DO PROTECTED AREAS REDUCE FOREST FRAGMENTATION? A MICROLANDSCAPES APPROACH

July, 2013

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ABSTRACT

Conservation policies influence both the amount of habitat loss and patterns of habitat fragmentation. This paper develops a "microlandscapes" approach that combines fragmentation measures with quasi-experimental evaluation methods in order to assess the effects of policy on habitat fragmentation. As an application, the paper estimates whether and to what extent wildlife sanctuaries and national parks in Thailand prevented forest loss and fragmentation. I find that both types of protected areas significantly increased forest cover, average forest patch size and maximum forest patch size. Comparisons between the two types indicate that wildlife sanctuaries were more effective than national parks in terms of protecting forest in the interior vs. exterior areas of parks and preventing fragmentation conditional on the level of forest cover. The differences are consistent with predicted differences resulting from spatial patterns of enforcement that are uniform or core-focused in the wildlife sanctuaries versus boundary-focused or include agglomeration penalties in the national parks. Given the greater effectiveness of wildlife sanctuaries in preventing fragmentation and the suggestive link to enforcement types, these results reinforce existing theoretical work urging conservation managers to consider how the spatial distribution of enforcement may affect patterns of resource use.

JEL classifications: Q23, Q24, Q56, Q57

Keywords: habitat fragmentation, land use, land conservation, protected areas, policy evaluation

1. Introduction

The fragmentation of large contiguous areas of habitat into smaller, dispersed patches is an important driver of biodiversity losses worldwide (Turner 1996, Bender et al. 1998, Fahrig 2002, Fischer and Lindenmayer 2007). Habitat fragmentation also reduces the productivity of ecosystems because smaller areas of habitat are often less resilient to severe weather or disease shocks (Boose et al. 1994, Rochelle et al. 1999, Allan et al. 2003, Opdam and Wascher 2004). The design of efficient and effective land conservation policies must therefore consider both the total amounts of habitat conserved and the spatial configuration of that habitat (Albers 1996, Bockstael 1996, Albers and Bu 2009, Lewis and Alig 2009, Albers et al. 2010). Previous research considers the costs and benefits of different habitat patterns and simulates the expected impacts of future policies on habitat fragmentation (Lewis and Plantinga 2007, Horan et al. 2008, Lewis et al. 2009, Ando and Shah 2010, Lewis 2010, Lewis et al. 2011), but does not rigorously evaluate the *ex post* impacts of land conservation policies.

The purpose of this paper is to develop and apply an approach for retrospective empirical evaluation of policy impacts on habitat fragmentation. To do so, I divide regional landscapes into smaller units, or "microlandscapes," so that impacts can be assessed by comparing areas affected by the policy to otherwise similar areas. This approach is novel in its combination of methods for measuring habitat fragmentation (Heilman et al. 2002, Riitters et al. 2002, Stanfield et al. 2002, Butler et al. 2004, Alig et al. 2005, Hawbacker et al. 2005, 2006) with methods for quasi-experimental policy evaluation. Quasi-experimental methods have previously been used to evaluate the impacts of conservation policies on loss of habitat or species (Ferraro et al. 2007, Costello et al. 2008, Andam et al. 2008, 2010, Greenstone and Gayer 2009, Pfaff et al. 2009, Sims 2010), but they have not been used to evaluate the impacts of conservation policies on habitat fragmentation.

I apply the microlandscapes approach to assess whether and to what extent protected areas in Thailand prevented forest loss and fragmentation. Protected areas are one of the most common land conservation policies worldwide (Chape et al. 2005) and have the potential to significantly reduce habitat fragmentation by conserving large, contiguous areas of land. Yet the majority of new protected areas are in developing countries (Zimmerer et al. 2004), where there is substantial pressure on natural resources and where legal restrictions are often incompletely enforced. To illustrate how spatial patterns of enforcement influence fragmentation in the context of protected forest areas, I use a simple von Thünen style model of clearing behavior on a sample microlandscape which is under pressure for conversion from forest to agricultural land. This framework illustrates that enforcement which is uniformly distributed in space or is focused on core areas of parks is likely to be more effective than enforcement which is stronger near park boundaries or includes agglomeration penalties, in terms of protecting forests in the interior areas of parks and preventing forest fragmentation conditional on the level of forest cover.

I then evaluate the impacts of strictly protected areas—wildlife sanctuaries and national parks—in North and Northeast Thailand. Wildlife sanctuaries are the most restrictive, aimed at preserving biodiversity and legally permitting only research or small-scale ecotourism activities. National parks are less restrictive, allowing for recreational use and tourism (Dixon and Sherman 1990, ICEM 2003, Chettamart 2003, FAO 2009). Both types of protected areas officially prohibit resource extraction and agricultural use, but in reality have been incompletely enforced and contain significant amounts of agricultural land (Vandergeest 1996, Roth 2004a,b, ICEM 2003, Chettamart 2003, FAO 2009).[1] Direct measures of enforcement are not available, but anecdotal evidence suggests that wildlife sanctuaries received more consistent funding and a clearer mandate to protect core habitat areas while national parks had fewer resources available for enforcement and were more lenient about subsistence use (Dixon and Sherman 1990, Albers and Grinspoon 1997, ICEM 2003, Chettamart 2003, Vandergeest 2003). This history raises the possibility of different enforcement levels and patterns between the two types of protected areas. The paper therefore examines both the overall effectiveness of wildlife sanctuaries and national parks in preventing forest loss and fragmentation as well as their differential effects on land-use patterns.

To estimate impacts, I use new panel data on protected area designations and forest cover between 1973 and 2000. I first divide regional landscapes into 5 km x 5 km microlandscapes and

[1] A 2003 report by the International Center for Environmental Management (ICEM) estimated that more than 500,000 people were living inside the strictly protected areas in Thailand. Legally, wildlife sanctuaries are similar to IUCN Category I areas while national parks are similar to IUCN Category II areas. The laws governing the two major types of protected areas and establishing the process for their designation were passed in the 1960's and remained in force without major changes throughout the period of this study.

use covariate matching to select control areas which were similar with respect to the characteristics that determined protected area locations and that might also influence fragmentation outcomes. I then estimate panel regressions including microlandscape fixed-effects, thereby identifying impacts using plausibly exogenous variation in the timing of protected area designations.

I evaluate the following outcomes for each microlandscape: the total amount of forest cover, average forest patch size, maximum forest patch size, cleared patch size, forest patch perimeter to area ratio, and the density of cleared patches. I select this set of fragmentation measures because they are believed to be ecologically relevant (Betts 2000, Leitao and Ahern 2002) and share a common framework based on habitat patches (Urban et al. 1987). Although there is debate about exactly which patterns of habitat maximize biodiversity or ecosystem productivity (Christensen 1997, Galvin et al. 2008), I define landscapes in this context as less fragmented if they have more forest cover and larger average and maximum forest patch sizes. In addition, I define them as less fragmented if they have larger cleared patch sizes, smaller forest patch perimeter to area ratios and a lower density of cleared patches, conditional on the level of forest cover.[2]

I find that both wildlife sanctuaries and national parks in Thailand have prevented significant forest loss and fragmentation, increasing forest cover by an estimated 19%, average forest patch size by 25%, and maximum forest patch size by 21%, compared to the counterfactual of no protection. However, comparisons between the two types indicate that wildlife sanctuaries were more effective at preventing forest fragmentation. Wildlife sanctuaries appear to have retained greater forest cover and forest patch size in the interior areas than national parks. They also resulted in marginally significantly larger forest patch sizes,

[2] The literature on habitat fragmentation contains numerous possible metrics for assessing outcomes, due to the many possible configurations of habitat (O'Neill et al. 1988, Turner 1990, Gustafson 1998, McGarigal and Cushman 2002). The chosen metrics measure both size and shape of habitat patches (Betts 2000, McGarigal and Cushman 2002). Larger forest patch sizes are thought to be important for species which cannot easily cross deforested areas to forage or reproduce. Habitat patch shape is important for species which require a safe distance from the edge of a patch and thus prefer patches with more core habitat. Smaller forest patch perimeter to area ratios indicate more core habitat relative to edge habitat (picture the difference between a circle and an amoeba shape). Larger cleared patch sizes and a lower density of cleared patches also indicate more core forest habitat, i.e. forested landscapes which are less broken-up by many small, dispersed areas of clearing (picture a solid block of forest versus one with lots of holes). Forest patch size tends to increase with forest cover, but forest patch perimeter to area ratio and the density of cleared patches do not necessarily vary monotonically with forest cover (see Section 3.1 for an example), so are compared here conditional on forest cover.

significantly larger cleared patch sizes, significantly smaller forest patch perimeter to area ratios and significantly lower densities of cleared patches, conditional on the level of forest cover. Such differences are consistent with predicted differences between spatial enforcement patterns that are uniform or core-focused in the wildlife sanctuaries versus boundary-focused or include agglomeration penalties in the national parks. Given the greater effectiveness of wildlife sanctuaries in reducing forest fragmentation and the suggestive link to different enforcement types, these results reinforce existing theoretical work urging protected area managers to consider how the spatial distribution of enforcement may affect patterns of resource use (Albers and Grinspoooon 1997, Robinson et al. 2002, 2005, 2008, Albers and Muller 2004, Albers 2010, Robinson and Lokina 2011).

The paper proceeds as follows. Section 2 briefly reviews previous literature and outlines the contributions of this paper in relation to that research. Section 3 describes the conceptual framework of clearing at the parcel level and applies this to a sample microlandscape to illustrate how different patterns of enforcement affect forest fragmentation. Section 4 describes the empirical strategy, including the data, the matching approach and the panel regression specification. Section 5 presents and discusses results and Section 6 concludes.

2. PREVIOUS LITERATURE

Conservation policies influence both the amount of habitat loss and patterns of habitat fragmentation. One strand of relevant previous literature explores the determinants of fragmentation and develops methods to predict and value policy outcomes. Another considers the effects of protected areas, including both theoretical and empirical assessments of their effectiveness. The microlandscapes approach brings these strands together by combining fragmentation metrics with policy evaluation techniques to analyze protected area impacts.

2.1 Empirical analysis of policies affecting fragmentation

Previous studies of the determinants of fragmentation suggest that land-owner type, zoning regulations and road building are important potential policy levers. In the United States, Butler et al. (2004) test whether forest fragmentation in western Oregon and Washington is linked to population density, income, agricultural profitability, distance to roads, slope, and federal land ownership. Alig et al. (2005) extend this approach by testing whether the spatial configuration of returns within each landscape (as measured by the pattern of low-quality land) is also a driver of fragmentation. Both find that more federal land ownership reduced fragmentation while proximity to highways increased it.[3] Turner et al. (1996) similarly shows a correlation between land ownership categories and land-cover change patterns in North Carolina and Washington. Albers, Ando and Chen (2008) find that federal ownership of conservation lands created crowding-in and crowding-out effects of private ownership (depending on the state), thereby also indirectly influencing fragmentation. The potential importance of zoning regulations for fragmentation patterns is suggested by the finding (Alig et al. 2005 and Butler et al. 2004) that fragmentation is positively related to population density and proximity to urban areas. Hawbacker et al. (2005, 2006) also find significant correlations between habitat patch size and housing density as well as biophysical characteristics and land use for Northern Wisconsin townships.

