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Complexity, land-use modeling, and the human dimension: Fundamental challenges for mapping unknown outcome spaces

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Abstract

Land-use systems are characterized by complex interactions between human decision-makers and their biophysical environment. Mismatches between the scale of human drivers and the impacts of human decisions potentially threaten the ecological sustainability of these systems. This article reviews sources of complexity in land-use systems, moving from the human decision level to human interactions to effects over space, time and scale. Selected challenges in modeling such systems and potential resolutions are discussed, including strategies to empiricize complex models and methods for linking models across human and natural systems. Illustrative examples from published literature and an ongoing research project focused on timber harvest and carbon sequestration are used throughout the paper. The paper concludes with a brief discussion of remaining challenges to modeling indirect and cross-scale linkages and of the potential utility of complex models of land systems.

Keywords: Land-use change; Land-use modeling; Complexity; Decision making; West Virginia; Carbon sequestration

1. Introduction

The sustainability of the global land system is increasingly threatened by anthropogenic pressures (GLP, 2005). This threat finds its foundation in the decisions of human actors and in the aggregation, interactions and indirect effects of these decisions with the biophysical environment. The complexity of human decision-making, coupled with mismatches between socioeconomic and biophysical systems, creates challenges to the sustainability of these systems. Mismatches span space, time, and scale, creating substantial challenges for mapping unknown outcome spaces. Multiple interdisciplinary modeling efforts are incrementally contributing to the challenge of mapping these outcomes spaces, potentially providing information on the functioning of such systems that may contribute to efforts towards sustainable management.

The goal of this paper is to offer a conceptual overview designed to provide context for models that strive to map the not-yet-understood outcome spaces created through humans' interactions with the land use system. Such mapping can serve multiple goals: linkage of socioeconomic drivers of resource use to their biophysical impacts, exploration of feedbacks between human and ecological systems, examination of the sustainability of the current land-use system, and design of policies to encourage more sustainable resource use. (More detailed discussions of potential roles for land-use models are provided by Briassoulis, 1999 and Verburg et al. 2006.) To meet this goal, we characterize sources of complexity for land-use systems

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and discuss the implications of mismatches and indirect linkages between individual land use decisions and their impacts.

The article is organized as follows. In the first half, we review key sources of human-induced complexity in land-use systems, moving from the human decision level to human interactions to effects over space, time and scale. For each increment, we address the following questions: (1) What sources of complexity are implied, and how do they create a potential challenge to the sustainability of the coupled human–environment system? (2) What questions can potentially be answered by building a model at that increment?

The second half of this paper discusses challenges commonly faced by research groups modeling coupled human-natural land use systems, and reviews some common means of addressing these challenges. First, practical challenges and resolutions related to building empirical models of complex land-use systems are discussed. This discussion focuses both on the need to gather data sufficient to measure all processes embedded in the model, and on strategies for simplifying the structure of the model where possible. Second, strategies for coupling natural and social system components of models are reviewed, illustrating each with examples of ongoing and completed research. Drawing on the conceptual framework developed in the first part of the article, a challenge is set forth to expand coupled models to acknowledge and include indirect cross-scale relationships.

Throughout the article, we also draw on research examples from the other projects represented in this special issue (Acevedo et al., 2007; Brown et al., 2007; Entwisle et al., 2007; Evans and Kelley, 2007; Olson et al., 2007; Walsh et al., 2007), on other recent applications of land-use change modeling, and specifically on the illustrative example of timber production, carbon sequestration, and global climate change, drawing in part on ongoing research by the authors (Parker et al., 2005). In the course of our current interdisciplinary project, we have encountered many of the complex relationships described in the first half of the paper, and we have been challenged to come up with concrete resolutions to the practical challenges we describe in the second half.

Human decisions are influenced by individual preferences, group dynamics, and top-down social, political, and economic forces. Decision-making therefore occurs simultaneously at multiple scales. Decisions at each scale interact with biophysical processes, often via imperfect linkages and feedbacks. In the next two sections, we outline sources of complexity related to human land-use decisions, scaling from the level of the human decision maker, to local landscape interactions, to potential global-scale influences. We also discuss key sources of complexity in the environmental systems with which the human systems interact. We discuss how features of humans' motivations, information, and incentives, combined with imperfect matching with natural systems, suggest that the dynamics of human decision-making may not be consistent with ecological sustainability, laying out the rationale for the study and management of coupled human–environment land-use systems (Acevedo et al., 2007).

2. Human information, perceptions, incentives and motivation

Individual land managers employ a diverse set of strategies and experience a range of influences in making land-use decisions. It is clear that no one simple model adequately captures the complexity of human decisions, and that decisions depend heavily on drivers and context that vary over space and time. In this section, we review key conceptual paradigms that can be used to describe sources of complexity relevant for land-use decisions.

We begin our discussion with the most optimistic of theoretical models, which predict that human decisions can lead to ecological sustainability, beginning with predictions of the standard economic model of an individual land manager who holds full property rights over his or her land. This model's relatively strong assumptions regarding land manager information and the properties of the natural system imply that individual choices will lead to economically and ecologically sustainable behavior. We then move to more complex and detailed decision models, which call sustainable behavior into question, moving from the level of the individual to interactions between that individual and other agents. Finally, we discuss ways in which the human and environmental drivers may operate independently, and how this independence introduces additional complexity into the coupled human-natural system.

2.1. "Homo-economicus", sustainability, substitutability, and discounting

While models of a rational economic decision maker (often titled "Homo-Economicus") are often expressed through relatively mathematically sophisticated optimization algorithms, the decision situation that they describe could be viewed as one of relatively low complexity. These models describe the objectives, resources, and constraints of a single agent with well-defined goals and complete information about the future. Interactions between the human and natural system are generally summarized in terms of a single, well-behaved dynamic processes that represents a stream of goods and services produced by the natural system. The decision maker is also assumed to have complete knowledge of this system (Clark, 1990). Thus, these models offer a potentially optimistic starting point from which to examine the feedbacks between human and natural systems, and whether these feedbacks lead to both ecological and economic sustainability.

