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1. Toward a Theory of Landscape Fire

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Landscape ecology is the study of relationships between spatial pattern and ecological process (Turner 1989; Turner et al. 2001). It is the subfield of ecology that requires an explicit spatial context, in contrast to ecosystem, community, or population ecology (Allen and Hoekstra 1992). One major theme in landscape ecology is how natural disturbances both create and respond to landscape pattern (Watt 1947; Pickett and White 1985; Turner and Romme 1994). Landscape disturbance has been defined *ad nauseum*, but here we focus on its punctuated nature, in that the rates of disturbance propagation are not always coupled with those of other ecological processes that operate more continuously in space and time. Disturbance can therefore change landscape pattern abruptly, and large severe disturbances can be a dominant structuring force on landscapes (Romme et al. 1998).

Fire is a natural disturbance that is nearly ubiquitous in terrestrial ecosystems (Fig. 1.1). Because fire is fundamentally oxidation of biomass, the capacity to burn exists virtually wherever vegetation grows. Occurring naturally in almost every terrestrial biome, fire and its interactions with ecosystems enable the study of landscape pattern and process under a wide range of climates and geophysical templates (Bowman et al. 2009).

Fire represents one of the closest couplings in nature of abiotic and biotic forces (Smithwick, Chap. 6). Fires are frequent, severe, and widespread enough in multiple regions and ecosystems to have served as a selective evolutionary force, engendering adaptive responses across a variety of plant and animal taxa (Bond and Midgley 1995; Hutto 1995; Bond and van Wilgen 1996; Schwilk 2003). Conveniently, the combustion process itself does not undergo evolutionary change. In that way it is unlike insects responsible for outbreaks, which evolve (and co-evolve) with host species over millennia (Royama 1984; Logan and Powell 2001). Fire as a physical and chemical process is fundamentally the same today that it was millions of years ago, and arguably will be the same a million years from now, although its behavior and effects on landscapes change with the development of ecosystems and vegetation.

Figure 1.1 here

Starting from simple triggers (lightning, striking a match), fire on landscapes develops into a complex spatio-temporal process both driven and regulated by abiotic and biotic factors (Johnson 1992; Johnson and Miyanishi 2001; Van Wagtendonk 2006). Fire behavior and fire effects reflect the relative strengths of multiple drivers, interacting at variable scales of space and time (Table 1.1). At fine scales (10^{-1}–10^{1}m^{2}), fire spread and intensity are conditioned by properties of fuel (mass, availability, spatial arrangement, and moisture), ignition (type, intensity, frequency, and spatial distribution), and ambient weather (air temperature, wind speed, and humidity). As a fire spreads over larger spatial scales (10^{1}–10^{3}m^{2}) other factors gain in importance, particularly topographic variation (aspect, slope, and slope position). As a result of these interactions, a fire can cover 5000 ha or more in a day, or smolder and creep through ground fuels for months.

The spatial and temporal scales of fire are intuitively observable and comprehensible by humans, although reconciling them quantitatively with the spatiotemporal domain of "normal" ecosystem processes introduces profound challenges, chiefly because of the different rates and scales at which processes occur. Fire can reset landscape processes and their spatial pattern, often across community and watershed boundaries, thereby forcing managers to take a landscape perspective. Planning at scales that are too fine will fail to account for disturbances that arise outside small management units; planning at scales that are too coarse, such as regional scales, will not account for local patterns of spatial and temporal variability and are in danger of applying one-size-fits-all solutions (Peterson et al., Chap. 10). Likewise, although fires occur as "events" over time spans of days to months, the postfire ecosystem response can unfold over decades to centuries. Landscape ecology provides a template for the analysis of both fire behavior and fire effects, and as a discipline offers the concepts and tools for understanding fire across scales (Turner et al. 2001; Falk et al. 2007).

A central concern in landscape ecology is the feedback that can exist between landscape pattern and ecological processes (White 1987; Turner 1989). In the case of fire, the mechanisms for this pattern-process dynamic are reasonably well understood at the fine scales for which fire behavior models were built (Johnson and Miyanishi 2001; Linn et al. 2006), albeit not always quantified accurately enough for reliable landscape predictions (Keane and Finney 2003; Cushman et al. 2007). As fire opens canopies, causes differential mortality, consumes standing biomass, affects watershed hydrology and soils, and prepares seedbeds, it acts as a powerful agent of landscape pattern formation. At the same time, however, the spread and behavior of fire depend explicitly on some of those very same landscape attributes, such as the distribution, type, age, and condition of vegetation. The spatial and temporal distributions of biomass and moisture influence the spread of fire, inhibiting the spread of fire where biomass is too scarce or too wet, and allowing fire to spread only where conditions are favorable to combustion. Fire is therefore a *contagious* disturbance (Peterson 2002), in that its intensity depends explicitly on interactions with the landscape.

The feedback between fire and landscape pattern is strong and ecosystem-specific, and provides a perfect illustration in nature of the interaction of pattern and process. Over time this pattern-process interaction creates *landscape memory*, a legacy of past disturbance events and intervening processes (Peterson 2002). This memory can be spatially sparse, but temporally rich, as with a spatial pattern of fire-scarred trees (Kellogg et al. 2008), or the converse, as with a landscape pattern of age classes and structural types (Hessburg and Agee 2005). Landscape memory extends to the less visible but no less important functional properties of ecosystems, such as biogeochemical processes (Smithwick, Chap. 6).

Fire effects illustrate this interaction of pattern and process. Fire consumes biomass as it spreads, producing a patch mosaic of burned areas on the landscape, whose heterogeneity reflects the combined effects of the spatial patterns of fuels, topographic variation, and microscale variation in fire weather. Burned areas produce characteristic patterns of spatial variability in severity and patch sizes. This tendency is the basis for the widespread use of remote sensing and geographic information systems (GIS) to quantify and evaluate fire as a patch-generating landscape process.

Remotely derived imagery has revolutionized the field of burn severity mapping, especially by greatly improving the precision and accuracy of characterizations of postfire environments (MTBS 2009). Both qualitative and quantitative metrics of burn severity can be derived from satellite imagery based on reflected and emitted electromagnetic radiation (Miller and Yool 2002; Holden et al. 2005; Key and Benson 2006). Although most burn severity work to date has used just two spectral bands from LANDSAT images at 30-m resolution, multi-spectral and panchromatic data are increasingly available at multiple resolutions as fine as 1m. Hyperspectral imaging (Merton 1999) and LiDAR (Lentile et al. 2006) also hold promise for more refined analysis of the three-dimensional structure of postfire landscapes.

A recently burned landscape is striking to look at. Spatial patterns of burn severity are often very heterogeneous, even within fires assumed to be stand-replacing (Fig. 1.2). Indices abound to quantify and interpret landscape spatial pattern (McGarigal et al. 2002; Peterson 2002), and have been used widely to understand spatial patterns specifically with respect to fire (Romme 1982; Turner et al. 1994). Our interest here, however, lies specifically in the processes that both generate and are controlled by that spatial pattern. For example, patterns of burn severity and the spatiotemporal structure of fire-scar records emerge from the cumulative effects of individual events and their interactions, but how these dynamic interactions play out over larger spatial and temporal scales is less well understood. A framework is needed for connecting these events and interactions that is conceptually and computationally feasible at the scales of landscapes. In this chapter we propose a theoretical framework that reduces the apparent complexity of ecosystem processes associated with fire. A full development of this theory would entail a formal structure for landscape fire dynamics and quantitative models for individual transformations of its elements (*sensu* West et al. 2009). Here we are content with suggesting a way of thinking about landscape fire that “streamlines” its complexity to a level that is tractable for both research and management.

Figure 1.2 here [but a and b should be on facing pages]

1.1 An Energetic Framework for Understanding Landscape Fire

Earth system processes reflect the distribution of energy across scales of space and time (Pielou 2001). The climate system, for example, is a direct manifestation of the flows of energy near the Earth's surface, including the uplift of equatorial air masses and major convection processes such as Hadley cells and atmospheric circulation, all of which redistribute incoming solar energy. Ocean circulation is likewise driven by system energetics, which are evident in three dimensions between deep and surface waters across thermohaline gradients and major quasi-periodic ocean-atmosphere couplings (El Niño Southern Oscillation, Pacific Decadal Oscillation, Atlantic Multidecadal Oscillation, North Atlantic Oscillation). Earth's fluxes of energy drive biogeochemical cycles that connect flows of materials and energy within and among ecosystems. Biogeochemical cycles, such as those of carbon and nitrogen, link the biotic and abiotic domains and reflect feedbacks between biological and non-biological components of the Earth system. Ecosystem ecologist H. T. Odum observed (1983) that biogeochemical cycles can be considered a form of energy flow at all scales, and that other ecological processes such as succession and productivity can be viewed as expressions of organized energetics.

