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How Landscape Ecology Informs Global Land-Change Science and Policy

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How Landscape Ecology Informs Global Land-Change Science and Policy

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Landscape ecology is a discipline that explicitly considers the influence of time and space on the environmental patterns we observe and the processes that create them. Although many of the topics studied in landscape ecology have public policy implications, three are of particular concern: climate change; land use–land cover change (LULCC); and a particular type of LULCC, urbanization. These processes are interrelated, because LULCC is driven by both human activities (e.g., agricultural expansion and urban sprawl) and climate change (e.g., desertification). Climate change, in turn, will affect the way humans use landscapes. Interactions among these drivers of ecosystem change can have destabilizing and accelerating feedback, with consequences for human societies from local to global scales. These challenges require landscape ecologists to engage policymakers and practitioners in seeking long-term solutions, informed by an understanding of opportunities to mitigate the impacts of anthropogenic drivers on ecosystems and adapt to new ecological realities.

Keywords: climate change, land use, landscape ecology, policy, urbanization

Landscape ecologists employ an interdisciplinary perspective to understand multiple natural and human-caused drivers of landscape change operating simultaneously and interactively, often focused on coupled human and natural systems with policy-relevant outcomes. In 2013, we surveyed all members of the US regional chapter of the International Association for Landscape Ecology (US-IALE), asking them to identify "the most pressing environmental policy issue to which the science of landscape ecology can contribute." The three most prevalent responses were (1) land use–land cover change (LULCC), (2) urbanization (as a particular case of LULCC), and (3) climate change. These emerging areas highlight global-change policy needs that are crucial to the preservation of the environment and human welfare. In this invited State of the Science report to *BioScience*, we first describe the discipline of landscape ecology and then focus on these three issues to showcase the contributions our discipline has made and can make to policy-relevant science.

What is landscape ecology?

Landscape ecology is a well-established subdiscipline of ecology (Turner 2015) that focuses on multiscale feedback between spatial pattern and ecological process (Urban et al. 1987, Forman 1995, Turner et al. 2001, Turner 2005). Patterns are quantified by a toolkit of metrics—such as patch size, connectivity, and shape (Riitters et al. 1995, McGarigal 2002)—which allow inferences about the dominant processes and scales operating over time. Ecological processes, such as nitrogen fixation, trophic interactions, or carbon sequestration, are driven, influenced, and constrained by their spatial context. Theories and methods of landscape ecology have been applied to many systems—terrestrial, nonterrestrial, and their interface—and along a natural to human-dominated gradient (Wiens 2002, Musacchio et al. 2005). Decades of work have enabled a broad understanding of the patterns that result from both natural disturbance and human use or management of ecosystems, of their interactions, and of the importance of scale-dependent spatial heterogeneity in structuring ecological processes (Turner and Gardner 2015). It is beyond the scope of this article to comprehensively review the field of landscape ecology (for that we refer the reader to the References cited); instead, we focus more narrowly on how the tenets of landscape ecology can help inform the policy decisions for managing landscapes in the face of global environmental challenges, such as the three prioritized by US-IALE members: LULCC, urbanization, and climate change.

Landscape ecology explicitly considers scale (Turner 1989, Turner et al. 2001, Urban 2005). *Scale* is defined by both

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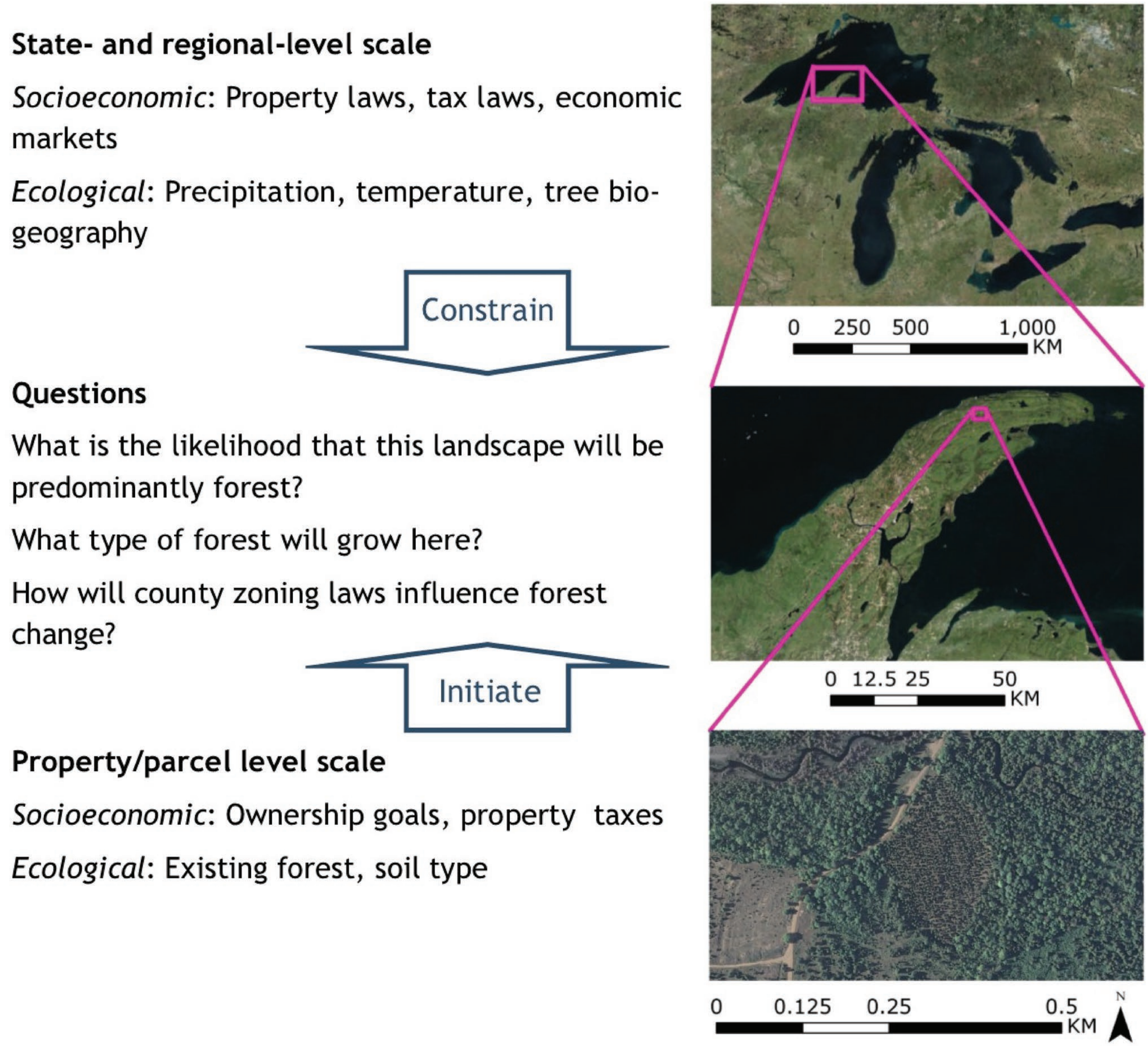

