

The Research Program of Historical Ecology

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Abstract

Historical ecology is a new interdisciplinary research program concerned with comprehending temporal and spatial dimensions in the relationships of human societies to local environments and the cumulative global effects of these relationships. Historical ecology contains core postulates that concern qualitative types of human-mediated disturbance of natural environments and the effect of these on species diversity, among other parameters. A central term used in historical ecology to situate human behavior and agency in the environment is the landscape, as derived from historical geography, instead of the ecosystem, which is from systems ecology. Historical ecology is similar to nonequilibrium dynamic theory, but differs in its postulate of human-mediated disturbance as a principle of landscape transformation. Such disturbances counterintuitively may involve anthropogenic primary and secondary succession that result in net increases of alpha and even beta diversity. Applied historical ecology can supply the reference conditions of time depth and traditional knowledge to restore past landscapes.

INTRODUCTION

Historical ecology is a research program concerned with the interactions through time between societies and environments and the consequences of these interactions for understanding the formation of contemporary and past cultures and landscapes (Balée 1998b; Balée & Erickson 2006a,b; Crumley 1994, 1998, 2003; Redman 1999; Sutton & Anderson 2004). A research program is a set of interdependent postulates on which only a portion of the scientific community agrees (Lakatos 1980, Stengers 2000 [1993]). It is therefore unlike a paradigm, which in the Kuhnian sense of normal science assumes conflicting models purporting to explain that the same phenomena cannot coexist (Kuhn 1970, Stengers 2000 [1993]). In anthropology, research programs include cultural ecology and sociobiology; a generation ago, research programs in the social sciences would have included psychoanalysis and Marxism (Lakatos 1999 [1973]). In ecology, systems theory and nonequilibrium dynamics constitute separate research programs (Zimmerer 2000).

Research programs consist of three to five hard-core postulates (Lakatos 1999 [1973]). In historical ecology, the postulates are the following: (a) Practically all environments on Earth have been affected by humans, including in a broad sense behavioral activities of the genus *Homo* (Kidder & Balée 1998, Redman 1999, Sauer 1956), although others would limit the wide-ranging effect of humans on the environment to only the entire time dating from the beginning of the Holocene, that is, the time coinciding with the beginnings of agriculture (Dickinson 2000); (b) human nature is not programmed genetically or otherwise to lessen or augment species diversity and other environmental parameters (Crumley 2001, Hayashida 2005); (c) it follows that kinds of societies defined by various socioeconomic, political, and cultural criteria impact landscapes in dissimilar ways, as some landscapes are less disturbed (and richer in species) than others; and (d) human inter-

actions with landscapes in a broad variety of historical and ecological contexts may be studied as a total (integrative) phenomenon (Balée 1998b, Egan & Howell 2001b, Rival 2006, Sutton & Anderson 2004).

Historical ecologists take a long view of history and landscapes and thus tend to be at variance with earlier established research programs of environmental anthropology (Balée & Erickson 2006b; Braudel 1980; Crumley 1994, 1998, 2003; Kidder & Balée 1998; Russell 1997). Historical ecology exemplifies revisionism of earlier regnant concepts in cultural ecology, cultural evolutionism, cultural materialism, and ecological systems theory (Dove 2001; cf. Headland 1997, Rival 2006). It is an interdisciplinary means of grappling with applications from both the social sciences and life sciences (Balée 1998a,b; Crumley 1994, 1998, 2003), the most important of which for strategic environmental concerns is restoration ecology, a synonym of applied historical ecology (Anderson 2001, Egan & Howell 2001a,b, Higgs 2003).

Historical ecology arose out of empirical studies that showed problems in the application of ecological anthropology to complex societies. Peasantries and other complex societies exhibiting different socioeconomic strata could not be analyzed according to methodologies developed in cultural ecology because cultural ecology referred only to classless or simple societies, wherein it was thought a linear relationship existed between key features of indigenous technology and the environment on the one hand and low population size and simple political organization of society on the other (Boglioli 2000, Cole & Wolf 1974). Such societies were not seen to exert so much a long-term effect on the local environment as to be adapted to its putative constraints (Adams 1998; Balée 1989; Balée & Erickson 2006b; Cole & Wolf 1974; Stahl 1996; Wolf 1982, 1999). Systems theory in ecological anthropology was an attempt to bring more mathematical rigor to the subject matter, especially by conceiving of human societies as populations having exchanges of

energy with other animal and plant populations in ecosystems (Rappaport 2000). Systems theory in anthropology, as in ecology, was ahistorical and excluded human agency and intentionality in the landscape (Dove 2001, Wolf 1999; cf. Biersack 1999).

A SECOND WORLD

In historical ecology the landscape is a place of interaction with a temporal dimension that is as historical and cultural as it is evolutionary per se, if not more so, upon which past events have been inscribed, sometimes subtly, on the land (Crumley 2003, Ingold 1993, Marquardt & Crumley 1987, Neves & Petersen 2006, Russell 1997). Historical ecology registers simple foragers and swidden horticulturalists as agents of history manifesting cultural pasts that defy placement in a stage of political evolution (Cormier 2003, Crumley 2003, Politis 2001, Rival 2002, Rival 2006, Zent & Zent 2004). Wolf (1982) showed that cultural ecology lacked a unified theory and could not rise above explanations of single cases because of its emphasis on human adaptations to the environment, rather than deeming society engaged with that environment and acting, effectively, to change it over time. Wolf (1999) argued that environmental anthropology needed to abandon systems theory and become both political and historical ecology to assess changes in relations between human societies and their landscapes.

Historical ecology has challenged the notion of “pristine primitives” (Wolf 1982) and virgin rainforests (Balée 1989, Denevan 1992) through different but ultimately convergent strands of interdisciplinary thinking in anthropology, geography, history, and ecology (Hayashida 2005). The notion that landscapes have history, and that natural things in given environments are historiographic indices of those environments, has several precursors in diverse fields, especially in history and geography. Historian Cronon’s (1983) classic study of the impact of precolonial Indians in New England in shaping the landscape

thought by the Puritans to be pristine was a careful empirical challenge to the concept of pristine primitives harnessed to the restraints of virgin forests (Turner 2005). The principal mechanism that created the parkland landscapes noted by the Puritans was controlled (or broadcast) fire (Cronon 1983, Pyne 1998).

