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LAND USE AND LAND COVER CHANGE PATHWAYS AND IMPACTS

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1 Introduction

Of the challenges facing the Earth over the next century, land-use and land-cover changes are likely to be the most significant. This anthropogenic process affects many parts of the earth's system (e.g., climate, hydrology), global biodiversity, and the fundamental sustainability of lands. Various estimates indicate that 50 percent of the ice-free land surface has been affected or modified in some way by human activity (Vitousek et al., 1997), while 10 to 55 percent of the net primary productivity has been captured by human land use activities (Rojstaczer et al., 2001). Over the next century, global population is projected to increase by 50-100% and it is likely that there will also be an increase in the global standard of living. Thus pressures to further convert or manage "natural" ecosystems for human needs as well as capturing more of the global net primary productivity are also likely to increase.[1]

Understanding of the patterns of land-use and land-cover change has increased significantly over the last decade (e.g., Turner, 2002a). This has been facilitated in part by increased awareness of the issues and by the large number of focused studies directed to understanding the nature of land-cover and land-use change (LCLUC). These studies have made significant advances in furthering our understanding of the socio-economic drivers of LCLUC, the impacts on natural and human systems, as well as feedbacks between natural and human systems. Given the large number of case studies that have been performed, we now have the opportunity to look broadly at the results of these studies to assess if there are fundamental patterns of land-use and land-cover change that consistently appear regardless of global location, social organization, economic state, etc. Furthermore, we can now assess whether there are persistent impacts of LCLUC that can be identified and related to the overall patterns.

Previous studies have attempted to assess whether there is a common pathway of landcover change, linked to common socio-economic drivers (e.g., Turner et al., 1990; Lambin et al. 2001). A recent study of tropical deforestation sought to assess common drivers from an analysis of the results of 150 case studies (Geist and Lambin, 2001). Here we develop a typology of (1) land-cover change (pathways), (2) link them to broad drivers (both land uses and their ultimate causes – policy, economics, social, environmental), and (3) address the major impacts consequences of the land-cover conversions. The typology is derived from examination of case studies results conducted under the NASA Land Use Land Cover Change (LCLUC) program since 1997 (which are summarized in this volume), and where appropriate, the results of studies conducted within the broader community of land change science.

The search for general principles from case studies is constrained by the limits of the various case studies that inform our analysis (i.e., they are specific to particular places, times). Searching for commonalities from the diverse environments and drivers of land-cover and -use change will necessarily be subject to uncertainty and error. However, case studies are essential for informing large-scale syntheses, and their results must contribute to syntheses describing general principles (Lambin et al., 2001). To a large extent the case studies from which we draw examples are focused on European colonization of western hemisphere regions, reflecting past orientations of the LCLUC program, and thus do not necessarily capture land-change processes in other parts of the world. One of the advantages of western hemisphere emphasis is that the time scale for significant changes in many landscapes is compressed relative to other parts of the globe which may extend over thousands of years. We recognize that the Americas were substantially altered by pre-Columbian societies (e.g. Turner and Butzer, 1992; Denevan, 2001). However, this region offers an excellent opportunity to understand the processes and impacts of the massive transformation that have affected this part of the world, particularly over the last 150 years.

The spatio-temporal scale of analysis strongly affects the results. A spatial scale that is too large (e.g., continental) will fail to capture the important interrelationships among processes and therefore lack specificity, while a scale that is too small (e.g. a village) will not encompass a sufficient number of interrelationships to understand the region. Likewise, a short temporal scale may miss past human-environment dynamics that reshaped the very landscape under study. We do not resolve the central issue of scale, but attempt to draw from studies cast at intermediate scales.

It is not possible within the scope of this chapter to put forth all the detailed case study results that have informed this synthesis as has been done elsewhere (Geist and Lambin, 2001). We present here representative case studies from a number of different ecological regions in the Americas, but the reader should recognize that much additional data is presented in the proceeding 25 chapters.

2 Representative land-cover and land-use histories

2.1 TROPICAL DEFORESTATION: THE AMAZON

2.1.1 *Scope of the change*

The Amazon region experienced deforestation prior to 1975, but on a small scale. The population collapse of indigenous communities by war and disease following European discovery resulted in a pattern of small communities practicing shifting cultivation and moving their settlements frequently (Beckerman 1991; Roosevelt 1989; Meggers 1971). Assessments using Landsat MSS found less than one percent of the Amazon Basin evidenced deforestation in 1975 (though the resolution of MSS probably hid many areas that were in secondary succession). Initiated in 1970, Brazil's Program of National Integration, associated with a major initiative to build roads across the Amazon and to settle land along these roads with colonists, began to change the rates of deforestation. The east-west Transamazon Highway, constructed in less than four years, cut a path from the northeast of Brazil to the frontier with Peru. The north-south Cuiaba-Santarem highway and the Belem-Brasilia highway linked, respectively, the central and eastern parts of the Amazon to the central part of Brazil (Moran 1981).

These roads were catalysts of land cover and land use change in the Amazon. Human settlements were promoted by a series of settlement schemes providing attractive incentives and virtually free land, attracted people who quickly began cutting forest in order to ensure their claims to land (Moran 1976, 1981; N. Smith 1982; Fearnside 1986). For the period up to 1988, Skole and Tucker (1993) were able to document that up to 15 percent of the Brazilian Amazon had been deforested and seriously fragmented—a rate close to 0.5 percent per year. This rate actually hides the real local rates of deforestation. In settlement areas the rates of deforestation were commonly in excess of one percent per year, while vast areas remained out of reach of human occupation by Brazilian society. Percentages, too, tend to hide the scope and magnitude of deforestation in the Amazon: one percent of the Brazilian Amazon is equivalent to 50,000 km^2 or an area the size of Belgium. Thus, while the percentage of deforestation is higher in Ecuador and Mexico's tropical forests, the area being deforested in Brazil is several orders of magnitude larger. Recent updates by EU scientists provide a needed reasssessment (Achard et al. 2002).

Rates of deforestation in the Brazilian Amazon reached a initial peak near 1987-88, followed by a notable decline. The drop was not a result, as some thought, of more effective conservation or of a more effective set of policies, and turned out to be temporary. It was, rather, the result of hyperinflation and a serious credit deficit in Brazil. After the introduction of the new currency, and effective control over inflation and exchange rates in 1994, the rate of deforestation surpassed (nearly doubled) the first peak of 1987-88, generating serious concern to policy-makers. This second spike in the rate of deforestation can probably be explained by the suppressed rates of deforestation from 1988 to 1993, and the opportunities that economic stabilization offered. Within two years, deforestation rates settled down to the more common rates of about 0.5 percent for the Basin, although in settlement areas the rates remained considerably higher, i.e. above 1 percent annually (Wood and Skole 1998; Moran et al. 2002; Brondizio et al. 2002; Lu et al. in press).

2.1.2 *Trajectories of Land Change*

Land change begins with the clearing of forest through slash-and-burn techniques, commonly followed by the planting of annual crops or the creation of pastures. In some cases, fields are kept in cultivation continuously, but this is rare. Only in areas with alfisols of relatively high fertility with favorable texture are there examples of continuous cultivation for over 25 years with some crop rotations in place (Moran et al. 2002). In most places the low nutrient conditions of oxisols and ultisols, dominant in over 75 percent of the Amazon Basin, present constraints to continuous cultivation without major fertilizer inputs—which remain prohibitively expensive throughout most of the Amazon basin. Without fertilizers, farmers have tended to plant pastures and graze cattle at very low densities as a preferred strategy. Cattle ranching has a long tradition in Latin America and receives favorable treatment by policy makers as a repository of value and a hedge against inflation and uncertain economic cycles. It is the traditional tool for occupying large areas of the vast frontiers of Latin America with few people and labor scarcity (Walker et al. 2000). Thus, Rondonia (predicted in the 1970's to become a center for cocoa production) and the Altamira region of Brazil, both of which have patches of high quality soils, are dominated by pasture land (Moran 1988). Less than ten percent of the land area is in crops, with less than four percent in annual or staple crops (e.g., rice, corn, beans, manioc), and the rest in some form of plantation or tree crop (e.g., cocoa, rubber, sugar cane, coffee) (Brondizio et al. 2002). Nevertheless, the typical nature of change is one from undisturbed forest to a landscape cleared for management for cultivation or ranching, with a significant component of secondary regrowth on abandoned land.