Several prior studies suggest the importance of roads as a driver of fragmentation. Heilman et al. (2002) is one of the first to develop fragmentation statistics for the U.S. based on road patterns which increase fragmentation over time. Albers et al. (2012) proposes a new "Road Network Agglomeration Index," which is a strong indicator of fragmentation in California's reserve network. Analyses of deforestation patterns in developing countries (Chomitz and Gray 1996, Pfaff 1999, Pfaff et al. 2007, 2011, Robalino and Pfaff 2012) also highlight the importance of roads. Although these papers do not measure forest fragmentation directly, they show that road density is a key driver of deforestation. Roads influence deforestation both directly, through decreased transportation costs and increased returns to deforestation (Chomitz and Gray 1996), and indirectly, through spatial or inter-temporal spillovers (Pfaff et al. 2007, Robalino and Pfaff 2012).

In addition to exploring the broad determinants of fragmentation, previous studies develop methods to predict how specific policies may affect fragmentation. A group of new studies (Lewis and Plantinga 2007, Lewis, Plantinga and Wu 2009, Lewis 2010, Lewis et al. 2011) forecasts ecosystem change using past econometric relationships between land uses and returns plus simulations of how policies change those returns. Lewis and Plantinga (2007) develop this method to analyze how afforestation subsidies with and without agglomeration

[3] An earlier study of Oregon watersheds also finds that land ownership structure is related to size of forest patches (Stanfield et al. 2002).

bonuses affect fragmentation in South Carolina's coastal plain. Lewis, Plantinga and Wu (2009) extend the method and analyze a situation in which afforestation subsidies can also vary between areas with different initial landscape conditions. Lewis (2010) predicts the probability of a local species going extinct depending on different housing densities on Northern Wisconsin lakes. Finally, Lewis et al. (2011) uses a similar methodology to assess the efficiency of different types of incentive-based policies depending on the degree of returns to scale in large habitat parcels.

These studies provide important tools for making forward-looking assessments of policies which are likely to potentially affect habitat fragmentation. However, as noted by Lewis and Alig (2009) in their review of methods for modeling landscape change, there is also a need for work which uses quasi-experimental methods to retrospectively evaluate how specific policies have influenced fragmentation. Actual responses to policy may differ from predictions because people and economic systems respond in unforeseen ways. The microlandscapes approach described here attempts to fill this methodological gap in the literature by combining fragmentation metrics with empirical evaluation methods.

2.2 The impacts of protected areas on forest fragmentation

This paper also seeks to fill a gap in our specific knowledge of the effects of protected areas on fragmentation patterns. Previous related literature includes theoretical modeling of extraction and enforcement patterns, empirical analyses of avoided deforestation benefits, and explorations of the association between protection and fragmentation.

Several theoretical models in the literature illustrate the idea that resource users and protected area managers are involved in a complex spatial and temporal game (Albers and Grinspoooon 1997, Robinson et al. 2002, 2005, 2008, Albers and Muller 2004, Albers 2010, Robinson and Lokina 2011). In general, these models analyze situations where resource users illegally enter protected forests to extract firewood or non-timber forest products. Robinson et al. (2002) show how changes in market access, minimum consumption needs, opportunity costs of labor and extraction costs affect the spatial patterns of resource extraction. They illustrate that policies which increase economic returns in extraction zones near protected areas may be more effective than increased enforcement because extraction decisions are very sensitive to labor costs and market access. Robinson et al. (2008) and Robinson and Lokina (2011) add a temporal dimension to the resource extraction decision and incorporate the possibility that communities may cooperate to manage resources. A key conclusion with respect to fragmentation is that snapshots of resource patterns may not accurately capture dynamic community decisions about regeneration or changing responses over time to protected areas. Albers and Grinspoon (1997) and Albers and Muller (2004) consider the success of different types of management regimes in protecting forests. Albers and Muller (2004) show that the effectiveness of enforcement strategies including direct punishment, conservation payments or integrated development and conservation projects depends on the context in which they are applied. Finally, Robinson et al. (2011), Robinson and Lokina (2011) and Albers (2010) bring together resource users and managers in a spatial game-theoretic model. Albers (2010) considers best responses by each side given constraints about different spatial patterns of enforcement including homogeneous patrols, boundary patrols, interior ring patrols and enforcement zones. She illustrates that managers can more efficiently protect core areas, thereby lessening fragmentation of the interior, if they are able to spatially target enforcement and allow extraction zones at the edge of protected areas. Unfortunately, resource managers in many countries (including Thailand) are legally prohibited from allowing extraction zones or explicit spatial targeting, possibly locking-in inefficient management regimes.

Within the empirical literature on protected areas, most research has focused on whether protected areas are effective in limiting the total amount of forest loss. These studies include evaluations of deforestation in Thailand (Cropper et al. 1999, 2001, Sims 2010), Costa Rica (Andam et al. 2008, Pfaff et al. 2009, Ferraro et al. 2011), Mexico (Deininger and Minten 2002), Belize (Chomitz and Gray 1996) and Brazil (Pfaff 1999). Together, they do indicate that protected areas have prevented significant amounts of deforestation, but less than would be implied by naïve inside-outside comparisons. In addition, these and other studies (Joppa and Pfaff 2009) document that protected areas were usually designated in more remote and rugged locations where the threat of deforestation was lower. Failing to account for this selection issue leads to overestimates of park effectiveness in maintaining forest cover.

To date, studies which have measured fragmentation inside and outside of protected areas do not account for this location bias. Papers by Sanchez et al. (2003, 2001, 1999) develop remote sensing methods for measuring tropical deforestation over time. They include calculations for forest fragmentation inside and outside of protected areas in Costa Rica, but do not attempt to derive causal impact estimates of protected area policy. Liu et al. (2001) do imply that changes in habitat fragmentation inside and outside of a panda reserve in China can be attributed to changes in protection status, but fail to account for other underlying differences. Several additional studies also measure forest fragmentation over time and draw possible links to community or state management regimes on the ground, but do not specifically examine protected areas. Examples of developing country analyses include Ochoa-Gaona (2001), Fox et al. (1995), Fox and Vogler (2005), and Thomas et al. (2004). In summary, previous empirical studies on the deforestation impacts of protected areas either do not measure fragmentation or compare fragmentation inside and outside of protected areas without accounting for other potentially confounding differences in land characteristics. This paper seeks to fill this gap by matching microlandscapes with similar characteristics and comparing changes over time using panel data analysis in order to rigorously analyze protected area impacts on fragmentation.

3. CONCEPTUAL FRAMEWORK

3.1 Clearing and fragmentation patterns on a sample landscape

Protected forest areas have the potential to reduce fragmentation by setting aside large contiguous areas of forest under public protection. Yet where those forests are under pressure for conversion to other uses and enforcement is not complete, deforestation and fragmentation will occur despite official restrictions on resource use. To illustrate the potential impacts of different incomplete enforcement patterns on forest loss and fragmentation, I construct a simple economic framework of clearing behavior on a sample two-dimensional microlandscape (illustrated in Figure 1). The microlandscape consists of multiple individual parcels; in this case 30 x 30 parcels. I follow the standard von Thünen style assumption that land-use choices on each parcel are driven by the relative returns to different land uses (e.g. Stavins and Jaffe 1990, Chomitz and Gray 1996, Pfaff 1999, Angelsen 2007, 2010). For simplicity, I assume that a parcel is cleared if the rents to agricultural use are weakly positive and otherwise remains forested. Agricultural rents are a function of the price of the agricultural product, p^a, times the yield to agriculture on each plot as a function of land quality, $f(q_i)$, minus labor costs (wage, w, times the quantity of labor, l), transportation costs as a function of distance to market $t(d_i)$ and clearing costs (c):

(1) $$R_i^a = p^a f(q_i) - wl - t(d_i) - c$$

Parcels within the microlandscape vary with respect to underlying land quality (q_i) and distance to market (d_i) as shown in panels a and b of Figure 1.[4] Wages are assumed to be exogenous and the amount of labor needed for each parcel is fixed.[5]

Figure 1c and d show examples of clearing patterns produced by this framework (specific parameter values are given below Figure 1).[6] The baseline scenario (Figure 1c) has a relatively low agricultural price and thus total amount cleared (10%). The second scenario (Figure 1d) has a higher agricultural price, modeling a rise in the value of the agricultural good over time which drives increased clearing (total amount cleared = 45%). A comparison of the patterns in Figures 1c and 1d illustrates changes over time consistent with the key stylized facts of tropical deforestation (e.g. Angelsen 2010) and similar to those observed in the actual Thai land-cover data. In general, deforestation increases where rents are higher, i.e. on higher quality land or land closer to markets. Forest fragmentation accompanies deforestation and is driven by the underlying patchy or branching pattern of land quality. Considering the specific set of fragmentation metrics which will be used in the empirical section, we note that as total clearing increases, forest patch size will generally decrease as the forest is divided into more, smaller patches. However, forest patch perimeter to area ratios and cleared patch densities may either increase or decrease with more clearing, depending on the underlying patchiness of land quality and the role of transportation costs.[7]

[4] For simplicity, land quality is assumed to be a function of distance to the nearest water source, but additional sources of heterogeneity in land quality such as soil type, slope, aspect, distance to local villages or paths, etc. could also be important. Adding more dimensions to the land quality function would change the specific patterns on the microlandscape but not the overall conclusion that fragmentation will be driven by patchy or branching distributions of land quality.

[5] This corresponds to an assumption that the microlandscape is small compared to the overall labor market and workers are freely mobile. Since this mobility assumption may not hold, the implications of relaxing it are also discussed in Section 3.3.

[6] Parameter values are illustrative and are not based on calibrations. The predictions described below are based on tendencies from experimenting with a variety of different parameter values. Excel model available on request.

[7] As an example, consider additional scenarios on the same landscape pictured in Figure 1. At p^a=.7, there would be is 1% cleared and 1 cleared patch; at p^a=.79, 5% cleared and 3 cleared patches; at p^a=.85, 10% cleared and 4 cleared patches; at p^a=1.01, 30% cleared and 5 cleared patches; at p^a=1.15, 50% cleared, 2 cleared patches and at p^a=1.36, 70% cleared and 1 cleared patch. This non-monotonic relationship happens because clearing initially spreads to more small dispersed areas and the density of cleared patches increases, but at high levels of deforestation the cleared patches merge together and cleared patch density decreases.

3.2 Protected areas and possible enforcement patterns

I now consider how clearing and fragmentation outcomes will vary as a result of protection with different spatial patterns of enforcement (Figure 2). Although protected area managers may be legally expected to fully enforce regulations, in reality enforcement is likely to be both incomplete and non-uniformly distributed across space (e.g. Albers 2010). I consider three possible patterns of enforcement: spatially uniform, boundary-focused and core-focused. In addition, I consider what happens if an agglomeration penalty is added to these patterns.

