2

Landscape Dynamics in the Wildland–Urban Interface

Wayne C. Zipperer

Abstract

The wildland–urban interface represents landscape change—changes brought about by urbanization, by shifts in forest management, and altered disturbance regimes, each having ecological, social, and economic ramifications. In this chapter, I will focus on some of the ecological ramifications associated with landscape change, primarily forest fragmentation and deforestation, resulting from urbanization. In particular, I will review forest fragmentation from a landscape and site perspective; examine the ecological aspects of edges, corridors, and roads; and discuss fragmentation in relation to human health issues as they pertain to the wildland–urban interface.

Forest Fragmentation

In 1967, MacArthur and Wilson (1967) published their seminal book on island biogeography. They proposed that the larger the oceanic island, the greater the species richness. Likewise, the closer an oceanic island is to the mainland, the greater the species richness. The application of island biogeographic theory to terrestrial systems (a forest fragment in a sea of forest management, agriculture or urban lands) has resulted in a plethora of research on forest fragmentation including several reviews (Forman, 1995; Laurance and Bierregaard, 1997; Lindenmayer and Fischer, 2006; Collinge, 2009).

Forest fragmentation and loss are landscape processes. A review of the fragmentation literature, however, reveals a set of terms—such as *habitat loss*, *habitat fragmentation*, and *habitat degradation*—also being used to describe forest fragmentation. Following Lindenmayer and Fischer (2006), *habitat* refers to a particular set of environmental factors needed by a specific species. Thus, for this chapter, habitat fragmentation, loss, and degradation refer to the alteration or loss of suitable habitat for a specific species, whereas forest fragmentation, loss, and degradation refer to alteration of forest cover at the landscape and site level. This distinction is important. A forested landscape contains multiple habitats, and the modification of that landscape by human activities can affect the availability and spatial arrangements of those habitats and subsequently the species they contain differentially.

Forman (1995) conceptualized human modification of forested landscapes into five processes: perforation, dissection, fragmentation, shrinkage, and attrition (Fig. 2–1). Overall,

W.C. Zipperer, USDA Forest Service, P.O. Box 110806, Bldg. 164 Mowry Rd., Gainesville, FL 32611-0806 (wzipperer@fs.fed.us).
doi:10.2136/2012.urban-rural.c2

Urban–Rural Interfaces: Linking People and Nature, David N. Laband, B. Graeme Lockaby, and Wayne Zipperer, editors

5585 Guilford Rd., Madison, WI 53711-5801, USA.

fragmentation increases the number of patches, decreases mean patch size, increases mean patch isolation, and increases the ratio of patch edge to area (Fahrig, 2003). The actual effect of human modification depends on the spatial configuration of the landscape and the type of modification. Fahrig (2003) illustrated this point nicely by showing five scenarios of landscape modifications (Fig. 2–2). It is important to distinguish among the types of effects because of their effects on biodiversity. Some species may be affected negatively due to forest and habitat loss (e.g., interior species), whereas other species may be affected positively because of an increase of edge habitat, such as edge species.

In addition to modifying the landscape, what is replacing forest cover—forest management, agricultural use, or urbanization—is a critical aspect of the modification. This is particularly important with respect to the ability of an organism to move or disperse across the transformed landscape. Specifically, how permeable or resistant is the landscape to a species' movement (Hobbs and Yates, 2003)? Within the wildland–urban interface, urbanization brings with it an increase in road density and potential for increase in fencing and ditching, all of which may affect species movement across a landscape (see Boundaries and Edges section below). Further, with urbanization, there is the addition of domestic pets (cats and dogs) (see Lepczyk et al., 2004) and the introduction of nonnative species that alters biodiversity (see Chapter 5, Huebner et al., 2012, this volume).

Reported values of forest loss are important because they provide insights into how the landscape is changing. The values, however, do not report what is lost and where and how it occurs at the site level. First, the loss of forest cover is seldom random, and second, the remaining forest cover often occurs on land unsuitable for human use such as steep slopes. Consequently, the remaining forest cover is seldom representative of the unmodified forest and may not be reflective of previous habitat diversity, forest productivity, landscape processes, and biodiversity (Lindenmayer and Fischer, 2006). Although our focus has been at the landscape level, modification at the site level also has ramifications with respect to forest and habitat losses. For the wildland–urban interface, acknowledgment of site level modification or development is important since development is often tucked into existing forest cover to hide its presence (see Yaro and Hiss, 1996), or forest is cleared around structures to create a firewise landscaping (see Chapter 16, Mercer and Zipperer, 2012, this volume).

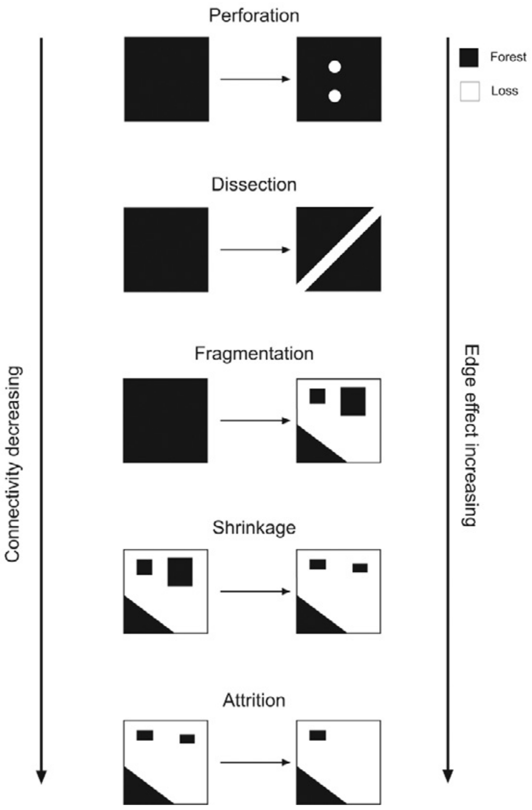

Fig. 2–1. Five ways of forest loss and changes in connectivity and edge effect in an urbanizing landscape (adapted from Lindenmayer and Fischer, 2006).

To identify the effects of site development at the site or patch level, Zipperer (1993) identified five site modifications that influence edge and interior habitats (Fig. 2–3). In the wildland–urban interface, development often creates internal and indentation development patterns because of single home development, whereas suburban development often results in cropping and removal. Indentation and internal patterns have the greatest significant effect on forest interior losses (Zipperer, 1993).

