# Running Head: Amazon Parks and Indigenous Reserves Inhibition of Amazon Deforestation and Fire by Parks and Indigenous Reserve

D. Nepstad[1,2*], S. Schwartzman[3], B. Bamberger[1,4], M. Santilli[2], D. Ray[1], P. Schlesinger[1], P. Lefebvre[1], A. Alencar[2], E. Prinz[5]

[1] *The Woods Hole Research Center, P.O. Box 296, 13 Church Street, Woods Hole, MA 02543, U.S.A.*

[2] *Instituto de Pesquisa Ambiental da Amazonia, Av. Nazaré 669, Centro. 66035-170 Belém, PA, Brasil*

[3] *Environmental Defense, 1875 Connecticut Avenue, NW, Suite 600, Washington, D.C. 20009, U.S.A.*

[4] *Yale School of Forestry & Environmental Studies, 205 Prospect Street, New Haven, CT 06511 U.S.A*

[5] *Cooperative Institute for Meteorological Satellite Studies, 1225 W. Dayton Street, Madison, WI, 53706, U.S.A*

***Correspondence and requests for materials should be addressed to Daniel C. Nepstad at The Woods Hole Research Center, P.O. Box 296, 13 Church Street, Woods Hole, MA 02543, U.S.A.*** ; email: dnepstad@whrc.org

Number of words: 3,734

**Abstract:**

Conservation scientists generally agree that many types of protected areas will be needed to protect tropical forests. But little is known of the comparative performance of inhabited and uninhabited reserves in slowing the most extreme form of forest disturbance: conversion to agriculture. We employed satellite-based maps of land cover and fire occurrence in the Brazilian Amazon to compare the performance of large (>50,000 ha) uninhabited (parks) and inhabited (indigenous lands, extractive reserves, and national forests) reserves. Reserves significantly reduced both deforestation and fire. Deforestation was 2 to 5 times higher along the outside *versus* the inside of the reserve perimeters and fire occurrence was 3 to 9 times higher. No significant differences in inhibition of deforestation or fire were found among reserve

types. However, the historical tendency of park planners to avoid the conservation risks associated with the active agricultural frontier has resulted in a skewed distribution of uninhabited reserves away from areas of high deforestation and burning rates. In contrast, indigenous reserves are often created in response to frontier expansion and land conflict, and many prevented deforestation completely despite high rates of deforestation along their boundaries. The inhibitory effect of indigenous reserves on deforestation is not lost over time through acculturation of indigenous residents or population growth, as previously postulated. Indigenous reserves occupy one fifth of the Brazilian Amazon—five times the area under protection in parks—and are currently the most important barriers to Amazon deforestation. As the Brazilian government proceeds to expand the park network from 5 to 10% of the Amazon over the coming years, the greatest challenge is successful park implementation in high-risk areas of frontier expansion, as indigenous reserves are strengthened. This success will depend upon political support from a broad constituency of grass-roots organizations.

## Introduction

There is growing agreement among conservation scientists that many types of protected areas, including those with resident human populations, are needed for an effective global strategy to preserve tropical forests. A recent synthesis concludes that protection of a substantial proportion of the world's remaining biodiversity is feasible in part because approximately two million km² of tropical forest are already protected for indigenous peoples and biodiversity (Pimm et al, 2001). Officially recognized indigenous lands of the Brazilian Amazon alone comprise half of this total. Most conservation literature and policy recommendations are still directed at uninhabited protected areas, however, which differ significantly from inhabited protected areas such as indigenous lands and extractive reserves in the processes by which they are created, in their long-term management needs, and, hence, in their role within conservation strategies. One rationale for this emphasis on uninhabited protected areas is the argument that the conservation value of indigenous lands is lower than that of parks because indigenous people ultimately adopt the cultural values, technology, and patterns of resource exploitation of their non-indigenous neighbors, a trend that is exacerbated by population growth within indigenous lands (Terborgh and van Shaik 2002, Terborgh 1999, 2000, Redford and Sanderson 2000).

