Annu. Rev. Anthropol. 2000. 29:493–524

CONSERVATION AND SUBSISTENCE IN SMALL-SCALE SOCIETIES

Eric Alden Smith
Department of Anthropology, University of Washington, Box 353100, Seattle, Washington 98195-3100; e-mail: easmith@u.washington.edu

Mark Wishnie
School of Forestry and Environmental Studies, Yale University, New Haven, Connecticut 06511; e-mail: mark.wishnie@yale.edu

Key Words collective action, sustainability, common property resources, biodiversity

■ **Abstract** Some scholars have championed the view that small-scale societies are conservers or even creators of biodiversity. Others have argued that human populations have always modified their environments, often in ways that enhance short-term gains at the expense of environmental stability and biodiversity conservation. Recent ethnographic studies as well as theory from several disciplines allow a less polarized assessment. We review this body of data and theory and assess various predictions regarding sustainable environmental utilization. The meaning of the term conservation is itself controversial. We propose that to qualify as conservation, any action or practice must not only prevent or mitigate resource overharvesting or environmental damage, it must also be designed to do so. The conditions under which conservation will be adaptive are stringent, involving temporal discounting, economic demand, information feedback, and collective action. Theory thus predicts, and evidence suggests, that voluntary conservation is rare. However, sustainable use and management of resources and habitats by small-scale societies is widespread and may often indirectly result in biodiversity preservation or even enhancement via creation of habitat mosaics.

INTRODUCTION

Beginning several decades ago, the idea that indigenous peoples and other small-scale societies were exemplary conservationists gained widespread currency in popular media as well as academic circles.[1] This indigenous conservationism has

---

[1] By "small-scale society" we mean one that maintains political autonomy at the level of one or a few local communities and, hence, numbers a few hundred to a few thousand inhabitants (Bodley 1996:12). There is little consensus on the meaning of "indigenous." We do not attempt to define it, but this has little bearing on the arguments we review here.

0084-6570/00/1015-0493$14.00

often been attributed to a spiritual respect for, and a practical understanding of, the natural world (e.g. Vecsey 1980, Martinez 1996, Berkes 1999). Evidence offered in support of this characterization includes culturally expressed conservation ethics, animistic religious beliefs that conceptualize other species as social beings, the high levels of biodiversity found in the homelands of these peoples, and the impressive environmental knowledge they possess (Nelson 1982, Durning 1992, Posey 1992, Gadgil et al 1993, Callicott 1994, Alcorn 1996, Bodley 1996, Nabhan 1997).

Although this view remains widely accepted, it is increasingly questioned in both popular (e.g. Diamond 1992, Redford 1991) and scholarly literatures. Much of the evidence cited by these critics is archaeological (Stahl 1996, Redman 1999) or ethnohistoric (Kay 1994, Krech 1999), but some is based on contemporary field studies by ethnographers or biologists (e.g. Stearman 1994, Alvard 1998b, FitzGibbon 1998). Specific examples of nonconserving behavior by small-scale societies include anthropogenic faunal extinctions and habitat degradation, as well as patterns of subsistence behavior that seem to conform to economic optimization rather than to resource or habitat conservation. In addition, some anthropologists have questioned the causal efficacy (Hames 1987, Low 1996), or even the indigenous existence (Bulmer 1982, Brightman 1987, Carrier 1987, Krech 1999), of conservation ethics.

This debate has become politically charged because those working on behalf of indigenous rights sometimes view critiques of indigenous conservation as attacks on their efforts to defend local resource control by indigenous groups (Alcorn 1995, Posey 1998, Sponsel 1992). Critics sometimes respond by defending indigenous land rights, while arguing that those rights will ultimately be undercut by naive or mistaken views on indigenous conservation (Hames 1991, Stearman 1994, Vickers 1994, Hill 1996). For these and other reasons, the literature on conservation in small-scale societies is often polarized into dichotomous viewpoints, with such peoples portrayed either as “guardians of the earth” (Durning 1992) living in harmony with pristine environments until invaded by an outside destructive force (agriculture, industrialization, capitalism), or as “primitive polluters” (Rambo 1985) with a record of environmental destruction stretching back millennia to the “Pleistocene overkill” (Martin 1973) with which the first peoples of America (and recently those of Australia) are sometimes charged.

With the accumulation of considerable empirical evidence on the topic, plus the development of a coherent body of theory that can guide the collection and analysis of such evidence, the opportunity now exists to present a more balanced and dispassionate evaluation of indigenous conservation, moving beyond such pejorative caricatures as “the ecologically noble savage” (Redford 1991). In this spirit, Krech (1999) and Redman (1999) have recently reviewed the archaeological and historic/ethnohistoric evidence. Here we concentrate on contemporary field studies, viewed in light of relevant theory from a number of disciplines.

The remainder of the paper is organized as follows. We begin with an outline of the ecological and evolutionary context in which environmental relations of small-scale societies take place, which include evolved features favoring dispersal abilities and colonization of new habitats, and what some scholars have called “ecological engineering.” We then summarize theory concerning environmental utilization and conservation from a number of disciplines, emphasizing such issues as sustainability, temporal discounting, collective action, and common property resource management. These contextual and theoretical sections provide the foundation for a review of the arguments and ethnographic evidence bearing on conservation in small-scale societies.

THE ECOLOGICAL AND EVOLUTIONARY CONTEXT

Although this review is concerned with ethnographic (as opposed to archaeological or ethnohistoric) instances of conservation (or lack thereof), it is instructive to place this phenomenon in a deeper historical/evolutionary perspective. It is equally important to look beyond our species to a broader ecological context, and to consider briefly how contemporary ecologists view issues of environmental change and perturbation.

Ecology Out of Equilibrium

For much of this century, the study of ecological communities and ecosystems was dominated by a set of concepts and theories emphasizing orderly succession to equilibrium (otherwise known as “climax” or “maturity”) (Smith 1984, Winterhalder 1994).[2] Beginning in the 1970s, this equilibrium-centered paradigm was increasingly questioned by ecologists impressed by the pervasiveness of dynamic processes involving perturbations, disturbance, unpredictability, complex nonlinear interactions, and the influence of history (or initial conditions) (Pickett & White 1985, DeAngelis & Waterhouse 1987, Botkin 1990, Perry & Amaranthus 1997). Currently, the nonequilibrium or dynamic paradigm is clearly ascendant in academic ecology circles, if not always in environmental management and social science (Scoones 1999).

This revised conception of nature has important implications for conservation biology, and for our view of humans in nature (Botkin 1990, Worster 1995). First, because it is now generally accepted that “species diversity is generally enhanced by disturbances that occur at intermediate levels of frequency and intensity” (Perry & Amaranthus 1997:33; but see Hubbell et al 1999), it seems likely that anthropogenic disturbances at such intermediate levels (such as swidden cultivation

---

[2] By “ecological community” we mean simply a set of organisms that coexist in a given area, without any necessary implication of temporal or spatial integrity. “Ecosystem” simply adds abiotic elements to a community defined in this sense.

at low population densities, or controlled burning to encourage herbaceous plant growth) can have biodiversity-enhancing effects.[3] Second, if ecological communities are always in flux, it is not clear what particular state or set of states anyone should attempt to conserve. As fire ecologist Stephen J. Pyne (1999:2) has recently argued,

> What advocates typically want restored is "natural" fire, but the historic landscape that supplies their vision of what restoration should produce was itself the outcome of thousands of years of culturally set burning. The charge to restore commits us to a state of permanent irony. We will never decide to what point in the past we should restore the land—and can never, in any event, actually get back there.

In fact, there is a double irony here because habitats seen as "natural" may in fact be only temporary states on a dynamic continuum, and the very places often celebrated as pristine nature—rich grasslands, for example, or a forest dominated by large, well-spaced trees—may be particularly dependent on a history of anthropogenic disturbance (Anderson 1990, Gomez-Pompa & Kaus 1992).

Human Dispersal and Ecosystem Engineering

The belief that indigenous peoples are unchanging, "frozen in time," living in the same spot since time immemorial, is widespread in popular culture. Anthropologists have long critiqued the notion that small-scale societies consist of people "without history" (Wolf 1982). But extension of this understanding to a recognition that such peoples are shapers of environmental history has been slower to come. Such recognition is now developing rapidly, and is changing our views of human ecology, adaptation, and conservation (e.g. Sponsel 1992, Crumley 1994, Kirch & Hunt 1997, Russell 1997, Balée 1998).

To be sure, many indigenous peoples have deep roots in particular locales, and virtually all have strong attachments to homelands. But this does not necessarily mean that they have continuously occupied a single spot on the planet and made it home for hundreds of generations. A case in point is provided by the first peoples of North America. An examination of the distribution of Native North American languages (a proxy for cultural ancestry) clearly shows that indigenous Americans engaged in extensive population movements that postdate the peopling of the Americas but that antedate the European invasion (Goddard 1996). Patterns such

---

[3] Biodiversity has a variety of meanings in the conservation biology literature (Meffe & Carroll 1997). Most commonly, it refers to species richness (number of species in a given area). This, in turn, can be divided into species richness within a given habitat type ("beta diversity") and that across habitats ("gamma diversity"). Habitat disturbance can increase gamma diversity, and in the long run it can preserve higher levels of overall biodiversity by allowing species to persist in early successional habitats that would otherwise perish from competition with late-successional species.

as these suggest that waves of dispersal or repeated immigration can lead to a patchwork of languages and cultural-ecological diversity.

The archaeological and historical records also reveal many episodes of replacement of one population (or at least lifeway) by another. As one example, the contemporary Inuit inhabitants of arctic Canada and Greenland are renowned for their ingenious adaptation to a challenging environment of low biological productivity (Damas 1984). Yet their immediate ancestors, the Thule peoples, arrived in the region (from North Alaska) only some six to nine centuries before Europeans first entered it. And the Thule colonization replaced the Dorset culture that had occupied the North American arctic for roughly two millennia—a culture that, in turn, had replaced the pre-Dorset paleoeskimos who first colonized the high arctic (McGhee 1984).

The available evidence indicates that everywhere on the planet, some version of this colonization/replacement scenario has played itself out at least once, if not repeatedly. Indeed, this basic pattern—waves of colonization spurred by technological and social innovations or environmental fluctuations, and consequent replacement of one lifeway by a competitively superior interloper—is now accepted by most specialists in modern human origins as the charter story of our species (Klein 2000).

Although there is little consensus on its causes, a plausible explanation for the evolution of this colonizing strategy in our lineage pinpoints ecological instability as a root cause (Potts 1997). The period in which the genus *Homo* emerged (the Pleistocene, lasting from approximately 1.7 million to 10,000 years ago) was marked by the greatest climatic instability of the past 65 million years, involving massive changes in sea level, ice movements, climatic fluctuations, and resultant habitat displacements (e.g. growth and shrinkage of forests) (reviewed in Potts 1997). Our hominid ancestors seem to have responded to such large-scale environmental instability by evolving capabilities that enhanced their ability to survive and reproduce under an increasingly wide variety of environmental conditions.

