Ann. N.Y. Acad. Sci. ISSN 0077-8923

ANNALS OF THE NEW YORK ACADEMY OF SCIENCES

Reassessing the forest impacts of protection

The challenge of nonrandom location and a corrective method

Lucas Joppa[1] and Alexander Pfaff[1,2]

[1]Nicholas School of the Environment, Duke University School of the Environment, Durham, North Carolina, USA. [2]Sanford School of Public Policy, Duke University Public Policy, Economics, Environment, Durham, North Carolina, USA

Address for correspondence: Lucas Joppa, Box 90328, Levine Science Research Building, Duke University, Durham, NC 27708. lnj@duke.edu

Protected areas are leading tools in efforts to slow global species loss and appear also to have a role in climate change policy. Understanding their impacts on deforestation informs environmental policies. We review several approaches to evaluating protection's impact on deforestation, given three hurdles to empirical evaluation, and note that "matching" techniques from economic impact evaluation address those hurdles. The central hurdle derives from the fact that protected areas are distributed nonrandomly across landscapes. Nonrandom location can be intentional, and for good reasons, including biological and political ones. Yet even so, when protected areas are biased in their locations toward less-threatened areas, many methods for impact evaluation will overestimate protection's effect. The use of matching techniques allows one to control for known landscape biases when inferring the impact of protection. Applications of matching have revealed considerably lower impact estimates of forest protection than produced by other methods. A reduction in the estimated impact from existing parks does not suggest, however, that protection is unable to lower clearing. Rather, it indicates the importance of variation across locations in how much impact protection could possibly have on rates of deforestation. Matching, then, bundles improved estimates of the average impact of protection with guidance on where new parks' impacts will be highest. While many factors will determine where new protected areas will be sited in the future, we claim that the variation across space in protection's impact on deforestation rates should inform site choice.

Keywords: protected areas; parks; deforestation; effectiveness; impact; avoided deforestation; biodiversity; conservation; matching; regression

Introduction

Species extinctions are estimated to be occurring at approximately 100 times the baseline rate,[1,2] and habitat loss is the leading cause of species endangerment.[2,3] Protected areas have been the principle defense of habitat to this point, and they surely will continue to play a role in efforts to prevent species loss.[4,5] In addition, protected areas may now be looked to for support of carbon storage goals.

While habitat protection through land preservation goes back thousands of years (e.g., sacred groves in Africa), the first call to protect areas for this purpose is conventionally credited to Frederick Von Mueller in 1890 (as cited in Scott[6]). Since that time, a considerable area has been protected.[7] Most of the currently protected areas were established in the 20th century.[8–10] Protected areas now cover approximately 12% of Earth's land surface, and assessing the success of these conservation efforts is vital.[11] Protected areas benefit many plants, animals, and people,[12,13] yet they cost money and other resources, such as time and political capital. An important question in conservation, then, is whether protected areas actually "work."

Which type of protection one is talking about, and what it means for a protected area to "work" are important points to clarify. Protection is established for a variety of reasons or goals and with a range of land-use strictures (both within and across countries[14]).[a]

[a]While terminology is important, given that there are differences in the descriptors used in this field, we will use "protected areas" and "park" interchangeably to refer to protected lands of any sort.

An effort in 1993 by the World Conservation Union (IUCN) created an international categorization system that differentiates categories by management objectives. This complements protected area agreements, such as the Convention on Wetlands of International Importance (Ramsar), the World Network of Biosphere Reserves under UNESCO's Man and Biosphere Programme, the Convention Concerning the Protection of the World Cultural and Natural Heritage, and others, but it does not erase differences among commentators in the definition of success. Some look mainly at economic opportunities for people around or in protected areas (see the review by West *et al.*[15]), while others focus almost exclusively on ecological outcomes, such as whether animal populations remain stable[16,17] or resistance to biological invasion.[18,19]

We focus on the impact of protection on rates of deforestation (two studies we review also report on fires). Deforestation reduces natural forest habitat and thus nonhuman species' populations.[20] It is also important for carbon sequestration, and hence, climate change and is readily observable using remotely sensed data. Here, we focus on methods for evaluating protected areas' deforestation impacts. First we review previous evaluation efforts from a rich literature base. In this, our paper complements and updates a 2005 review by Naughton-Treves *et al.*[8] as well as two more recent reviews by Nagendra[21] and Campbell *et al.*[22] Second, we stand apart from these prior reviews by emphasizing the hurdles faced by methods previously used and highlighting a new method that we believe better addresses those hurdles and generates different findings. In brief, the failure of most previous assessments to consider and explicitly control for the landscape characteristics within protected areas can significantly bias their conclusions.

In most countries, protected areas are not randomly distributed across the landscape. While there are often good reasons for this, if the resulting distribution means that protected areas are biased to favor areas of lower threats of deforestation then it also indicates that most previously used methods will overestimate the impact of protection on reducing deforestation.

Matching—a statistical impact-evaluation technique widely used in economic evaluations of policy impact—addresses such nonrandom location bias. It seeks to explicitly force "apples to apples" comparisons by pairing protected and unprotected locations that are similar in their landscape characteristics. It does that more directly than prior methods, which fail to consider nonrandom location or to carefully describe and control for it. Matching can evaluate impacts within the boundaries of protection as well as assess the prevalence of "spillovers." We discuss recent applications of the matching method to review evidence that controlling for landscape characteristics can make a big difference on impact estimates. Compared to methods that ignore location characteristics, emerging evidence suggests that matching can significantly reduce impact estimates on national[23–25] and global scales.[26]

Matching provides not only (reduced) average impact estimates, however. Results from matching also directly imply where a new protected area's forest impact could be relatively high or low. An example of this would be where even effective protection could not contribute that much given the lack of threat to the landscape. While many factors determine where new protected areas are sited, information on the variation across space in protection's impact on forest can inform site choice.

In section 2, we review past evaluations of the impacts of protected areas on forest systems. In section 3, we discuss hurdles for typical previous methods. We use section 4 to describe "matching," while in section 5, we review the few studies that employ matching or matching-like methods. Section 6 then adds discussion.

Literature evaluating protected areas' impacts

We have not tried to comprehensively include all studies that analyze deforestation in and around protected areas. Further, the studies that we do mention overlap those discussed in recent reviews.[8] Our novel goal is to categorize the methods that are predominant in such literatures, citing some examples. We are, then, trying to be comprehensive concerning estimation methods.

The evaluation methods that we review use measures of forest within the protected area (or "treated" location). Deforestation is observed through direct visual observation or through remote, sensing, such as satellite imagery, radar, and aerial photography. Increasing access to remotely sensed products and increased skills in analyzing these products has meant that the remotely sensed data dominate the literature.[b] On-the-ground observational data are rare because of the large scale of the study areas; plot-level vegetation surveys are impractical when considering the large areas often involved, even if useful when analyzing small protected areas or if considering practices, such as selective logging.[c] As our focus is on the methods used to compare such data, beyond these comments we do not focus on the data themselves, despite their importance for quality results.

Compare to nowhere

To start, when evaluating protected areas one may be tempted to consider only the current state of the protected area, without attention to the status of surrounding areas. For instance, at least in informal public discussions, many people consider Costa Rica's protected areas as successful because they are almost fully forested. Alternatively, some protected areas seem to be thought of as unsuccessful because significant deforestation has taken place there.

