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Community Forestry in Theory and Practice: Where Are We Now?*

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Key Words

community-based forest management, forests, forest communities, governance, livelihoods, sustainability

Abstract

Community forestry refers to forest management that has ecological sustainability and local community benefits as central goals, with some degree of responsibility and authority for forest management formally vested in the community. This review provides an overview of where the field of community forestry is today. We describe four case examples from the Americas: Canada, the United States, Mexico, and Bolivia. We also identify five hypotheses embedded in the concept of community forestry and examine the evidence supporting them. We conclude that community forestry holds promise as a viable approach to forest conservation and community development. Major gaps remain, however, between community forestry in theory and in practice. For example, devolution of forest management authority from states to communities has been partial and disappointing, and local control over forest management appears to have more ecological than socioeconomic benefits. We suggest ways that anthropologists can contribute to the field.

INTRODUCTION

Forests cover 30% of the earth's land area, totaling nearly four billion hectares (FAO 2006), but today represent about two-thirds of their historical expanse (Myers 1997) (**Table 1**). In the brief five-year interval between 2000 and 2005 scientists estimate an average annual net loss of 7.3 million hectares of forest occurred—5.8 million hectares (79.5%) of which were primary forest—mainly as a result of forest conversion to agriculture, with Africa and South America experiencing the greatest net losses (**Table 2**). Although deforestation rates have actually slowed since the 1990s, they remain a concern. Also worrisome are processes of forest degradation occurring within remaining forests, in many places resulting in simplification of forest structure, biodiversity loss, and alteration of forest ecosystem processes and functions. The causes of deforestation and forest degradation are many and complex (J.F. McCarthy 2006, Peluso 1992, Rudel 2005, Sponsel et al. 1996, Vandermeer & Perfecto 2005, Wood & Porro 2002).

Although reliable tallies of forest peoples are hard to come by, most estimates agree that 400–500 million people live in and around the world's forests, and many more—both rural and urban—depend on forest resources for cooking and heating (Arnold 1992, CIFOR 2006, Lynch & Talbott 1995, White & Martin 2002). Hunter-gatherers and shifting cultivators inhabiting forests derive their livelihoods directly from forests (Byron & Arnold 1999). People living in communities within and around forests use forest products for food, fuel, medicine, construction, fodder for livestock, and as a fallback when agricultural and other economic activities are inadequate to sustain the household economy. People also depend on the ecosystem services that forests provide, such as water for household uses and soil for agricultural production. In addition, many people depend on forests for income generation, for example, working in the wood products industry, developing small enterprises based on timber and nontimber forest products (NTFPs) (McLain & Jones 2002), or working as artisans (Byron & Arnold 1999). Finally, forests play an important role in peoples' sociocultural systems and profoundly influence their sense of place, ideologies, and identities. Some authors have noted a geographic correlation between the world's forests and the distribution of the world's rural poor (e.g., Sunderlin et al. 2005, Wunder 2001). Deforestation and forest degradation can exacerbate poverty among people who

Nontimber forest products (NTFPs): plants, lichens, and fungi used for food, medicine, floral greens, horticultural stocks, fiber, dye, oils, resins, and chemical extracts

Table 1   Global forest cover[a]. Source: FAO 2006

Region	Total area (million ha)	Forest area (million ha)[a]	% of land area forested	% of global forest
Africa	3031	635.4	21.4	16.1
Asia	3177	571.6	18.5	14.5
Europe[b]	2298	1001.4	44.3	25.3
North and Central America	2273	705.8	32.9	17.9
Oceania	856	206.3	24.3	5.2
South America	1784	831.5	47.7	21.0
Global total	**13,418.5**	**3952**	**30.3**	**–**

[a]"Forest land" as reported here is defined as land spanning more than 0.5 hectares (ha) and having trees higher than 5 m with a canopy cover of more than 10%, or trees able to reach these thresholds in situ. Forest land does not include land that is predominantly under agricultural or urban use, "other wooded land," and "other land with tree cover," which together account for 11% of the earth's total land area.
[b]The Russian Federation is reported as part of Europe; of the 1001.4 million hectares of forest land in Europe, 809 million hectares (80.8%) lie within the Russian Federation.

Table 2 Annual changes in forest area, 1990–2005. Source: FAO 2006

Region	Net change 1990–2000 (1000 ha)	% change 1990–2000	Net change 2000–2005 (1000 ha)	% change 2000–2005
Africa	−4375	−0.64	−4040	−0.62
Asia	−792	−0.14	1003	0.18
Europe	877	0.09	661	0.07
North and Central America	−328	−0.05	−333	−0.05
Oceania	−448	−0.21	−356	−0.17
South America	−3802	−0.44	−4251	−0.50
Global total	**−8868**	**−0.22**	**−7317**	**−0.18**

depend on forest products and services to support their livelihoods (Chomitz 2007).

To date, anthropologists have made major contributions to our understanding of the sociocultural and ecological relations between people and forests (e.g., Moran & Ostrom 2005). For example, they have demonstrated that the distribution of forests and indigenous populations is correlated in some parts of the world (e.g., Chapin 2002) and noted that areas of high biodiversity overlap with places where traditional communities maintain control over resource management (Maffi 2005). They have documented the extensive ecological knowledge regarding forests that many local and indigenous populations maintain, and forest management practices that are ecologically sound (e.g., Carlson & Maffi 2004, Fairhead & Leach 1996, Smith & Wishnie 2000). They have also highlighted struggles over forest access, use, ownership, and control, and associated human rights issues (e.g., Haenn 2005, Nygren 2005, Sturgeon 2005; on access, see Ribot & Peluso 2003). Although some colonial and postcolonial governments have recognized and formalized the customary forest management practices and institutions maintained by forest communities, in most of the world's forests, these have been replaced by state bureaucracies that control forest management, with negative consequences for forest peoples (e.g., Peluso & Vandergeest 2001, Sivaramakrishnan 1999).

Concern over the problems confronting forests and forest peoples has, since the late 1970s, led to efforts to address them simultaneously through community-based forest management approaches in various guises. Today these approaches are generally labeled "community forestry." Definitions and terms for community forestry abound in the literature (see Glasmeier & Farrigan 2005 for a review), and the forms it takes on the ground vary widely. In this review we use the term to refer to forest management that has three characteristics. In community forestry, (*a*) some degree of responsibility and authority for forest management is formally vested by the government in local communities; (*b*) a central objective of forest management is to provide local communities with social and economic benefits from forests; and (*c*) ecologically sustainable forest use is a central management goal, with forest communities taking some responsibility for maintaining and restoring forest health. By forest communities and forest peoples we mean communities and peoples that have social, cultural, and economic ties to nearby forests, recognizing that "community" is a debated concept that can be defined in multiple ways. Our emphasis in this review of community forestry as a field is on government-sanctioned, de jure forms of forest management by communities, rather than on customary or de facto forms of forest use and management by them. We do not address agroforestry, a form of resource management in which trees and other large woody plants are integrated into farming systems to produce increased social, economic, and

Access: the ability to derive benefit from things (in this case, forests), based in a "bundle of powers"

environmental benefits by diversifying production (see Schroth et al. 2004 for an overview of this topic).

The term community forestry has been used for community-based forest management initiatives associated with private forest lands (e.g., Best & Wayburn 2001), forests held as common property (e.g., Bray et al. 2005), and forests on indigenous peoples' lands (e.g., LTC/IES 1995). Here we focus on community forestry associated with state lands (also commonly referred to as government-owned lands or public lands). We also include some examples from other ownerships where communities and governments share management authority. Official reports of forest ownership across the spectrum of private, common, and state are limited by a lack of information and are often complicated by contested, overlapping, and unrecognized claims (White & Martin 2002). Nevertheless, public forests are estimated to comprise more than 80% of the world's forests (**Table 3**) (FAO 2006), and communal ownership of forest land appears to be more common in developing countries than in developed countries (**Table 4**) (White & Martin 2002).

Our purpose is to take stock of the field of community forestry both in theory and in practice some 30 years after its emergence.

We begin by tracing a brief history of community forestry. Then we look at how community forestry is currently being practiced in selected case examples from the Americas. The number of eligible cases is vast in both publications and in the gray literature; we focus on examples from the published, peer-reviewed literature. Next we identify a set of hypotheses underlying the concept and practice of community forestry, examining the evidence supporting these hypotheses to test whether community forestry is a viable approach to forest conservation and community development. We conclude by summarizing where the field of community forestry is now, exploring future directions, and suggesting ways that anthropology can better engage with and contribute to the field.

COMMUNITY FORESTRY: A BRIEF HISTORY

Anthropologists and other social scientists have documented many ways in which indigenous and local peoples have sustainably managed forests in the past through techniques such as practicing swidden agriculture for crop production (Conklin 1957, Denevan & Padoch 1988, Dove 1985); practicing succession management to produce food,

Table 3 Forest ownership 2000. Source: FAO 2006

Region	Forest area[a] (1000 ha)	% Total forested area shown in Table 1	Private (1000 ha)	Public (1000 ha)[b]	Other (1000 ha)
Africa	552,326	84.3	9951 (2%)	539,248 (98%)	3127 (<1%)
Asia	566,388	100	28,329 (5%)	534,845 (94%)	3214 (<1%)
Europe	998,071	100	99,631 (10%)	897,059 (90%)	1380 (<1%)
North & Central America	698,285	98.7	208,525 (30%)	462,477 (66%)	27,284 (4%)
Oceania	204,933	98.5	48,575 (24%)	125,527 (61%)	30,831 (15%)
South America	136,240	16	23,528 (17%)	103,379 (76%)	9333 (7%)
Global Total	**3,156,243**	**79.1**	**418,538 (13%)**	**2,662,534 (84.5%)**	**75,170 (2.5%)**

[a]Not all countries reported figures for forest ownership; thus, data are in relation to the total forested area of those reporting regions.
[b]Public ownership category includes government-owned and -managed forests and forests set aside for communities but ultimately owned by governments.

