Perils in Parks or Parks in Peril?
Reconciling Conservation in Amazonian Reserves with and without Use

CARLOS A. PERES* AND BARBARA ZIMMERMAN†

*School of Environmental Sciences, University of East Anglia, Norwich NR4 7TJ, United Kingdom, email c.peres@uea.ac.uk
†Conservation International–Brazil, and Faculty of Forestry, University of Toronto, Earth Sciences Centre, 33 Willcocks Street, Toronto, Ontario M5S 3B3, Canada

The debate over human-dominated versus strictly protected areas has surfaced once again in recent pages of this journal following a defensible critique by Schwartzman et al. (2000a) against pristine, people-free wilderness as the best servants of biodiversity conservation in the last remaining tropical forest strongholds such as Amazonia. As others have before, they argue that a narrow emphasis on old-fashioned parks (World Conservation Union categories I–IV) will be self-defeating in the long run because strictly protected areas (1) lack the necessary political support from local constituencies, (2) cannot carry the burden of a robust regional conservation framework single-handedly, and (3) ultimately depend on forest ecosystem functions, such as hydrological cycles and immunity to post-El Niño wildfires, well beyond park boundaries and afforded only by vast tracts of forest cover. They also argue that these services can be secured in human-occupied forests whether or not they can retain a full complement of the species in the original biota. In response to this essay, Terborgh (2000) and Redford and Sanderson (2000) argue that it would be both unfair and short-sighted to trust the future of biodiversity to the hands of newly empowered tribal and nontribal communities avid to reassert their land-tenure rights but whose extractive lifestyles may be rapidly changing and becoming an anachronism. At the heart of this debate is the persistence of harvested wildlife populations within communal areas controlled by rural people, or whether the loss of large-game vertebrates would amount to an appreciable reduction in the functioning of forest ecosystems. We wish to refocus this dispute, offer some suggestions as to how these apparently irreconcilable views could be amalgamated into a constructive approach to Amazonian land-use planning, and set the record straight in relation to whether subsistence hunters can coexist with a full complement of vertebrate species.

With an area 20 times larger than the United Kingdom, at least four-fifths of which remains intact, Brazilian Amazonia is by far the largest tropical forest region under the jurisdiction of a single country. Herein lies one of the last opportunities anywhere to redesign and consolidate a robust, basin-wide network of strictly protected, indigenous, and extractive reserves. The region is home to over 8.2 million rural Amazonians (mean human density = 1.61/km²), only one-quarter of which could be defined as forest dwellers, so in theory there are still ample opportunities to expand and consolidate the existing reserve system. This is a matter of urgency, however, because nowhere else is forest loss occurring faster in absolute terms, a process recently aggravated by a combination of El Niño events and selective logging, which catalyzes forest flammability by opening the canopy and increasing the amount of fuel in the understory (Holdsworth & Uhl 1997). In the one-third of Amazonia that repeatedly experiences strong seasonal droughts (Nepstad et al. 1999), recurrent wildfires are likely to drive rapid transitions in forest structure and composition which potentially will eliminate thousands of fire-intolerant organisms with limited ranges (Cochrane et al. 1999; Peres 1999a). To make matters worse, the relentless march of haphazard frontier development is expected to greatly increase access to hitherto remote parts of the region. The road network in Brazilian Amazonia is expected to increase nearly two-fold within this decade from its current total of 6,300 km to 11,000 km of paved roads (Instituto de Pesquisa Ambiental da Amazônia & Insti-
tuto Sócio-Ambiental 2000), generating access to as much as an additional one-fifth of the region. Time is thus rapidly running out for much of the southern and eastern flanks of Amazonia, where most of the selective logging, surface fires, deforestation, and forest fragmentation have been concentrated, whereas in some two-thirds of the region conservation efforts can go well beyond simply retaining forest cover.