Farmers experiment with a variety of strategies. They tend to clear more land than they can manage at the outset, and rates of six percent per year are not unusual when first arriving (McCracken et al. 2002). This rate quickly drops as farmers realize the high cost of managing regrowth through secondary successional dynamics ((Mesquita et al. 2001; Laurence et al. 2001; Zarin et al. 2001, 2002; Moran et al. 1994, 1996, 2001; Tucker et al. 1998; Steininger 1996). Those with more favorable biophysical initial conditions and some capital move towards plantations and pasture formation; those with less favorable conditions continue to combine annual crops with modest increments in pastures on lands with exhausted fertility as a way of combating the return of woody species by succession. Over time, those with favorable conditions tend to evolve a balance of crops and pasture, while those with unfavorable soil conditions and poor labor and capital resources tend to concentrate most of their land in pastures.

2.1.3 *Forest Conservation Efforts*

While legislation in Brazil has sought to protect up to 50 percent of the areas occupied by settlers, raising this figure to 80 percent more recently, there is little enforcement of this legislation even if it were wise to do so. Given poor enforcement and the likely fragmentation of these "back of the property" conservation areas, this legislation seems less than effective as a means of conserving flora and fauna biodiversity. Recent evidence from a study in Rondonia suggests that reserves, including extractive reserves, provide the only effective mechanism for conservation in areas of settlement (Batistella 2001). Reserves in themselves do not ensure conservation, but only where local people maintain a vested interest in protecting the forest for their own economic well-being—as in the extractive reserves in Machadinho, Rondonia—may forests be protected from the pressure for occupation and land clearing.

2.1.4 *The Amazon in the Context of Global Tropical Deforestation*

Research on causes and driving forces of tropical deforestation reveals that neither single factor causation (e.g., poverty, population growth) nor irreducible complexity adequately explain the dynamics of tropical deforestation (Geist and Lambin, 2001). Deforestation is driven by regional causes, of which the most prominent are economic, institutional, and policy factors which seem to drive agricultural expansion, logging, and infrastructure development (Angelson and Kaimowitz 1999; Lambin et al. 2001). Logging appears to be a more important driver at the outset of forest clearing in Africa and Asia than in Amazonia, where farming and ranching seem to precede logging activities. In Middle America, selective logging (not clear cutting) has provided road networks ultimately followed by farmers (e.g., Turner et al. 2001). The vastness of the Amazon, and the precariousness of infrastructure has probably mitigated the impact of logging in the Amazon as a primary driver of land change. Recent work by Cochrane (2000, 2001, 2002) and Nepstad and colleagues (1999, 2000) suggests that loggers are beginning to lead the way in places where some primary road infrastructure has been created.

2.2 FORESTATION: NEW ENGLAND

The environmental history of Massachusetts provides a representative case study for landscape experiencing all major techno-economic phases affecting land use (Foster et al., 1998; Hall et al., 2002). Prior to colonial settlement, the Massachusetts landscape was predominantly forested, though there is evidence for some manipulation of the landscape by native populations (Doolittle 2000; Mulholland, 1988). The colonial experience witnessed significant occupational growth, ultimately distributed somewhat evenly across those conditions that could sustain cultivation and/or resource extraction. With the expansion of the nineteenth-century industrial revolution, population concentrated in industrial towns, reducing rural population densities into the middle of the twentieth century, despite an over sharp rise in overall population numbers. Following World War II, industrial activity subsided as core manufacturing activities relocated (e.g., textiles to the south) and the regional economy shifted to high technology and service industries.

Land-cover and land-use change in Massachusetts followed these transformations, though not as a simple relationship with population (Figure 1). The initial colonization and movement towards the interior was accompanied by significant forest clearing, the majority of which was pasture. By the middle 1800s, the region experienced its greatest proportion of cleared land with only 20 to 40 percent of the land remaining forested. Those forested regions that remained were heavily managed for forest products. With the rise in industrial activity and the opening of the American west for settlement in the middle 1800s, there was a large decrease in rural populations and shift in agriculture to market crops to support the growing populations of the industrial town and cities of the region. Furthermore, higher efficiency agricultural practices in the west coupled with efficient rail transportation made the use of agricultural land in Massachusetts uneconomical for all but the highest value crops. Large-scale agriculture abandonment took place during this period, giving rise to an extensive period of afforestation such that by 1950 the region was 70 to 80 percent forested. The last 50 years has seen a decline and fragmentation of this forest cover associated with urban expansion and suburban/peri-urban development.

The environmental impacts from these enormous changes in land cover can only be broadly framed. Extensive measurements were not made with the exception of forest composition and structure (Forster et al., 1998; Hall et al., 2002). At the height of deforestation, the forest structure of remaining stands was one of relatively youthful, even aged stands. With afforestation, even-aged stands of early successional species (white and pitch pine, red maple, and birch) became established on abandoned agricultural land. Towards the end of the twentieth century, mature forest structures with long-lived shade tolerant species have become re-established (Hall et al., 2002). Associated with these changes in cover and stand properties have been changes in species composition. Except for the loss of chestnut, most of the changes have been in the relative abundance of species with a decline in the abundance of long-lived species (e.g, beech, sugar maple) and an increase in early successional species (e.g., red maple, poplars, white pine). Introduction of exotic pests and pathogens has probably wrought the most significant change on these forests. Chestnut was once a significant canopy species but is now present only as subcanopy sprouts because of a fungal pathogen introduced early in the 20th century (Paillet, 2002). Beech bark disease and hemlock wooly adelgid are additional examples of exotic species causing changes in forest structure and composition (Twery and Patterson, 1984; Orwig et al., 2002)

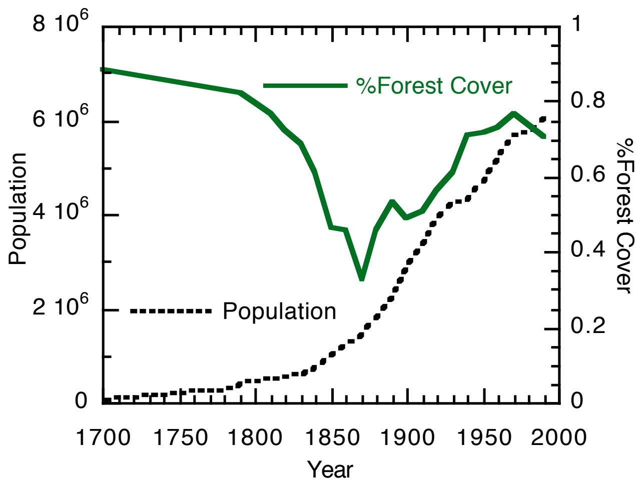

Figure 1. Relationship between forest cover and population, Massachusetts USA. See text for explanation (source??)