Put another way, one might say that our species has specialized in being an ecological generalist, consuming an amazing variety of foods, inhabiting an unprecedented range of habitats and climates, and developing a wide range of very flexible and opportunistic niches (Gamble 1994). We have developed these abilities not because they are intrinsically adaptive—an anthropocentric view that leaves mysterious the evolution and persistence of myriad relatively specialized and unchanging species—but because in our particular ecological context they conferred evolutionary advantage. In sum, ecological instability may well be the crucible from which our species emerged and the matrix in which we have thrived on the whole.

Ecological Engineering The secret of our species' readiness to colonize diverse habitats lies in the human ability to repeatedly and substantially alter our means of subsistence, in many cases literally remaking our habitats in the process. Even prior to agriculture, humans began altering ecosystems (e.g. through periodic burning) so as to capture a larger proportion of total biotic energy. This gambit has been so successful that it has produced astounding growth in the human population, and an unprecedented degree of ecological dominance. As an example of the latter, several recent analyses estimate that between 40% and 50% of the total food energy flowing through the biosphere is channeled into human sustenance or other consumptive uses (Vitousek et al 1997).

Although there are several means by which humans achieve ecological dominance, one involves diverse practices that have in common the physical reshaping of earth forms, waterways, plant and animal communities, and other aspects of ecological structure and function. Following the terminology of Jones et al (1994), we refer to this suite of practices as *ecosystem engineering*, a term used to describe the creation and maintenance of habitats by organisms. Thus, beavers engineer wetlands, species of coral engineer atolls, and earthworms engineer aerated soils; note that there is no necessary implication that the effects of engineering be intended or designed per se, though clearly some aspects may be.

Human ecosystem engineering on a significant scale has existed at least since our ancestors developed control of fire. The work of anthropologists, geographers, historians, and ecologists is increasingly revealing the extent to which small-scale societies, including preagricultural peoples, engage in habitat modification (Day 1953, Stewart 1956, Lewis 1982, Cronon 1983, Denevan 1992, Burney 1995, Fairhead & Leach 1996, Little 1996, Nabhan 1995, Russell 1997). The methods used for "domesticating" environments (Blackburn & Anderson 1993a) range from intentional burning (e.g. to increase growth of plants used by people or as browse for deer and other game) to sowing, pruning, and even irrigation of wild food plants, and from construction of fish weirs and hunting blinds to construction of monumental earthen mounds (e.g. Steward 1933, Lewis 1982, Roosevelt 1995, Anderson 1996, Pyne 1998). With the coming of agriculture, ecosystem engineering certainly added new techniques, including clearing of forested areas to make gardens, fields, and pastures; building irrigation systems, sometimes on a massive scale; terracing mountainsides (Evans & Winterhalder 2000); fertilizing, plowing, and various other forms of soil engineering; and of course the creation of new life forms via plant and animal breeding (reviewed in Redman 1999).

In engineered ecosystems, especially agricultural ones, humans have often been primary agents of biological disturbance (Chapin et al 1997, Noble & Dirzo 1997). As discussed above, evidence suggests that environmental perturbations of moderate intensity and frequency can often enhance biodiversity (Petraitis et al 1989, Perry & Amaranthus 1997). In forest and savanna environments in particular, a history of anthropogenic disturbance (such as clearing of small patches of forest, low intensity burning, soil modification, and moderate levels of predation) might have this effect. For example, a study of Mediterranean oak savanna woodlands (Pineda 1991) found that recurrent anthropogenic disturbance promoted spatial heterogeneity, and hence biodiversity, in this habitat. Similarly, Delcourt & Delcourt (1997) conclude that selective burning in southern Appalachia by Native Americans in pre-Columbian times created an "intermediate-scale disturbance regime that promoted a heterogeneous mosaic of different vegetation types" (p. 1013), thus heightening the contrast across vegetational boundaries and increasing habitat (beta) diversity. Other well-documented examples include the effects of pastoralist grazing and burning in East African savanna landscapes (Little 1996), the "forest islands" produced around villages in Guinea, West Africa (Fairhead & Leach 1996), the rich anthropogenic soils (*terra preta do Indio*) produced by Amerindian swidden horticulturalists in Amazonia (Smith 1980, Hecht & Posey 1989, Balée 1993, McCann 1999), and the burning, pruning, and foraging by Miwok Indians that helped shape the mosaic habitat of Yosemite, the crown jewel of American wilderness preservationists (Anderson 1993, Anderson & Nabhan 1991).

Although such moderate disturbances can be biodiversity-enhancing, introduction of periodic burning in areas where vegetation is not adapted to this disturbance may have more irreversible effects, including extinction of some plant species, and this has recently been suggested as the likely cause of a cascade of habitat changes and megafaunal extinctions during human colonization of Australia ca 50,000 years ago (Miller et al 1999). In any case, abundant evidence indicates that dense populations and intensive (particularly commercial) agricultural production tip the balance toward declining biodiversity (Chapin et al 1997, Matson et al 1997). In addition, intensification of agricultural production over the past several millennia has led to increasing human reliance on a few domesticated plant species (Matson et al 1997). In many instances, this has led to further biodiversity reduction in agroecosystems, and concurrently to population growth and hence a positive feedback cycle (Figure 1). When this cycle of intensification and population growth has been gradual enough, the trajectory has been sustainable (Redman 1999:122ff). In cases of more rapid change (either demographic or environmental), there is sometimes a trade-off between production yield and sustainability (e.g. Humphries 1993). The resultant unstable dynamic in many regions is cause for current widespread concern (McMichael 1993, Cohen 1995, Swanson 1995, Redford & Mansour 1996).

In sum, humans are colonizers, not homebodies; we are always pushing at the margins of our domain, expanding into new areas (even if occupied by other peoples) whenever changes in environment or mode of production allow. The characteristics supporting this colonizing strategy seem to have evolved in the

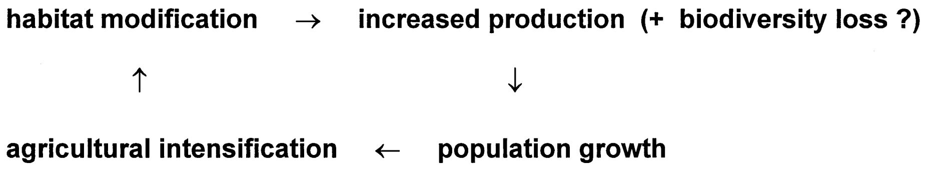

Figure 1 Positive feedback cycle in agricultural intensification, with biodiversity loss as a by-product.

context of very high levels of environmental instability during the Pleistocene. The strategy has been so successful that humans colonized virtually the entire terrestrial surface of the planet in a relatively short period and now are reworking the biosphere on an unprecedented scale. Our success as a species is based on cultural innovation, competitive dispersal, and ecosystem engineering; but this success is now spelling doom for thousands, and perhaps millions, of more venerable but less adaptable species (Kerr & Currie 1995, Norgaard 1995, Reaka-Kudla et al 1996, Vitousek et al 1997). In this context, it becomes crucial to understand the forces that have shaped human-environment relations in small-scale societies, and in particular the conditions under which sustainable use and perhaps even deliberate conservation have been possible.

THEORY OF ENVIRONMENTAL CONSERVATION

The primary role of theory in science is to guide the collection and evaluation of empirical knowledge. If our objective is to understand the existence and operation of environmental conservation, then a theory of this phenomenon should tell us the conditions under which it is likely to occur, the forms it is likely to take, and its environmental consequences. These theoretical predictions should be empirically testable, lest we become so enamored of our constructions that we fail to evaluate their actual explanatory power. In other words, an ideal theory would offer a coherent explanation for variation in the form and occurrence of environmental conservation that has been subject to rigorous empirical evaluation. Currently we do not have such an ideal theory. However, some elements of it are beginning to take shape, often in parallel in diverse disciplines. Here, we outline what we see as the most promising form of such a theory, synthesizing work done in microeconomics and decision theory, behavioral ecology, and political science.

What Is "Conservation"?

To begin with, we need to agree on the meaning of the term conservation. As noted in the introduction, this is of more than semantic interest because the ongoing and sometimes bitter debate over the existence of indigenous conservation turns in part on different conceptions of what this term means.

Many accounts of indigenous environmental relations implicitly or explicitly equate conservation with long-term coexistence of a people or culture with a suite of other species or an ecosystem. In other words, failure to overharvest or extirpate resident species, or to degrade habitat, are taken to be diagnostic of conservation. Sponsel et al (1996a:23) summarize this view succinctly when they take note of arguments that "many indigenes, like the Bora and Desana in Amazonia and the Mentawaians in Indonesia, are effective conservationists; their societies have used and managed the natural resources of their habitat in sustainable ways for centuries or even millennia."

But as many have pointed out, this view can be criticized for conflating conservation with sustainability (Hunn 1982, Bulmer 1982, Winterhalder 1983, Hames 1987, Alvard 1995, Smith 1983, 1995, Low 1996). In general, factors such as low human population density, low demand for a resource, or limited technology can be expected to contribute to many instances of sustainability, with any conservation being an incidental byproduct. To ascribe such sustainable use to conservation involves a causal misattribution, because one or more of the factors just mentioned is likely to be the actual cause(s), whereas labeling something "conservation" implies that it exists because it prevents the overexploitation or degradation that would otherwise occur. Such cases have been labeled epiphenomenal conservation (Hunn 1982), but might also be termed coexistence (for resident species) and sustainable use (for habitats).

With this issue in mind, Alvard (1995) proposed that practices must be "costly in the short term" to qualify as conservation. The essence of conservation, in this view, lies in restraining current resource harvests or land use (and thus lowering current benefits, which satisfies the cost criterion) in order to reap benefits in the future. However, the cost (or delayed benefit) criterion is not sufficient because, again, it does not exclude other causes for a practice (Smith 1995). The costs due to harvest restraint might be borne in order to achieve goals other than conservation, such as allowing prey to congregate in large numbers (Ruttan 1998:48), or avoiding enemy attack (Hickerson 1965, Kay 1994). Neither does short-term cost appear necessary for conservation because there might exist options that are no costlier than the alternatives but that are chosen because they yield conservation benefits (e.g. plowing in a contour fashion rather than downslope).

A second issue concerns effect: To qualify as conservation, a practice should have a measurable effect in limiting resource harvests or environmental degradation. The rationale for this criterion may seem self-evident, but it is worth emphasizing because of the widespread tendency to cite the existence of beliefs about conservation, stewardship, and reverence toward nature as evidence for effective conservation (Callicott 1994). The existence of such beliefs or ideologies is an interesting phenomenon worthy of ethnographic or philosophical investigation but does not in itself constitute evidence for conservation practices or outcomes (Hames 1991). To be sure, ideology may be a proximate cause of conservation practices and may be deployed in discourses surrounding environmental practices; our basic point is that if conservation refers to environmental actions, the effect of these actions should be demonstrated.

