LINKING THE LANDSCAPE: LEGAL AND POLICY TOOLS TO PROMOTE CONNECTED HABITATS IN FRAGMENTED LANDSCAPES

BY

JACQUELINE M. WILKOSZ

THESIS

Submitted in partial fulfillment of the requirements
for the degree of Master of Science in Natural Resources and Environmental Sciences
in the Graduate College of the
University of Illinois at Urbana-Champaign, 2010

Urbana, Illinois

Adviser:

Professor Eric T. Freyfogle

ABSTRACT

We have entered a sixth mass extinction period, and habitat loss due to human land uses has been named as one of its leading causes. By converting land to urban and agricultural land uses, humans have fragmented millions of acres of once-contiguous habitat. Fragmentation alters the spatial configuration and ecological processes of the remnant habitat fragments. These ecological changes impact plant and animal species, leading to population declines and, for some, local or total extinction. The impacts of fragmentation are projected to become more pronounced as the climate changes, hindering many species from adapting to novel climate conditions by shifting to a range with more hospitable climate conditions. Corridors can improve species viability in heavily-fragmented landscapes as well as in a changing climate by facilitating movement between separate habitat patches. Establishing broad linkages is logically feasible in areas with large reserves of habitat, primarily in the north-western region of the United States. The rest of the nation, however, lacks large habitat reserves and is dominated by private landownership. How do we establish corridors in landscapes like those in central Illinois or the sprawling metro-Chicago suburbs? Implementing linkages in these landscapes will require a coordinated, inter-governmental effort on landscape and regional scales. Legally, we must integrate stewardship into private landowner duties, update the common law meaning of "harm" to encompass ecological harm, and enhance government ability to curb harmful land uses. To achieve *real* conservation gains, however, we must move socially and culturally toward an ethic of stewardship within the private landscape.

For the badgers, and all species that seek to freely roam.

There is nothing impossible,
once shown the way
of perseverance.

ACKNOWLEDGEMENTS

First and foremost, innumerable thanks to my adviser Eric Freyfogle who has been my teacher, my guide, and my friend throughout my four years as a law and graduate student. Without his support, I would not have had the opportunity to undertake this project. His many comments, recommendations, corrections, and criticisms have molded this project into a clear, cohesive argument. Thanks for the many hours he dedicated to this project and for his continued and unwavering support, especially when I lacked faith in myself.

Thanks to my thesis committee, Jeff Brawn and Brian Deal, for reviewing this lengthy endeavor and for their thoughtful input. Their comments were invaluable to helping me clarify the scope and meaning of much of this project's content. Their challenging questions motivated me to more critically assess the substance of this material and my own relationship to this project.

A humble thanks to Drew Downing, who has actively supported me through every step of this project. His patience and his faith in me inspired me to keep "plugging away." His expert edits helped to improve the paper's clarity and flow. And special thanks for his help in animating the badger.

Thank-you to Sam Kramer for his support and encouragement, and especially for his thoughtful comments on my initial draft. He motivated me to think more deeply about the issues and to clarify the direction of my arguments.

A most deep and sincere thanks to my parents, John and Jolene Wilkosz, my sister Janine Wilkosz, and to Roger and Jody Downing for their warm support and encouragement. Thanks especially to Janine for sharing advice and lessons learned from her previous thesis-writing experience. And thanks to my friends, especially Isabel, Erin, and Emily, for cheering me on.

TABLE OF CONTENTS

Preface: An Animal Fable

A certain American Badger (*Taxidea taxus*) makes his home in a heavily farmed landscape in Illinois. He mostly lives in a small patch of grassland, though wandering outward from this patch in search of food. Because nearly all of the state's native prairie is gone--converted to agricultural lands, residential subdivisions, roads, and commercial space--the badger has few good options in establishing a home range.[1] Recently, two other badgers have entered his small grassland area. Given the resulting competition and guided by a desire for more reliable food and perhaps a mate, our badger decides to search for a new home.

Immediately upon setting out he is beset by challenges. At the edge of his grassland he encounters inhospitable terrain—a two-lane highway that provides no cover, food or places for burrows. He trots along this highway, attempting to stay hidden in the grassland edge. Perhaps, he thinks, more hospital vegetation lies ahead. As he journeys, though, the grassy strip ends abruptly in a newly plowed soybean field. The badger pauses at the unappealing terrain, then ambles backward to wait and watch. As dawn arrives he digs a protective burrow near the grassland edge to rest for the day. For two more nights he wanders along the various edges, searching for a route across the field or highway.

On the third night, our badger decides he must cross this strange strip of black, hard grass (asphalt) to try his luck on the other side. As he approaches the shoulder a semi-truck zooms by, followed by two SUVs and a van. Their intense headlights paralyze him momentarily. Soon he makes his move, darting at his fastest past across the two lanes and narrowly avoiding an oncoming car. Our badger scampers over the rocky, debris-strewn shoulder and bounds down into a grassy ditch, where he encounters a wire fence. He hastily digs under the fence to escape the highway noise. Once past the fence he encounters another farm field, freshly planted in corn. He moves onward, across the 150 open acres, despite the lack of cover and uncertain food.

At dawn, the badger spots a mouse. He bounds for it but the mouse outruns him easily. He follows the mouse to a hole and, digging rapidly, manages to capture it. As he swallows his last bite, however, he is spotted by the resident farmer, out on his tractor to finish his planting. The farmer dislikes badgers because their burrows unearth freshly-planted seed-corn, lowering

[1] Some badgers do live in farm fields, though that habitat likely poses more serious threats to a badger's life due to interactions with humans. This fable focuses on habitat that has not been altered by humans.

crop yields. Also, badger holes can cause cows to stumble and occasionally break legs. The farmer pulls out his shotgun, kept on hand for such occasions, and takes aim. Hearing the farmer's approach the badger sprints in the opposite direction, just avoiding a well-placed shot. Once across the open acres our badger burrows into a grassy pasture and beds down.

The badger will need to retain his sharp instincts. The following night his journey will take him across busy railroad tracks, two more roads, several backyards, and a large paved parking lot. Beyond that lies a gravel biking trail, bordered on both sides by unkempt grassy areas and with a small stream weaving through. As habitat it is only a small patch, just like the one he left, yet it is suitable enough in quality. At least for a time it will provide for the badger a decent home.

Will this certain American badger make it to this grassy patch and its meandering stream? Will he successfully cross the unfamiliar and hazardous terrain in his path, avoiding potentially deadly contact with humans? And if he reaches this grassland, will the humans who pass by on foot or bicycle realize how he arrived here? Will they understand the challenges their land uses have placed in his journey? Even more, might they learn what the badger needs in order to thrive and take steps to transform their human-occupied landscapes so that badgers, as well as people, can live in them?

The American Badger is listed by the state of Illinois as one of the mammals in "greatest need of conservation." (Illinois Department of Natural Resources, 2010). It is at risk in part because, as the above tale recounts, it faces in a typical Illinois landscape shortages of good habitats along with grave troubles dispersing from habitat patch to habitat patch. Although fictional, the tale reflects the actual spatial pattern of landscapes in the region. It is also draws upon the movements of a real cougar, tracked as it dispersed in southern California (Forman, 1995). In Illinois and elsewhere, dispersal can be dangerous and often unsuccessful for certain species[2], particularly in landscapes extensively altered by and divided among countless private owners. The dangers and high costs of dispersion are an important cause of declining populations for many species.

If wild species are to survive and thrive, performing their ecological roles and helping sustain fundamental ecological processes, they must somehow do so in our now-fragmented

[2] Some species are able to disperse across fragmented landscapes more easily than others. For some, fragmentation imposes a barrier to dispersal, confining them to the patch, or resulting in eventual death if they do attempt to disperse. See Chapters Two and Three for a more detailed discussion.

landscapes.[3] They will be able to do so, particularly in a world of rapidly changing climates, only if landscapes are suitable for their diverse needs. Whether landscapes meet these needs has come to depend heavily upon the human inhabitants of these landscapes. Will we make room for wildlife, preserving or creating habitats that meet their needs? And will we, in an era of shifting climates, adjust our own activities so that wildlife can move among and take advantage of these habitats? We can do that only through concerted land planning and by crafting new legal tools to implement our plans. Our success may also require, even more, a pronounced shift in our cultural values: in our attitudes toward nature, in our understanding of land and its functioning, and in the ways we understand our place in the scheme of life.

[3] It should be noted that many species that remain in heavily-fragmented landscapes are often tolerant of fragmented conditions and will likely continue to persist. This thesis advocates protecting those species that are less tolerant and likely to be harmed by fragmented conditions, which are often native species.

I. Introduction

No living man will see again the long-grass prairie, where a sea of prairie flowers lapped at the stirrups of the pioneer. We shall do well to find a forty here and there on which the prairie plants can be kept alive as a species.
– Aldo Leopold (1966, p. 265).

According to many scholars, the Earth is currently undergoing a sixth wave of species extinctions (Brown, 2009; Wake & Vredenburg, 2008). Every day, an estimated 100 species—four species every hour—go extinct[4] (Brown, 2009; Kahle, 2009). Unlike previous extinction periods, this present extinction wave is primarily due to the actions of a single species—humans. Whether we realize it or not, we are currently engaged in a one-sided war against biodiversity (Benedict & McMahon, 2006, p. 58). Unsurprisingly, we are winning, though very likely at great cost to ourselves and our descendents.

The Millennium Ecosystem Assessment defines biodiversity as "the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems" (Millennium Ecosystem Assessment, 2005, p. 18). Biodiversity influences, and is influenced by, climate and ecosystem processes. The Assessment emphasizes that biodiversity is directly and irrevocably linked to human wellbeing. Loss of biodiversity can result in ecosystem changes that can in turn impact the wellbeing of the humans that depend on those ecosystems (Millennium Ecosystem Assessment, 2005). According to the Assessment, loss of biodiversity "contribute[s]—directly and indirectly—to worsening health, higher food insecurity, increasing vulnerability, lower material wealth, worsening social relations, and less freedom of choice and action" (Millennium Ecosystem Assessment, 2005, p. 30).

Habitat loss and fragmentation due to land conversion, particularly conversion to urban and agricultural uses, is the dominant (though not sole) cause of species loss (Benedict &

[4] This estimation is for tropical deforestation alone.

McMahon, 2006; Collinge, 1996; Donald & Evans, 2006; Hilty, Lidicker, & Merenlender, 2006). Over the generations, agriculture in its many forms has been the prime force of habitat alteration. In recent decades, suburban and exurban sprawl has come forward as the predominant cause of further landscape transformation (Benedict & McMahon, 2006). Sprawling "development" transforms substantial tracts of prime farmlands and privately-owned forestlands. Between 1982 and 2003, developed land in the U.S. increased in area by 48%, a total land area larger than the state of Illinois (Natural Resources Conservation Service [NRCS], 2007). From 1997 through 2001, one million acres per year were converted to developed uses (Benedict & McMahon, 2006, p. 5). This change, chiefly at the expense of croplands and pasture, greatly exceeds in percentage growth the rate of population increase. From 1982 to 1997, the percentage of urbanized land in the U.S. increased by 47%, while population increased 17% (Benedict & McMahon, 2006, p. 6). White, Morzillo, & Alig (2009) have predicted that developed land in the U.S. will increase by another 51% between 2003 and 2030, or some 22 million hectares. Illinois, they estimate, will see a 27.7% increase in developed land, or a conversion of 374,700 hectares (White et al., 2009, p. 44). Nearly all of this development will take place on privately-owned land.

By converting and dividing landscapes into agricultural and urban uses, we have greatly transformed the habitat value of our landscapes for many species—nearly all species except those that thrive in and around humans. Large expanses of largely unaltered habitat today are rare, particularly in Illinois. Little-altered landscapes exist chiefly in relatively small, isolated pieces, surrounded by human uses. Habitat fragmentation of this type impacts all elements of landscapes, with effects that can permeate even seemingly unused habitat patches. In addition to directly harming individual plants and animals, fragmentation indirectly disrupts communities and impairs ecological processes, often leading to local and regional losses of particular species. In many ways, the effects of fragmentation are also felt by the resident humans.

Climate change is poised to exacerbate the challenges of preserving biodiversity in such fragmented landscapes. According to the IPCC's 2007 report, "[w]arming of the climate system is unequivocal" (Intergovernmental Panel on Climate Change, 2007). Current CO_2 concentrations in the atmosphere are higher than any level recorded over the past 650,000 years and continue to increase annually (National Oceanic and Atmospheric Association [NOAA], 2010; Williams, Jackson, & Kutzbach, 2007). In 2009, atmospheric CO_2 concentrations increased by 1.89 ppm to reach a level of approximately 387 ppm, up from approximately 355 ppm around 1990 (NOAA, 2010). Scientists generally agree that increasing levels of greenhouse gases in the atmosphere, including CO_2, will result in warmer climates. Global temperature could increase from 1.4 to 5.8 degrees Celsius (2.52 to 10.44 degrees Fahrenheit) by the end of this century (Williams et al., 2007). Human-caused climate change will also likely take the form of more frequent and severe weather events, such as hurricanes and floods (Karl & Trenberth, 2005; Williams et al., 2007). Certain existing climates may disappear completely and novel climates may develop in their place (Williams et al., 2007). Moreover, these climate changes with their attendant ecological impacts will occur in some degree over the course of this century regardless of mitigation efforts, given that the ill effects of elevated levels of CO_2 in the atmosphere apparently take many years to become manifest (Galatowitsch, Frelich, & Philips-Mao, 2009). In combination, these climate changes are capable of altering significantly the suitability of existing habitats to meet the needs of now-native plant and animal species. To survive, many species in many locations will need to seek new habitats, typically, scientists predict, by shifting their habitat ranges northward and to higher elevations (Brooker, Travis, Clark, & Dytham, 2007; Huntley, 2005; Opdam & Wascher, 2004; Parmesan, 2005).

The regional survival of many species seems likely to depend upon their ability to undergo such movements; their ability, that is, to find and move to new, suitable habitat. The current fragmentation of landscapes, however, poses substantial barriers to such adaptive dispersal. Given these realities, both of fragmentation and climate change, we should take measures to protect biodiversity and avert, or at least mitigate, the progress of the extinction crisis.

This thesis considers the challenges of protecting biodiversity in fragmented landscapes such as those that predominate in Illinois—landscapes that are fragmented legally among countless political jurisdictions and among many independent private landowners and fragmented spatially in that mosaics of human activities have left many species with only small, isolated patches of habitat suitable for their particular biological needs. This sizeable topic is considered in *ecological* terms, paying particular attention to the effects of habitat fragmentation and the possible ways of reducing that fragmentation through the use of protected habitat corridors. The issue is considered also in *legal* terms. The thesis looks at current legal tools that jurisdictions are using to promote better patterns of land use, probing their strengths and weaknesses. It goes further to consider alternative tools that could be used, highlighting in particular the need for action at larger spatial scales and thus by levels of government well above the local level—that is, well above the level that typically takes charge of land-use planning. The thesis ends with an examination of the ways in which we might update or more extensively revolutionize the current system of land-use law in the United States to more equally balance landowner rights with their attendant stewardship responsibilities. The reality is, however, that effective legal tools to achieve wildlife goals will encounter (and have already encountered) significant popular resistance due to their conflict with major elements of American culture. For wildlife to gain ground, in sum, wildlife advocates need to focus also and perhaps primarily on *cultural* change. What changes in culture are needed, and what types of concerted actions by engaged citizens might help stimulate such change?

By way of looking ahead, a few comments might be offered about overall conclusions.

First, if we are to provide room for wild species and remedy the effects of fragmentation we shall need to implement a conservation strategy at broad spatial scales, beginning with the landscape scale and moving up to regional and even national scales. A key component of such a strategy will entail establishing networks of corridors that connect habitat patches to form a more expansive, integrated habitat[5] system across otherwise fragmented landscapes. By connecting patches of habitat and effectively expanding the range of habitat for many species to traverse, corridors can alleviate many of the problems caused by fragmentation. When carefully planned, corridors have the potential to improve the ecological integrity of habitat patches, as well as the overarching landscape, benefiting wildlife as well as humans. In particular, corridors may provide a means by which various species can shift their ranges in response to climate change.

Second, in addition to establishing connections among habitat fragments, a landscape-scale conservation strategy must also address human land uses. Land uses surrounding habitat remnants can degrade their ecological functions and thus require control and sound planning. Of course, human land uses are ultimately the cause of much fragmentation and habitat loss. Thus, a sound conservation solution inevitably will need to incorporate humans as well as habitat,

---

[5] The term "habitat" is used broadly throughout this thesis to refer to the range of ecological conditions—spatial and functional, biotic and abiotic—suitable to sustain the biotic processes of a species or group of species. Because this thesis is a broad survey on conservation in fragmented landscapes, as opposed to a narrow study of one particular landscape, many scientific terms are used and applied broadly to encompass the full range of landscapes and plant and animal species that might be included within this paper's scope.

finding places for vital human activities that are less disruptive of wild species and ecological processes.

Conservation organizations such as the Wildlands Project are already undertaking or at least promoting landscape-scale conservation using corridors in the western United States. The work is difficult enough in parts of the West that comprise extensive tracts of federally owned lands. It is considerably more difficult to undertake in Midwestern and Southern regions of country, which lack large federal reserves. States east of the Rocky Mountains tend to be dominated by private land-ownership and intensive land uses. In them, planning for wildlife inevitably requires coordination of land uses over many thousands of tracts of privately owned land. Coordination on that scale, particularly given the inevitable resistance of many land owners, is only possible if undertaken through governmental means, likely with heavy reliance on land-use regulations.

This need for significant legal change is a third major conclusion of this thesis. Current laws and governmental processes are simply insufficient to perform conservation and land-use regulation on scales that promote wildlife needs on large spatial scales—on the scales needed for wild species to respond on their own to major climate shifts.

Like the physical landscape itself, the existing system of land-use law in the United States is fragmented on many levels. Land-use laws, regulations, and other governance mechanisms generally are not based upon the biological composition and ecological functioning of the underlying landscape. Typically they lack coordination and fail to provide for long-term habitat protection. More importantly, they fail to foster stewardship on the part of the landowner. Federal, state, and local land-use governance methods all fall short of providing any basis upon which we could establish and maintain networks of conservation corridors on a landscape scale.

This is not to say that we need to do away with our current system of laws and regulations and start over. Our existing governance methods offer some useful tools and legal concepts that, with updating and coordination, can be used to foster ecological conservation in tandem with needed human land uses. But changes are clearly needed, particularly in land-planning obligations and in the coordination of plans among jurisdictions. As explained below, a combined bottom-up and top-down approach is most likely to facilitate these land-use law updates. Updated laws must require governments at varied levels to prepare comprehensive, ecologically based land-use plans. Laws should also require coordination of these plans across physical, jurisdictional, and political boundaries. Governments can implement their plans using a combination of existing conservation planning tools, including conservation easements, transferable development rights, and conservation zoning. Regulation, likely on both a state and local level, will be necessary to compel stewardship from and to curb land abuses by private landowners. Additionally, we must update the common law concept of harm and expand the public trust doctrine to encompass ecological harm. Such legal changes should require governing bodies, courts included, to affirmatively acknowledge and protect the physical components of ecosystems as well as their functions.

While an updated system of land-use law has the potential to result in many conservation gains over the long-term, we will only achieve widespread conservation efforts and private land stewardship through social and cultural changes in the way we view private property rights and our relationship to the land. This is the fourth major conclusion of this thesis. Land abuses are in large part the result of our cultural orientation to the land. As explained in the final section, we must shift from a social attitude of conquest to one of stewardship, from short-term individual gain to long-term benefits for community and future generations. One way to summarize this needed cultural change is to say that we need to embrace, collectively, a shared land ethic at least roughly similar to the one proposed sixty-five years ago by Aldo Leopold, an ethic rooted in a vision of humans living, in perpetuity, on ecologically healthy land.

A well-grounded cultural orientation toward nature would help us see the steps that need to be taken to promote the land's health and to sustain biodiversity.

•We must preserve high-quality wild lands, pretty much whenever we can and certainly in all landscapes where wild lands are relatively scarce.

•We must stop land uses that seem bad because of their consequences, particularly when we can achieve our goals in less-damaging ways.

•And then we must take action to restore ecological systems, including their biological components.

These steps should aim, overall, at sustaining the land's ecological health—that is, some version of the conservation goal, land health, that Leopold crafted during the final years of his life. Indeed, land health can usefully serve as the thread that connects ecological science, sound moral thought, land-planning, and legal change. Ultimately, our laws must enable us to preserve biodiversity, critical ecosystem functions, and human wellbeing. To craft laws that achieve that lofty goal, however, we need follow the thread, beginning with ecology and moving ahead, step by step.

Before turning to the fundamentals of ecology, a few more words should be said about the main force driving land-cover change and habitat fragmentation today—urbanization in its various forms. These comments also provide background for later chapters, dealing with existing laws and the challenges and consequences of land-use fragmentation.

When considering suburban and exurban sprawl, one may envision vast expanses of asphalt parking lots, strips of big box stores stretched out in one-storey solitude, and single-family homes, flanked by ample front and back yards and collected in subdivisions that are separated from the rest of the community. Scholars have identified several traits that characterize these land uses. Sprawl is primarily characterized by low-density development (Pendall, 1999). Low land values, fragmentation in land ownership (particularly farmland), fragmentation in local government, and intermunicipal competition are all factors that have been found to contribute to low-density development (Pendall 1999). Sprawl is further characterized by segregated land uses, a lack of significant community or city centers, and poor accessibility (Pendall, 1999). Spatial patterns that constitute sprawl include scattered or leap-frog development, commercial strip development, uniform low-density development, and single segregated use development (Ewing, 1997; Pendall 1999, 556).

Sprawl occurs for a variety of reasons, some personal, some political, and some cultural. One explanation for sprawl is simply that demand for it exists, driven by individual choice and capacity. Rising incomes and economic prosperity have given increasing numbers of people the economic means to move farther from dense urban areas into single-family residences on the fringe or beyond (Deal & Shunk, 2004). Individuals affirmatively choose to move to low-density suburbs. A 1991 poll indicated that most respondents preferred single-family residential housing (Morrill, 1991 as cited in Deal & Shunk, 2004, p. 81). Demand for large lot sizes also drives low-density development (Benedict & McMahon, 2006).

Another explanation is that public policy promotes sprawl (Pendall 1999). Haphazard local government planning can result in poorly-structured, disjointed spatial organization of land uses. Local land-use controls, such as low-density zoning and building permit caps also produce sprawl. Furthermore, municipal dependence on local property taxes as a principal revenue source has been found to spur sprawl (Pendall, 1999).

Sprawl is also a result of American cultural traditions. According to Kunstler (1993), pervasive American individualism and individualistic views of property-ownership have fueled sprawl development. As the automobile gained prominence as a form of personal transport, developers began designing communities around roads and cars (Kunstler, 1993). Aiding them was an American conception of private property that placed little value on the public or communal aspects of property (Freyfogle, 2007b). Kunstler has further attributed the spread of sprawl development to Modernism and the Modernist view of architecture and society in functionalist terms (1993). In particular, Le Corbusier and the Bauhaus movement conceived of buildings as utilitarian structures, or machines; this view "did tremendous damage to the physical setting for civilization" (Kunstler, 1993, p. 84). In short, sprawl embodies the American individualistic cultural ethic.

The impacts of sprawl are many and far-reaching, affecting humans, society, wildlife, and entire ecosystems. These impacts do not remain within the bounds of the particular areas being developed; they extend far beyond physical and jurisdictional lines (Benedict & McMahon, 2006). The far-flung single-family homes and businesses that are typical of sprawl development—segregated from one-another by vast webs of roads, each sequestered in its own separate corner of town--have created extensive networks of "noplaces" that are detached from any sense of community and seem hardly worth caring about (Kunstler, 1993). Such a physical environment has reduced the attachment to place that residents might otherwise develop, weakening any sense of community (Deal & Shunk, 2004). Because residents develop little attachment to their community, they have little concern about the health of their local environment (Deal & Shunk, 2004). While individuals may benefit from sprawl development, it imposes high costs on the community and society (Deal & Shunk, 2004).

Because of the often substantial spatial distance between residences and stores, schools and other aspects of the community, these "no-communities" promote dependence on automobiles for every-day activities (Benedict & McMahon, 2006; Deal & Shunk, 2004). Thus, sprawl development makes accessibility more difficult, adds traffic congestion, and increases commute time. Automobile dependence in turn requires massive land-consuming infrastructure, including roads, parking lots, and driveways. The greater the spread of development, the higher the cost of infrastructure (Benedict & McMahon, 2006). Dependence on automobiles and road webs has adverse impacts on neighboring wildlife. Estimates show that, each day in the U.S., over one million animals are killed on roads. (Benedict & McMahon, 2006 , p. 58). Humans are also frequently harmed in wildlife-related in automobile accidents. The automobile dependence that drives and is driven by sprawl thus has serious repercussions for human health and safety of both humans and wildlife.

As it extends into natural areas, sprawl development impacts ecosystems and their resident wildlife. Due to its impacts on natural systems—some visible, some not—sprawl development contributes to a "gathering ecological calamity we have only begun to measure" (Kunstler 1993, p. 60). Sprawl has direct, physical impacts as wildlife habitat is converted to concrete and asphalt. Obviously, this diminishes habitat value and likely kills or harms wildlife in the converted areas. In larger regions, the total amount of useable habitat available for wildlife declines and becomes more fragmented (Benedict & McMahon, 2006). Changes occur also in ecosystem functions. For instance, increased impervious surface area impacts hydrologic flows. Many scholars consider habitat fragmentation "the most serious threat to biological diversity . . . and the primary cause of the present extinction crisis" (Wilcox and Murphy, 1985 as cited in Collinge, 1996, p. 60).

II. The Ecology of Fragmentation

What is habitat fragmentation and how does it occur? How does fragmentation impact species and ecosystems? As discussed in the section above, scholars have determined that habitat fragmentation is a leading cause of species loss. Using concepts drawn primarily from landscape ecology and secondarily from conservation biology, this chapter will discuss fundamental principles that explain how fragmentation alters landscape spatial structure and processes and how a fragmented landscape affects the viability of wildlife and ecosystems.

Healthy, Functioning Ecosystems are Valuable

Any landscape likely consists of several ecosystems, or "assemblages of organisms together with their physical and chemical environments" (Ricklefs, 2007, p. 3). Ecosystems are composed of communities of plant, animal, and insect species, as well as microbes, which are linked through "feeding relationships and other interactions, forming a complex whole" (Ricklefs, 2007, p. 400). Ecosystem types vary based upon the particular physical conditions of the landscape. Climate, soil composition, and hydrology all influence where organisms are able to survive and thrive (Ricklefs, 2007, p. 98-123). Species communities play a substantial role in ecosystem processes, which include energy flows, hydrological processes, and decomposition (Ricklefs, 2007).

Aldo Leopold's concept of "land health"[6] is centered on the roles these interrelationships play in maintaining viable ecosystems (Leopold, 1966). Land health, as Leopold generally conceived it, consists of protecting and maintaining the natural functioning of ecosystems and their capacity for ecological "self-renewal"[7] (Leopold, 1966, p. 221; Newton, 2006, pp. 320-21). Healthy ecosystems perform a variety of crucial functions. They maintain hydrologic flows within a landscape and provide balance to the hydrologic cycle by sustaining infiltration and evapotranspiration rates (Ricklefs, 2007; Ward & Trimble, 2004). By absorbing rainwater into

[6] "Leopold's land heath concept became his culminating vision of enduring prosperity and ecological harmony among humans and the entire community of life. Land health became for Leopold a yardstick for evaluating the ways people lived on land" (Newton, 2006, p. 321).

[7] Because of human interference, many ecosystems are unable to regulate themselves and will require careful management to maintain essential functions, such as proscribed fire regimes. However, in the long term, once we have implemented a system of conservation land use, ecosystems may once again regain the ability to self-regulate. It could be argued that due to human development, they will not be able to regain that ability. This question is debatable and its full scope is beyond the scope of this thesis.

the ground through infiltration, healthy ecosystems provide for groundwater recharge—which can in turn recharge stream flows—and mitigate the impacts of flooding by retaining the water and releasing it slowly over time (Ward & Trimble, 2004). Healthy rivers maintain their banks by establishing equilibrium between channel width and depth and speed of flow and sediment load (Ward & Trimble, 2004). Healthy wetlands retain and purify water, reducing peak flows in streams and rivers and keeping excess nutrients out of surface water supplies (Ward & Trimble, 2004).

Healthy ecosystems also maintain nutrient flows, regulating carbon, nitrogen, and phosphorous cycles, and decomposition rates (Aber & Melillo, 2001; Ricklefs, 2007). Vegetative cover within a healthy ecosystem regulates the amount of solar radiation reaching the ground, affecting surface temperatures (Aber & Melillo, 2001). This function is especially important for maintaining cool water temperatures in streams that support cold-water species of fish and other aquatic life. Healthy ecosystems also experience periods of natural disturbance, such as cycles of fire or flooding, that maintain the viability of many native species that are accustomed to those disturbance patterns and thrive because of them (Aber & Melillo, 2001). Furthermore, ecosystems provide shelter and food for different organisms, depending on vegetation and climate (Ricklefs, 2007).

The increasingly popular literature on "ecosystem services" attempts to place an economic value on the functions that healthy ecosystems perform or at least to explain their benefits for humans. Ecosystem services are sometimes divided into two categories: "benefits used indirectly through structural components derived from ecosystems" and "benefits used indirectly through the dynamic process of ecosystem functions" (Ruhl, Kraft, & Lant, 2007, pp. 28-29). Parsing those broad categories into their components, Costanza et al. (1997) identified seventeen major categories of ecosystem functions or services. Among these are climate regulation, water regulation, pollination, soil formation, and food production. Trees, soil, minerals, and even ecosystems constitute "natural capital" stock that generates the ecosystem services (Costanza et al., 1997, p. 254). Ecosystem services, therefore, "consist of flows of materials, energy, and information from natural capital stocks which combine with manufactured and human capital services to promote human welfare" (Costanza et al., 1997, p. 254).

The rationale for valuing ecosystem services along with goods and services in our human economy is that our human economy fails to incorporate such ecological services and thus fails to give them any value. Under our current system of national income accounting, most elements and functions of ecosystems are valueless[8] (Costanza et al., 1997; Freyfogle, 2007b). Too often, human actions that use up, damage, or destroy natural capital fail to include those costs when determining the value or profit of the actions. The costs of ecological damage are pushed to the side as externalities. Valuing ecosystem services can help incorporate them into accountings of costs and benefits of our actions and thereby improve economic decisions.

Costanza et al. (1997) estimate that the total value of global ecosystem services amounts to between US $16-54 trillion, or an average of US$33 trillion, annually (p. 259). The authors contend that this is a low estimate, with the actual value likely much higher. A 2002 study estimated that establishing a network of global nature reserves would produce $4.4 trillion annually in economic benefits (Balmford et al., 2002). Ingraham and Foster (2008) estimated the value of the ecosystem services provided by the U.S. National Wildlife Refuge System at $26.9 billion per year. Studies such as these indicate that ecosystem services constitute an asset of substantial value—value that our current economic system frequently fails to consider and respect.

By disrupting healthy ecosystem function, land conversion, particularly urbanization, has real and substantial economic costs. Urbanization often increases impervious surface area within a watershed and results in the removal of vegetation along streams and rivers. These spatial changes can result in reduced infiltration and increased rates of surface runoff, which raises peak stream flow levels and increases the frequency and intensity of floods (Ward & Trimble, 2004). Vegetation removal can cause stream bank erosion and warmer water temperatures, harming cold-water aquatic species and species that require high levels of water clarity. Land conversion can also disrupt natural disturbance regimes, which can alter the composition of species communities that are dependent upon disturbance over time (Forman, 1995; Ricklefs, 2007). By adding more nitrogen and phosphorous from backyard, golf course, or agricultural fertilizers, human land uses also alter nutrient flows, often increasing the availability of previously limiting nutrients and leading to algal blooms, eutrophication, and dead zones in estuaries and bays, including the Gulf of Mexico (Ricklefs, 2007). Agricultural land uses can increase erosion, which leads to further sedimentation in streams and rivers and degraded water quality (Ward &

---

[8] Aside from valuable extractable items such as oil, gold, or coal that have proven useful or desirable to humans and are able to be made into commodities.

Trimble, 2004). Human land uses can also introduce invasive species, which can outcompete and displace native ones. By degrading and disrupting ecosystem functions, certain human land uses disrupt the ecosystem services those functions provide. Farmers generally do not consider the impact of nitrogen runoff in annual fertilizer budgets, and subsidies do not take these impacts into account. Developers generally do not consider the cost of increased flood risk that their residential and commercial developments impose within watersheds. Yet their actions contribute to ecosystem disruptions that have tangible and often substantial economic consequences, particularly when calculating the loss in valuable ecosystem services.

Fragmentation: Causes and Characterizations

A landscape is not simply a particular spatial arrangement land or its uses, such as forest, single-family homes, and shopping centers. In addition to physical elements, it is composed of processes and deeply complex interrelationships among species (Fahrig, 2005; Forman, 1995). Because of these diverse interrelationships, Aldo Leopold likened land to a dynamic, living organism (Leopold, 1966; Netwon, 2006). In this sense, the concept of land health assumes heightened logical importance and validity. Health is integral for organisms to function properly and to heal themselves. The same is true of the land. To see how fragmentation impairs ecosystem health, we must first understand the attributes of the land community in spatial and functional terms.

Even when fragmentation occurs on a relatively small spatial scale, its impacts are often extensive, setting off chain-reactions in ecological functioning that can spread wide, spatially and temporally. This subsection begins by parsing the composition of landscape. It explains the spatial pattern of landscape—the arrangement of ecosystem units and land uses—in connection with ecological processes and the necessary relationship between pattern and process. After grounding the discussion of fragmentation in the concepts of scale and process, this subsection discusses how fragmentation occurs and how it disrupts and changes landscape pattern and process.

Viewed on a broad scale, land can be understood as a mosaic—an aggregation of multiple and varied ecosystem types and land uses marked by distinct boundaries and spread over a broad spatial scale[9] (Forman, 1995). The land mosaic is composed of regions, which Forman (1995) has defined as "broad geographical areas" which "often have diffuse boundaries determined by complex physiographic, cultural, economic, political, and climatic factors" (p. 13). Regions, in turn, are composed of multiple landscapes, which are compilations of local ecosystems and land uses across a distinct area and which are characterized by "structure, function, and change" (Forman, 1995, p. 5).

Landscapes are typically described in terms of pattern and process[10] (Fahrig, 2005; Forman, 1995). Pattern refers to landscape structure, particularly spatial heterogeneity, or the number of different habitat or ecosystem types and their spatial arrangement (Fahrig, 2005). Process refers to biotic and abiotic processes and flows within a landscape (Fahrig, 2005; Forman, 1995). Landscape pattern is intertwined with its processes and should be interpreted in the context of process (Fahrig, 2005; Haines-Young, 2005). Due to the reciprocal relationship between pattern and process, we should view landscapes as dynamic "process-response units" instead of static, structural units (Haines-Young, 2005, p. 108).

Scholars have organized landscape processes hierarchically into "discrete scales of interaction" (O'Neill, 2005, p. 23). Not only are processes, or flows, within one landscape hierarchically linked, but flows also link neighboring landscapes and regions (Forman, 1995; Shugart, 2005). Because of these interconnections, change to one element can initiate a feedback loop and result in changes to other, linked elements within a landscape. The feedback loop can also extend into other, neighboring landscapes through links between landscape flows (Forman, 1995). In sum, the concepts of landscape pattern and process, and the relationship between them, highlight the complexity and interconnectedness within and across landscapes and regions. Characterized by dynamic and interrelated spatial organizations and functions, land distinctly resembles a living organism, as Leopold so astutely recognized.

The spatial arrangement of unfragmented landscape is generally a large, contiguous expanse of land, consisting of one or multiple ecosystem types. Fragmentation is one of the

---

[9] A mosaic pattern is one type of spatial heterogeneity. A gradient pattern is a second type, characterized by a progression of ecosystem types that lacks distinct boundaries. (Forman, 1995). This mosaic theory is a fundamental concept of landscape ecology.

[10] Landscape ecology is commonly said to be "the study of the effect of pattern on process" (Fahrig, 2005, p. 3).

main ways such contiguous landscapes are transformed.[11] Fragmentation occurs when contiguous habitat is modified and broken into "smaller parcels," or "habitat patches that vary in size and configuration" (Foreman, 2004, p. 408; Hilty et al., 2006, p. 30). Fragmentation can be natural or human-induced (Hilty et al., 2006). Natural means of fragmentation include natural disturbances such as fire or hurricanes. Species have always had to endure these forms of fragmentation (Hilty et al., 2006). Human-induced fragmentation is relatively recent in biological terms and includes conversion of land for agricultural purposes and urbanization. Human-induced fragmentation occurs more frequently than natural fragmentation, has caused much higher rates of fragmentation than would naturally occur, and is often irreversible because it impedes the ability of natural systems to recover through succession (Hilty et al., 2006). The rest of this paper addresses human-induced fragmentation and its effects.

Depending on its underlying cause, fragmentation can occur in different ways and produce different types and configurations of patches. Habitat conversion can start at the edges and work toward the center of contiguous habitat. Conversion can also bisect the contiguous habitat or start at its center and move outward (Collinge, 1996). Patches can result from human alteration or natural disturbance of a small area within a larger, unchanged landscape; alternatively, they can result from the alteration or disturbance of a large area that leaves smaller pieces of land within that large area physically unchanged (Forman, 1995). Patches can also be created by localized environmental conditions, such as different soil or vegetation types, that cause a small area to differ from the surrounding landscape (Forman, 1995). Patches are surrounded by a matrix—different vegetation types or land uses that constitute the majority of the landscape that surrounds the patch (Hilty et al., 2006; Saunders, Hobbs, & Margules, 1991). The patch-matrix composition typifies the spatial configuration of a fragmented landscape.