2.2.1 *New England in the Context of Mid-Latitude Land-Use History*

The overall pathway of land-cover change in Massachusetts is not unidirectional in the face of ever increasing occupation from the Colonial era to present. Deforestation registered during the colonial frontier and subsequent agrarian phases of occupation. Forestation, however, marked the industrial phase, while forest fragmentation marks the advanced industrial-service sector phase and its suburban/peri-urban settlement patterns. This pattern is broadly representative of land-cover changes throughout the northeastern and upper Midwest of the United States, although the dates of the transformations and the duration of landscape states vary. For example, over the last 30 years the upper peninsula of

Michigan has witnessed an increase in forest cover from regrowth on abandoned farmland, but recently there has been an increase in fragmentation accompanied by a decrease in the size of ownership parcels driven by expansion of second home ownership and suburbanization (Drzyzga and Brown 2002). This 30-year experience is similar to the 150-year one in New England. Western Europe also displays a shift towards more forest during the industrial era, and increased landscape fragmentation recently, although land management policies there may reduce the scale fragmentation found in the United States.

2.3 WATER WITHDRAWAL IN ARID/SEMI-ARID LANDS: OWENS VALLEY

The land history in the semi-arid Owens Valley, California, mirrors in many ways that observed in mid-latitude temperate regions world wide, in this case, driven by competing demands on its water sources(Putman and Smith, 1995) and illustrated in a time-line of major events in Figure 2. Though situated in a high desert of the Great Basin, the Owens Valley's abundant and reliable water supply (from the surrounding mountains) encouraged establishment of agriculture beginning in the late 1800s, consisting of irrigated pasture and crop lands, including orchards. Water for agriculture was obtained by diversion of the Owens River, and by the early 20th century, Owens Lake had begun to decrease in size and volume due to this diversion. Agricultural activity peaked in the 1920s, followed by large-scale abandonment due to a reallocation of the water resources, through inter-basin transfer, for the agricultural, domestic, and industrial demands of Los Angles. Much of the abandoned agricultural land in the Valley was colonized by a mixture of perennial shrubs and annual grasses and plants. Water from Owens Valley, including the Mono Basin, makes up a significant fraction of the fresh water budget for Los Angeles, and all of the surface runoff has been exported from the valley since the 1920s. With the completion of a second aqueduct in 1968, the surface water export was supplemented by groundwater. With a diminished local supply of water, only a small fraction of the Owen Valley is cultivated today.

These transformations in water use and allocation have left a distinctive mark on the land cover of Owens Valley. The entire ecosystem downstream of the point where all surface water is diverted to fill the Los Angeles aqueduct has been transformed. The riparian and phreatophytic communities along the now dry Owens River have largely disappeared and the Owens Lake, once 280 km^2 in area, is now dry and constitutes the largest source of fine particulate aerosols (PM10) in the United States, posing significant health risk (Reheis and Kihl, 1995). The increased reliance on groundwater beginning in the 1960s caused many natural springs in the Valley to dry up, further reducing the amount of phreatophytic land cover (i.e., wetlands). Detailed studies (Elmore et al. 2003a; 2003b) of the resilience of the Owens Valley semi-arid ecosystems to the combined effects of a prolonged 6-year drought and the responses taken by resource managers show the following. (1) Phreatophytic communities are highly sensitive to depth to groundwater and show a threshold in response when water levels decrease below their rooting zone (3.3 m). Once this threshold is exceeded, the land is typically colonized by invasive shrubs and annuals, changing the ecosystem structure. (2) There is a legacy of land use. Abandoned agricultural land has lower species diversity and greater proportions of invasive shrubs and annuals, and this persists today nearly a century after abandonment.

The land-use and land-cover history of Owens Valley begins with an expansion of agricultural land use capitalizing on water resources. Agriculture contracted with the re-allocation of water resources for export from the region, outbid economically and politically by needs of Los Angeles. A period of relative stability in land cover and water abundance followed until additional demands were placed on the available water through groundwater extraction. The demands on water resources are now very close to the available supply, such that during periods of drought there is insufficient water for both natural and human needs. The net effect over the last hundred years has been the drying of Owens Lake, an expansion of invasive shrubs and annuals at the expense of native ecosystems, and a decline in wetlands. Periods of relatively stability have been punctuated by short periods of water stress in which demand exceeds supply This pulsed stress triggers important impacts(Elmore et al., 2003a).

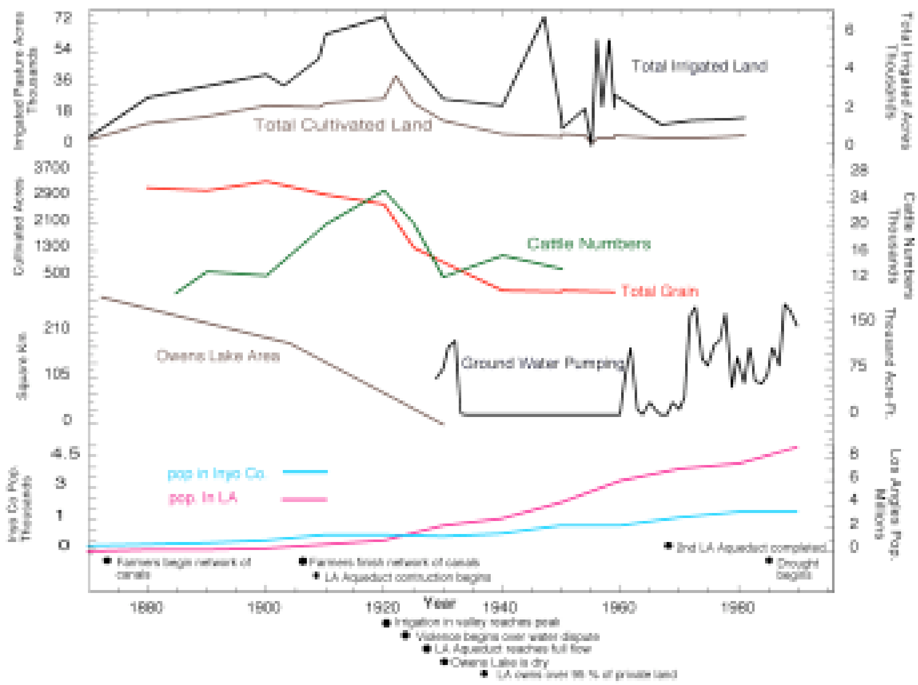

Figure 2. Land-use land-cover history of the Owens Valley CA, showing representative changes in land cover (irrigated and cultivated land area), effects of water policy (disappearance of Owens Lake, increase in the use of groundwater after completion of a second aqueduct), and major socio-economic events. (source)

2.3.1 *Owens Valley in the Context of Semi-Arid Land Use History*

There are parallels between this environmental history and other arid and semi-arid regions in the United States. For example, the Great Plains saw an expansion of population and land under cultivation in the early 1920s, followed by a collapse precipitated by the dust bowl of the 1930s and a contraction in the amount of land under cultivation (Worster 1979). Economic changes and government policies allowed for several periods of expansion and contraction over the last 50 years (Brooks and Emel 1999;Riebsame, 1990). While the specific processes and drivers differ from region to region, the common threads are anthropogenic land transformations driven by water re-allocation or access (e.g., irrigation, diversion, export)

3. Land-use land-cover change trajectories

Common LCLUC trajectories can be expected given an initial undisturbed state, three of which were detailed above. These trajectories involve four broad categories of land cover.

- Undisturbed: Landscapes dominated by "natural" cover types, where change is primarily by natural disturbance with little anthropogenic use (e.g. Amazonia in the 19th Century, New England in the 16th Century)(see note #1).

- Frontier: Landscapes experiencing transformations in "natural" cover, usually by extensive anthropogenic land uses (e.g., conversion to agriculture, forest re-growth through resource extraction) (e.g., Amazonia in the late 20th Century, New England in the 18th Century).