Boundaries and Edges

For this chapter, boundary is defined as the demarcation between two patch types, and an edge representing the biophysical environment created by the boundary. When we examine a map of a landscape, we observe a mosaic of different patch types, where patches are represented by an area that is relatively homogenous structurally, compositionally, or functionally (Forman, 1995). For terrestrial systems, the mosaic, itself, is often anthropogenically derived to help us interpret social, ecological, and economic patterns and processes of an area (Strayer et al., 2003). For instance, we map forest and nonforest lands, public and private ownerships, different types of private ownerships, and land use or land cover to better understand social and ecological patterns. So, the types of boundaries are often determined by the purpose or objectives of a study, management plan, or some other social, ecological, or economic need (Strayer et al., 2003). Nonetheless, in landscapes, boundaries exist naturally. For instance, they occur at the interface between land and water, rocky slopes and meadows, roots and soil, disturbed and undisturbed sites, and among different successional stages of a forest (see Belnap et al., 2003).

In the wildland–urban interface, we are converting forest cover to an urban land use, thus creating new boundaries and edges. Other boundaries created by human activities in the wildland–urban interface include those between managed and unmanaged forest lands, mined/excavated and unmined sites, agricultural fields and neighboring natural areas, and urban and nonurban lands. Often naturally occurring boundaries are more sinuous than human created boundaries, which tend to be more linear and sharp (Collinge, 2009). Therefore, it becomes important to distinguish the types of anthropogenic boundaries from each other to distinguish their social, ecological, and economic importance.

For our discussion, we will use the Strayer et al. (2003) classification of ecological boundaries to examine boundaries in the wildland–urban interface. Their classification is based on origin and maintenance, spatial structure, function, and temporal dynamics of the boundary. For urbanization, origin is anthropogenic and exogenous—arising from processes outside of the system. Unlike origin, maintenance is less well defined. Initially during the construction phase, a sharp boundary may exist between the urban land use and adjacent forest. Through time, a land owner may maintain the boundary between the developed site and forest, allow the canopy to close but the understory to remain cleared, or allow both the canopy and understory to develop and thus soften the distinction between the developed site and adjacent forest.

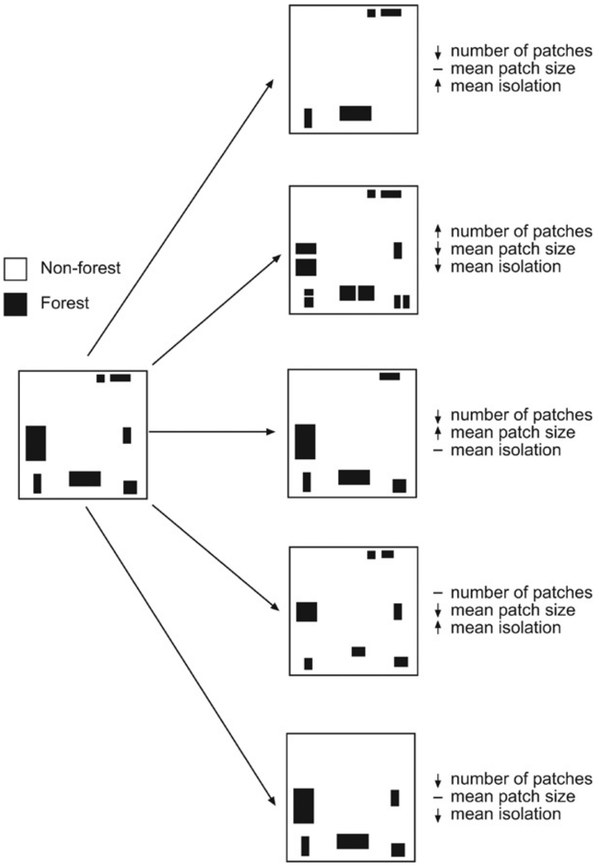

Fig. 2–2. Illustration of potential forest loss in an urbanizing landscape. Possible effects include changes to number of patches, mean patch size, and mean patch isolation. Arrows indicate actual changes (adapted from Fahrig, 2003).

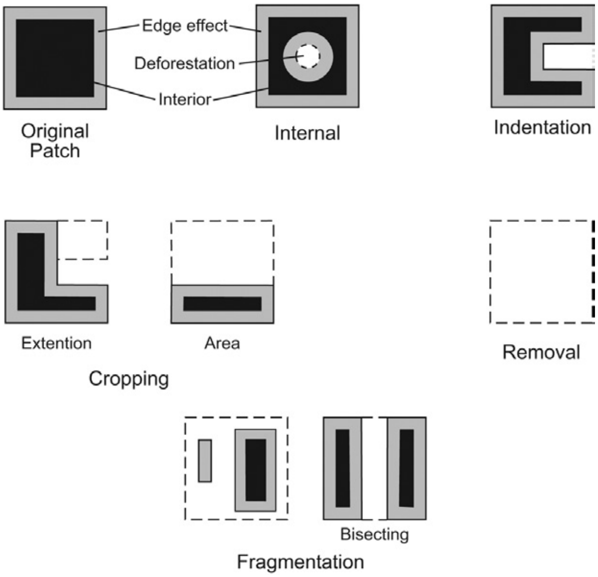

Fig. 2–3. Illustration of different deforestation patterns and their effects on forest interior and edge at the patch level (adapted from Zipperer, 1993).

Spatially, Strayer et al. (2003) identified 11 characteristics that describe the spatial structure of boundaries: grain size (resolution), extent, thickness and dimensionality, geometry of adjacency, interactive or noninteractive, abruptness/steepness, patch contrast, integrity, geometric shape and convolution, number of attributes, and offsets or congruencies of multiple attributes. All of these structural attributes are dictated by the focus of interest. Grain plays a particularly important role in the wildland–urban interface because it defines the resolution of analysis. At a broad-scale resolution, the canopy may seem contagious, with little indication of urban development, especially at low housing densities (see Riitters et al., 2004). By comparison, at a fine-scale resolution, the ecological interactions occurring at a site could be assessed. The presence of roads and housing becomes more predominant and their effects on ecological structure and function more discernable. For instance, at the residential lot scale, the presences of pets (cats and dogs) can be ascertained, and their ecological effects on local ground fauna can be assessed (Lepczyk et al., 2004). The other aspect of scale is extent, which is also defined by the question of interest and represents the landscape of interest. For instance, we may be interested in examining the effect of the occurrence of developed sites on different avian species. The extent for a wren would be quite different from that of a hawk (see Chapter 6, Reed et al., 2012, this volume). So, the types of boundaries and their frequency within a landscape will be determined by the type of information needed.

Another spatial element of a boundary is its dimensionality. What is the dimension of the boundary? Often, on maps, this dimensionality is represented by a thin line, but actuality boundaries are three-dimensional and represent a zone between contrasting patches, consist of a gradient which is greater than in the contrasting patches, and can be narrow or wide depending on that gradient (Cadenasso et al., 2003). For instance, the boundary between a developed site and adjacent forest can be quite sharp and narrow or soft and transitional, depending on its successional state and maintenance. Often in urbanizing landscapes, the dimensionality of a patch depends of the intensity and magnitude of maintenance.