Securing the remaining forest cover, whether or not in the form of conventional protected areas, therefore should be an immediate priority along the “deforestation arch” of Amazonia, and here we agree entirely with Schwartzman et al. As they point out, one powerful mechanism for securing forest cover is by building alliances with indigenous communities and strengthening their land claims. One example of how this can be done comes from our 8-year partnership with the Kayapó Indians of A’Ukrê, which has resulted in significant conservation gains and a better understanding of the ecology of key resource populations in this corner of Amazonia (Zimmerman et al. 2001). But it is at this juncture where staunch preservationists, who question the stewardship of local peoples, and conservationists, who extol the virtues of “soft” conservation areas, polarize this debate and part company.

The Kayapó Reserve consists of some 11 million ha of largely pristine forest occupied by fewer than 6000 Kayapós, and no other reserve of any kind in southeastern Amazonia rivals in importance for biodiversity conservation, including the most sensitive components of the vertebrate fauna (Zimmerman et al. 2001). Admittedly, some Kayapó leaders have become infamous for repeatedly liquidating the cream of their forest resources by selling off their broadleaf mahogany (Swietenia macrophylla) stands to logging companies in the nearest booming towns (Peres 1994). But those very timber revenues have also made possible expensive small-aircraft patrols during which Kayapó observers carefully scrutinize the activity of border trespassers and later ruthlessly enforce their reserve boundaries. As we had already pointed out to Steve Schwartzman, this partly explains how deforestation has been halted along the boundaries of the reserve, which is becoming increasingly visible from satellite images of the region (Zimmerman et al. 2001). Following a deliberation process, the Kayapó of A’Ukrê are protecting an unlogged, unhunted forest reserve of approximately 10,000 ha within their territory, which now perhaps contains one of the largest surviving mahogany stands in southern Pará. There are two key conservation points to occupation of forests by indigenous tribal communities. First, they have no cultural experience with and little present capacity to engage in large-scale agriculture, which is by far the greatest threat to tropical forests. Second, most indigenous communities fight for their rights for self-determination based on control over traditional lands and, in the process, provide unpaid protection services as demonstrated par excellence by the Kayapó. Our point is that, as shown by our experience with the Kayapó, it is feasible to work with native Amazonians to secure not only the forest and its key ecosystem services but also the most extinction-prone components of the biota by explicitly zoning extractive activities within traditional territories.

Although most groups of native Amazonians are far less assertive than the Kayapó, the bottom line is that legally recognized Indian reserves span tens of millions of hectares of intact Amazonian forests, which places them at the forefront of conservation issues in the Neotropics. Indigenous territories account for 248 of all 459 officially designated conservation areas of the Amazon basin, encompassing 52% of the entire area receiving some form of protection (Peres 1994). The 369 Indian reserves in Brazilian Amazonia, for instance, represent over 100 × 10^6 ha or 20.7% of the region, including 29 reserves larger than 1 × 10^6 ha (Ricardo 1999). Indian reserves also tend to be much larger and span a broader geographic representation than either nature or extractive reserves (Peres & Terborgh 1995). This makes them enormously valuable for biodiversity conservation because they retain a considerable fraction of the Amazonian biota that may otherwise remain unprotected, or serve as buffer zones for adjacent protected areas. Furthermore, Amazonian Indian populations are small (<1 person/km²) and for the time being still strongly regulated by density-dependent mechanisms. Village density thus tends to be low, and forest resources beyond 15 km from a village are unlikely to be depleted. In the long run, however, whether or not these areas will continue to function in the best interest of biodiversity conservation, or indeed retain most of their forest cover, will depend on strong stewardship by Indian leaders, government agencies, and conservation organizations alike; here we agree entirely with Terborgh (1999, 2000). A good example of how indigenous people can undermine conservation efforts comes from the recent “invasion” of Guarani Mbyua Indians from northern Argentina and Paraguay into four southeastern Brazilian parks (Superagüi National Park and state parks of Ilha do Cardoso, Juréia-Itatins, and Intervales). Meanwhile, an additional 15,000 Guarani are still hovering around the Argentina-Paraguay border ready to “migrate” to other Brazilian parks (M. Galetti, personal communication).