The ecosystem energy perspective offers a general framework for understanding landscape fire as a biophysical process. Fire redistributes energy, and in doing so, can dramatically transform landscape pattern. Here we outline a framework for understanding the landscape ecology of fire from an energetic perspective. In this energy—regulation—scale (ERS) framework we view fire as an ecosystem process that can be understood by examining how energy is transformed and redistributed, subject to regulation, across scales.

1. **E**nergy. Incoming solar energy is the ultimate basis for plant growth and thus the fuels involved in combustion. Solar energy is also the basis for atmospheric circulation and the weather that influences moisture conditions of fuels and fire behavior. Vertical energy transfer in the atmosphere generates lightning, the primary non-human source of ignitions. The preconditions for fire are thus related inextricably to energy sources and fluxes.
2. **R**egulation. Ecosystems apply controls on the energy flux rates important to landscape fire. Forests store energy (fuel) as living and dead biomass aboveground and in soils, and the time it takes to accumulate a storehouse of biomass that will burn is subject to biotic and abiotic controls on growth and decomposition that vary across ecosystems (Aber and Melillo 1991). The energy fluxes associated with the combustion process itself are facilitated or constrained by atmospheric humidity, temperature, and air-mass movement (weather). Topography works in a similar fashion with landscapes having regions of low resistance to fire spread (e.g., steep slope gradients in the direction of wind) or high resistance (cliffs, lakes, persistent fuel breaks). Indeed

all three elements of the traditional "fire triangle"—fuels, weather, and topography—can be interpreted as ecosystem components involved in regulating the flow of energy across a landscape (Table 1.2).

3. **S**cale. Flows of energy and mass (stored energy) are concentrated at characteristic scales of space and time (Holling 1992). For example, the main regulators of combustion at the space and time scales of millimeters and seconds (combustible fuel mass and moisture, a heat input source, and sufficient oxygen to sustain combustion) are different from those that regulate fire occurrence at subcontinental and decadal scales (interannual to decadal variation in winter precipitation, spring and summer temperature and humidity, prior fire history and regrowth of flammable biomass). Between these two ends of the scaling "gradient", fire dynamics play out across landscapes, in ways that are more complex and heterogeneous, and less tractable to analyze.

Within this "ERS" framework, we can recast the standard pattern-process polarity in landscape ecology (Turner et al. 2001) by examining energy in landscape fire. Following basic physics, we partition energy into potential and kinetic energy. Potential energy (PE) is stored mostly in biomass, in the form of molecular bond energy. Increases in biomass (productivity) are affected by kinetic energy (KE) in the form of photosynthetically active radiation (PAR), and regulated by levels of soil and foliar moisture. The potential energy in biomass is transformed rapidly into kinetic energy during a fire. Heat flux (radiative, convective, conductive) is basic to the physics of fire spread. The spatial interplay of heat flux with the connectivity of potential energy in fuels manifests as contagion on the landscape. Rates and directions of fire spread are determined by the interaction of heat flux, generated by the transformation of potential energy in fuels and driven by fire weather, with landscape pattern (*regulation*), producing the observed complex spatial patterns of landscape fire.

Energy fluxes associated with physiological processes of photosynthesis and respiration, and the ecosystem level processes of growth and decomposition involved in succession, proceed at very different rates from the energy fluxes associated with fire. The heat transfer in fire spread is pulsed, whereas the fluxes in growth and decomposition are more or less continuous, albeit time-varying. Fire therefore represents a dramatic and relatively instantaneous transformation of potential energy to kinetic energy, in contrast to the slower transformations associated with stand dynamics, which ultimately convert the kinetic energy from the sun into potential energy stored as biomass (Fig. 1.3).

Interactions among energy fluxes, and their cumulative effects over time, are evident in feedbacks to the process of landscape fire. These feedbacks can be negative, where fire is self-limiting, or positive, where fire is self-reinforcing. Fire as a landscape process is governed by available biomass, terrain properties that influence combustion, and meteorological variables that affect ignition, wind speed, temperature, and humidity. As a fire occurs, it effects a transformation of biomass (as potential energy) into thermal (kinetic) energy, which is then redistributed within and beyond the site. This transformation drives fire effects, including redistribution of organic and inorganic compounds (in foliage and soil) and water. The postfire environment integrates the legacy of the prefire landscape and the energy transformation from fire behavior to generate a new landscape on which stored energy has been redistributed. In this way, fire behavior, fire effects, and postfire ecosystem changes combine to create landscapes with unique self-regulating properties (Fig. 1.4).

1.1.1 Self-Limiting Properties of Landscape Fire

The behavior and spread of fire on a landscape depend in part on current conditions (e.g., today's weather), and in part on the legacy of past fire events and subsequent ecosystem processes (e.g., the mosaic of flammable vegetation). By definition, in an ERS framework each fire—each combustion event—alters the distribution of stored energy in the form of fuels to create a new postfire environment. In prescribed surface fires, fire intensity is controlled such that consumption is limited to herbaceous and dead woody fuels, whereas canopy consumption can approach 100% of foliage and even small branches in a high-energy crown fire (Stocks et al. 2004). How long the legacy of this redistribution of stored energy persists, and the extent to which the landscape fuel mosaic resembles the pre-fire mosaic, depend on many factors, including the type of vegetation, fire intensity (heat output per unit time and space), fuel conditions (e.g., moisture content) at the time of the fire, and the productivity of the site, which governs how quickly vegetation can regrow.

Each fire alters the conditions for the next fire in the same location. Fire managers know well that the intensity and rate of spread often moderate when a fire spreads into a recently burned area. Indeed, such understanding is the basis for the widespread application of prescribed fire and wildland fire use (Mitchell et al. 2009). Behavior of wildfires burning under all but the most severe weather conditions moderates when fuel conditions are altered by thinning or prescribed fire (Agee and Skinner 2005; Finney et al. 2005; Maleki et al. 2007).

A similar self-limiting dynamic can also be seen in unmanaged landscapes. For example, in a study in the central Sierra Nevada, Collins et al. (2009) found that under all but the most extreme conditions, the spread of a fire slows when it burns into recently burned areas, with the most noticeable effects arising when the previous fire occurred less than 20 years ago. Similar self-regulating landscape properties have also been inferred in pre-management historical fire regimes (Taylor and Skinner 2003; Scholl and Taylor 2010). In this way, any one fire exerts a negative-feedback regulatory influence on the subsequent fire event, with varying periods of persistence. As this dynamic is ramified across many patches on the landscape, the result is self-regulation, which may be a fundamental property of fire as an ecosystem process (Moritz et al., Chap. 3). From the energetic perspective, these self-limiting interactions might be viewed as an equilibrium—if an uneasy one—regulated by cycles of conversion between potential and kinetic (thermal) energy (Fig. 1.3).

1.1.2 Self-Reinforcing Properties of Landscape Fire

Another kind of landscape regulation also occurs, the self-reinforcing case. The clearest example of this is the tendency of many vegetation types—grasslands, ponderosa pine, chaparral, and lodgepole pine forest—to create fire regimes that favor their perpetuation and expansion. This occurs because dominant species create the physical environment and fuel complex that govern the fire regime, and in turn the fire regime reinforces a competitive hierarchy that favors these species (Rowe 1983; Agee 1993). For instance, the architecture of lodgepole pine (*Pinus contorta*) forests in the interior West—dense stands of trees with high canopy connectivity—tends to favor crown fire propagation, which kills most trees, giving an advantage to cohort reproduction by lodgepole due to its evolved capacity for serotiny (FEIS 2009).

Figure 1.3 here

Similarly, the open stand structure of many southwestern ponderosa pine (*P. ponderosa*) stands creates an open layer of surface fuels and grasses that carries relatively low intensity surface fires, killing seedlings and maintaining an open forest structure while generally causing relatively little or no mortality among canopy trees (Allen et al. 2002). Many grassland ecosystems have self-reinforcing fire regimes, with cured grasses providing fuel for fast-moving fires that burn off cured foliage and kill seedlings of woody species, while little heat penetrates to the apical meristem of the grasses, which has evolved to survive precisely such events (Brown and Smith 2000).

Whereas landscapes that are controlled by the self-limiting dynamic occupy a basin of attraction, under some conditions "escape" from this basin occurs, and the system moves into a new dynamic space (Gunderson and Holling 2001). Escape from an attractor may arise from stochastic rare events, including forcing by exogenous factors. For example, repeated fires at unusually short intervals may inhibit the recovery of certain plant species, allowing colonization by new species and a shift in the successional trajectory (Keeley et al. 1981; Suding et al. 2004). Weather conditions that promote an unusually severe or extensive fire, such as extended droughts, can also alter successional patterns. If the new vegetation is more flammable, slower growing, or more or less susceptible to a local insect or pathogen, the shift in the disturbance/succession dynamic may be sufficient to move the landscape to a stable state in a new basin of attraction (Keeley et al., Chap. 8).