Indigenous societies molded not only mosaic-like environments with patches rich in utilitarian natural resources, but also in some cases enhanced local (alpha) species diversity. Specifically, controlled (broadcast) fires are now considered to have enhanced local landscape heterogeneity as well as species diversity (especially of rare species), partly by preventing fuel buildups and the ensuing possibility of destructive wildfires in numerous indigenous areas of North America, South America, and Australia (Anderson 1999; Bird et al. 2005; Boyd 1999a,b; Lunt & Spooner 2005; Mistry et al. 2005; Posey 1985; Pyne 1991, 1998; Robbins 1999; Storm 2002; Winthrop 2001; cf. Foster et al. 2004). That fire by human agents in controlled cases amplifies diversity, whereas wildfires and combustion by fossil fuels tend to have degrading effects, refers one back to the notion of landscape, where humans and the environment meet in an analytic whole, with a temporal dimension that defines the relationship (Balée 1998a; Crumley 1994, 2003; Ingold 1993; Marquardt & Crumley 1987; Pyne 1998).

This notion of landscape in its most recent version originates in cultural and historical geography (Denevan 2001, Doolittle 2000, Kates et al. 1990, Olwig 2003, Rival 2006, Sutton & Anderson 2004). Geographers early derived the idea of an inseparability of humans and the environment in the context of a landscape (*Landschaft*) partly from German landscape gardeners and architects of the nineteenth century (Crumley 1994, Hall 2005, Wolschke-Bulmahn 2004) and from schools of nineteenth-century landscape painters in Europe, North America, and Australia who sought to capture their ideas of wilderness and its embedded humanity,

however savage, on canvas as well as in the Western psyche (Hirsch & O'Haulon 1995). The landscape in historical ecology is also influenced by the French *Annales* school of history in conceiving of the *paysage* as undergoing several forms of temporal change, both short- and long-term, as well as cyclical (Braudel 1980, Crumley 1998).

The thinking that humans are everywhere historical agents (apart from their consciousness of being so) of change in the landscape, by rendering it historical either by agriculture or some other recognizably human interference, dates from classical antiquity (Glacken 1967, Hall 2005, Hughes 1975). Herodotus proposed that historical events unfold in a physical place and that the characteristics of place, in turn, change through time—i.e., culture and the environment are in a sense intertwined and change together through time (Pitzl 2004). Cicero wrote of how through domestication, fertilization, and irrigation humans influenced the creation of a second world apart from the so-called natural one (Glacken 1967, Hughes 1975, Wolschke-Bulmahn 2004), a concept echoed 1700 years later in the Enlightenment (Roger 1997).

Cicero's second world was a built environment. He might not have recognized second worlds in sub-Saharan Africa, Australia, lowland South America, and much of North America, just as Renaissance and Enlightenment thinking did not, considering such regions to be wilderness (Raffles 2002, Roger 1997). The second world, from a nineteenth-century European perspective, incorporated natural and cultural things together, often in a garden-like setting, as seen especially in Italy and Germany (Hall 2005, Wolschke-Bulmahn 2004). The garden as a spatially defined landscape involving nature and culture antedates European civilization, having been borrowed by Hellenistic society from East Asia (Glacken 1967).

The garden is the underlying premise of a landscape, for there humans habitually interact with other living forms, both in a

cyclical fashion, and in the long-term (involving at least decades) and very long-term (involving centuries), a concept known as *longue durée* (Braudel 1980). Historical ecologists have discerned gardens in the midst of seeming wilderness in both the Neotropics and Paleotropics and have referred to these as forest gardens, forest fields, trail gardens, war gardens, man-made tropical forests, cultural or anthropogenic forests, and domesticated landscapes (Balée 1989; Clement 1999a; Denevan 1992, 2001, 2006; Erickson 2006; Gómez-Pompa et al. 1987, 1990; Heckenberger et al. 2003; Janzen 1998; Posey 1985; Posey & Balée 1989; Rival 2006). These are distinguished from cultural landscapes, which are not so much disturbed by humans as indexical (by iconic biota and places discernible to the naked eye) of local societies and their long-term history in situ (Stoffle et al. 2003). Historical ecologists also examine indexical functions of biota in specific landscapes affected by human activity over time (Feely-Harnik 2001, Verheyen et al. 2004, Walker 2000).

There is a long-standing division among European geographers and foresters between domesticated (or culturalized) and natural landscapes (Alexander & Butler 2004). The concept of pristine forests is gradually being replaced with a more hedged notion of old growth forest. The notion of old growth forest in Europe as well as North America includes forests that may have been disturbed by humans although not for long periods of time, so-called first nature (Rudel 2002). Increasing evidence, however, suggests intermediate disturbance may have lasting legacies, of the *longue durée* sort, in terms of redefining vegetation patterns (Turner 2005). Europe's culturalized landscapes run the gamut from treeless zones to mature forests similar to former woodland, yet none is primary, although relic species, such as lichens, bryophytes, and mycorrhizal fungi, as well as a few ancient trees, may still be found (Myers & Bazely 2003). Historical geographer Sauer proposed

that wherever humans had lived and impacted the environment by domestication, landscapes with determinate histories were the result (Olwig 2003, Rival 2006, Sauer 1956). In more recent times, a number of scholars have argued that agricultural impacts dating from the Holocene have essentially transformed the world so much that hardly any part of it is pristine per se and that, indeed, humans created the landscapes typically referred to as examples of Holocene environments (Denevan 1992, Dickinson 2000). The notion that certain species-rich forested landscapes of Greater Amazonia, Middle America, and West Africa were pristine wildernesses was challenged by new data and interpretations in the past two decades of the twentieth century from anthropologists, geographers, and biologists (Balée 1989; Balée & Campbell 1990; Denevan 1992, 2001; Fairhead & Leach 1996; Gómez-Pompa & Kraus 1992; Gómez-Pompa et al. 1987; Hayashida 2005; Leach & Fairhead 2000; Posey 1985; Posey & Balée 1989; Rival 2006; Stahl 1996; cf. Parker 1992), who supplied evidence of human activity in the origin of these landscapes.

In historical ecology, the concept of landscape transformation, resulting in so-called man-made forests (Campbell et al. 2006, Gómez-Pompa et al. 1987, Wiseman 1978), was derived initially from evidence of agriculture and agroforestry; more recent work suggests foraging and trekking societies have also influenced forest composition through activities such as sowing propagules of trees that attract honeybees without using fire for forest clearance (Zent & Zent 2004) and abandoning camp yet leaving changes in species composition that involve the coexistence of crops and noncrops (Politis 2001, Rival 2002, Rival 2006). The evidence of classless societies as disturbance agents that modified and managed environments earlier regarded as systemic concatenations of interactive, primeval biota and physical elements represents, in historical ecology, a divergence away from the core postulates of cultural ecology as well as equilibrium theory.