*Figure 1. To determine how forest cover changes over time on the Keweenaw Peninsula in northern Michigan, researchers must understand fine-scale social, economic, and ecological characteristics at the parcel level (e.g., management goals and decisions of the owner, soil types), as well as the broad-scale constraints at the state or ecoregion level (e.g., state property and tax laws, regional climate). Map data: ESRI, Digital globe, and US Geological Survey (*https://lta.cr.usgs.gov/high_res_ortho*). Abbreviation: km, kilometers.*

the grain (or resolution) of the data and the extent of the study region or period (Delcourt and Delcourt 1988). When conducting research to support policy decisions, the scale must be relevant to the focal organism, process, pattern, or policy of interest and to the organizational level at which it can be implemented (i.e., local, state/provincial, or federal jurisdictions). Ensuring a match between the policy scale and the ecological scale is not always straightforward. From an ecological perspective, the grain should provide enough detail for signal detection, whereas the extent should be large enough to encompass enough spatial and/or temporal heterogeneity to reflect the process or pattern of interest. From a policy perspective, the spatial and temporal scale should match those at which decisionmaking and implementation takes place (figure 1). Scale matching will minimize the costs associated with overly intensive, fine-scale research (expense, time, effort) while also minimizing the opportunity costs associated with collecting data either irrelevant or outside the scope of the policy realm. For example, focusing on watershed-scale drivers of water pollution (e.g., housing density) is relevant to policy focused at watershed management but less relevant to policy focused on single-household water-pollution prevention (in which housing density is less relevant). An explicit treatment of scale also helps to identify when more data are unlikely to yield dividends for landscape-related policy decisions—that is, the tradeoffs of costs for increased data acquisition versus the benefits of decreased uncertainty or increased accuracy in forecasts. With a tradition of working across diverse spatial and temporal scales, landscape ecologists (and others working at broad spatial extents) contribute to emerging concepts including macrosystems ecology (Heffernan et al. 2014), globalization (Mayer et al. 2005), telecoupling (Liu et al. 2013), and the sustainability of socioecological systems (Foley et al. 2005) and urban ecosystems (Grimm et al. 2008).

Advances in landscape ecology include applications of landscape and population genetics to measure habitat connectivity, percolation theory to test the influence of topography on the spread of disturbances, graph theory to measure the connectivity of habitat, and the use of acoustics and soundscapes to remotely monitor biodiversity (e.g., Risser et al. 1984, Gardner et al. 1987, Urban and Keitt 2001, Pijanowski et al. 2011). Landscape ecology also integrates methods and issues from the social sciences, including environmental history, geography, planning, anthropology, natural resource studies, and spatial economics (Turner 1989, Antrop 2001); this integration of ecological and social systems has led to a strong focus on drivers and feedback in coupled human and natural systems (Musacchio et al. 2005, Heffernan et al. 2014, Liu et al. 2015). In addition, landscape ecologists have sought to forge strong relationships between academic and nonacademic scientists; among researchers, practitioners, and policymakers; and between theory and applications. These partnerships have led to outcomes such as improvements in water management and policy in the restoration of the Florida Everglades (LoSchiavo et al. 2013) and strategies to enhance forest resilience and economic benefits through diversifying tree-species composition in Canadian forests (Dymond et al. 2014). Policy–research partnerships support a synergistic relationship between policy actions and knowledge building, each benefiting from advances in the other.

Landscape ecology uses a variety of tools and data that can support decisionmaking at multiple scales and provide a bridge between policy and research-based management actions (Opdam et al. 2013). The data sets and modeling frameworks that provide a foundation for informing decisions range from historical vegetation surveys, palaeoecological data, large-scale networks of spatially explicit survey information (such as the National Gap Analysis Program and the Long Term Ecological Research Network in the United States), aerial photographs, and remote sensing to spatial simulations of landscape change, including agent-based models, climate-change impact models, and land-use change scenarios. These data and models are integrated to identify mechanistic and spatial relationships, understand how abiotic gradients (e.g., precipitation or topography) alter those relationships, develop time series describing past responses of social and ecological systems to changes in land use and climate, and generate scenarios of future responses, providing guidance for management actions and decisionmaking.

In the following subsections, we provide an overview (including specific examples) of three priority areas identified by US IALE members as the most pressing environmental policy issues to which the science of landscape ecology can contribute. This is a small subset of theoretical and applied advances in landscape ecology research, but it serves to showcase the important policy-relevant contributions of landscape ecology used to focus this State of the Science report.

Land use–land cover change

Land use–land cover change (LULCC) is an important global change agent, and although the direct impacts of LULCC are relatively easy to identify, foreseeing the indirect and cumulative effects of landscape change is more difficult (Wiens et al. 2011). Many policy decisions must contend with these unintended and indirect consequences of land-use decisions, which are becoming more prevalent with globalization (Mayer et al. 2005, Liu et al. 2015). Explicit consideration of LULCC at multiple scales is a hallmark of landscape ecology and provides a source of information from which land-use policy can draw (Opdam et al. 2013, Table 1).

Land use is defined in terms of human activity, typically categorized into classes such as *industrial*, *agricultural*, and *forest plantation*. In contrast, *land cover* is not defined in terms of human activities and includes categories such as *forest*, *wetlands*, and *open water*. Changes in land use and land cover can have substantial landscape-level effects, including biodiversity loss, habitat fragmentation, and water-cycle interruptions, as well as broader-scale feedback to climate through interactions among the biosphere, hydrosphere, and atmosphere (Foley et al. 2005, Avila et al. 2012). Other LULCC can have positive effects on biodiversity and ecosystems, particularly intentional ecological restoration efforts such as reforestation. Finally, LULCC also includes changes that are not directly driven by human activities, such as transitions between woodlands and savannas resulting from changes in precipitation. A single LULCC, such as the clearcutting of forests or the planting of a monoculture tree plantation, can have long-term legacy effects on ecosystem dynamics, including succession and disturbance feedback (e.g., Loudermilk et al. 2013). Land use also affects the challenges organisms face when navigating or migrating through landscapes (e.g., Wegner and Merriam 1979, Scheller and Mladenoff 2008). Three of the most widespread LULCCs globally are deforestation, agricultural expansion, and urbanization.

Deforestation. Changes in forest cover due to deforestation and forest degradation continue to be a principal focus of LULCC studies globally. Forests are increasingly recognized for their crucial contribution to biodiversity, ecosystem services, and the global carbon cycle (Liu et al. 2015); however, the global trend in forest cover continues downward, with approximately 35% of the world's primary forest cover converted to other land uses since the advent of agriculture (Mackey et al. 2014). Remaining forests occupy about 34% of the earth's terrestrial surface, with primary forest, forest plantations, and forests managed for wood products accounting for approximately 36%, 7%, and 57%, respectively (FAO 2010). Landscape ecology has influenced forest policy along a continuum of land use: from the conservation of primary forests and the management of commercial forests to the restoration of converted or degraded forest lands (Liu and Taylor 2002, Lamb 2014, Table 1).

Increased understanding of scale-dependent effects of landscape composition and configuration on species habitat selection, movement, and demography has given rise to significant changes to forest policy. Habitat protection and recovery efforts for endangered and threatened species have shaped land-management decisions in forested landscapes. For example, the needs of the endangered northern spotted owl (*Strix occidentalis caurina*) and controversy over the harvest of their old-growth forest habitat culminated in the Northwest Forest Plan, ushering in a new era of forest ecosystem management on federal public lands in the United States (Yaffe 1994). Enhanced understanding of the landscape needs of the endangered boreal woodland caribou (*Rangifer tarandus caribou*) resulted in altering how their critical habitat is identified, widening the definition of their critical habitat to one that encompasses a majority of boreal forest in Canada (Environment Canada 2012).