- Agricultural/Managed: Landscapes in which management matches or supercedes nature in function, such as rangelands or cultivated lands sustained by intensive inputs. Land-covers may be relatively stable, and changes in them are slow.

- Urbanized/Industrialized:Landscape dominated by residential,commercial, and industrial land cover, and highly managed vegetation for services and recreation (e.g., parks, sports fields, and managed "natural areas"), but few resources of the land are utilized.

Most of the world's lands can be categorized according to this broad framework, or some version of it, and significant portions have experienced one or more the transitions from the undisturbed state Where LCLUC histories are sufficiently long and well documented, it is possible to track a region's transformation between these broad categories (Figure 3). A typical, full progression first involves a concerted movement of humans into the undisturbed landscape, motivated by push and pull factors, including natural resource extraction (forested systems) or agricultural colonization, or both. Where appropriate climates and soils exist, conversion to a managed landscape occurs, typically through explorations under extensive uses, followed by a contraction in the amount of land actively managed, due to poor economics and low returns, to that most economically viable. The abandoned land is usually re-colonized by natural cover, though with a species composition and ecosystem structure that is different than the undisturbed system. Intensification (greater inputs of labor, fertilizer, and other amendments) of the remaining actively managed land is a typical effect during this period. In those conditions favoring the emergence of an industrial-urban economy, non-agricultural land uses typically outbid agricultural uses, and a new period of land cover fragmentation may be driven by urban expansion and suburbanization. This last phenomenon is perhaps more common in North America and Western Europe, but examples appear elsewhere, such as in the Pearl River delta of southern China.

This framework can be used to understand current conditions, past evolution, and future possibilities for land-cover change (Table 1). It is important, however, to clarify that not all areas have experienced or necessarily will experience the last two states noted and the time periods for any given period or transition is elastic. For example, logging in the boreal forest regions of Canada and Siberia, or in the mountainous regions of the Pacific Northwest of the United States are not activities meant to open up land for agriculture, but rather the logged lands are to be replanted or reforested for future harvest. Such regions may never become widely settled and/or urbanized, but will remain in an anthropogenically driven cycle of natural cover, deforestation, and regrowth. Likewise highly productive agricultural lands distant from densely populated regions and centers are unlikely to witness a transition to urbanization and suburbanization in the near future and may exist in a stable managed state for long periods. In much of the developing world, where rural populations have few options for food production besides extensive farming of marginal lands, abandonment and transition to more intensive agriculture is unlikely to occur without major changes in land tenure and economic conditions. Finally, all regions do not move unidirectionally through the four states. Southern Yucatán and much of Petén, Guatemala, for example, transitioned into the agricultural/managed state before A.D. 900, only to revert to tropical forest for a millennium before experiencing a frontier state today (e.g., Turner et al. 2001).

Table 1. Regional land cover/state conditions

Region	Frontier	Agricultural/Managed	Settled/ Industrial	Post-industrial
New England (US)	1650	1850	1940	2000
Western Europe	0 ?	1100-1900	1850-1950	2000
Great Plains (US)	1860	1900-present		
Rondonia (Brazil)	1960	2000		
Yucatán	0-200	900		
	1960	2000		
Siberia	2000			

The transformation to a largely industrialized-urbanized state may be an endpoint in landscape evolution. There are no examples of an urbanized landscape of the magnitude having reverted to any of the previous states. Any large scale de-urbanization would have to be accompanied by large reductions in human populations perhaps by relocation, war, famine, or economic collapse. While these may occur in the future, we have no examples of previously urbanized landscapes of the scale that exist today.

It is the transition between these generalized landscape conditions where the largest impacts of land use-land cover change are manifested. The specific forces (drivers) of change that precipitate these transitions may vary by region and surely do in terms of their relative roles. For example, the transition from an undisturbed to a frontier landscape could be motivated at a national level by population pressures in a distant managed/urban landscape or the desire to secure sovereignty over remote land. Other drivers include policies to subsidize an extractive economy or to motivate individuals to develop subsistence or market agriculture. Specific impacts from the land transformations are documented in the context of case studies or cross cutting themes elsewhere in this book and include very evident biotic, biogeochemical and physical changes in the landscape (e.g., hydrology, nutrients, erosion, biodiversity, biomass carbon) as well as changes in the resilience of systems to interannual and interdecadal climate variability.

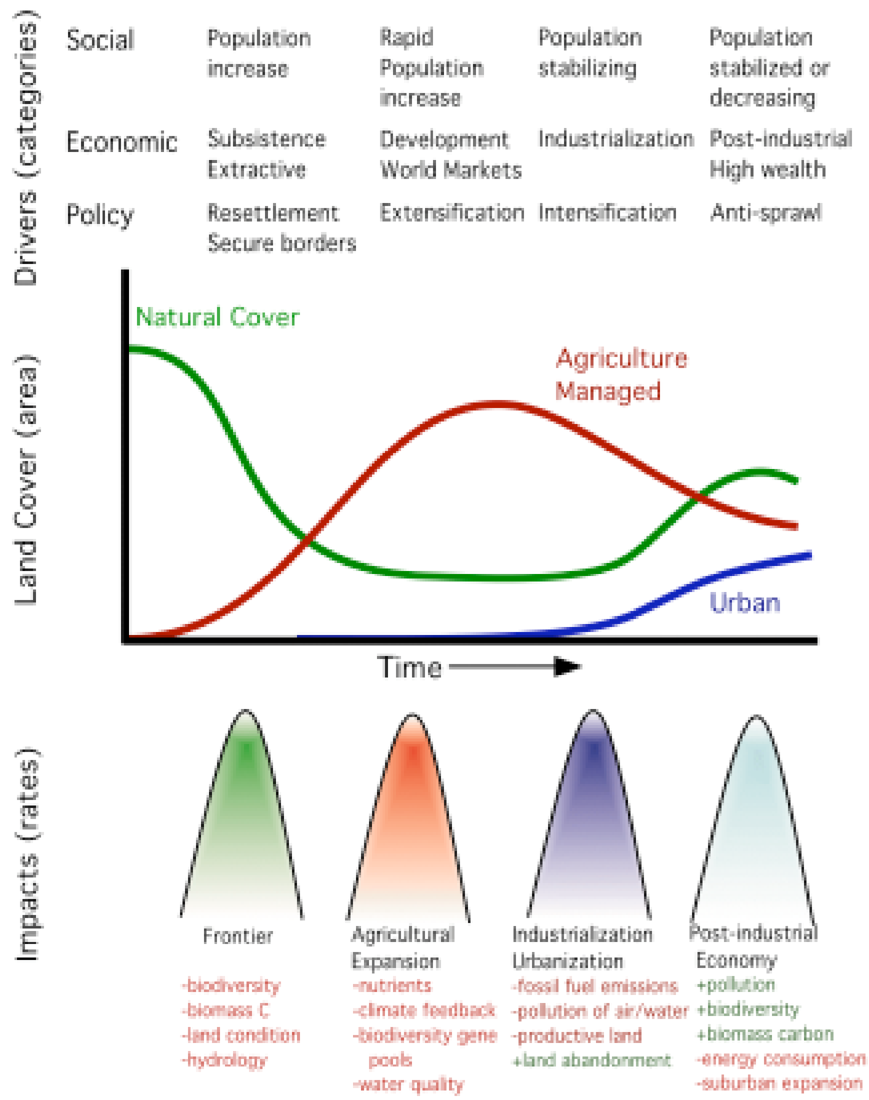

Figure 3. Synthesis of land cover trajectories, socio-economic drivers, and impacts of land-use land-cover change. Under impacts, the – sign indicates loss or declining quality, and the + indicates gain or improving quality.