That said, the forests managed by many folk communities of Amazonia that Schwartzman et al. speak of are often chronically overharvested and more reminiscent of collections of trees and lianas performing their climate and hydrological roles than theatres of the full array of ecological processes that will guarantee the continuity of these ecosystems. Indeed the “higher-level criteria of tropical forest integrity” advocated by these authors, such as “forest vulnerability to fire, fertility of forest
soils, forest carbon content, or the role of tropical forests in regional hydrological and climatic systems,” are more closely related to flows and cycles made possible by forest structure and biomass per se than to the species assembly of their plant and animal communities. Yet no one would think of a fast-growing tree monoculture such as a mature *Eucalyptus* or *Gmelina* stand (of say the Jari Project) as even reminiscent of an an old-growth *terra firme* forest, even if they both retained the same amount of aboveground phytomass and were functionally equivalent in terms of carbon storage and evapotranspiration. We may agree with these authors that the “depletion of large-animal populations does not threaten the majority of the other species that comprise these forests,” but from an aesthetic viewpoint alone most people—including indigenous, traditional, and rural folk of Amazonia—would agree that without the midsized to large-bodied vertebrates that are often overhunted, these forests would be reduced to a pallid skeleton of their former selves.

Yet we have never advocated that “human presence in tropical forests is ultimately incompatible with the conservation of biological diversity” (Schwartzman et al. 2000a). As each of us has >20 years of experience with fieldwork in Amazonia, it would be foolish to argue otherwise. Advocates of people-free reserves need not maintain that all reserves should be free of people. Peres and Terborgh (1995) set out a model recommending an interconnected system of complementary nature, production, and indigenous reserves allowing for various degrees of protection which together are more than the sum of their parts. And our recent work at the Kayapó Reserve has clearly shown how conservation of the full biotic integrity of forest ecosystems can be achieved within large Indian areas with the full collaboration of local communities (Zimmerman et al. 2001). Admittedly, strictly protected areas could still be seen as the backbone of this reserve network even if they are likely to represent a small proportion of the total conservation acreage, and to a large degree this is an inevitable political reality. In the long run, however, these largely uninhabited “jewels in the crown” (Redford et al. 1998) could not be viable in isolation without a basin-wide “crown” of national forests, sustainable forestry areas, private reserves, and Indian and extractive reserves inhabited by indigenous, traditional, and rural people which should collectively retain most of the region’s forest cover.

Unharvested areas could also be established as protected enclaves embedded within much larger harvested forests. Indeed, a number of traditional communities with whom we have worked over the years (e.g., Kaxinawá Indians of western Acre, Kayapó of southern Pará) have welcomed the notion of spatially structured harvest restrictions because they themselves understand the negative abundance responses of overexploited populations. This would maximize the benefits of source-sink dynamics in replenishing overharvested populations from adjacent unharvested areas, thus preventing local extinctions at large spatial scales. This would also maintain yields over time scales far longer than those predicted by sustainable harvest models applied to closed populations (Novaro et al. 2000; Peres 2000b).