Strategies to maintain or enhance forest ecosystem services via policy, including carbon storage and climate regulation, can also benefit from the principles and practices of landscape ecology. For example, the United Nations' REDD Program (Reducing Emissions from Deforestation and Forest Degradation) offers incentives for developing countries to reduce emissions from forested lands and invest in low-carbon paths to sustainable development (Mackey et al. 2014). These plans often include large-scale forest conservation, sustainable forest management strategies, and the enhancement of forest carbon stocks. Forest certification also offers consumer-driven incentives for forest stewardship. Although certification is voluntary, it has served to operationalize concepts of sustainable forest management in the marketplace and in international agreements and is increasingly reflected in government policies in Europe and North America (Cashore et al. 2003, Pulzl et al. 2013). Developing and updating the standards for REDD projects and certification programs, along with associated criteria and indicators, require robust landscape science to assure that these programs are having the intended effects.

Agricultural land use. Agricultural expansion often converts biologically diverse habitats to low diversity systems (e.g., crop monocultures) with high inputs of nutrients and synthetic chemicals (Foley et al. 2005, Fahrig et al. 2011). In addition to becoming sources of nutrient pollution, pest populations often increase in or are attracted to agricultural ecosystems, facilitated by the dominant practice of monoculture cropping. The simplification of industrial agricultural landscapes reduces the ecosystem services they provide not just at the field level but also over much larger areas (Prager et al. 2012). Nonpest species can face high mortality from pest eradication programs (e.g., use of pesticides and herbicides), turning agricultural areas into habitat sinks, where more individuals die than are produced through reproduction. Many of these impacts are either specifically exempt from existing laws (e.g., agricultural runoff is classified as a nonpoint pollution source and exempted from the US Clean Water Act of 1972) or are not addressed at all. Therefore, the strategic management of agricultural practices and expansion is important for both ecological and policy concerns.

The landscape configuration of monocultures can be designed to mitigate their negative impacts on biodiversity and ecosystem functions, mainly through land-use policy (Table 1). Early work in agricultural landscapes focused on creating corridors for species persistence and movement (Wegner and Merriam 1979), examining the edge effects on species diversity (Fry and SarlovHerlin 1997) and installing riparian buffers to limit nutrient runoff (Peterjohn and Correll 1984). More recent research has quantified landscape structure, such as the diversity of crop types (Fahrig et al. 2011), to better understand how agriculture affects terrestrial and aquatic ecosystems. Results from this research have informed the policies of the US Department of Agriculture's Conservation Reserve Program (CRP). These insights have also been integrated and extensively applied at a multinational scale via European Union agricultural and landscape policy, particularly through payment for ecosystem services programs (e.g., the 2000 European Landscape Convention; Cassatella and Peano 2011, Conrad et al. 2011, Prager et al. 2012). However, agroenvironmental programs in the US Farm Bill (e.g., CRP, the Grasslands Reserve Program, and the Wetlands Reserve Program) are primarily enacted at the farm scale, suggesting an obvious gap in addressing processes that take place within a broader landscape context. Landscape-ecology research can inform these agricultural policy gaps for regional, large-scale biodiversity conservation and preservation of ecosystem services, such as pollination and the biocontrol of pests (Kennedy et al. 2013, Office of the Press Secretary 2014).

Landscape ecology also helps shape mitigation measures that can be integrated into agricultural practices. The recent land sharing–versus–land sparing debate seeks to find a balance between ecosystem conservation and agricultural production across landscapes (Mastrangelo et al. 2014). A *land-sharing approach* supports production practices that provide ecological benefits, typically those that eliminate pesticides and synthetic fertilizers, and create habitat heterogeneity, supporting ecosystem functions. Shade-grown coffee plantations and polyculture cropping systems with perennials are examples of this approach (Railsback and Johnson 2014). Maintaining quality habitat

Table 1. Selected examples in which landscape ecology can address emerging challenges and inform policy development and implementation.

Emerging challenge	Principles from landscape ecology	Examples of current policy applications	Opportunities for better integration with policy
Land use–land cover change (LULCC)	Focus on scale and spatial pattern informs policy on pattern and intensity of land-use change (e.g., forestry). Ecological impacts are determined by LULCC patterns (extent, intensity, connectivity with surrounding matrix).	• US Roadless Area Conservation Rule of 2001 for National Forests • Regional and municipal land-use planning (e.g., the urban growth boundary of Portland, Oregon) • Land-management plans across scales (stand, tenure, and regional levels), such as the Landscape Conservation Cooperative Network in the United States	Use land sparing and sharing ideas to enable sustainable landscape planning. Examples: • *Agriculture:* USDA Conservation Reserve Program (CRP) • *Energy:* US Energy Independence and Security Act of 2007, state-level Renewable Portfolio Standards • *Forestry:* United Nations' Reducing Emissions from Deforestation and Forest Degradation (REDD)
	Emphasis on temporal scales and ecosystem dynamics provides insights on legacy effects or altered disturbance regimes	• Endangered Species Act (US) • Species At Risk Act (Canada)	• Management of disturbances over large landholdings (e.g., grazing land, Bureau of Land Management) • "Healthy Forests Initiative" (salvage logging), USDA Forest Service)
	Understanding links between spatial pattern and ecological processes can help predict how LULCC will affect biodiversity and ecosystem functions	• *Forests:* REDD program, Best Management Practices and Certification Schemes • *Agriculture:* Payment for Ecosystem Services programs, US Clean Water Act of 1972, 2000 European Landscape Convention, US Farm Bill/CRP	• Need for integrated planning across sectors (e.g., forestry, agriculture, mining). • Jurisdictional boundaries (state/provincial versus federal) create challenges and potential conflict in policy direction and the implementation of management strategies. • CRP impacts beyond the farm/parcel scale
LULCC: Urbanization	Predicting urban expansion based on past patterns and how landforms shape urban growth. How urbanization affects ecological processes such as species movements and local extinctions.	• *Zoning* (Greenbelts/urban growth boundaries and Conservation Subdivisions, or CSDs) • Natural landscaping ordinances (e.g., Ordinance 7522 in Tucson, Arizona)	Ecologically based building codes, vegetation and zoning ordinances
	Urbanization increases area of impervious surfaces, affecting hydrological systems and contributing to the urban heat island effect. Spatially explicit models predict how hydrological flows, water quality, and temperature will be affected by urban land cover and how these changes might be mitigated.	• Green infrastructure and brownfield redevelopment (e.g., the US Environmental Protection Agency's brownfields programs) • Urban agriculture/community gardens zoning (e.g., Chicago, Illinois)	Greenbelts and green infrastructure size and placement in landscape
	Nonnative species can increase in urban areas. Propagule pressures and responses of nonnative species to changes in landscape cover and composition can be modeled.	Management of invasive species that affect species at risk can be implemented through species Recovery Plans (Canada)	• Help craft policy for invasive or nonnative species management on private land. • Invasive species often become problematic when they cross geographical boundaries; therefore, cross-jurisdictional cooperation is a necessary challenge.
Climate change	Climate resilient landscapes are heterogeneous and well connected. Identify baseline levels of connectivity and heterogeneity for a region; how should these be configured to maintain resilience against climate change.	Kyoto Protocol (Ecologically relevant CO_2 emission targets)	• Scaling up effective local actions (e.g., US Conference of Mayors Climate Protection Agreement) • Disaster response policies (e.g., National Flood Insurance Program under US FEMA)
	Spatially explicit species distribution and movement models can predict how species ranges will shift, depending on species' niche, dispersal strategies, landscape connectivity, and landscape genetics.	Use of spatially dynamic reserves for species conservation in landscapes changing due to disturbances in forested landscapes	• Placement of migration corridors • Spatially dynamic reserves for climate-change adaptation through programs such as conservation easements on private property
	Forecast models must incorporate uncertainty and stochasticity. Landscape models with limited data and/or without experimental replicate units will be useful case studies for modeling "unknown unknowns."	Policy gap?	Disturbance response and climatic adaptation (National Flood Insurance Program)

Note: The policy examples draw mainly from the United States and Canada, which are regions with which the authors are familiar. Landscape ecology can inform policy in other regions of the globe; however, the specific applications may be different from those outlined here.