The remainder of this paper addresses remnant patches—the remaining, often scattered pieces of original, contiguous habitat—and the landscapes in which they are located. Remnant patches are smaller in size, disconnected from adjacent habitat, and, typically, widely and unevenly distributed across a landscape (Collinge, 1996; Forman, 1998). The matrix surrounding remnant patches typically consists of human agricultural or urban land uses.

---

[11] Other types of land transformation include: perforation, "the process of making holes" in habitat; dissection, "carving up or subdividing an area using equal width lines"; and shrinkage, "decreases in the size" of habitat. All of these methods of transformation result in habitat loss and isolation (Forman, 1995, p. 408).

Is it better to have one large remnant patch or several small patches of quality habitat? The "SLOSS," Single Large Or Several Small, debate poses this question. A large patch might be more ecologically valuable because it could contain multiple habitat types, while small patches are likely to contain only a single habitat type. However, several small patches distributed across a region may together contain more habitat types than a large patch (Forman, 1995). Some scholars assert that a large, contiguous patch maintains higher habitat diversity and produces greater ecological benefits than multiple small patches by providing room to roam through contiguous natural habitat (Forman, 1995; Saunders et al., 1991). Other scholars contend that multiple patches could provide habitat for a larger number of diverse species and could prevent extinction by spreading those species to multiple areas (Hilty et al., 2006). Some scholars challenge the basic notion of establishing reserves and parks, advocating conservation through landscape connectivity and land use change (Adams, 2009). The best practice, given our already highly fragmented and rapidly urbanizing landscape, may be to preserve as many habitat reserves and create as many linkages as possible (Hilty et al., 2006).

The conversion of forest or grassland through urbanization and sprawl development fragments the once-contiguous land, leaving behind smaller, isolated remnant patches within a matrix of suburban or exurban development. Due to the inherent interconnection between the ecological pattern and process, such changes to the spatial configuration, or pattern, of the landscape will also result in changes to the ecosystem processes that characterize the landscape. As explained in the next section, these various changes in combination can result in wide-ranging impacts on land health.

Ecological Impacts of Fragmentation

What happens when humans transform a landscape to create remnant patches of once contiguous, native habitat? "Fragmentation affects nearly all ecological patterns and processes, from genes to ecosystem functions" (Forman, 1995, p. 415). The effects of fragmentation permeate all aspects of the remaining habitat and can ripple throughout an entire landscape or region. Effects differ in type and degree, however, based on the spatial orientation, size, and shape of the remnant patches, as well as the degree of connectivity between patches and the heterogeneity among them (Collinge, 1996; Forman, 1995; Saunders et al., 1991). The following two subsections examine how fragmentation impacts ecological pattern and process and the species that inhabit the fragmented habitat.

When remnant habitat patches are formed, the spatial composition of the habitat within the patches changes. Altered spatial characteristics can in turn alter "abiotic and biotic processes within and between fragments" (Collinge, 1996, p. 69). Fragmentation has implications for soil, water, nutrient flows, and natural disturbance regimes (Collinge, 1996; Forman, 1995). To emphasize, the actual changes in these ecological processes that occur, as well as the magnitude of those changes, differ based on the characteristics of each particular patch and its orientation within a landscape or region (Collinge, 1996). The effects of fragmentation on ecological pattern and process are usefully addressed in three categories: edge effects; impacts due to size and shape of patch; and impacts due to the matrix context surrounding the remnant patch.

Edges occur where natural habitat meets the human-altered landscape that created the patch.[12] Edges are exposed to different environmental conditions than those that prevailed before fragmentation occurred (Collinge, 1996; Hilty et al., 2006; Saunders et al., 1991). These altered edge conditions are generally termed *edge effects*. Edge effects can impact ecological processes and species composition not just within the edge, but well into the interior of the patch. Depending on patch size and the extent to which edge effects permeate the patch, a patch could consist entirely of edge conditions, with no interior habitat conditions remaining (Saunders et al., 1991). Such a change could harm plant and animal species that depend on interior habitat conditions.

Opened to an altered landscape and exposed to environmental conditions unlike those in the interior, the edges of remnant patches undergo substantial changes in microclimate. Edges generally experience increased precipitation and snow load (Hilty et al., 2006). In amount and intensity, the solar radiation reaching the edges of a patch is generally greater than that reaching the interior (Collinge, 1996; Hilty et al., 2006; Saunders et al., 1991). Increased radiation can lead to higher air temperatures, which can in turn result in decreased relative humidity (Collinge, 1996; Saunders et al., 1991). Overall, in the northern hemisphere, southern-facing, windward edges are generally warmer, drier, and wider than north-facing edges (Collinge, 1996). Changes in vegetation at the edge, such as fewer tall trees and more shrubs, along with

[12] This can be referred to as a "hard edge," as opposed to a more natural transition of landscapes, or a "soft edge" (Hilty et al., 2006 41).

other edge conditions, can result in greater fluctuations in overall temperature, with higher daytime air temperatures and lower nighttime temperatures. Like dominos, changes in air temperature can set in motion myriad other ecological changes at the edge. Temperature change can alter nutrient cycles, which can impact soil microorganisms, soil moisture levels, and decomposition rates (Saunders et al., 1991). Changes in nutrient flows can change resource availability, affecting local fauna (Saunders et al., 1991).

Edges also generally experience increased exposure to wind and higher wind velocities (Hilty et al., 2006; Saunders et al., 1991). Increased wind exposure can result in higher tree mortality due to tree fall, which can in turn increase the accumulation of litter, changing the composition of ground habitat, thereby affecting ground-dwelling species (Hilty et al., 2006; Saunders et al., 1991). The roughness of edge vegetation can modify the velocity of the wind. Depending on changes in vegetation, wind velocities can increase or decrease within the edge and throughout the patch (Saunders et al., 1991). Increased exposure to wind can also result in increased deposition of particulate matter at the edge (Saunders et al., 1991).

These microclimatic changes can extend into the interior of the patch, often for distances of several meters or more (Collinge, 1996; Saunders et al., 1991). For instance, changes in wind velocity can extend into the patch up to 100-200 times the height of the outlying vegetation before reaching equilibrium with the air movement conditions surrounding the vegetation within the patch's interior (Saunders et al., 1991, p. 21).

Hydrology within patches can also change, particularly at the edge. If native vegetation is removed or disappears from the edge, edge vegetation will intercept less rainfall and evapotranspiration rates will decrease, altering soil moisture levels as well as the entire hydrologic cycle of the area (Saunders et al., 1991). Patches will likely encounter increased surface runoff from the neighboring matrix areas (Saunders et al., 1991). Due to increased surface runoff in the matrix, peak streamflows during storm events will likely be higher in the patch, and could result in streambank erosion. Increased runoff can also lead to sedimentation of streams (Saunders et al., 1991).

Due to the many changes in ecological processes, the composition of plant communities at the edge also changes. Changes in nutrients, such as the abundance of limiting nutrients, could facilitate invasion by non-native species (Saunders et al., 1991). Studies have found increased numbers of pioneer and xeric[13] plant species in edge habitat (Ranney et al., 1981 as cited in Collinge, 1996). Increased seed dispersal from the edge to the interior could change plant species composition throughout the patch. Weedy species can become more prevalent at edges, as well as throughout a patch, and, as a result, once-common and widespread species may begin to disappear from the patch (Hilty et al., 2006). Disappearance of certain plant species could trigger a host of other ecological changes within the patch, beginning a feedback loop of process changes.

Changed processes at edges can also impact soil conditions. The depth of humus at edges can differ from that inside the patch (Hilty et al., 2006). Soil moisture levels can change. Drier soils can lead to less plant available water, causing wilt and even mortality in some plant species that require high levels of moisture. Altered microclimate conditions can lead to increased freezing and thawing (Saunders et al., 1991). Soil composition can also change with differences in litter, decomposition rates, and moisture levels. Increased wind and reduced vegetative cover can cause an increase in erosion.

Overall, fragmentation compromises the ecological health of remnant patches, beginning with the edge. Changed edge conditions can affect interior conditions, rippling throughout a patch, affecting the plant and animal species within it.

Patch size and shape determine the relative amounts of edge versus interior habitat. The perimeter/core or perimeter/area ratio roughly measures the proportion of edge to interior habitat area (Collinge, 1996; Saunders et al., 1991). High perimeter/core ratios have higher proportions of edge to interior habitat. This ratio can also provide some indication as to the fragment's interaction with the surrounding matrix (Collinge, 1996). Round or square patch shapes have the least amount of edge while long and thin or irregularly-shaped patches have higher proportions of edge and less interior habitat (Hilty et al., 2006). Additionally, small patches often have more edge relative to size and can support fewer species than large patches (Collinge, 1996; Saunders et al., 1991). The size and shape of a particular patch, therefore, could determine the quality of available habitat[14] and the magnitude of change in ecological processes.

Land types and uses in the *matrix* surrounding a remnant habitat patch can also impact ecological conditions and contribute to changes in ecological processes in the patch. "The type,

---

[13] Species whose productivity is limited by water (Ricklefs, 2007, p. 87).

[14] It is important to note that habitat "quality" differs depending on the species in question. Generally, edge habitat is considered poorer "quality" than core habitat, but some species thrive within edge habitat.

intensity, and degree of dissimilarity of habitat types, land uses and human activities adjacent to habitat fragments may markedly influence the flow of nutrients and materials, and the persistence of plant and animal species in the fragments" (Collinge, 1996, p. 68). The matrix surrounding habitat remnants "squeezes, invades, and sometimes wipes out enclosed elements" (Forman, 1995, p. 281). Seeds, particularly seeds of non-native species, can blow into the patch from the matrix, become established at the edge, and permeate the patch as seeds blow farther into the interior (Saunders et al., 1991). In a matrix dominated by agricultural land use, pesticides and fertilizers can blow into the patch, damaging insect species and shifting plant-community composition by altering available nutrients (Collinge, 1996).

Human activities in the matrix can permeate as far as, or farther than, microclimatic changes in the edge (Collinge, 1996). Human land uses in the matrix may alter the hydrology of the patch. Impervious surfaces and agricultural drainage can speed surface runoff away from the matrix, often directing it toward the patch. Standing water could eventually accumulate, shifting soil composition and killing vegetation intolerant of saturated soils (Hilty et al., 2006). Roads can pollute nearby remnant patches through exhaust fumes, litter, and oil and other chemical runoff. They can also cause light and noise pollution that could harm sensitive species and cause the death of many mobile animals in collisions (Hilty et al., 2006). Humans themselves can enter remnant patches and degrade the habitat through recreational activities (Hilty et al., 2006).

To summarize: the biological composition and ecological health of a remnant habitat patch are dependent on many variables—the amount of edge and interior, the patch's spatial composition, human activities, and ultimately the condition of land surrounding the patch. All of these factors impact biotic and abiotic processes within a patch and thus plant and wildlife viability.

Impacts on species

Fragmentation can cause "changes in species composition, community structure, population dynamics, behavior, breeding success, individual fitness" (Donald & Evans, 2006, p. 211). Changes in patch habitat, especially in the edges, can translate into changes in plant and wildlife species composition within the patch (Collinge, 1996; Hilty et al., 2006). Species are affected directly and indirectly by changes in ecological processes that result from fragmentation, many of which are described in the subsection above. Increased wind can hinder the breeding success of birds, for example (Saunders et al., 1991). Certain species are affected more than others, some less than others. Major factors that influence species change include the degree of patch isolation, changes in ecological processes within the patch, the degree of connectivity among patches, and the composition of the surrounding matrix (Collinge, 1996; Hilty et al., 2006; Saunders et al., 1991). The resulting effects on resident species are usefully assessed in terms of four characteristics: *species density, dispersal, species interrelationships, and genetic diversity.*

Species density or abundance within a patch is not constant; it varies over time and is based in part on the types of species found within the patch. The species composition of a patch is often most dense when a patch is first established (Saunders et al., 1991). Mobile species from outside the patch may enter it because their former habitat has been converted and the matrix is inhospitable. This could lead to overcrowding within a patch, as the patch may lack sufficient resources or may simply be too small in size to support all of the species (Saunders et al., 1991). This initial period of high density is often followed by a species relaxation period, in which the number of species in the patch decreases over time (Hilty et al., 2006; Saunders et al., 1991). Changes in ecological processes also result in the local extirpation of species from the patch, further reducing density (Saunders et al., 1991). The earliest losses often occur in species "that depend entirely on native vegetation . . . require large territories and . . . exist at low densities" (Saunders et al., 1991, p. 22). Long-lived species can take longer to disappear simply due to their life spans, at least if patch conditions interfere with reproduction (Saunders et al., 1991). Species loss within a patch is dynamic and on-going and can continue for up to centuries (Saunders et al., 1991). If a patch is functionally connected to others, its species density[15] can shift over time as existing species die off or disperse to neighboring patches and others enter the patch from outside (Hilty et al., 2006). Patch density, therefore, depends on a number of interrelated factors including the species present in the patch, the patch's connections with others in the landscape, the matrix surrounding it, and the ecological changes within the patch.

Response to patch area varies based on species. In a review of studies on species' responses to habitat fragmentation, Debinsky & Holt (2000) determined from a lack of consistency in the results that the relationship between species richness and fragment area is species-specific. Overall, researchers have found that rare species are more sensitive to

[15] The topic of species dispersal between patches is discussed in greater detail in the following chapter.

decreases in habitat size (Collinge, 1996). Because of edge effects, species that are dependent on interior conditions can disappear from edges, and perhaps from entire patches if the patch is too small; edge-dependent species, however, will benefit in smaller patches (Hilty et al., 2006). Small populations are more prone to extirpation due to disease or stochastic events, such as fire or a severe storm (Hilty et al., 2006).

The ability of a particular species to move between and among patches—*dispersal*—is often an important influence on its viability. Again, movement is species-specific, and some species may not disperse far, if at all (Saunders et al., 1991). Species movement is sometimes referred to as "dispersal" and other times as "migration." Dispersal is unidirectional and is generally "permanent movement" away from a patch in which a species resides (Forman, 1995, p. 37). Migration, though often used broadly to mean dispersal, consists of the seasonal, cyclic movement of individuals of a particular species from breeding and nonbreeding ranges (Forman, 1995; Hilty et al., 2006).[16] Connectivity between patches is a key determinant of dispersal. Connectivity is the "ability of organisms to move among separated patches of suitable habitat" and can be viewed on multiple spatial scales (Hilty et al., 2006, p. 50). Connectivity enables new colonists to enter and repopulate a patch, preventing local extirpation of the species by allowing the species to return to a patch. Recolonization also helps to sustain the viability of other species that depend upon the dispersing species for survival. Additionally, dispersal promotes genetic diversity and allows species in degraded habitats to seek out higher quality habitats that may sustain them (Hilty et al., 2006).

The degree of connectivity among remnant patches is based on the type and quality of habitat or land use that dominates the matrix and the ability of a given species to move across that inter-patch space. In some circumstances, the matrix can be a barrier to movement; in others it can allow or even facilitate connectivity (Donald & Evans, 2006; Saunders et al., 1991). The permeability of the boundary between the patch and the matrix can also affect connectivity. For instance, some patches can be sealed off by impassable edge habitat (Collinge, 1996). For obvious reasons, roads are, for many species, effective barriers to dispersal (Collinge, 1996).

Corridors that link isolated habitat patches can enhance connectivity and therefore promote and guide dispersal through an otherwise hostile matrix. The term "corridor" generally implies a "linear landscape element composed of native vegetation which links patches of

[16] The remainder of this paper will deal primarily with dispersal.

similar, native vegetation"[17] (Collinge, 1996, p. 65). The functions, attributes, benefits, and drawbacks of ecological corridors will be discussed at length in the following chapter.

By changing ecological pattern and process, fragmentation can alter *species interrelationships*, shifting the fundamental biotic connections that sustain the patch ecosystem (Collinge, 1996). Patch isolation and edge effects are two primary causes of altered interactions (Collinge, 1996). Ecological changes within a remnant patch can interfere, for instance, with predation and interspecies competition. A patch, as noted, may simply be too small to support certain predators, such as large carnivores (Hilty et al., 2006). Also, generalist predators and exotic species often outcompete specialists and native species (Hilty et al., 2006). Edge-dominant species can move to the patch interior and harm interior-dependent species by interfering in predator-prey relationships and engaging in parasitism (Hilty et al., 2006). Temperature changes can also impact predator-prey relationships (Saunders et al., 1991).

Impacts on one species can often result in a cascade effect, causing a chain-reaction of impacts on multiple species (Hilty et al., 2006; Forman, 1995; Saunders et al., 1991). Pollination is one example of a cascade, where the loss of a pollinator species can result in the loss of plant species that depend upon the pollinator to reproduce. The disappearance of prey species can hurt predators that depend on the lost species for food. With the loss of large carnivores from a patch, small predator populations could increase, called "mesopredator release," thereby reducing prey species populations (Hilty et al., 2006; Noss. Beier, & Shaw, 1998). To cite a further example, increased herbivore populations could lead to extirpation of certain plant species within the patch. This could result in the loss of insects that depend on them for food or shelter and could cause process changes, such as shifts in nutrient flow, increased erosion, changes in sunlight perforation, and other changes that stimulate yet other impacts (Hilty et al., 2006).

Finally, fragmentation effects within a patch can lead to a decrease in the *genetic diversity* of species living within the patch. Loss of genetic diversity is essentially a loss of heterozygosity in a species' genetic make-up. This contraction of genetic material within a species can decrease the species' capability to adapt to changed conditions, particularly in the long term (Forman, 1995; Saunders et al., 1991). Small, isolated populations are especially vulnerable to genetic drift, genetic erosion, and inbreeding (Hilty et al., 2006). Low genetic

[17] Other, more specific definitions of corridor will be explained *infra*.

diversity can contribute to species decline and extinction by hindering adaptation, dispersal, and general species fitness (Hilty et al., 2006). As a population decreases in size, its genetic diversity can decline even further. Reconnecting isolated populations by improving connectivity, as discussed above, can increase genetic diversity and improve species' resilience (Hilty et al., 2006).

These four impacts on species within a remnant patch are all essentially connected. Limited connectivity and low population density can contribute to decreases in genetic diversity. Loss of a species through relaxation in a dense patch can impact species interrelationships by triggering cascading effects throughout the patch ecosystem. These impacts can all be traced back to the ecological changes in pattern and process within a patch due to fragmentation. These ecological changes, in turn (and as explained above), are the products of edge effects, isolation, and matrix interference. As Debinsky and Holt (2000) astutely noted, "Fragmentation effects cascade through the community, modifying interspecific interactions, providing predator or competitive release, altering social relationships and movements of individuals, exacerbating edge effects, modifying nutrient flows, and potentially even affecting the genetic composition of local populations" (p. 353). To mitigate the impacts of fragmentation on species, we must understand the land as a dynamic biotic community and preserve the relationships between its parts and processes.

<u>Fragmentation in a Changing Climate</u>:

Changes in climate conditions will undoubtedly alter ecological functions such as the hydrologic cycle. Warmer air holds more moisture, resulting in increased rates of evapotranspiration as well as more frequent and longer droughts (Karl & Trenberth, 2005). Climate-induced changes will vary depending on location; higher altitudes and northern regions in the U.S. are predicted to warm more than other areas (Committee on Environment and Natural Resources, 2008). Because of the complex relationship between a climate and ecological processes, we cannot predict the impacts of climate change in a particular habitat with complete certainty (McInerny, Travis, & Dytham, 2007). One general impact we *can* predict is that climate change will alter abiotic conditions, which will inevitably disrupt species that depend upon historic conditions.

Specific effects of climate change on species include "altered phenology, population density and community structure" (Brooker et al., 2007, p. 59). One overarching phenological change includes shifts in life-cycle timing (Root & Hughes, 2005). For instance, the onset of spring has been observed to occur 10 to 14 days earlier across temporal latitudes (Committee on Environment and Natural Resources, 2008). This temporal shift can interfere with migration and species interrelationships, such as pollination and predator-prey dynamics. By altering relationships among species, the phenological impacts of climate change can disrupt community structures and ultimately destabilize ecosystem functioning (Parmesan, 2005). Changes in precipitation could cause changes in vegetation, which could in turn change the species composition of an ecosystem (Karl & Trenberth, 2005; Parmesan, 2005). The effects of climate change, therefore, have the potential to permeate entire ecosystems.

Climate change will affect individual species differently. For a large number of species, changing climate conditions will result in harm, particularly for species whose specific climate conditions disappear and which are unable to adapt to new conditions or disperse to more suitable habitat (Williams et al., 2007). The habitat ranges for species may contract or they may lose habitat as a result of changing temperatures, disturbances, extreme climate events, or other climate change effects (Parmesan, 2005; Pimm, 2008). Existing habitat reserves may not contain sufficient habitat heterogeneity on their own to help species adapt to and recover from the impacts of climate change (Galatowitsch et al., 2009; Lovejoy, 2005). Some species, however, will likely benefit from climate change. Warmer temperatures can expand the ranges of some species, such as certain butteflies (Parmesan, 2005). Furthermore, a species' own "biotic processes" could determine its response to climate change and whether it will avoid global extinction (Brooker et al., 2007, p. 64). Overall, we know with very little certainty how climate change will affect individual species. Novel climate conditions and novel species associations could well develop, with consequences we can only roughly estimate (Williams et al., 2007).

Scientists predict that many species will shift their range distributions in an attempt to adapt to changing climate conditions (Brooker et al., 2007; Opdam & Wascher, 2004; Pimm, 2008). Dispersal is likely to be one of the few effective methods of adaptation available for most species,[18] particularly because of the rapid rate of climate change[19] (Huntley, 2005; Lovejoy & Hannah, 2005; McInerny et al., 2007). Studies across many locations indicate that a variety of species have already begun to shift their ranges to areas that were previously too cold (Pimm, 2008). Scientists predict that the direction of most range shifts will be northward and upward in altitude, as species move toward cooler climates in response to warming in their original ranges (Parmesan, 2005; Williams et al., 2007). Studies have shown that range boundaries are moving, on average, 6.1 km northward each decade (Williams et al., 2007). Although range shifting may enable species to move into more amenable climate conditions, it may also carry drawbacks. For instance, shifts to higher altitudes may result in smaller ranges of available habitat (Pimm, 2008). Rapidly dispersing species may function like invasive species in their new range, where remaining species have not yet dispersed (Galatowitsch et al., 2009). The ability to disperse to a new range also varies based on species. Some species are able to disperse rapidly and easily across large areas; other species are much less mobile or are easily stopped by physical barriers (McInerny et al., 2007; Opdam & Wascher, 2004). While range shifting provides a crucial means of adaptation to climate change for some, it is not a viable option for all species.

Fragmentation within a landscape, as explained, generally impedes the ability of species to disperse (McInerny et al., 2007; Opdam & Wascher, 2004). Some species, however, can cross inhospitable matrix habitat and successfully shift their ranges despite fragmentation (Opdam & Wascher, 2004). The functional impact of climate change in a fragmented landscape therefore depends upon species' dispersal abilities (McInerny et al., 2007). For less adept dispersers, landscape connectivity is important, if not essential, for enabling adaptation to climate change. Climate change may amplify the above-described impacts of urbanization and fragmentation. For instance, more frequent disturbances and extreme weather events could further isolate habitat patches (Opdam & Wascher, 2004). On the other hand, climate change in some settings might actually improve connectivity, by enhancing the growth of vegetation between patches, for instance (Opdam & Wascher, 2004). Fragmentation clearly creates a barrier that will impede, if not prevent, adaptation to climate change for many species. Despite

[18] Genetic adaptation on its own will likely occur too slowly to help species successfully adapt (Opdam & Wascher, 2004).

[19] Other periods of climate change have occurred throughout earth's history, but our current wave of change, brought on by human as opposed to natural causes, is two to five times faster than any previous rate of change (McInerny et al., 2007).

the many uncertainties regarding the impacts of climate change, we should act with caution and improve landscape connectivity to facilitate adaptation wherever possible, for as many species as possible.

The way to overcome barriers to range shifts and facilitate adaptation to climate change is to provide species with the room to follow their range boundaries northward and upward. This will require a dynamic conservation plan on the landscape scale (Huntley, 2005; Mawdsley, O'Malley, & Ojima, 2009; Opdam & Wascher, 2004). While the needs of individual species are relevant, we would do better to adapt the landscape to sustain as many species as possible by improving large-scale connectivity (Opdam & Wascher, 2004). Implementing networks of corridors across the landscape, particularly in north-south and elevational gradients, is an essential strategy to facilitate landscape scale connectivity and ultimately species range shifts (Mawdsley et al., 2009; Opdam & Wascher, 2004). Corridors will provide not only a spatial link between habitat patches, but also a temporal link, connecting current and future habitat.

III. Corridors: Pathways to Biodiversity Conservation

We have seen that the land resembles a dynamic organism, supported by interdependent relationships between species, spatial patterns, and ecological processes. Fragmentation alters or severs many of these integral relationships and leads to genetic impoverishment, local loss of resident species[20], and cascading impacts that spread across landscapes. We need a conservation method that facilitates and rebuilds crucial ecological relationships and processes and that makes them, as well as the plant and animal species within a patch, more resilient in the face of ecological stress.

Establishing and revising networks of ecological corridors, linking otherwise separate habitat patches, is a vital and necessary part of a landscape-scale conservation strategy. Corridors improve inter-patch connectivity, which can foster ecological interrelationships. They expand the amount of available habitat by linking isolated habitat patches and sometimes by providing habitat directly. A conservation approach centered on corridors thus can help remedy the problems of fragmentation. It can also facilitate dispersal in response to changed environmental conditions.

What constitutes a corridor, and how do corridors function, physically and ecologically? When and to what extent are they useful for conservation? This chapter presents two basic theories governing species movement in fragmented landscape that form the basis of corridor theory. Building upon those theories, it then discusses corridor types and functions, their benefits and potential drawbacks, and how they might function to enhance connectivity between patches and aid species dispersal in response to changed climate conditions. This chapter also illustrates corridor function by describing a study of black bear dispersal in Florida. Although no single corridor will benefit all species within a particular patch, corridors have proven useful overall for maintaining connectivity in an increasingly fragmented landscape.

Island Biogeography and Metapopulation Theory

Two fundamental theories explain how fragmentation impacts connectivity and species movement between patches. These theories establish basic ecological principles that explain

[20] It is important to keep in mind that fragmentation can lead to the local loss of certain native, resident species within a patch, particularly rare species or those dependent upon specific interior patch conditions. On the other hand, fragmentation can lead to an increase in other species such as edge-dwellers and exotics.

how species might behave in a fragmented landscape and how populations of species within such a landscape can remain viable. The first is the theory of island biogeography, first developed by R.A. MacArthur and E.O. Wilson in 1967. This theory was originally based on actual islands, land masses surrounded by water (MacArthur & Wilson, 1967 as cited in Hilty et al., 2006, p. 50). It has since been applied to habitat patches within a land matrix (Hess & Fischer, 2001). MacArthur and Wilson's study focused on species composition and richness within an island (Hilty et al., 2006). They concluded that the number of species on an island is a function of island size and species migration rates, or degree of isolation (Debinsky & Holt, 2000; Hess & Fischer, 2001). Larger islands have more species than smaller islands because they are likely to have a greater variety of habitats on them. Islands close to the mainland will be more diverse and will have lower local extinction rates due to a steady influx of immigrants than more distant, isolated islands (Hilty et al., 2006).

Some scholars argue that this theory is inadequate to explain species richness in habitat patches on land (Debinsky & Holt, 2000; Forman, 1995). Connections between patches in terrestrial landscapes are more complex than the island-mainland habitat connections (Hilty et al., 2006). In particular, the matrix, unlike an ocean, can function as habitat depending on the land uses dominating it and can sometimes facilitate species dispersal. It also interacts with habitat patches and has many diverse influences within a patch (Hilty et al., 2006).

The second theory is metapopulation dynamics, developed between 1969 and 1970 by Richard Levins. Levins's mathematical model has become the foundation for other, more complex and realistic models.[21] This theory addresses the "connectivity and interchange between spatially distributed populations" (Collinge, 1996, p. 62). A population is a group of individuals of a certain species that exist in one location (Forman, 1995). A metapopulation, therefore, is essentially a collection of spatially separated populations. According to the theory, individuals move from one patch to another, colonizing and re-colonizing different patches (Collinge, 1996; Hess & Fischer, 2001). Populations in some patches may be extirpated, but other patches will host individuals of the species, so that the species will remain present within the wider landscape. Eventually, a patch in which a population has disappeared may "blink on" again if individuals of the species re-colonize it (Hilty et al., 2006, p. 58). The processes of

[21] Levins's equation: $dp/dt=mp(1-p)-ep$ Essentially, this determines the change in the proportion of patches that a species occupies over time (Hilty et al., 2006, p. 57).

metapopulations influence the viability of a species within a region. These processes include the "quantity, quality, and timing" of dispersal, demographic processes, and genetic makeup (Hilty et al., 2006, p. 62). In sum, the metapopulation theory is a useful means by which to explain how a species might use corridors and the impact that corridors may have on populations.

Corridor Form and Function

The term "corridor" has been widely applied to many types of landscape structures that establish some form of connectivity between isolated habitat areas (Hess & Fischer, 2001; Hilty et al., 2006). "Corridor" has been used to refer to a narrow, linear structure containing native vegetation that connects two remnant patches (Beier & Noss, 1998; Collinge, 1996). "Corridor" can also refer to a wide expanse of habitat that functions like a habitat highway to link wildlife reserves (Foreman, 2004). In a third sense, the term can mean an interlaced network of such habitat highways (Foreman, 2004). Finally, "corridor" has been used more narrowly to refer to an individual structure that allows passage across a barrier, such as an underpass tunnel to facilitate road crossing (Hilty et al., 2006). The qualified terms "wildlife corridors," "habitat corridors," and "ecological structures" have also been used in the same general sense as "corridor" (Hilty et al., 2006, p. 90).

Because of its many meanings and manifestations, the term "corridor" has, in the view of some, become "contradictory and confusing" (Hess & Fischer, 2001, p. 196). This confusion can cause uncertainty as to a corridor's purpose and lead to poor design and management (Hess & Fischer, 2001). To prevent such confusion, it helps to define the intended meaning of "corridor" whenever the term is used (Hess & Fischer, 2001). This paper advocates corridors in the form of "landscape linkages," which are, according to Hess and Fischer (2001), the widest form of corridors and most useful in promoting regional connectivity (pp. 200-201). Due to their size and configuration, they are likely to support a wide range of community and ecosystem processes and to enable many species, including plants and small animals, to disperse between corridors consistently for generations (Hess & Fischer, 2001). Because this paper surveys landscapes in general as opposed to a particular landscape, it uses the term "corridor" in its broadest sense. An inclusive definition of "corridor" lends the flexibility needed to consider conservation needs in a wide variety of landscapes.

A corridor displays two main characteristics: structure and function (Hess & Fischer, 2001). Structure relates to the corridor's physical form and depends on its length, width, narrowness, and other physical attributes (Hess & Fischer, 2001). A corridor's function reflects whether plants or animals are able to move through the physical, spatial structure of the corridor. The functional qualities of a corridor are generally estimated using the theories of island biogeography and metapopulation dynamics (Hess & Fischer, 2001). When properly designed, both structure and function vary depending on the connectivity aims for the corridor—whether it is used to promote dispersal of select species or to enhance overall ecosystem processes, for instance. A corridor can also serve multiple functions, as will be explained below, even when designed chiefly to perform one (Hess & Fischer, 2001).

A corridor may appear to provide connectivity *structurally*, but may fail to provide *functional* connectivity (Hess & Fischer, 2001). Though a corridor may physically link two habitat patches, plants and animals may not travel through it if it is filled with nonnative species or is subject to human disturbance. Some species may even travel through the matrix instead of the corridor (Donald & Evans, 2006). Nonetheless, functional connectivity in a corridor depends in large part on its structural attributes. The vegetation and climatic conditions within a corridor, as well as the conditions of the surrounding matrix, can impact the corridor's suitability for use (Collinge, 1996; Hilty et al., 2006; Lindenmayer & Fischer, 2006). A corridor must therefore provide both structural and functional connectivity in order to facilitate species dispersal. The connection between corridor structure and function is another manifestation of the relationship between landscape pattern and process.

Scholars have identified five main types of corridor function: *habitat, conduit, filter/barrier, source, and sink* (Forman, 1995; Hess & Fischer, 2001; Hilty et al., 2006). Depending on its spatial structure, the composition of the surrounding matrix, and the species that dwell within the patches, among other factors, a particular corridor may perform a combination of these functions or may serve only one. We can also consider these functions in terms of the impact on ecological processes and the impact on species movement within and across the corridor. A corridor may assume one function with respect to process but another with respect to species movement.

A corridor functions as *habitat* when it contains "an appropriate combination of resources and environmental conditions for survival and reproduction of the species" (Hess & Fischer, 2001, p. 197). The wider the corridor, the more likely it will include interior conditions and provide habitat for interior-dwelling species. Narrow corridors are more likely to provide habitat only for edge species (Forman, 1995; Hess & Fischer, 2001).

Corridors that facilitate the movement of species but do not necessarily provide sufficient resources for survival and reproduction function as *conduits* (Hess & Fischer, 2001). Corridor length and other structural characteristics determine a corridor's conduit function (Forman, 1995). Animal species generally prefer short corridors that are relatively straight, have few entrances and exits, and are not frequently crossed by streams or roads (Forman, 1995). Herbaceous plant species with heavy seeds would not be able to disperse over long corridors (Wehling & Diekmann, 2009).

Filter and *barrier* corridors limit movement through the corridor to different degrees. Corridors that act as filters have some permeability but slow or block the movement of certain moving objects (Forman, 1995; Hess & Fischer, 2001). They can also filter out certain species (Hess & Fischer, 2001). Corridors that function poorly as conduits due to conditions within the corridor could function as filters; for instance, a corridor that is bisected by a large river could impede the movement of species that are unable to cross the river. Corridors that act as barriers completely block movement (Hess & Fischer, 2001).

A corridor may also function as a *source* or as a *sink*. Within a source corridor, survival exceeds mortality, and individuals of various species spread out from the corridor into the surrounding matrix (Hess & Fischer, 2001). A corridor may function as a source of plant seeds, nutrients, and bears (Forman, 1995). Source corridors successfully function as conduits or habitat to facilitate species survival. In a sink corridor, mortality rates exceed the survival rates of the species that enter it; in other words, more species enter the corridor than emerge from it (Hess & Fischer, 2001). Individual animals may be killed within the corridor due to high predation rates, human interference, or impassible streams that cross the corridor landscape (Forman, 1995).

A corridor's function is inherently species-specific (Beier & Noss, 1998; Collinge, 1996; Gilbert-Norton, Wilson, Stevens, & Beard, 2010; Hess & Fischer, 2001; Hilty et al., 2006). A corridor that functions as habitat for one species may serve merely as a conduit for another and may be unusable for a third (Hess & Fischer, 2001; Hilty et al., 2006). The effects of a corridor on various animal species will depend on the species' "foraging patterns, body size, home range size, degree of dietary specialization, mobility and social behavior" (Collinge, 1996, p. 65). Animals can and do avoid a corridor that does not seem to meet their habitat needs (Collinge, 1996). From a meta-analytic review of multiple corridor studies, Gilbert-Norton et al. (2010) concluded, as expected, that a corridor may be used by some species but not others. Some species are highly mobile and easily disperse across matrix habitat, while others are limited to short-distance movement. Dispersal is also frequently biased based on sex, age, or genetics of individuals within a species (Hilty et al., 2006). Young individuals, male mammals, and female birds are in general most likely to disperse (Hilty et al., 2006). In plants, seed dispersal can be limited by seed type and how seeds are moved. A study by Wehling & Diekmann (2009) in northwestern Germany showed that herbaceous forest plants with heavy seeds are unable to disperse great distances.[22] The study results showed that hedgerows beyond patch boundaries were predominantly populated by plant species with seeds dispersed by wind or animals (Wehling & Diekmann, 2009). In contrast, because of their high mobility, birds are less likely to use corridors (Beier & Noss, 1998; Gilbert-Norton et al, 2010).

Because individual species and structure play key roles in determining a corridor's function and effectiveness, both are important factors to consider when planning and establishing goals for corridors. Corridors, of course, can be unplanned; as such, they can be stretches of functional habitat that are primarily used for non-corridor purposes, such as hedgerows and railroad rights of way (Hilty et al., 2006). In the case of planned corridors, the planning process should typically aim to maximize its functional, and ultimately ecological, benefits. Spatial scale is a key design element (Hilty et al., 2006): Will the corridor provide connectivity on a local, regional, or even broader scale? Spatial scale can impact dispersal, as explained above, and can also impact the goals for the corridor. Potential goals include facilitating daily movement, seasonal migrations, one-way dispersal, or long-term viability through improved connectivity (Hilty et al., 2006). Given the species-specific responses to corridors, Hilty et al. (2006) recommends that corridor designers try to benefit entire biotic communities (p. 100). This thesis proposes that in addition to protecting as many species as possible, we should plan corridors to promote healthy ecological function across landscapes.

[22] Because of their long lives and limited dispersal, herbaceous forest plants are vulnerable to delayed extinction many years after fragmentation initially occurs (Wehling & Diekmann, 2009).

An Illustration: The Ocala-Osceola Corridor

The corridor linking Ocala and Osceola national forests in Florida provides an example of a functional corridor that links two large habitat reserves and performs multiple ecological functions.[23] Ocala and Osceola national forests are separate reserves, situated within a matrix characterized by public and private land ownership, urban development, and roads (Dixon et al., 2006). Each forest houses its own independent population of black bear, and a corridor of relatively open habitat stretches between the forest "patches" across a human-dominated matrix. Dixon et al. (2006) used genetic sampling from hair snags to identify the origin of bears using the corridor. Based on the samples, the researchers found that movement within the corridor was primarily in one direction, from Ocala toward Osceola.[24] This finding indicates that Ocala functions as a source population of bears for Osceola. The study results also indicated sex-biased dispersal, with more male bears found within the corridor as well as farther into the corridor. This finding correlates with other studies finding that dispersal is generally gender-biased. The researchers found bears of mixed genotypes, with parents from both Ocala and Osceola. Based on this finding, they concluded that the corridor is functional and likely allowed previously discrete populations to interbreed.[25] Finally, the researchers found that some bears were likely living for extended periods within the corridor.