The refinement of biodiversity conservation strategies in the tropics is hampered by a dearth of comparisons of the performance of inhabited vs. uninhabited protected areas in slowing the most extreme form of human disturbance: forest conversion to agriculture. Several studies have examined the effects of rural people on wild game populations (e.g. Peres 2000a, b, Robinson et al. 2000, Redford 1992) and others have analyzed the performance of uninhabited parks in protecting biological diversity (Bruner et al. 2001, Ferreira et al. 1999, Terborgh 1999, Terborgh and van Schaik. 2002). But we are unaware of studies that compare the inhibitory effects of inhabited and uninhabited protected areas on

forest clear-cutting. The assumption that the conservation value of uninhabited parks is higher than reserves with human residents remains untested.

We report on the results of a satellite-based comparison of the inhibitory effects of protected areas that prohibit human habitation (parks, biological reserves, ecological stations) and those that permit habitation (indigenous reserves, extractive reserves, and national forests) on deforestation and fire. Logging and hunting damage forests, but were omitted from this analysis because they are difficult to quantify (Nepstad et al. 1999a).

**Methods**

The quantification of reserve performance in slowing deforestation is a difficult undertaking. This performance is best measured against a baseline that describes the trajectory of deforestation in the absence of the reserve. This trajectory is influenced by the suitability of the land within the reserve for agriculture, logging, and other economic activities, by market trends for agricultural and forest products, by investments in transportation and energy infra-structure, and by agrarian reform. A reserve therefore inhibits deforestation only if (a) it slows the expansion of economic activities (i.e., protects natural resources that would otherwise be exploited), (b) it prevents or mitigates the effects of investments in roads and other infra-structure that cause direct environmental damages and/or that indirectly foster natural resource exploitation, and (c) it prevents agricultural settlements—either planned or spontaneous—motivated by agrarian reform pressures. (This third condition is not redundant of the first because agricultural settlements are often planned in places that are not suitable for agriculture.) Within this context, reserves that are far from the expanding agricultural and logging frontier and are not slated for infra-structural investments or agricultural settlement have a negligible short-term effect on deforestation, but may have a very important inhibitory effect as the frontier grows near.

A second challenge of measuring protected area performance is to distinguish between local and regional effects. To what extent is the inhibition of deforestation within a reserve compensated by an increase in deforestation elsewhere? In general, this "leakage" of the inhibitory effect of reserves should be greatest in expanding agricultural frontiers, diminishing over time as land scarcity begins to drive up land prices, inhibiting forest conversion regionally. This aspect of reserve performance is not addressed here.

We provide an initial comparative assessment of reserve performance in the Brazilian Amazon by using deforestation and fire occurrence along the reserve perimeter as a proxy for the threat of imminent deforestation. We assume that reserves with rapid rates of forest conversion to agriculture and/or high incidence of fire along the outside of their perimeters are more likely to suffer forest clear-cutting than reserves with low rates of forest conversion along their perimeters. Therefore, the ratio of deforestation and burning in buffer zones outside versus inside the reserve boundary provides a measure of reserve performance that normalizes the threat; when this ratio is greater than unity, the reserve has deflected forest conversion from the baseline trajectory. This approach overestimates the inhibition of deforestation in cases where the reserves were

established close to existing roads or to the boundaries of existing colonization projects.

The technical difficulties of quantifying forest damage in vast, remote forestlands further complicate the measurement of reserve performance. Interview-based analyses (e.g. Bruner et al. 2001) provide a qualitative indication of reserve performance, but are vulnerable to the biases of informants who have a vested interest in this performance. Park managers and conservation NGO personnel may have the best information about the status of reserves, but they also may have motives for overstating the success of the reserves. We therefore measured reserve performance using maps of landcover developed created from satellite images (Landsat TM) for 1992 and 1996 (Skole 2000) and maps of active fires derived from a geostationery weather satellite (GOES-8, Prins et al. 1998, Menzel and Purdom 1994). Deforestation is indicated by satellite detection of forest replacement by cattle pastures and agricultural systems, and by detection of the fires that are used as part of the forest clearing process, and in the maintenance of cattle pastures (Nepstad et al. 2001). Of the four major types of land use fire in the Amazon—including fire used to burn felled forest, fire used to improve forage quality in cattle pastures, accidental cattle pasture fire, and fires that burn standing forests—only the first three are registered as “hot pixels” by the thermal channels of satellites (Nepstad et al. 1999b, 2001). Our analysis does not capture understory fires in standing forests.