Even viewed through the lens of this traditional economic framework, theory and evidence offer mixed perspectives regarding the ecological sustainability of human choices regarding resource use. Not surprisingly, models that introduce more complexity are less likely to predict sustainable patterns of resource use. The most optimistic of economic models predict that human decisions will result in sustainable levels of natural resource consumption (Pezzey and Toman, 2002; Perman et al., 2003). However, these models rely on fairly strong assumptions: that resource users have perfect information regarding natural and social processes and act to maximize profits, that all resources are privately owned and can be traded within existing markets, that all costs and benefits of resource use accrue to the resource users, that man-made and natural resources that generate productivity (*capital*) can be substituted for each other, and that the productivity of man-made capital grows over time. Such models also predict that the natural resource stock will be gradually depleted over time, but will asymptotically approach a *steady state* in the very long run, at which point rates of natural resource consumption will equal rates of renewal. Translated into a land-use context, this model would correspond to a story of an individual land owner who acted to maximize profits from sale of products of the land (for example, some agricultural product) that was produced using a combination of the natural capacity of the landscape (soil productivity) and technological inputs (machinery, fertilizer, pesticides) whose capacity to increase production grew over time to compensate for decreasing soil fertility. This model would predict that the natural system would approach a steady state where the level of soil nutrient depletion reached its rate of regeneration. At this point, man-made inputs would substitute for much of the depleted productive capacity of the soil. Such conceptual models have been used as the basis to model decisions of individual farmers (Goetz, 1997; Carpentier et al., 2000).

The gradual depletion of resources predicted by this model results from the propensity of humans (and other animals) to discount consumption in the future in favor of consumption in the present (Sumaila and Walters, 2005). This tendency is influenced by financial incentives (most significantly prevailing or expected interest rates), but also operates independently of them, and can be a response to perceived risk—the risk that access to the resource will not be available in the future. Rather than being an artificial and restrictive assumption of economic models, the propensity to discount is a fact of human nature, one that implies fundamental challenges to sustainable behavior for humans. For example, Entwisle et al. (2007) discuss how high international cassava prices led Thai farmers to trade long-run decreases in soil productivity for short-run increases in profits. In recognition that this propensity for resource depletion may not be desirable from a long-run societal perspective, low or zero discount rates are often advocated for evaluating public projects, and alternatives to standard financial discounting have been proposed for such projects (Sumaila and Walters, 2007).

The realism of key assumptions included in such economic models has also been questioned. The most important assumption from the perspective of biocomplexity research is that natural capital has man-made substitutes. Many authors argue that natural capital that generates critical *ecosystem services*—flows of goods and services from ecosystems that provide life-sustaining benefits to humans—cannot be replicated by humans, and assumptions of optimistic models regarding substitutability of human and natural capital are incorrect (Pezzey and Toman, 2002; Ekins et al., 2003). Ecosystem services specifically provided by the land-use systems include food and fiber production, hydrological cycling and water purification, soil nutrient cycling, climate regulation, and biodiversity provision (Daily et al., 2000; GLP, 2005). As well, the functioning of critical natural capital may be compromised only after being depleted beyond some threshold, violating the assumption of continuous depletion and substitutability inherent in the most optimistic economic models (de Groot et al., 2002). Ecological evidence supports the hypotheses that the functioning of critical natural capital has been compromised. For example, evidence suggests that substitution of man-made inputs for soil fertility may not be viable in the long term, as artificial fertilizers eventually lead to acidification of soils, which can hinder plant growth and release aluminum into the soil (Matzner et al., 1983; Aber et al., 1989). Wackernagel et al. (2002) suggest that humans have already exceeded the regenerative capacity of land-use systems, based on examples including agricultural production, livestock grazing, timber harvest, and infrastructure related to human development.

Thus, even starting from the most optimistic framework, a story of human and natural processes operating as distinct, and potentially conflicting, systems emerges. First, the level of resource exploitation by humans is influenced by prevailing interest rates, whose levels may be independent of natural resource scarcity, and potentially by perceptions of risk. Next, this level of resource exploitation may lead to depletion of critical natural capital beyond a threshold at which life-sustaining ecosystem services are adequately provided.

In spite of their relative simplicity, abstraction, and high level of aggregation, many questions can be investigated by the models described in this section, including:

- How might rates of natural capital depletion be qualitatively affected by factors such as the rate at which decision makers weigh present vs. future payoffs and the development of substitutes for natural capital?
- What policies might be needed to protect ecosystem services generated by the land-use system?

2.2. Complexity in decision-making: motives, methods, and context

In reality, humans employ a variety of strategies when making decisions regarding land use beyond maximization of profits or satisfaction or minimization of risk, implying that economic decision models alone are insufficient to describe human behavior. For agricultural land managers, the need to meet subsistence requirements may drive decision-making, superceding financial incentives (Walker et al., 2002; Deadman et al., 2004). Cultural norms may also motivate cropping choices that are not economically optimal (Becu et al., 2003) as may imitative behavior (Berger, 2001; Polhill et al., 2001) and heuristic decision strategies (Deadman et al., 2004; Acevedo et al., 2007). Heuristic strategies may include *satisficing*: attempting to achieve a minimum threshold of returns or satisfaction, and trying new strategies only if this minimum threshold is not met. This diversity of strategies implies that various land managers may respond differently to the same set of opportunities. It also implies that decision strategies of groups of agents may be interdependent.

Even with fixed potential choices and activities, human decisions may change over time and context. There is a large amount of evidence that while humans do discount future payoffs, as discussed above, discounting does not occur according to an *exponential/logarithmic* model, as has been commonly assumed (Rabin, 2002). Such a mathematical model implies that having made a decision regarding a path of resource allocation today, a land manager would choose to continue that path in the future. In contrast, evidence indicates that human decision-making may be *time-inconsistent*. In other words, a land manager may initially follow a conservative resource allocation plan, but may later choose to exploit land resources at a higher level. These findings may explain why tasks with high costs are postponed, even when the postponement increases the cost (*procrastination*), and may have the potential to explain how unsustainable land-use decisions are made even in information and resource rich environments. As well, perceived satisfaction and rewards may also be highly context dependent. Kahneman and Tversky (2000) demonstrate that *framing effects* may influence perceived satisfaction. For example, satisfaction may be dependent not only on a household's absolute level of consumption, but on that household's level of consumption relative to other households (Brekke et al., 2003).

Perceived values of natural resources may also depend on individual experiences, knowledge, and goals. Greater experience with flood events may cause agents to shift concerns from property values to environmental impacts of development (Acevedo et al., 2007). A hiker may value a forest ecosystem for its aesthetic and spiritual value, while a hunter may gain utilitarian value from game animals housed in the forest, and a parent may view the ecosystem as something to be preserved for future generations. Alternatively, an individual with a severe poison ivy allergy may perceive the forest as a threat to human well-being. Perceived value also varies among humans who are spatially removed from the natural system according to information, values, and education (Stern et al., 1995; Dunlap et al., 2000). As a result, few individuals are likely to have knowledge of the full range of ecosystem values generated by the land-use system, and values are likely to be highly specific for a given individual.