Each protection scenario (Figure 2) uses the same framework and sample microlandscape as above. The microlandscape is now assumed to be set aside as part of a protected area after the baseline clearing scenario (Figure 1c) but before the rise in agricultural prices. The entire microlandscape is within the protected area, with the park boundary at the bottom of the microlandscape (dotted black line). Parcels close to the top of the microlandscape are thus more in the interior of the park while those at the bottom are nearer to the boundary. Enforcement is assumed to be incomplete, so new clearing will accompany the rise in agricultural prices. In order to consistently compare fragmentation patterns across scenarios, the amount of enforcement is assumed to be such that it results in the same total amount of deforestation across the cases.

Uniform enforcement

Spatially uniform enforcement is the baseline enforcement type. To model this case, I assume that parcel rents depend additionally on the chance of being caught and penalized for clearing, where π_u is the probability of detection and G_u is the fine that must be paid if caught:

(2) $E(R_i^a) = p^a f(q_i) - wl - t(d_i) - c - \pi_u G_u$

Since enforcement is uniformly applied, the probability of detection (π_u) is the same on every plot and thus rents are reduced evenly across space. Larger penalties (G_u) will result in less clearing. Figure 2a shows an example where spatially uniform enforcement results in 20% clearing.

Boundary-focused enforcement

A second possible enforcement pattern is boundary-focused enforcement. More frequent patrols near boundaries are likely where resources are limited and transportation is costly. Fujita (2003) reports clear resource limitations on patrols in Thailand: in 1989 there were about 240 forest protection units with 5-6 people per unit who were responsible for enforcing Royal Forestry Department policy for all of Thailand (some 517 thousand sq km). Road networks are likely to be better near the boundaries of parks and a reasonable manager might also expect more illegal activities near the boundaries. To model boundary-focused enforcement I assume that the probability of being caught and fined (π_b) varies with the distance to the boundary (b_i) of the protected area:

(3) $E(R_i^a) = p^a f(q_i) - wl - t(d_i) - c - \pi_b(b_i)G_b$

where π_b is decreasing as distance to the boundary increases. An example, again with 20% clearing, is shown in Figure 2b.

Core-focused enforcement

A third enforcement scenario (Figure 2c) is core-focused enforcement. This situation might occur if the regulator is intentionally trying to protect habitat at the center of protected areas and so levies greater punishment in the interior. In Thailand, the wildlife sanctuaries were created explicitly for habitat conservation and preservation of game (Dixon and Sherman 1990, Chettamart 2003); managers may therefore have focused on keeping core areas intact in those areas. In addition, a key goal of Thai protected areas was to safeguard national water resources and prevent downstream flooding (Ruhle 1964, Hirsch and Lohmann 1989, Vandergeest 2003, ICEM 2003). Enforcement resources may also have been differentially focused on sensitive upper watershed areas which are more likely to be in the interiors of parks. I model core-focused enforcement with the same Equation (3) above, except that the probability of enforcement (denoted π_c) is now increasing as distance to the boundary increases, and the penalty is denoted G_c.

Agglomeration penalties

Finally, an important additional aspect of enforcement might be agglomeration penalties, i.e. larger penalties for larger cleared areas. Such targeting might occur in situations where enforcement is limited by the political feasibility of taking away subsistence livelihoods. This is likely in Thailand, which has a long history of tension between protected area regulations which prohibit all use and traditional land laws which encouraged "productive" land use (Fujita 2003, Giné 2005). Fujita (2003) notes that Royal Forestry Department officials were reluctant to expropriate or destroy crops that had already been planted, even within protected forests. Albers and Grinspoon (1997) note that managers in Khao Yai National Park said they "rarely enforce regulations against collecting fruit, vegetables, grasses, and leaves in the park when the amounts collected imply that the goods are for family use rather than for sale."[8] Although not officially sanctioned, it is likely that protected area managers, particularly in the national parks, might preferentially overlook small cleared patches which are perceived as subsistence use while enforcing prohibitions on larger patches of agricultural clearing.

An agglomeration penalty might be combined with any of the other spatial patterns of enforcement above. I model it in a simple fashion as an additional penalty for patches which are next to those already cleared at baseline before the protected area was established (e.g. Figure 1c above). For the case of a uniform penalty plus the agglomeration penalty, the parcel rents are:

(4) $E(R_i^a) = p^a f(q_i) - wl - t(d_i) - c - \pi_u G_u - I * G_{ap}$

where I is an indicator function equal to one if a neighboring parcel was cleared at baseline and G_{ap} is the additional agglomeration penalty. This example is shown in Figure 2d.

3.3 Expected clearing and fragmentation as a function of enforcement type

Figure 2 shows representative patterns for each of the enforcement cases, following the equations given above (parameter values given at the bottom of Figure 2). While different parameter values produce different specific patterns, the equations above and the sample microlandscape can be used to gain a series of insights about how fragmentation measures tend to vary under different enforcement regimes. Table 1 summarizes these insights, first by comparing how different enforcement types will affect forest cover and forest patch size overall and in the interiors of parks (columns 1 and 2) and then by comparing how different types will affect fragmentation metrics conditional on the same level of forest cover (columns 3-6).

Compared to no protection, any enforcement will lower the rents on at least some parcels on the microlandscape, leading to less clearing and more forest cover (e.g. compare Figure 1d and Figure 2). In Table 1, this is noted in column 1 as (++) for all enforcement types. However, different enforcement patterns will differentially affect forest cover in the interior or core areas of the park versus at the periphery (Table 1, column 2). Spatially uniform or core-focused

[8] Apparently villagers responded by collecting less per trip to remain below the limits!

enforcement will most strongly protect core areas (++ and +++) while boundary-focused enforcement will be less effective at protecting forest cover in the interior (+). This is because higher transportation costs mean lower rents for interior parcels. Under uniform or core-focused enforcement, these interior parcels are the first set to become unprofitable and thus more likely to stay in forest. Under boundary-focused enforcement, the higher transportation costs for interior parcels are partially offset by lower expected penalties, so there is relatively more interior clearing.

The effects of an agglomeration penalty are considered as an addition to the other types of enforcement (Table 1). Compared to no additional enforcement, an agglomeration penalty will add to forest protection overall (++). This penalty will also increase forest protection in the interiors of parks compared to no protection, but will be relatively less effective at protecting interior vs. exterior areas of parks (+). This is because large clearings, which attract the most punishment under the agglomeration penalty, are likely to be near the boundaries of protected areas, where transportation costs are low.[9]

Although this framework predicts differences across enforcement types, we still expect all types of enforcement to result in more forest cover than no protection, even in interior areas of parks, because enforcement lowers rents everywhere on the microlandscape. This prediction may not hold, however, if there is a high degree of labor immobility combined with boundary-focused enforcement or enforcement with an agglomeration penalty. If labor cannot move easily across microlandscapes,[10] effective protection in one part of the landscape will raise the profitability of parcels in other parts through decreased wages. Lower wages combined with low penalties in the interior could be enough to actually decrease interior forest cover compared to the no-protection case (-).

Different patterns of enforcement may also affect other measures of habitat fragmentation. Table 1 (columns 3-6) summarizes how different enforcement types are likely to

[9] Note this assumes that transportation costs matter and are correlated with distance to the boundary of the protected area. If households are not well-integrated with markets (i.e. most agriculture is grown for household use and does not use market inputs), then clearing decisions would only be weakly related to transportation costs and the differences between enforcement types would be small.

[10] There are likely to be at least some factors which limit the mobility of labor. In Thailand these include the system of identification and political registration which is tied to village of birth, poorly defined private property rights, and the advantages of extended family networks in home villages.

affect forest and cleared patch size, forest patch perimeter to area ratios and cleared patch density, conditional on forest cover.[11] The basic conclusion is that compared to a uniform enforcement pattern, we expect to see less fragmentation with core-focused enforcement and more with boundary-focused enforcement or enforcement with an agglomeration penalty (e.g. Figure 2). More specifically, core-focused enforcement (e.g. Figure 2c) tends to consolidate clearing closer to the edge of the microlandscape, preserving larger forest patches. This consolidation also tends to lead to fewer, larger cleared patches and smaller forest patch perimeter to area ratios. In contrast, boundary-focused enforcement (e.g. Figure 2b) tends to push more of the clearing into the interior of the microlandscape due to lower penalties in the interior. This is likely to split the forest into more small areas, reducing forest patch size, and to create a more branching and patchy pattern with many smaller cleared patches and high forest patch perimeter to area ratios.

Adding an agglomeration penalty to any of these enforcement patterns will also tend to increase fragmentation, conditional on forest cover (e.g. Figure 2d). The agglomeration penalty protects forest next to the areas already cleared while new clearing tends to spread across the microlandscape, resulting in smaller patch sizes, larger forest patch perimeter to area ratios and higher densities of cleared patches. Finally, it is important to consider how relaxing the exogenous wage assumption might affect these predictions. Adding labor immobility to any of the enforcement patterns will tend to spread clearing and increase fragmentation. Again, lower wages induced by the restrictions on clearing will reduce labor costs relative to transportation costs, increasing the relative profitability of parcels in the interior of the microlandscape and dispersing clearing.

In summary, this framework illustrates that policymakers who are concerned with patterns of forest clearing in addition to total amounts should carefully analyze the possible implications of different enforcement types. Uniform or core-focused enforcement is more likely to be effective at preventing interior clearing and forest fragmentation than boundary-focused enforcement or enforcement with agglomeration penalties. The next section explores whether

[11] Uniform enforcement generally produces similar patterns of fragmentation conditional on forest cover as a no protection case with lower agricultural prices. If production is linear in land quality, the patterns are the same (both a price decrease and a uniform penalty reduce rents by a constant amount); if production has diminishing returns in land quality then uniform enforcement tends to create somewhat more fragmentation.

Thai protected areas have prevented forest loss and fragmentation and how the observed patterns may reflect possible differences in enforcement types.

4. Data and Empirical Strategy

This section describes the empirical application of the microlandscapes approach. Regional landscapes in North and Northeast Thailand are divided into smaller units of analysis and fragmentation outcomes are assessed as a function of landscape and policy variables. In order to ensure that comparisons are made between units which are as similar as possible, I first pre-match the microlandscapes on the basis of characteristics affecting the location of protected areas that could also influence fragmentation, then estimate impacts using panel regression with microlandscape fixed-effects.

4.1 Data sources

The microlandscapes are defined using a 5 km x 5 km grid (and as a robustness check, a 10 km x 10 km grid). Figure 3 illustrates the scale of these microlandscapes compared to two national parks in Northern Thailand (Doi Inthanon and Ob Luang). Cleared areas and forested areas are shown in 1973 and 2000, giving an example of the increasing levels of deforestation and forest fragmentation seen over time both inside and outside of the protected areas.