We conclude that to qualify as conservation, any action or practice should satisfy two criteria: It should (*a*) prevent or mitigate resource depletion, species extirpation, or habitat degradation, and (*b*) be designed to do so (Smith 1995, Alvard 1998a, Ruttan & Borgerhoff Mulder 1999). The first criterion, concerning effect, is conceptually straightforward but poses some practical difficulties not addressed in this article involving the operational definition of depletion or degradation (see Meffe & Carroll 1997, Cocklin 1989). The design criterion is more conceptually challenging, as it raises issues regarding historical processes, causal versus functional explanation, agency, and the like. But it nevertheless seems essential if we are to understand the historical processes that have shaped human-environment relations. It also has practical significance, for if a record of coexistence or sustainable use is taken as sufficient evidence of practices and institutions supporting environmental conservation, then we will be unprepared for the emergence of resource depletion or environmental degradation when factors such as population density or marketing opportunities change (Low & Heinen 1993, Henrich 1997, Galaty 1999).

Our use of “designed” is meant to allow for several distinct processes to play a role in shaping conservation practices. On the one hand, it can include design based on conscious beliefs and preferences, and hence an intentional form of explanation (Elster 1983). But evolutionary processes, whether cultural or genetic, can also produce complex design, and our definition also allows this. Thus, if it could be shown that a conservation practice had spread or was maintained in a population because its practitioners were either less likely to suffer extinction or more likely to prosper and spread (as compared with nonconservers), that would satisfy the design criterion. This would be the case even if the conscious reasons for carrying out the practice were not connected to conservation (but to propitiating spirits, for example). However, it is important to note that the conditions for this to occur seem quite restrictive because conservation not only involves a time lag in reaping a benefit (Rogers 1991), it also often produces a collective good (see below) that may benefit nonconservers as much as conservers. That is probably why no one has been able to demonstrate the evolution of systems of conservation in nonhuman populations, despite past claims for such in terms of “group selection” (Wynne-Edwards 1965) or “prudent predation” (Slobodkin 1968). On the other hand, it is possible that cultural evolution is more favorable than genetic evolution to the spread of group-beneficial traits (Boyd & Richerson 1985, Soltis et al 1995).

In sum, a definition of conservation that incorporates the two criteria of effect and design is logically coherent and perhaps of practical value. It serves to distinguish deliberate conservation from sustainability due to other factors, and it directs research toward measuring actual environmental effects in addition to beliefs or claimed effects. This definition also highlights some reasons why the debate over indigenous or premodern conservation has been so protracted and difficult to resolve. Some participants in this debate take sustainable use as the sole criterion for conservation whereas others take it to be an unintended byproduct of other factors (such as low population density); and some have seen statement of conservation beliefs or ethics as conclusive evidence of conservation whereas others demand data on environmental effects. Our definition does pose some difficult empirical and conceptual challenges. In particular, it means that conservation is inherently much harder to verify empirically than is depletion, for one must determine not only that coexistence or sustainable use occurs, but also that it is due to human actions designed to secure this end (as is the case for any valid claim involving adaptive functions of human behavior).

The Problem of Temporal Discounting

Conservation often involves taking a short-term loss in order to reap a long-term gain (Alvard 1995, Ruttan & Borgerhoff Mulder 1999). This trade-off between present and future raises the complex issue of temporal discounting, a crucial concern in determining whether conservation is rational (Clark 1973) or adaptive (Rogers 1991, Alvard 1998a). There are two reasons for discounting future gains relative to present ones: Delays increase the chance that the gain will fail to be realized at all (because environmental conditions change, or others make off with the resource, or the decision-maker dies); and delays reduce the opportunity to put a given gain to work in producing more gains. Such trade-offs are obvious in a commodified context, where immediate harvests allow the gains to be invested at the going rate of return. Thus, one can predict that slow-growing resources (like whales and redwood trees) that have a reproductive rate below the prevailing rate of return on capital investment will more likely suffer unsustainable harvest rates (Clark 1973, Alvard 1998a). Temporal discounting is likely to be present in subsistence economies as well, and it may be a significant deterrent to conservation because subsistence decisions always involve the two kinds of trade-offs mentioned above (Alvard 1998a). Indeed, both theory (Rogers 1994) and data on the foraging decisions of other species (Cuthill & Houston 1997, Kacelnik 1997) suggest that it is a ubiquitous product of evolution.

Despite this ubiquity, there are likely to be cultural and ecological sources of variation in temporal discount rates. Briefly, these might include the following:

1. Demographic growth rates: Stable or declining population size reduces the value of present versus future investment in reproduction (Fisher 1958, Rogers 1994); the converse is true for increasing population.
2. Mortality rates: Low mortality reduces future uncertainty; the converse is true for high mortality.
3. Resource renewal rate: As noted above, slowly-growing resources lower the gains that can be had in the future by conserving today.
4. Resource mobility: High mobility increases uncertainty about future harvests.
5. Capital mobility: High mobility decreases the incentive to bear short-term costs in order to realize long-term yields from conservation.
6. Control over future harvests: Those who have little control over who will actually obtain the deferred harvest have less incentive to forgo present harvests.

This last point leads directly to our next topic.

Conservation and Collective Action

The theory of collective action is a well-developed branch of political science and decision theory dealing with social choice (Hardin 1982). Collective action refers to any group of (two or more) individuals cooperating or coordinating their actions in order to produce some outcome. Based on the assumptions of methodological individualism and rational self-interest (Elster 1983), the logic of collective action is a logic of individual benefits and costs, not collective ones (Olson 1965).

A collective action problem arises whenever collective action is individually costly but collectively beneficial (or vice versa). A common scenario in which a collective action problem can arise involves the attempt to provide or safeguard a *collective good*—that is, any good or service that is available to all members of some collective (e.g. a village, an organization, a nation), but whose consumption by some reduces the benefit available to the remainder (Hardin 1982). A classic form of collective good is what economists and collective action theorists term a common-pool resource (CPR) (Gordon 1954, Ciriacy-Wantrup & Bishop 1975, Ostrom 1990).

Environmental conservation raises collective action problems because it usually requires individually costly actions that produce collective benefits, thereby providing the opportunity (and incentive) for free-riding. Hardin's (1968) famous essay on the "tragedy of the commons" involves just such a scenario, where individual herders are unwilling to pay the costs of conservation (limiting herd size) because the benefits of doing so are shared collectively. The logic of Hardin's argument is sound, but some of the assumptions on which it is based can be challenged (Feeny et al 1990, Ostrom et al 1999). In particular, Hardin fails to recognize the distinction between situations involving open access (absence of property rights) and those of communal ownership (Feeny et al 1990). This oversight is critical because both theory and data indicate that resources involving open access are much more vulnerable to overharvesting than those with restricted access (Ostrom 1990, McKean 1992).

Anthropologists and others have documented dozens of systems of communal ownership and management of CPRs, many of them quite long-lived, and some providing considerable evidence of explicit and effective conservation practices (Ostrom 1990, Feeny et al 1990). Examples are found in communities all over the world and range from pastures and forests to irrigation systems and fisheries. Various features of CPR management seem designed to solve key aspects of collective action problems: socially regulated access, management rules governing resource harvests, means of monitoring compliance to these rules, and sanctions to punish those who violate them. The ubiquity of monitoring and sanctions in successful CPR systems strongly supports the self-interest assumption of collective action theory; these would be superfluous if such systems sprang from collectivist motivations or reflected culturally specific rationalities (Froemming 1999).

If systems of CPR management are so effective at solving collective action problems involving resource utilization, and have apparently been invented independently many times, why is overharvesting or environmental degradation a widespread and recurrent problem? First, most systems of CPR management depend on relatively informal means of monitoring and sanctioning free-riding and other violations of management rules. But these become ineffective as group size increases, a well-studied phenomenon in the literature on collective action (Olson 1965, Hardin 1982, Taylor 1987, Boyd & Richerson 1988). As population density and political centralization increases, communities may exceed the size and homogeneity needed for endogenous systems of communal management (Singleton & Taylor 1992). Second, where they have managed to persist, small-scale CPR systems are vulnerable to competition from larger and more powerful political and economic systems.

The problem we face today is not so much a tragedy of the commons (though open-access systems do persist in important domains, such as the atmosphere and open-ocean fisheries) as it is the elimination of myriad small-scale systems of common property management involving socially regulated access (Feeny et al 1990). The twin juggernauts of centralized governments and expanding commercial interests—often working in collusion—can undermine local resource management and increase market-driven resource extraction at an accelerating rate worldwide. This dynamic has been well studied in forests in parts of South and Southeast Asia (Gadgil & Guha 1992, Alcorn & Molnar 1996, Padoch & Peluso 1996). Managed for centuries by villages or other collectivities, some systems have suffered expropriation by colonial and postcolonial governments intent on extracting commercially valuable resources or combating what they see as the dangers of deforestation due to local use. But such external control can destroy local incentives for communal management, leading to overexploitation by both outside commercial interests and local peoples (Why conserve what outsiders are going to take?). Once undermined in this way, it is hard to reestablish effective systems of CPR management, particularly given increased local demand for natural resources due to greater involvement in market economies, as well as population growth.[4]

Predicting Conservation

We have defined conservation as practices that are designed to prevent or mitigate species depletion or habitat degradation. This definition implies a design process, either intentional or evolutionary. In turn, these practices involve processes of decision-making or adaptation by natural selection (of culture or genes) that, in conjunction with the theory summarized above, allow one to make a series of predictions about the conditions under which conservation is likely to occur:

1. Controlled or exclusive access (stable land rights);
2. Distinct or confined resource populations (to which controlled access can apply);

---

[4] Although the above generalizations are widely applicable, the distinction between localized systems of forest management and larger external ones and the assumption that local forest management is always sustainable are currently the subject of debate and revision based on historical and anthropological research (e.g. Fairhead & Leach 1998, Sivaramakrishnan 1999).

3. Resource populations that are resilient or rapidly renewing (hence likely to respond to management controls);
4. Low discount rates, such that the value of sustained yield exceeds the value of immediate yield; and
5. Social parameters (e.g. small group size and stable membership) and institutions (monitoring and sanctioning) that counter free-riding.

Conversely, the following set of conditions make deliberate and effective conservation much less likely to emerge or be stable:

6. High demand from external markets;
7. Rapid population growth;
8. Acute resource scarcity;
9. Adequate substitutes for threatened resources;
10. Acquisition of novel technology or migration into novel habitats; and
11. Ease in relocating production (expandable frontiers, mobile capital).

With these expectations in mind, we turn now to a survey of the ethnographic evidence.

ETHNOGRAPHIC EVIDENCE

The theoretical arguments discussed above provide some expectations regarding the conditions under which local systems of environmental conservation would arise and be maintained endogenously. They suggest that although these conditions are fairly stringent, a broader range of conditions might result in sustainable resource use. Because of the paucity of studies that provide comparable data, it is not currently possible to systematically examine the ethnographic record to determine the empirical match with theoretical predictions. Hence we present summaries of a variety of cases, organized into two categories: practices designed to conserve species or habitats, and cases of sustainable use (coexistence and nondegradation) but without evidence of conservation design. The cases discussed here are representative of the range of possibilities, but not necessarily of the actual frequency of outcomes in each category.

Practices Designed to Conserve

Following the arguments summarized earlier, we include cases in this category if the actors clearly state that the practices in question are intended to conserve, or if these practices exhibit design features that are costly or complex and unlikely to have arisen for purposes other than conservation. There are a variety of such practices reported in the ethnographic literature, and we have grouped them into several categories. In almost no case, however, are data sufficient to determine whether or not the criterion of conservation effect is met.