The environmental social science decision models described in this section can be used to examine a broader range of questions than those addressed by traditional economic models, including:

- How do subsistence constraints determined by household composition influence decisions to grow subsistence vs. market crop varieties?
- To what degree does imitation behavior between managers influence adoption of new land-management technologies?
- How might relative inequality between households influence perceptions of household well being, and subsequently influence land management decisions?
- How might knowledge of their potential ecological effects influence a suburban resident's perceived values of native vs. non-native invasive ornamentals?

2.3. Collective behavior and the commons

Many land resources are commonly held and/or managed. In these cases, coordination between land managers may be required to ensure sustainable resource use. For example, an aquifer from which multiple landowners draw irrigation water is a classic example of a *common pool resource* (Feuillette et al., 2003). In general, such resources are described as *rivalrous* (one individual's use of the resources diminishes the benefits available to another user) but *non-excludable* (access to the resource by specific individuals cannot be limited). If land managers account for only private benefits and costs when making groundwater extraction decisions, from a social perspective, the groundwater will be overexploited, as in Hardin's tragedy of the commons (Hardin 1968). However, landowners may successfully coordinate to establish rules for management of the aquifer. Ostrom (1990) describes circumstances under which such coordination is more likely to be successful, and Acevedo et al. (2007) review the extent of success in establishment of local rules and institutions in four regions experiencing significant land-use change. Again, these circumstances imply that landowner decision making—in this case, collective rather than individual decision making—may be highly context dependent. Models of common pool resource use and management can be used to address questions such as:

- How do human incentives lead to degradation of common pool resources such as coastal wetlands that provide storm protection?
- How do factors such as trust and reciprocity affect the success of institutions that communally manage forests?

2.4. Dynamic drivers of change

The most influential drivers of change in a given natural resource may differ between human and natural systems.

As well, changes that occur in human drivers over space and time are often independent from the state of, or changes in, the natural resource itself. There are numerous examples of such independent drivers in land-use systems. Bennett and Tang (2006) discuss how changing perceptions of elk value over time and space have influenced management strategies, and how changes in ownership patterns have influenced elk habitat and migration. Lynch and Lovell (2003) demonstrate that factors such as agricultural profitability, off-farm employment and the extent of family involvement in farming may influence a landowner's decision to enter into a conservation easement. Household life stage, labor, and capital availability have been found to have a substantial influence on land clearing and cropping decisions (Mertens et al., 2000; Vance and Geoghegan, 2002; Walker et al., 2002; Deadman et al., 2004). Changes in international policy and international markets may have profound effects on the profitability of land-use strategies at a local level, inducing abrupt shifts in land-use strategies. Berger (2001) for example demonstrates the effects of new national trade policies on local adoption of irrigation technologies. Olson et al. (2007) discuss how new migration patterns, growing external markets for produce, and a growing tourism industry have led to changes in cropping and livestock decisions at a local level in East Africa. Entwisle et al. (2007) discuss how the rise and fall of European cassava demand has influenced agricultural development in Thailand. Walsh et al. (2007) discuss how increased accessibility following road building for oil exploration has lead to production of commercial agricultural crops in Ecuador. All of these examples demonstrate drivers of land use change that are partially or completely independent of the state of the natural system.

Social policies at a regional level may also bring about changes in biophysical processes at landscape scales. Suppression of wildfires in many locations in the Western US led to increased fuel buildup and explosive wildfires in the 1990's (Swetnam and Betancourt, 1998; Hessl et al., 2004). In this case, human institutions (suppression policies) and biophysical processes (fuel accumulation and climate variability) may have simultaneously driven the occurrence and extent of wildland fire, leading to ecological surprises—fires that burned outside the range of historical variability. Given recent increase in residential development at the wildland–urban interface, the risk to human lives and damage costs from wildfires has increased, and with that increase, pressure for suppression has also increased (Theobold, 2004; Radeloff et al., 2005).

A variety of questions related to varied natural and human drivers of land-use change can be formulated, including:

- How might increased popularity of new forms of outdoor recreation, such as mountain biking and ecotourism, affect the natural areas in which these activities take place?
- How might increased interactions between human and non-human species resulting from amenity-driven residential development affect the long-run viability of the non-human species?

2.5. Forests, carbon, and decision making in West Virginia

Our project, a collaboration between natural and social scientists, calibrates models of timber harvest and carbon dynamics in West Virginia, USA, and links these models to conduct sensitivity and scenario analysis of timber harvest and forest productivity. The study location is a temperate forest system where previous changes in land use may have had a measurable impact on global carbon (C) budgets (Vitousek and Mooney, 1997; Casperson et al., 2000) and where we expect future land use changes may also affect the global C budget. Our coupled timber/carbon model will be used to explore questions such as:

- How might changes in timber prices and ownership regimes lead to changes in carbon uptake?
- How might changes in climate affect carbon uptake?
- How might a "US Carbon Market" scenario (potentially including changes in climate, increased economic incentives for timber production, and resulting changes in land ownership patterns) affect both timber harvest and carbon sequestration?

Complexity in decision making: Complexity in decision-making strategies and drivers are seen in our timber harvest and carbon sequestration research application. West Virginia is the most heavily forested state in the nation, with over 76% of the state classified as timberland in 2000 (Griffith and Widmann, 2003). As in other forests in the US Appalachian, Northeast, and Ohio River Valley regions, West Virginia's upland hardwood forests are dominated by secondary growth on land that was almost completely deforested around the turn of the 20th century. These forests are now reaching economic maturity (Alig et al., 2002). In the presence of a healthy timber market and profit-maximizing land managers, this would lead to an expectation of increasing timber harvests. Economic theory suggests that the timing of harvest for profit-maximizing land managers would be determined according to *Faustman rotations* (Newman, 2002; Perman et al., 2003). Specific timing and targets for these harvests are likely to be dependent on drivers that operate independently from the ecological system, such as interest rates, trade policies related to timber, and the tastes and preferences for end users of hardwoods.

However, the majority of hardwood forests in West Virginia and the region as a whole are privately owned, and the majority of privately owned forests are owned by non-industrial private forest landowners (NIPFs) (Alig et al., 2002). Timber harvest by NIPFs owners accounts for a large and growing percentage of total harvest (Rosen and Kaiser, 2003; USFS, 2003). There is substantial evidence that NIPFs do not respond to economic incentives as industrial forest managers would. For most NIPFs, motivations for forest land ownership are non-pecuniary, and include such factors as recreational use, hunting, aesthetic enjoyment, ecological preservation, and preservation for future generations. Pecuniary motivations for timber harvest are often driven by life events (such as need for funding for a college education). There is also evidence that even when harvest occurs for pecuniary reasons, NIPFs may be less aware of the market value of their timber and may engage in less sustainable harvest practices than commercial foresters (Fajvan et al., 1998). In short, NIPFs are less likely to harvest trees than profit-maximizing commercial foresters, but more likely to harvest in ways that decrease the future productivity of the forests (Birch and Pywell, 1986; Jones et al., 1995; Koontz, 2001; Alig et al., 2002; Keefer et al., 2002; Rosen and Kaiser, 2003). Therefore, understanding differences in how NIPF owners respond to drivers of harvest and how these differences translate into carbon dynamics through management strategies is critical to understand the overall links between drivers of harvest and carbon dynamics in West Virginia, and more generally throughout Central US hardwood forests.