For each microlandscape, I calculate the following dependent variables: forest cover, average forest patch size, maximum forest patch size, forest patch perimeter to area ratio, average cleared patch size and the density of cleared patches. These are calculated for 1973, 1985, 1992 and 2000, using land cover data which is summarized in Table 2. Land use for 1992 and 2000 is based on Landsat 5 Thematic Mapper (TM) images and for 1973 and 1985 on Landsat Multi-Spectral Scanner images (MSS). All layers contain information on cleared land, forested land, and water bodies, but only the 2000 layer distinguishes between different types of forest.[12] With respect to the resolution of the data, the pixel scale of the Landsat TM is 30 m x 30 m and of the Landsat MSS 60 m x 60 m. In comparison, village-level data from the Thai community development department NRD2C survey for Chiang Mai province in the 1990's

[12] The satellite-based classifications were compared to an analysis based on aerial photos and fieldwork that was done for a small number of sites in Northern Thailand at a similar time (Thomas et al. 2004). This indicated that the classified layers are fairly accurate at picking up fully cleared areas for growing rice or other annual crops, but that tree crops and early secondary regrowth (fallow) may be classified as forest.

reports approximately 1-2 rai of agricultural land per person on average, where one rai is equivalent to a 40 m x 40 m plot. This means that most clearing is likely to be picked up in this data, especially because individual landholdings tend to be larger in upland areas where most parks are located and because most clearing is done by multiple households working together.[13]

The decision to use a 5 km x 5 km grid to define the microlandscapes is admittedly somewhat arbitrary but was made to balance two important factors. On the one hand we want a relatively fine scale of resolution for the analysis (i.e. several microlandscapes within each park) and on the other we want a landscape unit that is large enough to matter ecologically. Median protected area size in the dataset was approximately 450 sq km with a minimum of approximately 50 sq km, so the 5 km x 5 km grid gives at least one and on average about 20 microlandscapes per park. This choice does limit the maximum patch size to be at most 25 sq km, but this is large enough to be ecologically relevant for key species. For instance, the white handed gibbon, which is an endangered forest canopy dweller in Thailand, has an estimated home range of ~0.16-0.54 sq km and is estimated to need 10 sq km for a genetically viable population.[14] The average home range of the long-tailed Macqaw monkey, another relevant species, is approximately 1.25 sq km.[15] However, the home ranges or areas needed for minimum populations for larger mammals (such as the Asian elephant) would be greater. As a robustness check to ensure that we do not miss important spatial patterns using small cells, the full dataset was re-created using a 10 km x 10 km grid, which increases the maximum forest patch size to 100 sq km.

4.2 Empirical strategy

As discussed in Section 2, land which has been selected for protected areas is usually more remote and rugged. More remote lands may be less likely to undergo fragmentation, because pressures to develop it are less; on the other hand more rugged lands might be more

[13] Schmidt-Vogt (1998), for instance, documents areas equivalent to a square of 800m x 800m being cleared for one village. However, depending on terrain and ethnic group, individual households may also clear land (Schmidt-Vogt 1998) so it is possible that some very small clearings are not visible.

[14] Wild animals rescue foundation of Thailand (www.warthai.org), "Gibbon Rehabilitation Project." Accessed October, 2011; Brockelman, W. & Geissmann, T. 2008. "Hylobates lar: IUCN Red List of Threatened Species. Accessed October 2011.

[15] University of Michigan Museum of Zoology (http://animaldiversity.ummz.umich.edu) "Macaca fascicularis" Accessed October, 2011.

likely to undergo fragmentation because high quality land is very patchily distributed. Regardless of the expected direction, it is clear that in order to accurately assess the impacts of protected areas, we should identify effects based on a comparison to land with similar characteristics. Here, I first pre-process the data using matching to select a control group of microlandscapes which were never protected but have similar geographic characteristics to protected microlandscapes. I then estimate impacts using panel regression with microlandscape fixed effects, identifying effects by comparing changes over time between protected and unprotected landscapes.

Using matching to select control microlandscapes

To select a control group of microlandscapes which were never protected, I match on the basis of variables designed to capture factors which determined protected area locations and might also be correlated with deforestation or forest fragmentation.[16] As discussed in further detail in Sims 2010 (Section 4.2), the particular history of protected areas in Thailand means that their locations were selected largely by central planners on the basis of observable physical and geographic characteristics. In particular, lands were more likely to be protected if they were important for watershed protection, had higher historical forest cover, included less high quality agricultural or timber land, were further from cities, protected scenic areas, and were closer to national borders. I therefore calculate and match on the following characteristics for each microlandscape: average elevation, average slope, maximum elevation, maximum slope, distance to major city (all of which were established before 1973), distance to railroad line, distance to Thai national boundary, distance to large river, distance to nearest major and minor road prior to protection (1962 roads), distance to nearest waterfall, and percent area in each major ecotype.[17] Figure 4 shows the boundaries of the national parks and wildlife sanctuaries by period of implementation and illustrates that there is considerable spatial and temporal variation in protection.

The sources of data for the measures of protection and for each of the covariates are listed in Table 3. For the purposes of matching, protection must be a binary variable and is defined as

[16] Matching methods help to select a control group for which there are high levels of overlap between covariates (Ho et al. 2007). This can reduce bias due to functional form choices in standard regression models in cases where treatment and overall control groups are quite dissimilar (Imbens 2007, Rosenbaum and Rubin 1985).

[17] I use the logs of each of the slope, elevation and distance variables.

having any land in the 5 km x 5 km grid in a protected area by the year 2000. For the regression analysis, protection is defined continuously, as the share of each grid cell which is protected by 1973, 1985, 1992 or 2000. Table 4 gives summary statistics for the matched sample. The top half of Table 4 indicates that forest cover and average forest patch size have declined substantially across time, with the largest decrease in the earliest period (1973-1985). Median and maximum forest patch size also decline across time, while the average size of cleared patches increases. The average density of cleared patches in each microlandscape increases until 1992 and then decreases somewhat in the last time period, suggesting some consolidation of cleared areas in the 1990's.

I include in the sample available for matching all complete grid cells with more than 50% (majority) of forest cover in 1973,[18] with less than 10% cloud cover in any image, and with less than 20% water in 1973. Matching is conducted separately for the North and Northeast regions, using one-to-one Mahalanobis matching with replacement. The summary statistics of the covariates for landscapes with and without protection (bottom half of Table 4) illustrate that matching does achieve substantial similarity across these covariates. After matching, only one of the normalized differences (distance to any road in 1962) is greater than .25 standard deviations, which is the rule of thumb for reasonable covariate balance suggested by Imbens and Wooldridge (2009).

Regression model: protected area impacts

Despite the matching, there may still be remaining differences between treated and control units in observables as well as potentially important unobservables such as the patchiness of land quality within a microlandscape.[19] I therefore take advantage of the panel data in order to control directly for time-invariant unobservable characteristics by including microlandscape level fixed-effects. I estimate the following model:

$$\ln Y_{ijt} = \beta_1 WLS_{ijt} + \beta_2 NP_{ijt} + \alpha_i + \alpha_{jt} + \varepsilon_{ijt} \quad (5)$$

[18] This focuses the analysis on landscapes which have significant forest assets to start and thus would be the subject of concern about forest fragmentation. However, the results are robust to including all grid cells with more than 25% or 75% forest cover in 1973 (results available from author).

[19] For instance, Butler et al. (2004) finds that model fit improves when the spatial configuration of returns within each landscape is included.

where Y_{ijt} is the fragmentation outcome for microlandscape i in province j at time t. WLS_{ijt} and NP_{ijt} are the share of microlandscape i in province j which was protected at time t in wildlife sanctuaries and national parks. The microlandscape fixed effect is denoted α_i. I also include a set of province-year fixed effects (denoted as α_{jt}) to control for regional trends.[20] All specifications are run with robust standard errors, clustered at the level of the district (*amphoe*) to account for possible spatial and serial autocorrelation.[21]

5. RESULTS AND DISCUSSION

Following the predictions from the economic framework outlined in Section 3, I first discuss estimated impacts of protection on forest cover and forest patch size, overall and in the interiors of protected areas. I then discuss impacts on fragmentation metrics, conditional on forest cover.

5.1 Effects of protection on forest cover and forest patch size

Table 5 shows results from the fixed effects regression model (Equation 5) with forest cover and forest patch size as outcomes. Columns 1, 3, and 5 give the average impact for protected areas and columns 2, 4, and 6 allow effects to vary across wildlife sanctuaries and national parks. The results show, first, that protection did result in significant increases in forest cover. To interpret the magnitudes, recall that the regressions use the logs of the dependent variables and that the key policy variable is the share of each landscape protected. Therefore the coefficients indicate the approximate percentage change in the dependent variables corresponding to a change in the share of protection from 0 to 1 (i.e. a change from none of the microlandscape protected to all of it protected). Forest cover is thus estimated to be ~19% higher overall due to protection, or ~22% higher due to protection as a wildlife sanctuary and ~ 17% higher due to protection as a national park (columns 1 and 2). This suggests a possibly higher level of enforcement in the wildlife sanctuaries, but the difference is not significant (p-values for F-tests of equality of coefficients are in the last row).

[20] I check the robustness of the results to using lagged protection variables as well. The increases in forest cover and forest patch size are robust to using protection from the previous period and to including both current and lagged protection variables (results available from author).

[21] As an alternate robustness check to deal with possible spatial autocorrelation, I also take a random 20% sample of the dataset. The significant decreases in forest fragmentation are robust to this check (results available from author).

Second, Table 5 indicates that protected areas increased average and maximum forest patch size, by approximately 25% and 21% respectively. Wildlife sanctuary designation resulted in a more than 30% increase in average forest patch size, while national park designation increased average forest patch size by approximately 20%, although the difference between the two is again not statistically significant. Given that the mean of average forest patch size in the matched controls in 2000 was 10.3 sq km, a 20% increase represents an addition of approximately 2 sq km of forest to average patch size, while a 30% increase represents an addition of 3 sq km of forest. Maximum patch size increased by approximately 24% for the wildlife sanctuaries and approximately 19% for the national parks. As maximum patch size was on average 16.2 sq km in the matched controls, a 20% increase corresponds to a more than 3 sq km increase. These changes therefore represent potentially ecologically meaningful differences, given the home range sizes for the two primate species discussed previously. Table 6 shows the same set of regressions for the 10 km x 10 km microlandscapes, which allow for larger patch sizes. The results are similar, although the estimated percentage increases in average forest patch size and maximum forest patch size are higher. There is also a significant difference in the effect on average forest patch size between wildlife sanctuaries and national parks, with a greater impact on forest patch size due to wildlife sanctuary protection (column 4).

5.2 Effects of protection on interior vs. periphery areas

A key prediction of the economic framework described in Section 3 was that boundary-focused enforcement or enforcement with an agglomeration penalty would be less effective at protecting forest cover in the interior areas of parks. The summary statistics (Table 4) indicate potential differences between wildlife sanctuaries and national parks. When we divide protected microlandscapes into those which are fully in the interior of protected areas and those which overlap with protected area boundaries, we see that forest cover in the year 2000 is indeed somewhat higher in the interiors of wildlife sanctuaries (94.7%) versus the interiors of national parks (92.4%) and the difference in means (2.3%) is significant at the 5% level. However, forest cover is also higher at the exteriors of wildlife sanctuaries than national parks (1.8%).