Table 4   Distribution of forest ownership (expressed in percent of total forest)[a]. Source: White & Martin 2002

Category	Public[b]		Private[c]	
	Administered by government	Reserved for community and indigenous groups	Community/ indigenous	Individual/ firm
Developing countries	71%	8%	14%	7%
Developed countries	81%	1%	2%	16%
Global total	**77%**	**4%**	**7%**	**12%**

[a]Low-end estimates extrapolated from 24 countries of the top 30 forested countries. Data are derived from government reports on legal tenure and do not account for claims made by forest peoples without official sanction.
[b]Public ownership is defined as all lands owned by central, regional, or local governments; administration of public lands may be undertaken by government entities or local communities, including indigenous groups, on a semipermanent, but conditional basis.
[c]Private ownership is defined as rights over a specific area that cannot unilaterally be terminated by a government without due process and compensation; private rights (of both individuals and groups) are generally more secure and include rights to access, sell, manage, and exclude others.

construction materials, medicinal plants, and other desired species (Irvine 1989); burning to enhance the distribution and populations of plant and animal species important for hunting and gathering (Boyd 1999, Vale 2002); and planting and enriching soil to create forest patches (Fairhead & Leach 1996, Posey 1985). As European powers established colonial rule beginning in the sixteenth century, they appropriated much of the forest estate in countries around the globe, claiming it as state property and altering and often undermining customary forest tenure (Ribot & Peluso 2003) and management systems (e.g., Poffenberger 1999, 2000). Commercial timber extraction was largely unregulated until the nineteenth century, when colonial governments began to establish centralized, bureaucratic forest departments to implement "modern," scientifically based approaches to forestry following the European model (Guha 2000). Since World War II, industrial-scale timber extraction conducted by states and private-sector timber companies having logging concessions or other forms of access to state lands has been the dominant form of forestry practiced on state forest lands in many tropical and temperate regions.

Community forestry emerged in different places between the 1970s and 1990s as a response to different combinations of factors, but key drivers have been deforestation and forest degradation occurring as a result of decades of overexploitation from industrial logging (Poffenberger 2006); collective action and protest by local communities that have challenged centralized, bureaucratic forest governance structures and destructive resource extraction practices, often tied to broader national struggles for democratization and resource access (Bray 1991, Guha 1989, Rangan 2000); recognition that many state governments do not have the resources to enforce forest management laws and regulations and require assistance in carrying out forest management responsibilities (Wily & Mbaya 2001); pressure on national governments to address rural poverty and social inequality on the part of intergovernmental organizations such as the World Bank, United Nations Forum on Forests, and the Food and Agriculture Organization (Arnold 1992); resistance to the top-down approaches to development assistance, practiced in the 1960s and 1970s, that were seen by communities as unjust and irrelevant and a push for more decentralized, bottom-up approaches to development (Chambers 1983); and the availability of financial and technical assistance from international development agencies, foundations,

Tenure: relations of resource ownership and control sanctioned by social institutions

banks, and nongovernmental organizations to support community forestry worldwide (Colfer & Capistrano 2005). As such, the rise of community forestry can be viewed as part of a larger movement toward community-based natural resource management and conservation that began in developing countries in the 1980s and in developed countries during the 1990s (Borgerhoff Mulder & Coppolillo 2005, Brosius et al. 2005, Western et al. 1994), which was itself a response to similar ecological and social factors.

Some of the earliest examples of community forestry come from Asia, where they grew out of social forestry efforts that began in the 1970s (Arnold 1992, Glasmeier & Farrigan 2005, Pardo 1995). Asia's social forestry programs were a response on the part of state governments and international donors to the problems of deforestation and the ecological effects of industrial forestry, combined with growing rural populations who faced hardships obtaining fuel wood and other forest products. These programs generally entailed planting vast numbers of mostly fast-growing, exotic trees in wood lots, on plantations, and on farms to increase the supply of fuel wood, fodder, and small-diameter wood for timber, pulpwood, and construction to better meet the needs of forest communities. This strategy also aimed to relieve pressure on natural forests, which would be managed for conservation and industrial timber production, consistent with state interests (Poffenberger & McGean 1996). Social forestry programs were largely unsuccessful at relieving pressure on natural forests. Worse, they did not meet peoples' needs for a broad range of forest products and services; they did not address controversial issues of forest access, property rights, and governance; and they did little to alleviate rural poverty among forest peoples (Poffenberger 1999, 2000). Community forestry emerged as an alternative approach to addressing these needs. Nepal, the Philippines, and India were pioneers in the community forestry movement in Asia.

In Nepal, government legislation in 1978 formally recognized the rights of villagers to manage some government forest lands through territorially based political institutions called Panchayat (Acharya 2002). In 1993 a Forest Act was passed, which together with subsequent regulations and policies facilitated the transfer of forest use rights and management authority over state forest lands to local forest users. Community Forest User Groups were established and legally authorized to make forest management decisions. By the early 2000s, nearly one-quarter of the potential community forest area in Nepal was being managed by Community Forest User Groups (Acharya 2002).

In the Philippines, social forestry became the official government approach to forest conservation and development in 1982 (Donoghue et al. 2003). In 1989, a Community-Based Forest Management program was launched that proposed transferring forest management responsibility from the state to local communities. The Philippines adopted a community forestry program by executive order in 1995. Under the program, geographically defined communities obtain 25-year renewable leases on state forest lands previously held under timber license agreements by international timber companies. Community groups are responsible for forest management and protection in the area under leasehold, and they have the right to harvest, process, and sell forest products from these areas once management plans have been approved by the government. Income generated is reinvested in forest conservation and community development projects (Donoghue et al. 2003). More than one-third of the state forest land in the Philippines has been delegated to communities for management, a figure much higher than that of other southeast Asian countries (Poffenberger 2006).

India formally instituted Joint Forest Management in 1990, creating an opportunity for forest communities to participate in managing state forest lands through joint forest

management agreements (Poffenberger & McGean 1996). Under Joint Forest Management, state forest departments work in collaborative partnership with local village Forest Protection Committees to manage state forests, supervising their activities. Communities share some of the timber harvest and obtain other forest benefits and participate in forest planning, management, and protection (Sundar 2000). Since the early 1990s, 85,000 village committees have been formed that protect more than 17.3 million hectares of forest land, representing 27% of India's forests. The program is estimated to encompass 8.3 million families, half of which are scheduled castes and tribes (World Bank 2006).

In Africa, community forestry initiatives are relatively new; most began emerging in the 1990s to address problems of forest loss and degradation (Wily & Mbaya 2001). Wily & Mbaya (2001) characterize two forms of community forestry that exist in eastern and southern Africa: the benefit-sharing paradigm and the power-sharing paradigm. The benefit-sharing paradigm entails giving forest peoples access rights to forests; sharing the benefits derived from forests (mainly revenue generated from forest products, especially wildlife, or jobs associated with exploiting those products); and/or making local investments in community development projects. Community involvement in forest management is generally limited to assisting with protection functions, such as patrolling. The goal is to obtain the cooperation of forest peoples in state-dominated forest management schemes. The benefit-sharing approach prevails in southern Africa. The power-sharing paradigm involves forest communities as managers and entails devolving some degree of forest management authority to local forest users, and then deciding how the forest should be used. The goal is to provide incentive for communities to engage in sustainable forest use and management, given their strong vested interests in forest resources. This approach is more recent; is fairly well developed in Tanzania and Lesotho; and is emerging in Namibia, Malawi, and Uganda (Wily & Mbaya 2001).

Jeanrenaud (2001) and Wiersum et al. (2004) discuss the evolution of community forestry in Europe. In Australia, community forestry is a new concept and literature on the topic is limited (Bellinger & Gale 2004). Community forestry in the Americas is discussed in detail below.

COMMUNITY FORESTRY IN PRACTICE: EXAMPLES FROM THE AMERICAS

Community forestry initiatives show wide global variation. As with other community-based natural resource management and conservation initiatives, they are context-specific, making it difficult to generalize about them or to develop typologies (Brosius et al. 2005, Glasmeier & Farrigan 2005). Here we review selected examples of community forestry from the Americas (Canada, the United States, Mexico, and Bolivia) to give a better feel for what it looks like in practice. Owing to the limited space available for this review, we focus on the Americas because (*a*) there is little in the way of published literature that compares and contrasts the current status of community forestry in the Americas as a region; (*b*) examples from the Americas illustrate a broad range of forms that community forestry can take, from that in which formal devolution (Edmunds & Wollenberg 2003, Ribot et al. 2006) of forest ownership, responsibility, and management authority has been quite limited (the United States) to that in which it has been extreme (Mexico); and (*c*) we are most familiar with the literature from this region. Ferroukhi (2003) and Larson (2003) provide overviews of community forestry in Latin America. For more on specific Latin American countries see Elias & Wittman (2005) and Ferroukhi & Echeverria (2003) (Guatemala), Nygren (2005) and Tucker (1999) (Honduras), Larson (2002) (Nicaragua), and Toni (2003) (Brazil).