Dynamic drivers: The case study illustrates operation of potentially disconnected and dynamic drivers of change. Demand for particular hardwood species will be influenced by shifting tastes and preferences for furniture, the availability of substitute sources of supply, including both timber harvest from other regions and of engineered substitutes for hardwood, and even by consumers' perceptions of the ecological conditions under which the species are grown. If, as our preliminary findings suggest (Hessl et al., 2006), different species show different forest productivity and carbon sequestration patterns, then changes in land-use and harvest strategy driven by changes in economic opportunities and preferences will lead to significant changes in carbon sequestration. For instance, in one scenario, a landowner may be motivated to cut mature cherry due to demand for furniture and cabinet production, but may keep sugar maple for maple syrup production or for its colorful fall foliage. A shift to a regime in which light fine-grained hardwoods are in demand for furniture production could induce the opposite harvest and carbon uptake pattern.

3. Complexity and dynamic interactions: spatial and scale mismatch between actions and their impacts

As seen above, human decisions regarding land use are complex, and decisions may be influenced by drivers and context that do not necessarily embed feedbacks regarding the environmental impacts of land use. The land use system is also characterized by dynamic relationships between humans and between humans and their environment. While many of these relationships operate at similar scales or as direct cross-scale feedbacks, many feedbacks are indirect and may operate at different spatial and temporal scales, creating threats to the sustainability of the land-use system. Critical mass and threshold phenomena, which cause rapid, often unpredictable change of state in systems, provide additional levels of complexity in both social and biological systems.

3.1. Local spatial structure and dynamics

Spatial relationships, both human and biophysical, are an important source of potential disconnection between land use decisions and their impacts. Boundaries defining spheres of human influence and boundaries defining biophysical impacts (spatial extents) are often non-contiguous and non-overlapping. For example, neighboring land use activity threatens rare biotic resources in the US Potomac Gorge National Park (Alan and Flack, 2001). The watershed generating these uses is divided into multiple state and county level institutional jurisdictions, each with its own set of land-use regulations and policies.

Spatial processes, human and natural, may also transfer natural resource impacts across space and time. For example, road building for one purpose, such as logging, may facilitate movement of populations of new settlers into previously uncolonized areas (Nelson and Hellerstein, 1997; Deadman et al., 2004; Entwisle et al., 2007; Walsh et al., 2007). Removals at one location in a surface waterway influence water availability and ecological conditions for downstream users (Lansing and Kremer, 1993; Ray and Williams, 1999; Becu et al., 2003).

Spatial pattern and connectivity may affect landscape function. The pattern of surrounding land uses may affect the value of residential land (Geoghegan et al., 1997; Irwin and Bockstael, 2002). Brown et al. (2007) find that the appearance of neighbors' yards has a strong effect on residents' preferences among alternative new homes. The shape and connectivity of an organic agricultural parcel in relation to its neighbors may also affect the viability of production (Parker and Munroe, 2007). Human-induced changes in pattern and connectivity may also affect ecological function. Changes in ownership and management patterns have been shown to affect bird (Lewis and Plantinga, 2007) and elk (Bennett and Tang, 2006) habitat. Forest clearing and cultivation practices may also contribute to spread of invasive species (Silveri et al., 2001; Manson, 2004), as opportunities for invasion increase with disturbance and forest fragmentation.

Models of spatial structure and dynamics can be used to address questions such as:

- What patterns of cross-pollination of genetically modified (GM) and non-GM crops occur, and how do these patterns depend on the spatial relationships between growers of both crops?
- What spatial patterns of competition and survival between native and non-native species result when residential land managers adopt non-native invasive ornamental plants, and how are these patterns affected by the degree of landscape fragmentation?

- How might distinct land-use policies be developed and implemented in diverse political jurisdictions within a single ecoregion?

3.2. Critical mass and thresholds

Critical mass or *threshold* phenomena are observed in both social and biophysical systems, when a key variable exceeds a certain value causing a system to switch from one regime to another. Schelling (1978) describes a wide variety of critical mass phenomena in social systems, which generally depend on interdependent behaviors where one individual adjusts his or her decisions based on aggregate metrics of others' behavior. Using a simple dynamic spatial model, he demonstrates how both socially and individually undesirable levels of segregation could occur in residential neighborhoods, even though individual homeowners were content in diverse neighborhoods (Schelling, 1971). Entwisle et al. (2007) discuss how villages subdivide administratively once they reach critical sizes. Critical mass phenomena are also often present in agricultural systems, as adoption of a new agricultural technology, or creation of a new market, may require a minimum number of participants to be viable.

Thresholds are common phenomena in biological systems as well. Populations may face extinction thresholds at positive population numbers (*critical depensation*), due to reduction in gene pool sizes or mating opportunities (May, 1981; Clark, 1990; Limburg et al., 2002; Morris and Doak, 2002). As well, thresholds often occur in ecosystems when a single keystone species is removed or introduced. For example, wolf extermination from the greater Yellowstone Area during the early 1900's may have lead to unexpected changes in vegetation via a *trophic cascades effect*, where changes at one trophic level induce changes at lower trophic levels. In the absence of wolves, elk flourished and browsed extensively on aspen and willow populations, resulting in reduced aspen and willow regeneration (Ripple and Larsen, 2000; Hessl, 2002; Hessl and Graumlich, 2002). Similarly, the introduction of wolves to Isle Royale, Michigan lead to declines in moose, but increased growth of balsam fir trees (McLaren and Peterson, 1994).

Many of the most interesting questions related to critical mass in coupled human-natural land-use systems focus on how gradual changes on one side of the system can lead to abrupt changes on the other side. For example, Olson et al. (2007) explore when drought conditions would lead to a critical threshold at which farming-based livelihoods are no longer viable and land abandonment occurs. Additional representative questions include:

- When might adoption of genetically modified crop varieties in a region render production of certified non-GM crops infeasible, based on the spatial extent of pollen diffusion of the crop?
- What level of adoption of non-native invasive ornamental plants by residential land managers might lead to local extinction of native competitors?
- When might local changes in climate trigger political action by stakeholders resulting in carbon trading programs?