Dimensionality is also an important element of edges, and is often referred to as edge effect or zone of influence (Collinge, 2009; Forman et al., 1997). Matlack (1993b) defines the edge effect as a zone of abiotic, biotic, and social conditions that differ from those found in the interior of a patch. Examples of altered abiotic conditions include temperature, light, and wind regimes; humidity levels; and levels of precipitation and nutrient deposition (see Lindenmayer and Fischer, 2006). Examples of altered biotic conditions include elevated nest predation and parasitism, modified dispersal patterns for animals and plants, changes in levels of insect activities and soil organisms, and changes in density, reproduction, growth and mortality of plants (see Lindenmayer and Fischer, 2006). Examples of social differences include vandalism, trampling, and dumping often occurring within 40 m of the edge (Matlack, 1993a).

Because of the differential responses of species to forest edges, species have been classified into edge and interior species. Edge species are those species that occur or prefer edge habitat, whereas interior species are those species that occur principally at some distance away from the edge. For plant species in temperate forests, Ranney et al. (1981) observed an average distance of 30 m from an edge.

Ries and Sisk (2010) questioned the applicability of classifying species as either edge or interior for faunal species, especially given the different types of spatial, temporal, and functional boundaries. Rather than reporting a species' response (positive or negative) to an edge, Ries and Sisk (2010) recommended that researchers report on the species' sensitivity to the presence of edges. Given the different configurations of edges and the variety of species' sensitivity, it may be more beneficial to think of responses along a "sensitivity spectrum" rather than a dichotomy—edge loving or edge avoiding. Using a resource-based conceptual model, Ries and Sisk (2004) predicted species responses (positive, negative, and neutral) based on the quality of resource availability between the two patches forming the edge. This approach accounts for not only different patch configuration and structure of the vegetation but also temporal components, such as seasonality, and has been successfully applied to a number of faunal species (Ries and Sisk, 2004).

Following forest edges in three different landscapes for 30 yr, Laurance et al. (2007) reported that edges are acutely sensitive to local environmental factors and the landscape context of the vegetation matrix neighboring an edge. Laurance et al. (2007) hypothesized that fragments within a landscape will tend to have similar vegetation dynamics and trajectories. Subsequently, with time, a homogenization of species composition occurred in fragments in a landscape, whereas a divergence in species composition occurred in fragments in different landscapes. McKinney (2002) reported homogenization across urban landscapes, resulting primarily from forest loss and the introduction of nonnative species (see Chapter 5, Huebner et al., 2012, this volume). The homogeneity reported by Laurance et al. (2007) is different. It is a local phenomenon within a landscape rather than across landscapes and may be the result of the high diversity of flora species in the tropics. Because of the existing vegetation matrix (native and nonnative species) and local disturbance regime, patterns of convergence for forest fragments in the wildland–urban interface may vary by biome (i.e., nontropical and tropical), with regional convergence being observed in nontropical fragments and local convergence being observed in tropical fragments.

Another aspect of dimensionality is contrast. How distinct are adjacent patches (Strayer et al., 2003)? Ries et al. (2004) identified two primary attributes of contrast: differences in mean vegetation height and vegetation densities between adjacent patches forming the edge. For instance, the contrast between grass and forest is sharp (a high difference between mean vegetation heights). By comparison, the contrast between a grassland and a wheat field is less sharp. Nilon (1996) examined the influence of contrast of land development on bird richness in the Lake of the Ozark, Missouri, USA. In this study, they surveyed bird species richness for three habitat types: wildland (no development), low development (low contrast to wildlands), and traditional development (high contrast—the forest was cleared and the clearing was maintained). They found bird species richness, at least, for low density housing and wildlands did not differ significantly, but when traditional development was compared with wildland, bird richness was less in the developed lands, and there was a shift in composition. The traditional development had more species preferring open habitat than species observed in the wildland. Although the low density development did not appear to affect bird species richness significantly, it may affect other wildlife species, especially those sensitive to road density (see below). Responses to high and low contrast are species dependent, but additional research is needed to evaluate species responses (Ries et al., 2004). The other aspect of contrast is vegetation density, more specifically boundary permeability or integrity in response to energy, materials, and species (Strayer et al., 2003). In fact, Cadenasso et al. (2003) pointed out that boundaries occupy a relative small portion of the total area of a landscape but are critical control points for the flow of energy, material, and species. By manipulating vegetation density in a forest remnant adjacent to an abandoned agricultural field, Cadenasso and Pickett (2001) and Weathers et al. (2001) showed differential herbivory, seed dispersal, and nutrient deposition into the forest. Again, these examples indicate how maintenance of the boundary between a developed site and the adjacent forest in the wildland–urban interface may govern the movement of energy, material, and species across the landscape.

Another attribute of boundaries with application to the wildland–urban interface is temporal dynamics and legacy (Strayer et al., 2003). With time, the forest edge of a boundary may become less permeable as stem densities increase. Likewise, the boundary may shift by removing adjacent vegetation to increase the probability of the home surviving a wildfire (see Chapter 16, Mercer and Zipperer, 2012, this volume). Laurance et al. (2007) recognized that edges are highly variable in space and time. Through their long-term study of Amazon rain forest, they observed that edges were only relatively predictable across spatial scales, primarily because of windstorms and the proximity and number of neighboring edges (Laurance et al., 2007). In general, fragments neighboring two or more edges had significantly more tree mortality than fragments with just one edge. This finding suggested an additive effect—edge effect is compounded by multiple nearby edges. Laurance et al. (2007) also observed that tree mortality was highest after the creation of an edge, primarily because trees were not physiologically adapted to the new microenvironments of the edge. This mortality declined with time. A similar observation of windthrows and mortality has been observed in silvicultural practices in temperate zones, especially when openings were 2 to 10 times the height of the canopy (Somerville, 1980).Thus, in the wildland–urban interface, the size of the opening and its maintenance may have significant effects on tree mortality and edge effect.

Connectivity

In addition to configuration of a forest fragment, connectivity can affect species occurrence and abundance (Collinge, 2009; Fahrig, 2003). Lindenmayer and Fischer (2006) actually identified three types of connectivity: habitat, landscape, and ecological. Habitat connectedness refers to the connectivity between habitat patches for a given species. Remember, as discussed earlier, habitat refers to the environment suitable for a given species. Hence, habitat connectivity is often species specific (Lindenmayer and Fischer, 2006). By comparison, landscape connectedness refers to the connection of vegetation cover within a given landscape and is often what is perceived by humans on maps, aerial photographs, and remote images. A high amount of landscape connectivity does not always translate into higher levels of habitat connectedness, and the effect will vary between species (Lindenmayer and Fischer, 2006). Finally, ecological connectivity refers to the connectedness of ecological processes, such trophic levels, hydrologic flows, and disturbance processes (Lindenmayer and Fischer, 2006).