To more rigorously test how protection has affected forest cover and forest patch size in the interiors of protected areas, I re-run the regression specification given in Equation 5 above, but allow the marginal effects of protection to vary flexibly with protected area type and distance to protected area boundaries.[22] Figure 5 graphs these marginal effects. The top panel shows results for forest cover outcomes and the bottom panel for average forest patch size. Although the confidence intervals overlap, different patterns are evident between wildlife sanctuaries and national parks. The wildlife sanctuaries are more effective overall at protecting forest cover and increasing forest patch size, and this effectiveness is maintained even in core areas. National parks, by contrast, show a downward trend in effectiveness for forest cover in interior areas and appear to have actually decreased average forest patch size in interior areas, compared to no protection. Interestingly, national parks also appear to be slightly less effective close to park boundaries, suggestive of either the agglomeration penalties hypothesis or possibly a tacit acceptance by authorities of some encroachment near the boundaries.

In summary, although both national parks and wildlife sanctuaries were effective in increasing forest cover and forest patch size, differential effectiveness in the interior versus periphery areas is consistent with spatial patterns of uniform or core-focused enforcement in the wildlife sanctuaries and boundary-focused enforcement or enforcement with agglomeration penalties in the national parks. Yet the results also suggest a possibly greater level of enforcement in the wildlife sanctuaries. To further explore differences between the two, it is useful to consider impacts on additional fragmentation outcomes while controlling for the total amount of forest cover.

5.3 Effects of protection on fragmentation, conditional on forest cover

Table 7 shows protected area effects on forest patch size, cleared patch size, forest patch perimeter to area ratios, and cleared patch density, conditional on forest cover. The results indicate that protected areas did not significantly change average and maximum forest patch sizes, conditional on forest cover (column 1). This suggests that much of the unconditional increase in forest patch size can be explained by an increase in the total amount of forest cover rather than changes in patterns of clearing. Wildlife sanctuaries did, however, significantly increase average cleared patch size and significantly reduce forest patch perimeter to area ratios, conditional on forest cover (columns 3 and 4). This suggests a possible reduction in fragmentation through consolidation of agricultural patches in the wildlife sanctuaries. In

[22] Regressions are the same as in Equation 5 and Table 5 but include interactions between the share protected as national park or as wildlife sanctuary and distance, distance squared and distance cubed.

contrast, national parks significantly increased the density of cleared patches (column 5) and may have increased forest patch perimeter to area ratios (column 4), suggesting increased fragmentation through dispersion of clearing.

Comparing the two types of protected areas to each other, the results indicate that wildlife sanctuaries resulted in significantly less fragmentation, conditional on forest cover, than national parks (p-values for F-tests of equality of coefficients in the last row of Table 7). Wildlife sanctuaries retained marginally significantly larger average forest patch sizes (p-value = .084), significantly larger cleared patch sizes (p=.005), significantly smaller forest patch perimeter to area ratios (p=.020) and significantly lower density of cleared patches (p=.002) than national parks. As a robustness check, Table 8 shows the same regressions for the 10 km x 10 km grid. Note that the signs of the coefficients are the same in all cases as in Table 7. With the larger microlandscapes, wildlife sanctuaries retained significantly larger forest patch sizes (p=.042), significantly larger average cleared patch sizes (p=.009) and significantly smaller cleared patch density (p=.003) than national parks. Forest patch perimeter to area ratios are also estimated to be smaller in wildlife sanctuaries, although the difference is no longer statistically significant (p=0.137).

Taken together, the estimates of protected area effects on fragmentation conditional on forest cover indicate that wildlife sanctuaries resulted in significantly less fragmentation than national parks. Recalling the predictions from the economic framework (Table 1), these differences in patterns are consistent with spatial patterns of enforcement which are uniform or core-focused in the wildlife sanctuaries and boundary-focused enforcement approach or include agglomeration penalties in the national parks.

One potential concern is that these differences could be driven by differences in location rather than in enforcement patterns. As is shown in Figure 4, wildlife sanctuaries and national parks often share borders and were located in generally similar areas, but some differences remain.[23] Given the inclusion of microlandscape fixed-effects, the observed differences in fragmentation outcomes are not likely to be driven by fixed differences in landscape characteristics, but they could be driven by differential changes in access to transportation across

[23] Using the rule of thumb of .25 standard deviations as a substantial difference, landscapes with wildlife sanctuaries were sited further from 1962 roads and railways and had less ecoregion 3 forest type (tropical and sub-tropical dry broadleaf forest).

time. Two relevant robustness checks were performed: matching was conducting adding distance to late 1990's roads instead of and in addition to 1963 roads as a covariate, and the regressions were repeated including district-year fixed effects rather than province-year fixed effects to better control for possible local changes in access to transportation. The results are robust to both of these changes (available on request).

6. CONCLUSION

Functioning ecosystems provide goods and services that are valued directly or indirectly by humans, such as water supplies, plant products, wild game, and species biodiversity (Daily 1997, Boyd and Banzhaf 2007). Since the productivity of ecosystems depends on their spatial configuration in addition to overall quantities, resource managers need to understand how land-use policies will influence habitat fragmentation. Retrospective studies of past conservation policies can provide useful information to help understand the possible ramifications of current policy decisions. This paper has illustrated a method for retrospective analysis of fragmentation effects using program evaluation methods in combination with common fragmentation statistics. The method was applied to analyze how protected areas in Thailand affected forest fragmentation in the nearly 30 years between 1973 and 2000.

The results indicate that that protected areas in North and Northeast Thailand did significantly increase forest cover and forest patch size compared to the counterfactual of no protection. This is encouraging as it suggests these were not purely "paper parks"; they did make a real difference in preserving contiguous areas of forest habitat. However, differences in forest cover and forest patch size in the core vs. periphery areas of parks and in fragmentation metrics conditional on forest cover suggest different impacts of wildlife sanctuary and national park protection on spatial patterns of forest. Wildlife sanctuaries were more effective at maintaining forest cover and forest patch size in interior areas and conditional on the level of forest cover, resulted in marginally significantly larger forest patch sizes, significantly larger cleared patch sizes, significantly smaller forest patch perimeter to area ratios and significantly lower densities of cleared patches than national parks.

The differences in fragmentation outcomes between wildlife sanctuaries and national parks indicate possible differences in the spatial patterns of enforcement across protected area types. Given the lack of direct measures of enforcement, the empirical results are linked to possible enforcement patterns based on their consistency with predictions from a simple von Thünen style economic framework. The results are consistent with a uniform or core-focused spatial enforcement approach in the wildlife sanctuaries and a boundary-focused approach or enforcement with agglomeration penalties in the national parks. Despite the lack of direct evidence on enforcement, these results are important as they highlight that different types of protection may result in significantly different patterns of land use. Since these differences are potentially driven by different spatial patterns of enforcement, they reinforce existing theoretical work urging conservation managers to consider how enforcement affects both total deforestation and patterns of deforestation when allocating scarce enforcement resources.

The gap in data on enforcement patterns is one that holds globally and is due to the expense and political difficulties of collecting such information. Clearly, this type of data collection is a priority for future work on protected area effectiveness. Future work should also consider how protected area enforcement might be better structured to limit economic costs and maximize environmental benefits. Protected areas in Thailand have historically been managed under an awkward combination of legal prohibition of agricultural use and *de facto* recognition of community livelihoods. This gap between the legal system and day-to-day management and enforcement has caused considerable controversy and conflict (Vandergeest 1996, Albers and Grinspoon 1997, Laungaramsri 2000, ICEM 2003, Chettamart 2003, Emphandhu and Chettamart 2003, Roth 2004a,b, Delcore 2007). If, as the results for national parks suggest, incomplete enforcement which tries to protect boundaries or overlooks subsistence use has led to more clearing of small patches or more interior clearing, this is doubly unfortunate. Such clearing patterns likely indicate lower economic returns in addition to higher fragmentation. These results underscore previous arguments (Albers and Grinspoon 1997, Albers and Muller 2004, Albers 2010) that protected area managers should be given more flexibility in restrictions across space as well as more resources. Finally, future work should also use similar methods to determine whether the results hold for other countries. Thailand is a middle-income country which enjoyed relative stability throughout this period. Many developing countries likely have protected areas with even fewer enforcement resources available and higher potential for perverse fragmentation effects. A better understanding of empirical results across multiple cases could hopefully lead to enforcement patterns which simultaneously improve protection of ecosystems and local livelihoods.

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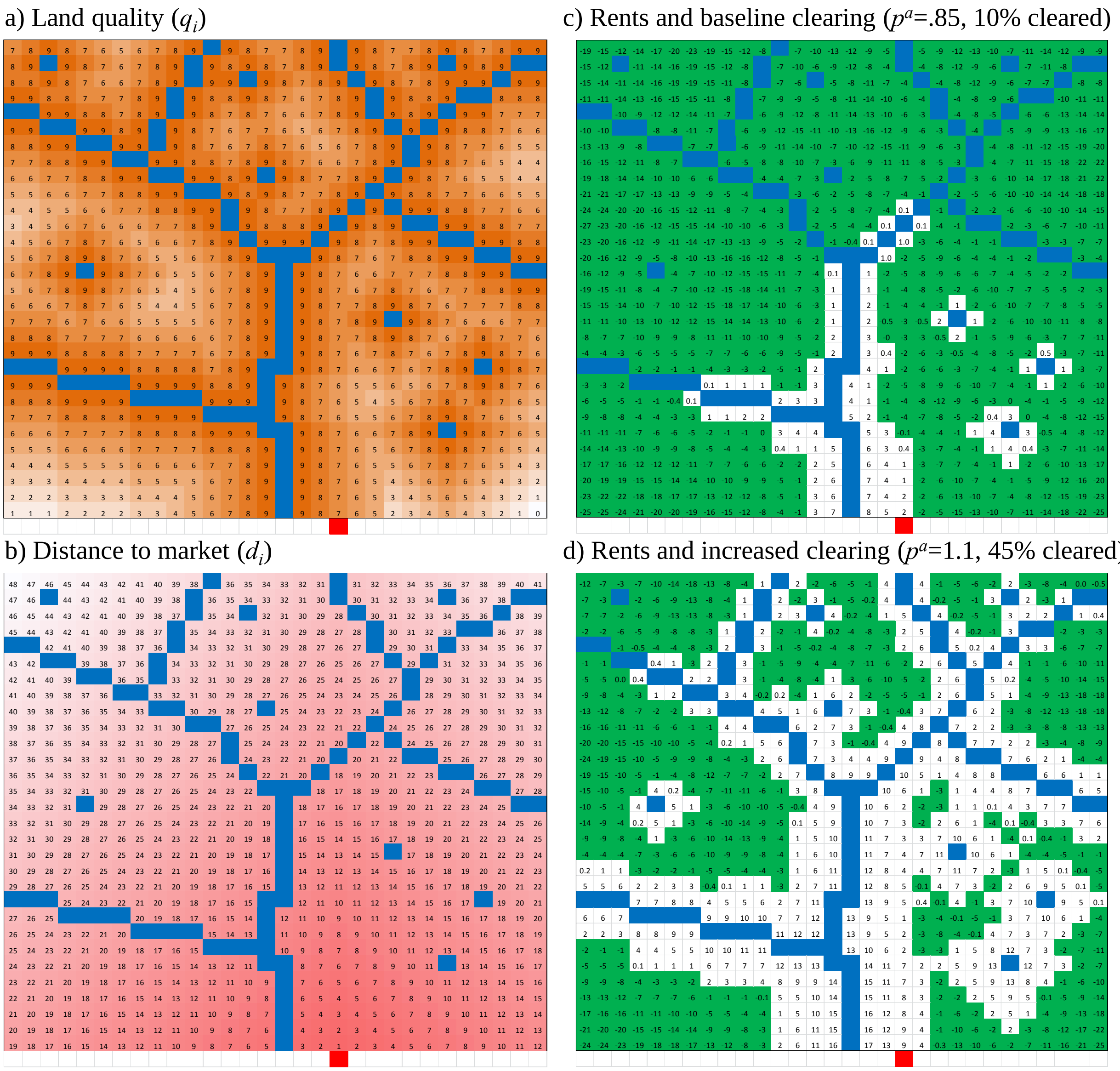
Figure 1: Economic framework for clearing on a sample microlandscape