Fuller *et al.* analyzed deforestation from 1996–2002 in Kalimantan (Indonesian Borneo).[27] The authors included all existing and proposed protected areas, comparing them with the unprotected portions of the region. Protected areas smaller than 100,000 ha had lost 23.6% of their forests, those between 100,000 and 250,000 ha had lost 52.4%, and areas larger than 250,000 ha had lost only 16.9%. Based upon these large percentages of forest loss, the authors suggest that Kalimantan's protected network was no longer viable.

However, it is problematic to condemn the viability of a network based on the forest loss in the protected areas alone. A reasonable perspective can only be achieved by comparing the status of the protected areas to unprotected areas. As an example in the case of the Kalimantan forests, if a study of the surrounding unprotected area showed that deforestation had occurred at a rate of 85%, then one might conclude that the protected network was working to reduce deforestation. Continuing hypothetically, if zero deforestation occurred within a protected area, one might conclude that protection deterred clearing. Yet the truth might be that protection had no impact at all, if deforestation likewise had not occurred elsewhere across the country either.

A 2001 study by Liu *et al.*,[28] which applies many assessment approaches (see below), appears to suggest that any degradation of Giant Panda habitat in the Wolong Nature Reserve (an important protected area for this species), could be considered a failure. At the end of Liu *et al.*'s study period in 1997, Zeng *et al.*[29] found human-impacted landscapes to cover approximately 30% of Wolong Nature Reserve. Yet if deforestation in unprotected locations that are similar to the protected area is higher than in the protected area, even imperfect protection had some gain.[d]

More generally, to assess impact one must compare what happened in protected areas to what would have happened in those same locations had the areas not been protected. It is not possible to observe what would have happened without protection. Thus, one must infer it from other locations. In the next three subsections we consider three methods of doing so, citing from prior studies, before moving on to the difficulties park location bias presents for these methods.

[b]A recent paper by Loarie *et al.*[74] provides an overview of the availability of these emerging technologies and their resulting data output.

[c]Banda *et al.*[75] used plot-level data to compare tree species richness across protected areas with differing management priorities in western Tanzania, finding that fully protected National Parks did not necessarily contain the highest diversity. Also, Hayes[37] used an extensive plot-level data set from 163 forest plots across 13 countries to examine vegetation density in protected (76 plots) and unprotected (87 plots), finding no difference in vegetation density between legally protected forests and those forests governed by users who establish and recognize their own forest management rules. Again in 2006, Bleher *et al.*[93] analyzed disturbance parameters at 22 plots across Kakamega Forest in western Kenya. Using these data they found high levels of human impact across the entire forest, but that logging was much less prevalent within highly protected National and Nature Reserves compared to high levels of logging within lesser protected Forest Reserves.

[d]The Chico Mendes extractive reserve located close to the InterOceanic Highway in Brazil provides an example of a protected area that has been cleared yet nevertheless has had an impact.

Compare to everywhere

One seemingly logical method to assess the impact of protected areas on deforestation is to compare deforestation inside a protected area to deforestation on all unprotected land outside. Thus, if no deforestation occurred inside, for instance, then one could estimate the amount of deforestation that was avoided by understanding the extent of regional deforestation outside.

Gaveau *et al.*[30] measured deforestation over the entire unprotected area on the Indonesian island of Sumatra and compared it to deforestation within three protected areas. From 1972 to 2002, the average rate of forest loss across the region was approximately 2.9% per year, with a total reduction of 84%. Because the deforestation rates inside the three protected areas they analyzed (2.7%, 2.1%, 0.64%) were lower than this outside rate, one would conclude that the protected areas reduced deforestation rates.

In similar work, albeit with a twist, Messina *et al.*[31] analyzed Cuyabeno Wildlife Production Reserve in the Ecuadorian Amazon. Parts of the area formerly protected had been re-gazetted to be "patrimony forests" where colonization and land titles were allowed. A comparison was made between the officially protected reserve and these patrimony forests. From 1986 to 1996, there was an increase in forest within the reserve but not within the various patrimony forests (which lost forest), and also not in the unprotected landscape, which experienced even higher deforestation than did the patrimony forests. This comparison suggests, at least implicitly, that the level of protection itself explains the differences.

In 1999, Sanchez-Azofeifa *et al.*[32] analyzed deforestation rates inside protected areas in the Sarapiqui region of Costa Rica and compared them with the rest of the region. Deforestation inside protected areas was low and decreased over time. The opposite was true outside protected areas. Sanchez-Azofeifa *et al.*[33] used the same approach for Corcovado National Park in the Osa Peninsula of Costa Rica. From 1979 to 1997, they detected no deforestation inside the park but high clearing outside, leaving the park with some of the region's only (43%) remaining forest cover.

The largest-scale assessment of protected area deforestation comes from DeFries *et al.*,[34] where the authors examined the isolation of protected tropical forests. This study used satellite imagery separated by 20 years to explicitly quantify deforestation inside versus 50 km zones outside 198 protected moist and dry tropical forests worldwide (the majority of which were also included in the Bruner *et al.*[35] analysis described below). They showed that approximately 70% of the regions outside of protected areas were cleared from 1982 to 2000, while only 25% of protected areas experienced deforestation.[e]

Compare to nearby land

The most common method of assessing the impact of protected areas on deforestation levels has been to compare deforestation within the protected area to that in a specific subset of land immediately outside protected areas—that is, the area immediately surrounding the site. In past studies this delineated area around the site is generally referred to as a "buffer zone."

In an early work, Bruner *et al.*[35] used survey data to analyze deforestation in and around 93 protected areas across 22 tropical countries. Survey respondents were asked to quantify the percentage of area cleared inside the protected area and within a 10 km buffer zone. Using these data, Bruner *et al.*[35] claimed that protected areas were largely effective (compared to the percent of forest cover at the time of park creation, more than 83% of protected areas had no net clearing, and more than 90% experienced less deforestation than their 10 km buffer). They also found that the results were predicted by the presence of management activities, such as law enforcement and boundary demarcation. This study has been widely cited, yet it also has been highly criticized.[36,37] Protected area managers likely have a vested interest in the perceptions of their "performance." Moreover, there is a lack of uniformity in how respondents measured or estimated deforestation, which can lead to widely varying deforestation estimates.

Liu *et al.*[28] provided not only the observations noted in 2.1 but also measured the difference in deforestation between the reserve and a 3 km buffer around the Wolong Reserve. In doing so, Liu *et al.*[28]

[e]This study could also fit Section 2.3 as it uses buffers. Yet, the 50 km buffer was used for a regional perspective we feel conforms more with "compare to everywhere" than the buffer-zone logic below.

found that deforestation inside the reserve was equal to or higher than in the buffer zone. Later, Viña *et al.*[38] revisited the Wolong analysis, analyzing habitat loss in temporal increments from 1965 to 2001. Across all time periods analyzed, "highly suitable" panda habitat was lost steadily both inside and outside the reserve.

Sanchez-Azofeifa *et al.*[39] analyzed 1960–1997 deforestation rates within Costa Rica, in and around 132 protected areas. Instead of comparing inside protected areas with all of the larger landscape (as in Sanchez-Azofeifa *et al.*'s[32] earlier analysis), this work compared land use within 0.5, 1.0, and 10.0 km buffer zones. Deforestation within all protected areas was negligible, and net forest growth actually occurred within the 1 km buffer zones. In the 10 km buffer, however, significant forest loss was found for all time periods. The authors used these results to claim that, in Costa Rica, "the boundaries of protected areas are respected."