4 Quantifying impacts

Terrestrial ecosystems provide many important goods and services on which human and other life depends, including regulation of climate, protection of watersheds, soil fertility, habitat to maintain diversity of plant and animal species, and cultural and aesthetic opportunities (Ayensu et al. 1999; Daily 1997; Daily et al. 1999). Primary among these goods and services is the provision of water, food and fiber. The vast majority of land-use change is associated with conversion of undisturbed landscapes to cropland, either for local consumption or export to market (Geist and Lambin 2002). Land-use change is essentially a trade-off between modifying terrestrial ecosystems for the positive benefit of providing food and fiber for human consumption and possible negative repercussions on other ecosystem services. These repercussions vary depending on the location and the state within the land-cover trajectory outlined in Figure 3. A focus of LCLUC research is to understand these impacts so that the trade-offs among ecosystem services can be quantified and assessed. The major impacts of LCLUC on ecosystem services are discussed below.

4.1 PROVISION OF FOOD AND FIBER

The predominant motive for land-use change is production of food and fiber. Although global food production is currently adequate to feed the world's population (Lappe et al. 1998), many throughout the world either do not have adequate food and/or rely on low-yielding, unproductive land for their subsistence. Other regions have a surplus which is exported or used to grow animals for meat production. Two significant trends indicate that land-cover change for food production will continue into this century: increasing population in most of the developing world where people do not have the means to purchase food to satisfy their requirements; and rising incomes associated with increased food consumption and diets richer in meat (Naylor 2000). The degree to which intensified agriculture with increasing yields can offset extensive agricultural expansion is a matter of debate (Tilman et al. 2001; Waggoner and Ausubel 2001), but is ultimately a major factor for determining the amount of land-cover change.

4.2 ALTERATION OF BIOGEOCHEMICAL CYCLES

Land-cover change plays an important role in the carbon cycle, which in turn regulates the concentration of the greenhouse gas carbon dioxide in the atmosphere (c.f. Houghton, et al., Chapter 14). Expansion into "frontier" landscapes generally results in extensive clearing of natural vegetation; consequent burning and enhanced soil respiriation results in release of carbon dioxide to the atmosphere. Model results estimate total carbon fluxes from human-induced land-cover change of 188-192 Pg globally, with approximately one-third occurring prior to 1850 (DeFries et al. 1999; Houghton 1999). Since the beginning of the industrial revolution, land-use change has contributed approximately one-third of the total carbon released to the atmosphere from human activities, with 250 Pg of carbon released from combustion of fossil fuels (Fung et al. 1997). In past centuries, the frontier landscapes were generally in temperate grasslands and forests, but in the late 20th Century the last remaining frontiers suitable for cultivation absent climate change are the vast expanses of tropical forests in Latin America, central Africa, and Southeast Asia (Sanderson et al. 2002). The high biomass of these tropical frontier forests is of particular significance for carbon fluxes, with tropical deforestation comprising a substantial portion of the contemporary global carbon budget (Prentice et al. 2001). Although the precise contribution of tropical deforestation and regrowth is a major uncertainty (Achard et al. 2002; DeFries et al. 2002), the transition from undisturbed to frontier landscapes is a significant factor in the human alteration of the global carbon cycle.

With a transition from "frontier" to the "managed" state of the land cover trajectory, higher-yield agriculture results from more intensive inputs of water and nutrients with profound impacts on nitrogen and phosphorus cycles. Use of synthetic nitrogen fertilizer, which supplements the natural processes that "fix" atmospheric nitrogen to biologically useful NH_3 and eventually to organic forms, has been one of the major factors responsible for increasing global food production and agricultural yields over the past several decades (Frink et al. 1999; Matson et al. 1997). Doubling of agricultural food production over the past 35 years was accompanied by a 7-fold increase in nitrogen fertilizer and a greater than 3-fold increase in phosphorus fertilization (Tilman et al. 2001). This anthropogenic alteration of the global nitrogen cycle has a number of repercussions, including the leakage of highly soluble nitrate (NO_3) from agricultural systems to cause eutrophication of surface waters, acidication of soil, groundwater pollution with nitrate, emissions of the greenhouse gas nitrous oxide to the atmosphere, and decrease of biodiversity as plants that favor a rich N supply displace other species. Release of phosphorus also results in eutrophication of freshwater streams and lakes. Regarding the carbon cycle, cropland abandonment in the "managed" state can sequester carbon from the atmosphere with regrowing forest (Caspersen et al. 2000).

4.3 ALTERED CLIMATE REGULATION THROUGH BIOPHYSICAL INTERACTIONS WITH THE ATMOSPHERE

Local, regional, and global climate are affected by land use and land cover through several types of interactions (c.f. Bonan et al., Chapter 17). The structure and density of vegetation influence the amount of absorbed incoming short-wave radiation (albedo) and the turbulent exchanges of momentum, heat, and moisture (surface roughness). Through the process of photosynthesis, plants transpire water vapor through their stomates and affect moisture fluxes to the atmosphere and consequently the balance between latent and sensible heat. Changes in vegetative cover can consequently alter surface fluxes of energy and water and modify surface climate.

Several modeling studies illustrate the sensitivity of climate to changes in vegetation. At the global scale, a simulation with extreme cases of unvegetated and vegetated land surfaces generated a two-fold difference in land precipitation and 8 K cooling in mean seasonal temperature with a vegetated relative to an unvegetated surface (Kleidon et al. 2000). In temperate and boreal regions, changes in vegetation may be responsible for a slight cooling owing to an increased albedo as brighter surfaces become exposed (Bonan 1997; Bonan 1999; Bounoua et al. 2002; Hansen et al. 1995). In the tropics, where forest clearing has predominantly occurred in the last few decades, the clearing likely leads to a warmer, drier climate (DeFries et al. 2002). Many model simulations of clearing the Amazon forest show increased temperatures and decreased precipitation (Nobre et al. 1991; Sud et al. 1996). Results of atmospheric general circulation models suggest that tropical deforestation may also influence climate through altered large-scale circulation patterns (Chase et al. 2000).

The feedbacks from land-cover change to climate through these biophysical mechanisms occur on spatial scales from local to regional, and possibly global through altered atmospheric circulation. The type of impact depends not only on the extent of the land cover change but also where it occurs. During the "frontier" stage in temperate latitudes, the predominant effect was to cool surface temperature from an increase in albedo with land cover clearing. In the current phase of frontier expansion in the tropics, the opposite is the case due to a large decrease in evapotranspiration associated with clearing of tropical forests (DeFries and Bounoua in press).

4.4 WATERSHED PROTECTION AND SOIL EROSION

Changes in land cover alter the water yield and discharge for watersheds at all spatial scales from 10's to 10,000's of km^2 (Sahin and Hall 1996). The canopy and root systems of vegetation affect a range of processes in the hydrologic cycle such as interception, percolation, surface retention, transpiration, and consequently surface and subsurface runoff and stream flow (Chang 2003). Rapid runoff, downstream flooding, soil erosion, and sedimentation are clear examples of local impacts of land cover change. With transformation from undisturbed to extensive agricultural expansion in the frontier stage, examples of these local impacts include cropland expansion in eastern North America accompanying European colonization (DeFries 1986) and current clearing in the Amazon Basin (Williams and Melack 1997). With a transition to more intensive production, these impacts would be lessened though nutrient exports would likely be enhanced (Mustard and Fisher, 2003). In the final urbanized stage, however, impervious surfaces will increase runoff, downstream flooding, and streambank erosion.

4.5 FRAGMENTATION OF THE LANDSCAPE AND HABITAT LOSS FOR BIODIVERSITY

Habitat loss is the single greatest threat to biodiversity and is likely to be more significant for biodiversity loss than climate change in this century (Sala et al. 2000). Biodiversity is fundamental to ecosystem services by providing a genetic library as the basis for modern agriculture, medicine, and industry (Myers 1997). A growing literature is also establishing the importance of biodiversity for maintaining healthy, stable, and functional ecosystems (Chapin et al. 2000), in addition to the intrinsic ethical concerns about human dominance over nature.