Climate change may accelerate these shifts to new basins of attraction, as disturbances such as fire change ecosystems abruptly. Coupled with other complicating factors like invasions, landscape self-regulation can become chaotic. For example, climate-driven changes in fire extent, severity, or frequency, in conjunction with an invasive species such as cheatgrass (*Bromus tectorum*), buffelgrass (*Pennisetum ciliare*), or less prolific annuals, can quickly reset the connectivity of a fire-prone landscape such that species composition and spatial structure accelerate away from the previous attractor into a very different system (Zedler et al. 1983; Fischer et al. 1996; Esque et al. 2006). Typically, such landscapes will exhibit more spatial homogeneity and simple structure—in the worst-case (so far) scenario, vast areas covered by invasive annuals in which there was formerly a mosaic of longer-lived shrubs and discontinuous fine fuels. These novel systems can be impressively resistant to change, however, as reflected in the difficulty of returning an invaded grassland to its pre-invasion composition. Part of the reason is that the new system includes a strong element of self-reinforcement in its new configuration. For example, desert grasslands that have been invaded by Old World grasses have greater fine fuel mass and continuity than the pre-invasion community; this new fuel complex promotes fire spread, which eliminates fire-sensitive native species while favoring the pyrophilic invaders (Zouhar et al. 2008; Stevens and Falk 2009).

Figure 1.4 here

1.1.3 Top-Down vs. Bottom-Up Controls

Energetic inputs and their regulation can be top-down or bottom-up, depending on the scale of spatial heterogeneity at which they act. For example, solar radiation, whether used to fix carbon between fires or to heat and dry fuels during a fire, is a top-down KE input. This energetic input is then subjected to further top-down regulation by locally homogeneous spatial fields of humidity, atmospheric pressure, temperatures, and precipitation. Fuels (stored PE) also become a source of thermal (kinetic) energy during a fire (Table 1.2). At finer scales, varying fireline intensity or flame length are associated both with fine-scale heterogeneity of fuels (spatial patterns of bottom-up inputs of PE), and with bottom-up regulation (e.g. by fuel moisture and topographic control of fire spread) of the PE→KE conversion associated with spatial variation in topography or fuel abundance at finer scales. Topographic barriers to fire spread shape and limit the size of individual fires, by creating spatial variation in flux rates, and over time produce spatiotemporal patterns of fire history of varying complexity (Kellogg et al. 2008). In general, variables with coarser resolution than these spatio-temporal patterns are associated with top-down controls, whether energetic or regulatory, whereas variables with finer resolution than this energy transfer are bottom-up controls.

In the language of pattern and process, energy flux represents process in landscape fire ecology, whereas regulation associated with the spatial distribution of energy represents landscape pattern. An obvious example of the latter is the spatial distribution of fuels (potential energy). Ideally we should be able to both quantify and predict landscape pattern change by measuring the relative strength of top-down vs. bottom up regulatory controls. For example, a dominance of top-down energy or regulation will homogenize and coarsen landscape pattern, whereas a dominance of bottom-up components will induce more complex (heterogeneous) spatial patterns to emerge. The spatial scale at which fire is "expressed" on the landscape is intermediate between the scales of variation of top-down vs. bottom-up components.

The expression of energy and regulation changes across scales, as some processes act cumulatively and others change qualitatively. For example, the energy transformed in the combustion process is a measurable physical property that is additive as a fire spreads, with output rates (e.g., joules $sec^{-1}m^{2}$) varying with external drivers and regulatory constraints such as fuel moisture and slope steepness. In contrast, topographic regulation across the landscape (e.g., ridges and valleys, barriers vs. corridors) changes combustion conditions and fire behavior in coherent spatial patterns correlated with aggregate patterns of slope and aspect. Similarly, with fuels, the expression of spatial heterogeneity changes from variation at fine scales (e.g., packing ratio) to larger-scale variation in landscape connectivity that influences fire shapes, sizes, and duration.

1.1.4 Landscapes and the Middle-Number Domain

The top-down and bottom-up organization implicit in the ERS framework might suggest that hierarchy theory could be a useful framework for studying landscape fire. Hierarchies are proposed to evolve in open dissipative systems, such as landscapes, establishing a regulatory structure (O'Neill et al. 1986). To our knowledge, however, hierarchy theory has not been applied successfully to landscape fire or similar landscape disturbances. We believe that the contagious and mercurial nature of fire, expressed as rapid temporal fluxes that greatly exceed the rates of other energy fluxes at both fine and coarse scales, confounds a hierarchical approach to the landscape ecology of fire. What works well for trophic structure in ecosystems, which can be studied over time scales of days to years, breaks down under the "metabolic" rates associated with fire: velocities can vary by orders of magnitude and temporal pulses of fire effects are far shorter than successional recovery. As such, fire is a "perturbing transitivity" (Salthe 1991) that melts hierarchical structure. Furthermore, hierarchy theory posits that ecosystem function is "driven" (forced) from lower hierarchical levels (finer scales) and constrained by upper levels (coarse scales). In our view, drivers (energy) and constraints (regulation) can issue from both coarser (top-down) and finer (bottom-up) scales than the level of interest, i.e., the landscape.

At the broadest scales, we can model fire occurrence and extent with aggregate statistics (e.g., Littell and Gwozdz, Chap. 5; Littell et al. 2009) and capture meaningful information about fire regimes. Broad-scale regulators such as climate or derived variables such as water deficit can explain much of the variance in flux rates that manifest as regional area burned (Gedalof, Chap. 4; Littell and Gwozdz, Chap. 5). At fine scales, fire's interactions with individual ecological objects (e.g., trees) are fairly straightforward to quantify. For example, individual tree mortality is closely associated with fireline intensity and flame length (energy flux) and tree resistance (e.g., bark thickness as a flux resistor) (Ryan and Reinhardt 1988). At both ends of the spectrum, both the energetic and regulatory components can be identified.

It is the intermediate scales that are problematic in the study of fire because of the interaction of bottom-up and top-down regulation. Recall that we have characterized a contagious disturbance as one whose properties depend on its interactions with landscape elements (Peterson 2002). The spatial heterogeneity of these interactions confounds attempts to predict fire area (or more importantly, fire severity) from spatially homogeneous top-down controls (e.g., weather), while also propagating and exacerbating estimation errors for many properties of fires that are computable at fine scales (Rastetter et al. 1992; McKenzie et al. 1996; Keane and Finney 2003). Fire as a contagious disturbance is thus inherently a multi-scale process.

This "modal" domain of fire, influenced by top-down and bottom-up controls on energy fluxes, which we refer to as the "landscape", is a middle-number system (O'Neill et al. 1986) with respect to ecological objects we can observe (growing trees, fuel transects, pixels, fire scars, animals—Fig. 1.5). We hypothesize here (and elsewhere—see McKenzie and Kennedy, Chap. 2) that in disturbance-prone landscapes, the physical limits to the extent of contagious disturbance coincide with the upper end of the middle-number domain. This is roughly equivalent to the spatial extent of the largest fires and the time frame of the fire cycle. At spatial scales much larger than the largest fires, and at time frames longer than the characteristic fire cycle, aggregate statistics suffice to characterize fire regimes. Indeed, for the purpose of understanding fire we define "landscape" as the spatial scale at which these middle-number relationships converge.

Figure 1.5 here

Ideally, analyses in the middle-number domain will be suitable for application of the ERS paradigm, if we can identify two thresholds. At the fine end of the gradient (near the origin in Fig. 1.5), what energetic and regulatory functions (Table 1.2) are in play up until a threshold at which spatial pattern starts to matter, where spatial contagion becomes a player in ecosystem dynamics? At the coarser end of the scale gradient, what are the energetic and regulatory components effecting the breakdown of contagion, such that top-down controls are in effect and simple aggregate statistics like means and variances suffice to capture variation in process and pattern? Between these thresholds, we would further seek some measure of how contagion changes across scale, as we have with more traditional properties of fire regimes such as fire frequency (Falk et al. 2007). To that end, we need to move from an *ad hoc* definition of contagion in relation to disturbance (see above) and establish a metric, or set of related metrics, to quantify it in a meaningful way. A spatially correlated physical process such as heat flux would be a good candidate for a covariate; i.e., the neighborhood effects of heat flux should modulate the strength of contagion, along with spatial variation in potential energy (fuel).