3.3. Indirect feedbacks between actions and their impacts

Much has been written about the importance of cross-scale feedbacks in land-use systems and the need to include these feedbacks in models (Wear et al., 1998; Clark et al., 2001; Verburg, 2006). Somewhat less attention has been paid to processes that cross scales but do not result in a direct feedback to the original temporal or spatial scale. Very often, individual land use decisions and their impacts are disconnected over space and time, and impacts may not feed back to the level of the individual making the land-use decision (Olson et al., 2007). In these cases, the mismatch across scale between actions and their impacts may compromise the sustainability of those systems, and the challenge faced by both modelers and policy makers may be to identify those indirect linkages and design mechanisms to create critical missing feedbacks.

For example, biodiversity may have prospective values, in terms of its potential to contribute to food crops and human medicines (Nunes and van den Bergh, 2001). However, an individual farmer in a tropical forest will not perceive this loss in value when clearing a forest plot, thereby reducing local biodiversity. Nutrient runoff from large-scale commercial farming in the US Midwest has been shown to lead to eutrophication in the Gulf of Mexico, impairing local fisheries (CENR, 2000). However, these effects do not perceptibly impact the well being or economic returns of the farmers generating the threats. While these sorts of problems have been recognized by economists as externalities that lead to a justification for intervention in markets (Baumol and Oates, 1988), the importance of scale mismatch has not been as clearly articulated. From a modeling perspective, this problem needs to be viewed not only in terms of a disconnection between actors and the impacts of actions, but also in terms of disconnection between the scale of actions and the scale of impacts. Daly and Farley (2004) recognize this view, and illustrate how decisions that are optimal at a fine scale may not be at higher scales.

Questions related to indirect and incomplete feedbacks may focus on either local-to-global or global-to-local relationships, such as:

- How do moves from traditional subsistence crop varieties to market varieties in developing countries affect global gene pools for food crops?
- How could spatially targeted fertilizer reduction incentives be designed to minimize the cost of alleviating hypoxia zones?

- How might carbon reduction targets set at a state or country level indirectly affect agricultural productivity in other states through changes in climate variability?

3.4. Complex dynamics and forests

The complexities of spatial and scale mismatches in dynamic human–environmental models are seen in forest systems through local spatial dynamics, critical mass phenomena, and indirect feedbacks.

Local spatial dynamics: Many complexities related to drivers and ecological impacts of timber harvest exist. Spatial dynamics on the human and biophysical side influence harvest rates and outcomes. For example, information about timber values and harvest opportunities may be passed from one neighbor to another. Timber harvest may generate ecologically negative off-site impacts through, for example, compaction and loosening of the soil, increased soil erosion, increased nutrient export, and loss of forest canopy, leading to increased turbidity, water temperatures, and eutrophication (US EPA, 2005).

Critical mass: Critical mass phenomena are also present in forest ecosystems. Even the removal of a single tree species in a species rich community may lead to long term consequences for the entire ecosystem. In eastern deciduous forests, chestnut blight and the subsequent loss of American chestnut in the 1930s may have resulted in an increase in oak dominance that is still obvious today (Abrams, 1992). The American chestnut blight is also an example of the complex feedbacks associated with human perceived thresholds. Before the actual impact of the blight was determined, humans perceived a pending loss of valued wood and conducted widespread harvests of the chestnut trees. One potential impact on the natural system was to eliminate genes that might have withstood the blight. Elimination of the chestnut also had a profound and unexpected effect on the human system, potentially leading to the decline of the subsistence lifestyle in Appalachia (Davis, 2006).

Changes in climate can also lead to rapid, threshold-type responses in forest ecosystems (reviewed in Burkett et al., 2005). In the southwestern US, threshold changes in drought conditions in the 1950's resulted in persistent shifts in the *ecotonal boundaries* (transition zones between distinct ecosystems) between ponderosa pine and piñon-juniper woodland via massive die back events (Swetnam and Betancourt, 1998). Alternatively, in the taiga-tundra transition zone, tree growth has shown a linear increase, coincident with increasing temperatures, while the rate of tree recruitment has been non-linear (Suarez et al., 1999), suggesting an advance of treeline that could persist. Such changes in timber cover can have subsequent effects on recreational and harvest value for humans, inducing more land-use change events. For example pine encroachment has altered land-use values for timber harvesters, grazers, and conservationists within a unique habitat area in the Causse Méjan region of France (Etienne et al., 2003).

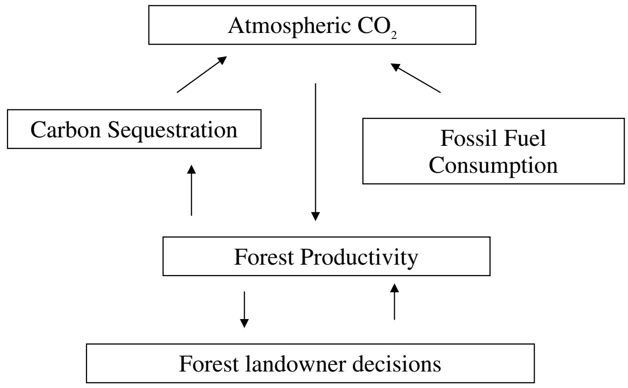

Fig. 1. Indirect linkages between landowner decisions, forest productivity, C sequestration, and atmospheric C.

Indirect feedbacks: The relationship between local landowner actions, carbon sequestration, global carbon pools and fluxes, and forest productivity provides a clear example of complex indirect, cross-scale linkages (see Fig. 1). Changes in land management at the local scale undoubtedly affect regional carbon budgets, especially if many landowners act in concert. However, landowners are currently largely unaware of how their forest management decisions affect the carbon cycle. Even if they were aware, current incentives may not encourage them to change their behavior, and landowners may (reasonably) perceive that their individual actions have relatively little influence on atmospheric carbon. For example, House et al. (2002) report that complete global reforestation would only reduce atmospheric carbon by approximately 40–70 ppm, an amount barely sufficient to offset past emissions (current levels are approximately 60 ppm above 1960's levels) and certainly insufficient to offset future emissions. Thus, though forests and forest soils serve as carbon reservoirs, individual contributions may seem insignificant when compared to the effects of fossil fuel emissions (Prentice et al., 2000).