Devolution: transfer of power and assets to nonstate bodies (e.g., citizens, forest user organizations) not created or controlled by the state

Canada

Nearly half of Canada is forested (Beckley 2003), 94% of this forest land being state (Crown) land, with 71% owned by provinces and 23% owned by the federal government (Bull & Schwab 2005). The dominant forest management paradigm on Crown lands has been industrial timber production through long-term leases to private forest products companies that have their own mills (Beckley 1998). Under this model, forest management decision-making power rests largely with forest industry and provincial and federal governments.

In the 1990s public critique of and controversy over this model and its ecological consequences led to experimentation with alternative approaches to forest management. Some of these approaches focused on increasing public participation in the management process, with a shift to socially, economically, and ecologically sustainable forest management as a priority. Examples are Canada's Model Forest Program, initiated in 1992 (Bull & Schwab 2005, Duinker et al. 2003), and forest industry stakeholder advisory committees (McGurk et al. 2006), which represented significant advances in public participation in forest management. Other approaches include Crown forest comanagement by communities and private industry (Bull & Schwab 2005, Chambers 2004) and community forests as a new form of tenure on Crown land (although some community forests date to the 1940s) (Teitelbaum et al. 2006). In Canada, comanagement occurs when local forest users share forest management power and responsibility with the government, industrial leaseholders, or forest owners (Beckley 1998). Forest comanagement by communities and the government exists in only a few cases in Canada (Beckley 1998).

Community forests have been established in Ontario, Quebec, and British Columbia (see Teitelbaum et al. 2006 for a review). In the Canadian context, a community forest can be defined as a "public forest area managed by the community as a working forest for the benefit of the community" (Teitelbaum et al. 2006, p. 417). British Columbia's Community Forest Agreement Program provides one example. Established as a pilot program in 1998 and becoming official in 2004, it gives communities tenure over defined areas of Crown land in a manner analogous to the leasing of Crown forests to private companies (J. McCarthy 2006). Under a community forest agreement, the forest remains in state ownership, but communities obtain significant management rights and responsibilities. After an initial five-year probation period the community can apply for tenure lasting 25–99 years. Community forests range from a few hundred to nearly 61,000 hectares in size (J. McCarthy 2006).

Communities as forest management institutions must be legal entities and place based. They have included First Nations, municipal governments, environmental nonprofit organizations, and local societies and cooperatives (J. McCarthy 2006), and their objectives vary from place to place. Community benefits from the program include greater control over forest management through development of management plans, decision-making authority over timber harvesting, more local jobs in the wood products industry, and revenues from timber production. The provincial government retains substantial control over forest management, however. Whether community forests should and do adhere to higher ecological standards than does the forest industry in timber production has been a subject of debate. To date, community forests occupy ~1% of the forest land in British Columbia and produce ~1% of the timber volume there, but are rapidly increasing in number (J. McCarthy 2006).

United States

Forest land covers 33% of the United States. Of this percentage, 43% is in government ownership and managed mainly by the U.S.

Forest Service (Smith et al. 2004). Community forestry initiatives in the United States have occurred in association with private (Communities Committee 2005), tribal (IFMAT 2003, Poffenberger 1998), and public (Kusel & Adler 2003, McCullough 1995) forest lands. Here we focus on community forestry associated with federal lands in the western United States.

Industrial timber production based on principles of sustained yield dominated public lands forestry in the United States from the 1940s through the 1980s, with professional foresters managing national forests with little citizen input, and private forest products companies gaining access to timber by bidding on timber sales (MacCleery & Le Master 1999). By the 1980s a shift in public values regarding forests, citizen concern over the environmental consequences of industrial timber production on public lands, and a call to increase public participation in federal forest management led to intense conflict and a series of lawsuits that ultimately brought about a paradigm shift in federal forest management in the 1990s. The results were (*a*) a dramatic cutback in timber harvesting on public lands (MacCleery & Le Master 1999); (*b*) "ecosystem management" as a new approach to forest management, which emphasized ecosystem conditions and biodiversity conservation (Cortner et al. 1999); and (*c*) a flurry of attempts to improve collaborative forest management processes (Conley & Moote 2003, Wondolleck & Yaffee 2000). This shift also created the space for experimentation with community forestry approaches. Community forestry associated with public forest lands arose largely in response to the decline in timber harvesting and timber industry restructuring, both of which caused job loss and industry divestment in many forest communities.

There has been widespread acceptance of the notion of increasing public participation in federal forest management in the United States through collaborative mechanisms as part of the trend toward democratizing environmental decision-making that has spread since the 1970s (most recently manifested in the 2005 Forest Planning rules). However, there has been resistance to, and lack of political support for, giving communities tenure rights to public forests on the part of government and environmental groups. Constraints include legal barriers, fear that community control will cause forest degradation, and concern that community management will favor local over national interests in public lands. This resistance, coupled with insufficient grassroots pressure from below, has resulted in little meaningful devolution of forest management to communities to date.

Instead, community forestry initiatives have relied heavily on collaborative partnerships between forest management agencies and local community groups to achieve the goals of conserving and restoring forest ecosystem health in a manner that simultaneously improves the well-being of forest communities (Baker & Kusel 2003, Brendler & Carey 1998, Gray et al. 2001, Wyckoff-Baird 2005). These collaborations have typically sought to link investments in forest stewardship on public lands with economic opportunities for forest communities. Some examples include creating forest-based jobs in restoration, conducting timber harvesting to achieve desired ecological conditions, supporting small value-added processing businesses (e.g., furniture making), developing uses and markets for restoration byproducts (e.g., small-diameter wood and biomass), increasing opportunities for NTFP production, and establishing markets for ecosystem services. Additional goals are local empowerment, community capacity building, and more equitable benefit sharing (Baker & Kusel 2003, Wyckoff-Baird 2005).

Cases of community forestry associated with public forest lands in the United States are described in Colburn (2002), Danks (2000, 2003), Kusel & Adler (2003), Wilson (2006), and Wyckoff-Baird (2005). Many innovative attempts at community forestry are being tried, but there is still a long way to go. The federal government still retains final

Ejido: Mexican common property system in which groups of individuals have legal rights to a given piece of land

Agrarian community: Mexican common property system based on lands granted to communities by the Spanish Crown, often correlated with ancestral lands

decision-making authority over federal forest management; devolution of authority to communities to manage federal forests has rarely occurred (but see London 2002); and it is often illegal to privilege local communities and workers in hiring, contracting, and timber sales, except under limited legal authorities in specific contexts.

Nevertheless, collaborative forest management has increased, partnerships between diverse community stakeholders and land management agencies have been built, some new forest restoration jobs have been created, some value-added businesses have been successful, and markets for restoration by-products are expanding. Little information is available regarding the impact of community forestry on poverty alleviation, social equity, and forest conservation and restoration in the United States (Glasmeier & Farrigan 2005). It is perhaps best viewed as an evolving movement for conservation and community development that is gaining political and economic momentum, holding hope for forest communities as they adapt to changing productive relations with forests and proactively take advantage of new opportunities associated with changing federal forest management paradigms.

Mexico

Mexico has 64.2 million hectares of forest land (more than 33% of the total land area), over half of which is primary forest (FAO 2006). An estimated 12 million people and 8000 communities live in and around Mexico's forests and depend on them for subsistence and cash income (Klooster 2003). Sixty to 80% of Mexico's forest land is located within common property regimes (Antinori & Bray 2005, Klooster 2003), and community forestry is the dominant mode of timber production in Mexico. In fact, Mexico has the largest sector of community-managed common property forests dedicated to timber production in the world, with ~2400 timber-producing community forests (Bray et al. 2006). Twelve

percent of the legal wood harvest in Mexico is currently certified as environmentally sustainable (Klooster 2006).

Policies of property ownership and forest management that provide the institutional basis for community forestry in Mexico are rooted in a dynamic history of peasant-state relations. Most of Mexico's forest peoples reside in *ejidos* or *comunidades agrarias* (agrarian communities), where members have rights to a combination of individual plots and communal holdings. Ejidos—comprised of agricultural lands and forests—were created for landless peasants following Mexico's postrevolution agrarian reform in 1917. Mexico's entrance into the North American Free Trade Agreement in 1992 ended the subsequent creation of new ejidos. Since then, existing ejidos have been undergoing a process of land titling, transferring a system of state property with community usufruct rights into secure common and individual private property (Klooster 2003, Merino Perez 2001). Agrarian communities hold common property with secure tenure rights based on prerevolution [1910] land grants and are often, but not always, composed of indigenous people. Forests in both ejidos and agrarian communities cannot be transferred to individual private ownership; if an ejido were to dissolve (agrarian communities cannot dissolve), the forests would convert to state property. Individual forest ownership is relatively uncommon in Mexico (Klooster 2003, 2006; Taylor 2003). Ejidos and agrarian communities (hereafter referred to as communities) have communal governance structures (largely consistent with traditional governance) that form the basis of community forestry enterprises (Bray et al. 2006).