Habitat, landscape, and ecological connectivity are not independent of each other but are interrelated. For instance, the loss of habitat connectedness for certain bird species may result in changes in seed dispersal, an ecological process. Similarly, altering landscape connectivity can affect tropic levels and the subsequent movement of energy across a landscape. In fact, isolation, through the loss of landscape connectivity, can affect species richness of a forest fragment (Collinge, 2009). In general, forest fragments connected by a forest corridor should have larger populations and potentially higher species richness than completely isolated fragments of equal size and shape (Collinge, 2009; Fahrig, 2003). Landscape context also plays an important role in landscape connectivity. Depending on the species, movement across different types of agricultural landscape may pose different risks to a species (see Gray et al., 2004). Likewise, the movement of a species across an urbanizing landscape poses different risks to a species than moving across an agricultural landscape (see discussion below).

To improve landscape connectivity, conservation strategies often recommend incorporating corridors into landscape planning and design. Corridors are purported to lower extinction rates, reduce demographic stochasticity, stem inbreeding depression, and fulfill an inherent need for movement (Simberloff et al., 1992). Yet, corridors often fail at achieving their management objective, primarily because landscape connectivity is being managed when habitat connectivity is needed (Lindenmayer and Fischer, 2006). Baudrey and Merriam (1988) actually identified the difference between structural connectivity (what is observed on the landscape) and functional connectivity (how it is being used). Hess and Fischer's (2001) reviewed the corridor literature and identified six ecological functions or roles that corridors play: filter, barrier, habitat, conduit, source, and sink (Fig. 2–4). When discussing the function of corridors it is important to recognize both biotic and abiotic influences. For example, corridors can act as filters and barriers (see the discussion below on roads and their effects on species), thus limiting species movement. For example, riparian fragments can be viewed as corridors that filter out nutrients (Schueler, 1995). Corridors often are viewed as conduits to aid species movements and the spread of diseases and invasive species. For examples, vehicles and trailers on roads, considered human corridors, were one of the primary reasons for the spread of Gypsy moth (*Lymantria dispar* L.) across the eastern United States (Sharov et al., 2002). Depending on the width, corridors can have a high edge/area ratio, creating habitat conditions favoring species that prefer edge habitats, thus increasing their reproductive success. An example would be nest parasitizers. Forman (1995) also discussed how roads can act as sources for contaminants. Similarly, corridors can act as sinks where species mortality exceeds reproduction success or places where materials such as nutrients are concentrated as in the case of riparian fragments. More research is needed to understand the ecological and economic costs and benefits

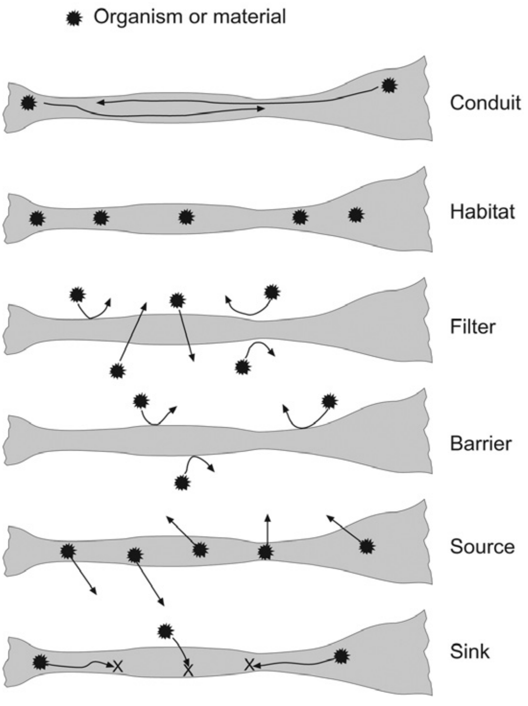

Fig. 2–4. Illustration of six different corridor functions as how they may affect the movement of an organism and material (adapted from Hess and Fischer, 2001).

of corridors and their function of linking forest fragments in an urbanizing landscape.

Roads

Roads are our primary corridors that connect our habitats. In 2009, there were approximately 6.7 million km of roads in the United States (USDOT Federal Highway Administration, 2009), and this road network has been designed for economic efficiency (Riiters and Wickham, 2003). Consequently, the travel distance to a road is minimized by an extensive road network. In the wildland–urban interface, there is an increase in road density to facilitate access and development. Riitters and Wickham (2003) reported that 18% of the total forest area in the conterminous United States is within 127 m of a road, and 81% is within 1061 m, or about 1 km. Only 1% of forest land is more than 5000 m away. In other words, very few forests are immune to the effect of roads. These effects include habitat loss, traffic mortality, resource inaccessibility, and the subdivision of flora and fauna populations (Jaeger et al., 2005).

To assess the effects of roads on ecological systems, Forman et al. (1997) suggested three indices: road effect or influence zone, road density, and road location. The influence zone of roads depends on what features—abiotic and biotic—are being considered (Collinge, 2009) (Fig. 2–5). Influences on abiotic factors generally range from tens to hundreds of meters and can include changes in humidity, temperature, water chemistry, and hydrology, and increases in dust and sediments from erosion, deposition of salt and heavy metals such as lead and zinc, nitrogen deposition, and site disturbances from vandalism, dumping, and fire (Botkin and Beveridge, 1997; Forman et al., 1997; Gatz, 1991; Lovett et al., 2000; Pouyat and McDonnell, 1991). These changes can alter stream dynamics and chemistry; forest decomposition, regeneration, and productivity; nutrient cycling; and ambient sound (Coffin, 2007). Influence on biotic factors generally range from tens to thousands of meters. Effects can include altered flora and fauna structure and composition, altered behavior, increased in windthrows and gap formation, occurrence of invasive species, and increased access by humans for recreational activities including hunting (Coffin, 2007; Forman, 1995; Laurance et al., 1997; Matlack, 1993a,b). These changes can alter the structure and composition of flora and fauna above and below ground, foraging and breeding behaviors, and dispersal and migration patterns (see Forman et al., 1997). Although extensive research has focused on many of these effects, additional research is needed to understand the effect of road juxtaposition to streams and forest patches in the wildland–urban interface.