OTHER ECOLOGIES, OTHER HISTORIES

The distinction between historical ecology and other ecological viewpoints and disciplines has to do with anthropocentrism in one guise or another (Balée & Erickson 2006b). Historical ecology differs from cultural ecology principally on the criterion of human agency, as well as adaptation to the environment. Cultural ecology holds that the environment is not transformable. Rather, humans must adapt their cultures, technologies, and populations to it. Typically cultural ecology cannot explain higher-order social phenomena such as cities, states, and their dependent hinterlands because the core postulates are based on the environmental determinism of societies with simple technologies (cf. Cole & Wolf 1974). Historical materialism as a research program (Lakatos 1999 [1973]) allows for human agency in initial appropriations from and transformations of nature (Wolf 1982) but does not conceive of the environment, once changed by human hands, exerting a longer-term effect on subsequent human cultures in the region of the changes (Balée 1998a,b). Historical materialism lacked the *longue durée* notion of the *Annales* historical school, which would be developed a century later (Crumley 1998).

Historical ecology differs from anthropological systems ecology—itself a critique of cultural ecology—by moving away from a concern with the functionalist adaptations of human behavior to given environmental conditions and steady states of the ecosystem (Wolf 1999; cf. Biersaack 1999). Although historical ecology underscores the importance of time and contingency in environmental change (Botkin 1990, Scoones 1999, Zimmerer 2000), as does the new ecology, it is not a formulated record of geological changes that took place in the absence of humans, a study of human response to natural catastrophes (cf. Bilsky 1980), or merely recorded history or prehistory of any environment(s). It differs from the new ecology,

moreover, which is not accepted as a term for a distinctive model in ecology (Zimmerer 2000), by emphasizing an anthropocentric history. Historical ecology involves a tripartite array of conceptions of human time, borrowed from the *Annales*, especially the following: (a) *événement* (event) as a short-term, episodic phenomenon; (b) *conjoncture* (cycle), involving repetitive statistical patterns over a decade, quarter-century, or half-century or so; and (c) *longue durée*, empirical patterns of history and prehistory occurring over centuries (Braudel 1980, Crumley 2003).

Historical ecology has been most often conflated with environmental history. Environmental history is a fairly well-established interdisciplinary subject (Beinart & McGregor 2003, Crosby 2004, Hughes 2001, Worster 1993), but it is not a perspective that articulates hard-core postulates, such as historical ecology does. In this sense, historical ecology is not a part of environmental history nor is it parallel to it as a separate way of thinking (cf. Moran 2000, Myllntaus 2001). Environmental history encompasses the following: the comparative history of human activity in widely separated but structurally similar environments having similar politico-economic and historical conditions seen as resulting in convergent behaviors, the history of green movements and the relation of these to government policy, the history of environmental sciences and forestry, and the historiography of environmental history writing (Beinart & Coates 1995, Crosby 2004, Hughes 2001, Worster 1994). Historical ecology of a landscape, such as the Llanos de Moxos of Bolivia or the Upper Xingu of Brazil (Erickson & Balée 2006, Heckenberger et al. 2003, Mann 2002), would not be coterminous with environmental history of the same because historical ecology subscribes to a single theory of history and offers a model of how and why the landscape underwent transformation, regardless of the unique chronology of events.

Historical ecology differs from landscape ecology (cf. Moran 2000). Landscape ecol-

ogy focuses on spatial heterogeneity reflected in clusters of ecosystems and, with notable exceptions (Hayashida 2005, Turner 2005), tends to exclude, as a principle, intermediate human disturbance of environments and temporal changes in them as a qualitative factor in landscape transformation (e.g., Forman & Godron 1986, Turner 2005; see critiques by Crumley 1998, 2003; Denevan 2006). The concepts of design, ecology, and architecture of landscapes in the modern senses seem to envision re-education of human beings so they can live more harmoniously with self-contained natural systems and processes. Basically landscape ecology does not involve humans recapturing indigenous or local knowledge that could be of use to restoration ecology (Gunn 1994), which, in at least one of its crucial theoretical aspects, authenticity, is close to or the same as historical ecology (Anderson 2001, Egan & Howell 2001a,b, Higgs 2003, Jones 2004).

Historical ecology is sometimes compared with, or thought to be the same as, political ecology (e.g., Wolf 1999). Many have deemed the term political ecology a misnomer by stating that it concerns only politics and not ecology (Vayda & Walters 1999) and that it doesn't increase knowledge relevant to ecology and the life sciences (Scoones 1999). In some ways political ecology is more similar to environmental history with its emphasis on the critique of conservationist movements (Dove 2001). Political ecology does, however, comprise one feature relevant to historical ecology, to wit: the possibility of understanding and applying the critique of regnant folk models of nature and the environment for building a more enlightened approach to the reconstruction of past landscapes. Political ecology could be synonymous with applied historical ecology, but the term itself is perhaps still used too widely in disparate senses to refer to a single field or theory.

The anthropological ecology of practice (influenced by the sociology of Pierre Bourdieu) instantiates the third hard-core

postulate of historical ecology by stressing the differential environmental results obtained from disjunct economic and political histories in given regions (Nyerges 1997, Porro 2005). The proposal of an evenemental (*sic*) or event ecology (Vayda & Walters 1999) is bound to the particularistic limitations of case-by-case studies, similar to cultural ecology (Wolf 1982), and the omission of human agency in landscape formation; therefore it does not represent a new concept. The notion of *événement* in the *Annales* is but the short-term episodic feature affecting the formation of new landscapes—the others are *conjonctures* and *longue durée* (Braudel 1980, Crumley 1998), concepts incorporated into the core postulates of historical ecology. These postulates are, moreover, at variance with equilibrium theory and systems theory—essentially synonyms—in ecology.

Historical ecology is unlike, and fundamentally at odds with, ecological systems theory by a similar logic—the logic of the behavior of sentient, sapient beings with cultural capacities not just to transform species-rich environments into barrens of low diversity and landscape homogeneity, which clearly humans can do and have done, but also in certain cases to heighten the species diversity of local environments through ongoing resource-management practices. Historical ecology answers the call for an anthropocentric as opposed to an ecocentric or geocentric ecology (Balée & Erickson 2006a,b, Erickson & Balée 2006). Some thinking in ecology interprets these practices as always destructive, but that view is derived from the misunderstanding of human agency as a principle of some disturbance of the environment, which at a given level of intensity may be essential to sustaining diversity itself, a finding comprehensible within a historical-ecological viewpoint, which has affinities, as such, with other models in contemporary ecology, including nonequilibrium dynamics (Botkin 1990, Huston 1994, Turner 2005, Zimmerer 2000).