As landscapes move through the trajectory from undisturbed and eventually to urbanized/industrialized, nature reserves and protected areas are critical for maintaining biodiversity, particularly in "hotspots" of endemic species (Myers et al. 2000). Reserves are generally successful in controlling land-cover change within their boundaries (Bruner et al. 2001), although they may be influenced by adjacent disturbed areas, particularly by atmospheric and hydrologic interactions. Even with the presence of nature reserves, rapid expansion of cropland in landscapes in the second stage of the trajectory can affect biodiversity, for example by altering critical seasonal habitat for wildebeest in east Africa (Serneels and Lambin 2001). Land-cover change in the third or fourth stages of the trajectory can also affect biodiversity, for example the effects on bird populations from the construction of affluent rural homes in the Greater Yellowstone Ecosystem (Hansen and Rotella 2002). As landscapes move along the trajectory described in Figure 3, resources, mobility, and interest in recreation increase, on one hand generating the demand for preserving landscapes but on the other hand placing heavy demands on the landscape for recreational use.

4.6 CULTURAL AND AESTHETIC OPPORTUNITIES

Land-cover change profoundly affects the aesthetic and cultural value associated with landscapes of all kinds. In the early stages, the cultural value largely derives from direct dependence on the ecosystem services (Gadgil and Guha 1992). In the latter stages [states], society values and has resources to invest in recreational and aesthetic opportunities.

The above discussion illustrates that the nature of the impacts of land-cover change and the spatial and temporal scale over which they occur depend largely on the stage within the general trajectory described in this paper. As landscapes move through the trajectory from the "undisturbed" to the "frontier" category, the extensive clearing provides food and fiber mainly for local consumption. The clearing, however, has global and regional repercussions by releasing carbon previously stored in the vegetation to the atmosphere, altering climatic patterns, and reducing biodiversity through habitat loss. More locally, the clearing can generate soil erosion and increase runoff from reduced vegetation in the watershed. With agricultural intensification, the biogeochemical cycles associated with nitrogen and phosphorus are affected to a greater degree, and a decrease in cropland area can sequester carbon from the atmosphere and benefit biodiversity. In the final urbanized/industrial state, the impacts are displaced in space as resources to support the population are obtained from afar, a spatial disjuncture that has proven difficult to incorporate into models.

5 Conclusions

Do LCLUC studies reveal broad commonalities in trajectories and impacts of land change? We conclude in the affirmative, and make the case for four general land cover/use conditions or states: Undisturbed, Frontier, Agricultural/Managed and Industrial/Urban. Many landscapes transition through these four states, though the timelines are elastic and there is no expectation that a given region is fated to experience all conditions. Furthermore the timeline is not unidirectional and through processes like abandonment land cover may revert from managed to undisturbed given enough time. The most profound impacts on land cover occur during transitions between conditions. This broad framework nevertheless masks many important details and its applicability to particular locations requires further investigation. This is particularly true with regards to the socio-economic drivers as the study of the linkages between land use drivers, biological and physical impacts, and feedbacks to land use decisions is yet in its infancy. Better understanding of these linkages, and the consequences for ecosystem services, will provide a basis for rational decisions about land use change.

An ultimate goal of the LCLUC program is to affect policy and the framework presented here represents a beginning model for decision makers. For example, once the condition of a landscape is assessed, the pathways and attendant impacts can be linked to policy choices, recognizing the abundant uncertainties involved . Through the specific examples of the case studies and the cross-cutting themes that emerge from these studies, the land-change research community is honing the ability to articulate options and their outcomes.

Scale becomes an extremely important issue for quantifying impacts. Impacts that can be identified and characterized at a global or regional scale have had a great effect in framing questions and pointing to the magnitude of some problems (e.g., deforestation, land degradation, drought). Nevertheless, detailed characterization and quantification of impacts have generally relied on higher resolution observations typically at the scale of one ha or less. Impacts can be divided into those that affect the local environment (e.g., water quality) and those that extend far beyond the local environment (e.g., carbon, climate). This dichotomy of scale clearly hampers the development of an integrated understanding of LCLUC processes and impacts across space and time. In the NASA LCLUC program, many of the analyses have been at the spatial scale of Landsat Thematic Mapper for the central reason that this resolution is a good compromise between high frequency, low-spatial resolution global sensors and high-spatial resolution and large data volume but low temporal resolution sensors. While this TM-based perspective (space and time) has clearly led to important advances in identifying LCLUC pathways and impacts, there is a critical intermediate scale, the regional view, that needs to be addressed. The advent of new high spatial resolution sensors and more frequent observations coupled with expanding capacity to analyze data will likely lead to a better merging of local and global approaches in the future.

It is important to asses what impacts of LCLUC can be quantified? When considering the range of case studies, it seems clear that in regions with rapid and distinct changes in land cover (e.g., forest to cleared/agriculture, agriculture to urban), including rates, patterns, and trajectories, can be quantified by current approaches. Changes in the biophysical properties of the surface (e.g., live cover in semi-arid regions, woody vegetation encroachment) can also be quantified with some measure of success. Impacts of intensification (e.g., water quality) and changes in some land use, as well as land cover are possible though this has not been widely demonstrated. Some critical measures of landscape health will not be amenable to analysis with remotely sensed data (Chapter 28). For these situations, and for incorporating socio-economic data, LCLUC analysis will have to rely on in situ data and models parameterized by empirical relationships instead of direct parameterization.

References

Angelsen, A., and Kaimowitz, D. (1999). "Rethinking the Causes of Deforestation: Lessons from Economic Models." *The World Bank Research Observer*, 14, 73-98.

Archard, F., H. Eva, et al., 2002. Determination of deforestation rates of the world's humid tropical forests, *Science* 297:999-1002.

Ayensu, E., D. V. R. Claasen, et al., 1999. International ecosystem assessment, Science 286(5440): 685-686.

Batistella,M. 2001. Landscape Fragmentation and Land Cover Dynamics in Rondonia, Brazilian Amazon. Ph.D. Dissertation. Indiana University: School of Public and Environmental Affairs. 367p.

Beckerman, S. 1994 Hunting and Fishing in Amazonia: Hold the Answers, What are the Questions? In *Amazonian Indians from Prehistory to the Present. Anthropological Perspectives*. Anna Roosevelt, ed. Univ. of Arizona Press.

Bonan, G. B., 1997. Effects of land use on climate of the United States, Climatic Change 37: 449-486.

Bonan, G. B., 1999. Frost followed the plow: Impacts off deforestation on the climate of the United States, Ecological Applications 9(4): 1305-1315.