Road density (kilometers per square kilometer) can have a significant effect on stream

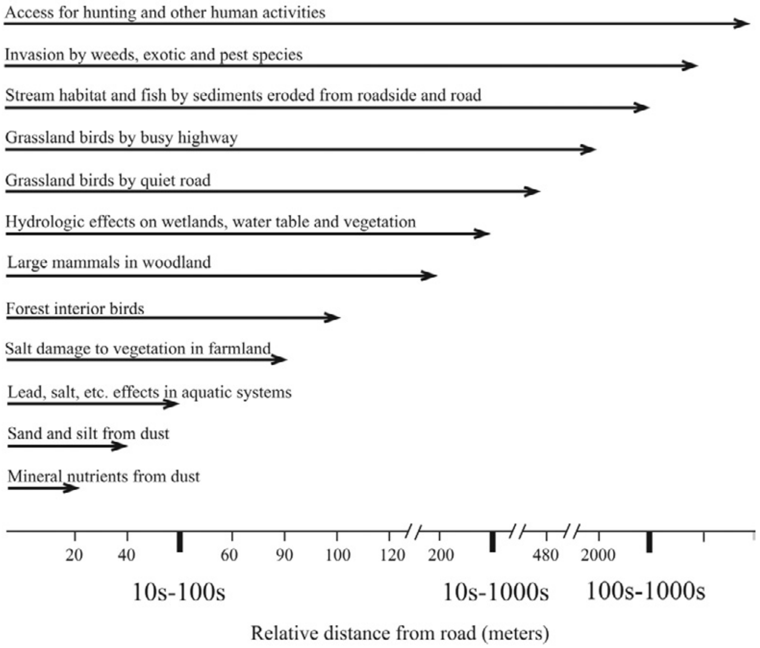

Fig. 2–5. Zone of influence for a road for abiotic and biotic attributes (adapted from Forman, 1995).

hydrology and on faunal populations because of avoidance, human access, and roadkills (Forman et al., 1997). A component of road density is its drainage network, the connection of culverts, ditches, and swales with adjacent streams (Coffin, 2007). As road density increases, its drainage network also increases, resulting in corresponding increases in storm runoff into connected streams. The resulting increase in stream peak flows from storm runoff is an increase in stream bank erosion, channel souring, and flooding (for more detail, see Chapter 3, Sun and Lockaby, 2012, this volume; and Paul and Meyer, 2001). Jones and Grant (1996) observed a significant increase in peak flows as road density increased in western Oregon. Alteration of stream banks, stream beds, and channel significantly affects stream biodiversity (Paul and Meyer, 2001).

From a faunal perspective, Forman et al. (1997) identified three mechanisms of how road density affects animal populations—road avoidance, human access, and traffic mortality—that have implications in the wildland–urban interface. Different species have different sensitivity to roads. As road density increases, species may alter their movements to avoid areas near roads. This avoidance, however, may be more related to human activities on those roads than the occurrence of the roads, themselves. For instance, Thurber et al. (1994) reported that gray wolves in Alaska avoid roads with high human activities but used closed roads for movement corridors. Mech (1989) reported a similar observation for wolves in Minnesota. Often a 0.6 km km^{-2} road density is reported as the maximum density for wolf survival. Above this density, wolf mortality increases. Mech (1989) reported that it was not necessarily the road density, but human activities (e.g., wolves being hit by cars) and increased human access (e.g., illegal hunting and trapping) that resulted in the higher mortality. Forman and Hersperger (1996) reported that a road density of 0.6 km km^{-2} appears to be the maximum threshold for other large mammal species, including deer, black bear, and elk. It should be noted that all of these species are also hunted legally or illegally and the observed density effect may be more related to greater human access than to the density itself. Nonetheless, Jaeger et al. (2005) identified three avoidance mechanisms—car, noise (light and smell), and road surface—that significantly affect species vulnerability to roads. Car avoidance behavior is associated with species that cross or use roads but try to avoid being hit, such as vultures feeding on carcass of species that did not avoid a vehicle. Although medium and large mammals are often seen dead along roadside, vulnerability to road mortality is especially pronounced in amphibians and reptiles. This group of species use roads for breeding habitats, thermoregulation, and often just do not avoid roads (Fahrig and Rytwinski, 2009). Species with high noise avoidance, such as birds (see Reijnen et al., 1996), will avoid noisy roads. Thus, the road becomes a barrier to movement, thereby limiting access to potential resources and breeding habitats. Noises, light, and smells are aspects of the zone of influence, as discussed by Forman et al. (1997). Similarly, for species with high road surface avoidance, such as small mammals (see MecGregor et al., 2008), the road becomes a barrier, thus limiting movement and potentially breeding opportunities. Unlike noise avoidance, road width becomes an important element of surface avoidance—the wider the road, the greater the avoidance (Jaeger et al., 2005). In their review, Fahrig and Rytwinski (2009) identified four primary groups of faunal species that are negatively affected by roads:

- any species that is attracted to roads and is unable to avoid individual vehicles (e.g., amphibians and reptiles);
- all species with large movement ranges, low reproductive rates, and low natural population densities (e.g., mountain lions);
- smaller mammals who avoid roads and suitable habitats near roads (i.e., resource inaccessibility);
- small animals that do not exhibit road avoidance and thus suffer from traffic mortality.

Fahrig and Rytwinski (2009) also identified two groups of species that are positively affected by roads:

- species that are attracted to roads for an important resource (e.g., food) and are able to avoid vehicles (e.g., vultures);
- species that do not avoid roads because of noises, smells, and other disturbances, who are able to avoid vehicles and whose main predators show a negative response to roads (predator release).

Harris and Scheck (1991) reported several reasons for increased mortality of species in association with roads. Roads can bisect territories, habitats, or migration routes. Roads can increase food sources, be attractive alternatives for movement, and create new habitats. These findings suggest that the juxtaposition of roads within a landscape is an important consideration when designing and placing roads (Coffin, 2007). In fact, Forman et al. (1997) reported that road location can have a significant effect on the zone of influence and species mortality. They also reported that road location is especially problematic when a road cuts through large forest patches or when there is a confluence of several habitat types. To test the importance of road placement and network, Jaeger et al. (2006) modeled different road densities and configurations. They observed that "bundling" roads reduced effects on wildlife, such as road mortality, and a grid network is less harmful than parallel roads when considering species with low road avoidance. Surprisingly, the grid network actually maintained core habitats better than the parallel roads. Further, they recommended that large forest patches should be protected from road construction. In the wildland–urban interface, new roads often cut into and through forest patches to facilitate access and development. The seasonality of human residence in the wildland–urban interface can also have an ecological effect by creating periods of low and high activity (see Chapter 14, Pickett et al., 2012, this volume). The ecological consequences of these actions, as outlined, can be significant for numerous species and across multiple scales. Pacing these recommendations into a wildland–urban interface context, development should be concentrated in one area rather than spread across the landscape. The number of roads should be minimized and bundled, and if necessary, a grid with larger undisturbed patches between roads should be used (see Theobald et al., 1997).