Perhaps the most elegant solution to quantifying how contagion changes across scale would involve deriving a scaling law linked to other sources of variability. For example, self-similar topography, if sufficiently dissected to produce bottom-up controls on energy flux (fire spread), produces scaling laws in fire-regime properties (Kellogg et al. 2008; McKenzie and Kennedy, Chap. 2). Characteristic scales, or correlation lengths, of bottom-up controls (Fig. 1.4) might determine at what spatial scale contagion must be produced solely by external (top-down) drivers such as fire weather unconstrained by topography or connectivity of fuels. In this reduced case, limits to contagion would depend only on the spatial extent of extreme weather events (combined with available fuel), which are known to drive the largest wildfires.

To motivate the ERS framework as a potential solution to the middle-number problem in landscape fire, we therefore need to demonstrate how explicit scaling laws can bridge the gap between simple means and variances, i.e. aggregate statistics that work at fine or broad scales, and the complexity of middle-number systems that varies across scales in non-obvious ways. Specifically, we need to specify scaling of energy and regulation in a way that reduces the dimensionality, and potential for error propagation, through calculations on middle-number data.

We take it as axiomatic that energy and regulation covary across scales of space and time. Scaling laws represent stochastic processes that have been codified from multiple realizations across spatial and temporal scales (Lertzman et al. 1998). They also preserve the total information in a system better than aggregate statistics. For example, historical fire regimes comprise multiple realizations of individual events, whose landscape memory is in fire-size distributions and the vegetation mosaic (Moritz et al., Chap. 3) or time-series of fire scars (Kennedy and McKenzie, n.d.). We may not be able to accurately reconstruct each individual realization (Swetnam et al., Chap. 7; Hessl et al. 2007), but we can back-engineer elements of the stochastic process (McKenzie et al. 2006; Kellogg et al. 2008) from the scaling relations, preferably in units of energy and regulation.

Falk et al. (2007) showed analytically how fire-regime information can be preserved in a scaling relation for fire frequency—the interval-area (IA) relation. Modeling across the middle-number domain under the ERS framework would explore analogous scaling patterns involving the more mechanistic "primitives" of fire regimes associated with the classic "fire triangle" (Table 1.2). Both energy-like and regulating elements are subject to scaling relations, and something akin to a covariance structure across scale is quantified, using these elements separately or in combination. An example of the latter would be the potential energy in a weighted combination of slope aspect, fuel loading, and packing ratio (*sensu* Rothermel 1972) represented in variograms.

We reiterate that the details of this theory are yet to be specified. Given such a theoretical framework, we need then to develop landscape experiments—probably simulation experiments—that not only test inferences but also demonstrate their tractability for quantifying landscape disturbance in the middle-number domain. Following this, we should attempt to "track" ERS components through ecosystem processes, beginning with the energetics themselves, from quantification of productivity and biomass pools (KE→PE and storage time) to heat-release dynamics (PE→KE). A later state of development could translate these energy fluxes to spatial fire-effects information, such as burn severity matrices, postfire patch characteristics, and other changes at scales useful to management.

1.2 Some Implications

Real improvements in landscape ecology theory will eventually be reflected in improved management of landscapes. The urgency for optimizing landscape management is heightened today with climate, land use change, and changing disturbance regimes that are affecting landscapes at ever broader scales (Peterson et al., Chap. 10; Miller et al., Chap. 11). The translation of theory and science into appropriate and realistic management is always imperfect (Schmoldt et al. 1999). How does an energetic framework for landscape disturbance inform and improve management?

First, there are profound examples of what can happen when we ignore these principles of energetics, regulation, and scale. Fire management and policy have ignored a basic principle of ecosystem energetics with an enormous impact on society. Energy is not permanently stored in biomass; at some point, that biomass must either decompose or combust, releasing energy. By ignoring this, suppression policies throughout the 20th century led to accumulated potential energy in biomass, and we should not have been surprised by the extent and severity of late 20th-century wildfires. The expectation that every fire could be suppressed became less and less realistic as the potential energy in ecosystems grew, such that now the synergy of increased kinetic energy in a warming climate with abundant fuel has jeopardized ecosystems as sustainable sources of goods and services (Peters et al. 2004; Baron et al. 2009; Joyce et al. 2009).

We have also been caught by surprise when we have ignored some powerful regulators and what can change when they are no longer in force. For example, before the establishment of exotic vegetation such as cheatgrass and buffelgrass (Fischer et al. 1996; Esque et al. 2006), fire extent in arid rangelands was limited by the patchiness of flammable vegetation. This "regulation" maintained spatially heterogeneous landscapes with a concomitant diversity of habitat for species such as sage grouse (Fischer et al. 1996). Cheatgrass and other similar invasives have taken the energy-regulation dynamic out of equilibrium in arid rangelands, leading to a self-reinforcing pattern of change in the fire regimes (Keeley et al., Chap. 8).

Ignoring the principles of scaling has contributed to misplaced focus (resolution) of management. For example, understanding how patch structure changes across scale is important for designing management plans, reserves, etc. (Baker 1989; Fahrig 1992; Parody and Milne 2004). Much energy has gone into documenting the importance of scale in landscape ecology (Peterson and Parker 1998; Turner et al. 2001; Wu et al. 2006), but considering the interplay of energy and regulation can be particularly cogent.

As an example, consider the tradeoff between maximizing C sequestration (in forests) and maintaining resilient landscapes under future fire regimes in a warming climate (North et al. 2009). Fuel treatments, and other practices in managed landscapes such as reduced planting densities, may remove biomass and release C but (if surface fuels are removed) can reduce the extent and intensity of subsequent fires (Peterson and Johnson 2007; Peterson et al., Chap. 10). How does the ERS framework inform our choices about where, when, and how much? Fuel management is only as effective as the top-down drivers of fire will let it be. How effective will our fuel treatments be under drier more extreme fire weather (Gedalof, Chap. 4; Littell and Gwozdz, Chap. 5)? How much of the landscape needs to be "treated" (in ERS terms, the spatial pattern of PE altered) to reduce the spread and growth of a fire? There are also temporal-scaling issues. For example, biomass (PE) accumulates at different rates in different ecosystems. How often do treatments need to be done to be effective?

We suggest that quantifying the potential energy stored in fuels, the strength of regulation (topographic complexity and fuel connectivity), and the kinetic energy associated with fire weather could provide valuable information for optimizing C sequestration over a chosen temporal domain. Alternatively one could jointly optimize C sequestration and other landscape metrics of interest (Kennedy et al. 2008), using inputs of energy and regulation. Identifying thresholds beyond which regulation breaks down (e.g., multiple megafires in the same year such as the Hayman, Rodeo-Chediski, and Biscuit fires of 2002) would also be essential (Littell and Gwozdz, Chap. 5). With limited resources for active management, a parsimonious model, such as we seek to enable with the ERS framework, could be a valuable tool to optimize the effectiveness of management.

1.3 Conclusions

We have proposed a theoretical model of landscape fire grounded in the interactions between energy fluxes and pools, and their controls, or "regulators", across spatial and temporal scales. If successful, an ERS framework could help identify the nature and strength of top-down vs. bottom-up controls on landscape fire, and help to solve two classic problems: the pulsed nature of fire *cf.* most ecosystem processes, and the middle-number problem, which can make landscape-scale analyses intractable at worst and fraught with uncertainty at best. A quantitative theory would need to compare favorably with existing paradigms in reproducing observed structures and processes on landscapes, while providing parsimony in both analysis and computation that could reduce uncertainty and increase the scope, both spatial and temporal, of inference. We return to this idea in Chapter 12, in which we suggest specifically how the analyses throughout this book might be transformed by an energetic perspective.

1.4 References

Aber, J.D., and J.M. Melillo. 1991. *Terrestrial ecosystems*. Philadelphia, PA: Saunders College Publishing.

Agee, J.K. 1993. *Fire ecology of Pacific Northwest forests.* New York: Island Press.

Agee, J.K., and C.N. Skinner. 2005. Basic principles of forest fuel reduction treatments. *Forest Ecology & Management* 211:83–96.

Allen, C.D., M. Savage, D.A. Falk, K.F. Suckling, T.W. Swetnam, T. Schulke, P.B. Stacey, P. Morgan, M. Hoffman, and J. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. *Ecological Applications* 12:1418–1433.

Allen, T.F.H., and T.W. Hoekstra. 1992. *Toward a unified ecology*. New York: Columbia University Press.

Baker, W.L. 1989. Landscape ecology and nature reserve design in the Boundary Waters Canoe Area. *Ecology* 70:23–35.

Baron, J.S., L. Gunderson, C.D. Allen, E. Fleishman, D. McKenzie, L.A. Meyerson, J. Oropeza, and N. Stephenson. 2009. Options for national parks and reserves for adapting to climate change. *Environmental Management.* 44:1033–1042

Bond, W.J., and J.J. Midgley. 1995. Kill thy neighbour: an individualistic argument for the evolution of flammability. *Oikos* 73:79–85.

Bond, W.J., and B.W. van Wilgen. 1996. *Fire and plants*. London: Chapman & Hall.