The Ocala-Osceola corridor study revealed that the corridor performs multiple functions. It serves a conduit, chiefly for bears traveling from Ocala to Osceola. It also functions as habitat at least for a few individuals. Because a road crosses through the corridor, making passage possible for a limited number of bears, that portion of the corridor, at least, acts as a filter, preventing many bears from making it all the way through. The road may function as a buffer for other species, completely preventing any individuals from crossing. Finally, the unidirectional dispersal of bears from Ocala into Osceola indicates that the corridor likely functions to provide a source of bears for Osceola. Additionally, the corridor illustrates the

[23] This study also illustrates what appears to be the common method of studying corridors—focusing on a single species to evaluate corridor functionality. A collection of broader, multi-species studies will be useful to determine the full range of species that may benefit from specific corridors.

[24] Dixon et al. (2006) acknowledged that this one-dimensional movement may the result of higher bear populations in Ocala due to a hunting ban.

[25] The researchers did note that several Ocala bears had been translocated to Osceola. They were able to rule out all bears as the potential parents of the mixed-origin bears except for one. As to that one, the researchers determined that it was highly unlikely that the one translocated bear was the parent of both mixed origin bears (Dixon et al., 2006). This suggests that at least one of the mixed-origin bears was parented by a bear that dispersed from Ocala.

theory of metapopulation dynamics, providing an example of one population mingling with another, likely enhancing its viability. The road presents a significant problem for successful movement across the full corridor and could threaten the corridor's functional connectivity. Overall, this study illustrates how a real corridor can assume multiple functions for one species. The corridor's functions would likely be different for other species.

Benefits and Detriments of Establishing Corridors for Species and Ecosystems

Corridors offer many benefits for the species within the remnant patches they link and for the ecological functioning of the landscape in which they reside. Spatially, corridors can increase the amount of available habitat (Dixon et al., 2006; Hilty et al., 2006). Functionally, by facilitating connectivity, corridors improve genetic diversity, facilitate or maintain species interrelationships, and enhance species persistence and population viability (Dixon et al., 2006; Hilty et al., 2006). They can also help animals avoid predation and harm that they might otherwise encounter crossing the matrix (Hilty et al., 2006). These benefits can ultimately lead to increased biodiversity and improved ecosystem function (Benedict & McMahon, 2006; Hilty et al., 2006).

In a review of multiple corridor studies, Beier and Noss (1998) determined that, while there is no answer to the question "do corridors provide connectivity," corridors do facilitate connectivity to some degree and provide some benefits to animals "in real landscapes" (pp. 1248-49). Gilbert-Norton et al. (2010) conducted a meta-analysis review of studies involving replicated control and corridor conditions, which were more scientifically sound than those reviewed by Beier and Noss. The review indicted that movement was generally greater between patches connected by corridors that between unconnected patches.[26] Their data also showed that, in general, invertebrates and plants benefit from corridors. Both reviews define corridor broadly as a linear linkage between two patches situated within dissimilar matrix (Beier & Noss, 1998; Gilbert-Norton et al., 2010). Both of these reviews of multiple corridor studies, separated by more than ten years, conclude that, overall, corridors do provide some degree of connectivity and ecological benefit.

[26] Twenty-three percent of the studies, however, indicated that corridors were actually less effective at facilitating dispersal than the matrix. The researchers conclude that this disparity is attributable to the species-specific nature of corridors.

Corridors can also provide benefits to humans. They offer recreational opportunities as well as aesthetically-pleasing living environments (Hess & Fischer, 2001; Hilty et al., 2006). By maintaining healthy ecosystems, corridors also help to ensure the continued receipt of beneficial ecosystem services. Corridors can aid in improving pollination.[27] Around urban areas, corridors can help to contain sprawl, sustain hydrologic flows, moderate flows of pollutants and nutrients into waterways, and reduce erosion (Hilty et al., 2006).

Some scientists assert that corridors in some circumstances may be detrimental to particular species or entire ecosystems. Critics of corridors point to gaps in the empirical evidence supporting corridors and highlight the many uncertainties surrounding how corridors function[28] (Beier & Noss, 1998; Collinge, 1996; Collingham & Huntley, 2000; Donald & Evans, 2006). Some critics anticipate that corridors may introduce or transport between patches ecological problems that would otherwise be contained within a single location. In a relatively early study, Simberloff and Cox (1987) raised the possibility that corridors could facilitate the spread of "contagious and catastrophic effects" such as fire, predators, and disease (p. 66). Corridors may increase an animal's risk of exposure to humans or predators and spread "weedy" or invasive species into habitat patches (Donald & Evans, 2006; Simberloff & Cox, 1987). While promoting genetic diversity is generally viewed as a benefit of corridors, it may be detrimental when it is ecologically valuable to preserve local genetic variations (Simberloff & Cox, 1987). Due to the uncertainties and potential risks associated with corridors, critics argue that the costs of implementing corridors in some settings might outweigh their benefits (Donald & Evans, 2006; Simberloff & Cox, 1987).

These criticisms rightfully warn us to implement corridors with caution. Because of the many uncertainties regarding benefits and detriments, we should anticipate all possible effects and plan thoroughly. The criticisms also highlight the need for further and more thorough study of corridor design and function. A greater number of more comprehensive empirical studies, considering multiple species over longer time periods, will substantially improve our understanding. Current inadequacies in the number and quality of studies generally arise from

[27] Hilty et al. (2006) has recounted a study of coffee plantations in Costa Rica in which plantations within 1 km of corridors had 20% higher yields than farther plantations due to improved pollination (Ricketts et al. 2004 as cited in Hilty et al., 2006, p. 115,).

[28] According to Donald and Evans (2006), reviews of corridor studies provide little evidence that corridors increase the functional connectivity between patches. Collingham & Huntley (2000) have stated that corridors are of questionable value for "sessile" or "sedentary" organisms (p. 1250).

the species-specific nature of corridor function and the challenges of conducting studies. Studies are financially and logistically challenging and particularly difficult to replicate (Beier & Noss, 1998).

The potentially detrimental "side-effects" of corridors that Simberloff and Cox (1987) highlight could occur even without corridors. A fire in one patch could likely spread through a permeable matrix to the next patch. A disease could easily travel in the same manner. Furthermore, if corridors establish healthy ecosystems that can regenerate and heal themselves, habitat patches connected by corridors will likely be better able to fend off catastrophes and recover from them. Ultimately, proper corridor design and management can likely fend off many of these potential problems.

These various criticisms give cause to evaluate the costs and benefits of a proposed corridor to determine whether it will provide value overall or if another conservation method would provide greater benefits for the cost.[29] A cost-benefit analysis, however, may not a reliable assessment. The analysis may overlook benefits, both long term and short term, and ignore ecological costs to the landscape if a corridor is not implemented. Any conservation measure will come with costs, possibly be high. But when we consider the ecological returns on the investment—the enhanced and sustained ecosystem services, the increased biodiversity—the benefits will likely win out. This is not to say we should ignore the costs of corridors. It is worthwhile to estimate costs and benefits as one factor when assessing corridor options. Corridors will not always provide the best solution, but their costs should not be determinative.

Where corridors may be impractical, ineffective or detrimental, other options might be more effective for achieving conservation goals. In some circumstances, acquiring patches of high conservation value may be more useful than creating corridors of questionable value (Beier & Noss, 1998, p. 1250). Collingham and Huntley (2000) have stated that the optimal size for a corridor to be functional is unclear. Because of such uncertainties, they advocate using a mix of corridors and smaller "stepping stone" patches. Collinge (1996) advocates the use of closely clustered stepping stone patches instead of corridors. Donald and Evans (2006) propose that improving the condition of the matrix would provide sufficient connectivity and that using the matrix for connectivity may eliminate some of the problems associated with corridors, include the spread of invasive species. These alternatives provide less intensive methods for improving

[29] This paper counters this cost-benefit argument in the following subsection.

connectivity that may be effective in some circumstances and may avoid some of the potential problems associated with corridors.

With respect to facilitating adaptation to climate change, corridors may not always be the most feasible or efficient option because they will require a substantial amount of time to plan and implement, especially in fragmented, urbanized landscapes (Galatowitsch et al., 2009). Their implementation may lag behind changes in climate and range boundaries. Because corridors alone may not meet adaptation needs, we should also aim to increase permeability across the matrix (Mawdsley et al., 2009). Increased permeability may actually facilitate adaptation for some species that would not benefit from corridors (Mawdsley et al., 2009). The combination of these two methods of improving permeability will provide the best opportunity for adaptation to the largest number of species. Essentially, corridors and increased permeability will give nature the space to shift and respond to climate change as conditions demand, enabling species to adapt at their own, different rates.

Although the true impacts of corridors may remain elusive, they are currently the best option available to improve the physical and functional condition of remnant habitat patches in an increasingly urbanized, fragmented landscape. In our already heavily fragmented landscape, corridors are essentially the only remaining option we have to restore some semblance of connectivity and healthy ecological function to isolated habitat patches (Noss et al., 1998). While corridors may not always be the best response to fragmentation, inter-patch connectivity is important to ecological health. Studies show that connectivity is a key element to species survival, especially when faced with an uncertain climate (Lovejoy, 2005; Mawdsley et al., 2009). Establishing corridors is better than doing nothing, as long as we approach planning and implementation intelligently and cautiously.

We can summarize the main conclusions as follows: By enhancing the spatial composition of remnant habitat patches, corridors can enhance the ecological processes within the patch, benefiting species communities. Those benefits can extend across the landscape through networks of linked patches and corridors. By maintaining crucial ecosystem processes and improving connectivity, corridors can promote species viability and protect biodiversity across a landscape. They can also facilitate adaptation in response to climate change. Despite their theoretical benefits, much uncertainty exists regarding the actual function and utility of corridors. Furthermore, the ecological impacts of corridors may not all be positive. For instance, corridors may facilitate the spread of disease, fire, or invasive species that can harm ecosystems. While these potential problems inspire a cautious approach to their implementation, corridors are one of the best options available to resuscitate healthy ecosystem function on a landscape scale. They provide us with an ecological insurance policy to help ensure the preservation of species and ecosystem functions. Overall, corridors can help the land and its resident plant and animal species better respond to climate- and human-induced stresses.

IV. Conservation in a Fragmented Landscape: What Strategies and Goals are Needed to Establish Healthy Ecological Systems on a Landscape Scale?

The previous chapters explained the importance of the abundant and complex interrelationships among the elements of an ecosystem, from plants to animals to processes to humans. Ecological pattern and process are inseparably linked; we cannot alter one element of an ecosystem with impacting other elements as well. Maintaining ecological interrelationships will be a critical determinant of land health within fragmented landscapes. What can we do to maintain or restore ecological interrelationships in fragmented landscapes containing increasingly splintered and isolated habitat? To answer simply: We need an array of actions (including new planning requirements and laws) that, in tandem, orchestrate our land uses so as to protect and enhance biodiversity while sustaining basic ecologically functioning. Underlying these many actions, giving them guidance and traction, must be an ecologically grounded, morally mature land ethic[30], practically defined.

Corridors in the Privately-Owned Landscape

Collectively, the previous chapters have delineated that fragmentation due to human land conversion triggers changes in both the spatial configuration and ecological processes of the affected landscape, as discussed above. These changes can be far-reaching, rippling throughout a region's ecological infrastructure. Within a remnant patch, edge effects, altered interior conditions, and isolation impact the plant and animal species that inhabit it. These changes benefit some species, most often edge-dwellers and "weedy" species, but harm many others. Fragmentation therefore presents a major challenge for conservation of landscapes, ecological processes, and species.

This challenge is growing as development expands ever farther across our landscapes. The rate of land conversion, particularly in the form of urbanization, is projected to increase over the next decades (White et al., 2009), resulting in a greater abundance of roads, buildings, and other urban land uses and inevitably leading to more fragmentation and increasingly isolated

[30] To Aldo Leopold, a land ethic was " a culturally agreed upon, cooperatively practiced idea that there was a moral right and wrong inland use, reaching beyond individual economic profit" (Newton, 2006, p. 149).

habitat remnants. With expanding fragmentation, a higher proportion of habitat will be exposed to increasingly hostile matrix conditions. For instance, higher runoff from growing urban areas will likely alter patch hydrological processes, increasing amounts of chemical fertilizers may destabilize nutrient flows and plant community structures, and a growing network of busy roads will likely result in higher roadkill rates and create more barriers between remnant patches. Additionally, expanding fragmentation will lead to higher species vulnerability to climate change.

What can we do to stop and even reverse the spread of habitat fragmentation? What can we do to maintain the viability of species and ecosystem processes already threatened by fragmentation? Initially, we must determine the appropriate spatial and temporal scales on which to pursue conservation. For reasons apparent in the chapters above, attempting to conserve on a patch-by-patch basis *only* would be largely ineffective and impractical. Individual patches are often besieged by edge effects and impacted by matrix conditions that impair vital connectivity and adversely impact ecological processes within the patch. Additionally, patches may often be too small on their own to support certain species, particularly large predators. Any potential solution will need to address both habitat patches and matrix conditions on the landscape scale. Faced with the uncertainties of climate change, we should aim for long-term viability for both species and ecosystem processes by linking current and future habitat.

To accomplish broad, landscape-scale conservation, we necessarily must establish networks of large linkages and ecological corridors.[31] Connectivity between habitat patches is likely to be a crucial component of any long-term conservation strategy, especially because many individual habitat patches are incapable of sustaining viable species populations as a single unit. Studies have shown that even Yellowstone National Park may not even be large enough to maintain species persistence long-term (Adams, 2009). In heavily-fragmented landscapes where it would not be feasible to establish additional, large reserves, landscape linkages and corridors are likely one of the only remaining options for establishing landscape-scale connectivity and long-term land health. The Wildlands Network has adopted this landscape- and regional scale approach to conservation. The organization aims to link large wildlife reserves

[31] To reiterate, by landscape linkage, this paper refers to large, wide-ranging corridors as described in the section above (Hess & Fischer, 2001). This paper uses "ecological corridor" broadly to encompass any strip of land that functionally connects two or more habitat patches, of any length, shape, or size. As stated in chapter 3, this paper contends that the benefits of corridors will generally outweigh their potential drawbacks.

through landscape linkages and corridors to establish habitat networks across North America (Wildlands Network, 2009). Such linkages are integral for "rewilding" the landscape, or maintaining viable populations of large carnivores, like wolves, that can establish and sustain ecological integrity (Foreman, 2004). Connecting ecosystems through networks of linkages and corridors will better enable populations and communities to adapt to ecological change, help restore natural disturbance regimes, and generally support crucial ecosystem processes (Foreman, 2004). Linkages are also necessary to foster self-sustaining ecosystems, Leopold's key characteristic of land health (Foreman, 2004).

Implementing a regional-scale network of habitat reserves connected by wide landscape linkages is feasible in the western United States, where a large amount of land is federally-owned and large expanses of contiguous habitat are protected in reserves like Rocky Mountain and Yellowstone national parks. In the southern, central, and midwestern regions of the U.S., however, the majority of land is privately owned and much of it is already converted to urban and agricultural uses (NRCS, 2007). According to the NRCS, seventy-one percent of the contiguous U.S. is non-federal, rural land (2007). Because of the dense populations and lack of unconverted natural areas in these regions, Foreman (2004) has claimed that establishing regional and landscape-scale linkages is "unlikely" and essentially disregards them, leaving the entire central and southern parts of the country out of his "national" rewilding plan. The Wildlands Network also conspicuously ignores these regions of the country. Their linkage map leaves the Midwest and the south blank, as if they are an ecological "no man's land" (Wildlands Network, 2009).

Certainly landscapes dominated by private land-ownership, intensive land uses, and severe fragmentation present a more substantial challenge than those in which large tracts of habitat are readily available, but is this a reason to deem these regions unsuited for landscape-scale conservation? These regions make up roughly two-thirds of the contiguous United States. Is it wise to condemn such a large portion of our nation to fragmentation and ecological degradation? This paper argues it is not wise, nor is such a concession necessary. The reality is that large portions of the U.S. are privately-owned, largely converted, and highly fragmented. We must address this reality and discover ways to implement conservation networks in these regions.

While landscape-scale conservation using linkages and corridors in such landscapes presents a challenge, it is by no means impossible. Conservation efforts may achieve success through coordinated planning based on the ecological composition of the landscape and an updated system of land-use laws and regulations. Landscape-scale conservation efforts in these regions are necessary if for no other reason than the importance of the ecosystem services that healthy natural landscapes provide. These regions of the U.S. depend heavily on ecosystem services to provide ample supplies of clean drinking water, reduce flooding, maintain nutrient flows, and sustain fertile soils. Such services are especially important in the Midwestern farmbelt, which relies upon soil, climate, and hydrological processes. People also value nature and want natural areas near their homes for aesthetic and recreational reasons. Furthermore, broad conservation networks may help prevent the eventual global extinction of native plant and animal species. Increased habitat area and increased connectivity may ultimately help to improve ecosystem functions.

There is hope for creating ecological networks in the private landscape. These regions do contain some reserves, including state parks and recreation areas. Missouri houses the Ozarks; Illinois has the Shawnee National Forest and some larger state parks, and Wisconsin is home to many protected natural areas. Foundational core areas, therefore, do exist here, though they may be small and most likely degraded. Private lands will ultimately play a large role in any landscape-scale conservation plan in the non-western United States, however. For example, riparian landowners may be called upon to convert their river front land to a vegetated buffer strip, and agricultural landowners may be required to provide wider and more abundant hedgerows. Private landowners may also need to alter their land uses to improve ecological conditions in and around corridors and core areas to improve overall matrix permeability.

Conservation on private lands is already taking place in the form of conservation easements with the help of land trusts and other conservation organizations. The organization Chicago Wilderness is implementing a green infrastructure vision for the Chicago area through restoration and monitoring efforts (Chicago Wilderness, 2010). Michigan Wildlink is a non-profit organization that works with private landowners to establish corridors across private lands in the northwestern region of Michigan's lower peninsula using conservation easements (Conservation Resource Alliance, 2010). These organizations show that it is possible to take at least initial steps in establishing networks of conservation corridors within landscapes dominated by private land ownership. With adequate planning, these efforts can be applied and extended on the landscape scale as well as the local scale.

Through coordinated efforts and an understanding of the ecology of fragmentation, we may be able to slow fragmentation, stop it completely, or at least require it to proceed in a less destructive manner. We need an overall plan to implement a network of landscape linkages and corridors across privately owned lands. This plan should provide for intergovernmental efforts on multiple scales, especially the landscape-scale, because conservation on that scale is sorely lacking. The plan should focus on establishing and maintaining the ecological health of entire landscapes and regions. This section sets forth three fundamental goals for land conservation in a land mosaic dominated by private land ownership. It then sketches out a strategy for implementing those three goals.

Land Health as an Overall Goal

In describing the land's ecological processes, Aldo Leopold explained "Land, then, is not merely soil; it is a fountain of energy flowing through a circuit of soils, plants, and animals. . . . When a change occurs in one part of the circuit, many other parts must adjust themselves to it" (1966, pp. 253-254). Leopold's conception of land in terms of energy flow within a circuit encompasses the many interconnections between elements of the landscape, discussed in the sections above. Ecological pattern and process are interrelated, and species depend upon one another as well as upon ecosystem processes. A disruption in any one of these interrelationships can cause a cascade effect, rippling throughout other ecological processes and impacting other species. Because the land's ecological functions consist of such complex, intricate relationships between its various parts and processes, Leopold appropriately analogized the land to an organism. Leopold "considered the land's healthful physical condition in terms of its functioning" (Newton, 2006, p. 321). The health of the land organism, therefore, depends on the ecological interrelationships that make up the organism. Sustained, dynamic interrelationships between ecosystem functions lead to healthy processes, which lead to a healthy land organism.

Leopold's concept of land health encompasses all ecological aspects of a landscape. Land health consists of "the cooperation of [land's] interdependent parts: soil, water, plants, animals, and people" (Newton, 2006, p. 322). While all ecological elements and functions are integral to the concept, this paper focuses only on the biodiversity aspect of land health. We cannot exclude the other components of land health from consideration, and they are implicitly included in future references of the concept, but these other aspects of land health—hydrological cycles, soil processes, fertility cycles—are outside the scope of this study. The purpose here is to highlight the role of biodiversity in achieving overall land health and the role of healthy ecosystem function in maintaining biodiversity.

Land health should be the overall goal for a landscape-scale conservation plan based on a network of corridors and linkages. Leopold understood that "[t]he goal of conservation, [ ] focused as it should be on the whole rather than the parts, was appropriately considered in terms of the health of the land community, or land health" (Newton, 2006, p. 320). Land health is a useful and appropriate goal largely because it necessarily encompasses the health of all parts of an ecosystem and is applicable to broad spatial scales. Maintaining land health means maintaining ecological processes, which in turn is related to spatial pattern, or structure. Structure health is where corridors can be effective. A network of linkages and corridors should establish and maintain long-term land health on local, landscape, regional, and even larger spatial scales. As mentioned in the sections above, corridors are akin to the as vascular system of the land organism. They transport materials vital to ecological health, including species and seeds. Like veins and arteries, corridors and linkages support the functions of all elements of the landscape. They are necessary to restore a land organism suffering from illnesses induced by fragmentation to health.

The goal of land health is especially relevant because land health ultimately impacts human health. Indeed, "the closest link you have with your environment may be the bloodstream that runs through your body" (Ward, 2005, p. 66). The ecological functions of the landscape mosaic provide us with drinking water, food, clean air, and aesthetic pleasure, among other benefits. By degrading natural systems, we degrade our own wellbeing. According to Ward (2005), to "replace self-destructive behaviors with sustainable behaviors" we must have opportunities to "feel and express affinity with the land" (p. 67). This implicates a normative element to land health, an emotional, moral, or sensory connection to the land-organism. Leopold incorporates this same normative requirement into his land ethic, stating "[w]e can be ethical only in relation to something we can see, feel, understand, love, or otherwise have faith in" (Leopold, 1966, p. 251). Essentially, we are the final link in the circuit of energy through the landscape; we are a part of the land organism ourselves, and we must acknowledge this interdependence in order to truly engage in conservation.

Above all, our interactions with the land, particularly our conservation efforts, should be guided by Leopold's maxim: "A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise" (1966, p. 262).

Conserving the Landscape Through Three Basic Actions

The fragmentation problem we face has several components, each of which should be addressed to ensure a successful conservation strategy. We first need to address the issue of continually expanding urbanization and our conversion of far more land than needed based on the rate of population growth. We next must address land uses within the matrix that harm wildlife and disrupt ecological processes within remnant patches. Finally, we must engage in some degree of restoration within patches and corridors.

Three main actions will guide us toward resolving these fragmentation-related issues in extensively altered landscapes such as those that predominate in the Midwest:

(1) Preserve existing habitat. We should preserve as much remaining habitat as possible and establish additional reserves where possible to incorporate into the landscape network. We should also attempt to increase the size of existing habitat patches, where possible and ecologically beneficial, to expand habitat areas as much as possible. Given the predicted high rates of future land conversion, land preservation will be a crucial element in mitigating its impacts.

(2) Stop bad land uses. As stated above, land uses within the matrix can have a substantial impact on the habitat it surrounds. They will also harm corridors that are established across it to link habitat patches. Agricultural lands can spread pesticides, herbicides, and nutrients into adjacent habitat areas, altering nutrient flows and potentially killing some plants and insects. Urban areas can spread pollutants, debris, noise, and light into adjacent habitat areas. Hydrological changes within the matrix can also change hydrological conditions within the matrix. Major diversions of water for

urban use or irrigation can deprive habitat areas downstream of adequate flow, and releases of sediment or pollutants can impair water quality, impacting aquatic and terrestrial species downstream. To truly promote land health, matrix conditions should play a supporting role to promote land health in combination with corridors and reserves.

(3) Restore degraded habitat. Patch habitat may be severely degraded in many places, especially if a patch is predominantly edge habitat or has undergone major species loss or shifts in ecological processes over sustained period of time. To restore land health, we will likely need to restore natural vegetation, hydrological conditions, and species interrelationships that improve ecosystem processes. We will also likely need to restore natural disturbance regimes, as disturbance is needed to sustain certain plant and wildlife species (Brawn, Robinson, & Thompson, 2001). Restoring connectivity between patches will also help to improve habitat conditions.

What's Needed: Considerations for Establishing Corridor Networks

To establish a functional, effective network of linkages and corridors, we should consider several important factors collectively. These factors are grounded in the elements of spatial scale, ecological context, and practical constraints that generally determine the function of any corridor, as explained above. Overall, these factors should provide us with a comprehensive and holistic approach to conservation (Steiner, 2008). We need to undertake five key actions to implement conservation corridor networks on a landscape scale: establish an overall plan, develop conservation priorities and goals, adopt an approach to guide implementation, determine the functions and purposes of corridors to be implemented, and provide for monitoring and management.

We first need an *overall plan* to guide and coordinate conservation efforts at various spatial scales. It could take the form of a comprehensive national conservation plan, like the UK Biodiversity Action Plan, which sets out in detail the elements, both ecological and logistical, to motivate and guide conservation efforts across the UK (Gummer, Lang, Redwood, Mayhew, & Chalker, 1994). The plan identifies important aspects of the country's ecology and biodiversity and uses this knowledge to inform the nation's approach to conservation. Such a plan would be ideal to ultimately facilitate a national network of protected habitat and provide a clear, coordinated approach to achieve nationwide connectivity. Alternatively, we could establish several state-wide or region-wide plans that, when combined, form linkages on an even broader, national level. To supplement large-scale plan, we could establish plans on smaller spatial scales, such as local plans, that more clearly define the terms of the broader plan and provide for conservation of local biodiversity in more detail.

It is helpful to have an overall conservation vision, but we also need to establish *conservation priorities and goals* to substantiate that vision and explain them within the overall plan. Priorities and goals are dependent upon the context of the landscape that is the subject of conservation. We should consider significant landscape spatial features, such as waterways, gradients, vegetation, and native species, a well as ecological processes, including community interactions and disturbance regimes (Peck, 1998). Conservation priorities may often consist of patterns, processes, or species that are particularly threatened or are particularly valuable to the landscape's ecological functions (Peck, 1998). Some general priority areas include: "community representation, . . . unusual abiotic features (soils, substrates . . .), functional associations of communities, abiotic gradients, . . . migratory species' routes, . . . [and] hydrologic processes" (Peck, 1998, p. 29). We may also have to determine whether to prioritize certain species or the habitat in general as the basis for conservation. Essentially, establishing priorities will provide a baseline for our conservation strategy that will guide future decisions and establish a general conservation framework within a landscape. These priorities and goals should largely stem from the three main conservation actions established above, expanding upon and applying the broad guidelines to the particular landscape context at hand. We will have to determine whether to primarily focus on species or habitat as the basis for conservation. This paper advocates incorporating both, but centers its conservation focus on habitat.

After establishing an overall plan and priorities, we should adopt a *general planning approach* for achieving landscape-scale connectivity. In the sections above, we have developed a strategy of creating conservation corridor networks on a landscape scale. The Green Infrastructure concept embodies this strategy in its central core-buffer-corridor approach to conservation planning. Green Infrastructure is an "interconnected green space network" as well as a planning process that "emphasizes the importance of open and green space as parts of interconnected systems that are protected and managed for the ecological benefits they provide"

(Benedict & McMahon, 2006, pp. 1-2). Benedict and McMahon (2006) set forth ten fundamental principles of green infrastructure that can help to ensure the creation of a successful conservation network. These principles establish the aims and scope of the green infrastructure approach. They provide that "green infrastructure should be grounded in sound science and development," and "green infrastructure requires long-term commitment" (Benedict & McMahon, 2006, pp. 37-52).

This approach to conservation can apply on multiple spatial scales, from local to national, and thus is capable of facilitating an inter-state or nationwide corridor network. It also recognizes existing and future human development within the landscape, which makes it a more practical approach than one that fails to consider the very real challenges development presents (Benedict & McMahon, 2006). The concept focuses on creating a network of habitat "hubs and links" across the landscape (Benedict & McMahon, 2006). This is essentially a version of the core-buffer-corridor concept that has been widely adopted as an important approach to landscape scale conservation (e.g, Foreman, 2004; Forman, 1995; Hilty et al., 2006). This structural approach is a more detailed conceptualization of our own conservation corridor network strategy. Our conservation strategy can therefore incorporate a conceptual approach based in the green infrastructure planning process and the spatial approach based in a core-buffer-corridor structure.

The Green Infrastructure and core-buffer-corridor concepts are useful to guide our approach to building an actual network. To construct a conservation network, we can incorporate existing state and federal reserve lands as core or hub areas, build buffers around them consisting of less intensive land uses and perhaps even expand the core areas. We can then plan linkages between these hubs to facilitate dispersal, particularly north-south linkages to facilitate northward range shifts that may be induced by climate change. Additionally, we can establish corridors along waterways to create another important ecological network, generating healthy streams and rivers that stretch across entire landscapes and regions. Guided by the core-buffer-corridor and green infrastructure concepts, our conservation strategy should ultimately focus on these two network systems—riparian corridor networks and linked reserve networks.

Though the principles provide that Green Infrastructure aims to benefit both humans and nature, in practice Green Infrastructure is primarily an anthropocentric approach to conservation, based in the need to conserve nature because of the important benefits it provides for humans. Viewing conservation solely through this perspective may result in our failure to account for and protect certain species or ecological processes that are of high ecological value but not of high value to humans. We therefore should establish an approach that also incorporates an understanding of the importance of healthy ecosystems and aims to benefit nature itself and the species within nature. This dual-purposed approach more closely correlates with our central goal of land health.

To ensure that the corridors at the heart of our conservation plan are functional and effective, we must clearly and explicitly define their *goals, purposes, and functions*. As stated in Chapter Three, it is vitally important to explain what form a corridor will take and what purposes and functions it will fulfill within a landscape (Hess & Fischer, 2001). We must therefore clarify whether a particular corridor within the system is a landscape linkage or a smaller ecological corridor, whether it will provide habitat or merely serve as a conduit, among other considerations. We must also anticipate any other functions that our corridor might perform beyond those we explicitly identify. Based on a corridor's functions and purposes, as well as its context, we must then define its appropriate size and shape. All of these considerations have been shown to be important to a corridor's effectiveness and proper functioning. They will also likely inform future monitoring and adaptive management efforts.

We must also address the species-specific nature of corridors by determining whether the corridor(s) we establish will target certain species, and if so, which species. Our corridor plans can follow one of two general paths: we can determine species that are integral to the landscape ecosystem and plan the corridor around conserving them; alternatively, we can plan to protect particular habitats that provide key ecological processes and anticipate that many species will be protected by virtue of those habitat protections. To plan around certain species, we could select one or more keystone species—species that have disproportionately significant impacts on the landscape—and plan the corridor to facilitate their dispersal (Hilty et al., 2006; Lindenmayer & Fischer, 2006). Large carnivores, such as wolves, are commonly designated as keystone species (Foreman, 2004; Hilty et al., 2006). We could also plan the corridors to accommodate an indicator species or an umbrella species. An indicator is a particular species whose status provides a measure of the status of other species, and an umbrella species refers to a particular species whose protection secures the protection of other species (Hilty et al., 2006, pp. 178-79). A keystone species could also act as an indicator and/or umbrella species. We would then determine the corridor's effectiveness by monitoring the keystone, indicator, or umbrella species' status (Hilty et al., 2006, pp. 178-79; Lindenmayer & Fischer, 2006, p. 213). Overall, it is best to select a species that can be easily identified and monitored (Hilty et al., 2006). Furthermore, planning for a multi-species assemblage may be a more effective approach than focusing on one species to the exclusion of others (Hilty et al., 2006).

Instead of planning corridors around particular individual species, we could plan them based on protecting the habitat itself. By protecting habitat, we can anticipate that at least some species will also be protected (Hilty et al., 2006). Planning corridors within or connecting biodiversity "hot spots," or physical locations within a landscape that support a large variety of species, may be one possibility (Forman, 1995). Selecting areas with high ecological value, such as riparian corridors or known species migration routes, may also be an effective strategy. Regardless of whether we plan corridors around particular species or particular habitat, we must realize that because dispersal abilities vary among species, we will likely not be able to plan corridors to conserve all species. Despite our best efforts, there will be ecological changes, and species loss will occur. We should aim to at least stabilize those losses and slow the rate of loss. At a later time, perhaps reintroduction efforts could increase species diversity within the landscape. Eventually, biodiversity could increase on its own, once habitat quality and ecological functions are improved and species begin to disperse through the network of linkages. The important thing is that the corridors actually facilitate dispersal for at least some species.

Once implemented, our corridor conservation strategy must provide for *long-term monitoring and adaptive management*. Monitoring yields information regarding a landscape's ecological response to conservation activities (Peck, 1998). It is a crucial activity to ensure that conservation management decisions that have been implemented in the landscape are promoting the desired healthy ecological functions. Adaptive management is an approach to conservation management that "uses the responses of a system to management actions to determine future actions" (Benedict & McMahon, 2006, p. 208). The purpose of adaptive management "is to acknowledge uncertainty, and then to develop a range of viable actions, each of which is designed to probe a different aspect of the system" (Peck, 1998, p. 154). It "emphasizes a scientific, rational process" (Peck, 1998, p. 155). Government agencies can undertake monitoring and adaptive management, as well as environmental organizations and perhaps even community organizations with help of local landowners (Peck, 1998). Because adaptive management requires a long-term view, it is important and useful for our long-term landscape scale conservation approach (Peck, 1998).

Learning is a key component of the process (Benedict & McMahon, 2006). Each management decision is considered an experiment that will yield valuable information and reduce uncertainty about future management decisions (Benedict & McMahon, 2006). The information about landscape ecological processes gained through monitoring and adaptive management will help to generate knowledgeable responses to any ecological problems that may arise from conservation activities and maintains flexibility to adjust conservation management to provide for improved ecological function, species movement, reduced edge effects and other beneficial ecological conditions. Monitoring and adaptive management, combined, is a way of addressing and working with the abundant uncertainty inherent in conservation efforts.

Moving Toward Implementation:

This chapter describes in general terms the conservation measures needed to interconnect a landscape through corridors and landscape linkages. Humans are a part of the interconnected, interdependent land community. Currently, we are functioning like virus cells, spreading farther throughout the organism, attacking its health. Guided and motivated by a land ethic grounded in the concept of land health, we must become land stewards and function like white cells, taking responsibility for reviving biodiversity and restoring the land's ecological functioning.

How do we enact this landscape scale and land health-based conservation strategy within our current system of land-use laws and regulations? Furthermore, how do we enact it in landscapes where most of the land is held in private ownership? Many scholars have criticized our land-use policies for failing to curb sprawl and inadequately addressing conservation concerns (e.g., Benedict & McMahon, 2006; Hidding & Teunissen, 2002). The chapters that follow will discuss current land-use laws and regulatory approaches and explain the many ways in which they are insufficient to implement a land-use system based in stewardship on the landscape scale, particularly on privately-owned lands. This paper will then propose how we can restructure our land-use laws to implement landscape scale conservation corridor networks and compel stewardship duties in landscapes dominated by private land ownership.

V. Governing the Landscape

The answer, if there is any, seems to be in a land ethic, or some other force which assigns more obligation to the private landowner.

– Aldo Leopold (1966, p. 250)

Understanding the ecological changes that are needed to create functional corridors is one thing. Achieving those changes within a human-dominated, urbanizing landscape is another. Particularly in states such as Illinois, privately-owned lands provide key pieces in the habitat linkages that wild species require to thrive. Indeed, portions of, or in some cases all, habitat for eighty-five percent of federally listed endangered species are found on private land (Rissman et al., 2007). Similarly, private lands contain seventy-five percent of the nation's remaining wetlands (Karkkainen, 1997). In short, without protections on private lands, wildlife cannot flourish. How, though, do we motivate or compel private landowners to effect the necessary land-use changes to sustain such linkages? What tools will enable us to create linkages and to preserve and restore habitat? This chapter will discuss the basic land-use laws and regulations that currently promote conservation on private lands, noting their principal features and paying particular attention to their limitations and failings. That inquiry begins, in this chapter, by examining the legal system and legal culture as a whole, considering the institution of private property and its links to laws emanating from all levels of government.

Viewed broadly, the ecological problems we face—fragmentation, species loss, landscape degradation—ultimately arise from how we use land[32] (Freyfogle, 2007b; Freyfogle, 2003; Noss, 1994). To create and protect wildlife corridors, therefore, we must first understand and evaluate why we use land the way we do.

Shifting our land uses to more ecologically-sound practices will require changes in our current land-use laws and policies. According to Karkkainen (1997), "If habitat destruction through land conversion is the principal cause of biodiversity loss for terrestrial species, then biodiversity conservation policy will necessarily implicate land use policy" (p. 70). Particularly in landscapes dominated by private land ownership, habitat conservation requires a system of

[32] By "land," I am referring to the landscape in broad terms, to include physical land as well as the wildlife, plants, and humans that inhabit the landscape. I am implicitly incorporating Aldo Leopold's concept of the "land" as ecosystems in broad terms, as a functional being made of integrated parts.

land-use law and policy that provides for and facilitates conservation. Such a system should consider the land's ecological function. It must also operate on a landscape-scale (Freyfogle, 2003). American law has begun to move in this direction; it already incorporates laws that promote conservative land uses. But measures taken to date are far from sufficient to accomplish the task at hand. Taken together they do not and cannot create and protect the wide-ranging migration corridors and other habitats that wild species need for long-term survival.

It will be no easy feat to modify the ways private landowners use their lands. Modification, however, is necessary and is, in fact, long past due. The place to begin is with the underlying foundation, with a re-evaluation and restructuring of the law governing private rights in nature.

What is Property Law?

If one has the occasion to think about private property and property law, an image of a "no trespassing—private property" sign might come to mind, or perhaps a house surrounded by a fenced-in yard. One might recall an annoying conflict years ago with a neighboring landowner over drainage problems, unsightly yard signs or disruptive backyard pets. The body of property law[33] in the United States is expansive, encompassing issues such as these and myriad others. It is the legal basis of the institution of private property, of individual property rights, and of land-use permissions and limitations.