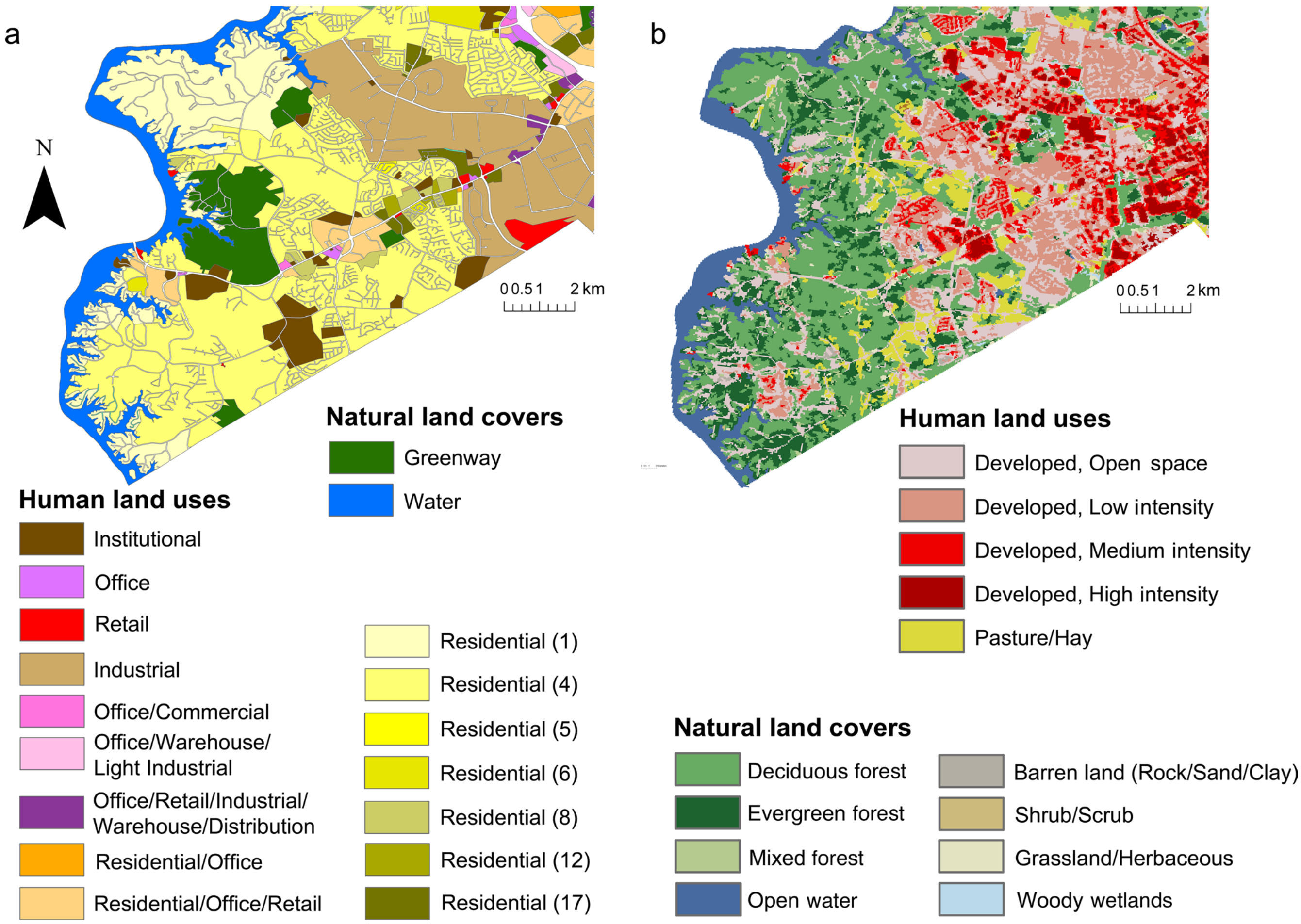

Figure 2. Urban planners and landscape ecologists use different vocabularies to describe land covers and uses. Planners have developed detailed classifications of human uses but relatively poor classifications of natural land covers (a). The opposite is true of landscape ecologists (b). In order for the disciplines to collaborate more effectively, a shared terminology must be developed. The data in panel (a) are from the City of Charlotte's (North Carolina) 2013 Adopted Future Land Use map. The residential land uses in (a) are differentiated by the number of dwellings per acre in parentheses. The data in panel (b) are from the United States Geological Survey's 2006 National Land Cover Database.

for crop pollination services can also ensure functional habitat connectivity throughout agricultural fields for wild pollinators (Kennedy et al. 2013), although bee foraging ranges limit these effects (Garibaldi et al. 2011). However, the land-sharing approach requires a larger amount of land to produce the same amount of crop biomass, with no guarantee of preserving biodiversity among shared areas.

A *land-sparing approach* (also known as land intensification, or a triad strategy in forest management) seeks to confine industrial production to as small an area as possible, with the implication that production areas will be inhospitable to most species (Phalan et al. 2011). The land-use intensification approach to conservation has a longer history in forest policy and management (Lindenmayer et al. 2012). Lessons learned in forested lands may be broadly applicable to agricultural and urban land uses, although the comprehensive effects of land-intensification policies need to be better understood (Tscharntke et al. 2012).

Landscape ecologists have not reached a consensus as to whether and where land-sharing and -sparing approaches should be considered for land management. For ecological services such as pollination and predation on agricultural pests, the land-sharing approach often leads to the most beneficial outcomes for both biodiversity and production (Railsback and Johnson 2014). In contrast, land sparing may result in better conservation outcomes for species with small distributional ranges (Phalan et al. 2011). For many landscapes, maximizing biodiversity and production may come from the judicious use of both approaches. Land managers will need to determine the appropriate balance between the two approaches to maintain the ecosystem services and functionality of a landscape (Phalan et al. 2011, Mastrangelo et al. 2014).

Urbanization. The vast majority of people in industrialized countries live in urban areas, and people in developing countries are rapidly becoming predominantly urban dwellers as well (Grimm et al. 2008). Urban areas are strongly dependent on their regional environments for energy, natural resources, and amenities and substantially influence their regions through these connections (Pickett et al. 2011). Cities influence ecosystems through habitat fragmentation and loss, changes to local and regional weather and climate (e.g., the urban heat island effect), alterations to nutrient and water cycling, the production of excess carbon dioxide, and the influx of nonnative species (Foley et al. 2005, Pickett et al. 2011). Cities can also be hotspots for new introductions of nonnative species, some through the globalized trade of exotic plant and animal species (McKinney 2006). Via escape, release, or spread through urban rivers and green space, introduced species can invade surrounding rural and natural areas, often causing additional strain on native species and their habitat. Through the loss of native species and the gain of invasive ones in cities with similar infrastructure design and pattern, urban areas become "homogenized" and lose their biological distinctiveness, attaining an ecological similarity to many other urban areas around the world (McKinney 2006, Groffman et al. 2014). *Urbanization* can lead to the production of algal blooms and eutrophic conditions through the addition of excess nitrogen and phosphorus from runoff (Grimm et al. 2008), and urban streams are affected by higher concentrations of toxic chemicals contributed by runoff from impervious surfaces. Furthermore, water flow becomes increasingly variable because of reduced infiltration.