As to the misleading claim that “no case of species extinction or severe depletion of large mammals has been reported from Amazonian indigenous and extractive reserves” (Schwartzman et al. 2000b), this is entirely a matter of how extinction is defined in space. An overwhelming body of evidence has shown that a number of large-bodied vertebrates can be driven to local extinction (at the scale of >100 km²) within the forest areas harvested by both tribal and nontribal subsistence hunters (e.g., Peres 1993, 1999b, 2000a, 2000b; Martins 1993; Calouro 1995; Nascimento 1999). For example, the Rio Jordão Kaxinawá Indians of western Acre have driven most of their key game species to local extinction within their village catchment areas, and now villagers rely primarily on domestic livestock as a source of protein (Peres 1993). These key species include several (white-lipped peccary [*Tayassu pecari*], tapir [*Tapirus terrestris*], black spider monkey [*Ateles chamaele*], woolly monkey [*Lagothrix lagotricha*]), pacarana [*Dinomys branickii*], Razor-billed Curassow [*Mitu turberosa*], Blue-throated Piping Guan (*Pipile cumanensis*) that no longer occur even in less accessible, rarely hunted areas (>10 km from the upper Tarauacá River). Nascimento (1999) was unable to detect any signs of tapir and white-lipped peccary within 6 km of the Kayapó village of A’Ukrê following extensive wildlife surveys conducted in 1996 and 1999. Several other species (e.g., bearded saki monkey [*Chiropteryx utabicki*], Razor-billed Curassow [*Mitu turberosa*], Red-throated Piping Guan [*Pipile cujubi*]), two species of tortoises (*Chelonoidis carbonaria* and *G. denticulata*) were also overhunted and severely reduced in numbers. Significant density differentials between hunted and unharvested areas exist for most game species at the Kayapó Reserve (Nascimento 1999; Peres 2000a, 2000b), and the high game biomass of the (protected) Pinkaiti study area, as noted by Schwartzman et al. (2000a), is primarily a reflection of its large ungulate stock nurtured by high forest productivity and release from hunting pressure for at least 10 years (Peres 2000a). In 1955 Calouro reported that local residents at the Antimâri State Forest, an extractive reserve in all but name, had effectively exterminated populations of spider monkeys and white-lipped peccaries, and several other species were facing “extensive hunting pressure.” The same species, as well as tapirs and Razor-billed Curassows, were virtually extinct in an area hunted by rubber tappers of the Rio Iaco, Acre (Martins 1993). One of us has also documented several cases of hunting-induced, local extinctions in other indigenous areas (Kulina do Médio Juruá, Kamamãni do Rio Jurúá) and three extractive reserves (Alto Cajari, Alto Ju-
ruá, and Tapajós-Arapiuns) (C. Peres, unpublished data). More ominously, interviews with young hunters from a number of old settlements in several parts of Amazonia show that they are simply unable to recognize several large-mammal species when presented with photographs and illustrations in field guides. Regional extinction events are, however, unlikely to occur at the scale of entire indigenus and extractive reserves because these tend to be large and, for the time being, sparsely populated. This could change as households become more sedentary and village density increases, thus gradually eroding source populations of large vertebrates that are currently subjected to little or no hunting pressure because access to them is prohibitive, rather than by choice.

Ultimately, whether or not game hunting is sustainable depends on the density of consumers living off the land, how long they have done so, and the wildlife productivity of a given forest. To use one of Schwartzman et al.’s examples, one of the main reasons the “co-management plan” endorsed by the local population of Jau National Park does not come into serious conflicts with biodiversity conservation is because this park of 2.2 million ha, for the time being, is home to only 17 caboclo households (Fundação Vitória Amazônica 1998).

In a heterogeneous region as colossal as Amazonia, a sweeping statement prioritizing conservation efforts into indigenous and extractive reserves, to the detriment of conservation of complete species assemblages in more remote parts of the region. A pristine and virtually uninhabited nature reserve (52,000 ha) was successfully created by the recent purchase of a defunct rubber state in the lower Rio Purús (M. van Roosmalen, personal communication), the recent purchase of a defunct rubber state in the lower Rio Purús (M. van Roosmalen, personal communication), and many large landholdings of centralwestern Amazonia are currently on the market at bargain prices (often US$2–10/ha) by the private sector, if not by conservation organizations. There is also a huge geographic imbalance between the combined conservation acreage designated as indigenous and extractive reserves and that of strictly protected areas (Fearside & Ferraz 1995). Federal national parks, biological reserves, and ecological stations now comprise only 3.25% of Brazilian Amazonia (Peres et al. 1999a), less than one-third of the 10% target proposed by the World Conservation Union and World Wildlife Fund and ratified by the Brazilian government. Moreover, most of these reserves exist only on paper and desperately require infrastructural and staffing investments (Peres & Terborgh 1995; Ferreira et al. 1999b).

Allocation of conservation investments should thus be seen as context-dependent, and different scales of conservation—ranging from efforts to merely avert deforestation to those aiming to protect the full integrity of forest ecosystems—should be acceptable to both Greeks and Trojans in this exchange and pursued in partnership and in keeping with the realities of different subregions.

### Literature Cited


analysis of the defensibility status of existing conservation units and design criteria for the future. Conservation Biology 9:34–46.