Whether species are avoiding roads because of human activities, the road itself, or the zone of influence (boundary and edge effects), there is a need for further research on the effect of roads, especially since the average road density in the United States is 0.73 km km^{-2}, and development in the wildland–urban interface adds to this density.

Infectious Disease

Land use change is a primary driver for infectious disease outbreaks and the risk they pose to humans and natural systems (Patz et al., 2004). In fact, land use changes, initiated by urbanization in the wildland–urban interface, can set the stage for an increase in prevalence of diseases and pathogens in several ways. The observed landscape transformation results in forest and habitat loss and degradation, an increase in the amount of edge, the creation of novel landscape configuration, and the alteration of connectivity (Collinge, 2009). All these changes affect the abundance and richness of not only diseases and pathogens but also their vectors and hosts. Similarly, land use changes may increase the susceptibility of a species to a disease due to stresses (limited resource accessibility), reduced genetic vigor (isolated breeding populations), and increased exposure created by a fragmented landscape (Collinge, 2009). In this section, I will focus on Lyme disease because of human susceptibility and its affinity to fragmented landscapes. The wildland–urban interface contributes to land conversions and thus increased human exposure through the construction of homes in affected areas.

Approximately 20,000 cases of Lyme disease are reported annually in humans in the United States and result from a bite primarily of the blacklegged tick [*Ixodes scapularis* (Say)] infected with the spirochete *Borrelia burgdorferi* (Diuk-Wasser et al., 2010). The disease often manifests itself as a circular rash, sore joints, and flulike symptoms. This disease, if left untreated, can be quite debilitating as it elicits an autoimmune response. Although infection is possible throughout the eastern United States, the disease is most prevalent in the fragmented forests of the Northeast and the upper Midwest (Diuk-Wasser et al., 2010).

The life cycle of blacklegged tick has three life stages—larval, nymphal, and adult—each requiring blood meals. Generally, larval and nymphal ticks feed on small forest floor animals such as chipmunks, mice, and birds. The adults commonly feed on white-tail deer (*Odocoileus virginianus*), but also will feed on small animals. Larval, nymphal, and adult ticks will feed on humans.

A tick becomes infected by feeding on a host with the spirochete. In a forest fragment, a common host for the larval and nymphal tick is the white-footed mouse (*Peromyscus leucopus*), which often is infected with the spirochete (Ostfeld, 1997). Field observations showed that the density of infected nymphs is inversely related to species richness (number of species). In other words, when species richness is high, the larval and nymphal ticks have greater probability of feeding on noninfected animals. This effect is known as the dilution effect—there are more uninfected animals for the tick to feed on than infected animals (Ostfeld and Keesing, 2000). Consequently, the disease risk is lower as humans are bitten less frequently by an infected tick. By comparison, when the density of white-footed mice is high and species richness is low, larval and nymphal ticks have a greater probability of feeding on an infected animal. Consequently, the disease escalates, and humans are bitten more frequently by an infected tick.

The high density of infected white-footed mice and low species richness is attributed to human-induced landscape changes. In the smaller forest fragments (<2 ha), predation may be suppressed and competitors may be absent because of habitat loss and degradation (Collinge, 2009). Under these conditions, the white-footed mouse population increases. In fact, the density of white-footed mice tends to be inversely related to fragment size (Krohne and Hock, 1999). In these smaller patches the white-footed mice becomes the dominant blood meals for the blacklegged tick (Allan et al., 2003). Additional research is needed to evaluate how human behavior may influence exposure, how different types of developments may affect white-footed mouse habitat (i.e., clumping development vs. dispersing development), and how different edge types play a role in the spatial heterogeneity of the landscape, tick occurrence, and species richness.

Summary

The wildland–urban interface sets up the precursors for major environmental changes because of land transformation by humans. Often in forested landscapes, development is the first permanent alteration in forest cover. Forest lands are fragmented, creating not only increased amount of edge but also different types of boundary edges that vary in their permeability and depth. The wildland–urban interface creates new road networks that increase human access and road mortality. The zone of influence by roads affects not only adjacent ecological systems but also those away from the road. Further, the combination of land use change and landscape legacy creates conditions that are favorable for many diseases that pose risks to humans, wildlife, forests, and agricultural lands (Patz et al., 2004).

When one considers that the wildland–urban interface represents more than 30% of the infrastructure in the United States (Radeloff et al., 2005), the ecological, economic, and social effects are significant. With the projected population growth for the United States, the wildland–urban interface will continue to grow and exert additional pressures on landscapes that are being stressed by a number of human activities, including climate change. With proper management and land use planning, these effects can be minimized. Whether we chose to continue on our current path of development or on a path that makes better use of our resources remains to be determined.

References

Allan, B.F., F. Keesing, and R.S. Ostfeld. 2003. Effect of forest fragmentation on Lyme disease risk. Conserv. Biol. 17:267–272. doi:10.1046/j.1523-1739.2003.01260.x

Baudrey, J., and H.G. Merriam. 1988. Connectivity and connectedness: Functional versus structural patterns in landscapes. In: K.F. Schreiber, editor, Connectivity in landscape ecology, Proceedings of the 2nd International Seminar of the International Association for Landscape Ecology. Munsteroche Geographische Arbeiten 29, Munster, Germany. p. 23–28.

Belnap, J., C.V. Hawkes, and M.K. Firestone. 2003. Boundaries in miniature: Two examples from soil. Bioscience 53:739–749. doi:10.1641/0006-3568(2003)053[0739:BIMTEF]2.0.CO;2

Botkin, D.B., and C.E. Beveridge. 1997. Cities as environments. Urban Ecosyst. 1:3–19. doi:10.1023/A:1014354923367

Cadenasso, M.L., and S.T.A. Pickett. 2001. Effect of edge structure on the flux of species into forest. Conserv. Biol. 15:91–97.

Cadenasso, M.L., S.T.A. Pickett, K.C. Weathers, S.S. Bell, T.L. Benning, M.M. Carreiro, and T.E. Dawson. 2003. An interdisciplinary and synthetic approach to ecological boundaries. Bioscience 53:717–722. doi:10.1641/0006-3568(2003)053[0717:AIASAT]2.0.CO;2

Coffin, A.W.. 2007. From roadkill to road ecology: A review of the ecological effects of roads. J. Transp. Geogr. 15:396–406. doi:10.1016/j.jtrangeo.2006.11.006

Collinge, S.K.. 2009. Ecology of fragmented landscapes. The Johns Hopkins Univ. Press, Baltimore, MD.