The word "property" encompasses a wide variety of objects. For instance, law recognizes personal property, real property, and intellectual property. The term "real property"—the type of property considered here--refers specifically to land and everything attached to land, whether growing trees, houses or parking lots. In the law, property is not understood chiefly as "a thing at all but a concept—the legal relationship among people in regard to" the item that is owned (Cribbet, Johnson, Findley, & Smith, 2002, p. 4). Thus, to own a tract of land is to have the legal power to limit how other people can use that land and to halt some or all interferences with one's own use of the land. To own land is, above all, to have the power to constrain the liberties of other people (Freyfogle, 2010).

[33] I use "property law" here as a broad term, to incorporate the bodies of property law, land use law, and the institution of private property and their attendant theories and interpretations.

U.S. property law is not a compendium of distinct laws that mandate what landowners can and cannot do. It is, instead, an amalgamation of many statutes, regulations, local ordinances, zoning schemes and comprehensive plans, all resting upon foundational common law principles and rules. The common law itself emerged gradually over time, mostly through court decisions with occasional modifications by Parliament. In the United States this common law of property resides at the state level. It thus varies somewhat from state to state, just as states vary in their statutes and local governments vary (even more) in their ordinances. When statutes are challenged as interfering unduly with individual property rights, courts routinely return to the common law as a point of beginning, as the United States Supreme Court did, for instance, in David Lucas' much-publicized challenge to a South Carolina beachfront protection law that prevented him from building residences on beachfront land that he owned,[34] in Anthony Palazzolo's challenge to Rhode Island wetlands regulations that caused the state to deny him a permit to build on wetlands[35], and in the Tahoe-Sierra Preservation Council's challenge to the Tahoe Regional Planning Agency's prohibition of new construction on sensitive lands around Lake Tahoe to protect the Lake's clarity[36].

As evidenced by its various components, property law is created by a variety of lawmaking entities including federal, state, and local governments, as well as by local land-use commissions and regulatory bodies. Some of these entities overlap, so that multiple levels of law—and laws from multiple sources originating at the same level--can and do apply to a single parcel. Historically, states have held the ultimate authority (sovereign power) to regulate land-use, power that they typically delegate in part (and subject to limitations) to local governments. Rising above these state and local laws are statutes enacted at the federal level, restricting the uses of private lands in the interest of achieving environmental or public health goals. Necessarily and appropriately, landowners in different places have, as a result of these laws, widely varied rights to use what they own.

Property rights are commonly referred to as a "bundle of sticks," meaning that ownership includes or entails multiple individual rights, or legal sticks[37] (Cribbet et al., 2002; Goldstein & Thompson, 2006). In the case of land it is appropriate that this popular metaphor derives from

[34] *Lucas v. South Carolina Coastal Council*, 505 U.S. 1003 (1992).
[35] *Palazzolo v. Rhode Island*, 533 U.S. 606 (2001).
[36] *Tahoe-Sierra Preservation Council, Inc. v. Tahoe Regional Planning Agency*, 535 U.S. 302 (2002).
[37] See Goldstein (1998) or a list of examples of types of rights in bundle. They include, "1.The right to possess; 2.The right to use; 3.The right to manage . . . 11.The prohibition of harmful use" (p. 375).

nature, from pieces of wood. Yet this natural metaphor also suggests the inherent irony of property rights as we typically understand them: private rights are grounded in nature yet cast nature in terms of commodities to be "bundled" up and owned by an individual. Critics of this metaphor argue that the "bundle" concept leaves out the responsibilities that accompany rights in property (Goldstein, 1998). Freyfogle (2003) argues further that this metaphor mischaracterizes the law; it overlooks how the rights of neighboring owners are intertwined and how lawmakers, when crafting private rights, necessarily must choose between and among alternative sticks. Goldstein (1998) has proposed the addition of "green wood" to the bundle of property rights in the form of "a duty of environmental context" (pp. 410-11). This proposed "stick," based in the ecological context of a particular land parcel, would recognize that the surrounding land community, not just the individual owner, has legitimate interests in how a land parcel is used. It would add to the bundle a requirement that an owner act only in ways consistent with the long-term welfare of that land community.

Property-rights advocates have asserted that property rights are inherent, natural rights at the core of individual liberties (Epstein, 2009). This cultural and political claim, though, is widely denied and is not supported by law, constitutional or other. Anglo-American law has never viewed property rights as existing independently of the particular positive law (the common law and statutes) that gives rise to the property rights (Goldstein, 1998). Similarly, property law is not chiefly a body of *limits* on how a person can use what he or she owns, despite a commonly held belief to the contrary. Rather, it is, at root, the *source* of property rights; the source of the legal power a person has to halt interferences with his or her use of the property. Property rights are thus "a product of law" and are valid only to the extent provided for and enforced by law; without such laws, property rights would not exist (Cribbet et al., 2002, p. 2; Freyfogle, 2007b, p. 13). Given this grounding in the work of lawmakers, generation upon generation, the institution of private property rights is in reality a "social convention," not an institution detached from time and place (Freyfogle, 2007b; Powell, 2009, § 2.01). Embedded as it is within a broader social framework, property ownership routinely comes with duties in addition to rights (Freyfogle, 2007b; Goldstein, 1998; Powell, 2009, §2.10). Ownership of property, in other words, "involves the rights and duties which the owners possess incidental to the status of title" (Goldstein, 1998, p. 347). Finally, like all bodies of law property law is subject to change (Goldstein & Thompson, 2006; Powell, 2009, §2.06). Throughout its history it has evolved based on the same cultural, political, and economic factors driving change in other bodies of law.

The Forms of Fragmentation

The American land ethic, if there is one, could be described as "private and divisionary" (Colburn, 2007). This "private and divisionary" orientation toward the land has contributed in a number of ways to the fragmentation of American land-use law, and as a result, of the American landscape. First, land-use law divides the landscape into many small, individual parcels owned by independent landowners. These landowners make their own land-use decisions based upon perceived self-interest, largely independently of one-another and quite often isolated from any conception of the common-good. As they seek maximum economic gains from their property, owners also often fail to care for the natural resources under their ownership. They frequently believe that ownership empowers them to do whatever they want, on the land or to the land, as long as they avoid physically harming neighbors.[38] This division of decision-making power over land results, predictably, in a "tyranny" of many "small decisions made singly;" a tyranny that flows when countless landowners, acting individually, disrupt their landscape's ecological functioning and harm common resources (Karp, 1993, p. 736; Theobald et al., 2000, p. 36). The division of an open-access commons into private shares might well diminish the "tragedy of the commons," but without adequate communal power to coordinate private actions privatization results in its own tragedy, the "tragedy of fragmentation" (Freyfogle, 2002). Without a balance between private rights and collective governance, the fragmentation of landscapes into individual private parcels can easily result in ecological degradation.

Fragmentation is also inherent in the laws themselves. One type of fragmentation arises because of the different levels of government that can and do enact overlapping laws. A municipal government, a county government, and a state government could all pass laws applicable to a given land parcel. Land-use laws are also fragmented in terms of the policies they promote. Many laws pay attention to specifically human or inter-personal concerns and show no concern for the natural attributes of the land itself. Such laws quite often authorize or encourage land uses that are inconsistent with the land's ecological functioning and the needs of

[38] As Freyfogle (2002) has explained, this is not a legally valid conception of private property rights, but is a commonly-held cultural belief harbored by many landowners who wish to maintain ultimate, absolute control over their lands (pp. 327-328).

wild species. Furthermore, land-use laws are based upon and defined by human-drawn political boundaries, and these political boundaries are most often not the same as ecological boundaries, such as watersheds (Colburn, 2007; Farrier, 1995; McElfish, 2004; Noss, 1994). Ultimately, effective land-use laws should take into account the natural features of the land and its ecological functioning, with private rights balanced with the common good. In some way, we must shift from a use-based and commodity-focused perspective toward private land, replacing it with an ecologically-informed perspective that helps humans meet their needs but also shows respect for biodiversity and ecosystem functions.

These basic points provide the legal and cultural framework—the points of beginning—for efforts to use law and public policy to protect biodiversity. Our landscapes are legally fragmented; land-use powers are diffuse; our laws for generations have paid little attention to nature; and our culture embraces and exalts individual liberty, including liberty for landowners. A sound conservation effort must understand these realities, finding ways either to work with them or change them, all with the goal of sustaining biodiversity, ecosystem functioning, and thus land health.

Land-Use Tools and Policies: A Brief Introduction

The following subsections examine particular elements of current law that attempt to enact conservation measures. Which legal tools are relevant for establishing corridors and protecting biodiversity on a landscape scale? How effective are these tools for conserving habitat? What are their benefits and drawbacks?

Three primary sources of property and land-use law are particularly applicable for regulating land uses to protect the land's ecological attributes: the basic common law principles of property; federal environmental statutes; and local zoning laws. All of these legal tools have some habitat and biodiversity conservation value. They also serve, however, other, much different policy goals, particularly the common law and local zoning laws. Further, they contain sizeable gaps and by no means overcome the various challenges of fragmentation noted in the previous chapters. Despite these defects, though, current law provides good places to begin, and its failings provide instructive lessons for crafting more effective legal tools.

Common Law of Property

Property law consists of several fundamental common law principles that secure a landowner's rights against the community and that also impose duties upon a landowner to prevent harm to neighbors and the community. One of the most fundamental common law powers is the landowner's ability to exclude anyone or anything she does not want on her property (Goldstein & Thompson, 2006). The ability to exclude, however, is far from absolute; exceptions apply depending upon context. As a legal matter, what is termed the right to exclude arises out of the law of trespass; it is, in fact, a product of trespass law in that a landowner can only exclude if and when trespass law provides a remedy. Under trespass law, a landowner can often sue and obtain relief if a person physically enters a landowner's parcel without the landowner's permission (Goldstein & Thompson, 2006). Trespass can also occur if a person places animals or other objects on a landowner's property without permission (Goldstein & Thompson, 2006). More severe and intentional trespasses can violate criminal statutes, subjecting trespassers to criminal penalties, including jail sentences (Goldstein & Thompson, 2006). As a result of trespass law a landowner can largely keep unwanted persons, animals, and objects off of her land, but is also responsible for staying off of other landowners' lands. Like two sides to a coin, a landowner's right comes with a reciprocal duty.

Perhaps *the* foundational principle of property law, from which all others logically arise, is *sic utere tuo*, or the do-no-harm rule. In its full Latin form *sic utere tuo* means "use your land so as to do no harm to others" (Freyfogle, 2007a, p. 257). Under this vague but vital principle, a landowner is inherently obligated to prevent harm to neighbors when undertaking any activity on her land. In actual disputes, courts can encounter difficulty in determining causation, or which landowner is injuring the other, when applying the do-no-harm rule. A baseline of acceptable behavior is needed to effectively apply the principle, otherwise it is "logically useless" (Freyfogle, 2007a, p. 259). This principle limits landowner A's activities to those that do not harm other landowners or the community. Because other landowners are equally bound and limited by the rule, it also protects landowner A from harm caused by other landowners. The rule is thus, at root, the *source* of property rights rather than merely a *limit* on them.

This do-no-harm principle[39] forms the core of the common law rule of nuisance, which gives landowners a legal remedy when they are materially harmed by outsiders under circumstances that seem unreasonable. Nuisance provides landowners and the public protection and recourse from activities on neighboring parcels of land that are, on balance, unreasonable under the circumstances and that on personal or public rights or impede the use and enjoyment of land. “The right of society to curtail the absoluteness of the individual’s dominion lies at the heart of nuisance” (Powell, 2009, §2.10). Prior to the federal environmental legislation of the 1970s, nuisance law provided much of the “strength and content” of U.S. environmental law (Ruhl, 2008). There are two types of nuisance: private nuisance and public nuisance (Restatement, §821A).

Private nuisance occurs between individual landowners, when one landowner’s actions materially and unreasonably impede another’s ability to conduct his own activities on his land. According to the Restatement of Torts, a private nuisance is “a nontrespassory invasion of another’s interest in the private use and enjoyment of land” (§ 821D). By “nontrespassory,” the definition implies a non-physical invasion, such as when smoke, noise, light, or runoff water travels from one landowner’s property onto another’s property. Private nuisance suits have included, for instance, a dispute over a landowner’s installation of a windmill that disturbed a neighboring land owner with loud, incessant noise (*Rose v. Chaikan*, 453 A.2d 1378 (Super. Ct. N.J. 1982)) and a dispute between a landowner who proposed constructing a residence that would obstruct sunlight from his neighbor’s rooftop solar panels (*Prah v. Maretti*, 108 N.W.2d 182 (Wis. 1982)).

Courts evaluate nuisance cases individually, based on the facts of each particular situation. A court will generally[40] find a person liable for private nuisance if his harm-producing invasion of another’s use and enjoyment of land is unintentional and constitutes reckless, negligent, or abnormally dangerous conduct (Restatement, §822). If the conduct is intentional, a court will generally find a person liable if the conduct is found to be unreasonable (Goldstein & Thompson, 2006; Restatement, § 825). The primary question in a nuisance claim is whether it is

[39] According to Guth, (2008) the nuisance balancing test largely overrules the do no harm principle—the do no harm rule once “defined an overarching standard that could not be invaded by cumulative impacts” unlike nuisance (p. 49).

[40] I use “generally” here because nuisance is a common law doctrine determined most often in state courts. As a result, different states can have slightly different approaches to determining nuisance. The elements for liability listed here are set forth in the Restatement, which expresses the common, general approach taken by most courts.

reasonable for the defendant, or the landowner whose conduct is challenged, to be doing what he is doing where he is doing it (Restatement, § 824; Ruhl, 2008). To determine "reasonableness," courts generally balance the "gravity of harm" against the "utility of the [disputed] conduct" (Goldstein & Thompson, 2006; Restatement, §826). Courts weigh several factors when determining the gravity of harm and the utility of conduct. Additionally, to qualify as a nuisance, the harm imposed must be significant harm, "involving more than slight inconvenience or petty annoyance" (Restatement, §821F cmt. c). For example, in *Rose v. Chaikan*, the New Jersey Superior court determined that the noise of defendant's backyard windmill constituted a private nuisance to the surrounding neighbors because of the unnatural "character, volume, and duration" of the noise and because the harm due to the noise outweighed the windmill's social utility as a form of renewable energy (453 A.2d at 218).

Public nuisances occur when a landowner's conduct results in an interference with the rights of the public (Goldstein & Thompson, 2006). Public nuisance has generally been defined as an "unreasonable interference with a right common to the general public" (Restatement, §821B). The fundamental issue underlying public nuisances is not the number of people injured but whether public rights have been adversely impacted (Goldstein & Thompson, 2006). Courts commonly find public nuisances where a landowner's conduct has resulted in a significant interference with public health, safety, peace, comfort, or convenience (Restatement, §821B, cmt. b). Many public nuisance disputes have involved law suits against a factory or other enterprise for pollution (Goldstein & Thompson, 2006). The person bringing suit for a public nuisance must have suffered a "particularized injury," or harm that is "different in kind" from the harm to the public at large; it is not sufficient if the plaintiff has been hurt merely in the same way as all other members of the local public (Goldstein & Thompson, 2006). To determine liability under public nuisance, courts have generally applied the same reasonableness-based balancing test described above for private nuisance (Restatement, §826 cmt. a). Additionally, it is possible for a landowner's conduct to qualify as both a public and private nuisance (Goldstein & Thompson, 2006).

Most courts allow concerned landowners or members of the public to bring a nuisance suit to pro-actively stop harm from occurring. In such "anticipatory nuisance" cases, plaintiffs are able to obtain an injunction to legally stop the landowner's proposed conduct (Restatement, §822 cmt. d; Ruhl, 2008). Historically, however, courts only allowed anticipatory nuisance suits if the conduct qualified as a nuisance *per se;* that is, if the conduct would be deemed a nuisance under any circumstance, without regard to the details of how and where it was undertaken (Goldstein & Thompson, 2006).

While common law nuisance enables landowners to stop harm, proactively or retroactively, the doctrine provides a remedy only for harms to humans, particularly human uses of land. It does not account for harm to ecological attributes of the land, whether in the balancing test to determine reasonableness or as its own independent action. The law does not acknowledge an "ecological public," for instance, that might be harmed by a landowner's actions; wildlife and plants are unable to bring their own nuisance suits against the landowners whose land uses harm them and their surroundings. Courts only consider the ecological aspects of a landscape in a nuisance suit with respect to their use by humans. Furthermore, the balancing test to determine reasonableness is based on the circumstances of each individual situation and is likely to give ecologically degrading uses that have high economic or social value more weight over concerns for ecological protection in its own right. As it currently exists, therefore, nuisance law provides limited protections against ecological harm apart from their use and enjoyment by humans. Aside from the these substantive defects in nuisance, nuisance litigation poses practical problems: suits are expensive to bring; the legal outcome of cases highly uncertain; and suits involved multiple plaintiffs (the typical case) are hard to organize because of difficulties in getting plaintiffs to work together and share the costs. It is thus unsurprising that nuisance suits are relatively rare, and have little effect in curtailing ecological degradation.

State Trust Duties

State governments have independent authority to protect natural resources and to prevent harm to the land and to the community under two related bodies of law: the public trust and wildlife trust doctrines. Both bodies of law empower states to take action to protect particular parts of nature and, in vague ways, also impose duties on states to do so.

According to the public trust doctrine, state governments hold title to publicly-available natural resources subject to a "trust" that benefits the people of the state[41] (Hudson, 2009;

[41] A state's public trust authority was granted as an inherent attribute of statehood once the state joined the Union, a remnant from the nation's English legal roots. Under English law, the king held authority over lands beneath navigable waters, and as this authority was not delegated to the federal government in the Constitution, it was reserved to the states (Freyfogle 2007a; Hudson, 2009).

Freyfogle, 2007a). This authority is in some way, not well articulated by courts, related to a state's police powers, or its inherent power to protect the community's health, safety, and welfare. In some states the doctrine is grounded in the state constitution, in provisions that specific vest the state with certain natural resources to be held in trust (Wood, 2009a). Public trust "simply means that the public owns in common certain property interests in natural resources and land within the territory, and that the government is the people's designated trustee with the obligation to protect such property on behalf of the citizens" (Wood, 2009a, p. 66). As trustee, the government manages the land, the subject of the trust, for the benefit of the state's citizens. *Illinois Central Railroad v. Illinois* is the seminal public trust ruling. In it, the Supreme Court held that the state of Illinois holds the shore of Lake Michigan in trust for the people of the state; as a result, Illinois could not sell the shore to a private company, the railroad, at least without firm protections for the public's continuing interests (146 U.S. 387 (1892)).

The public trust doctrine, despite its broad language, is quite limited in it effect because it applies in most states only to narrowly defined parts of nature. Typically, the lands held in trust are limited to tidelands and lands underlying navigable waters. These lands were deemed vital to the public because of the importance of navigable waterways to national commerce and, even more, the importance to the public of having access to fisheries (Freyfogle, 2007a; Hudson, 2009). Some scholars assert that the doctrine remains limited to lands under navigable waters (Freyfogle, 2007a). Others assert that the doctrine is broad and might properly encompass a wide range of natural resources, including biodiversity and wildlife habitat (Wood, 2009a; Hudson, 2009).

Similar to the public trust, the wildlife trust doctrine endows state governments with the authority to hold a state's wildlife in trust for the benefit of the state's citizens (Freyfogle & Goble, 2009). This trust arose because wildlife in England—or at least the valuable species that were of public concern—belonged to the Crown, which owned wild animals not as the King's personal property but in his capacity as sovereign head. The King was expected (by Parliament at least) to exercise his powers for the benefit of the people as a whole. These royal powers passed to the new states when Independence was declared (and to states created thereafter at the time they joined the union). The two doctrines have commonly been treated as interchangeable (Hudson, 2009) although the duties they impose on states as trustees would seem to vary based on the part of nature involved. Under the managerial responsibility imposed by these two doctrines, state governments are able to halt and prevent landowner actions that may cause harm to the resources and to seek money damages from actors who destroy trust property. In addition, as noted, the doctrines apparently impose some general duty on states to protect trust property for the benefit of the citizens.

Although state governments possess this authority in theory, they often fail to act in accordance with it, pursuing development and resource exploitation to generate revenue (Wood, 2009a). According to critics, state agencies have consistently acted contrary to their trust duties (Wood, 2009a). The public trust doctrine in particular could be applied more broadly to protect various vital parts of nature, both because the resources have direct value and also because of their roles in sustaining ecosystem services. If courts continue to interpret the public trust doctrine narrowly, however, this broader scope of protections may not be realized. At present, the wildlife trust does little to protect wildlife—and almost nothing to protect wildlife habitat—and the public trust doctrine is largely limited to protecting navigable waterways from unauthorized interference. The doctrines hold potential, but at present are not, except in a few settings, potent legal tools.

Underlying all of these fundamental common law principles is the premise that a landowner should not act in ways that cause harm to other landowners or to the community. Yet landowner activities regularly cause ecological harm, perceived or not, direct or indirect. Why this contradiction between legal requirement and landowner action? What is the malfunction within our land-use system that prohibits harm, yet permits it to occur? The problem, it would seem, is not with the do-no-harm principle or with its manifestations in nuisance law. It is with the ways we have traditionally defined "harm." Harm has rarely covered ecological degradation that did not cause direct financial loss to a landowner or to community members. Lawmakers over the years have often redefined harm to cover new situations and to reflect new values (Freyfogle, 2003). They could do so again, taking into account our environmental plight, by including significant ecological degradation within the coverage of the term.

The first step to re-evaluating our conception of harm and, with it, our system of private property, is to understand the existing laws and regulations that govern private land uses. What is the legal framework that has permitted urbanization to occur? What laws currently exist to curb private land uses in the name of environmental protection? Many laws and regulations exist at multiple levels of government, but none sufficiently address the conflicting conceptions of harm underlying our paradoxical system of private property.

Federal Environmental Statutes

The federal government has not adopted a central, overarching law that regulates land use on the national level, though it has contemplated doing so. Despite the lack of a national land-use regulatory system, several federal environmental laws regulate uses of land in the interest of protecting endangered species and sensitive lands such as wetlands and highly erodible areas. Land-use regulation on the federal level is thus interwoven with federal environmental protection laws. To control or otherwise influence land uses, some laws employ a system of financial incentives and disincentives while others directly mandate specific action, providing a permit option for exceptions. These diverse efforts in combination do supply some land and biodiversity protection but they are, overall, uncoordinated and ineffective. As one commentator has observed, current statutes demonstrate that "the articulated goal of biodiversity conservation has yet to develop into clear, effective, and coordinated policy in the United States" (Karkkainen, 1997, p. 6).

The Endangered Species Act indirectly regulates land use as it provides protections for listed species by prohibiting activities, including land uses, that would result in harm to listed species. Section 9 of the Act prohibits the "taking" of endangered species (16 U.S.C. § 1538). For threatened species, the default rule is that Section 9 prohibitions automatically and fully apply unless the agency issues a regulation providing lesser protections (Freyfogle & Goble, 2009). "Taking" includes direct and indirect killing of listed species, encompassing acts that "harass, harm . . . wound, [or] kill . . ." the species (16 U.S.C. § 1532). The meaning of "harm" sometimes includes habitat modification (Freyfogle & Goble, 2009). For habitat modification to constitute an unlawful taking, however, it must be shown that individual protected organisms have been killed or injured as a result. The ESA thus qualifies as a land-use regulation designed to protect imperiled biodiversity, although it does so only in very limited circumstances.

The Act provides an exception to this "take" prohibition, however, by authorizing incidental take permits (ITP). An ITP can authorize any person to engage in an otherwise unlawful take of a listed species so long as the take is "incidental to, and not the purpose of, the carrying out of an otherwise lawful activity" (16 USC § 1539). Such an exception to the ESA's take prohibition makes the statute more flexible, but also “weaker” (Freyfogle & Goble, 2009, p. 250). To obtain an ITP when the proposed take will be caused by habitat modification, the person seeking the take permit must prepare a Habitat Conservation Plan (HCP) that specifies, among other things, the impact the proposed action will have on the species and the actions to be taken that will mitigate the harm to the species (Freyfogle & Goble, 2009; Karkkainen, 1997). The U.S. Fish & Wildlife Service, which issues ITPs for all terrestrial and most fresh-water species, must ultimately review the HCP and approve or reject the permit application. Even if a landowner is allowed to take an endangered or threatened species under an ITP, the ESA can limit a landowner’s use of his land to mitigate impacts on the species through the HCP.

The ESA can also limit land use through its critical habitat provision. The Act requires the listing agency (usually the Fish & Wildlife Service) to designate critical habitat for a species upon its listing (16 U.S.C. § 1533(a)(3)). Critical habitat consists of the “physical and biological features essential for the species’ existence” within the species’ occupied range that “may require special management considerations or protection” (16 U.S.C. § 1532(5)). It also includes areas outside of the species’ range, if they are “essential for the conservation of the species” (16 U.S.C. § 1532(5)). Despite the importance of habitat and the fact that habitat loss is the most common reason species become endangered or threatened, the ESA allows for many exceptions to the designation rule, and little habitat is ever designated (Freyfogle & Goble, 2009). The listing agency can choose not to designate critical habitat if designation is “not determinable” or “prudent” or not beneficial for the species (Freyfogle & Goble, 2009, p. 250). The agency must also engage in cost-benefit analysis when deciding to designate, and if the costs of designation outweigh its benefits, the agency can decline designation (unless failure to designate will result in the species’ extinction) (16 U.S.C. § 1533(b)). As a result of a critical habitat designation on his land, a landowner’s use of those designated portions will be limited to ensure their protection. The ESA also requires the listing agency to prepare a recovery plan for each species, unless the plan would not “promote the conservation of the species,” and the Act provides for monitoring (16 U.S.C. § 1533(f)-(g)).

Although the ESA has the effect of limiting uses of lands inhabited by endangered or threatened species its limitations are largely ineffective, rarely restricting land use to any significant degree and thus failing to protect the at-risk species. The Act has benefited several species, including the bald eagle and the gray wolf, but mostly (as in these two instances) through restrictions on direct killing of animals, not by protecting species habitat (Freyfogle & Goble, 2009; Waller, 1996). Many species, listed and unlisted, remain inadequately protected, particularly species at risk due chiefly to habitat loss. The Act's species-by-species approach, focusing on individual species in isolation from other species and their ecosystems, unwisely ignores an individual species' role within the wider landscape (Karkkainen, 1997).

The ESA's focus on single species further ignores the issue of ecosystem health and ecosystem function—essentially ignoring the "forest for the trees." For political and cultural reasons this approach also favors charismatic megafauna, or visually-appealing, socially well-known species such as the wolf and the polar bear, over less *cute* or visible species, like the fairy shrimp, amphibians, and insects (Karkkainen, 1997). Charismatic species frequently receive more attention and protection than others, skewing the Act's protections toward a select subset of individual species. Furthermore, protections largely apply only after a species is formally listed by means of a process that is quite lengthy. The impending listing of a species could create a problem of "perverse incentives," prompting landowners to eradicate or drive away imperiled animals and to destroy suitable habitat, in an effort to avoid being constrained in their land-use options after the species is listed (Olson, 1996).

The exceptions and exclusions within the Act, particularly its inability to protect habitat, hinder the ESA's effectiveness. Encumbered by such limitations, the ESA has the effect of keeping charismatic species "on life support" as opposed to actually restoring ecosystems to health (Colburn, 2007, p. 267). As noted, the Fish & Wildife Service often fails to designate critical habitat or delays designation for long periods (in part because the designation process can also take considerable time and get snarled in litigation) (Freyfogle & Goble, 2009). When habitat is designated, the amount is often insufficient to sustain a species or help it to recover (Colburn, 2007). Additionally, the critical habitat provision only considers habitat in which a species currently is found. Climate change, however, may cause the appropriate habitat conditions for a species to shift. As written, in short, the ESA is ill-equipped to handle the potential future uncertainties regarding habitat ranges.

One of the most significant drawbacks of the ESA is that it is reactive, not pro-active. It only implements protections once a species is already threatened or endangered; it provides no protections to help species before they reach dangerously low population levels. Critics also complain that it is insufficiently funded, which restricts the agencies' implementation capabilities

(Freyfogle & Goble, 2009; Waller, 1996). Given these various limitations and challenges, it is apparent why the ESA falls far short of protecting biodiversity across large landscapes.

Shifting the federal focus from species to the land itself, the Conservation Reserve Program (CRP) allows farmers to take sensitive crop lands out of production and place them into "conservation reserve," or to use them less intensively, in return for payment from the U.S. Department of Agriculture (16 U.S.C. §§ 3831-3835a). The program is intended to "conserve and improve the soil, water, and wildlife resources" of the conserved land (16 U.S.C. § 3831(a)). CRP protections apply to erodible land, including erodible cropland, pastureland that has been converted to wetland or natural vegetation, land that would "pose an . . . environmental threat to soil, water, or air quality," and land that is used as a grassed waterway, buffer strip, hedgerow, or other conservation feature (16 U.S.C. § 3831(b)).

Under the standard conservation contract, the landowner's duties include planting vegetative cover and engaging in management activities. The landowner also must provide a plan for the land being converted to conservation status (16 U.S.C. § 3832(a)). In return for conserving portions of their lands, landowners receive annual rental payments from the Government and, for a period of two to four years, cost-sharing payments for up to 50% of the costs for planting and management activities that the Secretary of Agriculture determines are "appropriate and in the public interest" (16 U.S.C. § 3834). The statute clearly specifies that CRP payments also apply to land converted to wildlife corridors (16 U.S.C. § 3834). Though this program may enable some land to return to functional wildlife habitat, it is far too limited to provide any long-term, viable conservation gains.

Due to limitations in duration, scope, and application, the CRP cannot protect sensitive lands long term. The actual impact of the CRP is little more than a "short-term land retirement program" (Farrier, 1995, p. 333). By using contracts of limited duration, the CRP fails to provide permanent protections. The statute limits contracts terms to between ten and fifteen years (16 U.S.C. § 3831(e)). Once the contract term has expired, landowners are free to convert the conserved lands back to crop-production, reversing all of the conservation benefits that had been gained and to a significant extent wasting the taxpayers' financial investment. Furthermore, if income from crop production on the land reaches a price at which it outweighs the income from government payments, landowners will have a strong economic incentive to take lands out of reserve before the contract ends and forgo government payments. With increasing production of crops to make biofuels, it is anticipated that acreage in CRP will decrease, as sensitive lands are taken out of production to produce fuel crops (National Research Council, 2008). Additionally, the statute specifies that the program is effective through 2012 (16 U.S.C. § 3831). If Congress fails to renew the program, it will cease to exist after 2012, and all of the protections it has established will likely disappear.

The CRP is also a fragmented approach to land conservation, impeding the development of linkages and conservation on a landscape scale. The program looks at individual parcels of land in isolation and does not factor a land's overall ecological value when deciding which lands to conserve. The primary focus is preventing erosion, and while this is a noble goal in itself, it is not sufficient to conserve ecosystem functions across landscapes. The statute also caps the amount of land to be conserved to 32,000,000 acres. Because the program is voluntary, it protects only a subset of existing sensitive lands and fails to include unwilling landowners; because participation is voluntary, no agency can use the program to construct and protect a continuous wildlife corridor. The program also permits landowners to conduct some harvesting and grazing activities on conserved land (16 U.S.C. § 3832(a)), providing for further disturbance and compromising the conserved land's value as habitat. The CRP "is not a carefully planned attempt to conserve representative ecosystems by linking degraded areas in need of restoration with relatively undisturbed areas" (Farrier, 1995, p. 333). By conserving individual pieces of land one by one, with caps on the amount of acres to be conserved in any individual parcel, the CRP fails to provide an effective, landscape scale solution to fragmentation.

In addition to the Conservation Reserve Program for erodible cropland, Congress passed a wetland and buffer acreage conservation reserve program (16 U.S.C. § 3831b). A landowner can enroll existing wetland or land on which the landowner plans to develop a wetland. The statute also provides for enrollment of buffer land around wetland, including land contiguous to or needed to protect the wetland (16 U.S.C. § 3831b(b)). Under the program, landowners are required to "restore the hydrology of the wetland . . . to the maximum extent practicable" (16 U.S.C. § 3831b(e)), "establish vegetative cover" on the wetland area, and not use the land for commercial purposes (16 U.S.C. § 3831b(d)).

Like the CRP, the wetland and buffer reserve program's conservation impact is limited. The program extends only through 2012 (16 U.S.C. § 3831b(a)). If Congress declines to renew the program, the conservation value the program has been gained will likely be lost as farmers withdraw from the program and begin draining and planting in wetland areas. The statute sets a maximum limit for the number of acres to be enrolled. The program can only enroll up to 1,000,000 acres total, and a maximum of 100,000 in any one state (16 U.S.C. § 3831b(c)). The program also sets a maximum number of contiguous acres that any one landowner can enroll at 40 acres (16 U.S.C. § 3831b(d)). These maximum acreage limitations, particularly the limit on the number of contiguous acres any one landowner can enroll, perpetuate fragmentation. The contiguous acreage limitation essentially establishes a mandatory system of small wetland patches, located in isolation of different individual farms. There is no provision requiring or even recommending coordination in the location of enrolled wetland acres between neighboring farms. The statute aims for quantity of wetland preserved at the expense of ecological quality. There may be arguments that preserving a higher quantity of wetlands is preferable to quality, to theoretically preserve a more diverse number species in different areas, or a more diverse collection of wetlands instead of a large expense of only one type of wetland ecosystem. Still, the program overlooks the landscape scale and species' needs, focusing narrowly on the number of acres enrolled.

A second wetlands program, the Wetlands Reserve Program (WRP), has a broader scope and aims to establish more permanent wetland protections (16 U.S.C. § 3837). The program was established "to restore, protect, or enhance wetlands on [eligible] private or tribal lands" (16 U.S.C. § 3837(a)(1)). Overlapping with the wetlands conservation reserve program, the WRP applies primarily to farmed or converted wetland used for agricultural production, not pristine, unconverted wetland, unless unconverted areas border the converted wetland and contribute to the protection of the converted area (16 U.S.C. § 3837(c)). The WRP does, however, apply more broadly, establishing more permanent protections for vulnerable wetlands enrolled in the conservation reserve program "that are likely to return to production when they leave the conservation reserve" (16 U.S.C. § 3837(d)). It also can apply to other wetland if the Department of Agriculture determines that their protection will add to the functionality of other, qualified wetlands (16 U.S.C. § 3837(d)). In determining wetland eligibility, the statute requires the agency to consider lands that will "maximize[] wildlife benefits and wetland values and functions" (16 U.S.C. § 3837(d)).

To participate in the program, a landowner is required to grant an easement[42] to the Department of Agriculture, implement a conservation plan, and record a deed restriction (16 U.S.C. § 3837a). The easement terms prohibit the landowner from altering wildlife habitat and engaging in other destructive activities. The easement can also restrict activities on adjacent land owned by those with land enrolled in the WRP where these activities would diminish the functional value of the enrolled land (Farrier, 1995). The easements are for either 30 years in duration or are permanent (16 U.S.C. § 3837a). The program also permits the use of "restoration cost share agreements, or a any combination [of easement types and cost share agreements]" (16 U.S.C. § 3837(b)(2)). The program provides for payments to the landowner for the value of the easement in up to 30 annual payments. The value of the easement is determined based on fair market value of the property, corresponding to geographical cap, or based on an offer by the landowner (16 U.S.C § 3837a(f)). The Department of Agriculture also pays the landowner between 75 and 100 percent of the costs for establishing conservation practices and management for a permanent easement and between 50 and 75 percent of those costs for a 30 year easement (16 U.S.C. § 3837c).

The wetland protections that the WRP establish are more stringent and more permanent than a contract that can easily be broken. Like the other reserve programs, however, the WRP has significant ecological limitations. Though easements are a stronger form of protection, they are not always permanent and are vulnerable to being compromised, as will be discussed below. Additionally, temporary easements still carry the risk that the protected wetlands will be converted back to farmland or other uses after the easement term expires, destroying the ecological benefits that were gained. Like the others, this program sets a maximum limit for the number of acres that it will protect. Currently, the program can enroll no more than 3,041,200 acres (16 U.S.C. § 3837(b)(1)). The WRP also allows for "compatible economic uses" of land such as periodic grazing and timber harvest, which could compromise the habitat value of the wetlands (16 U.S.C. § 3837a(d)).

The WRP takes the same fragmented approach to wetlands protection as the conservation reserve programs. Through its landowner-by-landowner and parcel-by-parcel approach to protection, the WRP provides no coordinated, landscape-level system of wetland protection,

[42] An easement is a "nonpossessory" interest in another owner's land that allows the person holding the interest to use the land and reciprocally restricts the original landowner's use of the land subject to the easement (Cribbet et al., 2002, p. 492).

perpetuating habitat fragmentation. This program does, however, consider the ecological value and function of the wetlands to be protected, which can help distinguish wetlands with high habitat value for protection. This ecological assessment, however, does not seem to include consideration of the wetland's value to larger landscape scales and to wildlife needs at such scales. And, of course, the exclusive focus on wetlands offers no protections to other land types, which could be more ecologically beneficial or important to wildlife.

Two additional farmland conservation programs, called "Sodbuster" and "Swampbuster," aim to protect sensitive lands by using the threat of penalties to compel conservation. Swampbuster takes away federal aid and subsidies from farmers who convert wetlands into croplands (16 USC § 3821; Karkkainen, 1997). The scope of Swampbuster's impact is limited by exemptions. The statute exempts wetlands that were converted to cropland prior to 1985 as well as wetlands that result from artificial water retention or converted wetlands that were subsequently returned to wetland conditions (16 USC § 3822). In prairie frontier states settled in the early nineteenth century, like Illinois, most wetlands were converted prior to 1985; this statute thus has little effect. Additionally, Swampbuster's prohibitions apply only to wetlands converted into cropland. Presumably, if the landowner converts the wetland into something other than cropland—pasture for instance—subsidies would not be withheld (Farrier, 1995). Furthermore, if this same piece of wetland is enrolled in the wetlands conservation reserve program, the landowner will be paid an additional subsidy for preserving the wetland, on top of the existing subsidies.

