Second, given the interconnected nature of ecosystem products and processes, any attempt to quantify the effects of forest conversion must grapple with jointness in production. For example, the cost of losing a species from forest conversion must account for that species' role as both (1) a product that directly contributes to human well-being, and (2) as a component in an ecosystem process. Finally, the ecology and the human dimensions of ecosystems are highly specific to spatial-temporal circumstances. Consequently, the effects of forest conversion in one spatial-temporal context are likely to be quite different than effects elsewhere.

Although challenges exist in quantifying the effects of forest conversion on ecosystem services, it is nevertheless clear that private land development in forested ecosystems can have numerous adverse effects on ecosystem services. Projections based on geographical methods (e.g., Theobald 2005) have been quite useful in forecasting aggregate (e.g., county or census-tract scale) changes in land development—a necessary step in projecting changes in ecosystem services. However, these approaches are generally not suited for policy analysis because the models are typically not related to either economic returns or policy constraints such as zoning. As such, there is significant potential for economists to collaborate with geographers and ecologists to improve our analyses and understanding of potential policies to provide ecosystem services.

In general, the optimal design and spatial allocation of policy tools to conserve forest ecosystem services is not well-understood. Empirical analysis that improves our understanding of conservation policy toward ecosystem services can be

Further research on the nonmarket values of ecological products is necessary to better understand the public's demand for ecosystem services.

enhanced in two ways. First, further research on the nonmarket values of ecological products is necessary to better understand the public's demand for ecosystem services. Second, the supply side of ecosystem services can be better understood by continued development of spatially explicit land conversion models capable of linking landowner behavior and policy variables (e.g., Lewis and Plantinga 2007, Newburn et al. 2006, Stavins and Jaffe 1990). Such models are useful in documenting the costs of providing ecosystem services, as well as providing a platform for examining the cost-effectiveness of alternative policy design. Additional research should emphasize how ecosystem services are expected to change in the future under a range of uncertain market conditions, and how ecosystem services have been affected by past policy decisions (fig. 6).

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Figure 6—Future research could usefully emphasize how ecosystem services, such as open space, are expected to change in the future under a range of uncertain land market conditions.

Throughout this report we have emphasized the conceptual framework of delineating ecosystem services into products and processes. Such a distinction is important when considering the consequences of forest conversion because of the interconnectedness of products and processes, and the fact that a loss of a particular ecosystem component today implies not just a one-time loss of an ecosystem product, but also the potential for disrupting ecosystem processes that will limit

the potential production of other products. For example, losing a particular bird species means losing an ecosystem product valued by bird watchers today in addition to losing that bird's contribution in, say, pollinating commercially valuable fruit species. Nonmarket valuation techniques can be used to value ecosystem products, but information is needed on the relationship between products and processes to properly value all relevant services. Therefore, future research that quantifies the relationship between products and processes is a necessary step in fully accounting for the full range of consequences that arise from forest conversion. Further, policy design can be greatly aided by ecological research that quantifies the products and processes most affected by forest conversion.

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Forests on the Edge

Forests on the Edge is a project of the U.S. Department of Agriculture, Forest Service, State and Private Forestry, Cooperative Forestry staff, in conjunction with Forest Service Research and Development. The project aims to increase public understanding of the contributions of and pressures on America's forests, and to create new tools for strategic planning.

The first report, *Forests on the Edge: Housing Development on America's Private Forests* (Stein et al. 2005), identified private forested watersheds in the conterminous United States most likely to experience housing development. The second, *National Forests on the Edge: Development Pressures on America's National Forests and Grasslands* (Stein et al. 2007), identified national forests and grasslands most likely to experience increased housing density on rural private lands on their borders. A third report, *A Closer Look at Forests on the Edge* (White and Mazza 2008), examined projected future development on private forests in Georgia, Washington, and Maine. A fourth report, *A Sensitivity Analysis of "Forests on the Edge: Housing Development on America's Private Forests"* (White et al. 2009), examined sensitivity of earlier results in the first report to four factors.

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