HISTORICAL CONTINGENCY AND ECOLOGICAL SUCCESSION

Environments—ecosystems in systems theory—undergo histories of changes in their fundamental characteristics similar to suites or guilds of species of plants and animals over time. Disturbance is the nomothetic origin of change (called succession) in species composition in a locale, and without it, ecosystems do not evolve to display climax communities, defined as ecosystems in their most mature state, with the highest diversity of species. Equilibrium or systems theory in ecology holds that climatically stable, large-area (such as continental) environments support more readily climax communities that consist for the most part of organisms that are K-selected, that is, organisms with long life spans, low numbers of offspring, and slow growth rates. With regard to forest communities, these would be trees and other structural organisms (Huston 1994). In contrast, small (such as insular) ecosystems consist for the most part of r-selected organisms, that is, plants with short life spans, high numbers of offspring, and fast growth rates. These ecosystems are more easily invaded by species from continents. Because islands develop through genetic drift clines and endemism (as with Darwin's finches), they are also more prone to extinctions. The theory, called island biogeography theory (MacArthur & Wilson 1967), proposes that the further an island is from a continent, the higher its endemism and species diversity and the higher the number of K-selected organisms. In contrast, the closer an island is to a continent, all else being equal, the lower its diversity as a result of its susceptibility to invasions of organisms from the mainland that replace local biota in the same or similar niches and therefore cause their extirpation and possible extinction (i.e., an outcome of competitive exclusion). The theory is elegant as a qualitative model of the rise and fall of species diversity on islands (i.e., ecosystems)

but has been deemed problematic in quantitative prediction (Peters 1991, Walker & del Moral 2003) because it does not specify a human or other historical parameter in the transport of many invasive species, some of which, as with the brown tree snake on Guam, have cascading effects in new environments (Fritts & Rodder 1998). The theory excludes history, and partly for that reason it has been hard to replicate in the real world of island diversity and invasion biology (Huston 1994, Lomolino 2000, Simberloff 1997).

Species invasion is one kind of disturbance. Disturbance traditionally, in fact, can be biotic or abiotic. It can also be cultural and historical. When they are demonstrably natural and unrelated to global warming, to ill-conceived levee and dike construction, and to other sorts of human error, hurricanes and floods are abiotic disturbance agents that can account for the reductions of forests with many K-selected species and the near grassland and savanna environments replete with r-selected species (Huston 1994). Hurricanes and floods can also cause terrestrial environments to become marine ones, for example, by splitting islands in two (Walker & del Moral 2003). Biotic factors include not only invasive species and their effects on local biodiversity (whether to lessen or replace it with new species), but also organisms that demonstrably alter the landscape inhabited by other life forms (Schmitz et al. 1997, Simberloff 1997).

Humans effect and are influenced by changes in the landscape. The ancient Greek dichotomy between *physis* (nature) and *nomos* (culture) (Glacken 1967, Hughes 2001) that foreshadows the Cartesian dualism of the body (material world) versus the mind (thinking) is inapplicable in understanding ecological succession as modified or interfered with by humans as the disturbance agency. Historical ecology deals not with the synthesis of humans and the environment, but focuses on the result of their cyclical interaction (*conjoncture*).

Historically the more centralized the political regime (i.e., the more it is similar to a

state), the greater the potential for the reduction of species diversity. Advanced industrial technologies with fossil fuels have long been known to reduce the genetic diversity of crop plants (Kates et al. 1990). Ancient civilizations using intensive agriculture (with terracing, irrigation, and fertilizers) reduced the diversity of traditional cultivars in agricultural fields as a result of taxation exigencies on a narrow range of foodstuffs (Zimmerer 1993). Interestingly, human depopulation, as occurred in the Amazon (and in the Americas generally and Australia) after contact as a result of the introduction of new pathogens, can lower the agrodiversity of landraces in areas where local knowledge and behavior are key to the management of traditional crops, including tree crops (Clement 1999a,b). If genetic diversity below that of the species rank is considered part of a region's diversity, then gamma diversity has been diminished as a result of people being removed from the landscape. Depopulation and other consequences of contact have led to the loss of agriculture and other basic technology altogether, even the disappearance of a society's cultural ability to manufacture fire, a feature once thought a sociocultural universal (Balée 2000, Cormier 2003).

With regard to advanced industrial agriculture (Kates et al. 1990), fertilization alone tends to reduce biodiversity (Huston 1994) by increasing the competition for nutrients among species originally present on a plot. It is a paradox of enrichment: Areas of high primary productivity (rich in nutrients) are often impoverished (but not always) in species diversity (Huston 1994). Despite high rainfall, large tracts of tropical rainforests have been increasingly prone to wildfires as a result of deforestation and possibly global warming artifacts of the twentieth century to date. In the future, the spread of GMOs and their potential for uncontrolled gene flow with native crops (a result in part of efficient dispersal mechanisms, such as anemochory) may reduce agro-diversity (Altieri 2004, Burney 1995/1996, Pilson & Prendeville 2004). But the human impact on the environment is

highly variable, and historical ecologists recognize that each landscape needs to be understood in terms of its specific cultural and historical influences on succession without prejudice toward human nature.

Ecologists recognize two basic kinds of succession: primary and secondary. Primary succession refers to the initial colonization of a substrate that had no life on it before, such as the succession of organisms on newly formed volcanic atolls or emergent deltaic islands on the point bars of meandering rivers. On a longer timescale, the succession of organisms in areas of former glaciation is an example of primary succession, as is the replacement of entire phyla by new ones over millions of years (Huston 1994). Secondary succession refers to the replacement of organisms by other types of organisms (such as K-selected by r-selected organisms) on a substrate that has been disturbed, as is the case with well-drained forest lands when subjected to hurricanes, tornadoes, droughts, blowdowns (wind shear), and, as has been especially well studied in the tropics, human disturbance by slash-and-burn cultivation or some other form of extensive agriculture. In all cases, both in systems theory and cultural ecology, the idea is that following the disturbance, the succession of guilds of organisms proceeds anew: In the case of tropical forests, for example, the succession proceeds from dominance by the r-selected species to the climax, the dominance and highest diversity of what are basically K-selected communities. This is the climax community, an ecological systems concept dating from the early twentieth century (Huston 1994). Although equilibrium theory has many defenders (Lomolino 2000), an increasing number of ecologists recognize disturbance not as an alien agent of change in an ecosystem, but as a basic part of the function and maintenance of diversity (Botkin 1990, Huston 1994, Perry & Amaranthus 1997, Petraitis et al. 1989, Smith & Wishnie 2000). The disturbance they prognosticate is not the removal of many species guilds (highly intense disturbance), but an ongoing disturbance of a much