Sodbuster takes away payment supports, such as certain loans and contract payments, for crops produced on highly erodible land (16 U.S.C. §§ 3811-12). Because it is limited only to highly erodible lands, Sodbuster fails to conserve lands that might have high ecological value but that are not highly erodible (Farrier, 1995). Both of these statutes fail to foster any sense of stewardship in landowners. They are strictly economics-based statutes that compel behavior through the threat of decreased subsidies. There is no mention of the ecological value of the lands being preserved or the ecological benefits from conserving them. Furthermore, the statutes will not likely stop farmers from converting wetlands and highly erodible lands to cropland if the financial benefits of planting crops there outweigh the loss of the subsidies. From these two statutes, landowners gain only an understanding of how much money they can gain or lose in the short-term by choosing to conserve or plant crops in the wetlands or highly erodible areas on their property; they gain no underlying comprehension of the ecological value of the lands or the long-term losses incurred from farming these sensitive lands. Overall, both conservation reserve programs, the wetlands reserve program, sodbuster, and swampbuster are restrictive programs and, as such, fail to foster stewardship in the landscape (Farrier, 1995).

Private land uses that may impact water quality or wetlands are subject to the Federal Clean Water Act (CWA).[43] Section 404 of the Act protects certain wetlands by requiring landowners to obtain a permit from the Army Corps of Engineers before depositing dredge or fill material in wetlands and water bodies (33 U.S.C. §1344). The Act's nominal goal is to achieve no net loss of wetlands, either by denying requests to fill or by requiring mitigation (Farrier, 1995). It cannot and does not achieve that goal, however, because it does not protect wetlands lost to drainage; it protects them only when dredge or fill material is deposited in them. On the positive side, the dredge-and-fill permit requirements have been broadly interpreted to cover a wide range of activities that could threaten wetlands. For instance, courts have held that even moving soil from one location on a parcel of land to another within the proximity a wetland constitutes dredge and fill activities that require a permit (*Avoyelles Sportsmen's League, Inc. v. Marsh*, 715 F.2d 897, 923-24 (5th Cir. 1983)). Yet the Corps approves a large number of these permits annually. Indeed, out of the approximately 14,000 permits the Corps receives each year, only 500 are denied. The rest are withdrawn, granted, or approved without review as part of a category of requests that will result in minimal damage (Karkkainen, 1997 65). The wisdom of the Act's mitigation provisions is also questionable; many argue that instead of destroying a functional, existing wetland and recreating it elsewhere, it would be more ecologically sound to leave the functional wetland in place and send the development to another, better suited location (Farrier, 1995; Karkkainen, 1997).

Aside from wetlands protections, the CWA fails to establish effective measures for limiting private land uses that impact water quality. While pollution from point sources, such as factory effluent, are controlled under the NPDES[44] permit program, pollutants from non-point sources--erosion from croplands, fertilizers and pesticides, and runoff in urban areas--are generally not controlled (33 U.S.C. §§ 1313, 1342; Johnson, 2004). Nonpoint source pollutants can harm aquatic life through degraded water quality, clarity, and increased temperature.

[43] Because 75% of wetlands in the United States are located on private lands, the CWA is a prominent source of private land use regulation for conservation purposes (Karkkainen, 1997, p. 63).
[44] National Pollutant Discharge Elimination System (33 U.S.C. § 1342).

Additionally, nutrients that enter waterways from agricultural runoff can result in eutrophication, depriving aquatic organisms of oxygen (Ricklefs, 2007). Without enforced restrictions on nonpoint source pollutants, the CWA will remain largely ineffective at protecting aquatic ecosystems from a significant source of harm.

Though no federal statute governing land use currently exists, the federal government has previously considered whether it "should not only regulate, but also shape the planning of, private land use" (Wildermuth, 2005, p. 73). In its history, the federal government twice developed and nearly adopted federal land-use programs. During the New Deal, the Secretary of Agriculture, Henry Wallace, reorganized the USDA in a way that established the beginnings of national land-use planning, though it was confined to agricultural lands. "Under this scheme, each local community formed its own planning committee. These community committees then fed into a county committee. All county committees then fed into a single state planning committee, which reported to the USDA in Washington" (Wildermuth, 2005, p. 75). As part of the planning system, every county committee was required to develop a land-use map of the county, and all subsequent USDA decisions were required to be consistent with the appropriate land-use map. Four years after its initiation, however, the county planning[45] scheme was abandoned (Wildermuth, 2005).

The second effort at developing a national land-use planning program arose during the Nixon Administration in the early 1970s (Karkkainen, 1997; Wildermuth, 2005). Congress originated this land-planning effort in response to the rapid urbanization that was underway (Wildermuth, 2005). The proposed National Land Use Policy bill combined two separate bills introduced by Senator Henry Jackson and President Nixon. The combined bill sought to bring coordination and order to state land-use decisions and would have required states to review plans to build on sensitive lands and any project that would have region-wide environmental impacts. Parts of the bill were reviewed in Congress every year from 1971 through 1974, but no bill was ever passed (Wildermuth, 2005, p. 78). Despite these initial failures, it is pertinent to ask: Is a national land-use planning program something we should consider, particularly given the ecological challenges and increasing fragmentation our nation is facing? Is such a program a good idea for the future of American land-use planning? Despite their lack of success, these

[45] The planning program was also variously called "land use planning, cooperative planning, or agricultural planning" (Wildermuth, 2005, p. 75).

early attempts at federal land planning provide insight into future methods of land-use coordination among all states.

These various federal environmental statutes do provide conservation benefits; our predicament today would be worse without them. Their benefits, though, even considered in combination, are relatively modest compared with the needs of biodiversity. The land-use limits that they impose are not strong enough or broad enough to establish anything like a system of landscape-scale conservation of ecosystem function and functional corridors. The federal approach to land-use regulation is "piecemeal," with separate statutes for individual resources—endangered species, water, farmlands. In general, the statutes take a fragmented approach to conservation, perpetuating ecological fragmentation in the landscape. Many of the benefits are also specifically limited in duration. If wildlife is to thrive, particularly in fragmented landscapes and in the face of significant climate change, far more protection is needed.

State and Local Land-Use Regulation

Because land use activities necessarily occur within individual localities, land-use regulation has commonly been viewed as "quintessentially local" (Callies, Freilich, & Roberts, 2004, p. 19). Correspondingly, local governments[46], particularly municipal governments, are the primary source of private land-use regulation in the United States (Callies et al., 2004; Cribbet et al., 2002). While this localized system may seem practical, local governments commonly ignore the effects of their land-use decisions on neighboring localities, thus creating insular policies that favor local development while ignoring the larger landscape. This localism has been criticized for "contribut[ing] to the pervasive privatism that is the hallmark of contemporary American politics" (Briffault, as cited in Cribbet et al., 2002, p. 19).

The authority to regulate land uses lies in a government's police power, or inherent power to protect the health, safety, and welfare of the state's citizens (Powell § 79C.02). Because local governments lack inherent police powers, the state must delegate police powers to local governments and authorize them to regulate local land uses. States commonly use authorizing, or "enabling," legislation to accomplish this, but some use home rule or constitutional provisions (Breggin & George, 2003; Powell, 2009, § 79.03C[1][a]).

[46] The term "local governments" encompasses a variety of levels of government beneath within a state, including township, county, and municipalities. Commonly, all of these levels of government regulate land use to some extent.

Once granted authority, many local governments develop a comprehensive plan for land-use regulation. Commonly, a local government will establish a planning commission to develop a comprehensive plan using public input (McElfish, 2004). A comprehensive plan is "a written document that defines goals, objectives, and implementation strategies for the future growth and development of the jurisdiction" (McElfish, 2004, p. 31). The plan does not itself regulate local land use (McElfish, 2004). It essentially establishes the framework in which land-use decisions are made and guides the decision-making process for local land-use planning (Breggin & George, 2003; McElfish, 2004). States vary in the ways in which they authorize local planning. Some states grant localities optional authority to regulate land use. Most require a comprehensive plan if a local government chooses to regulate land use, while others make comprehensive plans optional. Some states require local governments to engage in regulation and mandate that localities adopt comprehensive plans (Breggin & George, 2003).

To implement the comprehensive plan and actually regulate land use activities, local governments use zoning ordinances (Breggin & George, 2003; Powell, 2009, § 79.01). Zoning ordinances divide the landscape into separate zoning districts, which often include such categories as residential, commercial, industrial, mixed use, agricultural, or conservation (Callies et al., 2004; McElfish, 2004). The districts specify the types of uses and the intensity of uses permitted in that area (McElfish, 2004). A zoning ordinance typically includes text that describes the districts and a map that lays out the location of the districts (Callies et al., 2004; McElfish, 2004). Most enabling laws require local governments to also provide for "variances," or a process for granting exclusions to the zoning ordinance in specific, individual instances of hardship (Callies et al., 2004). Despite their generally broad grant of authority to engage in zoning, local governments do not have absolute zoning power; their zoning regulations must be reasonable (Powell, 2009, § 79C.03[1][e]).

In addition to basic zoning districts, local governments can use overlay zones in their planning processes. Overlay zones are land-use regulations that apply on top of, or in addition to, the underlying regulations already in existence under the zoning ordinance. These zones can consist of parts of zoning districts or encompass several zoning districts. They enable local governments to supplement regulations to achieve particular land-use goals, such as ecological protections (McElfish, 2004).

State and local governments also regulate land use by acquiring land under their eminent domain power. Through eminent domain, a government may take a private owner's land, provided the government takes the land for a public use and compensates the private landowner for the value of the land (Callies et al., 2004; Powell, 2009, § 79.01F). If the government fails to meet one or both of these requirements, it will have violated the Fifth Amendment of the United States' Constitution, which prohibits the taking of private property "for public use, without just compensation."[47] The primary question in eminent domain cases is: what constitutes public use? Governmental cannot take property for purposes of conferring private benefits or under pretext of a private purpose (*Kelo v. City of New London*, 545 U.S. 469 (2005)). The Supreme Court has stated, however, that "private use" should not be narrowly construed to require that the property literally be open to and accessible by the public. To determine "public use," the Court uses the broader concept of "public purpose" and considers the public's needs, which can vary depending upon the circumstances (*Kelo*, 545 U.S. 479-480). The Supreme Court has given substantial deference to state and local governments in determining what constitutes a public purpose (Powell, 2009, § 79F.03).

Historically, the process of creating distinct zoning "districts" has led to urban sprawl and habitat fragmentation within the larger landscape. Over time, zoning ordinances generally pursued a planning scheme grounded in separation of uses, which meant that apartments could no longer be built above stores, as along the main streets of small towns in the early twentieth century. Clustering residences and commercial spaces into walkable, compact neighborhoods was no longer legal. The separation of uses resulted in urban landscapes that were spread out over greater distances, necessitating more roads, more infrastructure, and the use of automobiles to gain access to the different zones of the city. Other social and cultural forces also contributed to the spread of urbanization across the landscape, into the urban fringe and beyond. Zoning, however, made possible the vehicle-dependent, patchwork towns and cities with big-box dominated commercial areas and grid-like residential areas we see today. Given its significant culpability in the process of expansive urbanization and resulting fragmentation of the landscape, is zoning a worthwhile planning tool we should continue to use? Can traditional zoning be modified and used for conservation purposes or in a more ecologically-oriented manner? Are

[47] The prohibition against taking private land for public use without just compensation was applied to the states under the Fourteenth Amendment.

there other planning tools we can implement to create a more ecologically-minded planning process that will result in increased habitat conservation throughout urbanizing and urbanize areas?

Local Conservation Planning Tools:

State and local governments have developed a variety of tools to incorporate the ecological elements of the landscape into their planning methods and to provide room for biodiversity within their communities. The transition toward more environmentally-oriented planning correlates with the "Smart Growth" movement. Smart Growth has arisen from dissatisfaction with urban sprawl and a desire for new planning policies that preserve open space and decrease congestion. According to its proponents, Smart Growth "enhances a sense of community; . . . protects environmental quality" (Callies et al., 2004, p. 682).

Local governments can use general planning methods, including their comprehensive plans and zoning ordinances, to implement an overall conservation land-use strategy in their communities. They can adopt development-oriented tools that aim to mitigate the adverse impacts of new development on habitat and biodiversity. They can also implement tools to set aside and protect habitat within developed areas. While these tools have conservation potential, they are generally uncoordinated on the larger landscape-scale and insufficient to develop a broad system of linked corridors crossing multiple localities and regions.

The traditional, locally-based land-use regulation system described above can be updated to provide for ecological conservation, and many states and local governments have already begun to reshape local land-use into a more ecologically-based process. State governments have developed several methods of environmental planning that require habitat protections across the entire state landscape.

Acquiring and setting aside privately-owned land for strict conservation use through eminent domain is potentially the most effective and certain method for protecting habitat in the face of urbanization. Given the high cost of compensating all landowners whose lands were acquired, however, local governments are constrained financially from exercising their eminent domain power on a wide scale for habitat protection.

Although land-use regulation is essentially a local activity performed by local governments, states can establish a statewide land-use system, regulating land-use activities on the scale of the entire state. Regulation on a statewide scale may provide a promising alternative for implementing landscape-scale ecological protections through land-use planning laws. Hawaii is one state that has implemented such a system (Callies et al., 2004). Hawaii's state land-use law establishes a state land-use commission and divides the state into four overarching land-use districts consisting of urban, rural, agricultural, and conservation (HAW. REV. STAT. § 205 et seq.). Permissible land-use activities are based on district. The law gives county governments zoning authority within the districts, except for the conservation district. Zoning within that district is entrusted to the state's department of land and natural resources. The law also specifies the criteria by which the land-use commission should make its planning decisions. The criteria include a consideration of the impact any changes in the land-use system will have on the "preservation or maintenance of important natural systems or habitats" (§ 205-17). This system therefore provides an ecological perspective and lays a foundation for planning on the landscape scale.

States have also adopted growth-management laws that specifically require all local governments to consider the ecological impacts of land-use regulations. These laws are specifically directed at managing urban growth within the state to protect natural resources. On the state level, growth management laws can establish state conservation goals and policies and direct administrative agencies to take particular conservation-related actions (Breggin & George, 2003). The Maine growth-management law requires the state commissioner of conservation to develop a register of critical areas, for instance (Breggin & George, 2003). The laws can also provide for state land acquisition to protect important natural resources. On the local level, state growth-management laws can establish goals and priorities for local comprehensive plans. Generally, the laws require local governments to incorporate conservation and open space provisions in their comprehensive plans (Breggin & George, 2003). Growth-management laws are useful ways to establish conservation requirements statewide and apply them to all local governments. They create a method to conserve resources on a broader scale and highlight the importance of natural resources to the state. Because they are aspirational and goal-oriented, however, they may fail to enact real conservation gains on the ground.

If states choose not to adopt statewide conservation planning or growth-management laws, they can explicitly require, or at least authorize, local governments to provide for biodiversity protection in their planning decisions through state enabling laws (Breggin &

George, 2003; Powell, 2009, § 79.03[2][c][vi]). A state may require local governments to designate areas for biodiversity protection, or to develop specific plans or policies that provide for protection (Breggin & George, 2003). The requirements vary by state. For instance, Michigan requires that some or all local governments consider habitat and natural resources in their comprehensive plans. Illinois, on the other hand, does not require local governments to consider natural resources, open space, sensitive areas, or general environmental planning in their comprehensive plans (Breggin & George, 2003). Even if a state does not specifically provide for protections in an enabling statute, a local government may have inherent authority to establish biodiversity protections (Breggin & George, 2003). Alternatively, the states themselves can develop conservation plans that act as templates for local governments to follow. Ohio, for instance, requires a statewide plan (Breggin & George, 2003).

Whether or not state growth-management laws or enabling statutes explicitly require conservation goals and requirements, local governments have the ability to establish these measures in their comprehensive plans. Within its comprehensive plan, a local government can specify community conservation goals and identify sensitive land areas and important natural resources that need protection (McElfish, 2004). The plan can also identify contiguous areas of land and connections between the contiguous areas and other parcels of open space, highlighting these areas for protections or more rigorous land-use limitations. Zoning regulations then will have to conform to the conservation goals and protections established in the plan. In lieu of incorporating environmental protections into their comprehensive plans, local governments can establish a separate, "stand-alone" environmental plan (Berke, 2009). The effectiveness of the different plan types depends upon the local situation. Berke (2009) has noted, however, that communities generally only incorporate ecological protections into plans after environmental catastrophe has struck (Berke, 2009, p. 413). Conservation planning is also limited due to weak commitment to action, which has stymied implementation of other conservation plans (Berke, 2009). Furthermore, because environmental goals are developed to meet narrow local interests, the goals of different communities can be inconsistent and even conflicting.

In order to establish ecological protections through state and local comprehensive plans or growth-management laws, governments must take the necessary action to incorporate the protections or conservation goals into existing plans or create new conservation plans. As explained above, many state and local governments have failed to take such action and lack conservation plans or provisions. Furthermore, many lack the political will to enact such conservation goals. This failure on the part of multiple levels of government to identify conservation issues among broad planning goals hampers the development of further action, such as enactment of regulations or identification of sensitive lands, to implement a system of conservation-minded land-use regulation.

Tools to Guide Development Toward Conservation

Local governments have developed a variety of regulatory tools that control development and land uses with the intent of preserving habitat, natural resources, and general open space. These tools consist of zoning-based, voluntary, and economic approaches. Zoning-based techniques include an array ordinances and regulations that aim to limit the spread of urbanization, such as Planned Unit Developments. Voluntary approaches consist of land-use restrictions that landowners elect to implement on their property. One of the most common voluntary approaches is the conservation easement. Economic approaches include programs that pay landowners in some form in return for their relinquishment of future land use rights, such as the ability to develop. While each of these tools produces some conservation gain, they generally fail to establish the coordinated, landscape scale system of ecological protections necessary to implement corridor networks and protect biodiversity. There are many variants of the regulatory tools within these three categories. This chapter surveys a subset of these methods as representative examples.

Basic zoning techniques provide sufficient flexibility and structure to be used for conservation purposes. Local governments have adapted a variety of zoning methods to implement limits on development, require more open space, and even establish the foundations for a broad scale system green infrastructure. Through restrictions on "uses, density, and design," zoning can determine the impact of particular uses on the ecology of the landscape (McElfish, 2004, p. 39). Therefore, by restricting where development can go and how much development can occur through zoning, local governments can mitigate otherwise adverse impacts on the environment or prevent certain intensive uses completely. Local governments can also create separate conservation districts to protect open space, including forests and wetlands, or sensitive areas, such as steep slopes and coastal zones (McElfish, 2004). The town of Beverly Shores, Indiana, used a zoning ordinance to restrict development activities that might adversely affect the sand dunes along the lakeshore (McElfish, 2004). The town of Washington, New York, revised its residential zoning ordinance to increase density requirements in order to protect open space from fragmented development patterns (McElfish, 2004).

Overlay zones provide a way for local governments to integrate environmental protections into existing land-use regulations. Environmental protection overlay zones enable local governments to impose different levels of environmental protection on different zoning districts, targeting the areas that need the greatest protection (Callies et al., 2004). Local governments can adopt overlay zones to indicate sensitive areas or natural aspects of a particular area, such as steep slopes or certain soil types, which might render those areas unsuitable for the use specified in the underlying district (Callies et al., 2004). Through overlay zones, local governments can also identify and protect contiguous habitat that stretches across several districts (McElfish, 2004).

Other techniques have revised and evolved fundamental zoning concepts to require open space protections from new development. Cluster zoning protects open space by requiring developers to concentrate new development within a subset of the landscape in which development will occur (Benedict & McMahon, 2006; McElfish, 2004). The remaining open space receives permanent protection status. Between fifty and ninety percent of a landscape is generally left in open space (Benedict & McMahon, 2006, p. 165). Farmview, a development of 310 homes in Bucks County, Pennsylvania, preserves over 200 acres of a 431-acre landscape in agricultural fields and woodlands (Benedict & McMahon, 2006, p. 165). This technique benefits biodiversity because it enables a local government or developer to identify areas of a landscape that are ecologically important and to organize development around preserving those areas (McElfish, 2004). By preserving the aesthetic beauty of an area through open space protection, cluster zoning also benefits developers and residents because it often produces higher economic gains and higher home values (Benedict & McMahon, 2006; McElfish, 2004). Cluster zoning provides a practical method by using zoning to establish a balance between development and conservation.

The Planned Unit Development (PUD) is another, broader zoning-based tool that also balances the need for development with the need for conservation. It allows for more flexible and comprehensive development than zoning typically provides (Callies et al., 2004; McElfish, 2004; Steiner, 2008). The PUD consists of planning an entire development as a single unit and allows a range different uses in the development (Callies et al., 2004). Unlike traditional zoning, which focuses on separation of use, the focus of the PUD is intensity of use (Powell, 2009, § 53B.01). Usually PUDs include a mix of residential, commercial, industrial, institutional, and recreational uses. They generally use clustering to organize the uses and provide for open space (Powell, 2009, § 53B.01[2]; Steiner, 2008). A PUD gives the local government more oversight and control over an entire development, allowing the local government to determine which lands are appropriate to develop and in what ways (Callies et al., 2004). Developers are able to gain flexibility in planning by altering the use and density requirements for the tract of land in the underlying zoning ordinance (McElfish, 2004). PUD regulations are either included in the zoning ordinance or in other separate ordinances, such as the subdivision control ordinance (McElfish, 2004; Steiner, 2008). Like zoning, the state may authorize local governments to use PUDs through legislation[48] (Powell, 2009, § 53B.01[4]).

Because PUDs provide a comprehensive approach to planning—at least at the scale of the development--local governments find them useful tools for conserving habitat and biodiversity. Local governments can issue conservation requirements within the PUD ordinance (McElfish, 2004). Developers can in turn place land-use restrictions on the development, requiring the landowners to preserve natural elements of the landscape[49] (Steiner, 2008). The PUD can help protect sensitive areas such as steep slopes and wetlands, structuring the development around them. This ability to set aside certain lands is one of the benefits of planning a large tract of land as a whole (McElfish, 2004). This technique allows local governments to work with developers to establish environmental protections, as opposed to imposing restrictions upon developers that the developers will likely challenge.

General conservation zoning tools, including specific conservation zones and overlay zones, are readily available, useful ways for local governments to balance land use with conservation, but the scope of their impact is limited. They apply on the local scale and may not coordinate with the conservation protections in neighboring jurisdictions. If other municipalities fail to implement their own conservation oriented regulations, the open spaces that one municipality protects can easily be surrounded by traditional development. A lack of

---

[48] According to Powell (2009), states have authorized the use of PUDs. It is not clear, however, whether states must authorize local governments to use PUDs.

[49] Homeowners' associations often assume management of the restrictions once the development is complete (Steiner, 2008, p. 334).

coordination among municipal zoning patterns will thus perpetuate the problem of isolated and fragmented habitat. Because zoning laws are subject to change based on political pressures within local government, the conservation gains they achieve are tenuous (Benedict & McMahon, 2006). Local governments will also commonly lack the political will to implement and enforce these measures. Zoning ordinances provide for variances, and the variances in these cases could end up swallowing the conservation regulations (Benedict & McMahon, 2006). Additionally, overlay zones may violate the requirement in the Standard Zoning Enabling Act[50] that all regulations be uniform throughout each zoning district, though some courts give local governments considerable discretion (Callies et al., 2004).

Cluster zoning and PUDs provide for the protection of more open space than would be protected under traditional zoning methods, and they provide a useful means of balancing development with conservation. Despite their benefits, they are piecemeal approaches, proceeding development by development, not on broader landscape scales. It is not clear whether later developments are required to consider earlier developments in their allocation of open space to create links between the open spaces among multiple developments. Additionally, both of these techniques are only applicable to new development; they are unable to remedy the current lack of open space in already-developed areas (Benedict & McMahon, 2006; Steiner, 2008).

The definition of acceptable "open space" that cluster zoning or PUD regulations adopt can be detrimentally broad, encompassing areas that have little or no ecological value. The PUD regulation for the city of Bellevue, Washington, for example, includes sidewalks, gardens, and lawns in its definition of open space (McElfish, 2004). Many examples of cluster zoning preserve agricultural lands, including croplands, as open space (Benedict & McMahon, 2006; McElfish, 2004). Agricultural lands, however, are often intensively farmed and can function as barriers to wildlife dispersal. By preserving open space, these land-use tools diminish the impact of new development on biodiversity by reducing fragmentation, but they still insert human disturbances into the preserved open space areas. Generally, the human inhabitants of the developments are able to use the open spaces for recreation, gardening, or other uses. The expansion of human occupancy into protected areas could harm species that are intolerant of

---

[50] The U.S. Department of Commerce issued a draft of the Act in the 1920's, and states adopted it as their own zoning enabling law governing local government zoning methods (Callies et al., 2004).

human contact and could introduce invasive species. Instead of continuing the trend of new development, local governments should consider increased use of infill development and higher densities. Instead of spreading out, we can filter in and up. While they provide a useful foundation for future conservation efforts, cluster zoning and PUD techniques alone are insufficient techniques to establish corridor networks on a landscape scale.

Economic regulatory tools generally provide compensation to landowners in exchange for restrictions on future development. To channel development to certain areas and preserve others, local governments have developed Transfer of Development Rights (TDR) programs. Development rights are viewed as one “stick” in the “bundle” that represents property rights. Under TDR programs, a local government or some other management entity places development restrictions on a particular area, and to compensate the landowners for the loss of the future development rights, the owner can sell, or transfer, those rights to a developer in the area designated for development.[51] In other words, certain landowners give up their development rights “stick” in return for a transferable right, a theoretical piece of paper, that he or she can sell to developers in the development area.[52] The development rights are “sold in a *sending* or *preservation* zone to be used in a *receiving* or *development* zone” (Steiner, 2008, p. 363). TDR programs attempt to balance the benefits and burdens of environmental regulation by preventing the landowners who retain development rights from gaining a windfall while other landowners lose those rights, simply because zoning changed (Steiner, 2008).

These programs effectively limit development in ecologically sensitive areas and concentrate it in designated areas, preserving open land and its related ecosystem functions (McElfish, 2004). Through restrictions on development in major portions of the landscape, TDRs can effectively preserve large areas of contiguous habitat and maintain natural disturbance patterns (McElfish, 2004). TDRs therefore may constitute one viable way to facilitate conservation planning on a wider, regional scale.

TDRs are complex programs that require substantial organization and management to maintain, however. Because of this complexity, governments may find them logically infeasible to implement. TDRs are generally best suited to situations involving the protection of large areas

---

[51] Similarly, in Purchase of Development Rights (PDR) programs, local governments purchase the development rights directly from the landowners, either to supplement TDR programs or as a method of direct governmental acquisition to prevent future development (McElfish, 2004; Benedict & McMahon, 2006).

[52] The landowner also records a deed restriction to ensure the restrictions remain attached to the property in the future.

where the amount of developable land is limited (McElfish, 2004). These programs therefore require substantial coordination among government entities because of their large, interjurisdictional, scale and many organizational demands. For instance, governments will need to collectively establish the program's details, including where to locate sending and receiving areas (Benedict & McMahon, 2006). The surrounding community must also participate to ensure the program is widely accepted (Benedict & McMahon, 2006). Furthermore, TDRs are more easily implemented where the sending areas maintain multiple economically viable uses, as landowners in the sending areas are more likely to accept the program if they are still able to use their lands for some economic purpose (McElfish, 2004). The scope of retained economically viable uses, however, may include some land uses that are incompatible with habitat protections and the maintenance of healthy ecosystem function, such as intensive farming methods. TDR programs may successfully achieve conservation goals when designed and implemented well, but the substantial coordination, planning, and organization required to implement and maintain them limits their practical applicability. Finally and not insignificantly, TDR systems can achieve substantial protection only if implemented at an early stage, before a landscape is substantially developed. If a local government waits too long, after most of its area has been developed, there is a little a TDR program can accomplish.

Landowners may also voluntarily adopt land-use restrictions to protect habitat on their property, often with the help of government or a non-governmental conservation organization. Conservation easements are a widely-used method for establishing these voluntary restrictions. The Uniform Conservation Easement Act[53] defines a conservation easement as:

> a nonpossessory interest of a holder in real property imposing limitations or affirmative obligations the purposes of which include retaining or protecting natural, scenic, or open-space values of real property, assuring its availability for agricultural, forest, recreational, or open-space use, protecting natural resources, maintaining or enhancing air or water quality (§ 1(1)).

This definition means that conservation easements allow the holder of the easement, often a government entity or a non-governmental organization, the ability to prohibit certain bad land uses or compel certain stewardship actions. Although his use rights are limited, the landowner

[53] This is a model act that was developed by the National Council of Commissioners on Uniform State Laws, a non-governmental organization, in 1982.

subject to the easement maintains ownership in the underlying land (Benedict & McMahon, 2006). Most conservation easements are perpetual; the easement also binds subsequent landowners who attain title to the land (McLaughlin, 2005; Powell, 2009, § 34A.01). States must enact a statute to facilitate the creation of conservation easements and to override countervailing common law precedent that would otherwise invalidate them. Sixteen states and the District of Columbia have adopted the Uniform Conservation Easement Act, and thirty-three others have adopted their own statutes (Farrier, 1995, p. 343 n. 180; McLaughlin, 2005).

Conservation easements are considered easements "in gross"—an interest in a parcel of land that is not tied to ownership of another parcel that would benefit from the easement[54] (Cribbet et al., 2002; Powell, 2009, § 34A.01). This characteristic enables land trusts or governments to hold title to the easement based on their conservation interest, without the need to own an adjacent parcel of land (Farrier, 1995). As explained previously, the U.S. government uses conservation easements to protect wetlands and sensitive lands in its Conservation Reserve and Wetlands Reserve programs. Hundreds of national and local non-governmental conservation organizations and land trusts are actively protecting private lands through conservation easements (Farrier, 1995). Perhaps the largest land trust organization, the Nature Conservancy is an international organization and is active in all 50 U.S. states, often through affiliations with local land trusts. The organization asserts that it has protected over 119 million acres of land, focusing on the conservation of high-priority landscapes (Nature Conservancy, 2010).

Given their wide use across the United States, conservation easements are a potentially effective method of establishing landscape scale networks of protected habitat, if implemented in a coordinated manner across jurisdictional boundaries. They are flexible conservation tools that can be adapted to suit conservation needs in different ecological conditions (Benedict & McMahon, 2006). They have the capacity to promote stewardship on the part of the landowner subject to the easement, particularly through affirmative conservation or restoration obligations (Echeverria, 2005). By involving landowners in large-scale habitat conservation efforts,

[54] Typically, many easements are "appurtenant," or related to the use of another parcel of land. The holder of an appurtenant easement generally owns an adjacent or nearby parcel of land, and the easement facilitates the landowner's use of his own land, by granting a path for access, for instance, if the landowner's parcel is landlocked (Cribbet et al., 2002).

conservation easements can give them a stake in those efforts and may prompt landowners to develop other, innovative conservation strategies (Echeverria, 2005).

The voluntary nature of conservation easements and the methods in which they are often implemented, however, limit their utility as a tool for establishing networks of conservation corridors. The easements are generally created on an ad-hoc, reactive basis, often by several different land trusts, and often for widely different purposes. These implementation methods inhibit the integrated planning needed to link protected areas into corridors (Farrier, 1995). Many of the easement holders also fail to provide for continued monitoring and management to ensure that restrictions are enforced and conservation goals are realized. Because they are voluntary, one or several landowners who refuse to participate in easement programs could thwart the entire effort, leaving gaps in what would be a cohesive system of protected areas[55] (Farrier, 1995). Subsequent landowners may also resist the easement restrictions and attempt to challenge them in court as no longer practical or relevant (Farrier, 1995).

The purpose and function of a conservation easement could also undermine the quality of the habitat protections it might otherwise establish. In an attempt to balance land uses with conservation, the easement agreement could permit a wide range of uses within the protected area that may impair its habitat value. Rissman et al. (2007) surveyed 119 easements in eight states held by the Nature Conservancy. They found that while ninety-eight percent of the easements addressed development or fragmentation threats, eighty-five percent of the easements allowed some degree of development, including residences, commercial development, and even subdivision. Furthermore, they found that the Nature Conservancy did not specifically reserve the right of continued monitoring and maintenance in the terms of most of the easement agreements (Rissman et al., 2007). This data reveals that a substantial percentage of the conservation easements surveyed failed to limit private land uses that could adversely impact the ecological condition of the protected land. If this data is extrapolated to all conservation easements, it draws into question their true effectiveness as a viable tool to protect biodiversity at the landscape scale.

Essentially, Rissman et al.'s findings indicate that the ecological protections that conservation easements provide are as good as the terms of the agreement. Securing open land

[55] With respect to the ecological implications of landowner refusal, these gaps in a landscape containing otherwise connected and protected habitat areas may not affect the dispersal of some species, but for others, they could function as barriers, perpetuating fragmentation. These ecological ramifications are discussed in previous chapters.

from development is an important goal, but easement holders must also specify limits on land uses that could adversely impact the ecological functions of the land under easement and provide for ongoing management. While conservation easements are effective tools to protect open land from development, individual landowner resistance and lax development restrictions limit this voluntary conservation land-use tool's potential to secure adequate landscape scale habitat protections to ultimately establish networks of conservation corridors.

Challenges to Implementing Conservation Measures

At least some landowners will inevitably challenge conservation-based land-use regulations that restrict the uses of their property. Landowners often conceive of environmental land-use regulations as unfair because they think they are singled out to bear the conservation burden of the community at large. Also, the costs of conservation are localized to the landowner, but the benefits spread throughout the community, with the landowner seeing a small proportion of the benefit relative to the cost (Karkkainen, 1997). Landowners use a variety of methods to challenge land-use limitations, including suing the local or state government in court and taking initiative as citizens to restrict the scope of the government's regulatory ability.

One primary way in which landowners attempt to strike down a land-use regulation is to challenge the offending regulation as an unconstitutional taking of the property, in violation of the Fifth Amendment. As discussed above with respect to eminent domain, the Fifth Amendment prohibits the taking of private property "for public use, without just compensation." Two main types of takings can occur. The first is a direct, physical taking, or a "per se" taking, in which the government appropriates the entire parcel or a portion of it through eminent domain or physically enters and uses a portion or all of the property (Callies et al., 2004; Powell, 2009, § 79F.04). The Supreme Court has held that any "permanent physical occupation of property" is a taking (*Loretto v. Teleprompter Manhattan CATV Corp.*, 458 U.S. 419 (1982)). Such a physical occupation includes the installation of cable boxes on the roof of an apartment building. Physical takings have also been found where private landowners are forced to allow the public to access a natural feature of the property, such as a lagoon (*Kaiser Aetna v. U.S.*, 444 U.S. 164 (1979)).

Because the landowner will be challenging state or local government regulations on the use of his land, he will most likely challenge that regulation as a regulatory taking, the second type of taking. The Supreme Court cast an ominous warning that when a government regulation "goes too far" it will constitute a taking (*Pennsylvania Coal v. Mahon*, 260 U.S. 393 (1922); Powell, 2009, § 79F.05). In such a situation, the "government is deemed to have 'implicitly' exercised its power of eminent domain" (Powell, 2009, § 79F.01). In other words, a regulatory taking occurs when a regulation is so severely restrictive as to function like a like physical occupation of the property when applied. The determination of when a regulatory taking has occurred is complex, "a problem of considerable difficulty," with various factors and elements (*Penn Central Transportation Co. v. New York City*, 438 U.S. 104 (1978)).

Regulatory takings can fall into one of two classes. The first is a categorical or "per se" taking of the entire parcel. This type occurs when the regulation as applied deprives the landowner of any "economically beneficial or productive use" of the property (*Lucas v. South Carolina Coastal Council*, 505 U.S. 1003 (1992); Powell, 2009, § 79F.05(b)(ii)). In other words, when a regulation prevents the landowner from making any economically-based use of the property, it has the same effect as physically taking the parcel of property from the owner and constitutes a total taking (Powell, 2009, § 79F.05). In *Lucas v. South Carolina Coastal Council*, the Supreme Court found a categorical taking where a South Carolina statute aimed at protecting natural coastal areas prevented Lucas from building upon, or making any "economically beneficial use" of the property (*Lucas*, 505 U.S. 1003).

If the regulation affects less than a "complete elimination of value" and does not qualify as a categorical taking, courts determine whether it qualifies as a taking requiring "just compensation" on an ad hoc, case by case basis, focused on the facts of each situation (*Tahoe-Sierra Preservation Council, Inc. v. Tahoe Regional Planning Agency*, 535 U.S. 302 (2002)). In *Penn Central* and subsequent cases, the Supreme Court developed a multi-factor test to assess whether a regulation constitutes a taking (438 U.S. 104). The factors include the "economic impact of the regulation," the "character of the government action," and the "interference with reasonable investment-backed expectations" (*Kaiser Aetna v. U.S.*, 444 U.S. 164, 175 (1979); Powell, 2009, § 79F.05). If the purpose of the regulation is to prevent harm to the public health, safety, or welfare, it is not considered a taking. If the regulation instead confers a benefit to the public at the expense of a single landowner, it will be considered a taking. The theory underlying this "harm/benefit test" is that a select few individuals should not have to bear "public burdens, which in all fairness and justice should be borne by the public as a whole"

(*Penn Central,* 438 U.S. 104, 123 (1978)). This test is rather ambiguous, however, as certain regulations could be deemed to be both conferring a benefit and preventing a harm (Freyfogle, 2007b).

When assessing whether a taking has occurred, the Supreme Court has made clear that it evaluates the impact of the regulation on the “parcel as a whole,” as opposed to its impact on individual portions of the parcel. To analogize this rule to the bundle of sticks approach, "where an owner possesses a full 'bundle' of property rights, the destruction of one 'strand' of the bundle is not a taking" (*Tahoe-Sierra Preservation Council, Inc. v. Tahoe Regional Planning Agency,* 535 U.S. 302, 327 (2002)). In *Palazzolo v. Rhode Island*, the Supreme Court held that the state’s denial of Palazzolo’s permit to fill in wetlands on his property was not a taking when evaluated through the whole parcel rule (533 U.S. 606 (2001)). On remand, the Rhode Island state court held that the denial of the permit prevented a public nuisance and therefore was not a taking, as it fell under the nuisance exemption (Nagle, 2008).