Bounoua, L., R. DeFries, et al., 2002. Effects of land cover conversion on surface climate, Climatic Change 52: 29-64.
Brondízio, E.S., S. McCracken, et al., 2002. The Colonist Footprint:Toward a Conceptual Framework of Land Use and Deforestation Trajectories among Small Farmers in the Amazonian Frontier. Pp. 133-161 in *Deforestation and Land Use in the Amazon*. C.H. Wood and R. Porro, eds. Gainesville: University Press of Florida.
Brooks, E., and Emel, J. (1999). *The Llano Estacado of the US Southern Hign Plains: The Rise and Decline of a Modern Irrigation Culture*, United Nations University Press, Tokyo, JP.
Bruner, A. G., R. E. Gullison, et al., 2001. Effectiveness of Parks in Protecting Tropical Biodiversity, *Science* **291**(5501): 125-128.
Caspersen, J. P., S. W. Pacala, et al., 2000. Contributions of land-use history to carbon accumulation in U.S. forests, *Science* **290**: 1148-1151.
Chang, M., 2003. Forest Hydrology: An Introduction to Water and Forests, Washington, D. C., CRC Press.
Chapin, F. S. I., E. S. Zavaleta, et al., 2000. Consequences of changing biodiversity, *Nature* **405**(234-242).
Chase, T. N., R. A. S. Pielke, et al., 2000. Simulated impacts of historical land cover changes on global climate in northern winter, *Climate Dynamics* **16**: 93-105.
Cochrane, M.A. 2000. Using Vegetation Reflectance Variablitiy for Species Level Classification of Hyperspectral Data. *International Journal of Remote Sensing* 21: 2075-2087.
Cochrane, M.A. 2001a. Synergistic interactions between habitat fragmentation and Fire inEvergreen Tropical Forests. *Conservation Biology* 15(6):1515-1521.
Cochrane, M.A. 2001b. In the line of fire: Understanding the impacts of tropical forest fires. *Environment* 43(8):28-38.
Daily, G. E., Ed., 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Washington, D. C., Island Press.
Daily, G. C., T. Söderqvist, et al., (2000). “The value of nature and nature of value.” *Science*, 289, 395-396.
DeFries, R., 1986. Effects of land use history on sedimentation in the Potomac Estuary, Maryland. Denver, Co., *U. S. Geological Survey water-supply paper 2234-K.*
DeFries, R. and L. Bounoua, in press 2003. Consequences of land use change for ecosystem services: A future unlike the past, *Geojournal.*
DeFries, R., C. Field, et al., 1999. Combining satellite data and biogeochemical models to estimate global effects of human-induced land cover change on carbon emissions and primary productivity, *Global Biogeochemical Cycles* **13**(3): 803-815.
DeFries, R., R. A. Houghton, et al., 2002. Carbon emissions from tropical deforestation and regrowth based on satellite observations for the 1980s and 90s, *Proceedings of the National Academy of Sciences,* **99**(22): 14256-14261.
DeFries, R., L. Bounoua, et al., 2002. Human modification of the landscape and surface climate in the next fifty years, *Global Change Biology* **8**: 438-458.
Denevan, W. M. (2001). *Cultiated Landscapes of Native Amazoni and the Andes*, Oxford University Press, Oxford, UK.
Doolittle, W. E. (2000). *Cultivated Landscapes of Native North America*, Oxford University Press, Oxford, UK.
Drzyzga, S. A. and D. G. Brown, 2002. Spatial and temporal dynamics of ownership parcels and forest cover in three counties of Northern Lower Michigan USA, ca. 1970 to 1990. In S. J. Walsh and K. A. Crews-Meyer, Eds., Remote Sensing and GIS Applications for Linking People, Place, and Policy, Dordrecht: Kluwer, p. 155-185.
Elmore, A. J., J. F. Mustard, et al., 2003a. Regional patterns of great basin community response to changes in water resources: Owens Valley, California, (*in press*) *Ecological Applications,* 13(2).
Elmore, A. J., J. F. Mustard, et al., submitted 2003b, Agricultural legacies and their impacts on ecosystem structure and the stability of green cover in the Great Basin: Owens Valley, California. *Ecosystems*.
Fearnside, P.M. 1986. *Human Carrying Capacity of the Brazilian Rainforest*. New York: Columbia University Press.
Foster, D. R., G. Motzkin, et al., 1998. Land-use history as long-term broad-scale disturbance: Regional forest dynamics in central New England. *Ecosystems* **1**: 96-119.
Frink, C. R., P. E. Waggoner, et al., 1999. Nitrogen fertilizer: Retrospect and prospect, *Proceedings of the National Academy of Sciences* **96**: 1175-1180.
Fung, I., C. B. Field, et al., 1997. Carbon 13 exchanges between the atmosphere and biosphere, *Global Biogeochemical Cycles* **11**(4): 507-533.
Gadgil, M., and R. Guha, 1992. This Fissured Land: An Ecological History of India, Berkeley, CA, University of California Press.

Geist, H. J. and E. F. Lambin, 2002. Proximate causes and underlying forces of tropical deforestation, *BioScience* **52**(2): 143-150.
Geist, H. J. and E. F. Lambin, 2001. What drives tropical deforestation? A meta-analysis of proximate and underlying causes of deforestation based on subnational scale case study evidence. *LUCC Report Series No. 4.*, University of Louvain, Louvain-la-Neuve: pp. 116.
Hall, B., G. Motzkin, et al., 2002. Three hundred years of forest and land-use in Massachusetts, USA. *Journal of Biogeography* **29**: 1319-1335.
Hansen, A. J. and J. J. Rotella, 2002. Biophysical factors, land use, and species viability in and around nature reserves, *Conservation Biology* **16**(4): 1-12.
Hansen, A. J. and M. Sato, et al., 1995. Climate forcings in the Industrial era, *Proceedings of the National Academy of Sciences* **95**: 12753-12758.
Houghton, R. A., 1999, The annual net flux of carbon to the atmosphere from changes in land use 1850-1990, *Tellus* **Ser. B, 51B**: 298-313.
Kleidon, A., K. Fraedrich, et al., 2000. A green planet versus a desert world: Estimating the maximum effect of vegetation on the land surface climate, *Climatic Change* **44**: 471-493.
Lambin, E. F., et al., 2001. The causes of land-use and land-cover change: moving beyond the myths. *Global Environmental Change* **11**: 261-269.
Lappe, F. M., J. Collins, et al., 1998. World Hunger: Twelve Myths, New York, Grove Press.
Laurance, W. F., M. A. Cochrane, et al., 2001. The future of the Brazilian Amazon. *Science* **291**:438-439.
Lu, D.S. 2001. Estimation of forest stand parameters and application in classification and change detection of forest cover types in the Brazilian Amazon Basin. Ph.D. Dissertation. Indiana State University, Terre Haute, Indiana.
Matson, P. A., W. J. Parton, et al., 1997. “Agricultural Intensification and Ecosystem Properties.” *Science*, 280, 504-508.
McCracken, S.D., A.D. Siqueira, et al., 2002 Land Use Patterns on an Agricultural Frontier: Insights and Examples from a Demographic Perspective. Pp. 162-192 in *Deforestation and Land Use in the Amazon*. C. Wood and R. Porro, eds. Gainesville: University Press of Florida.
Meggers, B.J. 1971. *Amazonia: Man and Culture in a Counterfeit Paradise*. Chicago: Aldine, Atherton.
Mesquita, R. C. G., K. Ickes, et al., 2001. Alternative successional pathways in the Amazon Basin. Journal of Ecology **89**:528–537.
Moran, E.F. 1976. *Agricultural Development in the Transamazon Highway*. Bloomington: Indiana University Press.
Moran, E.F. 1981. *Developing the Amazon*. Bloomington: Indiana University Press.
Moran, E.F. 1982. Colonization in the Transamazon and Rondônia In *Frontier Expansion in Amazônia*. M. Schmink and C. Wood (eds.). Gainesville: University of Florida Press. Pp. 285-303.
Moran, E.F., E. Brondizio, P. Mausel and Y. Wu. 1994. Integrating Amazonian Vegetation, Land Use and Satellite Data. *BioScience* 44(5):329-338.
Moran, E.F., A. Packer, E. Brondizio, J. Tucker. 1996. Restoration of Vegetation Cover in the Eastern Amazon. *Ecological Economics* 18:41-54.
Moran, E.F. and E.S. Brondizio. 2001. Human Ecology from Space: Ecological Anthropology Engages the Study of Global Environmental Change. In *Ecology and the Sacred: Engaging the Anthropology of Roy A. Rappaport*. E. Messer and M. Lambeck (eds.). Ann Arbor: University of Michigan Press. Pp. 64-87
Moran, E.F., E.S. Brondízio, and S. McCracken. 2002. “Trajectories of land Use: Soils Succession, and Crop Choice.” Pp. 193-217 in *Deforestation and Land Use in the Amazon.* CH Wood and R Porro, eds. Gainesville: University Press of Florida.
Mullholland, M. T., 1988. Territoriality and horticulture, a perspective for prehistoric New England. In: G. P. Nichols, editor, Holocene Human Ecology in Northeastern North America. New York, Academic: 137-166.
Myers, N. 1997. Biodiversity's genetic library. Nature's Services: Societal Dependence on Natural Ecosystems, G. C. Daily, editor, Washington, D. C., Island Press, 255-274.
Myers, N., R. A. Mittermeier, et al., 2000. Biodiversity hotspots for conservation priorities, *Nature* **403**: 853-857.
Naylor, R. 2000. Agriculture and global change. in Earth Systems: Processes and Issues, edited by W.G. Ernst. Cambridge University Press. Cambridge, UK. pp. 462-475.
Nepstad, D., A. Moreiera, A. Alencar. 1999. *Flames in the Rainforest: Origins, Impacts, and Alternatives to Amazonian Fire*. IBAMA. Brasilia, D.F.
Nepstad, D., A. Verissimo, et al., 2000. O empobrecimento oculto da floresta Amzonica. (The Hidden impoverishment of Amazon forests). *Ciencia Hoje* **27**(157):70-73.
Nobre, C. A., P. J. Sellers, et al., 1991. Amazonian deforestation and regional climate change, *Journal of Climate* **4**: 957-987.