Despite strong communal forest tenure, timber production on community forests was controlled and operated by the central government and forest industry concession holders from the 1940s until the 1970s through a system that brought little benefit to communities (Klooster 2003). Grassroots protests and legal reforms led to the devolution of forest management and production in the 1970s

and 1980s and ushered in community forestry enterprises. Today, communities are heavily involved in all logging operations, some communities have their own sawmills, and others have created value-added processing facilities such as furniture production (Antinori & Bray 2005, Klooster 2003). De jure rights to forest management not withstanding, community forestry in Mexico today is best seen as a form of comanagement. Commercial timber and NTFP production on communal lands is subject to environmental regulations and forest use authorizations controlled by the government, which stipulate the species, quantity, method, and location of harvest (Bray et al. 2006, Mathews 2006).

Although Mexico's community forests are heralded as a successful world-class model of community forestry, challenges remain. These include problems of internal (community) conflict, corruption, covert privatization (e.g., illegal and clandestine logging by a group of community members), heavy and unstable regulatory frameworks, and distrust between communities (Bray et al. 2006). In addition, neoliberal restructuring since the 1990s has brought new pressures to privatize the forestry sector; government funding favors creating plantation forests through joint ventures between communities and commercial forestry companies on ejido lands, over community forestry (Klooster 2003). Despite these challenges, community forestry in Mexico has shown remarkable resilience in the face of ongoing forestry and agrarian policy shifts and will likely continue in the vanguard of community forest management (Mathews 2006).

Bolivia

More than half of Bolivia is forested (58.7 million hectares), and 20% of Bolivia's forests have been designated for conservation (FAO 2006). Up until the mid-1990s the state claimed ownership rights to all forested areas, but much of the valuable forest land was held under concession by private timber companies (Pacheco 2005). Commercial extractive activities in Bolivia's forests throughout the past century have focused on species such as mahogany (*Swietenia* spp.), Brazil nuts (*Bertholletia excelsa*), and rubber (*Hevea* spp.). Access to these economically valuable forest products was limited largely to powerful elites through the *barraca* system (forest concessions to private entrepreneurs) (de Jong et al. 2006, McDaniel 2003, Pacheco 2005). Growing concerns over forest sustainability, democratization of forest management, neoliberal reforms, and demands from peasant and indigenous groups have prompted a major overhaul of Bolivia's land and forest tenure regimes and management institutions over the past ten years (de Jong et al. 2006, Larson 2003), representing one of the most extensive efforts in Latin America to decentralize forest management (Pacheco 2005).

Two reforms have been crucial to Bolivia's decentralization (Ribot et al. 2006) and devolution efforts: the transfer of forest management to municipal governments (to which local residents can be elected), and the granting to indigenous communities of exclusive access to forest resources within their territories. Both have laid the groundwork for a shift toward community forestry (Larson 2005). In the former case, state forest lands have been decentralized to municipal governments, and municipal forest reserve areas have been created. Local user associations (such as small logging firms, peasant/farmer organizations, or any interested group) can request and be assigned access to forest resources in municipal forest reserve areas through concessions, contingent on approved forest management plans (de Jong et al. 2006, Larson 2003). Up to 20% of Bolivia's state forest lands can be transferred to municipal governments. In the latter case, Bolivian state forests become indigenous territories, representing devolution of ownership and giving indigenous peoples exclusive access to forest resources. In both cases, the central government retains the power to approve forest management plans for extractive uses; noncommercial forest uses

Decentralization: partial or complete transfer of power and assets from central states to lower-level branches of government or representative local institutions

do not require authorization. In 2002, ~1 million hectares of Bolivia's forest land had been approved for community forest management by indigenous groups, small logging firms, and other local forest users (Pacheco 2005). The remaining 80% of Bolivia's forests are national forests under central government authority (Larson 2003). In the remainder of this section we focus on municipal forests.

Prior to decentralization, municipal governments had limited discretionary power over forests and limited financial resources or other incentives to manage them. Today, municipal governments are responsible for managing municipal forest reserve areas. Municipalities propose areas to be protected as forest reserves, oversee activities that take place inside of municipal forest reserve areas, inventory forest resources, create forest and soil use plans, support and train local user groups, and promote local participation in forest management (Pacheco 2005). Municipal governments also have increased financial capacity to manage forests derived from forest resource use fees, property taxes, and national budget allocations (Pacheco 2005). Their power to make autonomous decisions is limited, however, because major decisions pertaining to rights, concessions, resource use, and taxation are reserved for the central government (Larson 2003).

To date, the socioeconomic outcomes of Bolivia's forest decentralization and devolution programs have been mixed. After ten years of legal reforms promoting community forestry, implementation on the ground has been slow (Larson 2003, Pacheco 2005). The creation of municipal forest reserve areas has been complicated by overlapping and uncertain rights to indigenous territories and protected areas, lack of information about the location of state forest lands, and conflicts between barraca entrepreneurs (former concession holders) and newly recognized indigenous and peasant communities (de Jong et al. 2006, Larson 2005). In addition, political patronage, political party dynamics, elite

stakeholders, and tenure conflicts still influence forest management (de Jong et al. 2006).

On the positive side, local people have legal access rights to forest resources that they did not have previously. They also have more opportunities to participate in forest management decision making. Municipal governments are more accountable to local community groups (Pacheco 2005), and some municipal funds derived from forestry are invested in local community development projects (Pacheco 2005). Capacity building among local groups has occurred (de Jong et al. 2006). And, where local people have accepted decentralization arrangements, property rights conflicts and illegal encroachment have diminished (Larson 2005). In addition, decentralization and devolution policies have promoted sustainable forest management (Pacheco 2005), with trends in management favoring certified forests (de Jong et al. 2006). Bolivia has become a model for forest certification in South America (McDaniel 2003).

COMMUNITY FORESTRY IN THEORY

As is apparent from the preceding examples of community forestry in practice, several hypotheses underlie the concept and efforts to implement it. These hypotheses hold that (*a*) a discrete "community" exists that can serve as the locus of community-based forest management; (*b*) devolution or decentralization of rights, responsibilities, and authority from the state to forest communities occurs to some extent; (*c*) forest utilization can occur in an ecologically sustainable way and be compatible with biodiversity conservation; (*d*) greater local control leads to healthier forests and more ecologically sustainable forest use; and (*e*) greater local control increases local community benefits associated with forests and forest management. In this section we examine these hypotheses more carefully and the evidence supporting them.

The Community

A key ingredient for the success of community forestry is said to be the presence of an identifiable community to which forest management authority can be devolved (Pardo 1995), suggesting that discrete communities exist and can serve as the locus of community-based forest management. The "community" in community forestry is defined differently in different cases. Most commonly the community is place based and geographically delineated, with local institutions functioning as the interface between community members and the state in forest management.

It is important to examine how the community as a unit is understood by community forestry policy makers and practitioners. "Community" has been portrayed in three main ways in the wider literature on community-based natural resource management and conservation, each of which is problematic: (*a*) community as a small spatial unit; (*b*) community as a homogenous social structure; and (*c*) community as shared norms (Agrawal & Gibson 1999).

The concept of community as a small or localized spatial unit can overlook important forest users. For example, there may be inconspicuous groups not recognized as members of forest communities, such as semi-nomadic or migratory people who harvest seasonal NTFPs (McLain 2001, McLain & Jones 1997). Others might be newcomers or immigrants who depend on and steward forest resources but are dismissed as community members by virtue of their ethnicity, family heritage, or failure to be "traditional" residents (Agrawal & Gibson 1999, Gupta & Ferguson 1997, Li 2000). Additional problems are that defining community boundaries for purposes of allocating forest use and management rights can be difficult and contentious (Edmunds & Wollenberg 2003). Focusing on community composition or shared space, therefore, may be misleading in terms of analyzing community forest management.

The concept of community as a homogenous social structure dovetails with ideas about its spatial boundaries and is reinforced when many community members share the same language, cultural practices, and ethnicity. Similarity within groups does not necessarily mean that resources will be managed in egalitarian ways. Social scientists caution against overlooking the heterogeneity and extant power hierarchies that exist within rural communities, which may serve to marginalize less powerful sectors of the community (Agrawal & Gibson 1999; Colfer 2005b; Leach et al. 1999; Li 1996, 2000; Mohan & Stokke 2000; Nygren 2005; Peluso et al. 1994). Actors experience varying degrees of inclusion and exclusion in community-based resource management because decision-making processes are not immune to social inequalities (Edmunds & Wollenberg 2003, Li 2000). One example can be found in Mexico where some individuals (disproportionately women) are not legally entitled rights holders despite being locally recognized as community members through kinship networks, residency, and other social factors (Bray et al. 2005). Identity politics—of gender, class, race, citizenship, age, etc.—within local communities coalesce in unique ways that empower some individuals and limit others. Community forestry can risk asserting primacy to local power elites, despite widespread celebration of its democratic principles (Rocheleau et al. 1996, Sundberg 2003, Zerner 2000).

Finally, the concept of community as shared norms and common interests suggests that all group members have similar values and sets of preferences. Although shared norms may facilitate resource conservation, these norms will not necessarily support conservation goals established elsewhere. Nor is it true that communities necessarily share common interests in resources (Nygren 2005). Women may use certain resources from living trees (such as leaves and resins) in cooking and medicinal applications, whereas men may want to cut down the same trees to sell

the wood (Peluso et al. 1994). Some residents may want forests managed for the so-called urban values of environmental preservation, whereas other residents see forests as containing resources to be exploited (Walker & Fortmann 2003). Such conflicts of interest must be addressed. Recent work emphasizes the importance of considering many values as a way to address diversity in community-based forestry (Anderson 2000, Menzies 2007, Wollenberg et al. 2005).