A taking could also potentially occur when a regulation temporarily prevents a landowner from using or developing her property. If a court determines that a temporary regulation constitutes a taking, using the tests described above, the governmental entity responsible for the regulation must pay the affected landowner damages as just compensation for the amount of time the regulation was in place and applied to the landowner’s property (*First English Evangelical Lutheran Church v. County of Los Angeles*, 482 U.S. 304 (1987); Powell, 2009, § 79F.05). However, the highest courts in New York and California have held that no compensation is due for temporary takings (Powell, 2009, § 79F.05). In *Tahoe Sierra Preservation Council, Inc. v. Tahoe Regional Planning Agency,* Lake Tahoe area landowners (the Preservation Council) challenged the Agency’s implementation of a temporary moratorium on development in the Lake Tahoe region during which it created a development plan to protect Lake Tahoe. The Supreme Court held that the moratorium did not constitute a *per se* taking and should be analyzed under the Penn Central test[56] (535 U.S. 302 (2002)). According to the Court, “a fee simple estate cannot be rendered valueless by a temporary prohibition on economic use, because the property will recover value as soon as the prohibition is lifted” (535 U.S. at 332). The Court noted, however, that if the plaintiffs had challenged the moratorium as applied to their particular parcels, it might have constituted a taking. Whether permanent or temporary, if a regulation is

[56] The Court also validated the whole parcel rule, declining to view rights in the parcel in severed time segments.

deemed a taking, the government must compensate the landowner. However, whether a taking exists depends intensely on the factual circumstances and the extent of the regulations impact on the landowner's rights.

Notably, regulations targeted to prevent public nuisances are held to be exceptions to the takings prohibition, even if the regulation substantially restricts use of the property (Powell, 2009, § 79F.05(4)(a)(iv)). The exception for prevention of harm discussed above encompasses nuisance-prevention regulations (Powell, 2009, § 79F.05). In *Lucas*, the Supreme Court maintained an exception for "background principles of [a] state's law of property and nuisance" (*Lucas*, 505 U.S. at 1029). The Supreme Court of Rhode Island also employed the nuisance exception in Palazzolo, holding that the state's denial of Palazzolo's permit to fill wetlands was an action to prevent a public nuisance (*Palazzolo v. State*, No. WM 88-0297, 2005 R.I. Super. LEXIS 108 (July 5, 2005)). This nuisance exception is an important tool for combating takings challenges to future environmental regulations that function to abate public nuisances on the ecosystem level.

Courts employ a more intensive takings analysis for regulations that place conditions on development approval, known as regulatory exactions (Powell, 2009, § 79F.05(4)(e)(i)). First, there must be an "essential nexus" between the regulation used and the public purpose the government was acting under (*Nollan v. California Coastal Commission*, 483 U.S. 825, 834 (1987)). Essentially, this essential nexus requirement attempts to ensure that the local government has a legitimate state interest behind itse actions and is not acting arbitrarily. Second, there must exist a "rough proportionality" between the exaction and the anticipated impact of the development (*Dolan v. City of Tigard*, 512 U.S. 374 (1994)). The heightened judicial scrutiny that these tests impose on development exactions may pose a barrier to procuring specific environmental protections from new developments (Karkkainen, 1997). However, the reach of the regulatory exactions test developed in *Nollan* and *Dolan* is limited. The Court has subsequently declined to broaden the scope of the test and has limited its application to cases involving exactions (Powell, 2009, § 79F.05(4)(e)(iii)).

In addition to takings challenges, there are also social and doctrinal obstacles to enacting conservation regulations and laws. The Property Rights Movement is one such obstacle (Karkkainen, 1997). The movement is grounded in the concept of individual liberty. It asserts the idea that property rights are "individual rights existing in the abstract, like free speech," and regulations that attempt to limit private land uses unlawfully infringe upon those inalienable rights inherent to property ownership (Freyfogle, 2007b). Proponents of the movement claim that development rights are overly restricted and land-use laws single out landowners for restrictions and ruin expectations for future land uses (Freyfogle, 2007b). According to Blumm and Grafe (2007), an unspoken view of the libertarian property rights movement is that "property rights equate to development rights, and that regulation . . . limiting a landowner's right to develop is impermissible without constitutionally required compensation" (p. 283). Clearly, the rhetoric of the movement lacks legal substance to support it. Even abstract rights like free speech are limited in some respects; fraudulent statements, for instance, are not protected under the first amendment. As explained above, private property rights are "created by the state" and "depend upon state enforcement" (Blumm & Grafe, 2007, p. 283). Property rights are only made possible by the existence of laws to protect property and, as a result, are subject to changes in the law.

Despite its lack of substantive validity, this movement has taken hold in many states and will likely generate much opposition to limitations of private property rights for environmental protection. It has already spawned efforts to oppose regulation in Oregon and Arizona through referenda. In Oregon, state citizens adopted Measure 37, later amended by Measure 49, which requires state and local governments to compensate landowners for any diminution in property value that results from any land-use regulation enacted after the landowner gained ownership of the property (Blumm & Grafe, 2007; OR. REV. STAT. § 197.352). The Measure allows governments to waive challenged restrictions instead of compensating landowners. It also includes two significant exceptions. It exempts regulations that prevent public nuisances and regulations that protect the public health and safety (Blumm & Grafe, 2007). The scope of the Measure and its specific applications are still uncertain (Blumm & Grafe, 2007). The impact of Measure 37, therefore, remains to be determined.

Efforts like Measure 37 that restrict state and local government ability to regulate land use could end up making environmental regulation costly and difficult. Such measures could result in the waiver of many regulations, perpetuating land uses that fragment and degrade habitat and natural resources. State and local governments could challenge such measures in court as unconstitutional. The Oregon Supreme Court, however, found that Measure 37 did not violate the state or federal constitutions. Governments could resort to using voluntary conservation efforts, such as reserve programs or conservation easements, but if citizens are so adverse to regulation that they voted to pass Measure 37, they will likely be unreceptive to adopting land-use restrictions voluntarily. Potentially the best defenses are to find and exploit ambiguities and logical holes in the measures and to advocate for narrow judicial interpretation of the measure's scope.

This chapter has surveyed some of the major federal, state, and local land-use laws and regulations that establish conservation protections in landscapes dominated by private landownership. Some of the various approaches include direct protections of endangered or threatened species, protections for wetlands through easements and restrictions on fill activities, preservation of contiguous open space through development restrictions, and protection of sensitive or ecologically significant land through government acquisition, easements, or restrictions on development rights. These various methods of land-use governance are for the most part uncoordinated, piecemeal efforts on small spatial scales, are short-term, and are subject to subsequent change or reversal. Moreover, they fail to compel stewardship on the part of landowners, the lack of which is a primary underlying cause of bad land use. The conservation measures that governments do pass are also vulnerable to being invalidated upon takings challenges or referenda initiated by recalcitrant landowners. While these laws and regulations may preserve open space in some locations or protect some sensitive lands from immediate cultivation or development, collectively, they are insufficient to establish the landscape scale system of conservation corridor networks necessary to provide for long-term biodiversity conservation.

How can governments create a land-use system that establishes conservation goals, implements effective ecological protections, and maintains those protections against landowner challenges? The following chapter offers some potential methods for achieving such a system.

VI. From Land *Use* to Land *Stewardship*: How to Establish Corridor Networks by Updating Land Use Laws and Regulations

We have already considered what basic actions are needed to conserve biodiversity within an increasingly fragmented and developed landscape. We must preserve as much remaining habitat as is feasible and set aside lands for corridors; in addition, we must stop bad land uses that are degrading ecosystems, curtail future detrimental land uses, and promote affirmative restoration. The challenge is to determine how to take these major steps, in ways that promote sound conservation, through well-designed planning requirements, common law changes, land-use laws and regulations, and other means. The previous chapter explains why our current legal regime is insufficient for achieving sound conservation goals: Our law fail to provide for planning across political boundaries, on a landscape scale; they take narrow, ad hoc approaches to conservation, and they are implemented in an uncoordinated manner at fragmented spatial scales. Are we simply doomed to poor land use? What kind of system might enable us to implement and ultimately achieve our ecological goals?

Wood (2009a) has asserted that we need a "revolutionary legal approach assuring natural resources protection and restoration" (p. 54). Her call for extensive change is warranted, but instead of a revolutionary *new* approach to land use, this paper argues that we need a land-use law revolution from *within* the system, brought about by a careful probing and retooling of the current system. We already possess the basic elements for a successful conservation-oriented land-use system; we simply need to reformulate and update these elements and use them more effectively.

This chapter identifies some of the important issues that will arise as we undertake this revolution. We first need to consider the type of approach—bottom-up or top-down—and settle upon an effective governance structure, deciding who is responsible for incorporating ecological considerations into land-use planning law and how should it be done? We also need to consider the types of legal tools that can help achieve our ecological goals effectively, tools that facilitate planning and then action across multiple spatial scales and jurisdictional boundaries. Finally, we need to anticipate and respond to the inevitable challenges to this land-use law revolution. Essentially and working from within, this revolution must transform our system of land *use* law into a system of land *stewardship* law.

A post-revolution system of land stewardship law should embrace a holistic, comprehensive approach to land use on the landscape scale. It should consist of a cascading, inter-jurisdictional approach that bridges spatial scales as well as physical and political boundaries. Most significantly, this rehabbed system of land-use law must be grounded in the ecological aspects of the landscape and its physical features and functions, and must compel stewardship on the part of private landowners as well as public land managers. The elements of such a system exist in basic form; our task is to revitalize them, help them to evolve, and bring them together into a coordinated, landscape-scale effort.

This chapter examines some of the elements that can help forge such an ecologically-oriented law of land stewardship. It identifies the key issues that any reform effort will encounter and suggests avenues for improvement. In particular, the chapter explores (i) approaches to governance, (ii) the appropriate roles various levels of government should assume, (iii) the benefits of using regulatory methods of governance, (iv) ways to use existing land-use tools to achieve conservation-based land uses, and (v) how to uphold and enforce the system in the face of resistance.

Development of a land stewardship law system

As noted already, to implement a network of biodiversity corridors successfully we need to reformulate our existing system of land-use law into one that is more holistic and ecologically based, that compels stewardship, and that limits bad land uses. How, though, do we go about this process? What are some of the key elements of this land stewardship law system?

An initial question relates to the basic approach used to develop the rules, laws, and regulations that will make up the system. Should we through law encourage local landowners to coordinate their efforts on their own, perhaps even adopting binding regulations, or should we rely instead on governments to craft, orchestrate and implement appropriate regulations? Put otherwise, do we leave the development and implementation of corridor networks to the landowners themselves, encouraged by government efforts, or do we instead give direct responsibility to some level or levels of government, which will then tell landowners what to do?

These questions highlight the two basic approaches that are possible: a top-down and a bottom-up approach. In a top-down approach, a government enacts and enforces laws, regulations, and other governing mandates (Bayne, 2009). A local government, for instance, could adopt a regulation requiring riparian landowners to leave a fifty-foot vegetated buffer on both sides of a local stream. All affected landowners would be expected to comply and perhaps penalized if they do not. When it works, the top-down approach can bring about rapid, large-scale change. With one central entity in charge the process can be controlled and reasonably efficient. Because the regulations carry the force of law, the top-down approach can also be reasonably effective. Overall, the top-down approach is appealing because governments “are better able to provide the funding, enforcement, education, and multi-jurisdictional authority their policy decisions rely upon” (Bayne, 2009, p. 4).

One limitation on this approach is that regulations may be too blunt, failing to take account of the particular physical features of landscapes and the circumstances of particular landowners. Similarly, the regulatory process may fail to take advantage of local knowledge about ecological patterns and processes, knowledge that could usefully inform the whole process. A top-down approach that fails to engage local landowners can more easily overlook local ecological conditions and the peculiar needs of affected landowners. This can heighten resistance from landowners, who may not understand the regulations and their purposes. Viewing the regulations as intrusive and perhaps arbitrary, landowners may resist and challenge them (Bayne, 2009; Dietz & Stern, 2008). Particularly when widespread, local resistance can frustrate enforcement.

In a bottom-up approach, individual landowners come together in a grassroots-style effort to establish their own governing mandates, which they in some manner impose upon themselves (Bayne, 2009). Bottom-up approaches can originate in different ways and follow different trajectories. Some are begun and guided by local citizens to address local needs. One such initiative was started by residents along the Chesapeake Bay who were concerned about the Bay’s ecological integrity. That effort developed into a sizeable, wide-ranging non-governmental organization (Chesapeake Bay Foundation, 2010). Other bottom-up efforts are started by non-governmental organizations that reach out to local residents, helping them either to practice conservation individually or to join with other local residents for collective local action. The Wildlink organization in Michigan illustrates this form. It enlists landowners in the northwestern corner of Michigan’s lower peninsula to establish habitat corridors on their lands, helping the landowners coordinate with one another (Conservation Resource Alliance, 2010).

The bottom-up approach offers several distinct benefits. It can incorporate the local ecological knowledge that landowners might possess, leading in some settings to more ecologically-informed decisions (Bayne, 2009). This approach might also heighten a sense of stewardship among participating landowners, thereby strengthening motivations. Landowners who participate in decision-making processes are more apt to appreciate the benefits and purposes of the actions they take and any regulations they may adopt. Better understanding can lead to broader acceptance and support (Dietz & Stern, 2008). Local landowners who impose expectations on one another, either in regulatory forms or in less formal ways, are perhaps more likely to abide by the norms. In some circumstances, top-down action may simply not be possible; if organized conservation is to occur, the bottom-up approach may offer the only option (Dietz & Stern, 2008).

Bottom-up approaches, however, have limits of their own. They can become too focused on narrow, local issues and ignore the broader landscape scale (Bayne, 2009). Even when local residents are aided by an outside organization, resources may be inadequate to study landscapes well and to craft and implement well-grounded conservation plans (Bayne, 2009). Informal organizations based on voluntary participation can lack good ways to deal with disagreements (Bayne, 2009; Gass, Rickenbach, Schulte, & Zeuli, 2009). Leaders of the process may have trouble maintaining coordination among participants, particularly when participants have different views and goals (Gass et al., 2009). Volunteer participants can and do drop out (Bayne, 2009). Given these various challenges and limits, bottom-up approaches appear unlikely to succeed in the work of large-scale, enduring conservation, except perhaps when sound conservation yields distinct economic gains for the participants (as it might in landscapes where recreation, tourism, and retirement residences are the chief economic drivers).

In light of these limits, which attach to both approaches, it appears that neither the top-down nor the bottom-up approach alone is likely to achieve ambitious conservation goals. These two approaches, however, are not mutually exclusive. They can blend together and stimulate one another. A bottom-up effort, for instance, might be used to evoke a top-down response; a top-down directive could stimulate bottom-up initiatives. A combination of the approaches appears more promising than the use of either approach alone.

Conservation planning involves science and policy, drawing upon professional expertise. But successful implementation of a conservation plan ultimately rests on culture and human values (Theobald et al., 2000). Citizen involvement, particularly the involvement of affected landowners, is therefore an important and perhaps necessary element of any approach (Farrier, 1995; Petrosillo, Zaccarelli, Semeraro, & Zurlini, 2009; Theobald et al., 2000). As explained above, citizen engagement facilitates acceptance and compliance. It can also help foster stewardship as affected landowners gain awareness of conservation issues and become invested in the conservation-planning process. In all likelihood, long-term solutions depend upon the knowing support of most affected landowners, even when conservation duties are imposed by binding regulations. For various reasons, then, a bottom-up component seems essential. What the top-down approach adds to the mix is the power of government to apply regulations broadly and to provide mechanisms for enforcement. It offers also better opportunities to draw upon professional expertise and stronger, more reliable mechanisms for raising the needed funding, year after year.

The best approach, then, would seem to be a holistic approach that employs valuable elements of both top-down and bottom-up models (O'Connell & Noss, 1992). Public input and local ecological knowledge are important attributes of the bottom-up approach and should be included within a combined approach. The approach should also encourage landowner participation and assist landowners in understanding the substance and purpose of the regulations. Because implementation and enforcement are often more effective from a top-down approach, the mixed or middle-ground approach should provide for government implementation to compel action on the landscape scale. A government entity could also provide guidance and coordination throughout the process, collating the views of landowners and providing consistency to a potentially ad hoc process.

For various reason, in short, this paper advocates a middle-ground approach that combines bottom-up development with top-down implementation. Developing the details of land-use plans and regulations may proceed primarily in a bottom-up manner, relying upon citizen input that is solicited by government and aided by professional involvement. This planning would be done in response to instructions mandated from above. Implementation would likely also take place primarily from the top down, through government enactment and enforcement of the regulations and other prescriptions. While this proposed middle ground approach might not remedy all problems, it could maximize the opportunities for effective implementation on the landscape scale.

In addition to an effective model for action, we also need to consider the spatial scale of that action. In particular, we need to determine how we will apply our laws and regulations at the landscape scale to coordinate land uses and ultimately establish corridor networks. As stated above, land-use planning is now generally undertaken at the local level, by individual municipal and county governments. This narrow, local focus hinders coordination at larger scales. Shifting the spatial focus from the local to a landscape spatial scale is needed, and it will require coordination in planning across jurisdictional boundaries. To facilitate this large-scale, inter-jurisdictional coordination, state and local governments must develop comprehensive landscape plans, which is to say key steps must be taken at quite large scales.

As stated in Chapter Four, establishing an overall conservation plan is one of five key steps in developing a system of corridors. Local government comprehensive plans can fulfill this requirement, either by incorporating conservation goals or in the form of a supplemental, stand-alone environmental plan (Berke, 2009). States, though, must *require* local governments to establish comprehensive plans, not merely extend them the option. States already authorize or require local governments to establish plans through state enabling laws, as explained in the previous chapter (Breggin & George, 2003). Currently, only fourteen of the fifty states require or encourage local governments to adopt comprehensive plans (as opposed to merely authorizing such plans) (Berke, 2009). Maryland, Michigan, and Vermont, for instance, are among the states that obligate local governments to adopt either a comprehensive plan or a growth-management plan (Breggin & George, 2003). The requirement proposed here, that local governments develop comprehensive plans, recognizes the importance of landscape-scale conservation. It compels local governments to be proactive in establishing land-use regulations that promote green infrastructure and maintain valuable ecosystem services. It also emphasizes the importance of each local government's role within the state-wide landscape. Comprehensive plans establish a framework, or an overall land-use vision, to guide land-use decisions and direct future development (Berke, 2009; Breggin & George, 2003). Without a plan, local governments lack this overarching vision and will likely perpetuate haphazard, uncoordinated land uses. The absence today of comprehensive plans in most parts of the country—particularly plans that reflect ecological knowledge--is a serious impediment to landscape-scale planning and conservation.

In addition to this work by local governments, each state government should itself prepare a comprehensive plan that addresses land uses on a statewide spatial scale.[57] Such a plan could effectively integrate the plans of local governments within the state while providing the basis for recommended changes in the local plans. This process of coordinating local plans into a single, state-wide plan can engage local governments in the state-wide process while also identifying local plans that need revision so as to make them compatible with regional and state-wide conservation goals. When plans are prepared at multiple spatial scales—state, municipal, county, and others—jurisdictions can see how their particular plans fit within the broader landscape, with the plans of neighboring governments, and with the state-wide conservation vision. Of course, state-level action can and should go beyond merely coordinating local plans. It should in various ways provide specific guidance for local governments to follow and specific goals for them to achieve. State plans, for instance, can identify areas of particular ecological concern that should be given conservation priority; this is particularly important for sensitive areas that cross municipal boundaries. As it mandates coordination among all affected local governments the state should insist that the local governments collectively protect important inter-jurisdictional ecological features.

To ensure that state and local comprehensive plans translate into effective conservation action, they must substantively engage the ecological elements and set forth clear goals. They must be holistic in scope and based on the ecological aspects of the landscape, both its physical components and its key ecosystem functions (Cooperrider, 1991). The plans should, generally speaking, shift the primary focus of land-use law and planning from an anthropocentric- and development-centered view to an ecological view. The days of new development as the dominant planning goal must necessarily come to an end, even as development retains a place in the planning mix. Within the local plans, as noted, governments should identify key landscape elements for particular conservation uses: sensitive wetland areas, for instance, areas rich in biodiversity, and potential corridor areas (Jooss, Geissler-Strobel, Trautner, Hermann, & Kaule, 2009; Steiner, 2008). By mapping out these areas, the plans can divert development away from

[57] The federal government might enact such a requirement to force states to prepare plans if they fail to do so voluntarily.

particularly important habitat.[58] In sum, clear, well-structured comprehensive plans at all levels of government are necessary to curb the increasing fragmentation and urbanization of landscapes across the country. Without such plans, and coordination among the plans, ambitious conservation goals are simply not achievable.

Requiring all governments to prepare comprehensive ecological plans in theory appears pragmatic and readily attainable. In practice, though, state and local governments will encounter problems that impede action. To plan well, government leaders need to understand ecological patterns and processes at the appropriate spatial scales. Government leaders often lack such expertise (Berke, 2009). To satisfy this need, government planners can engage the efforts of knowledgeable scientists. They could establish expert planning and management commissions consisting of policy makers, local residents, stakeholders, and scientists from various disciplines, including landscape ecology, conservation biology, and hydrology. The New Jersey Pinelands planning process relied upon this type of commission structure (Callies et al., 2004). Further expertise might come from local environmental groups whose leaders may have gained, through their work, intimate knowledge of various local ecosystems. This kind of multi-faceted, middle-ground approach seems likely to yield the most effective results.

Another problem, noted in earlier chapters, is the frequent lack of political will to act (Berke, 2009). Challenges from developers or local landowners, or even corporate capture, may block a government's effort to adopt a plan. Internal disagreements within government may also lead to gridlock and inaction. Inevitably, higher levels of government will need to motivate local action through some combination of incentives and penalties. They could threaten to cut off funding for local services. They could reward good work with additional funding or other incentives. Pressure to act can also come from outside the government, through bottom-up efforts. Citizens and environmental groups can organize to demand action from government leaders. They might even start the process by drafting and proposing sample comprehensive plans, at least if such groups are particularly well-funded themselves. This bottom-up option would not be available or effective in all cases, but could on occasion prompt needed government action.

[58] The "ecological planning" method proposed by Steiner (2008) provides a template for developing these comprehensive plans. The method particularly calls for an ecological inventory of the landscape, which will enable governments to identify ecologically important areas for conservation.

The land-use plan developed by the state of Baden-Wuerttemberg, Germany provides a useful illustration of coordinated action by state and local governments (Jooss et al., 2009). The German land-use system parallels that of the United States in that local governments are the primary source of land-use governance. In an effort to assign "conservation responsibilities" to local governments, the state of Baden-Wuerttemberg developed an overall land-use plan based upon the physical attributes of the state's landscape. The state identified ecologically important areas as conservation areas. Each local jurisdiction that contained all or a portion of a recognized conservation area was then obligated to protect that area by enacting appropriate laws. By adopting this "conservation responsibilities" approach, or a similar variant, state governments can facilitate coordinated action between local governments on larger spatial scales.

By clearly designating a municipality's permissible land uses, comprehensive plans can promote inter-jurisdictional transparency regarding spatial locations of land uses, development, and lands set aside for conservation. This transparency can help local governments link their plans so as to achieve large-scale conservation goals. Local governments, for instance, could link their conservation buffer zones along a riparian corridor to establish a continuous conservation corridor. They could similarly integrate their plans to create wildlife corridors and help meet the habitat needs of particular wide-ranging species. In general and to reiterate, these plans should structure land uses around the ecological features and functioning of the landscape and condition development upon its appropriateness to the conservation goals for the area in which it is proposed.

<u>New Use for Old Tools: Recasting existing land-use regulations to compel conservation and stewardship</u>

Once an overall approach and conservation vision are pieced together, how do we go about taking action? How do we enact a functioning, effective system of land stewardship law and with what legal tools? Answers to these questions should take account of the particular landscape at hand. They should also draw upon the three main conservation actions set forth in Chapter Four as well as the legal tools available to achieve those goals, including the tools described in Chapter Five.

Imagine, by way of illustration, a river flowing through central Illinois, bordered on both sides by riparian forests and grasslands that create a sizeable riparian buffer. The river and its surrounding lands serve as a corridor for many species, including coyote, deer, and cougars.[59] The mostly native vegetation within this corridor filters the runoff from the surrounding matrix and retains seasonal floodwaters. It also creates shallow pools that fish and amphibian species use for breeding. Migrating flocks of birds stop here for rest and nourishment during their journeys. This river corridor links to another further north, which links to yet another, extending north into Canada. In hard-to-predict ways that corridor would help various life forms, plants as well as animals, respond to a shifting climate.

The basic elements of this vision currently exist (although corridors in existence today in Illinois are, of course, distinctly fragmented). What is needed is to take the raw materials—the land, the science, and the planning and legal tools—and implement it.

The initial pieces for such a landscape-scale corridor could come from individual land parcels that are already protected. We can use state parks, wildlife preserves, recreation areas, restored lands held by land trusts, and lands protected by conservation easements as the cornerstones of potential corridors and as core areas within a future core-buffer-corridor conservation system. For example, we could begin establishing a corridor along the Illinois river by linking the Emiquon preserve; Starved Rock, Matthiessen and Buffalo Rock state parks; segments of several national wildlife refuges; and various lands contained in county forest preserves. Multiple state fish and wildlife areas, state nature preserves, and state recreation areas (e.g, Sanganois, Rice Lake, Sand Ridge, Goose Lake, Babb Slough) are either along or not far from the river. Various tracts along this corridor are likely already enrolled in various conservation programs, given the high proportion of wetland acres. Others may be held subject to privately controlled conservation easements. Large tracts of land are also owned by power companies and other businesses; some of these lands retain significant natural features that could contribute to corridor benefits and might be preserved in part without undue economic disruption. To be sure, most lands in the corridor are unprotected in any legal sense, but the protected or restricted lands provide a good foundation for further conservation efforts.

In highly-fragmented and intensively-used landscapes, state- and federally-owned parks, wildlife refuges, and nature preserves, however degraded, may provide the best or only

[59] Cougars have returned to the central Illinois area through the networks of corridors that link up with this one.

reasonable starting places to create viable conservation corridors.[60] Other private lands, not now protected, will necessarily make up the majority of the corridors that link the existing patches of protected land. On such private lands we can, using various tools, establish linkages through open lands, hedgerows, and even backyards where necessary. Even greater options are available in the case of private lands devoted to agriculture, particularly pastures, hayfields, and lands in forestry and tree crops. The aim of these conservation corridors, as discussed above, should be to promote biodiversity generally, taking into account the particular needs of key species, and to sustain basic ecological functioning. They should also be designed to facilitate adaptation to climate change, which will require (as best we can tell today) a substantial number of corridors that extend from north to south and from low to high elevation. Again, we should whenever possible build upon past and current conservation efforts, using conserved lands as starting points for our future corridor networks. In many landscapes (though not all), these currently conserved areas will contain the highest-quality remaining habitat patches and the populations of wild species that depend on these patches. From these lands we can work outward to conserve, to various degrees and in various ways, the surrounding private lands.

Just as we can utilize existing protected lands as cornerstones of a future network of conservation corridors, so too we can utilize existing legal tools as the foundation of a revolutionized system of land stewardship law. As described in the previous chapter, our current land-use conservation tools, such as easements and conservation-based zoning, have already been employed to preserve open space on a limited, local scale; these existing tools therefore provide a ready basis for a system of land stewardship law. We can revise, strengthen, update, and combine these various tools to make them more effective and apply them to broader spatial scales.

In order to transition our existing collection of land-use tools into a system of land stewardship law, we need to revise three primary characteristics. We first *must update the substance* of these tools, to incorporate the ecological attributes of the land to which they apply and to compel stewardship from private landowners. Some amount of stewardship responsibility and some degree of limitation on socially and ecologically acceptable land uses is inherent in the

[60] Of course there may be ecologically significant lands that are not protected and should be considered as cornerstones of a network. Some of state-owned lands may be less ecologically significant for preserving biodiversity than other unprotected lands. While we must take the ecological condition of a particular piece of land into consideration, state parks and preserves provide a simple default foundation for corridors.

bundle of landownership rights. The substance of these land-use tools requires revision to shift the balance of private landowner rights and responsibilities in ways that mandate greater stewardship responsibility. Second, we must *revise the ways we use* our existing land-use tools. Instead of on an ad-hoc, as-applied basis, we should combine them to achieve maximum habitat protection and apply them strategically, according to comprehensive conservation-based plans. Third (though perhaps the first step, logically), we must *update the purpose* of these tools. Their overall purpose should reflect the ecological needs of the landscape in addition to human needs and should aim to preserve land health as an overall goal.

In short, we must revolutionize the *substance, use*, and *purpose* of our existing land-use tools to transition to a stewardship-based land-use law system that balances land use with conservation. Only in that way will we have a system capable of establishing viable networks of conservation corridors.

The land-use policies and laws used to achieve these conservation aims will need to move in three basic directions, as noted earlier. They will need to (i) preserve as much remaining habitat as possible; (ii) stop bad land uses; and (iii) restore degraded habitat. All three of these basic actions deserve rather full exploration. We can turn to them.

<u>Preserving habitat.</u> The goal of preserving habitat requires that we set aside existing open spaces, particularly high-quality ones, and prevent those spaces from being developed or fragmented by roads or other human land uses. In practice, this goal can be applied broadly to incorporate any open space that will enhance the ecological pattern and process of a landscape, aid in the resilience of ecosystems generally or help particular species adjust to human disturbances.[61] Protection should, as noted, start with and include existing wildlife preserves, parks, and other natural areas that can supply the "cores" of the conservation network. We can also protect ecologically important lands that have not yet been protected[62], and we can secure key portions of corridors to link core areas. As explained in previous chapters, protecting open

[61] For purposes of clarity, this thesis divides the multiple types of "open spaces" into two categories—those that are undeveloped and function as habitat, such as privately owned woodlots or prairies, and those that are not technically developed but are under intensive human use, such as agricultural lands. This thesis advocates that those open spaces in the first category should be given higher priority under this preservation goal than those that are more intensively used. The distinction will often be ambiguous and should be left to ecologists and planners.

[62] Talk about section adjacent to protected Nachusa grasslands that is privately owned, ecologically important because it also contains prairies and borders protected prairie land, important to preserve contiguous nature of the area, but is currently unprotected.

land from development and further fragmentation maintains the land's ecological patterns and processes, yielding many ecological benefits for the species inhabiting the land and for humans.

Four general land-use tools will help to achieve this habitat-conservation goal: *acquisition, zoning, other regulation,* and *easements*.

Acquisition is the direct purchase of lands for conservation purposes. Governments as well as non-governmental conservation organizations, particularly land trusts, both can engage in the acquisition of lands as part of a broader conservation strategy. Because direct acquisition is an expensive its use is inherently limited. However, because it provides the purchaser such substantial control over land use it is a particular powerful tool. Given these characteristics, this method of preservation should be used strategically to focus on ecologically significant parcels of land. Acquisition can be used to preserve lands that demand the greatest protection as well as lands that are integral to a landscape-scale system of conservation corridors. In particular, acquisition can be a useful means of preserving core habitat areas and key links that lie within the "chain" of lands comprising planned corridors.

State and local governments[63] can engage in acquisition in two main ways: direct purchase as market participants and by means of their eminent domain power. Through direct purchase, a government can simply buy up parcels of ecologically significant open land as they are put up for sale, assembling a conservation network slowly over time, parcel by parcel. Frequently, direct purchase will not be a feasible option; owners of ecologically significant lands or of parcels integral to a cohesive landscape corridor may not be willing to sell. In these situations, a government can exercise its eminent domain power. As explained in Chapter Five, the eminent domain power enables a government to take private property for public use, provided the government gives the owner "just compensation" for the land it acquired (Chapter 5 *supra*; U.S. CONST. amend. V). The eminent domain power gives governments greater flexibility to preserve the lands that are key components of its corridor network or are of particular ecological importance. The eminent domain power is limited by the requirement that the acquisition be for a public purpose, but conservation use can clearly be construed as a public purpose, as conservation serves the public through aesthetic, recreational, and ecosystem services benefits.

---

[63] The federal government can also engage in acquisition, but for our purposes, state and local governments will be the primary actors.

Land trusts can also acquire lands through direct purchase, either by buying lands put up for sale or negotiating a sale with a willing landowner. Land trusts do not, however, have the flexibility of the eminent domain power. By working in concert, land trusts and governments can coordinate their preservation goals, so both are working toward establishing the same network of preserved land. In this way, they can expand the impact of their efforts, attaining greater preservation gains than they would individually. The acquisition powers of both land trusts and governments, however, are ultimately limited by funding, which is always limited and too often nearly non-existent. Paying landowners for the full value of their lands is an expensive business while raising elements of fairness to landowners if not carefully done (Freyfogle, 2007b). The cost of acquisition therefore restricts its impact. When used in conjunction with the other three techniques, acquisition can nonetheless serve an important role in an overall conservation land-use scheme by cherry-picking the most ecologically significant core areas for the greatest protection.

Local governments can also preserve sources of present and future habitat through *zoning*. As explained in the previous chapter, multiple zoning methods exist that enable local governments to protect open space by generally clustering development and providing specifically for the protection of surrounding open spaces. Cluster zoning and conservation zoning both specifically recognize and protect open lands from development directly within the local zoning system. Regulations governing Planned Unit Developments commonly incorporate the technique of clustering to establish a more holistic approach to new development centered on the protection of open space. Subdivision platting ordinances can contain similar requirements. As mentioned, however, the type of land that qualifies today as "open space" under such regulations can be overly broad, allowing developers to designate lands that have little conservation value. It helps little to leave patches of undeveloped space scattered in a region, particularly when developers pick the spaces chiefly to cut their costs. Lands that qualify as required open space should be limited by law to those that provide continuing ecological benefits through ecosystem services, habitat or both. Governments, that is, should clarify in their zoning ordinances the types and conservation values of acceptable open spaces. Clarification is particularly needed in regulations governing platting, subdivision development, planned unit developments, and other large-scale projects.

Building upon these basic tools, local governments could ultimately develop an overall system of *ecological zoning*, or zoning processes that are structured around the ecology of the local landscape to protect ecosystem function while providing for human land-use needs. This holistic zoning scheme would transition zoning from its traditional separation-of-use, grid-based approach to one centered on greater density, mixed uses in developed areas, and the preservation of open space, particularly (as just mentioned) space that helps to protect biodiversity and provides ecosystem services to residents.

Ecological zoning could also promote the core-buffer-corridor approach to planning through particularized zoning tools. To establish and protect corridors between habitat patches within this zoning system, governments could develop a new zoning "district"—the corridor zone. A form of conservation or open-space zoning, the corridor zone will facilitate the development of corridors by identifying lands integral to a network of connected habitat throughout the local landscape and beyond. A corridor zone will also protect corridors once they are established. By setting up a special district, ecological corridors will be given consideration as independently valuable land uses within the zoning system.

In addition to corridor zones, governments can implement "concentric zoning," or overlapping protections for core areas and surrounding buffer areas (O'Connell & Noss, 1992). In concentric zones, protections will increase moving inward toward the center, or core, and will decrease moving outward toward buffer and matrix zones. Together, corridor and concentric approaches to zoning will establish a locally-applied framework for preserving habitat and ultimately establishing broader networks of corridors. We can no longer approach land as collection of separate, unconnected parcels; we must regard the land as a dynamic, functional unit, in which all individual parcels are interconnected. An ecological approach to zoning prompts us to pursue our distribution of land uses through this perspective.

Ecological zoning, or any zoning to preserve habitat, must be undertaken transparently, logically, and systematically. To maximize the ecological benefits of open space protections, local governments should locate protected open spaces adjacent to one another wherever possible, creating large blocks of protected land as opposed to many small blocks. Furthermore, developed areas should be located near one another to reduce sprawl, the need for additional roads, and the overall impact on the surrounding habitat. Local governments must also coordinate their open-space protection plans with other jurisdictions to link preserved habitat areas across jurisdictional boundaries and achieve landscape connectivity. Clear, well-developed conservation and zoning plans—the kind recommended above--will help to facilitate this coordination and connectivity.[64]

In addition to zoning, governments can enact *other regulations* to require all landowners within a particular ecological area to preserve portions of their lands as vegetated, open areas of habitat. Regulations can be useful for establishing corridors across multiple private parcels of land or for establishing buffers abutting core habitat zones. Regulations can require landowners to maintain a portion of their backyard as forested habitat; they can compel agricultural landowners to create or maintain hedgerows in strategic locations that would help create a buffer or corridor; regulations can also prohibit landowners from removing vegetation along a stream flowing across their collective properties to maintain the stream's corridor values.[65] Regulations of this type mostly control intensive uses of lands, limiting the private activities in ways that promote conservation goals. Such detailed regulations do not preserve open expanses of habitat (except in rare cases of very large land parcels); they do not empower governments to establish large blocks of protected habitat like zoning or acquisition can. They are nonetheless highly valuable, supplementing other methods of conservation and significantly reducing the ill effects on biodiversity that intensive land uses might otherwise have.

Governments and land trusts can also establish *conservation easements* to preserve habitat. As explained in the previous chapter, a conservation easement gives the easement holder an interest in a parcel of land by placing specific restrictions or requirements on the use of the. Through these restrictions, the easement holder can maintain the open character of existing habitat or otherwise limit activities that would cause harm. Conservation easements are particularly useful tools for establishing corridors because they are flexible and the holder can link several easements together across multiple small parcels of privately-owned land. Voluntary, donated easements also cost a government or land trust nothing to acquire aside from lost tax revenues for charitable contributions and, in the future, declining real property taxes.

[64] The elements of such a planning process are beyond the scope of this discussion. Benedict & McMahon (2006), Peck (1998), and Steiner (2008) provide detailed discussions of planning for habitat preservation and the protection of biodiversity.
[65] Landowners will probably not accept these restrictions easily. They will likely challenge the validity of the regulations as takings in court. The next subsection presents a more detailed discussion of how governments might uphold regulations like this against takings challenges.