Harvesting Restraint That Raises Short-Term Production Cost The type of resource utilization that most clearly meets the conservation design criterion is harvesting restraint that raises short-term production cost. It has rarely been documented, however (with the obvious exception of husbanding of domesticated resources). Descriptions of Native American gatherers leaving smaller roots or corms (or portions thereof) behind during the gathering process (Anderson 1996) may qualify, although other reasons for that practice may apply.

Sometimes harvesting restraint is motivated by belief in supernatural sanctions. Thus, Kayapó honey-gatherers in the Brazilian Amazon leave a portion of honey, pollen, and broodcomb in the hive in order to placate the deity *Bepkôrôrôti,* which encourages bees to recolonize the hive (Posey 1998). Although intent to conserve is absent in this case, Posey argues that design for conservation is present. But he also reports that honey-gatherers access the hive by cutting down a large tree; this surely accelerates the decay of a prime nesting site (a hollow trunk), and leaves the partially destroyed hive accessible to ground-living predators, rendering the conservation effect of partial hive harvesting ambiguous at best.

Another recent study discussing subsistence harvest restraint describes limitations placed on hunting of mountain goats by Gitksan and Wet’suwet’en Indians of the Pacific Northwest, involving group defense of hunting territories and regulation of hunting within each group by the local chief (Gottesfeld 1994:468). Because mountain goats are localized and have low reproductive rates and dispersal ability, they are easily overhunted, and Gottesfeld argues that chiefly control on hunting was a necessary and effective form of conservation. Although no quantitative data on either hunting practices or mountain goat biology are presented, Gottesfeld does mention a local myth that recounts wanton slaughter of mountain goats, followed by their revenge on human hunters. Although crucial evidence is lacking, the existence of territorial control and centralized sanctions, plus the lack of adequate substitutes for goat wool in the traditional economy, provides some plausibility to the conservation hypothesis. Here, existence of design elements mitigating the free-rider problem, plus the vulnerability of mountain goats to extirpation, serve to strengthen the case for the conservation. In contrast, most of the other arguments Gottesfeld makes for Gitksan and Wet‘suwet’en game conservation fail to meet the design criterion for conservation (see below).

Many claims of harvesting restraint as conservation commonly fail to provide any evidence beyond very loose assertions of a likely conservation effect. For example, Posey (1998) presents the observation that Kayapó avoid schools of minnows out of fear of the mythological 20-m-long serpent *mry-ka’ak* to argue that they thereby preserve a basic component of the riverine food chain. Posey’s point is that intention or awareness of conservation is not needed for it to occur. Perhaps that is true, but surely we need more evidence than is provided in this instance. Although more detailed ecological study might confirm that such preservation is an effect of Kayapó beliefs about *mry-ka’ak,* there are many possible reasons why this belief exists. A cogent argument that conserving effects explain a practice requires close attention to design features and consideration of alternative hypotheses. But we find it hard to envision how a practice that sacrificed short-term gains to fishermen (assuming Kayapó would indeed be able to efficiently harvest minnows) in order to benefit an open-access common-pool resource (the riverine food chain) would arise and spread.

Even more problematic is Gottesfeld's (1994) interpretation of the bottleneck created by the labor demands of processing salmon for storage in the Northwest Coast Indian fishery as functioning to limit the harvest, and hence conserve salmon. Although it is true that the processing constraints meant that many more fish could be harvested than could be preserved, it does not follow that processing is designed to conserve fish stocks. Contrary to Gottesfeld, we argue that by developing means of preserving fish, Northwest Coast Indians increased rather than limited their harvest, being motivated to harvest enough fish to smoke and serve as a staple food for many months of the year (though there is no evidence that the increase was enough to deplete the salmon runs). Gottesfeld (1994:453) reports that when a Gitksan elder was asked about "traditional fish management and conservation practices," he answered that "all fish remains must be returned to the water. That was all that was necessary to renew the fish." This supports the inference that salmon conservation was neither practiced nor necessary in the indigenous fishery.

Inuit foragers one of us worked with described one conservation practice, avoiding the harvest of geese accompanied by goslings (Smith 1991:282), but no observations confirming or refuting this statement were made. Fienup-Riordan (1990) reports a contrary practice of harvesting flightless geese and goslings as a regular practice of Yup'ik Eskimos in western Alaska, and Burch (1994) cites a variety of nonconserving hunting practices of northern hunter-gatherers. Even simple statements that harvesting restraint occurs are rare in the ethnographic literature, occurring in only 5% of 122 cases in a recent cross-cultural study (Low 1996:359).

Extensive observational data on various tropical subsistence hunters uniformly indicate that their subsistence decisions are designed to maximize foraging efficiency (harvest per unit hunting time), usually contradicting predictions based on game conservation (Hames 1987, Kay 1994, Vickers 1994, Alvard 1995, FitzGibbon 1998). For example, hunters preferentially pursue species that yield high returns, regardless of vulnerability to depletion; they readily harvest females and prime-aged individuals; they often respond to game declines by increasing hunting effort; and they allocate hunting effort to habitat "patches" in proportion to profitability rather than prey abundance or population resilience. Similar results have recently been described for an artisanal fishery in the Solomon Islands (Aswani 1998, 1999).

It is important to note that just as sustainable use does not necessarily indicate conservation, lack of conservation does not necessarily indicate resource depletion. Sustainable harvest levels are a frequent outcome of Winterhalder & Lu's (1997) simulation model of predator (hunter-gatherer)-prey population ecology, despite the fact that the model assumes human choices to be guided simply by optimal foraging goals. Various quantitative studies of subsistence foraging indicate that harvest of most prey populations seem to be at sustainable levels. For example, Hill

(1983) tested the claim that the !Kung San were limiting their reproductive rate and foraging effort in order to avoid overharvesting the game supply (e.g. Harris 1979:81) and found that the !Kung harvest less than 1% of the biomass of local large herbivores, whereas the maximum sustainable yield of these species ranges from 10% to 20%. In this case, as in others, the likely explanation for low harvest rates of animal prey is that the returns from hunting using indigenous technology are too low to motivate higher harvest rates.

Some direct evidence on this is beginning to accumulate. Using source-sink models and careful assays of both hunting pressure and game abundance, Hill, Padwe, and Ache Indian coworkers have recently shown that Ache harvest a very small and sustainable proportion of the standing stock of their nine major game species (Hill et al 1997, 2000). Nevertheless, in the Ache case and others, detailed ethnographic evidence indicates that subsistence hunting by indigenous hunters can lead to localized depletion (Alvard 1994, Vickers 1991, Peres 2000) or even regional extirpation, as in Inuit hunting of musk oxen following introduction of rifles (Burch 1977, 1994). On balance, the evidence on faunal impacts of small-scale societies indicates that conservation is absent and depletion is sometimes a consequence. Exceptions to this can occur in cases where hunters control access to hunting territories (Feit 1987), congruent with one of the predictions listed above. Evidence that subsistence hunting depresses prey population levels (e.g. Kay 1994, Peres 2000) does not prove that the net effect of human predation on biodiversity is negative, however. First, localized depletion will often be balanced by prey population growth nearby (Hill et al 2000). Second, by serving as keystone predators (Paine 1966), humans might help prevent excessive ecological competition between prey species or (indirectly) other members of an ecological community (Simenstad et al 1978).

Protection or Propagation of Resource Species Another form of conservation involves practices designed to protect or propagate resource species. A variety of such practices are described for Native American foragers, including irrigation of wild grasses, deliberate dispersal of wild plants used for food and medicine through broadcast or planting of seeds or in some cases transplanting, and periodic burning to control outbreaks of plant diseases (reviewed in Anderson 1996, Blackburn & Anderson 1993a). It is interesting to note that all these examples involve plant resources, in contrast to the paucity of cases of harvest restraint or protection of fauna.

Avoidance of Harmful Habitat Modification As discussed above (and documented further below), habitat disturbance need not lead to environmental degradation or biodiversity loss and in fact may enhance biodiversity. But clearly some types of habitats are more sensitive to the effects of modification than others. Hence avoidance or mitigation of such habitat change can be a form of conservation, as recognized in environmental legislation and regulations in many industrialized societies. Do small-scale societies show analogous controls on habitat modification clearly designed to conserve? If so, they appear to be understudied in the ethnographic literature. One possible example is found among several Indian societies in Northwest Amazonia, who avoid swidden clearing at river margins, thus preserving gallery forests that enhance aquatic productivity (Chernela 1989, Beckerman & Valentine 1996). Although this is apparently undertaken in order to maintain fishing productivity, it is a plausible case of conservation involving short-term costs.

Another proposed example of indigenous habitat conservation involves various cases of sacred groves. Although many of these cases occur among long-standing state societies, and thus lie outside the orbit of this article, Olafson (1995) has recently published an interesting discussion of sacred groves in the Philippines and found that protection of groves was "a latent function of ... religion, and thus a weak form of conservation" (Olafson 1995:29). There was no indication that local protection was matched to overall forest preservation, but rather that it was matched to "protection of themselves and their children from disease-causing environmental spirits," a relationship he terms "incidental conservation" (Olafson 1995:29).

Restriction of Human Population Growth to Match Resource Base Although protection or enhancement of environmental resources is the obvious route to conservation, supply and demand can also be brought into line by reducing consumer demand, as advocates of contemporary population control frequently point out. Most claims we have examined of human population regulation to avoid resource exploitation confuse epiphenomenal effects with actual causes (as discussed in Bates & Lees 1979, Dewar 1984, Smith & Smith 1994). But under special circumstances—highly circumscribed populations, effective centralized or communal sanctions—such control over population growth can be achieved. One of the few adequately documented cases is the Polynesian chiefdom of Tikopia. Tikopia exhibited clear cultural population controls involving celibacy, contraception, infanticide, abortion, and periodic forced emigration (Firth 1936, Borrie et al 1957). This, coupled with intensive horticulture and arboriculture, as well as centralized (chiefly) controls on utilization of marine resources, limited consumption of resources to sustainable levels for many centuries (Kirch & Yen 1982, Kirch 1997).

Tikopian practices did not generally conserve native terrestrial species, however, and indeed initial occupation was followed by extensive deforestation, increasing soil erosion, extinction or extirpation of at least six bird species, and depletion of marine and terrestrial fauna (Kirch 1997:37). Eventually Tikopians converted to a sustainable production system emphasizing arboriculture, creating such a thoroughly anthropogenic habitat that "virtually the entire land surface is cloaked in such orchard stands of economic trees ... protecting and shading intensive undercanopy plantings" of yams and other cultigens (Kirch 1997:35). Thus, Tikopians did not conserve original biodiversity or specific resources so much as an entire highly modified agroecosystem (as well as marine resources of the reef and lagoon, whose exploitation was limited by chiefly authority and sanctions).

Sustainable Use Without Conservation

We suggested earlier that small-scale societies have developed many practices designed to enhance livelihood, for which habitat or biodiversity conservation is a byproduct—in other words, these practices would be carried out whether or not they had conservation effects. As with conservation by design, it is convenient to group ethnographic examples of these into several categories.