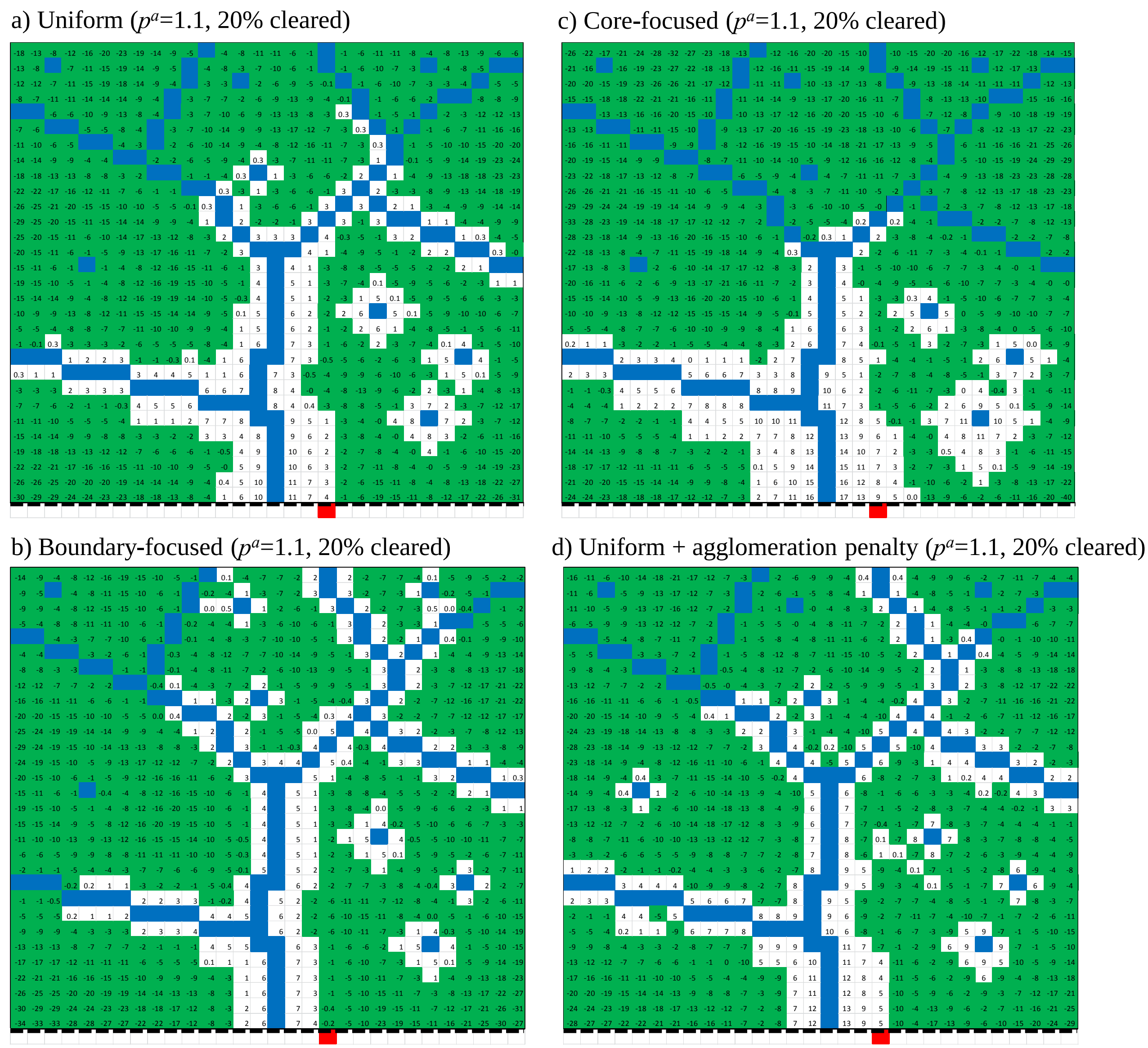
Land quality (a) and distance to market (b) provide underlying variation in rents which drive clearing at baseline (c) and as agricultural prices increase (d).



The sample microlandscape consists of 30 x 30 parcels. Solid blue parcels indicate water, red square at bottom indicates a road leading to the closest market. Panel a) indicates land quality as a function of parcel distance to water source (q_i = 10-dw_i); darker shading corresponds to higher quality. Panel b) indicates distance to market, d_i; darker shading corresponds to closer to market. Panel c) indicates parcel rents as determined by Equation 1 with agricultural price, p^a=.85 and parameter values: $f(q_i)$=3.8(q_i), l=1, w=5, t=.45, c=15. White parcels are cleared (rents >= 0) while green remain in forest (rents < 0). 10% of the landscape is cleared. Panel d) indicates parcel rents and clearing after an increase in agricultural prices (p^a=1.11, 45% cleared).

Figure 2: Clearing and fragmentation under different enforcement patterns

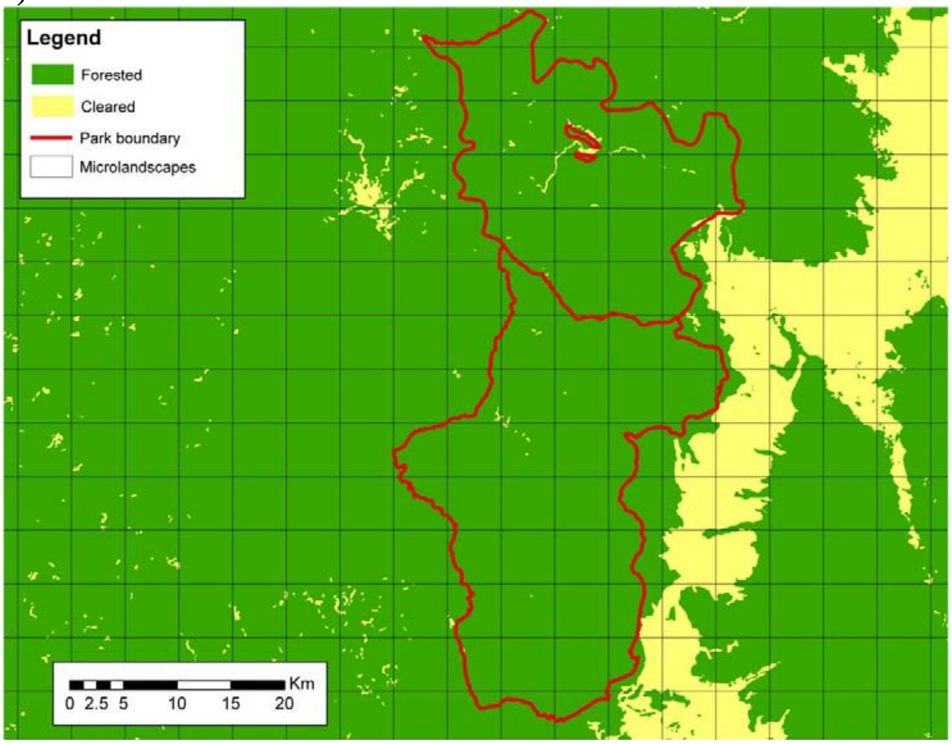
Same sample landscape as Fig. 1 with protected area established before the increase in ag. prices.



Microlandscape characteristics are the same as in Figure 1, but the landscape is now part of a protected area with the boundary at the bottom of the microlandscape (dotted black line). Parcel rents are determined by Equations 2-4. a) Uniform enforcement, with $\pi_u G_u$ = 5.5. b) Boundary-focused enforcement with $\pi_b(b_i)G_b$ = 1.4+0.26(31- b_i) where b_i is parcel distance from the park boundary. c) Core-focused enforcement with $\pi_c(b_i)G_c$ = 14.7-0.5(31- b_i). d) Uniform enforcement + agglomeration penalty with $\pi_u G_u$ = 3.6 and G_{ap}=10.4. Other parameter values are constant across all equations and same as Fig 1d: p^a=1.11, $f(q_i)$=3.8(q_i), l=1, w=5, t=.45, c=15. Sample parameters chosen so that clearing equals 20% across all four cases.