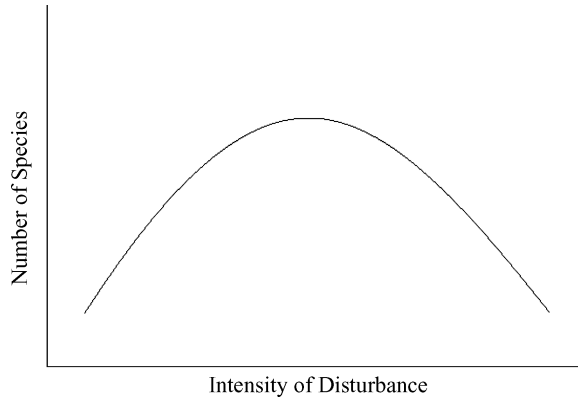


Figure 1

Simplified model of the intermediate disturbance hypothesis. Taken from Myers & Bazely 2003.

smaller scale, called intermediate disturbance, such as broadcast fire and chance treefalls in a forest without which local diversity (alpha diversity) may not be fully understood (**Figure 1**).

Intermediate disturbance (through broadcast burning, tree cultivation, settlement, and soil enrichment) has been seen to account for forest islands in West Africa (Fairhead & Leach 1996, Leach & Fairhead 2000). Evidence from Sierra Leone indicates resource exploitation and land use have varied over time, as have rangeland uses in southern Africa (Beinart & Coates 1995, Beinart & McGregor 2003), with the past use of forest sites for fortresses, the conversion of forests into charcoal for iron smelting, and the adoption of new cash crops (such as peanuts); such permutations of society explain forest composition and ecology today (Nyerges 1997, Scoones 1999; cf. Rudel 2002). Evidence from Namibia and Angola indicates that activities of historic kingdoms of the late nineteenth and early twentieth centuries expanded the frontier of orchards of fruit trees of palm, marula, bird-plum, fig, and baobab at the expense of savanna (Kreike 2003). Historical variation in landscape features is probably conditioned by changes in political complexity. Savannas sometimes seem to result from human choice (Erickson 2006, Scoones 1999). Intermediate

disturbance through controlled (broadcast) burning in African savannas, Brazilian *cerrados*, and the Bolivian Llanos de Mojos appears to promote the coexistence of trees and grasses by increasing landscape heterogeneity (Erickson 2006, Jeltsch et al. 1998, Mistry et al. 2005). Landscape heterogeneity induces edge effects that can reduce the diversity of species that require undisturbed forest for breeding, nesting, and reproduction (Renfrew et al. 2005), but sometimes habitat fragmentation leads to net increases of diversity (Fahrig 2003). Contrasting viewpoints hold that tropical forest islands on landscapes dominated by savanna species are relics of the Pleistocene and hence should be protected from human disturbance, the assumed cause of savannization, yet mechanisms of anthropogenic succession have been demonstrably associated with local structural and species diversity (Dove 2001, Fairhead & Leach 1996, Kreike 2003, Leach & Fairhead 2000).

Intermediate disturbance on a human scale involves the partial replacement of species of an episodic or cyclical nature on small plots of land, at times as small as treefall-sized light gaps and as large as 20 hectares (ha), as opposed to major perturbations such as clear-cutting, deforestation, selective logging, flooding, and eutrophication. Intermediate disturbance would not connote intensification, industrialization, or globalization, which can result in diminished species diversity per unit land area through overuse, fertilization, and erosion. The measurable effects of intermediate disturbance mediated by humans refer at least to one of the three types of species diversity recognized in ecology, alpha diversity. Alpha diversity is the number of species on a restricted locale with constant environmental parameters (such as drainage and soil type). Intermediate disturbance mediated by humans might also be seen in beta diversity, which is diversity over an environmental gradient such as slope or rainfall involving the distance between adjoining plots previously singled out by alpha diversity alone (Campbell et al. 2006, Erickson &

Balée 2006, Huston 1994). The third type of diversity, gamma, is the diversity of an entire region, such as the Amazon Basin. With regard to the Amazon Basin, because most of the species diversity predates the Pleistocene (Bush 1994, Vieira et al. 2001), prehistoric and early colonial societies probably had a negligible impact, except through the introduction of invasive species (biological invasions), these species occurring in inverse proportionality to pre-existing species diversity in the locales of introduction. Gamma diversity of Amazonia lacks a single, simple explanation (Bush 1994); it requires a complex model, grounded in historical ecology, capable of accounting not only for physical and temporal factors of diverse dimensions, but also for human-mediated ones impacting landscapes over time. The concept of ecological succession—the term originally used in systems and equilibrium theory—is still useful in a dynamic model of environmental change entailing increases and decreases of biological diversity (Huston 1994), the principal focus of inquiry in ecology, but for different reasons than in systems ecology. Ecological succession in historical ecology can be called landscape transformation (anthropogenic succession), of which there are several types.

Direct human impacts can be qualitatively assigned to a scale sensitive to time and place, indicative of least perturbation (e.g., by prehistoric foraging in the Arctic, the *puna* of the Andes, and the desert of the Great Basin), where a minimal human signature remains on the landscape from that time, to most perturbed (e.g., by advanced industrial agriculture and globalization), where prehistoric signatures are mostly erased as a result of complete replacements of guilds of species and the land-use intensity influenced by worldwide demand on agricultural labor and commodities (Kates et al. 1990, Rudel 2002). The intensity of human impacts on landscapes may lie in inverse proportionality to species diversity thereon (Burney 1995/1996), regardless of the rules that govern access to common resources of the landscape (Alvard & Kuznar 2001, Holt

2005, Smith & Wishnie 2000). The implicit corollary hypothesis is that where human-mediated disturbance was of the least intensity, species diversity would have been highest.

That viewpoint is a premise of wilderness areas. These are Western analogs to sacred groves (Russell 1997), and their implementation over time has passively affected species distribution and behavior on the landscape, for example, by converting hunted game to protected wildlife (Beinart & Coates 1995). Sacred groves have independent histories in sub-Saharan Africa, East and South Asia, and Mesoamerica (Byers et al. 2001, Gómez-Pompa et al. 1990, Russell 1997), and their function is to protect biotic diversity. Without sacred groves and wilderness areas, human-mediated disturbances may lessen diversity, especially if landscapes are not heterogeneous in situ (Turner 2005; cf. Renfrew et al. 2005). In other words, to maintain diversity, sacred groves in diverse cultural contexts suggest a folk belief that human-mediated disturbance ought to be excluded therein. Human disturbances of the environment, however, are ultimately scalar and temporal (Allenby 2000, Balée 1998b, Crumley 2001, Erickson 2000, Hayashida 2005, Peterson & Parker 1998, Sheuyange et al. 2005) and not in their genesis sociobiological.