Matching Harvest to Needs Statements that people harvest no more than they need or can fully utilize are commonplace in the popular literature on indigenous conservation. Nelson (1983:200) quotes a Koyukon Indian woman as saying "[p]eople never kill animals for no reason, because they know there's times when they'll really need to kill anything they can find." This is a rational stance, and it may well have a moral dimension involving a social contract with immortal animal spirits (see below), but we question Nelson's conclusion that this statement "sums up an important element of Koyukon interactions with nature—the conservation ethic and its associated practices." Avoiding wanton destruction of unneeded prey is not the same as restraining harvest below current desires, and it does not demonstrate design for conservation. After all, because harvesting additional resources generally requires effort, even with economies of scale, such avoidance will usually save time and effort. In turn, responding to acute shortages by intensifying hunting pressure—"killing anything they can find"—means that resource depletion is more likely to occur precisely when local prey populations are at a low ebb. Of course, a harvest that is greater than consumption demand might sometimes be very low cost or even unavoidable, as in bison cliff jumps, where the vagaries of herd size make the number killed difficult to control (Wheat 1972).

Matching harvest levels to current needs, then, does not qualify as conservation, although it may have the effect of minimizing off-take. Full utilization of each item harvested will also have that effect, but again, the conserving effect seems fortuitous. When resources are scarce, the marginal costs of harvesting them (due to increased search time) are higher, and the efficient response is then to invest increased labor in extracting more value per item harvested (Jones & Metcalfe 1988). Such practices as splitting bones to extract marrow and boiling them to extract grease are the commonly observed responses. Conversely, when resources are abundant, we can expect to see less frugal utilization; for example, during periods when caribou populations were high, Inuit sometimes killed a dozen or more animals per household each fall to obtain fresh skins for bedding and clothing, leaving the meat to rot (Kelsall 1968).

Regulating Onset or Duration of Harvests Controls governing timing of resource harvests, as well as who has the right to participate, are widespread in small-scale societies. They are found, for example, in many nearshore marine fisheries, such as the trochus shell fishery in the Kei Islands of eastern Indonesia (Ruttan 1998). As Polunin (1984) argued about such systems in general, in the cases Ruttan studied, the aim seems to be to maximize economic control by locals, rather than to conserve resources per se, although the latter may be a byproduct. On the other hand, regulating the onset of a harvest may sometimes be designed to maximize returns on labor time (e.g. waiting until schools of fish have gathered in large numbers), and thus may actually increase harvests rather than restrain them (Polunin 1984). A good example of this is provided by the communal bison hunt once practiced by Indian tribes of the North American Plains, such as the Cheyenne. Here, the onset of the summer tribal hunt was carefully regulated by specifically appointed "camp police," and anyone tempted to "jump the gun" (thus obtaining a large personal share, but scattering the herd before others could participate) was subject to severe sanctions, such as public beatings, and destruction of their horses and hunting equipment (Hoebel 1978:58f).

Limits on the duration of harvests are better candidates for conservation by design, but they seem to be correspondingly rare in the ethnographic literature. The related practice of restraint in foraging effort following on the heels of successful harvests has frequently been nominated as a conservation practice. Thus, Beckerman & Valentine (1996) report that among the Arawakan and eastern Tukanoan peoples of northwestern Amazonia, it is commonly believed that one should "rest" for a few days after successful harvests in order to avoid supernatural sanctions for immoderation. They propose that this practice protects vulnerable stocks of fish and game, but provide no data on actual conservation effects, nor do they consider alternative hypotheses. For example, because the social context of foraging in these communities involves communal sharing of most of the catch, it is possible that foragers engage in "turn-taking" to avoiding excessive free-riding on their efforts.

Patch-Switching to Maximize Overall Return Rates Coexistence of hunter-gatherers and their prey, rather than extinction or extirpation, is commonly observed. Many have used this as evidence for conservation, but as suggested above, this is not sufficient because such coexistence could be due to other causes. One prominent alternative invokes the advantages of leaving a patch (i.e. resource concentration) before it is fully depleted because of the likelihood of obtaining higher foraging returns elsewhere (Winterhalder 1981, Smith 1983). The foraging strategy model known as the marginal value theorem (MVT) (Charnov 1976) makes a number of predictions concerning patch use, movement, and time allocation. The key one for current purposes is that a forager wishing to maximize harvest per unit time spent foraging should leave any given patch when the marginal return rate (the expected rate of harvest over the next small period of time) falls below what can be obtained by traveling on to another less-depleted patch. Thus, assuming that there are sufficient alternative patches, and assuming that patch depletion follows a diminishing return curve, the MVT predicts that an efficient forager will generally leave a patch well before total exhaustion of resources has occurred.

Smith (1983) suggested this pattern of decision-making could account for the "rotation of hunting territories" described by Feit (1973) and others for Waswanipi Cree Indians. Feit (1987) has countered with more detailed evidence that he argues supports an active management (conservation) of moose, beaver, and possibly other resources by Waswanipi. Clearly, if hunters can be assured that managed hunting territories will not be "poached" by others, long-term management and short-term costly restraint in service of long-term sustainable yields could be in their interests. If such a system operates among Cree in recent decades, it certainly differs substantially from the much more mobile, opportunistic pattern of land use and hunting effort described in written accounts by Jesuit missionaries and explorers of the seventeenth and eighteenth centuries (Leacock 1954, Krech 1999:200ff).

Indeed, ethnohistoric evidence (Brightman 1987, 1993, Krech 1999) indicates that "today's conservation ethic and practices were largely absent among Northern Algonquians until certain historical conditions emerged in the wake of the arrival of European outsiders mainly interested in controlling Indians economically and spiritually" (Krech 1999:206). Instead, what prevailed was an animistic belief system that viewed animal prey as inexhaustible but success in capturing them as dependent on social reciprocity involving ritual forms of respect (e.g. in handling carcasses and caring for the bones of prey). Within this belief system, the way to ensure abundance of prey was to kill as many as possible and consume them respectfully, so that their reincarnated souls would animate future prey animals to yield themselves to hunters in greater and greater numbers. In some cases, hunters claimed that to refuse to kill animals that made themselves available to hunters was a form of disrespect that would lead to a decline in hunting success (Brightman 1993, Krech 1999). Fienup-Riordan (1990) has detailed a similar set of beliefs and practices among Yup'ik Eskimo thousands of miles to the west of the Algonquian area, although as in the Algonquian case, explicit conservation beliefs and practices have emerged as a result of Western influence (Zavaleta 1999).

Although the MVT is usually applied to foraging contexts only, its essential logic can be applied to any system of production. Thus, pastoralists often move their herds to better grazing areas before the current area is completely depleted (Ruttan & Borgerhoff Mulder 1999 and commentators thereon). Such decisions are analogous to foragers' decisions to leave a patch in order to pursue higher returns elsewhere, consistent with MVT expectations. Similarly, swidden cultivators generally cease cultivating a given plot after one or a few years, which allows soil nutrients to recover and forest to regenerate in the fallow phase. Clearly, this practice prevents environmental degradation, but again this patch-switching appears to be designed to maximize production returns, rather than to conserve forests or soil.

If patch-switching for efficiency reasons results in sustainable hunting, grazing, or swidden clearing, does it converge with conservation by design, making the distinction a sterile "academic" debate? We think not because the predictions from the MVT or similar logic is bound to diverge from those of conservation just when they matter most: when pressure on environmental resources is high. Thus, if for one reason or another (e.g. climate change, gradual over-hunting, forager population growth, territorial restrictions by encroaching state systems) the per-capita abundance of prey should decline, foragers following MVT logic will stay longer in each patch (there being fewer enticing alternatives to draw them out), and increasing depletion can result (Broughton 1994, Vickers 1994). Similarly, during regional droughts, efficient pastoralists will have little incentive to look for nonexistent greener pastures, and overgrazing is more likely, though in most cases pastures recover quickly once rain returns (Galaty 1999). And a scarcity of uncleared forest, whether it results from internal population growth or external competition, can drive swidden cultivators to shorten fallow length to unsustainable levels, resulting in conversion of forest to grassland or other degraded habitats (Uhl et al 1982, Richards 1996:284). In contrast, practices designed for conservation might be expected to prevent such scenarios from unfolding.

Anthropogenic Practices That Create Habitat Mosaics Although most anthropological discussion of conservation in small-scale societies has focused on the issue of restraints in resource harvests, a few researchers (primarily ethnobiologists and restoration ecologists) have taken a different approach. Drawing on the growing consensus among ecologists that moderate and repeated disturbance can enhance biodiversity at species, habitat, and landscape levels, these researchers have considered the ways in which small-scale and indigenous societies contribute to such disturbance regimes. As we noted earlier, recurrent controlled burning is one very widespread type of anthropogenic disturbance, but other types have been described as well. For example, by felling large trees to collect honey, Kayapó create gaps that are colonized by many gap-dependent species. Posey (1998:105) notes that "...the felling of one large tree offers an immediate source of honey and provides a new ecological niche for useful plants that develop into a long-term hunting ground." Similarly, data reviewed by Little (1996) indicate that East African savanna habitats have been strongly shaped by subsistence practices of pastoralists and foragers over the past several millennia. In particular, the maintenance of grasslands through burning and stock grazing has prevented encroachment of bush and has thereby helped maintain high densities and diversity of wild ungulates as well.

One of the most carefully documented cases of anthropogenic biodiversity enhancement via creation of habitat mosaics involves Ka'apor swidden fallows (Balée 1993). In a comparison of fallows with primary forest, Balée shows that they differ significantly in tree composition but have equal biodiversity. These findings strongly support the view that these anthropogenic forests have enhanced regional biodiversity. It is interesting that although fallows have a much higher proportion of significant food species than does primary forest, Balée finds that Ka'apor do not intentionally produce this result; instead, most of the species in fallows are introduced or dispersed by animals attracted to clearings. He concludes from this and other evidence that "the Ka'apor agroforestry complex ... appears to be incidental to developmental processes associated with semi-sedentary, egalitarian

Amazonian societies—it is, simply, not a product of long term design, indigenous or otherwise" (Balée 1993:248).

CONCLUSIONS

We have defined conservation as practices that are designed to prevent or mitigate species depletion or habitat degradation. This definition implies a design process, either intentional or evolutionary. In turn, these processes invoke mechanisms of decision-making or adaptation by natural selection (of culture or genes) that allow one to make a series of predictions about the conditions under which conservation is (or is not) likely to occur, as outlined above. Although the state of the ethnographic record does not allow a systematic test of these predictions, the survey presented in the previous section does seem congruent with most or all of them. In particular, deliberate conservation is rarely evident for animal prey, particularly large game, whose very mobility often prevents local control over access and hence diffuses any benefits from restraint. In contrast, most cases of deliberate conservation (ranging from harvest restraint to population enhancement) encountered in our survey of the literature apply to plant resources or habitats. Furthermore, because conservation is likely to be habitat or resource specific, the presence or absence of conservation practices is not necessarily an indicator of the overall human impact on ecosystem processes and biodiversity.