The northern Guatemalan Maya Biosphere Reserve is composed of nine national parks and reserves and one mixed-use area. In a 2001 study, Sader *et al.*[40] compared deforestation in each of them to a buffer zone surrounding the entire biosphere reserve, across four time periods. Deforestation had occurred in more than one-third of grid cells in the biosphere reserve, but in no period was any protected area cleared as extensively as the buffer zone.

Working in the same region as Fuller *et al.*,[27] Curran *et al.*[41] showed that Kalimantan's protected forests declined by 56% from 1985 to 2001. Further, the authors provided a detailed case study of Gunung Palung National Park, comparing deforestation in the park to that in a 10 km buffer zone. For a while, the buffer zone was deforested while the park maintained its forest cover. However, after less than 26% of the buffer forest remained, the park was deforested at a rate of 9.5% per year.

Also in Indonesia, Linkie *et al.*[42] analyzed deforestation around Kerinci Seblat National Park in Sumatra. The park itself is 13,300 km^2, makes up approximately 37% of the region within which it is located, and was the first officially recognized protected area in Indonesia. From 1995 to 2001, it lost on average 0.28% of its forest per year. The unprotected region lost 0.96%, causing the authors to speculate that the legal existence of the park "played an important role."

Nagendra *et al.*[43] compared Celaque National Park in Honduras with a 5 km buffer zone, finding that very little (approximately 4%) of the park was deforested—only 1 km in the park's core experienced deforestation. This contrasted sharply with the approximately 25% of the buffer zone that was converted between 1989 and 2000[43] (see also Southworth *et al.*[44]).

In the Tadoba Andhari Tiger Reserve in India, a 2006 study[45] compared the central core of the reserve with a 2 km inner buffer zone *inside* the park boundary, a 2 km buffer *around* villages *inside* the boundary, and a 5 km boundary *outside* the reserve. The 2 km inner buffer zone represented the section most threatened by villages located on the park periphery between 1989 and 2001. Considering "stable forest" (i.e., forest that remained unchanged) during the study period, 64% of the reserve core, 62% of the 2 km inner buffer, and 62% of the 2 km village buffers remained stable. This contrasts with 36% of the outer 5 km buffer that remained stable.

Nepstad *et al.*[46] looked at deforestation and fire incidence in a 10 km strip inside versus a 10 km strip outside 15 parks, 121 indigenous lands, 10 extractive reserves, and 18 national forests in the Brazilian Amazon from 1997 to 2000. They found that deforestation ranged from 1.7 (extractive reserves) to 20 (parks) times higher in the 10 km buffer zone *outside* protected areas than in the 10 km buffer zone *inside* protected areas, yielding their claim that: "Reserves significantly reduced both deforestation and fire."

Analyzing fire incidence, Wright *et al.*[47] used 5, 10, and 15 km buffer zones around 823 tropical and subtropical moist forest reserves. For all 823, the authors found fire detection density to be significantly lower inside the reserve than in any of the three buffer zones. Using the inside–outside differential, they then found that poverty and corruption levels of the countries in question strongly predicted the park's effectiveness in reducing fire density.

Maiorano *et al.*[48] asked if size affected the ability of 716 protected areas in Spain to halt deforestation from 1990 to 2000. Like Wright *et al.*,[47] they examined three buffer widths (1, 2.5, and 5 km). They found that protected areas experienced less deforestation than buffers and claimed "the majority of parks have been effective at protecting the ecosystems within their borders" (p. 1301).

Compare to nearby time

Finally, another option is to analyze only the protected location while using temporal variation in clearing to do so. Thus, one might compare deforestation rates within the area before protection with deforestation in the same exact area after it was given legal protection, with the assumption being that any change in rate is due to the change in its status. This was done in, e.g., Liu *et al.*'s 2001 analysis of Wolong Nature Reserve,[28] where the authors examined both pre- and postreserve deforestation, finding that deforestation actually increased after the reserve had been established, with some counter-intuitive reasons offered for the clearing increase.

Gaveau *et al.*[30] also contrasted pre- and post-protection deforestation rates in Sumatran reserves. Protection covered approximately 54% of the landscape, and deforestation in the protected areas varied. Gunung Raya Wildlife Sanctuary lost 81% of its forest (approximately 2.7% yr^{-1}), the forest of the Hydrological Reserve were reduced by 62% (approximately 2.1%/yr), and the most highly protected area, Bukit Barisan Selatan National Park, lost 19% of its forest cover (0.64%/yr). However, Gaveau *et al.*[30] found deforestation rates unchanged when they analyzed pre- and postreserve trends. While Bukit Barisan Selatan National Park experienced 0.64% deforestation per year, this rate was no more or less than had occurred on the landscape prior to its protection.

Challenges for evaluating protected areas' impacts

Claims of protected-area effectiveness based on studies like those described above have been hugely influential in the conservation community. We argue that many of these claims are insufficiently supported. Core challenges for accurate inference have not yet been addressed.

Three issues can confound evaluations of the consequences of assigning protective status on an area. The first one (and the focus of this review) is that protection is often distributed nonrandomly across the landscape. This is as one may expect and often even prefer, given the spatial distributions of conservation priorities. However, this location bias can confound the methods in sections 2.2 and 2.3. The second issue is that protection may cause spatial spillovers—for example, decreased deforestation in other locations, including within the immediately adjacent buffer zone. Any such spillovers can confound comparisons with nearby land. The third issue is that rates of deforestation tend not to be constant over time; this can confound purely temporal comparisons.

Nonrandom location

Protected areas may remain forested due to the protection itself (we will call this *de jure* protection) or because the landscape characteristics of the protected lands discourage deforestation (*de facto* protection). In the latter case, protection may have no impact at all. Impact evaluation aims to reveal a causal (i.e., *de jure* as opposed to *de facto*) impact to inform conservation choices.[f]

If protection were randomly distributed across a landscape, regressions or other analyses comparing protected to unprotected areas would reveal protection's causal impact. The landscape characteristics of protected and unprotected areas would be the same, so that differences in such characteristics would not confound inferences about the degree to which protection mattered.

Yet protected area networks are very often not representative of their region's landscape.[49,g] There

[f] For instance, if protected areas are located at high elevation, on steep slopes, and in places with poor agricultural suitability, there is likely to be less threat of deforestation. This is exactly what Kinnaird *et al.*[76] find in their analysis of Bukit Barisan Selatan National Park. Inside the park, deforestation occurred at a rate of 25 km^2/year in lowland forests, and at only 3.9 km^2/year in hill/montane forests. The same trend occurred with flatter slopes (0–20°, 16.5 km^2/year) compared to steeper ones (>40°, 0.8 km^2/year). Nagendra *et al.*[43] also recognized this, claiming that the likely reason the park had remained forested was its topographic inaccessibility.

[g] At larger scales, analysis using spatial data sets to find gaps in land protection (commonly known as GAP analyses) has furthered our understanding of the bias in protected area locations. Such biases have been found at national and regional scales. Scott *et al.*[77] showed for the coterminous United States that protected areas were predominately located on higher elevation and less productive soils than were private lands. Worryingly, in the United States the greatest number of species is found at lower elevations. Similar results were found for Nepal's protected area network, with a large portion of the

are many reasons for this. They may be very good reasons in terms of gains from and costs of protection. Conservation priorities, land prices, and politics affect where protection occurs.[50,51] The phrase "rock and ice" refers to the tendency for protected areas to be located more frequently on less desirable lands within a landscape.[h] Also, the species that many protected areas are meant to protect are very likely to be nonrandomly distributed. We simply argue that landscape characteristics must be taken into account when inferring the impact of protection on observed forest outcomes.