Conservation easements, particularly when donated, thus offer advantages over the more expensive option of direct acquisition. (Bray, 2010).

To use conservation easements effectively, however, governments or land trusts will need to proactively address the problems they might present. Easements in a given landscape should be coordinated spatially and in purpose. Additionally and ideally, easements should be implemented systematically. When possible, governments and land trusts should coordinate their efforts to link new easements with existing open spaces that have already been preserved. Easement agreements, of course, must clearly establish prohibited versus permitted uses and provide for ongoing monitoring and management. When developing a network of easements, governments or land trusts will likely encounter unwilling landowners. In these cases, a government may exercise its eminent domain power to acquire a needed easement, especially if it is a particularly important component of a corridor or possesses important ecological attributes. If planned well and coordinated with other preserved habitat, conservation easements offer useful and cost-effective tools to achieve landscape-scale goals without displacing all human land uses.

In addition to these existing land-use tools, Centner (2006) proposes a statutory tool for preserving habitat—an Undeveloped Lands Protection Act. In the form that Centner proposes, such an Act would establish a defense for undeveloped lands against nuisance claims. It would create a rebuttable presumption that undeveloped lands do not constitute nuisances. This proposed Act is chiefly aimed at protecting agricultural lands and other relatively "undeveloped lands" under human use; it specifically provides for continued uses of these undeveloped lands, including activities like forestry and crop planting. Such a statute, if enacted, would operate somewhat like conservation easements in that it would protect agrarian land uses from development pressures, at least so long as the landowners wanted to resist development.

More useful than Centner's proposed statute would be an Undeveloped Lands Protection Act that actively protects open spaces by recognizing the value of the ecosystem services they provide and requiring these services to be considered if development is proposed. Such an Act could require a balancing of the ecological harms of development against the human gains it might produce, allowing development to take place only if development resulted (or perhaps clearly resulted) in distinct overall benefits. The Act might conceivably also incorporate Noss's proposal that courts shift the burden of proof from environmentalists to developers; instead of requiring environmentalists to prove that developing a parcel of land would cause ecological harm, developers should be required to prove that development would *not* cause such harm (Noss, 1994). Such a shift in the proof burden has appeal but for various reasons is likely impractical for day-to-day administrative purposes.

Although preserving habitat is one integral element in the process of developing landscape-scale corridors, securing lands through which to locate corridors is not sufficient to ensure that those corridors are ecologically functional. As discussed, land uses in the matrix surrounding protected habitat areas can have significant impacts on ecological functions within those protected areas. Matrix conditions can also impair the permeability of the landscape, which influences dispersal for certain species. While this thesis emphasizes corridors as the primary method of establishing connectivity within a landscape, matrix permeability is an important secondary consideration to supplement corridors, especially in areas where corridors are not immediately feasible. As explained in previous chapters, land uses that impair the health of the land organism impair the health of biotic communities as well as the health of human communities. In addition to tools for preserving habitat, therefore, a system of land stewardship law must also include tools to facilitate ecologically responsible land uses while eliminating land abuses.

<u>Stopping bad land uses</u>. Our second overall conservation goal, stopping bad land uses, aims to curtail human land abuses within the land-use matrix. This goal is broadly stated and thus, by necessity, rather vague and flexible. As applied in different landscapes it can require varying degrees of landowner responsibility. At a minimum, it can require that landowners refrain from uses that directly result in material ecological degradation; more expansively it can compel active stewardship from landowners in the sense of taking steps to maintain habitat and ecological functioning. This goal applies to urban as well as rural and agricultural land uses, encompassing activities that result in, for instance, erosion, high levels of surface runoff, vegetation removal, and tillage. A system of land stewardship law can address unwanted land uses in two primary ways: (1) affirmatively compelling stewardship through a legal duty; and (2) prohibiting bad land uses through regulation.

In specific settings and ways land-use law could impose affirmative legal duties of stewardship upon private landowners (Karp, 1993). This idea is by no means a new one; commentators have proposed it for years. Such a duty would be based in the reality that property rights are the product of law and that landowners, accordingly, derive their rights from, and are controlled by, property law (Karp, 1993). As discussed earlier (and as reflected in the laws of private and public nuisance), the privilege of landownership comes with longstanding (albeit general) obligations to do no harm to neighbors or the community. Over time, we have tended to limit these obligations so as to permit more intensive, industrial land uses, but the general principles are very much alive and in place (Karp, 1993; Freyfogle, 2003). Moreover, many particular bodies of law require owners affirmatively to maintain what they own (public health laws, occupational safety laws, historic preservation rules, and many more). Urban landowners, for instance, are routinely obligated to control weeds, to avoid diverting water into streets, and to remove snow and ice off walkways. A legal duty of ecological stewardship would largely mirror such duties, making them applicable to rural lands. Central to a new ecological duty would be a landowner responsibility to take specified actions, perhaps on an on-going basis, to maintain the land's ecological functioning and preserve wildlife habitat. It might also require landowners, before altering their lands in material ways, to actively consider ecological impacts and perhaps effects on future generations.[66] Such a duty could be recognized by courts in broad terms; more detailed obligations, though, are properly set forth by legislative bodies.

State governments[67] could enact this duty as a stand-alone statute that applies, proactively and retroactively, to stop current abuses and forestall threatened abuses. A stewardship statute could delineate specific landowner responsibilities based on the type of land that is owned, its size, and its physical and biological characteristics. For example, it could require that landowners when landscaping use species native to their location, that landowners mitigate surface runoff from their property and that they maintain permanent vegetation on ecologically sensitive lands (e.g., riparian floodplains, steep slopes, unstable lands). Violations should incur penalties.

Such a state statute could help state governments counteract voter initiatives like Oregon's Measure 37, which limited local authority to regulate private land for conservation (and other) purposes (Freyfogle, 2007b). A stewardship statute would alter the status of private

---

[66] The exact scope of this duty, whether only ecological or also encompassing future generations, will be determined by the legislature or court that establishes it.
[67] The Federal Government can also enact such a statute. In this case, a general federal stewardship mandate could be administered by the states, akin to the Clean Water Act. Due to the degree of political controversy such a statute would entail as well as the administrative complexity on the federal level, it is much more likely to be implemented and more easily administered on a state level.

property rights —modestly but importantly— making them more fully subject to an updated version of the do-no-harm rule. The prospect of strong state action to promote conservation could well get citizens to reconsider their opposition to local land-use controls; it might seem preferable to many to have local governments work out the land-use details than have the state get involved. They may prefer specific limits on what they can do rather than face the uncertain of a vague, powerful duty to take good care of the land ecologically.

If legislatures fail to enact a stewardship duty by statute, courts could take the initiative by injecting the duty into basic common law principles and rules, most likely by drawing upon and clarifying the longstanding do-no-harm principle and the law of nuisance (Karp, 1993). In essence, a common law duty of stewardship would empower courts to judge the reasonableness of land uses based in part upon ecological considerations, in addition to economics and social impacts. Thus, in an action in private or public nuisance—brought by either a government agent or a private plaintiff—a court could find a land use unreasonable based on ecological considerations, leading to an injunction that restricted or terminated the land use. The ecological duty would also be significant when courts are called upon to judge the reasonableness and economic impact of new land-use regulations. A new regulation that largely duplicates the stewardship duty, giving clarity to a general constraint that already existed on private property, would not have much economic impact and would thus not raise serious questions under the regulatory takings doctrine. That is, detailed land-use regulations would be more readily upheld against constitutional attack when they build upon a preexisting common law rule that broadly limits landowner options.

In implementing this requirement, we must recognize that we humans need to use and change nature in order to live. In addition, we are likely wise to use many lands quite intensively so that other lands might be used modestly or even set aside. Legislatures and courts must therefore determine where to draw the line between land use and land abuse. Some land uses might be so destructive and so unnecessary that they could be banned anywhere. Other land uses might be acceptable in some places but inappropriate when undertaken in ecologically sensitive areas. Filling wetlands or destroying the habitat of an endangered species, for instance, should constitute *per se* violations given that activities conducted on such lands could almost always take place elsewhere, with less ecological effects. For other, less clear land uses, we can conduct a balancing test to determine whether a land use crosses the line from a permissible use to an impermissible abuse. A primary consideration within this balancing test, under the new legal regime, would be the ecological impact of the activity.

Instead of, or in addition to, compelling landowners to actively engage in stewardship and punishing them when fall short, state and local governments can enact regulations that prohibit particular, harmful land uses. In practice, regulation may be the only viable option given the suspicion if not hostility that might well greet a broadly worded stewardship duty. Given the political power of landowners, state legislatures might well lack the courage to pass a statute mandating stewardship on private lands. Getting courts to take action might be equally hard—or in any event time-consuming—given the conservatism of many courts and the widely held belief that legal change should be made by legislatures. Regulation provides a more direct, practical, and effective method of stopping bad land uses. It will likely be the more widely used method of pursuing our second overall conservation goal, even as efforts are made to promote the broader stewardship duty.

Because land abuses ultimately harm the public health, safety, and welfare, stewardship regulations are within the scope of state and local government police power[68] (Karp, 1993). Moreover, a greater degree of restriction would likely lead to less fragmentation and greater biodiversity protection. In a comparison of two Houston suburbs, Kim & Ellis (2009) found that more restrictive development regulations reduced fragmentation and improved the ecological framework within and surrounding a developed landscape.

By enacting stewardship-oriented regulations, state and local governments can identify specific land uses as improper, making clear to landowners what conduct is unacceptable. Governments can base their use/abuse determinations on the ecological impact of the activity in question, weighing the extent and severity of the ecological harm as the baseline for determining what uses qualify as improper. For example, regulations could prohibit mowing areas conducive to grassland bird nesting, during breeding times generally or at least until after one clutch is fledged. Other regulations could prohibit the conversion of privately-owned woodlots or wetlands. Still others might prohibit landowners from modifying lands near riverbanks to keep rivers connected with floodplains, thus providing room for flooding, migratory bird habitat, and

[68] The power to implement stewardship regulations is also within the scope of the state's public trust duty. In essence, the state not only has the authority to enact stewardship regulations, it also has the duty to do so as a trustee of the state's natural resources, including its biodiversity. This topic is explored in more detail in the following subsection.

fish spawning areas. Stewardship regulations could prohibit abuses outright, while also, as noted, requiring mandatory actions to mitigate existing harms. For instance, regulations could require agricultural land owners to establish hedgerows, to reduce subsurface drainage, to avoid regular tilling of sloping soils, and to leave a certain percentage of lands during any year in year-round cover. More is said below about such regulatory measures. They are offered here as illustrations of what is possible; as examples of regulations that could be developed for particular landscapes to achieve particular conservation goals.

The regulatory approach is flexible and enables governments to tailor individual regulations to specific problems in specific landscapes. By using detailed regulations, varying from landscape to landscape, governments could in effect begin to implement indirectly a broader duty of land stewardship. Once landowners accept and perhaps see the wisdom in such detailed constraints, they may become more receptive to a broader, general stewardship duty on all landowners.

One limitation on this regulatory method is that it gives landowners no particular incentive to do better. State governments might reduce this problem by adopting a land-stewardship statute. Alternatively, they could issue some sort of aspirational mandate or a set of stewardship guidelines, either through a state agency, like the Department of Natural Resources, or within its comprehensive land-use plan. Still, the limitation would remain. To get landowners to go further, beyond the required minimums, a state would need to instill landowners with a personal drive to do better. That kind of success—the ultimate aim of conservation policy—will likely require significant cultural change in the ways landowners interpret and value the land and in the ways they define personal success.

As an approach, regulation offers many advantages despite its limitations. A purely voluntary stewardship system, to be sure, might be ideal; in this arena as in other, government is a necessary burden. But voluntary action is infeasible given American culture and, in particular, prevailing ideas about private property rights. Regulatory efforts will likely be met with substantial resistance and takings challenges in court.[69] Yet, regulation can encourage proactive responses from landowners to avoid regulatory violations, instead of waiting to be caught and penalized (Echeverria, 2005). Regulations also allow for stronger and broader enforcement

[69] See the following subsection for a discussion on maintaining land stewardship regulations in the face of such takings challenges.

measures. Furthermore, the regulations passed in one jurisdiction may serve as a template for similar initiatives in other jurisdictions, or on the federal level. Like it or not, regulations appear necessary.

Even if these actions are taken—if we impose a legally-binding stewardship obligation and enact detailed regulations prohibiting ecological harm--many landowners might well continue present practices. At minimum, there will be a substantial lag time, not to mention political and judicial battles, between implementation of stewardship measures and changes in landowner behavior. Bad land use, in ecological terms, is ultimately a social and cultural problem, problem that is broad and tangled, encompassing elements of American and English history, economics, politics, and cultural values, among many others.

The American conception of private property presents a particular, substantial obstacle to full conservation. As commonly understood property law vests owners with the power to act pretty much as they see fit. This attitude has abetted poor land uses and absolved landowners of stewardship duties. Particularly exalted is the right of landowners to develop their lands. Americans consume land like they consume most things, greedily, wastefully, and in far larger amounts than needed. They buy and sell land as a discrete commodity with discrete boundaries. This understanding of private property must change substantially; if it does not, it will continue to impede communal efforts at land conservation. This particular challenge is beyond the scope of this thesis but directly relevant to it. So central is this issue—so important is it that Americans generally reconceive the meaning of land ownership—that progress on this issue may be essential for any land conservation effort. In some way, using some rhetoric, conservation advocates need to come forward with a new vision of what it means to own land, responsibly, in the twenty-first century.

<u>Restoration.</u> In landscapes already largely converted to human uses, like those that dominate Illinois, much land is ill-suited as habitat for vast numbers of animal species. As for plant species, the land is mostly managed to exclude them, typically with herbicides and regular tillage. Such lands thus do not and cannot serve as effective wildlife corridors, much less as homes for resident wild populations. Most of Illinois is dominated by row crops, which can extend to the edges of the state's tens of thousands of miles of waterways. These croplands commonly stretch from one protected area to another, many miles away. As for the habitat patches within the landscape—the state parks and wildlife reserves for instance—many are

ecologically degraded due to edge effects, surrounding land uses, internal roadways, and intensive recreational activities. Effective corridors can come about, therefore, only if vast tracts of land are returned to ecological conditions that provide good habitat.

This work of returning heavily used lands to more natural conditions is commonly termed restoration. The term is useful enough, so long as we recognize that restoration does not mean, and need not mean, returning lands precisely to some ecological condition that existed in the past. Nature changes on its own, and the reconstruction of a complex natural system from centuries ago is likely a challenge far beyond our means—even if we knew precisely the conditions that prevailed. Fortunately, nearly all wild species can inhabit varied landscapes, and we can make good habitat for them without replicating conditions that existed in the past. Indeed, exact replication may not even be what we want if our desire is to provide good habitat for various, particular species and their habitat needs differ somewhat from the precise conditions that prevailed in a given place. Mimicking nature is typically wise, given our ignorance. But we need not throw up our hands in frustration when we realize that our success can only be partial.

A network of functional, landscape-scale corridors can arise only if we engage in restoration as thus understood, as a necessary supplement to preserving (and enhancing) existing habitat and stopping bad land. Four basic land-use tools, drawn from other discussions above and tailored to the need, can guide our implementation of this goal : (i) the acquisition of key land parcels along proposed corridor; (ii) regulatory control of activities within the corridor; (iii) restoration zoning of the lands, and (iv) the imposition of duties to restore, by means of the common law of nuisance, on landowners whose lands are particularly degraded.

Restoration—that is, the creation of new corridors—must start with a sound grounding in science, drawing upon the ecological history of the landscape and taking into account the needs of particular species. This work should draw upon the science reviewed in the opening chapters, particular conservation biology. The aim, again, is not to restore lands to a state that pre-dates European settlement in North America, even if that were possible.[70] The aim is to support biodiversity generally, particularly native species, so that they can function as healthy biotic community. It is also, as considered earlier, to promote the ecological functioning of

[70] Perhaps with the help of rewildling projects and a successful land stewardship system of law, such a baseline might be feasible, but it is not reasonable to think that we would be able to easily achieve such a baseline now.

landscapes—to sustain the health of the land as a whole, or (to use current terminology) to sustain and enhance the ecological services they supply. To the extent possible land managers should encourage existing species to expand into newly restored ranges. It will doubtless be necessary, however, to take affirmative steps to introduce species into restored areas, always relying, to the extent possible, on plants and animals gathered from nearby so as to maintain local gene pools.

Aside from the ecological planning required for corridors, it is essential to assign responsibility for implementation. Governments—state or local—may assume control. Such high-level, landscape-scale ecological planning is properly the role of government to some degree, either a single state government, or a coalition of cooperating local governments. Governments have the ability to legally implement and enforce the restoration policies they develop. They can also bring in experts from within the state to assist with developing scientifically-sound ecological baselines. Governments could also establish restoration committees, consisting of government leaders, planners, scientific specialists including ecologists, and landowners.[71] However, restoration responsibilities could fall to the individual landowners, who would develop and implement their own restoration plans. We could also pursue a hybrid, middle-ground approach in which governments establish restoration goals and mandates, which landowners must implement. Landowners will likely resist such mandates, however, as invasions of their private property rights. Alternatively, local non-governmental conservation organizations, land trusts, and even universities could be given this task. These organizations are generally knowledgeable about local community and could easily become involved in local communities to implement restoration plans. Overall, the issue of who assumes restoration responsibilities determines what tools are appropriate to implement those responsibilities.

Governments or land trusts could use the tool of direct acquisition, discussed above, to gain title to lands that are degraded or have been converted to urban or agricultural uses but are critical components of proposed corridors. State or local governments could also invoke their eminent domain power to acquire a privately-owned parcel of ecological significance. By returning the parcel to a natural, ecological state, the government undertaking the acquisition

[71] Such an approach was taken in the New Jersey Pinelands. The governments organizing the Pinelands conservation effort established a Pinelands governance committee that was responsible for making the land use decisions that would impact the region (Callies et al., 2004).

could successfully argue that the land is being used for a public purpose—to protect the land's ecological function, which provides local citizens with ecosystem services, flood prevention, for instance. Additionally, the land could have some public recreational use; it could include a hiking trail at its edge or a small picnic area, for instance. In the case of acquisition, the party that attains title to the land, either the land trust or the government, would likely undertake restoration responsibility.

State or local governments could also enact regulations to compel private landowners to undertake restoration activities on portions of their lands.[72] In this case, landowners would be the ones responsible for implementing the restoration activities. One potential regulation might compel better management of private woodlots; in landscapes dominated by intensive agricultural land uses, a potential regulation might require landowners to create new or restore existing hedgerows to facilitate permeability across their otherwise uninterrupted acres of row crops. Another might require riparian landowners to restore buffer strips of native vegetation along the banks of any water course abutting their land. Restoration-based regulations should be based upon the ecological condition of the landscape—both its current condition and its proposed future condition once home to green infrastructure networks.

Public and private nuisance claims can also be used to compel restoration. The central argument in such a "restoration nuisance" action would rest on a claim that the degraded or converted parcel of land constitutes a nuisance in its unrestored state. In a nuisance action to compel restoration, a landowner, conservation organization, or a state or local government entity could bring suit against another private landowner, or in some cases the state, if the land at issue is state-owned. In a private nuisance claim, the plaintiff would claim that the degraded quality of a neighboring parcel of land is infringing upon his use and enjoyment his own land because the neighboring land impairs the ecological health of the surrounding landscape through spillover effects. The plaintiff could also argue that the poor ecological quality of the neighboring land inhibits the landscape's overall performance of ecosystem services, from which the plaintiff's

[72] State or local governments would have authority to impose these regulations subject to their police power. By prompting restoration, the regulations protect public health, safety, and welfare. They could also be construed to be stopping ecological harms. If the regulation singles out a particular landowner, however, it may be invalidated as a taking. The regulation should be general and applicable to all landowners equally. It also should be framed as preventing a harm to overcome takings claims. The next subsection discussed takings challenges to ecologically-based regulations in more detail.

land benefits.[73] Because these claims are broad and based on indirect impacts, a public nuisance suit may be more applicable to impose restoration obligations. In a public "restoration nuisance" suit, the plaintiff would claim that, in its unrestored condition, the neighboring land infringes upon the public health and welfare and impedes the public's rights to receive important ecosystem services. Again, these claims are broad and will ultimately depend upon the scope of a court's interpretation of harm and public rights as well as the degree to which it recognizes ecological harm as a valid, remediable claim. The responsibility to undertake restoration will rest with the defendant, or owner of the degraded property, if he or she loses the suit and affirmative restoration is ordered as a remedy.

Local governments can also require restoration in particular areas of the local landscape through zoning. By establishing a specific "restoration zone" district, local governments will create the capacity to target areas that are in specific need of restoration. A local government can choose to include a particular spatial configuration of land parcels within one of these "restoration zones" because of the ecological role the parcels, singly or collectively, might play or because they are located within a planned corridor area—or perhaps within or adjacent to an existing local "corridor zone." Among other considerations, these restoration obligations should be based upon the ecological composition of the landscape and the parcel's orientation within the landscape. Like any regulation, to be valid, this requirement will have to apply to all landowners of similar properties. In restoration zones, the local government would likely impose restoration obligations upon the landowners within the zone. Alternatively, the local government could implement restoration techniques itself or could pass the responsibility to a local conservation agency or land trust to implement and administer.

The corridors we establish will not be very effective if they consist of degraded lands with impaired ecological function. In addition to securing land for corridors and reducing the adverse ecological impacts of land uses within the matrix, our system of land-use law must also restore ecological function by restoring natural ecological pattern.

---

[73] To reach these findings, a court may be compelled to update its common law concept of "harm" within the nuisance doctrine to incorporate ecological harm and/or a recognition of ecosystem services as a valuable element of benefits obtained from land. This idea is discussed in the following subsection.

The Nature of the Common Law: Incorporating ecology into the meaning of harm and the scope of the public trust

Many private landowners will resist regulations that place increased restrictions on their current or future land uses and will challenge the regulations in court as legally invalid. To ensure that a system of ecologically based land-use laws, ordinances, and regulations is effective in practice, we will need to establish a clear basis of state and local government authority to enact such measures. We will also need to provide adequate enforcement measures and a means of upholding the ecological, land stewardship laws in the face of legal challenges. Two updates to the common law will go a long way to ensure that ecological land-use laws are effective, enforceable, and upheld in court: (1) update the public trust doctrine to apply to a broad range of natural resources, including wildlife habitat, and (2) update nuisance law to incorporate ecological harm.

With respect to legal challenges, landowners are likely to claim that particular regulations constitute unconstitutional takings without just compensation. As explained in a previous section, a taking may occur when a government entity physically invades private property, when a regulation results in total economic deprivation, or when a regulation "goes too far" such that it unreasonably burdens a private landowner or unfairly compels a single landowner to provide a public benefit. While some takings claims may be valid checks on abuse of regulatory power, many will likely be attempts to thwart land-use restrictions. A necessary component of a system of land stewardship law will therefore require an updated, ecological approach to takings challenges. Updating nuisance to incorporate ecological harm and broadening the scope of the public trust doctrine will provide the legal foundations necessary for overcoming takings challenges.

The first necessary common law update consists of expanding the scope of the public trust doctrine. The public trust doctrine, as explained previously, provides that state governments have an obligation to protect natural resources within the state for the benefit of state citizens and the common good (Wood, 2009a). As mentioned above, courts have traditionally interpreted the doctrine narrowly, to apply only to submerged lands and wildlife (under the wildlife trust doctrine), not to a broader range of natural resources (Hudson, 2009; Wood, 2009a). While some scholars contend the doctrine will remain narrow in application, there is no indication that it was intended to always be limited to only those two resources

(Hudson, 2009). Historically, private property rights have yielded to important public environmental interests (Hudson, 2009). Because it is an element of common law, the doctrine is inherently flexible and can, and should, change over time as social needs demand (Wood, 2009a). Some courts have begun to apply the doctrine more expansively to encompass other natural resources that have significant public value or serve important public functions (Hudson, 2009). The public trust doctrine logically should encompass a duty to protect ecological functions important for human well being, such as flood control, productive soils, clean water, and clean air. All state courts should expand the scope of the public trust doctrine to extend to all ecological resources within a state that contribute to important and necessary ecosystem functions.

The basis for an ecological trust already exists, within the current common law foundations of the public trust doctrine as well as in many state constitutional provisions that establish a healthful environment as a public right (Wood, 2009a). The Illinois state constitution provides, for example, "The public policy of the State and the duty of each person is to provide and maintain a healthful environment for the benefit of this and future generations" (IL Const. Art. XI Sec. 1). By incorporating "each person," this provision implicates not only the state but also private landowners as responsible for maintaining the health of the state's environment. It also incorporates a normative vision in its call to maintain a "healthful environment." Such a provision clearly indicates that the state's responsibility, and the responsibility of all citizens, including private landowners, is to take action to promote this normative vision. Because the state's public trust duties necessarily would incorporate other natural resources in order for it to fulfill this policy, this type of constitutional policy provision establishes a basis for expanding the public trust doctrine into an ecological trust.

An expanded public trust will provide state governments with a broadened scope of regulatory authority under which to protect ecological resources through land health regulation and restrictions on bad land uses. We might refer to this updated, broadened, ecologically oriented conception of the public trust as an Ecological Trust.[74] An expanded trust doctrine will serve to validate these necessary ecological regulations. Under this expanded trust duty, local governments will not only be able to regulate to compel stewardship, stop land abuses, and even preserve important core habitat, they will also be *obligated* to do so as part of their role as

[74] Wood (2009) has proposed a similar concept under the term "Nature's Trust."

ecological trustee. An Ecological Trust will give citizens the ability to sue to compel their governments to act under their public trust authority to protect ecosystems components and functions when necessary if they fail to do so (Wood, 2009a). A California appellate court determined that private parties do have a right to bring legal action against the state government to enforce its public trust duty (*Center for Biological Diversity, Inc. v. FPL Group, Inc.*, 166 Cal. App. 4th 1349 (2008)). An expanded public trust doctrine, or ecological trust, therefore will establish authority and obligation for governments to protect crucial components of ecological systems within the state and to preserve ecological function in the form of ecosystem services for human residents as well as for wildlife and ecosystems in their own right, and it will open the door to citizen enforcement to compel state action.

According to Wood (2009a), an expanded public trust duty must be "holistic, organic, and obligatory" (p. 67). A duty to protect ecological resources holistically necessarily implies a landscape scale approach and encompasses landscape pattern and process. Undertaking ecological trust duties will require state governments to engage with ecological *systems* as opposed to individual resources in isolation. The Ecological Trust, therefore, will provide the ecological foundation necessary to guide government land-use decision-making toward conservation and an appropriate balance between human uses and habitat.

In addition to broader authority and obligation to regulate ecological impacts, an ecological trust doctrine will likely provide greater latitude for government action by limiting the applicability of takings claims. The Ecological Trust can help governments overcome takings claims in two ways. First, a broadened public trust doctrine correspondingly will broaden the reach of the nuisance exception identified in the *Lucas* case. In most ecological regulation cases, the land will be subject to some additional environmental restrictions, but the landowner will still be able to maintain essentially the same underlying land use as before the regulations applied.[75] In some instances, however, regulations may result in complete economic deprivation, in which case they will be subject to a takings violation. As discussed previously, the Supreme Court in *Lucas v. South Carolina Coastal Council* identified an exception to regulations that deny all economically valuable use, or per se regulatory takings, for "background principles" of state law (505 U.S. 1003, 1029 (1992); Hudson, 2009). The Court equated these "background principles"

[75] The Clean Water Act, for example, applies additional restrictions with respect to the use of property, but does not impact the underlying land use.

with nuisance law, including a state's inherent power to prevent harm and abate public nuisances (Hudson, 2009). While the Court in *Lucas* did not define exactly what qualifies as a background principle, the public trust doctrine logically must constitute a background principle of state law, substantially because it defines a state government's authority to prevent harm (Hudson, 2009). In fact, during oral arguments in *Lucas*, the Supreme Court invited counsel for South Carolina to assert the public trust as a basis for the coastal protection statute and to claim the public trust as a background principle of South Carolina state law[76] (Hudson, 2009). A broadened public trust will incorporate government authority to regulate a broad scope of ecological resources into the background principles of state law. Expanding the scope of the public trust, therefore, will provide the basis for extending the *Lucas* exception to ecological regulations that limit land uses in order to protect important ecological resources for the community of state citizens.

An Ecological Trust doctrine can also prevent a court from finding that land stewardship regulations constitute takings by tipping the scales against takings in the multi-factor *Penn Central* test for regulatory takings. If a regulation does not deny all economically viable use, it may still constitute a taking under the *Penn Central* multi-factor test, which evaluates whether a regulation constitutes a taking under the circumstances surrounding its application. Two of the factors in the test include the character of the government action and the economic impact of the regulation (*Kaiser Aetna v. U.S.*, 444 U.S. 164, 175 (1979)). Generally, if the government action prevents harm, a court will not find it to be a taking. While regulations requiring stewardship could be construed as providing community benefits, their primary function is preventing further ecological harm. An Ecological trust doctrine implicitly characterizes land uses that fragment and degrade ecosystems as harmful and government action to protect important ecological components within the landscape as the prevention of social harm. We have developed more than sufficient scientific knowledge to understand that land uses that degrade ecological elements of the landscape or convert habitat into urban development bring harm to the landscape's ecological functioning; this ecological harm in turn harms human inhabitants of the landscape. Courts should consider such scientific knowledge as fitting within the "changed circumstances" that compel a rethinking and reformulation of the common law (*Lucas*, 505 U.S. at 1031). Because most government actions under the Ecological Trust duty

[76] According to Hudson (2009), if the state had made this argument, the statute may have survived.

will consist of efforts to protect valuable ecological attributes of the landscape, land stewardship regulations will fall on the side of preventing harm and should not constitute regulatory takings.

With respect to the economic consideration in the multi-factor takings test, courts should consider not only the economic loss that might result from the regulation, they should also consider the economic costs of the land uses that the regulation limits or prevents and the economic gains that might result from the regulation in terms of ecosystem services benefits. A consideration of only the economic costs to the regulated landowner is insufficient and fails to consider all costs and benefits associated with the regulation. Courts should therefore update the multi factor takings test to include the ecological implications of contested regulations and the land uses that are impeded.

Upholding ecological regulations against takings challenges will be less tenuous under an Ecological Trust than under the current, more narrowly construed public trust doctrine. Expanding the public trust will also encompass elements of the ecological composition and function of the various landscapes within the state as important natural resources subject to the public trust duty. An Ecological Trust doctrine provides state governments with a clear authority and obligation to act to protect important ecological features of the landscape. In addition to authority, an Ecological Trust doctrine provides grounds for upholding regulatory actions by effectively extending the Lucas exception and overcoming regulatory takings analysis.

Updating the definition of "harm" under the nuisance doctrine to incorporate ecological harm is the second necessary modification of the common law to ensure ecological regulations are enforceable. An updated nuisance doctrine will provide a strong legal basis for compelling landowners to stop ecologically unsound land uses as well as a strong legal argument for upholding regulations against takings challenges. As explained above, the nuisance doctrine is a common law tool that can invalidate land uses that interfere with public rights or with the "use and enjoyment" of private property. The remedies for nuisance generally include injunction to stop the harm and damages to compensate for them (Ruhl, 2008). Ecological nuisance claims would be most concerned with injunctive relief, which is consistent with compelling stewardship and enforcing land-use limitations.

The nuisance doctrine as it exists currently is "unabashedly anthropocentric" (Nagle, 2008, p. 810). Nuisance is generally defined in terms of harm to human interests, either to human uses of land or public rights; it fails to address impacts to the integrity of ecological systems or harm to other animal or plant species. To protect the underlying ecological components of the landscape, we need go beyond recognizing only harm to human uses or interests as nuisances to recognize landowner actions that destroy species or disrupt ecosystem processes as harmful as well. The nuisance doctrine provides a useful tool to stop bad land uses, but we need a legal basis within the doctrine to ecological systems independently, apart from human interest in them. We ultimately need to incorporate harm to ecological systems within the scope of the nuisance doctrine—essentially, we need to develop an ecological side to nuisance. Furthermore, the nuisance doctrine once provided much of the "strength and content" of environmental law, but declined in use after the federal environmental laws were enacted. The doctrine still provides a powerful tool to curb environmental harm and should be used to address land uses on private property that existing environmental laws are currently ill equipped to handle.

An ecologically-oriented nuisance doctrine could provide three significant benefits: a method for compelling land stewardship, a means of enforcing limitations on ecologically degrading land uses, and an exception to takings challenges. Through nuisance claims, landowners themselves can compel stewardship from other landowners and enforce limits on their land uses. This provides an important supplemental mechanism to ensure government regulations are effective in practice. Under a nuisance doctrine that encompasses ecological harm, one landowner would have the capacity to challenge a neighboring landowner's actions that adversely impact surrounding ecological systems through a private nuisance claim.[77] For instance, if one landowner fills a wetland or removes all buffer vegetation along a waterway, a neighbor could challenge the actions as ecological nuisances because they alter hydrological functions, leading to flooding or increased sediment loads, among other ecological harms. The neighbor who challenges these actions is, in effect, enforcing legal limits on what the landowner can do, thereby compelling the landowner to assume stewardship responsibilities.

A nuisance doctrine that recognizes ecological harm would allow one owner to challenge a neighbor's actions, based not only on harm to the plaintiff's land but on harm to the landscape's ecological functioning. Similarly, multiple landowners could join together to challenge the actions of one or several landowners whose land uses interfere with conservation standards established by law. Essentially, an ecologically based nuisance doctrine would enable

[77] Private nuisance and public nuisance are explained in more detail in Chapter 5.

residents of a landscape to uphold the landscape's ecological integrity for the benefit of the landscape itself and its living residents, humans included.[78] Going even further, nuisance law could allow landowners to challenge questionable land uses before they have begun—before the harm has taken place—under the doctrine of anticipatory nuisance actions. Many courts acknowledge anticipatory nuisance (Ruhl, 2008). Because anticipatory nuisance provides a useful tool, we should seek to expand its application to all jurisdictions.

An expanded nuisance doctrine might also include an expansion of the category of activities that qualify as public nuisances because they interfere materially with common rights and public health, including ecological health. Courts have generally held that interferences with the public health and welfare constitute public nuisances (Restatement, §821B cmt. b). Actions that alter hydrologic flows can impact the public access to clean and sufficient water supplies, thereby qualifying as a public nuisance. Once clear conservation goals are established through land-planning processes, actions by landowners that disrupt achievement of these goals could also qualify as public nuisances, even if they do not violate particular regulatory standards. Essentially, an ecological conception of nuisance should establish two strands of public ecological nuisance: impacts on public rights and impacts on ecosystem health.[79] While the impacts on ecological function or human health and wellbeing are likely to be less readily apparent than other types of public nuisances, such as a factory billowing smoke and soot over a town, they are nevertheless present and of significant public concern.

Ruhl (2008) has proposed the more narrow, economically-oriented approach of integrating ecosystem services into the nuisance doctrine to develop "ecosystem services nuisance." According to Ruhl, relying on the ecosystem services concept as a basis for nuisance is intuitive and easily applicable using existing nuisance analysis. This approach to nuisance opens up the possibility of legal recourse for landowners impacted by externalities from neighboring land uses. It also enables landowners to compel stewardship and enforce land-use limitations by stopping land uses that impair ecosystem services, which are in essence actions

---

[78] While nuisance may give landowners the capacity to sue to protect ecological aspects of the landscape, the next major issue is whether landowners would actually utilize this capacity. In general, these types of nuisance suits will likely result predominantly from local conservation groups, in the form of public nuisance, which will be discussed *infra*.

[79] Opponents of this idea will likely argue that updating public nuisance in this way will result in all land uses becoming public nuisances, because landowners must "use" their lands. Where is the line between land use and ecological nuisance? This topic is beyond the scope of this paper, but courts will need to draw the line between valid land uses that do not substantially harm ecological function and land uses that degrade and impede it.

that result in ecological harm. In an ecosystem services nuisance claim, a plaintiff will generally claim that, by damaging ecological resources on his property and impeding the provision of important ecosystem services, the defendant is causing harm to plaintiff's use and enjoyment of his property (Ruhl, 2008). For example, a plaintiff who owns a coffee plantation that benefits from pollination services provided by the surrounding forest land could sue a neighboring landowner who wants to cut down the forest in order to build his own coffee plantation in an anticipatory private nuisance action because it will impair the plaintiff's receipt of ecosystem services.

Ruhl asserts that ecosystem services nuisance can apply in both public and private nuisance actions. The doctrine is particularly "ready made" for public nuisance (Ruhl, 2008). In the *Palazzolo* case, for example, the Rhode Island Superior Court on remand held that Palazzolo's plan to fill a wetland on his property constituted a nuisance because of the important ecological benefits the wetland served for the surrounding landscape (*Palazzolo v. State*, No. WM 88-0297, 2005 R.I. Super. LEXIS 108 (July 5, 2005)). Private ecosystem services nuisance actions may present greater challenges, but recognizing the importance of ecosystem services under the allowance for changed circumstances will help to advance private nuisance claims in this area. Furthermore, Ruhl proposes that, under ecosystem services nuisance, harm to critical natural capital, or ecosystem services that humans are least able to substitute, should constitute a *per se* private nuisance (2008). Ecosystem services nuisance is a practical concept to begin introducing in courts right away because it is not a far leap from existing nuisance law, using most of the same basic principles. The concept, however, remains tethered to economic and anthropocentric considerations. We should ultimately aim to adopt ecosystem services nuisance as a subset of the broader concept of ecological nuisance because ecological nuisance does not require, to assert the claim, human harm or human interests.

As a further benefit, an ecological nuisance doctrine would help conservation advocates defend land-use regulations against claims that they unlawfully take private property without paying compensation. Regulations that ban nuisances or nuisance-like activities would not interfere materially with private rights and would thus not amount to improper takings. They would instead build upon basic principles of property law, giving specific content to basic limits, and thus not significantly deprive owners of rights they possess under basic property law.