Many will question a definition of conservation that requires evidence that it be intended or designed. But the alternative, labeling any actions that reduce harvest rates or environmental degradation as conservation, is problematic. First, it suggests a solution where there may be no problem, as in many cases where human environmental pressure is too mild or sporadic to cause irreversible biodiversity loss or habitat degradation; the evidence indicates that this is often the case in small-scale societies that are not engaged in commercial harvesting and that have low and stable populations. Second, without a design criterion for conservation, analyses are subject to the problems of functionalism that often plague social science (Elster 1983), including ecological anthropology (Smith & Winterhalder 1992). Any number of practices and beliefs may have the effect of reducing harvest rates or environmental impacts—going to church on Sundays, avoiding dangerous areas near enemy territory, or inventing a toggling harpoon that makes it more likely that a wounded seal will be captured rather than escape to die later. These effects are not why the practices are undertaken, however, and the environmental benefits in such cases are usually too diffuse to motivate individuals to undertake them for those reasons.

The dominant cultural meaning and practice of "conservation" at the current historical moment focuses on preservation of biodiversity (of species and higher levels) and ecosystem function. This approach to conservation is dependent on a set of political, economic, and intellectual realities with few parallels in human history. Conservation of biodiversity for its own sake, and preservation of wilderness for recreation or aesthetic admiration, are goals that may make sense to urbanized elites in industrial society (Guha 1989, 1997). But for subsistence-based societies, particularly small-scale ones, these appear to be alien concepts. Reviewing a large number of community-based conservation programs, Little (1994:350) concludes that "cases in which local communities in low-income regions manage their resource bases with the prime objective of conservation—rather than improved social and economic welfare—are virtually nonexistent." Instead, members of such societies are likely to pursue enhancement of the resources needed for livelihood, safeguarding of homelands from exploitation by outsiders, and allocation of subsistence effort to the most rewarding areas and resources currently available. These choices will often have the effect of conserving habitats and biodiversity, but they will not necessarily be designed to do so and may at times have the opposite consequence.

There are serious political issues at stake in debates about indigenous conservation, property rights, and economic change. There is no doubt that "coercive conservation" (Peluso 1993, Hitchcock 1995, Zerner 1996, Neumann 1998) can be a major threat to indigenous rights. Our critical examination of the debate over indigenous conservation is not meant to provide any support for those who believe that conservation agendas trump human rights, or that environmental protection justifies political and economic disenfranchisement of small-scale societies. Indigenous rights to traditional homelands and resource utilization should not be predicated on environmental conservation (Stearman 1994). Conservation is not the criterion for property rights employed within modern states, so it is hypocritical as well as unjust to impose it upon already marginalized and often impoverished people.

A more restricted and realistic understanding of indigenous conservation is not only better social (and environmental) science, it may also offer benefits in the arena of environmental policy and politics. Over-generalized or idealized expectations concerning the bases of environmental belief and practice in indigenous or small-scale societies can produce anger and mistrust when these are disappointed, as exemplified in the insightful discussion by Conklin & Graham (1995) of shifting relations between Kayapó Indians and environmentalists (see also Stearman 1994). As we move toward a more nuanced and realistic view of human-environment relations in small-scale societies, the actual accomplishments of these peoples in constructing their livelihoods in a generally sustainable fashion, sometimes deliberately conserving or enhancing species and habitats, become even more remarkable and worthy of understanding and respect.

ACKNOWLEDGMENTS

For constructive criticism of earlier versions of this article, not always heeded, we gratefully acknowledge Monique Borgerhoff Mulder, Don Grayson, Ray Hames, Kim Hill, Bobbi Low, Celia Lowe, Lore Ruttan, K Sivaramakrishnan, Linda Storm, and Bruce Winterhalder.

Visit the Annual Reviews home page at www.AnnualReviews.org

LITERATURE CITED

Alcorn JB. 1995. Comment on Alvard. *Curr. Anthropol.* 36:802–04

Alcorn JB. 1996. Is biodiversity conserved by indigenous peoples? In *Ethnobiology in Human Welfare*, ed. SK Jain, pp. 234–38. New Delhi: Deep Publ.

Alcorn JP, Molnar A. 1996. Deforestation and human-forest relationships: What can we learn from India? See Sponsel et al 1996b, pp. 99–121

Alvard MS. 1994. Conservation by native peoples: prey choice in a depleted habitat. *Hum. Nat.* 5:127–54

Alvard MS. 1995. Intraspecific prey choice by Amazonian hunters. *Curr. Anthropol.* 36:789–818

Alvard MS. 1998a. Evolutionary ecology and resource conservation. *Evol. Anthropol.* 7:62–74

Alvard MS. 1998b. Indigenous hunting in the neotropics: conservation or optimal foraging? See Caro 1998, pp. 474–500

Anderson MK. 1993. Native Californians as ancient and contemporary cultivators. See Blackburn & Anderson 1993b, pp. 151–74

Anderson MK. 1996. Tending the wilderness. *Restor. Manage. Notes* 14(2):154–66

Anderson MK, Nabhan GP. 1991. Gardeners in Eden. *Wilderness* 55:27–30

Anderson RC. 1990. The historic role of fire in the North American grassland. In *Fire in North American Tallgrass Prairies*, ed. SL Collins, LL Wallace, pp. 8–18. Norman: Univ. Okla. Press

Aswani S. 1998. Patterns of marine harvest effort in southwestern New Georgia, Solomon Islands: resource management or optimal foraging? *Ocean Coast. Manage.* 40:207–35

Aswani S. 1999. Common property models of sea tenure: a case study from Roviana and Vonavona Lagoons, New Georgia, Solomon Islands. *Hum. Ecol.* 27:417–53

Balée W. 1993. Indigenous transformation of Amazonian forests: an example from Maranhao, Brazil. *L'Homme* 33:231–54

Balée W, ed. 1998. *Advances in Historical Ecology.* New York: Columbia Univ. Press

Bates DG, Lees SH. 1979. The myth of population regulation. In *Evolutionary Biology and Human Social Behavior*, ed. N Chagnon, W Irons, pp. 273–89. North Scituate, MA: Duxbury

Beckerman S, Valentine P. 1996. On Native American conservation and the tragedy of the commons. *Curr. Anthropol.* 37(4):659–61

Berkes F. 1999. *Sacred Ecology: Traditional Ecological Knowledge and Resource Management.* Philadelphia: Francis & Taylor

Blackburn TC, Anderson K. 1993a. Introduction: managing the domesticated environment. See Blackburn & Anderson 1993b, pp. 15–26

Blackburn TC, Anderson K, eds. 1993b. *Before the Wilderness: Environmental Management by Native Californians.* Menlo Park, CA: Ballena

Bodley JH. 1996. *Anthropology and Contemporary Human Problems.* Mountain View, CA: Mayfield Publ.

Borrie W, Firth R, Spillius J. 1957. The population of Tikopia, 1929 and 1952. *Popul. Stud.* 10:229–52

Botkin D. 1990. *Discordant Harmonies: A New Ecology for the Twenty-First Century.* New York: Oxford Univ. Press

Boyd R, Richerson PJ. 1985. *Culture and the Evolutionary Process.* Chicago: Univ. Chicago Press

Boyd R, Richerson PJ. 1988. The evolution of reciprocity in sizable groups. *J. Theoret. Biol.* 132:337–56

Brightman RA. 1987. Conservation and resource depletion: the case of the Boreal Forest Algonquians. See McCay & Acheson 1987, pp. 121–41

Brightman RA. 1993. *Grateful Prey: Rock Cree Human-Animal Relationships.* Berkeley: Univ. Calif. Press

Broughton JM. 1994. Late Holocene resource intensification in Sacramento Valley, California: The vertebrate evidence. *J. Archaeol. Sci.* 21:501–14

Bulmer R. 1982. Traditional conservation practices in Papua New Guinea. In *Traditional Conservation in Papua New Guinea: Implicatins for Today*, ed. L Morauta, J Pernetta, W Heaney, pp. 59–77. Boroko, PNG: Inst. Appl. Soc. Econ. Res.

Burch ES Jr. 1977. Muskox and man in the central Canadian subarctic, 1689–1974. *Arctic* 30:135–44

Burch ES. 1994. Rationality and resource use among hunters. In *Circumpolar Religion and Ecology: An Anthropology of the North*, ed. T Iromoto, T Yamada, pp. 163–85. Tokyo: Univ. Tokyo Press

Burney D. 1995. Historical perspectives on human-assisted biological invasions. *Evol. Anthropol.* 4(6):216–21

Callicott JB. 1994. *Earth's Insights: A Multicultural Survey of Ecological Ethics from the Mediterranean Basin to the Australian Outback.* Berkeley: Univ. Calif. Press

Caro T, ed. 1998. *Behavioral Ecology and Conserv. Biol.* New York: Oxford Univ. Press

Carrier JG. 1987. Marine tenure and conservation in Papua New Guinea: problems in interpretation. See McCay & Acheson 1987, pp. 142–67

Chapin F, Walker B, Hobbs R, Hooper D, Lawton J, et al. 1997. Biotic control over the functioning of ecosystems. *Science* 277:500–4

Charnov EL. 1976. Optimal foraging, the marginal value theorem. *Theoret. Popul. Biol.* 9:129–36

Chernela J. 1989. Managing rivers of hunger: the importance of the blackwater river margin. See Posey & Balée 1989, pp. 238–48

Ciriacy-Wantrup SV, Bishop RC. 1975. "Common property" as a concept in natural resources policy. *Nat. Res. J.* 15:713–27

Clark CW. 1973. The economics of overexploitation. *Science* 181:630–34

Cocklin CR. 1989. Methodological problems in evaluating sustainability. *Environ. Conserv.* 16:343–51

Cohen JE. 1995. Population growth and earth's human carrying capacity. *Science* 269:341–46

Conklin B, Graham LR. 1995. The shifting middle ground: Amazonian Indians and ecopolitics. *Am. Anthropol.* 97:695–710

Cronon W. 1983. *Changes in the Land: Indians, Colonists, and the Ecology of New England.* New York: Hill & Wang

Crumley CL, ed. 1994. *Historical Ecology: Cultural Knowledge and Changing Landscapes.* Santa Fe: SAR Press

Cuthill IC, Houston AI. 1997. Managing time and energy. In *Behavioural Ecology: An Evolutionary Approach*, ed. JR Krebs, NB Davies, pp. 97–120. Oxford: Blackwell

Damas D, ed. 1984. *Handbook of North American Indians.* Vol. 5. *Arctic*. Washington, DC: Smithson. Inst.

Day GH. 1953. The Indian as an ecological factor in the Northeastern forest. *Ecology* 34:329–46

DeAngelis DL, Waterhouse JC. 1987. Equilibrium and nonequilibrium concepts in ecological models. *Ecol. Monogr.* 57:1–21

Delcourt HR, Delcourt PA. 1997. Pre-Columbian Native American use of fire on Southern Appalachian landscapes. *Conserv. Biol.* 11:1010–14

Denevan WM. 1992. The pristine myth: the landscape of the Americas in 1492. *Ann. Assoc. Am. Geogr.* 82:369–85

Dewar RE. 1984. Environmental productivity, population regulation, and carrying capacity. *Am. Anthropol.* 86:601–14

Diamond J. 1992. *The Third Chimpanzee.* New York: HarperCollins

Durning AT. 1992. *Guardians of the Earth: Indigenous Peoples and the Health of the Earth. Worldwatch Pap. No. 112.* Washington, DC: Worldwatch Inst.