The "compare to everywhere" approach in section 2.2 is particularly vulnerable to the problem of nonrandom distribution of protected lands. In this analysis, the deforestation rate on all unprotected lands provides the estimate for what would have happened to protected land were it not protected. "Everywhere" can be a poor comparison when the protected areas are located on landscapes with clearly different characteristics. Specifically, if the parks are on lands with lower deforestation probabilities, a "compare to everywhere" method likely provides an overestimate.

The "compare to nearby land" approach in section 2.3 can also be vulnerable to the nonrandom distribution problem.[i] A good example of this is Egmont National Park in New Zealand where the protected area contains a large volcanic cone and stops at the cone's lower edge. The extreme nature of the volcanic cone means that areas near the park may have a very different distribution of elevation. More generally, if there are thresholds or boundaries in the natural landscape, those who created protected areas were likely to be aware of them and even to have planned around them. This is commonly seen where protected area boundaries follow rivers, for example. Thus, adjacency may help with "apples to apples" comparisons but that is not guaranteed.

Spatial spillovers

The "compare to nearby land" approach is also vulnerable to the issue that deforestation in the buffer zone could be affected by the protected area nearby. For instance, if farmers were producing in an area that became protected, or were planning to, they may relocate production to the buffer zone. This "spillover" possibility, which can be local (as in this example) or global (if reduced production raises prices and thus production far away[52,53]), means that protected areas' total impacts in reducing deforestation are less than the observed impacts within their borders.

The problem for the "compare to nearby land" approach is that even if buffer-zone land is essentially the same as protected land, when factoring

---

network located at much higher elevations than most birds, mammals, or humans occupied.[78] In England, more than 60% of the land higher than 600 m has been protected in some way, but less than 20% of lands less than 400 m have been.[79] At a regional scale, in northeastern New South Wales, Australia, most protected lands are on lands with low soil productivity and/or steep slopes.[80] In an early analysis, Fearnside and Ferraz[81] found that of 111 vegetation zones in the Brazilian Legal Amazon, only 37 had any amount of protection. In the region of highest deforestation, only 1 of 10 zones was protected. In Costa Rica, Sanchez-Azofeifa *et al.*[39] found that two of the most dominant land-cover categories (tropical humid and wet forest premontane) were also two of the least protected. In an active example of this nonrandom bias in protected area location, Kerinci Seblat National Park's boundary was actually redrawn in 1999, from the original 1986 demarcation. Doing so allowed for the removal of valuable timber from protection.[82] As Messina[31] makes clear, this is not as uncommon as it may seem.

[h]Globally, there has yet to be a comprehensive analysis of the distribution of protected area networks relative to geophysical attributes, such as elevation, slope, and agricultural suitability (although Rodrigues *et al.*[83,84] do so for species coverage). We currently have such an analysis under way.[49] Our preliminary findings suggest that across multiple countries, protected and unprotected lands often differ in regard to variables affecting deforestation. This provides further proof that location must be accounted for when assessing the effectiveness of protected areas.

[i]It seems that 10 km has been chosen as a standard buffer. Intuitively, the shorter the distance one chooses from the boundary the more representative of the protected area the sample should be. Liu *et al.*[28] chose a 3 km buffer zone because it most closely matched the topology of the Wolong Nature Reserve, although their choice was also constrained by the lack of remotely sensed imagery beyond 3 km. Although Defries *et al.*[34] chose a 50 km buffer in their study on the grounds of analyzing regional trends, choosing such a large buffer most likely decreases the relevance of their conclusions on park effectiveness.

in leakage the buffer deforestation may be higher than what would have been observed in the buffer if there had not been a protected area. Thus, observed buffer deforestation is not a good estimate of what would have happened in the protected area without protection.[j] Specifically, with leakage, the buffer's observed deforestation would be an *overestimate* of what would have happened within the park without legal protection.

The buffer-comparison approach can also *underestimate* park impact if spatial spillovers *reduce* clearing. Creating a protected area could draw tourists and thus could lead farmers nearby to create forested supplementary tourist attractions (this could explain, e.g., Sanchez-Azofeifa *et al.*'s[39] increased forest growth in the 1 km buffer). In either direction, a nonzero spatial spillover from the protected area can confound estimates of avoided deforestation from comparing to the buffer.

Temporal shifts

The "compare to nearby time" approach in section 2.4 may have had all the challenges above in mind. Using solely the protected site seems an excellent response to the problems with comparing sites that differ in characteristics. Here land characteristics are the same but protection changes. Protection is assumed to be the causal mechanism in any change in deforestation rates.

This approach is severely hampered, though, if deforestation rates would not have been constant over time in the absence of protection. Since this approach uses past deforestation to estimate what would have happened, if other possible drivers can shift deforestation over time then this approach can mistakenly include their effects in evaluating the impacts of protection.

Consider, for instance, that annual clearing in the Brazilian Amazon over recent decades varied significantly.[54,55] Demand for timber can rise and fall drastically over 20 years, as can currency exchange rates, the productivity of new variants of soybeans, and any other number of potentially critical drivers of deforestation rates, including government subsidies. This evidence of shifts in clearing rates over time suggests limits on the "compare to nearby time" method.

Matching's explicit controls address challenges

Commonly used in the field of program evaluation,[56–58] matching methods that seek to compare "apples to apples" are useful here. In short, if we know deforestation is affected by land characteristics, such as elevation and slope, then comparisons of protected and unprotected lands that differ greatly along these dimensions are "apples to oranges." Matching selects the subset of the unprotected lands that have characteristics most similar to the land within the protected areas. The more similar the protected and the matched or selected unprotected groups, the cleaner or more "apples to apples" the comparison. This improves estimates of protection's impact.

The meaning of "similar"

People's opinions will differ about the meaning of "similarity." Two different matching estimators dominate recent applications and they use quite different definitions. The first is a propensity score estimator[59,60] defining similarity based upon an estimated probability of being treated.[k] In the case of matching for protected compared to unprotected lands, that probability is "how likely to be protected is a parcel?". That probability estimate is the index of similarity used to select which unprotected parcels are most similar to a given protected parcel. It is generated by a regression explaining the presence of

[j]One incremental improvement on the standard buffer method is to analyze several different buffer sizes to check for robustness of results across them.[39,47] A concern when using buffers is that an arbitrary buffer-zone size can result in a buffer with significantly more or less area than the park itself. This problem is particularly exacerbated when analyzing and comparing results across many protected areas simultaneously.

[k]Many choices remain to be made even upon choosing this form of defining similarity for matching. For instance, one might choose a fixed number of unprotected parcels, e.g., the three most similar, to construct the comparison set for the protected areas. Yet, a natural alternative would be to set an "acceptable similarity window" defining how good a match is required to be for inclusion and to let the number of matches be endogenous to the quality of the match for each treated observation.

a protected area using landscape characteristics. Regression followed by prediction of this probability is one way of collapsing the differences in the parcels along multiple dimensions into a single index in order to summarize similarity in a useable way.