Maps of reserves for the Brazilian Amazon were acquired from the *Instituto Socioambiental* (Capobianco et al. 2001). We refer here to those federal land designations that prohibit resource exploitation by people (national parks, biological reserves, ecological research stations) as “parks”. Indigenous reserves, extractive reserves and national forests permit human residents and subsistence agricultural activities, but restrict partially restrict deforestation.

Our metric for reserve performance—the deforestation and fire occurrence rates along the outside versus inside of the perimeter—is sensitive to co-registration errors between land cover maps of different years, and between land cover and park boundaries. We therefore excluded from the deforestation analysis those reserves that are small (<50,000 ha) and therefore had proportionally high image co-registration errors. We also inspected the superimposed land cover maps and reserve locations for each of the reserves and excluded from the analysis those for which co-registration errors could be detected visually. Reserves established after 1992, reserves established by state governments, and reserves for which satellite data were not available were also excluded from the analysis. Based on these selection criteria, we were able to quantify deforestation rates of 11 parks, 81 indigenous reserves, 2 extractive reserves and 6 national forests, representing 29%, 46%, 37% and 13%, respectively, of the total area of each reserve type under federal jurisdiction.

Fire inhibition was quantified for reserves that were at least 50,000 hectares in size and that had less than 20% classified as cerrado—Brazil’s savanna woodland vegetation that burns naturally. The spatial distribution of fires was compiled for 1998, a severely dry year. This sample consisted of 11 parks, 87

indigenous reserves, 4 extractive reserves, and 12 national forests, representing 35%, 51%, 74%, and 34% of the area of these federal reserves, respectively.

We measured deforestation inhibition by comparing average annual deforestation rates from 1992 to 1996 (difference between the amount of area classified as deforestation and recent regrowth in 1992 and 1996 divided by the original forest area) within 10-km-wide strips of land located along the inside and outside of the reserve perimeter. The influence of reserves on fire occurrence was measured by comparing fire density (number of fires $km^{-2}$ in 1998) within 20-km wide strips along the inside and outside of the reserve perimeter. We used larger buffer areas for the fire data because of the coarser spatial resolution of these data—the GOES pixels are 4 km wide, while the Landsat TM pixels are 30 m wide. Spatial accuracy of GOES fire detection is within one pixel (Menzel and Purdom 1994). Only those 'hot pixels' detected at mid-day and classified as having a high probability of being associated with fire were included in the analysis (Prins et al. 1998). For reserves <200,000 ha in size, fire density of the entire reserve was compared with that along the outside of the perimeter.

Due to limitations imposed by the distribution of reserves sizes, including a highly right-skewed distribution and unequal error variance, neither of which were adequately addressed by data transformation, we used non-parametric analyses to test for differences within and among reserve types. First, we compared the inhibition of deforestation and fire by the reserves (outside buffer vs. inside buffer values) within each reserve type using Wilcoxon Signed Ranks Test for dependent samples (one-tailed test). We then compared inhibition of deforestation and fire across reserve types using Kruskal-Wallis One-Way ANOVA for independent samples (two-tailed test). Our dependent variable for this analysis was the ratio between inside and outside buffer values. Because the research questions being asked focus on the effectiveness of the reserve boundary at minimizing disturbance pressures originating from outside the reserve, we did not include in these analyses those reserves for which no outside pressure was detected during the study period.

Deforestation caused by reserve residents, and not encroachers, might occur in the core of the reserve and, thereby, be missed by our buffer analysis. We therefore repeated the analysis using the deforestation rate of the entire reserve. This analysis was also less sensitive to co-registration errors described above, since the denominator of the deforestation term (original forest area) is larger.

Finally, we used regression analysis to test for a relationship between deforestation among indigenous reserves as a function of (a) time since first contact with non-indigenous groups and (b) reserve population density. The ratio between the annual deforestation rates, inside and outside of the reserve, determined between 1992 and 1996 provided the dependent variable for these analyses. A log-transformation was required to satisfy the assumptions of this analysis.