Socioeconomic drivers within cities create complex, scale-dependent patterns in biodiversity and ecosystem functioning (Wu 2014). Plant diversity increases with family income (i.e., the "luxury effect"; Hope et al. 2003), and urban biodiversity can peak in suburban residential areas as a result of higher habitat heterogeneity, higher primary productivity (driven by human-derived water and nutrient supplements), and the introduction of nonnative species (McKinney 2006). The associated ecological effects depend on the spatial and structural characteristics of urban growth, suggesting that policy outcomes will differ among cities of different sizes and at varying scales within cities.

Land-use policies can lead to more sustainable cities through the protection and addition of habitat patches, green space, and other vegetation (Lin and Fuller 2013, Wu 2014, Table 1). For example, vegetation features (or "green infrastructure") meant to mitigate stormwater runoff, such as green roofs, urban parks, street trees, and urban wetlands, can increase habitat availability and connectivity for terrestrial organisms (Braaker et al. 2014) and decrease urban heat island effects. Private and community gardens, rain gardens, and vacant lots also can boost local terrestrial and aquatic insect diversity while reducing the area of impervious surfaces (Philpott et al. 2014). Assuming the net-positive impact of gardens, policies aimed at legitimizing them would greatly support these activities. One example is a recent ordinance in Chicago, Illinois, that approved a new zoning code for urban agriculture, making it easier to establish larger community gardens. In combination with allied disciplines such as urban planning and landscape architecture, landscape ecology can inform land-use policies that dictate green infrastructure design, its distribution throughout urban landscapes, and the optimal locations for urban infill through brownfield redevelopment (Antrop 2001, Pickett et al. 2011, Felson et al. 2013).

Other policy efforts focus on setting the appropriate boundaries for cities, and the compact-versus-sprawling-city debate is similar to the land sharing–sparing debates in agriculture. The relative ecological impacts of promoting urban intensification by enacting "greenbelt" policies and increasing population density (land sparing) versus greening lower-density urban areas through the use of native plantings and connected green space (land sharing) are likely to depend on local conditions and legacies (Lin and Fuller 2013). For urban land use, limited research suggests that land sparing results in higher landscape-level biodiversity than land sharing does (Gagné and Fahrig 2010), although residents of high-density, compact developments may experience lower well-being because of reduced access to green space and other natural amenities.

At the urban fringe, the loss of agricultural and natural areas to exurban development is of growing concern (Theobald 2004). Amending zoning ordinances to allow for *conservation* (or "open space") *subdivisions* (CSDs) has been proposed as a way to restrain exurban development. In a CSD, 50% or more of the developable land is preserved in open space through homeowners associations or land trusts, and steps are taken to reduce habitat fragmentation, to protect riparian corridors and other ecologically fragile habitats, and to link multiple CSDs together to create an interconnected greenspace network (Arendt 2004). The few studies that have evaluated the success of CSDs from an ecological standpoint suggest that habitat connectivity for the benefit of a variety of species can be maintained (Freeman and Bell 2011), and the majority of CSDs focus protection efforts on representative ecosystems that are native to the area (Milder and Clark 2011).

It is important to note that cluster subdivisions (that maintain open space but not necessarily links between systems, native species, or use guidelines) do not seem to afford the same benefits as CSDs (Lenth et al. 2006). We encourage landscape ecologists to document the differences in conservation, cluster, and traditional subdivisions to assist planners in developing more ecologically protective zoning ordinances. Resolving these planning dilemmas will require harmonizing the way urban landscapes are described and investigated across disciplines (figure 2). Planners generally have limited means of incorporating ecological information into the planning process. Developing a shared terminology, in which ecological attributes are combined with zoning and land-use designations, will improve the translation of landscape ecological knowledge into planning practice. As a first step, urban landscape ecologists should consider adopting the zoning and land-use designations used by planners.

Climate change

Understanding the effects of climate on ecological patterns and processes requires the cross-scale perspective that is a mainstay of landscape ecology. Anthropogenic climate change has altered patterns of global temperature and precipitation, and nearly all climate projections predict accelerated changes through at least the end of this century (IPCC 2013). These changes will be manifest in myriad ways, including not only warmer temperatures but also longer growing seasons, rising sea levels, increased extreme events, and altered seasonality of precipitation and snowmelt. These changes are region dependent; will affect ecosystems and the socioeconomic systems reliant on and interacting with those ecosystems; and will be influenced and amplified by policy and LULCC (Avila et al. 2012).

Modern and paleo-distribution data indicate that most species have an affinity to a particular climatic and disturbance regimes, as well as the ability to shift distributions in response to climate change (Delcourt and Delcourt 1988). The capacity to adapt to climate change varies within and across species, often determined by life-history attributes and genetic variation (Foden et al. 2013). Whereas in the Pleistocene, plants and animals migrated relatively unimpeded across landscapes in response to natural climate changes, contemporary landscapes are fragmented by LULCC. Understanding the spatial connectivity needed to enable the biota to disperse, migrate, and adapt in response to a changing climate and LULCC is crucial. Data on landscape configuration combined with the biophysical requirements of organisms allow for explicit estimates of (a) how a species might be maintained in its current location, (b) whether natural migration is possible given the anticipated pace of change, (c) whether natural migration can be facilitated by specific LULCC policy, and (d) whether species translocation is feasible where natural migration is not possible (Iverson 1999, Scheller and Mladenoff 2008, Thomas 2011).

With climate change, many regions are likely to see changes in natural disturbance regimes: wildfire frequency, windstorm frequency and intensity, flooding, and forest insect and disease outbreaks (Dale et al. 2001). Changing disturbance regimes affect vegetation at broad spatial and temporal scales (Buma et al. 2013). Changes to disturbance regimes that progress outside of historical ranges may be more ecologically severe, although mortality will vary widely (Miller et al. 2011). In some cases, climate effects interact with long-term legacy effects from historic LULCC and forest recovery and may create unique conditions for future disturbances and management (e.g., Loudermilk et al. 2013, Buma 2015). In other cases, climate change may alter fundamental disturbance drivers; for example, fire disturbance frequency and intensity may switch from being primarily controlled by fuels to being primarily determined by drought. These changes may require major shifts in risk management and prediction (e.g., fire modeling, firefighting).

Notwithstanding legacy effects, future climate-driven disturbances and coupled interactions among disturbances will likely be the main catalyst for rapid vegetation change. Although the direct physiological effects of climate on organism survival and establishment may drive species' range changes in many areas where large disturbances are currently rare (Zimmermann et al. 2009), in areas where those large events currently occur (or will start to occur in the future climate), changes in disturbance regimes will likely overshadow and outpace these direct effects where and when they occur (Loudermilk et al. 2013, Syphard et al. 2013). Given the pace and magnitude of changes likely to occur to species' ranges and the distribution of their habitat (Zimmerman et al. 2015), conservation policy may need to move toward the greater use of spatially dynamic reserves to assist species' redistribution and adaptation (Strange et al. 2006, Moilanen et al. 2014, Table 1).

Visualizing resilience and response to climate change. Estimating the effects of climate change involves linking spatially coarse-scaled modeling projections of climate conditions to processes that occur at finer scales. For example, topography exerts a moderating effect on local climate (e.g., temperature regimes, water balance, snow pack) that drives species range dynamics and population dynamics (Ashcroft et al. 2009, Serra-Diaz et al. 2015). Spatial and topographic downscaling of climate data and models is required to bridge this gap (e.g., Franklin et al. 2013). Other temporal and spatial scaling factors affecting ecological responses to climate change include interannual climate variability and extreme events (Zimmermann et al. 2009).