Perhaps more intuitive is the nearest-neighbors covariate matching estimator.[61] This method defines similarity without a first-stage regression. It uses a multidimensional distance, within the space of all the land characteristics, between the treated and matched land parcels.[l]

For any definition of similarity, after the best possible control comparison set is chosen, one must check how equivalent or balanced are the resulting sets of parcels to be compared. A first check is to compare average characteristics in protected areas and the subset of unprotected areas chosen by matching. The selected subset should be significantly more similar to protected areas than is the full set of unprotected areas. Ideally, the subset should be statistically equal.

The importance of match quality

What happens if the matched unprotected lands remain different from the protected lands in terms of characteristics that affect deforestation? This might not be seen as a major problem, since a standard regression of deforestation on protection could control for such characteristics.

However, if the characteristics of protected and unprotected lands are quite different (e.g., if their distributions overlap somewhat but for the most part do not), the burden on a regression specification can be considerable. In an early work on matching, Rubin states:

> Unless the regression equation holds in the region in which observations are lacking, covariance will not remove all the bias, and in practice may remove only a small part of it. Secondly, even if the regression is valid in the no man's land, the standard errors of the adjusted means become large, because the standard error formula in a covariance analysis takes account of the fact that extrapolation is being employed. Consequently the adjusted differences may become insignificant merely because the adjusted comparisons are of low precision. When the groups differ widely in x, these differences imply that the interpretation of an adjusted analysis is speculative rather than soundly based" (p. 15).[62]

We emphasize that this challenge to control using a regression applies to matching as well when the quality of matching is poor—that is, if protected and matched unprotected lands are quite different. This makes it more likely that effects of landscape characteristics confound results.[m]

Revisiting the three challenges

Matching addresses the issue of nonrandom location of protection raised in section 3.1. To the extent possible, matching forces the characteristics of the unprotected control group to be the same as the protected areas being evaluated. If similar unprotected lands do not exist because the protected lands are simply too different, then matching explicitly documents that constraint.

Matching has no trouble with the issue of temporal shifts raised in section 3.3 because it compares within a time period. The challenge of spatial spillovers raised in section 3.2 can also be addressed with attention. If spatial spillovers may exist, then one must specify that the control points will not be drawn from the buffer zone, i.e., areas contiguous to the area being evaluated.

[l] The computation of standard errors is another difference in technique. Abadie and Imbens[85] show that bootstrapping standard errors is invalid with nonsmooth, nearest-neighbor estimators, such as the propensity score matching estimator with a fixed number of matches (contrasted with kernel versions that assign smoothly declining weights to progressively less-well-matched untreated observations). For propensity score matching, many follow Hill *et al.*[60] in calculating weighted standard errors while Abadie and Imbens[61] provide the covariate matching standard errors.

[m] Crump *et al.*[86] address the issue of a lack of covariate overlap, noting that many common estimators become sensitive to the choice of specification (much as Cochran[62] had noted for regression and following also related prior work that includes the studies by Heckman, Ichimura, and Todd.[56,57]) Crump *et al.*[86] characterize optimal subsamples for which treatment effects can be estimated most precisely, which under some conditions can be characterized by a rule based on the propensity score.

Controlling for characteristics in evaluating impact

Without matching

Controlling for distance from protection

Joppa *et al.*[11] examine four regions of moist tropical forest—the two largest remaining tracts of such forest (Amazon and Congo regions) and two of Norman Myers' original "biodiversity hotspots"[4] (the West Coast of Africa and the Atlantic Coast of South America). For areas within different management categories, they describe percent deforestation, at 2 km intervals, from 30 km inside protected areas to 30 km outside.[n] This shows change as the protected area boundary approaches. It helps to reveal "leakage,"[63] as disproportionate increases in deforestation on the nearby unprotected landscape can be readily observed.

Distance from protection as a summary characteristic is particularly revealing when more remote regions are compared to more human-dominated areas. Joppa *et al.* found that in regions (Amazon and Congo) where the majority of protected areas are large and remote, there is little deforestation inside *or* outside of the boundaries. These areas are most likely *de facto* protected due to their isolation. In the "hotspot" regions distinguished by both high species endemism and high rates of habitat loss, the story differs. Deforestation dominates at distances greater than several kilometers from protected areas, but land cover abruptly switches to natural vegetation at park boundaries. The transition to natural vegetation at boundaries suggests *de jure* protection, yet controlling for other characteristics would help make the case.

Regression to control for multiple characteristics

If protected areas differ in their land characteristics, one might strip out the effects of those differences using regressions.[o] One of the earliest studies to try this was done by Deininger and Minten[64] for the Mexican states of Chiapas and Oaxaca. The study combined biophysical data (slope, soil fertility) with property rights, irrigation, and socioeconomic data (e.g., distance to infrastructure, human population density) and protection. Looking at deforestation for a single point in time (1980), the authors found altitude, distance to infrastructure, slope, and protection to be the most important determinants of forest cover—these characteristics encompassed more than 70% predictive accuracy. Controlling for those characteristics permitted an estimate of what would have happened on currently protected lands without protection. The authors predicted that protected areas without protection would have been 43% deforested rather than the 9% observed.

For northern Thailand, Cropper *et al.*[65] estimated a bivariate probit model for separate probabilities that a location was cleared and protected. Combined, these allowed an estimate of the impact of protection on clearing. Steep slopes, high elevation, and poor agricultural suitability accurately predicted forest cover. Given the *de facto* protection derived from park locations, the estimated *de jure* effect of protected areas on forest was not significantly different from zero for national parks and wildlife sanctuaries together. Wildlife sanctuaries alone did have some effect.

For Tanzania, Pelkey *et al.*[66,67] compare protected and unprotected lands controlling for elevation, slope, and distances to roads and refugee camps. Logistic regression models find that national parks and game reserves increase vegetation gains. Forest reserves and the Ngorongoro Conservation Area showed insignificant vegetation gains, while game-controlled and open areas displayed insignificant losses.[66] Pelkey *et al.*'s 2003 results differed only slightly from this.[67]

Matching (and related) results

To convey the basic points, we focus on two papers that use matching to estimate avoided deforestation from protected areas in Costa Rica: Andam *et al.*[23] find much lower impacts using matching, having applied not only matching but also other methods for comparison; and Pfaff *et al.*[24] show that subsets of the protected areas based on observable characteristics of locations have very different forest impacts,

---

[n]Joppa *et al.*[87] also use this method also to study human population growth around parks.

[o]There is a rich literature modeling deforestation using regressions that include many such land characteristics, with several comprehensive reviews.[88,89] Although most studies we review below borrow significant methodologies from this literature, we wish to restrict our focus to those studies that have explicitly considered the role that protected areas play in reducing deforestation.

such that policy makers could target impact or additionality.[p] Other recent efforts examine Sumatra (Indonesia),[25] Mexico,[68] the InterOceanic Highway area,[69] the entire Brazilian Amazon,[70] and the entire globe (albeit with less detailed data[26]). In all of these cases, both a reduction in average impact and variation across space in protection's forest impacts are key results.[q] Prior to the discussion of applications of matching, however, first we review several studies that have used techniques similar to formal matching.

Matching-like analyses

Vogt *et al.*[71] analyzed protected areas in Uganda. Along the lines of matching's "apples to apples" comparisons, they assert that legal protection must be a factor in the stability of forest cover in the region because deforestation occurred less on the legally protected lands than on the unprotected lands even when there were similar soil characteristics between the two land areas.