## Results and Discussion

On average, deforestation from 1992 to 1996 was 2 to 5 times higher along the outside of the reserves than along the inside for all reserve types (Figs. 1A, 2A, 3A, 4). This inhibitory effect was significant for indigenous reserves (n=54; Z=-6.07; P<0.000), parks (n=9; Z=-2.55; P=0.006), and national forests (n=4; Z=-1.83; P=0.034). Differences among reserve types were not indicated (n=4; K-W=0.213; P=0.975). We were unable to test inhibition of deforestation by extractive reserves because of the small sample size (n=2). The same pattern of deforestation inhibition was found when we employed deforestation across the entire reserve as the dependent variable.

A similar inhibitory effect was found for fire (Figs. 1B, 2B, 3B). The average density of fires was 3 to 9 times higher along the outside of the reserves than along the inside (Figs. 1B, 2B). This effect was highly significant for indigenous reserves (n=87; Z=-6.84; P<0.000), parks (n=11; Z=-2.94; P=0.002), and national forests (n=12; Z=-2.76; P=0.003). For fires we also detected a marginally significant effect for small number of extractive reserves included in the sample (n=4; Z=-1.604; P=0.055). Comparable to what we found for deforestation rates, each of these reserve types exerted a similar degree of control over fire occurrence (n=4; K-W=3.253; P=0.354, Figs. 1B, 2B).

Indigenous lands strongly inhibited deforestation in the active agricultural frontier. Seven out of eleven indigenous reserves with high annual deforestation rates (>1.5% $yr^{-1}$) along the outside of their perimeters had inner deforestation rates of 0.75% or lower (Fig. 2A). Few parks have been established within the active agricultural frontiers of eastern and southern Amazonia (Fig. 3A), in part because of the historical tendency of park planners to avoid the conflicts and conservation threats associated with expanding frontier regions (Peres and Terborgh 1995). Only one of the parks in our sample (Gurupí, Fig. 2A, 3A) had an outer deforestation rate of >1% $yr^{-1}$, and exhibited a similar two-fold inhibition of deforestation (0.6 : 1.3 % $yr^{-1}$).

The proximity of indigenous reserves to the active frontier is also reflected in the fire data (Figs. 1B, 2B, 3B). The average density of fires was two times greater along the outside of indigenous reserves than it was along park perimeters (Figs. 2B). Through their location in lower risk regions of Amazonia, the park network has had a proportionally smaller effect on frontier expansion. Indigenous groups, in contrast, often live in the path of expanding frontiers, and fight to win legal recognition of their land rights while defending their forests from clearing by outsiders.

The high variability of reserve performance can be traced to individual reserve histories. High rates of deforestation in indigenous reserves were generally associated with exploitation or invasions from non-indigenous populations that had occurred prior to reserve demarcation. For example, the Sete de Setembro reserve of the Sururí indigenous group, and the Sararé Nambiquara reserve were the least effective in slowing deforestation in our sample (Fig. 1A, 3A). Both of these areas were legally protected as indigenous lands long after invasions by outsiders were established. In the Sete de Setembro area approximately 20,000 hectares of land were subdivided and sold to colonists by a private colonization firm, and occupied

by several hundred families (Instituto Socioambiental 2000). In the 1960's, the Sararé Nambiquara area was similarly invaded, illegally subdivided and sold off starting in the late 1960s. In 1991, the Sararé reserve was invaded by over 2,000 wildcat gold miners who were still operating in the area in 1996 (Mindlin 1985, Instituto Socioambiental 2000). The Gurupí Biological Reserve was also undermined historically. The state government of Maranhão actively encouraged land settlement of this park at the time of its creation in the 1980's.

Reserves that successfully inhibited deforestation within the active agricultural frontier were often inhabited by tribes who actively enforce legal restrictions on natural resource exploitation by outsiders. The Kayapó people have successfully defended their ancestral lands, expelling ranchers and settlers who invade their reserve (Zimmerman et al. 2001), maintaining deforestation rates at close to zero (Schwartzman et al 2000, Zimmerman et al 2001) (Figs. 1A, 3A). In recent years, various indigenous groups have taken intruders hostage to reinforce demands for reserve demarcation and government assistance in protecting boundaries.