Human-mediated disturbances of certain tropical forest landscapes in prehistory may have decreased alpha diversity, or had no measurable effect on it. Using the Shannon-Weaver index of biodiversity, Lentz et al. (2002) found that anthropogenic forests resulting from ancient Maya agriculture in Belize did not result in an increase in biodiversity, although forest composition is essentially anthropogenic on the study plots (see also Campbell et al. 2006, Gómez-Pompa et al. 1987, Wiseman 1978). Conversely, Lentz et al. (2002) do not show a reduction in alpha or beta diversity as a result of human impacts because prehistoric baseline inventories of all taxa are unavailable. Human-mediated disturbance of lowland Mesoamerica over time appears to have resulted in forests dominated

by a few species [in terms of their relative biomass and other components (e.g., relative density and frequency) of ecological importance]. These guilds of dominant species are referred to as oligarchies (Campbell et al. 2006, Peters et al. 1989).

Oligarchic forests (usually replete with fruit trees and other economic plants) contrast with sacred groves or other relatively undisturbed forests existing in areas used by traditional societies to the extent that traditional patterns of human-mediated disturbance specifically involving broadcast and swidden burning result in a mosaic of landscapes. Recent work suggests such mosaics—landscape heterogeneity—tend to increase not only density of wildlife, but also beta diversity of flora and fauna. Intermediate disturbance by traditional societies employing broadcast fire in Australia, Africa, and the Neotropics has demonstrably resulted in increases of alpha diversity of vegetation (Bird et al. 2005, Lunt & Spooner 2005, Mistry et al. 2005, Pyne 1998, Sheuyange et al. 2005). In other words, in these instances, human-mediated intermediate disturbance and management of tropical forest biotas may be essential to the explanation of their diversity in situ. *Homo sapiens* in certain socioeconomic contexts with historically determined land-use strategies may act as a keystone species in which the diversity of entire landscapes over time is dependent (Balée & Erickson 2006b, Denevan 2001, Erickson & Balée 2006, Mann 2002, Storm 2002).

PRIMARY AND SECONDARY LANDSCAPE TRANSFORMATION IN AMAZONIA

Primary landscape transformation in the Central and Lower Amazon Basin involved mound building as well as changes in river courses to effectuate the ease of transportation, with seemingly negligible effects on species diversity (Neves & Petersen 2006, Raffles 2002). The effects, if any, on alpha and beta

diversity are unknown from prehistoric manipulation of soils and drainage in the Upper Xingu (Lower Amazon) by approximately AD 1000 (Heckenberger et al. 2003), as well as in other areas of Amazon Dark Earth (anthropogenic soils) (Erickson 2003). Increases in alpha diversity of flora and fauna by indigenous resource management and use, however, have been reported in numerous environments in the Neotropics (Balée 1993, Fedick 1995, Politis 2001, Posey 1985, Rival 2002, Stahl 2000, Zent & Zent 2004). Landscapes of the Ka'apor, Guajá, and Tembé Indians of Pre-Amazonia (easternmost Amazonia, of approximately 10,000 km²) include high forests (relatively undisturbed) and old fallow forests [of intermediate indigenous disturbance, dating from 40 to 150 years ago (cf. Myers & Bazely 2003)]. Old fallow forests instantiate anthropogenic secondary succession. Forest inventories (completed using standard biological inventory procedures as discussed in Campbell et al. 2006) of 4 ha of old fallow and 4 ha each of nearby high forest across Pre-Amazonia showed that (a) tree alpha diversity measures between fallow forest and high forest were insignificantly different; (b) adjacent plots of high forest and fallow forest shared only approximately one-half the number of shared species within inventory plots of either type regardless of distance between those more similar plots in the same category; and (c) the effect is a net gain in both alpha and beta diversity, the latter involving the gradient of time (Huston 1994), the high forest being older than the fallow forests (Balée 1993, Balée 1998b).

Secondary forests are not necessarily more impoverished in diversity than primary forests (Schulze et al. 2004). Even if soils in many secondary forests are significantly more fertile than those of primary forests (Denevan 2001, 2006; Erickson 2003; Erickson & Balée 2006; Neves & Petersen 2006), the secondary forest soils have not noticeably been reported to suffer from the paradox of enrichment. The habitat of the Sirionó Indians of the Bolivian Amazon encompasses a heterogeneous land-

scape of well-drained forests on relic mounds, slightly inundated forests at the base of such mounds (called pampa forest), and seasonally inundated and poorly drained savannas, which account for approximately two-thirds of the landscape (Erickson 2003, 2006; Erickson & Balée 2006; Townsend 1996). Mound forests are all anthropogenic and date from approximately 500 to 1000 years since the time of their construction and continuous habitation; pampa forests are also anthropogenic, artifacts of mound construction. Two 1-ha tree inventories, one of a mound forest 18 m in height and the other of a nearby pampa forest, showed (a) a similarity in the number of species, with 55 on the mound forest and 53 in the pampa forest; (b) a shared 24 species between the two forests (yielding a large percentage compared with Pre-Amazonia); (c) a total number of species in both forests of 84 [(55 + 53) - (24) = 84]; (d) and a dominance of oligarchies in both forest types (the 10 ecologically most important species on the mound inventory constitute 65.4% of all importance values of all trees on the plot, and the top 10 on the pampa inventory represent 70.9% of all importance values of all trees on that plot). The oligarchies are, however, somewhat different (with one species of palm having an importance value of 46.21 on the mound and 9.21 on the pampa and another palm species having an importance value of 41.2 on the mound and 83.27 on the pampa), which is probably a result of slope (Erickson & Balée 2006). Savannas are the original, oldest landscape, and these are maintained by periodic burning (Erickson 2006). Flooding occurs in the savanna every year, and it alternates with a marked dry season. The total number of vascular plant species in the savanna is less than 20 (Townsend 1996), with sedges and grasses by far most dominant.

The primary landscape transformation that accrued at the Ibibate Mound Complex (the mound and adjoining pampa forest) would be, in ecological terms, a primary succession, although that term most commonly excludes anthropogenesis of the

landscape (e.g., Huston 1994). The transformation involved a sweeping replacement of savanna species with flood-intolerant trees, having an alpha diversity of several orders of magnitude higher than the savanna. Comparable mounds, specifically prehistoric shell middens (including gastropods and bivalves of many different species) in southwest Florida and southeastern Louisiana, which support higher diversity than any nearby marshes regardless of salinity levels, represent human-mediated increases in alpha diversity (Kidder 1998, Marquardt 1992). Therefore forest ecology in these cases of primary succession without natural causes is actually an artifact of culture and society. Perhaps to represent better and distinguish the impact of human-mediated disturbance of the environment, given that it is scalar and temporal (Peterson & Parker 1998, Sheuyange et al. 2005), one should therefore refer to primary and secondary landscape transformation when discussing biotic and environmental change on a human scale of time. In other words, alpha and beta diversity are amenable to analysis in all three modes of historical time (Braudel 1980) and are hence the definitive material of historical ecology.