Oliveira *et al.*[72] examined land around Peruvian protected areas. They quantified both deforestation and forest disturbance; we refer to *damage* as a combination of the two. While Peru averaged approximately 1277 km^2 of damage per year, only 1–2% of this occurred in protected areas. Noting the important connection between roads and forest damage in Peru (approximately 75% of deforestation occurred within 20 km of roads), Oliveira *et al.*[72] analyzed land within 20 km of a road inside and outside of protected areas. They found that protected forests near roads were more than four times as likely to remain forested than were unprotected lands near roads.

Mas[73] estimated the effectiveness of Calakmul Biosphere Reserve in Mexico. First, he used a chi-square test and logistic regression to determine five variables (soil type, elevation, slope, distance from settlement, distance from roads) that most influenced deforestation in the region surrounding the reserve. Next, he quantified the area belonging to each of the unique combinations of the variables (variables were binned to decrease the possible combinations). Mas then set a buffer area and retained any unprotected land that contained a combination of the five deforestation drivers in the same proportion as any piece of land within the reserve. The rest were discarded, resulting in what he referred to as a "similar buffer area." Inside the reserve, deforestation was 0.3%/year. The buffer lost forest at a rate of 1.3%/year. However, deforestation within the similar buffer area was only 0.6%/year, indicating that the estimated impact of protection was considerably lower than would have been predicted by other methods.

Matching results for Costa Rica

For Costa Rica, Andam *et al.*[23] used matching for more than 150 protected areas over the period of 1960–1997. During the 1960s and 1970s, Costa Rica had one of the world's highest deforestation rates. Andam *et al.*[23] controlled for land productivity and distances from forest edge, roads, and cities, and required a perfect match for land productivity (a dominant factor in protected area location, as most parks were on medium/medium-low productivity lands). Matching on these covariates greatly increased the similarity values of each covariate from

[p]Related work exists on avoided deforestation from payments for ecosystem services. Sanchez-Azofeifa *et al.*[90] use a regression approach while Pfaff *et al.*[91] for 1997–2000 and Robalino *et al.*[92] for 2000–2005 apply matching methods to address nonrandom payment allocation in Costa Rica. Juxtaposed, the latter show potential to shift policy implementation as nonrandomness vanished.

[q]Just as no global analysis has yet to be completed on quantifying location bias in protected area networks, no global matching effort has been attempted on protected area impact. Similar to our location bias efforts, we have such a global matching analysis underway.[26] Our preliminary results from a random sample of countries suggest that results vary significantly across space. We have verified existing results from Costa Rica, Brazil, and Mexico and show that compared to the "compare to everywhere" method, matching techniques significantly lower estimated impact of protected areas on avoided natural land conversion. Moving to countries with no previous results to which to compare ours, we find similar trends in Congo, Mongolia, Tanzania, and the United States. In some countries, such as Niger and Mozambique, matching provides little improvement on the "Compare to Everywhere" technique. In Chad and Sudan, however, matching estimates increase the estimated impact. Further analysis of all countries, as well as a combined global estimate, are required before any significant conclusions can be drawn. Besides the general impact estimates, it will be interesting to see the role country-level attributes, such as size, play in determining the impact of a country's protected network.

the unmatched to matched samples, creating a much more representative control sample.

Estimating avoided deforestation using a covariate-balanced sample, Andam *et al.*[23] calculated that approximately 11% of the protected sites would have been deforested in the absence of protection. This result was robust to use of a "caliper," which is a technique to set a threshold concerning the minimal acceptable similarity in the matching. For the same data, the "compare to everywhere" method estimated that 44% of protected locations would have been deforested, while a "compare to nearby land" approach with 10 km buffer zone yielded an impact estimate of 38%.[r]

With an eye to guiding future protected area siting, Pfaff *et al.*[24] revisited Costa Rica protected areas using matching methods for 1986–1997. As in Andam *et al.*,[23] matching impact estimates were found to be less than a third of other methods' findings, but the focus of this study was on variation in park impacts.[50]

Pfaff *et al.*[24] found that protected areas within 85 km of San Jose, a major metropolitan area, had an impact of 3% avoided deforestation during the period studied, while those further away had an impact of only around 1%. For parks within 6 km of a national road, they find that 5% of the forest was saved while in forests farther away the impact was essentially zero. Slope, a good proxy for agricultural suitability, was a critical factor in assessing the impact of protection. Parks on flatter land blocked 14% deforestation but protecting steeper lands had close to zero impact.[s] As these dimensions that affect impact are easily observed, one may use them to target areas where protection's impacts might be greatest.

[r]In Andam *et al.*[23] the percent differences between matching and traditional results are large. In actual areas, matching estimates that of the ∼483,000 ha of land protected from 1960 to 1980, ∼54,000–60,000 ha remained forested due to protection. Conventional estimates are ∼ 181,000–240,000 ha.

[s]The authors note that one need also take care in interpreting results. Pfaff *et al.*[24] get a result that smaller parks have greater impact than larger ones, but smaller parks tend to be on flatter lands, nearer to roads and nearer to San Jose. Additionally, it may be financially or politically difficult to put a larger park on more highly pressured lands.

Discussion

Protected areas are often not distributed equally across landscapes. This location bias, no matter why it comes about, affects the potential impact of protected areas upon deforestation. Only by controlling for landscape characteristics can the best possible estimates of impact be obtained. To this point, improved controls for landscape characteristics considerably lower estimates of average impact. However, the same methods of explicitly characterizing protection's context can guide future protected area siting with an eye toward raising potential impact.

Good estimates are important because protected areas have been and may remain the leading global conservation tool. Nations have relied on them heavily in the past and very likely will continue to do so into the foreseeable future for biodiversity preservation and also for climate change mitigation.

That new methods of assessment lower the estimated impact of existing protected areas does not mean that protection is incapable of preventing deforestation. Rather, it highlights the importance of variation across space in the potential for avoiding deforestation; in some places, protection has little to no impact since almost no deforestation would have occurred even in the absence of protection. In light of this variation in potential impact, one might preferentially locate protected areas where their impact would be highest. However, in doing this there will be tradeoffs with benefits (like ecosystem uniqueness) and costs (like land prices) of protection. Local management practices, funding levels, effective law enforcement, and community involvement and support all affect impact. When allocating these scarce or costly resources, planners can consider these tradeoffs and choose to apply them in the places they can be most effective.

Conflicts of interest

The authors declare no conflicts of interest.

References

1. Pimm, S. *et al*. 1995. The future of biodiversity. *Science* **269:** 347–350.
2. Pimm, S. & P. Raven. 2000. Biodiversity: extinction by numbers. *Nature* **403:** 843–845.
3. Pimm, S. *et al*. 2001. Can we defy nature's end? *Science* **293:** 2207–2208.