It has been postulated that the tendency of indigenous people to protect their forests from deforestation is lost as these groups adopt the values of market-based society, and as their population densities increase (Redford and Sanderson 2002, Terborgh 2000, Terborgh and van Shaik 2002). We tested this prediction by examining the response of deforestation inhibition by indigenous reserves to (a) the time since first contact with non-indigenous groups and (b) population density. Neither of these comparisons showed a significant relationship, and, in fact, high inhibition ratios, suggesting weaker regulation of disturbance, were relatively more common among the recently contacted reserves (Fig. 5A, B). We do not suppose that the absence of these relationships in the past, of itself, predicts anything about the future. It does show that contact with the national society and resource degradation are not inevitably linked. The ecological integrity of the indigenous lands will ultimately depend in large measure on what sorts of economic alternatives are available to indigenous peoples.

We did not measure reserve performance in protecting forests from impoverishment through logging and hunting. Many indigenous groups have opened their lands to mahogany extraction. Logging is also common in those parks that are located in the active frontier. The Xikrin do Cateté indigenous group, in contrast, has experimented with a timber management system, and the Panará people have actively expelled mahogany loggers from their area. The A'Ukre group of the Kayapó people has initiated a promising resource management system that may have prevented depletion of game species (Zimmerman et al 2001). The 13-million-ha reserve of the Kayapó and the Upper Xingu peoples in south-central Pará and Mato Grosso is larger than any tropical forest park in the world, and is the main barrier between the forest and business-as-usual frontier expansion in the heavily-settled eastern Amazon (Fig. 2a, b). In fact, the Xingu was one of 15 indigenous reserves facing deforestation pressure from the outside that actually registered an increase in forest cover between 1992-96 (Fig. 1A). These indigenous groups have been strengthened through collaboration with

conservation organizations (Zimmerman et al 2001, Instituto Socioambiental, 2000).

Indigenous reserves occupy one fifth of the closed-canopy forests of the Brazilian Amazon, which is twice the area targeted by the Brazilian government for preservation in parks (http://www.mma.gov.br/port/sbf/dap/parqbras.html), and five times the area currently designated as parks (Fig. 3). They also contain larger blocks of forest than parks, as represented in our sample (Fig. 4). In Amazonia, where 85% of the forest is still standing, protection of the regional forest-climate system is critical to the long-term protection of biodiversity, and this will demand forest cover over most of the region (Silva-Dias et al. 2002, Nobre et al. 1991, Nepstad et al. 1999). Recent advances in the enforcement of environmental legislation in the Brazilian Amazon (Nepstad et al. 2002) demonstrate the potential feasibility of maintaining forest over most of this region. Government regulation of deforestation, fire and logging is essential to this strategy, as is an expanding network of forest reserves.

The establishment of parks in regions that are largely inaccessible to humans is an important component of a long-term strategy to defend nature in places like the Amazon, but such risk-avoiding reserves must be complimented by reserves within the active frontier. The Brazilian government's efforts to expand the network of parks in the Amazon will have the greatest conservation value if protected areas are successfully established in active frontier regions, where 20,000 $km^2$ of forest are currently being converted to agriculture each year. This will require strong political support for such initiatives (Dourojeanni 2002, van Shaik and Rao 2002, Brandon 2002), and improvements in government capacity to protect such reserves. Recent experiences in the four-million hectare, *Terra do Meio* (Land in the Middle) forest complex north of the Kayapo reserve complex (Fig. 3A) demonstrate that uninhabited parks can gain broad political support within active frontier regions of the Brazilian Amazon if they are advanced within the context of a regional conservation and development planning process that addresses the needs and aspiration of local indigenous groups, agro-extractivist populations (e.g. rubber tappers), and colonist farmers. This proposed mosaic of reserves encompasses the largest remaining block of intact forest in southeastern Amazonia, and was conceived by the organization of the smallholder farmers of the Transamazon Highway--the *Movimento pelo*

*Desenvolvimento do Transamazonico e Xingu* (MDTX--Movement for the Development of the Transamazon and Xingu). With support from MDTX, grass-roots organizations representing the rubber tappers (Conselho Nacional de Seringueiros, CNS—National Rubber Tappers Council) and indigenous groups (COIAB), and non-governmental research institutes (Instituto Socioambiental, Instituto de Pesquisa Ambiental da Amazonia), there is the potential for creating a vast, interconnected network of indigenous reserves, extractive reserves, national forests, and parks in the heavily deforested eastern Amazon that could encompass more than 16 million hectares. The prospects of such an achievement are enhanced by recognition among conservationists of the existing and potential role

of a diverse alliance of the Amazon's rural stakeholders as proponents of nature conservation (Diegues 1992, Hall 1997).