Although there is currently no consensus on the characteristics of a climate-resilient landscape, landscape ecology can provide insight into future range of uncertainty and options for management, as well as guide long-term policy decisions. Historically resilient systems are becoming less informative because of shifting baselines; past reference systems for management are no longer attainable (Thomas 2011). Landscape ecology holds promise for ameliorating the effects of climate change by identifying locations of inherent resilience that can be preserved or restored to resilient conditions (e.g., Buma and Wessman 2013, Duveneck and Scheller 2015) and vulnerable areas where restoration or protection is difficult. For example, changing climate and disturbance regimes may substantially diminish the capacity of the US Federal Emergency Management Agency to effectively implement disaster-management policies, such as the National Flood Insurance Program. Landscape analyses can determine which areas are likely to experience greater flooding risks because of the loss of ecosystems that can mitigate the effects of storms and sea-level rise.

Interactions among global-change drivers and policy

We have discussed LULCC and climate change separately, and although existing policies often address them separately, these drivers are connected. One example of an emerging issue linking LULCC and climate change is biomass-based bioenergy production (generating energy from crops, tree plantations, or residues from agricultural and natural habitats). Advocates suggest that bioenergy can mitigate climate change by providing carbon-neutral energy sources (Dale et al. 2011). In the United States, biofuel production is incentivized by policies such as the federal Energy Independence and Security Act of 2007 and many state-level renewable portfolio standards. However, if this transition to a biomass-based energy system is to occur at all, it must be accomplished without further reducing biodiversity and ecosystems health through the use or development of truly carbon-neutral energy sources and in a way that does not complicate landscape management in other areas (Dale et al. 2011, Liu et al. 2015). One policy choice concerns the use of the roughly 15 million hectares of marginal cropland enrolled in the CRP to grow bioenergy crops (Werling et al. 2014). These lands are currently managed as highly diverse grasslands and young forests (Wiens et al. 2011). Ultimately, the policy choice among options such as using marginal land to produce biofuels to offset carbon-intensive fossil fuels or restoring natural habitats as carbon sinks on that land will be improved by our knowledge of the interactions between LULCC and its climate impacts (Liu et al. 2015).

Landscape ecologists are keenly interested in how and why resilience is spatially variable across landscapes and in creating more climate-resilient landscapes through land management (Hobbs et al. 2014). The pace at which species are exposed to climate change (Serra-Diaz et al. 2014) will be influenced in part by the structure of the landscape. Landscapes that have already undergone extensive LULCC and are highly fragmented or dominated by nonnative species may have reduced resilience to climate change. It can be difficult to test policy and management options to mitigate these global-scale changes or manage for them. Therefore, landscape modeling is a priority. Via modeling, landscape ecology offers a unique capacity for scientists and land managers to work together to "test" different management options using computer simulation, mitigating the need for resource-intensive landscape-manipulation experiments (e.g., Buma and Wessman 2013, Duveneck and Scheller 2015). Models have the capacity to include a variety of social, ecological, and climatic drivers, including LULCC, restoration scenarios, and disturbances, as well as their interactions for long-term landscape management (e.g., Syphard et al. 2013). The ability to test complex scenarios at a variety of scales is one of the more practical approaches brought by landscape ecology to the management community. Nevertheless, if the theories and tools derived from landscape ecology are to inform LULCC and climate-change policies, associated uncertainties must be clearly communicated and incorporated into policy recommendations (Nassauer and Corry 2004).

Conclusions

Landscape ecologists in the United States recently identified climate change, LULCC (including deforestation and agriculture), as well as a particular type of LULCC, urbanization, as the most pressing issues to which the discipline of landscape ecology can contribute policy-relevant science. As a result, among the many objectives addressed by the discipline, these three formed the focus of this State of the Science essay.

The world is entering a period of significant climate departure for which there is no analog in our historical data sets (Williams and Jackson 2007). Many recent climate and LULCC changes may be irreversible, and scientists, managers, and policymakers have begun to adapt to this new reality. Proactive management and restoration plans must identify habitats that will be resilient under future climate regimes; these may include novel ecosystems when restoration or maintenance of historic conditions are infeasible because of legacy effects (Hobbs et al. 2014).

The policy challenges posed by these conditions are place based: They occur in specific spatial and social contexts with local and regional constraints on inputs and outcomes. For example, a critical examination of spatial patterns in ecosystem services can contribute to locally appropriate zoning and green infrastructure policies to support urban ecosystems and enhance the quality of life for urban residents (Felson et al. 2013). At broader scales, long-term data sets (i.e., the 30-year Landsat satellite image series) can be used to evaluate and improve the implementation of existing land-use policies, such as the 2001 Roadless Area Conservation Rule for US national forests and the national No Net Loss goal for wetlands (Mayer and Lopez 2011). Finally, studies that reveal the links among social, economic, and ecological drivers of land-use change are well positioned to provide guidance on policies (such as REDD) that affect both LULCC and climate change (Liu et al. 2015). However, policy-sensitive problems must be differentiated from policy-insensitive problems: Some problems will be responsive to management activities or behavioral changes, whereas others will not.

Landscape ecology is well positioned to examine the causes, consequences, and interactions of LULCC, urbanization, and climate change at multiple scales. The field is at the forefront of providing information on the complex interactions between the environment and ecosystems—and between urban and nonurban land uses and climate—at multiple spatial and temporal scales. The focus on pattern and process allows for the exploration of future scenarios of specific policy actions, often through simulations and modeling exercises (Nassauer and Corry 2004). However, we must "move past the map" and toward discussions of the natural and anthropogenic mechanisms involved in LULCC and climate change so that policies may be developed or revised to address these mechanisms directly (rather than only their impacts).

The field is well placed to be a bridge between other specialized disciplines and policymakers, putting science in a spatial and temporal context and supporting scientifically informed place-based policy. There are many opportunities to collaborate across disciplines to address these problems. This can be done by bridging terminology and methodology gaps, encouraging cross-attendance of all parties (including policymakers) at professional-society meetings, learning from and sharing approaches and tools, and training students in interdisciplinary thinking. Landscape ecologists also have a responsibility to better integrate stakeholders and public interests into their research (Conrad et al. 2011, Opdam et al. 2013), communicate findings to policymakers, and accurately and responsibly portray the uncertainty in those findings while also conveying the risk of inaction. Its unique perspective across time, space, and disciplines means that landscape ecology is well positioned to contribute policy-relevant knowledge for solutions to the world's most crucial environmental problems.

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References cited

Antrop M. 2001. The language of landscape ecologists and planners: A comparative content analysis of concepts use in landscape ecology. Landscape and Urban Planning 55: 163–173.

Arendt R. 2004. Linked landscapes: Creating greenway corridors through conservation subdivision design strategies in the northeastern and central United States. Landscape and Urban Planning 68: 241–269.

Ashcroft MB, Chisholm LA, French KO. 2009. Climate change at the landscape scale: Predicting fine-grained spatial heterogeneity in warming and potential refugia for vegetation. Global Change Biology 15: 656–667.

Avila FB, Pitman AJ, Donat MG, Alexander LV, Abramowitz G. 2012. Climate model simulated changes in temperature extremes due to land cover change. Journal of Geophysical Research: Atmospheres 117: D04108.

Braaker S, Ghazoul J, Obrist MK, Moretti M. 2014. Habitat connectivity shapes urban arthropod communities: The key role of green roofs. Ecology 95: 1010–1021.

Buma B. 2015. Disturbance interactions: Characterization, prediction, and the potential for cascading effects. Ecosphere 6 (art.70).

Buma B, Wessman CA. 2013. Forest resilience, climate change, and opportunities for adaptation: A specific case of a general problem. Forest Ecology and Management 306: 216–225.

Buma B, Brown C, Donato D, Fontaine J, Johnstone J. 2013. The impacts of changing disturbance regimes on serotinous plant populations and communities. BioScience 63: 866–876.