To enforce and maintain land stewardship regulations, we will need to update the common law public trust and nuisance doctrines to incorporate ecological considerations, including the duty to protect ecological resources and the ecosystem services they provide as well as a recognition that damage to ecological systems constitutes harm. Updating both of these doctrines will enable governments to compel stewardship, enforce regulations, and overcome takings challenges, thereby upholding conservation gains in the face of certain landowner resistance. A land use that perpetrates ecological harm should be seen as a public harm, and abatement of ecological harm constitutes the prevention of public nuisance, and thus should not constitute a taking or else should be subject to the *Lucas* exception. These common law updates are therefore important to ensure the effectiveness of a system of land stewardship law. The ecological trust and ecological nuisance doctrines will open up avenues of enforcement for both landowners and governments. A nuisance doctrine ecologically defined will provide a key method by which landowners will be able to assert their interest in the ecosystem services of the surrounding landscape and protect their interest in an asset that benefits the entire community. Once updated, these doctrines will require landowners to consider the impacts of their land use decisions on the human community and the ecological community and will finally begin to account for ecological externalities of land uses. In *Lucas*, Justice Scalia acknowledges the possibility of changed circumstances or new knowledge—it's time to exercise this contingency. Circumstances have changed and continue to change, and we have acquired much new knowledge that confirms the ecological harms that unrestricted land use can cause. It's time to ground land-use law in its most fundamental, basic element, an element that it currently lacks—the land itself.

VII. Summary

Habitat loss and fragmentation resulting from land conversion for human use is a primary cause of biodiversity loss, substantially contributing to what many scholars perceive as the sixth great mass extinction in earth's history. Currently in the United States, the driving force of land conversion is urbanization, often in the form of urban and exurban sprawl. The intensive land uses that accompany sprawl degrade remaining habitat patches by isolating them and otherwise threatening the plant and animal species living in the patches. Climate change further threatens biodiversity; warming temperatures and shifting weather patterns may alter existing habitat conditions so much that they become uninhabitable for many species. To adapt to a changing climate, species will likely attempt to shift their ranges to more hospitable conditions. Fragmentation will likely impede if not prohibit this shifting, compounding the ills of existing fragmentation.

Our challenge is to protect biodiversity by providing room for nature[80] to adapt and respond on its own to human and environmental stresses. To do that, we must find ways to address human land–use needs while also protecting biodiversity and maintaining healthy ecological systems. This is the work of conservation, undertaken on a landscape scale.

This thesis addresses the challenge of biodiversity protection in fragmented landscapes paying particular attention to landscapes, such as Illinois, where private lands predominate. It explains the ecological strategies and goals that should properly guide landscape-scale conservation. It then proposes how we might best utilize land-use laws and policies to implement these strategies and goals.

Conservation work in fragmented landscapes necessarily begins with the landscape itself, which is best understood by drawing upon several scientific fields (chiefly landscape ecology) and paying particular attention to the landscape's spatial patterns and the ecological processes interconnected with these patterns. Fragmentation—the splintering of once-contiguous habitat into separate, isolated patches—alters spatial patterns and biotic and abiotic processes, not just at the landscape scale but within each patch. For instance, changes in the spatial composition of a patch can affect its hydrological and soil conditions. Fragmentation also alters the microclimatic conditions at the edges of a patch, causing "edge effects." These changes in turn

[80] This use of "nature" is broad, encompassing ecological processes, biotic communities, and individual species.

affect the species that live in the patch by altering the density of species, the ability of species to disperse to neighboring habitat, interrelationships among species, and even the genetic diversity within particular species. Conditions in the matrix surrounding habitat patches also alter ecological conditions within a patch, permeating patches and reducing connectivity between patches.

In various ways corridors that connect isolated habitat patches can help mitigate the adverse effects that fragmentation has on various species. Corridors come in many forms, from narrow strips of land to wide, broad landscape linkages. How a particular corridor functions is largely determined by its spatial structure and the vegetation it supports. A corridor's effects also vary among species; a corridor might facilitate dispersal for one species while doing little to help another. Corridors can function in various ways: they can facilitate movement as a conduit; they can provide additional habitat; they can act as a buffer or barrier to hinder movement. Scientists debate the ecological value of corridors, yet studies have shown that, overall, they do facilitate movement for many species and otherwise help mitigate the ill effects of fragmentation. We should exercise caution when planning and implementing corridors, clarifying their intended purposes and functions. Still, in heavily-fragmented landscapes, corridors are one of the few viable options for improving connectivity and thereby promoting large-scale conservation. This thesis proposes that we develop networks of corridors within and across landscapes, and that such corridor networks form the center of landscape-scale conservation strategies.

Corridor networks are easiest to imagine in regions of the United States with large blocks of contiguous habitat that are federally-owned or otherwise in public hands. Corridor-creation is much harder in landscapes that are heavily fragmented and dominated by private land ownership, like those found throughout much of Illinois. Some conservation organizations, such as Michigan Wildlink and Chicago Wilderness, have begun to assemble corridors in such landscapes. We must greatly broaden these efforts and also move, through legal means, to protect the resulting corridors and give them greater permanence.

The first step in landscape-scale conservation is to craft an overall goal for the work. A sound goal is properly framed in terms of the land's ecological functioning; its ability to sustain primary productivity, to cycle and retain nutrients, and to maintain the hydrological flows and biological populations that give the community resilience. An attractive goal, used in this thesis, is the one proposed by Aldo Leopold some decades ago, the goal that he summed up as "land health."[81] We can best promote land health by pursuing three courses of action: (1) preserve existing habitat; (2) stop bad land uses; and (3) restore degraded habitat. When undertaken concurrently, these three courses of action can form the building blocks of landscape-scale networks of conservation corridors.

A holistic strategy, consisting of several interrelated elements, is needed to promote conservation at the landscape scale. A sound strategy should include: a comprehensive land-use plan, based on the ecological patterns and processes of the landscape or region; specific conservation priorities and goals; and a planning approach grounded in the Green Infrastructure concept of linked core-buffer-corridor systems. A strategy should also specify the purposes and functions of the corridors to be implemented. Corridors can be designed to meet the needs of select target species, such as keystone or umbrella species. They can be designed instead or in addition to preserve specific habitats, such as biodiversity "hot spots." Finally, a strategy should provide for adequate monitoring and adaptive management to learn from mistakes and respond to changed conditions. Combined, these elements can establish a clear purpose and structure for corridors, facilitating implementation and maximizing that chance that corridor networks actually work as intended.

A landscape-scale conservation strategy of this type is ecologically feasible. In practice, however, its implementation would doubtless encounter resistance, particularly in the case of the privately-owned lands that are necessary elements of corridor networks. As they address this resistance, conservation advocates will inevitably have to grapple with the institution of private property and individual property rights. Property rights are not inherent, "natural" rights that arise by virtue of land ownership. They are a product of law, and like all law, subject to change. Furthermore, property rights come with attendant responsibilities--to the surrounding human and land communities. Currently, the balance of rights is tipped heavily in favor of private uses, at the expense of surrounding communities.

Land laws in the United States are highly fragmented, consisting of diverse laws, regulations, ordinances, and court decisions from multiple authorities--the federal government, state and local governments, and state courts. At the moment, a variety of legal tools limit

[81] This thesis recognizes that Leopold's concept of land health is broad, encompassing all biotic and abiotic processes, including hydrologic functions and soil processes. While recognizing that all elements are integral to the establishing land health, this thesis addresses only the biodiversity element of the concept; the other factors are beyond its scope.

private land uses for conservation purposes although even in combination they fall far short of promoting full conservation. On the federal level, the Endangered Species Act limits certain habitat modifications that immediately harm an endangered or threatened species. Several other federal schemes, including the Conservation Reserve Program, Swampbuster, and the Clean Water Act, include protections for ecologically sensitive lands, particularly wetlands. On the local level, municipal or county governments utilize zoning tools, including cluster zoning and planned unit development laws, to preserve open space and limit the spread of urban development. Some local governments have employed large-scale transfer of development rights schemes to restrict development and protect ecologically significant lands. Additionally, local governments, as well as land trust organizations, use conservation easements to prevent intensive land uses in protected areas. In the common law (that is, the body of accumulated judicial precedent), the nuisance doctrine operates as a limit on unreasonable activities that infringe upon a landowner's ability to use and enjoy his or her own land. All governments can also exercise eminent domain authority to directly acquire land--yet another legal conservation tool--provided land is used for public purpose and the government pays the landowner just compensation.

Even if a government enacts land-use limitations to protect sensitive lands, habitats, or species, landowners may put up substantial resistance. Landowners can challenge conservation-based restrictions as unconstitutional "takings" in violation of the Fifth Amendment's just compensation clause. In some states, landowners have successfully promoted referenda to enact laws that effectively keep local governments from further restricting land uses. Such challenges, fueled by a pervasive conception of broad, inherent property rights, impose a substantial barrier to new, conservation-based land-use regulations.

Overall, existing land use laws are insufficient to implement and sustain corridor networks on a landscape scale. In general, they take an uncoordinated, ad hoc approach to conservation; the proceed parcel-by-parcel rather than following an overall, landscape–scale plan. Too often they merely protect isolated islands of habitat in a landscape otherwise hostile to wildlife. Many current laws permit a range of land uses within protected areas that may reduce the quality of the land as functional habitat or as a functional corridor. Additionally, they often provide weak or temporary protections. They also fail to promote stewardship on the part of landowners.

We need to significantly update our system of land use law, transitioning it from land *use* to land *stewardship* law. This thesis proposes using existing lands and existing legal tools to implement a landscape-scale system of conservation through corridor networks. Existing state parks, wildlife preserves, recreation areas, as well as lands that have been protected by local land trusts form a foundation, however meager, upon which to build networks of corridors that span entire states, and eventually could link between states. Existing land use tools can also form the basis of a system of land stewardship law. To transition them into a system of land *stewardship* law, we must update their substance and purpose and also revise the ways in which we use them.

Essentially, we must ground existing land-use tools in the ecological aspects of the landscape. Additionally, we must promote greater inter-jurisdictional coordination: such coordination will be particularly important in establishing corridors across local government boundaries. To facilitate such coordination, state governments should require all governments, state and local, to establish comprehensive plans that map out the landscape, identifying particularly sensitive habitats. A middle-ground governance approach that combines elements of top-down and bottom-up approaches will facilitate effective development and implementation of land stewardship tools by involving local landowners in the process and providing for effective implementation and enforcement mechanisms.

We can use updated land use tools to implement the three main conservation goals outlined above. To *preserve habitat*, state and local governments can acquire lands directly through purchase or through their eminent domain authority. They can also establish conservation easements, making sure these easements are created with a clear and coordinated purpose. Local governments can utilize zoning tools to specifically create corridor zones and concentric, or core-buffer, zones—creating a system of "ecological zoning." They can also use other regulatory measures to require landowners to maintain open habitat, prohibiting the conversion or intensive use of such habitat on private lands. Additionally, state governments could pass a special statute to preserve open lands, such as an Undeveloped Lands Protection Act. To *stop bad land uses*, governments can compel stewardship by enacting a legal stewardship duty by statute, or courts could require stewardship by updating common law nuisance. Local governments could also enact regulations that prohibit bad land uses, requiring landowners to engage in stewardship indirectly by compelling them to refrain from ecologically harmful activities. To restore degraded habitats, local governments could develop "restoration zones" that require landowners within the zone to undertake restoration activities; they could also implement other regulations to require restoration activities. Courts could hold that unrestored lands constitute a nuisance. State and local governments could also acquire ecologically significant land and undertake restoration efforts as a state-sponsored activity, or allow land trusts to undertake restoration. When coordinated between multiple jurisdictions, these land use tools will establish the legal infrastructure necessary to implement corridor networks.

To enforce and maintain the conservation-based limitations on private land uses implemented through the various land stewardship tools described above, we will need to update the common law public trust and nuisance doctrines. Courts will need to expand the scope of the public trust doctrine to encompass a wide range of natural resources, including wildlife habitat. Essentially, we need an "Ecological Trust" doctrine that authorizes as well as obligates state and local governments to protect the ecological health of the landscape. Courts also need to update the nuisance doctrine to incorporate ecological harm as a valid basis for a nuisance action, establishing a form of "ecological nuisance." Alternatively, courts could adopt ecosystem services as a basis for nuisance actions, recognizing a form of "ecosystem services nuisance." (Ruhl, 2008). Both ecological and ecosystem services nuisance would provide a method of enforcing limitations on land uses that harm ecological functions within a landscape. Moreover, both an ecological trust doctrine and an updated nuisance doctrine will provide the legal basis for upholding conservation-based regulations against takings challenges.

Our challenge in today's changing climate and fragmented landscapes is to protect biodiversity from potentially massive waves of extinction. This will benefit the species protected as well as humans. To protect biodiversity in fragmented landscapes dominated by private ownership, we need a landscape-scale conservation strategy. At the center of this conservation strategy should be a plan to develop networks of corridors that will connect isolated patches of habitat within and across landscapes. To implement these corridor networks, however, we will need to enact changes to our current system of land use law, transitioning it, as noted, from land *use* to land *stewardship* law. A system of land stewardship law should reestablish a balance between landowner rights and responsibilities, requiring landowners to assume their rightful roles as stewards, as well as users, of the lands they own. Conservation of habitat and biodiversity should be a primary, not a secondary, consideration in the way we use land.

Even if we establish a landscape-scale conservation strategy and update our system of land-use law, the scope of our conservation efforts will always be limited in some way because our inability to use land well is, at its core, a cultural and social problem. True landscape-scale conservation will ultimately require cultural and social change, particularly in the way Americans view private property. To successfully develop and implement a landscape-scale conservation system across privately-owned lands, Americans, as a society, must embrace a collective ethic of land stewardship; we must accept the notion that private property rights come with related responsibilities to surrounding human and ecological communities. We must understand and enact the truth that Aldo Leopold long-ago recognized: "A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise." (1966, p. 262).

REFERENCES

Aber, J. D., & Melillo, J. M. (2001). *Terrestrial Ecosystems* (2d ed.). San Diego, CA: Academic Press.

Adams, J. (2009). Parks and protected areas: Conserving lands across administrative boundaries. In R. Knight & C. White (eds.), *Conservation for a new generation: Redefining natural resources management* (61-76). Washington, DC: Island Press.

Avoyelles Sportsmen's League, Inc. v. Marsh, 715 F.2d 897, 923-24 (5th Cir. 1983).

Bailey, S. (2007). Increasing connectivity in fragmented landscapes: An investigation of evidence for biodiversity gain in woodlands. *Forest Ecology and Management, 238,* 7-23.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R., . . . Turner, R. K. (2002). Economic reasons for conserving wild nature. *Science, 297,* 950-953.

Bayne, D. (2009). Bioregionalism and environmental regulation: A policy consideration for future environmental reforms. *University of Baltimore Journal of Environmental Law, 17*, 1-31.

Beardsley, K., Thorne, J. H., Roth, N. E., Gao, S., McCoy, M. C. (2009). Assessing the influence of rapid urban growth and regional policies on biological resources. *Landscape and Urban Planning, 93*(3-4)*,* 172-183.

Beier, P., & Noss, R. F. (1998). Do Habitat Corridors Provide Connectivity? *Conservation Biology, 12*(6), 1241-1252.

Benedict, M. A. & McMahon, E. T. (2006). *Green infrastructure: Linking landscapes and communities*. Washington D.C.: Island Press.

Berke, P. R. (2009). Integrating bioconservation and land use planning: A grand challenge of the twenty-first century. *Vermont Journal of Environmental Law, 10,* 407-433.

Blumm, M. C., & Grafe, E. (2007). Enacting libertarian property: Oregon's Measure 37 and its implications. *Denver University Law Review, 85*, 279-368.

Brawn, J. D., Robinson, S. K., & Thompson, F. R. III (2001). The Role of disturbance in the ecology and conservation of birds. *Annual Review of Ecological Systems, 32*, 251-276.

Bray, Z. (2010). Reconciling development and natural beauty: The promise and dilemma of conservation easements. *Harvard Environmental Law Review, 34*, 119-177.

Breggin, L., & George, S. (2003). Planning for biodiversity: Sources of authority in state land use laws. *Virginia Environmental Law Journal, 81*, 81-121.

Brooker, R. W., Travis, J. M. J., Clark, E. J., & Dytham, C. (2007). Modelling species' range shifts in a changing climate: The impacts of biotic interactions, dispersal distance and the rate of climate change. *Journal of Theoretical Biology, 245*(1), 59-65.

Brown, L. (2009). Where the wild things are. *Berkeley Science Review*. Retrieved from http://sciencereview.berkeley.edu/articles.php?issue=11&article=newspecies.

Bryant, M. M. (2006). Urban landscape conservation and the role of ecological greenways at local and metropolitan scales. *Landscape and Urban Planning, 76*, 23-44.

Callies, D. L., Freilich, R. H., & Roberts, T. E. (2004). *Cases and materials on land use* (4th ed.). St. Paul, MN: Thomson West.

Center for Biological Diversity, Inc. v. FPL Group, Inc., 166 Cal. App. 4th 1349 (2008).

Centner, T. J. (2006). Creating an 'Undeveloped Lands Protection Act' for farmlands, forests, and natural areas. *Duke Environmental Law & Policy Forum, 17*, 1- 61.

Chesapeake Bay Foundation. (2010). Our History. Retrieved from http://www.cbf.org/Page.aspx?pid=392.

Chicago Wilderness. (2010). Our Initiatives. Retrieved from http://www.chicagowilderness.org/initiatives.php.

Clean Water Act. 33 U.S.C. §§ 1251-1387 (2010).

Colburn, J. (2007). Bioregional conservation may mean taking habitat. *Environmental Law, 37*, 249-300.

Collinge, S. K. (1996). Ecological consequences of habitat fragmentation: Implications for landscape architecture and planning. *Landscape and Urban Planning, 36*(1), 59-77.

Collingham, Y. C., & Huntley, B. (2000). Impacts of habitat fragmentation and patch size upon migration rates. *Ecological Applications, 10*(1), 131-144.

Committee on Environment and Natural Resources, National Science and Technology Council. (2008). *Scientific assessment of the effects of global change on the United States*. Washington, DC: U.S. Climate Change Science Program.

Conservation Reserve Program. 16 U.S.C. §§ 3831-3835a (2010).

Conservation Resource Alliance. (2010). WildLink: How it works. Retrieved from http://www.rivercare.org/wildlink.

Cook, E.A. (2002). Landscape structure indices for assessing urban ecological networks. *Landscape and Urban Planning, 58*(2-4), 269-280.

Cooperrider, A. (1991). Introduction. In W. E. Hudson, (Ed.), *Landscape Linkages and Biodiversity* (141-148). Washington, DC: Island Press.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., . . . van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature, 387*, 253-260.

Cribbet, J. E., Johnson, C. W., Findley, R. W., & Smith, E. E. (2002). *Property* (8th ed.). New York, NY: Foundation Press.

Deal, B., & Shunk, D. (2004). Spatial dynamic modeling and urban land use transformation: A simulation approach to assessing the costs of urban sprawl. *Ecological Economics, 51*(1-2)*,* 79-95.

Debinsky, D. M., & Holt, R. D. (2000). A survey and overview of habitat fragmentation experiments. *Conservation Biology, 14*(2), 342-55.

Dietz, T., & Stern, P. C. (2008). *Public participation in environmental assessment and decision making*. Washington, DC: National Academies Press.

Dixon, J. D., Oli, M. K., Wooten, M. C., Eason, T. H., McCown, J. W., & Paetkau, D. (2006). Effectiveness of a regional corridor in connecting two Florida Black Bear populations. *Conservation Biology, 20*(1), 155-162.

Dolan v. City of Tigard, 512 U.S. 374 (1994).

Donald, P. F., & Evans, A. D. (2006). Habitat connectivity and matrix restoration: The wider implications of agri-environment schemes. *Journal of Applied Ecology, 43*, 209-218.

Echeverria, J. D. (2005). Regulating versus paying landowners to protect the environment. *Journal of Land, Resources, and Environmental Law, 26*, 1- 46.

Endangered Species Act. 16 U.S.C. §§ 1531-1544 (2010).

Epstein, R. A. (2009). Property rights, state of nature theory, and environmental protection. *New York University Journal of Law and Liberty, 4*, 1-35.

Ewing, R. (1997). Is Los Angeles-style sprawl desirable? *Journal of the American Planning Association, 63* (1), 107-120.

Fahrig, L. (2005). When is a landscape perspective important? In J. A. Wiens & M. R. Moss (eds.), *Issues and perspectives in landscape ecology* (3-10). Cambridge, UK: Cambridge University Press.

Farrier, D. (1995). Conserving biodiversity on private land: Incentives for management or compensation for lost expectations? *Harvard Environmental Law Review, 19*, 303-408.

First English Evangelical Lutheran Church v. County of Los Angeles, 482 U.S. 304 (1987).

Foreman, D. (2004). *Rewilding North America: A vision for conservation in the 21st century*. Washington, DC: Island Press.

Forman, 1995, R. T. T. (1995). *Land Mosaics: The ecology of landscapes and regions.* Cambridge, UK: Cambridge University Press.

Freyfogle, E. T. (2002). The tragedy of fragmentation. *Valparaiso University Law Review, 36,* 307-337.

Freyfogle, E. T. (2003). *The land we share: Private property and the common good*. Washington, DC: Island Press.

Freyfogle, E. T. (2007a). *Natural resources law: Private rights and collective governance*. St. Paul, MN: Thomson West.

Freyfogle, E. T. (2007b). *On private property: finding common ground on the ownership of land*. Boston, MA: Beacon Press.

Freyfogle, E. T. (2010). Property and liberty. *Harvard Environmental Law Review, 34,* 75-118.

Freyfogle, E. T., & Goble, D. D. (2009). *Wildlife law: A primer.* Washington, DC: Island Press.

Galatowitsch, S., Frelich, L., & Philips-Mao, L. (2009). Regional climate change adaptation strategies for biodiversity conservation in a midcontinental region of North-America. *Biological Conservation, 142*(10)*,* 2012-2022.

Gass, R. J., Rickenbach, M., Schulte, L. A., & Zeuli, K. (2009). Cross-boundary coordination on forested landscapes: Investigating alternatives for implementation. *Environmental Management, 43*(1)*,* 107-117.

Gilbert-Norton, L., Wilson, R., Stevens, J. R., & Beard, K. H. (2010). A meta-analytic review of corridor effectiveness. *Conservation Biology*. DOI: 10.1111/j.1523-1739.2010.01450.x

Gildor, D. L. (2005). Preserving the priceless: A constitutional amendment to empower congress to preserve, protect, and promote the environment. *Ecology Law Quarterly, 32*, 821-861.

Groom, M. J., Meffe, G. K., Carroll, C. R. (2006). *Principles of conservation biology* (3rd ed.). Sunderland, MA: Sinauer Associates.

Goldstein, P., & Thompson, B. H. (2006). *Property law: Ownership, use, and conservation*. New York, NY: Foundation Press.

Goldstein, R. J. (1998). Green wood in the bundle of sticks: Fitting environmental ethics and ecology into real property law. *Boston College Environmental Affairs Law Review, 25,* 347-430.

Gummer, J., Lang, I., Redwood, J., Mayhew, P. Sir, & Chalker of Wallasey, B. (1994). *Biodiversity: The UK action plan*. London: HMSO.

Guth, J. H. (2008). Cumulative impacts: Death-knell for cost-benefit analysis in environmental decisions. *Barry Law Review, 11*, 23-57.

Haines-Young, R. (2005). Landscape pattern: Context and process. In J. A. Wiens & M. R. Moss (eds.), *Issues and perspectives in landscape ecology* (103-111). Cambridge, UK: Cambridge University Press.

Harte, J. (2001). Land use, biodiversity, and ecosystem integrity: The challenge of preserving life's support system. *Ecology Law Quarterly, 27*, 929-965.

HAWAII REVISED STATUES § 205 et seq. (2010).

Hawkins, V., & Selman, P. (2002). Landscape scale planning: Exploring alternative land use scenarios. *Landscape and Urban Planning, 60*(4), 211-224.

Hay, K. G. (1991). Greenways and biodiversity. In W. E. Hudson, (Ed.), *Landscape Linkages and Biodiversity* (162-175). Washington, DC: Island Press.

Hess, G. R., & Fischer, R. A. (2001). Communicating clearly about conservation corridors. *Landscape and Urban Planning, 55*(3), 195-208.

Hicks, G. A. (2008). Protecting and promoting wildlife habitat on state and private land in Washington's arid interior. *Hastings West-Northwest Journal of Environmental Law and Policy, 14,* 781-830.

Hidding, M. C., & Teunissen, A. T. J. (2002). Beyond fragmentation: New concepts for urban-rural development. *Landscape and Urban Planning, 58,* 297-208.

Highly Erodible Lands Conservation Program. 16 U.S.C. §§ 3811-12 (2010).

Hilty, J. A., Lidicker, W. Z. Jr., & Merenlender, A. M. (2006). *Corridor ecology: The science and practice of linking landscapes for biodiversity conservation*. Washington, DC: Island Press.

Hudson, B. (2009). The public and wildlife trust doctrines and the untold story of the Lucas remand. *Columbia Journal of Environmental Law, 34,* 99-147.

Huntley, B. (2005). North temperate responses. In T. E. Lovejoy & L. Hannah (eds.), *Climate change and biodiversity* (109-124). New Haven, CT: Yale University Press.

Illinois Central Railroad v. Illinois, 146 U.S. 387 (1892).

ILL. CONST. art. XI § 1.

Illinois Department of Natural Resources. (2010). Species in Need of Greatest Conservation. Retrieved from http://dnr.state.il.us/ORC/WildlifeResources/theplan/species.htm.

Ingraham, M. W., & Foster, S. G. (2008). The value of ecosystem services provided by the U.S. National Wildlife Refuge System in the contiguous U.S. *Ecological Economics, 67*(4), 608-618.

Intergovernmental Panel on Climate Change (IPCC). (2007). *Climate change 2007: Synthesis report, Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Geneva, Switzerland: Intergovernmental Panel on Climate Change.

Johnson, S. J. (2004). It all comes out in the wash: *Sierra Club v. Meiburg*: Nonpoint source pollution continues unabated as the Eleventh Circuit refuses to permit implementation of total maximum daily loads through citizen suits. *Arkansas Law Review, 57,* 349-382.

Jooss, R., Geissler-Strobel, S., Trautner, J., Hermann, G., & Kaule, G. (2009). 'Conservation Responsibilities' of municipalities for target species: prioritizing conservation by assigning responsibilities to municipalities in Baden-Wuerttemberg, Germany. *Landscape and Urban Planning, 93*(3-4), 218-228.

Kahle, R. (2009). The 6th extinction and protected areas. *World Resources Institute Earth Trends*. Retrieved from http://earthtrends.wri.org/updates/node/356.

Kaiser Aetna v. U.S., 444 U.S. 164 (1979).

Karkkainen, B. C. (1997). Biodiversity and land. *Cornell Law Review, 83*, 1-104.

Karl, T. R., & Trenberth, K. E. (2005). What is climate change? In T. E. Lovejoy & L. Hannah (eds.), *Climate change and biodiversity* (15-28). New Haven, CT: Yale University Press.

Karp, J. P. (1993). A private property duty of stewardship: Changing our land ethic. *Environmental Law, 23*, 735-762.

Kelo v. City of New London, 545 U.S. 469 (2005).

Kim, J., & Ellis, C. D. (2009). Determining the effects of local development regulations on landscape structure: Comparison of The Woodlands and North Houston, TX. *Landscape and Urban Planning*, *92*(3-4), 293-303.

Kunstler, J. H. (1993). *The geography of nowhere: The rise and decline of America's man-made landscape*. New York: Touchstone.

Leopold, A. (1966). *A sand county almanac, with essays on conservation from Round River*. New York, NY: Ballantine Books.

Lindenmayer, D. B., & Fischer, J. (2006). *Habitat fragmentation and landscape change: An ecological conservation synthesis*. Washington, DC: Island Press.

Loretto v. Teleprompter Manhattan CATV Corp., 458 U.S. 419 (1982).

Lovejoy, T. E. (2005). Conservation in a changing climate. In T. E. Lovejoy & L. Hannah (eds.), *Climate change and biodiversity* (325-328). New Haven, CT: Yale University Press.

Lovejoy, T. E., & Hannah, L. (2005). Greenhouse gas levels and the future of biodiversity. In T. E. Lovejoy & L. Hannah (eds.), *Climate change and biodiversity* (387-396). New Haven, CT: Yale University Press.

Lucas v. South Carolina Coastal Council, 505 U.S. 1003 (1992).

Margles, S. W., Peterson, R. B., Ervin, J., & Kaplin, B. A. (2010). Conservation without borders: building communication and action across disciplinary boundaries for effective conservation. *Environmental Management, 45*, 1-4.

Mawdsley, J. R., O'Malley, R., & Ojima, D. S. (2009). A review of climate-change adaptation strategies for wildlife management and biodiversity conservation. *Conservation Biology, 23*(5), 1080-89.

McElfish, Jr., J. M. (2004). *Nature-friendly ordinances*. Washington, DC: Environmental Law Institute.

McInerny, G., Travis, J. M. J., & Dytham, C. (2007). Range shifting on a fragmented landscape. *Ecological Informatics, 2*, 1-8.

McLaughlin, N. A. (2005). Rethinking the perpetual nature of conservation easements. *Harvard Environmental Law Review, 29*, 421-521.

Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being: Biodiversity synthesis*. Washington, DC: World Resources Institute.

Mueller, J. M., Hellmann, J. J. (2008). An assessment of invasion risk from assisted migration. *Conservation Biology, 22*(3), 562-67.

Nagle, J. C. (2008). Common Law Environmental Protection: From Swamp Drainage to Wetlands Regulation to Ecological Nuisances to Environmental Ethics. *Case Western Reserve Law Review, 58*, 787-812.

National Oceanic and Atmospheric Administration (NOAA). (2010). The NOAA Annual Greenhouse Gas Index (AGGI). Retrieved from http://www.esrl.noaa.gov/gmd/aggi/.

National Research Council. (2008). *Water implications of biofuels production in the United States*. Washington, DC: National Academy of Sciences.

Natural Resources Conservation Service. (2007). *National resources inventory: 2003 annual NRI*. Washington, DC: U.S. Department of Agriculture.

Nature Conservancy, The. (2010). About Us. Retrieved from http://www.nature.org/aboutus/?src=t5.

Newton, J. L. (2006). *Aldo Leopold's odyssey*. Washington DC: Island Press.

Nollan v. California Coastal Commission, 483 U.S. 825 (1987).

Noss, R. F. (1994). Some principles of conservation biology, as they apply to environmental law. *Chicago Kent Law Review, 69*, 893-909.

Noss, R., Beier, P., & Shaw, W. (1998). *Evaluation of the Coal Canyon Biological Corridor*. Oregon State University.

O'Connell, M. A., & Noss, R. F. (1992). Private land management for biodiversity conservation. *Environmental Management, 16*(4), 435-450.

Olson, T. G. (1996). Biodiversity and private property: conflict or opportunity? In W. J. Snape III (ed.), *Biodiversity and the law* (67-79). Washington, DC: Island Press.

O'Neill, R. V. (2005). Theory in landscape ecology. In J. A. Wiens & M. R. Moss (eds.), *Issues and perspectives in landscape ecology* (23-28). Cambridge, UK: Cambridge University Press.

Opdam, P., & Wascher, D. (2004). Climate change meets habitat fragmentation: Linking landscape and biogeographical scale levels in research and conservation. *Biological Conservation, 117*(3), 285-297.

OREGON REVISED STATUTES § 197.352 (2010).

Orr, D. (1991). Biological diversity, agriculture, and the liberal arts. *Conservation Biology, 5*(3), 268-270.

Palazzolo v. Rhode Island, 533 U.S. 606 (2001).

Palazzolo v. State, No. WM 88-0297, 2005 R.I. Super. LEXIS 108 (July 5, 2005).

Parmesan, C. (2005). Biotic response: Range and abundance changes. In T. E. Lovejoy & L. Hannah (eds.), *Climate change and biodiversity* (41-55). New Haven, CT: Yale University Press.

Peck, S. (1998). *Planning for biodiversity: Issues and examples*. Washington, DC: Island Press.

Pendall, R. (1999). Do land-use controls cause sprawl? *Environment and Planning B: Planning and Design, 26,* 555-571.

Penn Central Transportation Co. v. New York City, 438 U.S. 104 (1978).

Pennsylvania Coal v. Mahon, 260 U.S. 393 (1922).

Petrosillo, I., Zaccarelli, N., Semeraro, T., & Zurlini, G. (2009). The effectiveness of different conservation policies on the security of natural capital. *Landscape and Urban Planning, 89*(1-2)*,* 49-56.

Pimm, S. L. (2008). Biodiversity: Climate change or habitat loss—Which will kill more species? *Current Biology, 18*(3), R117-119.

Pirnat, J. (2000). Conservation management of forest patches and corridors in suburban landscapes. *Landscape and Urban Planning, 52,* 135-143.

Powell, R. R. (2009). *Powell on real property*. (Vols. 1, 4, 9-13) New Providence, NJ: Matthew Bender.

Prah v. Maretti, 108 N.W.2d 182 (Wis. 1982).

Restatement (second) torts (1979).

Ricklefs, R. E. (2007). *The economy of nature* (5th ed.). New York, NY: W.H. Freeman and Company.

Rissman, A. R., Lozier, L., Comendant, T., Kareiva, P., Kiesecker, J. M., Shaw, M. R., & Merenlender, A. M. (2007). Conservation easements: Biodiversity protection and private use. *Conservation Biology, 21*(3), 709-718.

Rolstad, J. (2005). Landscape ecology and wildlife management. In J. A. Wiens & M. R. Moss (eds.), *Issues and perspectives in landscape ecology* (208-216). Cambridge, UK: Cambridge University Press.

Root, T. I., & Hughes, L. (2005). Present and future phonological changes in wild plants and animals. In T. E. Lovejoy & L. Hannah (eds.), *Climate change and biodiversity* (61-69). New Haven, CT: Yale University Press.

Rose v. Chaikan, 453 A.2d 1378 (Super. Ct. N.J. 1982).

Ruhl, J. B. (2008). Common law environmental protection: Making nuisance ecological. *Case Western Reserve Law Review, 58*, 753-785.

Ruhl, J. B., Kraft, S. E., & Lant, C. L. (2007). *The law and policy of ecosystem services*. Washington, DC: Island Press.

Saunders, D. A., Hobbs, R. J., & Margules, C. R. (1991). Biological consequences of ecosystem fragmentation: A review. *Conservation Biology, 1*(5), 18-32.

Shearer, K. S., & Ning-Xiang, W. (2009). Representing multiple voices in landscape planning: A land suitability study for a park land banking program in Concord, North Carolina, USA. *Landscape and Urban Planning, 93*(2), 111-122.

Shugart, H. H. (2005). Equilibrium versus non-equilibrium landscapes. In J. A. Wiens & M. R. Moss (eds.), *Issues and perspectives in landscape ecology* (36-41). Cambridge, UK: Cambridge University Press.

Simberloff, D., & Cox, J. (1987). Consequences and costs of conservation corridors. *Conservation Biology, 1*(1), 63-71.

Steiner, F. (2008). *The living landscape: An ecological approach to landscape planning* (2d ed.). Washington, DC: Island Press.

Tahoe-Sierra Preservation Council, Inc. v. Tahoe Regional Planning Agency, 535 U.S. 302 (2002).

Tarlock, D. A. (1993). Local government protection of biodiversity: What is niche? *University of Chicago Law Review, 60*, 555-613.

Theobald, D. M., Hobbs, N. T., Bearly, T., Zack, J. A., Shenk, T., & Riebsame, W. E. (2000). Incorporating biological information in local land-use decision making: Designing a system for conservation planning. *Landscape Ecology*, *15*(1), 35-45.

U.S. CONST. amend. V.

Wake, D. B. & Vredenburg, V. T. (2008). Are we in the midst of the sixth mass extinction? A view from the world of amphibians. *Proceedings of the National Academy of Sciences, 105*(Supp. 1), 11466-11473. Retrieved from http://www.pnas.org/content/105/suppl.1/11466.full.

Waller, D. M. (1996). Biodiversity as a basis for conservation efforts. In W. J. Snape III (ed.), *Biodiversity and the law* (16-32). Washington, DC: Island Press.

Ward, A. D., & Trimble, S. W. (2004). *Environmental hydrology* (2d ed.). Boca Raton, FL: CRC Press.

Ward, C. (2005). Symposium: Coming home: Instilling a connection to the land. *Journal of Land, Resources, and Environmental Law, 26,* 63-72.

Wehling, S., & Diekmann, M. (2009). Importance of hedgerows as habitat corridors for forest plants in agricultural landscapes. *Biological Conservation, 142*(11), 2522-2530.

Wetland Conservation Program. 16 U.S.C. §§ 3821-3824 (2010).

Wetlands Reserve Program. 16 U.S.C. §§ 3837-3837f (2010).

White, E. M., Morzillo, A. T., & Alig, R. J. (2009). Past and projected rural land conversion in the US at state, regional, and national levels. *Landscape and Urban Planning, 89*(1-2), 37-48.

Wildermuth, T. A. (2005). National land use planning in America, briefly. *Journal of Land, Resources, and Environmental law, 26*, 73-81.

Wildlands Network. (2009). Wildways: Creating landscapes for life. Retrieved from http://www.twp.org/wildways.

Williams, J. W., Jackson, S. T., & Kutzbach, J. E. (2007). Projected distributions of novel and disappearing climates by 2100 AD. *Proceedings of the National Academy of Sciences, 104*(14), 5738-5742.

Wood, M. C. (2009a). Advancing the sovereign trust of government to safeguard the environment for present and future generations (Part I): Ecological realism and the need for a paradigm shift. *Environmental Law, 39*, 43-89.

Wood, M. C. (2009b). Advancing the sovereign trust of government to safeguard the environment for present and future generations (Part II): Instilling a fiduciary obligation in governance. *Environmental Law, 39*, 91-139.