In fact, emissions produced from worldwide fossil fuel consumption are major contributions to atmospheric CO_2, contributing 6.3×10^{-15} g C per year (in the 1990's) in the form of carbon dioxide, a greenhouse gas, to the atmosphere (Schlesinger, 1997). Human decisions resulting in the burning of fossil fuels are generally distinct from those related to forest management. Greenhouse gases from fossil fuel consumption are well mixed in the atmosphere, and subsequently may have important implications for ecosystem productivity. For example, parts of Appalachia have been disproportionally affected by high levels of nitrogen deposition from human fossil fuel burning ("acid rain" from Midwestern coal-fired power plants and industry). This indirect anthropogenic influence may be altering baseline forest productivity (Peterjohn et al., 1996; Gilliam et al., 2001), which may subsequently have affects on timber harvest incentives and land values. Although some models that estimate forest productivity incorporate fertilization effects through generalized inputs of atmospheric

CO_2 and/or nitrogen deposition (Wofsy et al., 1993; Lovett 1994; Aber et al., 1997; Luo et al., 2004), these models do not connect fuel burning and forest management decisions with their impacts. To our knowledge, few models of these indirect feedbacks exist.

4. Challenges in building empirical models of complex land-use systems

As is evident from the above discussion, many potential sources of complexity influence human decisions and feedbacks between human decisions and the natural environment in land-use systems. These complexities are complicated by the differences between biophysical and socio-economic models. Biophysical and socio-economic models often work at different scales and may require input or produce output with vastly different resolutions, creating challenges for linking these models (Antle et al., 2001; Evans et al., 2005; Melton et al., 2005). Models of human decision-making need to be linked to landscape, community, or ecosystem models, yet field-based ecological datasets may contain more information about fewer points while socio-economic data may contain large numbers of observations about relatively few variables. Climate data may be particular sparse and/or unevenly spatially distributed (Olson et al., 2007).

In spite of the complexity of the systems under study, empirical models must be constructed to be as parsimonious as possible, so that relationships between model inputs and outputs can be traced and understood (Parker et al., 2003). Thus, any strategies for simplifying empirical models, that do not at the same time move the model away from a clear representation of the research question it investigates, are of practical use. Even a simplified model, however, may contain many free parameters, meaning that complex models can be very data hungry. Below, we briefly discuss three current strategies that may help address these challenges: modeling at the highest possible level of aggregation, using case-specific information to reduce model complexity by distilling the number of state variables in the model, and combining multiple data sources.

4.1. Modeling aggregates

One strategy to assist in linking models across scales, and to maintain model tractability, is to model populations of agents at a more aggregate level when such aggregation does not compromise the model's ability to address key research questions, or bias the results. In this case, the aggregate entity may have properties that reflect the potential heterogeneity of the agents who compose it. For example, in a model studying linkages between agricultural land use, leasing of hunting rights on agricultural lands, and duck populations, Mathevet et al. (2003) model hunting groups as single agents that are characterized by their group size and preferences for hunting experience quality, and they model duck populations in terms of spatial density. Land owner decisions are often modeled at a household level, representing household size and composition through attribute variables (Mertens et al., 2000; Staal et al., 2002; Vance and Geoghegan 2002; Walker et al., 2002; Deadman et al., 2004). Ideally such aggregation should be a modeling decision. However, often key data are only available at an aggregate level, and must be entered as regional or spatial averages, rather than the less attractive option of omitting the influence altogether (Deininger and Minten, 2002).

4.2. Identifying a minimum set of state variables

Model complexity can also potentially be reduced by restricting possible outcomes (such as agent decisions and/or land-use trajectories) to those appropriate for a particular research case, rather than allowing all logically feasible outcomes. Such paring down ideally occurs by bringing additional real-world information into the model. For example, the CLUE-S model (Verburg et al., 2002) uses suitability and fixed cost information to dampen statistically estimated transition probabilities. Land-use models developed for the Brazilian Amazon (Carpentier et al., 2000; Walker et al., 2002; Deadman et al., 2004) focus on investment trajectories related to available labor and household life cycles. Mertens and Lambin (2000) and Batty and Xie (2005) identify feasible land use trajectories to constrain model transitions. By accounting for co-occurrence of land uses, or acknowledging real-world limits to land-use transitions, these strategies reduce model complexity by reducing the number of outcomes variables in the model.

4.3. Combining multiple data sources

Because they strive to represent both human and natural influences, and often focus on drivers of individual decisions, coupled models of land-use systems can be very data hungry. Many projects combine both spatial data—to represent key biophysical inputs and spatial relationships and survey data—to represent decision-making influences at an individual or household level. Initiatives such as the NSF biocomplexity program facilitate such large-scale projects, as seen by the large number of papers in this issue that combine spatial and survey data (Acevedo et al., 2007; Brown et al., 2007; Evans and Kelley, 2007; Walsh et al., 2007). This approach is seen in a wide number of other land-use modeling initiatives as well (Mertens et al., 2000; Müller and Zeller, 2002; Staal et al., 2002; Vance and Geoghegan, 2002; Verburg et al., 2002; Walker et al., 2002; Lynch and Lovell 2003; Deadman et al., 2004). Data from laboratory decision-making experiments, participant observation, and companion modeling can provide additional inputs to land-use models (Olson et al., 2007; Robinson et al., 2007). While significant challenges may still exist to link multiple data sources, their availability may help modelers to build better structural representations of the system that they are studying, and to subsequently choose between competing process models.

4.4. Empirical challenges for modeling timber harvest and carbon sequestration and land use

Modeling aggregates: Ideally a timber harvest model would be estimated at the spatial scale corresponding to the manager unit over which a harvest decisions is made. Due to data limitations, timber harvest models are more often estimated at more aggregated scale, such as a county level. For example, Ahn et al. (2000) link county-level drivers of land-use to carbon sequestration outcomes, but the scale of their model does not allow modeling impacts of local-scale biophysical, accessibility, and ownership drivers. It is an open question whether models that include such fine-scale drivers will provide greater insights than models that operate at a more aggregated level—one which we hope to answer through our own modeling efforts. Evidence exists that, on the ecological side, finer-scale inputs change model predictions. Pan et al. (2004) found that refined scaling of foliar nitrogen parameters significantly altered forest productivity estimates, and other ecological modeling efforts have found similarly important differences in modeling according to input parameters from different scales (Luo et al., 2003; Shipley et al., 2005).