Cashore B, Auld G, Newsom D. 2003. Forest certification (eco-labeling programs and their policy-making authority: Explaining divergence among North American and European case studies. Forest Policy and Economics 5: 225–247.

Cassatella C, Peano A, eds. 2011. Landscape Indicators: Assessing and Monitoring Landscape Quality. Springer.

Conrad E, Christie M, Fazey I. 2011. Is research keeping up with changes in landscape policy? A review of the literature. Journal of Environmental Management 92: 2097–2108.

Dale VH, et al. 2001. Climate change and forest disturbances. BioScience 51: 723–734.

Dale VH, Kline KL, Wright LL, Perlack RD, Downing M, Graham RL. 2011. Interactions among bioenergy feedstock choices, landscape dynamics, and land use. Ecological Applications 21: 1039–1054.

Delcourt HR, Delcourt PA. 1988. Quaternary landscape ecology: Relevant scales in space and time. Landscape Ecology 2: 23–44.

Duveneck MJ, Scheller RM. 2015. Climate suitable planting as a strategy for maintaining forest productivity and functional diversity. Ecological Applications 25: 1653–1668.

Dymond CC, Tedder S, Spittlehouse DL, Raymer B, Hopkins K, McCallion K, Sandland J. 2014. Diversifying managed forests to increase resilience. Canadian Journal of Forest Research 44: 1196–1205.

Environment Canada. 2012. Recovery Strategy for the Woodland Caribou (*Rangifer tarandus caribou*), Boreal Population, in Canada. Species at Risk Act Recovery Strategy Series. Environment Canada. (8 March 2016; *www.registrelep-sararegistry.gc.ca/virtual_sara/files/plans/rs_caribou_boreal_caribou_0912_e1.pdf*)

Fahrig L, Baudry J, Brotons L, Burel FG, Crist TO, Fuller RJ, Sirami C, Siriwardena GM, Martin J-L. 2011. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. Ecology Letters 14: 101–112.

[FAO] Food and Agricultural Organization of the United Nations. 2010. Global Forest Resources Assessment 2010: Main Report. FAO. Forestry Paper no. 163. (8 March 2016; *www.fao.org/docrep/013/i1757e/i1757e.pdf*)

Felson AJ, Oldfield EE, Bradford MA. 2013. Involving ecologists in shaping large-scale green infrastructure projects. BioScience 63: 882–890.

Foden WB, Butchart SH, Stuart SN, Vié J-C, Akçakaya HR, Angulo A, DeVantier LM, Gutsche A, Turak E, Cao L. 2013. Identifying the world's most climate change vulnerable species: A systematic trait-based assessment of all birds, amphibians and corals. PLOS ONE 8 (art. e65427).

Foley JA, et al. 2005. Global consequences of land use. Science 309: 570–574.

Forman RT. 1995. Land Mosaics: The Ecology of Landscapes and Regions. Cambridge University Press.

Franklin J, Davis FW, Ikagami M, Syphard AD, Flint A, Flint L, Hannah L. 2013. Modeling plant species distributions under future climates: How fine-scale do climate models need to be? Global Change Biology 19: 473–483.

Freeman RC, Bell KP. 2011. Conservation versus cluster subdivisions and implications for habitat connectivity. Landscape and Urban Planning 101: 30–42.

Fry G, SarlovHerlin I. 1997. The ecological and amenity functions of woodland edges in the agricultural landscape: A basis for design and management. Landscape and Urban Planning 37: 45–55.

Gardner RH, Milne BT, Turner MG, O'Neill RV. 1987. Neutral models for the analysis of broad-scale landscape pattern. Landscape Ecology 1: 19–28.

Garibaldi LA, Steffan-Dewenter I, Kremen C, Morales JM, Bommarco R, Cunningham SA, Klein AM. 2011. Stability of pollination services decreases with isolation from natural areas despite honey bee visits. Ecology Letters 14: 1062–1072.

Groffman PM, et al. 2014. Ecological homogenization of urban USA. Frontiers in Ecology and the Environment 12: 74–81.

Heffernan JB, et al. 2014. Macrosystems ecology: Understanding ecological patterns and processes at continental scales. Frontiers in Ecology and the Environment 12: 5–14.

Hobbs RJ, et al. 2014. Managing the whole landscape: Historical, hybrid, and novel ecosystems. Frontiers in Ecology and the Environment 12: 557–564.

Hope D, Gries C, Zhu W, Fagan WF, Redman CL, Grimm NB, Nelson AL, Martin C, Kinzig A. 2003. Socioeconomics drive urban plant diversity. Proceedings of the National Academy of Sciences 100: 8788–8792.

[IPCC] Intergovernmental Panel on Climate Change. 2013. Climate Change 2013: The Physical Science Basis. Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press.

Iverson LR, Prasad AM, Hale BJ, Sutherland EK. 1999. Atlas of Current and Potential Future Distributions of Common Trees of the Eastern United States. US Department of Agriculture. General Technical Report no. NE-265.

Kennedy CM, et al. 2013. A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. Ecology Letters 16: 584–599.

Lamb D. 2014. Large-Scale Forest Restoration. Earthscan–Routledge.

Lenth BA, Knight RL, Gilgert WC. 2006. Conservation value of clustered housing developments. Conservation Biology 20: 1445–1456.

Lin BB, Fuller RA. 2013. Sharing or sparing? How should we grow the world's cities? Journal of Applied Ecology 50: 1161–1168.

Lindenmayer D, Cunningham S, Young A. 2012. Perspectives on land use intensification and biodiversity conservation. Pages 137–150 in Lindenmayer D, Cunningham S, Young A, eds. Land Use Intensification: Effects on Agriculture, Biodiversity, and Ecological Processes. CRC Press–Taylor and Francis.

Liu J, Taylor WW, eds. 2002. Integrating Landscape Ecology into Natural Resource Management. Cambridge University Press.

Liu J, et al. 2013. Framing sustainability in a telecoupled world. Ecology and Society 18 (art. 26).

Liu J, Mooney H, Hull V, Davis SJ, Gaskell J, Hertel T, Lubchenco J, Seto KC, Gleick P. 2015. Systems integration for global sustainability. Science 347 (art. 1258832). doi:10.1126/science.1258832

LoSchiavo AJ, Best RG, Burns RE, Gray S, Harwell MC, Hines EB, McLean AR, St. Clair T, Traxler S, Vearil JW. 2013. Lessons learned from the first decade of adaptive management in comprehensive Everglades restoration. Ecology and Society 18 (art. 70). (8 March 2016; *www.ecologyandsociety.org/vol18/iss4/art70*)

Loudermilk EL, Scheller RM, Weisberg PJ, Yang J, Dilts TE, Karam SL, Skinner C. 2013. Carbon dynamics in the future forest: The importance of long-term successional legacy and climate-fire interactions. Global Change Biology 19: 3502–3515.

Mackey B, et al. 2014. Policy options for the world's primary forests in multilateral environmental agreements. Conservation Letters 8: 139–147.

Mastrangelo ME, Weyland F, Villarino SH, Barral MP, Nahuelhual L, Laterra P. 2014. Concepts and methods for landscape functionality and a unifying framework based on ecosystem services. Landscape Ecology 29: 345–358.

Mayer AL, Lopez RD. 2011. Use of remote sensing to support forest and wetland policies in the USA. Remote Sensing 3: 1211–1233.

Mayer AL, Kauppi PE, Angelstam PK, Zhang Y, Tikka P. 2005. Importing timber, exporting ecological impact. Science 308: 359–360.