Although communities are rarely homogenous and conflict-free, a number of social scientists have documented instances where simplified community identities have been mobilized for important political objectives (e.g., Cromley 2005, Hale 2006, Li 2000). Harmonious, traditional, and egalitarian concepts of community, although idealized and romanticized, have sometimes had the practical political economic effect of helping people maintain claims to territory, resources, and subsistence livelihoods (Cederlof & Sivaramakrishnan 2006, Li 1996).

The discrete community that can serve as a tidy locus of community-based forest management rarely exists; nevertheless, communities and institutions that purportedly represent them will continue to be defined for purposes of community forestry. Whether and how these definitions are ascribed endogenously and exogenously matters. Here we emphasize the need for careful understanding of the social, political, and spatial relations and intricacies of communities as they affect the practice of community forestry.

Governance

An underlying hypothesis of community forestry is that some decentralization or devolution occurs from central governments to forest communities of rights, responsibilities, and authority related to forest management. We define decentralization as circumstances when centralized governments transfer some degree of forest management (e.g., administrative functions) to lower branches in a government bureaucracy, such as municipalities or other local institutions. Decentralization needs to be distinguished from devolution, which we define as circumstances when power (e.g., discretionary authority and forest access) is vested in either local authorities with downward accountability to local forest users or directly in nonstate forest user groups (Fisher 1999, Ribot et al. 2006). Decentralization and devolution can occur simultaneously, but these are distinct processes that imply different outcomes.

What governments decentralize or devolve varies by case and can include rights of access to forest products and the benefit streams they generate, the work of forest management such as forest protection and restoration functions, opportunities for a greater role and voice in forest management and planning, administrative functions, decision-making power and authority, and in some cases property rights, with forests transferring from government to community ownership. The purpose of the transfer is similarly variable, ranging from conflict reduction, more efficient completion of work, economic development and poverty alleviation, and ecological restoration, to democratization and formal ceding of expropriated lands and resources. Devolution or decentralization can occur via several mechanisms (Glasmeier & Farrigan 2005, Pardo 1995). For example, authority to manage a specific area of forest can be granted to a community (Wily 1999), or forest land can be leased to a community for production purposes (Gauld 2000). Alternatively, collaborative management, partnership, or co-management agreements between the state and communities can be established (Sundar 2000).

On-the-ground experiences with community forestry have often found that the policy mechanism for devolution is in place, but in reality it has only partially been realized. One is rather hard pressed to find cases in which government authorities have fully transferred control over forest use and management to

local communities (de Jong et al. 2006, Gauld 2000, Ribot et al. 2006, Sarin et al. 2003, Sundar 2000, Sunderlin et al. 2005). The transfer of secure forest ownership from states to communities is even more rare (White & Martin 2002). Another challenge occurs when forest management is decentralized to local governments without adequate resources to carry out their new responsibilities (Fisher 1999). Forest management authority in most cases ultimately remains in the hands of the state. In Asia for example, Fisher (1999, 2003) finds that the administrative functions associated with forest management are often decentralized, but devolution of power over forest resources and their management is extremely limited. Devolution of rights to use and manage nontimber forest products is much more common than devolution of rights to use and manage more commercially valuable timber resources (Menzies 2007). This failure to transfer power can produce tension when state promises fall short of community expectations. Some researchers suggest that devolution and decentralization policies may simply represent a change in the way central governments control forest management and forest peoples (Edmunds & Wollenberg 2003, Elias & Wittman 2005, Larson 2005). Other theorists advocate for a continued, if limited, role for central states in community forestry, with a shift from control to support functions (Ascher 1995, Shackleton et al. 2002). Examples of such functions include conflict mediation; law enforcement; provision of legal, technical, and financial assistance; support of local community capacity building; and defending of communities against powerful external interests.

Although full devolution and an equitable balance of power in forest decision making and control are rare, community forestry projects have sometimes successfully improved forest governance in other ways that highlight the processural dimensions of governance. Such improvements include improved rights of forest access and management responsibilities, increased participation in forest management decision making by people who previously had no voice, creation of mechanisms for management accountability, or increased visibility and recognition of the most marginalized forest constituents, providing ground on which to negotiate their interests in the future (Edmunds & Wollenberg 2003, Li 1996, Shackleton et al. 2002). As forest management devolution and decentralization policies are relatively new compared with central bureaucratic and industrial forestry, assessing outcomes and long-term patterns of social and political change related to governance is an ongoing research need.

Ecologically sustainable forestry: forestry that perpetuates ecosystem integrity while continuing to provide wood and nonwood values

Ecologically Sustainable Forestry

A fundamental hypothesis of community forestry is that forest utilization can occur in an ecologically sustainable way and be compatible with biodiversity conservation. Some scientists argue that biodiversity can be maintained only in the absence of human use (Oates 1999, Terborgh 1999). Others argue that biodiversity evolved in the context of human use and depends on it, evidenced by the fact that the world's most biodiverse regions are also the world's most culturally diverse regions (Anderson 2005, Deur & Turner 2005, Maffi 2005). In 2005 ~11% of the world's forests were designated for the conservation of biological diversity as their primary function (**Table 5**) (FAO 2006). Most of the world's forests still lie outside of protected areas, and some scientists stress that finding ways to use them sustainably may be the only way to save them (Hartshorn 1995, Johns 1997). If community forestry is to meet this challenge, it is important to examine the potential for timber and nontimber forest product harvesting to be conducted sustainably.

Timber Harvesting. The literature contains dissenting opinions regarding whether commercial timber production can occur in an ecologically sustainable way, and in a manner compatible with forest biodiversity conservation. If sustainable forestry is possible, it has

Table 5   Forest land managed for biodiversity protection as main function. Source: FAO 2006

Region	Total forest area[a] (1000 ha)	% of total forested area shown in Table 1	Forest area designated for conservation (1000 ha)	% of total forest area designated for conservation
Africa	455,129	71.6	74,585	16.4
Asia	571,567	100	71,541	12.5
Europe	991,192	99	37,776	3.8
North and Central America	703,364	99.6	88,927	12.6
Oceania	203,467	98.6	29,366	14.4
South America	831,540	100	119,742	14.4
Global total	**3,756,260**	**95**	**421,936**	**11.2**

[a]Not all countries reported figures on forest lands designated for biodiversity protection. Therefore, data are reported in relation to the total forested area of those reporting regions.

rarely occurred on an industrial scale (Bowles et al. 1998, Seymour & Hunter 1999). The reasons are many, but primary among them is the relative economic profitability of conventional compared with sustainable timber harvesting (Hartshorn 1995, Pearce et al. 2003, Putz et al. 2000).

Little is known about the direct impacts of timber harvesting on tropical moist forests (Vandermeer & Perfecto 2005). Our understanding of how tropical forests work remains limited (Bowles et al. 1998, Hartshorn 1995), most economically valuable timber species that occur there are not well understood, our knowledge of forest regeneration and recovery processes following logging is similarly small (Chazdon 1998), and undertaking forest restoration after logging is difficult in tropical forests because of high species diversity (Laurance & Bierregaard 1997). The impacts of logging depend on the intensity with which harvesting occurs and the nature of the extraction process (Bawa & Seidler 1998). Selective logging is the most common type of commercial logging in tropical moist forests (Vandermeer & Perfecto 2005). Most of the destructive impacts associated with it are a result of road building and damage to non-target trees caused by the extraction process (Johns 1997). Even a low level of tree removal can substantially alter forest structure, how-

ever, with negative effects on biodiversity at the landscape scale (Bawa & Seidler 1998). Few studies have examined the impact of logging on genetic diversity and ecosystem processes. Over the long term, logged tropical moist forests are likely to lose substantial biodiversity (Bawa & Seidler 1998).

Nevertheless, tropical moist forests are resilient, with ecological succession setting in following logging, resulting in forest regeneration (Chazdon 1998, Vandermeer & Perfecto 2005). How long it takes and what conditions are necessary for the recovery of high species diversity are the critical issues. Several scientists believe tropical moist forests can be managed for sustainable timber production, the limiting factors being social and economic rather than technical (Dawkins & Philip 1998, Johns 1997, Johnson & Cabarle 1993).

Regarding temperate forests, the North American and European models of industrial timber production have focused on sustained-yield harvesting based on two key concepts: conducting harvest rotations of trees when they are the age at which the average annual yield is maximized; and regulating forest structure so that it has equal areas of age classes up to the rotation age (Seymour & Hunter 1999). This approach lacks an ecosystem perspective, has rarely been implemented

at a sustainable level, and has had negative effects on biodiversity. Since the 1990s, sustained-yield timber production in temperate forests has been replaced in some areas by "ecological forestry." Ecological forestry entails manipulating forest ecosystems to maintain a full range of natural disturbance patterns that are mimicked through management activities (Seymour & Hunter 1999). Although ecological forestry may be better at maintaining biodiversity than earlier forestry methods, it is too soon to tell how it will affect biodiversity (Hansen 1997).

Forest certification emerged in the 1990s as a market-based mechanism to create financial incentive for more sustainable timber production (Vogt et al. 2000). In 2004 the amount of forest managed under the world's four main certification systems was estimated at 164 million hectares (∼4%) and accounted for more than 20% of global industrial roundwood production (MEA 2005). By early 2007, the amount of forest managed under certification had risen to 8.9% (D. Irvine, personal communication, 2007). Most certified forests are not tropical forests, however, where at least 50% of the world's biodiversity is found (Myers 1997).