Bowman, D.M.J.S., J.K. Balch, P. Artaxo, and 16 others. 2009. Fire as an earth system process. *Science* 324:481–484.

Brown, J.K., and J.K. Smith, eds. 2000. Wildland fire in ecosystems: effects of fire on flora. General Technical Report RMRS-GTR-42-vol. 2. Ogden, UT: U.S. Forest Service.

Collins, B.M., J.D. Miller, A.E. Thode, M. Kelly, J.W. van Wagtendonk, and S.L. Stephens. 2009. Interactions among wildland fires in a long-established Sierra Nevada natural fire area. *Ecosystems.* 12:114–128.

Cushman S.A., D. McKenzie, D.L. Peterson, J.S. Littell, and K.S. McKelvey. 2007. Research agenda for integrated landscape modeling. General Technical Report RMRS-GTR-194. Fort Collins, CO: U.S. Forest Service.

DigitalGlobe. 2010. Digital globe: an imagery and information company. http://www.digitalglobe.com. Accessed 25 Jan 2010.

Esque, T.C., C.R. Schwalbe, J.A. Lissow, D.F. Haines, D. Foster, and M. Garnett. 2006. Buffelgrass fuel loads in Saguaro National Park, Arizona, increase fire danger and threaten native species. *Park Science* 24:33–37.

Fahrig, L. 1992. Relative importance of spatial and temporal scales in a patchy environment. *Theoretical Population Biology* 41:300–314.

Falk, D.A., C. Miller, D. McKenzie, and A.E. Black. 2007. Cross–scale analysis of fire regimes. *Ecosystems* 10:809–823.

FEIS (Fire Effects Information System). http://www.fs.fed.us/database/feis/plants/tree/pinconl/all.html. Accessed 8 October 2009.

Finney, M.A., C.W. McHugh, and I.C. Grenfell. 2005. Stand and landscape effects of prescribed burning on two Arizona wildfires. *Canadian Journal of Forest Research* 35:1714–1722.

Fischer, R.A., K.P. Reese, and J.W. Connelly. 1996. An investigation on fire effects within xeric sage grouse brood habitat. *Journal or Range Management* 49:194–198.

Gunderson, L.H., and C.S. Holling. 2001. *Panarchy: understanding transformations in systems of humans and nature*. Washington, DC: Island Press.

Hessburg, P.F., and J.K. Agee. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre–settlement and modern eras. *Forest Ecology and Management* 211:117–139.

Hessl, A.E., J. Miller, J. Kernan, and D. McKenzie. 2007. Mapping wildfire boundaries from binary point data: comparing approaches. *Professional Geographer* 59:87–104.

Holden, Z.A., A.M.S. Smith, P. Morgan, M.G. Rollins, and P.E. Gessler. 2005. Evaluation of novel thermally enhanced spectral indices for mapping fire perimeters and comparisons with fire atlas data. *International Journal of Remote Sensing* 26:4801–4808.

Holling, C.S. 1992. Cross–scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs* 62:447–502.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain (U.S.A.) conifer forests. *Conservation Biology* 9:1041–1058.

Johnson, E.A. 1992. *Fire and vegetation dynamics: studies from the North American boreal forest.* Cambridge, UK: University Press.

Johnson, E.A., and K. Miyanishi. 2001. *Forest fires: behavior and ecological effects*. San Diego: Academic Press.

Joyce, L.A., G.M. Blate, S.G. McNulty, C.I. Millar, S. Moser, R.P. Neilson and D.L. Peterson. 2009. Managing for multiple resources under climate change: national forests. *Environmental Management* 44:1022–1032.

Keane, R.E., and M.A. Finney. 2003. The simulation of landscape fire, climate, and ecosystem dynamics. In *Fire and climatic change in temperate ecosystems of the Western America,* ed. T.T. Veblen, W.L. Baker, G. Montenegro, T.W. Swetnam, 32–68. New York: Springer-Verlag.

Keeley, S.C., J.E. Keeley, S.M. Hutchinson, and A.W. Johnson. 1981. Post-fire succession of the herbaceous flora in southern California chaparral. *Ecology* 62:1608–1621.

Kellogg, L.-K.B., D. McKenzie, D.L. Peterson, and A.E. Hessl. 2008. Spatial models for inferring topographic controls on low-severity fire in the eastern Cascade Range of Washington, USA. *Landscape Ecology* 23:227–240.

Kennedy, M.C. and D. McKenzie. [N.d.]. Using a neutral model to estimate spatial structure in historical low-severity fire regimes. Manuscript in review. *Landscape Ecology*.

Kennedy, M.C., E.D. Ford, P. Singleton, M. Finney, and J.K. Agee. 2008. Informed multi-objective decision-making in environmental management using Pareto optimality. *Journal of Applied Ecology* 45:181–192.

Key, C.H., and N.C. Benson. 2006. Landscape assessment: sampling and analysis methods. In *FIREMON: Fire effects monitoring and inventory system,* eds. D.C. Lutes, R.E. Keane, J.F. Caratti, C.H. Key, N.C. Benson, S. Sutherland, and L.J. Gangi. General Technical Report RMRS-GTR-164-CD . Ogden, UT: U.S. Forest Service.

Lentile, L.B., Z.A. Holden, A.M.S. Smith, M.J. Falkowski, A.T. Hudak, P. Morgan, S.A. Lewis, P.E. Gessler, and N.C. Benson. 2006. Remote sensing techniques to assess active fire characteristics and post-fire effects. *International Journal of Wildland Fire* 15:319–345.

Lertzman, K., J. Fall, and B. Dorner. 1998. Three kinds of heterogeneity in fire regimes: At the crossroads of fire history and landscape ecology. *Northwest Science* 72:4–23.

Linn, R., J. Winterkamp, J. Canfield, J. Sauer, J. Colman, J. Reisner, C. Edminster, F. Pimont, J. Dupey, and P. Cunningham. 2006. Versatility of FIRETEC, a physics-based wildfire model. *Forest Ecology and Management* 234 (Supplement 1):S94.

Littell, J.S., D. McKenzie, D.L. Peterson, and A.L. Westerling. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. *Ecological Applications* 19:1003–1021.

Logan, J.A., and J.A. Powell. 2001. Ghost forests, global warming, and the mountain pine beetle. *American Entomologist* 47:160–172.

Maleki, S., C. Skinner, and M. Ritchie. 2007. Tested by fire: the Cone Fire and the lessons of an accidental experiment. Science Perspectives PSW-SP-008. Albany, CA: U.S. Forest Service.

McGarigal, K., S.A. Cushman, M.C. Neel, and E. Ene. 2002. FRAGSTATS: Spatial Pattern Analysis Program for Categorical Maps. http://www.umass.edu/landeco/research/fragstats/fragstats.html. Accessed 25 Jan 2010.

McKenzie, D., D.L. Peterson, and E. Alvarado. 1996. Extrapolation problems in modeling fire effects at large spatial scales: a review. *International Journal of Wildland Fire* 6:65–76.

McKenzie, D., A.E. Hessl, and L.-K.B. Kellogg. 2006. Using neutral models to identify constraints on low-severity fire regimes. *Landscape Ecology* 21:139–152.

Merton, R.N. 1999. Multi-temporal analysis of community scale vegetation stress with imaging spectroscopy. Ph.D. thesis. Aukland, New Zealand: University of Auckland.

Miller, J.D., and S.R. Yool. 2002. Mapping forest post-fire canopy consumption in several overstory types using multi-temporal Landsat TM and ETM data. *Remote Sensing of Environment* 82:481–496.

Mitchell, S.R., M.E. Harmon, and K.E.B. O'Connell. 2009. Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. *Ecological Applications* 19:643–646.

MTBS (Monitoring Trends in Burn Severity). 2009. http://mtbs.gov/. Accessed 1 November 2009.

North, M., M. Hurteau, and J. Innes. 2009. Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. *Ecological Applications* 19:1385–1396.

Odum, H.T. 1983. *Systems ecology: an introduction*. Chichester, UK: Wiley and Sons Ltd.

O’Neill, R.V., D.L. deAngelis, J.B. Waide, and T.F.H. Allen. 1986. *A hierarchical concept of ecosystems*. Princeton, NJ: Princeton University Press.

Parody, J.M., and B.T. Milne. 2004. Implications of rescaling rules for multi-scaled habitat models. *Landscape Ecology* 19:691–701.

Peters, D.P.C., R.A. Pielke Sr., B.T. Bestelmeyer, C.D. Allen, S. Munson-McGeed, and K.M. Havstad. 2004. Cross-scale interactions, nonlinearities, and forecasting catastrophic events. *Proceedings of the National Academy of Sciences* 101:15130–15135.

Peterson, D.L., and M.C. Johnson. 2007. Science-based strategic planning for hazardous fuel treatment. *Fire Management Today* 67:13–18.