McGarigal K. 2002. Landscape pattern metrics. Pages 1135–1142 in El-Shaarawi AH, Piergorsch WW, eds. Encyclopedia of Environmetrics, vol. 2. Wiley.

McKinney ML. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127: 247–260.

Milder JC, Clark S. 2011. Conservation development practices, extent, and land-use effects in the United States. Conservation Biology 25: 697–707.

Miller AD, Roxburgh SH, Shea K. 2011. How frequency and intensity shape diversity–disturbance relationships. Proceedings of the National Academy of Sciences 108: 5643–5648.

Moilanen A, Laitila J, Vaahtoranta T, Dicks LV, Sutherland WJ. 2014. Structured analysis of conservation strategies applied to temporary conservation. Biological Conservation 170: 188–197.

Musacchio L, Ozdenerol E, Bryant M, Evans T. 2005. Changing landscapes, changing disciplines: Seeking to understand interdisciplinarity in landscape ecological change research. Landscape and Urban Planning 73: 326–338.

Nassauer JI, Corry RC. 2004. Using normative scenarios in landscape ecology. Landscape Ecology 19: 343–356.

Office of the Press Secretary. 2014. Presidential Memorandum: Creating a Federal Strategy to Promote the Health of Honey Bees and Other Pollinators. The White House, Office of the Press Secretary. (8 March 2016; *www.whitehouse.gov/the-press-office/2014/06/20/presidential-memorandum-creating-federal-strategy-promote-health-honey-b*)

Opdam P, Nassauer JI, Wang Z, Albert C, Bentrup G, Castella J-C, McAlpine C, Liu J, Sheppard S, Swaffield S. 2013. Science for action at the local landscape scale. Landscape Ecology 28:1439–1445.

Peterjohn WT, Correll DL. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. Ecology 65: 1446–1475.

Phalan B, Onial M, Balmford A, Green RE. 2011. Reconciling food production and biodiversity conservation: Land sharing and land sparing compared. Science 333: 1289–1291.

Philpott SM, Cotton J, Bichier P, Friedrich RL, Moorhead LC, Uno S, Valdez M. 2014. Local and landscape drivers of arthropod abundance, richness, and trophic composition in urban habitats. Urban Ecosystems 17: 513–532.

Pickett STA, et al. 2011. Urban ecological systems: Scientific foundations and a decade of progress. Journal of Environmental Management 92: 331–362.

Pijanowski BC, Farina A, Gage SH, Dumyahn SL, Krause BL. 2011. What is soundscape ecology? An introduction and overview of an emerging new science. Landscape Ecology 26: 1213–1232.

Prager K, Reed M, Scott A. 2012. Encouraging collaboration for the provision of ecosystem services at a landscape scale: Rethinking agri-environment payments. Land Use Policy 29: 244–249.

Pülzl H, Hogl K, Kleinschmit D, Wydra D, Arts B, Mayer P, Palahí M, Winkel G, Wolfslehner B, eds. 2013. European Forest Governance: Issues at Stake and the Way Forward. European Forest Institute. (8 March 2016; *www.efi.int/files/attachments/publications/efi_what_science_can_tell_us_2_2013.pdf*)

Railsback SF, Johnson MD. 2014. Effects of land use on bird populations and pest control services on coffee farms. Proceedings of the National Academy of Sciences 111: 6109–6114.

Risser PG, Karr JR, Forman RTT. 1984. Landscape Ecology: Directions and Approaches. Illinois Natural History Survey. Special Publication no. 2.

Riitters KH, O'Neill R, Hunsaker C, Wickham JD, Yankee D, Timmins S, Jones K, Jackson B. 1995. A factor analysis of landscape pattern and structure metrics. Landscape Ecology 10: 23–39.

Scheller RM, Mladenoff DJ. 2008. Simulated effects of climate change, fragmentation, and inter-specific competition on tree species migration in northern Wisconsin, USA. Climate Research 36: 191–202.

Serra-Diaz JM, Franklin J, Ninyerola M, Davis FW, Syphard AD, Regan HM, Ikegami M. 2014. Bioclimatic velocity: The pace of species exposure to climate change. Diversity and Distributions 20: 169–180.

Serra-Diaz JM, Scheller RM, Syphard AD, Franklin J. 2015. Disturbance and climate microrefugia mediate tree range shifts during climate change. Landscape Ecology 30: 1039–1053.

Strange N, Jellemark Thorsen B, Bladt J. 2006. Optimal reserve selection in a dynamic world. Biological Conservation 131: 33–41.

Syphard AD, Regan HM, Franklin J, Swab R. 2013. Does functional type vulnerability to multiple threats depend on spatial context in Mediterranean-climate regions? Diversity and Distributions 19: 1263–1274.

Theobald DM. 2004. Placing exurban land-use change in a human modification framework. Frontiers in Ecology and the Environment 2: 139–144.

Thomas CD. 2011. Translocation of species, climate change, and the end of trying to recreate past ecological communities. Trends in Ecology and Evolution 26: 216–221.

Tscharntke T, Clough Y, Wanger TC, Jackson L, Motzke I, Perfecto I, Vandermeer J, Whitbread A. 2012. Global food security, biodiversity conservation and the future of agricultural intensification. Biological Conservation 151: 53–59.

Turner MG. 1989. Landscape ecology: The effect of pattern on process. Annual Review of Ecology and Systematics 20: 171–197.
———. 2005. Landscape ecology: What is the state of the science? Annual Review of Ecology, Evolution, and Systematics: 319–344.
———. 2015. Twenty-five years of United States landscape ecology: Looking back and forging ahead. Pages 43–54 in Barrett GW, Barrett TL, Wu J, eds. History of Landscape Ecology in the United States. Springer.
Turner MG, Gardner RH. 2015. Introduction to Landscape Ecology and Scale. Pages 1–32 in Turner MG, Gardner RH, O'Neill RV, eds. Landscape Ecology in Theory and Practice: Pattern and Process. Springer.
Turner MG, Gardner RH, O'Neill RV. 2001. Landscape Ecology in Theory and Practice: Pattern and Process. Springer.
Urban DL. 2005. Modeling ecological processes across scales. Ecology 86: 1996–2006.
Urban DL, Keitt T. 2001. Landscape connectivity: A graph-theoretic perspective. Ecology 82: 1205–1218.
Urban DL, O'Neill RV, Shugart HH Jr. 1987. Landscape ecology. BioScience 37: 119–127.
Wegner JF, Merriam G. 1979. Movements by birds and small mammals between a wood and adjoining farmland habitats. Journal of Applied Ecology 16: 349–357.
Werling BP, Dickson TL, Isaacs R, Gaines H, Gratton C, Gross KL, Liere H, Malmstrom CM, Meehan TD, Ruan LL. 2014. Perennial grasslands enhance biodiversity and multiple ecosystem services in bioenergy landscapes. Proceedings of the National Academy of Sciences 111: 1652–1657.
Wiens JA. 2002. Riverine landscapes: Taking landscape ecology into the water. Freshwater Biology 47: 501–515.
Wiens J[A], Fargione J, Hill J. 2011. Biofuels and biodiversity. Ecological Applications 21: 1085–1095.
Williams JW, Jackson ST. 2007. Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment 5: 475–482.
Wu JG. 2014. Urban ecology and sustainability: The state-of-the-science and future directions. Landscape and Urban Planning 125: 209–221.
Yaffe SL. 1994. The Wisdom of the Spotted Owl. Island Press.
Zimmermann NE, Yoccoz NG, Edwards TC, Meier ES, Thuiller W, Guisan A, Schmatz DR, Pearman PB. 2009. Climatic extremes improve predictions of spatial patterns of tree species. Proceedings of the National Academy of Sciences 106: 19723–19728.

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