Elster J. 1983. *Explaining Technical Change:*

A Case Study in the Philosophy of Science. Cambridge, UK: Cambridge Univ. Press

Evans T, Winterhalder B. 2000. Modified solar insolation as a potential agronomic benefit of terracing in steep-slope habitats. *Land Degrad. Dev.* In press

Fairhead J, Leach M. 1996. *Misreading the African Landscape: Society and Ecology in the Forest-Savanna Mosaic.* Cambridge, UK: Cambridge Univ. Press

Fairhead J, Leach M. 1998. *Reframing Deforestation: Global Analyses and Local Realities, With Studies in West Africa.* London: Routledge

Feeny D, Berkes F, McCay BJ, Acheson JM. 1990. The tragedy of the commons: twenty-two years later. *Hum. Ecol.* 18:1–19

Feit HA. 1973. The ethno-ecology of the Waswanipi Cree; or how hunters can handle their resources. In *Cultural Ecology*, ed. B Cox, pp. 115–25. Toronto: McClelland & Stewart

Feit HA. 1987. Waswanipi Cree management of land and wildlife: Cree ethno-ecology revisited. In *Native People, Native Lands: Canadian Indians, Inuit and Metis*, ed. B Cox, pp. 75–91. Ottawa, Can.: Carleton Univ. Press

Fienup-Riordan A. 1990. Original ecologists? The relationship between Yup'ik Eskimos and animals. In *Eskimo Essays*, pp. 167–91. New Brunswick, NJ: Rutgers Univ. Press

Fisher RA. 1958. *The Genetic Theory of Natural Selection.* New York: Dover. 2nd ed.

FitzGibbon C. 1998. The management of subsistence harvesting: behavioral ecology of hunters and their mammalian prey. See Caro 1998, pp. 449–73

Froemming SJ. 1999. *Rational choice and collective action in an Andean community.* PhD thesis. Univ. Washington, Seattle. xxi + 952 pp.

Gadgil M, Berkes F, Folke C. 1993. Indigenous knowledge for biodiversity conservation. *Ambio* 22:151–56

Gadgil M, Guha R. 1992. *This Fissured Land: An Ecological History of India.* Berkeley: Univ. Calif. Press

Galaty J. 1999. Comment on Ruttan and Borgerhoff Mulder. *Curr. Anthropol.* 40:39–41

Gamble C. 1994. *Timewalkers: The Prehistory of Global Colonization.* Cambridge, MA: Harvard Univ. Press

Goddard I, ed. 1996. *Handbook of North American Indians.* Vol. 17. *Languages*. Washington, DC: Smithson. Inst.

Gomez-Pompa A, Kaus A. 1992. Taming the wilderness myth. *BioScience* 42:271–79

Gordon HS. 1954. The economic theory of a common property resource. *J. Polit. Econ.* 62:124–42

Gottesfeld LMJ. 1994. Conservation, territory, and traditional beliefs: an analysis of Gitksan and Wet'suwet'en subsistence, Northwest British Columbia, Canada. *Hum. Ecol.* 22:443–65

Guha R. 1989. Radical American environmentalism and wilderness preservation: a third world critique. *Environ. Ethics* 11:71–83

Guha R. 1997. The authoritarian biologist and the arrogance of anti-humanism. *Ecologist* 27(1):14–20

Hames RB. 1987. Game conservation or efficient hunting? See McCay & Acheson 1987, pp. 92–107

Hames RB. 1991. Wildlife conservation in tribal societies. In *Biodiversity: Culture, Conservation, and Ecodevelopment*, ed. M Oldfield, J Alcorn, pp. 172–99. Boulder, CO: Westview

Hardin G. 1968. The tragedy of the commons. *Science* 162:1243–48

Hardin R. 1982. *Collective Action.* Baltimore, MD: John Hopkins Univ. Press

Harris M. 1979. *Cultural Materialism: The Struggle for a Science of Culture.* New York: Vintage/Random House

Hecht SB, Posey DA. 1989. Preliminary results on soil management techniques of the Kayapó Indians. See Posey & Balée 1989, pp. 174–88

Henrich J. 1997. Market incorporation, agricultural change, and sustainability among the Machiguenga Indians of the Peruvian Amazon. *Hum. Ecol.* 25(2):319–51

Hickerson H. 1965. The Virginia deer and intertribal buffer zones in the upper Mississippi valley. In *Man, Culture, and Animals: The Role of Animals in Human Ecological Adjustment*, ed. A Leeds, AP Vayda, pp. 43–65. Washington, DC: Am. Assoc. Adv. Sci.

Hill K. 1983. *Male subsistence strategies and time allocation to activities among the Ache hunter-gatherers of Paraguay.* PhD thesis. Univ. of Utah, Salt Lake City. 233 pp.

Hill K. 1996. The Mbaracayú and the Aché of Paraguay. See Redford & Mansour 1996, pp. 159–96

Hill K, Padwe J, Bejyvagi C, Bepurangi A, Jakugi F, et al. 1997. Impact of hunting on large vertebrates in the Mbaracayu reserve, Paraguay. *Conserv. Biol.* 11(6):1339–53

Hill K, Padwe J, Bejyvagi C, Bepurangi A, Jakugi F, et al. 2000. Sustainability of Ache hunting in the Mbaracayu Reserve, Paraguay. In *Sustainability of Hunting in Tropical Forests*, ed. J Robinson, E Bennet, pp. 79–105. New York: Columbia Univ. Press.

Hitchcock RK. 1995. Centralization, resource depletion, and coercive conservation among the Tyua of the northeastern Kalahari. *Hum. Ecol.* 23:169–98

Hoebel EA. 1978. *The Cheyennes: Indians of the Great Plains.* New York: Holt, Rinehart & Winston. 2nd ed.

Hubbell SP, Foster RB, O'Brien ST, Harms K, Condit R, et al. 1999. Light-gap disturbances, recruitment limitation, and tree diversity in a neotropical forest. *Science* 283:554–57

Humphries S. 1993. The intensification of traditional agriculture among Yucatec Maya farmers: facing up to the dilemma of livelihood sustainability. *Hum. Ecol.* 21:87–102

Hunn ES. 1982. Mobility as a factor limiting resource use in the Columbia Plateau of North America. See Williams & Hunn 1982, pp. 17–43

Jones CG, Lawton JH, Shachak M. 1994. Organisms as ecosystem engineers. *Oikos* 69:373–86

Jones KT, Metcalfe D. 1988. Bare bones archaeology: bone marrow indices and efficiency. *J. Archaeol. Sci.* 15:415–23

Kacelnick A. 1997. Normative and descriptive models of decision making: time discounting and risk sensitivity. In *Characterizing Human Psychological Adaptations* (*Ciba Found. Symp. 208*), ed. G Bock, G Cardew, pp. 51–67. Chichester, UK: Wiley

Kay CE. 1994. Aboriginal overkill: the role of Native Americans in structuring western ecosystems. *Hum. Nat.* 5:359–98

Kelsall JP. 1968. *The Migratory Barren-Ground Caribou of Canada.* Ottawa, Can.: Dep. Indian Affairs North. Dev.

Kerr J, Currie D. 1995. Effects of human activity on global extinction risk. *Conserv. Biol.* 9(6):1528–38

Kirch PV. 1997. Microcosmic histories: island perspectives on "global" change. *Am. Anthropol.* 99:30–42

Kirch PV, Hunt TL, eds. 1997. *Historical Ecology in the Pacific Islands: Prehistoric Environmental and Landscape Change.* New Haven, CT: Yale Univ. Press

Kirch PV, Yen DE. 1982. *Tikopia: The Prehistory and Ecology of a Polynesian Outlier. Bernice P. Bishop Mus. Bull. 238.* Honolulu, HI: Bernice P. Bishop Mus.

Klein RG. 2000. Archaeology and the evolution of human behavior. *Evol. Anthropol.* 9:17–36

Krech III S. 1999. *The Ecological Indian: Myth and History.* New York: Norton

Leacock EB. 1954. *The Montagnais "Hunting Territory" and the Fur Trade. Am. Anthropol. Assoc., Mem. No. 78.* Menasha, WI: Am. Anthropol. Assoc.

Lewis HT. 1982. Fire technology and resource management in aboriginal North America and Australia. See Williams & Hunn 1982, pp. 45−67

Little PD. 1994. The link between local participation and improved conservation: a review of issues and experiences. In *Natural Connections: Perspectives in Community-Based Conservation*, ed. D Western, PM Wright, pp. 347–72. Washington, DC: Island

Little PD. 1996. Pastoralism, biodiversity, and the shaping of savanna landscapes in east Africa. *Africa* 66(1):37–51

Low BS. 1996. Behavioral ecology of conservation in traditional societies. *Hum. Nat.* 7(4):353–79

Low BS, Heinen JT. 1993. Population, resources, and environment: implications of human behavioral ecology for conservation. *Popul. Environ.* 15:7–41

Martin PS. 1973. The discovery of America. *Science* 179:969–74

Martinez D. 1996. First people, firsthand knowledge. *Sierra* 81(6):50–51, 70–71

Matson PA, Parton WJ, Power AG, Swift MJ. 1997. Agricultural intensification and ecosystem properties. *Science* 277:504–9

McCann JM. 1999. Before 1492: the making of the pre-Columbian landscape. Part I: The environment. *Ecol. Restor.* 17:15–30

McCay BJ, Acheson JM, eds. 1987. *The Question of the Commons: The Culture and Ecology of Communal Resources.* Tucson: Univ. Ariz. Press

McGhee R. 1984. Thule prehistory of Canada. See Damas 1984, pp. 369–76

McKean MA. 1992. Success on the commons: a comparative examination of institutions for common property resources management. *J. Theoret. Polit.* 4:247–81

McMichael A. 1993. *Planetary Overload: Glob. Environ. Change and the Health of the Human Species.* Cambridge, UK: Cambridge Univ. Press

Meffe GK, Carroll CR, eds. 1997. *Principles of Conservation Biology.* Sunderland, MA: Sinauer

Miller GH, Magee JW, Johnson BJ, Fogel ML, Spooner NA, et al. 1999. Pleistocene extinction of *Genyornis newtoni:* human impact on Australian megafauna. *Science* 283:205–8

Nabhan GP. 1995. Cultural parallax in viewing North American habitats. See Soulé & Lease 1995, pp. 87–101

Nabhan GP. 1997. *Cultures of Habitat.* Washington, DC: Counterpoint

Nelson RK. 1982. A conservation ethic and environment: the Koyukon of Alaska. See Williams & Hunn 1982, pp. 211–28

Nelson RK. 1983. *Make Prayers to the Raven.* Chicago: Univ. Chicago Press

Neumann RP. 1998. *Imposing Wilderness: Struggles Over Livelihood and Nature Preservation in Africa.* Berkeley: Univ. Calif. Press

Noble IR, Dirzo R. 1997. Forests as human-dominated ecosystems. *Science* 277:522–25

Norgaard RB. 1995. Biodiversity: processes of loss. In *Encyclopedia of Environmental Biology*, 1:201–9. Orlando, FL: Academic