Peterson, D.L., and V.T. Parker, eds. 1998. *Ecological scale: theory and applications*. New York: Columbia University Press.

Peterson, G.D. 2002. Contagious disturbance, ecological memory, and the emergence of landscape pattern. *Ecosystems* 5:329–338.

Pickett, S.T., and P.S. White. 1985. *The ecology of natural disturbance and patch dynamics*. New York: Academic Press.

Pielou, E.C. 2001. *The energy of nature*. Chicago: University of Chicago Press.

Rastetter, E.B., A.W. King, B.J. Cosby, G.M. Hornberger, R.V. O’Neill, and J.E. Hobbie. 1992. Aggregating finescale ecological knowledge to model coarser scale attributes of ecosystems. *Ecological Applications* 2:55–70.

Romme, W.H. 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. *Ecological Monographs* 52: 119–221.

Romme, W.H., E.H. Everham, L.E. Frelich, and R.E. Sparks. 1998. Are large infrequent disturbances qualitatively different from small frequent disturbances? *Ecosystems* 1:524–534.

Rothermel, R.C. 1972. *A mathematical model for predicting fire spread in wildland fuels*. Research Paper INT-115. Ogden, UT: U.S. Forest Service.

Rowe, J.S. 1983. Concepts of fire effects on plant individuals and species. In *The role of fire in northern circumpolar ecosystems*, ed. R.W. Wein and D.A. MacLean, 135–154. New York: Wiley and Sons.

Royama, T. 1984. Population dynamics of the spruce budworm *Choristoneura fumiferana*. *Ecological Monographs* 54:429–462.

Ryan, C.K., and E.D. Reinhardt. 1988. Predicting postfire mortality of seven western conifers. *Canadian Journal of Forest Research* 18:1291-1297.

Salthé, S.N. 1991. Two forms of hierarchy theory in western discourses. *International Journal of General Systems* 18:251–264.

Schmoldt, D.L., D.L. Peterson, R.E. Keane, J.M. Lenihan, D. McKenzie, D.R. Weise, and D.V. Sandberg. 1999. *Assessing the effects of fire disturbance on ecosystems: a scientific agenda for research and management.* General Technical Report PNW-GTR-455. Portland, OR: U.S. Forest Service.

Scholl, A.E., and A.H. Taylor. 2010. Fire regimes, forest change, and self-organization in an old-growth mixed-conifer forest, Yosemite National Park, USA. *Ecological Applications* In press.

Schwilk, D.W. 2003. Flammability is a niche construction trait: canopy architecture affects fire intensity. *American Naturalist* 162:725–733.

Stevens, J., and D.A. Falk. 2009. Can buffelgrass (*Pennisetum ciliare* (L.) Link) invasions be controlled in the American Southwest? Using invasion ecology theory to explain buffelgrass success and develop comprehensive restoration and management. *Ecological Restoration* 24: 417-427.

Stocks, B.J., M.E. Alexander, and R.A. Lanoville. 2004. Overview of the international crown fire modelling experiment (ICFME). *Canadian Journal of Forest Research* 34:1543–1547.

Suding, K.N., K.L. Gross, and G.R. Houseman. 2004. Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology and Evolution* 19:46–53.

Taylor, A.H. and C. N. Skinner. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. *Ecological Applications* 13:704–719.

Turner, M.G. 1989. Landscape ecology: the effect of pattern on process. *Annual Review of Ecology and Systematics* 20:171–197.

Turner, M.G., R.H. Gardner, and R.V. O'Neill. 2001. *Landscape ecology in theory and practice: pattern and process*. New York: Springer.

Turner, M.G., and W.H. Romme. 1994. Landscape dynamics in crown fire ecosystems. *Landscape Ecology* 9:59–77.

Turner, M.G., W.H. Hargrove, R.H. Gardner, and W.H. Romme. 1994. Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. *Journal of Vegetation Science* 5:731–742.

van Wagtendonk, J.W. 2006. Fire as a physical process. In *Fire in California's ecosystems*, ed. N.G. Sugihara, J.W. van Wagtendonk, J. Fites-Kaufman, K.E. Shaffer, and A.E. Thode, 38–57. Berkeley: University of California Press.

Watt, A.S. 1947. Pattern and process in the plant community. *Journal of Ecology* 35:1–22.

West, G.B., B.J. Enquist, and J.H. Brown. 2009. A general quantitative theory of forest structure and dynamics. *Proceedings of the National Academy of Sciences* 106:7040–7045.

White, P.S. 1987. Natural disturbance, patch dynamics, and landscape pattern in natural areas. *Natural Areas Journal* 7:14–22.

Wu, J., K.B. Jones, H. Li, and O.L. Loucks. 2006. *Scaling and uncertainty analysis in ecology*. Dordrecht, The Netherlands: Springer.

Zedler, P.H., C.R. Gautier, and G.S. McMaster. 1983. Vegetation change in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub. *Ecology* 64:809–818.

Zouhar, K., J.K. Smith, S. Sutherland, and M.L. Brooks. 2008. Wildland fire in ecosystems: fire and nonnative invasive plants. General Technical Report RMRS-GTR-42-v6. Ogden, UT: U.S. Forest Service.

Table 1.1. Spatiotemporal properties of fire regimes and drivers of fire behavior and effects. Drivers act on means, variances, and extremes of properties. Adapted from Falk et al. (2007).

	Climate, weather	Vegetation, fuels	Topography, landform
Temporal distribution			
Frequency or fire interval	Ignition availability and flammability; wind, humidity, and temperature patterns; fuel moisture	Vegetation productivity, postfire recovery and fuel buildup	Interaction of fire size with fuel availability; topographic barriers to fire spread
Duration	Drought or days without rain; frontal and synoptic climatic dynamics	Fuel biomass, condition, size distribution, connectivity; consumption rates	Topographic controls on rate of spread; fire spread barriers; rain shadows
Seasonality	Seasonal progression and length of fire season; effects on fuel phenology	Fuels phenology: green up, curing, and leaf fall	Topographic effects on fuel types, moisture, and phenology
Spatial distribution			
Extent	Local and synoptic weather control of ignition and fire spread	Vegetation (fuels) abundance and connectivity	Topographic influences on fire spread; fire compartments
Pattern (patch size, aggregation, contagion)	Orographic and frontal atmospheric instability, wind vectors, spatial distribution of ignitions	Spatial pattern of landscape fuel types (fuel mosaic)	Topographic influences on fire spread and spatial distribution of fuel types and condition
Intensity and severity	Microclimate and weather influences on spatial patterns of fuel moisture and abundance	Vegetation (fuel) mass, density, life-history traits, configuration; vertical and horizontal connectivity of surface and canopy layers	Slope and aspect interactions with local microclimate and weather

Table 1.2. Some important energetic and regulatory functions of elements of the "fire triangle" that are particularly relevant to landscape fire. Energy can be in kinetic (KE) or potential (PE) form. Energy storage and regulation of energy fluxes in landscape fire involve myriad ecosystem components.

Fire triangle component	Energy sources and fluxes	Regulation of energy conversion
Weather and climate	Solar energy is the primary KE input, driving temperature and precipitation patterns that provide preconditions for ignition.	Fuel moisture and fuel temperature affect the rate of PE→KE conversion, regulating ignition of fuels, fire intensity, and fire spread.
	KE is distributed to ecosystems via circulation (wind, convection, and turbulence) contributing to fire spread.	Energy regulation in the climate system is expressed in temporal and spatial patterns of precipitation, temperature, seasonality, and ocean-atmosphere teleconnections.
Fuels and vegetation	Photosynthetic plants convert solar energy to PE in the form of chemical-bond energy in biomass.	Abundance, compactness, and arrangement of fuels affect ignition, heat-transfer rates, and fire spread.
	PE is stored on the landscape, measured as living and dead biomass and productivity (Table 1.3). During combustion, these energy pools become sources of energy (KE) redistributed to the system.	Tree density and canopy cover affect regulation by fuel moisture and temperature. Rates of postfire plant growth and decomposition influence how often fires occur.
Topography and landform	N/A (By themselves they do not provide nor convert energy)	Slope steepness affects heat-transfer rates and fire spread.
		Solar incidence varies with aspect, affecting fuel moisture and fuel temperature, and thus the ignition of fuels, fire intensity, and fire spread.
		Shape of terrain and topographic barriers influence connectivity and the spatial pattern of fire spread.