Figure 3: Illustration of 5 km x 5 km microlandscapes and forest fragmentation over time

a) 1973 forest cover



b) 2000 forest cover

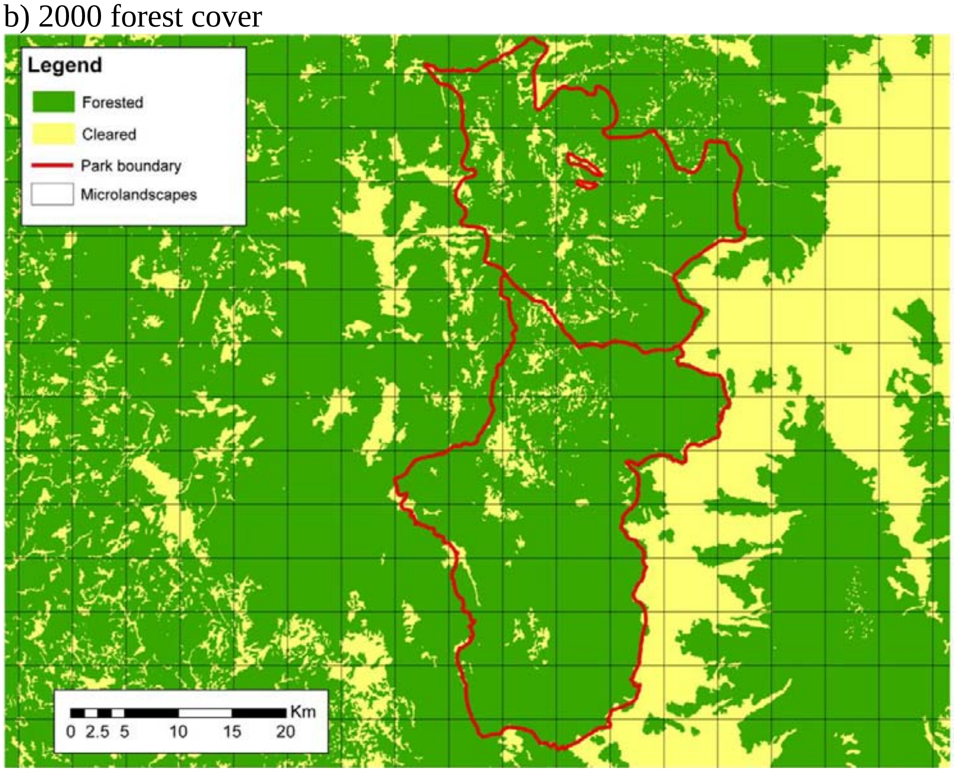
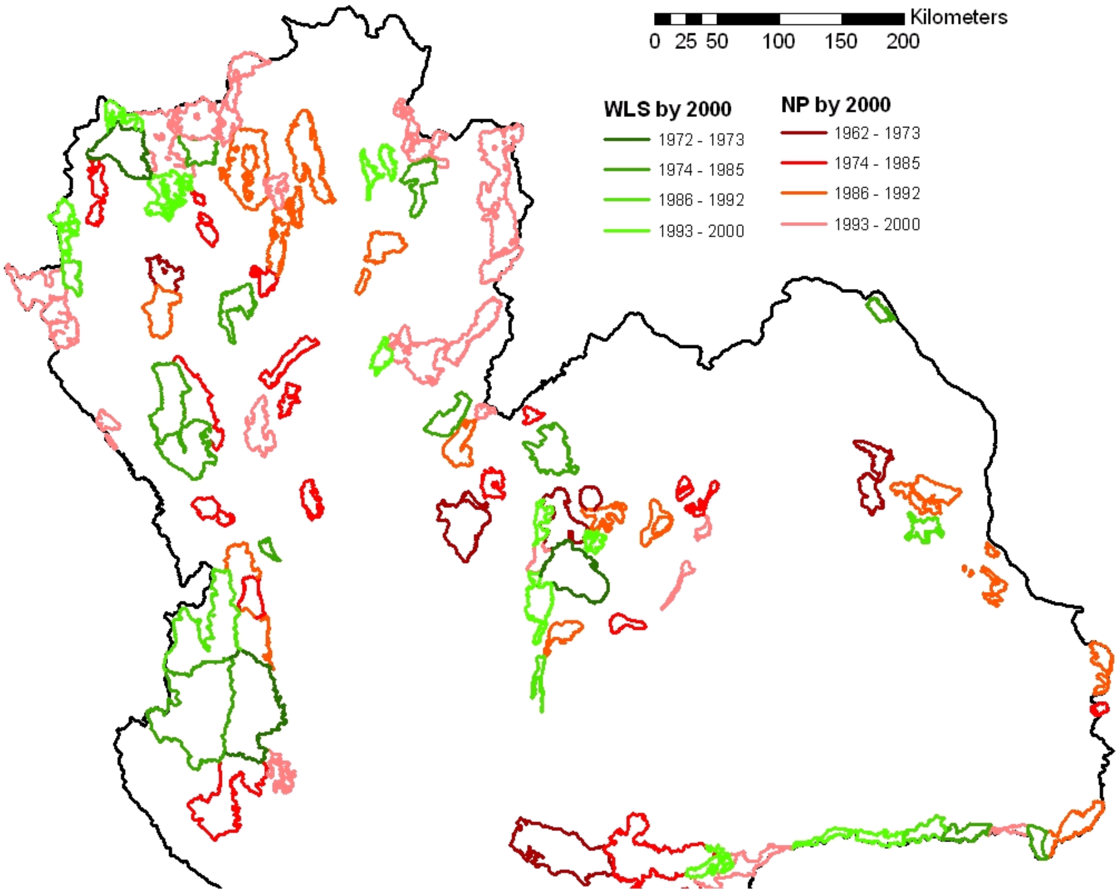
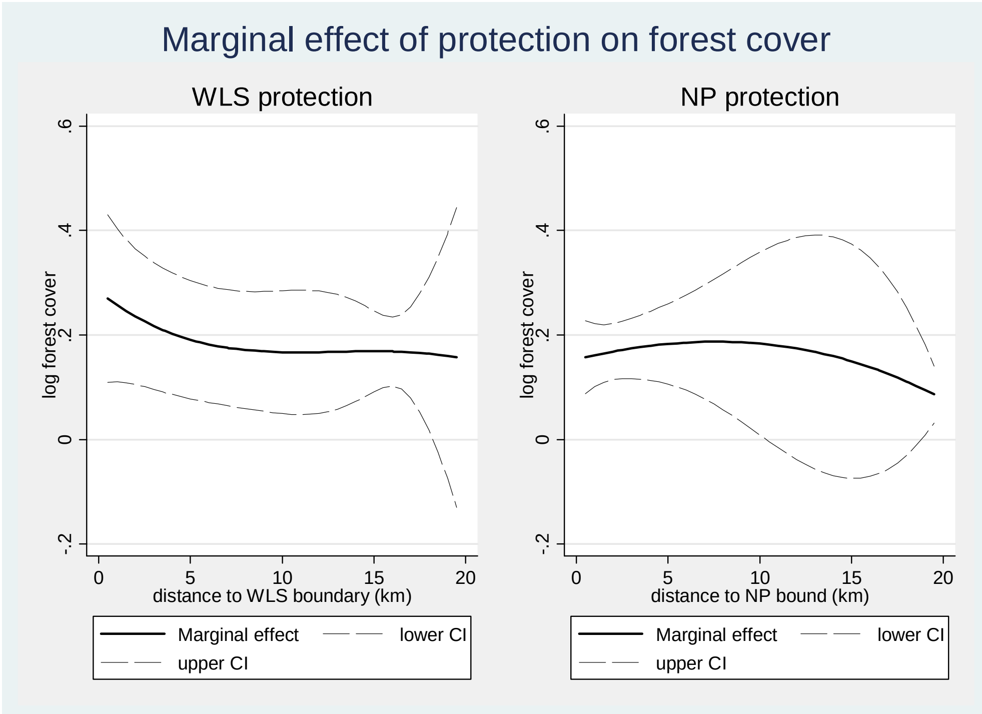


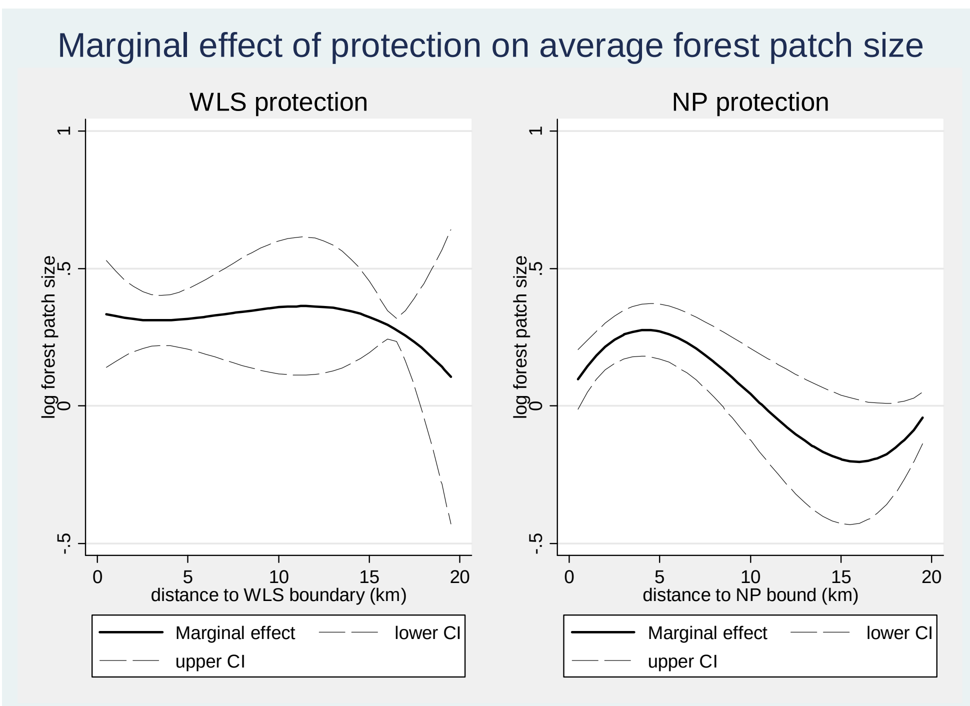
Figure 4: Variation in the timing of protected area designations (North and Northeast Thailand)



Source: IUCN Database of Protected Areas and Thai National Parks Division

Figure 5: Marginal effects of protection on forest cover by distance to park boundaries





Graphs show estimated marginal effects of protection from the fixed effects regression in Equation 5 and Table 5, with added interactions allowing the effect to vary by distance to park boundary (interactions added between share protected as NP and WLS and distance, distance^2 and distance^3).

Table 1: Comparison of predicted forest cover outcomes and fragmentation metrics by enforcement type

Outcomes	**Forest cover / forest patch size (compared to no protection)**		**Fragmentation metrics conditional on forest cover (compared to uniform enforcement case)**			
	(1)	(2)	(3)	(4)	(5)	(6)
Enforcement type	All	Interior areas	Forest patch size	Cleared patch size	Forest patch perimeter to area ratio	Density of cleared patches
Spatially uniform	++	++	0	0	0	0
Boundary-focused	++	+	-	-	+	+
Core-focused	++	+++	+	+	-	-
+ Agglomeration penalty	++	+	-	-	+	+
+ Labor immobility	+	-	-	-	+	+

Table summarizes the directional predictions from von Thünen framework in Section 3. The top panel compares three enforcement types: spatially uniform, boundary-focused, and core-focused. The bottom panel considers additional effects from adding an agglomeration penalty or assuming that labor is relatively immobile. Recall that less fragmentation corresponds to more forest cover and larger patch sizes in the interior. Conditional on forest cover, less fragmentation corresponds to larger forest and cleared patch sizes, smaller forest patch perimeter to area ratios and a lower density of cleared patches.

Table 2: Land cover data sources and remote sensing technologies

Year	**1973**	**1985**	**1992**	**2000**
Source	TRFIC*	TRFIC*	TRFIC*	Thai RFD‡
Satellite sensor type	Landsat MSS	Landsat MSS	Landsat TM	Landsat TM
Satellite resolution	~60 m	~60 m	~30 m	~30 m
Original data	raster	raster	raster	vector
Processing by author	georeferenced to match year 2000 features; spline transformation (RMS error 189.4) †	georeferenced to match year 2000 features; spline transformation (RMS error 196.0) †	georeferenced to match year 2000 features; spline transformation (RMS error 274.7) †	processed and georeferenced by Thai RFD

* Tropical Rain Forest Information Center, Michigan State University. These data were created by a project supported by NASA's Landsat Pathfinder Humid Tropical Forest Project (HTFP) (Samek et al. 2004). The goal was to monitor forest cover in Southeast Asia across time for the purposes of calculating deforestation rates and assessing contributions to global carbon stocks. A detailed description of the process of combining the images into mosaics and interpreting them can be found in Skole et al. 1998.