State variables: Opportunities exist to reduce the number of state variables in timber harvest models, since particular species are often harvested together, implying that species-level harvest decisions are not independent. Often, species of lesser value will be harvested at the same time as species of higher value, in order to take advantage of fixed costs of harvest and transport. As well, proximity to certain processing facilities may lead to harvest of species that can be used to create similar products (hardwood veneer, pulp, etc.). Potential information sources to identify species groups that are co-harvested include expert opinion, the use of cluster analysis to statistically identify species groups that are co-harvested, and the use of statistical techniques to account for positive correlation between independent variables in separately estimated equations. However, it should be noted that some of these aggregation strategies may reduce options for coupling socioeconomic and biophysical models. For example, if the ultimate goal of modeling is to compare species-specific carbon uptake, then species-level harvest models must be estimated, and thus the third option above would be followed.

Combining data sources: Many timber harvest models combine multiple data sources in order to more closely link land-manager heterogeneity and land management decisions. Koontz (2001) conducted in-depth land-manager interviews to gather detailed information regarding spatial management strategies and their motivations. Using this information, they characterize the broad range of pecuniary and non-pecuniary motivations for private forest land owners in South-Central Indiana, USA. Stevens et al. (1999) incorporate hypothetical information on parcel contiguity to analyze the willingness of NIPFs to engage in collective ecosystem management programs. Vokoun et al. (2006) use a stated preference approach, which implicitly combines survey and experimental methodology, to examine sensitivity of spatially and demographically diverse land owners to hypothetical timber price offers. Through this approach, they are able to estimate responsiveness of harvest to prices outside the current range of market prices, thus expanding the range of applicability of their empirical model. Multiple data sources (remote sensing and ground measurement) may also provide important validation information for carbon models, allowing researchers to better assess the potential out-of-sample performance of carbon models at various scales that use only remotely-sensed data (Reich et al., 1999).

5. Approaches for linking human–environment models in land-use systems

Models of human–environment systems are linked through a common variable or variables in order to represent hypothesized chains of causality and feedbacks. (We use the term "model linkages" to distinguish this discussion from related discussion of model coupling, both conceptually; Antle et al., 2001 and in computer code; Castle and Crooks, 2006.) Three approaches to linking such systems in models are discussed here: natural science models as inputs to social systems; natural-social-natural linkage in a one-way chain, with natural system input and output models potentially differing; and endogenous determination of common variables through interactions between social and natural systems (see Fig. 2). Focusing on the human component in land-use models, we review applications that progress from single uni-directional linkages to fully coupled systems. This description encompasses the complex processes described through Section 3.2. In Section 5.4, we discuss the implications of the indirect and incomplete linkages described in Section 3.3 for model coupling, and following this, we put forth a challenge to researchers to develop models that cross-scale and incorporate indirect linkages.

5.1. Single linkages

Given the demonstrated influence of biophysical suitability on land-use transitions, the majority of land-use change models developed in recent years input spatial data layers that represent a broad spectrum of biophysical inputs. Deininger and Minten (2002) fit a land-cover change model with and without biophysical influences in order to demonstrate possible bias in identification of key drivers that can result if biophysical influences are excluded. In this issue, Evans and Kelley (2007) initialize their model using topography and forest cover layers. Several applications demonstrate the importance of neighboring land cover for land-use transitions. Mertens and Lambin (2000) show that the extent of local forest frag-

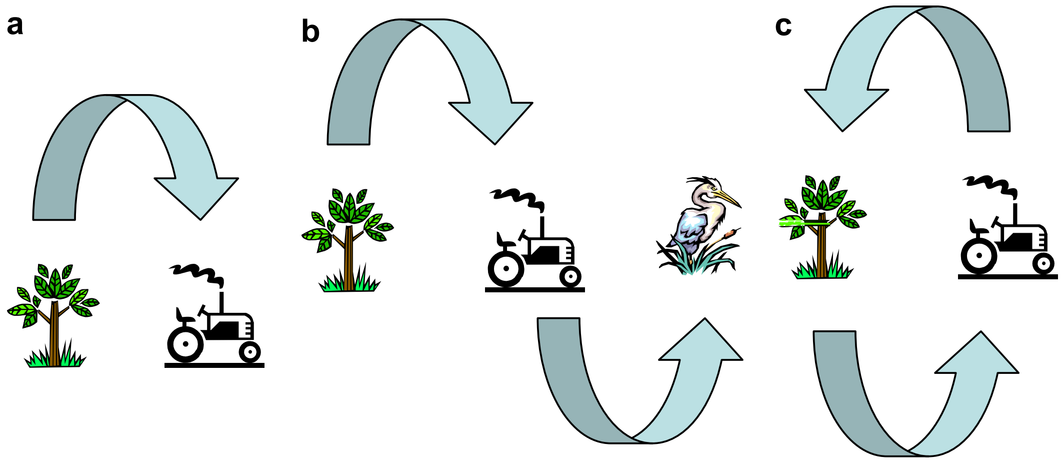

Fig. 2. Three approaches to linked systems: natural to social (a), natural, social, nature, (b), and fully linked.

mentation influences deforestation probabilities. Irwin and Bockstael (2002) show that local open space can generate amenity values that influence land value and land-use transitions. Antle et al. (2003) link regional climate changes to induced changes in cropping patterns in order to understand the extent of potential adaptation by farmers. Lewis and Plantinga (2007) link suitability factors, forest land conversion, and distribution of forest fragmentation outcomes. Our West Virginia research employs single linkages first, developing and validating a harvest model based on biophysical plot characteristics and a carbon model based on ecophysiological characteristics and historical harvest. This allows us to validate the socio-economic model for harvest decisions separately from the ecological process model.

5.2. Multiple uni-directional linkages

Many models illustrate a complete trajectory without fully endogenous feedbacks, linking biophysical inputs to land-use change output (as described above), and then using these outputs to calculate biophysical effects. In many cases, the biophysical input and output targets may differ. Tang et al. (2005), for example, link projected increases in urbanization to concentrations of nutrients, oil and grease, and heavy metals through a watershed model. The Environment Explorer model, developed by Engelen et al. (2003), estimates changes in traffic congestion, air quality, noise pollution, and flooding risk that result from local and regional land-use changes.

Our West Virginia project takes this approach by coupling the two separately-developed models described in Section 5.1. Biophysical inputs, including slope, elevation, and volume of standing timber, are a subset of independent variables representing drivers of timber harvest in an econometric model of timber harvest. Biomass removal estimates from this model will feed into PnET-CN, a model that predicts above and below ground productivity (Aber et al., 1997) in order to estimate changes in carbon uptake induced by estimated harvest.