The failure to find models of sustainable commercial timber production has spawned interest in small-scale, community-based forestry enterprises as an alternative approach. A number of strategies are being tried (e.g., Bray et al. 2005, Freese 1997, LTC/IES 1995, Primack et al. 1998, Zarin et al. 2004). We examine evidence of their success in the next section.

Nontimber Forest Products. NTFP extraction has often been thought of as being more ecologically sustainable and compatible with biodiversity conservation than is timber harvesting because it is often assumed to have little or no deleterious ecological impact (Belcher et al. 2005, Nepstad & Schwartzman 1992, Plotkin & Famolare 1992, Ticktin 2004). However, the ecological and biodiversity impacts of NTFP harvest-

ing are mostly unknown (Belcher et al. 2005). Studies of the ecological consequences of harvesting have focused on select life forms (trees, shrubs, and herbs) and plant parts (leaves, fruits, and seeds), with little information regarding other life forms (e.g., vines, lianas, bryophytes) and plant parts (e.g., roots, bulbs, bark, resin, branches) (Ticktin 2004).

The effects of commercial NTFP extraction depend on the way in which production is carried out (e.g., frequency and intensity of harvest, timing of harvest, size of harvested individuals, harvest technique) and the context in which harvesting takes place (Belcher et al. 2005). Low levels of extraction can have a low impact on biodiversity at the species and landscape scales. As harvesting intensity increases, so does the likelihood of negative ecological effects (Belcher et al. 2005). Without regulation, commercial NTFP extraction often leads to resource depletion (Kusters et al. 2006). The most direct effects of NTFP harvesting are changes in the rates of survival, growth, and reproduction of harvested individuals, which in turn can affect the structure and dynamics of species populations (Ticktin 2004). Less information is available about the effects of NTFP harvesting at the community and ecosystem levels, but significant harvest effects can occur including alteration of forest structure, composition, and regeneration and changes in plant-animal and plant-plant interactions (Ticktin 2004).

In the absence of an enforceable regulatory system, heavy harvest pressure often results in unsustainable harvest levels. A comparative analysis of 61 cases of commercial NTFP production from Africa, Asia, and Latin America found that most commercially harvested wild NTFP species were declining in number, with overexploitation at the species level (Belcher et al. 2005). The better the markets, pricing, infrastructure, and demand for a species were, the more likely increased harvesting and overexploitation were, especially where forest resources were open access. However, neutral or positive ecological effects from harvesting have been documented in cases where

Open access: the absence of defined property rights, whereby access to resources is uncontrolled and open to anyone, and resource use is unregulated

institutions that increase local control and management over forests and NTFP harvesting exist (Neumann & Hirsch 2000).

Some initiatives promote sustainable NTFP production. Brazil's extractive reserves—natural tropical forest areas deliberately set aside as reserves where people engage in the commercial production of certain renewable NTFPs—are one example (Allegretti 1990, Murrieta & Rueda 1995). In their review of the case-study literature on extractive reserves, Agrawal & Redford (2006) find that the evidence for their success at biodiversity conservation is insufficient to be conclusive. Positive conservation outcomes associated with extractive reserves have been documented, however (Ruiz-Perez et al. 2005). The potential for developing NTFP certification systems has been explored (Shanley et al. 2002), but to date NTFP certification is relatively rare.

Local Control and Ecological Sustainability

The preceding discussion indicates that the ecological effects of timber and nontimber forest product harvesting are influenced by the manner in which harvest is carried out and the presence of forest management institutions that regulate it. A hypothesis implicit in the community forestry literature is that greater local control over forest management will result in more ecologically sustainable forestry. This hypothesis is based in part on the observations that governments and private industry have generally done a poor job of managing forests (Colfer 2005a) and that national governments may not have the resources, or the political will to allocate the resources, needed to implement existing plans and enforce regulations (Ascher 1995). This hypothesis is also based on several assumptions about the relations between people and forests, summarized as follows. The people who live closest to and are most dependent on forests are more likely than are central states or corporate stakeholders to

manage forests sustainably because they have the greatest vested interest in them, and therefore the most incentive to do so (Wily & Mbaya 2001). In addition, people are more likely to take responsibility for the health of forest resources if they have a sense of ownership and control over them (Glasmeier & Farrigan 2005, Wily 1999). Local communities, being geographically close to forests, may also be able to carry out protection and rule-enforcement functions more cheaply (Wily 1999), although their ability to do so effectively may be problematic; local enforcement is subject to peoples' social, economic, and political interests. Finally, many forest peoples have traditional and local ecological knowledge about forests, and customary institutions for managing them, which have resulted in useful examples of sustainable management (Clay 1988, Colfer et al. 1997, Lawes et al. 2004, Posey & Balée 1989, Redford & Padoch 1992).

Although community-based management does not ensure conservation outcomes (Menzies 2007), there is evidence that supports the hypothesis that greater local control over forest management will result in more ecologically sustainable forestry. Satellite imagery and geographic information systems combined with more traditional social science research methods have made it possible for researchers to analyze how forest cover is correlated with different forest ownership categories and governance structures. For example, Gautam et al. (2002) examined how the amount of total forested area (defined as areas with at least 10% crown cover, referred to as "high forest," and shrubland), and the spatial extent of high forest, had changed between 1978 and 1992 in villages that had formally devolved management authority over community forests, and those that had not, in one watershed in Nepal. They found that total forested area had decreased in both categories of village but that it had decreased much more in villages that lacked community forestry than in those that had implemented community forestry policies. They also found

that high forest cover increased six times more in villages with community forestry than in those without as a result of plantation establishment and natural succession. They concluded that community forestry policies in Nepal have positive benefits for forest cover (Gautam et al. 2002).

Two studies from Latin America also demonstrate positive effects on forest cover associated with community-based forest management. Nepstad et al. (2006) show that indigenous lands in the Brazilian Amazon (managed by the indigenous peoples who live there) have been as effective at inhibiting deforestation as have uninhabited protected areas (parks), and they have experienced deforestation rates much lower than those of forested lands outside protected areas. This finding is particularly significant given that indigenous lands, unlike parks, tend to be located at the frontier of agricultural expansion where the pressure for deforestation is high. Bray et al. (2004) show that in ejidos in the Mayan zone of Quintana Roo, Mexico, where community forestry has been implemented and logging and farming are practiced, deforestation rates between 1984 and 2000 were almost imperceptible and were comparable with or lower than those of regions dominated by protected areas. They attribute this finding to the establishment of community-based forestry enterprises during this period, which entailed setting up permanent forest areas where logging would be conducted sustainably, and creating community-based forest stewardship institutions. Other areas of Mexico outside of protected areas that lacked community forestry institutions experienced much higher rates of deforestation.

An analysis of forest vegetation density (a proxy measure for forest condition) based on forest plot mensuration data from 163 forests located inside and outside protected areas found no significant difference in forest conditions (Hayes 2006). The most important factors correlated with vegetation density were the presence of forest product rules and the ability of forest users to make those rules,

which were more common in areas not legally protected. Other studies have produced similar findings regarding the importance of community involvement in forest management. Salafsky et al. (2001) found that more involvement by community members in the design and management of development projects linked to biodiversity conservation was correlated with greater ecological success. Colfer & Byron (2001) found that people who have a voice in forest management and use are more likely to embrace forest stewardship than are those who do not. A review of 28 conservation and development projects found that greater levels of community input into decision making were associated with better conservation outcomes (Brooks et al. 2006). Increasing community input did not guarantee ecological success, however; effective community-level institutions for regulating resource use were also needed. In their review of 69 case studies of community forestry, Pagdee et al. (2006) found that two variables important to success were (a) well-defined property rights over forests, and (b) strong community institutions for forest use and management. Both are more likely to exist where forest management authority and responsibility have been devolved to local communities. Other studies that document positive ecological effects from community forestry are Poffenberger (2006) for southeast Asia, Wily (1999) for Tanzania, and Nittler & Tschinkel (2005) for Guatemala.

Although this body of work supports the hypothesis that greater local control over forest management will result in more ecologically sustainable forestry, coarse-level assessments can obscure the nature of these ecological outcomes. For example, Edmunds & Wollenberg (2003) found that forest cover increased after forest management devolution policies were implemented in India, China, and the Phillipines. However, increased tree cover was the result of reforestation and afforestation projects in which timber and commercial NTFP species favored by the government were planted, at the expense of

species valued by forest communities and for biodiversity.

If devolving forest management to local communities increases the likelihood of conservation success, which local-level institutions for forest management are associated with the best ecological outcomes? Common-property theorists in particular have addressed this question at length (Agrawal 2001). They have found that institutional arrangements, rather than specific forms of property rights (public, private, common), are what is important for sustainable forest use (Ostrom & Nagendra 2006, Tucker & Ostrom 2005). The forms that local forest management institutions take are highly variable and necessarily context specific (Gibson et al. 2000). Whatever the form, to be effective they must have rules regarding who has access and use rights to forests, which forest products are harvested and when, harvest technologies, and forest guardianship; sanctions for rule breaking; and enforcement mechanisms (Gibson et al. 2000). When local forest users play a role in developing these rules, or consider the rules legitimate, they are much more likely to follow them and to participate in monitoring and enforcement (Ostrom & Nagendra 2006). In turn, increased participation can make local people stronger supporters of forest conservation (Agrawal 2005).