INVASIVE SPECIES AND LANDSCAPE HISTORIES

Biological invasions sometimes refer only to invasive species that replace other (usually) structurally similar species in the new environment, but the term here refers both to invasive species in the conventional sense and invasive diseases (Turner 2005), including bacterial, protozoan, viral, and prion infections, that take on epidemic characteristics in regard to previously unexposed native flora and fauna, including humans (**Table 1**). The integration of landscape ecology and epidemiology (Turner 2005) is analogous to the recognition in historical ecology that human activity has been associated with a variety of new pathogens and their distribution and that human societies' political organization mirrors

their susceptibility to epidemic disease, as well as their potential to generate biological invasions in new environments (Newson 1998).

Biological invasions that involve the transfer and spread of invasive species from one point to another have been termed succession in action (Myers & Bazely 2003). Invasive species are introduced (exotic) species of plants and animals that have become weedy supplanters of existing (native) flora and fauna. Definitions vary, but usually weedy organisms are considered out of place; they multiply and spread rapidly at the expense of other organisms (Crosby 2004, Myers & Bazely 2003). Only a minority of introduced species became invasive. The success of invasive species, as weeds, depends on biotic and historical factors, alone or in combination, specific in each case. The invasive species may have no natural enemies in the place of introduction (as with the Brazilian rubber trees in Malaysia), a view originally proposed by Darwin (Hierro et al. 2005), or they may fill an empty niche in the place of introduction (Hierro et al. 2005). They may have higher numbers of duplicate chromosomes (such as tetraploids), which give them greater reproductive success, as is the case of all invasive grass species from Europe in North America (Myers & Bazely 2003). Structurally, invasive species are likely to be ruderal (Hierro et al. 2005, Huston 1994) rather than treelike, but there are many exceptions. They may exude secondary metabolites toxic to native biota but not to others in their place of origin (Hierro et al. 2005). Invasive species may have coexisted with humans longer than the affected (replaced) species in the place of their introduction and subsequent expansion (Burney 1995/1996, Hierro et al. 2005). Finally, propagules of many invasive species are located closer to shipping lanes in their points of origin and in their initial places of dispersion (Hierro et al. 2005) and hence are easily transported often as ballast (Burney 1995/1996, Crosby 2004, Russell 1997). Biological invasions since the emergence of modern humans usually have occurred with historical agency; these are called

Table 1 Some modern human-mediated invasive species and diseases

Taxon	Common name	Origin	New environment	Date established	Use (if any)
<i>Agave</i> sp.	Century plant	Mexico, Southwest United States	Southern Africa	Twentieth century	Fencing
Avian influenza (H5N1 virus type A influenza)	Bird flu	East Asia	?	Early twenty-first century	–
<i>Boiga irregularis</i>	Brown tree snake	South Pacific	Guam	1945–1950	–
<i>Casuarina equisetifolia</i>	Australian pine	Australia	South Florida	Early twentieth century	Windbreak Scenic byway
<i>Cerato stomauchinii</i>	Dutch elm disease	Europe	Northern United States	Late nineteenth century	–
<i>Cryphonectria parasitica</i>	Chestnut blight	Europe	Northeastern United States	Late nineteenth century	–
<i>Melaleuca quinquenervia</i>	Melaleuca	Australia	South Florida	1906	Ornamental
<i>Mycobacterium bovis</i>	Bovine tuberculosis	North America	Southern Africa	Late twentieth century	–
<i>Myocastor coypus</i>	Nutria	South America	Louisiana, Florida	1941	Fur
<i>Rattus rattus</i>	Black rat	Asia via Europe	North America	Sixteenth century	–
<i>Schinus terebinthifolius</i>	Brazilian pepper	Southern Brazil	South Florida	Nineteenth century	Ornamental
<i>Ulex europaeus</i>	Orse	Europe	New Zealand	Nineteenth century	–

Sources: Beinart & Coates 1995, Burney 1995/1996, Caron et al. 2003, Fritts & Rodder 1998, Hierro et al. 2005, Kiddler 1998, Myers & Bazely 2003, Russell 1997, Schmitz et al. 1997, Simberloff 1997, Simberloff et al. 1997.

human-mediated invasions (Myers & Bazely 2003). [Note that not all of these biological invasions have been human mediated: For example, because of their superior swimming abilities, hippopotami, elephants, and deer hopped from one island to another across open seas during the Quaternary (Burney 1995/1996).] Regardless of the transport agent, biological invasions have been the principal proximate cause of extirpations and extinctions of native flora and fauna (Burney 1995/1996, Myers & Bazely 2003, Pimm 1991) when compared with others such as overexploitation (as with extinctions of the Carolina parakeet and the Giant Auk) and locally cascading effects of the removal of singularly important species (keystone species) on which other species depend.

Invasive species are often transported to new destinations accidentally (Crosby 2004, Myers & Bazely 2003) as in ship ballast (Crosby 2004, Myers & Bazely 2003, Pilson & Prendeville 2004, Russell 1997, Simberloff et al. 1997), the case, for example, with red tides (toxic dino-flagellates) (Burney 1995/1996), a phenomenon that increased significantly after approximately 1500 (Crosby 2004). Perhaps just as often, however, invasive species have been introduced intentionally (before their invasive features in the new environment were known) to fulfill desired functions, including as ornamentals, windbreaks, scenic byways, erosion control, fencing, and livestock feed (Beinart & Coates 1995, Myers & Bazely 2003, Simberloff et al. 1997). In some cases, these species are indicative of landscapes inhabited and modified by human societies in ancient times, as with the Kentucky coffee tree, American chestnut, and butternut, propagated at long distances from their original distribution in prehistoric North America, and the walnut and sweet chestnut in the British Isles, taken there by the Romans in the first millennium AD (Russell 1997).