Olofson H. 1995. Taboo and environment, Cebuano and Tagbanuwa: two cases of indigenous management of natural resources in the Philippines and their relation to religion. *Philipp. Q. Cult. Soc.* 23(1):20–34

Olson M. 1965. *The Logic of Collective Action: Public Goods and the Theory of Groups.* Cambridge, MA: Harvard Univ. Press

Ostrom E. 1990. *Governing the Commons*: *The Evolution of Institutions for Collective Action.* Cambridge, UK: Cambridge Univ. Press

Ostrom E, Burger J, Field CB, Norgaard RB, Policansky D. 1999. Revisiting the commons: local lessons, global challenges. *Science* 284:278–82

Padoch C, Peluso NL, eds. 1996. *Borneo in Transition: People, Forests, Conservation, and Development.* Kuala Lumpur/New York: Oxford Univ. Press

Paine RT. 1966. Food web complexity and species diversity. *Am. Nat.* 100:65–76

Peluso NL. 1993. Coercing conservation? The politics of state resource control. *Glob. Environ. Change* 3:199–217

Peres CA. 2000. Effects of subsistence hunting on vertebrate community structure in Amazonian forests. *Conserv. Biol.* 14:240–53

Perry DA, Amaranthus MP. 1997. Disturbance, recovery, and stability. In *Creating a Forestry for the 21st Century: The Science of Ecosystem Management,* ed. KA Kohm, JF Franklin, pp. 31–56. Washington, DC: Island

Petraitis P, Latham R, Niesenbaum R. 1989. The maintenance of species diversity by disturbance. *Q. Rev. Biol.* 64:393–418

Pickett ST, White PS, eds. 1985. *The Ecology of Natural Disturbance and Patch Dynamics.* Orlando, FL: Academic

Pineda F. 1991. Conservación de la naturaleza y diversidad biológica: el caso de España. *Panda* 36:19–26

Polunin NVC. 1984. Do traditional marine “reserves” conserve? A view of Indonesian and New Guinean evidence. *Senri Ethnol. Stud.* 17:267–83

Posey DA. 1992. Interpreting and applying the “reality” of indigenous concepts: what is necessary to learn from the natives? In *Conservation in Neotropical Forests: Working from Traditional Resource Use,* ed. KH Redford, C Padoch, pp. 21–34. Irvington, NY: Columbia Univ. Press

Posey DA. 1998. Diachronic ecotones and anthropogenic landscapes: contesting the consciousness of conservation. See Balée 1998, pp. 104–18

Posey DA, Balée W, eds. 1989. *Advances in Economic Botany. Vol. 7. Resource Management in Amazonia.* New York: NY Botan. Gard.

Potts R. 1997. *Humanity’s Descent: The Consequences of Ecological Instability.* New York: Avon

Pyne SJ. 1998. Forged in fire: history, land, and anthropogenic fire. See Balée 1998, pp. 64–103

Pyne SJ. 1999. Attention all keepers of the flame. *Whole Earth Rev.* Winter: 1–2

Rambo AT. 1985. *Primitive Polluters: Semang Impact on the Malaysian Tropical Rainforest Ecosystem. Anthropol. Pap., Vol. 78.* Ann Arbor: Mus. Anthropol., Univ. Mich.

Reaka-Kudla ML, Wilson DE, Wilson EO, eds. 1996. *Biodiversity II: Understanding and Protecting Our Biological Resources.* Washington, DC: Joseph Henry/Natl. Acad.

Redford KH. 1991. The ecologically noble savage. *Orion* 9:24–29

Redford KH, Mansour JA, eds. 1996. *Traditional Peoples and Biodiversity Conservation in Large Tropical Landscapes.* Washington, DC: Nat. Conserv.

Redman CL. 1999. *Human Impact on Ancient Environments.* Tucson: Univ. Ariz. Press

Richards PW. 1996. *The Tropical Rainforest.* Cambridge, UK: Cambridge Univ. Press. 2nd ed.

Rogers AR. 1991. Conserving resources for children. *Hum. Nat.* 2:73–82

Rogers AR. 1994. The evolution of time preference by natural selection. *Am. Econ. Rev.* 84:460–81

Roosevelt AC. 1995. Educating natural scientists about the environment. *Pract. Anthropol.* 17:25–28

Russell EWB. 1997. *People and the Land Through Time: Linking Ecology and History.* New Haven, CT: Yale Univ. Press

Ruttan LM. 1998. Closing the commons: cooperation for gain or restraint? *Hum. Ecol.* 26(1):43–66

Ruttan LM, Borgerhoff Mulder M. 1999. Are East African pastoralists truly conservationists? *Curr. Anthropol.* 40:621–52

Scoones I. 1999. New ecology and the social sciences: what prospects for a fruitful engagement? *Ann. Rev. Anthropol.* 28:479–507

Simenstad CA, Estes JA, Kenyon KW. 1978. Aleuts, sea otters, and alternate stable-state communities. *Science* 200:403–11

Singleton S, Taylor M. 1992. Common property, collective action and community. *J. Theoret. Polit.* 4:309–24

Sivaramakrishnan K. 1999. *Modern Forests: Statemaking and Environmental Change in Colonial Eastern India.* Stanford, CA: Stanford Univ. Press

Slobodkin LB. 1968. How to be a predator. *Am. Zool.* 8:43–51

Smith EA. 1983. Anthropological applications of optimal foraging theory: a critical review. *Curr. Anthropol.* 24:625–51

Smith EA. 1984. Anthropology, evolutionary ecology, and the explanatory limitations of the ecosystem concept. In *The Ecosystem Concept in Anthropology*, ed. E Moran, pp. 51–85. Boulder, CO: Westview

Smith EA. 1991. *Inujjuamiut Foraging Strategies: Evolutionary Ecology of an Arctic Hunting Economy.* Hawthorne, NY: Gruyter

Smith EA. 1995. Comment on Alvard. *Curr. Anthropol.* 36:810–11

Smith EA, Smith SA. 1994. Inuit sex ratio variation: population control, ethnographic artifact, or parental manipulation? *Curr. Anthropol.* 35:595–624

Smith EA, Winterhalder B. 1992. Natural selection and decision-making: some fundamental principles. In *Evolutionary Ecology and Human Behavior*, ed. EA Smith, B Winterhalder, pp. 25–60. Hawthorne, NY: Gruyter

Smith NJH. 1980. Anthrosols and human carrying capacity in Amazonia. *Ann. Assoc. Am. Geogr.* 70:553–66

Soltis J, Boyd R, Richerson PJ. 1995. Can group-functional behaviors evolve by cultural group selection? An empirical test. *Curr. Anthropol.* 36:473–83

Soulé ME, Lease G, eds. 1995. *Reinventing Nature? Responses to Postmodern Deconstruction.* Washington, DC: Island

Sponsel LE. 1992. The environmental history of Amazonia: natural and human disturbances, and the ecological transition. In *Changing Tropical Forests: Historical Perspectives on Today's Challenges in Central and South America*, ed. HK Steen, RP Tucker, pp. 233–51. Durham, NC: Forest Hist. Soc.

Sponsel LE, Bailey RC, Headland TN. 1996a. Anthropological perspectives on the causes, consequences, and solutions of deforestation. See Sponsel et al 1996b, pp. 3–52

Sponsel LE, Headland TN, Bailey RC, eds. 1996b. *Tropical Deforestation: The Human Dimension.* New York: Columbia Univ. Press

Stahl PW. 1996. Holocene biodiversity: an archaeological perspective from the Americas. *Ann. Rev. Anthropol.* 25:105–26

Stearman AM. 1994. "Only slaves climb trees." Revisiting the myth of the ecologically noble savage in Amazonia. *Hum. Nat.* 5:339–57

Steward JH. 1933. Ethnography of the Owens Valley Paiute. *Univ. Calif. Publ. Am. Archaeol. Ethnol.* 33:233–350

Stewart OC. 1956. Fire as the first great force employed by man. In *Man's Role in Changing the Face of the Earth*, ed. WL Thomas, pp. 115–33. Chicago: Univ. Chicago Press

Swanson TM, ed. 1995. *The Economics and Ecology of Biodiversity Decline: The Forces Driving Global Change.* New York: Cambridge Univ. Press

Taylor M. 1987. *The Possibility of Cooperation.* Cambridge, UK: Cambridge Univ. Press

Uhl C, Clark K, Clark H. 1982. Successional patterns associated with slash-and-burn agriculture in the upper Rio Negro region of the Amazon Basin. *Biotropica* 14:249–54

Vecsey C. 1980. American Indian environmental religions. In *American Indian Environments: Ecological Issues in Native American History*, ed. CT Vecsey, RW Venables, pp. 1–37. Syracuse, NY: Syracuse Univ. Press

Vickers WT. 1991. Hunting yields and game composition over ten years in an Amazon Indian territory. In *Neotropical Wildlife Use and Conservation*, ed. JG Robinson, KH Redford, pp. 53–81. Chicago: Univ. Chicago Press

Vickers WT. 1994. From opportunism to nascent conservation: the case of the Siona-Secoya. *Hum. Nat.* 5:307–37

Vitousek PM, Mooney HA, Lubchenco J, Melillo JM. 1997. Human domination of the earth's ecosystems. *Science* 277:494–99

Wheat JB. 1972. *The Olsen-Chubbuk Site: A Paleo-Indian Bison Kill. Soc. Am. Archaeol. Mem. 26*. Washington, D.C.: Society for American Archaeology

Williams NM, Hunn ES, eds. 1982. *Resource Managers: North American and Australian Hunter-Gatherers.* Boulder, CO: Westview

Winterhalder B. 1981. Foraging strategies in the boreal environment: an analysis of Cree hunting and gathering. In *Hunter-Gatherer Foraging Strategies*, ed. B. Winterhalder, E Smith, pp. 66–98. Chicago: Univ. Chicago Press

Winterhalder BP. 1983. The boreal forest, Cree-Ojibwa foraging, and adaptive management. In *Resources and Dynamics of the Boreal*

Zone, ed. RW Wein, RR Riewe, IR Methven, pp. 331–45. Ottawa, Can.: Assoc. Can. Univ. North. Stud.

Winterhalder BP. 1994. Concepts in historical ecology: the view from evolutionary theory. In *Historical Ecology: Cultural Knowledge and Changing Landscapes*, ed. CL Crumley, pp. 17–42. Santa Fe, NM: Sch. Am. Res.

Winterhalder B, Lu F. 1997. A forager-resource population ecology model and implications for indigenous conservation. *Conserv. Biol.* 11:1354–64

Wolf ER. 1982. *Europe and the People Without History.* Berkeley: Univ. Calif. Press

Worster D. 1995. Nature and the disorder of history. See Soulé & Lease 1995, pp. 65–85

Wynne-Edwards VC. 1965. Self-regulating systems in populations of animals. *Science* 147:1543–48

Zavaleta E. 1999. The emergence of waterfowl conservation among Yup'ik hunters in the Yukon-Kuskokwim delta, Alaska. *Hum. Ecol.* 27(2):231–66

Zerner C. 1996. Telling stories about biodiversity. In *Valuing Local Knowledge: Indigenous Peoples and Intellectual Property Rights*, ed. SB Brush, D Stabinsky, pp. 68–101. Washington, DC: Island