Fig. 1.1. Global compilation of MODIS fire detections between 19–28 June 2004. Image courtesy of MODIS Rapid Response System. http://rapidfire.sci.gsfc.nasa.gov/firemaps/

Fig. 1.2. (a) Fire-severity classes on the 2006 Tripod Complex Fire in northcentral Washington, USA. Fire severity classes are identified from LANDSAT imagery using the algorithm of Key and Benson (2006). (b) Photos demonstrate low-mixed severity as crown scorch (above), and mixed severity as juxtaposed high- and low-severity patches (below). Fire-severity data are from the Monitoring Trends in Burn Severity (MTBS) project. http://www.mtbs.gov. Accessed 1 November, 2009. Photos courtesy of C. Lyons-Tinsley.

Fig. 1.3. The familiar landscape fire cycle is shown in black. Elements in boxes are things fire scientists (top portion) and landscape ecologists (bottom portion) are accustomed to measuring or modeling. In red is the energetic perspective. Short pulses of potential to kinetic energy (KE) occur during a fire, and kinetic energy is transformed into potential energy (PE) over long periods of time by plants. The spatial pattern of PE is continually being redistributed, subject to regulatory controls.

Fig. 1.4. Examples of energetic vs. regulatory emphasis in dynamics of self-limiting properties on the firescape. (a) In moderate topography, fires may not carry through an entire area depending on the connectivity of fuels and the characteristic scale of variability in potential energy (correlation length). (b) The physical template (steep topography) regulates the energetic dynamics by introducing physical barriers that create resistance to fire spread. In theory, one could have the same correlation length in these two systems, with different dynamic underpinnings. (c) In a very different system subject to top-down controls (climate), correlation length is much larger, reflected in patch-scale variation in age classes.

Fig. 1.5. Spatial scaling domain of landscape fire. Landscape fire regimes occupy the middle-number domain for objects of analysis—trees, stands, pixels, etc. A middle-number domain is “in between” the finer scales at which the number of observations and computations on them are still analytically tractable and the coarser scales at which aggregate statistics can explain sufficient proportions of system variability for meaningful inferences. This is the spatial domain of maximum complexity (O’Neill et al. 1986), where bottom-up (BU) and top-down (TD) controls converge. The Gaussian-like curve represents a mean of many processes whose individual “complexity curves” may be less regular, e.g., perhaps even monotonic or bimodal.

Figure 1.1

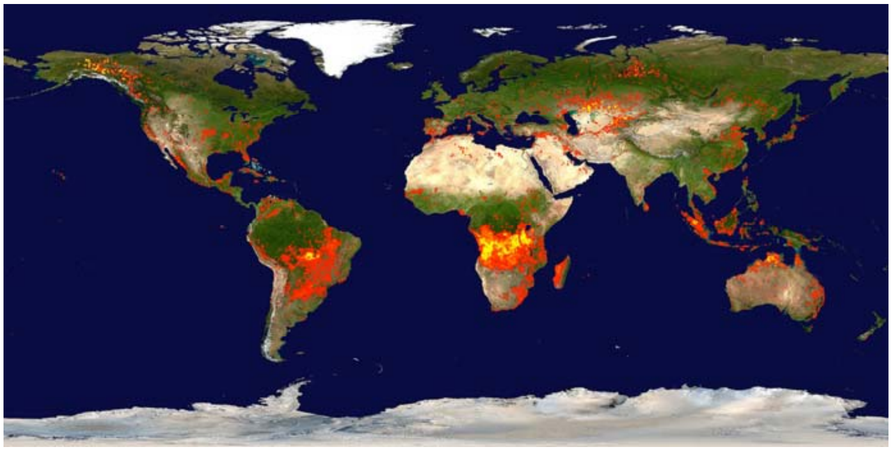
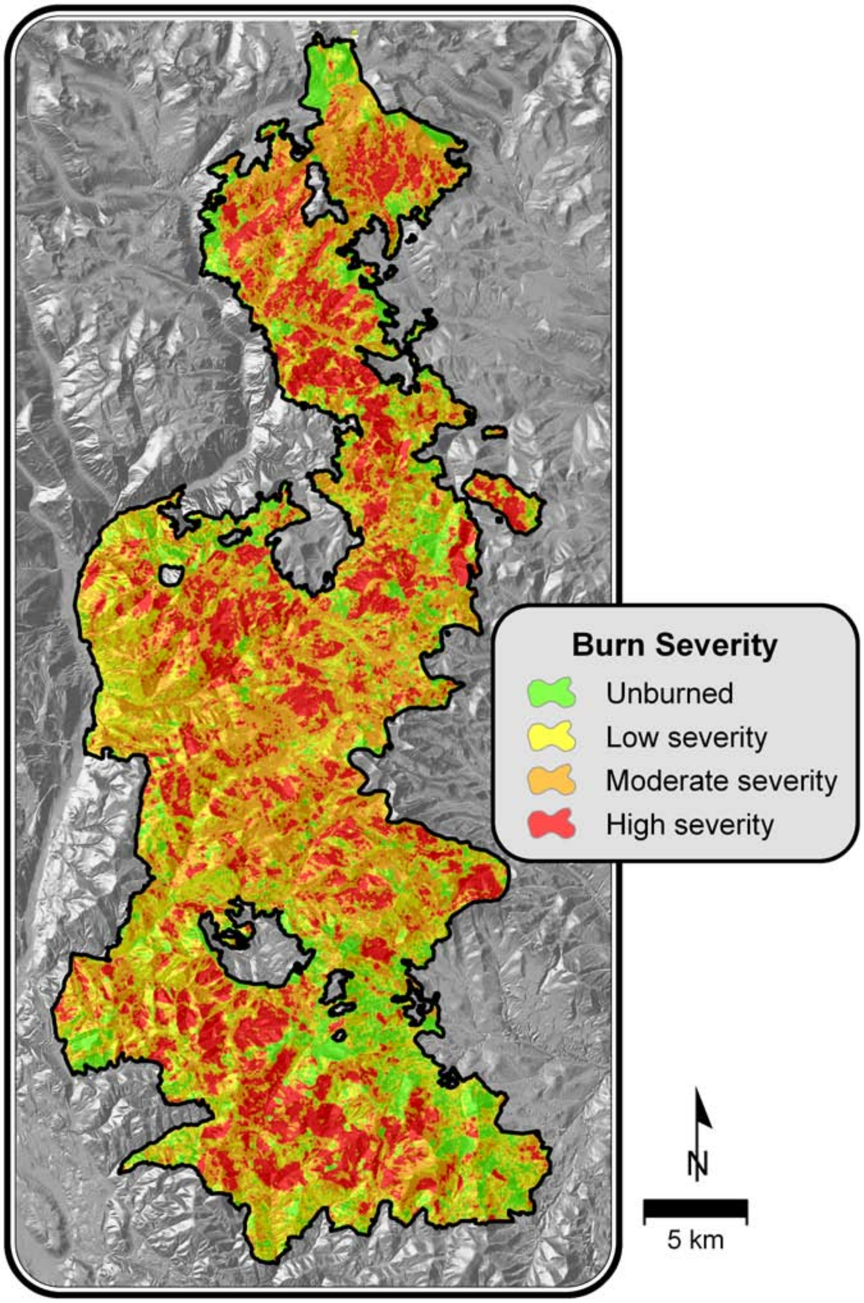
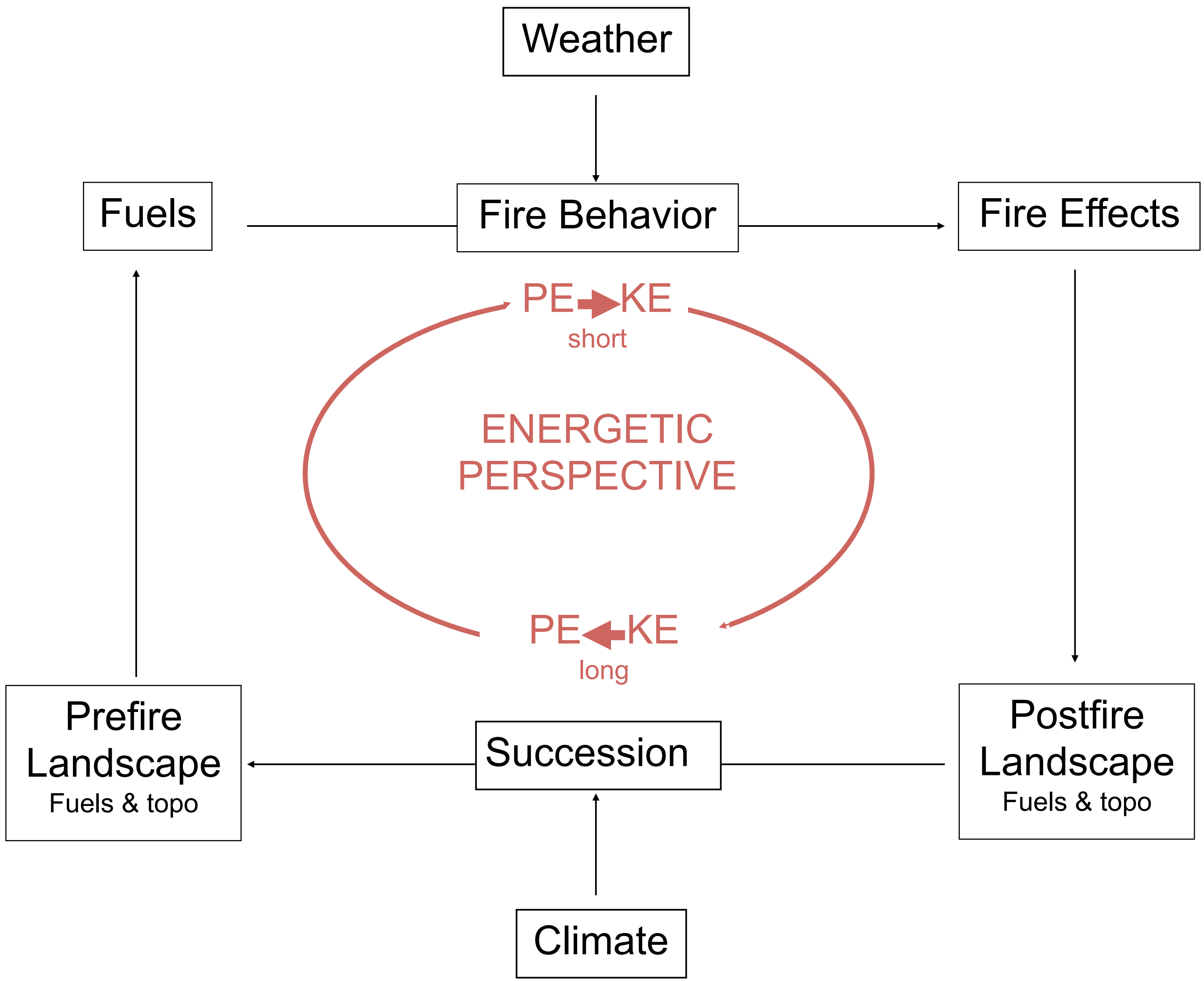