The restorationist (and conservationist) notion that native species in situ are superior and preferable to introduced species

is traceable to nineteenth-century European thinking that conjoined culture and wilderness into a single landscape (Hall 2005), such as the German *Naturgarten* (Wolschke-Bulmahn 2004). Specifically regarding diversity, goals of restoration ecology that include eradication of exotic species can be traced to antiquity, with Plato's doctrine on the desirability of high species diversity—"the world is the better, the more [living] things it contains" (Glacken 1967)—insofar as the richer the species diversity, the greater the resilience of the area to biological invasion (Pimm 1991; cf. Simberloff 1997). Plato further thought the state of nature was only bountiful when actively managed by humans (Hall 2005). More recent study of Mediterranean so-called ruined landscapes suggests many represent the anthropogenic expansion of forests rich in species diversity in the region, not the reverse (Grove & Rackham 2003).

Biological invasions have caused reductions and extirpations of numerous species through mechanisms including direct competitive exclusion (Burney 1995/1996, Simberloff et al. 1997). In the case of introduced pathogens, their success is only mitigated by the extent to which a host population survives and can be a reservoir for future endemic propagation (Newson 1998). The species barrier between humans and other animals is effectively broken down by diseases that are anthrozoonotic (the vector human, infecting other animals), such as tuberculosis, measles, and human herpes virus (Karesh & Cook 2005), and by habitat loss [an extreme example of habitat fragmentation, which in a general sense does not always cause reductions in species diversity (Fahrig 2003)], accounting for changes in relations between pathogens and hosts, as with chronic wasting disease of mule deer, white-tailed deer, and Rocky Mountain elk (Farnsworth et al. 2005). Malaria seems to be both zoonotic (the vector an animal, infecting humans) and anthrozoonotic in the relationship between the Guajá people and their pet monkeys in Amazonian Brazil. Specifically, the reservoir

for malaria in this case seems to alternate between humans and monkeys (Cormier 2005). Some pathogens developed in livestock domesticates have jumped species barriers several times. Bovine tuberculosis (from domestic cattle) has infected wild bison in Canada, deer in Michigan, and Cape buffalo, lions, leopards, cheetahs, greater kudu, and chacma baboons in South Africa (Caron et al. 2003, Karesh & Cook 2005). Avian influenza (H5N1 type A influenza virus), which has potential to become pandemic, is notable for high morbidity as well as for having the potential for multiple vectors: wild birds, house cats, big cats, chickens, pigs, and humans, all of which have human-mediated distributions and interactions (Normile 2005). Scrapie (a prion disease) in sheep jumped the species barrier and became mad cow disease in cattle, a strain of which appears to have crossed the species barrier to humans as variant Creutzfeldt-Jakob disease (Karesh & Cook 2005). Disease ecology to the extent that it links humans and other biota, by affecting distributions on the landscape of both, becomes more fully comprehensible within the temporality of historical ecology (Newson 1998, Turner 2005).

The morphology and behavior of invasive biota may be predictable (they may be weedy, opportunistic, and genetically plastic with a capacity to mutate rapidly and, in some cases, to infect other organisms and jump species barriers), but biological invasions do not necessarily all result in net reductions of alpha, beta, or even gamma diversity. Indeed, many *r*-species invaded *K*-dominated environments before the advent of the *Homo* migrations out of Africa at the beginning of the Pleistocene, and initial modern humans did not have many invasive species other than head lice and a few others, none of which were domesticated, to transport with them (Burney 1995/1996). The question then is how long must a species exist in a given environment to no longer be considered invasive? An arbitrary classification for invasive plants is used in European forestry, whereby archaeotypes existed on the continent before 1500 and neophytes are those

plants arriving after 1500; other species' origins are simply unknown, and the species are denoted as cryptogenic (Myers & Bazely 2003). Some invasive species in fact function as keystone species, and "even their removal may not cause a return to the 'uninvaded' state" (Myers & Bazely 2003). This is arguably the case with invasive species that have altered the fundamental structural characteristics of landscapes, such as melaleuca trees, which form woodlands in Florida where previously there were none, hence altering the distribution of numerous other species of flora and fauna (Schmitz et al. 1997, Simberloff 1997).

DISCUSSION AND CONCLUSIONS

On the basis of the notion that native species are more desirable than exotic ones, not only for aesthetics but for reasons related to protecting biotic diversity, efforts in restoration ecology have tended to focus on the removal and eradication of invasive species. These efforts have met with mixed results. Restoration ecology (applied historical ecology) essentially requires the knowledge of reference conditions of a past state of the landscape to attain authenticity (Egan & Howell 2001b, Hall 2005, Hayashida 2005, Higgs 2003, Jones 2004).

Historical ecology can supply the reference conditions needed for authenticity of landscape reconstruction (Egan & Howell 2001a,b, Hayashida 2005, Higgs 2003). The sources vary and are derived from research in paleoecology, ethnohistory, history, and archaeology (Crumley 1994, 2003; Erickson 2003; Hayashida 2005; Heckenberger et al. 2003; Kidder 1998; Turner 2005); from ethnography and ethnobiology (Balée 1993, Posey 1985, Posey & Balée 1989, Rival 2002, Zent & Zent 2004); from biological inventory work (Campbell et al. 2006, Erickson & Balée 2006, Turner 2005); and from research on symbols and language. Landscapes, as gardens, communicate meaning about their users and owners in the Peruvian Andes (Finerman

& Sackett 2003). Knowledge of archaic cultigens is retained in memory and reflected in ritual in Borneo and elsewhere in Southeast Asia (Dove 1999). Language catalogs past states of landscapes by current topographic terms (Russell 1997); it also embodies past methods of resource exploitation, such as agriculture in prehistoric equatorial Africa, reconstructed by historical-linguistic study of living Bantu languages (Vansina 1990). Language retains evidence of former economic valorization of species and landscapes, as with the cacao export cycle from the Amazon in the eighteenth century, reflected in the indigenous borrowing of a nonnative term for cacao even though the tree is native (Balée 2003). Marking reversals for biota represent a chronology of landscape transformation inscribed in vocabulary, such as the change, in many Mesoamerican languages, in the name for sheep, introduced from Spain, to the original name for white-

tailed deer, a native forest animal but increasingly rare as a result of the invasion of pasture for sheep grazing, and vice versa (Witkowski & Brown 1983). Historical ecology is interdisciplinary, and in one of its disciplines, anthropology, it is clearly intersubdisciplinary.

Applied historical ecology may become the holistic engagement of knowledge from diverse disciplines for the benefit of human societies and selected biota and landscapes. It is derived from several fields with the objective of determining reference conditions of past landscapes with the highest degree of authenticity for the period chosen for restoration. The remaining problem, in terms of the applications of historical ecology, concerns political questions as to who will be privileged in determining the desired time depth (Higgs 2003, Jones 2004) and the associated state of historical knowledge (Anderson 2001, Hall 2005) about the landscapes to be restored.

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