Diuk-Wasser, M.A., G. Vourc'h, P. Cislo, A. Gatewood Hoen, F. Melton, S.A. Hamer, M. Rowland, R. Cortinas, G.J. Hickling, J.I. Tsao, A.G. Barbour, U. Kitron, J. Piesman, and D. Fish. 2010. Field and climate-based model for predicting the density of host-seeking nymphal *Ixodes scapularis*, an important vector of tick-borne disease agents in the eastern United States. Global Ecol. Biogeogr. 19:504–514.

Fahrig, L.. 2003. Effects of habitat fragmentation on biodiversity. Ann. Rev. Ecol. Evol. Systematics 34:487–515. doi:10.1146/annurev.ecolsys.34.011802.132419

Fahrig, L., and T. Rytwinski. 2009. Effects of roads on animal abundance: An empirical review and synthesis. Ecology and Society 14(1):21. http://www.ecologyandsociety.org/vol14/iss1/art21/ (accessed 27 Feb. 2012).

Forman, R.T.T. 1995. Land mosaics. Cambridge Univ. Press, New York.

Forman R.T.T., D.S. Friedman, D. Fitzhenry, J.D. Martin, A.S. Chen, and L.E Alexander. 1997. Ecological effects of roads: Toward three summary indices and an overview for North America. In: K. Canters, A. Piepers, and D. Hendriks-Heersma, editors, Proceedings of Habitat fragmentation and infrastructure. Ministry of Transport, Public Works and Water Management, Delft, The Netherlands. p. 40–54.

Forman, R.T.T., and A.M. Hersperger. 1996. Road ecology and road density in different landscapes, with international planning and mitigation solutions. In: G.L. Evink et al., editors, Trends in addressing transportation related wildlife mortality. Publ. FL-ER-58-96. Florida Dep. of Transportation, Tallahassee, FL. p. 1–22.

Gatz, D.F. 1991. Urban precipitation chemistry: A review and synthesis. Atmos. Environ. 25B:1–15.

Gray, M.J., L.M. Smith, and R.I. Leyva. 2004. Influence of agricultural landscape structure on a southern High Plains, USA, amphibian assemblage. Landscape Ecol. 19:719–729. doi:10.1007/s10980-005-1129-3

Harris, L.D., and J. Scheck. 1991. From implications to applications: The disperal corridor principle applied to the conservation of biological diversity. In: D.A. Saunders and R.J. Hobbs, editors, Nature conservation 2: The role of corridors. Surrey Beatty, Chipping Norton, Australia. p. 189–220.

Hess, G.R., and R.A. Fischer. 2001. Communicating clearly about conservation corridors. Landscape Urban Plan. 55:195–208.

Hobbs, R.J., and C.J. Yates. 2003. TURNER REVIEW No. 7 Impacts of ecosystem fragmentation on plant populations: Generalising the idiosyncratic. Aust. J. Bot. 51:471–488. doi:10.1071/BT03037

Huebner, C.D., D.J. Nowak, R.V. Pouyat, and A.R. Bodine. 2012. Nonnative invasive plants: Maintaining biotic and socioeconomic integrity along the urban–rural–natural area gradient. In: D.N. Laband, B.G. Lockaby, and W. Zipperer, editors, Urban–rural interfaces: Linking people and nature. ASA, CSSA, and SSSA, Madison, WI. p. 71–90, this volume.

Jaeger, J.A.G., J.B. Bowman, L. Fahrig, D. Bert, J. Bouchard, N. Charbonneau, K. Frank, B. Gruber, and K.T. von Toschanowitz. 2005. Predicting when animal populations are at risk from roads: An interactive model of road avoidance behavior. Ecol. Model. 185:329–348. doi:10.1016/j.ecolmodel.2004.12.015

Jaeger, J.A.G., L. Fahrig, and K.C. Ewald. 2006. Does the configuration of road network influence the degree to which roads affect wildlife populations? In: L.C. Irwin, P. Garrett, and K.P. McDermott, editors, Proceedings of the International Conference on Ecology and Transportation. North Carolina State University, Raleigh, NC. p. 149–163.

Jones, J.A., and G.E. Grant. 1996. Peak flow responses to clearcutting and roads in small and large basins, Western Cascades, Oregon. Water Resour. Res. 32:959–974. doi:10.1029/95WR03493

Krohne, D.T., and G.A. Hock. 1999. Demography of Peromyscus leucopus populations on habitat patches, the role of dispersal. Can. J. Zool. 77:1247–1253.

Laurance, W.F., and R.O. Bierregaard Jr., editors. 1997. Tropical forest remnants. Univ. of Chicago Press, Chicago.

Laurance, W.F., S.G. Laurance, L. V. Ferreira, J.R.-d. Merona, C. Gascon, and T.E. Lovejoy. 1997. Biomass collapse in Amazonian forest fragments. Science 278:1117–1118. doi:10.1126/science.278.5340.1117

Laurance, W.F., H.E.M. Nascimento, S.G. Laurance, A. Andrade, R.M. Ewers, K.E. Harms, R.C.C. Luizão, and J.E. Ribeiro. 2007. Habitat fragmentation, variable edge effects, and the landscape-divergence hypothesis. PloS One 2:E1017. doi:10.1371/journal.pone.0001017.

Lepczyk, C.A., A.G. Mertig, and J. Liu. 2004. Landowners and cat predation across rural-to-urban landscapes. Biol. Conserv. 115:191–201. doi:10.1016/S0006-3207(03)00107-1

D.B. Lindenmayer, and J. Fischer. 2006. Habitat fragmentation and landscape change. Washington: Island Press.

Lovett, G.M., M.M. Traynor, R.V. Pouyat, W. Zhu, and J.W. Baxter. 2000. Atmospheric deposition to oak forests along an urban–rural gradient. Environmental Science and Technology 34:4294–4300. doi:10.1021/es001077q

MacArthur, R.H., and E.O. Wilson. 1967. The theory of island biogeography. Princeton: Princeton Univ. Press.

Matlack, G.R. 1993a. Sociological edge effects: Spatial distribution of human impact in suburban forest fragments. Environ. Manag. 17:829–835. doi:10.1007/BF02393903

Matlack, G.R. 1993b. Microenvironment variation within and among forest edge sites in the eastern United States. Biol. Conserv. 66:185–194. doi:10.1016/0006-3207(93)90004-K

McKinney, M.L. 2002. Urbanization, biodiversity and conservation. BioScience 52:883–890.

MecGregor, R.L., D.J. Bender, and L. Fahrig. 2008. Do small mammals avoid roads because of traffic. J. Appl. Ecol. 45:117–123.