References

Brandon K. 2002. Putting the right parks in the right places. In:Terborgh J, van Schaik C, Davenport L, Rao M, editors. Making Parks Work. Washington, D.C.: Island Press; p 443-467.
Bruner AG, Gullison RE, Rice R, Fonseca GA. 2001. Effectiveness of parks in protecting tropical biodiversity. Science 291(5501):125-128.
Capobianco JPR, Verissimo A, Moreira A., Sawyer D, Santos I, Pinto LP. 2001. Biodiversidade na Amazônia Brasileira: avaliação e ações prioritárias para a conservação, uso sustentável e repartião de benefícios. Sao Paulo, Brasil: Estação Liberdade: Instituto Socioambiental.
Diegues A. 1992. O Mito da Natureza Intocada. University of São Paulo, Brazil.
Dourojeanni MJ. 2002. Political will for establishing and managing parks. In:Terborgh J, van Schaik C, Davenport L, Rao M, editors. Making Parks Work. Washington, D.C.: Island Press; p 320-334.
Ferreira L, Sá R. 1999. Áreas protegidas na Amazonia Brasileira. WWF, Brazil.
Menzel, W. P. and J. F. W. Purdom, 1994: Introducing GOES-I: The first of a new generation of geostationary operational environmental satellites. Bulletin of the American Meteorological Society, 75, 5, 757-781.
Mindlin B. 1985. Nós Paiter: Os Surui de Rondonia. Vozes: Petrópolis.
Nepstad DC, Veríssimo A, Alencar A, Nobre CA, Lima E, Lefebvre PA, Schlesinger P, Potter C, Moutinho PRdS, Mendoza E, Cochrane MA, Brooks V. 1999. Large-scale impoverishment of Amazonian forests by logging and fire. Nature 398:505-8.
Nepstad DC, Carvalho GO, Barros AC, Alencar A, Capobianco JP, Bishop J, Moutinho P, Lefebvre PA, Silva UL, Prins E. 2001. Road paving, fire regime feedbacks, and the future of Amazon forests. Forest Ecology and Management 154:395-407.
Nepstad D, McGrath D, Alencar A, Barros C, Carvalho G, Santilli M, del C. Vera Diaz M. 2002. Frontier governance in Amazonia. Science 295:629-31.
Nobre CA, Sellers PJ, Shukla J. 1991. Amazonian deforestation and regional climate change . Journal of Climate 4:957-88.
Peres, CA 2000a. Effects of subsistence hunting on vertebrate community structure in Amazonian forests. Conservation Biology 14:240-253.
Peres, CA 2000b. Evaluating the impact and sustainability of subsistence hunting at multiple Amazonian forest sites. Pp 83-115. In: Hunting for sustainability in Tropical Forests (JG Robinson & EL Bennett, Eds) Columbia University Press, New York.
Peres CA, Terborgh JW. 1995. Amazonian nature reserves: an analysis of the defensibility status of existing conservation units and design criteria for the future. Conservation Biology 9(1):34-46.
Pimm SL, Ayres M, Balmford A, Branch G, Brandon K, Brooks T, Bustamante R, Costanza R, Cowling R, Curran LM, Dobson A, Farber S, da Fonseca GAB, Gascon C, Kitching R, McNeely J, Lovejoy T, Mittermeier RA, Myers N, Patz JA, Raffle B, Rapport D, Raven P, Roberts C, Rodríguez JP, Rylands AB, Tucker C,