5.3. Full integration (the biocomplexity approach)

The biocomplexity approach envisions fully coupled systems that integrate two-way feedbacks between human and natural systems, often through closed-loop feedbacks at a particular scale. Such models have been successfully constructed for a variety of land-use applications. Deadman et al. (2004) model feedbacks between smallholder land clearing, cropping, and soil fertility. Soil fertility subsequently affects crop yields, a driver of cropping decisions. Mathevet et al. (2003) model feedbacks between farmer's land use decisions, duck habitat, and duck populations. Agricultural land use affects duck populations through habitat and population dynamics models. Feedbacks to the human system come through economic opportunities to lease land for duck hunting. Silveri et al. (2001) examine the feedbacks that occur between logging decisions and biological structure of a forest in the context of invasive species establishing after logging disturbance and then competing with native timber species for resources. Brown et al. (2005,2007) and Acevedo et al. (2007) model the influence of open space amenities on attractiveness for residential development. Residential development subsequently reduces available open space, increasing relative attractiveness of less developed areas, and pushing the urban-rural fringe outward. Olson et al. (2007) model the effects of land use on climate at a regional level. Induced changes in climate feed back to affect regional agricultural productivity, a driver of cropping decisions. Each of these examples describe fully integrated human–environmental feedbacks. Although these examples focus on single feedback mechanisms, key outcomes of interest may also be determined through multiple direct feedbacks (Olson et al., 2007).

Although our West Virginia project takes a uni-directional approach to model coupling, important feedbacks between timber harvest events, carbon uptake, and subsequent incentives for harvest exist that would justify additional feedbacks. In a fully linked model, timber harvest may alter the age or species composition of the forest, leading to changes in carbon uptake and timber growth, inducing further subsequent changes in harvest incentives. The ability to develop a fully integrated model for the West Virginia application depends on the success of our unidirectional coupled model. A successful model of such feedbacks would likely need to be calibrated over a long time period, in order to capture variation in timber prices and institutional influences. Even with a model calibrated using a longer time series, the human decision process captured in the estimated coefficients is less likely to be stationary over space and time than the calibrated carbon sequestration model, especially as the carbon model is dynamic. This issue could be addressed by building a more detailed structural model of land-manager decision making. With a rich set of data with which to parameterize a model of forest manager harvest decisions (potentially provided through multiple data sources as discussed in Section 4.3), an agent-based model could be developed that contains a structural representation of drivers of harvest, including both economic and non-economic motivations for harvest and preservation. Such a structural model is likely to be more robust outside the range of calibration data (Grimm et al., 2005).

5.4. Indirect linkages: modeling across peoples, places and scales

The majority of research examples discussed in this article model directly linked systems—one in which feedbacks can be traced through a single variable, and in which human-induced changes in the environment produce feedbacks closely linked enough in time and space to influence subsequent human land-use decisions. These models represent important advances beyond models with only one-way linkages. However, as discussed in Section 3.3, linkages between land-use decisions and their impacts are often indirect, crossing spatial and temporal scales. The next challenging modeling frontier is likely to involve tackling these indirect, cross-scale linkages. Such modeling will provide even broader challenges, especially with respect to up-and-down scaling of meso- and micro-level models that may produce global non-point scale inputs.

Linkages between sequestered carbon, global anthropogenic carbon emissions, and atmospheric carbon illustrate important indirect linkages between the human and natural systems. They illustrate that the more important impacts of carbon sequestration are experienced at a global scale, and for the most part, impact human and natural populations that are disconnected in space and time from the particular land parcel (or power-plant) where decision-making occurs. These complexities indicate that a fully coupled model focusing on either forest productivity or global carbon budgets would require careful consideration of global effects on local decision-making, of indirect linkages, and likely modeling of decisions made by multiple populations of human actors.

Yet, the value of information from such coupled models is increasing, especially as participation in carbon limit and trading programs increases. Connections between biological function and global policy are beginning to be made through assessment of the terrestrial carbon budget for defining forest carbon sequestration allowances in the Kyoto protocol (Steffen et al., 1998). These allowances are linked to human decisions regarding locations of emissions through participation in both voluntary and mandatory carbon markets, such as the Chicago Climate Exchange and the EU Emissions Trading Scheme (Hopkin, 2004; Victor et al., 2005). Trades can be directly linked to forest carbon stores, providing new linkages between land-use decisions and global carbon levels. Despite these advances in creating linkages between land manager incentives and forest carbon, estimates of ecosystem carbon are still uncertain, and such markets may be poorly informed, or may not operate effectively if limits are set too high (Hopkin, 2004). Many carbon sequestration models still require input variables from human activities but do not account for the fine-scale dynamic nature of human management decisions in response to socioeconomic factors. For example, carbon flux from land use changes has been calculated (Houghton 1999, 2003), but the drivers of these land use changes are not explicitly modeled, making future scenarios difficult to forecast. Major challenges remain for scaling up local-level models of land-use change and carbon sequestration, especially if those models need to be linked through carbon market trades.

6. Concluding thoughts

This paper has attempted to make concrete the often-nebulous concept of ''complex human–environment interactions,'' using the context of human–environment interactions across the land-use system. We have reviewed specific sources of complexity in human and ecological processes, and have also reviewed complex interactions between humans and their environment that may cross spatial and temporal scales. Much of our discussion has focused on linkages that are indirect and imperfect, implying a potential disconnection between actions and their impacts that may compromise the sustainability of the land-use system. These linkages also make modeling, projection, and scenario analysis challenging. We hope that the perspectives outlined in the article will be useful to multiple groups—certainly to land-use change modelers, but more generally to those non-modelers interested in assessing the potential contributions of modeling to understanding these dynamics.

With human–environment interactions characterized by such high degrees of complexity, it may be natural to wonder if the task of model-building is justified, given that we can be confident that our model will fall short of an exactly accurate representation of the real-world system. It is convenient to fall back on Box's wisdom ''All models are wrong, but some are useful'' (Box, 1979). The relevant question is whether we can learn from our models, and whether that learning may contribute to insights that improve human welfare and the sustainability of our natural resource base. In our minds, both questions have affirmative answers. We also believe that we will not improve the success of our models in meeting these goals if we fail to understand and incorporate the complexities that drive such systems. Human-environment models may highlight scale differences between social and biophysical processes, different driving variables for each side/model, non-linearities at the intersection of human–environment relations, or gaps in our knowledge about how each system relates to the other. Models that reveal such relationships between human and environmental systems, in the biocomplexity tradition, may facilitate more concrete communication between scientific discovery and policy, and may shed light on how human incentives can be modified to move the land-use system towards ecological sustainability. Finally, the process of model building can also be a process of knowledge building, especially as members of interdisciplinary teams come to understand the diverse perspectives, concerns, and knowledge of their colleagues (Olson et al., 2007). As the mandate of the biocomplexity program communicates, ultimate progress in modeling human–environment interactions and their effects on ecological sustainability will only be made with insights from both social and natural scientists.

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