Orwig, D. A., D. R. Foster, et al., 2002. Landscape patterns of hemlock decline in New England due to the introduced hemlock woolly adelgid. *Journal of Biogeography* **29**:1475-1530.

Paillet, F., Chestnut: history and ecology of a transformed species. *Journal of Biogeography* **29**:1517-1530.

Putman, F. and G. Smith, Deepest Valley, 1995. University of Nevada Press, Reno NV: pp. 280.

Prentice, I. C., G. D. Farquhar, et al., 2001. Chapter 3: The Carbon Cycle and Atmospheric Carbon Dioxide. Climate Change 2001: The Scientific Basis. J. T. Houghton et al., editors, Cambridge, UK, Cambridge University Press.

Reheis, M. C., and R. Kihl, 1995. Dust deposition in southern Nevada and California, 1984-1989: Relations to climate, source area, and source lithology: *Journal of Geophysical Research*, **v. 100**, no. D5, p. 8893-8918.

Rojstaczer, S., Sterling, S. M., and Moore, N. J. (2001). “Human Appropriation of Photosynthesis Products.” *Science*, 294, 2549-2552.

Roosevelt, Anna. 1989. Resource management in Amazonia before the conquest: beyond ethnographic projection. *Advances in Economic Botany* 7: 30-62.

Sahin, V., and M. J. Hall, 1996. The effects of afforestation and deforestation on water yields, *Journal of Hydrology* **178:** 293-309.

Sala, O. E., F. S. I. Chapin, et al., 2000. Global biodiversity scenarios for the year 2100, *Science* **287**: 1770-1774.

Sanderson, E. W., M. Jaiteh, et al., 2002. The human footprint and the last of the wild, *BioScience* **52**(10): 891-904.

Serneels, S. and E. Lambin, 2001. Impact of land-use changes on the wildebeest migration in the northern part of the Serengeti-Mara Ecosystem, *Journal of Bigeography* **28**: 391-408.

Skole, D., and C. Tucker. 1993. “Tropical Deforestation and Habitat Fragmentation in the Amazon: Satellite Data from 1978-1988.” *Science* 260 (5116): 1905-1910.

Smith , N. 1982. *Rainforest Corridors*. Berkley: University of California Press.

Steininger, M. K. 1996. Tropical secondary forest regrowth in the Amazon: age, area and change estimation with Thematic Mapper data. *International Journal of Remote Sensing.* 17 (1): 9-27.

Sud, Y. C., G. K. Walker, et al., 1996. Biogeophysical consequences of a tropical deforestation scenario: A GCM simulation study, *Journal of Climate* **9**: 3225-3247.

Tilman, D., J. Fargione, et al., 2001. Forecasting agriculturally driven global environmental change, *Science* **292**: 281-284.

Turner, B. L., 1990. The rise and fall of Maya population and agriculture 1000 B.C. to present: the Malthusian perspective reconsidered. In: Newman, L. (ed.) Hunger and History: Food Shortages, Poverty and Deprivation. Basil Blackwell, Oxford: 178-211.

Turner, B. L., W. C. Clark, et al., 1990. The Earth as Transformed by Human Action: Global and Regional Changes in the Biosphere over the Past 300 Years, Cambridge Univ. Press, Cambridge: pp. 713.

Turner II, B. L., and Butzer, K. W. (1992). “The Columbian Encounter and Land Use Change.” *Environment*, 34(8), 16-44.

Turner, B. L., S. C. Villar, et al., 2001. Deforestation in the southern Yucatán peninsular region: an integrative approach. *Forest Ecology and Management* **154**: 353-370.

Turner, B. L., II. (2002a). “Toward Integrated Land-Change Science: Advances in 1.5 Decades of Sustained International Research on Land-Use and Land-Cover Change.” *Challenges of a Changing Earth: Proceedings of the Global Change Open Science Conference, Amsterdam, NL, 10-13 July 200*, W. Steffan, J. Jäger, D. Carson, and C. Bradshaw, eds., Springer-Verlag, Heidelberg, GR., 21-26.

Twery, M. J. and W. A. Patterson III, 1984. Variations in beech bark disease and its effects on species composition and structure of northern hardwood stands in central New England. *Canadian Journal of Forest Research* **14**: 565-574.

Vitousek, P. M., H. A. Mooney, et al., 1997. Human domination of earth’s ecosystems. *Science* **277**: 494-499.

Waggoner, P. E. and J. H. Ausubel, 2001. How much will feeding more and wealthier people encroach on forests? *Population and Development Review* **27**(2): 239-257.

Walker, R., E. Moran, and L. Anselin 2000. Deforestation and Cattle Ranching in the Brazilian amazon: External Capital and Household Processes. World Development 28 (4): 683-699

Williams, M. R. and J. M. Melack, 1997. Solute export from forested and partially deforested catchments in the central Amazon, *Biogeochemistry* **38**: 67-102.

Wood, C. and D. Skole 1998. Linking Satellite, Census, and Survey Data to Study Deforesation in the Brazilian Amazon. In People and Pixels: Linking Remote Sensing and Social Sciences. Edited by D. Liverman, E. Moran, R. Rindfuss, and P. Stern. Washington DC: National Academy Press. Pp. 70-93

Worster, D. (1979). *Dust Bowl: The Southern Great Plains in the 1930s*, Oxford Univesrity Press, Oxford, UK.]

Zarin, D.J. ed. 2001. New Directions in Tropical Forest Research. Forest Ecology and Management (special issue). Vol 154(3).

Zarin, D.J., G. Huijun and L. Enu-Kwesi. 2002. Guidelines on the assessment of plant species diversity in agricultural landscapes. Pp. 57-69 in H. Brookfield, C. Padoch, H. Parson and M. Stocking, eds. Cultivating Biodiversity: The Understanding, Analysis and Use of Agrodiversity. ITDG Publications, London.

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[1] The terms natural and undisturbed are used throughout this chapter in reference to landscapes and environments which are only ephemerally managed and used, even if they were significantly altered by human action in the past.