‡ Thai Royal Forestry Department data; made available online by Marc Souris, IRD.

† The 1973-1992 dataset was originally collected for large area analysis. To ensure a higher degree of local spatial accuracy, I geo-referenced each of the layers to match the year 2000 land-use layer. More than 200 control points were selected for each year based mainly on water bodies that can be seen across time in the images. Images were then rectified using a spline transformation. Similar methodology has been used in previous assessments of forest change across time in tropical countries (see e.g. Sanchez-Azofeifa et al. 2001, Viña et al. 2004). Some error does remain after this process, this is summarized by the root mean squared error terms in Table 1. This is classical measurement error and should not bias the regression coefficients.

Table 3: Data sources for protected areas and other covariates

Variable	Description	Source
Protection variables		
Share WLS	Share of microlandscape protected as wildlife sanctuary (IUCN I)	IUCN World Database on Protected Areas (2007)†
Share NP	Share of microlandscape protected as national park (IUCN II)	
Matching variables		
Avg. slope	Average slope (degrees)	National Geospatial Intelligence Agency-Digital Terrain Elevation Data from USGS Global GIS (1999)
Avg. elevation	Average elevation (meters)	
Max. slope	Maximum slope (degrees)	
Max. elevation	Maximum elevation (meters)	
Distance major city	Distance to nearest major city (pop > 100,000)	ESRI World Cities (2000)
Dist. 1962 major road	Distance to 1962 major road (km)	Digitized East Asia Road Map, U.S. Map Service (1964); data from 1962
Dist. 1962 any road	Distance to 1962 minor road (km)	
Distance rail	Distance to railroad line (km)	Vector Map Level 0 / USGS Global GIS (1997)
Distance major river	Distance to major river (km) (flow accumulation > 5000)	USGS EROS Data Center, Hydro 1k dataset
Distance border	Distance to Thai national border (km)	Vector Map Level 0 / USGS Global GIS (1997)
Ecoregion 2	Share tropical and sub-tropical coniferous forest	WWF Conservation Science Program / USGS Global GIS
Ecoregion 3	Share tropical and sub-tropical dry broadleaf forest	
Waterfall	One or more waterfall points in microlandscape	Mapguide Thailand (2009)

†Years of establishment for protected areas were cross-checked with information from Thailand's Department of National Parks.

Table 4: Summary statistics (matched sample†)

	1973	1985	1992	2000
Protected area (share of microlandscape):				
WLS (IUCN I)	0.052	0.127	0.160	0.196
NP (IUCN II)	0.034	0.094	0.195	0.284
Forest cover (share of microlandscape)				
All microlandscapes (N=3510)	0.947	0.824	0.798	0.774
Any protection:				
WLS-interior (N=334)	0.993	0.953	0.960	0.947
WLS-exterior (N=693)	0.945	0.794	0.778	0.785
NP-interior (N=329)	0.982	0.924	0.923	0.924
NP-exterior (N=1255)	0.937	0.813	0.784	0.767
Fragmentation metrics:				
Average forest patch size (sq km)	20.93	16.26	11.46	13.54
Maximum forest patch size (sq km)	23.39	20.19	19.32	18.58
Forest patch perimeter to area ratio (km/km^2)	7.843	15.53	52.72	20.80
Average cleared patch size (sq km)	0.367	1.700	1.343	2.153
Density cleared patches (# patches/ sq km)	0.103	0.205	0.492	0.289
N	3510	3510	3510	3510

Fixed characteristics:	**Any Protection**		**No Protection**		**Normdiff***
	Mean	SD	Mean	SD	
Avg. slope (degrees)	7.047	3.573	6.442	3.314	0.124
Avg. elevation (m)	644.3	277.0	595.7	277.1	0.124
Max. slope (degrees)	14.05	6.691	12.89	5.867	0.131
Max. elevation (m)	867.5	342.1	800.7	334.6	0.140
Distance major city (km)	112.7	40.02	115.3	41.04	-0.045
Dist. 1962 major road (km)	19.98	17.98	15.83	14.47	0.180
Dist. 1962 any road (km)	26.46	20.64	19.01	15.03	0.292
Distance rail (km)	87.67	38.18	85.60	41.24	0.037
Distance major river (km)	35.70	20.50	29.17	16.38	0.249
Distance border (km)	61.69	44.04	63.13	45.19	-0.023
Ecoregion 2 (share)	0.018	0.118	0.021	0.134	-0.020
Ecoregion 3 (share)	0.340	0.451	0.447	0.483	-0.161
Waterfall (dummy)	0.037	0.189	0.030	0.172	0.026
N	2459		1051		

†Matched control microlandscapes (5 km x 5 km) chosen by one-to-one Mahalanobis matching on the covariates listed above. Matching is conducted with replacement and is done separately for the North and Northeast regions. The beginning universe of microlandscapes includes all those which were complete grid cells, which had more than 50% of forest cover in 1973, which had less than 10% cloud cover in any image and less than 20% water in 1973.

*Normdiff is the difference in average covariate values, normalized by the standard deviation (Imbens and Wooldridge 2009).

Table 5: Protected area effects on forest cover and forest patch size (5 km x 5 km microlandscapes)

	Forest cover		Average forest patch size		Maximum forest patch size	
	(1)	(2)	(3)	(4)	(5)	(6)
Share PA	0.187*** (0.041)		0.245*** (0.041)		0.210*** (0.040)	
Share WLS		0.220*** (0.074)		0.325*** (0.068)		0.239*** (0.071)
Share NP		0.169*** (0.038)		0.200*** (0.050)		0.193*** (0.037)
Microlandscape fixed effects	Y	Y	Y	Y	Y	Y
Province-year fixed effects	Y	Y	Y	Y	Y	Y
N	14040	14040	14040	14040	14040	14040
Adj R^2	0.287	0.287	0.402	0.402	0.329	0.329
F-test (WLS=NP)		0.447		0.152		0.478

Table shows coefficients and standard errors from the fixed effects regression model in Equation 5.
*** p < .01 ** p < .05 * p < 10. Standard errors (in parentheses) are robust and clustered at the district level.

The dependent variables are the log of each outcome. Protection is measured as the share of each microlandscape protected. Fixed effects are included for each microlandscape and for province-year, to capture regional deforestation trends.

The last row gives the p-values for an F-test that the coefficients on share WLS and share NP are equal.

Table 6: Protected area effects on forest cover and forest patch size (10 km x 10 km microlandscapes)

	Forest cover		Average forest patch size		Maximum forest patch size	
	(1)	(2)	(3)	(4)	(5)	(6)
Share PA	0.220*** (0.068)		0.314*** (0.105)		0.319*** (0.088)	
Share WLS		0.251** (0.100)		0.575*** (0.116)		0.350*** (0.115)
Share NP		0.202*** (0.075)		0.168 (0.129)		0.301*** (0.097)
Microlandscape fixed effects	Y	Y	Y	Y	Y	Y
Province-year fixed effects	Y	Y	Y	Y	Y	Y
N	4384	4384	4384	4384	4384	4384
Adj R^2	0.387	0.387	0.521	0.522	0.413	0.413
F-test (WLS=NP)		0.633		0.027		0.654

Table shows coefficients and standard errors from the fixed effects regression model in Equation 5.
*** p < .01 ** p < .05 * p < 10. Standard errors (in parentheses) are robust and clustered at the district level.

The dependent variables are the log of each outcome. Protection is measured as the share of each microlandscape protected. Fixed effects are included for each microlandscape and for province-year, to capture regional deforestation trends.

The last row gives the p-values for an F-test that the coefficients on share WLS and share NP are equal.

Table 7: Protected area effects on fragmentation metrics, conditional on forest cover (5 km x 5 km microlandscapes)

	Average forest patch size	Max forest patch size	Average cleared patch size	Forest patch perimeter to area ratio	Cleared patch density
	(1)	(2)	(3)	(4)	(5)
Share WLS	0.052	0.004	0.082***	-0.167**	-0.011
	(0.032)	(0.013)	(0.028)	(0.075)	(0.016)
Share NP	-0.023	0.001	-0.009	0.076	0.040**
	(0.030)	(0.012)	(0.032)	(0.080)	(0.016)
Forest cover	3.080***	2.657***	-2.499***	-1.565***	0.066
	(0.117)	(0.080)	(0.072)	(0.353)	(0.063)
Microlandscape fixed effects	Y	Y	Y	Y	Y
Province-year fixed effects	Y	Y	Y	Y	Y
N	14040	14040	14040	14040	14040
Adj R^2	0.669	0.868	0.659	0.324	0.339
F-test (WLS=NP)	0.084	0.840	0.005	0.020	0.002

Table shows coefficients and standard errors from the fixed effects regression model in Equation 5, adding forest cover as a control.
*** p < .01 ** p < .05 * p < 10. Standard errors (in parentheses) are robust and clustered at the district level.

The dependent variables are the log of each outcome. Protection is measured as the share of each microlandscape protected. Fixed effects are included for each microlandscape and for province-year, to capture regional deforestation trends.

The last row gives the p-values for an F-test that the coefficients on share WLS and share NP are equal.

Table 8: Protected area effects on fragmentation metrics, conditional on forest cover (10 km x 10 km microlandscapes)

	Average forest patch size	Max forest patch size	Average cleared patch size	Forest patch perimeter to area ratio	Cleared patch density
	(1)	(2)	(3)	(4)	(5)
Share WLS	0.257**	0.033	0.214***	-0.236	-0.017
	(0.109)	(0.054)	(0.071)	(0.162)	(0.022)
Share NP	-0.083	0.051	-0.011	0.140	0.061**
	(0.116)	(0.040)	(0.077)	(0.194)	(0.025)
Forest cover	3.156***	3.148***	-2.993***	-0.053	0.141**
	(0.212)	(0.187)	(0.165)	(0.365)	(0.065)
Microlandscape fixed effects	Y	Y	Y	Y	Y
Province-year fixed effects	Y	Y	Y	Y	Y
N	4384	4384	4384	4384	4384
Adj R^2	0.615	0.775	0.570	0.405	0.419
F-test (WLS=NP)	0.042	0.730	0.009	0.137	0.003

Table shows coefficients and standard errors from the fixed effects regression model in Equation 5, adding forest cover as a control.
*** p < .01 ** p < .05 * p < 10. Standard errors (in parentheses) are robust and clustered at the district level.

The dependent variables are the log of each outcome. Protection is measured as the share of each microlandscape protected. Fixed effects are included for each microlandscape and for province-year, to capture regional deforestation trends.

The last row gives the p-values for an F-test that the coefficients on share WLS and share NP are equal.