Mech, L.D. 1989. Wolf population survival in an area of high road density. American Midland Naturalist 121:387–389. doi:10.2307/2426043

Mercer, E., and W. Zipperer. 2012. Fire in the wildland–urban interface. In: D.N. Laband, B.G. Lockaby, and W. Zipperer, editors, Urban–rural interfaces: Linking people and nature. ASA, CSSA, and SSSA, Madison, WI. p. 287–304, this volume.

Nilon, C. 1996. Wildlife conservation issues in ecological restoration of urban areas. Abstract. Society for Ecological Restoration, Madison, WI.

Ostfeld, R.S. 1997. The ecology of Lyme disease risk. Am. Sci. 85:338–346.

Ostfeld, R.S., and F. Keesing. 2000. Biodiversity and disease risk: The case of Lyme disease. Conserv. Biol. 14:722–728. doi:10.1046/j.1523-1739.2000.99014.x

Patz, J.A., P.Daszak, G.M. Tabor, A.A. Aguirre, M. Pearl, J. Epstein, N.D. Wolfe, A.M. Kilpatrick, J. Foufopoulos, and D. Molyneux. 2004. Unhealthy landscapes: Policy recommendations on land use change and infectious disease emergence. Environ. Health Perspect. 112:1092–1098. doi:10.1289/ehp.6877

Paul, M.J., and J.L. Meyer. 2001. Streams in the urban landscape. Ann. Rev. Ecol. Syst. 32:333–365. doi:10.1146/annurev.ecolsys.32.081501.114040

Pickett, S.T.A., M.L. Cadenasso, P.M. Groffman, and J.M. Grove. 2012. Importance of integrated approaches and perspectives. In: D.N. Laband, B.G. Lockaby, and W. Zipperer, editors, Urban–rural interfaces: Linking people and nature. ASA, CSSA, and SSSA, Madison, WI. p. 259–274, this volume.

Pouyat, R.V., and M.J. McDonnell. 1991. Heavy metal accumulations in forest soils along an urban–rural gradient in the southeastern New York, USA. Water Air Soil Pollut. 57–58:797–807. doi:10.1007/BF00282943

Radeloff, V.C., R.B. Hammer, S.I. Stewart, J.S. Fried, S.S. Holcomb, and J.F. McKeefry. 2005. The wildland–urban interface in the United States. Ecol. Appl. 15:799–805. doi:10.1890/04-1413

Ranney, J.W., M.C. Bruner, and J.B. Levenson, 1981. The importance of edge in the structure and dynamics of forest islands. In: R.L. Burgess and D.M. Sharpe, editors, Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York. p. 67–96.

Reed, S.E., H.E. Kretser, M.J. Glennon, L. Pejchar, and A.M. Merenlender. 2012. Faunal biodiversity at the urban–rural interface: Current knowledge, research priorities, and planning strategies. In: D.N. Laband, B.G. Lockaby, and W. Zipperer, editors, Urban–rural interfaces: Linking people and nature. ASA, CSSA, and SSSA, Madison, WI. p. 99–114, this volume.

Reijnen, R., R. Foppen, and H. Meeuwsen. 1996. The effects of traffic on the density of breeding birds in Dutch agricultural grasslands. Biodivers. Conserv. 75:255–260.

Ries, L., and T.D. Sisk. 2004. A predictive model of edge effects. Ecology 85:2917–2926. doi:10.1890/03-8021

Ries, L. 2010. What is an edge species? The implications of sensitivity to habitat edges. Oikos 119:1636–1642. doi:10.1111/j.1600-0706.2010.18414.x

Ries, L., R.J. Fletcher, J. Battin, and T.D. Sisk. 2004. Ecological Responses to habitat edges: Mechanisms, models, and variability explained. Ann. Rev. Ecol. Evol. Syst. 35:491–522. doi:10.1146/annurev.ecolsys.35.112202.130148

Riitters, K.H., and J.D. Wickham. 2003. How far to the nearest road? Front. Ecol. Environ. 1:125–129. doi:10.1890/1540-9295(2003)001[0125:HFTTNR]2.0.CO;2

Riitters, K., J. Wickham, and J. Coulston. 2004. Use of road maps in national assessments of forest fragmentation in the United States. Ecol. Soc. 9:13. www.ecologyandsociety.org/vol19/iss12/art13 (accessed 28 Feb. 2012).

Schueler, T., 1995. Environmental land planning series: Site planning for urban stream protection. The Center for Watershed Protection, Silver Spring, MD.

Sharov, A.A., D. Leonard, A.M. Liebhold, E.A. Roberts, and W. Dickerson. 2002. “Slow the Spread” a national program to contain the Gypsy Moth. J. Forest. 100:30–35.

Simberloff, D., J.A. Farr, J. Cox, and D.W. Mehlman. 1992. Movement corridors: Conservation bargains or poor investments. Conserv. Biol. 6:493–504. doi:10.1046/j.1523-1739.1992.06040493.x

Somerville, A. 1980. Wind stability: Forest layout and silviculture. N.Z. J. Forest Sci. 10:476–501.

Strayer, D.L., M.E. Power, W.F. Fagan, S.T.A. Pickett, and J. Belnap. 2003. A classification of ecological boundaries. Bioscience 53:723–729. doi:10.1641/0006-3568(2003)053[0723:ACOEB]2.0.CO;2

Sun, G., and G. Lockaby. 2012. Water quantity and quality at the urban–rural interface. In: D.N. Laband, B.G. Lockaby, and W. Zipperer, editors, Urban–rural interfaces: Linking people and nature. ASA, CSSA, and SSSA, Madison, WI. p. 29–48, this volume.

Theobald, D.M., J.R. Miller, and N.T. Hobbs. 1997. Estimating the cumulative effects of development on wildlife habitat. Landscape Urban Plan. 39:25–36. doi:10.1016/S0169-2046(97)00041-8

Thurber, J.M., R.O. Peterson, T.R. Drummer, and S.A. Tomasma. 1994. Gray wolf response to refuge boundaries and roads in Alaska. Wildlife Soc. Bull. 22:61–68.

USDOT Federal Highway Administration. 2009. Highway statistics 2009. http://www.fhwa.dot.gov/policyinformation/statistics/2009/ (accessed 22 Feb. 2012).

Weathers, K.C., M.L. Cadenasso, and S.T.A. Pickett. 2001. Forest edges as nutrient and pollutant concentrators: Potential synergisms between fragmentation, forest canopies, and the atmosphere. Conserv. Biol. 15:1506–1514.

Yaro, R.D., and T. Hiss. 1996. A region at risk: The third regional plan for the New York–New Jersey–Connecticut metropolitan area. Island Press, Washington, DC.

Zipperer, W.C. 1993. Deforestation patterns and their effects on forest patches. Landscape Ecol. 8:177–184. doi:10.1007/BF00125349