4. Myers, N. *et al*. 2000. Biodiversity hotspots for conservation priorities. *Nature* **403:** 853–858.
5. Pimm, S. 2001. *The World According To Pimm: A Scientist Audits the Earth*. McGraw-Hill. New York, NY.
6. Scott, J. 1999. A representative biological reserve system for the United States. *Conserv. Biol. Newsl.* **6:** 1.
7. Brooks, T.M. *et al*. 2004. Coverage provided by the global protected-area system: is it enough? *Bioscience* **54:** 1081–1091.
8. Naughton-Treves, L., M.B. Holland & K. Brandon. 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annu. Rev. Environ. Resour.* **30:** 219–252.
9. Zimmerer, K.S., R.E. Galt & M.V. Buck. 2004. Globalization and multi-spatial trends in the coverage of protected-area conservation (1980–2000). *Ambio* **33:** 520–529.
10. Chape, S. *et al*. 2005. Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philos. Trans. R. Soc. B: Biol. Sci.* **360:** 443–455.
11. Joppa, L., S. Loarie & S. Pimm. 2008. On the protection of "protected areas". *Proc. Natl. Acad. Sci. USA* **105:** 6673.
12. Hansen, A. & R. DeFries. 2007. Ecological mechanisms linking protected areas to surrounding lands. *Ecol. Appl.* **17:** 974–988.
13. Karanth, K. 2007. Making resettlement work: the case of India's Bhadra Wildlife Sanctuary. *Biol. Conserv.* **139:** 315–324.
14. Locke, H. & P. Dearden. 2005. Rethinking protected area categories and the new paradigm. *Environ. Conserv.* **32:** 1–10.
15. West, P., J. Igoe & D. Brockington. 2006. Parks and peoples: the social impact of protected areas. *Annu. Rev. Anthropol.* **35:** 251–277.
16. Newmark, W. 1995. Extinction of mammal populations in Western North American National Parks. *Conserv. Biol.* **9:** 512–526.
17. Woodroffe, R. & J. Ginsberg. 1998. Edge effects and the extinction of populations inside protected areas. *Science* **280:** 2126–2128.
18. Pauchard, A. & P. Alaback. 2004. Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of south-central Chile. *Conserv. Biol.* **18:** 238–248.
19. Pysek, P., V. Jarosỉk & T.S. Kucera. 2002. Patterns of invasion in temperate nature reserves. *Biol. Conserv.* **104:** 13–24.
20. Baillie, J., C. Hilton-Taylor & S. Stuart. 2004. *IUCN Red List of threatened species. A global species assessment*. IUCN. Gland, Switzerland and Cambridge, UK.
21. Nagendra, H. 2008. Do parks work? Impact of protected areas on land cover clearing. *Ambio* **37:** 330–337.
22. Campbell, A., S. Clark, L. Coad, *et al*. 2008. Protecting the future: carbon, forests, protected areas and local livelihoods. *Biodiversity* **9:** 117–122.
23. Andam, K. *et al*. 2008. Measuring the effectiveness of protected area networks in reducing deforestation. *Proc. Natl. Acad. Sci. USA* **105:** 16089–16094.
24. Pfaff, A. *et al*. 2009. Location affects protection: observable characteristics drive park impacts in Costa Rica. *B.E. J. Econ. Anal. Pol.* **9:** 1–24.
25. Gaveau, D.L.A. *et al*. 2009. Evalutating whether protected areas reduce tropical deforestation in Sumatra. *J. Biogeogr* **36:** 2165–2175.
26. Joppa, L. & A. Pfaff. 2009. Global Park Impacts: How Much Deforestation Has Protection Avoided? Duke University Nicholas School of the Environment Working Paper.
27. Fuller, D., T. Jessup & A. Salim. 2004. Loss of forest cover in Kalimantan, Indonesia, since the 1997–1998 El Nino. *Conserv. Biol.* **18:** 249–254.
28. Liu, J.G. *et al*. 2001. Ecological degradation in protected areas: the case of Wolong Nature Reserve for giant pandas. *Science* **292:** 98–101.
29. Zeng, H., D.Z. Sui & X.B. Wu. 2005. Human disturbances on landscapes in protected areas: a case study of the Wolong Nature Reserve. *Ecol. Res.* **20:** 487–496.
30. Gaveau, D.L.A., H. Wandono & F. Setiabudi. 2007. Three decades of deforestation in southwest Sumatra: have protected areas halted forest loss and logging, and promoted re-growth? *Biol. Conserv.* **134:** 495–504.
31. Messina, J.P. *et al*. 2006. Land tenure and deforestation patterns in the Ecuadorian Amazon: conflicts in land conservation in frontier settings. *Appl. Geogr.* **26:** 113–128.
32. Sanchez-Azofeifa, G.A. *et al*. 1999. Protected areas and conservation of biodiversity in the tropics. *Conserv. Biol.* **13:** 407–411.
33. Sanchez-Azofeifa, G.A. *et al*. 2002. Dynamics of tropical deforestation around national parks: remote sensing of forest change on the Osa Peninsula of Costa Rica. *Mountain Res. Dev.* **22:** 352–358.
34. DeFries, R. *et al*. 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. *Ecol. Appl.* **15:** 19–26.
35. Bruner, A. *et al*. 2001. Effectiveness of parks in protecting tropical biodiversity. *Science* **291:** 125–128.

36. Vanclay, J.X. 2001. The effectiveness of parks. *Science* **293:** U1–U2.
37. Hayes, T.M. 2006. Parks, people, and forest protection: an institutional assessment of the effectiveness of protected areas. *World Dev.* **34:** 2064–2075.
38. Viña, A. *et al*. 2007. Temporal changes in Giant Panda habitat connectivity across boundaries of Wolong Nature Reserve, China. *Ecol. Appl.* **17:** 1019–1030.
39. Sanchez-Azofeifa, G.A. *et al*. 2003. Integrity and isolation of Costa Rica's national parks and biological reserves: examining the dynamics of land-cover change. *Biol. Conserv.* **109:** 123–135.
40. Sader, S. *et al*. 2001. Forest change monitoring of a remote biosphere reserve. *Int. J. Remote Sensing* **22:** 1937–1950.
41. Curran, L. *et al*. 2004. Lowland forest loss in protected areas of Indonesian Borneo. *Science* **303:** 1000–1003.
42. Linkie, M., R.J. Smith & N. Leader-Williams. 2004. Mapping and predicting deforestation patterns in the lowlands of Sumatra. *Biodivers. Conserv.* **13:** 1809–1818.
43. Nagendra, H. *et al*. 2004. Monitoring parks through remote sensing: studies in Nepal and Honduras. *Environ. Manage.* **34:** 748–760.
44. Southworth, J. *et al*. 2004. Assessing the impact of Celaque National Park on forest fragmentation in western Honduras. *Appl. Geogr.* **24:** 303–322.
45. Nagendra, H., S. Pareeth & R. Ghate. 2006. People within parks—forest villages, land-cover change and landscape fragmentation in the Tadoba Andhari Tiger Reserve, India. *Appl. Geogr.* **26:** 96–112.
46. Nepstad, D. *et al*. 2006. Inhibition of Amazon deforestation and fire by parks and indigenous lands. *Conserv. Biol.* **20:** 65–73.
47. Wright, S.J. *et al*. 2007. Poverty and corruption compromise tropical forest reserves. *Ecol. Appl.* **17:** 1259–1266.
48. Maiorano, L., A. Falcucci & L. Boitani. 2008. Size-dependent resistance of protected areas to land-use change. *Proc. R. Soc. B: Biol. Sci.* **275:** 1297–1304.
49. Joppa, L.N. & A. Pfaff. 2009. High & Far: How the World's Protected Areas Have Avoided Threat. Duke University Nicholas School of the Environment Working Paper.
50. Pfaff, A. & G.A. Sanchez-Azofeifa. 2004. Deforestation pressure and biological reserve planning: a conceptual approach and an illustrative application for Costa Rica. *Resour. Energy Econ.* **26:** 237–254.
51. Ando, A. *et al*. 1998. Species distributions, land values, and efficient conservation. *Science* **279:** 2126–2128.
52. Sohngen, B., R. Mendelsohn & R. Sedjo. 1999. Forest management, conservation, and global timber markets. *Am. J. Agric. Econ.* **81:** 1905–1910.
53. Sohngen, B. & S. Brown. 2004. Measuring leakage from carbon projects in open economies: a stop timber harvesting project in Bolivia as a case study. *Can. J. Forest Res.* **34:** 829–839.
54. Skole, D. & C. Tucker. 1993. Tropical deforestation and habitat fragmentation in the Amazon: satellite data from 1978 to 1988. *Science* **260:** 1905.
55. Achard, F. *et al*. 2002. Determination of deforestation rates of the world's humid tropical forests. *Science* **297:** 999–1002.
56. Heckman, J., H. Ichimura & P. Todd. 1997. Matching as an econometric evaluation estimator: evidence from evaluating a job training programme. *Rev. Econ. Stud.* **64:** 605–654.
57. Heckman, J., H. Ichimura & P. Todd. 1998. Matching as an econometric evaluation estimator. *Rev. Econ. Stud.* **65:** 261–294.
58. Dehejia, R. & S. Wahba. 2002. Propensity score-matching methods for nonexperimental causal studies. *Rev. Econ. Stat.* **84:** 151–161.
59. Rosenbaum, P. & D. Rubin. 1983. The central role of the propensity score in observational studies for causal effects. *Biometrika* **70:** 41–55.
60. Hill, J., J. Brooks-Gunn & J. Waldfogel. 2003. Sustained effects of high participation in an early intervention for low-birth-weight premature infants. *Dev. Psychol.* **39:** 730–744.
61. Abadie, A. & G. Imbens. 2006. Large sample properties of matching estimators for average treatment effects. *Econometrica* **74:** 235–267.
62. Cochran in Rubin, D. 2006. William G. Cochran's contributions to the design, analysis, and evaluation of observational studies. In *Matched Sampling for Causal Effects*. D. Rubin (Ed.): 7–29. Cambridge University Press. New York.
63. Ewers, R. & A. Rodrigues. 2008. Estimates of reserve effectiveness are confounded by leakage. *Trends Ecol. Evol.* **23:** 113–116.
64. Deininger, K. & B. Minten. 1996. Determinants of Forest Cover and the Economics of Protection: An Application to Mexico. Research Project on Social and Environmental Consequences of Growth-Oriented Policies Working Paper.
65. Cropper, M., J. Puri & C. Griffiths. 2001. Predicting the location of deforestation: the role of roads and protected areas in North Thailand. *Land Econ.* **77:** 172.
66. Pelkey, N., C. Stoner & T. Caro. 2000. Vegetation in Tanzania: assessing long term trends and effects of protection using satellite imagery. *Biol. Conserv.* **94:** 297–309.