Safina C, Samper C, Stiassny MLJ, Supriatna J, Wall DH, Wilcove D. 2001. Can We Defy Nature's End? Science 293:2207-8.
Prins EM, Menzel WP, Feltz JM. 1998. Characterizing Spatial and Temporal Distributions of Biomass Burning Using Mult-Spectral Geostationary Satellite Data. In: Ninth Conference on Satellite Meteorology and Oceanography. Paris, France, 25-29 May: p 94-97.
Redford KH. 1992. The empty forest . Bioscience 42(6):412-22.
Redford KH, Sanderson SE. 2000. Extracting humans from nature . Conservation Biology 14(5):1362-4.
Robinson JG, Bennett EL 2000. Hunting for sustainability in tropical forests. Columbia University Press, New York.
Silva Dias MAF, Rutledge S, Kabat P, Silva Dias PL, Nobre C, Fisch G, Dolman AJ, Zipser E, Garstang M, Manzi AO, Fuentes JD, Rocha HR, Marengo J, Plana-Fattori A, Sá LDA, Alvalá RCS, Andreae MO, Artaxo P, Gielow R, Gatti L. 2002. Cloud and rain processes in biosphere-atmosphere interaction context in the Amazon region. Journal of Geophysical Research 107(D20):8072, doi:10.1029/2001JD000335.
Schwartzman S, Moreira AG, Nepstad DC. 2000. Rethinking tropical forest conservation: perils in parks. Conservation Biology 14(5):1351-7.
Skole DL. Tropical Rainforest Information Center, http://www.bsrsi.msu.edu/trfic/index.html . April, 2000.
Terborgh J. 1999. Requiem for nature. Washington, DC: Island Press.
Terborgh J. 2000. The fate of tropical forests: a matter of stewardship. Conservation Biology 14(5):1358-61.
Terborgh J, van Schaik C. 2002. Why the world needs parks. In:Terborgh J, van Schaik C, Davenport L, Rao M, editors. Making Parks Work. Washington, D.C.: Island Press; p 3-14.
van Schaik C, Rao M. 2002. The frontier model of development and its relevance to protected area management. In:Terborgh J, van Schaik C, Davenport L, Rao M, editors. Making Parks Work. Washington, D.C.: Island Press; p 424-440.
Zimmerman B, Peres CA, Malcolm JR, Turner T. 2001. Conservation and development alliances with the Kayapó of south-eastern Amazonia, a tropical forest indigenous people. Environmental Conservation 28(1):10-22.

Acknowledgements- This study was funded by the Gordon and Betty Moore Foundation through grants to the Conservation International Center for Applied Biodiversity Science and Woods Hole Research Center, NASA (LBA-ECO) and the US Agency for International Development. G. Fiske assisted with spatial analysis.
Figure captions:
Figure 1. Inhibition of (A) deforestation and (B) fire by individual reserves. Boundary effectiveness is illustrated relative to the 1:1 line. Letters indicate reserves that are referred to in the text: Gu=Gurupí Biological Reserve, Ka=Kayapó Indigenous Reserve (IR), Sa=Sararé IR, SS=Sete de Setembro IR, Xi=Xingú Indigenous Park, XC=Xikrin do Cateté IR.
Figure 2. Reserve performance in slowing Amazon deforestation and fire. A. Average annual deforestation rates (% of original forest area) from 1992 to 1996 within 10-km strips of land along the inside and outside of each reserve boundary

n=55 indigenous reserves, 9 parks, 2 extractive reserves, and 4 national forests. B. Cumulative fire density for 1998 within 20-km-wide strips of land along the inside and outside of each reserve boundary (n=87 indigenous reserves, 11 parks, 4 extractive reserves, and 12 national forests). Fire data were obtained from the GOES satellite (Prins et al. 1998), and were restricted to one fire day$^{-1}$ 16 km$^{-2}$ pixel.

Figure 3. Reserves and human-caused disturbance in the Brazilian Amazon. (A) Reserves, highways and deforestation (as of 1992) of the Brazilian Amazon. Parks and biological reserves are rare in the eastern and southern margins of Amazonia, where most highways and deforestation activities are concentrated. Paved highways are gray; those to be paved are hatched. (B) Reserves and active fires registered by the GOES satellite for 1998 (10). Like deforestation, fires are more common in eastern and southern Amazonia. Reserves in the Cerrado woodland can burn as part of their natural disturbance regime.

Figure 4. Reserve size and deforestation rate in the 10-km-wide buffer along the outer perimeter. Indigenous reserves are larger than other reserves and many have high rates of deforestation along their perimeters.

Figure 5. The relationship between deforestation inhibition by indigenous reserves and (A) time since contact with white populations, and (B) population density. The ratio between the annual inside and outside deforestation rates determined between 1992 and 1996 provided the dependent variable for these analyses. Log-transformed regressions were used to test for a significant slope parameter, and in both cases the null hypothesis was accepted (P=0.91 for time since contact, and P=0.89 for population density), providing evidence that neither of these factors exerted much control over reserve performance during this time.