Local institutions for forest management are not the only local variable influencing forest conditions, however. Biophysical, economic, demographic, and socio-political variables also play a role in shaping the ecological outcomes of community-based forest management systems (Agrawal & Chhatre 2006). For these systems to be effective, the root causes of forest destruction and degradation must also be considered (Wolvekamp 1999). As Sikor (2006) notes, studies that adopt a local perspective on how people manage forests, including the relationship between local-level governance institutions and forest conditions, often fail to consider the influence of larger-scale political and economic forces. These forces can be more important than local institutions for forest management in shaping forest conditions (Tucker & Southworth 2005). Sikor (2006) reminds us to look at how local forest relations (including property, governance, and access) are influenced by both local-level processes and larger political and economic forces and at how these linkages affect ecological outcomes on the ground. Community forestry may not always be an appropriate response, or the only response needed, to achieve more ecologically sustainable forestry (see also Glasmeier & Farrigan 2005).

Local Control and Community Benefits

Not only do writings on community forestry hypothesize that local control over forest use and management will result in better ecological outcomes for forests, but also they suggest that local control will produce more social and economic benefits for forest communities (Colfer 2005a,b). The logic underlying this hypothesis is that (*a*) central governments are likely to prioritize national and private-industry interests over local interests in forest management (which often differ), whereas local communities are likely to prioritize their own interests; (*b*) local institutions can respond to local needs more efficiently and effectively than can central government institutions because they have more information about these needs, understand them better, and are accountable to local people (Ribot et al. 2006); and (*c*) local control creates more opportunity for marginalized groups to influence policy (Larson 2003).

What do the social and economic benefits of community forestry consist of? A review of 69 case studies of community forestry categorized these benefits as follows: more equitable sharing of forest management rights and responsibilities, more equitable access to and control over forest resources, more equitable distribution of forest benefits among community members, increased investment in the future productivity of forests, better

meeting of local peoples' needs for forest resources, improvement of the standard of living, alleviation of poverty, reduction of conflict between local communities and government authorities, control of corruption, resolution of forest mismanagement problems, and reduction of forest misuse by individuals (Pagdee et al. 2006). Another benefit is community capacity building; where communities organize to manage local forests, the skills they develop can increase their capacity to organize to address other areas of community life (Baker & Kusel 2003). Here we focus on the effectiveness of community forestry in helping people meet household subsistence needs for forest products, providing income generation and employment opportunities, promoting equitable distribution of forest benefits, and increasing democratic participation in forest management.

Assessments of the effectiveness of community forestry in providing community benefits are limited. In general, the equity and social welfare outcomes of conservation and development projects are difficult to evaluate (Saterson et al. 2004), a task complicated by a lack of conceptual clarity around key measures, such as poverty (Agrawal & Redford 2006). Monitoring and assessment to determine whether community forestry does more to contribute to social and economic well-being in forest communities than does state-controlled forest management have rarely been done. The limited evidence that does exist suggests that the community benefits of forest management decentralization and devolution policies have been mixed.

Meeting subsistence needs. The ability to maintain access to forest products is critical for poverty prevention (Wunder 2001). Forest products help households meet their subsistence needs, provide a safety net in times of emergency, and help fill seasonal economic gaps. Therefore, securing, increasing, or restoring access rights to forests is often a main objective of community forestry initiatives, at least from the perspective of forest

communities. Where the main desired benefit is increased access to forest resources to fulfill basic livelihood needs—common in many developing countries (Glasmeier & Farrigan 2005, Mallik & Rahman 1994)—local control is more likely to deliver than is government-dominated forestry (Ascher 1995). The devolution of forest management authority to local communities has been significantly correlated with increased tenure security over forest resources, but the devolution of forest management responsibility alone has not (Pagdee et al. 2006). Devolution also increases the likelihood that effective local-level institutions for regulating access to forests will develop, with positive ecological outcomes that can in turn increase livelihood security.

Exceptions exist, however. In Nepal, for instance, community forestry policies have caused poor households in particular to experience reduced access to forest products needed for subsistence because harvest regulations have become more stringent, and resource allocations are equally distributed between households without considering the different needs of rich and poor households (Malla 2000). Elsewhere, community forestry enterprises that emphasize the production of commercially valuable forest products may do so at the expense of managing forest products important for subsistence (Peluso et al. 1994). And in places where community forestry initiatives emphasize afforestation and reforestation, communities may not control decisions about which species to plant, resulting in plantations that do not contain species valued for community subsistence (Edmunds & Wollenberg 2003).

Income generation and employment. Both timber and nontimber forest products can make a significant contribution to income generation and employment in forest communities. Historically, forest peoples have seldom benefited from timber wealth because national governments and international markets favor industrial style and scale timber production by large private companies, and they

cannot compete under these conditions. Most people hired for jobs in the wood products industry are nonlocal; extracting timber is skill intensive, and local people often lack the required skills, especially those who are poor (Bliss & Bailey 2005, Sunderlin et al. 2005, Warner 2000). Moreover, employment in logging and primary processing is often short term and undergoes boom and bust cycles (Power 1996, Warner 2000). In addition, increased mechanization has caused timber production to become capital rather than labor intensive, meaning forestry employment is unlikely to be a source of jobs for many forest peoples (Wunder 2001). Plantation forestry also provides little in the way of jobs and income for local communities (Charnley 2005). Although there may be ways of redistributing profits made by forest industry to local communities, most forestry companies, states, and politicians are unlikely to want to equitably distribute these profits (Wunder 2001).

Findings regarding whether community forestry increases forest peoples' opportunities for income generation and employment associated with timber production vary by case. Often community forestry initiatives are associated with forests that have low timber value and are in poor ecological condition (Edmunds & Wollenberg 2003, Menzies 2007). States either do not devolve control over forests that have high commercial timber value (Ribot et al. 2006), or they devolve control after the most lucrative timber extraction has already occurred and other investors have moved on (Li 2005). Alternatively, community forestry policies emphasize environmental protection and prohibit commercial production from forests (Malla 2000) or devolve control only if local people agree to conserve and protect remaining forests (Li 2005).

In other cases, community forestry has facilitated income generation and employment associated with commercial timber production. Several attempts have been made to establish community forestry enterprises in which communities control timber harvesting and processing (Bray et al. 2005,

LTC/IES 1995, Primack et al. 1998, Zarin et al. 2004). Successful examples come from Mexico and Guatemala. The Mexican case was described earlier in this article. In the Maya Biosphere Reserve of Guatemala, 13 community forestry enterprises have concessions to manage roughly a half million hectares of forest land for timber production (Nittler & Tschinkel 2005). Although they required substantial financial and technical support from donor organizations and nongovernmental organizations to get established, and although they have experienced a number of problems, these enterprises are economically viable, are expected to generate some five million dollars in wood products sales annually, have created local jobs, and have provided a source of income for many families. Many have been certified by the Forest Stewardship Council (Nittler & Tschinkel 2005). Elsewhere, as in the United States, community forestry initiatives have emphasized developing local, value-added processing businesses and businesses that use wood that is otherwise uneconomical for the timber industry to extract (Wyckoff-Baird 2005). All these approaches hold promise but require secure access to timber or timberlands, capital investment, capacity building, markets, and market access. Moreover, the economic benefits generated are not distributed to everyone.

NTFPs make an important contribution to the subsistence economy of most forest peoples. In addition, a small subset of NTFPs are commercially traded in local, regional, and international markets and are especially valuable in circumstances where there are few alternate employment options. A review of 61 cases of commercial NTFP production from Asia, Africa, and Latin America (Belcher et al. 2005) found that commercial NTFP harvesters are typically quite poor and that the sale of commercial NTFPs generally contributes only a portion of the household income, which is usually small. This is partly due to the seasonal nature of NTFP gathering and partly because most profits go to middlemen that operate at positions along the marketing chain

in between harvesters and consumers. For most producers, commercial NTFPs are not a primary revenue-generating source, although they often provide important supplementary income (Neumann & Hirsch 2000, Thadani 2001). Although NTFPs can generate significant revenues, these revenues tend to accrue to individuals rather than to communities as a whole, to be unevenly distributed among community members, and may not be sustainable over the long term (Neumann & Hirsch 2000). An analysis of 55 cases of commercial NTFP production from Africa, Latin America, and Asia found that the better the livelihood outcomes were, the worse the environmental outcomes were (Kusters et al. 2006).

The context in which NTFP production occurs is important in influencing their economic contribution to rural households (Belcher et al. 2005, Neumann & Hirsch 2000). In cases where markets are good, a product is valuable, and land and resource tenure are secure, households may intensify management of, or cultivate, NTFPs, which can bring higher returns and incomes. In remote areas where there is poor infrastructure and poor market access, and forest resources are open access, competition for NTFPs may cause overexploitation, and low prices and low bargaining power are common. For NTFPs to generate more income and employment in forest communities, increased or more efficient commercial production and trade are needed (Belcher et al. 2005).

Have community forestry policies enhanced commercial NTFP production opportunities? The evidence is limited and mixed. In the Maya Biosphere Reserve of Guatemala, community forestry enterprises have promoted commercial NTFP harvesting (chicle, xate, allspice), and income from NTFPs has the potential to exceed that from timber (Nittler & Tschinkel 2005). In contrast, community forestry policies in Nepal prohibit people from using forest products from community forests for commercial purposes, causing poor people to lose income from NTFPs that they previously received (Malla 2000). A

review of the case literature on extractive reserves found inconclusive evidence regarding their success at alleviating poverty (Agrawal & Redford 2006).