67. Pelkey, N., C. Stoner & T. Caro. 2003. Assessing habitat protection regimes in Tanzania using AVHRR NDVI composites: comparisons at different spatial and temporal scales. *Int. J. Remote Sensing* **24:** 2533–2558.
68. Zepeda, Y. *et al*. 2009. Evaluating the Impacts of Mexican Protected Areas on Deforestation from 1993–2000. Resources for the Future Working Paper.
69. Delgado, C. *et al*. 2008. Will Nearby Protected Areas Constrain Road Impacts On Deforestation? Presentation at the NASA LBA conference 'Amazon In Perspective', Manaus.
70. Pfaff, A. 2009. Evaluating deforestation impacts of protected areas. Presented at Connecting Amazon Protected Areas and Indigenous Lands to REDD Frameworks, Stanford, CA.
71. Vogt, N.D. *et al*. 2006. Understanding the stability of forest reserve boundaries in the West Mengo region of Uganda. *Ecol. Soc.* **11:** 1–22.
72. Oliveira, P. *et al*. 2007. Land-use allocation protects the Peruvian Amazon. *Science* **317:** 1233–1236.
73. Mas, J.F. 2005. Assessing protected area effectiveness using surrounding (buffer) areas environmentally similar to the target area. *Environ. Monit. Assess.* **105:** 69–80.
74. Loarie, S., L. Joppa & S. Pimm. 2007. Satellites miss environmental priorities. *Trends Ecol. Evol.* **22:** 630–632.
75. Banda, T., M.W. Schwartz & T. Caro. 2006. Woody vegetation structure and composition along a protection gradient in a miombo ecosystem of western Tanzania. *Forest Ecol. Manage.* **230:** 179–185.
76. Kinnaird, M.F. *et al*. 2003. Deforestation trends in a tropical landscape and implications for endangered large mammals. *Conserv. Biol.* **17:** 245–257.
77. Scott, J. *et al*. 2001. Nature reserves: do they capture the full range of America's biological diversity? *Ecol. Appl.* **11:** 999–1007.
78. Hunter, M. & P. Yonzon. 1993. Altitudinal distributions of birds, mammals, people, forests, and parks in Nepal. *Conserv. Biol.* **7:** 420–423.
79. Oldfield, T. *et al*. 2004. A gap analysis of terrestrial protected areas in England and its implications for conservation policy. *Biol. Conserv.* **120:** 303–309.
80. Pressey, R. *et al*. 2002. Effectiveness of protected areas in north-eastern New South Wales: recent trends in six measures. *Biol. Conserv.* **106:** 57–69.
81. Fearnside, P. & J. Ferraz. 1995. A conservation gap analysis of Brazil's Amazonian vegetation. *J. Soc. Conserv. Biol.* **9:** 1134–1147.
82. Linkie, M. *et al*. 2008. Evaluating biodiversity conservation around a large Sumatran protected area. *Conserv. Biol.* **22:** 683–690.
83. Rodrigues, A. *et al*. 2004. Effectiveness of the global protected area network in representing species diversity. *Nature* **428:** 640–643.
84. Rodrigues, A.S.L. *et al*. 2004. Global gap analysis: priority regions for expanding the global protected-area network. *Bioscience* **54:** 1092–1100.
85. Abadie, A. & G. Imbens. 2006. On the Failure of the Bootstrap for Matching Estimators. NBER Working Paper.
86. Crump, R. *et al*. 2008. Moving the Goalposts: Addressing Limited Overlap in the Estimation of Average Treatment Effects by Changing the Estimand. NBER Working Paper.
87. Joppa, L., S. Loarie & S. Pimm. 2009. On population growth near protected areas. *PLoS ONE.* **4:** e4279.
88. Kaimowitz, D. & A. Angelsen. 1998. *Economic Models of Tropical Deforestation: A Review*. Center for International Forestry Research. Jakarta.
89. Lambin, E. 1997. Modeling and monitoring land-cover change processes in tropical regions. *Progr. Phys. Geogr.* **21:** 375–393.
90. Sanchez-Azofeifa, G.A. *et al*. 2007. Costa Rica's payment for environmental services program: intention, implementation, and impact. *Conserv. Biol.* **21:** 1165–1173.
91. Pfaff, A., J. Robalino & G.A. Sanchez-Azofeifa. 2009. Payments for Environmental Services: Empirical Analysis for Costa Rica. Terry Sanford Institute for Public Policy Working Paper.
92. Robalino, J. *et al*. 2009. Changing the Deforestation Impacts of Ecopayments: Evolution (2000–2005) in Costa Rica's PSA Program. Terry Sanford Institute for Public Policy Working Paper.
93. Bleher, B., D. Uster & T. Bergsdorf. 2006. Assessment of threat status and management effectiveness in Kakamega Forest, Kenya. *Biodivers. Conserv.* **15:** 1159–1177.