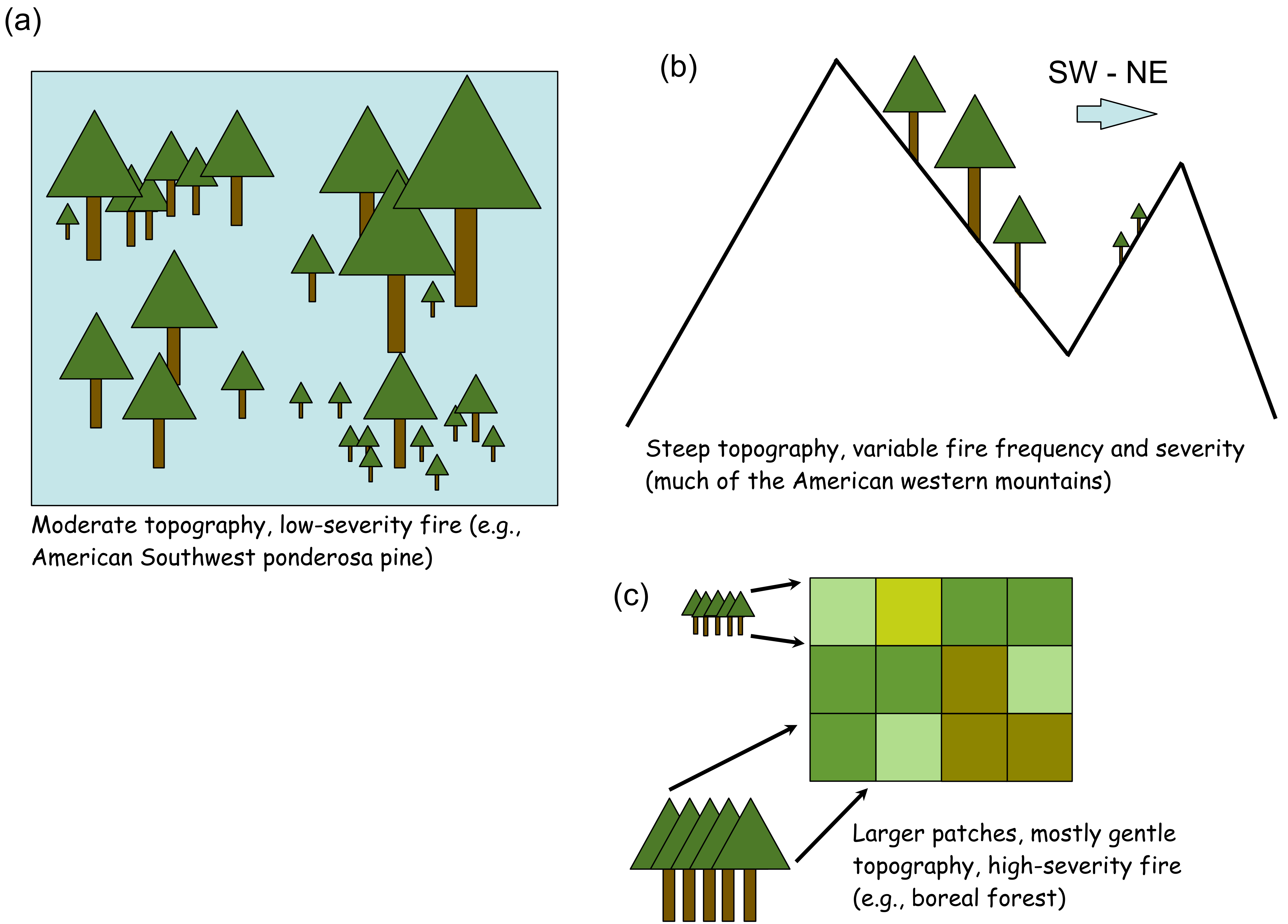
Figure 1.2(a)



Burn Severity
Unburned
Low severity
Moderate severity
High severity
N
5 km

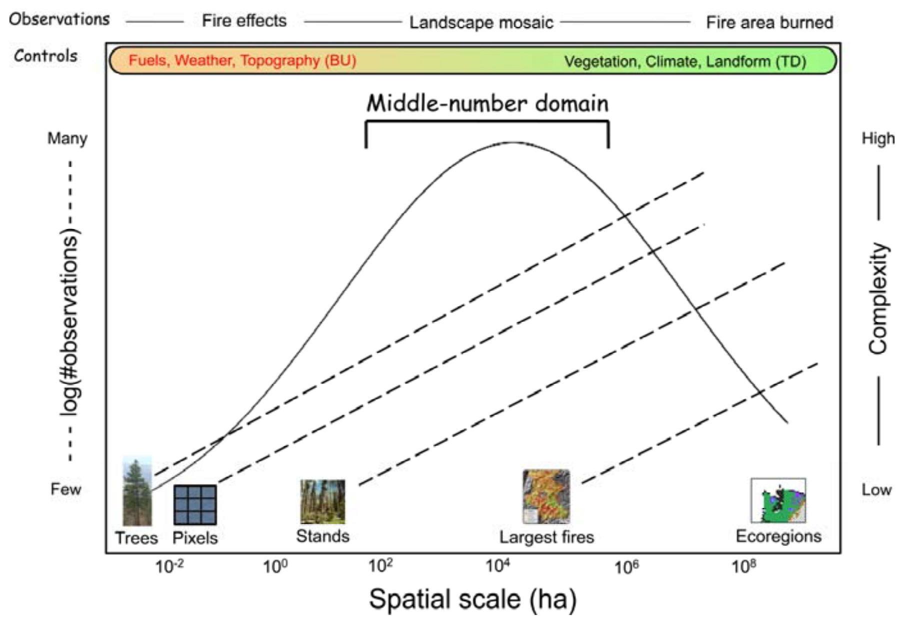
Figure 1.2 (b)


Weather
Fuels
Fire Behavior
Fire Effects
PE KE
short
ENERGETIC
PERSPECTIVE
PE KE
long
Prefire
Landscape
Fuels & topo
Succession
Postfire
Landscape
Fuels & topo
Climate



(a)
Moderate topography, low-severity fire (e.g., American Southwest ponderosa pine)
(b)
SW - NE
Steep topography, variable fire frequency and severity (much of the American western mountains)
(c)
Larger patches, mostly gentle topography, high-severity fire (e.g., boreal forest)

Figure 1.5


Observations
Fire effects
Landscape mosaic
Fire area burned
Controls
Fuels, Weather, Topography (BU)
Vegetation, Climate, Landform (TD)
Middle-number domain
Many
Few
log(#observations)
High
Low
Complexity
Trees
Pixels
Stands
Largest fires
Ecoregions
10-2
100
102
104
106
108
Spatial scale (ha)