Payments to protect ecosystem services [e.g., provision of biodiversity habitats, maintenance of clean water, soil protection, preservation of watershed functions, modulation of climate, mitigation of global warming (Myers 1997)] are other ways in which forests can be a source of revenue to communities. Conservation and development initiatives using market-based mechanisms for the protection of forest ecosystem services are emerging (de Jong et al. 2004, Pagiola et al. 2002), but to date, payments to protect ecosystem services are rare as a basis for community forestry projects. Nevertheless, paying people to protect environmental services may be an important strategy for community forestry initiatives that focus on forest protection and biodiversity conservation.

Equitable distribution of benefits. A common critique found in the literature on community-based natural resource management is that these initiatives can serve to reinforce existing power hierarchies and inequalities in communities rather than bring about social justice and more equitable sharing of social and economic benefits associated with natural resources (Brosius et al. 2005). This problem has been documented in the community forestry literature. For example, a study of forest management devolution policies in China, India, and the Philippines found that these policies provided more benefits to some forest users—such as better access to forest products for subsistence and income generation—but that most of these benefits were captured by local elites and non-local residents, and rarely by the poorest and most marginalized forest users (Edmunds & Wollenberg 2003).

One way to promote more equitable sharing of economic benefits is to make provisions for investing at least a portion of the profits generated by commercial forestry activities

in community development projects, rather than allowing them to accrue only to a subset of community members. Community forestry enterprises in Mexico and Guatemala have such provisions (Antinori & Bray 2005, Nittler & Tschinkel 2005), although they are not always implemented.

Democratization of forest management. Studies of whether community forestry has led to increased public participation in forest management, vested more decision-making power in local people, and brought about more bottom-up or downwardly accountable forest management report mixed results. In some cases, community members have gained a stronger voice in forest management and enjoy increased power and authority to manage local forests (e.g., Wily 1999). In the United States, community forestry has emphasized increased public participation in forest management, and this has been one of its main achievements.

In other cases, community forestry policies have increased rather than decreased state control over forest management and forest peoples. The Gambia provides one example, where responsibility for the work associated with forest management was devolved to forest communities, with access to forest resources made contingent on performance of these tasks as specified through contracts, and authority over forest management was retained by the state (Schroeder 1999). Another example comes from Guatemala's western highlands, where decentralization of forest management has increased state power and eroded local control over communal and municipal forests by establishing new forest governance institutions that override traditional and customary management practices and supplant historic claims to forest resources and territory, disempowering indigenous peoples (Elias & Wittman 2005, Sundberg 2003, Wittman & Geisler 2005).

In sum, the literature on community forestry provides mixed evidence regarding the hypothesis that devolving or decentralizing control over forest management leads to better social and economic outcomes for forest communities. To a large degree, its success depends on the extent to which devolution or decentralization has actually occurred in practice and what exactly has been devolved—e.g., rights of access to forest products, forest management tasks, decision-making power, or property rights. Other factors that contribute to success include the capacity of communities to create local institutions that are accountable and fairly represent the interests of all community members; the capacity of communities to represent their collective interests in wider social and political arenas; their technical capacity to manage forest resources; their capacity to participate in markets in an informed way; and their capacity to build community assets using benefits generated from forest management (Menzies 2007).

It may be too soon to evaluate fairly the social and economic outcomes of community forestry. Devolution/decentralization policies take time to implement, and bringing about change in forest governance is a slow process.

CONCLUSIONS: WHERE ARE WE NOW?

Differing national political and economic contexts, forest and forestry histories, and local relations to forests have caused community forestry to manifest differently in different countries and cases. Although it varies by place, community forestry shares the common goals of improving ecological conditions in forests and encouraging ecologically sustainable forest use practices; increasing social and economic benefits from forests to local communities; and increasing forest communities' access to and control over nearby forests. Community forestry is relatively young—beginning in the mid-1980s to 1990s in most countries—although forest communities have been using and managing forests for centuries, if not millennia. Outcomes remain inadequately documented, however.

On the one hand, community forestry has been accused of being another mechanism for increasing state control over forest communities and forests (on and off state lands) and for exploiting the cheap labor of forest peoples who are coerced into doing work that states do not have the money or resources to do, such as restoring forests these peoples did not degrade in the first place, as well as policing (Schroeder 1999, Sundar 2000). On the other hand, community forestry represents an important step for communities toward reclaiming access to and control over forests that were appropriated by colonial and postcolonial states and whose management has historically been dominated by central governments and the private industrial forestry sector, with little citizen input. In this regard, it represents a major paradigm shift in how state forest lands are managed and is significant in terms of the "politics of possibility" (J. McCarthy 2006) because it opens a door to increasing forest peoples' rights to forest resources.

Our review of community forestry in theory has several main findings. First, we need careful analysis of the social, political, and spatial relations within communities that are defined for purposes of community forestry initiatives and how these relations affect its practice.

Second, legal and policy rhetoric and mechanisms for decentralizing or devolving rights, responsibilities, and power often exist. In reality, however, decentralization and devolution have only partially been realized, with many states retaining significant authority over forest management. Without real devolution of power, the goals of community forestry will be difficult to achieve because they are premised on this transfer.

Third, whether ecologically sustainable forestry is possible from an ecological and social standpoint is still a matter of debate, clouded by scientific uncertainty. Studies have tried to assess whether biodiversity conservation and community development can be achieved simultaneously (e.g., Agrawal & Redford 2006, Bowles & Prickett 2001, Freese 1997, Kusters et al. 2006, McShane & Wells 2004, Naughton-Treves et al. 2005, Primack et al. 1998, Zarin et al. 2004), prompting the question of whether it is realistic to expect community forestry to help conserve forests and also produce social and economic benefits for forest peoples. An important finding that emerges from some of this work is that biodiversity conservation and community development have multiple dimensions, and trade-offs between specific aspects of one may be needed to achieve specific aspects of the other. Making trade-offs need not undermine the entire endeavor; communities and other stakeholders must consciously negotiate and choose which trade-offs to make.

Fourth, evidence demonstrates that local control over forest management on state and communal lands can have positive ecological outcomes where effective local-level institutions for forest management exist, especially when local people play a meaningful role in developing these institutions. These outcomes include reducing rates of deforestation, maintaining and increasing forest cover, and maintaining forest vegetation density. However, the social and economic benefits associated with local control over forest management have been mixed and unequally distributed, influenced by the degree to which devolution and decentralization have occurred and what has been devolved or decentralized.

We conclude that community forestry in theory holds promise as a viable approach to forest conservation and forest community development. Gaps remain, however, between community forestry in theory and in practice. Further research is needed to better understand the workings of "community" in community forestry projects, including who participates and who does not, why, and how power and benefits are distributed and captured at the community level. Also needed is a better understanding of how newly created institutions for forest management established

by state community forestry policies articulate with preexisting, customary institutions for forest management in communities, and how states and communities negotiate their interests in the decentralization and devolution process. In addition, we need more empirical evidence to sustain or reject the hypotheses outlined here. Better understanding of community forestry outcomes, and how the gaps between theory and practice can be closed, is still needed.

How can anthropologists contribute to this field? Anthropologists to date have contributed much to our understanding of related topics: (*a*) the underlying social, political, and economic causes of deforestation and its consequences for forest peoples (e.g., Painter & Durham 1995, Sponsel et al. 1996); (*b*) the extensive ethnoecological and ethnobotanical knowledge of forests maintained by forest peoples (e.g., Balee 1994, Carlson & Maffi 2004, Posey 2004); (*c*) the forest use and management practices of forest peoples, including customary institutions for forest management, and their ecological effects (e.g., Anderson 1990, Clay 1988, Fairhead & Leach 1996, Posey 2002, Posey & Balée 1989); (*d*) the effects of conservation and development interventions on communities (e.g., Brosius et al. 2005, Tsing 2005, West et al. 2006); (*e*) the internal dynamics and power relations at play within communities, including the micropolitics of resource access and control (e.g., Li 1996); and (*f*) culture and sense of place as integral to peoples' struggles over rights to land and resources (e.g., Moore 2005).

These kinds of analyses applied to community forestry initiatives can contribute in several ways. They can help situate studies of community forestry in a broader political and economic context, making it possible to identify the kinds of interventions needed in conjunction with community forestry initiatives to address problems of deforestation, forest degradation, rural poverty, and social injustice. They can further efforts to integrate traditional and local ecological knowledge into forest management. They can increase understanding of the relationship between local-level governance institutions and their ecological outcomes. And, by exposing the power relations and social processes that underlie community forestry initiatives, they can provide insight into how community forestry can be implemented in ways that promote more equitable participation and distribution of benefits.

Anthropologists can also contribute by better addressing the policy arena and incorporating more serious environmental analysis in their work. Regarding the former, anthropological methods that emphasize participatory action research with marginalized groups hold promise for helping people in their struggles for rights to forests and forest resources (e.g., Hale 2006). Regarding the latter, anthropologists who collaborate with natural scientists in interdisciplinary research will be well positioned to enhance our understanding of the social-ecological dynamics of community forestry that result from state, community, and forest interactions. As community forestry continues to evolve, we call for more engagement by anthropologists, who can advance our understanding of why community forestry does or does not achieve its promise, and help find ways for it to do so.

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The authors are not aware of any biases that might be perceived as affecting the objectivity of this review.

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Contents

Sociocultural Anthropology

Indexes

Errata

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